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1 Introduction

The mission of the Environmental Protection Agency (EPA) is to protect human health and the environment. During its history, EPA has focused much of its decision-making expertise on the first part of this mission, in particular the risks to human health from chemical stressors in the environment. Although protecting human health is the bedrock of EPA's traditional expertise, the broad mission of the EPA goes beyond this. EPA's Strategic Plan (U.S. Environmental Protection Agency [EPA], 2006b) explicitly identifies the need to ensure "healthy communities and ecosystems" as one of its five major goals. Agency publications and independent sources document EPA's efforts in protecting ecological resources – and its authority for doing so (EPA, 1994; EPA Risk Assessment Forum, 2003; EPA Science Advisory Board, 2000; Hays, 1989; Russell, 1993).

EPA's mission to protect the environment requires that the Agency understand and protect ecological systems. Ecologists use the term "ecosystem" to describe the dynamic complex of plant, animal, and microorganism communities and non-living environment interacting as a system. For example, a forest ecosystem consists of the trees in the forest plus the birds, insects, soil microorganisms, and streams that inhabit or run through it. Ecosystems provide basic life support for human and animal populations and are the source of spiritual, aesthetic, and other human experiences that are valued in many ways by many people.

There has been a growing recognition of the numerous and varied services that ecosystems provide to human populations through a wide range of ecological functions and processes (e.g., Daily, 1997). Ecosystems not only provide goods and services that are directly consumed by society such as food, timber, and water; they also provide services such as flood protection, disease regulation, pollination, and the control of diseases, pests, and climate. There is, too,, increasing recognition of the impact of human activities on ecosystems (e.g., Millennium Ecosystem Assessment Board, 2003; Millennium Ecosystem Assessment, 2005). Among the examples of this impact are traditional air and water pollution (such as sulfur dioxide emissions, ground-level ozone, and eutrophication), as well as global warming; changes in the nitrogen cycle; invasive species; aquifer depletion, and land conversions that lead to deforestation or loss of wetlands and biodiversity.

Given the vital role that ecosystems play in our lives, the state of these systems and the flow of services they provide have important human implications. EPA actions, including regulations, rules, programs, and policy decisions, can affect the condition of ecosystems and the flow of ecosystem services. These effects can occur narrowly, at a local or a regional scale, or broadly, at a national or global scale.

Despite the importance of these ecological effects, EPA policy analyses have tended to focus on a limited set of ecological endpoints, such as those specified in tests for pesticide regulation (e.g., effects on the survival, growth, and reproduction of aquatic invertebrates, fish, birds, mammals, and terrestrial and aquatic plants) or specified in laws administered by the Agency (e.g., mortality to fish, birds, plants, and animals) (EPA Risk Assessment Forum, 2003).[1] Given EPA's responsibility to ensure healthy communities and ecosystems, the Agency should consider the full range of effects that its actions will have: on human health; on individual organisms and plant and animal populations; and on the key structural and functional characteristics of communities and ecosystems. Such consideration should be comprehensive and integrated.

To promote good decision making, policy makers also require information about how much ecosystems contribute to society's well-being. EPA increasingly recognizes this need. The stated goal of EPA's recently released *Ecological Benefits Assessment Strategic Plan* is to "help improve Agency decisionmaking by enhancing EPA's ability to identify, quantify, and value the ecological benefits of existing and proposed policies" (2006a, p. xv). Information about the value of ecosystems and the associated effects of EPA actions can also help inform the public about the need for ecosystem protection, the extent to which specific policy alternatives address that need, and the value of the protection compared to the costs.

Despite EPA's stated mission and mandates, a gap exists between the need for understanding and protecting ecological systems and services and EPA's ability to address this need. This report is a step toward filling that gap. It describes how an integrated and expanded approach to ecological valuation can help the Agency describe and measure the value of protecting ecological systems and services, thus better meeting its overall mission.

This report was prepared by the Committee on Valuing the Protection of Ecological Systems and Services (C-VPESS) of EPA's Science Advisory Board (SAB). The SAB saw a need to complement the Agency's ongoing work by offering advice on how EPA might better value the protection of ecological systems and services and how that information could support decision making to protect ecological resources. Therefore, in 2003, the SAB Staff Office formed C-VPESS,[2] a group of experts in decision science, ecology, economics, engineering, law, philosophy, and psychology, with a particular understanding of ecosystem protection. The committee's charge was to undertake a project to improve the Agency's ability to value ecological systems and services.[3] The SAB set the following goals:

- Assessing Agency needs for valuation to support decision making
- Assessing the state of the art and science of valuing protection of ecological systems and services
- Identifying key areas for improving knowledge, methodologies, practice, and research at EPA

This report provides advice for strengthening the Agency's approaches for valuing the protection of ecological systems and services, facilitating the use of these approaches by decision makers, and investing in the research areas needed to bolster the science underlying ecological valuation.[4] It identifies the need for an expanded and integrated approach for valuing EPA's efforts to protect ecological systems and services. The report also recognizes and highlights issues that need to be addressed in using and improving current valuation methods and recommends new research to address these needs. It provides advice to the Administrator, EPA managers, EPA scientists and analysts, and other staff across the Agency concerned with ecological protection. It addresses valuation in a broad set of contexts, including national rule making, regional decision making, and site-specific decisions that protect ecological systems and services.

This report appears at a time of lively interest internationally, nationally, and within EPA in valuing the protection of ecological systems and services. Since the establishment of the SAB C-VPESS, a number of major reports have focused on ways to improve the characterization of the important role of ecological resources (Silva and Pagiola, 2003; National Research Council [NRC], 2004; Pagiola, von Ritter et al., 2004; Millennium Ecosystem Assessment, 2005).[5] In addition, the Agency itself has engaged in efforts to improve ecological valuation. The most recent product of these efforts is the *Ecological Benefits Assessment Strategic Plan* noted above

(EPA, 2006a). EPA also has sought to strengthen the science supporting ecological valuation through the extramural Science to Achieve Results (STAR) grants program.

The committee has both learned from and built upon these recent efforts. However, C-VPESS distinguishes its work from many of the earlier efforts in several key ways. First, C-VPESS considers EPA its principal audience. In particular, C-VPESS analyzes ways in which EPA can value its own contributions to the protection of ecological systems and services, so that the Agency can make better decisions in its eco-protection programs. Many of the recent studies, including the Millennium Assessment and National Research Council report, do not consider the specific policy contexts or constraints faced by EPA. Second, most, but not all, of the previous work has concentrated on economic valuation, and monetary valuation in particular. C-VPESS, by contrast, is inter-disciplinary and does not focus solely on monetary or economic methods or values.

The report is structured as follows. Chapter 2 provides an overview of the conceptual framework and general approach advocated by the committee. It discusses fundamental concepts as well as the current state of ecological valuation at EPA. Most importantly, it identifies the need for an expanded and integrated approach to ecological valuation at EPA and describes the key features of this approach. Subsequent chapters develop in more detail the basic principles outlined in chapter 2, focusing on implementation. Chapter 3 discusses predicting the effects of EPA actions and decisions on ecological systems and services. Chapter 4 examines a variety of methods for valuing these changes. Chapter 5 covers cross-cutting issues related to deliberative approaches, uncertainty, and communication. Recognizing that implementation of the process can vary depending on the decision context, chapter 6 discusses implementation in three specific contexts where ecological valuation could play an important role in EPA analysis: national rule making, regional partnerships, and site-specific decisions (looking specifically at cleanup and restoration). Finally, chapter 7 provides a summary of the report's major findings and recommendations.

2 Conceptual framework

2.1. An overview of key concepts

2.1.1. The concept of ecosystems

As noted in chapter 1, the term "ecosystem" describes a dynamic complex of plant, animal, and microorganism communities and their non-living environment, interacting as a system. Ecosystems encompass all organisms within a prescribed area, including humans, who are often the dominant organism. Ecosystem functions or processes are the characteristic physical, chemical, and biological activities that influence the flows, storage, and transformation of materials and energy within and through ecosystems. These activities include processes that link organisms with their physical environment (e.g., primary productivity and the cycling of nutrients and water) and processes that link organisms with each other, indirectly influencing flows of energy, water, and nutrients (e.g., pollination, predation, and parasitism). These processes in total describe the functioning of ecosystems.

2.1.2. The concept of ecosystem services

Ecosystem services are the direct or indirect contributions that ecosystems make to the well-being of human populations. Ecosystem processes and functions contribute to the provision of ecosystem services, but they are not synonymous with ecosystem services. Ecosystem processes and functions describe biophysical relationships that exist whether or not humans benefit from them. These relationships generate ecosystem services only if they contribute to human well-being. Thus, ecosystem services cannot be defined independently of human values.

The Millennium Ecosystem Assessment uses the following categorization of ecosystem services:

- Provisioning services – services from products obtained from ecosystems. These products include food, fuel, fiber, biochemicals, genetic resources, and fresh water. Many, but not all, of these products are traded in markets.
- Regulating services – services received from the regulation of ecosystem processes. This category includes services that improve human well-being by regulating the environment in which people live. These services include flood protection, human disease regulation, water purification, air quality maintenance, pollination, pest

control, and climate control. These services are generally not marketed but many have clear value to society.

- Cultural services – services that contribute to the cultural, spiritual, and aesthetic dimensions of people's well-being. They also contribute to establishing a sense of place.
- Supporting services – services that maintain basic ecosystem processes and functions such as soil formation, primary productivity, biogeochemistry, and provisioning of habitat. These services affect human well-being indirectly by maintaining processes necessary for provisioning, regulating, and cultural services.

As this categorization suggests, the Millennium Ecosystem Assessment adopts a very broad definition of ecosystem services, limited only by the requirement of a direct or indirect contribution to human well-being. This broad approach recognizes the myriad ways in which ecosystems support human life and contribute to human well-being. Boyd and Banzhaf (2006) propose a narrower definition that focuses only on those services that are "end products of nature, i.e., "components of nature, *directly* enjoyed, consumed or used to yield human well-being" (emphasis added). They stress the need to distinguish between intermediate products and final (or end) products and include only final outputs in the definition of ecosystem services, because these are what affect people most directly and consequently what people are most likely to understand. Under this definition, ecosystem functions and processes, such as nutrient recycling, are not considered services. Although they contribute to the production of ecological end products or outputs, they are not outputs themselves. Likewise, because supporting services contribute to human well-being indirectly rather than directly, they are not included.

Regardless of the specific definition used, ecosystem services play a key role in the evaluation of policies that affect ecosystems because they reflect contributions of the ecosystem to human well-being. Simply listing the services derived from an ecosystem, using the best available ecological, social, and behavioral sciences, can help ensure appropriate recognition of the full range of potential ecological responses to a given policy and their effects on human well-being. It can also help make the analysis of the ecosystem more transparent and accessible.

The committee recognizes that ecosystems can be important not only because of the services they provide to humans directly or indirectly, but also for other reasons including respect for nature based on moral, religious, or spiritual beliefs and commitments. The committee's name includes reference to the protection of both ecosystem services and the ecosystems themselves. Thus, although much of this report focuses on ecosystem services, the discussion of ecological protection and valuation applies both to ecosystem services and to ecosystems per se.

2.1.3. Concepts of value

People assign or hold all values. All values, regardless of how they are defined, reflect either explicitly or implicitly what the people assigning them care about. In addition, values can be defined only relative to a given individual or group. The value of an ecological change to one individual might be very different than its value to someone else.

People might use many different concepts of value when valuing the protection of ecosystems and their services. People have material, moral, religious, aesthetic, and other interests, all of which can affect their thoughts, attitudes, and actions toward nature in general and, more specifically, toward ecosystems and the services they provide. Thus, when people talk about environmental values, the value of nature, or the values of ecological systems and services, they may have different things in mind. Furthermore, experts trained in different disciplines (e.g., decision science, ecology, economics, philosophy, psychology) understand the concept of value in different ways. These differences create challenges for ecological valuations that seek to draw from and integrate insights from multiple disciplines.

In short, value is a complex concept. Nonetheless, in considering concepts of value, a fundamental distinction can be made between those things that are valued as ends or goals and those things that are valued only as means. To value something as a means is to value it for its usefulness in helping bring about an end or goal that is valued in its own right. Things or actions valued for their usefulness as means are said to have instrumental value. Alternatively, something can be valued for its own sake as an independent end or goal. While a possible goal is maximizing human well-being, one could envision a range of other possible social goals or ends including protecting biodiversity, sustainability, or protecting the health of children. Things valued as ends are sometimes said to have intrinsic value. This term has been used extensively in the philosophical literature but there is not general agreement on its exact definition.[7]

Ecosystems can be valued both as independent ends or goals and as instrumental means to other ends or goals. This report therefore uses the term "value" broadly to include both values that stem from contributions to human well-being and values that reflect other considerations, such as social and civil norms (including rights), and moral, religious, and spiritual beliefs and commitments.

The broad definition of value used here extends beyond what are sometimes called the benefits derived from ecosystem services. Even the term "benefits," however, means different things in different contexts. In some contexts (e.g., Millennium Ecosystem Assessment Board, 2003, Millennium Ecosystem Assessment, 2005), benefits refers to the contributions of ecosystem services to human well-being. In contrast, the term has a very precise (and narrower) meaning in the context of EPA regulatory impact analyses conducted under guidance from the U.S. Office of Management and Budget (OMB). In that context, benefits are defined by the economic concept of the willingness to pay for a good or service or willingness to accept compensation for it.

Table 1 lists the various concepts of value considered by the committee, categorized as either preference-based or bio-physical. Although people assign or hold all values, preference-based values reflect individuals' preferences across a variety of goods and services, including (but not limited to) ecosystems and their services. In contrast, bio-physical values reflect explicit or implicit bio-physical goals or standards determined to be important prior to the valuation. The goal or standard might be chosen directly by decision makers or based on the preferences of the public or relevant groups of the public. Separating values into preference based and biophysical categories is not the only way to categorize values, but it has proven useful for the committee in understanding the various concepts of value used by different disciplines and how they relate.

These value concepts are not mutually exclusive. For example, values expressing attitudes or judgments can be based on the same utilitarian goals as those underlying economic values or on the same considerations that underlie civic values. Likewise, constructed preferences can relate to self-interested attitudes or judgments (as economic values do) as well as expressed civic values.

Table 1: A Classification of Concepts of Value as Applied to Ecological Systems and Their Services

<u>Preference-based values</u>
Economic values
Constructed preferences
Community-based values

Attitudes or judgments

<u>Bio-physical values</u>

Bio-ecological values

Energy-based values

Economic values are based on individuals' preferences and reflect the tradeoffs that individuals are willing to make, given the constraints they face. For example, the economic value of a change in one good (or service) can be measured by the amount of another good that an individual is willing to give up in order to get the change in the first good, given his income and the prices he faces. Alternatively, it can be defined as the change in the amount of the second good that would compensate him for foregoing the change in the first good. Economic values are based on utilitarianism and assume substitutability, i.e., that different combinations of goods and services can lead to equivalent levels of utility for an individual. People are assumed to be rational and have well-defined and stable preferences over alternative outcomes, which are revealed through actual or stated choices. Economic values can include both use and nonuse values, and they can reflect both self-interest and altruistic motives.

The tradeoffs that define economic values need not be defined in monetary terms (willingness to pay for a change, or willingness to accept monetary compensation for foregoing it), although typically they are. Expressing economic values in monetary terms allows a direct comparison of the values of ecosystem services with the values of other services produced through environmental policy changes (e.g., effects on human health) and with the costs of those policies.

Constructed preferences. In contrast to the assumption underlying economic values, values reflecting constructed preferences are based on the premise that, particularly when confronted with unfamiliar choice problems, individuals do not have well-formed preferences and hence values. This implies that simple statements of preferences or willingness to pay are unreliable (Gregory and Slovic, 1997; Lichtenstein and Slovic 2006). Some have advocated using a structured or deliberative process as a way of assisting respondents in learning about the ecosystem services to be valued and in constructing their preferences and values. This report refers to values arrived at by these processes as "constructed values." The difference between economic values and constructed values can be likened to the difference between the work of an archeologist and that of an architect. Economic methods assume preferences exist and simply need to be "discovered" (implying the analyst works as a type of archeologist), while constructed value methods assume that preferences need to be built through the valuation process (similar to the work of an architect). As a result, the values expressed by individuals (or groups) engaged in a constructed-value process are expected to be influenced by the process itself. Constructed values can reflect both self-interest and community-based values.

Community-based values are based on the assumption that, when consciously making choices about goods that might benefit the broader public, individuals make their choices based on what they think is good for society as a whole rather than what is good for them as individuals. In this case, individuals could place a positive value on a change that would reduce their own individual well-being (e.g., Jacobs, 1997; Costanza and Folke, 1997; Sagoff ,1998). In contrast to economic values, these values may not reflect tradeoffs that individuals are willing to make, given their income. Instead, an individual might express value in terms of the tradeoffs (perhaps, but not necessarily, in the form of monetary payment or compensation) that he feels society as a whole – rather than he as an individual – should be willing to make.

Attitude or judgment-based values are based on empirically derived descriptive theories of human attitudes, preferences, and behavior. These values are not necessarily defined in terms of tradeoffs and are not typically constrained by income or prices, especially those that are outside the context of the specified assessment process. Rather, the values are derived from individuals' judgments of relative importance, acceptability, or preferences across the array of changes in goods or services presented in the assessment. Preferences and judgments are often expressed through responses to surveys asking for choices, ratings, or other indicators of importance. The basis for judgments may be individual self-interest, community well-being, or accepted civic, ethical, or moral obligations.

Human preferences directly determine all of the concepts of value described above. In contrast, bio-physical values do not depend directly on human preferences. Bio-physical values reflect the contribution of ecological changes to a pre-specified bio-physical goal or standard identified or set prior to measuring the contribution of those changes. This goal or standard can be defined in ecological terms (e.g., biodiversity or species preservation) or based on a bio-physical theory of value (e.g., energy theory of value).

Bio-ecological values depend on known or assumed relationships between targeted ecosystem conditions (e.g., biodiversity, biomass, energy transfer, and transformation) ecosystem functions, and the pre-specified bio-physical goal or standard. Scientists can determine bio-ecological values in several different ways that contribute to the goals. For example, contributions to a biodiversity goal could be based on individual measures such as genetic distance or species richness, or on more comprehensive measures that reflect multiple ecological considerations.

Energy-based values, which reflect an energy theory of value, are based on the effect of an ecological change on energy or materials flows into and out of ecological systems. They are defined as the free energy (or "exergy") required directly and indirectly to produce a good or service. Note that, although energy-based methods were designed to provide an alternative way to define value independently of human preferences, choosing to use energy as the basis for defining value implicitly reflects the preferences of those who use it.

The committee considered all of these various concepts of value in its deliberations. To date, EPA analyses have primarily sought to measure economic values, as required by some statutes and executive orders (see section 6.1). However, the committee believes that information based on other concepts of value can also be an important input into Agency decisions affecting ecosystems. Recognizing the significance of multiple concepts of value is an important first step in valuing the protection of ecological systems and services.

2.1.4. <u>The concept of valuation and different valuation methods</u>

Although ecosystems per se and their associated services have value, the term "valuation" generally refers to the process of measuring the value of a *change* in an ecosystem, its components, or the services it provides – i.e., it is predicated on a comparison of "with" and "without" or "before" and "after" scenarios. In its simplest form, valuation requires, first, a prediction of a change in the ecosystem or the flow of ecosystem services, and then, the assignment of a value to that change. When a valuation exercise seeks to measure the value of an entire ecosystem or its services, it should be interpreted as a comparison of the world with the ecosystem (and hence the services provided) and the world without the ecosystem – a world that might be difficult to describe in any meaningful way.

An important issue in ecological valuation is the extent to which individuals expressing values understand the contributions of related ecological goods and services to human well-being. In many cases, an ecological change may have important implications that are not widely recognized or understood by the general public. For example, Weslawski et al. (2004) indicated that the invertebrate fauna found in soils and sediments are important in remineralization, waste treatment, biological control, gas and climate regulation, and erosion and sedimentation control. However, the general public had no understanding or appreciation of these services (although the public may have an appreciation of the higher-level services or end-point services, such as clean water, aesthetics, and foods that could be derived from the system). Likewise, although individuals might understand the recreational contributions to human well-being associated with a given EPA action to limit nutrient pollution in streams and lakes, they might not recognize or fully appreciate the associated nutrient-cycling or water-quality implications. The policy preferences or values they express will reflect that incomplete information. Individuals might respond to a survey or behave as if they place no value on an ecosystem service if they are ignorant of the role of that service in contributing to their well-being or other goals.

Ideally, valuation should seek to measure the values that people hold and would express if they were well informed. When the public is not well informed, the valuation process should provide information about ecological responses to policy options based on the best available science. More generally, public agencies have an obligation to aggressively pursue public education when a gap exists between public knowledge (and hence, expressed values) and scientific understanding. Although valuation should be informed by the best available science, it ultimately seeks to reflect the values that would be held by a fully informed general public, not merely the personal values or preferences of scientists or experts. Basing valuation on the personal preferences of scientists or experts rather than those of the general public would undermine the usual presumptions that, in a democratic society, values held individually and collectively within society should be considered in public policy decisions, and that public involvement is central to democratic governance (e.g., Berelson, 1952).

Even when valuation is informed by the best available science, the valuation process will almost always involve uncertainty. Uncertainty arises in the prediction of changes in ecosystems, in the resulting change in the flow of services, and in estimating the values associated with those changes. The valuation process needs to recognize, assess, and communicate the various sources of uncertainty (see section 5.1 for further discussion).

The valuation process should also recognize that information about different types of value may be important for decision making and identify appropriate methods for characterizing or measuring those values. There are a number of valuation methods that can be used to try to estimate or measure values. Some of these methods are well developed and have been applied extensively in different contexts; others require further development and testing. Chapter 4 discusses specific methods. However, applying any a method to value changes in ecological systems and services poses significant challenges beyond those that might exist in other, less complex contexts.

The methods considered here differ on a number of dimensions, including the type(s) of value they attempt to measure (and hence their theoretical foundations and assumptions) and the type(s) of metrics or outputs produced. In addition, some valuation methods yield a single metric of value, while others yield multiple metrics. Methods that produce a single metric are not necessarily preferable to those that do not. Which approach is more appropriate or useful depends, in general, on the decision context. For example, if the context requires a ranking or choice based on a single criterion (e.g., net benefits), a valuation approach that yields a single (aggregate) metric is needed. In contrast, in a decision context where multiple values are involved (e.g., human health, threatened species, aesthetics, social equity, and other civil obligations) and decision makers themselves are charged with appropriately weighing and balancing competing interests and resolving trade-offs, a multi-attribute approach is preferable. Depending upon the context, this weighing and balancing might be done through political discourse or through a deliberative, decision-aiding process (see the discussions in section 5.3).

Finally, a fundamental distinction exists between valuation methods that assume individuals have well-defined preferences and those based on the premise that preferences – and hence values – must be constructed through the valuation process. As discussed above, the concept of constructed values is based on the premise that, for complex and relatively unfamiliar goods such as ecosystems and some of their associated services, an individual's preferences may not be well-formed and may be subject to intentional or unintentional manipulation or bias (e.g., by changes in the wording or framing of surveys). The extent to which this is true has been the subject of scholarly debate both within the committee and outside, and most likely varies with the context. If preferences and values regarding ecological systems and services are not well-formed and must instead be constructed, they cannot be accurately measured or characterized by valuation methods that assume well-formed preferences. For example, some individuals have strongly held values that they find difficult, impossible, or inappropriate to express in monetary units. Requiring these individuals to express such values in monetary equivalents (e.g., in a survey) may compel them to assume a perspective that is unfamiliar or even offensive. Valuation methods based on discourse and deliberation are designed to make explicit and facilitate the construction of preferences in such contexts.

2.2. Ecological valuation at EPA

As noted in chapter 1, this report is focused on ecological valuation within EPA. This necessitates consideration of some issues that might not be considered in more general discussions of ecological valuation. EPA operates in a variety of different decision contexts where valuation might be useful. Although much of the interest in ecological valuation at EPA has focused on valuation needs in national rule making, valuation can also be useful in other decision contexts. Different parts of the Agency need valuation for different purposes and for different audiences. Some contexts closely prescribe how valuations are to be conducted; other contexts are less prescriptive. In addition, EPA faces institutional constraints that influence and limit how it typically conducts valuation in different contexts.

This section of the report describes the committee's understanding of the Agency's needs and constraints related to ecological valuation. It then discusses the committee's understanding of how ecological valuation is typically done at EPA, using an illustrative example. The committee's observations from this example form the basis of its recommendations for an expanded and integrated approach to valuation discussed in the remainder of this report.

2.2.1. Policy contexts at EPA where ecological valuation can be important

As noted, much of the interest in ecological valuation at EPA stems from the need to better value the ecological effects of EPA actions in national rule makings. Two of EPA's governing statutes (the Toxic Substances Control Act and the Federal Insecticide, Fungicide and Rodenticide Act) require economic assessments for national rule making. In addition, Executive Orders 12866 and 13422 have similar requirements for "significant regulatory actions." An Office of Management and Budget circular on "Regulatory Analysis" (OMB Circular A-4) issued in September 2003 identified key elements of a regulatory analysis for "economically significant rules."[8]

EPA's regional offices may also find valuation important in their partnerships with other governments and organizations where the contributions of ecological protection to human welfare are potentially important. Regional offices, for example, may find valuation useful in setting priorities, such as targeting projects for wetland restoration and enhancement or identifying critical ecosystems or ecological resources for attention. Valuation may also assist state and local governments, other federal agencies, and non-governmental organizations in deciding how best to protect lands and land uses and in communicating the value of the approach chosen.

Valuation can also be useful to EPA in making site-specific decisions, such as those related to the remediation, restoration, and redevelopment of contaminated sites. By demonstrating the value of the ecosystem services that could be obtained from site redevelopment, ecological valuation can enhance decisions at cleanup sites, including hazardous waste sites listed on the Superfund National Priority List and other cleanup sites (e.g., sites that are the focus of EPA's Brownfields Economic Redevelopment Initiative, Federal Facilities Restoration and Reuse Program, Underground Storage Tank Program, and Research Conservation and Recovery Act).

Although many of the issues and recommendations throughout this report apply across decision contexts, specific valuation needs and opportunities vary across these contexts. For this reason, chapter 6 of this report discusses the implementation of the report's general recommendations in these three specific decision contexts: national rule making, regional partnerships, and site-specific restoration or redevelopment. Ecological valuation may also be useful for EPA in other contexts and for other purposes, including:

- Assessing programs as mandated by the Government Performance and Results Act (GPRA) of 1993[9]
- Setting Supplemental Environmental Protection penalties for enforcement cases where those penalties involve protection of ecological systems and services
- Reviewing Environmental Impact Statements prepared by other federal agencies, under the National Environmental Protection Act
- Executing ecological protection duties otherwise delegated to states, such as issuing permits to protect water quality, for those specific states that have not applied for or been approved to run programs on their own

Although this report does not explicitly discuss these contexts, the approach and selected valuation methods described can be useful in the such contexts.

2.2.2. Institutional and other issues affecting valuation at EPA

EPA must conduct ecological valuation within a set of institutional, legal, and practical constraints. These constraints include procedural requirements relating to timing and oversight, as well as resource limitations (both monetary and personnel). To better understand the implications of these issues for its work, the committee conducted a series of interviews with Agency staff.[10] The interviews focused on the process of developing economic analyses as part of Regulatory Impact Assessments (RIA) for rule making and on the relationship between EPA and the Office of Management and Budget (OMB). The interviews also proved beneficial in better understanding strategic planning, performance reviews, regional analysis, and other situations where the Agency needs to assess the value of ecosystems and ecosystem services.

EPA has a formal rule-development process involving several stages, each of which imposes demands on the Agency. Despite the rigidity of the process, Agency analysts assess the value of protecting ecosystems in different ways. Practices vary considerably across program offices, reflecting differences in mission, in-house expertise, and other factors. Program offices have different statutory and strategic missions and have primary responsibility for developing the rules within their mission-specific areas. The organization, financing, and skills of the program offices differ. Although the National Center for Environmental Economics (NCEE) is the Agency's centralized reviewer of economic analysis within the Agency,[11] the primary expertise and development of the rules resides within the program offices.

The timing of the process largely determines the kinds of analytical techniques that are employed. Court-imposed deadlines on the rule process, as well as Paperwork Reduction Act requirements related to the collection and analysis of new data, influence the timing. By contrast, the scientific community is accustomed to much longer time horizons for their analyses.

Collecting new data poses a significant bureaucratic problem for the Agency. To collect original data, the Agency must submit an Information Collection Request, which is reviewed within the Agency and by OMB. The Paperwork Reduction Act requires this hurdle and imposes the review responsibility on OMB, adding a significant amount of time to the assessment process. With a time limit of one or two years, at most, to conduct a study, this kind of review significantly limits the scope of analysis the Agency can conduct. Because EPA most often has not been able to collect new information, the Agency has, by necessity, relied heavily on transferring ecological and social values information from previous studies to new analyses.

OMB also acts as an oversight body to review EPA's economic benefit analyses. EPA must justify its claims regarding the economic benefits of its actions, including any analyses of willingness to pay or willingness to accept for ecological protection. As noted above, OMB's Circular A-4 provides explicit guidance for valuation. For a contribution to human welfare or cost that cannot be expressed in monetary terms, the circular instructs Agency staff to "try to measure it in terms of its physical units," or, alternatively, to "describe the benefit or cost qualitatively" (p. 10).[12] Thus, although Circular A-4 does not require that all economic benefits be monetized, it does require, at a minimum, some scientific characterization of those contributions. However, little guidance is provided on how to carry out this task. The circular instead urges regulators to "exercise professional judgment in identifying the importance of non-quantified factors and assess as best you can how they might change the ranking of alternatives based on estimated net benefits" (p. 10).

In conducting benefit assessments, EPA has an incentive to use valuation methods that have been accepted by OMB in the past. This creates a bias toward the status quo and a disincentive to explore new or innovative approaches, both when monetizing values using economic valuation and when quantifying or characterizing values that are not monetized. The committee recognizes the importance of consistency in the methods used for valuation, but also sees limitations from relying solely on previously approved methods when innovative or expanded approaches might also be considered.

A related issue involves review of RIAs by external experts. The Agency does not take a standardized approach to RIA review. EPA staff and managers reported that peer review was focused only on "novel" elements of an analysis, meeting the requirements of EPA's peer review policy (EPA, 2006c). This raises the question of how the term novel is defined by the Agency, and perhaps by OMB. More importantly, the novelty standard, ironically, creates another incentive to avoid conducting innovative analyses because the fastest, cheapest option is to avoid review altogether.

Finally, the Agency relies, to varying degrees, on a variety of offices to develop assessments, including individual program offices and NCEE. It is not clear what form of organization is most effective. The Agency's *Ecological Benefits Assessment Strategic Plan* (2007) contains suggestions for addressing some of the limitations on ecological valuation resulting from the Agency's internal structure. It advocates the creation of a high-level Agency oversight committee and a staff-level ecological valuation assessment forum. The committee endorses these recommendations.

The Agency will continue to face significant external constraints when considering ecological valuation. The committee recognizes the practical importance of these constraints and urges the Agency to be as comprehensive as possible in its analyses within the limitations imposed by these constraints.

2.2.3. An illustrative example of economic benefit assessment related to ecological protection at EPA

To better understand the current state of ecosystem valuation at EPA, the committee thoroughly examined one specific case in which assessment of economic benefits was undertaken: the environmental and economic benefits analysis that EPA prepared in support of new regulations for Concentrated Animal Feeding Operations (CAFOs) (EPA, 2002a).[13,14] In communications with the committee, the Agency indicated that this analysis was illustrative in form and general content of other EPA regulatory analyses and assessments of the economic benefits of ecological protection.

EPA proposed the new CAFO rule in December 2000 under the federal Clean Water Act, to replace 25-year-old technology requirements and permit regulations. EPA published the final rule in December 2003. The new CAFO regulations, which cover more than 15,000 large CAFO operations, require the reduction of manure and wastewater pollutants from feedlots and land applications of manure and remove exemptions for stormwater-only discharges.

Because the proposed new CAFO rule constituted a significant regulatory action, Executive Order 12866 required EPA to assess the economic costs and benefits of the rule.[15] An intra-agency team at EPA, including economists and environmental scientists, worked with the U.S. Department of Agriculture on the economic benefit assessment. Before publishing the draft CAFO rule in December 2000, EPA spent two years preparing an initial assessment of the economic costs and benefits of the major options. After releasing the draft rule, EPA spent another year collecting data, taking public comments, and preparing assessments of new options. EPA published its final assessment in 2003. EPA estimates that it spent approximately $1 million in overall contract support to develop the assessment, with approximately $250,000-$300,000 allocated to water-quality modeling.

EPA identified a wide variety of potential "use" and "non-use" benefits as part of its analysis.[16] Using various economic valuation methods, EPA provided monetary quantifications for seven benefit categories.[17] Approximately 85 percent of the estimated monetary benefits quantified by EPA were attributed to recreational benefits. According to Agency staff, EPA's analysis was driven by what EPA could monetize. EPA focused on those contributions for which data were known to be available for quantification of both the baseline condition and the likely changes stemming from the proposed rule, and for translation of those changes into monetary equivalents.

EPA's final assessment provides only a brief discussion of the contributions to human welfare that it could not monetize. A table in the Executive Summary lists a variety of non-monetized contributions [18] but designated them only as "not monetized." EPA did not quantify these "contributions" in non-monetary terms (e.g., using bio-physical metrics) or present a qualitative analysis of their importance. Instead, it represented the aggregate effect of these "substantial additional environmental benefits" simply by attaching a "+B" placeholder to the estimated range of total monetized benefits. Although the Executive Summary gives a brief description of these "non-monetized" benefits, the remainder of the report devotes little attention to them.

The CAFO economic benefits assessment illustrates a number of limitations in the current state of ecosystem valuation at EPA. First, as noted above, the CAFO analysis did not provide the full characterization of ecological contributions to human welfare using quantitative and qualitative information, as OMB Circular A-4 would appear to require. The report instead focused on a limited set of economic benefits, driven primarily by the ability to monetize these benefits using generally accepted models and existing value measures.[19] These benefits did not include all of the major ecological contributions to human welfare that the new CAFO rule would likely generate.[20] The circular requires that an assessment identify and characterize all of the important benefits of a proposed rule, not simply those that can be monetized. In this case, the monetized benefits alone exceeded the cost of the rule and hence the focus on benefits that could be readily monetized did not affect the outcome of the regulatory review. However, in a different context an economic benefit assessment based only on easily monetized benefits could inadvertently undermine support for a rule that would be justified based on a more inclusive characterization of contributions to human welfare.

Second, the monetary values for many of the economic benefits were estimated through highly leveraged benefit transfers that often were based on dated studies conducted in contexts quite different from the CAFO rule application. [21] This was undoubtedly driven to a large extent by time, data, and resource constraints, which made it very difficult for the Agency to conduct new surveys or studies and virtually forced the Agency to develop benefit assessments using existing value estimates. Nonetheless, reliance on dated studies in quite different contexts raises questions about the credibility or validity of the benefit estimates. This is particularly true when values are presented as point estimates, without adequate recognition of uncertainty and data quality.

Third, EPA apparently did not develop a comprehensive conceptual model of the rule's potentially significant ecological effects. The report presents a simple conceptual model that traces outputs (a list of pollutants in manure – Exhibit 2-2 in the CAFO report) through pathways (Exhibit 2-1) to environmental and human health effects. [22] This model provided useful guidance, but was not sufficiently comprehensive to assure identification of all possible significant ecological effects. A conceptual model of the relevant ecosystem(s) at the start of a valuation project, as discussed in section 3.1, can help in identifying not only important primary effects but also important secondary effects – which frequently may be of greater consequence or value than the primary effects. [23]

Fourth, the CAFO analysis demonstrates the challenges of conducting required economic benefit assessments of ecological protection at the national level. [24] National rule making inevitably requires EPA to generalize away from geographic specifics, in terms of both ecological responses to policy options and associated values. It is, however, possible (and desirable) to use existing and ongoing research at local and regional scales to conduct intensive case studies (e.g., individual watersheds, lakes, streams, estuaries) in support of the national-scale analyses.[25]

Fifth, although EPA invited public comment on the draft CAFO analysis as required by Executive Order 12866, there is no indication in the draft CAFO report that the Agency consulted with the public for help in identifying, assessing, and prioritizing the effects and values addressed in its analysis. Nor is there discussion in the final CAFO analysis of any public comments that might have been received on the draft CAFO analysis. Early public involvement can play a valuable role in helping the Agency to identify all of the systems and services affected by proposed regulations and to determine the regulatory effects that are likely to be of greatest value.

Sixth, EPA did not conduct a peer review of the benefit estimates used in the analysis of the CAFO rule. While the Agency appropriately emphasized peer review in its analysis and report, EPA did not seek peer review in deriving benefit estimates for the CAFO rule. Once again, this shortcoming is undoubtedly a function of time and resource constraints. However, peer review, especially early in the process, could help EPA staff identify relevant and available data, models, and methods to support its valuation efforts. An effective method would be to review not only individual components of an analysis (e.g., watershed modeling, air dispersal, human health, recreation, and aesthetics) but also the overall conceptual model and analytic scheme as well.

Finally, EPA's analysis and report closely adhered to the requirements of Executive Order 12866. Although the Executive Order provided the proximate reason for preparing the analysis and report, the Agency did not have to limit itself to the goals and requirements contained therein. The Executive Order does not preclude EPA from adopting broader goals and hence conducting other analyses in addition to the required benefit-cost analysis. Assessments such as the CAFO study can serve many purposes, including helping to educate policy makers and the public more generally about the economic benefits and other values that stem from EPA regulations. It is important for EPA to recognize this broader purpose and to regularly consider it.

2.3. An integrated and expanded approach to ecosystem valuation: key features

The CAFO example highlights a number of limitations to the current state of ecosystem valuation at EPA. The committee's analysis points to the need for an expanded, integrated approach to valuing the ecological effects of EPA actions. This approach focuses on the effects of greatest concern to people and integrating ecological analysis with valuation. The remainder of this chapter describes the approach to ecological valuation developed and endorsed by the committee. The approach should serve as a guide to EPA staff as they conduct RIAs and seek to implement Circular A-4, as well as in decisions on regional and local priorities and activities. Subsequent chapters provide a more detailed discussion of the implementation of the approach..

The committee recommends that, when conducting ecological valuation, the Agency use a valuation process that has three key, interrelated features:

- Early consideration of effects that are socially important
- Prediction of ecological responses in value-relevant terms
- Consideration of the possible use of a wider range of valuation methods to provide information about multiple types of values

2.3.1. Early consideration of effects that are socially important

The first key component of the proposed approach is the early identification and prediction of the ecological responses that contribute to human welfare and are likely to be of greatest importance to people, whether or not the contributions are easily measured, monetized, or widely recognized by the public. These could include ecosystem responses that people value directly or the resulting responses in the services provided by the ecosystem. The importance of a given response will depend on both the magnitude and bio-physical importance of the effect and on the resulting importance to society. Early in the valuation process EPA needs to obtain information about the ecosystem services or characteristics that are of greatest concern, so that efforts to quantify and characterize values can focus on the related ecological response.

Identifying socially relevant effects requires a systematic consideration of the many possible sources of value from ecosystem protection and an identification of the values that may be relevant to the particular policy under consideration. Such a systematic consideration will likely lead to expanding the types of services to be characterized, quantified, or explicitly valued. Previous valuation assessments have often focused on what can be measured relatively easily, rather than what is most important to society. This can diminish the relevance, usefulness, and impact of the assessment.

An obvious question is how to assess the likely importance of different ecological responses prior to completion of the valuation process. A main purpose of conducting a thorough valuation study is to provide an assessment of the importance of ecological responses to different policy options. Nonetheless, in the early stages of the process, preliminary indicators of likely importance can serve as screening devices to provide guidance on the types of responses that are likely to be of greatest concern. EPA can obtain relevant information in a variety of ways. These range from in-depth studies of people's mental models and how their preferences are shaped by their conceptualization of ecosystems and ecological services, to more standard survey responses from prior or purpose-specific studies. In addition, early public involvement [26] or the use of focus groups or workshops, composed of representative individuals from the affected population and relevant scientific experts, can help identify ecological responses of concern.

In identifying what matters to people, it is important to bear in mind that people's preferences depend on their understandings of causal processes and relationships and the information at hand. As noted previously, people's expressions of what is important or of the tradeoffs they are willing to make can change with the amount, the manner, and the kind of information provided. Collaborative interaction between analysts and public representatives can help to ensure that respondents have sufficient information when expressing views and preferences. In fact, EPA can use the ecological valuation process as a mechanism for increasing and augmenting public discourse about ecosystem services and how EPA actions affect those services, thereby narrowing the gap between expert and public knowledge of ecological effects.

The committee's approach to valuation envisions consideration of a broader set of ecological effects, but in most cases, the purpose of the ecological valuation is to help answer specific questions that the Agency faces. Often, the analyses do not necessarily have to be complete to provide the information needed to answer a particular question. For example, a state agency partnering with EPA must decide about whether to allow logging at a particular site. An analysis focused solely on the recreational value of the unharvested site shows that these values alone exceed the net commercial value of logging. Therefore, the agency can conclude that logging will lead to a net social loss without valuing other ecological effects of logging. Of course, the converse is not true. If the recreation value is less than the net commercial value of logging, the agency cannot conclude that logging would lead to a net social gain. However, if the sole purpose of the valuatione exercise is to determine whether the logging would generate a net social gain or loss and tht determination can be based on a subset of values, then it would be unnecessary to expend a large effort to analyze the full suite of values..

2.3.2. Predicting ecological responses in value-relevant terms

The second major component of the C-VPESS process is to predict ecological responses in terms relevant for valuation. This should begin with a conceptual model, followed by quantification (where possible) using specific ecological and related models. It requires both the prediction of bio-physical responses to EPA actions and the mapping of those responses into effects on ecosystem services or features that are of direct concern to people – first conceptually and then quantitatively. Ideally, this would be done using an ecological production function that is specified and parameterized for the ecosystem and associated services of relevance.

Numerous mathematical models of ecological processes and functions are available. These models cover the spectrum of biological organization and ecological hierarchy (e.g., individual level, population level, community level, ecosystem level, landscape level, and global biosphere). In principle, models can provide quantitative predictions of ecological responses to a given EPA action at different temporal and spatial levels. Some models are appropriate for specific contexts, such as particular species or geographic locations, while others are more general.

Ecological models provide a basis for estimating the ecological changes that could result from a given EPA action or policy (e.g., changes in net primary productivity or tree growth, bird or fish assemblages) and the associated changes in ecosystems or ecosystem services. However, many have been developed to satisfy research objectives and not EPA policy or regulatory objectives. Using these models to assess the contributions of EPA actions to human welfare thus poses challenges..

The first challenge is to link existing models with Agency actions that are intended to control chemical, physical, and biological sources of stress. The valuation framework outlined here requires an estimation of the bio-physical responses to a specific EPA action. To be used for this purpose, ecological models must be linked to information about stressors. This link is often not a key feature of ecological models developed for research purposes. Existing models may need to be modified or new models developed to address this need.

Ecological models also need to be appropriately parameterized for use in policy analysis. Numerous ecological studies have been conducted at various levels, for example, at Long-Term Ecological Research Sites (Farber et al., 2006). These might provide a starting point for parameterizing policy-relevant models. A key challenge is to determine whether and to what extent parameters estimated from a given study site or population can be transferred for use in evaluating ecological changes at a different location, time, or scale. In many cases, data do not currently exist to parameterize existing models for use in assessing EPA's actions. Such data may need to be developed before the Agency can use these models fully. To the extent that transferable models and parameter estimates exist, a central repository for this information would be extremely valuable.

The final, but perhaps most important, challenge is translating the responses predicted by standard ecological models into responses in ecosystem services or features that can then be valued. If adapted properly, ecological models can connect material outputs to stocks and services flows (assuming that the services have been well-identified). Providing the link between material outputs and services involves several steps. These steps include: identifying service providers; determining the aspects of ecological community structure that influence function; assessing the key environmental factors that influence the provision of services; and measuring the spatial and temporal scales over which services are provided (Kremen, 2005). However, most ecological models currently are not designed with this objective in mind. In particular, they do not predict bio-physical responses to stressors in ways that lay individuals can understand or that directly link to value.

2.3.3. Use of a wider range of valuation methods

Given predicted ecological responses, the value of these responses needs to be characterized and, when possible, measured or quantified. As noted above, a variety of valuation methods exist. To date, economic valuation methods have been the mainstay of ecological valuation at EPA, not only in the context of national rule making (as required by OMB Circular A-4) but also in decision contexts not governed by OMB guidance. The committee's approach envisions drawing on a wider range of methods than EPA has typically used to capture a broader array of values. It recognizes that there are many sources and types of value and many valuation methods. Different methods provide different ways of characterizing information about values, and multiple methods may be needed to sufficiently capture all types or sources of value.

A key tenet of the valuation process proposed by the committee is consideration of both economic valuation methods and other valuation methods. The suite of methods used should vary with the specific policy context because of differences in information needs, legal and regulatory requirements, the underlying sources of value being captured, data availability, and methodological limitations.

In the context of national rule making, non-economic valuation methods may be useful in three possible ways. The first is to help in identifying the ecological responses that people care about. For example, surveys in which individuals indicate the importance of different environmental and other concerns or small focus groups might provide information about the ecological effects of a specific rule that are likely to be viewed as important.

Second, some non-economic methods could provide a proxy for an economic benefit that the Agency cannot monetize using economic valuation. For example, metrics that are primarily bio-physical or social-economic indicators of impact, such as acres of habitat restored or the number and characteristics of individuals or communities affected, can serve as proxies for at least some contributions of ecosystem protection to human welfare. As noted earlier, OMB Circular A-4 requires that benefits be quantified when they cannot be monetized; bio-physical or social-economic metrics provide potentially useful forms of quantification in such circumstances. Although they would not provide full information about the magnitude of benefits, they might be expected to generally correlate with benefits. Thus, when properly chosen, higher levels of a particular bio-physical or socio-economic metric would signal higher benefits.

