UNITED STATES
ENVIRONMENTAL PROTECTION AGENCY

A FRAMEWORK FOR THE ECONOMIC ASSESSMENT OF ECOLOGICAL BENEFITS

February 1, 2002

Acknowledgements

The preparation of *A Framework for Ecological Benefits Assessment* (henceforth *Framework)* was managed under the direction of the Social Sciences Discussion Workgroup, a working group under the Science Policy Council. Initial work on the *Framework* began in 1996 and concluded in 2001.

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Prepared for

Ecological Benefit Assessment Workgroup
Social Sciences Discussion Group
Science Policy Council
U.S. Environmental Protection Agency

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1.0 INTRODUCTION

1.1 BACKGROUND

The Social Sciences Discussion Group (SSDG), convened under the auspices of the EPA Science Policy Council, was initiated to address issues related to the conduct of economic and other social science analyses at EPA. One of their efforts focused on improving the Agency's ability to conduct economic benefit analyses for regulatory cost-benefit or relative benefit assessments. The SSDG identified a need to "improve the Agency's ability to quantify, and, where possible, monetize ecological benefits, including quality of life." A workgroup representing all major EPA programs and environmental media was established to meet that charge.

The workgroup began by surveying EPA offices for completed or ongoing analyses of ecological benefits to determine the current state of the practice within EPA. During this exercise, the workgroup identified the need for a common approach to analyzing ecological benefits and a better understanding of both the scientific and economic techniques used in these analyses.

1.2 PURPOSE AND SCOPE

This document represents a joint effort of ecologists and economists. This document is intended to address the two needs identified above by (1) proposing a common framework for the economic analysis of ecological benefits and (2) discussing the elements of ecological risk assessment and economic benefit analysis. In addition, this document is intended to:

- Promote greater coordination between ecologists and economists working on such efforts;
- Provide an understanding of the approaches and techniques currently in use;
- Suggest additional sources of information; and
- Provide a starting point for individuals who need to assign economic values to changes in ecosystems that have or might result from human activities.

This document is intended to provide general information to EPA staff and others who are interested in the concepts and techniques used to assess and quantify ecological effects of an environmental decision and to monetize ecological impacts and benefits. An important aspect of the document is an introduction of a framework for collaboration between economists and ecologists. The framework presented is not intended as Agency guidance and should not be considered to be promoting any particular benefits methodology.

The document is not designed to be either a "cookbook" or a "how to" manual — it does not provide a step-by-step guidance on the application of specific techniques. Because this document is a framework for estimating the economic value of ecological benefits, it also does not address other possible effects of an action or other perspectives. Specifically, this document does not

discuss non-ecological effects, such as human health impacts or socio-economic effects (e.g., employment, local revenue, growth).

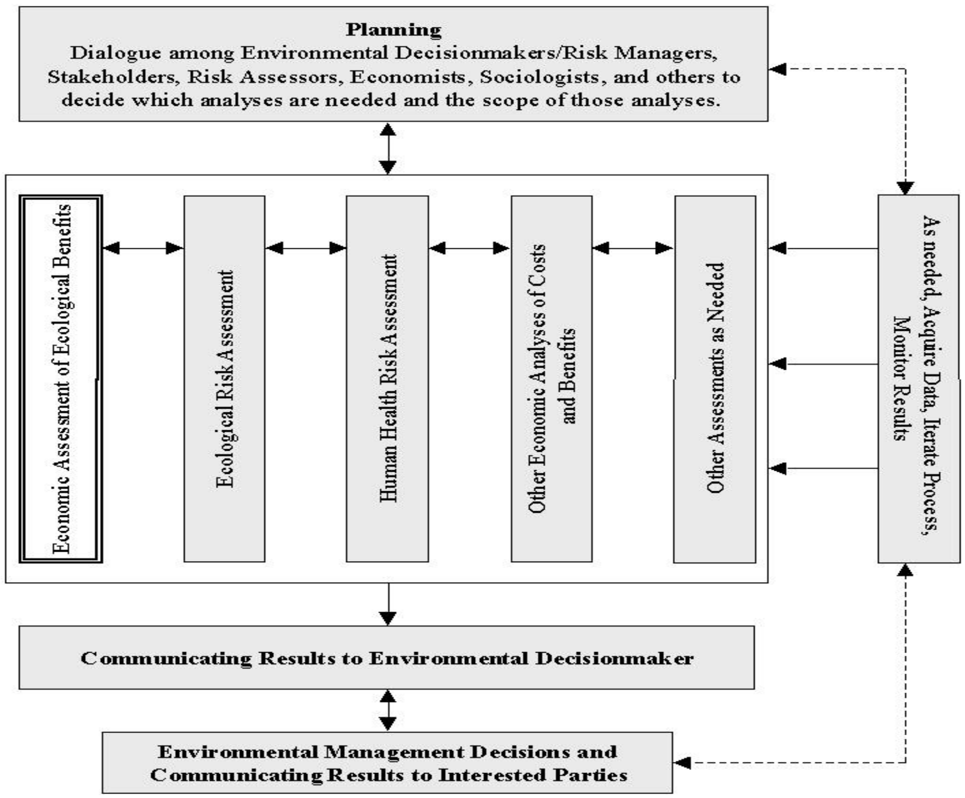
The Framework presented in this document represents one part of a larger process of environmental decisionmaking at EPA, as illustrated by the white box with a double-line border in Exhibit 1. The goal of the Framework is to provide a structure for conducting benefits assessments for the purpose of informing risk management decisions and to meet risk management objectives. It is most applicable for determining, as part of a benefit cost analysis, the ecological benefits of policies or regulatory actions commonly undertaken by governmental agencies such as the EPA. Other types of analyses that might be conducted to inform a decision, such as human risk assessment, environmental justice assessments, and other types of economic assessments, are beyond the scope of this document, but may be included in the decisionmaking process. Discussion of other types of economic analyses can be found in EPA's (2000) *Guidelines for Preparing Economic Analyses*. Other activities included in environmental decisionmaking, such as monitoring and program evaluation, also are not addressed in this Framework.

For an economic analysis of ecological benefits of a decision to be conducted, an assessment of the ecological changes that might result from that decision is needed. In essence, an ecological risk assessment is required, where both beneficial changes as well as potentially adverse changes (the usual implication of the word "risk" in many risk assessment contexts) in ecological endpoints, are evaluated. We have therefore chosen to develop a framework for the economic evaluation of ecological benefits around the framework in EPA's (1998) *Ecological Risk Assessment Guidelines*, which consists of planning, problem formulation, analysis of stresses and effects, and risk characterization.

The Framework proposed in this document identifies the major points in the ecological and economic assessment processes where coordination between the two assessments is needed. This Framework is intended to provide a starting point for approaching such analyses; it does not prescribe a particular method of research or interaction. An important aspect of the Framework is the recognition that ecological risk assessment and economic benefits assessment are distinct disciplines. It focuses on the phases of the respective assessment processes during which communication and coordination between ecologists and economists are needed to ensure an adequate benefits assessment. While it has been assembled based on the experience and judgment of EPA environmental economists and ecological risk assessors, this Framework *per se* has not undergone testing – such as through a program of case studies.

Efforts to manage ecological resources using a place-based (or community-based) approach can differ from a government agency-based approach and may require additional tools not described in this document. Place-based environmental management situations may be characterized by multiple parties; varied or competing objectives; weak or decentralized authorities; and a broad range of potential actions which analyses must seek to narrow and define. Economic approaches involving scenario simulation, multiple agents, or multiple objectives, as well as conventional valuation approaches, may be useful. An EPA, Office of Research and Development effort to integrate place-based ecological risk assessment and economics, which explores the utility of a range of economic tools, is underway (U.S. EPA, 2000b). In addition, an EPA and Science

Exhibit 1
Environmental Decisionmaking Process



Note: This Framework focuses only on the economic assessment of ecological benefits depicted in the clear box with a double outline.

Advisory Board workshop exploring the potential contribution of non-economic approaches to ecological valuation, including approaches from psychology, anthropology and decision science, occurred in May 2001 and will be followed by a report.

1.3 ORGANIZATION OF THIS DOCUMENT

The remainder of this document is organized in seven sections.

Chapter 2: **The Framework:** Provides an overview of the Framework and a description of each of its components, emphasizing the points of coordination between the ecological risk assessment team and the economic assessment team.

Chapter 3: **Important Principles of Ecology and Ecological Assessments:** Defines ecosystems and biological levels of organization. Describes the interactive nature of ecosystems and cascading effects. Provides an overview of prospective and retrospective assessments.

Chapter 4: **The Ecological Risk Assessment Process:** Provides an overview of EPA's Framework and guidelines for ecological risk assessment, emphasizing where coordination with the economists is needed to ensure an adequate economic assessment of possible ecological changes.

Chapter 5: **Background Theory on Valuing Changes to Ecological Resources:** Provides an introduction to how economists define the value of ecological resources and the theoretical basis for estimating changes in these values to measure economic benefits.

Chapter 6: **Economic Assessment of Ecological Benefits:** Provides an overview of the economic assessment of ecological benefits, including detailed information on the types of benefits that might be identified and the techniques available for valuing changes.

Chapter 7: **Issues:** Discusses some additional issues relevant to the economic analysis of ecological benefits, including uncertainty, discounting, aggregation, and equity.

Chapter 8: **References:** Provides a complete listing of the materials used to develop the Framework as well as suggested readings for additional information.

References and Further Reading

U.S. EPA. 1998. *Guidelines for Ecological Risk Assessment*. Risk Assessment Forum. EPA/630/R-95/002F.

U.S. EPA. 2000a. *Guidelines for Preparing Economic Analyses*. Office of the Administrator. EPA 240-R-00-003. September.

U.S. EPA. 2000b. Research Plan for Integrating Ecological Risk Assessment and Economics in Place-Based Decision Making. NCEA-C-0633. National Center for Environmental Assessment, Office of Research and Development, Washington, DC.

2.0 A PROPOSED FRAMEWORK

This section describes a proposed framework for the economic assessment of ecological benefits. It describes the relationship between ecological and economic analyses, identifying many of the points of interdisciplinary coordination between the ecological risk assessment team and the economic analysts. This chapter provides an overview of this Framework (Section 2.1) and a discussion of the key phases and elements of the process, including the planning phase (Section 2.2), several elements of the problem formulation phase (Sections 2.3 through 2.6), conducting the analyses (Section 2.7), and characterizing and presenting results (Section 2.8).

2.1 OVERVIEW

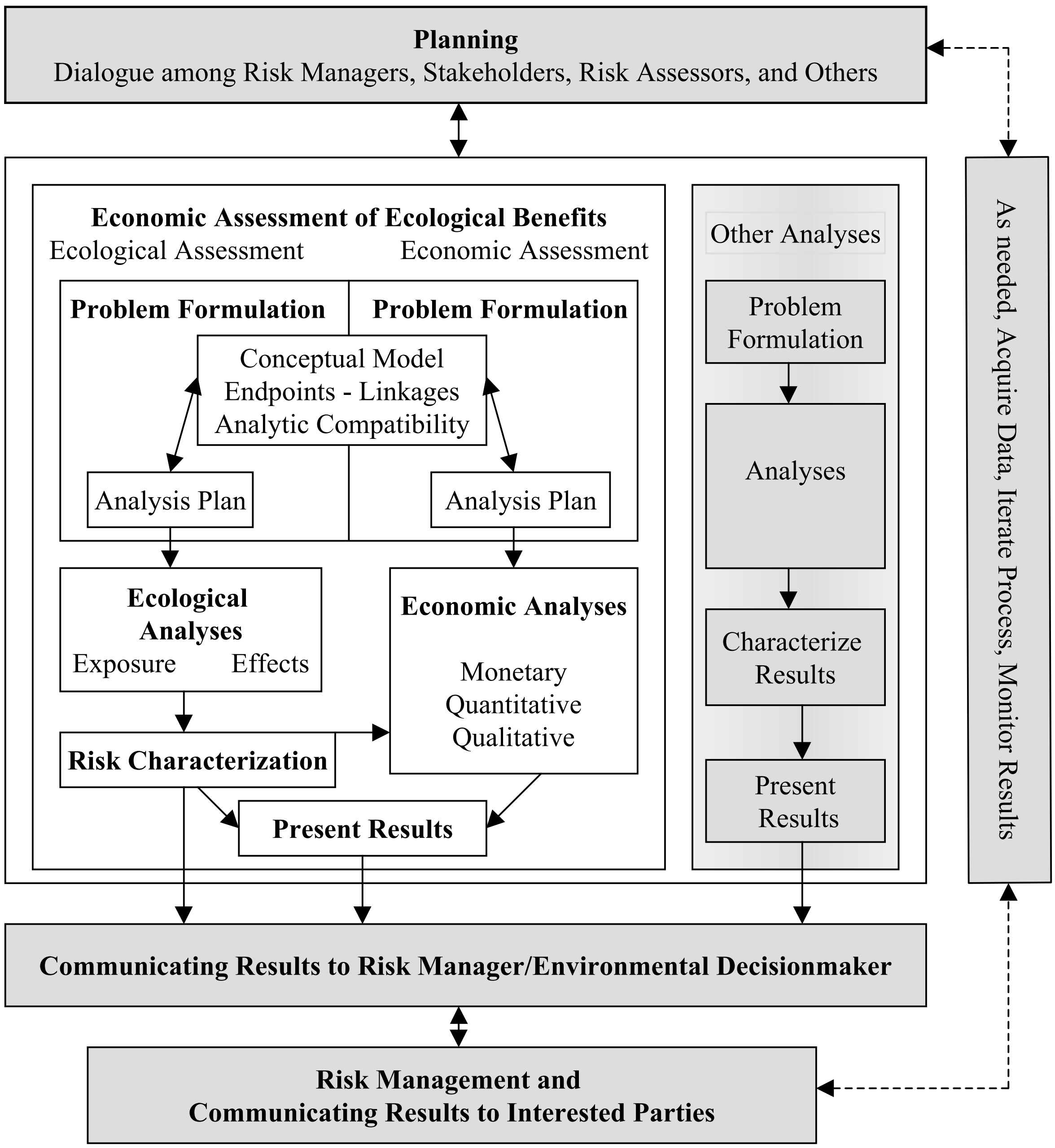
The proposed Framework describes a process by which the ecological risk assessors and economic analysts conduct and coordinate their assessments as recommended by recent EPA guidance. (See U.S. EPA, 1997; 2001.) Interdisciplinary coordination promotes:

- Development of better information for risk managers;
- Greater utility of ecological assessments for economic assessment of ecological benefits;
- Greater relevance of economic assessments to ecological resource issues; and
- Timely and streamlined collection of necessary data.

Exhibit 2 illustrates this Framework within the context of the larger environmental decisionmaking process. While this Framework follows the general phases outlined in EPA's (1998) *Guidelines for Ecological Risk Assessment*, it focuses on the coordination between the ecologists and economists during those phases. EPA's ecological risk assessment process can readily be adapted to an ecological benefits assessment and consists of a problem formulation phase, analysis phase, and risk characterization phase (U.S. EPA, 1998). The steps of the economic benefits analysis, as identified in EPA's (2000) *Guidelines for Preparing Economic Analyses*, are matched with the ecological risk/benefit assessment process. The economic benefits assessment consists of identifying the potentially affected benefit endpoints, quantifying the significant changes to these endpoints, and estimating the economic value of those changes.

As a precursor to conducting an economic assessment of ecological benefits, a planning step occurs where risk managers, stakeholders, risk assessors, economists, and other parties each share their perspectives on the problem to help guide planning of goals, scope, and resources for the assessment. Planning for and beginning an economic benefit analysis simultaneously with the ecological benefit assessment allows for the coordination called for by the Framework and can greatly improve the economic assessment of ecological benefits. The majority of the interdisciplinary coordination between the ecological risk assessors and the economic analysts occurs during the planning and problem formulation phases and again at the end, in the

Exhibit 2
A Framework for Economic Assessments of Ecological Benefits



presentation of the results. The planning and problem formulation steps of the economic assessment of ecological benefits include the following:

- During the planning phase of the overall assessment, the planning team agrees on the scope of the proposed action, the risk management objectives, the management alternatives and policy options that will be explored by the assessments, the types of studies and activities that will occur as part of each assessment, and the basic information required to support each of those studies and activities (Section 2.2). This document only discusses planning as related to the economic assessment of ecological benefits.

- At the beginning of problem formulation, ecologists and economists discuss the information agreed to during the planning phase as it relates to their analyses and begin the design of their respective conceptual models (Section 2.3).

- After identifying the direct and possible indirect ecological changes that might result from options under assessment, economists and ecologists work together to link ecological changes to economic endpoints (Section 2.4).

- Based on the resources available for the assessment and other appropriate criteria, ecologists and economists prioritize the assessment endpoints to identify a subset for quantitative analysis (Section 2.5).

- As ecologists and economists develop their assessment plans, they confer on several issues to ensure that the ecological assessment and the economic benefit analysis are analytically compatible before finalizing their assessment plans at the end of problem formulation (Section 2.6).

- The ecological risk assessment team conducts its assessment. Using results from the ecological risk assessment, economists complete their assessment of the economic benefits of the ecological changes (Section 2.7).

- Both groups characterize and present the results of their assessments (Section 2.8). The ecological risk assessment team might also present results for some questions unrelated to the economic analysis (e.g., which species might serve best as indicators for future monitoring of ecological changes).

Because portions of the ecological risk/benefit assessment and economic analysis of the ecological changes will occur simultaneously, along with any other studies included in the overall assessment, certain parts of the process outlined in this Framework might be repeated until the information is sufficiently precise to be of use in decisionmaking.

2.2 PLANNING

Environmental decisionmaking begins with a planning step. The goal of planning is to identify the context of the environmental decision, the risk management objectives, the options under assessment, the individuals involved, the types of analyses that are needed, what resources are available, and to resolve other questions concerning scope and process. Typically, an ecological risk management objective has an entity, an attribute, and a desired state or direction of preference. The risk management objective helps to focus the assessments on risks that are susceptible to management. The decisionmakers or risk managers meet with staff who can represent the disciplines that might be required, including toxicologists, sociologists, economists, ecologists, risk assessors, engineers, as well as potentially interested or affected parties. The assessment teams might include representatives of Federal, State, local, and tribal governments, commercial, industrial, and private organizations, leaders of constituency groups, and other sectors of the public.

Several types of questions may be addressed during the planning phase:

- What kind of decision is involved (e.g., siting a facility; regulation development)?
- Who is involved in decisionmaking (i.e., who are the risk managers)?
- Who are the stakeholders (those affected and those who are needed to help define the value of ecological benefits)?
- What analyses will be required (e.g., socioeconomic, human health, ecological)?

Products of planning dialogues can include several types of information:

- The type of management goals at issue (e.g., benefits exceed costs, restore striped bass population in the Potomac to its 1940 level);
- The exact nature and timing of the decisions to be made and who will make them;
- The analyses to be performed and which analyses must be coordinated;
- Agreement on scope, complexity, and focus of the assessments, including the expected output; and
- The timeline and technical and financial support available to conduct the assessments.

Depending on the context, decision, management goals, and other attributes of the decision at hand, one or more of several different types of analyses might be called for, including human health risk assessment, ecological risk assessment, economic assessment of health risks, economic assessment of ecological changes, environmental equity or justice, and others (Exhibit 1). This Framework addresses only the economic assessment of ecological benefits, and thus focuses on a subset of the possible analyses that might be conducted during the environmental decisionmaking process.

During planning, the economists, ecologists, toxicologists, and staff representing other involved disciplines can contribute to a definition of the scope of the problem and alternatives to be considered. For example, economic analysts might provide preliminary information from market research efforts that indicate what chemicals/products are being manufactured and used and in what amounts. Producers and consumers and chemical uses can be identified. Risk assessors might use this information to determine where (geographically) exposures might take place, the magnitude and frequency of exposure (how chemicals/products are used), and exposed populations (both human and ecological).

Economic analysts might also provide insights on economically feasible substitutes that might be used if a chemical or product is no longer available, projecting potential market shares and production and use volumes. Socioeconomic information on the conditions of affected communities and/or populations can provide insight as to what the concerns and potential risk management alternatives are for a particular community. Economic factors, in addition to others such as political and cultural factors, also come into play when identifying feasible risk management alternatives.

Ecologists can provide information on likely direct and indirect ecological impacts, sensitive ecosystems or receptors, and likely spatial and temporal scales over which ecological changes might be expressed. With information on environmental persistence and bioaccumulation potential for chemicals, ecologists can evaluate whether food-chain analyses might be important to the assessment. They also can provide initial judgments on likely magnitude, severity, and reversibility of anticipated effects.

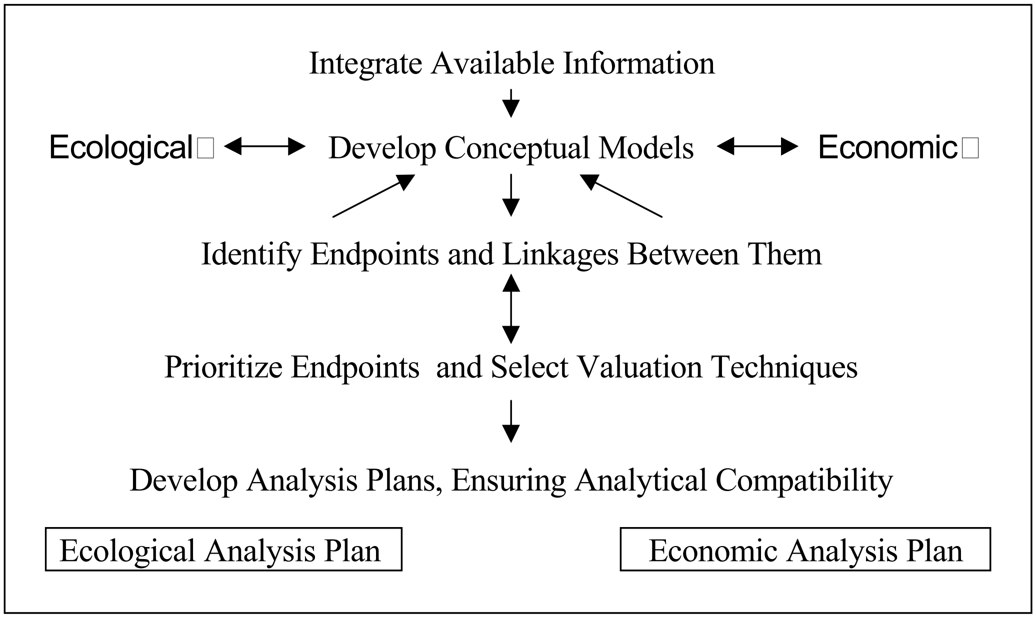
By the completion of the planning step, agreements have been made about several aspects of the assessment: (1) the management goals, (2) the range of management options that the ecological and economic analyses are intended to support, (3) characterization of the decisions to be made given the management goals, (4) the focus, scope, and complexity of the assessments, and (5) resources available to conduct the assessment. At a general level, there must be agreement on the spatial and temporal scale of the assessments. At this point, the groups that will need to coordinate with each other also are identified.

2.3 PROBLEM FORMULATION – INITIAL COORDINATION

2.3.1 Problem Formulation

Once planning is complete, the risk and economic assessments can begin. In its (1998) *Guidelines for Ecological Risk Assessment*, EPA describes problem formulation as the first phase of an ecological risk assessment. The same principles described in those *Guidelines* can apply to problem formulation phase for an economic assessment. Problem formulation in this Framework includes integrating available information, selecting ecological and economic assessment endpoints, developing of a conceptual model linking the proposed actions to ecological and economic endpoints, and developing analysis plans for the ecological and economic benefit assessments (Exhibit 3).

Exhibit 3
Problem Formulation for Economic Assessment of Ecological Benefits



This document focuses on four parts of the problem formulation phase: (1) initial coordination in defining the problem; (2) developing the conceptual models that identify linkages between ecological and economic endpoints; (3) prioritizing endpoints for quantitative and qualitative assessment, and (4) ensuring analytic compatibility between the ecological risk and economic assessment plans. Exhibit 4 provides a detailed illustration of the problem formulation phase.

Questions Asked During Problem Formulation

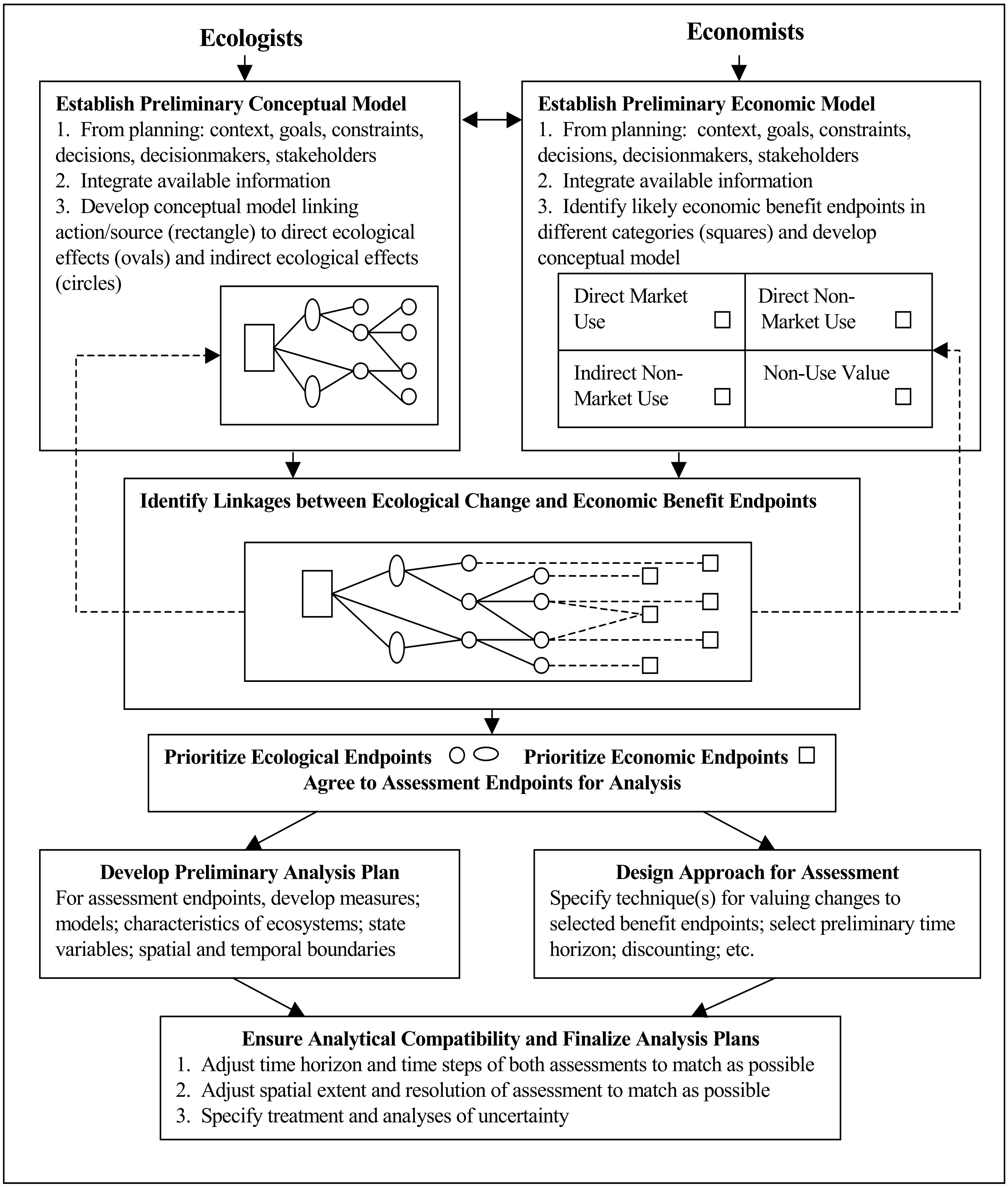
- What laws already protect what entities?
- What are the policy considerations (law, corporate stewardship, societal concerns, intergenerational equity)?
- What is the nature of the problem?
- What is the context of the assessment?
- What is the likely scale of the assessment in time and space?
- What is the starting information like?
- Develop a conceptual model of the problem, context, scale, then:
  -What are the most obvious assessment endpoints (i.e., relevance to management goals)?
  -What additional endpoints are ecologically and socially relevant (e.g., laws can be good indicators of societal values) or important to stakeholders?

2.3.2 Initial Coordination

During the preliminary dialogues among the risk managers, risk assessors, economic analysts, and stakeholders in the planning stage, the ecological assessors and economic analysts will have begun to formulate ideas about the tasks ahead of them. Fundamental to beginning problem formulation is a clear description of the environmental decision at hand, the options under evaluation, the types of actions, and the initial changes those actions would cause in the abiotic environment (e.g., land use, water flow, chemical concentrations). The ecologists will begin to formulate an understanding of the proposed options and the types of ecosystem structures and services that might be affected by those options; the economists will develop some ideas as to the most obvious economic benefits that might accrue. The discussions initiated during the planning stage between ecologists and economists to define the scope of the overall assessment will continue throughout the problem formulation process. The sharing of information and ideas can be particularly helpful in developing comprehensive conceptual models.

For both the ecological and economic assessments, problem formulation begins by integrating the information from the initial discussions of the problem, including the context of the assessment, its goals and constraints, the decisions to be made, and which stakeholders are involved. Separate ecological and economic assessment teams are assembled to include the required areas of expertise and also possibly to include stakeholder representatives. Careful consideration about who will participate and how they will participate is best done up front.

Exhibit 4
Problem Formulation



2.4 LINKING ENDPOINTS DURING PROBLEM FORMULATION

After the initial coordination, each assessment team begins to develop an explicit conceptual model of their part of the analysis. The ecologists begin by tracing the consequences of the proposed actions from the sources through the initial changes produced in the physical and chemical characteristics of the environment, direct effects on ecological entities, and then the cascade of secondary ecological effects that might follow.

The economic benefit assessment is based on the premise that actions affecting the state of an ecological resource, measured in terms of changes to the ecological assessment endpoints, will result in changes to the goods and services provided by that resource (i.e., changes to the economic benefit endpoints). Because of this connection, economists need to work with ecologists and other scientists in determining what economic benefit endpoints are likely to be affected and estimating the magnitude of those effects. By working with economists to define the economic benefit endpoints, ecologists can help ensure that ecologically significant but less obvious or less direct effects are not overlooked by the economic benefit analysis. Furthermore, as ecologists gain a better understanding of the objectives and process of the economic benefit analysis, they might be able to provide information and data that are better suited to the needs of the economist.

What Is An Endpoint?

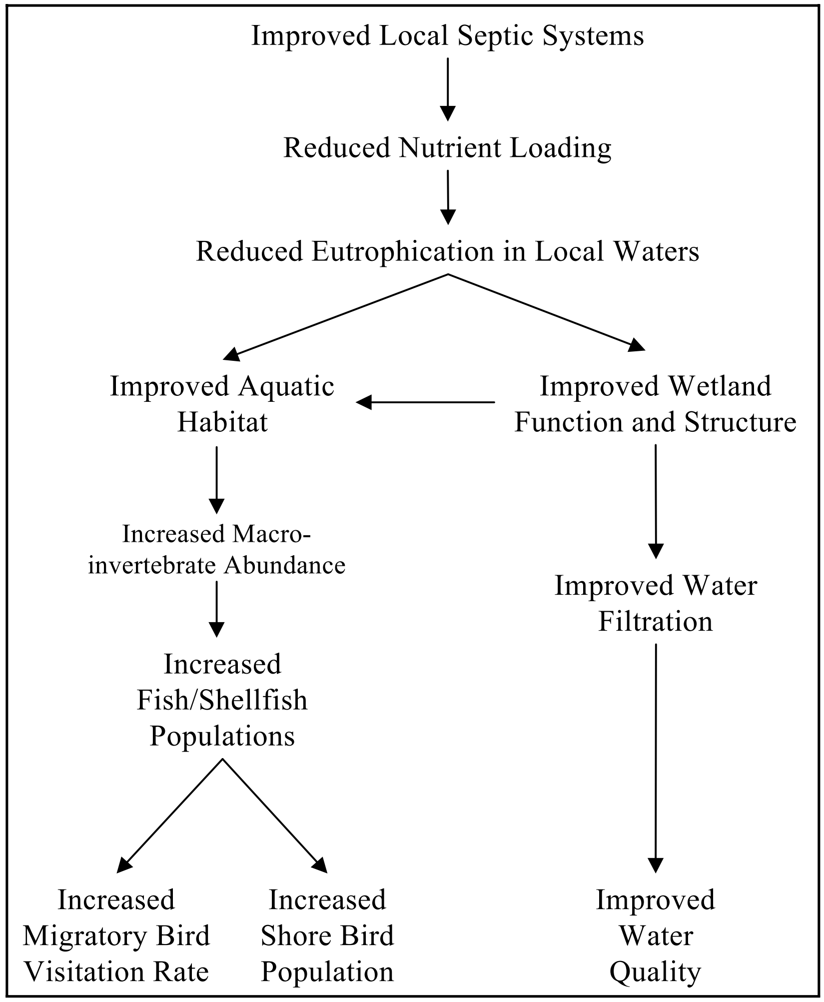
Endpoints differ by discipline. Ecological assessment endpoints are explicit descriptions of the actual environmental attribute that is expected to change in response to an action. Ecological assessment endpoints are operationally defined by an ecological entity and its attributes. Changes to ecological assessment endpoints are estimated from analyses of both direct and indirect effects of the action and in the context of a benefits assessment, are used to estimate changes in the economic benefit endpoints. Economic benefit endpoints are the goods or services provided or supported by the ecological resource, directly or indirectly, that have *economic value* (see Section 5.3) to society, such as recreational fishing.. Changes in the economic benefit endpoints are used to assess the economic value of the action under study.

2.4.1 Conceptual Model of a Cascade of Ecological Effects

The ecologist outlines ecological changes that might result from one or more decisions and actions. Exhibit 5 illustrates a simple, preliminary conceptual model that might be drawn up to depict possible ecological benefits of improving local septic systems, one of many possible risk management actions. The diagram in Exhibit 5 is a substantial oversimplification of a conceptual model for purposes of illustration only. For additional examples and explanation of the development of conceptual models for ecological risk assessments, see Chapter 4.

The conceptual model traces the sequence of changes from the initial direct effect of reduced nutrient loading to surface waters to the consequences of that effect, here described as reduced eutrophication (i.e., nutrient enrichment) of local waters, which in turn would lead to improved aquatic habitat for fish and shellfish populations, and that, in turn, might support larger populations of breeding and migratory shorebirds. The reduced eutrophication also would be expected to improve the condition and areal extent of wetland vegetation and the ability of that vegetation to filter sediments, nutrients, and contaminants out of the water before the water reaches rivers and lakes.

Exhibit 5
Diagram Linking Action to Cascade of Ecological Effects



The cascade of indirect effects and interactions among the affected entities, which is characteristic of ecosystems, can be difficult for economists and non-ecologists to envision. As the ecological risk assessment team develops their conceptual model, they should meet frequently with the economic analysts to explain the ecological relationships and interactions represented by the conceptual model. Communicating with economists during the process of developing the ecological conceptual model, starting with the most simple preliminary model, will greatly improve economists' understanding of the ecological assessment and help economists to define appropriate economic benefit endpoints.

One difference between the ecological conceptual models for risk assessments designed to identify environmental concentrations of concern or to set cleanup levels and those conducted to support economic analyses deserves note here. A conceptual model used to design an ecological risk assessment to establish environmental concentrations of concern or cleanup levels at a specific site often focuses on the most sensitive and exposed (i.e., vulnerable) receptors or processes in the ecosystem at the site. The goal is to identify contaminant concentrations in environmental media that are unlikely to cause adverse ecological effects. In Exhibit 5, the most vulnerable endpoint might be shellfish populations. To document an environmental concentration of concern, for example, an ecological risk assessment might be able to stop there, and consider only increased shellfish populations as a goal. A conceptual model that will be linked to an economic assessment of ecological benefits should attempt to identify all of the direct and indirect effects of an environmental decision. The additional ecological effects depicted in Exhibit 5 are added to allow the economic analysis to capture a more complete spectrum of ecological benefits.

When developing a conceptual model to support an economic assessment of ecological benefits, the ecologists should also consider the possibility for adverse effects. What is beneficial for one species can be detrimental to another, as has sometimes occurred when land is managed for the benefit of a single species (e.g., game species such as deer).