Third, non-economic methods could be used to provide supplemental information (outside the strict benefit-cost analysis) about types of value that might not be reflected in benefit measures that come from economic valuation, such as moral or spiritual values. This is consistent with the EPA's call in its *Ecological Benefits Assessment Strategic Plan* for exploring supplemental approaches to valuation. Even if not part of a formal benefit-cost analysis, information about non-economic values may be of significant interest to both EPA and the public.

Circular A-4 dictates the role that other valuation methods can play in national rule makings. However, in other contexts not governed by legislative or executive rules, the use of particular valuation methods is less prescribed, and hence, if desired, non-economic valuation methods can play a more central role. There may be more scope in particular for exploring the use of methods that are relatively novel and in the developmental stage.

Regardless of the decision context, in all cases, only valuation methods that meet appropriate validity and related criteria should ultimately be used. Section 4.1 provides a discussion of criteria for assessing validity. The validity of some methods has already been subjected to considerable scrutiny and the strengths and weaknesses of these methods are fairly well understood. For methods that are still in the developmental stage, exploration of the method's potential should include an assessment of the validity of the method using a scientifically based set of criteria.

The use of an expanded suite of methods could allow EPA analyses to better capture the full range of contributions stemming from ecosystem protection and the multiple sources of value derived from ecosystems. In addition, information regarding the similarities or differences in estimates stemming from alternate assessment methods of the value of an ecological change could be an important input into a policy decision. For example, the use of multiple methods to characterize the same underlying value can, in some cases, increase the confidence that decision makers, policy makers, and the public have in the value estimates. However, this form of validity check requires conceptual consistency across methods. In addition, because most ecological valuation methods are resource-intensive, providing multiple estimates of the same value may not always be feasible, or in some cases even a wise use of funds, given resource constraints.

Using multiple methods to estimate values raises the question of aggregation across methods. Values cannot be aggregated across methods that yield value estimates in different units. However, even when units are comparable (e.g., both methods yield monetary estimates of value), aggregation across methods may not be appropriate. One reason is the potential for double counting. As noted previously, the concepts of value outlined in Table 1 are not mutually exclusive. Therefore, adding estimates of different concepts of value from different valuation methods could lead to double counting. Also, because of their different assumptions, different methods can measure quite different things and yield values that are conceptually different and hence not comparable. As a result, simple aggregation across methods is generally not scientifically justified. For example, it would be conceptually inconsistent to add monetary value estimates obtained from an economic valuation method and monetary estimates obtained from a deliberative process in which preferences are constructed, because the two are not based on the same underlying premises. Nonetheless, information about both estimates of value may be of interest to policy makers. In such cases, value estimates should be reported separately rather than aggregated across methods. This is consistent with the suggestion above that, in the context of national rule makings where benefit assessments are conducted under Circular A-4, information about non-economic values should be considered separately (as supplemental information) rather than "added to" the economic benefit estimates to obtain a measure of total value.

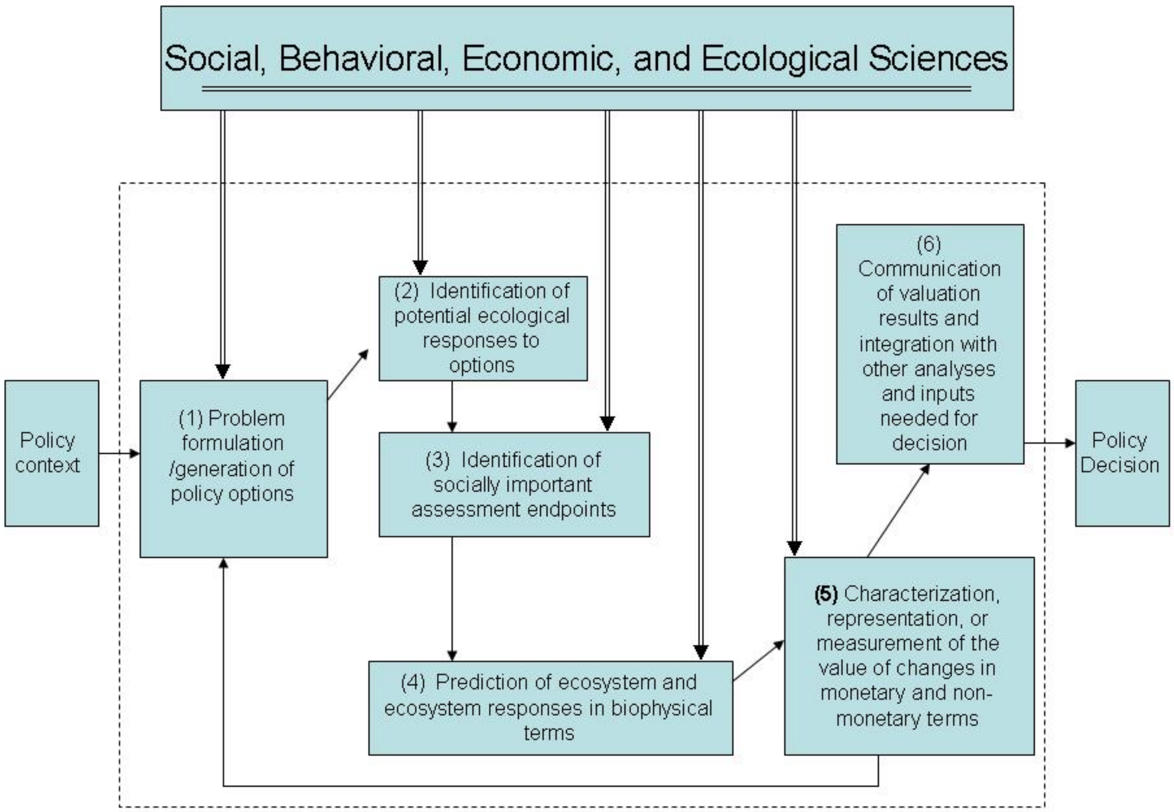
2.4. <u>Steps in implementing the proposed approach</u>

The previous section provides an overview of an integrated and expanded approach to ecological valuation proposed by the committee. The process for implementing the proposed framework would involve the following steps, depicted in Figure 1:

1. Formulating the valuation problem and choosing policy options to be considered, given the policy context
2. Identifying the significant bio-physical responses that could result from the different options
3. Identifying the responses in the ecosystem and its services that are socially important
4. Predicting the responses in the ecosystem and relevant ecosystem services in biophysical terms
5. Characterizing, representing, or measuring the value of responses in the ecosystem and its relevant services in monetary or non-monetary terms
6. Communicating results to policymakers for use in policy decisions

Figure 1 depicts these steps as sequential. However, in practice numerous feedbacks should occur with interactions and iterations across steps. For example, information about the value of responses in ecosystem services to a given set of policy options might cause a reformulation of the problem or identification of new policy options that could be considered. Also, a projected bio-physical effect might suggest human-social values that were not initially considered.

Figure 1: Process for implementing an expanded and integrated approach to ecological valuation



As depicted in Figure 1, the implementation of the approach is contingent upon the specific policy context and intended to provide input for a particular policy decision. As noted above, ecological valuation can play a key role in a number of different decision contexts, including national rule making and regional or local decisions regarding priorities and actions. The valuation problem should be formulated within the specific EPA decision context. Different contexts will generally be governed by different laws, principles, mandates, and public concerns. These contexts can differ not only in the required scale for the analysis (e.g., national vs. local) but possibly also in the type of valuation information that may be needed. For example, in contexts requiring an economic benefit-cost analysis, benefits need to be monetized whenever possible. In contrast, expressing contributions to human welfare in monetary terms might be of little or no relevance to EPA analysts in other contexts. The policy context therefore influences the appropriateness of data, models, and methods.

Figure 1 also highlights the need for information and input from a wide range of disciplines at each step of the process, beginning with problem formulation and the identification of the ecosystem responses that matter and continuing through the valuation of those responses. Instead of ecologists working independently from economists and other social scientists, experts in those disciplines should collaborate throughout. Ecological models need to be developed, modified, or extended to provide usable inputs for value assessments. Likewise, valuation methods and models need to be developed, modified, or extended to address important ecological and bio-physical effects that may be underrepresented in value assessments.

Figure 1 suggests a structure that in many ways parallels the Agency's Framework for Ecological Risk Assessment (EPA Risk Assessment Forum, 1992; EPA Risk Assessment Forum, 1998). This framework underlies the ecological risk guidelines developed by EPA to support decision making intended to protect ecological resources (EPA Risk Assessment Forum, 1992). Ecological valuation is a complement to ecological risk assessment. Both processes begin with an EPA decision or policy context requiring information about ecological effects. Next follows a formulation of the problem and an identification of the purpose and objectives of the analysis, as well as the policy options that will be considered. In addition, both ecological risk assessment and ecological valuation involve the prediction and estimation of possible ecological responses to an EPA action or decision. They also both ultimately use this (and related) information in the evaluation of alternative actions or decisions.

Although they are similar, ecological valuation goes beyond ecological risk assessment in an important way. Risk assessments typically focus on predicting the magnitudes and likelihoods of possible adverse effects on species, populations, and locations, but do not provide information about the societal importance or significance of these effects. In contrast, ecological valuation seeks to characterize the importance to society of predicted ecological effects by providing information on either the value that society places on ecological improvements or the loss it experiences from ecological degradation. By incorporating human values, ecological valuation is closer to risk characterization than risk assessment. Many of the principles that should govern risk characterization outlined in the 1996 National Research Council Report *Understanding Risk: Informing Decisions in a Democratic Society* pertain to ecological valuation as well. For example, both should be the outcome of an analytical and transparent process that incorporates not only scientific information but also information from the various interested and affected parties about their concerns and values.

2.5. Conclusions and recommendations

Ecosystems provide a wide array of services that directly or indirectly support or enhance human populations. People also can value them in their own right for reasons stemming from ethical, religious, cultural, or biocentric principles. EPA's broad mission to protect human health and the environment includes the protection of ecosystems.

Many EPA actions affect the state of ecosystems and the services derived from them. To date, ecological valuation at EPA has focused primarily on a limited set of contributions to human well-being from ecological protection. This stems primarily from the difficulty of predicting the responses of ecological systems and services to EPA actions and the difficulty of quantifying, measuring, or characterizing the resulting contributions to human welfare and associated values. The presumption that contributions need to be monetized in order to be carefully characterized also restricts the range of ecological effects that are typically considered in EPA analyses, particularly at the national level.

EPA's current efforts to improve its ability to value ecological systems and services is very important and timely. The committee recommends that the Agency take the following steps.

- EPA should cover an expanded range of important ecological effects and human considerations using an integrated approach. Such an approach should:
    - Identify early in the process the ecological responses or contributions to human welfare that are likely to be of greatest importance to people and focus valuation efforts on these responses. This would likely expand the range of ecological responses that are valued, recognizing the many sources of value.
    - Predict ecological responses in value-relevant terms. To do so, the valuation process should highlight the concept of ecosystem services and provide a mapping from responses in ecological systems to responses in services or ecosystem components that can be directly valued by the public.
    - Allow for the use of a wider range of possible valuation methods to provide information about multiple types of values. EPA should evaluate methods using a scientifically-based set of criteria and apply methods where legally permissible in a manner that is consistent with their conceptual foundations.

- Valuation analyses should involve, from the beginning and throughout, an interdisciplinary collaboration among physical/biological and social scientists, as well as input about public concerns.
- Beyond the use of economic valuation methods (the current mainstay of ecological valuation at EPA), EPA should experiment with the use of other valuation methods. EPA has not used these other methods for ecological valuation in any significant way in the past. Because some are still in the developmental stages, the committee believes that it would be wise for the Agency to experiment with the use of these other methods in different valuation contexts. In the context of national rule making, the Agency should conduct one or two model analyses (perhaps one prospective and one retrospective) of how the use of a wider range of methods might be applied. This experience could then guide the Agency's valuation efforts as it conducts subsequent benefit assessments. In addition, the Agency should experiment in local and regional decision contexts, which are less prescriptive, and noneconomic methods could play a more central role.

Through the use of the expanded and integrated valuation framework recommended in this report, EPA can move toward greater recognition and consideration of the effects that its actions have on ecosystems and the services they provide. This will allow EPA to improve environmental decision making at the national, regional, and site-specific levels and contribute to EPA's overall mission regarding ecosystem protection. EPA can also better use the ecological valuation process to educate the public about the role of ecosystems and the value of ecosystem protection. The remainder of this report discusses in more detail how to implement the ideas embodied in the C-VPESS integrated value assessment approach.

3 Building a foundation for ecological valuation: predicting effects on ecological systems and services

Chapter 2 presented an overview of an integrated and expanded approach to valuing ecological responses to EPA actions or decisions. This chapter focuses on one part of that approach: predicting ecological responses in value-relevant terms. In every context where the need for valuation arises, information about the magnitude of ecological effects will be a key component of value assessment. No matter what valuation method is used, the valuation process first requires an assessment of the responses of ecosystems and ecosystem services to the relevant EPA action or decision. Even where valuation is not possible, an assessment of ecosystem and ecosystem responses can provide valuable information to decision makers and the public. OMB in Circular A-4, for example, provides that, where a benefit cannot be expressed in monetary terms for a major national rule making, EPA "should still try to measure the benefit in its physical units."

This chapter begins with a discussion of the importance of developing an initial conceptual model of the relevant ecosystem and its services that can guide the entire valuation process. Section 3.2 discusses the steps needed to estimate the response of ecosystem and ecosystem services to EPA actions or decisions, including the key importance of ecological production functions. Section 3.3 highlights the challenges that currently exist in trying to implement ecological production functions in specific contexts. These challenges include understanding and modeling the relevant ecology, identifying the relevant ecosystem services, and mapping ecological responses into changes in the relevant ecosystem services. To a large extent, these challenges stem from the underlying complexity and site-specificity of ecosystems. Section 3.4 discusses the strategies for evaluating the effects of EPA actions on ecosystem services in the absence of a comprehensive ecological production function. Section 3.5 examines the problem of data availability and conditions where transfer of ecological information might be appropriate. Section 3.6 briefly addresses the importance of new ecological research to support valuation efforts. Finally, section 3.7 summarizes the committee's conclusions and recommendations.

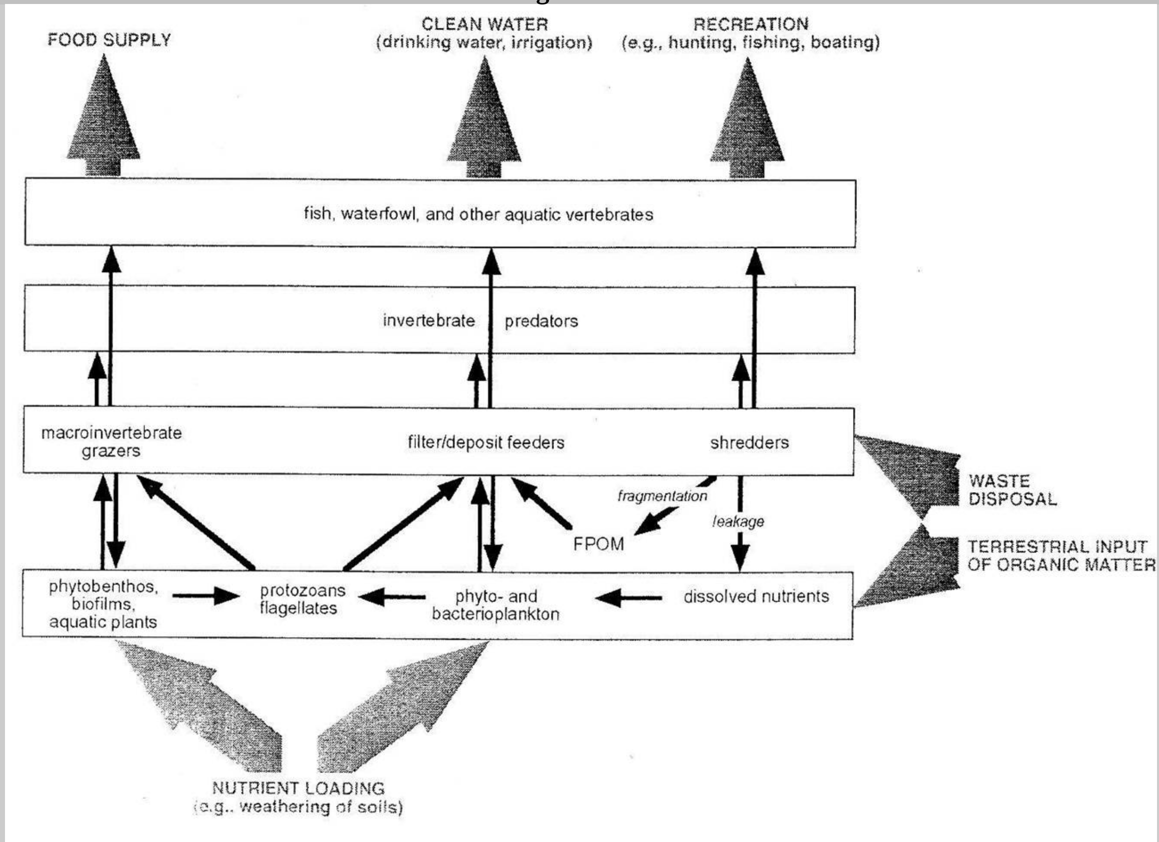
3.1. The road map: a conceptual model

The key first step in predicting the effects of EPA actions and decisions on ecological systems and services is the formulation of a conceptual model of the relevant ecosystem(s) and its associated services that can guide the valuation effort. The committee recommends that EPA start each ecological valuation by developing such a model. Because the purpose of the model is to guide the valuation process, the model should be context-specific and constructed at a general level. The conceptual model should diagram, using boxes and arrows, the predicted relationships among the relevant EPA actions, affected ecosystems, and associated services. The conceptual model is fundamentally a decision tool to help characterize and predict the ecological and social consequences of the relevant EPA actions and thereby help guide the full valuation process.

Later in the valuation process, EPA will need to use ecological production functions to generate more detailed analyses of key interactions, specific ecological responses to EPA decisions or actions, and resulting consequences to ecosystem services using ecological production functions. As discussed in section 3.3, these analyses will typically require the use of appropriately scaled and parameterized ecological models with a narrower focus. The conceptual model provides a framework for planning for the use of these predictive models at the start of the process and for integrating the more specific analyses into the overall valuation exercise.

The conceptual model should clearly identify the relevant functional levels of the ecosystem, the interrelationships among ecosystem components, and how they contribute to the provision of ecosystem services, either directly or indirectly. Figure 2 provides an example illustrating some aspects of ecosystem services related to nutrient pollution, adapted from Covich et al. (2004).

Figure 2: Illustration from Covich et al., 2004, Showing Relationships of Major Functional Types to Ecological Services



As Figure 2 highlights, the conceptual model should include both information about the underlying ecology and a link to ecological services that are of importance to society. The conceptual model, for example, should include: the impacts of environmental stressors, such as waste disposal, on organisms at different trophic levels; key interactions among species at different levels; and changes at different levels that affect ecological services, such as the food supply, clean water, or recreation.

Not surprisingly, ecologists often focus on underlying ecological relationships (depicted in the lower part of Figure 2), and valuation experts tend to focus on the later, value-oriented stages of the process, starting with ecosystem services (shown at the top of the figure). A key principle of this report is the need to consider and integrate both aspects of the process. For ecological valuation aimed at improved decision making, a detailed analysis of ecological responses is insufficient unless those responses are mapped to responses in ecosystem services or system components that can be valued. Valuation exercises that do not reflect the key ecological processes and functions are similarly insufficient. Both parts of the valuation process are essential. The development of a conceptual model at the outset of the valuation process can help ensure that the process is guided by this basic principle.

The development of the conceptual model is a significant task that deserves the attention of EPA staff throughout the Agency, experts in relevant topics from the bio-physical and social sciences, and the public. Involving all constituents, including the public, at this stage will enhance transparency, provide the opportunity for more input and better understanding, and ultimately give the process greater legitimacy. Participatory methods such as mediated modeling, described in section 5.1, can play a valuable role in the development of the conceptual model. To promote transparency and understanding, the conceptual model, the process for developing and completing it, and the decisions embedded in it should also be part of the formal record.

The conceptual model should allow for iteration and possible model changes and refinement over time. For example, analysts may initially believe that an action at a local site has local ecological effects, but, on further analysis of the stressors, realize that effects reach to more distant regions downstream or downwind, requiring a change in the conceptual model. Similarly, analysis of the relevant ecological system may show that stressors originally considered insignificant should be added to the conceptual model. As an example, a relatively non-toxic chemical effluent, normally seen as insignificant, might become significant if it is determined that low stream flows or intermittent streams effectively increase the concentration of the chemical to toxic levels during some parts of the year. The need for iterative model changes and refinements is critical and should be part of all valuation efforts.

3.2. The important role of ecological production functions in implementing the conceptual model

While the conceptual model serves as a guide for the overall valuation process, the individual components and linkages embodied in that model must be operationalized. The goal is to provide, to the extent possible, quantitative estimates of the responses of ecosystem components or services that can then be valued. Operationalizing the conceptual model requires mapping or describing:

1. How the relevant EPA action will affect the ecosystem

2. How the effects on the ecosystem will, in turn, affect the provision of ecosystem services
3. How people value that ecosystem service response.

The third step, valuation, is the subject of chapter 4. The remainder of this chapter considers how to implement the first two steps, estimating how the EPA actions will affect the ecosystem, and how the ecosystem response will affect ecosystem services.

The first step requires describing how the EPA action – by reducing or eliminating a stressor or by otherwise protecting or altering an environmental factor – will affect important aspects of ecosystem structure or function. Would a stressor that EPA can eliminate otherwise cause a species to disappear or change in abundance? Would the stressor result in a change in biogeochemistry? For any important effects, EPA should make a quantitative estimate.

The ecological production function is a critical tool for implementing the second step – estimating how the ecological response will affect the provision of ecosystem services. Ecological production functions are similar to the production functions used in economics to define the relationship between inputs (e.g., labor, capital equipment, raw materials) and outputs of goods and services. Ecological production functions describe the relationships between ecological inputs and outputs, i.e., between the structure and function of ecosystems and the provision of various ecosystem services. These functions capture the biophysical relationships between ecological systems and the services they provide, as well as the inter-related processes and functions, such as sequestration, predation, and nutrient cycling. Coupled with information about how alternative EPA actions or management scenarios will affect the ecological inputs, ecological production functions can be used to predict the effects of the actions or scenarios on ecosystem services.

Ecological production functions could describe the relationship between a broad suite of inputs and ecosystem services. An ecological production function could describe the relationship between inputs for an individual service or, to the extent that two or more services are linked (e.g., produced jointly or in competition), a multiple-output function could capture these linkages.

The analogy between ecological production functions and economic production functions is not perfect. Economic production functions generally involve inputs over which humans have direct control, and the relationship between inputs and outputs is frequently well studied and defined. Ecological production functions, by contrast, involve inputs over which humans have variable and often limited control, and the relationship between inputs and outputs is complex and often very uncertain. Nonetheless, economic production functions provide a useful analogy for the type of relationships and models needed in order to effectively estimate the effect of EPA actions or scenarios on ecosystem services of importance to the public.

Scientists are making rapid progress in understanding and defining ecological production functions for certain ecosystem services. One such service is pollination. Animal pollination is essential for the production globally of about one-third of agricultural crops and the majority of plant species (Kremens et al., 2007). Ecologists have recently built spatially explicit models incorporating land use and its effect on habitat and foraging behavior of pollinators (Kremens et al., 2007). Such models can link changes in ecosystem conditions to the level of pollination of agricultural crops and their yields. Empirical studies using such models have shown the effects of proximity to natural forest on coffee productivity (Ricketts et al., 2004) and the interaction of wild and honey bees on sunflower pollination (Greenleaf and Kremens, 2006).

A second ecosystem service for which considerable progress has been made in developing ecological production functions is carbon sequestration. Agricultural systems, forests, and other ecosystems contain carbon in soil, roots, and above-ground biomass. Rapidly growing markets for carbon sequestration and the potential for generating carbon credits are pushing interest in accurately assessing the carbon sequestration potential of agricultural and other managed ecosystems (Willey and Chamaides, 2007). It is possible to fairly accurately quantify above-ground carbon stores in various types of ecosystems such as (e.g., Birdsey, 2006; Smith et al., 2006; EPA Office of Atmospheric Programs, 2005), but greater uncertainty remains about stocks of soil carbon that make up the majority of carbon in agricultural and grassland systems (e.g., Antle et al., 2002, EPA Office of Atmospheric Programs, 2005).

Despite this progress, our current understanding of ecological production functions for most ecosystem services remains limited (Balmford et al., 2002; Millennium Ecosystem Assessment, 2005; NRC, 2004). Although many ecological models exist, most do not predict ecosystem service responses. The next section discusses some of the challenges in developing complete ecological production function models for use in ecological valuation.

3.3. Challenges in implementing ecological production functions

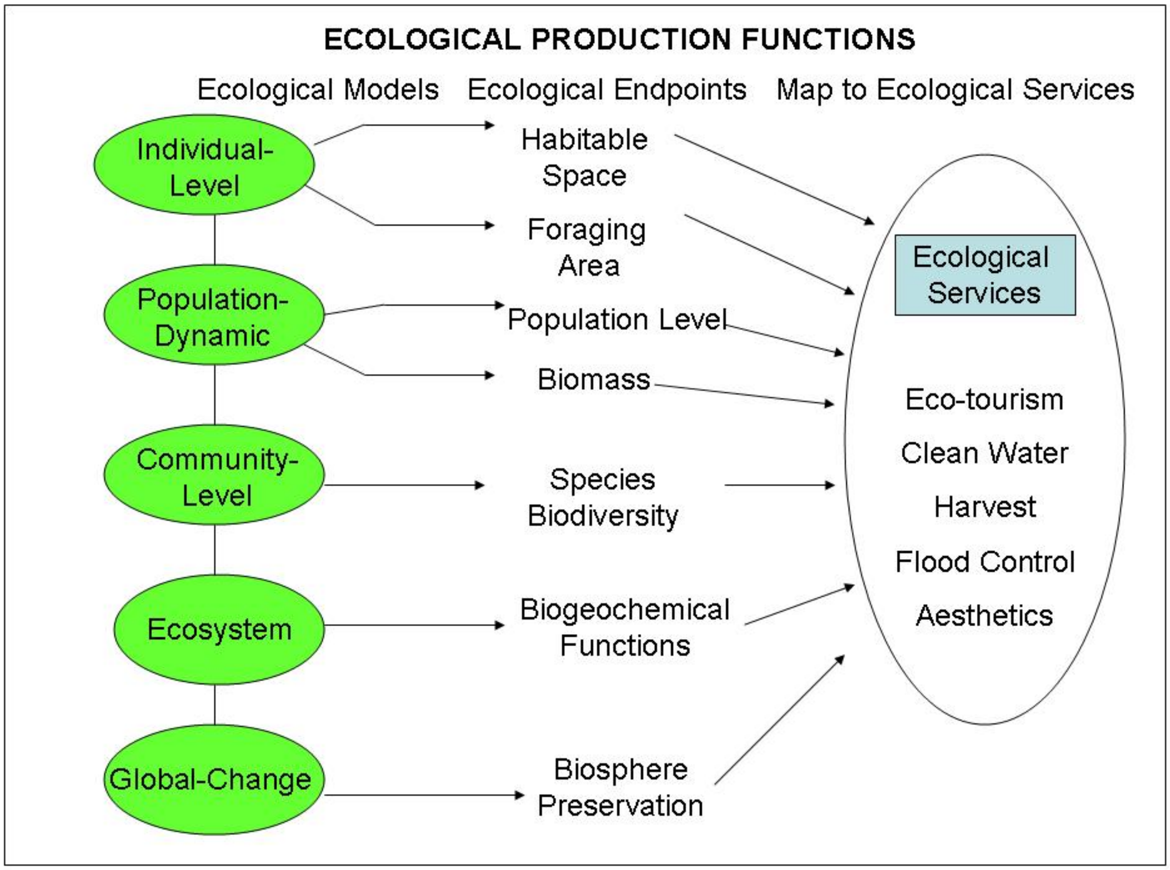
Developing and implementing an ecological production function requires:

- Characterizing of the ecology of the system
- Identifying of the ecosystem services of interest
- Developing a complete mapping from the structure and function of the ecological system to the provision of the relevant ecosystem services

Figure 3 provides a graphical representation of the necessary elements of an ecological production function. On the left side of the figure, ecological models at various organizational levels predict ecological elements or attributes – ecological endpoints – that can be linked to ecosystem services of interest. These ecological models are important components of an ecological production function, but they are not the complete function. An ecological production function requires that the endpoints of these ecological models be mapped or translated into corresponding predictions regarding the ecosystem services of interest.

Each of these three key steps in developing and implementing ecological production functions face challenges that EPA should work to address. This section elaborates on the challenges.

Figure 3: Graphical Depiction of Ecological Production Functions



3.3.1. Understanding and modeling the underlying ecology

As noted, the first step in developing an ecological production function is to understand the components, processes, and functioning of the ecosystem that underlie and generate the ecosystem services. Analysts must have a strong understanding of the underlying ecology. Although much is known about ecological systems, current knowledge is still very incomplete, largely because ecosystems are inherently complex, dynamic systems that vary greatly over time and space.

As an example of the complexity of ecological functions, consider the ecological services associated with the activities of soil organisms that might be affected by disposal of waste on that soil. These organisms thrive on organic matter present or added to the soil. By breaking down that organic matter, certain groups of organisms maintain soil structure through their burrowing activities. These, in turn, provide pathways for the movement of water and air. Other kinds of organisms shred the organic material into smaller units that microbes then utilize. The microbes release nutrients in a form that higher plants can use for their growth or in a dissolved form that can move hydrologically from the immediate site into groundwater or a stream. Other groups of specialized microbes may release various nitrogen gases directly to the atmosphere. The nature of soil organisms and the products that they utilize, store, or release all help to regulate the biogeochemistry of the site, as well as the site's hydrology, productivity, and carbon-storage capacity. Predicting the effect of particular actions on ecosystem services such as waste processing and the provision of clean water requires an understanding of these complex ecological relationships.

Complexity also stems from the fact that ecological effects may persist for different periods of time, affecting both the temporal and spatial scales that are relevant for any analysis. The ecological effects from carbon dioxide in the atmosphere, for example, are likely to persist far longer and require a larger temporal and spatial analysis than the effects from acute toxic exposures to hazardous chemicals.

Because of the complexity of most ecosystems, analysts need ecological models to organize information, elicit the interactions among the variables represented in the models, and reveal outcomes under different sets of assumptions or driving variables. Some models are statistical; others are primarily simulation models. Some statistical and theoretical models are relatively small, containing a few equations. Other ecological models are very large, involving hundreds of interacting calculations. Models may be valuable in many of the steps of assessing ecological value including:

- Estimating stress loading
- Estimating the exposure pattern of stress (especially the spatial and temporal implications)
- Identifying ecological elements receiving exposure
- Estimating the exposure-response function of ecological elements
- Estimating the change in stress from potential Agency actions
- Estimating the response of ecosystem services or functions to change in stress

Ecological models can describe ecological systems and ecological relationships that range in scale from local (individual plants) to regional (crop productivity) to national (continental migration of large animals). These models frequently focus on specific ecological characteristics, such as the populations of one or more species or the movement of nutrients through ecosystems, and can cover the full spectrum of biological organization and ecological hierarchy. For instance, a hydrological model might describe possible changes in the timing and amount of water in streams and rivers. A biogeochemical model might predict effects on the levels of various chemical elements in soils, groundwater, and surface waters. A terrestrial carbon cycle model might project changes in plant growth and in carbon sinks or sources. Population and community models might project changes in specific animal and plant populations of concern.

Inevitably, models suffer from limitations. Although many ecological models are well established and used routinely for describing ecological systems, ecological models can only represent the current state of knowledge about the dynamics of an ecological system and generate outputs only as reliable as the data the models use. The dynamism of a system adds to the challenge of modeling, as does the non-linear response of system components. The model outputs are subject to known, and sometimes unknown, levels of statistical uncertainty. Chapter 5 of this report discusses the issue of uncertainty and how EPA should address uncertainty in its valuation efforts. At the moment, the important point to emphasize is that uncertainty pervades the entire valuation process, including the modeling of ecological processes.

Moreover, no ecological model can include all possible interactions. Some ecological models explicitly or implicitly incorporate human dimensions, but most focus primarily on ecological functions. In addition, models capture historical relationships and typically are not able to predict ecosystem patterns for which no modern counterpart exists. For example, if a stressor such as climate change leads species to "reshuffle into novel ecosystems unknown today" for which there is no analog, current models will not predict the effect (Fox, 2007).

Data insufficiency also frequently constrains the applicability, and to some degree formulation, of ecological models. Even when a full theoretical model of an ecosystem exists, applying the model to a specific context of interest will require determining the parameters of the model for that context. However, parameterization is generally difficult because of the complexity of ecological systems and their dependence on an array of site-specific variables. As a result, many ecological models are site specific. The relatively large amounts of site-specific data required to build and parameterize models means that transferability of the models is limited, either because the model has been developed using spatially constrained data or because inadequate data are available at specific sites with which to drive or parameterize the model. This site-specificity can significantly limit models' applicability to the spatial and temporal complexities required in valuing ecosystem services, especially at regional and national scales.

Ecological models incorporate the best available scientific knowledge of how ecosystems will respond to a given perturbation and the sensitivity of various ecosystem components. The committee therefore recommends that EPA support all of its ecological valuations with ecological models and data sufficient to understand and estimate the likely ecological response to major alternatives being considered by decision makers. Ecological models are essential in representing and analyzing ecological production functions. Guided by the conceptual model described in section 3.1, the Agency should use ecological models to quantify the likely effects of an action on the ecosystem and the resulting effect on ecosystem services.

Given the limitations of many current models, however, the committee also recommends that EPA make developing effective ecological models one of its research priorities. EPA is already strengthening its approach for developing and using models for decision making. For example, EPA has established the Council for Regulatory Environmental Modeling (CREM), a cross-Agency council of senior managers with the goal of improving the quality, consistency, and transparency of models used by the Agency for environmental decision making. The committee endorses this effort and advises EPA to continue to strengthen its work in this area.

Because many ecological models exist and a variety of models might be used for any particular valuation context, the Agency will often be faced with a choice among one or more predictive models. In identifying and choosing models for particular purposes and contexts, primers on ecological theory and modeling such as the *Primer of Ecological Theory* (Roughgarden, 1998b) can provide a useful starting point.

The appropriate choice of models, and the availability and appropriateness of supporting databases, will depend in part on the scale of analysis (e.g., local vs. national) and the precision of the analysis needed for the relevant policy decision.

The committee recommends that EPA identify clear criteria for selecting ecological models and apply these criteria in a consistent and transparent way. Several existing reports discuss the selection and use of models for environmental decision making and can provide valuable guidance to EPA in the valuation context. In 2005, EPA's Council for Regulatory Environmental Modeling prepared a "Draft Guidance on the Development, Evaluation and Application of Regulatory Environmental Models." In 2006, an EPA Science Advisory Board panel reviewed the draft report and provided recommendations on revisions (EPA Science Advisory Board, 2006). Until EPA publishes final guidance, the draft guidance and SAB review can provide EPA with valuable advice in selecting models. A 2007 report of the National Research Council Board on Environmental Studies and Toxicology entitled "Models in Environmental Regulatory Decision Making" also provides valuable guidance on selecting appropriate ecological models for use in valuation exercises. The criteria in these reports and the SAB review can guide the Agency both in selecting among models and in setting priorities for future model development.

These reports address environmental modeling in general and do not focus on the use of ecological models for valuation purposes. For valuation purposes, EPA should use the criteria from these reports and choose models that generate outputs either directly in terms of relevant ecosystem services or that are easily translatable into effects on such services. The ultimate goal is to provide a measure of the value of the effects of an action on ecosystem services. The models chosen must advance that goal.

3.3.2. Identifying ecosystem services

Another key challenge in implementing ecological production functions is identifying the relevant ecosystem services to be evaluated in any given context. As already emphasized, ecological production functions must ultimately link ecological responses to effects on ecosystem services. This requires that EPA identify the relevant services in a consistent and appropriate way.

Identifying the relevant ecosystem services cannot be done deductively. The relevant services depend on what is important to people in the specific context, once they have been informed about potential ecological effects. The objective is to identify what in nature matters to people and to express this intuitively and in terms that can be commonly understood. Technical expressions or descriptions meaningful only to experts are not sufficient; however, underlying ecological science must inform the identification of relevant services. Identifying relevant services requires a collaborative interaction among ecologists, social scientists, the public, and stakeholders.

The Millennium Ecosystem Assessment (2005) provides a good starting point for identifying potentially relevant ecosystem services by providing an extensive discussion and classification of ecosystem services. In each specific context, however, EPA should also seek input from the general public and from individuals or entities particularly affected by the relevant EPA decision as to what is important. In doing so, EPA can use a variety of sources, such as the valuation methods described in chapter 4 (e.g., surveys, individual narratives, mental model research, and focus groups), content analysis of public comments, solicitation of expert opinion and testimony, and summaries of previous decisions in similar circumstances.

Moving toward a common understanding of ecosystem services is important for the success of future valuation efforts. The relative success of EPA efforts to translate air quality problems into human health-related social effects is due in part to the development of agreements about well-defined health outcomes that can be valued. In order to value the health effects of air pollution, it has been necessary to move from describing effects in terms such as oxygen transfer rates in the lung to terms that are more easily understood and valued by the public, such as asthma attacks. Although the search for common health outcomes that can be used for valuation has been difficult, the lesson is clear: If health and social scientists are to productively interact in assessing the economic value of improved environmental quality, measures of health outcomes that are understandable and meaningful to both groups of scientists are necessary. These outcomes are now understood by disciplines as divergent as pulmonary medicine and urban economics (EPA Science Advisory Board, 2002a). The search for common outcomes that can be valued will be equally important in the ecological realm, where biophysical processes and outcomes can be highly varied and complex.

Some authors have advocated the development of a common list of services to be collectively debated, defined, and used by both ecologists and social scientists across contexts (e.g., Boyd and Banzaf, 2007). Such a list might include:

- Species populations – including those that generate use value, such as harvested species and pollinator species, and those that generate existence values
- Land cover types – such as forests, wetlands, natural land covers and vistas, beaches, open land, and wilderness
- Resource quantities – such as surface water and groundwater availability
- Resource quality – such as air quality, drinking water quality, and soil quality
- Biodiversity

Although only a subset of the services on a common list might be relevant in any particular context, the list would provide some standardization in the definition of ecosystem services across contexts. Advocates argue that development of a common list is the best way to debate and convey a shared mindset, foster the integration of biophysical and social approaches, and provide greater transparency, legitimacy, and public communication about what in nature is being gained and lost. Achieving agreement on a common list might be an important goal, but it is likely to be difficult for complex ecological systems. Converging prematurely on a limited list of services could misdirect valuation efforts and miss important intermediate and end services.

To ensure that the services can be readily and accurately valued, the identification of relevant ecosystem services, either as a common list or for a specific analysis, should follow some basic principles. First, it is important to avoid double counting. All things that matter should be counted, but only once.[27] Second, the ecosystem services should have concrete outcomes that can be clearly expressed in terms that the public can understand. If ecological outcomes are to provide useful input into valuation, they must be described in terms that are meaningful to those whose values are to be assessed.

EPA has launched several initiatives to develop common and useful endpoints for ecological models. These endpoints, however, are typically not themselves ecosystem services. The endpoints instead are often ecological attributes or elements, such as biomass, that serve as inputs to the production of ecosystem services. Although these endpoints often link to the Agency's statutory responsibilities and policy concerns, social scientists typically cannot use them by themselves to value effects on ecosystem services. Looking at Figure 3, social scientists need information on the ecosystem services at the right side of the diagram.

Most endpoints, shown in the center column of Figure 3, are at least one step removed and must still be translated into responses in ecosystem services.

EPA's generic ecological assessment endpoints (GEAEs) (EPA Risk Assessment Forum, 2003) provide a valuable example. The GEAEs are based on legislative, policy, and regulatory mandates. If expanded to include landscape-, regional-, and global-level endpoints (see EPA Risk Assessment Forum, 2003, Table 4.1; Harwell et al., 1999; EPA Science Advisory Board 2002b), they can serve as a first step in characterizing relevant ecological systems and quantifying responses to stressors. Although the GEAEs are a valuable starting point, they also illustrate how far EPA must go in estimating responses in ecosystem services. First, the GEAEs are expressed in technical terms and not in terms of concrete outcomes that the public can understand. These technical terms are certainly appropriate for some regulatory purposes, but most of the public is unlikely to be familiar with them. Therefore, they will have limited use in valuation.

Second, the GEAEs do not necessarily reflect the things in nature that people care about. Although the endpoints reflect policy and regulatory needs (EPA Risk Assessment Forum, 2003, p.5), they depict a narrow range of ecological outcomes, confined to organism, population, and community or ecosystem effects. They do not relate to water availability, aesthetics, or air quality, but rather to kills, gross anomalies, survival, fecundity and growth, extirpation, abundance, production, and taxa richness. These effects are clearly relevant to biological assessment. However, for anglers who care about the abundance of healthy fish in a particular location at a particular time, lost value depends not on the number of kills or anomalies but rather on the abundance of healthy fish.

Another important ecological endpoint initiative is EPA's Environmental Monitoring and Assessment Program (EMAP). Created in the early 1990s, EMAP is a long-term program to assess the status and trends in ecological conditions at regional scales (Hunsaker and Carpenter 1990; Hunsaker, 1993; Lear and Chapman, 1994). Once again, the endpoints developed in EMAP are generally not direct measures of ecosystem services. EMAP does, however, emphasize the importance of developing endpoints that are understandable and useful to decision makers and the public. As EPA has recognized, if an endpoint is to serve as a useful indicator of ecological health, it "must produce results that are clearly understood and accepted by scientists, policy makers, and the public" (Jackson et al., 2000). One study that used focus groups to examine the value of EMAP endpoints as indicators of environmental health similarly concluded that there is a need "to develop language that simultaneously fits within both scientists' and nonscientists' different frames of reference, such that resulting indicators [are] at once technically accurate and understandable" (Schiller et al., 2001). The committee agrees with this conclusion and urges EPA to move further toward this goal.

The Agency is aware of the limitations of current endpoints. The committee emphasizes the limitations for two reasons: to highlight the difference between the Agency's current approach to defining relevant ecological endpoints and the need to identify effects on ecosystem services; and to encourage the Agency to move toward identifying and developing of measures of ecosystem services that are relevant and directly useful for valuation.

The identification of relevant ecosystem services will require increased interaction between natural and social scientists within the Agency. The committee urges the Agency to foster this interaction through a dialogue related to the identification and development of measures of ecosystem services. One means of doing this is through encouraging greater coordination among the Agency's extramural research programs, including the Decision-Making and Valuation for Environmental Policy grant program. A joint research initiative focused on the development of measures of ecosystem services will address a critical policy need and provide a way for the Agency to concretely integrate its ecological and social science expertise.

3.3.3. Mapping ecosystem responses to ecosystem services

Once the underlying ecology is understood and modeled and the relevant ecosystem services are identified, ecological production functions still require a correlation of the ecosystem responses to the relevant ecosystem services. As noted above, although numerous ecological models exist for modeling ecological systems, most of them fall short of estimating effects on ecosystem services. Many of the models have been developed to satisfy research objectives, rather than Agency policy or regulatory objectives. The outputs of these models have not generally been cast in terms of direct concern to people and thus are not useful as inputs to valuation techniques. For example, evapotranspiration rates, rates of carbon turnover, and changes in leaf area are important for ecological understanding, but are not values of direct human importance. Some models exist with outputs directly related to human values and include models that predict fish and game populations or forest productivity. These models, however, address only a limited set of ecosystem services.

3.4. Strategies to provide the ecological science to support valuation

Although development of a broad suite of ecological production functions faces numerous challenges, EPA can employ several other approaches at this time to gain a better understanding of how ecosystem services respond to its actions. These approaches include using proxies or "indicators" for ecosystem services, and meta-analyses. Proxies represent a form of simplification; meta-analysis is based on data aggregation.

3.4.1. Use of indicators

As noted above, an ecological production function describes the relationship between ecological inputs and ecosystem services. When a full characterization of this relationship is not available, some indication of the direction and possible magnitude of the changes in the services that would result from an Agency action might still be obtained using indicators. "Indicators," as the term is used here, are measures of key inputs whose changes are correlated with changes in ecosystem services. In general, an indicator approach involves selecting and measuring key predictive variables rather than defining and implementing a complete ecological production function. Because of the complexity of the interactions between economic and ecological systems, economists frequently take a similar simplified approach that focuses on effects only in the relevant markets, assuming that the effects on the broader market are negligible and can be ignored (Settle et al., 2002).

Indicators can provide useful information about how ecological responses to EPA actions or decisions might affect ecosystem services. If it is known that an indicator is positively or negatively correlated with a specific ecosystem service, predicting the change in the indicator can provide at least a qualitative prediction of the change in the corresponding ecosystem service. Indicators may be important even where models exist that can provide more sophisticated ecological analysis. The use of large, complex ecological models to make numerous or rapid evaluations can be difficult, especially given the quantities of required data and the short time in which assessments generally must be made (Hoagland and Jim, 2006). In these situations, simplification can be far more practical. The use of indicators that simplify and synthesize underlying complexity can have advantages in terms of both generating and effectively communicating information about ecological effects.

Ecologists and environmental scientists have sought to identify indicators of ecosystem condition that might be linked to specific services. Many ecosystem indicators have been proposed (EPA, 1996; NRC, 2000; EPA 2002b; EPA, 2007), and several states have sought to define a relatively small set of indicators of environmental quality. Indicator variables have been established for specific ecosystems such as streams (e.g., Karr, 1993) and for entire countries (e.g., The H. John Heinz III Center for Science, Economics, and the Environment, 2002). The committee acknowledges EPA's work in developing indicators for air, water, and land and for ecosystem condition and encourages the Agency to see where those indicators can be linked to specific services relevant to the valuation of EPA decisions.