2.4.2 Identifying Preliminary Economic Endpoints

EPA primarily relies on an effect-by-effect approach for estimating the benefits of a policy option (U.S. EPA, 2000a). This approach involves identifying the major beneficial effects of an action (e.g., various types of improvements to activities or functions of ecological resources), assessing the economic value of each of these improvements independently, and summing up the individual values to provide an estimate of the total benefits. Identifying the major beneficial effects that will be examined in detail in the benefit analysis involves several steps. Before economists determine what effects will be examined in the benefits analysis, they first try to identify all possible effects. Based on early discussions with ecologists during planning and early in the problem formulation stage, economists begin identifying potential economic benefits by thinking about the action under study, reviewing analyses of similar actions, and working with the ecological assessment team and their preliminary conceptual models to understand what ecological changes are expected.

Economists might identify various types of benefits stemming from changes to ecological resources. The economic benefit endpoints are generally viewed as services or uses provided by ecological resources. The types of benefit endpoints include direct market uses, direct non-market uses, indirect non-market uses, and non-use values. Chapter 5 describes this categorization of economic benefit endpoints and provides example services and uses that might be considered. This categorization of potential economic benefit endpoints reflects how directly each service or use is experienced by an individual and the extent to which an individual can be restricted from enjoying the service or use. Characterizing the economic benefit endpoints in this way helps economists identify appropriate valuation techniques for each endpoint.

One method of identifying economic benefit endpoints is to develop a table that links likely ecological changes to impacts on human uses and values (e.g., see U.S. EPA, 1995; King, 1997, and the U.S. Army Corps of Engineers (USACE) approach described in Cole *et al.,* 1996). For example, groundwater discharges contribute to the flow or stock of water in wetlands, streams, rivers, and lakes. As a result, a policy that changes the quality or quantity of groundwater might affect the services provided or supported by these surface water resources, such as drinking water supply and recreational boating, fishing, and hunting. The economist looking at changes to groundwater thus might list increased availability of drinking water, increased opportunities for river recreation, or improved quality of recreational fishing as potential economic benefit endpoints.

Exhibit 6 provides a simple illustration of how such an approach might be used to develop a preliminary list of economic benefit endpoints. Some of the potential economic effects listed in

Exhibit 6 are not very specific and may need to be refined before the economic value of the effect
can be estimated, but this listing provides a starting point for economists to work with ecologists in identifying the economic benefit endpoints that are linked to the policy or action. As discussed in the next subsection (2.4.3), the Framework presented here recommends using the conceptual model developed for the ecological risk assessment to identify appropriate economic endpoints and to facilitate coordination between ecologists and economists.

Exhibit 6
Hypothetical List Linking Ecological Changes to Potential Economic Effects

Ecological Change	Economic Effects
Reduced turbidity of water body	Increased commercial and recreational fish harvests Reduced water treatment costs Improved aesthetic quality of the water
Increased wetland acreage	Reduced costs of storm damage Improved recovery after storm-induced combined sewer overflows Reduced water treatment costs Increased commercial and recreational fishery and shellfish harvests

2.4.3 Identifying and Defining Linkages

With the preliminary work described above, the ecologists and economists can now work together to extend the conceptual model developed by the ecologists to include corresponding economic benefit endpoints that might be affected. The thoroughness of the economic benefit analysis depends on identifying and defining as many of the linkages between changes to ecological resource(s) and changes to the economic benefit endpoints as possible. Identifying and defining these linkages begins with a qualitative understanding of the relationships and interactions that occur within the natural system. As noted earlier, the ecological risk assessment team can help the economist understand these relationships by communicating with the economist during the development of the conceptual model. The conceptual model should clearly identify the direct and indirect effects of an action and form the basis for explaining the ecological cascade of effects to the economic analysts.

The first attempt to link economic benefit endpoints to ecological endpoints is likely to result in four types of findings: (1) good matches between some ecological changes and some economic benefit endpoints; (2) a series of ecological changes that the economists had not considered; (3) a series of potential economic valuation endpoints for which there are no clear connections to the ecological changes represented in the conceptual model; and (4) a series of endpoints whose economic benefits are ambiguous or unmeasurable. Economists can then consider the additional ecological changes represented in the conceptual diagram and identify appropriate economic benefit endpoints to reflect those changes. Ecologists can consider whether they have overlooked any effects suggested by the economic endpoints and whether some of the ecological endpoints could be redefined to provide a more clear connection to the remaining economic endpoints.

Example of How this Step Might Work

It might be reasonable for economists to list recreational and commercial fishing as a potential economic benefit endpoint for any change in water quality. To define linkages between ecological and economic endpoints, economists work with ecologists to determine if a proposed change in water quality will actually have any impact on fish populations. Once a link is identified, the nature of the relationship needs to be defined. For example, ecologists and economists might discuss which fish species are most sensitive to the change (e.g., game fish such as trout), a threshold for effects, and the relationship of the magnitude of the change to likely population size.

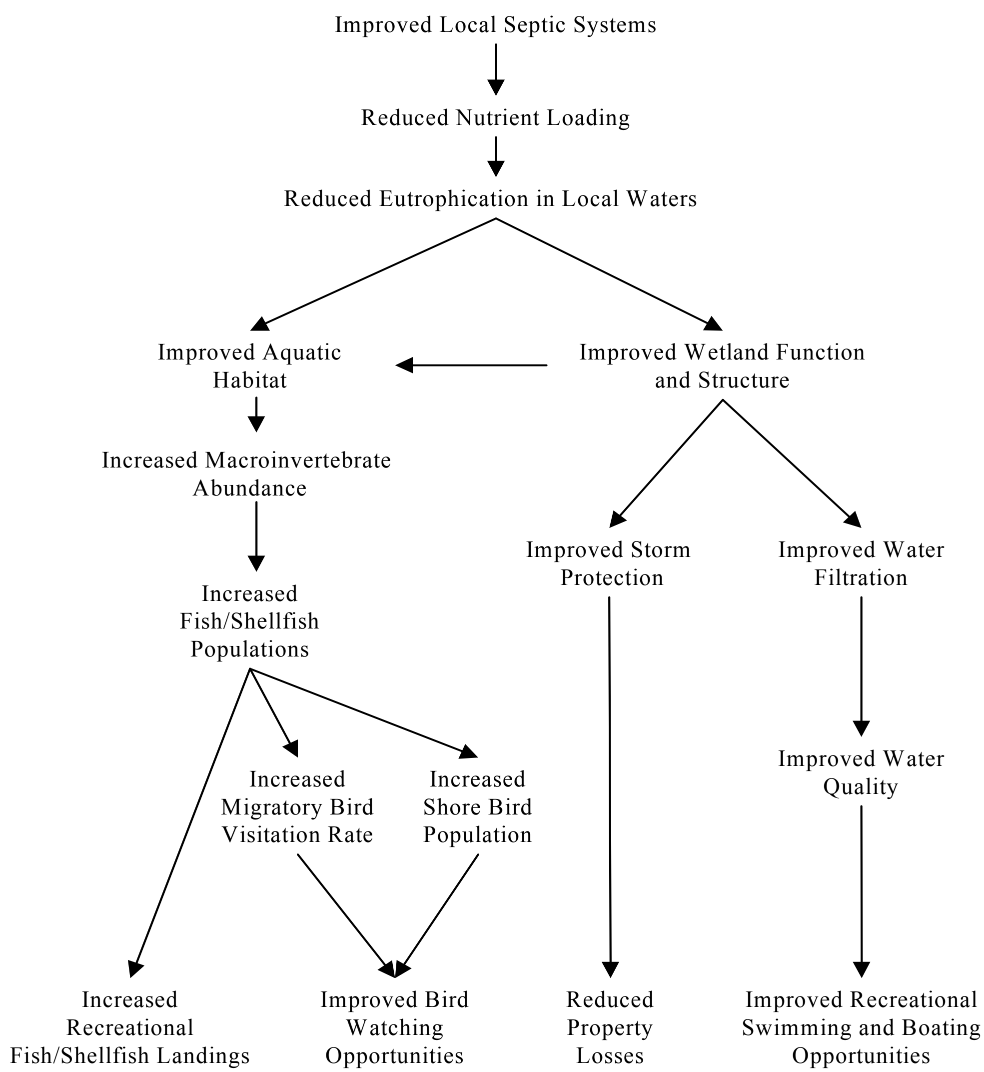
Importance of "Obscure" Ecological Changes

The process of linking economic benefit endpoints to ecological endpoints can be challenging. Economic valuation expresses benefits in terms of human values. Improvements considered important by ecologists (e.g., increased biodiversity of a macroinvertebrate stream community, see Exhibit 5) might not necessarily be appreciated by the public. Therefore, it can be helpful to describe the cause-and-effect relationship between seemingly unimportant ecological changes and changes with obvious implications for humans. For example, an increase in fishing opportunities can result from increased fish populations that occur because the macroinvertebrate community is healthier. Indirectly, then, the change in the macroinvertebrate community is valued by the increase in fishing opportunities.

The ecologists and economists continue to work together to refine the match between ecological and economic endpoints. When a new connection (or linkage) is identified, the economic valuation endpoint is added to the expanded conceptual model along with a diagrammatic explanation of the connection to one or more of the ecological changes represented in the ecologists' conceptual model. At this point in problem formulation, the goal of collaboration is to be all inclusive and to extend the conceptual diagram to include as many linkages between ecological changes and economic benefit endpoints as is reasonable. It is an iterative process, as the dotted lines looping from the joint conceptual diagram back into the individual disciplines in Exhibit 4 indicate.

The expanded conceptual model produced through this collaborative process should identify the economic benefit endpoints that are likely to be affected and the pathways by which these effects are realized. Exhibit 7 provides an example of how the conceptual model presented in Exhibit 5 might be expanded to include economic benefit endpoints. For example, stormwater protection is added to the outcomes of improved wetland function and structure, which in turn reduces property losses during storms. Other economic use endpoints are linked to several ecological effects. Exhibit 7 illustrates the linking of economic benefit endpoints to ecological effect endpoints for a very limited set of endpoints. This simplified example considers only a single change, reduced nitrogen loading from local septic systems, and only some of the potential linkages between the ecological effect of reduced eutrophication and changes experienced by some of the economic benefit endpoints. The iterative process of refining the list of economic benefit endpoints and their links to ecological effects endpoints continues until agreement is reached that the important elements of the problem are represented.

Exhibit 7
Expanding a Conceptual Model to Include Linkages to Specific Economic Benefit Endpoints



The economists and ecologists also should consider whether there are any feedback loops between ecological and economic endpoints. For example, if improved water quality results in increased fish populations and increased recreational fishing, the potential effects of overfishing on the same fish populations should be considered.

For each economic benefit endpoint, economists and ecologists must define the linkages and relationships between the ecological changes and the economic endpoints in sufficient detail to move forward with prioritization of the endpoints and development of the analysis plans. In defining the linkages, the economist must gather sufficient information from the ecological risk assessment team to be able to estimate the potential magnitude of the change to each economic endpoint and to determine what techniques might be appropriate for estimating the monetary value of the change.

2.5 PROBLEM FORMULATION – PRIORITIZING ENDPOINTS AND SELECTING VALUATION TECHNIQUES

Time and resource constraints generally require that the ecological and economic benefit analyses focus on fully explaining and quantifying changes to only a limited number of endpoints. This section discusses where coordination between ecologists and economists is needed to prioritize ecological and economic endpoints for analysis.

2.5.1 Prioritization Criteria

From the ecologist's perspective, EPA has defined several criteria that can help identify the most important and useful endpoints for an ecological risk or benefit assessment (U.S. EPA, 1998):

- Ecological relevance of an endpoint (e.g., importance to maintaining ecosystem structure or function);

- Susceptibility of the endpoint to the proposed actions. Susceptibility depends on the sensitivity of the endpoint to the action (e.g., plants are particularly sensitive to herbicides) and on the likelihood of exposure (e.g., is there a pathway by which the stress can reach the organisms?); and

- Relevance to the management goals established during the planning phase.

EPA's (1998) *Guidelines for Ecological Risk Assessment* discuss these criteria, and examples are provided in Chapter 4 of this document. Sometimes, the endpoints that best fit the ecological selection criteria may not be those best suited for economic valuation. The ecologists might include endpoints that the economists do not plan to address (e.g., which ecosystem process offers the highest signal-to-noise ratio for purposes of monitoring change after an environmental decision is implemented).

The prioritization of benefit endpoints for the economic benefit analysis requires consideration of several factors:

- Type of information required by decision-makers;

- Expected magnitude of the change in the economic value of one benefit endpoint relative to other endpoints;

- Anticipated uncertainty associated with the predicted change and value of the change for the benefit endpoint relative to other endpoints;

- Variation in the change to each benefit endpoint under alternative policy scenarios; and

- Analytical feasibility considerations.

Economists generally want to estimate the dollar value of those changes that represent the greatest economic benefits. However, economists must also consider whether there are likely to be significant differences in the change to the benefit endpoint under alternative policy options and whether stakeholders or decision-makers will need information on the benefit endpoint even if the magnitude of the economic value of the change is relatively small (U.S. EPA, 2000a). If the purpose of conducting the benefit analysis is to choose between alternative policy options, the ecological changes that experience the greatest variation in economic value across policy options might be the most important to address in detail in the economic assessment of benefits. Similarly, there may be particular benefit endpoints that are of interest to stakeholders or decision-makers that are given priority for detailed consideration in the economic assessment of benefits.

The potential economic value of the change to each economic endpoint will depend on the magnitude of the ecological change or changes linked to that economic endpoint. The ecological risk assessment team can help economists determine which benefit endpoints are likely to experience the largest or most wide-spread changes or vary most significantly across policy options. In some cases, an ecological change that is relatively small in magnitude may provide large economic benefits. By working with the ecologists to develop the expanded conceptual model and to prioritize endpoints, the economists can be sure that those ecological changes are included in the conceptual model. Similarly, ecologists can ensure that economists do not overlook ecological changes that might appear to be relatively minor but in fact have widespread or long-term consequences. Thus, by working together, the ecologists and economists can make sure that the "joint" conceptual model encompasses a comprehensive suite of economic benefit endpoints.

What can be measured in the ecological assessment will dictate, in part, what ecological changes and economic effects are captured by the economic benefit analysis. Economists need to understand the data traditionally collected and developed by an ecological assessment and determine how well these data address the data needs of the economic analysis. Ecologists also need to better understand the data needs of the economic benefit analysis. Better communication between the disciplines during problem formulation allows both the economists and the ecologists to identify opportunities to slightly expand the scope of the conceptual model, possibly adding additional ecological endpoints or altering slightly the types of information developed to provide for significant improvements in the economic benefit analysis.

Changes to endpoints that are better understood and more certain are given higher ranking than changes to endpoints that are less well understood or more variable. Economists want to provide a more certain estimate of the benefits of an action to better support policy decisions. However, where changes are potentially very large, they need to be considered even though they might be highly variable or not well understood.

The number of benefit endpoints that can be evaluated in detail in the economic benefits analysis depends on the type of assessment conducted for those endpoints as well as the time and resources available for the economic assessment. Toward the end of this prioritization process, the ecological and economic benefit endpoints have been roughly ranked according to ecological and economic importance. The ecologists and economists will need to confer and compare their rankings of linked endpoints. Where the rankings agree (e.g., an ecological endpoint ranked as high priority has an explicit linkage to an economic endpoint that also is ranked as high priority), the discussions will be short. Where the rankings disagree (e.g., an ecological endpoint is listed as high priority, but the linked economic benefit endpoint is listed as low priority), further discussion might help one or the other group change their ranking. Or, the ecologists might decide to evaluate a high priority endpoint even though the economists will devote little attention to it. Any high priority economic endpoints will need the supporting ecological analyses to be conducted. At the end of this step, both groups have identified those endpoints on which they will focus their efforts.

The next section discusses the decision criteria used to determine if a monetary, quantitative, or qualitative economic benefit assessment is appropriate and how communication with the ecological assessment team supports that decision.

2.5.2 Monetized, Quantitative, and Qualitative Assessments

The following issues are considered in determining whether a monetized, quantitative or qualitative assessment of the economic benefits associated with each endpoint is appropriate:

- Need for a dollar value estimate of the ecological benefits associated with the action;
- Availability of appropriate economic assessment techniques (e.g., techniques for non-use values currently not available);
- Compatibility of available benefit assessment techniques with the data and outputs of the ecological assessment; and
- Availability of relevant ecological and economic data.

The appropriate type of assessment of the economic value of ecological changes is often determined by answering the questions posed during the planning process: "Why is the analysis being conducted?" "What are the questions the analysis will address?" and “What are the decisions that these analyses will inform?” In some cases, a qualitative or quantitative assessment of the economic value of ecological benefits, rather than a monetized assessment, may be all that is necessary to support the decisionmaking process.

If a monetary estimate of economic benefits is needed, economists must determine if they can provide a monetized measure of value of the change for each economic endpoint within the time and resource constraints of the overall analysis. The ability of economists to provide a monetized measure of benefits associated with any particular endpoint depends on the applicability of economic valuation techniques to the situation and the availability of the data necessary to support the analysis. Most often, the ability of EPA to provide a monetized measure of benefits will depend on the applicability of existing value estimates in the literature for use in a benefits transfer analysis (see Section 6.3.9 on Benefits Transfer). Once economists identify the economic benefit endpoints for which appropriate economic data and techniques are available, they can work with ecologists to determine if the ecological assessment can provide the information needed on the ecological changes. For example, to apply a value from a previous economic study to the current economic benefit assessment (i.e., to use benefits transfer) the economist will need a comparable measure of the ecological change as used in the original valuation study.

A thorough economic benefit assessment focuses not just on the effects that can be monetized, but on the full scope of effects. Many ecological services are not provided through markets or are not readily associated with market transactions. As a result, it may be more difficult or impossible to provide a dependable monetized measure of the benefits associated with many ecological changes. For those benefits that are not monetized, a qualitative, and when possible quantitative, assessment of the economic value of the changes provides a measure of a service's importance and the degree of change, even when a dollar value cannot be assigned to that change. For those benefits that are monetized, including a thorough qualitative and quantitative discussion of the changes that are valued supports the dollar valued generated by the analysis. Again, the ecological assessment team can help economists determine if some type of quantitative assessment of the change is possible, or if a qualitative assessment must suffice.

It might not be possible for the ecological assessment or economic analysis to assess the change to some of the ecological endpoints considered key by the ecologists' selection criteria. For example, certain ecological services are too complex and too poorly understood to quantify or monetize potential changes (e.g., carbon cycle, nitrogen cycle). Other ecological benefits are difficult to characterize and quantify (e.g., species habitat, pollination, microclimate control). Principe (1995) termed these "neglected" benefits because they are seldom included in benefit assessments. EPA's resource book on *Assessing the Neglected Benefits of Watershed Management Practices* (U.S. EPA, 2000b), provides further examples of ecological benefits that, although important ecologically and economically, are rarely included in benefit assessments because they are hard to characterize and quantify. Nonetheless, these services are extremely important to our economic and human welfare (Dailey *et al.,* 1997; Dailey, 1997).

Even though the ecological assessment and economic analysis are not able to estimate a specific change, the economic benefit assessment should recognize that the potential impact or improvement to the ecological service might have great value to society. In these situations, the economic benefit analysis should include a detailed qualitative discussion on the effect of the policy or action on the ecological and economic endpoints and, if possible, describe the potential economic significance of these changes to society.

2.6 PROBLEM FORMULATION – ENSURING ANALYTICAL AND DATA COMPATIBILITY IN THE ANALYSIS PLANS

Problem formulation for both the ecologists and the economists culminates in the development of the analysis plan for the assessment. Since the ecological assessment is often the main source of information for the economist regarding how a specific action or change has affected or is likely to affect an ecological resource, it is imperative that the analysis plans are compatible. Having compatible analysis plans means having a common understanding of the baseline from which effects are measured and the scenarios or policy options to examine, ensuring that the outputs of the ecological assessment meet the needs of the economic analysis, defining consistent spatial and temporal scales for the analyses, and determining how uncertainty will be treated by the analyses.

Having compatible analysis plans ensures that:

- Outputs from the ecological assessment are compatible with the needs of the economic benefit analysis;
- Findings of the ecological assessment and economic benefit analysis are analytically consistent; and
- Conclusions of the ecological assessment and economic benefit analysis meet the needs of the decisionmakers as defined during the planning stage.

During the prioritization of ecological and economic endpoints, ecologists and economists will have discussed what information is required by the economist and if that information can be derived from or developed during the ecological assessment. During the analysis design phase, ecologists and economists formalize what information is needed by economists and determine how and when that information will be provided. They must also agree on how changes will be described or measured (e.g., from what baseline, under what scenarios, at what level of spatial or temporal detail) and how any limitations or uncertainties will be represented. The following subsections address each of the issues that must be discussed when designing the analysis plan to ensure compatibility between the ecological assessment and the economic analysis.

2.6.1 Establishing the Baseline and Alternative Scenarios

The baseline from which effects are measured and the specific scenarios or policy options to consider must be consistent between the ecological assessment and the economic benefit analysis. EPA's (2000) *Guidelines for Preparing Economic Analyses* provides detailed guidance on specifying the baseline for economic analyses. According to those guidelines, the baseline must be appropriate for the question or policy option addressed, identify a particular point in time from which point forward the effects of the policy or action are to be assessed, and define assumptions about underlying conditions or factors that are unknown or uncertain but will affect the conclusions of the assessment (e.g., number of alternative fishing opportunities available). Baseline specification might also include determining what alternative assumptions might be examined as part of a sensitivity analysis. Because the baseline must be consistent with any other analyses conducted as part of the overall study, the parameters for defining a baseline for the ecological assessment and economic benefit analysis may be determined during the planning stage for the overall study. However, issues relating to baseline specification specifically related to assessing changes to ecological resources may arise and must be resolved when developing the analysis plan.

Defining the scenarios to examine includes defining the action or change to be evaluated, the area(s) expected to be affected, the time period over which effects will be evaluated, and identifying any additional factors or actions (e.g., other regulations) that might affect the outcome and determining how they will be accounted for in the ecological assessment and economic benefit analysis. Many of these questions will have been addressed during the planning stage. However, it may be necessary to reexamine the decisions made during the planning stage in light of the joint conceptual model.

2.6.2 Measuring and Modeling Linkages

The analysis plans put in writing the assessment design, the analyses that will be conducted, data needs, measures, models to be applied, and statistical techniques to use. Both the ecological and economic analysis plans should specify what will be measured and how changes in endpoints will be expressed. In developing an analysis plan for the economic benefit assessment, economists must determine how they will represent the ecological changes in the economic analysis. Most economic valuation approaches will require some measure of the ecological change associated with the economic endpoint assessed (e.g., an assessment of the value of improved swimming opportunities requires a measure of the change in water quality, an assessment of improved wildlife viewing opportunities requires information on the estimated change in the wildlife population). During the analysis design phase, economists must determine if the information required by the economic analysis is specified in the ecological analysis plan. If not, the ecologists and economists must confer until agreement is reached on the how ecological changes will be characterized at the end of the ecological risk/benefit assessment. At this point, an initial screening-level assessment might be planned to (a) assess the likelihood of certain linkages between ecological and economic endpoints and (b) determine the sensitivity of those relationships. If any of the results are unexpected, the endpoints for assessment might be reprioritized.

To the extent that the conceptual model identified feedback loops between the economic and ecological endpoints, the analysis plans must specify how those interactions will be modeled. Such interactive models will require more extensive coordination and cooperation between the ecologists and economists than models without feedback loops. For example, in assessing the economic benefits associated with an increase in fish populations resulting from improved water quality, economists may want to account for market adjustments in response to the increased supply of fish, namely lower commercial prices and increased consumption. The long-term net effect of these reactions may be a slightly smaller increase in fish populations than estimated without accounting for this market adjustment. The change in the estimated increase in the fish population may have an impact on other ecological resources, such as piscivorus birds, that rely on the fish population as their food source. Thus, by accounting for the economic market response associated with the commercial fishery, the ecological assessment may provide a better estimate of the benefits to the bird populations (which also may be associated with a separate economic benefit endpoint - bird watching).

Until recently, most economic valuation models focused on only a single change, ignoring interactions inherent to the natural system and failing to account for interactions between multiple economic and ecological endpoints simultaneously. However, with advances in ecological and economic models there will be greater opportunities for using models that capture the interactions and feedback loops both within the ecological system and between ecological and economic endpoints (e.g., incorporate role of human action in ecological model and reflect effect of economic changes on human actions).

In the coordination between the ecological and economic analysis plans, there are two areas that require substantial attention: the spatial and temporal scales of the assessments. These must be matched, and issues associated with each are described in the next two subsections.

2.6.3 Matching Spatial Scales

The benefits of many ecological processes and services "play out" at a much larger spatial scale than specific projects under consideration (Limburg, 1999). The proposed actions and alternatives often can be delineated geographically and often are limited to small portions of watersheds or landscapes. Ecosystems, on the other hand, can be difficult to delineate geographically, and the spatial scale of the change in ecological benefits often is much larger than the spatial scale of an implemented management practice. Also, because the benefits play out at a larger spatial scale, those services are impacted by other projects, land uses, etc., in ways that will affect the outcome. All ecosystems lose, gain, or exchange some types of materials and energy with neighboring ecosystems through one or more processes.

The spatial area over which ecological benefits might occur is generally larger or different from the spatial area over which the proposed action/alternatives can be delineated because of the cascading nature of the ecological effects. One way to overcome this analytic difficulty and to ensure a good match between elements of the analyses spatially is to view the natural systems as being organized in hierarchies (O'Neill *et al.,* 1986). In this view, the coarser-scale entities (i.e., aggregates of finer-scale entities) can be separated into manageable sets of relatively homogeneous subgroups (Costanza *et al.,* 1995; Vatn *et al.,* 1999). Watersheds or catchments often are the largest unit of analysis for many assessments, with finer scale units being comprised of areas with similar slope, soils, vegetation, and microclimate. Once units of analysis have been defined in a series of hierarchies, the next analytic task is to maintain the necessary finer-scale variations as one moves from the finer- to the more coarse-scale entities/aggregations (Vatn *et al.,* 1999). For a discussion of conducting analyses using hierarchies based on different spatial scales, see Costanza *et al.* (1995), Vatn *et al.* (1999).

An advantage of developing the conceptual model that describes both the direct and indirect ecological effects of an action is that each node in the model (i.e., the endpoint described in a box with arrows entering and leaving the box) is likely to be associated with a change in the spatial scale of ecological effects. Thus, the ecologists can examine each node and estimate whether the spatial scale of effects at that node is likely to be larger than the spatial scale for analysis of the preceding node. To increase the utility of the conceptual model, it is helpful to provide a description of the geographic scale and location associated with each node in the model.

Defining the spatial area of consideration for the economic analysis is an important step that can have significant impacts on the conclusions of the analysis. The spatial area of consideration defined by the ecological risk assessment serves as the starting point for defining the spatial limits of the economic analysis. Because the economic analysis focuses on human uses associated with ecological resources and humans are more mobile than plant and animals, the economic analysis might consider a broader spatial area than that defined by the ecological assessment. The economic analysis must define an appropriate spatial area from which humans may still make use of or otherwise benefit from the services provided by the ecological resource. If the economic analysis can monetize benefits for only a portion of the entire area affected, the economic assessment's qualitative discussion should address the entire area affected and recognize that the monetary benefits estimate represent only a portion of the benefits expected over the entire area.

2.6.4 Matching Temporal Scales

As indicated above for the spatial scale, the benefits of many ecological processes and services "play out" over a longer time period than a specific project under consideration (Limburg, 1999). The activities associated with a management practice might require only weeks or months to implement, or might occur at specified points of time each year. The ecosystem responses can require years to decades to develop and often are reflected in changes throughout the entire annual seasonal cycle. For the purposes of assessing the economic value of the ecological changes, effects may need to be assessed over shorter time intervals. A thorough benefits assessment needs to consider the role of lagged or future effects and determine how best to account for these types of effects. This may include a better characterization of the stream of benefits based on scientific information on changes in environmental conditions over time. It also may include determining an appropriate discounting scheme for comparing future effects against current effects (see Chapter 7 for further discussion on discounting).

Again, it can be helpful to establish a hierarchy of time steps for the analysis and compartmentalize the analysis accordingly (Vatn *et al.,*1999; Costanza *et al.,* 1995).

2.6.5 Data Limitations and Uncertainty

Evaluation of uncertainty should be a theme throughout the analysis phase that follows problem formulation. The analysis plans should specify how uncertainty will be addressed. Several sources of uncertainty need to be considered. These include human error, natural variability in parameters, data gaps, uncertainty about a parameter's true value, uncertainty introduced by models that attempt to predict real-world processes, and other sources. An important distinction to maintain throughout the analysis is the difference between natural variability, which can be quantified using various statistics, and uncertainties due to lack of information.

The economic benefits analysis should recognize the uncertainties in the ecological assessment process as well as the uncertainties inherent in economic analysis. The level of uncertainty in the ecological assessment process as well as the economic valuation process is often substantial.

As discussed in detail in EPA's (2000) *Guidelines for Preparing Economic Analyses*, the issue is not to avoid uncertainty but to recognize and account for uncertainty and provide information that is useful to decision makers. As noted in the *Guidelines*, to adequately address uncertainty, the analysis should: use the expected or most plausible outcomes; discuss all key assumptions, biases and omissions; include sensitivity analyses for key assumptions; and justify the inputs and assumptions used based on the results of the sensitivity analyses. (See Chapter 7 for further discussion on accounting for variability and uncertainty.)

2.7 CONDUCTING THE ASSESSMENTS

Once the analysis plans are complete, the actual analyses can begin. In general, the ecological assessment must be conducted first to provide the inputs on predicted changes in ecological endpoints for the economic assessment. The ecological exposure and response assessments are conducted by the ecological risk assessment team independent of the economists. In other words, if problem formulation and planning for the analyses are carefully conducted, there should be little need for communication between the economists and the ecologists during the ecological analyses. Often, however, unexpected data gaps or unexpected interim modeling results might require discussions between the ecologists and economists to resolve such issues.

2.7.1 Ecological Risk/Benefit Assessment

During the analysis phase, the ecological risk/benefit assessment team collects the data specified in the ecological analysis plan. The team then conducts both an exposure assessment and an ecological response assessment. The exposure assessment evaluates the potential sources of stress or change, their distribution in the environment, and their overlap with ecological receptors. The response analysis attempts to quantify exposure-response relationships and the relationship between measures of response and the ecological assessment endpoints. For economic assessments of ecological benefits, it is insufficient to identify thresholds for effects, as sometimes is done in ecological risk assessments to identify environmental levels of concern or cleanup goals for contaminated waste sites. The ecological risk assessment conducted for an economic benefits assessment must estimate the type and magnitude of ecological changes to allow the economists to predict the economic benefits (i.e., positive economic changes) from an environmental decision.

Because the ecological endpoints can include both direct and indirect effects, the exposure and response analyses generally will include two types of assessments. The first is an analysis of the relationship between the magnitude and extent of the initial action/stressor and the magnitude and extent of direct ecological effects of that stressor. The second is an analysis of the relationship between the magnitude and extent of changes in those initial ecological endpoints to the magnitude and extent of responses in the endpoints down the cascade of effects depicted in the conceptual model. The analysis phase concludes with descriptions of the findings from the exposure and response assessments. Additional discussion of this phase of the ecological risk assessment is provided in Chapter 4.

The final phase of an ecological risk assessment is risk characterization (see Exhibit 2). This phase integrates the findings from the exposure and response assessments to characterize the predicted changes in ecological endpoints. For ecological risk assessments for contaminated sites, risk characterization often consists of a simple question. Does the exposure level exceed a threshold for effects? For ecological risk assessments conducted to support economic benefit analyses, the emphasis of risk characterization needs to be predicting the magnitude and extent (both spatial and temporal) of changes in the ecological assessment endpoints.

Ecological risk assessment conducted to support economic benefit assessments also differ from ecological risk assessments for contaminated sites in how uncertainty in the assessment is handled. For risk assessments for contaminated sites, data gaps are generally addressed using conservative assumptions. Moreover, exposure assessments generally focus on possible high-end exposures (e.g., upper 90^{th} percentile). That is because the risk management goal at hazardous waste sites often is to be reasonably sure that a site is "clean" (i.e., unlikely to cause adverse effects to the assessment endpoints) after site remediation is complete. In other words, the assessment is designed to ensure a "reasonable margin of safety". For an economic benefit analysis, on the other hand, uncertainty might be handled in other ways.

For economic benefit assessments that address large areas (e.g., national, regional, or state assessments), best estimates, instead of high-end estimates, often are the most useful for characterizing ecological risks. Estimated mean values for the change in an ecological endpoint, with some type of confidence interval on those estimates, provides the economists with numbers that can be added or multiplied in the economic assessment without compounding conservative biases. Lower and upper percentile estimates of the degree of change in an ecological endpoint also are useful to the economists. Plausible worst or best case scenarios are generally only useful as bounding exercises for the assessment.

2.7.2 Qualitative and Quantitative Economic Assessment

Following the effect-by-effect approach to benefit analysis discussed above, economists proceed with the qualitative, quantitative, or monetized assessment of changes for each endpoint. Economists begin by gathering economic data and developing their models, as called for by their analysis plan, in anticipation of the final input from the ecological risk assessment. At this point, economists also collect any additional information and data required for their qualitative and quantitative assessments. Once the monetized benefits associated with the various economic endpoints are estimated, the dollar values are summed together. It is important, however, that economists emphasize that the monetized benefits estimate reflects only a portion of the total economic benefits expected to accrue from the action. This statement must be supported by strong qualitative and quantitative assessments of other benefits not captured by the monetized assessment.

If models or value estimates from other studies are used in the assessment, the analysis must describe the source and provide some assessment of the confidence associated with the source.

For example, if multiple high-quality studies have produced a similar value estimate, the economists can have more confidence in using this value estimate in their benefit calculation (U.S. EPA, 2000a).

Chapter 6 discusses in detail the various economic techniques for estimating the economic value of changes to different types of economic benefit endpoints. Economists will likely apply more than one valuation technique in estimating the total benefits. Care must be taken in designing and implementing the economic analysis to avoid double counting of benefits, particularly when applying more than one method to estimate the value of changes to related benefit endpoints. Additionally, the economic analysis should recognize any negative consequences of the action or policy under study that may offset some of the beneficial improvements.

2.8 CHARACTERIZING AND PRESENTING RESULTS

For many assessments, ecologists and economists will present their results separately, usually with the results of the ecological assessment first. Because characterization of ecological risks/benefits provides the input to the economic analysis, it is important that the presentation of ecological changes address several factors: the types of ecological changes expected, the magnitudes of those changes spatially, temporally, and per unit area (i.e., severity), and the certainty associated with those estimates. The presentation of the ecological changes should present the conceptual model and assessment endpoints, review and summarize major areas of uncertainty and potential bias, discuss the degree of scientific consensus in key areas of uncertainty, identify major data gaps, describe any assumptions used to bridge information gaps (U.S. EPA, 1998), and indicate how the uncertainty in the results might be magnified through the cascade of ecological effects considered.

In addition to the formal report of the findings for the ecological risk assessment, collaboration between the ecological risk assessment and economics teams might help to explain the relationships between the ecological endpoints addressed by the ecological assessment and the economic endpoints identified in the economic benefit analysis.

The results from the economic benefit analysis will present the prioritized list of economic effects, discuss the criteria used to select the economic endpoints examined in detail by the benefit analysis, and discuss how the economic value of the effects was assessed. The monetary benefits estimated for some of the changes will be accompanied by the qualitative and quantitative assessment of other benefits that were not monetized. If possible, the qualitative assessment should discuss the potential magnitude of the economic benefits for any priority endpoints that are not accounted for by the quantitative and monetized assessment. The final report should also discuss to some degree the other effects identified that were deemed less important to the economic analysis. Finally, the results of the economic analysis must disclose any source of error in the analysis and the potential impact of such error on the results. The presentation of results should identify any possibility of double-counting of benefits, any limitations of the analysis, and any potential imprecision and uncertainty associated with the benefit estimates. In discussing the potential impact of any source of imprecision or uncertainty, economists should discuss whether the analysis is likely to over- or under-estimate the economic value of benefits.

The results presented also may include information and details that are needed for other analyses. For example, an equity analysis may require information on the geographic distribution of effects, the distribution of ecological effects and economic benefits across different ethnic or economic classes of the human population, or the distribution of ecological effects and economic benefits over time.

2.9 CONCLUDING REMARKS

Economists and ecologists have different views and perspectives that are important to recognize. Closer coordination can be encouraged by understanding how the disciplines differ and acknowledging these differences. In conclusion, this section identifies some areas in which economists and scientists may find they have different approaches or interpretations.

- ***Perspective.*** Economists approach the identification and valuation of changes to ecological resources differently than ecologists and other scientists. For example, human activities and welfare are the focus of economists while ecologists are concerned with complete ecological systems and the interactions between ecological components, which may or may not include effects on humans.

- ***Terminology.*** Each discipline has its own terminology, including different units of measure. Even common words such as "value," "benefit," and "function" have different meanings across disciplines. To improve interdisciplinary coordination, care needs to be taken to define and use terms consistently.

- ***Scale.*** Part of interdisciplinary coordination is understanding how a change will be measured. This requires that ecologists and economists agree on the units of measurement and discuss the spatial and temporal boundaries of the analysis.

- ***Focus***. The ranking of endpoints will likely differ between the ecological and economic assessment. Additionally, the approach for assessing changes may differ (e.g., economists may measure only the direct change, without addressing system or feedback effects). Such differences are partially a consequence of the training associated with each discipline but also reflect important differences in the characteristics of the systems studied by the respective disciplines.

- ***Metrics.*** Economists focus on the effect of changes to human welfare and typically want to standardize effects or welfare changes into dollars to compare effects that may be dissimilar. Other metrics may be appropriate for a qualitative and quantitative description of ecological and economic effects.

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3.0 IMPORTANT PRINCIPLES OF ECOLOGY AND ECOLOGICAL ASSESSMENT

This chapter defines some basic terms and concepts used by ecologists and explains how these concepts can be applied in conducting an ecological assessment. First, this chapter defines an ecosystem and levels of ecological organization and examples of endpoints at each level (Section 3.1). Next, this chapter describes interactions that occur within ecosystems, including the concepts of "food chain," "food web," and "energy flow," competition, predation, and symbiosis (Section 3.2). The chapter then examines what attributes of ecosystems and ecosystem entities are of value both to society and to sustaining ecosystems themselves (3.3). This chapter concludes by describing different standard approaches to ecological assessments and why EPA's ecological risk assessment process is used in the proposed framework for the economic assessment of ecological benefits (Section 3.4).

3.1 DEFINING ECOSYSTEM AND OTHER LEVELS OF ECOLOGICAL ORGANIZATION

3.1.1 Definitions

According to the Institute of Ecosystem Studies "Ecology is the scientific study of the processes influencing the distribution and abundance of organisms, the interactions among organisms, and the interactions between organisms and the transformation and flux of energy and matter." An *ecosystem* can be defined in various ways, but one definition that is particularly useful is "a spatially explicit unit of the Earth that includes all of the organisms, along with all components of the abiotic environment within its boundaries" (Likens, 1992). The concept of an ecosystem can be applied at any scale ranging, for example, from a small pond to an entire mountain range. Because ecology is concerned not only with organisms but with energy flows and material cycles on land, in water, and in air, ecology is often defined as the "study of the structure and function of nature."

Ecosystem Concepts

Ecosystems refer to a system formed by the interaction of a group of organisms and their environment. An ecosystem may be a pond or the entire globe. It can be natural or artificial. All ecosystems are composed of components, structure, and processes (functions). Components are the plants, animals, soil, air, and water. Structure refers to spatial and temporal distribution of those components. Processes are the flow of energy and the cycling of materials and nutrients through space and time.

Ecosystems occur in geographic arrangements. Smaller ecosystems exist within larger ones. The scale selected and the boundaries used to define an ecosystem depend on the problem or question to be addressed.

Source: Appendix A in U.S. Department of the Interior. 1994. *Ecosystem Management in the National Park Service:*

3.1.2 Levels of Biological Organization

There are five levels of biological organization that are conventionally recognized and potentially useful in ecological risk/benefit assessments:

- Individual,
- Population,
- Community,
- Ecosystem, and
- Landscape.

Ecological assessments do not address the sub-organismal levels of organization (organ systems and cells), nor do they generally address the larger scales of organization of biomes and the biosphere. The levels of organization listed above are not defined by the environment. Rather they are defined by scientists to facilitate our understanding of relationships within and among ecological systems. Thus, these levels can be described as criteria for observation and analysis (Allen and Hoekstra, 1992).

A species is a group of individuals that are able to successfully interbreed. In a species, slight biological variations, both genetic and apparent, will exist among individuals. A population is a group of organisms of the same species that live in the same place, and have the potential to reproduce with one another during their lifetimes. A community is an organized assemblage or association of species in a prescribed area or a specific habitat. An ecosystem, defined above and described in more detail below, can be viewed as a biotic (i.e., living) community functioning within its abiotic (i.e., nonliving) environment. A landscape comprises a group of spatially contiguous ecosystems and is usually defined in geographic terms, such as a watershed.

Ecosystems are often defined in terms of their structural and functional components. Structural components are physical elements present in the environment. Examples include soil, nutrients, water, and biological entities such as plants, animals, and microorganisms. Functional components are processes or interactions that support the structural components, such as nutrient cycling and energy flow. It is the pathways of energy flow and cycles of matter that help to determine ecosystem boundaries for our purposes of observation.

3.1.3 Interactions Within Ecosystems

In an ecosystem, the biological community and the abiotic elements of the environment (e.g., water, soil) are bound together by action and reaction, defined by the reciprocal effects of the physical environment on an organism and an organism on the physical environment. Temperature, moisture, light and other kinds of radiation, texture and chemical composition of soil or water, the presence or absence of gases and chemicals, gravity, pressure, and sound can all have profound effects on organisms. Examples of interactions between an organism and its physical surroundings would be rising river levels forcing muskrats to abandon burrows and move to higher ground or the use of sunlight by plants as an energy source. Organisms themselves can also affect the physical environment through their activities, thereby indirectly affecting other organisms. Examples include beavers damming streams, which changes the aquatic community, and earthworms burrowing through and aerating soil, which improves plant growth.

In an ecosystem, interactions also occur among individuals within a population and between individuals of different species. For example, the social behavior exhibited by different members of a wolf pack is an example of interactions occurring between individuals within a population. Predator-prey interactions between wolves and mice are interactions that occur between members of different species.

To understand the influence of the various types of interactions described above on ecosystem structure and function, it is important to view more than one level of biological organization. Individual-level effects, such as mortality and reduced reproductive success, can have population-level effects, such as decreasing or increasing population size and density. On the other hand, population-level processes can compensate for individual-level effects. For example, in some types of species, increased adult mortality might be compensated for by increased survivorship of younger animals to maturity.

Population Growth. Interactions among individuals within a population and harvesting of energy and materials from food allow animals to reproduce and increase in local abundance. Population growth rate is a function of birth rates, death rates, time to maturity, and reproductive success. Population growth rate also depends on the immigration of individuals from other geographic areas into the population and the rate at which individuals emigrate in search of better habitat. Reproduction by some species that have just recently invaded a new geographic area (e.g., introduced exotic or invasive species) is not hampered by predation or competition for resources. Such populations increase in abundance without check (i.e., exhibit *density-independent* growth) for some period of time. During that period, the species "intrinsic rate of a natural increase" (i.e., the maximal rate at which offspring can be produced) governs the population growth rate. Species that mature quickly and produce large numbers of offspring that can survive under favorable conditions have a high intrinsic rate of natural increase. Initially, when population density is low, population growth might follow an exponentially increasing function. Eventually, however, resources available for growth and reproduction will become limiting, and birth and death rates will depend on population density. This situation is called *density dependent* population regulation. Such populations are characterized by relatively stable population densities (i.e., an equilibrium situation). When population growth rates are density dependent, an increase in the rate of loss of juveniles (e.g., harvesting eggs) or adults (e.g., hunting) is compensated for (within limits) by increased survival and/or reproductive rates by the remaining population. This is the basic principle underlying management of fisheries and game populations.

Life History Strategies. Interactions between a species and its environment over time result in the evolution of traits in the species that are adapted to that environment. Some species that have evolved in an environment that is densely populated tend to have few offspring in which they invest heavily to improve the offspring's ability to compete with other individuals. Species with this type of life history have been called *K-selected* (Wilson and Bossert, 1971). Facing resource limitations, these species have evolved to mature relatively late and attempt to reproduce repeatedly (e.g., annual) over their lifespan. Moreover, at each reproductive effort, they produce only a few offspring that exhibit a high survivorship. These species also tend to be characterized by longer life spans and larger body size. Whales, elephants, and seabirds are good examples of K-selected species. At the other end of the spectrum of life history strategies are the *r-selected* species (Wilson and Bossert, 1971). Such species evolve where resources for reproduction and growth are not limiting but might exhibit a patchy distribution in the environment. Under these circumstances, the best strategy is simply to maximize the number of offspring produced, investing little in each of the individual offspring. These species also tend to reproduce one time during their life, producing large numbers of offspring that exhibit poor survivorship except where they encounter an "empty" patch of habitat. Weeds are classic examples of r-selected species.

Competition. Competitive interactions are those in which two or more species tend to depress each other's population growth rates and abundance (Gotelli, 1998). There are several different types of interactions between species that fit this general definition. *Exploitation* competition occurs when species compete with each other for the same resource (e.g., food). For example, domestic cattle and bison compete for food (grasses) on open range lands in several areas of the mid-western United States. *Interference* competition occurs when one species interferes with the ability of another species to exploit a resource (Gotelli, 1998). An example would be a plant species that releases toxic chemicals into the soil, thereby preventing other plants from germinating and growing in that soil. Another example of interference competition is introduced (i.e., non-native) species of vines that cover other plants, thereby reducing the solar energy available to the covered plants, stunting their growth and eventually killing them. *Pre-emptive* competition occurs where species compete with each other over space (Gotelli, 1998). Examples include competition for anchorage in the rocky intertidal zone of New England coastal areas by barnacles and mussels.

If environmental conditions were constant over space and time, the population density of those species best suited for those conditions would increase at the expense of other species. However, environmental conditions vary substantially over time and with geography (e.g., altitude, exposure to the sun, soil conditions, rainfall patterns). This environmental variation helps to maintain a number of species in competition with each other because the competitive edge among species changes as the environmental conditions change. One species might have a competitive advantage under some conditions, but be at a competitive disadvantage under other conditions.

Predation. Predation is a direct interaction between two species in which an individual of one species consumes an individual of the other species (e.g., as when a hawk captures and consumes a rabbit). This interaction results in removal of individuals and biomass from the prey species' population. The relationship between predator and prey often results in oscillations in their relative abundances. For example, when a particular prey species prospers under favorable environmental conditions, it tends to increase in abundance. The population of predators can respond in two ways. In the short-term, the predator can change its behavior (i.e., a functional response) and include more of that prey species in its diet. Over the longer-term, the predator population abundance can increase as its death rate from starvation decreases and reproductive success increases. This increase in predator abundance is self-limiting, however. Increasing predation pressure tends to reduce the prey populations, resulting in starvation, reduced reproductive success, and a reduction in the abundance of the predator species. This feedback system usually results in a reasonably stable equilibrium ratio of predator to prey, although, minor to moderate oscillations in that ratio can occur in response to changing environmental conditions.

Some species of animals (e.g., mice, deer) are typically held at densities lower than their food resource base could support because of predation. Removal of the predators in such systems generally results in increases in the prey population until some other factor becomes limiting (e.g., food resources). Where prey are considered pests (e.g., many insect species), removal or loss of the predators for any reason can result in an outbreak of the pest species. Such outbreaks often cause serious economic losses ingriculture and silviculture.

Herbivory. Herbivores consume plant materials, usually in a way that does not kill the individual plant. Animals that feed on grasses (e.g., bison, cattle) tend to have digestive systems adapted for extracting nutrients from very fibrous plant materials and must consume large quantities of food to satisfy their metabolic (i.e., caloric) needs. Species that feed on new growth (e.g., rabbits), with high protein and low fiber content, can survive on lower total quantities of plant material. A different type of herbivory is the consumption of plant seeds, which are rich in both protein and lipids. Many species of birds and small rodents specialize in harvesting seeds. In general, however, seed eaters (i.e., gramnivores) generally have little impact on the abundance of many plant species because of the prolific production of seeds by those plant species.

Herbivores generally can detoxify a wider array of chemicals than can carnivores (i.e., predators). This ability results from an evolutionary arms race between plants and herbivores. The plants cannot "escape" from an herbivore; instead, they must develop defenses that work *in situ*. Over evolutionary time, plants have developed a substantial array of defenses, including toxic substances stored in their tissues, to discourage herbivores from eating them. The toxic substances often cause an herbivore to be acutely ill. Animals that recover from that experience will avoid eating that species of plant in the future. In response, however, over evolutionary time, herbivores have been developing metabolic pathways to detoxify the toxic chemicals found in plants. Carnivores (at least north temperate species), on the other hand, have not needed to develop elaborate detoxification pathways because their prey usually cannot afford to use chemicals in their body tissues as a defense.

Pollination. Pollination is a mutually beneficial interaction between a flowering plant and its pollinator (e.g., bee, butterfly). The flowering plant offers rewards (e.g., nectar with a high sugar content) to attract the pollinators. The flower shape of the plant has evolved so that as the pollinator feeds on the reward, it becomes covered in the pollen of that plant. When the pollinator moves on to the next individual of that plant species, the pollen can be transferred and fertilize the eggs of that individual. Thus, pollinators help plants reproduce sexually, maintaining genetic variability among individuals. Many species of plants cannot self-fertilize, and thus require their pollinators in order to reproduce at all.

Symbiosis. Another beneficial interaction occurs in symbiotic relationships, where two organisms in close association with each other benefit from the association. Examples include the relationship between corals and the algae that grow in their tissues, the relationship between an alga and its host fungus to form a lichen (one of the few organisms that can live on bare rocks and begin the process of soil formation), between "cleaner" shrimp and the fish that they clean, and between acacia ants and the acacia trees they protect in return for nectar and shelter. In many symbiotic relationships, death of one member of the pair sooner or later results in illness or death of the other. For example, death of the algae living in the tissues of corals, which can be recognized by the "bleached" appearance of the coral, often is followed by death of the coral organisms themselves.

In summary, there are many types of interactions within and among species that occur in all ecosystems. For this reason, direct impacts (or benefits) of an activity on one species tend to produce a cascade of effects through an ecosystem because of the interactions among species in the ecosystem.

3.2 UNDERSTANDING ECOSYSTEM STRUCTURE AND FUNCTION

Ecosystems may be as large as unbroken tracts of forest and grassland or smaller than a pond. The ecosystem is an energy-and-material-processing system, receiving abiotic and biotic inputs. The driving force is the energy of the sun. Abiotic inputs include oxygen, carbon dioxide, and nutrients. Nutrients become available via weathering of the Earth's crust and precipitation. Biotic inputs include organic materials, such as living organisms and detritus matter (i.e., dead and/or decaying organisms).

The ecosystem itself consists of three components:

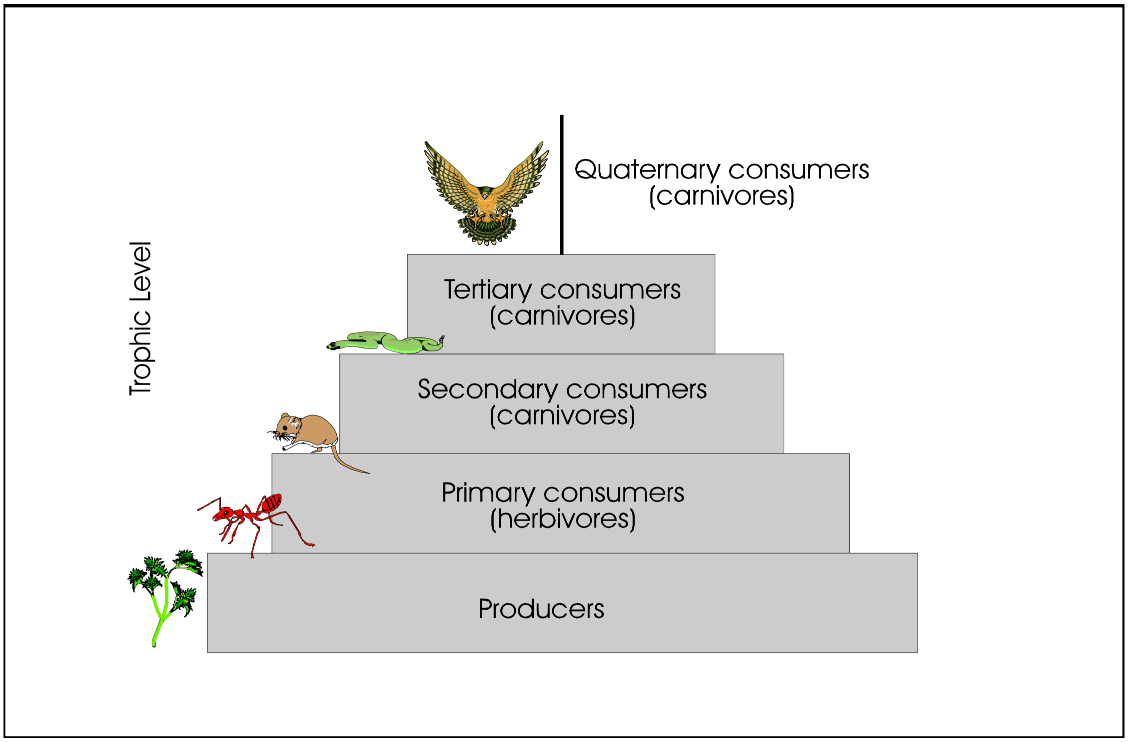
- Producers that derive their energy from the sun (i.e., photosynthetic plants);

- Consumers and decomposers that use the energy fixed by the producers and eventually return nutrients to the ecosystem; and

- Dead organic material and inorganic substrates that act as short-term nutrient pools and support the cycling of nutrients within the ecosystem.

The most basic functions of the ecosystem are photosynthesis and decomposition. Photosynthesis is the process by which green plants utilize the energy of the sun to convert carbon dioxide and water into carbohydrates. Through photosynthesis, plants are able to capture the sun's energy and drive the majority of metabolic activities in the living world. Decomposers are responsible for the return of nutrients to the ecosystem and a final dissipation of energy to the environment.

Exhibit 8
Trophic Level Organization



The "food chain" is a concept that describes the movement of energy and nutrients from one feeding group of organisms to another in a series that begins with producers and ends with consumers. The food chain concept specifies a sequence of organisms, each of which feeds on the preceding group. The trophic structure ("trophic" means "feeding") of a community is based on the food chains in the community (see Exhibit 8). A simple food chain might be: oak leaf ⇨ caterpillar ⇨ small bird ⇨ hawk. One useful approach in defining relationships among organisms is to group organisms based on their trophic levels (i.e., their position in the food chain).

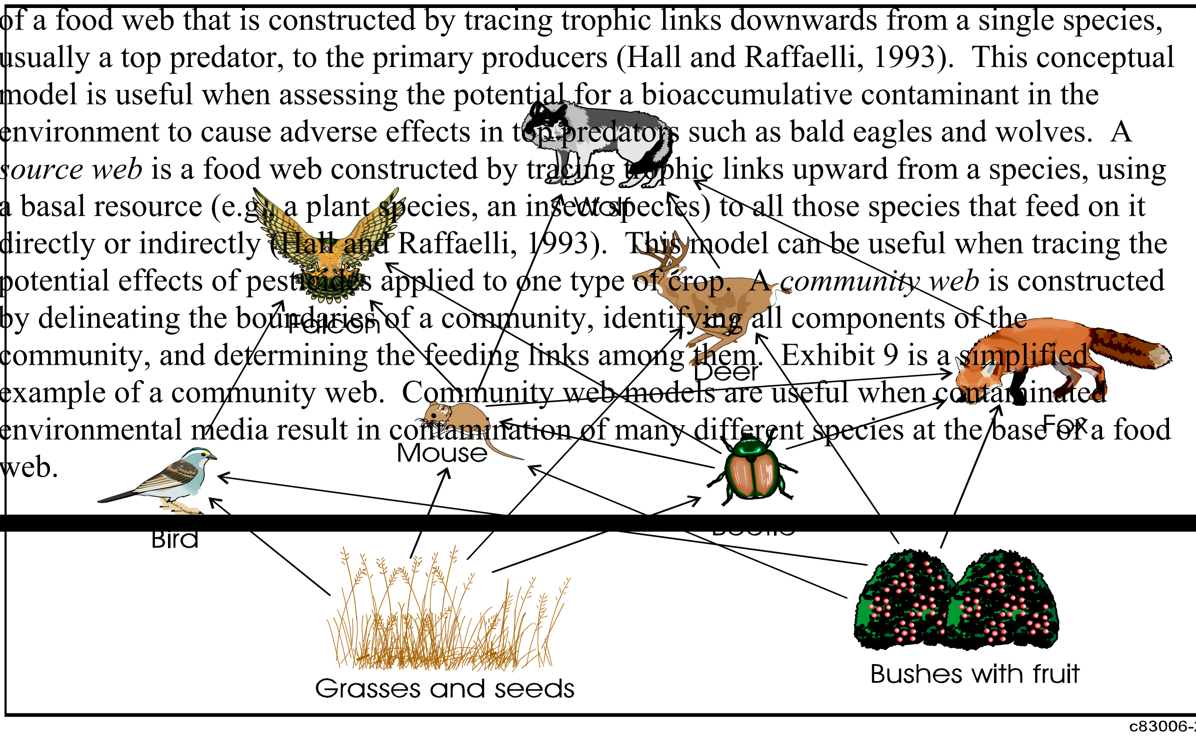
The major categories for trophic organization are producers, primary consumers, and secondary consumers. However, ecosystems are too complex to be characterized by a single, unbranched food chain. Instead, the transfer of materials and energy from one type of organism to another is better described as a food web (see Exhibit 9).

The food web for most communities is very complex, including many species and trophic groups. Several primary consumers may feed on the same plant species. For example, several insect species might feed on one tree. On the other hand, one species may feed on several different plants. Also, some species may feed at more than one trophic level. For instance, owls may eat primary consumers, such as field mice, and also prey on higher level organisms like snakes. It is more correct, then, to draw relationships between these trophic levels, not as a simple chain, but as a more elaborate interwoven food web. Complexity of the food creates

Food Web Models

Food webs can be described from various points of view. A *sink web* is a conceptual model of a food web that is constructed by tracing trophic links downwards from a single species, usually a top predator, to the primary producers (Hall and Raffaelli, 1993). This conceptual model is useful when assessing the potential for a bioaccumulative contaminant in the environment to cause adverse effects in top predators such as bald eagles and wolves. A *source web* is a food web constructed by tracing trophic links upward from a species, using a basal resource (e.g., a plant species, an insect species) to all those species that feed on it directly or indirectly (Hall and Raffaelli, 1993). This model can be useful when tracing the potential effects of pesticides applied to one type of crop. A *community web* is constructed by delineating the boundaries of a community, identifying all components of the community, and determining the feeding links among them. Exhibit 9 is a simplified example of a community web. Community web models are useful when contaminated environmental media result in contamination of many different species at the base of a food web.

Exhibit 9
Terrestrial Food Web



opportunities for amelioration of impacts on a particular food chain, but also makes it possible for there to be indirect effects that are larger than the direct effects.

Two processes occur in an ecosystem through the food web: energy transfer and nutrient cycling. Both energy and nutrients are transferred from plants (producers) to herbivores to carnivores (primary and secondary consumers) and from all preceding levels to the decomposers through the food web. By tracing the energy transfers and nutrient cycles, the ecologist is able to analyze the changes in an entire ecosystem.

Terrestrial and aquatic ecosystems tend to have somewhat different patterns of energy flow and nutrient cycling. In terrestrial systems, the major energy input is from the sun, while in some aquatic systems, for example small streams within forests, additional energy inputs come from the terrestrial environment from plants, insects, and other animals. In forest ecosystems, a substantial proportion of the organic matter in the system can be in the form of dead and decomposing organic matter, the leaf litter. Hairston and Hairston (1993) estimate that 95 percent of the net primary production (NPP) by plants in temperate forests reaches the forest floor, while only about 15 percent of the NPP in a lake reaches the lake bottom. Once on the bottom, only a fraction of the detritus in a lake is consumed; the rest accumulates annually in the sediments where it might be permanently “lost.” In terrestrial ecosystems, detritivore-based food webs tend to be at least as important as herbivore-based food webs, whereas herbivore-based food webs tend to predominate in aquatic ecosystems (Hairston and Hairston, 1993).

The length of food chains in terrestrial ecosystems tends to be shorter than the length of food chains in aquatic ecosystems (Hairston and Hairston, 1993; Oksanen, 1991). Some ecologists have even proposed that the length of food chains in terrestrial ecosystems typically is two steps, while in large open water systems (e.g., large lakes), the food chain length typically is three steps (Briand and Cohen, 1987; Pimm, 1980, 1982). As the difference in food chain length would suggest, bioaccumulation of contaminants in terrestrial ecosystems is less prevalent than bioaccumulation of contaminants in aquatic ecosystems.

Example of Translation and Magnification of the Effects of Pollutant Discharges Through the Food Web

Pollutant discharges can affect not only the health, behavior, and survival of individual organisms, but they can also adversely influence the vital interactions and energy flow of the food web. This could lead to an adverse change in the structure or function of a population or community. When energy and materials flow from one trophic level to the next, contaminants in plant or animal tissues consumed are also transferred to the next trophic level. If a contaminant is retained in the consumer's body tissues, its concentration will be higher in each succeeding trophic level, because an organism eats many times its own body weight during its lifetime. In this way certain pollutants can bioaccumulate as they travel up the food web, reaching toxic concentrations in the upper trophic levels of a food web even though concentrations in environmental media (e.g., soils, surface water) are relatively low. Methylmercury is an example of a bioaccumulative compound. Methylmercury is generally present in small amounts in surface waters. It is absorbed directly from the water by aquatic organisms, but, more importantly, it bioaccumulates as it is passed up the food web, beginning with algae and ultimately passing to fish-eating animals such as gamefish and certain mammals, including humans. Mercury concentrations in consumers near the top of the food web can reach toxic levels, thousands of times greater than that of the ambient

Given the discussion above of interactions within ecosystems, it should be clear that defining the boundaries of an ecosystem for purposes of assessment can be difficult. Margalef (1968) suggested that an investigator might operationally define ecosystem boundaries as locations where energy flows are near zero or negligible in comparison with energy flows within the system (Suter, 1993). Examples of ecosystem boundaries by this definition would include coastlines, forest edges, or watersheds. This definition is consistent with hierarchical descriptions of ecosystems, where components within the system are strongly coupled with each other and form systems or subsystems that are weakly coupled to other subsystems (Suter, 1993; Hoekstra, 1992). A similar definition of ecosystem boundaries might be constructed using nutrient cycles instead.

3.3 VALUED ECOLOGICAL ENTITIES

Over the course of many years and activities, EPA has examined the issue of what environmental entities should be considered priorities for protection. EPA's (1997) *Priorities for Ecological Protection: An Initial List and Discussion Document for EPA* states:

Environmental legislation has a long history of protecting certain groups of animals such as fish, shellfish, migratory songbirds and waterfowl, and large mammalian game species. More recently, legislation has sought to protect entire ecosystems and to ensure their "integrity" for the foreseeable future.

The following paragraphs describe what is meant by ecological integrity and what characteristics of ecosystems are diagnostic of ecological integrity.

3.3.1 Definitions

Ecological integrity has been defined by EPA (1994a) as "the interaction of the physical, chemical, and biological elements of an ecosystem in a manner that ensures the long-term health and sustainability of the ecosystems." This definition encompasses the concepts of *sustainability, resiliency*, and *biodiversity* (U.S. EPA, 1997).

Sustainability is the ability of an ecosystem to support itself over a long time. In the context of human use of resources, the concept indicates the ability of an ecosystem to support itself despite continued harvest, removal, or other types of losses. For example, the National Marine Fisheries Services uses measures of fish reproduction, growth, and recruitment to determine allowable fish harvests.

Resiliency is the ability of an ecosystem to adapt to or to recover from a stress. The stress might be natural (e.g., flood, fire, pest outbreak) or anthropogenic (e.g., timber harvest, chemical releases, development of land). A system subject to a permanent alteration will not return exactly to its original state. Resiliency is a natural attribute of most undisturbed ecosystems, reflecting the natural stresses (e.g., drought, temperature extremes, cycles in population abundance) to which the ecosystem has adapted over evolutionary time. However, human activities often move such stresses beyond the range found naturally and beyond the level of resilience developed by the ecosystem during its evolution.

Biodiversity has been variously defined. One useful definition is that of Norse (1990), who describes biodiversity as "the variety of life on all levels of organization, represented by the number and relative frequencies of items (genes, organisms, and ecosystems)." Biodiversity is one of the keys to an ecosystem's sustainability and resiliency. Ecosystems that contain more species and include higher levels of genetic variation within species often are better able to recover from disturbances than other ecosystems. This is because biodiversity tends to reflect internal structural and functional redundancies in an ecosystem, such that the loss of some individuals or species is compensated for to some extent by other individuals and species (U.S. EPA, 1997).

3.3.2 Identifying Valued Ecological Entities

Publicly valued ecological functions, services, and entities are evidenced by current laws, by private and government actions, and by expressed human values and philosophies. The values range from immediate human utility to values that are independent of humans. At one end of the spectrum are utilitarian values include the use of natural or manmade resources for direct human consumption and in the marketplace. Somewhere in the middle are recreational and aesthetic uses and human-derived preservation values. At the other end of the spectrum are those associated with moral, religious, and spiritual values (U.S. EPA, 1997).

For EPA, several statutory provisions direct EPA's attention to several specific ecological concerns. These include ecosystem components, ecosystems, and special places (U.S. EPA, 1997):

- **Ecosystem components**: The Clean Water Act (CWA) specifies fish, shellfish, and wildlife as entities for protection.; the secondary National Ambient Air Quality Standards (NAAQS) under the Clean Air Act specifies soils, water, crops, vegetation, animals, wildlife, etc. as entities for protection..

- **Ecosystems**: The Clean Air Act (CAA) refers to "regionally representative" and "critical" ecosystems, and the CWA specifies rivers, lakes, and estuaries. The CAA gives the Administrator the authority to assess risks to ecosystems from criteria pollutants.

- **Special places**: The CWA and CAA identify the Chesapeake Bay, the Great Lakes, and Lake Champlain. The CAA also makes special provisions for national parks and wilderness areas. The CWA identifies "Outstanding Natural Resource Waters" for enhanced protection.

In its discussion document for *Priorities for Ecological Protection*, EPA (1997) has proposed four criteria for prioritizing ecological entities to be protected (see Chapter 3): mandated protection, other societal values, rare or under threat, and ecological significance.

Mandated protection: Protection for certain types of entities is mandated by law (e.g., endangered species are protected by the Endangered Species Act; fish, shellfish, and wildlife by the Clean Water Act; and special places such as the Great Lakes by the CWA and CAA). These laws codify some of the indirect use and non-use values that humans place on ecological entities.

Other societal value: As evidenced in its laws, practices, and community projects, society values organisms, places, ecosystems, and their structures and functions for commercial, recreational, spiritual, or other reasons. Economic assessments regularly address commodities that are used in commerce (market) and the recreational (non-market direct-use) values of ecosystems. Although techniques to monetize indirect-use and non-use values of ecological entities are not yet available, these values can be addressed at least qualitatively in the economic assessment.

Rare or under threat: Species need not be designated as threatened or endangered to be at risk of local or regional extinction. Many species of both plants and animals are declining or are already so rare that some additional stresses might easily lead to their extinction. An example is neotropical migrant songbirds. Although few of these species are federally designated as threatened or endangered, populations of most species in this group are declining as their habitats in both the northern and southern hemispheres are degraded, fragmented, and lost. Even rare species contribute to overall biodiversity and can provide functional redundancies within ecosystems, contributing to resilience. Economic valuation of the protection of rare or threatened species, communities, and ecosystems can be achieved through the economic valuation of biodiversity.

Ecological significance: Ecological entities that help to sustain ecosystems include plants and animals that provide a significant food base, promote nutrient cycling, assist in regenerating critical resources, or through competition or predation are "key" to maintaining the balance of species in a community. These are often referred to as "keystone" species or functions. If the ecological assessment team identifies potential impacts or benefits to keystone species, the team should continue the conceptual model to include the important functional and structural attributes of the ecosystem or community that might be affected by changes in the abundance or presence of a keystone species.

In response to one of the recommendations of EPA's (1994b) report *Managing Ecological Risk*, the Agency has begun a process of trying to reach consensus on a list of ecological concerns or entities that should be considered in every EPA decision where relevant (U.S. EPA, 1997). Exhibit 10, from EPA's (1997) discussion document *Priorities for Ecological Protection*, provides the proposed list of ecological entities in three categories: (1) animals, plants, and their habitats; (2) whole ecosystems; and (3) special places and species. For each entity, the table provides examples of attributes of the entities that deserve protection. The combination of an ecological entity and an attribute of concern for the entity represents an assessment endpoint. Also for each entity, the table provides examples of the management objectives for each specific ecological entity. Finally, each ecological entity is evaluated relative to the four criteria listed above.

3.3.3 Neglected Benefits

Of particular importance to economic assessments of ecological benefits is the entity "ecosystem functions and services" which have very high ecological significance, but values to society that often are not recognized.

"Neglected" Ecological Benefits

Biotic Resources
- Species habitat
- Biotic productivity
- Species fitness
- Food chain support
- Biodiversity
- Pest control
- Pollination

Processes/Infrastructure
- Microclimate control
- Geomorphological control
- Water supply
- Energy and nutrient exchange
- Purification of resources

Sources: U.S. EPA, 1993; Principe, 1995

Biotic Resources

- **A species habitat** is defined as the environment that a given species uses over the course of its life history. It includes biotic (e.g., assemblages of plant and animal species) and physical (e.g., rainfall and temperature range) components. For animal species, suitable habitat is essential for both resident and transient animal populations. Of particular value is habitat for endangered, threatened, and rare species and habitat that is vital to important animal species activities (e.g., reproduction, foraging, migration, and overwintering).

- **Biotic productivity** refers to the total amount of growth among organisms at any level of the ecosystem. It includes primary productivity, which accounts for the growth of autotrophic organisms (primarily plants) that manufacture their own organic materials from inorganic sources. Biotic productivity at all trophic levels is essential for energy transfer and for maintaining the integrity of natural food webs.

- **Species "fitness"** refers to the ability of a species to sustain its populations over the long term.[1] Attributes that are closely related to species fitness include reproductive success, survivorship, and genetic diversity. Genetic diversity within a species is needed to allow adaption of the population to changing environmental conditions.

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[1] The more traditional use of the word "fitness" is to designate reproductive fitness, which is an individual trait, not a species attribute.

Exhibit 10
EPA's 1997 Proposed List of Ecological Entities to be Considered in EPA Decisionmaking

Category	Ecological entity	Examples of attributes	Examples of objectives	Criteria			
				Mandated	Societal Value	Rare, under threat	Ecological significance
Animals, plants, and their habitats	1. Aquatic communities in lakes, streams, and estuaries	Survival, development, reproduction of aquatic species; habitat extent for key species	Protect 95 percent of aquatic species, or maintain population of a key species	Some (CWA)	High for fish and shellfish	Some	Relatively high
	2. Regional populations of native species and their habitats - terrestrial and aquatic	Survival and recruitment; habitat extent	Maintain viable regional population of native species; maintain or restore habitat for native species	Some in CAA, CWA	High for some	Some	High for some
	3. Groups of native or migratory species exposed to severe or acute threat	Survival without visible damage	Avoid widespread and recurring or massive die-offs	Some (e.g. Migratory Bird Treaty Act)	Usually high	Not usually	Varies, often unknown
Whole ecosystems	4. Ecosystem functions and services	Nutrient recycling, ability to filer pollutants, habitat extent for diversity of species	Maintain or restore function or service to some standard	Some general authority	Not always recognized	A few	Very high

Exhibit 10
EPA's 1997 Proposed List of Ecological Entities to be Considered in EPA Decisionmaking

				Criteria			
Category	**Ecological entity**	**Examples of attributes**	**Examples of objectives**	**Mandated**	**Societal Value**	**Rare, under threat**	**Ecological significance**
Whole ecosystems	5. Wetlands and stream corridors	Extent	Maintain extent of wetland	Yes	High for many	Some	High
Special places and species	6. Endangered ecosystems (e.g., old-growth forests, tall-grass prairies)	Extent	Maintain extent of endangered ecosystem types	Some	High for some	All	Important for biodiversity
	7. Endangered species and their habitats	Survival, development, reproduction, and recruitment	Maintain and restore populations	Yes	Potential for some	High	Usually low
	8. Other places with high ecological or societal value, as appropriate	Species diversity; nutrient levels, etc., appropriate to the type of ecosystem; landscape measures	Restore biodiversity, maintain as oligotrophic lake, maintain extent of certain habitat	Some (e.g., Great Waters by CAA)	High for many	Some	High for many

[a]The term objective is used here to refer to a specific objective for an ecological entity.
[b]Special places do not necessarily fit the definition of entity in EPA's (1998) *Guidelines for Ecological Risk Assessment*.
[c]An oligotrophic lake is one with low (but adequate) nutrient and high oxygen levels.