There is currently no agreement on a common set of indicators that can be consistently applied and serves the needs of decision makers and researchers in all contexts (Carpenter et al., 2006). However, there are guidelines for specific issues. For example, in evaluating the economic consequences of species invasion, Leung et al. (2005) have developed a framework for rapid assessments based on indicators to guide in prevention and control, simplifying the ecological complexity to a relatively small number of easily estimated parameters.

One potentially useful approach to indicators is to incorporate multiple dimensions into a coherent presentation that describes the status of ecosystems within a region, especially as the ecosystems relate to social values and ecosystem services. For example, the "ecosystem report card" in South Florida (Harwell et al., 1999) uses an array of indicators designed to provide information about the status and trends associated with the ecological services provided by the South Florida ecosystem. The report card identifies seven ecosystem characteristics thought to be important: habitat quality, integrity of the biotic community, ecological processes, water quality, hydrological system, disturbance regime (changes from natural variability), and sediment/soil quality. These characteristics are then related to the goals and objectives for the report card.[28] The outputs are not monetized, but rather described by narratives or quantitative/qualitative grades that are scientifically credible and understandable by the public. The report card is designed to:

- Be understandable to multiple audiences

- Address differences in ecosystem responses across time
- Show the status of the ecosystem
- Transparently provide the scientific basis for the assigned grades on the report card

This simplified approach to ecological modeling cannot identify all the possible consequences of EPA actions. The challenge is building ever more complex models that address a wide array of issues over multiple spatial and temporal scales. It may well be that, with accumulated experience, it may be more practical to adopt the simplified approach of selecting a few key indicators or ecological processes that are correlated with specific ecosystem services and can be valued. The committee advises EPA to continue research to develop key indicators for use in ecological valuation. This is likely to be particularly fruitful when those indicators can be used for key repeated rule makings or other repeated decision contexts. Such indicators should meet ecological science and social science criteria for effectively simplifying and synthesizing underlying complexity while still providing scientifically based information about key ecosystem services that can be valued. Use of the chosen indicators should also be accompanied by an effective monitoring and reporting program.

3.4.2. Use of meta-analysis.

A second promising approach to providing information about effects on ecosystem services is the use of meta-analysis. Meta-analysis, or data aggregation, involves collecting data from multiple sources and attempting to draw out consistent patterns and relationships from those data about the links between ecological functions or structures and associated services. For example, Worm et al. (2006) attempted to measure the effects of biodiversity loss on ecosystem services across the global oceans. They combined available data from multiple sources, ranging from small-scale experiments to global fisheries. In these analyses, the impossibility of separating correlation and causation is a severe limitation. But examining data from site-specific studies, coastal regional analyses, and global catch databases allowed researchers to draw correlative relationships between biodiversity and decreases in commercial fish populations – variables that can be monetized.

In a similar data aggregation approach, de Zwart et al. (2006) noted that ecological methods for measuring the magnitude of biological degradation in aquatic communities are well established (e.g., Karr, 1981), but determining probable causes is usually left to a combination of expert opinion, multivariate statistics, and weighing of evidence. As a result, the results are difficult to interpret and communicate, particularly because mixtures of potentially toxic compounds are frequently part of these assessments. To address this issue the authors used a combination of ecological, ecotoxicological, and exposure modeling to provide statistical estimates of probable effects of different natural and anthropogenic stressors on fish. This approach links fish, habitat, and chemistry data collected from hundreds of sites in Ohio streams. It assesses the biological condition at each site and attributes impairment (e.g., loss of one or more of 117 fish species) to multiple probable causes. It then provides the results of the analyses in simple-to-interpret pie charts. When data were aggregated from throughout Ohio, 50 percent of the biological effect was associated with unknown factors and model error; the remaining 50 percent was associated with alteration in stream chemistry and habitat. Although the results do not fully explain the biological effect, the technique combines multiple data sets and assessment models to arrive at estimates of the loss of fish species based on broad patterns. Like the Worm et al., (2006) study of the relationship of biodiversity to ocean productivity, this study aggregates data from many sources and uses various models to arrive at estimates that can be easily interpreted and, at least in the case of game fish species, monetized.

3.5. <u>Data availability</u>

Data availability is a serious problem in the development of ecological production functions. However, data on the structure and function of ecological systems are becoming more available and better organized across the country. Part of the increased availability is simply that Web-based publication now enables authors to make data and analysis readily available to other researchers in electronic format. Also, as government agencies are being held more accountable, these agencies are increasingly making the data they collect and use available to constituents.

The committee recommends that EPA work with other agencies and with scientific organizations such as the National Science Foundation (NSF) to encourage the sharing of ecological data and the development of more consistent ecological measures that are useful for valuation purposes. EPA should also encourage strong regional initiatives to develop information needed for valuations. Within the ecological research community, the NSF's Long-Term Ecological Research (LTER) program has emphasized organizing and sharing data in easily accessible electronic datasets. Although these data have rarely been collected for valuing ecosystem services, they measure long-term trends and therefore can be particularly valuable in separating short-term fluctuations from longer-term patterns in ecological properties. In addition, the LTER program recently has focused on regionalization, in which data are collected from sites surrounding a primary site, providing a regional context for site-based measurements and models. Planning for NSF's forthcoming National Ecological Observatory Network also includes a networking information and baseline design component aimed at connecting the key scientific questions to the data required to answer the questions.

3.5.1. Transferring ecological information

Despite the increasing availability and organization of ecological data, there is rarely enough available information to support many desired analyses. In addition, the costs of collecting extensive data from all the sites in which EPA is considering action would be prohibitive. An important issue that therefore arises is the reliability of transferring ecological information from one site to another or over different spatial or temporal scales. The information can include tools or approaches, data on properties of an ecosystem or its components, and services or contributions to human well-being provided by an ecosystem.

There are no hard and fast rules for when ecological information can be transferred. Confidence in doing so depends on the type of information and the system in question. Given the complexity, the richness of interactions, and the propensity for non-linearity, extrapolation of ecological information requires caution. However, certain generalizations are possible. Information is more likely to be transferable when there is greater similarity between ecosystem contexts. Also, aggregate information, such as data on ecosystem properties, is more likely to be transferable than information on particular species or the interactions of particular species. Thus, the ecosystem properties (e.g., leaf area index, primary productivity, or nitrogen-cycling patterns) of an oak-hickory deciduous forest in

Tennessee might be transferable to oak-hickory forests in other parts of the eastern United States that are at similar stages of development. To a lesser extent, the information might be transferable to other types of deciduous forests.

Information may be transferable to other spatial or temporal scales if the dynamics over time and space are known for the ecosystems. For instance, if data are available on how the characteristics of an oak-hickory forest change as it develops or goes through cycles of disturbance, data transfers from one point in time to another should be possible. Similarly, if information is available on how the properties of the system vary with spatial environmental variation (e.g., local climate, soil type, or land-use history), the extension of information from one spatial context to another should be possible. EPA and other national and international agencies have sponsored extensive research on the scaling up of data from particular sites to regions [CITATION NEEDED]. The results from these analyses are applicable to the transfer of information on ecological properties and services.

To some extent, the same generalizations apply to transferring tools such as models, although success depends on how generally applicable the tool is and how difficult in terms of data requirements it is to parameterize for other situations. For example, forest ecosystem models can often be transferred to other forests using available information from sources such as LTER sites.

3.6. <u>Directions for ecological research to support valuation</u>

EPA has briefed the committee on its plans to redesign a major part of its intramural and extramural research program to forecast, quantify, and map production of ecosystem services (see briefings to the C-VPESS, EPA Science Advisory Board, 2006c and 2007b). The committee welcomes these efforts as a way to strengthen the foundation for ecological valuation but notes with concern EPA's limited and shrinking resources for ecological research (EPA Science Advisory Board, 2007). Although the committee has not received any details about Agency plans, it encourages the Agency to carefully focus its research program because the cost of implementing ecological production functions in multiple places on multiple issues may be significant. The committee commends EPA for asking for additional science advice on its Ecological Research Program Strategy and Multi-year Plan and believes this advisory activity should be a priority for an SAB panel of interdisciplinary experts in ecological valuation, drawing on information in this report.

3.7. Conclusions and recommendations

Implementation of the integrated valuation process recommended by this report requires the Agency to predict the ecological responses to its actions, identify the relevant ecosystem services of importance to the public, and linkthe predicted ecological responses to the effect on those services. Estimating the responses of relevant ecosystem services to EPA actions is an essential part of valuation and must be done before the value of those responses can be assessed.

With regard to predicting the responses of ecosystems and ecosystem services, the committee recommends the following:

- EPA should begin each valuation with a conceptual model of the relevant ecosystem and the ecosystem services that it generates. This model should serve as a road map to guide the valuation. EPA should formalize a process for constructing the initial conceptual model, recognizing that the process must be iterative and respond to new information and multiple points of view. The conceptual model should reflect the ultimate goal of valuing the effect of EPA's decision on ecosystem services. The model and its documentation should also clearly describe the reasons for decisions about the spatial and temporal scales of the chosen ecological system, the process used to identify stressors associated with the proposed EPA action, and the methods to be used in estimating the ecological effects. In constructing the conceptual model, the Agency should involve staff throughout the EPA, outside experts from the bio-physical and social sciences, and seek information about relevant public concerns and needs.
- EPA should identify and develop measures of ecosystem services that are relevant to and directly useful for valuation. This will require increased interaction between natural and social scientists within the Agency. In identifying and evaluating services for any specific valuation effort, EPA

should count all things that matter once and only once, and describe them in terms that are meaningful and understandable to the public.

- EPA should seek to use ecological production functions wherever practical to estimate how ecological responses (resulting from different policies or management decisions) will affect the provision of ecosystem services.
- All ecological valuations conducted by EPA should be supported by ecological models and data sufficient to understand and estimate the likely ecological responses to major alternatives being considered by decision makers. There are many ecological models. Building on recent efforts within the Agency and elsewhere, EPA should develop criteria or guidelines for model selection that reflect the specific modeling needs of ecological valuation, and EPA should apply these criteria in a consistent and transparent way.
- Because of the complexity of developing and using complete ecological production functions, EPA should continue and accelerate research to develop key indicators for use in ecological valuation. Such indicators should meet ecological and social science criteria for effectively simplifying and synthesizing underlying complexity and be associated with an effective monitoring and reporting program. The Agency should also support the use of methods such as meta-analysis that are designed to provide general information about ecological relationships that can be applied in ecological valuation.
- EPA should work with other agencies and with scientific organizations such as the National Science Foundation to encourage the sharing of ecological data and the development of more consistent ecological measures that are useful for valuation purposes. EPA should similarly encourage strong regional initiatives to develop information needed for valuations. EPA should also promote efforts to develop data that can be used to parameterize ecological models for site-specific analysis and case studies, or that can be transferred or scaled to other contexts.

- EPA should carefully plan and actively pursue research to generate ecological production functions for valuation, including Science to Achieve Results program research on ecological services and support for modeling and methods development. EPA should make the development of ecological models that can be used in valuation efforts one of its research priorities.
- Finally, EPA should foster interaction between natural scientists and social scientists in identifying relevant ecosystem services and developing and implementing processes for measuring and valuing them. As part of this effort, EPA should more closely link its research programs on evaluating ecosystem services and valuing ecosystem services

4 Methods for assessing value

In advocating an expanded and integrated approach to valuing the protection of ecological systems and services, the committee urges the Agency to consider and experiment with a broader set of valuation methods. This chapter provides an overview of the methods that the committee discussed for possible use in implementing its approach, including methods and approaches for transfer of valuation information.

As noted in chapter 2, the methods considered by the committee vary in the roles that they might play in different decision contexts. For example, as noted previously, benefit assessments for national rule makings must be conducted under the guidance of Office of Management and Budget Circular A-4, which implies that, in that context, monetized methods must be based on economic valuation methods. Some non-economic valuation methods can still provide useful information in this context, but the role of these methods is limited by the need to follow the guidance in the circular (see section 6.1). In other, less-prescribed decision contexts, non-economic valuation methods can play a more central role in analysis and can even be the primary source of value information (see sections 6.2 and 6.3). Thus, as the Agency considers alternative methods that might be used, it must consider the context of the information needs defined by the particular policy context in which the valuation exercise will be done.

4.1. <u>Criteria for choosing valuation methods</u>

The methods discussed by the committee differ in a number of important respects. These include: the underlying assumptions; the types of values they seek to measure or characterize; the empirical and analytical techniques used to apply them; their data needs (inputs) and the metrics they generate (outputs); their involvement of the public or stakeholders; the degree to which the method has been developed or utilized; their potential for future use at EPA; and the issues involved in implementing the methods.

Any method used by the Agency must meet relevant scientific standards. Before relying on any given method in a particular valuation process, EPA must determine if there is a sound scientific basis for the method's use in that context. Methods that are in their early stages of development and application to valuation must be evaluated both for their scientific merit and for their appropriateness in the given context of interest. Methods that are well-developed, have been extensively used for valuation, and have been validated in other contexts should still be evaluated for their suitability in valuing ecosystems and services, because a given context may pose challenges that might not exist in other situations. The committee has not developed a full set of criteria for evaluating methods, nor has it applied criteria comprehensively to the methods discussed here. The committee urges EPA to develop criteria and evaluate methods by those criteria prior to use in valuation. Some suggestions for criteria that EPA should consider for inclusion are described briefly in section 4.1.2.

In developing criteria for evaluating valuation methods, a distinction should be made between criteria for evaluating the suitability of a particular method in a given context (i.e., evaluating the scientific merit and suitability of the method) and criteria for evaluating the manner in which the method is actually applied (i.e., evaluating the implementation of the method). For example, the question of whether survey methods in general can appropriately be used to estimate or elicit value(s) in a particular context is a different question, requiring different criteria, than the question of whether a specific survey was properly designed and executed to estimate or elicit the intended value(s). If not properly implemented, any method can yield results that are not useful for the intended purpose. For any individual method, EPA can develop criteria to ensure that the method is carefully implemented. Criteria of this type exist for many of the methods described here, and committee members have described criteria for many valuation methods (see valuation method descriptions on the SAB Web site at www.epa.gov/sab/XXXXX). The committee recommends that EPA develop a higher-order list of criteria designed to evaluate the suitability of specific methods for a specific valuation context, assuming that any method chosen would be implemented according to best practices.

4.1.1. Suggested criteria

While not prescribing the specific criteria that EPA should use to evaluate methods before using them in a specific context, the committee offers some suggested criteria. These draw on the literature cited below, as well as the committee's own deliberations.

A primary consideration in evaluating a method should be the extent to which the method seeks to elicit or measure a concept of value that has a consistent and transparent theoretical foundation appropriate for the intended use. Different valuation methods measure different concepts of value. For a method to be appropriate in a valuation context, it must seek to measure a concept of value that is well-defined, theoretically consistent, and relevant for the particular valuation context. For example, a method derived from a biodiversity-based theory of value would not be relevant in a context where biodiversity is not important. Similarly, legal requirements may prescribe a theory of value that must be used in a particular valuation context (most notably, national rule making). Thus, the Agency should consider the theory of value underlying a particular method and its relevance when evaluating the appropriateness of using that method in a specific context.

Assuming a method seeks to elicit or measure a well-defined and relevant concept of value, another over-arching criterion for evaluation is validity – i.e., how well the method measures the underlying construct that it is intended to measure (Gregory et al. 1993; Freeman, 2003; Fischhoff 1997). Ideally, a method should measure only what it is supposed to measure. Although the underlying construct of value is not directly observable, it can be estimated through the use of valid methods. EPA should use criteria to assess the extent to which a given method is likely to yield a measure, or at least an unbiased estimate, of the underlying construct of value. Examples of criteria that provide information about the validity of a method include:

- Does the method capture the critical features of the relevant population's values, including how deeply they are held? Does it yield value estimates that reflect the intensity of people's preferences or the magnitude of the contribution to a given goal?
- Does the method impose demands on respondents that limit their ability to articulate values in a meaningful way? For example, does the method impose unrealistic cognitive demands on individuals expressing values? Does it allow those individuals to engage in the process that they would normally undertake to identify or formulate and then articulate their values?
- Does the method yield value estimates for individuals that those individuals would, if asked, consent to have used in the proposed way? Fischhoff (2000)

suggests that this form of implied informed consent can help to ensure the quality of valuation data generated by a given method and avoid inappropriate use of the resulting value estimates, by ensuring that individuals would "stand behind researchers' interpretation of their responses" (p. 1439).

- Does the method ensure that measured or elicited values reflect relevant scientific information? A basic premise of the valuation approach proposed by the committee is that a method should elicit or measure values that individuals would hold when well-informed about the relevant science. This does not require that all individuals expressing values know as much as scientific experts in the field, but rather that they understand as much of the science as necessary to make informed judgments. For example, they should be aware of the magnitude of the changes in ecosystem services or characteristics that would result from the ecological changes being valued, as well as the implications of those changes for themselves and for others.
- Does the method yield value estimates that are responsive to changes in variables that the relevant theory suggests should be predictors of value, and invariant to changes in variables that are irrelevant to the determination of value? For example, under an economic theory of value, an increase in the quantity of the good or service being valued should result in an increase in the magnitude of expressed values. This form of validity has been termed construct validity (Fischhoff, 1997; Mitchell and Carson, 1989).
- Are the expressions of value resulting from the method stable (i.e., reliable) in the sense that they do not change upon further reflection (Fischhoff, 1997) and are not unduly influenced by specific researcher or group characteristics?
- To what extent does the information elicited from participants in the application of the method (e.g., survey respondents or focus group participants) provide information that can be used to reliably infer something about the values of a broader group within the relevant population?

Methods can be evaluated on the extent to which the resulting value estimates can be transparently communicated in a useful format to those who will use the value information.

Decision makers and the public should be able to understand how the value measures relate to and inform the decision that needs to be made.

4.2. An expanded set of methods

This section provides an overview of, and introduction to, the wide array of methods considered by the committee for possible use in implementing the valuation process proposed in chapter 2. **Error! Reference source not found.**2 provides a listing of these methods, along with an overview of the form of output from each method and the concept(s) of value that it seeks to measure or elicit. General descriptions of the categories of methods follow. Although these methods are not easily categorized, the committee has grouped the methods based on the premises underlying the methods.

The following discussion of methods is illustrative and introductory rather than comprehensive. The goal is to provide the reader with sufficient information about the methods to allow a preliminary assessment of the role that various methods can play in implementing the proposed valuation process and to direct the interested reader to the relevant scientific literature for further information. The brief descriptions in this section are based on more detailed information about individual methods supplied by individual committee members, which can be found on the SAB Web site at www.epa.gov/sab/XXXXX. Appendix A also provides detailed information about the use of survey methods for ecological valuation.

Table 2: Alternative methods considered by the committee for possible use in valuation

Method	Form of output/units	Related concepts(s) of value from Table 1
Biophysical ranking methods		
Conservation value method	Spatially-differentiated index of conservation values across a landscape	Bio-ecological value
Embodied energy analysis	Cost of the total (direct plus indirect) energy required to produce an ecological or economic good or service	Energy-based value
Ecological footprint	Area of an ecosystem (land and/or water) required to support a consumption pattern or population	Bio-ecological value
Ecosystem benefit indicators		
Ecosystem benefit indicators	Quantitative spatially-differentiated metrics or maps related to supply of or demand for ecosystem services	Correlated with economic value and/or community-based values
Measures of attitudes, preferences, and intentions		
Survey questions eliciting information about attitudes, preferences, and intentions	Attitude scales, preference or importance rankings, behavioral intentions toward depicted environments or conditions	Attitudes and judgments; community-based values
Individual narratives and focus groups	Qualitative summaries and assessments from transcripts	Attitudes and judgments; community-based values
Behavioral observation	Inferences from observations of behavior by individuals interacting with actual or computer-simulated environments	Attitudes and judgments; community-based values
Economic methods		
Market-based methods	Monetary measure of willingness-to-pay (WTP) for ecosystem services that contribute to the provision of marketed goods and services	Economic value
Travel cost	Monetary measure of WTP for ecosystem services that affect WTP to visit different locations	Economic value
Hedonic pricing	Monetary measure of marginal WTP or willingness-to-accept (WTA) as revealed by WTP for houses or WTA jobs with different environmental characteristics and prices	Economic value
Averting behavior	Monetary or other measure of WTP as revealed by responses to opportunities to avoid or reduce damages, for example, through expenditures on protective goods or substitutes	Economic value
Survey questions eliciting stated preferences	Monetary or other measures of WTP or WTA as expressed in survey questions about hypothetical tradeoffs	Economic value
Civic valuation		
Referenda and initiatives	Rankings of alternative options, or monetary or other measure of tradeoffs a community is willing to make, as reflected in community choices	Economic value under some conditions
Citizen valuation juries	Rankings of alternative options, or monetary or other measures of required payment or compensation, based on jury-determined assessments of public values	Community-based values; constructed values
Decision science approaches		
Decision science approaches	Attribute weights that reflect tradeoffs individuals are willing to make across attributes, including ecological attributes, for use in assigning scores to alternative policy options	Constructed values

Method	Form of output/units	Related concepts(s) of value from Table 1
Cost as a proxy for value		
Replacement cost	Monetary estimate of the cost of replacing an ecosystem service using the next best available alternative	Lower bound on economic value only under limited conditions
Habitat equivalency analysis	Units of habitat (e.g., equivalent acres of habitat) or other compensating changes needed to replace ecosystem services lost through a natural resource injury	Bio-physical value; economic value only under very limited conditions

4.2.1. Biophysical ranking methods

In some contexts, policy makers or analysts are interested in values based on quantification of biophysical indicators. Possible indicators include measures of biodiversity, biomass production, carbon sequestration, or energy and materials use. Quantification of ecological changes in biophysical terms allows these changes to be ranked based on individual or aggregate indicators for use in evaluating policy options based on biophysical criteria.

Use of a biophysical ranking does not explicitly incorporate human preferences. Rather, it reflects either a non-anthropocentric theory of value (based, for example, on energy flows) or a presumption that the indicators provide a proxy for human value or social preference. This latter presumption is predicated on the belief that the healthy functioning and sustainability of ecosystems is fundamentally important to the well-being of human societies and all living things, and that the contributions to human well-being of any change in ecosystems can be assessed in terms of the calculated effects on the overall health and sustainability of ecosystems. Opinion is mixed – among both committee members and the broader scholarly community – on whether it is an asset or a drawback that these ranking methods are not tied directly to human preferences.

The committee discussed two types of biophysical rankings. The first is a ranking method based on conservation value. The **conservation value method** develops a spatially-differentiated index of conservation value across a landscape based on an assessment of rarity, persistence, threat, and other landscape attributes, reflecting the contribution of these attributes to sustained ecosystem diversity and integrity. Policy makers or stakeholders can use these values to prioritize land for acquisition, conservation, or other purposes, given relevant biophysical goals. Based on geographic information system (GIS) technology, the method can combine information about a variety of ecosystem characteristics and services across a given landscape and overlay ecological information with other spatial data. Conservation values have been used in various contexts by federal agencies (e.g., the U.S. Forest Service, Fish and Wildlife Service, National Park Service, and Bureau of Land Management), non-governmental organizations (e.g., The Nature Conservancy and NatureServe), and by regional and local planning agencies.

The second group of biophysical methods that the committee discussed quantify the flows of energy and materials through complex ecological systems, economic systems, or both. Ecologists have used these methods to identify the resources or resource-equivalents needed to produce a product or service, using a systems or life-cycle ("cradle to grave") approach. For example, **embodied energy analysis** measures the total energy, direct and indirect, required to produce a good or service. Similarly, **ecological footprint analysis** measures the area of an ecosystem (e.g., the amount of land and/or water) required to support a certain level and type of consumption by an individual or population. These methods can provide estimates of the cost of producing a given good or service based on required inputs.[29]

In addition to using these methods to measure required inputs, some ecologists have advocated using the resulting cost estimates as a measure of value, based on an energy (or other biophysical input) theory of value. However, making policy decisions based on whether the total energy available for use increases or decreases in a given system could yield drastically different decisions than those based on whether human welfare increases or decreases. For example, an analysis based on an energy theory of value might imply that global warming is of value to society if it increases the energy content of the global system from which organisms draw.

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4.2.2. <u>Ecosystem benefit indicators</u>

Ecosystem benefit indicators offer quantitative metrics that are generally correlated with ecological contributions to human well-being and hence can serve as indicators for these contributions in a specific setting. They use geo-spatial data to provide information related to the demand for, supply (or scarcity) of, and complements to particular ecosystem services across a given landscape, based on social and biophysical features that influence – positively or negatively – the contributions of ecosystem services to human well-being. Examples of these indicators include the percentage of a watershed in a particular land use or of a particular land type, the number of users of a service (e.g., water or recreation) within a given area, and the distance to the nearest vulnerable community.

Ecological benefit indicators can serve as important quantitative inputs to valuation methods as diverse as citizen juries and economic valuation methods. Ecosystem benefit indicators provide a way to illustrate factors influencing ecological contributions to human welfare in a specific setting. The method can be applied to any ecosystem service where the spatial delivery of services is related to the social landscape in which the service is enjoyed. However, although the resulting indicators can be correlated with other value measures, such as economic values, they do not themselves provide measures of value.

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4.2.3. Measures of attitudes, preferences, and intentions

There are a variety of valuation methods that seek to measure attitudes, preferences, and intentions. These methods are sometimes referred to as social psychological methods. They seek to characterize the values people hold, express, and advocate, focusing mainly on individuals' judgments of the relative importance of, acceptance of, or preferences for ecological states or changes. These methods can elicit value-relevant perceptions and judgments about a wide array of objects, events, and conditions. They typically focus on choices or ratings among sets of alternative policies and may include comparisons with potentially competing social and economic goals. Individuals making the judgments may respond on their own behalf or on behalf of others (society at large or specified subgroups). The basis for judgments can be changes in individual well-being or in civic, ethical, or moral obligations.

Social psychological methods primarily elicit information about preferences and values through surveys, individual narratives, and focus groups. Recently, experts have also been experimenting with eliciting this type of information through observations of behavioral responses by individuals interacting with either actual or computer-simulated environments.

Survey questions eliciting information about attitudes, preferences, and intentions related ecosystems and ecosystem services can be well-conveyed in perceptual surveys (e.g., assessments based on photographs, computer visualizations, or multimedia representations of targeted ecosystem attributes) and conjoint surveys (e.g., requiring choices among alternatives that systematically combine multiple and potentially competing attributes). Quantitative analyses of responses are usually interpreted as ordinal rankings or rough interval-scale measures of differences in assessed values for the alternatives offered. Survey questions about social and psychological constructs may be especially useful when the values at issue are difficult to express or conceive in monetary terms, or where monetary expressions are viewed as ethically inappropriate. Federal agencies have extensively used surveys to elicit value-related information.[30]

In contrast to surveys based on large representative samples, individual **narrative methods –** including mental-model analyses, ethnographic analyses, and other relatively unstructured individual interviews - generally employ small samples of informants and analyze responses qualitatively. Similarly, **focus groups** can be used to elicit information about values and preferences from small groups of relevant stakeholders engaging in group discussion led by a facilitator. Rigorous qualitative analyses of transcripts from individual narratives or focus groups can expose subtle differences in individual beliefs and perspectives and the inferential bases of participants' values.

Given the small number of participants, the goal of individual narratives and focus groups is rarely to assess the public's values per se. Rather, these methods seek to identify the types and range of value perspectives, positions, and concerns of individual participants, and to use this information to identify the ecosystem effects that might be particularly important to the public. The open-ended nature of these methods can reveal perspectives and concerns that more structured methods might miss. Thus, these methods can provide useful input early in a valuation process, regardless of the valuation methods later used. In addition, for some valuation methods, focus groups can play a critical role in guiding the use of the method itself. For example, focus groups can be extremely useful in the early stages of designing a survey to elicit value information from a broader sample of the relevant population.

Researchers have recently explored the use of **behavioral observation methods** for obtaining information about people's values. Observing how the activities of people change as environmental conditions change can reveal information about the importance of these changes to those people. Researchers can observe changes in actual behavior (e.g., visitation rates) or virtual behavior (e.g., responses in interactive computer simulation games). Behavioral observation methods are relatively new and untested, particularly in the context of valuing ecosystem services. Nonetheless, they show promise for sue in this context.

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4.2.4. Economic methods

Economic valuation methods seek to measure the tradeoffs individuals are willing to make for ecological improvements or to avoid ecological degradation, given the constraints they face. An ecological change that an individual values will increase that person’s utility. The value or economic benefit of that change is defined to be the amount of another good that the individual is willing to give up to enjoy that change (willingness-to-pay) or the amount of compensation that a person would accept in lieu of receiving that change (willingness to accept). Although these tradeoffs are typically expressed in monetary terms, economic methods that express tradeoffs in non-monetary terms (such as conjoint analysis or other choice-based methods) are increasingly being used.

Economic methods can estimate values not only for goods and services for which there are markets but also for non-market goods and services. Economic methods can also value both use and non-use (e.g., existence) values. Thus, economic valuation captures values that extend well beyond commercial or market values. However, it does not capture non-anthropocentric values (e.g., biocentric values) and values based on the concept of intrinsic rights. In addition, because the tradeoffs people are willing to make generally depend on their income (as well as market prices), economic valuation typically yields value estimates that are higher for individuals with higher incomes. Many view this as a drawback in the context of public policy decisions.

There are multiple economic valuation methods that can be used to estimate economic values. These include methods based on observed behavior (market-based and revealed-preference methods) and methods based on information elicited from responses to survey questions about hypothetical tradeoffs (e.g., stated-preference methods). Some of these methods are more applicable to some contexts than to others.

Market-based methods seek to use information about market prices (or market demand) to infer values related to changes in marketed goods and services. For example, when ecological changes lead to a small change in timber or commercial fishing harvests, the market price of timber or fish can be used as a measure of willingness to pay for that marginal change. If the change is large, the current market price alone is not sufficient to determine value. Rather, the demand for timber or fish at various prices must be used to determine willingness to pay for the change. In general, market-based methods can value only those provisioning services supplied in well-functioning markets.

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Revealed-preference methods exploit the relationship between some forms of individual behavior (e.g., visiting a lake or buying a house) and associated environmental attributes (e.g., of the lake or the house) to estimate value. For example, **travel cost methods** (including applications using random utility models) use information about how much people implicitly or explicitly pay to visit locations with specific environmental attributes, including, specific levels of ecosystem services, to infer how much they value changes in those attributes. **Hedonic pricing** uses information about how much people pay for houses with specific environmental attributes (e.g., visibility, proximity to amenities or disamenities) to infer how much they value changes in those attributes. It also may use information about the wages people would be willing to accept for jobs with differing mortality or morbidity risk levels to infer how much they value changes in those risks. In contrast, **averting-behavior methods** use observations on how much people spend to avoid adverse effects, including environmental effects to infer how much they value or are willing to pay for the improvements those expenditures yield.

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<u>Travel costs</u>

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4.2.5. Civic valuation

Civic valuation seeks to measure the values that people place on changes in ecosystems or ecosystem services when explicitly considering or acting in their role as citizens. These valuation methods often seek to value changes that would benefit or harm the community at large. They purposefully seek to assess the full public regardedness value, if any, that groups attach to any increase in community well-being attributable to changes in the relevant ecosystems and services.

Civic valuation, like economic valuation, can elicit information about values either through revealed behavior or through stated valuations. One source of information based on revealed behavior is votes on public referenda and initiatives involving the provision of environmental goods and services (e.g., purchases of open space). Another source is community decisions to accept compensation for permitting environmental damage (e.g., by hosting noxious facilities). Where revealed values are difficult or impossible to obtain, citizen valuation juries or other representative groups can be charged with determining the value they would place on changes in particular ecological systems or services when acting on behalf of, or as a representative of, the citizens of the relevant community.

Referenda or initiatives can be used to determine how a majority of the voting population values a particular governmental action involving the environment. Analysis of referenda or initiatives can reveal whether the majority of the voting population feels that a given environmental improvement is worth what it will cost the relevant government body, given a particular means of financing the associated expenditure (and hence, an anticipated cost to the individual who is voting). In casting their votes, individuals may consider not only what they personally would gain or lose but also what the community as a whole stands to gain or lose if the proposal is adopted. Similarly, analyses of public votes about whether to accept an environmental degradation (e.g., through hosting a noxious facility) seek to determine if the majority of the voting population in that community feels that the environmental services that would be lost are worth less than the contributions to well-being the community would realize (e.g., in the form of tax revenues, jobs, monetary compensation).

These approaches provide information about the policy preferences of the median voter and, under certain conditions, provide information about the mean valuations of those who participate in the voting process. To the extent that voters consider their own budget constraints when voting, these valuations reflect economic values, i.e., willingness to pay or willingness to accept. As with all economic values, the revealed economic value reflects both personal benefits and costs, as well as any altruistic motivation (public regardedness) individual voters have when casting their votes.

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In contrast to initiatives and referenda, **citizen valuation juries** measure stated rather than revealed value. They also incorporate elements of the deliberative valuation process (see chapter 5). The jury is given extensive information and, after a lengthy discussion, usually asked to agree on a common value or make a group decision. To date, citizen juries have typically been asked to develop a ranking of alternative options for achieving a given goal. Although citizen juries have been used in other contexts, experience using citizen juries for ecological valuation is very limited. Nonetheless, in principle, a jury could be asked to generate a value for how much the public would, or should, be willing to pay for a possible environmental improvement, or, conversely, willing to accept for an environmental degradation. In contrast to estimates of willingness to pay derived from economic valuation methods, the estimates from citizen juries would not reflect the budget constraints of the individual participants and would reflect community-based values rather than economic values. To the extent that a citizen jury engages in group deliberation, resulting value estimates also would reflect constructed values.

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4.2.6. Decision science methods

Decision science valuation methods presume that individuals' preferences need to be constructed through a deliberative process that helps individuals understand and assess tradeoffs among multiple attributes. The ultimate goal is to have an individual or group assign scores to alternatives (e.g., different projects) that can then be used to choose among those alternatives, recognizing that those alternatives will differ along a number of relevant dimensions or attributes. Generally, one alternative will dominate along some dimensions but not others, suggesting that tradeoffs must be made when choosing among alternatives.

Decision science valuation methods are typically embedded in a decision-aiding process. As part of the process, an expert facilitator helps the individual or group decompose the choice problem by identifying and operationalizing objectives as well as relevant attributes. For example, people may feel that the value of a project to protect an estuary depends on attributes such as the estuary's ability to provide nutrient exchange and nursery habitat for anadromous fish, the opportunities it provides for recreation, and the cost of the project. The facilitator leads the individual or group through a process by which they assign weights to each of the attributes. A variety of approaches to assigning weights have been used, including assigning importance points, eliciting ratio weights, determining swing weights, and pricing out attributes. These weights reflect the tradeoffs that the individual or group is willing to make across attributes, and hence reveal information about values. Because this method is based on a deliberative process, the resulting values are constructed values.

Once the attribute weights are determined, an aggregating function (or utility function) is used to combine the weights and attribute levels into a score (or measure of multi-attribute utility) for each alternative. Ranking alternative projects or options based on these scores can provide information about which option (and hence which combination of attributes) is viewed as more valuable.

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4.2.7. Methods using cost as a proxy for value

A fundamental principle in economics is the distinction between benefits and costs. In the context of ecosystem services, economic benefits reflect what is gained by increasing the amount of a given service relative to some baseline, while costs reflect what must be given up in order to achieve that increase. Costs can provide information about benefits or value only under specific and limited conditions. Nonetheless, several methods based on costs have been used in the valuation of ecosystem services.

One such method is **replacement cost**. Under this method, the value of a given ecosystem service is viewed as the cost of replacing that service by some alternative means. For example, some studies have valued clean drinking water provided by watershed protection by using the cost savings from not having to build a water filtration plant to provide the clean water (National Research Council, 2000 and 2004; SAgoff 2005). This type of cost savings can offer a lower-bound estimate of the value of an ecosystem service, but only under limited conditions (Bockstael et al., 2000). First, there must be multiple ways to produce an equivalent amount and quality of the ecosystem service. In the above example, the same quantity and quality of clean water must be provided by both the watershed protection and the filtration plant. Second, the value of the ecosystem service must be greater than or equal to the cost of producing the service via this alternative means, so that society would be better off paying for replacement rather than choosing to forego the ecosystem service. In the example, the value of the clean water provided must exceed the cost of providing it via the filtration plant. When these two conditions are met, it is valid to use the cost of providing the equivalent services via the alternative as a lower-bound estimate of the economic value of the ecosystem service.

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Another cost-related concept is **habitat equivalency analysis** (HEA), which has been used in Natural Resource Damage Assessments under the Comprehensive Environmental Response, Compensation and Liability Act and the Oil Pollution Act. HEA seeks to determine the restoration projects that would provide ecosystem or other related services (including capital investments such as boat docks) sufficient to compensate for a loss from a natural-resource injury (e.g., a hazardous waste release or spill). In principle, to determine whether a set of projects provides sufficient compensation for a loss, HEA should determine the tradeoffs required to make the public whole using utility equivalents of the associated losses and gains – i.e., it should use a value-to-value approach (see Roach and Wade, 2006; Jones and Pease, 1997). However, in practice HEA is often based on a service-to-service approach specified in biophysical equivalents (e.g., acres) rather than utility equivalents (value). Restoring habitat far from where people live and recreate, however, may not create value equivalent to nearby lost habitat, even if the replacement habitat is of the same size.

Although HEA can provide dollar estimates of the cost of providing replacement services or projects, these estimates do not necessarily satisfy the two conditions noted above that are necessary for replacement cost to provide a lower bound on value. For example, the value of the ecosystem or other services provided by the restoration projects may not exceed the cost of providing those services. Even if it does, several other assumptions are needed to ensure that HEA will provide an actual estimate of the economic value of the lost ecosystem services and these assumptions will often not be met in practice. These include fixed proportions between services and values, as well as unit values that are constant over time and space (Dunford et al., 2004).

Because costs and benefits are two distinctly different concepts, the committee urges caution in the adoption of any methods using costs as a proxy for value. The above conditions for valid use must be satisfied. Analyses of costs should not be interpreted as measures of benefits unless these conditions are met. Nonetheless, when appropriately applied, methods such as replacement cost and HEA may be useful to EPA in policy contexts where there are multiple ways of providing an ecosystem service.

The price of tradable emissions permits under cap-and-trade systems will almost never meet the requirements for using cost as a proxy for value. The price of an emission permit in a well-functioning market will reflect the incremental cost of pollution abatement. This price does not reflect the value of pollution reduction unless one of two conditions is met: a) the number of permits is set optimally, so that the incremental cost of pollution equals the incremental benefit of pollution reduction; or b) there are significant purchases of permits for purposes of retiring rather than using the permit, which indicates the willingness-to-pay for pollution reduction by the purchaser. Absent these exceptions, the price of tradable emissions permits should not be used as a proxy for value.

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4.3. Value transfer

This section examines economic benefits transfer. Chapter 3 previously considered the transfer of ecological information.

4.3.1. Transfer of economic benefits

Economists often use information about economic benefits derived from a previous valuation study to assign values to changes in another context. This process or method is known as benefits transfer. As an example, suppose that a hedonic property value study used data from the sales of residential homes in Chicago (the study site) to estimate the incremental change in housing prices associated with variations in the air quality conditions near these homes. Given a variety of theoretical and statistical assumptions, measures adapted from the estimates of these price equations can be used to estimate the marginal value of small improvements in air quality in another city, such as New York or Los Angeles (the policy site). The adjustments necessary to use benefit information from a previous study in a new context depend on a number of factors, including the needs of proposed policy application, the available information about the policy site, and the added assumptions the analyst is prepared to make.

In light of the time and money needed to generate original value estimates, EPA relies heavily on benefits transfer. In fact, benefits transfer is the primary method EPA uses to develop the measures of economic trade-offs used in its policy evaluations. Most regulatory impact assessments and policy evaluations rely on adaptation of information from the existing literature. Recent examples of policy evaluations that used benefits transfer methods include EPA's *Economic and Benefits Analysis for the Final Section 316(b) Phase III Existing Facilities Rule, June 1, 2006* (EPA, 2006), EPA's *Final Report to Congress on Benefits and Costs of the Clean Air Act, 1990 to 2010*. (EPA, 1999), and the economic benefit-cost analysis of the CAFO regulations offer.

EPA's heavy reliance on benefits transfer raises a significant issue regarding its validity: Under what conditions can the findings derived from existing studies be used to estimate values in new contexts? Inappropriate benefits transfer often is a weak link in valuation studies. A number of environmental economists and other policy analysts have devoted considerable attention to benefits transfer (e.g., Wilson and Hoehn 2006).

The evaluations of benefits transfer in the literature have been mixed. For example, Brouwer (2000) concludes that "no study has yet been able to show under which conditions environmental value transfer is valid" (p. 140). Similarly, Muthke and Holm-Mueller (2004) urge analysts to "forego the international benefit transfer" and remark that "national benefit transfer seems to be possible if margins of error around 50% are deemed to be acceptable" (p. 334). On the other hand, Shrestha and Loomis (2003) conclude that, "Overall, the results suggest that national BTF can be a potentially useful benefit transfer function for recreation benefit estimation at a new policy site" (pp. 94-95).

Because benefits transfer constitutes a wide collection of methods that arise from the specific needs of each policy application, broad conclusions regarding validity are not meaningful. Rather, assessment of the validity of the approach requires case-by-case evaluation of the assumptions used in the specific application of interest and must consider the similarities and dissimilarities between the study site and the policy site(s). For this reason, overall the committee believes that general conclusions regarding the validity of the application of these methods are not possible. However, some applications of benefits transfer by EPA have been valid, while others have not.

4.3.2. <u>Transfer methods</u>

As noted, benefits transfer refers to a collection of methods rather than a single approach. Values derived from one or more study sites can be transferred to a policy site in three ways. The first is the transfer of a unit value. A unit-value transfer usually interprets an estimate of the tradeoff people make for a change in environmental services as locally constant for each unit of change in the environmental service. For the policy site, the relevant and available values for these factors are used to estimate an adjusted measure for the unit value based on the specific conditions in the policy site (see Brouwer and Bateman 2005 for an example in the health context). As noted above, the required adjustments will depend on a number of factors.

A second approach is the function transfer approach, which replaces the unit value with a summary function describing the results of a single study or a set of studies. For example, a primary analysis of the value of air-quality improvements might be based on a contingent valuation survey of individuals' willingness to pay to avoid specific episodes of ill health, such as a minor symptom-day (e.g., a day with mildly red watering itchy eyes) or one day of persistent nausea and headache with occasional vomiting (e.g., Ready et al. 2004). A value function in this context relates willingness to pay to respondent characteristics and other factors that are likely to influence it, such as income, health status, demographic attributes, and the availability of health insurance. This value function is then used to estimate willingness-to-pay for populations with different characteristics. Alternatively, the original study might estimate a demand function or discrete choice model based on an underlying random utility model describing revealed preference choices. The demand function or discrete choice model is transferred and then used to estimate economic benefits at the policy site. In this case, the function being transferred is an estimated behavioral model rather than a value function.

Meta-analyses, which statistically combine results from numerous studies, can also involve a type of function transfer. Meta-analyses can be undertaken when there is accumulated evidence on measures of economic tradeoffs for a common set of changes in resources or amenities, provided that the benefit concept that is measured and the resource change that is valued are consistent across the studies that are combined in the analysis (Smith and Pattanayak, 2002). One area with a large number of applications is water quality relevant to recreation (e.g., Johnston et al., 2003; Smith and Kaoru, 1990a, 1990b). EPA recently used this approach in its assessment for the Phase III component of the 316(b) rules.

Some meta-analyses combine unit values to produce a weighted average unit value. While this might be appropriate in some valuation contexts (EPA Science Advisory Board 2008), in the context of ecological valuation it can be problematic because it ignores the site-specific variation in ecological values that stems from heterogeneity in both ecosystem and population characteristics. Alternatively, meta-analyses can combine studies to estimate a meta-regression function, which can be used to identify both site and population characteristics as well as methodological characteristics that influence benefit estimates. Such a function has the potential to be used for benefits transfer and allows an adjustment for characteristics of the policy site, if based on a structural approach that ensures that basic consistency properties are satisfied in order for the results to yield reliable benefit estimates (Bergstrom and Taylor, 2006). These approaches to benefit transfer have not yet been widely used. They need to be evaluated before it would be possible to describe a set of practices for applications, for example, in national in rule making (see further discussion in section 6.1).

A third approach to benefits transfer is preference calibration. It uses information from the study site to identify the parameters that describe underlying preferences, with the objective of then using the resulting preference relationship to estimate benefits at the policy site (see Smith et al 2002). With calibration, not all relevant parameters (in this case relating to preferences) are estimated directly from the data. Rather, some are calculated or inferred from available estimates of other parameters and assumed or observed relationships and constraints. When the parameters can be calibrated or estimated from the existing literature, the transfer uses the calibrated preference function, together with the conditions at the policy site, to measure the tradeoff for the change associated with the policy application.

4.3.3. Challenges regarding benefits transfer

Several challenges arise when using benefits transfer. The first stems from possible differences between the study and policy sites. Regardless of the type of transfer method used, economic benefits or economic value functions derived from a particular ecosystem study site will not necessarily be relevant for a different policy site. How people value the preservation or alteration of an ecosystem depends on two dimensions: their preferences and the nature of the biophysical system. Differences in both biophysical characteristics and human values and preferences dictate that great care must be taken in deciding whether the valuation of benefits in one context can be validly used in another context.

Similarities or differences in preferences are likely to depend on how close the stakeholders in the two cases are along social and economic dimensions that influence marginal willingness to pay. For example, income levels or age profiles are sometimes relevant, as in many cases of valuing recreational opportunities. The particular cultural characteristics of the community also may be relevant. For example, in locations where salmon are seen as iconic species reflecting the entire ecosystem (e.g., Seattle), people are likely to value more highly both salmon and water quality important for preserving the salmon.

When only information on willingness to pay per unit of improvement is available, the analyst must be sensitive to the types of differences that would render a transfer inappropriate. If all the differences between the study site and the policy site are such that one is likely to have a higher value per unit of improvement than the other, the study site can provide either a floor or ceiling for the policy site. When the information from the study site is in functional terms (e.g., willingness to pay as a function of income levels or age), social-economic differences between the study site and the policy site can be accommodated if these specifications are valid.

Although it may be possible to adjust for differences in social-economic characteristics of the populations, the capacity to adjust for biophysical differences is typically more limited. For example, even if the affected populations have identical characteristics (or adjustments can be made for their differences), the value of improving the water quality of one small lake in Minnesota is likely to be quite different from improving water quality in a small lake in Texas, because the effects on the overall provision of ecosystem services are likely to be quite different and not captured by a single relationship.

The challenge of transferring benefit estimates is exacerbated by the fact that often few economic benefit studies are available for use. One consequence is that analysts sometimes rely on benefits estimates that are too old to be reliable for new applications. For example, the regulatory impact assessment conducted for the concentrated animal feeding operations (CAFO) rule based its willingness-to-pay estimates for improved water quality on indices taken from a contingent valuation survey conducted by Mitchell and Carson that was more than 20 years old. In addition, due to lack of suitable previous studies, analysts sometimes inappropriately use values or functions derived from studies designed for purposes other than those relevant for the policy site. For example, the Mitchell-Carson study used in the CAFO rule was not intended to apply to specific rivers or lakes. Moreover, the water quality index used by Mitchell and Carson was highly simplified, with no intention of capturing ecosystem services beyond those related to fishing.

An additional challenge stems from the difficulty of finding the most appropriate unit values to carry over from the study site to the policy site. In the example below, illustrating willingness to pay for an improved fishing catch rate, several different metrics of value are possible, and the different metrics will have very different implications for the valuation at the policy site. The choice of unit values also has to be appropriate to the scale and context. For example, willingness to pay for increased wilderness areas in a study site may have been expressed in terms of dollars per absolute increase in area (e.g., $100 per taxpayer annually for a 100-acre increase in area, or $1 per acre). This unit value may be reasonable for a small, heavily populated municipality, but far too high for a municipality with substantially more existing wilderness area.

Willingness to pay for an improved catch rate: The challenge of choosing a unit value for economic benefits transfer

Suppose estimates from the literature imply that the average value of the willingness to pay for a 10% improvement in the catch rate (i.e., fish caught per unit of effort) for a sport fishing trip is $5 per trip. This estimate could be from a study describing specific types of fishing trips by a sample of individuals or it could be an average of several studies.

One approach for developing a unit value transfer would divide $5 by 10% to generate a unit value of $0.50 for each 1% improvement. This strategy implicitly assumes the benefit measure is not influenced by the level of quality – i.e., to be constant for each proportionate improvement.

Another approach would take the same information on average tradeoffs and calculate a unit value using the level of the quality variable, in this case a catch rate that itself embeds another economic decision variable – the effort a recreational fisher devotes to fishing. In this example, the quality or number of fish caught per hour of effort must be known. Suppose that in the study providing the estimated economic benefit, the average number of fish caught with an hour of effort before the improvement was 2. Thus a 10% improvement means that the typical recreationist would catch 0.2 more fish with an hour's effort, implying a unit value of $5 for every additional 0.2 fish caught per hour of effort, or (assuming a linear relationship in terms of the catch rate rather than the proportionate change in this quality measure) $25 for every additional fish caught per hour of effort.

Finally, the unit value could be expressed in terms of improved fishing trips. Suppose the average recreational trip involves 5 hours of fishing over the course of a day. Then the improvement of 0.2 fish per hour implies an average of one more fish caught during a trip. These additional data might be used to imply that the improvement makes typical trips yield incremental economic benefits of $5 per trip (the value of catching 0.2 additional fish per hour for a period of five hours).

There are other ways this estimate could be interpreted. These examples are not intended to be the only "correct" ones or the best. They illustrate that the information on the baseline conditions, the measurement of quality, and the measurement and terms of use all can affect how a given set of estimates is used in a benefits transfer.

For the study site, all three interpretations are simply arithmetic transformations of the data describing the context for the choices that yield the tradeoff estimates. However, the same conclusions do **not** hold when they are transferred to a different situation. Suppose the policy site involves a case where we wish to evaluate the effects of reducing the entrainment of fish in power plant cooling towers. Assume further it is known from technical analysis that this regulation would lead to 5% improvement in fishing success along rivers affected by a rule reducing fish entrainment. Table 3 shows the alternative unit value transfers if these areas have 2000 fishers, each taking about 3 trips per season and currently they catch 1 fish per hour.

Table 3: Table of alternative unit value transfers

Assumption	Unit Value	Interpretation of Policy	Aggregate Value
Constant unit value for a 1% improvement	\$0.50 per 1% improvement	5% improvement per trip	\$0.50 × 5 × 3 × 2000 = \$15,000
Constant unit value for an extra fish caught per hour of effort	\$25 per additional fish per hour	Added fish caught	\$25 × .05 × 1 × 3 × 2000 = \$7,500
Constant value for an improved trip	\$5 per trip	Improved fishing trips	\$5 × 3 × 2000 = \$30,000

Clearly these examples deliberately leave out some important information. Trips may be different – longer, requiring more travel time, or involving different features such as different species or related activities. These added features are aspects omitted in the example. These estimates also do not allow for the possibility that fishing success induces current recreationists to take more trips or that people who never took trips may start taking them after the improvement. Under each of these possible outcomes, the sources for error in the transfer compound. Even without such details, these simple examples illustrate how the aggregate economic benefit measures can differ by a factor of four.

4.3.3.1 Use of screening processes

Screening processes can help address the challenges of determining whether and how to conduct a benefits transfer. This procedural approach assumes that a deliberate effort to examine the similarities and differences between study sites and the policy site, by both EPA analysts and those overseeing their work, will help flag problematic transfers and clarify the assumptions and limitations of the study-site results. Several procedures can be considered, one of which is to contact experts familiar enough with both the previous and current contexts to determine whether to proceed with the economic benefits transfer. These experts can apply the criteria that they regard as relevant, even if the set of criteria are not explicit. Experts knowledgeable in both the study case and the policy case can suggest the most appropriate functional forms and unit values (e.g., Desvousges, Johnson and Banzhaf, 1998). Experts may also be able to suggest other existing valuations that would be better candidates for transfer of willingness-to-pay or willingness-to-accept information.

Another procedure is to make a detailed examination of the appropriateness of the study case a regular part of EPA review of analyses using benefits transfer. Such oversight of the use of case studies would require analysts to clarify the assumptions, purposes, and units of the study-site analysis so that EPA reviewers can judge the appropriateness of the transfer. Analysts must also be fully transparent regarding the origin and context, including the date, of the original valuation.

More thorough cataloguing of existing valuation studies, with careful descriptions of the characteristics and assumptions of each, would be helpful in increasing the likelihood that the most comparable existing valuations will be identified. This is a compelling rationale for developing databases of valuation studies. The establishment and development of a Web-based platform for data and models focusing on valuation estimates would be very worthwhile. Comparable to the Web sites developed and maintained for other large-scale social science research surveys such as the Panel Study on Income Dynamics (PSID) and the Health and Retirement Study (HRS), such a platform could expand the ability of Agency analysts to search for the most appropriate study cases and to supplement these records with related data for transfers. Some efforts along these lines are currently underway. These include the Environmental Valuation Reference Inventory (EVRI), which was developed by Environment Canada in conjunction with other agencies including EPA (see http://www.evri.ca/) , and a database currently being developed for recreational use values (see http://www.cof.orst.edu/cof/fr/research/ruvd/Recreation_Letter.html). However, a more systematic effort across a wide range of ecosystems services is needed (see Loomis and Rosenberger, 2006).

In addition to development and maintenance of a comprehensive database of existing valuation studies, more original valuation studies across a wider range of ecosystem services are needed to increase the Agency's capacity to conduct benefit transfers. The committee urges the Agency to support research of this type. This research will be most useful if conducted with the explicit intention of developing value estimates that EPA can use for benefits transfer. Such an intention can influence how the original valuation studies are conducted and documented. For example, Loomis and Rosenberger (2006) suggest a number of ways of designing original studies to facilitate benefits transfer, such as the use of objective, quantitative measures of quality changes within realistic ranges and the consistent and full reporting of project details.

4.4. Conclusions and recommendations

The valuation approach proposed in this report calls for EPA to allow for the use broader suite of methods than EPA has typically employed in the past for valuing ecosystems and their services. There are a variety of methods that could be used and the committee urges EPA to experiment with the use of alternative methods, where legally permissible and scientifically appropriate. Some of the methods considered by the committee have been used extensively in specific decision contexts (e.g., the use of economic methods in national rule making), while others are still relatively new and in the developmental stages (e.g., citizen juries). The methods also differ in a number of important ways, including the underlying assumptions, the types of values they seek to characterize, the empirical and analytical techniques used to apply them, their data needs (inputs) and the metrics they generate (outputs), and the extent to which they involve the public or stakeholders. For these reasons, the potential for use by EPA in ecological valuation will be different for the different methods. However, the Agency should only use methods that are scientifically based and appropriate for the particular decision context at hand. Thus, as an important first step in implementing the valuation approach proposed in this report, the Agency should develop a set of criteria to use in evaluating methods to determine their suitability for use in specific decision contexts.

Because of time and other resource constraints, EPA is likely to continue to rely heavily on benefits transfer in the ecological valuations that it conducts. The committee advises EPA to explicitly identify relevant criteria related to societal preferences and the nature of the biophysical system of the cases being considered for economic benefits transfer to determine the appropriateness of the transfer. Both EPA analysts and those providing oversight of their work must take into account the differences between study site and policy site to flag problematic transfers and clarify the assumptions and limitations of the study site results. The committee also advises EPA to support efforts to develop Web-based databases of existing valuation studies across a range of ecosystem services, with careful descriptions of the characteristics and assumptions of each, to assist in increasing the likelihood that the most comparable existing valuations will be identified. Finally, because of the importance of benefits transfer to the Agency, additional original research on valuation that is designed to be used in subsequent benefit transfers is particularly important for the Agency.

5 Cross-cutting issues

This chapter addresses three topics important to multiple stages of ecological valuation: analysis of uncertainties related to ecological valuation; communication of ecological valuation information; and the role of deliberative processes.

5.1. Analysis and representation of uncertainties in ecological valuation

5.1.1. Introduction

All aspects of ecosystem valuation efforts – from the estimation of ecological impacts to valuation – are subject to uncertainty, regardless of the methods used. Assessment of this uncertainty allows for a more informed evaluation of proposed policies and of comparisons among alternative policy options. Because any given policy may result in a range of different outcomes, decision makers should have sufficient information regarding what is known about the distribution of possible outcomes in order to take uncertainty into account when they make their policy choices. Identifying key uncertainties can also provide potentially important insights regarding the design of research strategies that can reduce uncertainty in future analyses.

When addressing uncertainty in ecological valuation, four key questions arise: First, what are the major sources of uncertainty and what types of uncertainty are likely to arise when using alternative valuation methods? Second, what methods are available to characterize uncertainty in ecological valuations? Third, how should information regarding uncertainty be communicated to decision makers? Fourth, what types of new research – data collection, improvements in measurement, theory building, theory validation, and others – can reduce uncertainty for particular sources in specific applications? Section 5.2.2 briefly describes the major sources of uncertainty in the valuation of ecosystems and ecosystem services. The overview of specific valuation methods available at www.epa.gov/sab/xxxx discusses the uncertainty arising from the use of individual methods. Section 5.2.3 then discusses two approaches to characterizing uncertainty regarding ecological values: Monte Carlo analysis and expert elicitation. Section 5.2.4 addresses the communication of uncertainty information. Section 5.2.5 discusses how EPA can use uncertainty analysis to set research priorities.

Historically, efforts to address uncertainty in ecological valuations and in all economic benefit assessments that are part of regulatory impact analyses have been limited. Providing greater information about uncertainty is consistent with the need for transparency and can improve decision making. In the context of regulatory impact analyses, Office of Management and Budget Circular A-4 explicitly calls for analysis and presentation of important uncertainties. To assess the level of confidence to attribute to projections used in a valuation, decision makers must know the analyst's judgment of the uncertainty of the valuation and its component steps, as well as the assumptions underlying the valuation analysis.

5.1.2. Sources of uncertainty in ecological valuations

As discussed in chapters 3 and 4, ecological valuation entails several analytic steps, each potentially subject to uncertainty. These steps include predicting ecological impacts of the relevant Agency decision or action predicting the effects of these impacts on ecosystem services and valuing the consequences of these effects.[31] Although it might be tempting to limit attention to uncertainty in the final step, uncertainties in each stage of the analysis are of potential importance, and there is no reason – on the basis of theory alone – to judge one to be more important than the other. Rather, the relative magnitude of the uncertainty involved in each step is fundamentally an empirical question.

At each stage, uncertainty can arise from several sources:[32] First, some of the physical processes might be inherently random or stochastic. Second, there can be uncertainty about which of several alternative models of the process best captures its essential features. Finally, there are uncertainties involved in the statistical estimation of the parameters of the models used in the analysis.

At the biophysical level, for example, any characterization of current or past ecological conditions will have numerous interrelated uncertainties. Any effort to project future conditions, with or without some postulated management action, will magnify and compound these uncertainties. Ecosystems are complex, dynamic over space and time, and subject to the effects of stochastic events (such as weather disturbances, drought, insect outbreaks, and fires). Also, our knowledge of these systems is incomplete and uncertain. Errors in projections of the future states of ecosystems are thus unavoidable and constitute a significant and fundamental source of uncertainty in any ecological valuation.

All social, economic, or political forecasts are also based on implicit or explicit theories of how the world works, either represented by the mental models of the forecasters or by the mental models underlying the formal and explicit methods used in econometric modeling, systems dynamics modeling, and other forms of modeling. Theories and their expressions as models are unavoidably incomplete and may simply be incorrect in their assumptions and specifications.

Valuation methods are also subject to data and theory limitations. They unavoidably rely on assumptions that introduce uncertainty. In addition, analysts are often required to apply estimated values to contexts that differ from those in which the values were developed. The possibility that appropriate adjustments have not been made in transferring estimates to different contexts introduces another source of uncertainty.

In identifying the types of uncertainty most likely to be of concern for individual valuation approaches in specific contexts, two issues are relevant: the sensitivity of the approach to the potential sources of uncertainty listed above, and the magnitude of uncertainty thereby generated. The consequence of data limitations can be assessed by determining the variation in results implied by variations in data. Vulnerability to theoretical limitations is more difficult to assess, but can be gauged in some cases by comparing predictions based on alternative models.

5.1.3. Approaches to assessing uncertainty

Probabilistic uncertainty analysis, by its very nature, is complex, particularly in the context of ecological valuation. The simplest and probably most common approach to evaluating uncertainties is some form of sensitivity analysis, which typically varies one parameter or model assumption at a time and calculates point estimates for each of the different parameter values or assumptions. The results provide a range of estimates of the "true" value, including lower and upper bounds. No effort is made to assign probabilities to the calculated values or estimate the shape of the distribution of values within the range.

Although sensitivity analysis may be sufficient for some simple problems, its use in the context of ecological valuation is likely to give an incomplete and potentially misleading picture of the true uncertainty associated with the value estimates. Due to the number of sources of uncertainty in many ecological valuations, sensitivity analysis is unlikely to account for the implications of all the sources of uncertainty. In addition, sensitivity analysis becomes unwieldy when the outcomes relevant to the value assessment themselves consist of multiple interrelated variables. For example, it is extremely difficult at the biophysical level to calculate the uncertainty in projecting outcomes from a complex ecological system composed of multiple interacting variables subject to the influence of external stochastic events.

Because of the limitations of simple sensitivity analysis, other approaches to characterizing uncertainty have been developed. These include Monte Carlo analysis and the use of expert elicitation. These approaches can provide a more useful and appropriate characterization of uncertainty in complex contexts such as ecological valuation.

Monte Carlo analysis is an approach to characterizing uncertainty that allows simultaneous consideration of multiple sources of uncertainty in complex systems. It requires the development of a model to predict the system's outputs from information about inputs (including parameter values). The underlying inputs that are uncertain are assigned probability distributions. A computer algorithm is then used to draw randomly from all of these distributions simultaneously (rather than one at a time, as in sensitivity analysis) and to predict outputs that would result if the inputs took these values. By repeating this process many times, the analyst can generate probability distributions for outputs that are conditional on the distributions for the inputs.

Developments in computer performance and software have substantially reduced the effort required to conduct calculations for a Monte Carlo analysis once input uncertainties have been characterized. Widely available software allows the execution of Monte Carlo analysis in common spreadsheet programs on a desktop computer. In developing probability distributions for uncertain inputs, uncertainty from statistical variation can also often be characterized with little additional effort relative to that needed to develop point estimates. Much of the needed data already will have been collected for the development of point estimates (although characterizing other sources of uncertainty in inputs can require more effort).

The use of Monte Carlo analysis can provide a more reliable and richer characterization of uncertainty than simple sensitivity analysis. In contrast to sensitivity analysis, Monte Carlo analysis provides information on the likelihood of particular values within a range, which is essential to any meaningful interpretation of that range. Without such an understanding, the presentation of a range of possible outcomes may lead to inappropriate conclusions. For example, a reader may assume that all values within the range are equally likely to be the ultimate outcome, even though this is rarely the case. Others may assume that the distribution of possible values is symmetric. This, also, is often not the case.

Because of its ability to characterize uncertainty in a more meaningful way, Monte Carlo analysis has become common in a variety of fields, including engineering, finance, and a number of scientific disciplines. It has been useful in policy contexts. EPA recognized as early as 1997 that it can be an important element of risk assessments (EPA, 1997). Circular A-14, in calling for the analysis and presentation of uncertainty information as part of regulatory analyses, also notes the potential use of Monte Carlo analysis. However, efforts to quantify uncertainties through Monte Carlo analyses rarely have been undertaken in ecological valuations. More often, uncertainty has been addressed qualitatively or through sensitivity analysis.

One of the challenges in applying Monte Carlo methods is that reliable application requires not just specification of variances on key variables, but also co-variances across the variables. Without appropriate co-variances, the method is less reliable. Positive co-variance increases the spread of results, while negative co-variances decrease the spread.

Where Monte Carlo analysis can be reliably used in the estimation of ecological values, the analysis is unlikely to address all sources of uncertainty. Thus, the results will likely understate the range of possible outcomes that could result from the relevant public policy. Nonetheless, the ranges produced will still provide more reliable information about the implications of known uncertainties than simple sensitivity analysis. In turn, these ranges can better inform judgments by policy makers as to the overall implications of uncertainty for their decisions. The committee therefore urges EPA to move toward greater use of Monte Carlo analysis, where feasible, as a means of characterizing the uncertainties associated with estimating the value of ecological protection.

A host of expert elicitation methods can also provide indications of the amount and nature of uncertainty associated with estimates of specific values or predictions regarding the impacts of a given activity or change (e.g., Morgan and Henrion, 1990; Cleaves, 1994). In its simplest form, an expert elicitation is a single expert's assessment of the uncertainty of an estimate, forecast, or valuation, whether it is based on implicit judgment or a more explicit approach like the Monte Carlo technique. Policy makers can elicit more information from the expert, such as the assumptions underlying his or her analysis or the bases for uncertainty, to better understand the reliability of the expert's input and the nature of the uncertainty.

Although an elicitation can rely on a single expert, the bulk of expert elicitation methods involve multiple experts, which allows for a comparison of their judgments and an assessment of any disagreements. If the experts are of equal credibility, so that no judgment can be discarded in favor of another, the range of disagreement reflects uncertainty. If top scientists strongly diverge in their estimates, forecasts, or valuations, the existence of a high level of uncertainty is irrefutable. This relationship, however, is asymmetrical because narrow disagreement does not necessarily reflect certainty. The experts may all be equally wrong, a somewhat common occurrence given that experts often pay attention to the same information and operate within the same paradigm for any given issue (Ascher and Overholt, 1983). When experts interact before providing their final conclusions (e.g., by exchanging estimates and adapting them to what they learn from one another), errors due to incompleteness can be reduced. For example, biologists may benefit from the kind of information that atmospheric chemists can provide, and vice versa. Such interactions, however, run the risk of "groupthink" – the unjustified convergence of estimates due to psychological or social pressures to come closer to agreement (Janis, 1982).

For many expert elicitation methods, translation into probabilities is difficult. Simple compilations of estimates (e.g., contemporaneous estimates of species populations) from different experts can generate a table with the range of estimates. However, these compilations are unable to convey the degree of uncertainty that each expert would attribute to his or her estimate. Including confidence intervals can provide this information.

The committee believes that expert elicitation should be used to characterize uncertainty when more formal uncertainty analysis (e.g., Monte Carlo analyses) is not feasible. In addition, the committee recommends that EPA use expert elicitation to obtain estimates of parameters and their uncertainty for use in Monte Carlo analysis, if suitable information about the relevant range for the parameter values is not available based on observation (e.g., field work or experiments).

5.1.4. Communicating uncertainties and ecological valuation

It is important not only to analyze the sources and size of uncertainty involved in a valuation but also to effectively communicate that uncertainty to decision makers. In the past, point estimates have been given far greater prominence in public documents such as regulatory impact assessments and other government valuations than discussions of the uncertainty associated with them. Uncertainty assessments are often relegated to appendices and discussed in a manner that makes it difficult for readers to discern their significance. This result may be inevitable, given that single-point estimates can be communicated more easily than lengthy qualitative assessments of uncertainty or a series of sensitivity analyses. The ability of Monte Carlo analysis to produce quantitative probability distributions, however, provides a means of summarizing uncertainty that can be communicated nearly as concisely as point estimates. If a summary of uncertainty is not given prominence relative to an estimate itself, decision makers will lose both the context for interpreting the estimate and opportunities to learn from the uncertainty.

Some resistance to the use of formal uncertainty assessments such as through Monte Carlo analysis, and to the prominent presentation of the results, may be due to the perception that such analysis requires greater expert judgment and therefore renders the results more speculative. Also, some might argue that, given the inevitably incomplete nature of any uncertainty analysis, prominently presenting its results could incorrectly lead readers to conclude that the results of an ecological valuation are more certain than they actually are. Both concerns are generally unfounded. As described above, developing characterizations of uncertainty, such as for inputs in a Monte Carlo analysis, often simply involves making explicit and transparent expert judgments that already must be made to develop point estimates for those inputs. To the extent that an uncertainty analysis is incomplete in its characterization of uncertainty, that fact can be communicated qualitatively.

5.1.5. Using uncertainty assessment to guide research initiatives

Over time, additional research related to data collection, improvements in measurement, theory building, and theory validation can reduce the uncertainties associated with ecological valuation. For example, research can improve our understanding of the relationships governing complex ecological systems and thereby reduce the uncertainty associated with predicting the biophysical impacts of alternative policy options. Even stochastic uncertainty can sometimes be addressed by initiating research that focuses on factors previously treated as exogenous to the theories and models. For example, an earthquake-risk model based on historical frequency will have considerable random variation if detailed analysis of fault-line dynamics is excluded; bringing fault-line behavior into the analysis can lead to reductions in such uncertainty (Budnitz et al., 1997).

Assessments of the magnitude and sources of uncertainty can help to establish research priorities and to inform judgments about whether policy changes should be delayed until research reduces the degree of uncertainty associated with possible changes. Determining whether the major source of uncertainty comes from weak data, weak theory, randomness, or inadequate methods can help guide the allocation of scarce research funds. Some data needs will simply be too expensive to fulfill, and some methods have intrinsic limitations that no amount of refinement will fully overcome. Uncertainty analysis can provide insight into whether near-term progress in reducing uncertainty is likely, based on the sources of uncertainty and the feasibility of addressing these limitations promptly. However, it is important to avoid the pitfall of delaying a necessary action simply because some uncertainty remains – because uncertainty always will remain.

5.2. Communication of ecological valuation information

The success of an integrated and expanded approach to ecosystem valuation depends in part on how EPA obtains information about public concerns during the valuation process and then communicates the resulting ecological valuation information to decision makers and the public. Although the committee has not extensively discussed the communication challenges presented by ecological valuation, it believes that generally accepted practices for communication of technical information apply to the valuation context. Section 5.2.1 discusses general practices of particular relevance to valuation. Section 5.2.2 addresses the special communication challenges that arise for ecological valuation.

Three essential functions of communication in valuing the protection of ecological systems and services are:

- Communication among and between technical experts and the public within the valuation process itself

- Communication of valuation information by analysts to decision makers
- Communication of the results of the valuation and decision making processes to interested and affected members of the public.

Although these communication functions may appear to be separate steps, they overlap. The success of the overall valuation process and any communication step within it, for example, depends on understanding how decision makers use valuation information. Spokespersons must understand how different public groups and experts frame valuation issues before they can effectively communicate the results of a formal valuation analysis.

5.2.1. Applying general communication principles to ecological valuation

Effective communication should be designed for the relevant audience of the valuation information. The potential pool of interested parties include decision makers, interested and affected members of the public, and experts in social, behavioral, and economic sciences and ecological sciences. A broad public audience is likely to be interested in better understanding the value of protecting ecological systems and services. Also important is an intermediate audience of analysts, who serve as important mediators for valuation information through their analyses and activities. This latter audience needs to access not only value estimates but also technical details and models. To support decisions effectively, communications must be designed to address a recipient's goals and prior knowledge and beliefs, taking into account the effects of context and presentation (Morgan et al., 2002). The committee recommends that EPA formally evaluate the communication needs of the users of valuations and adapt valuation communications to those needs.

An effective communication strategy also requires interactive deliberation and iteration (NRC, 1996). Effective communication of values requires systematic interactions with interested parties, where the interaction will differ depending on the technical expertise and focus of the parties. In general, interactive processes are critical for improving understanding, although reports (such as EPA's *Draft Report on the Environment*) are also important, especially in the context of assessment.

Basic guidelines for risk and technical communication are generally applicable to communicating ecological values. Linear graphs, for example, are likely to convey trends more effectively than tables of numbers (Shah and Miyake, 2005), and text that incorporates headers and other reader-friendly attributes will be more effective than text that does not (Shriver, 1989). In developing effective communication approaches for ecological valuation, EPA can look to guidelines developed for risk and technical communication. Two useful examples of such guidelines are the communication principles in EPA's *Risk Characterization Handbook* (EPA Science Policy Committee, 2000) and the guidelines for effective web sites (Spyridakis, 2000). The principles in the *Risk Characterization Handbook* include transparency, clarity, consistency, and reasonableness. Spyridakis, in turn, provides guidance in five categories: content, organization, style, credibility, and communicating with international audiences. Spyridakis provides a concise table for communicating information via Web sites and provides generally accepted guidance useful for communication of valuation information, including:

- Select content that takes into account the reader's prior knowledge.
- Group information in such a way that it facilitates storing that information in memory hierarchically.
- State ideas concisely.
- Cite sources appropriately, and keep information up to date.

As in the case of any type of communications, it is difficult to predict the effects of communicating ecological valuations. Good communications practice requires formative evaluation of the communications as part of the design process. Summative evaluation after the fact will enable assessments of effectiveness, leading to continued improvement in communications (e.g., Scriven, 1967; Rossi et al., 2003). The committee recommends that EPA evaluate its communication of ecological valuations to assess the effects of the communication and to learn how to improve upon Agency communication practices.

5.2.2. Special communication challenges related to ecological valuation

Although application of these general communication principles will improve communications of ecological valuations, special challenges arise in this context.

First, communicating the value of protecting ecological systems and services requires conveying not only value information (in terms of metrics such as monetized values and rating scales), but also information about the nature, status, and changes to the ecological systems and services to which the value information applies. The EPA Science Advisory Board review of EPA's *Draft Report on the Environment* (EPA Science Advisory Board, 2005) and other reports (e.g., Schiller et al., 2001; Carpenter et al., 1999; Janssen and Carpenter, 1999) emphasize that people need to understand the underlying causal processes in order to understand how ecological changes affect the things they value, such as ecosystem services.

The causal processes can be conveyed using such visual tools as mapped ecological information, photographs, graphs, and tables of ecological indicators. To the extent that such visual outputs – especially outputs from integrated geographic information systems using best cartographic principles and practices (Brenner, 1993) – can be interactive, the outputs will facilitate sensitivity analysis that can address audience questions about scale and aggregation and may be more effective as communication tools. The EPA Science Advisory Board has proposed this kind of framework for reporting on the condition of ecological resources (EPA Science Advisory Board, 2004). EPA's *Draft Report on the Environment* (EPA, 2002) and Regional Environmental Monitoring and Assessment Program reports illustrate a range of representational approaches.[33]

Second, the many uses and definitions of the term "value" complicate the communication of ecological values. The broad usage of the term in this report includes all the concepts of value described in Table 1. Context and framing can strongly influence how people rank, rate, and estimate values (Hitlin and Piliavin, 2004; Horowitz and McConnell, 2002), as well as how they interpret value-related information (e.g. Lichtenstein and Slovic, 2006).

As discussed elsewhere in this report, value measures are required or useful in a variety of regulatory and non-regulatory contexts, ranging from national rule makings, to site-specific decision making and prioritization of environmental actions, to educational outreach in regional partnerships. In some cases monetization is required, whereas in others (e.g., educational outreach by regional partnerships), narratives and visual representations of values may play a more important role.

Little direct evidence exists about how people perceive alternative value measures. However, survey and decision research is suggestive. Because survey response scales tend to promote responses congruent with their structure, asking people for ecological value in dollars will likely elicit those values that are most readily expressed in dollars and not those that are difficult to express in dollars. However, numerical information alone provokes weak, if any, effect and is unlikely to significantly influence respondents' estimates of the value of the stimulus (e.g., Dunn and Ashton-James, 2007), as demonstrated by studies on scope insensitivity. Further, visual information often dominates other representations. Taken together, this suggests that monetized values will more strongly influence quantitative benefit-cost analyses than qualitative or non-monetized quantitative information that is not readily included in a benefit-cost calculus. It also suggests that attitudes, opinions, and values elicited based on qualitative and visual stimuli could dominate those elicited based on numbers alone, unless the numbers have special significance (such as money).

One mechanism for mitigating disconnects when reporting ecological values in different metrics is to employ an iterative, interactive approach to eliciting, studying, and communicating values and tradeoffs, where values are represented in multiple ways. Verbal quantifiers (e.g., "many" or "very likely") , for example, may make technical information more accessible, but the wide variability with which these terms are interpreted (Budescu and Wallsten, 1995) makes it critical to make the underlying numerical information readily available. Appropriate use of graphical and visual approaches, including geographic information systems, can aid interpretation of quantitative information. Visualization can facilitate new insights (MacEachren, 1995).

Third, in many circumstances, interactive communication of ecological valuation information is likely to be more effective than static displays. Interactive communication allows users to manipulate the data or representations of the data, such as with sliders on interactive simulations. Interactive visualization has the potential to allow users to tailor displays to reflect their individual differences and questions. Even with exactly the same presentation, people's understandings of content vary because of differences in educational or cultural background, and different intellectual abilities. Interactive exploration tools give the audience a chance to investigate freely the part in which they are interested or about which they have questions.

As Strecher, Greenwood, Wang, and Dumont (1999) argue, the advantage of interactivity lies in:

- Allowance for active, instead of passive, participation of the audience
- The ability to tailor information for individual users
- The ability to assist the assessment process
- The ability to visualize possible risks under different hypothesized conditions (allow users to ask "what if" questions).

Interactivity is a good solution if the complexity of the visualization has the potential to overwhelm users (Cliburn, Feddema, Miller, and Slocum, 2002). Interactive visualization nonetheless poses challenges as well. 3-D visualization, which has become increasingly popular in visualization practice (Encarnacao et al., 1994), both necessitates interactivity and at the same time challenges it because of the sheer computational power required.

5.3. Deliberative Processes

Deliberative processes, in which analysts, stakeholders, decision makers, and/or other members of the public meet in facilitated interactions, can be useful in estimating and valuing the potential effect of EPA actions on ecosystems and their related services. Such processes can assist at several steps of an assessment, ranging from developing conceptual models and determining the ecosystem services on which the Agency should focus its assessment to valuing those services. For example, where the public is not familiar with key ecosystem services, deliberative processes can provide the public with expert information that may better enable them to identify what services are important to them and to value those services. Similarly, where the public is not accustomed to valuing particular ecosystem services, deliberative processes may again help members of the public estimate the value that they would place on those services. Deliberative processes also can increase public understanding and acceptance of a valuation effort and, where appropriate, permit the public to play a more active role in shaping and analyzing options.

Two specific types of deliberative processes of potential use to EPA in particular valuation efforts are mediated modeling and constructed value processes. In mediated modeling, analysts work with stakeholders to develop a model representing a particular environmental system of interest, ranging from watersheds or local ecosystems to large regions or even the globe (for example, Higgins et al., 1997; Cowling and Costanza, 1997; Van den Belt, 2004). Stakeholders participate in all stages of the modeling process, from initial problem scoping to model development, implementation, and use. The resulting model can be used for multiple purposes, including determining the ecosystem services that are potentially important to the public and evaluating alternative scenarios or options of interest. If the model is to be used to consider tradeoffs, the model must incorporate values drawn from methods described in Chapter 4. Because of the stakeholder involvement in the modeling process, the model and any results derived from it are likely to enjoy stakeholder buy-in and reflect group consensus.

Constructed value processes can help in both estimating values and, in some cases, making policy decisions. A central premise of constructed value processes is that people's preferences and values for complex, unfamiliar goods, such as many ecosystem services, are multi-dimensional and that people sometimes construct their preferences and values for such goods during the process of elicitation. This premise contrasts with the premise underlying some valuation methods, most notably economic valuation methods, that assume preferences are given and that values or contributions to well-being can be measured using a single metric such as willingness to pay or accept.

Constructed value processes can be used either as part of an evaluation process or directly in decision making. In both situations, constructed value processes involve a number of steps, including identifying objectives, defining the attributes to be used to judge progress toward the objectives, specifying the set of management options, and measuring changes in relevant attributes under the options (Gregory et al., 1993; Gregory et al., 2001; Gregory and Wellman, 2001). Objectives are diverse and often multi-dimensional. Examples include maintaining some requisite level of ecological services, protecting endangered or threatened species, producing particular resources, increasing tourism or recreational opportunities, and supplying a sense of pride or awe (Gregory et al. 2001). The final output is either a judgment about the current state of the system relative to an alternative state (if the context is evaluative) or the selection or identification of a preferred management option (if the context is decision making). Constructed value processes draw on inputs from a variety of disciplines, including economics, ecology, psychology, and sociology.

EPA, however, must use considerable care in using stakeholder processes, as previously noted by the Science Advisory Board (SAB 2001). To be effective, deliberative processes involving stakeholders must adequately address and incorporate relevant science and must receive necessary support. This in turn requires the provision of substantial financial resources, adequate time, and high-quality staff. EPA must take special care in using deliberative processes involving stakeholders to 1) decide how stakeholder processes should be used as input for valuation and 2) to distinguish use of stakeholder processes as input for valuation from other uses of stakeholder processes as aids to decision making. As noted in the SAB report cited above, stakeholder processes are appropriate for making decisions in only a modest subset of environmental regulatory decisions if at all and only under careful conditions.

5.4. Conclusions & Recommendations

Providing information to decision makers and the public about the level of uncertainty involved in ecosystem valuation efforts is critical for the informed evaluation of proposed policies and alternative policy options. The committee makes the following recommendations to ensure the effective analysis and representation of uncertainties in ecological valuations:

- :In assessing uncertainty, EPA should go beyond simple sensitivity analysis and make greater use of approaches, such as Monte Carlo analysis and expert elicitation, that provide a useful and appropriate characterization of uncertainty for the complex contexts of ecological valuation. Sensitivity analysis is unlikely to account for all sources of uncertainty in ecological valuation and can become unwieldy when value outcomes consist of multiple interrelated variables.
    - Where feasible, the Agency should use Monte Carlo analysis to characterize uncertainties.
    - Where more formal uncertainty analysis such as Monte Carlo analysis is not feasible, EPA should use expert elicitation. EPA should also use expert elicitation to obtain estimates of parameters and their uncertainty for use in Monte Carlo analysis, if suitable information about the relevant range for the parameter values is not available based on observation.
- The Agency should not relegate uncertainty analyses to appendices but should ensure that a summary of uncertainty is given as much prominence as the valuation estimate itself. EPA should also explain qualitatively any limitations in the uncertainty analysis.

- EPA should invest in additional research designed to reduce the uncertainties associated with ecological valuation through data collection, improvements in measurement, theory building, and theory validation. Assessments of the magnitude and sources of uncertainty can help to establish research priorities inform judgments about whether policy changes should be delayed until research reduces the degree of uncertainty associated with possible changes. The Agency, however, should not delay a necessary action simply because some uncertainty remains, because uncertainty always will remain.

The success of ecological valuations also depends on how EPA obtains information about public concerns during the valuation process and then communicates the resulting ecological valuation information to decision makers and the public. To promote effective communications, the committee recommends the following steps:

- EPA should evaluate the users of valuation information and their needs and adopt communications that are responsive to those needs.
- In communicating ecological valuation information, the Agency should follow basic guidelines for risk and technical communication. EPA's *Risk Characterization Handbook* (U.S. EPA Science Policy Committee 2000) provides one set of useful guidelines, including transparency, clarity, consistency, and reasonableness.
- EPA should evaluate its communication of ecological valuations to assess its effects and to learn how to improve upon its practices.
- Where feasible, the Agency should communicate not only value information but also information about the nature, status, and changes to the ecological systems and services to which the value information applies. Visual tools such as mapped

ecological information, photographs, graphs, and tables of ecological indicators can be very useful in conveying causal processes.

- Where appropriate, the Agency should employ an iterative, interactive approach to communicating values.

6 Applying the approach in three EPA decision contexts

This chapter discusses implementing the C-VPESS ecological valuation approach in three specific EPA decision contexts: national rule making, site-specific decision making, and regional partnerships. The committee believes that improved ecological valuation in each context can contribute to improved policy analysis and decisions. The committee examined a number of illustrative examples for each decision context and used these examples to inform its views about application of the approach advocated in this report.

The discussions below elaborate on the three key features of the valuation approach advocated in this report as they relate to the specific decision contexts:

- Identifying and focusing on impacts that are likely to be most important to people early in the process
- Predicting ecological changes in value relevant terms
- Using multiple methods in the valuation process.

The discussions are meant to be illustrative rather than comprehensive and the exclusion of a particular method from discussion in a specific context is not intended to suggest inappropriateness. Note that the general principles and concepts used in the discussions below are described in more detail elsewhere in this report (see, for example, chapter 4 and appendix A for descriptions of individual methods).

6.1. Valuation for national rule making

6.1.1. Introduction

This section examines the valuation of ecosystem services as part of the assessment of the economic benefits and costs of national rules promulgated by the Agency and recommends how to implement the committee's framework in this context. As background for this discussion, the committee examined three examples of previous Agency economic benefit assessments:

- The Agency's assessment for the final effluent guidelines for the aquaculture industry (EPA, 2004)
- The Agency's assessment for the 2002 rule making regarding concentrated animal feeding operations (CAFOs) (EPA, 2002; chapter 2 also discusses this benefit assessment
- The prospective analysis of the benefits of the Clean Air Act Amendments of 1990 (EPA, 1999).[34]

Brief descriptions of the three benefit analyses are presented later in this section. These examples provide insights reflected in the discussion and recommendations throughout this section.

6.1.2. Valuation in the national rule making context

As noted previously, valuation by EPA in the national rule making context is typically subject to constraints imposed by statute, executive order, and/or guidance from the Office of Management and Budget (OMB). Most of the environmental laws administered by the Agency require that regulations such as environmental quality standards and emissions standards be based on criteria other than economic benefits and costs. In some cases, the legislation explicitly precludes consideration of costs or benefits in the standard-setting process. For example, under the Clean Air Act, primary ambient air quality standards for criteria air pollutants must be set to protect human health with an adequate margin of safety. Even where a law, such as the Safe Drinking Water Act, allows consideration of benefits and costs, adherence to a strict "benefits must exceed costs" criterion is not required.

However, even when national EPA rules are not determined by a strict benefit-cost criterion, assessments of the benefits and costs of EPA actions, conducted under prescribed procedures, can be important for a number of reasons. First, Executive Order 12866 (as amended by Executive Order 13422), requires federal agencies to “assess both the costs and the benefits of the intended regulations, and … propose or adopt a regulation only upon a reasoned determination that the benefits of the intended regulation justify its costs" (Executive Order 12866, October 4, 1993). These assessments are commonly referred to as regulatory impact assessments (RIAs). They generally evaluate, in economic terms, the form and stringency of the rules that are established to meet some other objective, such as protection of human health.

Second, in some cases, an assessment of economic benefits and costs can be mandated by law. For example, Section 812 of the Clean Air Act Amendments of 1990 requires the Agency to develop periodic reports to Congress that estimate the economic benefits and costs of various provisions of the Act.

Finally, the benefit and cost estimates developed in national rule making can help in setting research or legislative priorities.

In summary, a complete, accurate, and credible analysis of the benefits and costs of a given rule can have broad impacts, even if the analysis does not determine whether the current rule is enacted.

In conducting RIAs, EPA is subject to requirements specified by OMB guidance, and all EPA benefit assessments are subject to OMB oversight and approval. As noted in chapter 2, OMB’s Circular A-4 (OMB, 2003) makes it clear that Executive Order 12866 requires an economic analysis of the benefits and costs of proposed rules conducted in accordance with the methods and procedures of standard welfare economics. In the context of national rule making, the terms benefit and cost thus have specific meanings. To the extent possible, EPA must assess the benefits associated with changes in goods and services as the result of a rule, judged by the sum of the individuals’ willingness to pay for these changes. Similarly, the costs associated with regulatory action are to be evaluated as the losses experienced by people, and measured as the sum of their willingness to accept compensation for those losses. EPA must begin the analysis by specifically describing environmental conditions in affected areas, both with and without the rule. EPA must then value these changes based on individual willingness to pay and to accept compensation, aggregated over the people (or households) experiencing them. Although other valuation methods described in chapter 4 may yield monetary estimates of value, monetizing values using multiple methods and then aggregating the resulting estimates would mean combining estimates that are based on quite different theoretical constructs, as well as diverse underlying assumptions. Thus, for both theoretical and empirical consistency – as well as compliance with OMB guidance – monetization of benefits in the context of an RIA should be based on economic valuation.

Circular A-4 recognizes that it may not be possible to express all benefits and costs in monetary terms. In these cases, it calls for measurement of these effects in biophysical terms. If that is not possible, there should be a qualitative description of the benefits and costs (OMB, 2003, p. 10). Circular A-4 is clear about what should be included in regulatory analyses, but it does not preclude the inclusion of information drawn from non-economic valuation methods. Nonetheless, it implies that when conducting ecological valuation in the context of national rule making, EPA must seek to monetize benefits and costs using economic valuation methods as much as possible.

Although economic valuation methods are well-developed and there is a large literature demonstrating their application, applying these methods to the ecological benefits of a national-level rule raises significant challenges. A key challenge is the difficulty of deriving a national estimate of the effect of an EPA rule on ecosystems and the services derived from these ecosystems. Such a national estimate requires information about changes in stressors resulting from the action, as well as information about how the changes in stressors will affect ecosystems and the flow of services. In many rule-making contexts, predicting the changes in stressors is difficult. Often, the rule prescribes adoption of a particular technology or a particular behavior (e.g., adoption of best management practices) rather than a specific change in stressors (e.g., discharge limits). The aquaculture rule associated with the Clean Water Act, described below, provides an example. In those cases, to estimate associated benefits, EPA must predict the changes in stressors that would likely result from the required behavioral change.

Valuation and the Aquaculture Effluent Guidelines

Title III of the Clean Water Act gives EPA authority to issue effluent guidelines that govern the setting of national standards for wastewater discharges to surface waters and publicly owned treatment works (municipal sewage treatment plants). The standards are technology-based, i.e., they are based on the performance of available treatment and control technologies. The proposed effluent guidelines for the concentrated aquatic animal production industry (aquaculture) would require that all applicable facilities prevent discharge of drugs and pesticides that have been spilled. In addition, facilities must minimize discharges of excess feed and develop a set of systems and procedures to minimize or eliminate discharges of various potential environmental stressors. The rule also includes additional qualitative requirements for flow-through and recirculating discharge facilities and for open water system facilities (EPA, 2004).

The Agency identified the following potential ecological stressors that might be affected by the rule: solids; nutrients; biochemical oxygen demand from feces and uneaten food; metals (from feed additives, sanitation products, and machinery and equipment); food additives for coloration; feed contaminants (mostly organochlorides); drugs; pesticides; pathogens; and introduction of non-native species. Some of these (e.g., drugs and pathogens) were thought by the Agency to be very small in magnitude and not require further analysis. To this list, C-VPESS would add habitat alteration from changes in water flows.

For most of these stressors, it is not possible to specify the change that would result from the rule for two reasons. First, the rule called for adoption of "best management practices" rather than imposing specific quantitative maximum discharge levels. Second, for most of these stressors, baseline data on discharges in the absence of the rule were not available.

The Agency analyzed the effects of the rule on dissolved oxygen, biochemical oxygen demand, total suspended solids, and nutrients (nitrogen and phosphorus). There appear to have been three reasons why the remaining endpoints were not quantified:

- The Agency lacked data on baseline stressor levels.
- The rule called for adoption of “best management practices” rather than imposing specific quantitative maximum discharge levels, and the Agency lacked information on how these requirements would change the levels of stressors.

- The Agency did not use a model capable of characterizing a wide range of ecological effects. The Agency used QUAL2E rather than the available AQUATOX model. The choice of QUAL2E appears to have been driven largely by the ability to link its outputs with the Carson and Mitchell valuation model (1993).