Source: U.S. EPA, 1997

- **Food chain support** refers to the support of the trophic structure of communities through adequate primary productivity and balanced predator-prey relationships that maintain the species diversity and abundance of organisms in each trophic level naturally associated with an ecosystem. These relationships are essential for energy transfer and for maintaining the overall integrity of the food web.

- **Preservation of biodiversity** includes maintaining genetic diversity within populations, species richness within communities and ecosystems, and ecosystem variety within landscapes. Biodiversity is generated and maintained in natural ecosystems, where organisms encounter a wide variety of living conditions and chance events that shape their evolution in unique ways. Overall, biodiversity provides a reservoir for change, enabling life to adapt to changing conditions.

- **Pest control** refers to natural pest control, which includes the control of pests by their natural enemies (e.g., predators, parasites, and pathogens), by genetic resistance in host plants, and by natural conditions or man-made environmental modifications (e.g., fallows, hedge rows, flooding) that interrupt reproductive cycles of pest species, including weeds. Natural pest control helps maintain the stability and diversity of ecosystems and reduces societal reliance on chemical pest control.

- **Pollination** refers to the dependence of many plants on insects or other wild animals for sexual reproduction (i.e., transfer of pollen). Successful pollination contributes to the overall maintenance of both plant and animal diversity in an ecosystem. Pollination is related to species habitat because the availability of pollinators can be affected by the availability of their foraging and reproductive habitats.

Process/Infrastructure

- **Microclimate control** includes processes such as shading and wind breaking that provide local and regional temperature control. Microclimate control is essential to maintaining the structure and quality of many wildlife habitats.

- **Geomorphological control** describes the services that maintain the physical integrity and structure of ecosystems and wildlife habitats. Specific processes include the following: organic production and export, sediment trapping, soil generation, flood control and desynchronization, storm surge protection, wave and wind buffering, shoreline anchoring, erosion control, and disturbance recovery.

- **Water supply** includes the quantity and quality of groundwater and surface water, which influence both the amount and quality of available aquatic habitat and characteristics of terrestrial vegetation. For example, groundwater recharge protects aquifers from saltwater intrusion in coastal areas, which otherwise could alter the species composition of the local plant, and therefore animal, communities. Terrestrial animals also rely on water for basic life support functions.

- **Energy and nutrient exchange** refers to processes that control the flow of energy, minerals, and nutrients (such as nitrogen, phosphorus, carbon, and sulfur). For example, photosynthesis by primary producers captures energy from the sun and converts inorganic carbon to organic carbon. Decomposition of dead biotic materials is essential to the provision of essential raw materials. These energy and nutrient exchange processes make energy and essential raw materials available to other organisms.

- **Purification of resources** includes the retention and detoxification of pollutants as well as the removal of excess nutrients. The retention and detoxification of pollutants can reduce adverse effects on survival, growth, and reproduction in wildlife. The removal of excess nutrients by microorganisms can maintain the integrity of aquatic ecosystems by preventing algal blooms and anoxic conditions where they do not occur naturally.

These neglected benefits can prove particularly useful in developing conceptual models for economic assessments of ecological benefits, as described in Chapter 4.

3.4 TYPES OF ECOLOGICAL ASSESSMENTS

Ecological assessment is a process used to evaluate changes to ecological resources resulting from natural or manmade events. Ecological assessments rely on the principles of ecology, discussed above, to identify, describe, and estimate the consequences of a change to any component(s) of an ecosystem. The changes may be biological (e.g., introduction of a nonnative predatory species), chemical (e.g., presence of a toxic chemical), or physical (e.g., loss of habitat). The ecosystem effects of the changes depend on those components of the ecosystem that are directly impacted and interactions of those components with the rest of the ecosystem.

3.4.1 Assessment Models

Prospective (stressor-driven) and Retrospective (effects-driven). Ecological risk/benefit assessments can be used to estimate the likelihood of future adverse effects or improvements (prospective) or to evaluate the likelihood that existing effects are caused by past exposure to stresses or removal of stresses (retrospective). Prospective risk assessments are stressor driven, and prospective benefit assessments are driven by proposed management options. Retrospective risk assessments are impact driven. For example, most watershed-level ecological risk assessments have been driven by observations of loss of water quality and degradation of aquatic communities, with the associated loss of recreational and fisheries values of the waters. Retrospective benefit assessments could be conducted to determine the efficacy of actions taken to help improve ecosystem condition.

Individual-level ecological risk or benefit assessments are used only for endangered species. The ecological endpoints for assessing risks or benefits to such species include individual survivorship, growth, health, and reproductive success. This level of assessment ignores the rest of the species in an ecosystem and higher-level ecological entities. This approach can be useful in assessing threats or benefits to endangered species in the context of actions that would affect that species. Because individuals of endangered species are considered valuable, this approach generally looks for a threshold for individual-level effects rather than dose-response information.

This level of assessment generally misses ecosystem resources and services that should be addressed in economic assessments of ecological benefits.

Population-level assessments can be approached from the bottom up using data on individual-level endpoints in or from the top down by modeling interactions among species in a community. The bottom-up approach uses individual-level endpoints, including mortality, growth, and reproductive success, to model population growth or sustainability. The top-down approach uses models of competition or predatory-prey relationships. Population-level assessment endpoints include the number of organisms that can be harvested on a sustainable basis, population density, and the probability of extinction. Where sustainable populations of a species are valued because of their direct use (e.g., consumption) by humans, a population-level assessment is key to an economic assessment of benefits. Information on life history strategies is important to population-level assessments. As Suter (1993) points out, populations of long-lived vertebrates (e.g., whales, seabirds) are more sensitive to changes in adult mortality than are shorter-lived species (e.g., quail, grasshoppers) that produce large numbers of offspring at each reproductive effort. The shorter-lived species can be more sensitive to short-term catastrophic events that coincide with and affect critical life stages (Suter, 1993).

Many approaches have been developed to conduct population-level assessments, including modeling reproductive potential (e.g., using the "Leslie matrix"), aggregated models, and individual-based models (noted above) using logistic growth or age-structured population models (Suter, 1993, Gotelli, 1998). Aggregated models use aggregate components, such as population size or adults and juveniles, to assess population-level effects. These models are simplified versions of models of age-structured populations, where age is divided into one or two classes.

A difficulty with modeling population growth or harvest potential is that the factors limiting a population (e.g., food resources or predation) often are difficult to identify or measure in the field (Suter, 1993). Most populations exhibit density-dependent mortality and growth, which complicates the modeling process. However, simple density-independent models have been used successfully by fish and wildlife managers where time horizons are short and expected changes are small (Suter, 1993). Gulland (1977) provides helpful discussions of surplus production models and various stock-recruitment models developed by fisheries biologists.

Community-level assessments focus on the interactions among species in the community, including predator-prey and competitive relationships. These interactions can be much more sensitive to a stress than the individual-level endpoints (e.g., mortality, growth, and reproduction) used to estimate population-level effects. For example, chemical contaminants might impair the ability of a prey species to detect (e.g., sensory impairment) or escape from a predator (e.g., motor impairment) at levels that do not otherwise affect the growth and reproduction of the species. Many species interactions are of direct economic importance, such as the predatory behavior of biocontrol agents and the symbiosis between legumes and nitrogen-fixing bacteria (Suter, 1993). Assessment endpoints at the community-level include community trophic structure and indices of community species composition (e.g., Index of Biotic Integrity, IBI). Difficulties with community-level assessments include selecting the interactions on which to focus and data availability. To the extent that an action or a stressor impacts the relationship of some species with their abiotic environment, ecosystem-level assessments can be needed.

Ecosystem-level assessments use ecosystem properties, such as eutrophication, changes in biodiversity, and net productivity, as the assessment endpoints. They also can be used to predict changes in community or population-level endpoints (Suter, 1993). For example, an ecosystem model can be used to predict the effects of changes in populations at lower trophic levels on species at higher trophic levels and changes in community structure that result. Ecosystem-level endpoints are needed to assess changes in many of the beneficial ecosystem services (e.g., flood protection, water filtration, microclimate control). In general, ecosystem-level observations and data are needed to support stressor-response profiles for ecosystem-level responses. Relatively few laboratories run such tests on toxic chemicals. Stressor-response data for ecosystem services tends to be derived from field studies in which parameter measures (e.g., areal extent of wetland water recharge rate) are correlated with indicators of the ecosystem service (e.g., flood control). A key uncertainty in using ecosystem-level stressor-response data in prospective risk/benefit assessments is in extrapolating from the observed ecosystem to other ecosystems (Suter, 1993).

The most common practice in ecological risk assessments at this time is to predict ecosystem-level responses from the bottom up, extrapolating from lower-level responses to the ecosystem level. Because of functional and structural redundancies in ecosystems, significant effects can occur in single species without affecting ecosystem structure or function. An ecosystem cannot be more sensitive than its most sensitive component. Thus, it is a common practice in risk assessments to identify thresholds for adverse ecosystem effects based on individual-level effects on one of the most sensitive species. For example, EPA's National Water Quality Criteria identify a threshold for adverse ecosystem-level effects as the best estimate (fiftieth percentile) of a concentration that would protect 95 percent of the species in the system (Stephan *et al.,* 1985). Use of individual-level effects to predict a threshold for ecosystem-level effects is not helpful, however, for the economic assessment of ecological benefits because it does not allow estimates of the magnitude of response, i.e., it does not use exposure-response data to predict the degree of change in response to an action of a specified magnitude.

Landscape-level assessments are needed where watersheds are at issue or where the geographic distribution, connectivity, and diversity of different ecosystems or habitats over a large geographic area can be affected by a proposed action. MacArthur and Wilson's (1963, 1967) theory of island biogeography states that the low species diversity characteristic of oceanic islands reflects a dynamic equilibrium between rates of extinction and rates of colonization of individual species' populations. This theory has been used to assess the effects of habitat loss and fragmentation in terrestrial environments on animal species, including neotropical migrant songbirds, wolves, turtles, and many others. Landscape-level assessments are also needed to assess "edge effects," such as higher bird nest predation rates by blue jays and crows at the edge of forests rather than in the interior of forests (Terborgh, 1989; Wilcove, 1985). Nest parasitism by brown-headed cowbirds also occurs at the edge of forested habitats (Terborgh, 1989). Edge effects can penetrate several hundred meters into forested habitats. Thus landscape-level measures such as the ratio of the length of forest edge to the area of the forest interior can be useful in predicting population-level responses of valued species.

3.4.2 Standardized Approaches to Ecological Assessments

Ecological assessments have been conducted under a variety of statutes by federal and other agencies for some decades. Standardized approaches and nomenclature for those assessments have developed somewhat independently between the different agencies and offices. Differences among the approaches can be attributed to different statutory requirements, the type of information generally available at the start of an assessment, and the experience of the agency or office responsible for the assessment. The remainder of this section briefly describes the primary different types of ecological assessments and explains why EPA's (1998) *Guidelines for Ecological Risk Assessment* was used as the starting point for development of this Framework.

Types of Standardized Approaches

A variety of different "standardized" approaches to ecological risk and impact assessments have developed over the past few decades in response to various legal mandates administered by different agencies. Key attributes of some of the more well-recognized approaches are listed in Exhibit 11.

Ecological Risk Assessment. As defined by EPA (1998), an ecological risk assessment is a process that evaluates the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors. Ecological effects can be evaluated both qualitatively or quantitatively in terms of structural and functional changes at one or more levels of biological organization. In EPA's paradigm, the part of the ecosystem that is affected by the change is called a "receptor(s)" and is usually a structural component. The natural or anthropogenic (i.e., manmade) event causing the effects is called a "stressor." Stressors can be chemical, physical, or biological. The "effects" of the stressor include direct changes to the receptor(s) as well as indirect changes to other structural or functional components that are affected through the interconnections that define the ecosystem (e.g., energy flows through the food web).

Environmental Assessment (EA). An EA is frequently required under the National Environmental Policy Act (NEPA), prior to, and in some cases, in lieu of, the preparation of an Environment Impact Statement. EAs are concise documents prepared on a case-by-case basis by government agencies. They describe the environmental impacts of a proposed government action, provide a listing of agencies or persons consulted, and discuss possible alternative actions. There must also be an evaluation of the probable cumulative, long-term environmental effects including any beneficial impacts.

Environmental Impact Statement (EIS). An EIS is a type of assessment that attempts to reveal the consequences of a proposed action as an aid to governmental decisionmaking. In the United States, federal agencies are required by NEPA to prepare an EIS for any "major federal action." Similar requirements exist for some states as well as for a few other nations. The scope

Exhibit 11
Standardized Approaches to Ecological Risk Assessment

Standard Name for Assessment	Statute	Key Attributes	Process
Ecological Risk Assessment (ERA): national-level	various, e.g. RCRA, CAA	prospective/stressor driven, traditionally chemical, now being adapted to other stressors	tiered
ERA: regional, landscape, or watershed level	various, including CWA	prospective/stressor or impact driven, usually chemical	tiered
ERA: site-specific	various, e.g., CERCLA, RCRA	prospective/stressor driven, usually chemical	tiered
Environmental Assessment (EA)	NEPA	prospectiv/stressor driven, all types	screening
Environmental Impact Assessment (EIA)	NEPA	prospective/stressor driven, predominantly physical stressors	refined
Habitat Assessment (and Habitat Suitability Index)	NEPA and land management	prospective/stressor driven, land management changes	refined
Hazard Assessment	various, e.g., TSCA and FIFRA	prospective/stressor driven, traditionally chemicals	screening, and tiered
Natural Resource Damage Assessment (NRDA)	CERCLA	retrospective/impact driven, chemicals only	refined

Acronyms for Statutes: CAA - Clean Air Act; CWA - Clean Water Act; CERCLA - Comprehensive Environmental Response, Compensation, and Liability Act; FIFRA - Federal Insecticide, Fungicide, and Rodenticide Act; RCRA - Resource Conservation and Recovery Act; NEPA - National Environmental Policy Act; TSCA - Toxic Substances Control Act.

and content of an EIS depend on the type of activity under consideration. An EIS is required to predict any or all future effects on the environment. Consequently, it devotes considerably more attention to identifying the full range of affected environmental components, defining the geographic and temporal changes, and identifying secondary and tertiary effects than an EA. NEPA explicitly states a policy of preserving the quality-of-life benefits of natural areas and resources for future generations and evaluating cumulative impacts of activities over time.

Habitat Assessment. Habitat assessments evaluate the suitability of a local habitat to support a given species. The most well known example is the U.S. Fish and Wildlife Service's Habitat Evaluation Procedure (HEP). HEP provides a framework for determining habitat quality for specific fish and wildlife species by quantifying many characteristics of the environment, including physical, chemical, and biological characteristics (Scodari, 1992). The relationships of different values of those characteristics to the species' population densities and reproductive success (implied suitability) have been developed from previous field studies. Use attainability analyses performed under the Clean Water Act are also considered habitat assessments. They determine what uses of a water body are attainable (e.g., swimming, fishing, water supply), the extent to which pollution is impacting these uses, and the necessary pollution control measures that are needed. Use attainability analyses must consider habitat limitations such as frequency of low tides, natural water quality, and physical structure of the habitat.

Hazard Assessment. Hazard assessments determine the existence of a hazard. This type of assessment identifies the types of effects a particular stressor might have on different groups of organisms based on experimental exposures of organisms to the stresses. The hazard assessment helps to identify particularly sensitive groups of organisms (or functions), which in turn affect the selection of assessment endpoints. The phrase hazard assessment has been used to describe a comparison of the magnitude of expected levels of stress in the environment to thresholds of effect in groups of organisms (or functions). Currently, that activity is more appropriately called a screening-level ecological risk assessment.

Natural Resource Damage Assessment (NRDA). Natural Resource Damage Assessments are retrospective assessments that address both ecological and economic damages. Standard methodologies promulgated by the Department of the Interior (DOI) and the National Oceanic and Atmospheric Administration (NOAA) require an assessment of injury to an ecological resource and an evaluation of the economic damages. In an NRDA, federal or state officials, acting as trustees for natural resources, can seek compensation from responsible parties under the Oil Pollution Act, CERCLA, and other statutes for damages to natural resources (e.g., loss of shellfish beds) caused by releases of oil and other toxic materials. Trustees have used NRDA regulations to seek monetary compensation for natural resource injuries associated with accidental releases, such as the Exxon Valdez oil spill. A NRDA may be conducted at a Superfund site at the discretion of natural resource trustees. An injury assessment, which documents the adverse effects associated with a release, is the basis for the NRDA. An injury assessment is basically a retrospective risk assessment to link injuries to particular contaminant sources.

Selection of Approach for this Framework

Of the various standardized approaches noted above, EPA's (1998) *Guidelines for Ecological Risk Assessment* is the most general and flexible, because the *Guidelines* document was designed to encompass the broad range of statutory requirements that different EPA Offices administer. Although historically, EPA has focused on assessments of chemical contaminants in the environment, the Agency intentionally included information on risk assessment for physical and biological stressors as well to further broaden the scope and utility of the *Guidelines* within and beyond the Agency. That *Guidelines* document, for example, is general enough to be adapted to benefit analyses (U.S. EPA, 1998). That *Guidelines* document was developed by EPA's Risk Assessment Forum over a period of years with substantial input from EPA and other federal agencies. Drafts of the proposed Guidelines received extensive scientific peer review and interagency committee review. For these reasons, the framework described by EPA's (1998) *Guidelines for Ecological Risk Assessment* was used as the starting point for building the framework for the economic assessment of ecological benefits proposed in this document. The adaptation of EPA's (1998) *Guidelines for Ecological Risk Assessment* to the ecological benefits assessment addressed in this document is discussed in the next chapter.

The next chapter goes into more detail on conducting an ecological risk/benefit analysis. The purpose of Chapter 4 is to help the economist better understand the scientific framework for analysis and type of information that may be generated through an ecological assessment. Improved understanding of the ecological risk assessment process will facilitate communication between economists and ecologists during planning of ecological assessments and thereby increase the utility of assessment results for economic analyses.

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4.0 ECOLOGICAL RISK/BENEFIT ASSESSMENT

As indicated in Chapter 3, the framework used for an ecological benefits assessment in this document is based on EPA's (1998) *Guidelines for Ecological Risk Assessment*. This section describes the adaptation of that framework to build the proposed framework for the economic assessment of ecological benefits.

4.1 OVERVIEW OF EPA'S GUIDELINES FOR ECOLOGICAL RISK ASSESSMENT

An ecological risk assessment determines the likelihood, potential nature, and magnitude of an adverse ecological effect resulting from exposure to a stressor (U.S. EPA, 1998). Some examples of ecological stressors are listed below:

Physical

- Erosion
- Heat
- Turbidity
- Impoundments
- Habitat alterations

Biological

- Disease-causing organisms (*Pfiesteria*, diatoms)
- Genetically-engineered microorganisms
- Non-native species (kudzu, zebra mussels)

Chemical

- Hazardous substances (e.g., pesticides, industrial wastes)
- Salinity
- Air pollutants (CO, NO_x, ozone, hazardous air pollutants)

The description of potential ecological effects should include magnitude, duration, spatial distribution, time to recovery, and other relevant parameters. As indicated in Chapter 3, ecological risk assessments may be predictive (i.e., estimate the probability and magnitude of *future* ecological changes in response to a given stressor), or they may be retrospective, (i.e., assess the probability that a past event caused this present problem). A predictive benefits assessment may take the form of modeling the effects of an activity (e.g., removal of a stressor), such as the effects of reducing atmospheric nitrogen levels on the Chesapeake Bay. Such assessments depend on applying previously collected data from similar events and ecosystems to a new situation. In a retrospective ecological benefits assessment, an effect may be well defined, such as reduced sedimentation, but the potential of other environmental changes contributing to the improvement must be considered.

Benefits can be expressed quantitatively (e.g., a probability, such as an 80 percent chance that a population will not go extinct in the next 100 years) and/or qualitatively (e.g., low, medium, or high). In addition, the uncertainty associated with the probability needs to be addressed, either quantitatively or qualitatively.

Steps in an Ecological Benefits Assessment

As illustrated in Exhibit 2 of Chapter 2, an ecological benefits assessment starts with planning. During planning, the environmental decision makers, risk assessors, economists, and others determine the need for and scope of the ecological benefits assessment. It is during this stage that societal and political issues are considered, and a key to success is determining the involvement of different individuals in planning the risk assessment. Participants in the planning phase may include risk assessors (including scientists), risk managers (e.g., government regulators), economists, and, if appropriate, interested outside parties (e.g., environmental and industry groups, those whose land may be affected by risk assessment decisions, state and/or local government officials, etc.). Collaborative planning can help foster a consensus on which ecological benefits are most valuable to the stakeholders and the goals, scope, and timing of the ecological benefits assessment.

Stakeholder Involvement in Planning

Waquoit Bay provides an excellent example of how stakeholder involvement can be instrumental in developing management goals for an ecological risk or benefits assessment. It has been generally agreed by all involved parties that the Bay is changing — eelgrass is disappearing and is being replaced by thick mats of macro algae, fish kills are occurring, and scallops have disappeared. Something must be done to prevent further degradation and restore what has already been damaged. Three steps were used to develop management goals for Waquoit Bay:

- A public meeting of all stakeholders;
- An evaluation of written goals by organizations having jurisdiction or an interest in the ecology of the watershed; and
- A meeting of members of these organizations to review and approve the management goals.

The public meeting was advertised in local newspapers. The meeting was designed to determine what the public viewed as valuable in the bay and what the main stressors were on these values. The participants found the bay to be valuable for a number of reasons including open space, scenic views, flyways for waterfowl, shellfishing, navigation, wildlife, and human serenity. Stressors were many. They included physical, chemical, and biological impacts to the bay such as the introduction of non-native species, man-made noise, fertilizers, ~~ignorant tourists,~~ habitat loss, and boat wake disturbance.

Numerous governmental (federal, state, and local) and non-governmental organizations were involved in the review and approval of the management goals. The groups involved in developing these goals are considered the risk management team for the watershed and will be principally responsible for implementing the management plan in Waquoit Bay.

Planning might be considered complete once the following objectives have been met.

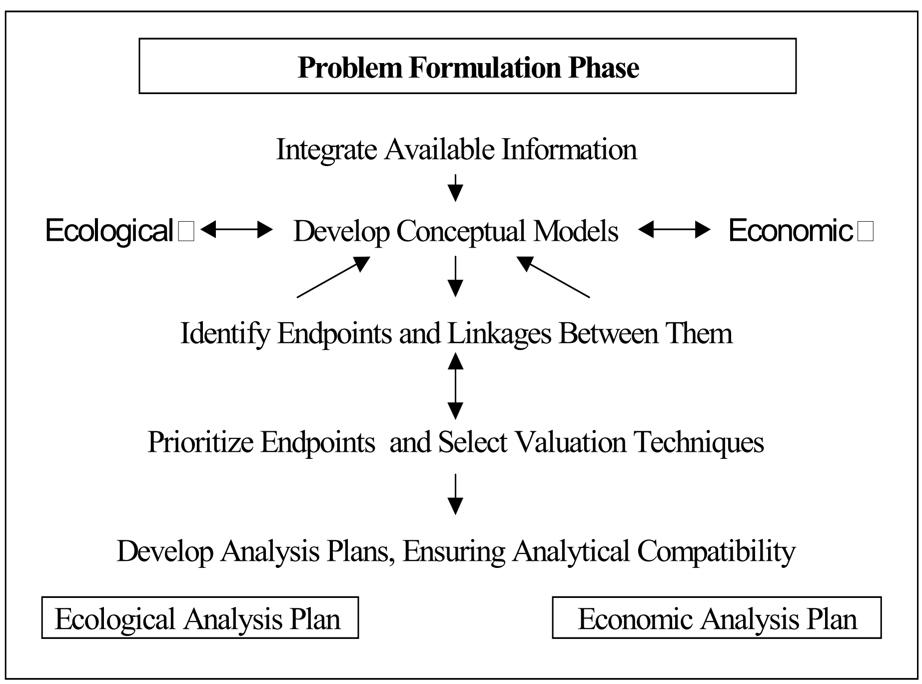
- Objectives of the risk assessment have been defined (including criteria for success);
- Goals for ecological values have been established;
- The range of options under consideration has been developed;
- Focus and scope of the assessment have been agreed upon; and
- Resources to conduct the assessment have been provided (U.S. EPA, 1998).

As described in Chapter 2, EPA defines three phases for conducting ecological risk assessments (U.S. EPA, 1998) that can be adapted for ecological benefits assessments:

- Phase 1 — Problem Formulation;
- Phase 2 — Analysis (exposure assessment and ecological effects characterization); and
- Phase 3 — Risk Characterization.

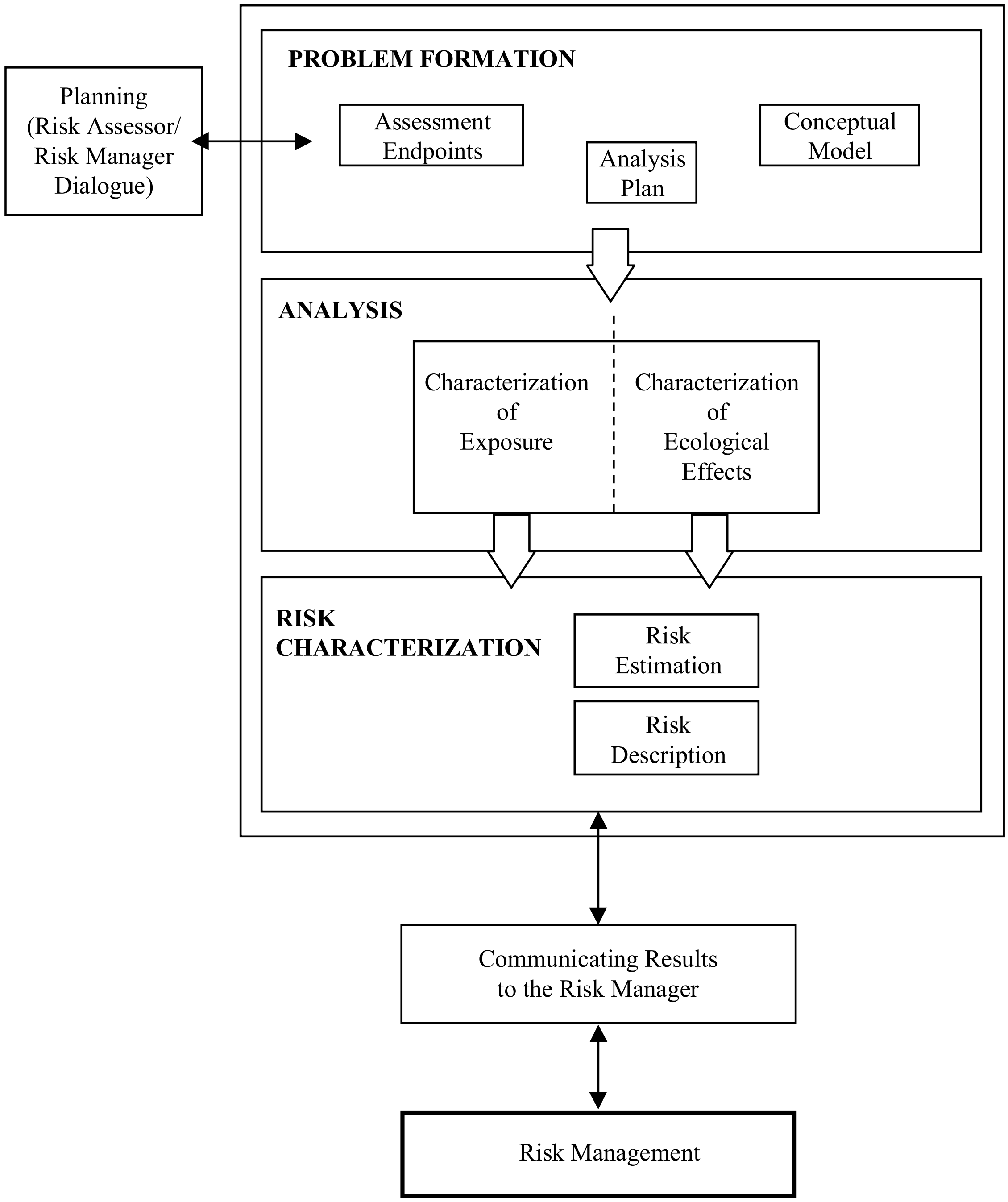
A somewhat simplified diagram of EPA's risk assessment process is provided in Exhibit 12. Readers interested in EPA's (1998) *Guidelines* as they pertain to *risk* instead of benefit assessment are referred to that document. The remainder of this Chapter describes how those *Guidelines* have been adapted to the proposed framework for economic assessments of ecological benefits.

4.2 PHASE I: PROBLEM FORMULATION



Problem formulation in an ecological benefits assessment is the process for generating and evaluating preliminary hypotheses about what benefits might accrue from a proposed action and its alternatives. It provides the foundation for the entire economic assessment of ecological benefits. Problem formulation involves the development of three products: assessment endpoints, a conceptual model, and an analysis plan.

Exhibit 12
Ecological Risk Assessment Framework



The development of these products requires several activities:

- Integrating available information;
- Using an iterative process and coordination between the ecological assessment team and the economic analysts to develop a joint conceptual model that links ecological assessment endpoint to economic benefit endpoints;
- Prioritizing endpoints jointly with the economists; and
- Developing the analysis plan, ensuring analytic compatibility with the economic assessment analysis plan.

4.2.1 Conceptual Model

As described in Chapter 2, the ecologists develop their preliminary conceptual model based on the cascade of ecological effects expected from the action itself to direct effects and indirect effects. The conceptual models presented in Chapter 2 are examples of preliminary models in which little attention has been given as yet to what might be considered as assessment endpoints. The ecologists and the economists work together in an iterative process refining the model to ensure comprehensive coverage and appropriate linkages between ecological endpoints and economic benefit endpoints.

In contrast to the goals of many site-specific ecological risk assessments (e.g., to protect an aquatic community by protecting the most sensitive species in the community), a goal of the economic assessment of ecological benefits is to capture as many of the potential benefits that might result from an action as possible. The emphasis in the benefits conceptual model is to identify the full cascade of ecological effects that might result. In developing the conceptual model of likely ecological benefits, the framework described in EPA's (2000) report *Assessing the Neglected Ecological Benefits of Watershed Management Practices* can be helpful. The report discusses linkages between watershed management practices and the neglected benefits (see Chapter 3), including a general estimate of the strength of the linkages. Exhibit 13 below provides an example of the neglected ecological benefits that can accrue from the forestry management practices of revegetation and forest regeneration. Exhibit 13 also provides specific examples of benefit endpoints within each category of neglected benefits.

The conceptual model is accompanied by hypotheses about how the initial action causes both direct and indirect effects. The complexity of the conceptual model depends on the complexity of the problem (e.g., number and types of stressors, number of assessment endpoints, nature of effects, and characteristics of the ecosystem).

A *conceptual model diagram* (see Exhibit 14) is a useful way to visually express the relationships described by the benefit hypotheses. Conceptual model diagrams can communicate important exposure pathways in a clear and concise way, facilitating the coordination between ecologists and economists in problem formulation. These diagrams and hypotheses also are useful tools to aid communication with the environmental decisionmakers and interested parties.

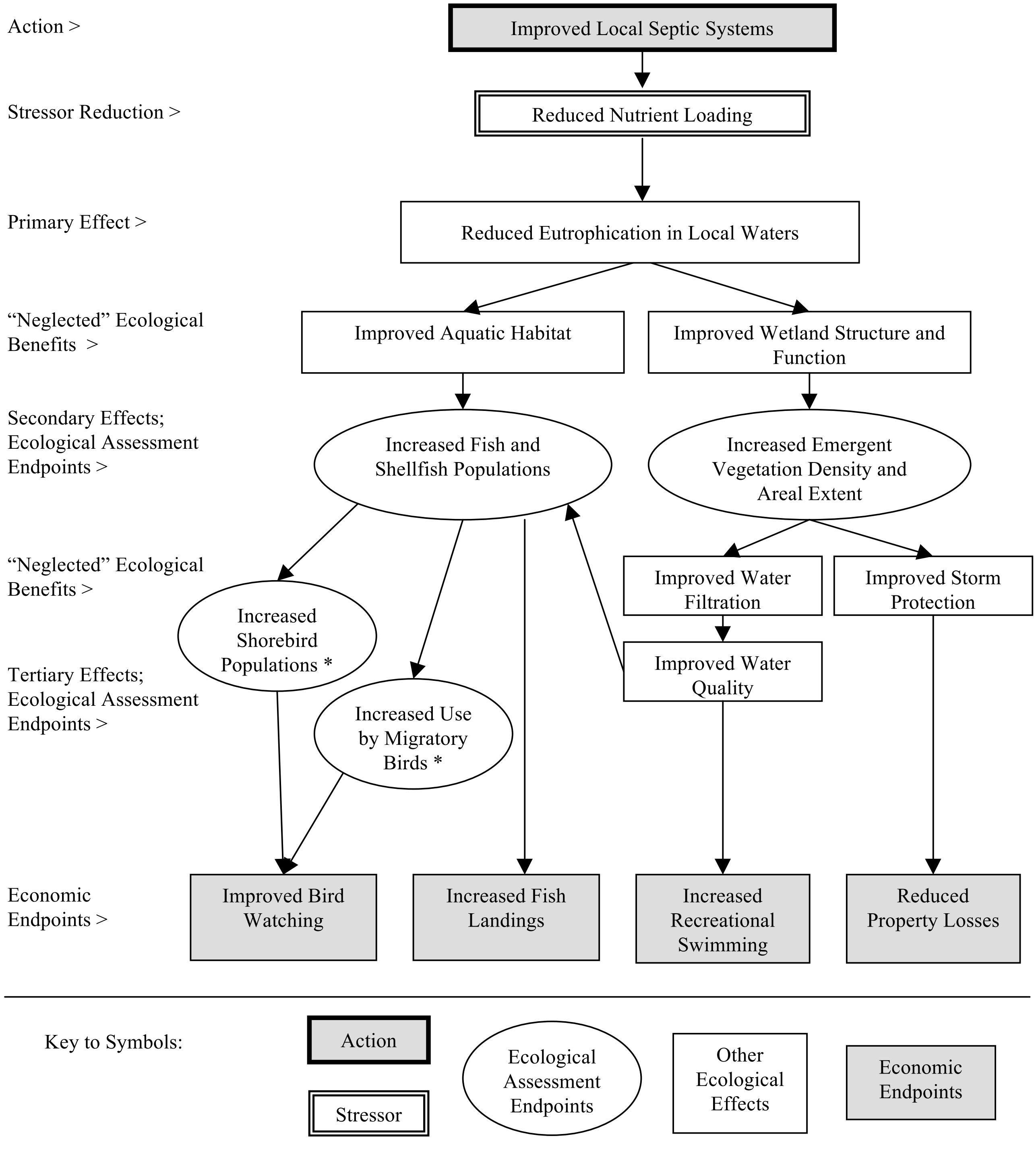
The number of relationships that can be depicted in one flow diagram depends on the comprehensiveness of each relationship. The more comprehensive the relationship, the fewer relationships that can be shown with clarity in one diagram, thus separate diagrams may be required. There is no set configuration for conceptual model diagrams.

In developing the conceptual model, the ecologists can consider each proposed action in relation to each category of neglected benefits, trying to identify and specify ecological benefits in that category that might result from the proposed action. Using the list of neglected benefits should help to ensure that the ecological conceptual model is comprehensive and that as many ecological benefits of a proposed action as possible are identified for the economic assessment. Exhibit 14, illustrates where use of "neglected" ecological benefits in the conceptual model helps to link the action to important and quantifiable ecological benefits.

Exhibit 13
Examples of Benefits Associated with Forestry Practice of Revegetation and Forest Regeneration (from U.S. EPA, 2000)

Neglected Benefit Category	Example(s) of Benefit(s) Provided by Watershed Management Practice(s)
Species Habitat	• increase in species habitat provided by new trees
Biotic Productivity	• contribution of new trees to primary productivity • maintenance of aquatic plant productivity through decreased water column turbidity and increased light penetration
Biodiversity	• maintenance of natural species assemblages by planting native species • maintenance of or increase in biodiversity by increasing available forest habitat for threatened or endangered species
Food Chain Support	• maintenance of species and population balance within food web
Microclimate Control	• increase in stream shading due to new trees in riparian areas • maintenance of wind speed
Geomorphological Control	• prevention of landslides and erosion by stabilizing soil
Water Supply	• decrease in soil erosion that could otherwise increase water turbidity and nutrient loading
Energy and Nutrient Exchange	• preservation of soil microbes by maintaining leaf litter levels
Purification of Resources	• preservation maintenance of decomposition organisms in leaf litter and soil • interception and retention of airborne pollutants

Exhibit 14
Conceptual Model for an Economic Assessment of Ecological Benefits



* Indicates assessment endpoints that might not be needed for an ecological risk assessment, but are added to assist the economic benefit analysis.

4.2.2 Assessment Endpoints

The conceptual model of the ecological benefits assessment should include well-defined assessment endpoints that will be linked to the economic benefit endpoints. In the conceptual models in Chapter 2, a series of vague endpoints were linked, with "improved wetland structure and function" leading to improved water filtration and storm protection. However, the concept of improved wetland structure and function is too vague to allow quantitative links to those benefits. Thus, an assessment endpoint specified as "increased emergent vegetation density and areal extent" was inserted to allow quantification.

Assessment endpoints may be defined for both structural and functional aspects of an ecological resource at any level of organization ranging from a single individual to an entire landscape. Exhibit 15 provides examples of assessment endpoints at different levels of biological organization. Most benefit endpoints, however, are likely to be defined at the population level or higher. This is true because a population is the lowest level of biological organization that can be meaningfully protected (Suter, 1993). An effect on one or several individuals will not necessarily result in significant population changes. Exceptions to this premise are threatened and endangered species for which each individual is valuable to the survival of the population.

Exhibit 15
Examples of Assessment Endpoints

Level of Organization	Ecological Entity	Assessment Endpoints
Individual	bald eagle	survivorship, reproductive success
Population	fish populations	population density, population growth rate
Community	aquatic communities	survival, development, and reproduction of fish, aquatic invertebrates, and plants
Ecosystem	lake	eutrophication, nutrient flux
Landscape	habitat	connectivity, ratio of length of edge to area of interior, patch size

To assess interactions, it is important to use the most appropriate biological level of organization at which the interactions can be observed. For example, community-level effects can be obscured at the ecosystem-level of observation, some ecological changes can only be effectively evaluated from a landscape perspective, and so on. Assessment using a population -, community-, ecosystem-, watershed-, or landscape-level assessment may be most appropriate depending on the situation. For bioaccumulative compounds, exposure of the populations must be traced through the food web in order to understand the full magnitude of potential impacts to the community and ecosystem. The selection is based on the conceptual model on which the assessment is based. The conceptual model depicts the endpoints selected for assessment and the interactions (or linkages) among endpoints.

Both structural and functional changes should be considered in a benefits assessment. Improvements of a specific habitat, such as an increase in wetland acreage resulting from the removal of a barrier to water flow, might yield both benefits that are both structural and functional in nature. In Exhibit 14, the conceptual model indicates that increased emergent vegetation and areal extent can provide improved water filtration services and improved storm protection because of the structural attributes of the emergent vegetation.

Criteria for Prioritizing Assessment Endpoints

As indicated in Chapter 2, there are both ecological and economic criteria that will be important in prioritizing the potential assessment/benefit endpoints for quantitative assessment. This section focuses on the ecological criteria.

By addressing the needs for ecological assessments under a variety of different programs, the Agency has identified some general criteria to assist in selecting and prioritizing assessment endpoings. EPA's (1998) *Guidelines for Ecological Risk Assessment* specify three criteria by which to select assessment endpoints in a risk assessment: (1) have ecological relevance, (2) be susceptible to the stressor, and (3) and be relevant to the management goals (see text box). These criteria are particularly useful in assessments of the risk of adverse ecological effects. For an ecological benefits assessment, additional considerations and criteria for identifying high priority ecosystems and ecological components can be useful. As described in Chapter 3, EPA (1997a) has proposed four criteria for prioritizing ecological entities to be protected in its discussion document for *Priorities for Ecological Protection*: mandated protection, other societal values, rare or under threat, and ecological significance. For a benefits assessment, it is more appropriate to consider these attributes as prioritization criteria, instead of selection criteria. The remainder of this section discusses EPA's 1998 criteria as they might apply to an ecological benefits assessment.

Example of High Priority Assessment Based on EPA's 1998 Criteria

Salmon abundance would rank high among the possible assessment endpoints for evaluating the benefits of not constructing a hydroelectric dam on a river in the Pacific Northwest. Salmon have ecological relevance, because they are a food source for many aquatic and terrestrial species, and they eat many aquatic invertebrates. Salmon are also sensitive to changes in sedimentation, water temperature, and substrate pebble size. Most importantly, salmon are valued by society as a source of food, for recreational fishing, and for their ceremonial and symbolic significance to Native Americans.

Ecological relevance. The ecological relevance of an endpoint refers to its importance in relation to other components of a specific community or ecosystem. For example, if the change in abundance of certain benthic invertebrate species can affect the abundance or productivity of fish in the community, the benthic (i.e., bottom-dwelling) invertebrates are ecologically important. As another example, honeybees are ecologically significant in prairies because they pollinate many plants. Abundance or age structure of a certain population of fish may be selected as an assessment endpoint because of the critical role the species plays in maintaining the functional integrity of the ecosystem (i.e., top consumer that exerts control over lower trophic levels in the food web). Effects on primary producers, such as green algae in a lake, may be critical to higher trophic levels, such as insect larvae, that feed upon the algae, and fish, that feed upon the insect larvae.

Susceptibility to the ecological stressor. For an ecological risk assessment, EPA's (1998) *Guidelines* indicate that an assessment endpoint must be based on a link between a susceptible ecological component (or receptor) and exposure to an ecological stressor. Susceptibility is based on the sensitivity of the ecological receptor to the stressor and attributes of the life history of the receptor that might influence the likelihood (and magnitude) of exposure. Some organisms are more sensitive to stress at certain times in their life cycle such as during molting or during seed germination. In conducting an ecological risk assessment, it is desirable to determine effects on sensitive species and effects during sensitive life stages. For an ecological benefits assessment that evaluates the benefits of removing stressors from the environment, this criterion will be useful in selecting assessment endpoints.

In a benefits assessment, the susceptibility of the different types of receptors included in the conceptual diagram need to be considered. In addition, the potential for some changes to be adverse while others are beneficial needs to be considered. For example, application of a pesticide might kill the target organisms such as mosquitos in a stagnant pond (primary effect) but it might also cause a decrease in the dragonfly population that feeds upon the mosquitos as a primary food source (the dragonflies may starve or leave the area seeking food elsewhere). Another factor to consider is that multiple stressors can increase the sensitivity of ecological components to any given stressor. A seal with a virus may be weakened and thus become easier prey for a shark.

Relevance to management goals. An ecological risk assessment is most useful when the assessment endpoints are related to an ecological component or process that is valued by both the public and decisionmakers such as clean air in parklands. An ecological benefits assessment also requires that the economic benefit endpoints can be clearly linked to ecological endpoints. Ecological benefit assessments should include those ecological components that are used directly by humans, such as sport fish, groundwater, or timber land. Ecological benefit endpoints might also reflect those components and processes that indirectly benefit humans, such as water filtration, climate control, or flood protection. What is actually measured might be different from what is important to the economic valuation study, but the relationship between the two elements should be clearly defined. It also is important to remember that in many cases, selection of an assessment endpoint is constrained by an environmental law, such as the Clean Air Act, or a policy goal, such as pollution prevention via water permitting.

Although not a specific criterion for selecting assessment endpoints, it is important that changes to the assessment endpoint can be predicted or measured, particularly if a stressor-response

Linking the Assessment Endpoints to the Management Goals in Waquoit Bay

The goal (or expected benefits) of the Waquoit Bay watershed management plan is to reestablish and maintain water quality and habitat conditions in the Bay to support diverse native fish and shellfish populations as well as reverse degradation of ecological resources in the watershed. One way to help accomplish this is to reestablish viable eelgrass beds and associated aquatic communities in the Bay. Therefore, an assessment endpoint was eelgrass abundance and distribution. Eelgrass is a rooted plant in the shallows of the Bay that decreases erosion and increases sedimentation, which in turn, provides food and habitat for a variety of marine organisms, such as juvenile scallops, invertebrates, and forage fish.

Eelgrass is a good assessment endpoint because it has great ecological significance (i.e., it provides habitat for fish and shellfish.). The disappearance of eelgrass might have resulted from reduced light due to shading by algal blooms and turbidity from suspended sediments. Stressed eelgrass beds are also more susceptible to disease from slime mold. In addition, its distribution and acreage covered can be readily measured.

Scallop abundance in the Bay is not a good assessment endpoint. Although it might have societal value, it would be difficult to assess whether changes in abundance resulted from natural variability or the effects of the stressor. A qualitative discussion of this endpoint might be useful, however, for benefits analysis.

relationship can be established. If the assessment endpoint cannot be measured directly, appropriate surrogate components or qualitative values will need to be identified as well as methods for extrapolating effects to the assessment endpoints. For example, in many cases, a stressor-response relationship could be impossible to quantify even though the existence of a stressor-effect relationship is well established. This is the case when an affected population cannot be tested, such as an endangered species, or a situation where a synergist is involved. In these situations, it may be appropriate to simply indicate that an effect has been observed without indicating the intensity of the stressor.

Combining Ecological and Economic Criteria for Prioritizing Endpoints

In an economic assessment of ecological benefits, ecological selection or prioritization criteria alone are insufficient for selecting the ecological assessment endpoints that will be examined quantitatively. In addition, economic criteria, as described in Chapter 2, must also be considered. When prioritizing endpoints for assessment, the ecologists and economists must work together to agree to the relative importance of their respective criteria for the overall assessment. The economic selection criteria include:

- Expected magnitude of the change in the economic value of one benefit endpoint relative to other endpoints;

- Anticipated uncertainty associated with the predicted change and value of the change for the benefit endpoint relative to other endpoints;

- Variation in the change to each benefit endpoint under alternative policy scenarios; and
- Analytical feasibility considerations.

Analysis Plan

In an analysis plan, the ecologists describe the data and measures that will be used to evaluate the benefit hypotheses, i.e., the relationships between proposed actions and assessment endpoints. Measures are identified for exposure, ecosystem and receptor characteristics, and effects. Exposure can be quantified or qualitatively characterized (e.g., for chemical contaminants, how much enters the environment and how it is distributed, including its possible degradation or reaction products). Measures of ecosystem and receptor characteristics identify important life history traits (e.g., reproductive cycles, migration patterns, and habitat types) that affect the receptors' potential exposure or the response of assessment endpoints to the change in the stressors. Measures of effects quantify the response of the receptors to changes in the stressors (e.g., survival, growth, reproduction, and community structure) and help link the effects with the assessment endpoints. The analysis plan also specifies how risks will be characterized (e.g., qualitative vs. quantitative).

As indicated earlier, there are two aspects of an assessment endpoint, the entity (e.g., eelgrass) and an attribute of that entity (e.g., spatial extent). It is the latter aspect that must be measured (quantitatively or qualitatively and directly or indirectly). Spatial extent of eelgrass can be quantified by aerial photography; however, for some assessment endpoints, such as songbird population (an entity) and abundance (an attribute) as a result of pesticide ingestion, it may be difficult to estimate population losses due to mortality if the birds are able to fly away before dying.

Assessment Endpoints and Measures Specified in the Analysis Plan

An ecological benefits assessment is to be conducted for adding a waste-water treatment process and sediment retention ponds at a pulp mill on a river in the Pacific Northwest. One assessment endpoint may be Coho salmon breeding success and fry survival. Possible measures of the effects of reduced loading of toxic substances to the river on the fish may include: egg and juvenile response to low dissolved oxygen, response of adults to change in river currents and flow, and adult spawning behavior and egg survival in response to sedimentation and contamination. Measures of the ecosystem and receptor (fish) characteristics include: water temperature and turbidity, abundance and distribution of breeding substrate, food sources for juveniles, variations in abundance, reproductive cycles, and laboratory tests for reproduction, growth, and mortality. Measures of exposure could include contaminant concentrations in water, sediment, and fish, and dissolved oxygen levels in the water.

As described in Chapter 2, during the prioritization of ecological and economic endpoints, ecologists and economists will have discussed what information is required by the economist and if that information can be derived from or developed during the ecological assessment. During the analysis design phase, ecologists and economists formalize what information is needed by economists and determine how and when that information will be provided. They must also agree on how changes will be described or measured (e.g., from what baseline, under what scenarios, at what level of spatial or temporal detail) and how any limitations or uncertainties will be represented. In addition, coordination is required to ensure analytic compatibility between the ecological and economic assessments:

- The baseline from which effects are measured and the specific scenarios or policy options to consider must be consistent between the ecological assessment and the economic benefit analysis.

- The analysis plans put in writing the assessment design, the analyses that will be conducted, data needs, measures, models to be applied, and statistical techniques to use. Both the ecological and economic analysis plans should specify what will be measured and how changes in endpoints will be expressed.

- The spatial area of consideration defined by the ecological benefits assessment serves as the starting point for defining the spatial limits of the economic analysis. It generally will be true that the spatial scale defined for the ecological analysis will be larger than the spatial scale at which the proposed actions can be described. Because the economic analysis focuses on human uses associated with ecological resources and humans are more mobile than plants and animals, the economic analysis might consider a broader spatial area than that defined by the ecological assessment.

- Both the time horizon and the time step for analysis need to be compatible between the ecological and economic analyses. That requirement does not mean that they must be the same. For example, the time steps over which economic values of the ecological changes are assessed could be shorter than the time steps over which changes in the ecosystem are assessed. It is important to account for benefits for future generations in both assessments.

- The economic benefit's analysis should recognize the uncertainties in the ecological assessment process as well as the uncertainties inherent in economic analysis. The level of uncertainty in the ecological benefit assessment process is often substantial because secondary and tertiary indirect effects are more difficult to estimate than primary or direct effects.

4.3 PHASE II: ANALYSIS PHASE

Once the analysis plan is complete and the ecologists and economists agree that the analyses and measures will be compatible, the actual analyses can begin. In general, the ecological assessment must be conducted first to provide the inputs on predicted changes in ecological endpoints for the economic assessment. The ecological exposure and response assessments are conducted by the ecological risk assessment team independent of the economists. In other words, if problem formulation and planning for the analyses are done correctly, there should be no need for communication between the economists and the ecologists during the ecological analyses. Often, however, unexpected data gaps or unexpected interim modeling results might require discussions between the ecologists and economists to resolve such issues.

The analysis phase consists of the technical evaluation of data to reach conclusions about ecological exposure to the stressor and the relationship between the stressor and ecological effects (U.S. EPA, 1998). During analysis, risk assessors use measures of exposure, effects, and ecosystem and receptor attributes to evaluate questions and issues that were identified in problem formulation.

The analysis phase is composed of two activities: characterization of exposure and characterization of ecological effects (U.S. EPA, 1998). These assessments are usually conducted simultaneously, and interaction between the scientists conducting them is recommended.

Exposure Assessment

Characterization of exposure in a risk assessment identifies the source(s) of the stressor, the spatio-temporal distribution of the stressor in the environment, and the contact or co-occurrence of the stressor with ecological receptors. Many benefits assessments will evaluate the removal of a stressor from the ecosystem; however, similar analyses are required to estimate which existing exposures might be reduced or eliminated by a proposed action. The exposure assessment should identify the source of the stressor and the complete pathway by which it is acting upon the receptor. A complete pathway indicates that a stressor is released from a source, is present at a level that may cause an effect, and that the receptor is present and susceptible in the ecosystem and co-occurs in time and space with the stressor. Exposure assessment may start with the source when it is known, but in cases where the source is unknown, the analysis may attempt to link observed contact of the stressor (e.g., a chemical contaminant) with the receptor (e.g., fish) to a source. Contaminant residue levels in fish are examples of observed contact.

Nitrogen Loading in the Chesapeake Bay

The Chesapeake Bay is eutrophic, with excess algal growth causing declines in fish populations. Several possible sources of excess nutrients have been identified:

- Atmospheric deposition
- Run-off from agricultural land
- Industrial waste streams

Although fertilizer runoff is the most obvious source of the pollution, atmospheric deposition, which may originate many miles from the watershed, has been demonstrated to be a significant loading factor. Any activity proposed to reduce nitrogen loading from one of these sources should be evaluated in conjunction with estimates of the loading from the other sources.

In addition to establishing the original or current source of the stressor, the stressor should be described in terms of its distribution in time and space. Several factors that may be considered in describing a stressor include:

- **Intensity** - How much of the stressor is in the environment and at what concentration or magnitude? It may be necessary to determine the persistence of the stressor if the concentration is not the same at the source as it is at the receptor.

- **Duration** - Is the stressor present for a short time or an extended period of time, and how is the time defined (hours, days, years)?

- **Frequency** - Is the stressor occurring as a single event (chemical spill or volcanic eruption), intermittent (pesticide spraying twice a growing season), or continuous ?

- **Timing** - What is the occurrence of the stressor relative to biological cycles (e.g., if it affects reproduction, is it present during the breeding season or is it present when animals are in hibernation)?

Source of Stressors in Waquoit Bay

Multiple potential sources were identified for the many stressors acting upon the Bay. Some of the sources were local, others were regional. Among the sources of the stressors to the Bay are:

- Cranberry cultivation, which releases nitrogen fertilizers, animal wastes, and pesticides;
- Local and regional atmospheric deposition of nitrogen and toxic contaminants, including mercury;
- Residential development, which results in releases of nutrients from fertilizer and septic systems, habitat loss from housing and road construction, and altered groundwater flow due to increased impervious surfaces and the number of wells;
- Industrial discharges to groundwater from a military installation;
- Sewage treatment facilities and runoff of nutrients and contaminants entering the surface waters; and
- Marine activities that alter habitat, increase contamination, disturb sediments and shorelines, dredging, and increased fish and shellfish harvesting.

Thus to remove many of the stresses on the Bay, actions will be needed in many of the community sectors. A screening-level economic assessment of the ecological benefits of reducing the stresses might be conducted for the purpose of helping the local governments to prioritize activities to reduce the stresses.

- **Location** - What is the physical area over which the stressor acts? The stressor may act over a very limited area (application of a pesticide in a specific area), or it may act over a large area (tropospheric ozone). What types of habitats are affected (e.g., nesting or spawning habitat).

Many stressors have natural counterparts (e.g., biogenic sources) or multiple sources. The characterization of these other sources can be an important component of the analysis. Whether alternative sources are analyzed in a given assessment, however, depends on the objectives articulated during problem formulation.

Describe the Distribution of the Stressor in the Environment

The spatial and temporal distribution of a chemical stressor(s) in the environment is described by evaluating the pathways the stressors take from the source to the receptor (e.g., what is the medium to which the stressor is released — air, soil, or water — and does it move from one medium to another? For example, if a chemical is released to water, does it vaporize?). For physical stressors that directly alter or eliminate natural habitats, the temporal and spatial distribution of the changed environment should be described (e.g., for how many miles downstream from the dredging is turbidity in the water column evident?). For biological stressors, the distribution may be more complex. These stressors have the ability to reproduce in suitable environments, and do not necessarily rely on passive transport by wind, water, or gravity to disperse or move to a suitable habitat. Therefore, when identifying the exposure pathways for biological stressors, both active and passive modes of dispersal need to be considered. Furthermore, the ability of the biota to reproduce in favorable habitats can alter the relative importance of alternate exposure routes.

Examples of Biotic Interaction

Metabolism: Several bacteria have been genetically engineered to be particularly useful in degrading petroleum. These organisms are able to use petroleum as a food source and break down the oil to more environmentally benign compounds. In some cases, metabolism of a compound may result in a toxic substance. For example, inorganic mercury compounds may be metabolized by microorganisms to methylmercury, which is very toxic.

Bioaccumulation: Many chemicals that are lipophilic (fat-loving) such as polychlorinated biphenyls (PCBs), dioxins, mercury, and cadmium, are readily absorbed and are retained in fatty tissues. This way, these chemicals can enter the food chain and affect organisms that have been directly exposed.

The environmental fate of a stressor depends on several factors:

- **Distribution**: Once in the environment, where does the stressor go? Stressors may be released to or formed from various environmental media. A pollutant released to water may partition to the sediment, remain in the water column, or concentrate in the biota. Different physical forms of a stressor may partition to different media.

- **Transport**: When released or formed, a stressor may be transported from the source. Transport occurs via air, water, soil, or biological carrier. Distribution and transport are closely related, and are frequently modeled to provide an estimation of where a stressor can be found in the environment. The physical, chemical, and biological characteristics of both the stressor and the receiving environment determine the transport and distribution of the stressor in the environment.

- **Degradation or Transformation**: Degradation may occur via biotic processes (metabolism), or abiotic processes (transformation by exposure to light or water). Degradation implies that a stressor is being physically changed into another simpler entity. Transformation may be a gradual or incomplete process (precipitation of a crystal from a complex solution). Degradation products and transformation products can also be toxic, perhaps more so. For example, elemental mercury released into the environment is transformed into methylmercury by microbes in certain aquatic environments. Methylmercury is more toxic than elemental mercury, and it is more readily bioaccumulated.

Identifying the distribution, transport, degradation, or transformation processes to which a stressor is subject provides an indication to what extent the stressor is likely to act upon a potential receptor. It may be possible to show that a stressor is unlikely to affect a receptor given its environmental fate and transport.

The formation and subsequent distribution of secondary stressors may be important depending on the objectives of the assessment. For chemicals, the evaluation of secondary stressors usually focuses on metabolites or degradation products. Physical disturbance of the environment can also lead to secondary stressors. Several methods may be used to understand the distribution and environmental fate of a stressor and characterize the potential exposure of specific receptors to the stressor. Ideally, direct monitoring by collecting and analyzing environmental (including biological) samples is preferred. Monitoring should be designed to define the area over which the stressor may be acting and characterize spatial and temporal variability in the level of stressor (including its degradation products).

Examples of Secondary Stressors

Chemical: Aldicarb is toxic to mammals but not very persistent in the environment. However, it is rapidly degraded to aldicarb sulfone, which is toxic, very persistent, and moves through the soil to the groundwater where it may remain for years.

Physical: Dredging of a waterway not only causes loss of habitat for the organisms at the site of the activity, but may result in severe turbidity of the water and excessive sedimentation down-current.

Where monitoring information is lacking or difficult to obtain, models may be used to estimate exposure to a stressor. Fate and transport models are commonly used to predict the amount that is distributed over a geographic area or the amount of degradation that may be expected over a period of time. These models, preferably based on or verified by actual monitoring data, generally use the physical, chemical, and biological properties of the stressor as well as the environment of concern to characterize the exposure of the stressor to a receptor. This characterization should include spatial extent, intensity, frequency, timing, and location of exposure. Typically, a combination of monitoring and modeling is used to determine the stressor levels.

Describe the Contact or Co-occurrence with the Receptors

The exposure assessment must also include an analysis of how the receptors are exposed to the stressor (i.e., a pathway by which the stressor acts upon the receptor must be identified). In many cases, it is not possible to establish direct causality due to the lack of appropriate information. Therefore, it may be necessary to extrapolate or assume that a pathway exists. However, if the analyst can demonstrate that a pathway from source to receptor is not plausible, then it may be assumed that the receptor will not be affected by the stressor.

Characterizing the ecosystem on which the stressor is expected to have an impact will assist in determining the nature and extent of exposure, and ultimately the adverse effects that may occur. If a chemical affects only hardwood trees, but the surrounding area has only softwood trees, any observed damage to the softwood trees is unlikely to be the result of the chemical.

Ecological components may be characterized in many ways, including: habitat, predator/prey or feeding relationships, reproductive cycles, and cyclic/seasonal activities. An important consideration is at what level of biological organization should an assessment be conducted to yield the most useful information? Selection of the best level of organization for an assessment must take into account many factors, including tools available for economic analysis if an economic analysis is to be conducted. For example, a stressor may cause adverse effects in many species in a community, and those effects may be exacerbated or reduced at higher trophic levels, depending on the nature of the stressor. A classic example of an ecological stressor that is best assessed at the community-level is the bioaccumulation of DDT through the food chain. Chapter 3 provides more information on the strengths and limitations of a benefits assessment at each biological level of organization.

It is also important to know the characteristics of the potential receptors. For example:

- Are they present on a permanent basis (e.g., trees), or are they migratory (e.g., many species of birds)?
- Can and do receptors avoid exposure (i.e., are they capable of detecting the stressor and of movement to avoid it)?
- What are population parameters, such as the size and distribution of the receptors?
- Is the population particularly vulnerable (e.g., nesting or molting) when exposure is most likely to occur?
- What are the most important physical and temporal parameters (e.g., seasonal and diurnal changes in temperature; does the lake freeze in the winter?)?

Exposure can be described in several different ways, depending on how the stressor causes adverse effects:

- **Co-occurrence of the stressor with receptors**. Co-occurrence is particularly useful for evaluating stressors that can cause effects without actually contacting ecological receptors.

- **Contact of a stressor with receptors**. Many stressors must contact receptors to cause an effect. For example, fish must come in contact with the bacterium *Pfiesteria piscicida* before they become sick or die.

- **Uptake of a stressor into a receptor**. Some stressors must not only be contacted, but also internally absorbed. For example a chemical that causes liver tumors in fish must first be absorbed through the gills to reach the liver to cause the effect. Uptake can vary on a situation-specific basis, because it depends on the properties of the stressor (e.g., its chemical form), the properties of the receptor (e.g., its physical characteristics and health), and the location where contact occurs.

When the analyses and supporting documentation have been completed, the exposure assessment should provide a description of the amount of the stressor that is in the environment, how it is able to act on a receptor, and a characterization of the receptor that would or could be affected.

Exposure assessment is one of the more difficult aspects of an ecological risk or benefits assessment, and often introduces the largest uncertainties into the assessment. EPA guidelines and other reference materials on conducting exposure assessment, including the use of probabilistic methods, should be consulted (e.g., U.S. EPA, 1988, 1989a, 1989b, 1992, 1993a, 1997b, 1997c; Suter, 1993).

Effects Characterization

An *ecological effects characterization* describes the relationship between the stressor characteristics (e.g., timing, frequency, magnitude, spatial extent) and the magnitude of the resulting ecological effects. The ecological effects characterization indicates the levels of exposure that elicit different responses (i.e., the stressor-response relationship). Many stressors do not affect all receptors in the same way. In Waquoit Bay, for example, nitrogen loading is a significant stressor. Increased nitrogen levels in the Bay result in excessive algal abundance that has two effects: (1) shading of eelgrass by the algae, which prevents photosynthesis and kills the eelgrass, and (2) decreased oxygen levels in the water that causes physiological stress, suffocation, and increased predation on the finfish. In this case, there are several ecological effects that can be attributed directly or indirectly to nitrogen loading and that are expected to reverse once the nitrogen loading is reduced to more natural levels. The ecological effects characterization involves three steps: determining (e.g., quantifying) the stressor-response relationship(s), evaluating causality, and linking the measure of effects to assessment endpoints.

Determine the Stressor-Response Relationship

Once the receptors and stressors of concern have been defined and plausible exposure scenarios have been identified, the next step is to identify those receptors for which stressor- response information would be most useful for the ecological effects assessment. Stressor-response analysis is often used for chemical stressors such as toxic substances. However, the technique can be applied to many stressors and effects, such as increasing levels of microorganisms and disease, increasing water temperature and enzyme inactivation, or habitat loss and reproductive failure. This type of analysis is particularly valuable, because it describes effects as a function of the level of stress. For example, a slight increase in temperature (stressor) in a given stream may lead to a significant decline in trout abundance (response), but only a minor decline in algal abundance. If the temperature continues to increase, however, algae will also eventually decline in abundance.

Measuring Stressor-Response Relationships

It is difficult to determine whether algae are alive or dead. However, it is relatively easy to measure chlorophyll content both in the laboratory and in the field. Therefore, a change in chlorophyll content is often used to measure algal response to stressors, such as increased temperature, decreased light, or toxic chemicals.

Certain types of pesticides are toxic to birds and animals, because they inhibit the enzyme cholinesterase, which is necessary for proper neurologic function. It is possible to establish a dose-response relationship between the amount of pesticide ingested and the effects of cholinesterase inhibition. Relationships may range from changes in blood cholinesterase levels with no obvious nerve effects to relatively mild tremors to convulsions and death.

Stressor-response analysis often provides a quantitative characterization of the stressor and effect. For an ecological benefits assessment where the expected magnitude of the change in ecological assessment endpoints is needed, full stressor-response curves are needed. In other words, a single point on such a curve (e.g., a 50^{th} percentile or 90^{th} percentile response) is not useful for benefits assessments.

Stressor-response relationships are not always linear (e.g., an increase in the stressor will not necessarily result in an equal increase in receptor response). For some stressors, a threshold may exist below which no response is evident. For example, small increases in water temperature may not adversely affect trout – growth may actually be enhanced – but progressively higher temperatures will impair growth and, if high enough, result in death. Some stressors may have disproportionate ecological effects if the receptors are already subject to another stressor. If deer are starving because of deep snow covering their food, the introduction of wolves may reduce the deer population by greater numbers than expected.

Stressor-response data are needed at the biological level of assessment, for example, at the population level, the relationship between the stressor and a population-level measure such as population density can provide an adequate basis for an assessment of population changes. In some cases, stressor-response profiles are estimated from measures at lower levels of biological organization (e.g., individual level) based on models (e.g., various population models). For some stressors, a quantitative characterization may be difficult to develop. In these cases, a qualitative characterization may be used. See Chapter 3 for more discussion on assessments conducted at different levels of biological organization. Stressor-response information is typically obtained from laboratory or field studies.

Evaluate Causality

Without a sound basis for linking cause and effect, the uncertainty associated with the conclusions of the ecological risk assessment is likely to be high. For example, many seal populations have suffered from epidemics of a distemper-like disease. While several causes (stressors) have been suggested and studied, including pollution-impaired immune systems, warm ocean temperatures, reduced food supply, and pollution-impaired reproductive systems, none have been definitively linked to declining seal populations (U.S. EPA, 1992b). Therefore, while the assessment endpoint can be identified for the receptors (i.e., a change in seal abundance or reproductive success), the potential benefits of removing any single stressor cannot be quantified.

The following criteria may be used for evaluating causality (U.S. EPA, 1998):

<u>Criteria strongly affirming causality</u>:

- Strength of association
- Predictive performance
- Demonstration of a stressor-response relationship
- Consistency of association

Identifying Causes for Declines in Neotropical Migrant Bird Species

Populations of neotropical migrant bird species appear to be in decline in many areas of the United States. These birds, such as the Blackburnian warbler, eat insects and live in the interior of large blocks of forest where they breed. They migrate south in the winter, following their food supply.

The risk hypothesis is that population decline is caused by forest fragmentation in North America and deforestation in tropical South America. Forest fragmentation results in loss of core forest areas and creation of additional forest edge habitat which in which the breeding success of the birds is lower due to predation and parasitism by species adapted to open and edge habitats.

Data (taken from previous studies) were gathered to assess the susceptibility of neotropical migrant species to edge effects, island effects, and the loss of wintering habitat in the tropics. Further monitoring was recommended, including the development of databases to collect additional data on these birds.

Criteria providing a basis for rejecting causality:

- Inconsistency in association
- Temporal incompatibility
- Factual implausibility

Other relevant criteria:

- Specificity of association
- Theoretical and biological plausibility

Link the Measures of the Effects to the Assessment Endpoints

Assessment endpoints express the environmental values of concern for a risk assessment, but cannot always be measured directly. When the measures of effect differ from assessment endpoints, sound and explicit linkages between the two are needed.

The following are examples of extrapolations that risk assessors may use to link measures of effect to assessment endpoints (U.S. EPA, 1998):

- Between similar organisms (e.g., bluegill to rainbow trout);
- Between responses (e.g., mortality to growth or reproduction);
- Between different sources of data (e.g., laboratory to field data);
- Between geographic areas (e.g., northeastern U.S. to northwestern U.S.);
- Between spatial scales (e.g., stream to river); and
- Between temporal scales (e.g., data for short-term effects to longer-term effects).

During the development of the analysis plan in the problem formulation phase (Phase I), the ecological assessment team identified the extrapolations that would be required between assessment endpoints and measures of effect. Decisions about specific extrapolations are usually based on the scope and nature of the risk assessment, resources available for conducting the assessment, and the amount of uncertainty that is acceptable. During the analysis phase, the assessors implement these extrapolations. However, they should reconsider all available data to determine whether the plan should be modified. For example, the exposure characterization may indicate different spatial or temporal scales than originally anticipated. If a stressor persists for an extended time in the environment, it may be necessary to extrapolate short-term responses over a longer exposure period and population-level effects may become more important.

The goal of the analysis phase is to provide sufficient information such that it is possible to characterize the changes in ecological assessment endpoints specified during Problem Formulation.

Characterizing Uncertainty

Uncertainty evaluation is an ongoing issue throughout the analysis phase. The purpose of an uncertainty analysis is to formally recognize that the ecological risk assessment is constructed upon incomplete knowledge and to explain the implications. Specifically, the uncertainty analysis characterizes both the qualitative and quantitative uncertainties associated with the input values and carries those uncertainties through to the estimated exposure and ecological effects.

Any uncertainty analysis need not always be expressed mathematically. Instead, a qualitative description may be used, such as indicating that the animal tested may not be the best surrogate for animals actually exposed to a stressor. This frequently occurs in wildlife toxicity testing where the laboratory animal may be more or less sensitive than other species in the wild.

> **Uncertainty Factors**
>
> Uncertainty factors may be quantitative or qualitative depending on their application. In the development of a conceptual model for the benefits assessment, there may be uncertainty associated with the assumptions used for the model. Examples may be the use of a well characterized species as a surrogate for a species that is less well characterized (e.g., use of coyotes rather than wolves). A pathway may not be clearly defined from the source of the stressor to the receptor. For example, a species of bird exhibit impaired reproduction. The initial risk hypothesis was that loss of habitat from timber cutting was responsible for the impairment. Alternative hypotheses (e.g., the birds are exhibiting reproductive effects as a result of runoff from the timber cutting exposing contaminated soil) also should be considered.

Each of the extrapolations listed in the previous subsection also introduces uncertainty. Other sources of uncertainty in an ecological risk/benefit assessment include, but are not limited to:

- Sampling variability;
- Inability to obtain appropriate samples (this may be of concern if the organism is endangered or difficult to identify or collect);
- Lack of knowledge about combined effects of multiple stressors; and
- Nonlinear behavior of complex systems.

Quantitative measures of uncertainty are often difficult (and sometimes impossible) to provide; when this is the case, the assessors should try to characterize uncertainty in a qualitative manner as completely as possible. This ensures that economists, policy makers, and others who use the results of the ecological risk assessment have a sense of the assessment's strengths and weaknesses.

Methods for analyzing and describing uncertainty associated with an ecological risk assessment range from simple to complex and are beyond the scope of this document. For further reading on the topic of uncertainty analysis, see EPA publications (1992, 1997a, 1997b) and discussions by Suter (1993) and Suter *et al.* (2000).