The Agency estimated benefits for recreational use of the waters and non-use values. To estimate these values, the Agency estimated changes in six water quality parameters for 30-mile stretches downstream from a set of representative facilities and calculated changes in a water quality index for each facility. The Agency then used an estimated willingness-to-pay function for changes in this index taken from Carson and Mitchell. Carson and Mitchell had asked a national sample of respondents to state their willingness to pay for changes in a water quality index that would move the majority of water bodies in the United States from one level on a water quality ladder to another, resulting in improvements that would allow for boating, fishing, and swimming in successive steps. The aggregate willingness-to-pay for the change in the water quality index for each representative facility was then used to extrapolate to the population of facilities of each type affected by the rule.

A rule will often involve many stressors with complex interactions, which greatly complicates the development of quantitative estimates of changes in stressors. The CAFO rule, described in chapter 2 and below, is an illustration.

Valuation and the CAFO Effluent Guidelines

In December 2000, in response to structural changes in the industry, EPA proposed a new rule to govern discharges from CAFO facilities. The new rule, which was finalized in December 2003, requires facilities to implement comprehensive nutrient management plans designed to reduce the runoff of pollutants from feedlots and from the land application of manure. The rule focuses on the largest operations that represent the greatest environmental threats.

Manure from livestock operations produces a variety of potential pollutants that can migrate to ground water, streams, rivers, and lakes. These pollutants include nitrogen, phosphorus, sediments and organic matter, heavy metals, salts, hormones, antibiotics, pesticides, and pathogens (over 150 pathogens found in manure are human health risks). CAFO facilities also release a variety of gases and material into the atmosphere including particulates, methane, ammonia, hydrogen sulfide, odor-causing compounds, and nitrogen oxides.

Of the water-polluting materials covered in the CAFO rule, excess nutrients can directly affect human water supply through excess nitrates, impacts on agriculture through excess salts in irrigation waters, and cause eutrophication of water bodies, anoxia, and toxic algal blooms. These latter effects can result in fundamental changes in the structure and functioning of aquatic ecosystems, including cascading effects that reduce water quality and species diversity. Uncontrolled releases of animal wastes have resulted in massive fish mortality.

Pathogens in polluted waters are a health hazard, both directly and through the food chain. The potential human health impacts of antibiotics and hormones in wastes have not been well identified but are of concern.

Of all the potential environmental impacts, the CAFO economic benefits analysis focused to a large extent on the nutrient runoff from land where manure has been applied and the economic benefits that would accrue from the manure management requirements of the CAFO rule. To estimate the benefits, the analysis utilized the GLEAMS model (Groundwater Loading Effects of Agricultural Management Systems). The outputs include nutrients, metals, pathogens, and sediments in surface runoff and ground-water leachate. This model was applied to model farms of different sizes, animal types, and geographic regions. From this model the reductions in pollutant loading of nutrients, metals, pathogens, and sediments were estimated for large- and medium-sized CAFO.

Seven categories of economic benefits were estimated: water-based recreational use (by far the largest category), reduced numbers of fish kills, increased shellfish harvest, reduced ground water contamination, reduced contamination of animal water supplies, and reduced eutrophication of estuaries. Reductions in fish kills and animal water supply contamination were valued using replacement cost. Increased shellfish harvests were valued using estimated changes in consumer surplus. Water-based recreation was valued using the Carson and Mitchell (1993) study. Ground water contamination was valued using economic benefits transfer based on a set of stated-preference studies. There was no national estimate of the economic benefits of reduced eutrophication of estuaries, but there was a case study on one estuary focusing on recreational fishing and using economic benefits transfer based on revealed-preference random utility models.

A number of potential impacts were not included in the economic benefits analysis relating to the water quality improvements of the rule including human health and ecological impacts of metals, antibiotics, hormones, salts, and other pollutants; eutrophication of coastal and estuarine waters due to nitrogen deposition from runoff; nutrients and ammonia in the air; reduced exposure to pathogens due to recreational activities; and reduced pathogen contamination of drinking water supplies. These impacts were not monetized mainly because of a lack of models and data to quantify the impacts and, in some cases, the lack of methods to perform the monetization.

Changes must also be defined relative to a baseline, and few national-level databases useful for this purpose exist. For example, in the RIA for the aquaculture rule, it was difficult to quantify the changes in stressors because, in some cases, baseline data on stressor levels were not available.

Even if changes in stressor levels can be predicted at the national level, mapping these into national-level changes in ecosystem characteristics or services using ecological production functions is generally very difficult. There may be a long chain of ecological interactions between the stressors and the ecosystem services of interest – and often many of links in that chain are not fully understood by scientists, particularly at the level required for comprehensive national analysis. Scientific knowledge is especially lacking on the ecological impacts of substances such as heavy metals, hormones, antibiotics, and pesticides. However, these substances can have important and far-ranging impacts at the national level. In addition, the nature and magnitude of impacts can be very site-specific because they vary substantially both within and across regions of the country. As a result, predictions of biophysical impacts in one region generally cannot readily be transferred to other regions where the characteristics of the relevant ecosystems, as well as the affected population, are different.

Even if the national impact of the rule can be estimated, the Agency must then seek to monetize the value of that impact using economic valuation techniques if possible. Because EPA generally does not have the time or resources required to conduct significant original economic valuation research for specific national assessments of benefits and costs, the Agency typically must rely heavily on benefits transfer, i.e., using results from previous studies and adapting those results for the specific valuation context of interest. However, most of the previous ecological valuation studies that might serve as study sites for benefits transfer are not national in scope and generally have focused on only a limited number of ecosystem characteristics or services. Because they were designed for different purposes, previous studies have not selected either the study sites or the assessed services to facilitate national assessments of ecological benefits that might be important in a rule making context. Rather, they usually have involved specialized case studies selected because data were available or a specific change was readily observable. In addition, the studies generally measure tradeoffs for small, localized changes affecting a limited regional population.[35]

Perhaps the most relevant area for which considerable economic valuation has been conducted is recreation demand. Many economic valuation studies have estimated the recreation benefits stemming from hypothetical or predicted changes in environmental characteristics of recreation sites. For example, several studies have used random utility models (a revealed-preference approach) to link physical descriptors of water quality to recreation behavior and estimate the willingness-to-pay or willingness-to-accept per recreational trip for a given change in water quality (CITATIONS). However, these studies value only localized changes and cannot be directly used to provide national-level benefit estimates.

Previous studies have also estimated the benefits associated with changes in ecological services that affect the well-being of homeowners living near the ecological systems. Examples include water regulation, flood control, and the amenities associated with healthy populations of plants and animals. The willingness of residents to pay for these services is capitalized into housing prices and can be estimated using hedonic property value methods. Examples illustrating this approach to valuing ecosystem services include Leggett and Bockstael (2000), Mahan et al. (2000), Netusil (2005), and Poulos et al. (2002). Estimates from such studies could also be candidates for use in an economic benefits transfer. However, as with the recreation studies, these studies are almost exclusively local rather than national in scope, which makes extrapolation to national-level benefit assessments difficult.[36]

If sufficient high-quality original valuation studies are available, it might be possible to combine estimates of economic benefits from local studies in meta-analyses for use in benefits transfer (e.g. Smith and Pattanayak, 2002; Bergstrom and Taylor 2006; and chapter 4). However, using meta analysis to estimate benefits at a specific policy site can raise a number of issues. These include issues of consistency and those related to the scope of the resource changes valued in the original studies (e.g., whether they valued localized changes or changes at the national level). A meta-analysis of studies that valued localized changes can, at best, generate values for similar localized changes. It cannot generate values for changes that would occur at the national level unless individuals care only about localized effects. Therefore, even structurally based meta functions from local studies generally do not provide a functional relationship that can be used to estimate benefits at the national level, based on characteristics of the affected population. For example, using a meta function of unit values for a localized ecosystem change to predict average willingness-to-pay per person (e.g., evaluating the meta function using mean population characteristics) and then multiplying the resulting value by the relevant national population would generally not provide a valid measure of national-level benefits..

Despite the challenges described, the Agency has, in some cases, generated defensible estimates of economic benefits at the national level for a limited set of ecosystem services. For example, in the prospective benefit assessment of the Clean Air Act Amendments, EPA used the best available economic and ecological models to estimate commercial forestry and agricultural benefits. However, in other cases, the Agency's efforts to provide monetized ecological benefit estimates using benefits transfer have generated benefit estimates that are much less defensible or have led the Agency to focus on a very limited set of ecosystem services.

Ecological benefit assessment as part of the prospective study of the economic benefits of the Clean Air Act Amendments

The first prospective benefit-cost analysis mandated by the 1990 Clean Air Act Amendments included estimates of the ecological benefits resulting from the expected reductions in air pollutants (EPA, 1999). The Agency included qualitative discussions of several potential ecological effects of atmospheric pollutants based on a review of the peer-reviewed literature (chapter 7, and pp. E-2 to E-9), including acid deposition, nitrogen deposition, mercury and dioxins, and ozone.

The Agency used two criteria to narrow the scope of work for quantification of impacts: First, the endpoint must be an identifiable service flow. Second, a defensible link must exist between changes in air pollution emissions and the quality or quantity of the ecological service flow, and quantitative economic models must be available to monetize these damages.

The Agency provided estimates of three categories of economic benefits related to ecosystems based on standard economic models and methods: a) benefits to commercial agriculture associated with reductions in ozone; b) benefits to commercial forestry associated with reductions in ozone; and c) benefits to recreational anglers in the Adirondacks lake region due to reductions in acidic deposition.

For agriculture, the Agency used crop-yield loss functions to estimate changes in yields, which were then fed into a model of national markets for agricultural crops (AGSIM) to estimate changes in consumers' and producers' surplus.

For commercial forestry, the PnET-II model was used to estimate the effects of elevated ambient ozone on timber growth. The PnET-II model relates ozone-induced reductions in net photosynthesis to cumulative ozone uptake. Analysis of welfare effects used the USDA Forest Service timber assessment market model (CITATION) to translate the increased tree growth from a reduction in ozone to an increase in the supply of harvested timber and computed the changes in consumers plus producer surplus based on the associated price changes. Because of the lack of data and relevant ecological models, the Agency did not quantify or monetize aesthetic effects, energy flows, nutrient cycles, or species composition in either commercial or non-commercial forests.

To estimate the recreational economic benefits of reducing acid deposition in Adirondacks lakes, the Agency used a published study of recreational angling choices of households in New York, New Hampshire, Maine, and Vermont (Montgomery and Needelman, 1997). Measured pH of lakes was used as an indicator of the level of ecological services from each lake. The literature on the economics of recreational angling shows that likelihood of success as measured by numbers of fish caught is a major determinant of demand for recreational angling (Phaneuf and Smith, 2005; Freeman, 1995). To the extent that populations of target species are correlated with pH levels, pH is a satisfactory proxy for fish populations and angling success rates. There was no attempt to quantify other ecosystem services of water bodies likely to be affected by acid deposition.

The Agency also presented an estimate of the economic benefits of reducing nitrogen deposition in coastal estuaries along the east coast of the United States. Although the Agency was able to estimate changes in nitrogen deposition for the three estuaries covered in the prospective analysis, it was not able to establish the necessary ecological linkages to quantify the effects on recreational and commercial fishing. The assumed avoided costs were the costs of achieving equivalent reductions in nitrogen reaching these water bodies through control of water discharges of nitrogen from point sources in these watersheds. As noted in chapter 4 of this report, avoided cost is a valid measure of economic benefits only under certain conditions, including a showing that the alternative whose costs are the basis of the estimate would actually be undertaken in the absence of the environmental policy being evaluated. Because it was not possible to make this showing in the case of controlling nitrogen deposition, the Agency chose not to include the avoided cost benefits in its primary estimate of economic benefits, but only to show them as an illustrative calculation.

Chapter 2 of this report addresses the benefit assessment for the CAFO rule and highlights the committee's concern about EPA's approach. As discussed in that chapter, EPA estimated recreational benefits using a water quality survey conducted in the early 1980s (Carson and Mitchell, 1993). The principal advantage of this approach was that it utilized a national survey and presented a simple willingness-to-pay relationship for improvements in water quality that allowed national-level benefits to be estimated relatively quickly without new research. This study was also used in the assessment of EPA's aquaculture rule (described above). However, in addition to being more than 20 years old, the survey was not designed for those uses. The water quality index employed in the study was highly simplified and not intended to reflect ecological services related to water quality (other than those related directly to fish). Thus, the benefits transfers were considerably outside the domain of what was envisioned in the design of the original survey and what could have been known by the people who responded to it in the early 1980s.

The desire to use value estimates from the Carson and Mitchell study apparently also influenced the choice of ecological models used to predict water quality impacts. In both the CAFO and aquaculture assessments, EPA chose to use the QUAL2E water quality model (Barnwell and Brown, 2987) apparently because it could readily be linked to this valuation study. Although this model can estimate the interactions among nutrients, algal growth, and dissolved oxygen, it is not capable of ascertaining the impacts of total suspended solids, metals, or organics on the benthos and the resulting cascading effects on aquatic communities that might have important water quality impacts. (CHECK) Thus, even for the limited set of ecosystem services for which these assessments provided monetized benefits, the resulting benefit estimates were not very reliable.

A further concern with prior EPA estimates relates to the Agency's concentration on a limited set of ecosystem services, driven by the focus on monetization. If the benefit estimates derived from a limited set of services are sufficiently large to "justify" the costs (as required by Executive Order 12866) and the only objective of the analysis is to make this determination, omitting detailed consideration of other impacts can save scarce resources without affecting the conclusion. However in some cases, the benefits from a limited set of services might not justify the costs, but a more complete assessment of benefits very well might. In these cases, focusing on only a subset of services could lead to incorrect conclusions or inferences about relative benefits and costs. Perhaps more importantly, even if estimated benefits based on a limited set of services are sufficient to justify costs, a more complete assessment of benefits could provide useful information about whether a more stringent rule is warranted. In addition, representing the benefits from a limited set of ecosystem services as the total economic benefits associated with a rule can be misleading and confusing to policy makers and the public if they have a broader conception of the rule's possible benefits.

6.1.3. Implementing the proposed approach

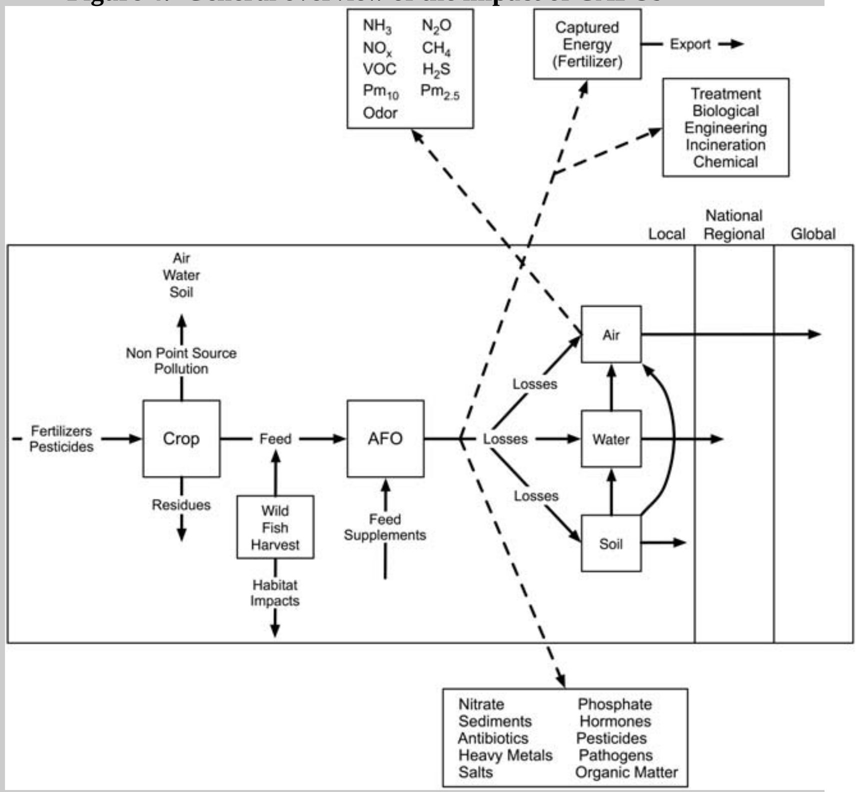
While recognizing the many challenges posed by ecological valuation in the context of national rule making, the committee believes that the valuation approach proposed in this report can be usefully applied in this context and can improve on the Agency's current approach to these challenges. Implementing the proposed approach would entail some short-term steps that could be incorporated into EPA's valuation processes using the existing knowledge base, as well as some longer-term strategies for research and data/method development that would improve ecological valuation for national rule makings in the future.

6.1.3.1 Implementation in the short run

A key premise of the committee's approach is that valuation should include early identification of the socially important impacts of an EPA rule. This requires information about both the potential biophysical effects of the Agency's actions and the ecological services that matter to people. As discussed in chapter 3, the Agency should develop a conceptual model early in the valuation process and then use that model to guide the valuation process. Conceptual models can allow the Agency to take a broad view of the complexities involved in ecological changes and ensure that impacts that are potentially important to people are included in the analysis. It should be standard practice for the Agency to develop such a conceptual model before other analytical work begins on an ecological benefit assessment.

Development of a conceptual model requires both an interdisciplinary team of experts and input from the public. To determine the relevant ecological effects to include in the conceptual model, EPA can draw on technical studies of impacts and their magnitudes. It can also solicit expert opinion regarding the physical and biological effects of a regulatory change. Figure 4 illustrates the type of conceptual model that EPA could have used for the CAFO assessment. It gives a general overview of the ecological impacts of CAFOs, enabling a comprehensive evaluation of those impacts. As shown, the environmental impacts of CAFOs extend beyond water quality impacts. For example, CAFOs generate interactive pollutants that affect air as well as water. A conceptual model helps provide a comprehensive overview of the potential ecological services that might be affected by an EPA rule.

Figure 4: General overview of the impact of CAFOs



The conceptual model should reflect not only ecological science but also information about the changes that are likely to be of greatest importance to people. This information cannot be derived deductively. Rather, it requires input from the public. Although Circular A-4 requires use of economic valuation methods when estimating benefits and costs, at this early stage, EPA can use a variety of methods to identify the public concerns associated with a given rule. For example, EPA can glean this information from the existing knowledge base or actively solicit it through an interactive process.

Possible approaches for obtaining information about public preferences and concerns from the existing knowledge base include:

- Inventorying the reasons invoked in similar rule-making processes in other jurisdictions (e.g., state and local)

- Inventorying the concerns expressed in public hearings at various governmental levels (perhaps with weightings based on the frequency of concerns raised), through, for example, content analysis of transcripts of public comments
- Studying previously conducted surveys providing information about related public concerns
- Analyzing relevant initiatives, referenda, or community decisions revealing preferences for various types of ecosystem services or the avoidance of various risks

An important consideration in identifying socially important impacts is the extent to which the public understands the role that ecosystems play in providing services that contribute to human well-being. When relying on information from public expressions of preferences (e.g., surveys, public hearings, community decisions) to identify socially important impacts, the Agency should assess whether the public, when expressing preferences, understood the ecosystem contributions sufficiently well to provide informed responses. Many ecosystem services – although well known to the scientific community – are little known or misunderstood by the general public (Weslawski et al., 2004). For example, the public generally does not understand or appreciate the full chain of connections described in Figure 4. Nor does the public typically understand the organisms and processes involved in breaking down waste products or the services provided by those processes. Lack of public understanding can be more problematic in national-level analyses, where ecological impacts and vulnerabilities can vary substantially across locations.

EPA can at least partially mitigate this information problem in national assessments by seeking public input through an interactive or participatory process. Such a process could take a number of forms, including focus groups, active solicitation of comments on a preliminary list of potentially important ecosystem services, or mediated modeling. A participatory process could also educate the participants about the underlying science and thus increase the likelihood that individuals expressing value judgments are well-informed. Although time and resource constraints may preclude use of a participatory process in many contexts, the committee suggests that EPA experiment with such processes (e.g., by holding open meetings with the public and Agency staff) to aid in identifying ecological changes that are important both biophysically and socially.

When properly conducted, the development of the conceptual model should identify a list of ecosystem effects or changes in ecosystem services that are potentially important to people, as well as the associated complexities, interactions, variability, and sources of uncertainty, including gaps in information. The Agency should ensure that the call for monetization, coupled with the need to generate national-level benefit estimates, does not unduly restrict the types of ecosystem impacts considered in the economic benefit assessment or lead to inappropriate application of economic valuation methods or benefits transfer.

Toward this end, a key step in implementing the committee's approach should be to categorize potentially important effects identified in the conceptual model into four categories:

- Category 1: Effects that can be monetized using available ecological models and appropriate benefits transfer or other method
- Category 2: Effects that cannot be monetized but can be quantified in biophysical terms using available ecological models and for which some indicator(s) of economic benefits can be derived using previous research and existing databases
- Category 3: Effects for which the baseline data or ecological models needed to quantify the biophysical effects are not available
- Category 4: Effects that are likely to generate important non-economic values (based, for example, on moral or spiritual convictions)

Some effects might fall into multiple categories. For example, the effect of a rule on a given fish population might have not only benefits for commercial fishing that can be monetized but also non-economic value for based on moral convictions, such as in the case of salmon protection in the Pacific Northwest..

The first three categories compose the benefits of a rule as defined by OMB; the fourth category can yield supplemental information about associated ecological values. To the extent possible, EPA should try to include benefits in the first category and should support the research needed to include more benefits in that category in the future. Nonetheless, explicit identification of benefits in the second two categories can help ensure that non-monetized benefits receive sufficient attention in benefit assessments.

The analysis of benefits or values under the committee's proposed approach differs across these different categories. With regard to the first category, estimation of monetized benefits requires three elements: a prediction of the change in relevant stressors resulting from the rule; a prediction of how that change will affect the ecosystem and ultimately the provision of ecosystem services; and an estimation of the benefits associated with the effect. To do this, the conceptual model must be linked with one or more ecological models that capture the essential linkages embodied in the conceptual model and are parameterized to reflect the range of relevant scales and regions. These ecological models must generate outputs that can be used as inputs in a benefits transfer or other valuation method. Because many existing ecological models do not satisfy this requirement, in the short run this requirement represents a significant constraint on the ecosystem effects that can be monetized and highlights the need for research to develop new ecological models.

Even when ecological models directly link to valuation, using these models to generate national-level estimates of the biophysical impacts of an EPA rule is still very challenging, given the variability of ecosystem impacts within and across regions. The SAB has noted and discussed this point in other benefit assessment contexts, including the impact of Superfund sites (U.S. Science Advisory Board 2006d). To address variability across sites, the Agency should explore the use of a bottom-up approach to valuation. Under this approach, a number of case studies that reflect different types of ecosystems could be conducted. If information is available in a given rule making about the distribution of the ecosystem types and populations affected, EPA could aggregate these case studies to provide national-level estimates of changes in ecosystem services resulting from the rule. Even without full information about the distribution of ecosystem types and populations, the case studies could still provide information about the range of impacts and their dependence on ecosystem characteristics. This information could be useful not only for the specific policy decision at hand, but also in guiding future research. For example, it could suggest key ecosystem characteristics that would be useful in categorizing ecosystems for future valuation analyses and for which additional distribution information is needed.

Once changes in ecosystem services are estimated, those changes must still be valued to generate national benefit estimates. The appropriate valuation approach will depend on the nature of the ecosystem services. For services that generate only local benefits, benefits transfer based on comparable previous studies of localized impacts can be used, provided the benefits transfer is conducted appropriately (see discussion in chapter 4). The local or regional benefit estimates can then be appropriately aggregated to the national level. However, for ecosystem services for which local impacts generate broader national benefits, use of localized studies for benefits transfer can be problematic, as noted above. For these services, benefits transfer should instead be based on studies that have generated value information at the national level, such as national surveys of willingness to pay for national-level changes in ecosystem services. However, the only existing survey of this type is the water quality survey of Carson and Mitchell (1993), which should not be used for ecological valuation, for the reasons discussed earlier. Additional research to generate benefit estimates that could appropriately be used for national benefits is needed.

Although in principle economic valuation methods can fully capture the benefits associated with changes in ecological systems and services, in practice there are significant limitations that can make this very difficult, particularly at the national level. For benefits that cannot be monetized using available ecological models and reliable information about economic values (category 2 above) ecological impacts may still be quantifiable. Here again, EPA should focus on quantifying ecological changes that are potentially important to people, choosing from among a broader set of ecological models than those used for category 1, because the category 2 ecological models need not directly link to existing information about economic values. As with monetized values, EPA can address the site-specificity of ecological impacts by using a bottom-up approach, and – if the relevant information about the distribution of ecosystem types or characteristics exists – aggregate the resulting estimates to the national level.

When monetization is not possible, the Agency should also seek to identify scientifically-based indicators of those benefits, to the extent possible. Some of the valuation methods discussed in chapter 4 might be useful for this purpose. To the extent these methods generate non-monetary measures that economic theory suggests are likely to be correlated with benefits, they can provide useful information about benefits when direct monetary estimates of those benefits are not available. For example, economic theory suggests that total economic benefits associated with an increase in wetlands in a specific area will depend, among other things, on the number of people who visit the area for recreational purposes. Other things being equal, the more people who visit the area, the higher the benefit associated with an increase in wetlands acreage. Likewise, the more people who live in the vicinity of an affected ecosystem, the greater the benefit associated with protecting that ecosystem. Similarly, if more people judge the protection of a given ecosystem service to be "somewhat important" or "very important" in a survey of attitudes and judgments, then it follows that willingness-to-pay to protect that service is likely to be higher. Although these indicators do not provide monetary estimates of benefits that can be compared to cost, they can provide important information or signals about possible benefits.

Care must be taken to avoid misinterpreting these indicators. For example, just because a large population lives in the vicinity of an affected ecosystem does not necessarily mean that a change in that ecosystem has a large value. If the change relates to a service that is not important to people, the value of that change (i.e., the willingness to pay for it) would be low regardless of the number of people living in the vicinity. To draw correct inferences, the Agency needs information not only about the number of people affected but also about the importance that individuals attach to the service, as revealed through surveys or other methods.

For potentially important benefits for which quantification of the associated ecological changes is not possible (category 3 above), the Agency should characterize the changes as carefully as possible. It should discuss in detail why the changes are potentially important but not quantifiable, citing relevant literature. A carefully developed and scientifically-based conceptual model can serve as the basis for a qualitative but detailed description of the ecological impacts of a given change. A simple summary of possible impacts is not sufficient. EPA should also provide justification based on the conceptual model and associated theoretical and empirical scientific literature. To the extent possible, the Agency should use the existing literature to draw inferences about the likely magnitude or importance of different effects, even if only qualitatively (e.g., high, medium, low).

Although benefit-cost analysis requires the use of economic valuation to estimate benefits, regulatory impact assessments need not be limited to information generated for use in comparing benefits and costs. Information about other sources of value that do not fit within the theoretical framework underlying benefit-cost analysis (category 4 above) can still be of interest to policy makers when making decisions on ecological protection. For example, the religious, spiritual, or cultural value of some ecosystems and services may be an important consideration not adequately captured by standard measures of willingness to pay. Non-economic valuation methods can provide information about these other sources of value.

In addition to its implication for how ecological valuations are conducted, the committee's valuation approach also has implications for reporting value estimates in national benefit assessments. To increase transparency EPA should document in economic benefit assessments and RIAs the conceptual model used to guide the analysis and how decisions underlying the model were made. The assessments should describe how the ecosystem services were identified and the rationale for key choices regarding the focus of the assessment.

Consistent with the guidance in Circular A-4, benefit assessments should also clearly identify the four categories of values outlined above. If methods other than economic valuation are used to provide non-monetary quantitative or qualitative information about benefits, the RIA should include a discussion of the extent to which the methods provide indicators of willingness to pay or to accept. If non-economic methods are used to capture sources of value other than those typically reflected in willingness to pay, the RIA should describe the methods used and the results as supplemental information in a separate section.

When monetized economic benefits are aggregated, the resulting sum should always be described as "total economic monetized benefits" rather than "total benefits." In the past, EPA has sometimes reflected non-monetized benefits in aggregate measures of benefits by including an entry such as +X or +B in the summary table of benefits and costs to indicate the unknown monetary value that should be added to the benefits if the value could be determined. Although this approach indicates that the measure of monetary benefits is incomplete, the +X or +B designation provides insufficient information and can be easily overlooked in using the results of the benefit assessment. Designating the sum as "total monetized economic benefits" provides a continual reminder of what is, and is not, included in this measure. By also including key quantified but non-monetized impacts that are measured in biophysical units, along with indicators of economic benefits and a detailed description of the non-quantifiable impacts, the Agency can provide a more accurate and complete indication of total benefits as called for by Circular A-4.

Because of the difficulties of estimating the biophysical impacts of an EPA rule and the associated benefits or costs, the Agency must also characterize the uncertainty associated with its assessment. EPA should include a separate chapter on uncertainty characterization in each benefit assessment and RIA. This chapter should discuss the scope of the benefit assessment, the different sources of uncertainty (e.g., biophysical changes and their impacts; social information relevant to values; valuation methods, including transfer of willingness-to-pay or willingness-to-accept information), and the methods used to evaluate uncertainty. At a minimum, the chapter should report ranges of values and statistical information about the nature of uncertainty for which data exist. For each type of uncertainty, EPA should report information similar to that reported in the Agency's prospective analysis of the benefits and costs of the Clean Air Act Amendments (EPA, 1999) and should provide a summary of this information in the executive summary of the RIA or benefit assessment. Specifically, EPA should report potential sources of errors, the direction of potential bias for the overall monetary benefits estimate and the likely significance relative to key uncertainties in the overall monetary estimate.

<u>6.1.3.2</u> <u>Research needs for improvements in future valuation</u>

EPA can takes the steps suggested above in the short run to improve ecological valuation, but additional improvements will require longer-term investments in research at least three areas: national-level databases to support prediction of ecological impacts; means of mapping changes in stressors to changes in ecosystem services; and benefits transfer.

Research is needed to develop national-level databases to predict ecological impacts (including baseline data on ecological conditions) and to value those impacts (including data on affected populations). The current availability of national-level databases with this information is limited. In addition, research is needed on the distribution of relevant ecosystem and population characteristics across local or regional sites that can be used to aggregate case studies in a bottom-up approach to national-level benefit assessment. As noted above, case studies provide a means of incorporating heterogeneity regarding both ecological impacts and values into benefit assessments. However, to generate national-level estimates for use in national rule makings, results from case studies must be aggregated using weights that reflect the distributions of the relevant combinations of biophysical and population characteristics. Research to identify both the key relevant characteristics and their joint distributions is needed.

As discussed in chapter 5, research is needed to develop ecological production functions and associated models that can map changes in stressors to changes in ecosystem services. In the past, EPA has often been unable to estimate certain benefits because the Agency was not able to predict how a given rule would change stressors and how those changes would in turn affect ecosystem services. Both baseline data and the development of ecological models that focus on ecosystem services, as well as other ecosystem characteristics of importance to people, are needed. The datasets and models should support aggregate, national-level benefit assessments.

Finally, additional research related to benefits transfer is needed, including both research on methodological issues that arise in using benefits transfer and additional original valuation studies that the Agency can use for benefits transfer. These new studies should focus on benefit estimates that can be applied in multiple contexts (e.g., recurring rule makings) and across a broader geographical scale. Loomis and Rosenberger (2006) suggest features of study design that facilitate the use of a study's results in benefits transfer. These include use of objective quantitative measures of quality, measured in policy-relevant physical units; the evaluation of realistically small changes; the provision of information about relevant baselines; and the full and consistent reporting of results. New studies should also expand the range of ecosystem services valued using economic valuation methods so that benefits transfer can be applied to a wider range of services and/or ecological impacts.

In addition to localized studies that could be used as study sites, national-level studies are also needed for ecological valuation in national rule making. National economic valuation surveys (such as that one conducted by Carson and Mitchell [1993]) that have recent data and a specific focus on ecosystem services, have the potential to contribute significantly to the Agency's ability to conduct ecological benefit assessments to support national rule making, provided they are conducted in accordance with state-of-the-art survey procedures (see appendix A). Because conducting surveys for individual rule makings is prohibitively costly in both time and resources, the Agency should focus on conducting a limited number of surveys designed to provide value information usable in multiple rule makings.

Toward this end, the Agency should develop an extramural grant program focused on developing methods and value estimates specifically for use in recurring rule makings (e.g., for rule making associated with National Ambient Air Quality Standards or Effluent Guidelines). In past years, EPA has targeted some of its Science To Achieve Results (STAR) grant resources toward benefits transfer, but a larger and more concerted effort focused on its use in national rule making is clearly needed.

6.1.4. Summary of recommendations

To develop more comprehensive estimates of the ecological benefits associated with national rules and regulations, the Agency needs a broader approach to ecological valuation than it has typically used in the past. The expanded approach to valuation proposed in this report can and should be applied to national rule making. This would entail challenges, but important opportunities for improvement as well. EPA can implement some of the committee's recommendations using the existing knowledge base. Other recommendations call for research to enhance the Agency's future capacity to conduct high-quality, scientifically-based ecological valuation for national rule making.

The Agency can improve ecological valuation for national rule making in the short run by incorporating the following recommendations:

- The Agency should begin each valuation exercise with the development of a conceptual model of the ecological system being analyzed and the ecosystem services that it generates. This model should serve as a guide or road map for the benefit assessment.
- EPA should develop the model using input from both experts and the public, to ensure that it incorporates important ecological functions and processes as well as related ecosystem characteristics and services that are potentially important to people. Public concerns can be identified through a variety of methods, drawing on either existing knowledge or an interactive process to elicit public input. The Agency should experiment with an open, interactive public forum for identifying issues of concern.
- Once the Agency has identified a list of potentially important ecological effects and associated services, it should categorize those effects according to the extent to which they can be quantified and monetized at the national level using economic valuation techniques (primarily benefits transfer).
- To address site-specific variability in the impact of a rule, the Agency's benefit assessments should include case studies for important ecosystem types, and aggregate across these case studies if information about the distribution of ecosystem types is available. This bottom-up approach would establish separate estimates for each locality or region and then add them together to obtain a national estimate.
- For ecosystem services for which the benefits are primarily local, EPA can use benefits transfer using prior valuations at the local level, provided the benefits transfer is conducted appropriately. However, for services with broader benefits, the Agency should use benefit transfers that draw from studies with broad geographical coverage (in terms of both the changes that are valued and the population whose values are assessed).
- EPA should not compromise the quality of a benefits assessment by inappropriately applying benefits transfer to effects that cannot be monetized at the national level using scientifically sound benefits transfer, nor simply list such effects in a category of "non-monetized benefits." The Agency instead should seek to provide a scientific basis for the belief that these benefits are

important. EPA could include quantifications of biophysical impacts using ecological models, metrics that provide information about the likely magnitude of the associated benefits (and hence are indicators of benefits), and detailed qualitative discussions based on existing scientific literature.

- EPA should also consider estimating non-economic values for at least some ecosystem services, where appropriate. Even though these values do not properly fit within a formal economic benefit-cost analysis, they can provide important additional information to support decision making. When such value estimates are included in RIAs, the RIA should discuss both the valuation method and the results in a separate section.
- To ensure that benefit assessments do not inappropriately focus only on those impacts that have been monetized, EPA should report non-monetized ecological effects in appropriate units in conjunction with monetized economic benefits. Aggregate monetized economic benefits should be labeled as "total monetized economic benefits" rather than "total benefits."
- EPA should include a separate chapter on uncertainty characterization in each economic benefit assessment and RIA.

To enhance the Agency's capacity to conduct future ecological valuations, EPA should support research specifically designed to facilitate ecological valuation for national rule making, particularly for recurring rule makings. The committee recommends that EPA focus on at least three areas of research:

- EPA should support the development of national level databases to support valuation, including data on the joint distribution of ecosystem and population characteristics that are important determinants of ecological benefits.
- EPA should support the development of quantitative ecosystem models and baseline data on ecological stressors and ecosystem service flows that can support national-level predictions of the consequences of changes in ecological stressors on the production of ecosystem services.
- EPA should support the development of additional methodological and original valuation studies designed to enhance national-level ecological benefits transfer, including national surveys relating to ecosystem services

with broad (rather than localized) benefits that can generate value estimates for use in multiple rule making contexts.

6.2. Valuation in regional partnerships

6.2.1. EPA's role in regional-scale value assessment

Significant opportunities exist to use regional-scale valuations of ecosystem services to guide decision making by EPA and local governments to protect and restore the environment. Many important ecological processes take place at a landscape scale. For example, habitat connectivity on landscapes, water and nutrient flows through watersheds, and patterns of exposure and deposition from air pollution in an airshed pose issues larger than a particular site and thus require regional-scale analysis.

An increase in data and methods, supported by EPA research, has also opened new frontiers for regional-scale analysis of ecosystems and their services. Publicly available, spatially explicit data on environmental, economic, and social variables have increased dramatically in recent years. At the same time, the ability to display data visually in maps and to analyze spatially explicit data using a variety of analytical models and statistical methods has expanded. An active EPA extramural program in ecological research is underway for regional-scale analysis of ecosystems and services. As part of that program, EPA has funded research relating to restoration of water infiltration in urbanizing watersheds in Madison, Wisconsin, restoration of multiple ecosystem functions for the Willamette River in Oregon, decision-support tools to meet human and ecological needs in New England rivers, and the provision of multiple services from agricultural landscapes in the upper Midwest. As discussed in section 6.1.3.2, EPA Region 4: Southeast has developed a tool for regional ecological assessment. Other regions have also undertaken assessments of ecosystem services.

There is great potential – largely untapped to date – to use this type of analysis to aid regional decision making. Municipal, county, regional, and state governments make many important decisions affecting ecosystems and the provision of ecosystem services. Examples include land-use planning and watershed management.

Unfortunately, local and state governments rarely have the technical capacity or the necessary resources to undertake regional-scale analyses of the value of ecosystems or their services or to incorporate these values into their decision making processes.

Regional partnerships among EPA, other governmental agencies, and the private sector offer the potential for expanding national, state, and local capacity to value and protect ecosystems and their services. EPA regional offices have many opportunities to collaborate at a regional scale with local and state governments, regional offices of other federal agencies, environmental non-governmental organizations, and private industry. Through collaborating with such groups, EPA can enhance environmental protection by engaging important local stakeholders, gaining access to regional expertise, and promoting effective decision making on important regional-scale environmental decisions. Local and state partners can gain from access to EPA technical expertise and resources. Such partnerships can expand the knowledge base for decision making and improve the analysis of the value of ecosystems and services.

Unlike national rule making, where specific statutes or regulatory mandates often constrain analysis, regions have freedom to experiment with novel approaches to valuing ecosystems and their services. Such experimentation may lead to improved methods and practices of valuation with potential positive impacts well beyond the region that pioneers the innovations. For example, EPA can use regional-level partnerships as a mechanism for testing and improving various valuation methods that might ultimately be used at the national level.

Because of the absence of legal or statutory requirements that EPA value ecosystems or services at the regional scale, there have been few regional ecological valuation efforts to date. In addition, regional offices may have lacked the time, resources, and expertise to undertake some of the crucial steps recommended in this report for valuing ecosystems and their services. For example, few regional offices have economists on staff who can work on valuations. Partly for these reasons, many of the potential advantages of regional partnerships for valuing ecosystems or their services have not been realized to date.

To analyze opportunities for regional partnerships for valuation, the committee, through the SAB Staff Office, surveyed regional offices for examples of where the Agency or other governmental agencies have engaged in regional valuation efforts (EPA Science Advisory Board Staff, 2004). This section explores three case studies from Chicago; Portland, Oregon; and the Southeast. The case studies illustrate several general lessons about regional-scale analysis of the value of ecosystems and services and the potential usefulness of regional partnerships.

6.2.2. Case study: Chicago Wilderness

Chicago Wilderness is an alliance of more than 180 public and private organizations. The overall goal of Chicago Wilderness, as stated in its *Biodiversity Recovery Plan,* is "to protect the natural communities of the Chicago region and to restore them to long-term viability, in order to enrich the quality of life of its citizens and to contribute to the preservation of global biodiversity" (Chicago Wilderness, 1999, p. 7). Chicago Wilderness is a bottom-up organization. No single decision maker or agency controls or guides Chicago Wilderness. It pursues objectives, as defined by its members, through consensus. Chicago Wilderness pursues its goals and objectives by promoting a green infrastructure to support biodiversity and to maintain ecosystems and services linked to quality of life in the Chicago metropolitan area.

As a member of Chicago Wilderness, EPA Region 5 (serving Illinois, Indiana, Michigan, Minnesota, Ohio, Wisconsin, and 35 Tribes) has provided technical and financial assistance and facilitates the partnership. EPA expertise in Region 5, particularly in natural sciences, has contributed to quantifying ecosystem services and understanding how potential stressors affect ecosystems and the provision of services. Chicago Wilderness has produced several reports, including its *Biodiversity Recovery Plan* and a green infrastructure map for the region.[37] The Chicago Wilderness Web site (http://www.chicagowilderness.org/) contains a chronology and links to many relevant documents, including the *Biodiversity Recovery Plan.*

Chicago Wilderness has been interested in valuing ecosystems and services, but has only begun to explore the opportunities. Although no specific legal authority mandates valuation of ecosystems or services as part of the work of Chicago Wilderness, quantifying values associated with the conservation of green space and biodiversity could help Chicago Wilderness meet its own stated objectives and communicate its analysis to other groups and the general public. The possible uses of valuation identified by Chicago Wilderness members include:

- Informing decisions on the establishment of green infrastructure, including priorities for acquisition of land by, for example, forest preserve districts or soil conservation districts
- Assessing the value of preserving ground water and ecosystem services related to clean water
- Assessing the relative value of conventional versus alternative development and demonstrating conditions in which development decisions that have positive impacts on the environment might be in the financial interest of the developer
- Communicating effectively with residents of the Chicago region regarding the value of green infrastructure and biodiversity and how these relate to residents' quality of life
- Assessing the relative value of investing in different research projects to establish priorities for funding decisions

Members of Chicago Wilderness, however, possess only limited technical expertise and practical experience in valuing the protection of ecological systems and services. EPA Region 5 also has limited capacity to economically value ecosystem services.

In sum, Chicago Wilderness, like many regional partnerships, would gain much from the ability to analyze the value of ecosystems and services, but it is constrained by lack of expertise and resources.

6.2.2.1 An example of how valuation could support regional decision making

Valuation of ecosystems and services is most useful when done in the context of specific decisions affecting the environment. The committee therefore chose a specific decision context – county open space referenda in the Chicago metropolitan area – to explore how this report's approach to valuation could support regional decisions.

Voters in four counties in northeastern Illinois have passed referenda authorizing bonds to purchase land for open space preservation or watershed protection. In November 1997, voters in DuPage County passed a $70 million open space bond. In November 1999, voters in Kane County and Will County passed bond issues totaling $70 million for open space acquisition or improvement. In 2001, the voters in McHenry County passed a $68.5 million bond for watershed protection. Although these multi-million dollar bond proposals have provided substantial funding to preserve open space and ecological processes in the region, the funds are insufficient to protect all worthwhile open space and watersheds. Given this shortfall, input about the most important lands to purchase or management actions to undertake to maintain or restore natural communities would help ensure that counties invest these funds wisely.

This section of the report looks at how valuation could help inform conservation investments under the local county bonds. The section examines three types of values derived from protecting natural systems:

- Conservation of species and ecological systems
- Water quality and quantity
- Recreation and amenities

The discussion of water quality and quantity focuses on McHenry County because the bond issue there related directly to watershed protection. Following the process outlined in chapter 2 of this report, the section explores: the role of stakeholder involvement and input in determining ecosystem services of interest; predicting ecological impacts in terms of effects on these ecosystem services; and assessing and characterizing the values of these effects on the ecosystem services.

6.2.2.2 Stakeholder involvement, scientific and technical input, and public participation

The planning documents and activities of Chicago Wilderness illustrate several of the themes from chapter 2 of this report, including broad public involvement and interdisciplinary collaboration. Chicago Wilderness has made extensive efforts to engage the local community in determining the most important features of regional ecosystems and services. Two of the great strengths of the organization are the broad range of groups involved and its commitment to open processes. Chicago Wilderness participants themselves define the objectives, goals, and priorities of the organization. As a result of the open, democratic process and the extensive efforts to include multiple views and voices, the group's goals and objectives largely reflect what people in the region view as important to conserve. Engaging local communities is a vital first step in the process of valuing ecosystems and services. Engagement helps to focus scarce agency resources on issues of prime local importance, as well as to promote partnership and dialogue.

The inclusive planning process followed by Chicago Wilderness has included developing a common statement of purpose, setting up three working groups (steering, technical, and advisory committees), and working through nine planning steps (from visioning, development of inventories, assessment of alternative actions, to adopting a plan). In its early stages, Chicago Wilderness conducted workshops and meetings to define implementation strategies and to prioritize among its long- and short-term goals, which focus on the restoration and conservation of biodiversity. For priority setting, several of the workshops used non-monetary valuation exercises with qualitative rankings of importance. Chicago Wilderness also referenced other valuation measures, such as polls and The Nature Conservancy's global rarity index.[38]

Chicago Wilderness conducted eight workshops to assess status and conservation needs for natural communities in the area: four workshops on species, addressing birds, mammals, reptiles, amphibians, and invertebrates; and four consensus-building workshops on natural communities, addressing forests, savanna, prairie, and wetlands. The natural-communities workshops developed overall relative rankings based on the amount of area remaining, the amount protected, and the quality of remaining areas (incorporating fragmentation and current management). The workshops assessed relative biological importance for community types, based on "species richness, numbers of endangered and threatened species, levels of species conservation, and presence of important ecological functions (such as the role of wetlands in improving water quality in adjacent open waters)" (*Biodiversity Recovery Plan*, chapter 4, p. 41). The workshops identified visions of what the areas should look like in 50 years.