4.4 PHASE III: RISK/BENEFIT CHARACTERIZATION

The final step in an ecological benefits assessment is estimation and characterization of the changes in the ecological assessment endpoints specified by the conceptual model and the analysis plan. In this step, the characterization of exposure and characterization of ecological effects (Phase III - Analysis) are integrated to provide an indication of the changes in the ecological assessment endpoints and associated uncertainty. For a benefits assessment, single point estimates of risk (e.g., threshold for effect) is not useful; the hazard quotient approach is not applicable. Instead, the risk and benefit estimation should be based on the entire stressor-response relationship and the probability of exposure, and decided based on the process models. In other words, the ecological risk assessment team estimates the likely degree of change in the assessment endpoint from the probability estimates of exposure-probability function and from the stressor-response curves.

Stressor-Response Relationships

Examination of a time series of indicators of the health of the forests in the northeastern United States suggested that gypsy moths might be playing a significant role in the observed decline in forest condition. To estimate the costs and benefits of controlling gypsy moths, information on the relationship between gypsy moth abundance and forest condition was sought. The literature revealed that a small number of gypsy moth larvae may cause minor damage to the foliage on some trees. However, a larger infestation can result in stunted tree growth or even tree death if the larvae eat enough leaves where trees cannot sustain their photosynthetic requirements. The density of gypsy moths can be directly related to tree damage, up to and including death.

Monte Carlo simulations or other probabilistic approaches for incorporating variability and uncertainty can be used to estimate the probability of exposure at various levels. The Team also should attempt to provide uncertainty bounds on that estimate and the likelihood that actual responses would be greater or less than those predicted. This determination can be qualitative or quantitative. For further discussion of probabilistic assessments, see other EPA documents (1997a, 1997b).

Process models are mathematical expressions that represent our understanding of the mechanistic operation of a system under evaluation. A major advantage of using process models for a benefits assessment is the ability to consider “what if” scenarios, and to forecast beyond the limits of the observed data that constrain risk estimation techniques based on empirical data. For example, process models may be used to extrapolate from individual-level to population- and ecosystem-level effects. These models may also be of use in estimating indirect effects on the assessment endpoints and the probable rate of recovery. A variety of process models are available for both terrestrial and aquatic ecosystems (e.g., RAMAS, Aquatox). Because process models are only as good as their assumptions, they should be treated as hypothetical representations of reality until appropriately tested with empirical data.

After the magnitude and likelihood of changes in the ecological assessment endpoints have been estimated, the results are ready to supply to the economic analysts. The ecological assessment team should characterize both the beneficial and adverse effects that could accrue from an action so that the economists can estimate "net" benefits. The ecological assessment team needs to provide the economic analysts with full descriptions of the potential range in natural variability that might be expected in the assessment endpoint and the uncertainties associated with their estimates of changes to the assessment endpoints. Where the analysis plan called for quantitative assessments of variability and uncertainty for specific assessment endpoints, the ecological assessment team should provide quantitative estimates to the extent possible. For all other assessment endpoints, the ecologists and economists agreed to pursue only a qualitative assessment of variability and uncertainty.

There are several other parameters that must be addressed to characterize changes in the ecological assessment endpoints with sufficient specificity to be used in the economic assessment. The magnitude of effect needs to be specified in terms of geographic coverage, the degree of change per unit area, the probability of changes of that degree or higher (or lower), and the time-frame over which the change would be expected to occur. Again, quantitative or qualitative error bounds should be provided for these parameter estimates.

This document has focused on the economic assessment of ecological benefits. The ecological assessment team might have included assessment endpoints that were not going to serve as inputs to the economic analysis. The team would prepare and present these results to the audience for which they were intended.

The next chapter discusses how economists define the value of changes in an ecological assessment endpoint, and subsequent chapters discuss the steps associated with economic benefits analysis and the specific methods for the economic valuation of ecological benefits.

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5.0 BACKGROUND THEORY ON VALUING CHANGES TO ECOLOGICAL RESOURCES

This chapter discusses how economists define the value of an ecological resource and then estimate the value of a change to such a resource. This chapter provides a basic introduction to welfare economics as applied to the valuation of environmental changes -- it is not, however, a comprehensive review of welfare economics. References for further reading on the subjects discussed below are provided at the chapter's end.

Subsequent chapters will discuss the steps associated with economic benefits analysis and the specific methods for estimating benefits, applying the general concepts introduced in this chapter. Readers should be aware that subsequent discussions presume an understanding of basic economic concepts (e.g., supply, demand).

5.1 WELFARE ECONOMICS AND THE VALUE OF AN ECOLOGICAL CHANGE

Welfare economics provides the theoretical basis for estimating the economic benefits of an action. Welfare economists assess the value of an action based on its effect on the well-being, or level of welfare, of humans. Economic value is based on what people want -- that is, their preferences -- and is measured by examining how people trade-off different goods and services. Economic theory is based on the assumption that the decisions people make regarding what they choose to consume and what they choose to do for activities reflect the values they hold for the various goods and services available.

The anthropocentric perspective of welfare economics implies that the economic value of an ecological resource depends on the value humans derive from the resource. Some people take affront to this basic premise of welfare economics, arguing that decisions should be based on the broader values of community, the impacts to future generations, or the inherent rights of natural resources (see for example, Sagoff, 1988). Nonetheless, welfare economic analysis provides useful information for making decisions and presently serves as the basis for most economic/policy analyses. For this reason, this document focuses on describing the basic concepts and techniques of welfare economists. Decisionmakers must, however, keep in mind that a welfare analysis is just one approach among many for evaluating a change and should consider other perspectives and information when making policy choices.

The type, quantity, and quality of goods and services available to an individual determine the individual's level of well-being, or level of welfare. Some goods and services are produced by industry and purchased by individuals in markets, some are produced within the household, some are provided by government, and some are provided by nature or ecological resources. (See Chapter 6 for a discussion of the different types of goods and services provided by ecological resources).

The condition of an ecological resource determines the type, quantity, and quality of goods and services provided by that resource. As a result, any action that affects an ecological resource, such as an environmental regulation or natural resource management activities, will likely also affect the goods and services the resource provides, and subsequently the level of welfare of the individuals who enjoy those goods and services. Typically, economists estimate the welfare change associated with a policy or action using an "effect-by-effect" approach (U.S. EPA, 2000). Under this approach, economists measure the change in welfare associated with the change to each good and service provided by the ecological resource and sum these individual measures to estimate total benefits. The next section describes how economists measure changes in welfare.

5.2 MEASURING THE BENEFITS OF IMPROVEMENTS TO ECOLOGICAL RESOURCES – THE CONCEPT OF WILLINGNESS-TO-PAY

The economic value of a good is determined by the maximum amount of something else (usually money) that an individual is willing to pay to obtain the good. This measure of economic value is called "willingness-to-pay" (WTP). For an environmental improvement, WTP is the amount an individual is willing to pay to obtain the improvement. An alternative measure, "willingness-to-accept" (WTA) is defined as the minimum amount of money an individual is willing to receive in compensation to forgo a benefit, such as an environmental improvement, they would otherwise receive.

The choice between using WTP or WTA to value changes in environmental quality implies different assumptions regarding the property rights of individuals experiencing the change. Using WTP implies that polluting entities have a right to pollute, so the public must pay them not to pollute. Using WTA implies that the public has a right to a clean environment and must be compensated for pollution. There also can be significant differences in the estimated value of a change measured in terms of WTP or WTA. One reason for this difference is that for an environmental improvement, WTP is based on an individual's level of welfare without the improvement, while WTA is based on the level of welfare achieved with the improvement. (See Hanley, Shogren, and White, 1997 for further discussion of this issue).

Additionally, measuring economic value in terms of WTP also does not allow for the possibility that certain goods may be "incommensurable" for some individuals, because their WTP is constrained by their income level. This constraint of welfare economics imposes an ethical assumption that people will always be willing to substitute other goods for ecological resources. Although WTA is typically the theoretically correct measure for estimating the benefits of environmental improvements, WTP is more commonly used in practice because it is easier to measure and estimate (U.S. EPA, 2000).

For consistency with how goods and services are traded through markets and comparability with the estimated dollar costs of an action, economists measure the benefits of an action, such as a regulation, in dollar terms using WTP. WTP values reflect individual's preferences for exchanging goods and services. Because preferences are likely to vary from one individual to another, WTP values for a change to a particular good or service will vary from one individual to another. The total social value of an improvement in a good or service is the sum of the WTP across all individuals.

Although economists are most often asked to value the change in social welfare (measured by WTP) associated with a change in a particular good or service provided by an ecological resource, they will also sometimes be asked to value the availability or existence of the ecological resource itself. For example, an economist might be asked to determine the change in social welfare associated with the complete loss of a wetland and all the goods and services it provides. In this circumstance, the economist will likely have to value the individual goods and services lost separately and sum these benefit estimates. Alternative approaches that attempt to estimate the total value of such a resource based on replacement cost or embodied energy (e.g., Costanza *et al.,* 1997; Ehrlich and Ehrlich, 1997; Pearce, 1998; Pimentel *et al.*, 1997) have been discussed recently, but are not appropriate for an economic benefit analysis because the methods are not grounded in economic theory (U.S. EPA, 2000).

5.3 How Economic Benefits of Improvements to Ecological Resources are Realized

The economic benefits of an action that affects an ecological resource depends on how the state of the ecological resource influences the supply or consumption of the goods and services provided or supported by that resource. In evaluating the economic benefits of an action affecting an ecological resource, economists consider two possible relationships between the resource and the goods and services enjoyed by society:

- The ecological resource is an input to the production of a good or service, such that the state of the ecological resource directly affects the production (or supply) of the good or service; or

- The state of the ecological resource is a characteristic of the good or service, such that the state of the ecological resource directly or indirectly affects the demand for (or value of) the good or service.

Ecological Resource as Input to Production

When an ecological resource serves as an input in the production of another good or service, changes affecting the quality or state of the ecological resource can have direct impacts on the production or supply of the good or service. "Production" might consist of a natural or bioeconomic process, or a man-made or industrial process. For example, a change in the ozone concentration in the air will affect the growth rate of plants and, thus, the productivity of agricultural crops. Alternatively, an improvement in the water quality of a river that provides water used for paper production may reduce the processing costs and, thus, increase the productivity of the paper mill. That same change in the water quality of a river might also affect the non-market recreational opportunities provided, or "produced," by the river. The effect of the change in the productive process may be realized through changes in the flow of a non-market good or service, a change in the price of a marketed good or service, or a change in the wage rates or earnings of workers in the affected sector.

The State of the Ecological Resource as Characteristic of a Good or Service

When the state of an ecological resource is a characteristic of a good or service, a change in the state of the ecological resource affects the demand for that good or service. For example, the demand for recreational fishing days at a particular lake is likely to change if the water quality of the lake is improved. Similarly, the demand for hiking days or scenic views may increase as a result of an improvement in air quality that increases visibility. The change in demand may be realized through increased number of visitation days, increased number of users, or increased spending to make use of a good or service provided by the ecological resource.

5.4 ESTIMATING WILLINGNESS-TO-PAY

There are several techniques used to estimate WTP for changes to goods and services provided by ecological resources. The technique employed depends on the type of good or service affected. The techniques used by economists may:

- Estimate demand and supply curves for the good or service in question;
- Estimate demand and supply curves for a related good or service; or
- Estimate WTP based on other observations.

Market Goods and Services

The change in social welfare for a given change in the supply or price of a good and service that is sold through a market is often approximated by the sum of predicted change in consumer and producer surplus. Consumer and producer surplus is represented as the area above the supply curve and below the demand curve. These surplus measures are standard and widely accepted terms of applied welfare economics. Consumer and producer surplus is derived from market data on how much of the good is demanded and produced in the aggregate at various price levels and can be easier to estimate than individual WTP.

Although surplus measures do not, in general, provide a theoretically correct estimate of the change in social welfare, they can provide a reasonably accurate estimate of social WTP for relatively small price changes (Willig, 1976). The estimate is less reliable, however, for changes in the quality or quantity of goods and services. Nonetheless, measures of changes in consumer and producer surplus are often used as indicators of the economic magnitude of impacts when more precise measures are not feasible or practical.

Non-Market Goods and Services

The lack of markets and prices for many of the goods and services provided or supported by ecological resources often makes it more difficult to estimate WTP for changes to these goods and services. For goods and services that are not traded through markets, economists measure changes in economic welfare, or WTP, based on changes in human behavior and the decisions people make under different circumstances. Economic techniques for non-market goods and services estimate individual WTP using either market information for related goods and services (revealed preference methods) or direct statements of people's preferences (stated preference methods). Individual WTP estimates are generally averaged and multiplied by the total number of affected individuals.

Chapter 6 provides more information on specific methods for estimating WTP for market and non-market goods and services.

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6.0 ECONOMIC ASSESSMENT OF ECOLOGICAL BENEFITS

The first section of this chapter discusses the various components or steps of an economic assessment of ecological benefits ("economic benefit analysis"). The steps of an economic benefit analysis described here are based on EPA's (2000) *Guidelines for Preparing Economic Analyses*. Later sections of this chapter provide additional information on the types of economic benefit endpoints that might be considered by an economic benefit analysis and the different approaches available for estimating the economic value of those benefits.

6.1 COMPONENTS OF AN ECONOMIC ASSESSMENT OF ECOLOGICAL BENEFITS

As discussed in Chapter 2, there is an extensive planning phase that precedes the ecological risk assessment and the economic benefit analysis. The decisions made during that planning process regarding the nature of the decision under study and other criteria guide the economic analysis. Of particular importance in designing and conducting the economic benefit analysis is a good understanding of the type of information needed by decisionmakers and the nature of the information that will be made available from the ecological risk assessment.

Following the planning phase, the economic benefit analysis begins. The economic benefit analysis generally follows an "effect-by-effect" approach (U.S. EPA, 2000). As noted in Chapter 2, under this approach economists estimate the benefits associated with the major effects of a policy or action separately and then sum together the value estimates for the individual effects to arrive at an estimate of the total benefits.

An economic benefit analysis can be broken down into four steps:

- Identify and prioritize economic benefit endpoints
- Describe and quantify effects of the policy or action on the economic benefit endpoints
- Estimate the value of those effects
- Summarize and present the results

The key elements of each of these steps are discussed below. Chapter 2 discusses in detail the opportunities for improving the economic benefit analysis by coordinating with the ecological risk assessment team in each of these steps. Chapter 2 also discusses the variety of issues that must be coordinated with the ecological risk assessment in designing an economic benefit analysis: establishing the baseline from which changes are measured, measuring changes to the economic endpoints based on the changes to the ecological endpoints, determining the appropriate spatial and temporal scale for the analysis, and determining how uncertainty will be treated. These discussions are not repeated here.

6.1.1 Identify and Prioritize Economic Benefit Endpoints

The process of identifying and prioritizing the economic benefit endpoints affected by a policy or action is described only briefly here. A more detailed discussion of this step is provided in Chapter 2.

The identification and prioritization of relevant economic benefit endpoints involves:

- Developing a list of preliminary economic endpoints;
- Linking changes to ecological resources to changes in the economic benefit endpoints;
- Prioritizing the economic benefit endpoints for consideration by the benefit analysis; and
- Determining how the economic analysis will assess the changes for high priority economic benefit endpoints.

Economists identify potential economic benefit endpoints by understanding the policy or action under study, reviewing analyses of similar policies or actions, and working with the ecological risk assessment team to understand what ecological changes are expected.Linking ecological changes to changes to specific economic benefit endpoints involves extending the conceptual model developed by the ecological risk assessment team to include the economic benefit endpoints that are expected to be affected. As noted in Chapter 2, the ecological risk assessment team can provide valuable assistance to the economists in determining how various economic benefit endpoints might be affected.

Time and resource constraints generally require that the economic benefit analysis focus on fully assessing and valuing changes to a limited number of endpoints. Each endpoint's consideration priority is based on decisionmakers' needs, the expected magnitude of the predicted change to that endpoint, the uncertainty associated with the predicted change and anticipated value of the change, and the variability of the change to each benefit endpoint under different policy options. This prioritization method ensures that important changes that typically are not amenable to a monetary assessment are given equal attention by the benefit analysis. For the high priority economic benefit endpoints, the economic analyst must determine how the changes to each endpoint will be analyzed -- with a qualitative, quantitative, or monetized assessment of the change. The choice of assessment method depends upon the relative need for a dollar value estimate of benefits, the availability of the necessary data and appropriate quantification and/or economic valuation techniques, and the time and resource constraints of the economic benefit analysis. (See Chapter 2 for an extensive discussion of these steps.)

6.1.2 Describe and Quantify Changes to the Economic Benefit Endpoints

Using results from the ecological risk assessment and information from other data sources, the economist describes the changes to the ecological services and values affected (i.e., the economic benefit endpoints) and provides information on the magnitude of the changes. The ecological risk assessment is responsible for estimating the likely changes to the ecological resources. If care has been taken to coordinate during problem formulation (see discussion in Chapter 2), the information provided by the risk assessment will be compatible with the needs of the economic benefit analysis.

In addition to the information from the risk assessment, economists may collect additional economic data to describe and estimate the effects and economic value of changes to the economic benefit endpoints. For example, in describing the impact of a change to an economic benefit endpoint, the economist might consider the estimated number of users of the goods and services provided by the resource (e.g., number of fishermen, number of visitors, local population, national population), the quantity of the good or service provided or used (e.g., timber production, commercial fish landings), or some measure of the magnitude of the ecological resource itself (e.g., acres, productivity).

A thorough qualitative, and when possible quantitative, discussion of the changes to the economic benefit endpoints is an essential component of the benefit analysis. For endpoints for which a monetized assessment is not possible, the qualitative or quantitative assessment provides a measure of the good or service's importance and the degree of change experienced under the policy or action. For those endpoints for which a monetized assessment is conducted, the qualitative and quantitative discussion of the change that is valued supports the dollar value generated by the benefit analysis. Some ecological improvements may also result in economic losses. The net of positive and negative economic changes must be calculated in determining the benefits of any action.

6.1.3 Estimate the Value of the Changes

There are a wide variety of techniques available to estimate the value or change in value of specific attributes or services provided by ecological resources. In this step, the economist selects the approach that is most appropriate given the attribute or service being analyzed, the data available regarding the production or demand for the attribute or service, and the time and resource constraints of the study. The most common approach used by EPA analyses is benefit transfer, in which value estimates from one study are applied to another situation.

Regardless of the technique used to estimate the value of changes to the economic endpoints, the benefit analysis must describe the source of the value estimate and the degree of confidence in the estimate. This is particularly important when using benefits transfer because transferring a value estimate to a new situation can only increase the uncertainty associated with the estimate.

Several different techniques may be used to estimate the benefits associated with changes to multiple endpoints. Although using multiple methods may provide more information on the value of changes experienced by the economic endpoints, care must be taken to avoid double-counting of benefits. A careful understanding of the relationships among the various endpoints considered is important for identifying potentially overlapping benefits that could lead to double-counting.

Many economic valuation techniques estimate the value or benefits associated with a change for an individual person (i.e., individual willingness-to-pay (WTP)). To develop an aggregate estimate of the social benefits of an action, economists sum individual WTP for the action across all affected individuals. For many actions that affect ecological resources, some individuals will benefit while other individuals may experience a decline in their individual welfare. For example, while removing a dam might improve opportunities for kayaking and other whitewater activities, it might also result in the loss of a boating area behind the dam for water skiing and fishing. In this type of situation, the benefit of the action is the net total of all gains minus all losses in individual welfare experienced by members of society.

In aggregating benefits estimates across all affected individuals, each individual's WTP is given the same weight in the summation. As discussed in EPA's *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2000), an equity assessment and impact analysis may be conducted to assess the impact of an action on any populations of concern.

Typically, the economic benefit analysis will examine changes that occur over an extended period of time (i.e., longer than a single year). The economic benefit analysis, therefore, may need to describe when these changes occur over the time period considered by the analysis as well as summarize the value of the change over the whole time period. The standard approach for summarizing the value of changes that occur over an extended time period is to sum the value of all changes over the time period using discounting. Discounting is commonly used to express future costs or benefits in present monetary value. The use of discounting and the choice of an appropriate discount rate are complex and highly debated issues that are beyond the scope of this document. Of particular concern is the potential effect of discounting ecological benefits on resource conservation and intergenerational equity issues. References for additional information on discounting are provided in Chapter 7.

6.1.4 Summarize and Present the Results

As discussed in Chapter 2, the results of the economic benefit analysis will include the prioritized list of economic effects, discuss the criteria used to select the economic endpoints examined in detail by the benefit analysis, and discuss how the economic value of the effects was assessed. The monetary benefits estimated for some of the changes will be accompanied by the qualitative assessment of other benefits that were not monetized. If possible, the qualitative assessment should discuss the potential magnitude of the economic benefits for any priority endpoints that are not accounted for by the quantitative and monetized assessment. The final report should also discuss to some degree the other effects identified that were deemed less important to the economic analysis.

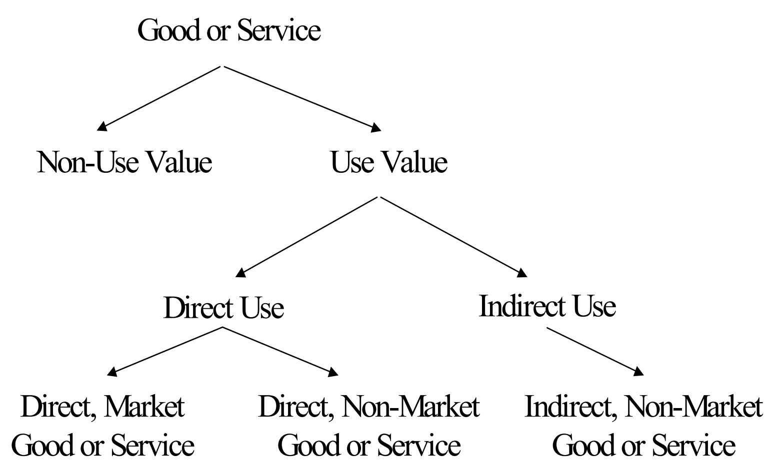
Finally, the results of the economic analysis must disclose any source of error in the analysis and the potential impact of such error on the results. The presentation of results should identify any possibility of double-counting of benefits, any limitations of the analysis, and any potential imprecision and uncertainty associated with the benefit estimates. The analyst should note uncertainties in the ecological assessment that are relevant to the economic assessment and discuss how these uncertainties may be compounded. In discussing the potential impact of any source of imprecision or uncertainty, economists should discuss whether the analysis is likely to over- or under-estimate the economic value of benefits.

The results presented also may include information and details that are needed for other analyses. For example, an equity analysis may require information on the geographic distribution of effects, the distribution of ecological effects and economic benefits across different ethnic or economic classes of the human population, or the distribution of ecological effects and economic benefits over time.

6.2 IDENTIFYING THE SERVICE FLOWS AND OTHER VALUES PROVIDED BY AN ECOLOGICAL RESOURCE

There are numerous types of goods and services provided by ecological resources that have economic value to some or all individuals in society (see ***Chapter 5*** for a discussion on defining the economic value of ecological resources). This section discusses the various types of goods and services and offers their taxonomy, which may be useful in developing a comprehensive list of specific economic benefit endpoints. The proposed taxonomy for generally characterizing the goods and services provided by ecological resources is presented in Exhibit 16.

Exhibit 16
Proposed Taxonomy of Goods and Services Provided by Ecological Resources



Some of goods and services provided by ecological resources are obvious because they are directly used or enjoyed by society, such as the fish provided by a fishery, the timber/lumber provided by a forest, or the swimming and boating opportunities provided by a coastal area. These types of goods and services are defined as direct, market uses, when the good or service is bought and sold through open markets, and direct, non-market uses, when the good or service is not bought and sold through a market.

The direct, market uses of an ecological resource are typically the most obvious and most easily valued goods provided by an ecological resource because price and quantity information for each good and service is generally available. The direct, non-market uses of an ecological resource may be readily apparent, such as recreational opportunities, although more difficult to value. Valuation of changes to direct, non-market uses is more difficult because the goods or services are not sold through markets, making it more difficult to obtain information on the "price" of the service and the number of people enjoying the service (i.e., how many people benefit from the resource through a specific use).

Ecological resources will also provide some services and ecological processes that indirectly benefit society. For example, a coastal wetland provides services as a wildlife habitat and fish nursery, as a means for flood control, and as a filtering system for run-off waters. These types of services, which are not bought and sold through markets, are referred to as indirect, non-market uses. Individuals may value these services even though they are not directly using the resource. Sometimes these types of services can be connected to other activities that humans value and, therefore, are valued through that relationship (see the discussion on identifying economic benefit endpoints in Chapter 2).

Economists also recognize several different categories of non-use values. As the term implies, non-use values represent the value that an individual places on the ecological resource that does not depend on the individual's current use of the resource. Existence value, for example, refers to the value people place on knowing that a particular resource exists, even if they have no expectation of using the resource. Other examples of non-use values include bequest value, which refers to the value people place on a maintaining a resource for future generations, and altruism, or the value people place on maintaining resources that are important to their family and friends.

As described in Chapter 5, the benefits of an action that improves a specific ecological resource can be estimated by estimating people's willingness-to-pay (WTP) for improvements to the various types of goods and services provided by the resource. For example, in estimating the benefits of an action to improve the quality of a wetland area, one might consider that the wetland area serves as a primary breeding area for several species of birds. Therefore, one might estimate the change in the value of bird watching and recreational fowl hunting to the individuals using the area. To capture the total value or benefits of a change to a specific ecological resource, one also needs to consider the value of its role in supporting the ecosystem and the indirect benefits it provides to mankind. That is, one needs to also identify and evaluate the indirect, non-market uses and non-use values associated with an ecological resource.

The economic benefit analysis should identify as many different goods and services (and values) affected by the policy or action. For example, if a policy is expected to improve ecological resources that support various bird populations, the economic benefit analysis might consider potential impacts on the following goods and services society derives from birds:

- Food source (direct, market use);
- Hunting, bird watching, and contributing to the aesthetic environment for hikers, campers, anglers, and other recreationists (direct, non-market use);
- Component to an ecosystem that supports or provides other goods and services and contribute to maintaining biodiversity (indirect, non-market use); and
- As an endangered species or to maintain the bird species for future generations (non-use value).

The following four subsections elaborate on the types of goods and services that might be provided by an ecological resource and identify the economic techniques that might be appropriate for estimating the economic value of changes to these goods and services.

6.2.1 Direct, Market Uses

Direct, market uses refer to those goods and services provided by an ecological resource that are directly used by society and are bought and sold through the market system. Direct, market uses primarily refer to those goods produced by an ecological resource that are consumed by humans or serve as inputs in the production of other goods, such as food products, water, fuel sources, and building materials. Prices and quantities produced for these goods and services are directly observable.

For example, one benefit of a policy to improve air quality might be measured through the value (i.e., change in welfare) of the increased productivity of commercial crops and timber production. Similarly, the benefit of an action to improve water quality might be measured through the value of the increased production of a commercial fishery (i.e., more fish caught and sold).

It is important to remember, however, that the change in value of the direct, market uses (e.g., timber, crops, or fish) provided by an ecological resource (e.g., air, water) may represent only a portion of the total benefits of the change experienced by the ecological resource.

Examples of Direct, Market Uses Provided by Ecological Resources

❑ Food Source
  - Fish (specific species) -- commercial fishery
  - Crops (specific type: corn, beans, apples, etc.) -- commercial and home production
  - Animal (fowl, deer, etc.) -- commercial consumption

❑ Building Materials
  - Timber (specific species)
  - Stone

- ❑ Fuel
  - Timber (specific species)
  - Coal
  - Oil
- ❑ Drinking Water Supply
  - Ground water reservoir
  - Surface water reservoir
- ❑ Medicine
- ❑ Chemicals/Minerals

Valuing Direct Market Uses

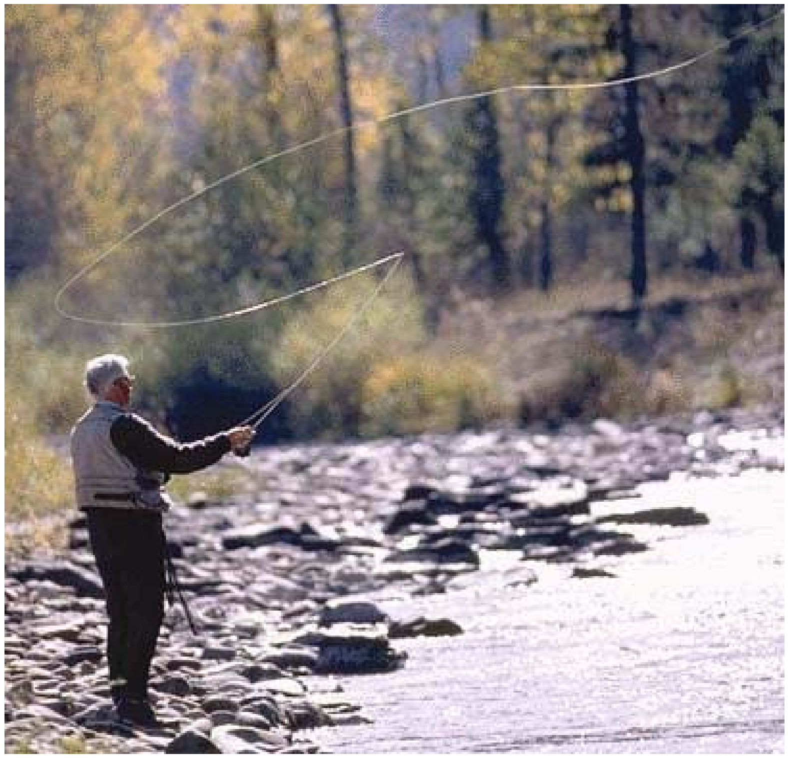
There are a number of market-based approaches that may be useful in estimating the value of changes to a direct market use provided by an ecological resource. In most cases, a market-based approach is used to estimate the demand and supply functions for the good or service. For some market goods, such as commodity crops and timber, detailed general and partial equilibrium models have been developed, which estimate demand and/or supply responses to changes in productivity, prices, and other variables. Impacts or changes to the ecological resource that affect the quantity or quality of the goods and services provided by the resource can be measured by estimating the change in the demand and supply functions resulting from the change and measuring the welfare change or change in willingness-to-pay.

For relatively small changes that do not change the supply or demand for the good or service provided by the resource, the change in the value of the goods and services provided by the resource can be measured based on the increase (or decrease) in the quantity of the good or service provided and the market price of the good or service (see later section on estimating benefits using market price and supply/demand relationships for additional discussion of this issue). Other market-based valuation approaches, such as examining the cost of alternatives or the spending to provide similar goods or services, may also be useful when price or quantity information is not readily available. Although these second-best approaches can provide an estimate of the magnitude of the potential benefits, they do not directly reflect welfare changes.

Specific techniques that can be used to value changes in market-based goods include:

- Market-Price and Supply/Demand Relationships
- Market-Based Valuation Approaches.

6.2.2 Direct Non-Market Uses



Direct, non-market uses of an ecological resource include those goods and services that are directly observed and used by humans, but are not sold or traded through an open, competitive market. Direct, non-market uses include both consumptive uses (e.g., recreational fishing and hunting) as well as non-consumptive uses (e.g., bird watching or boating). Direct, non-market uses are generally considered quasi-public/quasi-private goods because access or use of the resource can be controlled but is often not strictly regulated and the benefit or value to one individual does not affect the benefit or value to others up to a point (i.e., congestion reduces the benefit/value to all users).

Examples of Direct, Non-Market Uses Provided by Ecological Resources

- ❑ Fishing
  - Recreational Fishing (specific species, area)
  - Subsistence Fishing (specific species, area)
- ❑ Beach Use (sunbathing, swimming, walking)
- ❑ Recreational Hunting (specific species) - for sport and/or personal consumption
- ❑ Bird Watching (general, specific species)
- ❑ Tourism
- ❑ Boating
- ❑ Hiking/Camping
- ❑ Animal Viewing, Photography, Feeding (general, specific species)
- ❑ Sightseeing
- ❑ Aesthetic Value

Valuing Direct, Non-Market Uses

These types of services are not bought and sold through observable markets and therefore, do not have market prices associated with their use. For most of these types of goods and services, however, the change in the quantity and/or quality of the service being provided is quantifiable (e.g., increased number of fish caught per fishing trip, increased number of beach or boating days, increased chance of viewing wildlife). Because these types of goods and services do not have market prices, non-market valuation techniques must be used to estimate the implicit prices for the goods and services provided by the resource. Some methods rely on the explicit transactions (e.g., entrance or licensing fees, spending to protect a resource) or observed choices that people make (e.g., travel decisions, home location) that are associated with the use of the goods and services provided by the ecological resource. These methods assume that people demonstrate, or reveal, the value they place on a good or service through the choices they make. Other methods rely on the responses of individuals using the resource to proposed choices or questions regarding the value they place on their use of the resource. In some cases, more sophisticated techniques and models, which combine information on engineering and biophysical processes with economic information, are used to estimate ecosystem changes and impacts to specific uses or services.

Specific methods that may be useful in valuing changes to direct, non-market uses include:

Revealed Preference Methods:

- Hedonic Price Methodologies
- Travel Cost Methodologies
- Random Utility Models

Stated Preference Methods:

- Contingent Valuation
- Contingent Activity and Combining Contingent Valuation with Other Approaches
- Conjoint Analysis and Contingent Ranking.

6.2.3 Indirect, Non-Market Uses

Indirect, non-market uses of an ecological resource include those goods and services that provide an observable benefit to mankind but are not directly consumed or used by individuals. Indirect, non-market uses include many ecological processes that indirectly benefit mankind by supporting other ecological resources, maintaining viable ecosystems, and protecting the local environment. Indirect, non-market goods and services are usually public in nature because access or use of the ecological resource cannot generally be excluded and any number of individuals can benefit from the use of the ecological resource through these services without reducing the benefits accruing to anyone else. These goods and services are not sold or traded through an open, competitive market, although a community may pay for replacement or substitute goods (often through taxes) that provide the same public services as provided by the ecological resource.

Examples of Indirect, Non-Market Uses Provided by Ecological Resources

- ❑ Flood Control
- ❑ Storm Water Treatment
- ❑ Ground Water Recharge
- ❑ Climate Control
- ❑ Pollution Mitigation
- ❑ Wave Buffering
- ❑ Soil Generation
- ❑ Nutrient Cycling
- ❑ Habitat Value
- ❑ Biodiversity

Valuing Indirect, Non-Market Uses

These types of services are not bought and sold through observable markets and therefore, do not have market prices associated with their use. Because these types of goods and services do not have market prices, non-market valuation techniques must be used to estimate the implicit prices for the goods and services provided by the resource. Some methods rely on the observed choices that people make that are related to the indirect, non-market goods and services provided by the resource. These methods assume that people demonstrate, or reveal, the value they place on the goods and services provided by ecological resources through the choices they make. In some cases, expenditures for replacement or substitute goods that provide the same public services as the ecological resource may be used to estimate the minimum value of the indirect, non-market services supported by the ecological resource. Other methods rely on the responses of individuals to proposed choices or questions regarding the value they place on the goods and services provided by the resource.

Specific techniques that may be useful in estimating the value of changes to indirect, non-market uses include:

Revealed Preference Methods:

- Hedonic Price Methodologies
- Replacement/Alternative Cost
- Avoidance Expenditures

Stated Preference Methods:

- Contingent Valuation
- Contingent Activity and Combining Contingent Valuation with Other Approaches
- Conjoint Analysis and Contingent Ranking.

6.2.4 Non-Market, Non-Use Values

Non-market, non-use values of an ecological resource are the values that individuals hold for the resource unrelated to their current use of the goods and services provided by the resource. Individuals may value the existence of the ecological resource or the availability of the goods and services provided by the ecological resource although they do not directly consume or use the resource themselves. Non-market, non-use values may stem from the desire to ensure the availability of the resource for future generations, benevolence toward relatives and friends, sympathy for people and animals adversely affected by environmental degradation, or a sense of environmental responsibility. Additionally, the specific non-use values associated with a particular ecological resource may not be mutually exclusive: when asked directly, people are unlikely to be able to separately identify the non-use values they hold or distinguish between the value they place on direct or indirect uses and their non-use value(s).