Two different groups of scientists and land managers developed a classification scheme for aquatic communities based on physical characteristics. The groups assigned recovery goals (i.e., protection, restoration, rehabilitation, and enhancement) to streams and priority levels (i.e., exceptional, important, restorable, and other, based on Garrison, 1994-95) to lakes. The groups assessed streams using the index of biotic integrity, species or features of concern, the macroinvertebrate biotic index, and abiotic indicators. The groups also assessed threats and stressors to streams, lakes, and near-shore waters of Lake Michigan.

However, Chicago Wilderness' strengths in engaging local communities highlight some of the difficulties in doing so. Different individuals and member groups define value differently. A priority for some groups is restoring pre-settlement ecosystem conditions. Open space and recreation are the primary motivation for others. Yet others focus on maintaining water quality or conserving the region's biodiversity. Because Chicago Wilderness is an organization based on consensus, the group often cannot make choices involving tradeoffs among worthwhile objectives. Protecting biodiversity, protecting water quality, and providing open space and recreational opportunities are all seen as good things. The choices become more difficult when getting more of one goal implies getting less of another. The inability to make tradeoffs limits the ability of Chicago Wilderness to make policy recommendations or have an influence on decision making. Valuation could help highlight which goals are of greater importance and help decision makers navigate among difficult choices.

Another disadvantage of Chicago Wilderness' broad engagement of local communities is the time-consuming nature of community involvement processes. The organization is not well placed to make rapid analyses or provide feedback on decisions that occur over a short time period.

6.2.2.3 Predicting ecological impacts in terms of changes in ecosystem services

Because Chicago Wilderness is committed to the value of protecting biodiversity, it is interested in predicting impacts on the conservation of species and ecological systems at the landscape scale. It has collected spatially explicit information relevant to land use, open space, recreation, biodiversity conservation, water quality, and water quantity. It has also successfully applied a variant of the conservation value method to identify and prioritize conservation actions through spatial representation and analysis of unique and threatened species and ecosystems. Use of the method demonstrates how conservation science can be used for planning, and how a transparent approach to mapping conservation goals can be useful in a regional partnership.

However, for this spatially-explicit information to be relevant to decisions affecting ecosystems, Chicago Wilderness needs cause-and-effect relationships that can predict how policy choices will affect ecosystems and services. It does not have the information to estimate ecological production functions. Although it can be effective in providing descriptive information – particularly in the form of maps – it is limited in its ability to analyze alternative policies and make recommendations about which alternatives are preferable. For example, Chicago Wilderness would be able to provide only limited guidance to a decision maker in McHenry County concerning how to invest the $68.5 approved by voters for watershed protection in a way that would maximize the value of ecosystems and services, because it would not be able to martial information about how particular actions affect systems and services.

Watersheds figure prominently in Chicago Wilderness' work. The protection or restoration of watersheds can have a number of impacts on ecosystem services, including water quality, water quantity, and the support of ecological communities.

Possible ecological impacts and provision of services from the protection or restoration of watersheds

<u>Surface water</u>

- Availability – More water will be retained in the watershed because there is less runoff from impervious surfaces.
- Periodicity of flows – Changes in the hydrograph are mitigated because precipitation will be captured in the soil and vegetation, and subsequently released more slowly.
- Maintenance of minimum flows – There is a greater chance of maintaining adequate minimum flows because of the dampening effects of intact watersheds and continuation of subsurface flows.
- Flooding – Flooding is reduced because of the retention capabilities of the intact watershed.

<u>Subsurface water</u>

- Availability for domestic and industrial use – Availability will be increased because percolation and subsurface recharge will be enhanced by natural soil surface and vegetation.

- Maintenance of wetlands – Those habitats that depend on the water table or subsurface flow will be enhanced because natural percolation and recharge processes will be maintained.

Water quality

- Pollution dilution – Increased flows will dilute concentrations of organic and inorganic pollutants.
- Assimilation of biotic pollutants – Increased stream flows will permit greater opportunity for the assimilation of biological materials.
- Biological communities – Habitats that depend on increased quantities of water in the watershed and containing protected species will enjoy increased persistence.
- Specific habitats – Increased water quantity and more uniform stream flows will support regionally important ecological communities, e.g., in-stream communities, bottomland forests, wetlands, and wet prairies.

To illustrate how Chicago Wilderness might characterize impacts on these ecosystem services in McHenry County, suppose that Chicago Wilderness, using the deliberative processese discussed in section 6.1.3 decides that the most important ecosystem services for comparing the value of watersheds within the county are: minimizing flooding; maintaining or increasing groundwater recharge; and maintaining or increasing wetland communities. To predict impacts related to flooding, Chicago Wilderness could make use of a geographic information system (GIS) database it developed that includes layers depicting rivers, streams, wetlands, forest lands, and floodplains. As a first approximation, Chicago Wilderness could use historical records of flooding in McHenry County watersheds to identify watersheds with the greatest potential for flooding. it could then evaluate the potential for restoring floodplain forests and wetlands for mitigating flooding. To estimate whether a development option would adequately maintain or increase groundwater resources, it could use the maps of aquifers and soils in the GIS database that describe run-off and percolation rates for each soil type. Watersheds could be compared in terms of potential for aquifer recharge. Chicago Wilderness could then consider the effects of alternative land use decisions on recharge (Arnold and Friedel, 2000). To address whether wetland communities would be maintained or increased, topographic maps and GIS data on rivers, streams, floodplains, forests, wetlands, and land cover could be used to rank watersheds within McHenry County in terms of potential wetlands minus current wetlands. The potential for expanding existing wetlands or restoring wetlands within watersheds could then be measured.

A number of GIS data files for McHenry County thus could assist in understanding how the protection of a given part of a watershed contributes to ecosystem processes and services. What is often lacking, however, is a cause and effect relationship that could be used to predict how alterations in management or policy would change the provision of ecosystem services. It might be possible to transfer results from studies of ecological services from other regions. For example, Guo et al. (2000) measured the water flow regulation provided by various forest habitats in a Chinese watershed. If these relationships are transferable, estimates of the effect of a policy of restoring forest habitat on water flow could be generated. Changes in water flow could then be used to predict impacts on aquatic organisms and their production functions such as waterfowl, fisheries, and wildlife viewing (Kremen, 2005).

In trying to predict how policy choices will affect ecosystems and the provision of services, experts must be careful not to substitute their own values for those of the stakeholders and community. Different judgments used in models may give rise to different recommendations. Making sure that the results of the analysis reflect the values of the community rather than the values of the experts requires honest communication as well as commitment on the part of experts to carry out the stated desires of the community.

6.2.2.4 Valuation of changes in ecosystems and services

Government decisions about what lands to conserve can involve tradeoffs among different ecosystem services of importance to the public. A study conducted in the Chicago metropolitan area, for example, found a tradeoff between desires to locate open space access close to people's homes and desires to locate open space to conserve species (Ruliffson et al., 2003). When there are such tradeoffs among different services, decision makers need information about the value of various aspects of ecosystems and services in order to determine what alternatives are more beneficial for the community. This information about relative values goes beyond understanding the ecological impacts of the management and policy alternatives.

This section begins with a discussion of the potential contributions that valuation could make to Chicago Wilderness and briefly examines possible valuation methods that could be applied for different types of ecosystem services. This discussion goes well beyond what Chicago Wilderness has actually done in the valuation realm. The organization has conducted very few quantitative valuation studies and largely lacks the resources and the expertise to do so.

In one sense, however, Chicago Wilderness carried out an important valuation exercise at its very outset when it engaged its member organizations and gathered feedback on what the community felt was important. This process resulted in an important statement about the values held by the collection of organizations that constitute Chicago Wilderness. As noted earlier, its overall goal is "to protect the natural communities of the Chicago region and to restore them to long-term viability, in order to enrich the quality of life of its citizens and to contribute to the preservation of global biodiversity."

Given this clear goal statement, formal valuation studies that try to quantify the monetary value of alternatives may be of secondary importance. Of primary importance is to understand how various potential strategies contribute to the protection and restoration of natural communities and the ecosystem services they provide. As noted earlier, Chicago Wilderness has used a variant of the conservation value method to identify and prioritize conservation actions that would contribute to this goal through spatial representation and analysis of biodiversity and conservation values. Not surprisingly, Chicago Wilderness has devoted most of its attention to biophysical measures of the status of natural communities. It has devoted much less attention to quantitative measures of value, monetary or otherwise.

With a clearly stated overall goal "to protect the natural communities of the Chicago region and to restore them to long-term viability," economic analysis may be largely restricted to estimating the cost of various potential strategies to achieve that objective. Cost-effectiveness analysis addresses how best to pursue a specific objective, given a budget constraint. Information about how potential strategies contribute to the protection and restoration of natural communities and about the cost of these strategies is the main information needed. There is no need to estimate the value of protecting natural communities or other ecosystem services.

Of course, things are rarely so clear. Even with a single overall goal, there are often multiple dimensions and tradeoffs among those dimensions that require an analyst to go beyond cost-effectiveness analysis. For example, in protecting natural communities, there may be tradeoffs between protecting one type of natural community versus another. When there are multiple natural communities or ecosystem services of interest, it becomes important to address questions of value – a practical matter when investment of bond monies is at stake. Is it more valuable to allocate resources to restoring upland forest or wetlands? Is it more valuable to mitigate flood risk or improve water quality? Such questions can be addressed only by comparing the relative value attached to different natural communities or services.

Economic valuation of the protection of natural communities may be important for Chicago Wilderness and the public at large for several reasons. First, when there are multiple sources of value generated by protecting natural communities (e.g., species conservation, water quality, flood control, recreational opportunities, aesthetics, etc.), monetary valuation provides a way to establish the relative importance of various sources of value. With prices or values attached to different ecosystem services, one can compare alternatives based on the overall economic value generated. Second, some biological concepts such as biodiversity are multi-faceted. How one makes tradeoffs among different facets of biodiversity conservation or among different natural community types is ultimately the same question as how one makes tradeoffs among multiple objectives. Establishing prices on different components of biodiversity or on different natural communities allows for analysis of tradeoffs among components and an assessment of the overall value of alternatives. Finally, monetary valuation may facilitate communication about the importance of protecting and restoring natural communities in terms readily understood by the public.

Non-monetary valuation can also be useful. If decisions involve tradeoffs among different natural communities or services, surveys containing attitude questions may be helpful. In some cases, people may find it easier to say whether they think it more important to provide additional protection of forests versus wetlands than to state the monetary value of protecting forests rather than wetlands.

People may value natural communities because of the ecosystem services they provide or because of their existence or intrinsic values. Of these two sources of value, the ecosystem services are generally the easier to value. Consider how Chicago Wilderness might value protecting wetlands and other watershed lands for flood control and water quality. To measure the value of flood control, it might measure avoided damages. Several studies of the value of preserving wetlands for flood control have been undertaken in Illinois, including studies of the Salt Creek Greenway (Illinois Department of Conservation, 1993; USACE, 1978) and the value of regional floodwater storage from forest preserves in Cook County (Forest Preserve District of Cook County, Illinois, 1988). The Cook County study found estimated flood control benefits of $52,340 per acre from forest preserves. The value of providing clean drinking water to the public is extremely high, far exceeding the costs of supplying it either by natural or human-engineered means. Because the question is how, not whether, to supply clean drinking water, replacement cost (e.g., the cost of building a filtration system to replace lost water purification services provided by wetlands) can be used to value the contribution of ecosystems to the provision of clean drinking water.

A large literature in environmental economics exists on estimating the values of various recreational opportunities and environmental amenities created by the natural environment. As discussed below, typical methods used to estimate the monetary value of recreation and environmental amenities include hedonic property price analysis, travel cost, and stated preference. A smaller literature uses referenda voting to infer values for open space and other environmental amenities.

Hedonic property price analysis is a common method for estimating the value of environmental amenities, especially in urban areas because of the availability of large data sets on the value of residential property values. Analysts have used the hedonic property price model to estimate the value of air quality improvements (e.g., Ridker and Smith, 1967; Smith and Huang, 1995), living close to urban parks (e.g., Kitchen and Hendon, 1967; Weicher and Zeibst, 1973; Hammer et al., 1974), urban wetlands (Doss and Taff, 1996; Mahan et al., 2000), water resources (e.g., Leggett and Bockstael, 2000), urban forests (e.g., Tyrvainen and Miettinen, 2000), and general environmental amenities (e.g., Smith 1978; Palmquist 1992). Although

Chicago Wilderness has not used this method to date, the large number of residential property sales in the Chicago area and spatially explicit databases on many environmental attributes offers great potential to use hedonic property price analysis to estimate the values of environmental amenities.

A large literature has used the travel cost method to value recreation sites. With the large number of visitors to Lake Michigan beaches, forest preserves, and parks in the Chicago metropolitan area, Chicago Wilderness could also apply travel cost to estimate the value of recreational activities. Several studies have applied the travel cost method in urban areas (e.g., Binkley and Hannemann ,1978; Lockwood and Tracy, 1995; Fleischer and Tsur, 2003).

Stated-preference methods can also be used to estimate the value of recreational opportunities and environmental amenities. In one such study completed for Chicago Wilderness, Kosobud (1998) used a contingent valuation survey to estimate willingness to pay for the recovery or improvement of natural areas in the Chicago region. Kosobud found an average willingness to pay for expanded natural areas of approximately $20 per household per year. Extrapolating over the number of households in the region, expansion of natural areas in the region would generate about $50 million per year in benefits.

Finally, there is a small but growing literature that estimates values from voting behavior in referenda involving environmental issues. In particular, studies have analyzed the value of open space using results of voting on open-space referenda (Kline and Wichelns, 1994; Romero and Liserio, 2002; Vossler et al., 2003; Vossler and Kerkvliet, 2003; Schläpfer and Hanley, 2003; Schläpfer et al. 2004; Howell-Moroney, 2004a, 2004b; Solecki et al.. 2004; Kotchen and Powers, 2006; Nelson et al., 2007). As noted earlier, several counties in the Chicago metropolitan area have passed referenda authorizing bonds to purchase open space or protect watersheds. Although the number of referenda is relatively small, making it difficult to generalize or make comprehensive statements about values, analysis of these referenda could provide insights into the values different segments of the public place on various environmental amenities.

The only methods currently accepted by economists for estimating non-use values, such as the existence value of natural communities or biodiversity, are stated-preference methods such as contingent valuation and conjoint analysis. To estimate the existence value of protecting species and ecological systems, Chicago Wilderness could survey respondents in the Chicago area. Alternatively, it could attempt to use economic benefits transfer by applying the results of relevant surveys done in other locations. The advantage of obtaining a monetary value for the conservation of species and ecological systems through contingent valuation or conjoint analysis is that it would allow Chicago Wilderness to calculate a total economic value for alternative strategies. Without contingent valuation or conjoint analysis, non-use value could not be included, and only a partial economic value estimate for each strategy could be derived.

Any effort to estimate a monetary non-use values raises the questions of whether monetary values are commensurate with the types of values that Chicago residents attach to protecting natural communities. In discussing the importance of protecting biodiversity, Chicago Wilderness emphasizes that a survey of Chicago focus groups found that “responsibility to future generations and a belief that nature is God’s creation were the two most common reasons people cited for caring about conservation of biodiversity” (*Biodiversity recovery plan*, p. 14). Contingent valuation of the bequest value of biodiversity might be consistent with measuring responsibility to future generations, although the respondents in the focus group were presumably thinking in moral rather than monetary terms. Strong differences of opinion exist on whether it is appropriate to try to capture such notions as stewardship or moral values in monetary terms using stated preference methods (CITATIONS).

Citizen juries or decision-science methods also provide a useful means of evaluating tradeoffs among potential strategies in the Chicago Wilderness context. With citizen juries, experts could work with a small group of selected individuals in the Chicago area to determine comparative values for parcels of land through a guided process of reasoned discourse. These methods might enable participants to develop more thoughtful and informed valuations, better analyze tradeoffs among multiple factors, and engage in a more public-based consideration of values. Decision science methods could provide either monetary comparisons of the values of alternative properties or weights that could be used to aggregate multiple layers of data.

Monetary values derived through citizen juries or decision science approaches may differ considerably from traditional private values, both because of the consent-based choice rules employed and the explicitly public-regarded nature of the valuation exercise. Recent analysis suggests that deliberative valuations may aggregate individual values in a manner that systematically departs from the additive aggregation procedures of standard benefit-cost analysis (Howarth and Wilson, 2006). Monetary values from deliberate processes, in short, do not necessarily yield traditional economic benefit measures.

Although valuation information could be of great use to decision makers in evaluating alternative strategies and in communicating consequences of the alternatives to the public, Chicago Wilderness has undertaken very little valuation research or analysis. Despite some attempts to collect information about the value of protecting natural communities and ecosystem services (e.g., Kosobud 1998), Chicago Wilderness' efforts have not been comprehensive or systematic. This contrasts with its major efforts to garner broad stakeholder involvement and input in setting the goals for the organization and its large-scale effort to collect technical and scientific knowledge to characterize the status of ecosystems and species. In part, the lack of valuation activity has been the result of the mix of expertise of the individuals involved in Chicago Wilderness. In part, the lack of valuation activity is the result of the organization's choice regarding the set of activities most important to it (which is a different sort of revealed preference). Chicago Wilderness is interested in using economic and other social-science approaches to study the value of protecting natural communities but has not yet enjoyed the right mix of expertise and circumstances to make this a reality.

6.2.3. Other case studies

6.2.3.1 Portland, Oregon's assessment of the value of improved watershed management

In the early 2000s, Portland, Oregon, decided to analyze the ecosystem benefits and ecosystem-service values that would result from improved watershed management. Portland officials hoped to find more effective approaches to watershed management that could both save the city money and improve the welfare of its citizens. The city was particularly interested in impacts on flood abatement, water quality, aquatic species (salmon in particular), human health, air quality, and recreation. The city's Watershed Management Program requested David Evans & Associates and ECONorthwest to undertake the study, completed in June 2004 (David Evans & Associates and ECONorthwest, 2004). Although not an example of a regional partnership with EPA, the project provides one of the best examples of the kind of landscape-scale analysis of the value of ecosystems and services recommended by this report.

City officials realized that they understood only a portion of the contributions to well-being from improved watershed management. To be able to make more intelligent decisions about watershed management, these officials wished to have a more complete accounting. The project aimed to expand the range of ecological changes that were valued, focusing on those changes in ecosystems and services likely to be of greatest concern to the population. The study monetized the economic benefits from a variety of ecosystem services, including flood abatement, biodiversity maintenance (represented by improvement of avian and salmon habitat), air quality improvement, water quality improvement (measured by reduction of water temperature), and cultural services (which the study defined as including the creation of recreational opportunities and increase of property values).

The project commissioned both biophysical and economics analyses. The biophysical analyses included studies of hydrology and flooding potential, water quality, water temperature, habitat for salmon and other aquatic species, habitat for birds and other terrestrial species along riparian buffers, and air quality impacts (ozone, sulfur dioxide, carbon monoxide, carbon, and particulates). The economic analyses included studies of the impact of ecosystem changes on property values (including public infrastructure and residential and commercial property), flood risks, recreation, and human health.

The project used an approach that closely resembles the ecological production function approach advocated in this report The approach linked management changes, such as flood project alternatives, to a range of ecological changes. These ecological changes were then analyzed for their effect on various ecosystem services. Finally, the analysis attempted to economically value the changes in ecosystem services. Although conducted by separate teams, the project closely linked the ecological analyses and economic valuation.

Of particular note was the emphasis on estimating the change in values that would occur under various management alternatives. Rather than provide a static description of current conditions, which is the predominant form of information collected by Chicago Wilderness, Portland's approach tried to estimate cause-and-effect relationships that would allow the systematic appraisal of alternative policy or management decisions. This focus, along with a systems approach capable of incorporating multiple economic benefits, made this an effective vehicle to study the net economic benefits of alternative management options.

The Portland study illustrates a number of good practices in conducting an integrated, regional-level analysis. The project solicited input from the public and important stakeholder groups in the design of the project so that it captured the impacts of greatest interest to the public. The project presented its results with a graphical interface that allowed stakeholders to run scenarios and see the resulting impacts based on underlying biophysical and economic models. The analysis effectively deployed existing methods and estimates, although it did not attempt to develop or test new approaches or methods.

The project also illustrates some of the potential problems and limitations in undertaking detailed quantitative landscape-scale analysis. Inevitably, there are gaps in data and understanding in this type of analysis. Gaps in understanding include how changes in management actions will affect ecological systems, and how this will affect the provision of ecosystem services and consequent value. For example: How will songbird populations change in response to changes in the amount and degree of habitat fragmentation? What is the value to residents of Portland of changes in songbird populations? Because of a lack of local information, the study often had to use economic benefits transfer, drawing on cases quite different from the Portland context to generate estimates of values.

The project was commissioned by the City of Portland and although it had minimal EPA involvement, the project is a good example of the type of systematic and integrated approach to valuing the protection of ecosystems and services recommended by this report. The project aptly illustrates the sequence of steps, from stakeholder input, to characterizing change in ecosystem functions under various policy and management options, to valuation of services under these alternatives. The project shows the great potential that this type of analysis offers in providing important and useful information to decision makers.

6.2.3.2 Southeast ecological framework project

The Southeast ecological framework (SEF) project represents a regional geographic information system (GIS) approach for identifying important ecological resources to conserve. The Southeast region, which encompasses Alabama, Florida, Georgia, Kentucky, Mississippi, North Carolina, South Carolina, and Tennessee, is one of the fastest growing regions in the country, yet it still harbors a significant amount of globally important biodiversity and other natural resources. The SEF seeks to enhance regional planning across political jurisdictions and help focus federal resources to support state and local protection of ecologically important lands. The Planning and Analysis Branch of EPA Region 4 and the University of Florida completed the work in December 2001.

The SEF created a new regional map of priority natural areas and connecting corridors, along with GIS tools and spatial datasets. The project also identified 43 percent of this land that should be protected and managed for its specific contributions to human well-being. The project developed additional applications for conservation planning at the sub-regional and local scales.

The SEF offers a good tool to carry out regional analysis of ecological components, particularly habitat conservation. The SEF focused narrowly on conservation value, defined as the ability to sustain species and ecological processes. Because of its focus, the level of scientific knowledge underpinning the SEF is, in general, far higher than in the other case studies examined here.

The SEF, however, does not reflect the broad, integrated approach to valuation recommended by this report. The SEF focuses almost exclusively on habitat conservation rather than on a broad suite of ecosystem services. The SEF did not undertake extensive stakeholder involvement to determine its objective; it started with a focus on habitat conservation. It also did not attempt to combine its ecological analysis with an effort to value the protection of ecosystems or services in monetary or other terms. An important challenge facing regional analysis, particularly at a broad scale like the eight-state Southeast region, is how to incorporate all of these essential elements – a rigorous ecological approach capable of showing the range of ecological impacts from alternative policy and management decisions; stakeholder involvement and input on what consequences are of greatest importance to them; and rigorous evaluation of changes in value under alternative decisions.

6.2.4. Summary and recommendations

Regional-scale analysis holds great potential to inform decision makers and the public about the value of protecting ecosystems and services. Recent increases in publicly available, spatially-explicit data and a parallel improvement in the ability to display and analyze such data make it feasible to undertake comprehensive regional-scale studies of the value of protecting ecosystems and services. Municipal, county, regional, and state governments make many important decisions affecting ecosystems and the provision of ecosystem services at a regional scale, but local and state governments rarely have the technical capacity or the necessary resources to undertake regional-scale analyses of the value of ecosystems or services. Regional-scale partnerships between EPA regional offices, local and state governments, regional offices of other federal agencies, environmental non-governmental organizations, and private industry could aid both EPA and regional partners. Such partnerships offer great potential for improving the science and management for protecting ecosystems and enhancing the provision of ecosystem services.

At present, however, this potential is largely unrealized. Valuation of ecosystems and services has not been a high priority for EPA regional offices largely because of tight agency budgets and the lack of specific legal mandates and authority. To date, regional offices have not undertaken the valuation of ecosystems and services at a regional scale in a comprehensive or systematic fashion. As the case studies have shown, however, various regional EPA offices and local governments have pursued some innovative and promising directions despite limited budgets and lack of specific mandates.

The committee sees great value in undertaking a comprehensive and systematic approach to valuing ecosystems and services at a regional scale. Realizing the great potential of regional-scale analyses, however, will require a significant increase in resources for regional offices and, in some cases, a somewhat different mode of operation. To reach the potential for regional-scale analysis of the value of ecosystems and services, the committee recommends that:

- EPA should encourage its regions to engage in valuation efforts to support of environmental decision making, following the recommendations of this report.
- EPA regional staff should be given adequate resources to develop expertise necessary to undertake comprehensive and systematic studies of the value of protecting ecosystems and services. Increased expertise is needed in several areas:
    - Economics and social science: Expertise is very limited at the regional level to undertake economic or other social assessments of value. A pressing need exists to increase expertise in this area among regional offices.
    - Stakeholder involvement processes.
    - Ecology: Regional staffs have greater expertise in ecology than in stakeholder involvement, economics or other social sciences, but doing systematic valuations of ecosystem services will require additional ecological staff. Of greatest utility would be ecologists with expertise in assessing impacts on ecosystem services through ecological production functions to evaluate alternative management options.
- A systematic and comprehensive approach to valuing the protection of ecosystems and services requires that ecologists and other natural scientists work together with economists and other social scientists as an integrated team. Regional-scale analysis teams should be formed to undertake valuation studies. Teams composed of social scientists and natural scientists should participate from the beginning of the project to design and implement plans for stakeholder involvement, ecological production functions, and valuation.
- Gathering stakeholder input is of great importance in establishing the set of ecological consequences of greatest importance to the community. All regional-scale analyses of the value of ecosystems and services should involve stakeholders at an early stage to ensure that subsequent ecological, economic, and social analyses are directed toward those ecosystem components and

services deemed of greatest importance by affected communities. Generally, the process should proceed bottom-up, as opposed to top-down. Rather than asserting what is valuable, EPA must seek to understand what various communities view as being valuable. An important question that should be addressed by EPA regional offices is how to develop effective stakeholder involvement at broader regional scales.

- Some EPA staff have expressed a desire to be provided a value for an ecosystem component or service that they can then apply to their region (e.g., a constant value per acre of wetland or wildlife habitat). Such short cuts to the valuation process are uninformed by local social, economic, and ecological conditions and can generate results that are not meaningful. This approach to valuation should be avoided.
- Regional staffs need to be able to learn effectively from valuation efforts being undertaken by other regional offices. EPA regional offices should document valuation efforts and share them with other regional offices, with EPA's National Center for Environmental Economics, and with EPA's Office of Research and Development. Each regional office should also publish their studies.
- Future calls by the Agency for extramural research should incorporate the research needs of regional offices for systematic valuation studies. Doing so will maximize the probability that future grant funding will be useful for EPA's regional offices.
- Regional staff should form partnerships with local and state agencies or local groups where doing so advances the mission of EPA directly or indirectly by promoting the ability of partner organizations to value the effect of their actions on ecosystems and services and to protect environmental quality.

6.3. Valuation for site-specific decisions

6.3.1. Introduction

The Environmental Protection Agency makes many decisions at the local level, including the issuance of permits (air, water, and waste), policies that influence the boundaries for establishing permits (e.g., impaired water bodies designations), and administrative orders related to environmental contamination. The social and ecological implications of such decisions, like the decisions themselves, generally are local in nature, affecting towns, townships, and counties rather than entire states or regions. Therefore, the decision processes need to rely on valuation approaches that also are local in nature and are robust enough to adapt to a range of local stakeholder interests.

In this section, the committee focuses on the regulatory processes associated with one set of local decisions: the remediation and redevelopment of historically contaminated sites. That focus includes the Superfund program and its efforts to assess the contributions to human well-being from ecosystem services related to site remediation and redevelopment efforts (Davis, 2001; Wilson, 2005). As part of this committee's study, the SAB staff, with assistance from the Agency's National Regional Science Council, surveyed the regional offices to assess their needs for valuation information. Seven of the eight responding regions indicated that they need information to help value the protection of ecosystems in the management and remediation of contaminated sites (EPA Science Advisory Board Staff , 2004). The discussion that follows is applicable to any remediation and redevelopment processes for contaminated properties that contain the following elements:

- Identification, selection, and prioritization of sites
- Site characterization – establishment of site condition
- Site assessment – evaluation of risks and impacts
- Selection of remedial and redevelopment approaches
- Performance assessment of clean up and redevelopment
- Public communication of assessment results as well as proposed actions and outcomes

This section explores how valuation methods can positively influence individual steps in a remediation and redevelopment process and lead to a better outcome. As appropriate, the section identifies and discusses individual valuation approaches or methods relevant to specific steps. The section builds on a white paper funded by EPA's Superfund Program to evaluate the potential of valuation for redevelopment of contaminated sites (Wilson, 2005). The white paper assessed the improvement in ecosystem services and implied ecological value from the remediation and redevelopment of Superfund sites. Although the Wilson paper did not perform a formal valuation for any redeveloped property, it provides a useful starting point for exploring the utility of valuation methods in the remediation and redevelopment process. For his analysis, Wilson reviewed approximately 40 Superfund cases before selecting three case studies that represent urban (Charles-George landfill), suburban (Avtex Fibers), and exurban (Leviathan mine) environments. This section analyzes and relies on these same three cases, as well as an additional urban example, the DuPage landfill, which provides a useful counterpoint to the Charles-George landfill example. The DuPage example shows how an early focus on ecosystem services can better identify potential ecosystem services that can be targeted during the remediation and restoration phases. A brief overview of each of these cases appears in section 6.2.3.

6.3.2. Opportunities for using valuation to inform remediation and redevelopment decisions.

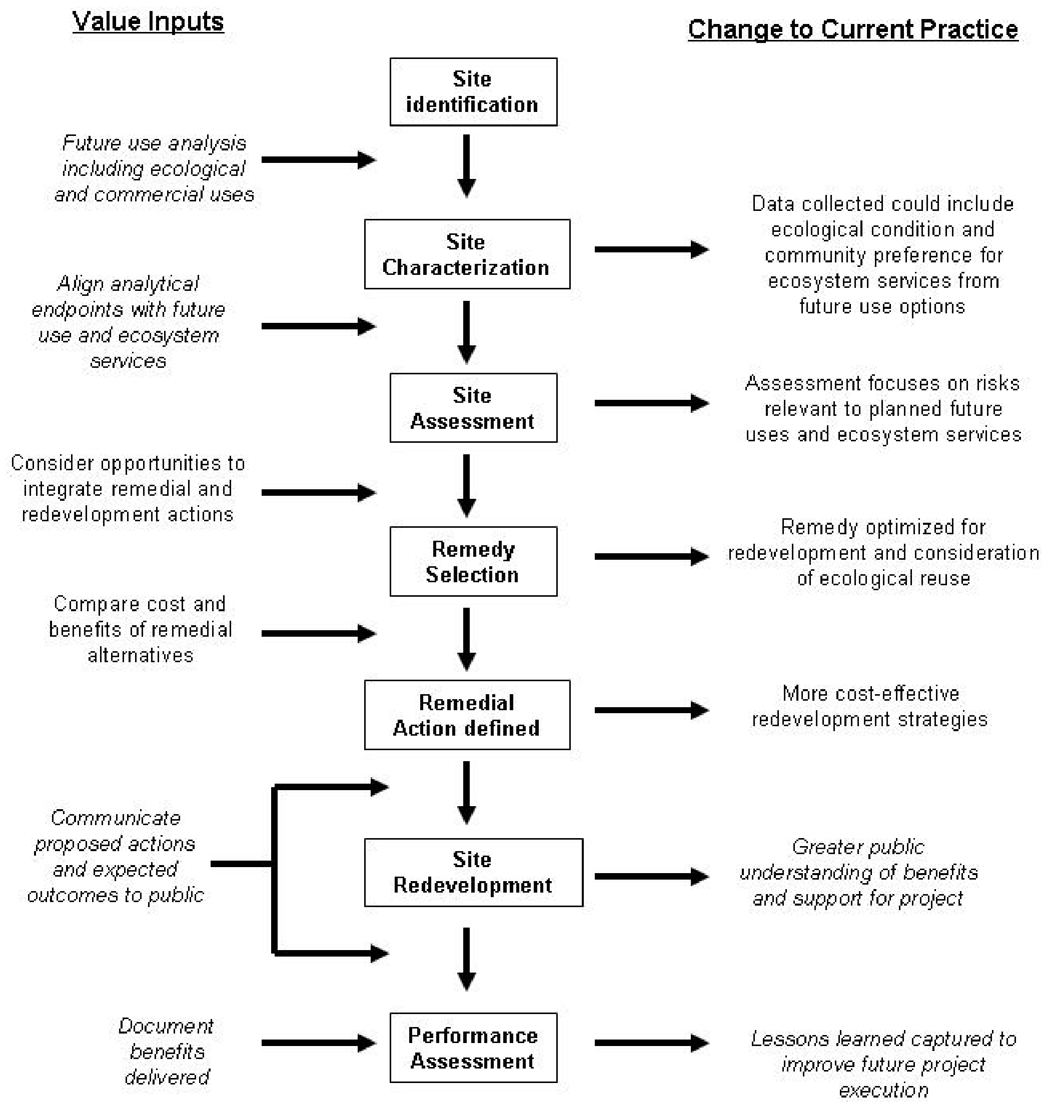
The Superfund process and its individual steps or stages are well defined (EPA CERCLA Education Center, 2005). Superfund and related remediation processes are focused on first defining a problem, then characterizing and assessing its potential and actual human health and environmental impacts, and finally developing and executing a technical strategy to alleviate or avoid those impacts. Since 1985, EPA's Brownfield Program has integrated consideration of upstream redevelopment into the remediation process (EPA, 2004). The Agency developed the reuse assessment tool to integrate land use into the Superfund process (Davis, 2001). Integrating remediation and redevelopment demonstrates the need to consider ecological valuation into all steps and stage from the very beginning.

Figure 5 illustrates how valuation information can be integrated into the traditional process for remediation and redevelopment. In the committee's view, EPA and the community should define at the outset what the potential site should be after remediation and redevelopment and what ecological services are to be preserved, restored, or enhanced for use by the local community. This differs from the more traditional practice. This practice initially focuses on the type, degree, and extent of chemical contamination, and then on the human and ecological receptors currently exposed and therefore at risk under current chemical conditions.

In the traditional approach, the data collection for site characterization captures the degree and pattern of chemical contamination but does not collect information about the ecological condition of the site or the value of any services associated with the site in its current or proposed future conditions. In the traditional approach, moreover, the conceptual model that defines the exposure pathways to key receptors and therefore guides the design of the risk analysis is based on current rather than future conditions. This can lead to a risk assessment that selects receptors that are sensitive under current conditions but may not be sensitive or important under alternative future uses. This logic focuses remedy evaluation and selection on controlling the risks under current use. In the end, the traditional approach assumes that risk reduction and management, rather than the optimized reuse value for the community, are the ultimate performance goals. Such an approach may leave the community feeling that the risk is gone but still dissatisfied with the values gained by the cleanup.

Integrating future use considerations into the remediation process and focusing on value generation will lead to outcomes that better satisfy the public. To accomplish this metamorphosis, it is essential to find ways to introduce estimates of ecosystem services and values into management strategies and associated analytical processes. Early recognition of future uses and the ecosystem services that matter to people can inform site assessment and the ultimate selection of remedial actions and redevelopment options. Identifying expected or actual contributions to human well-being can also lead to more effective communication with the affected public. The rest of this section discusses the opportunities and utility of adapting valuation methods to this more integrated and forward-looking assessment and redevelopment process.

Figure 5: Integration of valuation information with the traditional remediation and redevelopment process



Valuation methodologies are important first in identifying how a site and its current or potential ecosystem services matter to the surrounding community. EPA should use valuation methods to determine how the site has contributed and can contribute to human well-being and how potential effects on ecological components may diminish those contributions. When the ecosystem services that matter to people are well-defined and when ecological risk assessments are coupled with these services, the remediation and redevelopment plan can target what matters to the local community. A key recommendation, therefore, is that EPA consider ecosystem services and their contributions to human well-being and other values from the earliest stages of addressing contaminated properties.

Even as early in the management process as site selection or prioritization, tools that can compare the potential of sites to provide ecosystem services could be informative. The contribution of ecological protection to human well-being should be considered in the design of any site characterization plan. A typical site characterizations focuses on the aerial extent of chemicals and their range of concentration in site media (e.g., ground and surface water, soil, and biological tissue). A plan that also collects information to define and assess ecosystem services would better align ecological-risk assessments with economic benefits and other contributions to human well-being. Aligning risk assessments and assessments of contributions to human well-being should be a critical objective for the Agency. Alignment will help ensure that remedial actions address the restoration of contributions to human well-being derived from important ecosystem service flows that have been diminished or disrupted. Aligning risk assessment endpoints with ecosystem services should also result in multiple benefits, including:

- Improved alignment with community goals
- Improved ability to perform meaningful assessments of economic benefits and other contributions to human well-being
- Improved ability to communicate proposed actions
- Improved ability to monitor and demonstrate performance

Successfully remediating and redeveloping contaminated sites depends in great part on the degree to which efforts either protect or restore ecosystem services that contribute to human well-being. If values have been broadly explored and effectively integrated into site assessment and remedy-selection processes, appropriate measures of performance will be apparent. Ecological measures of productivity or the aerial extent of conditions directly linked to valued ecosystem services will be useful in tracking the performance of remediation and redevelopment processes. Advancing the Agency's capability to evaluate performance both in real time and retrospectively will help the Agency better justify its overall performance record in the remediation and redevelopment of contaminated sites.

Finally, the remediation and redevelopment of a property encompasses more than just the science and engineering that historically have underpinned the remediation process. Effective communication with stakeholders actively participating in the remedial and redevelopment process and with the general public is a critical element in the success of the management process. Both of these audiences bring values to the table when they evaluate proposed actions or the results of any action taken. A strong alignment between the ecosystem services valued by these audiences and expected or actual outcomes will facilitate effective remediation and redevelopment.

6.3.3. Illustrative site-specific examples

The following analysis applies the general recommendations of chapter 2 to the site-specific level. The committee illustrates these site-specific recommendations with lessons gleaned from a series of Superfund examples in urban (Charles-George and DuPage landfills), suburban (Avtex Fibers) and ex-urban (Leviathan Mine) contexts. The backgrounds on each of these cases appear below..

6.3.3.1 Determining the ecosystem services important to the community and key stakeholders.

The urban examples of the Charles-George and DuPage County landfills show the value of engaging with the community at an early stage to determine the ecosystem services of importance to them. Although neither landfill apparently used formal valuation methods at the outset, DuPage County's focus on ecosystem services and the inclusion of additional experts (i.e., forestry experts) led to a more positive outcome.

Charles-George Landfill: An urban example

From the late 1950s until 1967, the Charles-George Reclamation Trust Landfill, located one mile southwest of Tyngsborough and four miles south of Nashua, New Hampshire, was a small municipal dump. A new owner expanded it to its present size of approximately 55 acres and accepted both household and industrial wastes from 1967 to 1976. The facility had a license to accept hazardous waste from 1973 to 1976 and primarily accepted drummed and bulk chemicals containing volatile organic compounds (VOCs) and toxic metal sludges. Records show that over 1,000 pounds of mercury and approximately 2,500 cubic yards of chemical wastes were landfilled. The state ordered closure of the site in 1983. That same year, EPA listed the site on the National Priorities List (NPL) and the owner filed for bankruptcy. Samples from wells serving nearby Cannongate Condominiums and some nearby private homes revealed VOCs and heavy metals in the groundwater. Approximately 500 people lived within a mile of the site in this residential/rural area; 2,100 people lived within three miles of the site. The nearest residents were located 100 feet away. Benzene, tetrahydrofuran, arsenic, 1,4-dioxane, and 2-butanone, among others, had been detected in the groundwater. Sediments had been shown to contain low levels of benzo(a)pyrene. People faced a potential health threat by ingesting contaminated groundwater. Flint Pond Marsh, Flint Pond, Dunstable Brook, and nearby wetlands were threatened by contamination migrating from the site.

EPA's involvement at the site began with groundwater testing conducted by an EPA contractor during 1981 and 1982. The site was proposed for the NPL on October 23, 1981, and finalized on the NPL in September 1983. In September 1983, EPA also allocated funds for a removal action at the site to replace the state's Department of Environmental Quality Engineering temporary water line with another temporary but insulated water line. Other removal work included construction of a security fence along the northwestern entrance to the landfill, regrading and placement of soil cover over exposed refuse, and installation of twelve gas vents. A remedial investigation and feasibility study was also begun in September 1983. The basis for the removal action was documented in the first record of decision issued on December 29, 1983.

EPA Web Site History:
http://yosemite.epa.gov/r1/npl_pad.nsf/f52fa5c31fa8f5c885256adc0050b631/ABD286D719D254878525690D00449682?OpenDocument

At the Charles-George landfill, EPA did not consider ecological values or future uses at the start. The human health risks at this site were so salient at the time that they were the focus of subsequent decisions. EPA addressed the health and safety risks by capping the landfill site and extending the water system from the city of Lowell, Massachusetts, to the affected community. Although EPA published the record of decision more than 20 years ago, the site is still a fenced-off no-man's land, and the potential for ecosystem services remains untapped.

By contrast, the remediation and redevelopment of the DuPage County landfill site appears to have been motivated largely by the need to address existence values (e.g., the presence of hawks and other rare birds) and recreational values (e.g., hiking, bird watching, boating, camping, picnicking, and sledding). The remediation effort succeeded, and the site is now part of the Blackwell Forest Preserve. Listed as a Superfund site in 1990, "a once dangerous area is now a community treasure, where visitors picnic, hike, camp, and take boat rides on the lake" (CITATION NEEDED).

DuPage County Landfill: An urban example

The 40-acre tract of land that is the Blackwell Landfill was originally purchased by the DuPage County Forest Preserve District (FPD) in 1960 and is centrally located within the approximately 1,200-acre Blackwell Forest Preserve, about 30 miles outside Chicago, Illinois. The landfill was constructed as a honeycomb of one-acre cells lined with clay. Approximately 2.2 million cubic yards of wastes were deposited in the landfill between 1965 and 1973. The principal contaminants of concern for this site were the volatile organic compounds (VOCs) 1,2-dichloroethene, trichloroethene and tetrachloroethene, detected in onsite groundwater at or slightly above the maximum contaminant level (MCL). Landfill leachate contained all kinds of VOCs and semivolatiles including benzene, ethylbenzene toluene, and dichlorobenzene, as well as metals such as lead, chromium, manganese, magnesium, and mercury. VOCs and agricultural pesticides had also been detected in private wells down gradient of the site but at low levels. Some metals (manganese and iron) had been detected above the MCLs in down-gradient private wells. Post-remediation, the site now consists mainly of open space, containing woodlands, grasslands, wetlands, and lakes, used by the public for recreational purposes such as hiking, camping, boating, fishing, and horseback riding. There are no residences on the FPD property, and the nearby population is less than 1,000 people. The landfill created Mt. Hoy, which is approximately 150 feet above the original ground surface.

EPA Web Site History:
http://www.epa.gov/superfund/programs/recycle/impacts/pdfs/dupage.pdf)

The urban examples show that even the most rudimentary dialogue about future use can lead to an outcome with greater service to the community. At the DuPage landfill site, a qualitative focus on the utility of ecosystem services led to the recognition that in a very flat landscape, even a 150-foot hill, if properly capped and planted, would be a welcome refuge for people as well as wildlife. The DuPage Forestry District understood the ecological potential of the area, particularly for hawks, and recognized that, where hawks abound, birders will come to watch them. The difference was one not of methodology but of conception.

In working with the Avtex Fibers site, a suburban location, EPA also engaged key stakeholders. After the site was listed and a management process established, EPA undertook a clear effort to engage stakeholders through a multi-stakeholder process in the development of the master plan. Although there was some consideration of ecosystem services, EPA does not appear to have engaged in any systematic efforts to assess the services that people cared the most about.

Avtex Fibers Site: A suburban example

The Avtex Superfund site consists of 440 acres located on the bank of the Shenandoah River within the municipal boundaries of Front Royal, Virginia. The site is bordered on the east by a military prep school, on the south by a residential neighborhood, and on the west by the Shenandoah River. From 1940 to its closure in 1989, industrial plants on the site manufactured rayon and other synthetics. Tons of manufacturing wastes and byproducts accumulated on the site, infiltrated into groundwater under the site, and escaped into the Shenandoah River. The Avtex Fibers site was proposed for inclusion on the National Priorities List on October 15, 1984, and the site was formally added to the list on June 10, 1986. EPA began removal activities at the site in 1989 to address various threats to human health and the environment. The cleanup and restoration plan called for most remaining wastes to be consolidated on site, secured with a protective material where needed, and covered by a thick cap of soil and vegetation.

Front Royal is close to the Appalachian Trail, Shenandoah National Park, and George Washington National Forest, and a number of significant Civil War sites, making it a major tourist center for the Blue Ridge Mountains. Biologically, the Avtex site contains some residual forested areas, open meadows, small wetland areas, and more than a mile and a half of frontage along the Shenandoah River. The proposed master plan for redevelopment, created through a formal multi-stakeholder process, divides the site into three areas: a 240-acre river conservancy park along the Shenandoah River combining ecological restoration and conservation of native habitats; a 25-acre active recreation park with boat landings, picnic shelters, and a developed recreation area including a visitor center and soccer fields; and a 165-acre eco-business park, featuring the refurbished historic former Avtex administration building. Cleanup of the Axtex site is ongoing, and the redevelopment plan is being actively pursued by local government agencies and private industry groups.

EPA Web Site History:
http://www.epa.gov/superfund/accomp/success/avtex.htm

Stakeholders' Avtex Fibers Conservancy Park Master Plan
http://www.avtexfibers.com/Redevelopment/avtexWEB/avtex-Mp.html\

For sites like Avtex Fibers, deliberative group processes involving stakeholders and relevant experts, including historians, could help identify and document ecosystem services of most concern to stakeholders. In framing the dialogue with stakeholders, methods such as ecosystem benefits indicators or the conservation value method might help EPA's site managers understand the ecosystem-service potential of future uses. Those methods could also provide inputs for further valuation using other methods described in chapter 4 (e.g., economic methods or decision science approaches).