Examples of Non-Market Non-Use Values Provided by Ecological Resources

- ❑ Scarcity Value
- ❑ Option Value (although some consider this a use value)
- ❑ Existence Value
- ❑ Cultural/Historical Value
- ❑ Intrinsic Value
- ❑ Bequest Value
- ❑ Altruistic Value
- ❑ Philanthropic Value

Valuing Non-Market, Non-Use Values

These types of services are not bought and sold through observable markets and, therefore, do not have market prices associated with their use. Because these types of goods and services do not have market prices, non-market valuation techniques must be used to estimate the implicit prices for the goods and services provided by the resource. Furthermore, by definition, the non-use value associated with an ecological resource cannot be estimated based on observed actions or choices made by individuals. Thus, to estimate non-use values economists must rely on people's responses to proposed choices or questions regarding the value they place on certain ecological resources (known as contingent valuation). Determining the total non-market, non-use value associated with a change to an ecological resource is often difficult because the total value depends not only on the value each individual holds, but also on the appropriate number of such individuals to count in the valuation process. Additionally, as discussed in the later technique sections, the use of contingent valuation is very controversial and continues to be refined by economists, sociologists and psychologists.

The following techniques are applicable for estimating non-market, non-use values:

- Contingent Valuation
- Contingent Ranking
- Conjoint Analysis.

6.3 APPROACHES TO MEASURING RESOURCE VALUES

This section introduces the reader to the different types of approaches available to estimate the economic value (i.e., change in social welfare or willingness-to-pay) of a change in the quality and/or quantity of the goods and services provided by an ecological resource. Each valuation method has a different approach to eliciting the value that society places on such changes. Most, if not all, techniques require sophisticated econometric analysis to employ.

This section organizes and explains the general types of valuation techniques and discusses, generally, what data might be required to implement each type of approach. A framework for understanding the similarities and differences between the techniques is presented, followed by a brief description of each category of techniques.

More detailed descriptions of the individual techniques are provided in later sections. The information provided on the individual techniques is based on findings from the literature; the reader is encouraged to independently evaluate any technique for their own use. An additional reference document, EPA's *Guidelines for Preparing Economic Analyses* (U.S. EPA, 2000), also reviews the various techniques available for benefits valuation.

Valuation Techniques

Valuation techniques can be grouped into four general categories according to the means by which preferences are revealed and the process by which these preferences are translated into monetary values (Mitchell and Carson, 1989; Freeman, 1993). To determine into which category a method falls, it is necessary to ask the following two questions:

1. Does the technique use data or observations from people acting in real-world situations (i.e., revealed preferences) or from people responding to hypothetical situations (i.e., stated preferences)?

2. Does the technique yield monetary values directly (i.e., direct estimation of willingness-to-pay) or must monetary values be inferred based on a model of individual behavior (i.e., indirect estimation of willingness-to-pay)?

Exhibit 17 illustrates the matrix and the corresponding organization of the valuation techniques available for developing original valuation estimates (Mitchell and Carson, 1989; Freeman, 1993). Benefits transfer analysis, which is not listed in the following table, relies on the results of previous analyses to develop a valuation estimate for a new policy case or study site. Following the table is a discussion of the four categories of approaches and benefits transfer analysis.

Exhibit 17
Categorization of Valuation Techniques

	Direct Estimation of WTP	Indirect Estimation of WTP
Revealed Preferences Approach	Market Price/Quantity (Estimated Supply/Demand) Market Simulation Models User Fees Replacement Costs	Travel Cost Studies Random Utility Model Hedonic Studies Avoidance Expenditures Referendum Voting
Stated Preferences Approach	Contingent Valuation Studies	Contingent Ranking Contingent Activity Contingent Referendum Conjoint Analysis

Note: Benefits Transfer Analysis relies on estimates developed using one or more of the techniques listed in this table.

Direct, Revealed Preference Approaches

Direct, revealed preference approaches require data on real-life choices made by individuals regarding their consumption or use of a particular good or service. These approaches assume that an individual who is free to choose the quantity of good or service they desire at a specific price will choose the quantity that maximizes their welfare (or benefits), given the constraints placed upon them by the market (e.g., limited individual income, availability of substitutes and other goods, limited availability of specific goods or services). Thus, these types of approaches can only be applied for goods and services bought and sold through markets. Competitive market prices and production cost information, for example, can be used to estimate supply and demand relationships, that can then be used to estimate the consumer and producer surplus associated with the goods or services provided by a resource. Alternatively, more complex market simulation models might be used to mimic market conditions in an effort to determine the value (or change in value) placed on a good or service. Estimating market relationships for a good or service requires, at a minimum, time series or cross-sectional data on the price of the good or service, the quantity sold and consumed, detailed cost and revenue information for representative producers, as well as data on the environmental change affecting the supply and/or demand for the marketed good or service.

In some circumstances, market data may be useful in providing a lower bound estimate of the value of a good or service. User fees, or the amount paid to use the services provided by the resources at that site, indicate a lower bound for the value that individuals place on the use of a specific site. The replacement cost technique infers the value of goods and services from the cost of replacing the goods and services or of providing alternatives.

Indirect, Revealed Preference Approaches

Indirect, revealed preference approaches rely on the relationships between the value placed on a good or service not traded through markets that is affected by environmental quality and the other real-world choices that individuals make. These approaches typically require modeling of these relationships to infer values for the non-marketed good or service. Because of the need to model complex relationships in order to infer values for a specific good or service, these techniques tend to have fairly significant data needs, which may include: price and quantity information for consumption of related market goods and services; use or consumption information for the good or service one wants to value; characteristics of the goods or services as well as substitute goods and services; and characteristics of users.

Travel cost studies, for example, have been used to estimate the value of a particular recreational activity, such as fishing, based on the time and expense required to partake in that activity. Similarly, in using the avoidance expenditures approach, the cost of a particular event (or benefits of preventing an event), such as flooding, is estimated based on current expenditures to prevent or reduce the negative impact of the event. Random utility models estimate recreational demand by focusing on an individual's choice among substitute sites for any given recreational trip. Hedonic property and wage models attempt to identify the value of environmental quality implicit in purchasers' willingness-to-pay for property and in the monetary value placed on working conditions, respectively. Referendum voting offers an individual a fixed quantity of a good or service at a fixed price. If the individual accepts the offer, it can be assumed that the person values the resource by at least that amount. Thus, referendum voting data (e.g., approval for new regulation or management scheme) can also be used to indicate the minimum value placed on protecting the resources affected by the outcome of the vote.

Direct, Stated Preference Approaches

Direct, stated preference approaches, or contingent valuation approaches, involve asking a sample group of people directly about the values they place on certain effects or changes. Some direct approaches used to determine an individual’s willingness-to-pay for a specific improvement include:

- Asking each individual directly how much they would be willing to pay to ensure or prevent a change;
- Asking each individual whether they would be willing to pay some specific amount of money to ensure or prevent a change, varying the amount of money across the sample; and
- Conducting a bidding game with each individual to determine the maximum amount each would be willing to pay to ensure or prevent a change.

By aggregating over the sample, an analyst can estimate a demand curve for the specific change, which can then be used to estimate total WTP for the change. Both the degree of environmental change and the cost of the change can be varied in a contingent valuation analysis. Contingent valuation analysis requires conducting a survey of a representative sample of individuals affected by the environmental change. Good survey design and implementation are critical to the success of a contingent valuation analysis. Unfortunately, these activities, as well as the analysis of the resulting data, are typically very time and resource intensive.

Indirect, stated preference approaches

Indirect, stated preference approaches are also contingent valuation studies, except that the individuals questioned are not asked directly about the value they place on a specific change. Instead, individuals are asked to make a decision about another situation that depends or otherwise relates to the value they would place on the specific change to be valued. The responses to these questions are then used to draw inferences about the value of changes to the non-market good or service of interest. For example, individuals may be asked:

- To rank combinations of varying quantities or qualities of goods, including both market goods, which have prices associated with their use, and non-market goods, for which the analyst wants to estimate the value (Contingent Ranking); or

- To estimate the change in their current level of activity or use of a specific good or service under alternative scenarios in which the availability and quality of the good or service is varied (Contingent Activity).

Contingent ranking asks individuals to rank combinations of varying quantities and qualities of non-marketed environmental goods and services as well as other marketed goods. In a contingent activity study, individuals are asked hypothetical questions about their level of activity under alternative levels of availability and quality of an environmental good or service. In a contingent referendum study, respondents are asked whether they would vote yes or no for a policy or action that would impose a specific cost on them and provide or ensure a hypothetical quality or quantity of an environmental service. Values for the environmental goods or services are then inferred from the choices made by the individuals. Conjoint analysis uses data gathered from survey respondents concerning the relative importance of various features of a product to determine the willingness-to-pay for a particular feature. For any of these indirect, stated preference approaches, the data requirements and concerns will be the same as those associated with the direct, stated preference approaches.

Benefits Transfer Approach

Benefits transfer analysis can often be used to estimate the value of a particular change when the resources or time to conduct original research are not available. Benefits transfer is also a desirable approach in cases where good information already exists from previous studies of the good or service in question, particularly when studies exist for similar types of locations and resource users. This approach involves identifying other valuation studies of similar changes at similar sites and using, or transferring, the value from the previous study(ies) to the new site of concern. In some instances, additional data might be used to adjust the value estimate to better suit the new situation or to correct for errors introduced in the original study. More advanced benefits transfer analysis involves transferring a benefit function, demand function, or valuation model to a new study site.

Data Sources

In addition to selecting a valuation technique, it is also necessary to identify data sources that can be used in the valuation of public goods and services. Some of the data, such as the ecological components affected, will come from the ecological assessment. Other data will also need to be obtained from other sources. The type of data required depends upon which valuation technique is chosen. Data might include market data on the prices of various goods, data on the number of users (e.g., the number of fishermen using a specific fishery), the quantity used (e.g., acres of forests cut down in a given year's lumber production), or some measure of the ecological resource itself (e.g., acres of wetlands). The individual valuation technique sections provide a detailed discussion of the types of data required to implement each technique.

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6.3.1 Market Price and Supply/Demand Relationships

The "market value" of a good or service that is conveyed through the market system is the price placed on the good or service. The price of a good represents the value of an additional unit of that good, assuming the good is sold through an undistorted, perfectly competitive market (i.e., a market with properly assigned property rights, full information, and no taxes or subsidies). Market prices can be used to value small changes in the quantity of a good or service being provided (i.e., small effects or changes that do not affect the supply of or demand for the product or service). For example, the value of increased commercial fish harvest in a specific bay could be estimated based on the market value of the additional fish caught (i.e., pounds of additional fish caught multiplied by market price per pound of fish), assuming that the increased harvest for the area under study will not affect the market price.

The value (i.e., cost or benefit) of larger scale changes that are likely to affect the supply or demand for a good or service cannot be correctly valued using market prices. Using market price ignores the change in the extra value provided by the good or service to consumers (e.g., the amount consumers would be willing to pay above the market price, known as consumer surplus). For the same reason, the change in the total consumer expenditures for a good or service (market price times the quantity purchased) is generally not a good indicator of the benefits associated with a change in the use of that good or service. Where such a bias matters, other approaches are necessary for estimating the benefits or the change in willingness-to-pay resulting from a change in the goods or services provided by an ecological resource.

Estimating Supply and Demand Relationships

One approach is to estimate the supply and demand relationships for each service or product before and after the environmental change to estimate the benefits of a specific action. Depending on the good or service considered, the change to the ecological resource will cause a shift in the supply curve or the demand curve. The change in the willingness-to-pay, or benefits, associated with the action can then be estimated based on the change in the area above the supply curve and below the demand curve. The demand and supply curves, or functions, are estimated using past data on prices and quantities of the good sold, the cost of production inputs, and information on production relationships (i.e., the quantity of output produced with a given amount of inputs).

Market Simulation Models

Economists have developed market simulation models that combine economic, engineering, and biophysical information to estimate changes in market supply and/or demand relationships, and thus, the benefits, of an environmental change. Such models can be used to examine the relationship between changes in environmental quality, such as the amount of acid deposition, and "material damage," including reductions in stocks of physical assets such as buildings, bridges, roads, and art, or changes in biological outputs, such as agriculture and vegetation. Environmental changes that affect the level of output or production will affect the price and quantity of the good on the market that can lead to further changes in output or production. Although simple estimates of changes in supply and demand relationships can be used to estimate the initial change in price and quantity, a more complex market simulation model is needed to estimate further changes that result from market interactions and feedback relationships. Market simulation models are regularly used to estimate the effects of changes in environmental quality on agricultural and timber production. Simulation models have also been used in material damage assessments to identify changes in production and consumption caused by environmental changes, identify the responses of input and output to these changes, and identify the adaptations affected factors can make to minimize losses or maximize gains from changes in opportunities and prices (Adams and Crocker, 1991).

Valuing the benefits of a change to an ecological resource based only on a single or a few market goods or services provided by that resource is unlikely to capture the full benefits of the change because many other services provided by the resource that are not sold through markets may also be affected. In the case of an action that improves the quality of a forest, for example, the forest will provide improved habitat for other species of flora and fauna and better scenic views and recreational opportunities, in addition to the increased value of the forest as a supply of timber. Therefore, when using changes to market goods and services to estimate benefits, one should also consider the potential benefits associated with additional services provided by the resource that are not sold through markets.

Advantages

➤ For established markets, price, quantity, and input cost information should be readily available.

➤ Actual consumer preferences are measured using observed data.

Disadvantages

➤ Market data may only be available for a limited number of goods and services provided by an ecological resource and may not reflect the value of all productive uses of a resource.

- ➤ It may be difficult to correctly estimate demand and/or supply relationships if limited data on prices and quantities are available.

- ➤ It may be difficult to separate the supply and demand effects and to isolate the effects of the environmental change.

- ➤ Market-based analyses do not capture non-use value.

Data Requirements

This technique requires time series data on market prices for the resource, the quantity sold and consumed, and detailed cost and revenue information for representative producers, as well as environmental data for both before and after the change.

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Revealed Preference Approach

6.3.2 Market-Based Valuation Approaches

Although the goods and services provided by an ecological resource may not be bought and sold through the market, there may be other market transactions occurring that provide information regarding the value of the environmental good or service under study. When estimating the value of specific goods or services, for example, it may be useful to look at other market transactions, such as fees paid for use of similar services or spending on projects or activities designed to provide similar goods or services. When estimating the value of changes to an ecological resource (or the goods and services it provides) it may be useful to consider the estimated cost of alternative actions undertaken to produce similar results or, alternatively, the level of spending to prevent or reduce the negative impacts resulting from damage to an ecological resource.

Although these measures cannot generally be expected to provide an exact measure of the benefits of a change to an ecological resource, they can be useful in developing preliminary or order-of-magnitude estimates. This section describes how the cost of alternatives or replacements, avoidance expenditures, simulated markets, referendums, and user fees might be useful in estimating the benefits of improvements to ecological resources.

Alternative/Replacement Costs

The cost of providing or replacing the goods or services that an ecological resource could provide can be used to estimate the value of those goods and services and, in some cases, the benefits of an action to protect or restore that ecological resource. This approach is based on the concept of revealed preference: by choosing to undertake an action to provide or replace certain goods and services, society demonstrates (or reveals) that it values the goods and services provided by the ecological resource (and correspondingly value the resource itself) by at least as much as the cost of the project. In other words, it is assumed that if society invests in a project to provide similar services to those provided by an ecological resource, then the value of the services provided can be assumed to be at least as great as the dollar amount spent on the project. Therefore, the cost of the project might also be used to approximate a lower bound for the value of the ecological resource that provides the same services. Specific examples include:

- Using the cost of building a retaining wall to estimate the value of wave buffering services provided by a wetland or coastal marsh area;

- Using the cost of fish breeding and stocking programs to estimate the value of fish nursery services provided by estuaries or upland streams; or

- Using the cost of constructing and operating a storm water filtration plant to estimate the value of water filtration by wetland areas.

In using this approach, however, it is important to keep in mind that because the goods or services replaced probably represent only a portion of the full range of services provided by the ecological resource, this approach is likely to underestimate the benefits of an action to protect or restore the ecological resource. In addition, this approach should only be applied if the project has been implemented or if society has demonstrated its willingness-to-pay for the project in some other way (e.g., approved spending for the project). Otherwise, there is no indication that the value of the good or service provided by the ecological resource to the affected community is greater than the estimated cost of the project.

In a similar context, the cost or estimated value of alternative approaches to achieving an environmental goal (e.g., reduced pollution levels) can be used to estimate the value of changes (most often improvements) to an ecological resource. Under this approach, the estimated benefits of one program designed to protect or improve an ecological resource would be used to estimate the benefits of a different program that is also intended to protect the same resource. For example, the value of reducing NO_x emissions, in terms of reduced nitrification of surface water bodies, might be estimated based on the estimated benefits of reducing the flow of nutrients from non-point source run-off to surface water bodies (see also ***Benefits Transfer***).

The concept and approach discussed above is different from the restoration/replacement cost approach used commonly in Natural Resource Damage Assessments (NRDA) (and incorporated in damage assessment models developed by the Department of the Interior (DOI) and the National Oceanic and Atmospheric Administration (NOAA)). The NRDA restoration/replacement cost approach uses the cost to restore, rehabilitate, or replace the damaged natural resource, in addition to the value of lost uses during the period when the resource is damaged, to determine how much the polluter should pay in compensation. The problem with using the cost of restoration or replacement as a valuation technique is that there is no direct link between the cost of the restoration activities and the value of the services provided by a natural (ecological) resource that would be lost without restoration. As a result, the estimated cost to restore or replace the natural resource will likely bear little relationship to the true social value or change in the value of the resource.

Avoidance Expenditures/Averting Behavior

Averting behavior and defensive or avoidance expenditure analyses are more commonly applied in efforts to estimate the benefits of actions that protect or improve human health. However, such approaches also may be applicable in estimating the benefits of actions that improve the state of an ecological resource. This approach is also based on the concept of revealed preference: by choosing to undertake the action, society demonstrates (or reveals) that it values the resource or the improvement of the resource at least as much as the cost of the action designed to protect or improve the resource. Some argue that this approach is inconsistent because few environmental actions and regulations are based solely on benefit-cost comparisons (particularly at the national level). As a result, the cost of a protective action may actually exceed the benefits to society. It is probably more likely, however, that the cost of those actions already taken to protect an ecological resource will underestimate the benefits of a new action to improve or protect the resource.

Using this approach to estimate the benefits of an action that protects an ecological resource, one might look at the expenditures by society to prevent or reduce the negative impacts to the resource as a measure of the value or benefits of that action. For example, the cost of alternative controls to reduce effluent emissions to a water body could be used to estimate the value or benefits of reducing pollutant concentrations in the water body.

Bartik (1988) shows formally how changes in defensive expenditures by households to alleviate the negative effects of pollution can be used to estimate the benefits of reducing pollutants. Exhibit 18 presents some of the possible measures for estimating the benefits of reducing pollutant levels using defensive expenditures:

Exhibit 18
Estimating the Benefits of Reducing Pollutants Using Defensive Expenditures

Pollutant	Defensive Expenditure Measures
Air Pollution	Clean or repaint exterior of house; install air purifiers or new air conditioners; visit the doctor more frequently; move away from pollution source
Water Pollution	New well; bottled water; water purifiers; move away
Hazardous Waste	Similar to both water and air pollution depending upon medium by which hazardous waste affects households
Noise Pollution	Storm windows; thicker walls; move away
Radon in well water	Filter or aerate water; bottled water; increase house air ventilation; move away
Radon in Soil Underneath House	Ventilate crawlspace of house; seal foundation of house; use thicker concrete in basement; increase house air ventilation; move away

Source: Bartik (1988).

Referenda

Referenda provide an institutional basis for asking individuals' preferences for certain goods and services and may provide a basis for estimating the value of a particular change. A typical referendum might ask voters if they are willing to pay a specified amount to support a program that increases the supply of a public good. The decision to vote "yes" is based on the individual voter's assessment of whether the added benefit of the program exceeds the added cost of the payment. One of three conditions must exist to use an actual referendum to value a good or service (Mitchell and Carson, 1989):

- The same people must vote for different levels of the public good at a fixed tax level or for a fixed level of the good at different tax levels. For example, a situation where a referendum fails and the supporters modify it for the next election;

- Different jurisdictions vote on the same level of a good; or
- Different jurisdictions vote on different levels of a good.

User Fees

User fee information, such as entrance fees or other fee receipts, can be used to infer the value individuals place on the use of a specific site, such as a national park. User fees indicate a lower bound for the value that individuals place on the use of a specific site. At a minimum, it can be assumed that each visitor values their visit (or use) of the ecological resource by at least as much as they paid as an entrance fee or other charge to use the services provided by the ecological resource. User fee data alone, however, is likely to significantly underestimate WTP, because it misses values such as existence and option value and does not capture other "travel costs."

If one assumes that visitors react to increases in entrance fees in the same manner as to increases in travel costs, entrance parameters can be used to trace a demand curve for the site, much in the same way as under a travel cost study where the area under the demand curve is the measure of the value of the ecological resource. In addition, it is possible to use entrance fees or other charges assessed on users as a component of a broader travel cost or random utility model study.

Simulated Markets

Simulated market studies estimate what a person would pay for a good that is not sold on the market by creating market conditions for that good. Under market conditions, the price that a person will pay for a good or service is the value that the person places on that good or service. Therefore, by mimicking market conditions, one should be able to estimate the value that a person places on public goods and services.

This technique can have advantages over other valuation methods, such as contingent valuation and travel cost. Like simulated market studies, these techniques attempt to attach value to public goods; however, they do not simulate market conditions, and therefore certain biases exist that affect their ability to estimate value.

There may also be biases associated with simulated market studies, however, due to the potentially limited scope and artificial nature of the study. Additionally, conducting a simulated market study could be potentially costly. Simulated market studies may be most useful in limited contexts for interpreting the results and biases of contingent valuation, travel cost, and other valuation techniques (Bishop and Heberlein, 1979).

Example Simulated Market Study (Bishop and Heberlein, 1979)

This study used simulated markets, contingent valuation, and travel cost to estimate the value of goose hunting permits. Goose hunting permits were readily available -- hunters wrote in and requested permits for a specific season. Each permit allowed the hunter to take one goose and no fees were charged for the permits. Three samples of hunters were drawn from the total number of hunters who were issued permits. For the simulated market approach, the first sample of hunters received cash offers for their permits by mail; if the hunter accepted the offer, they were to send the permit back, otherwise, the check. The cash offers ranged between $1 and $200. A second sample of hunters received contingent valuation questionnaires in the mail designed to measure the value of the permits. The third sample received travel cost questionnaires designed to estimate a travel cost demand curve.

Responses to cash offers yielded a total consumer surplus for the permits of $800,000 total, or $63 per permit. The contingent valuation survey estimated that the average willingness-to-sell was $101 per permit, while the average willingness-to-pay was $21 per permit. The travel cost study estimated costs per permit of $11, $28, or $45 based on the assumptions regarding the value of time (time value equals zero, 1/4 median income rate, and ½ median income rate, respectively).

In theory, the simulated market study approximates the true value of the permit more closely than a contingent valuation study would because real money was used, and people were asked to make a choice similar to the market choices that are made each day.

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6.3.3 Travel Cost Methodologies

The travel cost method was developed as a technique to value public recreation sites. This technique incorporates the assumption that individuals visiting a recreational site pay an implicit price for the site's services that includes the cost of travel to the site and the time spent visiting the site. Travel cost models pay special attention to the value of time.

To illustrate the concept behind travel cost models, consider, for example, that on a particular day a person chooses to go to work or to a park (or engage in some other activity). The person must first decide whether or not to go to work and, if the person decides to go to the park, he or she must decide how much time to spend there. The cost of the visit to the park includes the cost of getting to the park, any entry fee that is paid, plus the foregone earnings, or opportunity cost, one could have earned by going to work. If these costs and the number of trips made in one season were assembled for a large population, the unit willingness-to-pay for a certain number of visits could be estimated (Pearce and Turner, 1990).

In calculating the average willingness-to-pay for a trip using the travel cost method, it is important to note factors that require careful attention. In determining the number of trips taken by individuals, it is necessary to recognize that some visits may be multi-purpose trips and some trips may be taken by holiday-makers while others may be taken by residents. Furthermore, it may be difficult to accurately calculate distance costs and the value of time associated with visiting the site. These factors may require special attention in order to accurately estimate the value of the resources at the site (Hanley and Spash, 1993).

To determine a demand curve for recreation at a specific site, it is necessary to understand that trip costs are like prices. Theory dictates that if prices are lower, people will consume a higher quantity of the good, or, in this case, if trip costs are lower, people will take more trips to the site. By plotting trip cost against the number of trips to the recreation site from different areas, a demand-curve for recreation days can be traced (Loomis, 1993).

Typically, travel cost models are used to estimate the demand curve for an individual, although aggregate or market demand for a site might also be modeled. The consumer surplus for an individual visitor is the area under the estimated demand curve but above the trip cost. Because people come from different distances to use the site, consumer surplus is different for each user. People living close to the site "buy" more trips and pay less per trip. Hence, these people receive a much larger consumer surplus than people living farther from the site who "buy" fewer trips and pay more per trip. In other words, people living close to the site are willing to pay more than those living further away to have access to the site or to prevent deterioration of its environmental quality. Total consumer surplus for all individuals is found by adding up all of the trips from all locations and adding up each individual's consumer surplus. The average consumer surplus per trip can be used as an estimate of the average willingness-to-pay for a trip (Loomis, 1993).

Shifts in the demand curve due to an improvement in the quality of the site can be used to estimate the change in the value of the site, or the benefit resulting from the improvement.[2] Similarly, because environmental quality is expected to influence demand for a site, changes in visitation rates for sites with different levels of environmental quality, holding travel costs constant, can be used to estimate the benefit of changes in environmental quality (Pearce and Turner, 1990). However, the random utility model may be a more appropriate technique when examining the choice between multiple sites.

Advantages

- ➤ The travel cost method can provide benefit measures for changes in environmental quality from the observed behavior of visitors to recreation sites.
- ➤ The method can be adapted to many environmental quality issues where changes in quality affect the desirability of a recreation site.
- ➤ The method can be implemented using mail, phone, interview surveys, or site registration data. In some cases, data are available from state and federal resource management agencies.

Disadvantages

- ➤ Travel cost studies may over- or under-estimate the value of a good or service if they use an inappropriate estimate for the market price of the time that people spend traveling to a recreation site. Economists continue to disagree about whether the value of travel time should be based on the person's wage rate, some fraction of their wage rate, or valued at zero.
- ➤ The method can provide benefits information only on changes in environmental quality that have a direct effect on the site preferences of recreationists. Quality characteristics that users are indifferent to or unaware of cannot be evaluated.
- ➤ Exclusion of alternative recreation sites and their characteristics (environmental quality and other site features) from the travel cost model may bias the benefit estimates.
- ➤ Excludes non-use values.
- ➤ Environmental quality and other site characteristics may be difficult to describe in quantitative terms.

Data Requirements

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[2] Because the travel cost method does not provide for estimation of the theoretically correct measure of WTP for a site or for changes in the environmental quality at a site, such estimates should be used cautiously. Furthermore, because of this potential limitation, one might consider the appropriateness of utilizing a method of exact welfare measurement, where the functional form for the travel cost demand curve is derived from an explicit specification of the individual's utility function (Freeman, 1993).

Travel cost models typically have the following data needs: (1) the county of residence or zip code for users of the recreation site, population size, and summary measures for features of the population in each origin zone; (2) round-trip mileage to the site and to substitute sites; (3) mode of transportation; (4) vehicle operating costs per mile and implicit time cost of travel; and (5) data on on-site characteristics, such as size, number, location, and type of facilities. Typically, this information is collected through surveys using phone, on-site, or mail surveys, or by acquiring site registration data.

Example Travel Cost Study (Bockstael *et al.*, 1989)

A travel cost model was used to estimate the value of improved water quality to Maryland beach users on the western shore of the Chesapeake Bay. Water quality was measured as the product of the concentrations of nitrogen and phosphorous in the water at the monitoring site nearest to the beach in question. Data for the model was obtained from a survey of 484 people at 11 public beaches in the study area.

The model was used to calculate the willingness to pay for a 20 percent improvement in water quality -- that is, a 20 percent reduction on total nitrogen and phosphorus. The average annual aggregate benefits to beach users of water quality improvement were estimated to be $35 million (1984 dollars). The long-run benefits to beach users of water quality improvements may be higher than the estimates reported, however, for several reasons. First, as people learn that the Bay has become cleaner, they will adjust their preferences toward beach recreation. People who do not currently use the Bay beaches will be especially likely to make this change. Additionally, the population and income of the area have grown and are likely to continue growing, increasing the demand for and value of the water quality improvements. Finally, the estimates given ignore households outside the Baltimore-Washington Statistical Metropolitan Sampling Area.

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6.3.4 Random Utility Model

The random utility model is a popular method to estimate consumers' recreational demand. The random utility model is also known as a "discrete choice model" because it is used to study people's choices between one or more alternatives. The term "random" refers to the fact that the model cannot directly observe people's decision processes. The economist observes the final decision but must assume the decision process is logical, with people choosing the alternative providing the greatest possible level of satisfaction. The lack of direct observation is what makes the process "random" to an economist.

With respect to valuing changes to ecological resources, the use of random utility models focuses on the choices individuals make among substitute sites for any given recreational trip rather than the number of trips a recreationist takes to a given site in a season, as with the travel cost model. The random utility model is especially suitable when the selection of alternatives or substitutes differ in terms of their quality or other characteristics. The random utility model is particularly appropriate when there are many substitutes available and when the change being valued is a change in a specific quality characteristic of one or more sites, such as catch rates or water quality. The random utility model can also be used to value the benefits of introducing a new site (U.S. EPA, 1995).

The characteristics of the alternative sites that are used in the estimation of the model are instrumental in explaining how people allocate their trips across sites. Sometimes information on the characteristics of the individuals making the choices are also used in estimating a random utility model.

Advantages

- The random utility model can provide benefit measures for changes in environmental quality from the observed behavior of visitors to recreation sites.
- The method can be adapted to many environmental quality issues where changes in quality affect the desirability of a recreation site.
- This technique is preferred over the travel cost model for handling the issues of substitute sites and environmental quality considerations.

Disadvantages

- An inappropriate estimate for the value of time that people spend traveling to a site can adversely affect the estimated value of the good or service.
- The method can provide benefits information only on changes in environmental quality that have a direct effect on the site preferences of recreationists. Quality characteristics that users are indifferent to or unaware of cannot be evaluated.
- Model specification, as with all techniques and estimation procedures, can have a significant impact on benefit estimates.

- This technique requires a significant amount of data.
- Benefit estimates may be biased if: (1) known substitute sites are not included in the model or (2) additional substitute sites are included in the model that are unknown to the individuals surveyed.
- Excludes non-use values.

Data Requirements

The random utility model has data needs similar to those of the travel cost model, including the cost of travel to the site or information to estimate the cost (i.e., distance traveled, any fees paid, plus the value of the individual's time) and characteristics of the chosen site and alternative sites. In addition, the researcher needs to know what alternative sites are considered by recreationists and may want to collect information on the characteristics of the individuals (e.g., education, income, other socio-demographic information). Additionally, accurate measurement of the characteristics of the alternative sites is necessary.

Example Random Utility Study (Englin *et al.*, 1991)

This study uses both the random utility model and the travel cost method to estimate the damages to recreational trout fishing in the Upper Northeast due to acidic deposition. Data were collected on freshwater recreational trips made during the summer of 1989 by 5,724 randomly selected individuals in four Northeastern states: Maine, New Hampshire, New York, and Vermont. Changes in acidic deposition were expected to impact fish populations by changing acidic stress levels, thereby changing catch rates of various species. An angler's well-being should change when a change in the catch rate causes him or her to enjoy a site less (more) or results in a decision to change sites and travel farther (closer). The two models are based on the premise that the cost of travel to a site can be used to represent the price of a recreational fishing site.

The random utility model provides estimates of changes in the value per choice occasion based on the relevant changes in the quality characteristics of the sites available to anglers. The model estimates that damages from acidic deposition are approximately $0.12 per trip. The travel cost model estimates the willingness to pay for a marginal increase in each attribute. With this technique, the willingness to pay for no damages from acidification was found to be $0.02 per trip.

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Revealed Preference Approach

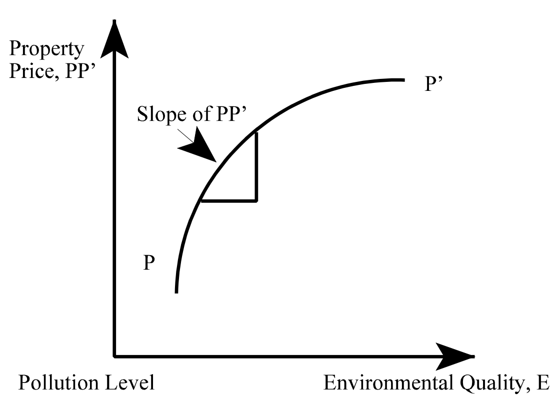
6.3.5 Hedonic Price and Hedonic Wage Methodologies

Hedonic methods typically use residential housing prices or labor wage rates, as well as other data, to measure the value of specific characteristics of a home, property, or job. These analyses identify the indirect linkage between environmental quality and the market price of a good or service, such as a residential property or employment opportunity, and use this linkage to estimate the implicit price, or benefit, of improvements in environmental quality. Under appropriate conditions, this implicit price can be interpreted as an individual's willingness-to-pay for environmental quality. In other situations, however, it is very difficult, if not impossible, to measure the welfare effects of a change to a specific characteristic, such as environmental quality. Nonetheless, the hedonic approach can still be useful for estimating a demand function for an environmental quality characteristic, such as the demand for proximity to a water body or distance from a hazardous waste facility.

Hedonic Price

The hedonic price, or property valuation, technique uses the assumption that the price of a house is a function of the characteristics of the home such as the quality of the surrounding neighborhood, the location of the home relative to business centers, and environmental characteristics including local air and water quality. The hedonic property model focuses on how changes in environmental quality affect property values by studying data from housing markets in different areas. Studying the relationships between changes in property values and differences in environmental quality can sometimes be used to determine an individual's willingness-to-pay for improved environmental quality (Palmquist, 1991). The graph below illustrates the relationship between environmental quality and property values that might be uncovered by the hedonic property model. It shows that property values rise at a declining rate as the pollution level decreases or environmental quality improves. Other shapes of the hedonic function may be possible.

Figure 1
Graphic Illustration of a Hedonic Price Equation
for an Environmental Quality Attribute (Pearce and Turner, 1990)



When a change in environmental quality affects a large population, however, the hedonic property model alone may not be adequate to measure the change in welfare, or willingness-to-pay, and a more complicated analysis is required. In this case, some knowledge of the consumers' preferences and a knowledge or a forecast of the change in the hedonic price equation (represented by PP' in Figure 1) is necessary (Palmquist, 1991). A full discussion of this issue is beyond the scope of this document.

In valuing changes in environmental quality, the hedonic approach attempts to do two things:

- Identify how much of a property price differential is due to a particular environmental difference between properties; and

- Infer how much people are willing to pay for an improvement in the environmental quality they experience (Pearce and Turner, 1990).

For example, all other things being equal, one would expect prices of houses in neighborhoods with clean air to be higher than prices for houses in neighborhoods with polluted air. By comparing the market values of similar houses in areas with different levels of air quality, one may be able to determine that part of the difference in the price of housing in the two areas can serve as a measure of the value of clean air (Tietenberg, 1992).

Example Hedonic Pricing Study (Palmquist *et al.*, 1997)

Palmquist, Roka, and Vukina used the hedonic pricing model to analyze the effects of hog operations on nearby houses. The authors examined the amount of hog manure located at varying distances from residential properties. Their purpose was to determine whether the presence of hogs influenced property values.

Results from the hedonic model show that the presence of hog operations had a statistically significant negative effect on nearby property values. Changes in house values decreased as much as approximately $5,000 for a home located within ½ mile of a projected hog operation and as little as $1 for homes located 2 miles from the projected site in an area with higher concentrations of hog operations. The results indicated that the strongest negative impacts occurred closest to the hog operation and that effects on property value decreased as distance to the operations increased. Furthermore, in areas of high concentrations of hog operations, growth of hog operations experienced smaller negative effects than those areas with low concentrations of hog operations.

Example Hedonic Pricing Study (Edwards and Anderson, 1984)

Edwards and Anderson performed a hedonic price analysis that related the value of a house and its lot to characteristics of the house such as square footage, number of bathrooms, age, size of lot, and the following coastal zone characteristics: distance to a salt pond or ocean, frontage on a salt pond or ocean, and the presence of a view of the pond or ocean from the property. Their purpose was to determine the lost economic value to property owners associated with a zoning restriction in Narragansett Bay, Rhode Island.

The results of the study suggest that the saltwater view and proximity to a salt pond are valued attributes of houses in the region. Using the estimated hedonic price equation, an approximate value of a water view was $5,790. It was further estimated that a land use policy restricting residential zoning in the salt pond region to protect groundwater supplies and water quality would result in lost opportunities for water view and water frontage locations valued at approximately $407,200.

Hedonic Wage

The hedonic wage technique is based on the presumption that, other things being equal, workers will prefer jobs with more pleasant working conditions. As the number of people seeking out the more pleasant jobs increases, the wages offered for such jobs will fall. Conversely, employers will have to raise the wage they offer for jobs with less pleasant working conditions to attract employees to these jobs. Therefore, at an equilibrium, the monetary value of better working conditions will be reflected in the difference in wages between two jobs with different working conditions (Freeman, 1993).

Hedonic models are generally used to perform two types of valuations. The first, and more common, usage concerns the value of reducing the risk of death, injury or illness. In labor markets, workers that face higher levels of environmental or other job-related risk are compensated for that risk with higher than average wages. By estimating the dollar amount by which wages are increased to compensate workers for the greater risk, one can value the benefits that would be conferred by a reduction in the risk of death, injury, or illness (Tietenberg, 1992; Viscusi, 1993). Hedonic wage studies used to value the risk of illness or mortality may produce inaccurate results, however, if they do not account for the possible self-selection of less risk averse individuals into riskier jobs.

Hedonic wage techniques can also be used, however, to value the environmental, social, and cultural amenities that vary across regions. This usage assumes that those cities and regions that are more desirable places to live and work in will attract workers from less desirable regions. As a result, employers in more desirable locations will pay lower wages, on average, than employers in less desirable locations for a worker with the same training and experience. Hedonic wage models try to measure the differences in wages between regions, or the “compensating wage differential,” to estimate the monetary value of differences in amenities (Freeman, 1993).

Example Hedonic Wage Study (Bayless, 1982)

Bayless used a hedonic wage analysis to relate the wages paid to academic professors and the air quality of the surrounding area. Bayless estimated a hedonic wage equation for salary of professors that incorporated pollution measures, as well as factors of productivity and locational characteristics. Bayless then used the hedonic wage equation to estimate a demand function for clean air, which was then used to estimate the willingness-to-pay for better air quality.

This analysis found that the professors would be willing to pay approximately $100 to $400 per year in salary to move from areas of low air quality to high air quality. Willingness-to-pay values increased as the disparity in air quality between locations increased.

Advantages

➤ The hedonic techniques use market data on property sales prices and labor wages, these data are usually available through several sources and can be related to other secondary data sources to obtain descriptive variables for the analysis.

➤ The technique is versatile and can be adapted to consider several possible interactions between market goods and environmental quality.

➤ The hedonic method provides estimates of individuals' preferences for changes in environmental quality, which, under special conditions, can be interpreted as benefit measures.

Disadvantages

➤ The assumptions necessary to interpret the results of the hedonic technique as benefit measures are restrictive and, in many real world settings, implausible. Market equilibrium conditions require full knowledge of environmental effects that may be imperceptible to the physical senses. For example, if there are subtle long-term changes in water quality associated with some housing sites but people are unaware of the causal link of the effects to the housing site, their willingness-to-pay to avoid the effects will not be reflected in housing price differences.

➤ Benefit estimates from a single product class will likely only capture a part of an individual's preferences for environmental quality. Property value models, for instance, are based on the consequences of individuals' choices of residence and therefore do not capture willingness-to-pay for improvements in environmental amenities at other points in the area, such as parks and recreational areas.

➤ The estimating equations used for the hedonic technique may be statistically sensitive to model specification and estimation decisions. Appropriate tests for unbiasedness in housing and wage studies are still being developed.

- Complete data on property or job characteristics may be difficult and expensive to gather, especially environment related characteristics. The omission of relevant characteristics and/or interactive environmental effects may reduce the validity of benefit estimates.

- It may be difficult to isolate the specific amenity or environmental characteristic that is of interest.

- Excludes non-use values.

Data Requirements

Data needs include sales or income, prices, wage data, characteristics of houses sold or jobs, and environmental amenity characteristics for each area of interest. These data can be collected from organizations such as multiple listing agencies, local tax assessors, and federal government agencies. Environmental quality data may be available from state, regional, or federal agencies and databases. Data collection, therefore, can often be time-consuming because of the effort required to gather data from a range of sources. The data sets can be gathered from markets that are separated either spatially or temporally or from a single market, although data from multiple markets tend to capture variation in price schedules, which may yield more accurate results. Additionally, while the data may be available, another problem may be the question of how individuals perceive their environment and whether individuals are aware of the quality of their environment. Most hedonic analysis use objective measures of environmental quality. However, some researchers have used subjective indicators, such as visibility, to determine environmental quality (Palmquist, 1991).

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6.3.6 Contingent Valuation

Contingent valuation studies rely on surveys to ascertain respondent preferences for environmental goods and services by determining how much someone is willing to pay for changes in the quantity or quality of the good or service. These methods do not depend on market data; instead they establish a hypothetical market that gives survey respondents the opportunity to purchase the good or service. The dollar value that individual respondents are willing to pay for the good or service, when aggregated, can provide a means to value the good or service "sold" in the hypothetical market (Mitchell and Carson, 1989; Bateman and Willis, 1998; Cummings *et al.,* 1986). Because this method does not rely on market data, it can be applied to a variety of environmental quality issues for which market-based information is not available, including the elicitation of non-use values.

Contingent valuation is a technique whereby people are asked what they are willing to pay for a benefit or what they are willing to receive by way of compensation to tolerate the loss of a good or service. The individual responses are aggregated to derive a demand curve for the good or service.

A contingent valuation study is conducted either by written survey, interview, or some combination, and it generally consists of four parts:

- Background information on the situation and possible changes to be made.
- A detailed description of the good(s) or change to the good(s) being valued and the hypothetical method of payment.
- Questions to elicit the respondents' willingness-to-pay for the good(s) or the change being valued.
- Questions to collect socio-demographic (e.g., age, income); to validate the WTP response (e.g., what are their preferences relevant to the good(s) being valued, why did they give that dollar value); and to model their use of the good(s) (e.g., how frequently they visit the site).

The aim of contingent valuation is to elicit valuations, or "bids," that are close to those that would be revealed if an actual market existed. The questioner, questionnaire, and respondent therefore, must represent as real a market as possible. For example, the respondent should be familiar with the good in question, such as improved scenic visibility, and with the hypothetical means of payment, such as a local tax or entry charge.

Several individuals and groups have identified specific criteria for conducting reliable contingent valuation studies (Bjornstad and Kahn, 1996; Arrow *et al.*, 1993; and Carson *et al.*, 1996). Generally, these criteria include (at a minimum):

- Interview a large sample of the affected population;
- Achieve a high response rate;

- Conduct in-person interviews when feasible;

- Pre-test the questionnaire for interview effects and other potential biases;

- Provide an accurate description of the event, program, or policy choice and the commodity to be valued; and

- Remind interviewees of their budget constraints and the availability of comparable goods and services.

While these guidelines are useful in assessing the reliability of a contingent valuation study, less restrictive and less costly approaches may be appropriate for informing policy decisions. In addition, some studies have found some of the above criteria (e.g., budget reminders) to have no effect on value estimates (Loomis, *et al.,* 1994).

Advantages

➤ The contingent valuation method can be used to estimate the benefits of a variety of environmental effects for which market or secondary data are not available.

➤ Comparisons of benefit measures from well-done contingent valuation studies with benefit estimates from other direct and indirect market techniques suggest that respondents can generally provide reasonable and consistent values for changes in environmental quality.

➤ Contingent valuation methods are the only methods available for estimating non-use values (e.g., existence values).

➤ The willingness-to-pay estimates from contingent valuation include both the use and nonuse value of the good or service.

➤ Survey-based contingent valuation methods can capture respondents' attitudinal and behavioral information that are not available in other non-survey based valuation techniques.

➤ Useful for estimating non-use values.

Disadvantages

➤ The contingent valuation method is based on hypothetical situations in which it is difficult to verify whether expressed preferences are consistent with actual or planned behavior. Attempts to minimize the hypothetical nature of the process may only be partly successful.

➤ Survey participants learn about their preferences for environmental quality during the valuation exercise. Survey design features may have a significant effect on this learning process and lead to responses that may not represent participants' true preferences.

Conditional choice settings that are not at least partly familiar to the respondent may lead to uncertain responses.

➤ Survey research is costly and time-consuming. National benefit estimates require properly designed sampling and enumeration procedures. Respondent refusals to consider environmental tradeoffs, as discussed in the choice exercises can raise questions regarding the validity of the benefit estimates.

➤ Participants may answer strategically (high or low). Participants may provide unrealistically high answers if they believe that they will not have to pay for the good or service, but expect that their answer may influence the resulting supply of the good. This could lead to an overestimate of the actual willingness-to-pay. On the other hand, if participants believe that they might have to pay for the good or service based on the results of the survey, they might answer in such a way to keep the price low, and thereby cause surveyors to underestimate the value of the good or service.

➤ Contingent valuation studies do not always find that WTP increases when the quality or quantity of a good or service increases.

Example Contingent Valuation Study (Whittington *et al.*, 1994)

A contingent valuation survey was conducted of randomly selected households in the Greater Houston-Galveston Area to assess residents' willingness-to-pay for improvements in Galveston Bay's environmental quality and ecological health. In total, 234 interviews were completed in a mail/in-person follow-up survey, and 393 interviews in a mail-only survey. The analysis of responses showed that high-income respondents were more likely to vote for the management plan at a given price than low-income respondents; that users of the Bay were more likely to support the plan than passive users; and that people in general were less likely to vote for the management plan as the price of the plan presented as a monthly surcharge on their water bill increased.

Based on the results of the mail-only contingent valuation survey, after adjusting results to account for differences between the socioeconomic profiles of respondents and the population of the study area, the authors estimated that the average household in the Greater Houston-Galveston Area is willing to pay approximately $7 per month, or about $80 per year, over five years for the management plan described in the questionnaire.

Elicitation Methods

There are several elicitation methods that are used in contingent valuation studies to determine an individual's willingness-to-pay. These methods represent different approaches for asking the respondent about their willingness-to-pay.

The four methods discussed here include:

- Direct, open-ended questioning
- Payment card
- Referendum/dichotomous choice
- Iterative bidding games

Each of these approaches is described below.

Direct Open-Ended Questioning

When using the direct open-ended questioning method, respondents are asked directly,

> **"How much would you be willing to pay for the change in the good or service described?"**

Although the most obvious approach, it is also one of the most problematic methods.

Advantages

➤ Does not require pre-testing to determine an appropriate range for values as do the payment card and referendum voting methods.

➤ Appears to provide conservative estimates of WTP.

Disadvantages

➤ Difficult for people to respond to an open-ended question because they are usually not accustomed to valuing environmental goods and services and typically do not face this type of question in a market situation.

➤ May result in a high non-response rate and high number of extreme values (e.g., zeros and very large values).

Payment Card

The payment card method incorporates properties similar to the direct questioning approach but increases the response rates of willingness-to-pay questions. The payment card method asks the respondent to choose a willingness-to-pay amount from a card with a range of possible willingness-to-pay amounts usually starting from $0. The card sometimes indicates the average amount households of the same income range are willing to pay for other public goods (Mitchell and Carson, 1989). The payment card method, particularly with an average amount from other households, is no longer used in contingent valuation studies, but is described here for reference in reviewing older studies.

Under this approach the respondent is asked:

> **"What amount on this card or any amount in between is the most that you would be willing to pay for the level of good being proposed?**

Advantages

- Provides more of a context for the respondent to provide a value.
- Easier for respondent to select a value than to respond to an open-ended question.

Disadvantages

- Susceptible to biases associated with the ranges shown on card and the benchmark values provided by other households in the same income range**.**

Referendum Voting/Dichotomous Choice

Referendum voting/dichotomous choice is a technique where an individual is offered a fixed quantity of a good at a fixed price on a take-it-or-leave-it or yes-no basis. This is currently the favored approach for eliciting willingness-to-pay (WTP) estimates from survey respondents. Referendum voting as an elicitation method for contingent valuation differs from the use of actual referendum data described under Market-Based Valuation Approaches, in that a contingent valuation study referendum vote is a hypothetical scenario. While often referred to interchangeably, referendum style format and dichotomous choice can be distinct approaches. A survey could use a referendum scenario with more than two voting options (see example from contingent referendum section) and dichotomous choice could be used without a referendum scenario. Observing and analyzing the choices that individuals make through these techniques reveals the value of the good as it relates to the offered price (Freeman, 1993). For example, if someone accepts an offer to pay $10 a year in additional property taxes to preserve a wilderness area, it can be assumed that the person values the area by at least $10. If the resource was valued at less than $10, the person would not have accepted the $10 fee in a vote. However, the person may value the resource at more than $10 a year, and unless iterative voting is permitted, it would be impossible to determine the maximum that the voter is willing to pay. For this reason, referendum or dichotomous choice questions are often presented with one or two follow-up questions that present the respondent with a second choice scenario. This two-stage, or double-bounded, approach increases the statistical efficiency of the valuation estimate and reduces the necessary sample size.

Advantages

- Voting is a familiar social context, therefore respondents are likely to feel comfortable answering this type of question.

- A vote provides a simple decision problem (either "yes" or "no").

- If the referendum questions are asked without an implied value judgment, there should be no starting point bias affecting the answers (Freeman, 1993).

- Recent analysis has found the referendum question format to be strategic compatible (i.e., respondents are not expected to provide unrealistically high or low values for strategic purposes of supporting or suppressing the proposed (action).

Disadvantages

- Referendum voting requires more data and a larger sample size than direct questioning.

- The outcome of referendum voting may be dependent on the distribution of offered bids, particularly the highest offered bid, because some respondents may be yea-saying or agreeing to pay any bid, no matter how high.

- Outcomes of referendum voting may be dependent on the statistical methods used to analyze the responses. (See Haab and McConnell, 1997.)

Example Contingent Valuation Study (Carson *et al.*, 1996)

A contingent valuation study using the referendum voting elicitation method was conducted by the National Opinion Research Center in 1993 of 1,182 residents in 12 U.S. cities to estimate the willingness-to-pay of individuals for a plan to provide two Coast Guard ships to escort oil tankers through the Prince William Sound to prevent future accidents and injuries due to oil spills. Willingness-to-pay was measured in terms of a one-time addition to Federal income taxes. During personal interviews, respondents were asked if they would be willing to pay a $10, $30, $60, or $120 one-time payment (each respondent was randomly assigned a dollar value). Based on the number of individuals willing to pay each dollar amount, the expected willingness-to-pay per individual was estimate to range from $50.61 to $52.81.

Iterative Bidding Games

Generally bidding games are conducted through personal interviews where the interviewer iteratively questions the respondent. Although this approach is generally no longer used because of bias issues, it is described here for reference in reviewing older studies.

Questions are structured to lead to a "yes" or "no" response. For example, to estimate the value of environmental improvements, the interviewer might ask,

> **"Would you continue to use this area if the cost to you was to increase by X dollars?" or**
>
> **"Would you be willing to pay an increase in your monthly electric bill of X dollars for Y reduction in air pollution?"**

The amount is varied with the same individual and the highest "yes" answer is recorded.

Advantages

- ➤ Able to get maximum willingness-to-pay from each individual surveyed.
- ➤ May not require as large a sample as other approaches.

Disadvantages

- ➤ The outcomes of bidding games have been found to be highly dependent on the starting point, or first offered bid.
- ➤ It can be difficult to develop a credible bidding game; the situation presented to survey respondents must be realistic and credible to the participants. Because of these difficulties, few recent contingent valuation surveys use bidding games to elicit values.

Data Requirements

The primary data for a contingent valuation analysis are acquired from a clearly defined and pretested survey of people who are representative of an affected population. A representative sample of the affected must be identified to allow extrapolation to the full affected population. Some additional research may also be required to determine the extent of the affected population or market for the good or service affected by the proposed action.

The survey should generate data on respondents' willingness-to-pay for (or willingness-to-accept) a program or plan that will affect their well-being, as well as socio-demographic information and other data required to test for potential biases. A critical component of the data collection or survey implementation is the transfer of information to respondents about the resource, resource service, or action they are being asked to value. Photographs, verbal descriptions, video, and other multimedia techniques are commonly used to convey this information. In conducting a contingent valuation survey, the quality of the results depends in large part on the amount of information that is known beforehand about the way people think about the good or service in question.

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Stated Preference Approach

6.3.7 Combining Contingent Valuation with Other Approaches: Contingent Activity

In a contingent activity or contingent behavior study individuals are asked how they would change their behavior in response to a change in an environmental amenity. For example, one could use a contingent activity to estimate how a demand function for visits to a recreational site would shift with a change in one of the site's environmental attributes. Assuming that one has already estimated the demand for visits to a site under current conditions, the analyst asks visitors how their visitation behavior would change as a result of a change in an environmental attribute of the site (e.g., change in water quality). This information can then be used to estimate a shift in the demand curve for visits to the site (Freeman, 1993).

In essence, a contingent activity approach combines the technique of contingent valuation with other valuation approaches used to model demand for a particular good or service to extend the application of these models to other scenarios. Recently, analysts have explored more advanced approaches for using travel cost data in combination with contingent valuation data to estimate a single joint model of individual's preferences and demand for a particular good or service (Cameron, 1992). Future analysis is expected to also explore the use of travel cost information and contingent valuation responses to estimate a random utility or discrete-choice model. Jointly soliciting contingent valuation responses with other data, such as travel cost data or site-selection data, both (1) expands the ability of the model to account for both current users and non-users in characterizing demand and (2) lends credibility to the contingent valuation information.

Advantages

➤ Can expand the applicability of existing valuation analyses.

➤ Potentially will allow for more complete characterization of demand by accounting for both current users of the resource and non-users.

Disadvantages

➤ The theoretical models and applied approaches for estimating demand using combined data are technically complex and not thoroughly developed.

➤ It is not clear how to reconcile data from the different approaches if they do not correspond well.

Example of Combining Contingent Valuation and Travel Cost Data (Cameron, 1992)

In this study, Cameron combines contingent valuation responses and travel cost data on actual behavior collected through a single survey instrument to estimate a joint model of individual's preferences and demand for fishing days. The in-person survey, conducted by the Texas Department of Parks and Wildlife, asked 3,366 respondents:

(1) If they would have participated in salt-water fishing if their total annual cost was $X more, where the additional dollar amount was randomly chosen from $50 to $20,000;

(2) How much they will spend on their current fishing trip; and

(3) How many trips they took over the last year.

The estimated model of demand for fishing days was then used to value greater and lesser restrictions on days of access. Specifically, Cameron estimated that a 10 percent reduction in fishing days would result in a $35 loss in welfare, on average. The complete loss of access was estimated to result in a $3,451 loss in welfare, on average.

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6.3.8 Conjoint Analysis and Contingent Ranking

This section introduces the reader to conjoint analysis, a technique applied fairly recently to the valuation of environmental quality, and the more familiar approach of contingent ranking, which actually represents one type of conjoint analysis.

Conjoint Analysis

Conjoint analysis is a technique developed by marketing analysts used to value consumer preferences for specific features of goods or services. First, a composite good is separated into its constituent attributes. Then, individuals are surveyed regarding their relative preferences for alternative bundles of attributes, with multiple attributes varying simultaneously. The information gathered from survey respondents can then be used to calculate the marginal rates of substitution between the constituent attributes. By including price as one of the attributes, it is possible to rescale the utility index in dollars and derive estimates of willingness-to-pay for particular attribute bundles.

Conjoint analyses generally fall into one of three types: ranking (same as contingent ranking approach discussed below), paired rating, and discrete choice. In a ranking study, respondents are often given several cards. Each card shows a unique product or program composed of specific attribute levels. Respondents are asked to put these cards in order -- from their most preferred to least preferred product or program. Alternatively, with the pairwise rating technique, respondents are shown two different products or programs simultaneously. Respondents are asked which product they prefer, and answer by supplying a rating within some range of number, for example, 1 to 9, where 1 indicates a strong preference for the first program, 9 indicates a strong preference for the second program, and 5 indicates indifference between the two programs. Finally, the discrete choice technique provides respondents with several different products or programs simultaneously and simply asks them to identify the most-preferred alternative in the choice set.

Conjoint analysis can be a useful technique in the valuation of improvements to ecological resources, given that several service flows are often affected simultaneously. For example, improved water quality in a lake will improve the quality of several services provided by the lake such as a cleaner drinking water supply, increased fishing and boating usage, and increased biodiversity. Conjoint analysis allows the valuation of these service flows both individually and as a whole. The technique also allows respondents to systematically evaluate trade-offs between multiple environmental attributes or between environmental and non-environmental attributes (Johnson *et al.*, 1995).

Example Conjoint Analysis Study (Mackenzie, 1992)

This study develops a conjoint measure approach to evaluate unpriced attributes for recreational waterfowl hunting trips in Delaware. First, focus interviews were conducted with various hunters to identify major attributes of hunting trips that influence trip preferences. Four plausible levels were chosen for each of the following attributes:

- Travel time (1, 2, 4, or 8 hours each way)
- Trip cost per day ($25, $50, $100, or $200)
- Type of hunting party (alone, with casual acquaintances, with close friends, or with family)
- Site congestion (none, slight, moderate, heavy)
- Hunting success (none, one duck, three ducks, three ducks and one goose)
- Annual license fee (for state residents: $15, $20, $25, or $30; else: $45, $50, $60, or $80)

A mailback survey questionnaire was designed to measure the relative preferences for these attributes by asking respondents to rank trip options providing alternative levels of each of the attributes. For example, respondents may have been asked to choose between (1) a trip with family to a slightly congested site two hours away, costing $100 per day, resulting in three ducks and requiring a $20 license, and (2) a trip with close friends to a heavily congested site one hour away, costing $25 per day, resulting in one duck and requiring a $15 license. The survey was administered in 1989 to 3,351 hunters who purchased Delaware hunting licenses for the 1988-1989 hunting season. The survey generated 1,384 usable responses; of these, 696 respondents had hunted waterfowl at least one day during Delaware's 1988-1989 waterfowl season.

A logistic model was then used to model these responses, and the marginal value of the various trip attributes could be calculated. The implied value of ducks bagged, for example, was found to be $81.35 per duck. The value of travel time was found to be $37.07 per hour.

Advantages

➤ Conjoint analysis allows the valuation of an action as a whole and the various attributes or effects of the action.

➤ Respondents are allowed to systematically evaluate trade-offs among multiple attributes.

➤ The trade-off process may encourage respondent introspection and facilitates consistency checks on response patterns (Johnson *et al.*, 1995).

➤ Respondents are generally more comfortable providing qualitative rankings or ratings of attribute bundles that include prices, rather than dollar valuations of the same bundles without prices. By de-emphasizing price as simply another attribute, the conjoint approach minimizes many of the biases that can arise in open-ended contingent valuation studies where respondents are presented with the unfamiliar and often unrealistic task of putting prices on non-market amenities (Mackenzie, 1992).

- Because the technique has been so widely used in marketing literature, many of the statistical issues in the design and analysis of this type of study have been resolved.
- Allows for questions regarding how one resource might be traded-off against another resource (rather than estimating WTP in terms of dollars).
- Allows for the assessment of situations where some attributes of a resource improve while other attributes decline.

Disadvantages

- Respondents may find some trade-offs difficult to evaluate or unfamiliar to them.
- A large number of trade-off questions may frustrate respondents.
- Pairwise comparisons impose strict assumptions on preferences.
- Although conjoint analysis has been used widely in the field of market research, its validity and reliability for valuing non-market commodities is largely untested (Johnson *et al.*, 1995).
- If the number of attributes or levels of the attributes is increased, the sample size and/or the number of comparisons each respondent makes must be increased.

Contingent Ranking

Contingent ranking asks respondents to hypothetically rank alternative choices or bundles of goods or services, where the alternatives vary in terms of their characteristics (e.g., representing different qualities or quantities of a good or service and different costs), in order of preference. These rankings can be analyzed to determine each respondent's preferences for the various attributes of the goods or services. If a monetary value can be assigned as one of the attributes, then it is possible to compute the respondent's willingness-to-pay for the environmental quality characteristic of the good or service on the basis of the ranking of the alternatives (Freeman, 1993).

One benefit of contingent ranking studies (compared to other contingent methods) is that respondents may be able to give more meaningful answers to questions about their behavior (e.g., they prefer one alternative over another) rather than to direct questions about the value of a good or service or the value of changes in environmental quality. The major challenge with contingent ranking is how to translate the answers into a dollar value. It may be necessary to imply a value from the relative ranking of other goods and services that do have a monetary value, which may lead to greater uncertainty in the actual value that is placed on the good or service of interest.

For example, contingent ranking could be used to value a proposed change in the environmental quality of a recreational site. Respondents would be asked how they rank a set of sites that vary in two or more characteristics, where one characteristic is distance and another is level of environmental quality. Based on the ranking of the sites, the value of changes in environmental quality could be implied based on how distance (and therefore, the cost of travel) is traded off for other characteristics, including environmental quality (Mitchell and Carson, 1989).

Advantages

➤ Respondents may be more comfortable ranking alternative options rather than answering a willingness-to-pay question.

Disadvantages

➤ Contingent ranking requires more sophisticated statistical techniques to estimate WTP.

➤ The respondents' behavior underlying the results of a contingent ranking study is not well understood.

➤ Contingent ranking tends to extract preferences in the form of attitudes instead of behavior intentions, and by only providing a limited number of options, it may force respondents to make choices that they would not voluntarily make (Mitchell and Carson, 1989).

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6.3.9 Benefits Transfer

Benefits transfer is often used in benefit-cost analysis when limited resources or time constraints make it difficult to conduct an original valuation study. Benefits transfer involves obtaining an existing estimate of an economic use value (e.g., unit willingness-to-pay per individual) or demand function from a previous study to estimate the value associated with a similar use being provided by a similar ecological resource under another policy case or at a new study site. The benefit estimate from the original valuation study is scaled by the level of change under the new policy case or level of use at the new study site (e.g., number of users) to estimate the benefits of a similar change in the services provided by the ecological resource under study.

Benefits transfer is most reliable when (U.S. EPA, 1995):

- The original policy case or site and the new policy case or study site are very similar;
- The environmental change is very similar for the original and new analyses; and
- The original valuation study was carefully conducted and used sound valuation techniques.

The reliability of the benefits estimate developed using the benefit transfer technique depends primarily on the similarity between the original and the new policy case or study site. With respect to benefits transfer between sites, large differences in quality, location, visitor characteristics, availability of substitutes, or object of valuation between the original and the new site have been found to impact the reliability of the benefit estimates derived through benefit transfer (Kirchhoff *et al.*, 1997).

There are three commonly used benefit transfer techniques:

- Mean unit value transfer;
- Adjusted unit value transfer; and
- Benefit/demand function or model transfer.

When possible, the transfer of demand functions or models is generally preferred to the use of unit value estimates. Both Loomis (1992) and Kirchhoff *et al.* (1997) conducted empirical analyses that found the transfer of a benefit or demand function was more reliable (e.g., smaller percentage errors) than a unit value transfer approach.

Mean Unit Value Transfer

Average unit values are generally used in benefits transfer analysis when either the demand function or model used for the original study is unavailable or the input data for a demand function or model is not available for the new policy case or study site. Average unit values are often used for regulatory analysis because the broad, typically regional or national, scope of the analyses makes it impossible, and often inappropriate, to reestimate a demand function or model developed for a specific location. The mean unit value technique assumes that the use value of a resource change under the original policy case or at the original site can be applied directly to the new policy case or site without adjustment. In this case, the unit value estimates generally apply to a specific use of the resource (e.g., recreational fishing, duck hunting, fresh water swimming) and represent an average or median value developed from a wide range of studies.

Adjusted Unit Value Transfer

The unit value estimate may be adjusted before it is applied to the new study situation to correct for any bias or inaccuracies associated with the original valuation study or to adjust for differences in the attributes of the policy case or study site that would affect the value estimate. Under the adjusted unit value technique, adjustments are generally made to account for three types of differences between the original and the new policy case or study site (U.S. EPA, 1995):

- Differences in attributes of the policy case or site, level of use, or in the socioeconomic characteristics of users affected by the change;

- Differences in the environmental policy, change, or resulting effects; and

- Differences in the availability of substitute goods and services.

Additional adjustments may also be made to the nominal value from the original study(ies) to update the value estimate to current year dollars. If the benefits transfer application is using multiple primary valuation studies from different study years, the estimates will need to be converted to the same year dollars to allow comparisons to be made.

Benefit/Demand Function or Model Transfer

A final option under benefits transfer is to transfer the entire demand function or valuation method estimated by another valuation study to the new policy case or study site. In most circumstances, as with transferring a unit value estimate, the demand function may need to be adjusted to better suit the characteristics of the new policy case or study site. The transferred demand function can then be used to estimate the willingness-to-pay or benefits associated with improving the service provided by the ecological resource. When the demand function is transferred, the benefit estimate captures both changes in the level of use and unit value benefit estimate for the new study site (Loomis, 1992). Recent research suggests that when conducting a benefits transfer analysis for a new study site, benefit or demand functions that account for a larger number of site characteristics may provide for more accurate benefit transfer analysis (Kirchhoff *et al.*, 1997). Unfortunately, the use of more detailed benefit or demand functions increases the need to collect site-specific data for both the original study site and the new study site (or policy-specific data in case of a benefits transfer analysis for a new policy case), which increases the time and resource costs of benefits transfer analysis.

Models for valuing ecological resources and damages to ecological resources can also be transferred in their entirety. Any valuation model being considered should be evaluated to determine its applicability to the new study situation, much in the same way as a unit value estimate or demand function must be reviewed for appropriateness before it is used to estimate the value of a change in a service under a different policy case or at a different site.

Advantages

- Economic benefits can be estimated more quickly than if undertaking an original valuation study.
- Benefits transfer is typically less costly than conducting an original valuation study.
- Can be used as a screening technique to determine if a more detailed, original valuation study should be conducted.

Disadvantages

- It may be difficult or impossible to find high-quality, well-documented original studies from which to obtain unit value estimates that can be appropriately applied to the new study site. The use of lower quality unit value estimates will adversely affect the accuracy and reliability of the benefit transfer analysis.
- Unit value estimates can quickly become dated.

Example Benefits Transfer Study (Bowen *et al.*, 1993; U.S. EPA, 1995)

In order to estimate the value of recreational fishing in Massachusetts Bays, Bowen *et al.* reviewed several studies of different types of marine recreational fishing experiences, largely using the travel cost model. They chose to use estimates reported by Rowe (1985) ranging from $13 to $104 (1981 dollars) per fishing day. They then inflated these estimates to 1989 dollars ($18 to $142) and applied them to the 2.5 million marine recreational fishing trips estimated to have been taken in the Massachusetts Bays region in 1989. This calculation yielded a net benefit value range of all recreational fishing trips in the Massachusetts Bays of $45 to $355 million.

This estimate is only reliable as an indication of the order of magnitude of the likely net recreational fishing benefits generated by the Bays, because the data on the number of trips conducted in the Bays system are subject to considerable uncertainty. In addition, an assumption was made that the range of recreational fishing values developed in a variety of different settings for a variety of different species reported by Rowe are applicable to the Bays system. The use of fishing day values from these other studies to value Massachusetts Bays recreation is also subject to criticism because of the use of estimates from a distinctly different geographic region.

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7.0 ISSUES AFFECTING THE ECONOMIC VALUATION OF ECOLOGICAL BENEFITS

This section identifies and briefly discusses some additional issues that should be addressed by an economic benefit analysis. These issues include:

- uncertainty and variability;
- discounting; and
- equity.

Please refer to EPA's (2000) *Guidelines for Preparing Economic Analyses* and the other references provided for a complete treatment of these topics.

7.1 UNCERTAINTY AND VARIABILITY

The variability and uncertainty associated with specific estimates is an important consideration in a thorough benefits assessment. As EPA's (2000) *Guidelines* note, the issue is "not how to avoid uncertainty, but how to account for it and present useful conclusions to those making policy decisions."

Variability and uncertainty are inherent to ecological and economic assessment, stemming from multiple potential sources including estimating the effect of the policy, modeling the fate and transport of a pollutant (e.g., air modeling), estimating effects, and valuing the effects (or changes in the effects). Variability and uncertainty arise from the inherent variation of natural processes as well as from limited knowledge about the many relationships between emissions and exposures and effects.

To assess and present uncertainty, EPA (2000) instructs the analyst to:

- Present outcomes or conclusions based on expected or most plausible values;

- Provide descriptions of all known key assumptions, biases, and omissions;

- Perform sensitivity analyses for key assumptions; and

- Justify the assumptions used in the sensitivity analysis.

If this initial assessment of uncertainty is not sufficient, then a more sophisticated analysis is required. The appropriate approach depends on the objectives of the analysis and the needs of the decisionmakers. Uncertainty and variability might be addressed by:

- Using Monte Carlo analyses or other probabilistic techniques to estimate a probability distribution for the output (e.g., benefits);

- Discussing and/or incorporating expert judgement regarding the potential range of effects and/or benefits (e.g., Delphi methods); or

- Using meta-analysis to combine estimates of inputs (e.g., risks, values) or outputs (e.g., benefits estimates) from multiple studies.

Accounting for uncertainty and variability can provide a more complete characterization of the distribution of benefits than point-estimates. Nonetheless, many sources of uncertainty will likely remain unquantified. Any remaining omissions, biases, and data gaps should be described qualitatively.

7.2 DISCOUNTING

When the benefits of an action accrue over time, such as with lagged and/or cumulative effects, the role and importance of discounting needs to be considered in the context of the benefits assessment. The discount rate used and time period for comparison can have significant effects on the magnitude of the benefits estimate and the conclusions of the benefits assessment, especially if benefits and costs occur in different points in time. Discounting can be applied to monetary values as well as quantitative assessments of benefits.

Traditionally, present value costs and benefits have been calculated using the shadow price of capital or the consumption rate of interest as the discount rate. These may be appropriate or inappropriate discount rates, depending on the assumptions made regarding the flow of capital and the value of future consequences (e.g., are future values adjusted upward to reflect increased value due to increased scarcity). Furthermore, a different discount rate (or even no discounting) might be appropriate for inter-generational effects. With respect to the time period of comparison, the analysis might choose to translate future values into present ones -- the traditional approach -- or alternatively, annualize costs and benefits or accumulate benefits (and costs) forward to some future time period.

Chapter 6 of EPA's (2000) *Guidelines* provides a lengthy and detailed discussion of discounting as well as numerous references for further reading on this topic.

7.3 DISTRIBUTIONAL AND EQUITY ANALYSES

Distributional and equity analyses examine the realized impacts or improvements across different sectors of society. Determinations regarding whether a policy or action is "equitable" rely on ethical and moral principles, rather than economic principles. In measuring changes in social welfare, economists most often implicitly assume that the welfare of all individuals is weighted equally. This assumption implies that if a positive change, or benefit, experienced by a wealthier individual is determined to be greater in value than the cost, or negative effect, experienced by a poorer person, social welfare is said to be "improved" by the change. However, such a change may not be "equitable" from an ethical or moral perspective.

To support a distributional or equity analysis, an ecological benefit analysis should provide information on the distribution of costs and benefits (i.e., track who in society is benefitting from the change and who is not) in addition to the total or net benefit estimate. The elements of a distributional or equity analysis include:

- Identifying the groups and entities of concern (e.g., race, income) for an equity evaluation;

- Ensuring that data are developed for the groups and entities of concern; and

- Estimating the distribution of changes across each group and entity of concern.

Decisionmakers then use the results of the distributional or equity analysis, along with the results of the ecological benefit analysis, other analyses conducted, and moral, legal, and/or philosophical considerations to evaluate the proposed policy or action.

Chapter 9 of EPA's (2000) *Guidelines* provides a lengthy and detailed discussion of distributional analyses as well as numerous references for further reading on this topic.

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