The Leviathan Mine case illustrates how EPA often must consider a complex array of competing interests. The Agency in this case faces a clear dichotomy between the ecosystem services valued by the full-time resident population of American Indians and by occasional recreational users. Recreational users would gain from services associated with hiking, fishing, and camping. The Washoe Tribe, however, values the ecosystem as a provisioning service for food as well as for its spiritual and cultural services.

Leviathan Mine Superfund Site: An ex-urban example

In May 2000, the EPA added the Leviathan Mine site in California to the National Priority List of Superfund sites. The site is currently owned by the state, but from 1951 until 1962 the mine was owned and operated by the Anaconda Copper Mining Company (a subsidiary of ARCO) as an open pit sulfur mine. The mine property is 656 acres in a rural setting near the Nevada border, 24 miles southeast of Lake Tahoe. The mine itself physically disturbed about 253 acres of the property plus an additional 21 acres of National Forest Service land. The site is surrounded by national forest. In addition, it lies within the aboriginal territory of the Washoe Tribe and is close to several different tribal areas.

The mine has been releasing hazardous substances since the time that open pit mining began in the 1950s. Releases occur through a number of pathways, including surface water runoff, groundwater leaching, and overflow of evaporation ponds. In particular, precipitation flowing through the open pit and overburden and waste rock piles creates acid mine drainage (AMD) in the form of sulfuric acid, which leaches heavy metals (such as arsenic, cadmium, copper, nickel, and zinc) from the ore. These releases are discharged into nearby Leviathan Creek and Aspen Creek, which flow into the East Fork of the Carson River. Pollution abatement projects have been underway at the site since 1983. Despite these efforts, releases continue today.

The releases of hazardous substances from the mine have significantly injured the area's ecosystem and the services it provides. In the 1950s, structural failures at the mine that released high concentrations of AMD into streams resulted in two large fish kills, and the trout fishery downstream of the mine was decimated during this time. More recently, data have documented elevated concentrations of heavy metals in surface water, sediments, groundwater, aquatic invertebrates, and fish in the ecosystem near the site. This suggests that hazardous substances have been transmitted from abiotic to biotic resources through the food chain, thereby affecting many trophic levels. A recent assessment identifies seven categories of resources potentially impacted by the site: surface water resources, sediments, groundwater resources, aquatic biota, floodplain soils, riparian vegetation, and terrestrial wildlife. The assessment identified five types of ecosystem services that might be provided by these resources: aquatic biota (including the threatened Lahontan cutthroat trout) and supporting habitat; riparian vegetation; terrestrial wildlife (including the threatened bald eagle); recreational uses (including fishing, hiking, and camping); and tribal uses (including social, cultural, medicinal, recreational, and subsistence).

The process of determining compensatory damages and developing a response plan involves a number of different stages for which information about the value of these lost services would be a useful input. For example, in accordance with the Natural Resource Damage Assessment (NRDA) regulations under the Comprehensive Environmental Response, Compensation and Liability Act, the trustees for the site conducted a pre-assessment screening to determine the damages or injuries that may have occurred at the site and whether a natural resource damage assessment should be undertaken. This required a preliminary assessment of the likelihood of significant ecological or other impacts from the contamination (corresponding to step 2 in figure 1 of this report).

The decision was made in July 1998 to move forward with a Type B NRDA and thus to assess the value of the ecosystem services that have been lost as a result of the site contamination. A Type B assessment involves three phases: an injury determination to document whether ecological damages have occurred; a quantification phase to quantify the injury and reduction in services (corresponding to step 4 of figure 1); and a damage determination phase to calculate the monetary compensation that would be required (corresponding to step 5 of figure 1).

In the Leviathan Mine case, the trustees proposed using resource equivalency analysis based on a replacement cost estimate of the lost years of natural resource services to determine damages for all affected services other than non-tribal recreational fishing. For this latter ecosystem service, they proposed using economic benefit transfer to estimate the value of lost fishing days. In the decision by EPA whether to list the site on the NPL and in the subsequent record of decision selecting a final remedy for the site, information about the value of the ecological improvements from cleanup could play an important role, although these decisions have often been based primarily on human health considerations.

EPA Web Site History:
http://www.epa.gov/superfund/sites/npl/nar1580.htm

Leviathan Mine National Resource Damage Assessment Plan,
http://www.fws.gov/sacramento/ec/Leviathan%20NRDA%20Plan%20Final.pdf

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The Leviathan Mine case also highlights the need to consider the existence or intrinsic values of an ecosystem. The ecosystem near the Leviathan Mine provides a habitat for threatened species such as the Lahontan cutthroat trout and bald eagle. In considering site restoration or remediation, or in measuring damages from contamination at the mine, the

Agency could miss the primary sources of value if it limited consideration to use value and did not consider existence or intrinsic value.

For the Leviathan Mine example, EPA could obtain information about the impacts of greatest concern to affected individuals in at least three ways. The first would be to gather information about the relative importance of the various services directly from affected individuals through focus groups, mental models, mediated modeling, deliberative processes, or anthropological or ethnographic studies based on detailed interviews. The second approach would be to gather basic information that could indicate the importance of different services. This information might be of the type used to construct ecosystem benefit indicators: water use data for the Washoe tribe and others in the vicinity of the site (e.g., sources, quantities, and purposes), harvesting information for the Washoe (e.g., what percent of their harvesting of nuts, fish, etc., comes from the area affected by the site), recreational use data (e.g., the number of people visiting the local national forest for hiking, camping, fishing, and wildlife viewing), data on flooding potential and what is at risk in the vicinity of the site, and data on spiritual/cultural land-use practices by the Washoe. The third approach would be to review related literature and previous studies to learn about impacts of concern in similar contexts. For example, previous social/psychological surveys not specific to this site or other expressions of environmental preferences (e.g., outcomes of referenda or civil court jury awards) might provide insight into what people are likely to care about in this context. Similarly, previous contingent valuation studies of existence value might provide some, at least partial, indication of the likely importance of impacts on species such as bald eagles. Likewise, previous studies of the value of recreational fishing (e.g., from travel cost models) could be coupled with use data to provide an initial indication of the importance of the impact on recreational fishing.

<u>6.3.3.2</u> <u>Involving interdisciplinary experts appropriate for valuation.</u>

Interactions among experts and the affected public form a key component of any program of hazardous site assessment, planning, and implementation. Ideally, collaboration among all relevant experts, including physical, chemical, and biological scientists (e.g., ecologists and toxicologists) and social scientists (e.g., economists, social psychologists, and anthropologists), as well as communication with affected stakeholders, must begin very early in the planning stages of remediation and redevelopment and continue throughout implementation and post-project monitoring and evaluation. Key areas for collaboration among experts are the development of alternative management scenarios and the translation of physical and biological conditions and changes into value-relevant outcomes that can be communicated to stakeholders.

The Leviathan Mine illustrates the need for collaboration among multiple disciplines to understand how the population's values are affected. Because of the unique cultural and spiritual values associated with the site, anthropologists could play an important role in characterizing the value of the ecosystem services to the Washoe Tribe. Economists or others seeking to estimate existence value for an affected species would need to work closely with ecologists to determine the likely impact of any change or proposed project on that species so that the change could be readily valued.

6.3.3.3 Constructing conceptual models that include ecosystem services

Ecological assessments associated with the remediation and redevelopment of contaminated property will better aid decision making if they incorporate ecological production functions that link remediation and redevelopment actions to ecosystem services. None of the four sites chosen by the committee conducted such assessments. Both the DuPage County landfill and the Aztex Fibers cases appear to have qualitatively considered ecosystem services, with commendable results, illustrating how more formal assessments using ecological models and production functions could further improve site-specific remediation and redevelopment efforts.

Although it is now standard practice to develop a conceptual model in performing ecological risk assessments for contaminated sites, EPA's analyses of adverse impact have generally not linked to ecosystem services. The primary focus of the Agency's remediation efforts has been to control anthropogenic sources of chemical, biological, and physical stress that could lead to adverse impacts to human health or the environment. Developing conceptual models that incorporate the linkage between ecological endpoints and community-identified services would better guide both for the valuation of ecological protection and site remediation and redevelopment.

The Avtex Fiber case highlights what EPA could gain from developing the capacity to use conceptual models that integrate ecological effects and ecosystem services. A noteworthy feature of the Avtex Fiber process was the development of a master plan, which included some consideration of ecosystem services. For example, early concerns about contamination of groundwater and the discharge of toxic substances into the Shenandoah

River focused attention on water quality. Aquatic basins constructed to contain contaminants on site were designed to restore important ecosystem services, including safe habitat for waterfowl, runoff control, and water purification services. In this regard, the plan implied but failed to quantify or document a rudimentary ecological production function.

The development of a conceptual model that incorporated ecosystem services would have systematically facilitated greater integration of ecosystem services into remedial design and future uses. Recreational and aesthetic services were clearly important considerations for many features of the plan. However, because no comprehensive ecological model identifying ecosystem services apparently guided redevelopment at the site, it is unclear whether the particular pattern of restored forests and wetlands, recreation areas, and industrial parks produced the most valuable protection for ecosystem services. Different siting and design of soccer fields, for example, might have provided the same recreational value while achieving greater wildlife habitat, water quality, or aesthetic values for visitors, nearby residents, or both. The master plan's declared green focus for the industrial park implied that ecological concerns were important in the selection of industrial tenants and in the siting and design of facilities, but no ecological model for achieving this goal, or monitoring progress toward it, was presented. This omission leaves open the prospect that future industrial, recreational, and tourist developments and uses at the Avtex site might simply substitute one set of damages to ecosystems and ecosystem services for another.

6.3.3.4 Predicting effects on relevant ecosystem services

As discussed in chapter 3, development of a conceptual model should be followed with predictive analyses of the effects of EPA's actions on ecosystem services. Expanding ecological risk assessments to include assessments of the services that matter to people may present technical challenges, given that current focus of ecological risk assessments on toxicological data for a limited range of species and for toxic responses from individuals in those species. Such data will rarely link well to the ecosystem services that matter to a particular site-specific decision.

The Agency will need to develop its capacity to adapt and apply models that incorporate ecological production functions. These models are the real bridge between risk estimates and subsequent injury or damage projections and provide a major piece of the puzzle to quantify and value the impacts of chemical exposures under different remedial and restoration alternatives.

Incorporating ecological production functions into EPA's risk assessments will be important not only for EPA decisions on site remediation and redevelopment but also for natural resource damage assessments (NRDAs). Although trustee agencies, such as the National Oceanic and Atmospheric Administration and the U.S. Fish and Wildlife Service, are the regulatory leads for NRDAs, the ecological risk assessments and conceptual models produced by EPA in the remediation process are often the basis for damage assessment. If EPA could effectively conduct assessments that use ecological production functions to predict impacts on ecosystem services, those assessments would enhance the ability of resource trustees to appropriately assess injury, define restoration goals, and calculate damages. Predictive ecological production functions can play a critical role in such assessments.

The Leviathan Mine example illustrates how ecological impacts and damages are currently assessed. The Leviathan Mine natural resource damage assessment plan gives detailed information on concentrations of key pollutants (particularly heavy metals such as cadmium, zinc, copper, nickel, and arsenic) in surface water samples, groundwater samples, sediment samples, samples of fish tissues, and insect samples at various distances from the mine site. These concentration levels can be compared to concentration levels at reference sites (because historical information for the site itself is not available), toxicity data from the literature, and existing regulatory standards (e.g., water quality criteria or drinking water standards) to evaluate the potential for impact. Importantly, none of these comparisons is a direct demonstration of injury, which can only be measured through field observation and tests. EPA must rely on surrogates for estimating impact.

Once the impacts on water quality, sediments, etc., have been determined, ecological production functions could translate these impacts into predicted changes in ecosystem services. If recreational fishing is important, for example, EPA must estimate the site's impact on the fish population in the nearby water body. Such an analysis would require estimating the impacts of changes in water quality, streambed characteristics, bank sediments, and riparian vegetation on fish population, both directly and through impacts on the insects on which fish feed. If elevated levels of arsenic, copper, zinc, or cadmium exist in insects and fish tissue, EPA must also be able to use this information to predict an overall impact on the fish population.

EPA has already developed complex ecological risk assessment modeling tools (e.g., TRIM, EXAMS, and AQUATOX) to estimate the fate and effects of chemical stresses on the environment. In some cases, EPA has even coupled such exposure-effects models with ecological production models to estimate population level effects (CITATION).

In many cases, an ecological model that links ecological processes at a site to ecosystem services of interest to that site do not currently exist, although it might be possible to adapt models from the literature to fit local conditions with site-specific field data if the scale and ecological components of the site are similar (using the criteria for selecting among existing models described in section 3.3.1). In the absence of such a site-specific model, EPA might look to the scientific literature for guidance on how sensitive the insects and fish species are to these types of stressors. It could then ask expert ecologists to judge the likely magnitude of the impacts in the specific case. As for transfer of ecological benefits, however, scientists must take into account the differences between the reference site and the contaminated site and define and communicate the assumptions and limitations of transferring the information.

The Leviathan Mine Natural Resource Damage Assessment Plan also suggests studying the fish population downstream from the mine and comparing it to the population in a reference location, assuming a realistic reference site can be identified. More generally, it suggests comparing riparian vegetation, the composition of the benthic community, and wildlife populations near the mine and at an acceptable reference site. Such a comparison can help frame the types of damages resulting from the mining activity. Because reference sites and exposed sites may differ for a number of reasons not related to the contamination, such a comparison may not directly estimate the injury and will not take into consideration the impact of proposed remedial actions. Decisions about remediation and restoration require analysis of proposed actions, and it may not be reasonable to assume that remedial actions will be 100 percent effective in restoring relevant ecosystem services to their original level.

Comparative analyses of remedial actions using ecological production functions are needed and can be facilitated through comparative tools such as net environmental benefit analysis (Efroymson et. al., 2004). This analysis provides a framework for using valuation tools to compare alternative remedial strategies based on net impacts on ecological services.

Net Environmental Benefit Analysis

As described by Efoymson et al. (2003), “Net environmental benefits are the gains in environmental services or other ecological properties attained by remediation or ecological restoration, minus the environmental injuries caused by those actions. Net environmental benefit analysis (NEBA) is a methodology for comparing and ranking the net environmental benefit associated with multiple management alternatives. A NEBA for chemically contaminated sites typically involves the comparison of the following management alternatives: (1) leaving contamination in place; (2) physically, chemically, or biologically remediating the site through traditional means; (3) improving ecological value through onsite and offsite restoration alternatives that do not directly focus on removal of chemical contamination; or (4) a combination of those alternatives.

NEBA involves activities that are common to remedial alternatives analysis for state regulations and the Comprehensive Environmental Response, Compensation and Liability Act, response actions under the Oil Pollution Act, compensatory restoration actions under Natural Resource Damage Assessment, and proactive land management actions that do not occur in response to regulations, i.e., valuing ecological services or other ecological properties, assessing adverse impacts, and evaluating restoration options.”

Figure 6, taken from Efroymson et al. (2003), “depicts the high-level framework for NEBA. It includes a planning phase, characterization of reference state, net environmental benefit analysis of alternatives (including characterizations of exposure of effects, including recovery), comparison of NEBA results, and possible characterization of additional alternatives.” Dashed lines indicate optional processes; circles indicate processes outside the NEBA framework. Only ecological aspects of alternatives are included in this framework. “The figure also depicts the incorporation of cost considerations, the decision, and monitoring and efficacy assessment of the preferred alternative, although these processes are external to NEBA.”

Because NEBA is a framework, the needed resources, data inputs, and limitations are associated with whatever ecological models and valuation tools are selected.

Currently, NEBA is being applied at a local scale, although the size of some contaminated properties and their impacts can extend to the regional scale (e.g., impact of releases from a contaminated site to a watershed). NEBA should be highly adaptable to different levels of data, detail, scope, and complexity.

Figure 6: Framework for Net Environmental Benefit Analysis (from Efoymson et al., 2003)

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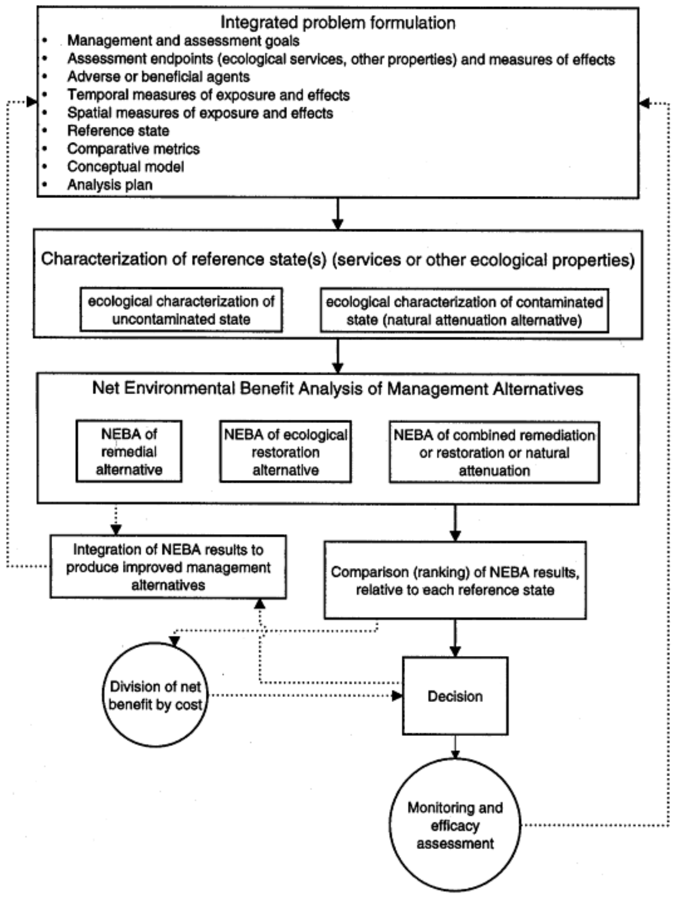

6.3.3.5 Defining, cataloging, and accounting for ecosystem services.

Accounting rules are needed to avoid double counting or undercounting the contributions to human well-being from ecosystem services. Ecosystems and their numerous components are linked in an intricate and complex network of biological, chemical, and energy flows. A focus on isolated impacts to individual organisms or components and their associated services can lead to double counting or undercounting contributions to human well-being generated by Agency actions.

For example, the listing of services (aquatic biota and habitat, riparian vegetation, terrestrial wildlife, recreational uses, and tribal uses) in the Leviathan Mine case does not seem to be useful for sorting out the different services to be valued. The listing fails to identify mutually exclusive services and presents a high likelihood of double counting. It also does not adequately distinguish between inputs and outputs. The significance of protecting habitat and riparian vegetation, for example, is not clearly addressed. Is it because society cares about the populations they support? Or is it because these populations are an input into something else of value, such as recreation? Consider insect populations. If society cares about the insects for their own sake, the insects generate unique existence value. If they are valued as a food source for fish and society cares about fish, there is value in the change in fish brought about by the change in insects. But in the latter case, insects should not be valued separately.

A better delineation of ecosystem services might involve identifying directly experienced, measurable, and spatially and temporally explicit services. For the Leviathan Mines example, such a list of ecosystem services might consist of the following:

- Water used by Washoe Tribe members and others for washing and drinking
- Non-consumptive uses of wildlife (e.g., viewing bald eagles and other species)
- Harvesting (hunting, fishing, and collecting fish) by Washoe tribal members
- Cultural, spiritual, and ceremonial values of land used by Washoe tribal members
- Flood control (e.g., reduction in flooding from snowmelt or runoff)
- Recreational services (e.g., fishing, hiking, and camping)

6.3.3.6 Expanding valuation methods.

The typical comparison of remedial strategies currently includes two tests: whether a remediation action controls risk to an acceptable level, and if so, whether it is cost effective. Under this scheme, if a proposed remediation action is adequate with regard to risk reduction, the least costly alternative is the obvious choice. Such an approach decouples remediation and redevelopment, delays the development process, and may not maximize what matters to key stakeholders or the public.

If remediation and redevelopment alternatives are to be compared based on their contributions to human well-being, EPA must be able to value the effect of each alternative on ecosystem services. As mentioned previously, NEBA offers a conceptual framework for comparing remedial and redevelopment alternatives on the basis of their net contributions to human well-being, whether monetized or non-monetized. Chapter 4 in turn describes a broad range of methods for valuing ecosystem services.

Habitat equivalency analysis (HEA) provides one approach for comparing contributions to human well-being associated with different remedial and redevelopment alternatives. HEA reports results in ecological units over time (e.g., discounted service acres years). The cost of creating or replacing those ecological units in monetary terms provides a replacement cost. Although these approaches do not provide direct measures of the value of the ecosystem services, they support a comparison of the services provided under different options. Alternatively, impacts of alternatives could be compared purely in ecological or biophysical terms through a method such as the conservation value method

EPA could also compare remediation and redevelopment alternatives using economic valuation. For example, EPA could use hedonic pricing studies to determine the economic impacts of the cleanup and redevelopment options on adjacent residential property values. New contingent valuation studies or studies using travel cost models could capture in monetary terms recreational or aesthetic values. Models might be used to compare expected gains to the local economy across the feasible set of redevelopment scenarios. Ecosystem benefit indicators, as discussed above, might also be used to evaluate the impacts of different remediation or redevelopment options.

If stakeholders are involved in testing remediation and redevelopment alternatives, EPA could use decision-aiding processes to assess their preferences for or weighting of alternatives. Formal social-psychological surveys of potential recreational users, visitors, and tourists could measure the relative preferences of these groups among remediation and redevelopment plans. Parallel economic or monetary assessments, perhaps using contingent valuation or travel cost methods, could extend and cross-validate survey results. Decision science methods could provide weights to facilitate analyses of tradeoffs among recreation, tourism, and industrial development at a site.

<u>6.3.3.7</u> <u>Communicating information about ecosystem.</u>

EPA should explicitly address ecosystem services in communications about site remediation and redevelopment. Managers will be able to better communicate the reasoning behind their selection of preferred options if analyses effectively integrate ecosystem services and their contributions to human well-being. A focus on the ecosystem services that matter to the public should lead to greater public understanding of options and acceptance of the proposed remediation and redevelopment plan. Performance measures defined in terms of contributions to well-being that the interested public understands and accepts as important should help facilitate communications about progress in the remediation and redevelopment process.

Because non-technical audiences often find scientific information unclear and difficult to understand, information about ecosystem services should often include visual communication techniques. For example, EPA might use perceptual representations (e.g., visualizations of revegetation options as viewed from adjacent homes and prominent tourist and recreation sites and passageways) to improve stakeholders' understanding of the implications of the various restoration and redevelopment alternatives under consideration. Consider the restoration plan for the Avtex site, which included replanting and encouraging re-growth of three different forest types on appropriate locations within the site. Accurate visualizations of the reforestation projects, including their expected growth over time, would have been useful for communicating the implications of alternative plans to stakeholders. Effectively developing and using such visualizations would require collaboration between forest ecologists and visualization experts (such as landscape architects). These collaborations could lead to the creation of accurate and realistic representations of how the different forests would look from significant viewpoints at different stages of the restoration program for each management alternative. Psychologists, communications experts, and other relevant social or decision scientists might create appropriate vehicles and contexts for presenting the visualizations to relevant audiences. Computer graphics experts might also be helpful. Further interdisciplinary collaboration would be required if the visualizations were to be accompanied by information about expected wildlife or other ecological effects associated with each visualized forest condition. While this example may seem to be an intricate, exhaustive process, many contaminated properties are under redevelopment for years (or decades in the case of Superfund projects). With proportional resource allocations, this level of effort may be appropriate.

<u>6.3.3.8</u> <u>Fostering information-sharing about ecological valuations at different sites.</u>

The committee recommends that EPA pursue the broad and rapid transfer of experience within the Agency of integrating valuation concepts and techniques into the remediation and redevelopment of contaminated sites. The Agency can build its capacity to utilize valuation to inform local decisions through a systematic exchange of information about site-specific valuations. The lessons learned from trial efforts, whether successes or failures, need to be shared widely across the Agency with the regions, program offices, and tool-builders in research organizations. The Agency can catalog and share such experiences in a number of ways, such as reports, databases, or computer-based networks of users sharing best practices. The Agency is in the best position to know how to build off existing information exchange systems. Regardless of how it is done, information should be shared broadly.

6.3.4. Summary of recommendations for valuation for site-specific decisions

Incorporation of ecological valuation into decisions about site remediation and redevelopment can help maximize the ecosystem services provided in the long run by such sites and the sites' contributions to local welfare. To effectively value the protection of ecological systems and services in this context, the committee recommends that EPA:

- Provide regional offices with the staff and resources needed to effectively incorporate ecological valuation into the remediation and redevelopment of contaminated sites.
- Define the ecosystem services and values important to the community and key stakeholders at the beginning of the remediation and redevelopment process.
- Involve the mix of interdisciplinary experts appropriate for valuation at different sites.
- Construct conceptual models that include ecosystem services.
- Adapt current ecological risk assessment practices to include ecological production functions to predict effects on relevant ecosystem services.
- Define ecosystem services carefully and develop a standard approach for cataloging and accounting for ecosystem services for site remediation and redevelopment.
- Expand the variety of methods the Agency uses to assess in monetary and non-monetary terms the services lost or gained from current conditions or proposed Agency action.
- Communicate information about ecosystem services in discussing opti.ons for remediation and redevelopment of sites

- Create formal systems and processes to foster information-sharing about ecological valuations at different sites.

7 Conclusion: Findings and recommendations

7.1 Findings of the committee

- Through an expanded and integrated approach to valuation, EPA can move toward greater recognition and consideration of the effects that its actions have on ecosystems and the services they provide.
    - This will allow EPA to improve environmental decision making at the national, regional, and site-specific levels and contribute to its's overall mission regarding ecosystem protection.
    - EPA can also use this expanded and integrated process to educate the public about the role of ecosystems and the value of ecosystem protection.
- Implementation of the integrated valuation process recommended by this report requires the Agency to predict the ecological responses to its actions, identify the relevant ecosystem services of importance to the public, and linkthe predicted ecological responses to the effect on those services. Estimating the responses of relevant ecosystem services to EPA actions is an essential part of valuation and must be done before the value of those responses can be assessed.
- EPA should allow for the use of a broader suite of methods than EPA has typically employed in the past for valuing ecosystems and their services. There are a variety of methods that could be used and the committee urges EPA to experiment with the use of alternative methods where legally permissible and scientifically appropriate. Some of the methods considered by the committee have been used extensively in specific decision contexts (e.g., the use of economic methods in the context of national rule making), but others are still relatively new and in the developmental stages (e.g., citizen juries). The methods also differ in a number of other important ways, including: the underlying assumptions; the types of values they seek to characterize; the empirical and analytical techniques used to apply them; their data needs (inputs) and the metrics they generate (outputs); and the extent to which they involve the public or stakeholders.
- Providing information to decision makers and the public about the level of uncertainty involved in ecosystem valuation efforts is critical for the informed evaluation of proposed policies and forcomparisons among alternative policy options. However, EPA

should not delay a necessary action simply because some uncertainty remains, because uncertainty will always remain.

- The success of ecological valuations depends on how EPA obtains information about public concerns during the valuation process and how it communicates the resulting ecological valuation information to decision makers and the public.
- The expanded approach to valuation proposed in this report can and should be applied to national rule making. This would entail challenges but would also present important opportunities for improvement. EPA can implement some of the committee's recommendations using the existing knowledge base. Others call for research to enhance EPA's future capacity to conduct high-quality, scientifically-based ecological valuation for national rule making.
- Regional-scale analysis holds great potential to inform decision makers and the public about the value of protecting ecosystems and services. Recent increases in publicly available, spatially-explicit data and a parallel improvement in the ability to display and analyze such data make it feasible to undertake comprehensive regional-scale studies of the value of protecting ecosystems and services. Municipal, county, regional, and state governments make many important decisions affecting ecosystems and the provision of ecosystem services at a regional scale, but local and state governments rarely have the technical capacity or the necessary resources to undertake regional-scale analyses of the value of ecosystems or services. Regional-scale partnerships between EPA regional offices, local and state governments, regional offices of other federal agencies, environmental non-governmental organizations, and private industry could aid both EPA and regional partners. Such partnerships offer great potential for improving the science and management for protecting ecosystems and enhancing the provision of ecosystem services.
- Incorporation of ecological valuation into decisions about site remediation and redevelopment can help maximize the ecosystem services provided in the long run by such sites and the sites' contributions to local welfare.

7.2 Recommendations

The tables below identify recommendations to implement the ideas embodied in the integrated and expanded approach. Table 4 identifies changes that can be made in the short term to in EPA's valuation processes. Table 5 identifies longer-term recommendations for research, information sharing, planning for resource needs for regional partnerships and site-specific decision making.

Table 4: Recommendations for implementation of an expanded and integrated approach to ecological valuation at EPA

	EPA should cover an expanded range of important ecological effects and human considerations using an integrated approach.			*Valuation analyses should involve, from the beginning and throughout, an interdisciplinary collaboration among physical/biological and social scientists, as well as input about public concerns.*
	Identify early in the process the ecological responses or contributions to human welfare that are likely to be of greatest importance to people and focus valuation efforts on these responses. Expand the range of ecological responses that are valued, recognizing the many sources of value.	*Predict ecological responses in value-relevant terms. Highlight the concept of ecosystem services and provide a mapping from responses in ecological systems to responses in services or ecosystem components that can be directly valued by the public.*	*Allow for the use of a wider range of possible valuation methods to provide information about multiple types of values. Evaluate using a scientifically-based set of criteria and apply methods in a manner consistent with their conceptual foundations.*	
Applicable to all EPA ecological valuations	• EPA should begin each valuation with a conceptual model of the relevant ecosystem and the ecosystem services that it generates. This model should serve as a road map to guide the valuation. • EPA should formalize a process for constructing the initial conceptual model, recognizing that the process must be iterative and respond to new information and multiple points of view. The conceptual model should reflect the ultimate goal of valuing the effect of EPA's decision on ecosystem services. • The model and its documentation should clearly describe the reasons for decisions about the spatial and temporal scales of the chosen ecological system, the process used to identify stressors associated with the proposed EPA action, and the methods to be used in estimating the ecological effects.	• EPA should identify and develop measures of ecosystem services that are relevant to and directly useful for valuation. This will require increased interaction within EPA between natural and social scientists. In identifying and evaluating services for any specific valuation effort, EPA should: count all things that matter once and only once; and describe them in terms that are meaningful and understandable to the public. • EPA should seek to use ecological production functions wherever practical to estimate how ecological responses (resulting from different policies or management decisions) will affect the provision of ecosystem services. • All ecological valuations conducted by EPA should be supported by ecological models and data sufficient to understand and estimate the likely ecological responses to major alternatives being considered by decision makers. • Building on recent efforts within the Agency and elsewhere, EPA should develop criteria or guidelines for model selection that reflect the specific modeling needs of ecological valuation, and EPA should apply these criteria in a consistent and transparent way. • EPA should support the use of methods such as meta-analysis that are designed to provide general information about ecological relationships that can be applied in ecological valuation.	• Beyond the use of economic valuation methods (the current mainstay of ecological valuation at EPA), EPA should experiment with the use of other valuation methods. EPA has not used these other methods for ecological valuation in any significant way in the past. Because some are still in the developmental stages, the committee believes that it would be wise for EPA to experiment with the use of these other methods in different valuation contexts. • EPA should only use methods that are scientifically-based and appropriate for the particular decision context at hand. • EPA should develop a set of criteria to use in evaluating methods to determine their suitability for use in specific decision contexts • Because EPA is likely to continue to rely heavily on benefits transfer in the ecological valuations that it conducts, EPA should explicitly identify relevant criteria related to societal preferences and the nature of the biophysical system of the cases being considered for economic benefits transfer to determine the appropriateness of the transfer. EPA analysts and those providing oversight of their work must take into account differences between study site and policy site to flag problematic transfers and clarify assumptions and limitations of the study-site results. • In assessing uncertainty, EPA should go beyond simple sensitivity analysis and make greater use of approaches, such as Monte Carlo analysis and expert elicitation that provide a more useful and appropriate characterization of uncertainty in complex contexts such as ecological valuation. • EPA should not relegate uncertainty analyses to appendices but should ensure that a summary of uncertainty is given as much prominence as the valuation estimate itself. EPA should also explain qualitatively any limitations in the uncertainty analysis.	• In constructing conceptual models, EPA should involve staff throughout the EPA, outside experts from the bio-physical and social sciences and seek information about relevant public concerns and needs. • In communicating ecological valuation information, EPA should follow basic guidelines for risk and technical communication. • EPA should evaluate its communication of ecological valuations to assess its effects and learn to improve practices. • Where feasible, EPA should communicate not only value information but also information about the nature, status, and changes to the ecological systems and services to which the value information applies. Visual tools such as mapped ecological information, photographs, graphs, and tables of ecological indicators can be very useful in conveying causal processes. • Where appropriate, EPA should use an iterative, interactive approach to communicating values. • EPA should consider the appropriate use of deliberative processes, in which analysts, stakeholders, decision makers, and/or other members of the public meet in facilitated interactions. Two particular types of deliberative processes that can be useful in ecological valuations are mediated modeling and constructed value processes.

	•		•	•
Additional recommendations for valuations for national rule making	•		• In the context of national rule making, EPA should conduct one or two model analyses (perhaps one prospective and one retrospective) of how the use of a wider range of methods might be applied. This experience could then guide EPA's valuation efforts as it conducts subsequent benefit assessments. • Once EPA has identified a list of potentially important ecological effects and associated services, it should categorize those effects according to the extent to which they can be quantified and monetized at the national level using economic valuation techniques (primarily benefits transfer). • To address site-specific variability in the impact of a rule, EPA's benefit assessments should include case studies for important ecosystem types, and aggregate across these case studies if information about the distribution of ecosystem types is available. This bottom-up approach would establish separate estimates for each locality or region and then add them together to obtain a national estimate. • For ecosystem services for which the benefits are primarily local, EPA can use benefits transfer using prior valuations at the local level, provided the benefits transfer is conducted appropriately. However, for services with broader benefits, EPA should use benefit transfers that draw from studies with broad geographical coverage (in terms of both the changes that are valued and the population whose values are assessed). • EPA should not compromise the quality of a benefits assessment by inappropriately applying benefits transfer to effects that cannot be monetized at the national level using scientifically sound benefits transfer, nor simply list such effects in a category of "non-monetized benefits." EPA instead should seek to provide a scientific basis for the belief that these benefits are important. EPA could include quantifications of biophysical impacts using ecological models, metrics that provide information about the likely magnitude of the associated benefits (and hence are indicators of benefits), and detailed qualitative discussions based on existing scientific literature. • EPA should also consider estimating non-economic values for at least some ecosystem services where appropriate. Although these values do not properly fit within a formal economic benefit-cost analysis, they can provide important additional information to support decision making. When such value estimates are included in RIAs, the RIA should discuss both the valuation method and the results in a separate section. • To ensure that benefit assessments do not inappropriately focus only on those impacts that have been monetized, EPA should report non-monetized ecological effects in appropriate units in conjunction with monetized economic benefits. • Aggregated monetized economic benefits should be labeled as "total monetized economic benefits" rather than "total benefits." • EPA should include a separate chapter on uncertainty characterization in each economic benefit assessment and RIA.	•

Additional recommendations for valuations in regional partnership			• EPA should experiment with a range of valuation methods because regional decision contexts do not prescribe use of economic methods. • Some EPA staff have expressed a desire to be provided a value for an ecosystem component or service that they can then apply to their region (e.g., a constant value per acre of wetland or wildlife habitat). Such short cuts to the valuation process are uninformed by local social, economic, and ecological conditions and can generate results that are not meaningful. This approach to valuation should be avoided.	• Regional-scale analysis teams should be formed to undertake valuation studies. Teams composed of social scientists and natural scientists should participate from the beginning of the project to design and implement plans for stakeholder involvement, ecological production functions, and valuation. • All regional-scale analyses of the value of ecosystems and services should involve stakeholders at an early stage to ensure that subsequent ecological, economic, and social analyses are directed toward those ecosystem components and services deemed of greatest importance by affected communities. The process should proceed bottom-up, as opposed to top-down. Rather than asserting what is valuable, EPA must understand what various communities view as being valuable. An important question that should be addressed by EPA regional offices is how to develop effective stakeholder involvement at broader regional scales. • Regional staff should form partnerships with local and state agencies or local groups where doing so advances the mission of EPA directly or indirectly by promoting the ability of partner organizations to value the effect of their actions on ecosystems and services and to protect environmental quality.
Additional recommendations for valuations for site-specific decision making	• .	• Adapt current ecological risk assessment practices used in site remediation to include ecological production functions to predict effects on relevant ecosystem services • Develop a standard approach for cataloging and accounting for ecosystem services for site remediation and redevelopment.	• EPA should experiment with a range of valuation methods because regional decision contexts do not prescribe use of economic methods. •	• Define the ecosystem services and values important to the community and key stakeholders at the beginning of the remediation and redevelopment process. • Involve the mix of interdisciplinary experts appropriate for valuation at different sites. • Communicate information about ecosystem services in discussing options for remediation and redevelopment of sites.

Table 5: Recommendations for research, information sharing, and planning for resource needs for regional partnerships and site-specific decision making

Research	Information Sharing	Planning for Resource Needs for Regional Partnerships and Site-specific Decision Making
• Because of the complexity of developing and using complete ecological production functions, EPA should continue and accelerate research to develop key indicators for use in ecological valuation. Such indicators should meet ecological and social science criteria for effectively simplifying and synthesizing underlying complexity and be associated with an effective monitoring and reporting program. • EPA should encourage strong regional initiatives to develop information needed for valuations. • EPA should also promote efforts to develop data that can be used to parameterize ecological models for site-specific analysis and case studies or transferred or scaled to other contexts. • EPA should carefully plan and actively pursue research to generate ecological production functions for valuation, including STAR research on ecological services and support for modeling and methods development. EPA should make the development of ecological models that can be used in valuation efforts one of its research priorities. • EPA should foster interaction between natural scientists and social scientists in identifying relevant ecosystem services and developing and implementing processes for measuring and valuing them. As part of this effort, EPA should more closely link its research programs on evaluating ecosystem services and valuing ecosystem services. • Because of the importance of benefits transfer to EPA, EPA should conduct or fund original research on valuation that is designed to be used in subsequent benefit transfers. • EPA should invest in research designed to reduce the uncertainties associated with ecological valuation through data collection, improvements in measurement, theory building, and theory validation. Assessments of the magnitude and sources of uncertainty can help to establish research priorities and inform judgments about whether policy changes should be delayed until research reduces the degree of uncertainty associated with possible changes. • EPA should support the development of quantitative ecosystem models and baseline data on ecological stressors and ecosystem service flows that can support national-level predictions of the consequences of changes in ecological stressors on the production of ecosystem services. • EPA should support the development of methodological and original valuation studies designed to enhance national-level ecological benefits transfer, including national surveys relating to ecosystem services with broad (rather than localized) benefits that can generate value estimates for use in multiple rule making contexts. • Future calls by EPA for extramural research should incorporate the research needs of regional offices for systematic valuation studies. Doing so will maximize the probability that future grant funding will be useful for EPA's regional offices.	• EPA should work with other agencies and with scientific organizations such as the National Science Foundation to encourage the sharing of ecological data and the development of more consistent ecological measures that are useful for valuation purposes. • EPA should support efforts to develop Web-based databases of existing valuation studies across a range of ecosystem services, with careful descriptions of the characteristics and assumptions of each, to assist in increasing the likelihood that the most comparable existing valuations will be identified. • EPA should support the development of national-level databases to support valuation, including data on the joint distribution of ecosystem and population characteristics that are important determinants of ecological benefits. • Regional staffs need to be able to learn effectively from valuation efforts being undertaken by other regional offices. EPA regional offices should document valuation efforts and share them with other regional offices, with EPA's National Center for Environmental Economics, and with EPA's Office of Research and Development. Each regional office should also publish their studies. • Create formal systems and processes to foster information sharing about ecological valuations at different sites.	• EPA regional staff should be given adequate resources to develop expertise necessary to undertake comprehensive and systematic studies of the value of protecting ecosystems and services. Increased expertise is needed in: o Economics and social science: Expertise is very limited at the regional level to undertake economic or other social assessments of value. A pressing need exists to increase expertise in this area among regional offices. o Stakeholder involvement processes. o Ecology: Regional staffs have greater expertise in ecology than in stakeholder involvement, economics or other social sciences, but doing systematic valuations of ecosystem services will require additional ecological staff. Of greatest utility would be ecologists with expertise in assessing impacts on ecosystem services through ecological production functions to evaluate alternative management options. • Provide regional offices with the staff and resources needed to effectively incorporate ecological valuation into the remediation and redevelopment of contaminated sites.

Appendix A: Survey issues for ecological valuation: current best practices and recommendations for research

Survey methods support many of the approaches for eliciting and measuring information about values discussed in the C-VPESS report. Although scientific and technical issues concerning survey design and administration can affect some aspects of ecological valuation, they are distinct from the science and value assessment issues that are the main focus of the C-VPESS report.

The C-VPESS recognizes, however, that issues related to survey methods are important to some methods of ecological valuation and learned they were of particular concern to EPA representatives participating in the SAB's workshop "Science for Valuation of EPA's Ecological Protection Decisions and Programs," held December 13-15, 2005. After that workshop, the committee requested that this appendix be commissioned to supplement the main body of the committee's report. This appendix provides an introduction for EPA staff to questions posed to the C-VPESS pertaining to survey use for ecological valuation. It provides an overview of how recent research and evolving practice relating to those questions might assist the Agency.

<u>Defining survey research</u>

Survey research entails collecting data via a questionnaire from a sample of elements (e.g., individuals or households) systematically drawn from a defined population (see Babbie, 1990; Fowler, 1988; Frey, 1989; Lavrakas, 1993; Weisberg, et al., 1996).[29] Conducting a survey involves: drawing a sample from a population; collecting data from the elements in that sample; and then analyzing the data generated. Survey research is a well-established and respected scientific approach to measuring the behavior, attitudes, and beliefs (and much more) of populations of individuals.[39] Surveys are usually done:

- To document the prevalence of some characteristic in a population
- To compare the prevalence of some characteristic across subgroups in a population
- To document causal processes that produce behaviors, beliefs, or attitudes

Because scientific surveys involve probability sampling, their results can be used to estimate population parameters. This appendix addresses issues of survey methodology that cut across many different applications including: monetary valuations (e.g., CVM); measures of preference, importance, or acceptability; and determinations of the assumptions, beliefs, and motives that might underlie these expression of value.

<u>Designs of surveys</u>

Surveys can take on a variety of designs to address various types of research questions. For example, cross-sectional surveys are useful for measuring a variable at a given point in time, whereas repeated cross-section surveys are more useful for observing change over time in a population. Panel surveys are more useful for examining change over time in a sample of respondents, and surveys that implement experiments may be more useful for establishing causality. Many types of information can be derived from the data from each of these types of surveys.

Cross-sectional surveys involve the collection of data at a single point in time from a sample drawn systematically from a population, and are often used to document the prevalence of particular characteristics in a population. Cross-sectional surveys allow researchers to assess relations between variables and differences between subgroups of respondents. Data from cross-sectional surveys can also be used to provide evidence about causal hypotheses using statistical techniques (e.g., two-stage least squares regression or path analysis; Baron and Kenny, 1986; James and Singh, 1978; Kenny, 1979) by identifying moderators of relations between variables (e.g., Krosnick, 1988) or by studying the impact of an event occurring in the middle of data collection (e.g., Krosnick and Kinder, 1990).

Repeated cross-sectional surveys involve collecting data from independent samples drawn from the same population at two or more points in time. Such data can be used to provide evidence about causality by gauging whether changes in an outcome variable parallel changes in a purported cause of it. Repeated cross-sectional surveys can also be used to study the impact of social events that occurred between the surveys (e.g., Weisberg, et al., 1995).

Panel surveys involve collecting data from the same sample of respondents at two or more points in time and can be used to gauge the stability of a construct over time and identify the determinants of stability (e.g., Krosnick, 1988; Krosnick and Alwin, 1989). Panel surveys can also be used to test causal hypotheses. This can be done by examining whether changes over time in a purported case correspond to changes in an outcome variable, by assessing whether changes over time in the outcome variable can be predicted by prior levels of the purported cause, or by testing the effects of events that occur between waves (see, e.g., Blalock, 1985; Kessler and Greenberg, 1981, on the methods; see Rahn, et al., 1994, for an example).

There are a number of challenges associated with conducting panel surveys, including respondent attrition (or "panel mortality"). This occurs when some of the people who provide data during the first wave of interviewing cannot or choose not to participate in subsequent waves. Attrition reduces a panel's effective sample size and it is particularly undesirable if a non-random subset of respondents drop out. However, the literature suggests that panel attrition minimally affects sample composition (Becketti et al., 1988; Clinton, 2001; Falaris and Peters, 1998; Fitzgerald et al., 1998a; 1998b; Price and Zaller, 1993; Rahn et al., 1994; Traugott, 1990; Zabel, 1998; Zagorsky and Rhoton, 1999; and Ziliak and Kniesner, 1998 ; although see Groves et al., 2000; Lubin et al., 1962; and Sobel, 1959).

A second methodological issue in panel research is panel conditioning, or the possibility that interviewing people repeatedly may change them and thereby make the sample less representative of the larger population to which investigators wish to generalize. But again, the literature is, for the most part, reassuring. A number of studies have found either no evidence of panel conditioning effects or very small effects (Clinton, 2001; Cordell and Rahmel, 1962; Himmelfarb and Norris, 1987; Sobol, 1959; Willson and Putnam, 1982). Particularly if repeated interviews with panel members touch on a wide variety of topics, each wave may blend in with memories of prior waves via what psychologists call "retroactive interference," thus minimizing the likelihood of stimulated interest in any one topic. However, some evidence suggests that interviewing people on a particular topic may cause them to become more cognitively engaged in that topic (Bridge et al., 1977; Granberg and Holmberg, 1992; Kraut and McConahay, 1973; Willson and Putnam, 1982; Yalch, 1976; although see Mann, 2005). Other studies have documented that asking people just one question about their behavioral intentions can affect their subsequent behavior (see, e.g., Greenwald et al., 1987; Gregory, et al., 1982).

Interestingly, membership in a long-term panel survey may actually be beneficial to the quality of data collected because of "practice effects" (e.g., Chang and Krosnick, 2001). The more a person performs any task, the more facile and effective he or she becomes at doing so. In our case, the tasks of interest include question interpretation, introspection, recollection, information integration, and verbal reporting (see Tourangeau, et al., 2000).

Mixed designs are used when researchers can capitalize on the strengths of more than one of these designs by incorporating elements of two or more into a single investigation. For example, a researcher interested in conducting a two-wave panel survey but concerned about conditioning effects, could concurrently administer the second-wave questionnaire to both the panel and to an independent cross-sectional sample drawn from the same population. Differences between the data collected from these two second-wave samples would suggest that carry-over effects were a problem in the panel survey.

Experiments can also be implemented in surveys to test causal hypotheses. If respondents are randomly assigned to "treatment" and "control" groups that are asked different versions of a question or question sequence, differences between the two groups can then be attributed to the treatment.

Elements of a well-defined survey

Sampling

When designing a survey's sample, the sampling frame (the complete list of elements in the population to which one wishes to generalize findings) must be defined, and the subset of elements (the individual unit about which information is sought) in the population to be interviewed must be selected. These decisions have important implications for the results of the survey because they may impact both coverage and sampling error (see, e.g., Laumann, et al., 1994). Coverage error occurs when the sampling frame excludes some portion of the population. For example, telephone surveys usually exclude households without telephones. Sampling error is the discrepancy between the sample data and the true population values that is due to random differences between the sample and the sampling frame.

There are two broad classes of sampling methods: nonprobability and probability sampling. Nonprobability sampling refers to selection procedures such as haphazard sampling, purposive sampling, snowball sampling, and quota sampling in which elements are not randomly selected from the population or in which some elements have zero or unknown probabilities of selection. Probability sampling refers to selection procedures such as simple random sampling, systematic sampling, stratified sampling, or cluster sampling in which elements are randomly selected from the sampling frame and each element has an independent, known, nonzero chance of being selected. Unlike nonprobability sampling, probability sampling allows researchers to be confident that a selected sample is representative of the population from which it was drawn and to generalize beyond the specific elements included in the sample. Probability sampling also allows researchers to estimate sampling error, or the magnitude of uncertainty regarding obtained parameter estimates. Therefore, the best survey designs (and virtually all scientific surveys) use some form of probability sampling.

Sampling error can be minimized by surveying large samples. However, the relation between sampling error and sample size is not linear. A moderate sample size reduces sampling error substantially in comparison with a small sample size, but further increases in sample size produce smaller and smaller decrements in sampling error. Thus, researchers should recognize that beyond a moderate sample size, the funds necessary to produce a large sample might be better spent reducing other types of error.

Questionnaire design

High-quality, scientific surveys typically provide respondents with several key pieces of information when introducing the survey, whether through an introductory letter, an e-mail, or an introduction from a telephone or face-to-face interviewer. This information protects respondents' rights, helping to ensure that the survey is being conducted ethically. It may also help to increase the perceived validity of the survey and, as a result, respondent participation. The introduction usually includes information about the sponsor of the survey, a brief description of the survey topic, and how the data from the survey will be used. It should also include a reassurance to respondents that their survey responses will be kept confidential and descibe any other measures in place to protect respondents. Finally, the burden being placed on respondents and any risks to the respondent should also be described. This information allows respondents to give informed consent. That is, knowing this information, respondents can make an informed choice about whether or not to participate in the survey. However, it is important to also keep this introduction as short as possible, because longer introductions place a greater burden on respondents and may reduce survey participation.

Survey questions. All surveys include questions, and a series of decisions must be made to achieve optimal designs of those questions. First, a researcher must decide if each question will be open- or closed-ended. For closed-ended questions, a researcher interested in obtaining rank orders of objects must decide whether to ask respondents to report those rank orders directly or to rate each object separately. If respondents are asked to rate objects, the researcher must decide how many points to put on the rating scale, how to label the scale points, the order in which response options will be offered, and whether respondents should be explicitly offered the option to say they "don't know" or have no opinion. Once the questions are written, the researcher must determine the order in which they will be administered. Researchers must also decide how to optimize measurement on sensitive topics, where social desirability response bias may lead respondents to intentionally misreport answers in order to appear more respectable or admirable. There is a large body of relevant scientific studies about the questionnaire design that, when taken together, clearly suggest strategies for designing questionnaires to maximize the quality of measurement. Although a description of the entire literature is beyond the scope of this review, we provide a few examples here about survey questions using rating scales to provide a flavor of what this literature has to offer.

When designing a rating scale, one must begin by specifying the number of points on the scale (for a review of relevant literature, see Krosnick and Fabrigar, forthcoming). For bipolar scales that have a neutral point in the middle (e.g., running from positive to negative), reliability and validity are highest for about seven points (e.g., Matell and Jacoby, 1971). In contrast, the reliability and validity of unipolar scales, with a zero point at one end (e.g., from no importance to very high importance), seem to be optimized for somewhat shorter scales, approximately five points long (e.g., Wikman and Warneryd, 1990). [40]

A number of studies show that data quality is better when all points on a rating scale are labeled with words, rather than just some of the points (e.g., Krosnick and Berent, 1993). Researchers should try to select labels that have meanings that divide up the continuum into approximately equal units (e.g., Klockars and Yamagishi, 1988). For example, "very good, good, or poor" is a poor choice, because the meaning of "good" is much closer to the meaning of "very good" than it is to the meaning of "poor" (Myers and Warner, 1968).[41]

Researchers also must decide how to order the response alternatives, and people's answers to rating scale questions are sometimes influenced by this order. With most rating-scale questions, respondents are likely to begin to formulate a judgment after reading the question stem. For example, the question, "How effective do you think the cleanup plan will be?" would induce respondents to begin to generate an assessment of effectiveness. As respondents read or listen to the answer choices presented, some may settle for the first acceptable response option they encounter rather than considering all the response options and selecting the answer choice that best reflects their judgment. This results in primacy effects in ratings, which have been observed in many studies (e.g., Belson, 1966; Carp, 1974; Chan, 1991; Matthews, 1929). To minimize bias, it is usually best to rotate the order of response choices across respondents and to statistically control for that rotation when analyzing the data.[42]

<u>Pretesting</u>. Even the most carefully designed questionnaires sometimes include items that respondents find ambiguous or difficult to comprehend, or items that respondents understand but interpret differently than the researcher intended. Researchers can conduct pretests of a draft questionnaire to identify these kinds of problems. Pretesting methods include conventional pretesting, in which interviewers conduct a series of interviews and report any problems with question interpretation or comprehension (see, e.g., Bischoping, 1989; Nelson, 1985); behavior coding, in which a researcher notes the occurrence of verbal events during the interview that might indicate problems with a question (e.g., Cannell et al., 1981); and cognitive interviewing, in which a questionnaire is administered to individuals who either "think aloud" while answering or answer questions about the process by which they formulated their responses (e.g., Forsyth and Lessler, 1991). Each of these methods has advantages and disadvantages. When resources are available, researchers can use multiple methods to pretest questionnaires because different methods identify different types of problems (see Presser et al., 2004).

<u>Mode of data collection</u>

Survey data can be collected in one of four primary modes: mail, telephone, face-to-face, and Internet. Interviewers administer telephone and face-to-face surveys, whereas mail and Internet surveys involve self-administered questionnaires. Mode choice can produce notable differences in survey findings. So mode choice must be made carefully in light of each project's goals, budget, and schedule. Each survey mode has advantages and disadvantages. When choosing a mode for a particular survey, researchers must consider cost, characteristics of the population, sampling strategy, desired response rate, question format, question content, questionnaire length, length of the data-collection period, availability of facilities, the purpose of the research, and the resources available to implement it.

Aspects of the population, including literacy, telephone coverage, and familiarity with and access to computers, are important in the decision about mode. Literacy is necessary for self-administered questionnaires. Broad telephone coverage of the population is necessary when conducting a telephone survey. Internet access and familiarity with computers is important for an Internet survey.

Coverage error is minimized in face-to-face household surveys, but is larger in random digit dial (RDD) telephone household surveys, because they exclude respondents without telephones and those with only cell phones. Coverage error for mail and Internet surveys depends upon the sampling strategy used and with list samples, the quality of the list that is used as the initial sample frame.

Although probability sampling is possible in all modes, mode affects the ease with which it can be implemented. Telephone and face-to-face surveys routinely use probability household sampling strategies, but mail and other self-administered surveys are more commonly used when a list of the entire population is available. In some Internet surveys, nonprobability sampling methods are used (e.g., inviting individuals to opt in through websites). This does not yield results that can be generalized to the population of interest (Malhotra and Krosnick, in press). Some researchers, however, have implemented probability sampling to recruit respondents to complete questionnaires weekly via the Internet and provided Internet access to respondents who do not have it.

Mode also influences the response rates achieved in a survey, with face-to-face surveys typically achieving the highest response rates. Telephone surveys achieve somewhat lower response rates, and self-administered mail surveys achieve low response rates unless a sequence of multiple contacts are implemented at considerable cost and with considerable implementation time (see Dillman, 2006).

The types of information and questions researchers wish to present may also influence the choice of mode. If a survey includes open-ended questions, face-to-face or telephone interviewing is preferable because interviewers can probe incomplete or ambiguous respondent answers. If complex information will be presented as part of the survey, face-to-face interviewing or Internet questionnaires allow the presentation of both oral and visual information. If the researcher needs to ask questions about sensitive topics, self-administered questionnaires and computers provide respondents with a greater sense of privacy and therefore elicit more candid responses than interviewer-administered surveys (e.g., Bishop and Fisher, 1995; Cheng, 1988; Wiseman, 1972). Face-to-face interviewing is likely to elicit more honest answers than telephone interviewing because face-to-face interviewers can develop better rapport with respondents and more easily implement private response methods.

Face-to-face data collection permits interviews of an hour or more, whereas telephone interviews usually last no more than 30 minutes. With self-administered questionnaires, response rates typically decline as questionnaire length increases, so they are generally kept even shorter.

Telephone and Internet surveys can be completed in very short field periods, often within a matter of days (though at the cost of lower response rates). In contrast, mail surveys require significant amounts of time, and follow-up mailings to increase response rates further increase the overall turnaround time. Similarly, face-to-face interview surveys typically require a substantial length of time in the field.

Face-to-face interviews are usually considerably more expensive than telephone interviews, which are usually about as expensive as self-administered questionnaire surveys of comparable size using methods necessary to achieve high response rates. The cost of Internet data collection from a probability sample is about equivalent to that of telephone RDD interviewing.

These differences between modes also contribute to differences in data quality. Face-to-face surveys have the highest response rates, are the most flexible in terms of interview length and presentation of complex information, and acquire more accurate reports than do telephone surveys (Holbrook et al., 2003). Internet surveys allow presentation of complex information, and reporting accuracy appears to be higher in Internet surveys than in telephone surveys (Chang and Krosnick, 2001). Although response rates from Internet surveys based on initial RDD telephone samples are quite low and have similar coverage error to telephone surveys, such difficulties may be reduced by recruiting probability samples of respondents face-to-face in their homes.

Assessing survey accuracy

In order to optimize survey design or to evaluate the quality of data from a particular survey, the accuracy (or conversely error) in survey data needs to be assessed. If optimal procedures are implemented a high level of accuracy can be achieved, but departures from such procedures can compromise the accuracy of a survey's findings. Usually, researchers have a fixed budget and must decide how to allocate those funds in order to maximize the quality of their data. The "total survey error" approach enables researchers to consider survey design issues within a cost-benefit framework that allows them to maximize data quality within budget constraints (cf. Dillman, 1978; Fowler, 1988; Groves, 1989; Hansen and Madow, 1953; Lavrakas, 1993).

The total survey error perspective recognizes that the goal of survey research is to accurately measure particular constructs in a sample of people who represent the population of interest. In any given survey, the overall deviation from the ideal is the cumulative result of several sources of survey error. The total survey error perspective disaggregates overall error into four components: coverage error, sampling error, nonresponse error, and measurement error. Coverage and sampling error are described above. Nonresponse error is the bias that can result when data are not collected from all members of a sample. Measurement error refers to all distortions in the assessment of the construct of interest, including systematic biases and random variance that can be brought about by respondents' own behavior (e.g., misreporting true attitudes), interviewer behavior (e.g., misrecording responses), and the questionnaire (e.g., ambiguous or confusing question wording).

Nonresponse occurs when data are not collected from all of the eligible sample elements. Nonresponse occurs either because sampled elements are not contacted (e.g., no one is available at the time of contacted) or because members of sampled households decline to participate. The response rate for a survey is the proportion of eligible sample elements from whom data were collected and is almost always less than 100%. Lower response rate increase the risk that the sample is not representative of the population.

To maximize response rates, researchers implement various procedures. For example, the field period during which potential respondents are contacted can be lengthened (e.g., Groves and Lyberg 1988; Keeter et al. 2000), the number of times an interviewer tries to contact a household member can be increased (Merkle, et al., 1993; O'Neil, 1979), financial incentives can be offered for participation (e.g., Singer et al., 1999; Singer, et al., 2000), advance letters can be mailed to households to inform residents about the survey (e.g., Camburn et al., 1995; Link and Mokdad 2005), and the questionnaire can be kept as short as possible (e.g., Collins et al. 1988). All of these strategies have been found to increase response rates in at least some studies in which these factors were considered one by one. However, some strategies, such as sending advance letters or leaving messages on answering machines of potential respondents, may not always be successful because they give advance notice that interviewers will try to contact respondents, and respondents may use this knowledge to avoid being interviewed.

Low response rates increase only the potential for nonresponse error, because nonresponse error is a function of two variables: the response rate and the size of the difference between respondents and nonrespondents. If respondents and nonrespondents do not differ substantially, response rates will be unrelated to nonresponse bias. That is, it is possible to conduct a survey with a response rate of 20 percent and end up with data that describe the population quite accurately.

A number of publications using a variety of methods have shown that as long as a representative sample is scientifically drawn from the population and professional efforts are made to collect data from all potential respondents, variation in response rates (between 20 percent and 65 percent) does not substantially increase the accuracy of the survey's results (Curtin et al., 2000; Holbrook et al., in press; Keeter et al., 2000). Furthermore, although many surveys manifest substantial nonresponse error, there is little evidence that the observed amount of nonresponse error is related to the response rate for the survey.

Measurement error includes any distortion or discrepancy between the theoretical construct of interest and the concrete measurement of that construct. One method for assessing measurement error is to compare responses to a survey to a known standard to assess their validity. For example, reports of whether or not a respondent voted in an election can be compared to public records of voting, or reports of drug use can be compared to the results of drug tests performed on hair, urine, or saliva samples. However, surveys often measure constructs for which there are no available standards. In these cases, the reliability or predictive validity of survey measures is often used to judge the quality of the measurement. One method for comparing different survey questions or question orders is to use split-ballot experiments. In these, half the respondents are randomly assigned to receive one form of a questionnaire (using one question wording or order) and the other half are randomly assigned to receive a different form of the questionnaire (using a second question wording or order). One or more of the approaches described above (e.g., comparison to a known standard, reliability, or predictive validity) can then be used to compare the reliability and/or validity of responses across questionnaire forms to determine if one question wording or order is better.

The total survey error perspective advocates explicitly taking into consideration each of these four sources of error and making decisions about the allocation of resources with the goal of reducing the total error. Many steps that do not cost real dollars can be taken to reduce error, but other steps to reduce error do cost money, and the more money spent on reducing one type of error, the less money is available to reduce other types of error. Researchers should make such tradeoffs explicitly, recognizing the opportunity costs they pay when making a particular move to maximize quality in a particular way, selecting approaches likely to yield the greatest overall impact.

<u>Challenges in using surveys for ecosystem protection valuation</u>

Introduction. One application of the survey method is in assessing the value of ecosystems and services. A variety of techniques have been developed to assess the monetary value of ecosystems, and these values can be used as input to required benefit-cost analyses by EPA in the policy-making process. When monetary values are not required, are too difficult to attain, or are deemed ethically or otherwise inappropriate to the problem at hand, surveys can be used effectively to determine quantitative measures of preference, importance, or acceptance of alternative policies, actions, and outcomes. When surveys are used for valuation, many respondents are asked to rank, rate, or place a monetary value on a change in the condition of ecosystems or services with which they may not be familiar prior to the survey. However, this does not mean that respondents lack a value for the ecosystem in question. Respondents' experiences have cumulated into beliefs and attitudes that influence their orientation toward objects or situations they encounter. Therefore, an important component of valuation survey design is to describe the ecosystem as fully as possible so that respondents can use these beliefs and attitudes to determine its value. Doing so helps to maximize the extent to which the values that respondents report validly reflect these underlying beliefs and opinions. This means that valuation surveys will be different from most other surveys because they must devote a considerable amount of time to educating the respondent about the ecosystem in question. This may require respondents to listen to or read relatively long passages of text and perhaps to observe visual presentations of nonverbal information as well, such as charts, maps, drawings, or photographs.

Conveying a large amount of information. The survey needs to provide all the information that respondents want in order to make the judgments being asked of them, and present that information in a way that is understandable to all respondents. To achieve these goals, researchers can conduct research with pretest respondents to assess what information they want to know and their understanding and interpretation of information presented to them. These procedures can be used iteratively to refine the presentation to enhance understanding and sufficiency of the information set.

In order to present a sizable set of information to respondents, a variety of techniques can be implemented to maximize comprehension. The principles of optimal design can be used to construct graphical displays of information (e.g., Kosslyn, 1994; Tufte, 2001). A single visual display can convey a great deal of information if an interviewer can explain or the respondent has the opportunity to study the display. Information can also be presented in narrative form – for example, by telling respondents about: the state of an ecosystem as it used to exist 50 years ago; changes that have occurred to the ecosystem in the intervening years; the causes of those changes; what could be done to reverse those changes; and how this could be implemented. Instead of a long lecture, a questionnaire can maintain respondent engagement by presenting information in small chunks, separated by questions that allowing them to react briefly to the information they've been given (e.g., "Had you ever heard of the Golden River before today?"). As the story progresses, respondents can also be asked periodically to verbalize any information that they would like to have, to allow them to express their cognitive responses to the presentation.

The choice of survey mode also has an impact on the presentation of information about an ecosystem. Face-to-face interviewing is optimal because it allows any type of visual display and interviewers can create a strong sense of interpersonal connection with respondents. Telephone interviewing permits a similar connection, though probably less strongly, and visual displays are usually not possible. Computer administration of a questionnaire can include static and dynamic presentation of visual and verbal information, and questions can be interspersed with this information, but it may not be possible to create the strong sense of connection between the respondent and the researcher. Self-administered paper and pencil questionnaires allow only visual presentation of information and do not allow information to be presented in small chunks (because respondents can look ahead in the questionnaire). A large volume of information presented densely on a many pages may be intimidating or dispiriting, thus minimizing respondent motivation and provoking superficial processing of the information. For this reason, the self-administered mode may be the least desirable. For all modes, it is important to pretest the final instrument to be sure it is working as intended.

Communicating uncertainty. Because of the uncertainty inherent in estimating the effect a policy might have on an ecosystem or service (see section 5.1.4), researchers using surveys for valuation may not only want to convey large amounts of information to respondents, but they may also want to convey their level of certainty or uncertainty about that information. Such uncertainty could be conveyed to respondents in a number of ways, including: providing ranges or confidence intervals for the information provided (e.g., the estimated cost of maintaining the ecosystem is between 1 million and 3.3 million dollars per year); providing a verbal description of scientists' confidence in the information (e.g., scientists are very confident that a policy will protect an ecosystem); communicating the degree of consensus about the information among scientists (e.g., 75 percent of scientists agree that a particular policy will protect the ecosystem); or conveying the probability that an outcome or benefit will occur (e.g., scientists believe this policy has a 75 percent probability of protecting the ecosystem). There is substantial evidence that people have difficulty accurately interpreting this last type of evidence (e.g., Tversky and Kahneman, 1974), but the EPA may want to explore these various methods for conveying uncertainty to determine the extent to which people understand and use different types of information about uncertainty in valuation.

Scale and spatial issues. Because the spatial and temporal scale of ecological systems and services may affect valuation processes, these dimensions should be incorporated into the communication of information and the measurement of value. For example, the information that respondents receive during the survey interview should, if possible, explicitly describe the scale of a proposed policy or the ecosystem or service for valuation. This is particularly true if the scale is fixed and can be described consistently across presentation of information, evaluation of policies, and valuation of ecological systems and services. In other cases, the physical or temporal scale may be variables of interest, so researchers may want to measure whether these features affect respondents' evaluation of the policy. This could be accomplished by manipulating the physical or temporal scales of a proposed policy (either between- or within-subjects) to determine whether and how these features influence support for the policy.

Transfer issues. The most effective way to use surveys for valuation applicable to a particular ecosystem is to use a survey tailored specifically to that situation. However, this requires that time and material resources be devoted each time EPA must complete a value assessment. A more efficient approach might be to design studies to test whether the findings from a survey about one set of environmental conditions can be extrapolated to a different set of environmental conditions. For example, if a survey measures the ecosystem values affected by one oil spill, would it be possible to multiply these losses by three to anticipate the comparable losses caused by three comparable oil spills to three comparable ecosystems? Even if such extrapolations must be done using more complex transformations, it may be possible to conduct parametric research to ascertain how such predictions can be made.

Implementing survey research at EPA. Whatever the value measure being sought, the design and conduct of surveys is best done when informed by the literature on survey methods. Therefore, it is important that EPA surveys be implemented at least partly by individuals who are well-versed and up-to-date in this literature. This is probably best accomplished by teams of researchers composed partly of EPA employees who specialize in surveys and outside consultants who are experts in survey methods. EPA may, therefore, want to assess its current capacity to conduct or oversee contractor design and implementation of high-quality surveys.

OMB clearance is required for all EPA surveys, and achieving this clearance requires that a survey meet high standards of quality. In order to maximize the likelihood of approval, it is important that a proposed survey meet a set of criteria:

- Representative sampling of the population of interest with minimal non-coverage error
- A very high response rate or a plan to assess the presence of nonresponse bias
- A measuring instrument that has been developed according to optimal design and pretesting practices
- A measurement approach for which a body of empirical evidence documents validity

Probability sampling is relatively easy to do for general population samples, but more challenging for smaller, more specific subpopulations which require specialized sampling procedures currently under development (e.g., Blair and Blair, 2006; Rocco, 2003). If EPA is interested in conducting surveys of such specialized subpopulations, it may be of value to commission a group of sampling statisticians to develop a series of guidelines that can be consulted and followed when conducting sampling for such studies.

The recent literature on response rates has focused on exploring the impact of response rates on data accuracy and exploring the effectiveness of various data-collection techniques for enhancing response rates. Although lower response rates are generally not associated with substantially decreased accuracy, it may be useful for EPA to reanalyze a set of its own past surveys simulating lower response rates and observing the impact on the survey results. If systematic bias is detected, it may be possible to build correction algorithms to adjust the results of future surveys to correct for such bias.

It might seem obvious that when EPA conducts surveys, all possible steps should be taken to increase response rates. According to federal convention, that cannot include offering financial incentives to respondents, but EPA can implement other techniques to enhance response rates, including lengthening the field period during which data are collected, and more attempts to contact potential respondents. However, to justify resources to implement such techniques, it is important to have empirical evidence documenting the effectiveness of these techniques for EPA surveys. It is also important to be sure that efforts to increase the response rate of a survey do not inadvertently decrease the representativeness of the sample. For example, telling respondents that a survey is about the environment may increase response rates among people interested in the environment and may decrease response rates by a smaller margin among less-interested people, thus increasing nonresponse bias. EPA may want to conduct studies assessing whether efforts to increase response rates unintentionally decrease sample representativeness.

Another approach to facilitating OMB approval may be to gather evidence documenting the effectiveness of particular measurement techniques. For example, there is considerable controversy surrounding the use of contingent valuation (CV) methods in surveys. However, a National Oceanic and Atmospheric Administration blue ribbon panel concluded that CV is a viable method of valuation (Arrow et al, 1993). It may be of value for EPA to identify the optimal elements and implementation of a CV survey and to assess the validity of CV measurement in surveys by comparisons with other monetary measures (e.g., from revealed-preference studies) or with measures based on judgments of preference, importance, or acceptability. This same sort of developmental work can be conducted with other valuation techniques such as conjoint analysis, about which there is little consensus (e.g., Dennis, 1998; Stevens, et al., 2000; Wainright, 2003). This may help to reassure OMB evaluators of the merit of value measurements produced by the various methods when they are implemented well. EPA could also consider conducting research comparing the validity of value assessments by these and other techniques to identify the technique(s) that yield the most valid data.

Finally, new OMB guidelines on surveys suggest that when a survey is expected to obtain a relatively low response rate, investigators should plan to implement techniques to assess sample representativeness. Rather than outlining what such procedures would look like, OMB has left it to investigators to propose and justify such techniques. EPA could therefore commission work to design procedures for this purpose and conduct studies to validate the effectiveness of the procedures.

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Endnotes

[1.] Laws include: the Clean Air Act, Clean Water Act, Comprehensive Environmental Response, Compensation, and Liability Act, Federal Insecticide, Fungicide and Rodenticide Act, Toxic Substances Control Act, and Resource Conservation and Recovery Act.

[2.] Although C-VPESS was initiated by the SAB, senior EPA managers supported the concept of this SAB project and participated in the initial background workshop that launched the work of the C-VPESS.

[3.] The SAB Staff Office published a *Federal Register* notice on March 7, 2003, (68 FR 11082-11084) announcing the project and calling for the public to nominate experts in the following areas: decision science, ecology, economics, engineering, psychology, and social sciences with emphasis in ecosystem protection. The SAB Staff Office published a memorandum on August 11, 2003, documenting the steps involved in forming the new committee and finalizing its membership.

[4.] The committee developed the conclusions in this report after multiple public meetings, teleconferences and workshops: a) an Initial Background Workshop on October 27, 2003, to learn the range of EPA's needs for science-based information on valuing the protection of ecological systems and services from managers of EPA Headquarters and Regional Offices; b) a Workshop on Different Approaches and Methods for Valuing the Protection of Ecological Systems and Services, held on April 13-14, 2004; c) an advisory meeting focused on support documents for national rule makings held on June 14-15, 2004; d) an advisory meeting focused on regional science needs, in EPA's Region 9 (San Francisco) Office on Sept. 13, 14, and 15, 2004; e) advisory meetings held on January 25-26, 2005, and April 12-13, 2005, to review EPA's draft *Ecological Benefits Assessment Strategic Plan;* and f) a Workshop on Science for Valuation of EPA's Ecological Protection Decisions and Programs, held on December 13-14, 2005, to discuss the integrated and expanded approach described in this paper. The committee also discussed text drafted for this report at public meetings on October 25, 2005; May 9, 2006; October 5-6, 2006, and May 1-2, 2007 and on eight subsequent public teleconferences.

[5.] The committee also notes a report published shortly before this report was finalized (United Kingdom Department for Environment, Food and Rural Affairs, 2007).

[6.] Likewise, this definition would not include goods or services such as recreation that are produced by combining ecological inputs or outputs with conventional inputs (such as labor, capital, or time). In addition, Boyd and Banzhaf (2006) advocate defining changes in ecosystem services in terms of standardized units or quantities, which requires that they be measurable in practice. Such an approach is consistent with the concept of "green accounting," which extends the principles embodied in measuring marketed products to the measurement and consideration of the production, or changes in the stock, of ecological or other environmental "products" (reference NRC report by Nordhaus-CITATION NEEDED).

[7.] There is controversy over the meaning of intrinsic value (Korsgaard, 1996). Many people take intrinsic value to mean that the value of something is inherent in that thing. Some philosophers have argued that value or goodness is a simple non-natural property of things (see Moore 1903 for the classical statement of this position), and others have argued that value or goodness is not a simple property of things but one that supervenes on the natural properties to which we appeal to explain a thing's goodness (this view is defended by, among others, contemporary moral realists; see McDowell (1985), Sturgeon (1985), Sayre-McCord (1988), and Brink (1989).

[8.] One of these elements is an evaluation of willingness to pay for or willingness to accept a proposed regulatory action and the main alternatives identified and the related costs. The circular explicitly defines benefits using the economic/utilitarian concept of willingness to pay (or willingness to accept). The circular contains general guidance on how to provide monetized, quantitative, and qualitative information to characterize contributions to human welfare as fully as possible.

[9.] Under GPRA, the Office of Management and Budget requires EPA to periodically identify its strategic goals and describe both the social costs and budget costs associated with them. EPA's Strategic Plan for 2003-2008 described the current social costs and willingness-to-pay or willingness-to-accept analyses of EPA's programs and policies under each strategic goal area for the year 2002 (EPA, 2003). This analysis repeatedly points out that EPA lacks data and methods to quantify willingness-to-pay or willingness-to-accept associated with the goals in its strategic plan. In addition, GPRA established requirements for assessing the effectiveness of federal programs, including the outcomes of programs intended to protect ecological resources. EPA must report annually on its progress in meeting program objectives linked to strategic plan goals and must engage periodically in an in-depth review [through the Program Assessment Rating Tool (PART)] of selected programs to identify their net contributions to human welfare and to evaluate their effectiveness in delivering meaningful, ambitious program outcomes. Characterizing ecological contributions to human welfare associated with EPA programs is a necessary part of the program assessment process.

[10.] These interviews were conducted by one committee member, Dr. James Boyd, in conjunction with the Designated Federal Officer Dr. Angela Nugent, over the period September 22, 2004, through November 23, 2005. In seven sets of interviews, Dr. Boyd spoke with staff from the Office of Policy, Economics and Innovation, Office of Water, Office of Air and Radiation, and the Office of Solid Waste and Emergency Response.

[11.] NCEE is typically brought in by the program offices to help both design and review RIAs. NCEE can be thought to provide a centralized "screening" function for rules and analysis before they go to OMB. NCEE is actively involved in discussions with OMB as rules and supporting analysis are developed and advanced.

[12.] In addition, the circular states (p. 27) "If monetization is impossible, explain why and present all available quantitative information" and "If you are not able to quantify the effects, you should present any relevant quantitative information along with a description of the unquantified effects, such as ecological gains, improvements in quality of life, and aesthetic beauty" (p. 26).

[13.] The Committee reviewed and critically evaluated the CAFO Environmental and Economic Benefits Analysis at its June 15, 2004, meeting. As stated in the Background Document for SAB Committee on Valuing the Protection of Ecological Systems and Services for its session on June 15, 2004, the purpose of this exercise was "to provide a vehicle to help the Committee identify approaches, methods, and data for characterizing the full suite of ecological 'values' affected by key types of Agency actions and appropriate assumptions regarding those approaches, methods, and data for these types of decisions." The Committee based its review on EPA's final benefits report (EPA, 2002a) and a briefing provided by the EPA Office of Water staff.

[14.] In December 2000, EPA proposed a new CAFO rule under the federal Clean Water Act to replace 25-year-old technology requirements and permit regulations (66FR 2959). EPA published its final rule in December 2003 (68 FR 7176). The new CAFO regulations, which cover over 15,000 large CAFO operations, reduce manure and wastewater pollutants from feedlots and land applications of manure and remove exemptions for stormwater-only discharges.

[15.] Prior to publishing the draft CAFO rule in December 2000, EPA spent two years preparing an initial assessment of the costs and benefits of the major options. After releasing the draft rule, EPA spent another year collecting data, taking public comments, and preparing assessments of new options. EPA published its final assessment in 2003. An intra-agency team at EPA, including economists and environmental scientists in the Office of Water, Office of Air and Radiation, Office of Policy Economics and Innovation, and Office of Research and Development, worked on the benefit assessment. EPA also worked with the U.S. Department of Agriculture in developing the assessment. Dr. Christopher Miller of EPA's Office of Water estimated that EPA spent approximately $1 million in overall contract support to develop the benefit assessment. EPA spent approximately $250,000-$300,000 on water-quality modeling as part of the assessment.

[16.] The potential "use" benefits included in-stream uses (commercial fisheries, navigation, recreation, subsistence, and human health risk), near-stream uses (non-contact recreation, such as camping, and nonconsumptive, such as wildlife viewing), off-stream consumptive uses (drinking water, agricultural/irrigation uses, and industrial/commercial uses), aesthetic value (for people residing, working, or traveling near water), and the option value of future services. The potential "non-use" values included ecological values (reduced mortality/morbidity of certain species, improved reproductive success, increased diversity, and improved habitat/sustainability), bequest values, and existence values.

[17.] These benefits were recreational use and non-use of affected waterways, protection of drinking water wells, protection of animal water supplies, avoidance of public water treatment, improved shellfish harvest, improved recreational fishing in estuaries, and reduced fish kills.

[18.] These include reduced eutrophication of estuaries; reduced pathogen contamination of drinking water supplies; reduced human and ecological risks from hormones, antibiotics, metals, and salts; improved soil properties from reduced over-application of manure; and "other benefits".

[19.] EPA apparently conducted no new economic valuation studies (although a limited amount of new ecological research was conducted) and did not consider the possible benefits of developing new information where important benefits could not be valued in monetary terms based on existing data.

[20.] For example, while the report notes the potential effects of discharging hormones and other pharmaceuticals commonly used in CAFOs into drinking water sources and aquatic ecosystems, the nature and possible ecological significance of these effects is not adequately developed or presented. Similarly, the report does not adequately address the well-known consequences of discharging trihalomethane precursors into drinking-water sources.

[21.] EPA used estimates based on a variety of public surveys in its benefit transfer efforts, including: a national survey (1983) that determined individuals' willingness to pay for changes in surface water quality relating to water-based recreational activities (section 4 of the CAFO Report); a series of surveys (1992, 1995, 1997) of willingness to pay for reduced/avoided nitrate (or unspecified) contamination of drinking water supplies (section 7); and several studies (1988, 1995) of recreational fishers' values (travel cost, random utility model) for improved/protected fishing success related to nitrate pollution levels in a North Carolina estuary (section 9).

[22.] Although EPA later prepared more detailed conceptual models of the CAFO rule's impact on various ecological systems and services, EPA did not prepare these models until after the Agency finished its analysis.

[23.] Contamination of estuaries, for example, might negatively affect fisheries in the estuary (a primary effect) but might have an even greater impact on offshore fisheries that have their nurseries in the estuary (a secondary effect).

[24.] The goal of EPA's analysis was a national-level assessment of the effects of the CAFO rule. This involved the effects of approximately 15,000 individual facilities, each contributing pollutants across local watersheds into local and regional aquatic ecosystems. A few intensive case studies were mentioned in the report and used to calibrate the national scale models (e.g., NWPCAM, GLEAMS), but there was no indication that these more intensive data sets were strategically selected or used systematically for formal sensitivity tests or validations of the national-scale model results.

[25.] A key question, of course, is whether case studies are representative. Both representative and non-representative case studies can nonetheless provide useful information. Representative case studies offer more detailed data and models that can fill in gaps in broad-scale national analyses and check the validity of these analyses systematically. In general, systematically performing and documenting comparisons to intensive study sites can indicate the extent to which the national model needs to be adjusted for local or regional conditions. It also can provide data for estimating the range of error and uncertainty in the projected national-scale effects. Non-representative case studies can provide valuable information about the extent to which certain regions or conditions may yield impacts that vary considerably from the central tendency predicted by the national analyses.

[26.] This could include either a robust public involvement process following Administrative Procedures Act requirements (e.g., publication in the *Federal Register*), or some other public involvement process (see EPA's public involvement policy [EPA Office of Policy, Economics and Innovation, 2003] ; the SAB report on science and stakeholder involvement (EPA Science Advisory Board, 2001).

[27.] In theory, one can value a final product *either* directly (output valuation) or indirectly as the sum of the derived value of the inputs (input valuation), but not both, because separately valuing both intermediate and final products leads to double counting. In some cases, it may be easier or more appropriate to value the intermediate service, while in other cases the change in the final product can be directly valued.

[28.] Note that these essential ecosystem characteristics are very similar to the seven ecological indicators in EPA's report on assessing ecological systems: landscape condition, biotic condition, chemical and physical characteristics, ecological processes, hydrology and geomorphology and natural disturbance regimes (EPA Science Advisory Board 2002b).

[29.] Both embodied energy analysis and ecological footprint analysis use a consistent set of accounting principles based on input-output analysis to compute these costs. An alternative biophysical method, emergy, on the other hand, also seeks to measure the energy cost of producing a good or service, but it does not follow these principles, and hence, does not generally satisfy basic adding-up properties. Rather, it focuses on converting inputs of varying quality to a common energy metric – usually solar energy equivalents – so that they can be combined into a cost estimate measured in those units.

[30.] The U.S. federal government is one of the largest producers of survey data, which form the basis of many government policy making decisions (see the table below for examples of federally funded surveys).

Examples of federal surveys		
Continuously Funded Surveys	Agency Sponsor	Years
Survey of Income and Program Participation	Census Bureau	1984-

		present
Consumer Expenditure Surveys	Census Bureau	1968-present
Survey of Consumer Attitudes and Behavior	National Science Foundation	1953-present
Health and Nutrition Examination Surveys	National Center for Health Statistics	1959-present
National Health Interview Survey	National Science Foundation	1970-present
American National Election Studies	National Science Foundation	1948-present
Panel Study of Income Dynamics	National Science Foundation	1968-present
General Social Survey	National Science Foundation	1972-present
National Longitudinal Survey	Bureau of Labor Statistics	1964-present
Behavioral Risk Factor Surveillance System	Centers for Disease Control and Prevention	1984-present
Monitoring the Future	National Institute of Drug Abuse	1975-present
Continuing Survey of Food Intake by Individuals	Department of Agriculture	1985-present
National Aviation Operations Monitoring System	National Aeronautics and Space Admin.	2002-present
National Survey of Drinking and Driving	National Highway Traffic Safety Admin.	1991-present
National Survey of Family Growth	National Center for Health Statistics	1973-present
National Survey of Fishing, Hunting, and Wildlife-Associated Recreation	Census Bureau	1991-present
National Survey of Child and Adolescent Well-Being	Department of Health and Human Services	1997-present
Survey of Earned Doctorates	National Science Foundation	1958-present
National Survey on Drug Use and Health	Department of Health and Human Services	1971-present

Youth Risk Behavior Surveillance System	Department of Health and Human Services	1990-present
National Crime Victimization Survey	Bureau of Justice Statistics	1973-present
Schools and Staffing Survey	National Center for Educational Statistics	1987-present
Educational Longitudinal Survey	National Center for Educational Statistics	2002-present
Current Employment Statistics Survey	Bureau of Labor Statistics	1939-present
Other Major Federally-Funded Surveys	Agency Sponsor	
National Survey of Distracted and Drowsy Driving	National Highway Traffic Safety Administration	
National Survey of Veterans	Department of Veteran Affairs	
National Survey of Children's Health	Health Resources and Services Administration's Maternal and Child Health Bureau	
National Survey of Recent College Graduates	National Science Foundation	
National Survey of Speeding and Other Unsafe Driving Actions	Department of Transportation	

[31.] Valuations also require a variety of other predictions, including predicting the anthropogenic response to EPA actions or decisions. Valuations sometimes ignore the need for such predictions. For example, many valuations assume that the regulated community will comply fully with regulations and not adjust other behavior in response to the regulation. In many cases, this assumption is incorrect. Where valuations do incorporate additional predictions, however, they again are subject to uncertainty.

[32.] For a more detailed discussion of the sources and possible typologies of uncertainty, see Krupnick, Morgenstern et al. (2006).

[33.] The discussion of value in the National Research Council report (2001) and SAB review of the EPA's Draft *Report on the Environment* (EPA SAB, 2005) and related literature (e.g., Failing and Gregory, 2003) tends to focus more on qualitative rather than quantitative expressions. However, issues of scale and aggregation are important. Both the NRC report (2001) and the SAB review of the EPA's Draft Report on the Environment (EPA SAB, 2005) emphasize the importance of using regional and local indicators. Over-aggregating information can obscure critical ecological threats or problems. In general, allowing sensitivity analysis on disaggregated data is desirable if the data are aggregated at a regional or higher level. So while some authors recommend simple summary indicators (e.g., Schiller et al., 2001; Failing and Gregory, 2003), others emphasize disaggregating indicators (EPA SAB, 2003).

34. This analysis evaluated the benefits and costs of amendments to the Clean Air Act passed by Congress in 1990. Its effort to evaluate the ecological benefits of these amendments raises many of the same issues that arise in evaluating the benefits of national rules. The prospective analyses compare the sequence of increasingly stringent rules called for under the 1990 Clean Air Act Amendments with a situation where the rules were held constant at their 1990 levels (e.g., with the regulatory regime prior to the amendments).

35. The one exception is the national survey on water quality conducted in the 1980s by Carson and Mitchell (1993), but this survey is not appropriate for use by the Agency in valuing ecosystem services, for reasons discussed below.

36. One exception is Greenstone 2008.

37.

Major Chicago Wilderness reports and chronology of valuation effort

Decision/document	Date	Source/URL
Biodiversity Recovery Plan	1999 (Award from APA in 2001 for best plan)	http://www.chicagowilderness.org/pubprod/brp/index.cfm Executive summary available at http://www.chicagowilderness.org/pubprod/brppdf/CWBRP_chapter1.pdf
Chicago Wilderness Green Infrastructure Vision	Final report, March 2004	http://www.nipc.org/environment/sustainable/biodiversity/greeninfrastructure/Green%20Infrastructure%20Vision%20Final%20Report.pdf
Green Infrastructure Mapping		http://www.greenmapping.org/
A Strategic Plan for the Chicago Wilderness Consortium	17 March 2005	http://yosemite.epa.gov/SAB/sabcvpess.nsf/06347c93513b181385256dbf00541478/72c1b26a9d2087568525713f005832e1!OpenDocument
Chicago Wilderness Regional Monitoring Workshop final report by Geoffrey Levin	February 2005	http://yosemite.epa.gov/SAB/sabcvpess.nsf/06347c93513b181385256dbf00541478/8c33ee9115d706e68525713f005784e6!OpenDocument
Center for Neighborhood Technology (CNT) – green	Copyright 2004-2007	http://greenvalues.cnt.org/calculator

infrastructure valuation calculator		

[38.] In one 1996 poll, only two out of ten Americans had heard of the term "biological diversity." Yet when the concept was explained, 87% indicated that "maintaining biodiversity was important to them" (Belden and Russonello, 1996, as cited in the Chicago Wilderness *Biodiversity Recovery Plan*, p. 117).

[39.] The use of surveys has also been growing in the private sector and the academic world (Presser, 1984; Saris et al., 2003), which likely reflects that: (1) surveys are now capable of generating much more interesting data, via implementation of multifactorial experimental designs and complex measurement procedures; (2) cross-national comparisons are of increasing interest; and (3) social scientists want to collect data on more heterogeneous and representative samples. There is also substantial evidence that the quality of optimally-collected survey data are generally quite high. For example, in the Monthly Survey of Consumer Attitudes and Behavior, a representative national sample of American adults has been asked each month what they expect to happen to the unemployment and inflation rates in the future. Their aggregated answers have predicted later changes in actual unemployment and inflation remarkably well (correlations of .80 and .90, respectively, between 1970 and 1995).

[40.] Presenting a seven-point bipolar rating scale is easy to do visually but is more challenging to do aurally. Such scales can be presented in sequences of two questions that ask, first, whether the respondent is on one side of the midpoint or the other or at the midpoint (e.g., "Do you like bananas, dislike them, or neither like nor dislike them?"). Then, a follow-up question can ask how far from the midpoint the respondents are who settle on one side or the other (e.g., "Do you like bananas a lot or just a little?"). This branching approach takes less time to administer than offering the single seven-point scale, and measurement reliability and validity are higher as well (Krosnick and Berent, 1993).

[41.] A common set of rating scale labels assesses the extent of agreement with an assertion: strongly agree, somewhat agree, neither agree nor disagree, somewhat disagree, strongly disagree (Likert, 1932). Yet a great deal of research shows that these response choices are problematic because of acquiescence response bias, whereby some people are inclined to agree with any assertion, regardless of its content (see, e.g., Couch & Keniston, 1960; Jackson, 1967; Schuman and Presser, 1981), which may distort the results of substantive investigations (e.g., Jackman, 1973; Winkler et al., 1982). Although it might seem that the damage done by acquiescence can be minimized by measuring a construct with a large set of items, half of them making assertions opposite to the other half, doing so requires extensive pretesting, is cumbersome to implement, is cognitively burdensome for respondents, and frequently involves asking respondents their agreement with assertions containing the word "not" or some other such negation, which increases both measurement error and respondent fatigue (e.g., Eifermann, 1961; Wason, 1961). Acquiescers also presumably end up at the midpoint of the resulting measurement dimension, which is probably not where most belong on substantive grounds. Most importantly, answering an agree/disagree question always involves first answering a comparable rating question in one's mind. For example, respondents who are asked their agreement with the assertion "I am not a friendly person" must first decide how friendly they are and then translate that conclusion into the appropriate selection. It would be simpler and more direct to ask respondents how friendly they are on a scale from "extremely friendly" to "not friendly at all." Every agree/disagree question implicitly requires a respondent to make a mental rating of an object on the construct of interest, so asking about that dimension is simpler, more direct, and less burdensome. Not surprisingly, then, the reliability and validity of rating scales that do so are higher than those of agree/disagree rating scales (e.g., Ebel, 1982; Mirowsky and Ross, 1991; Ruch and DeGraff, 1926; Wesman, 1946).

[42.] This recommendation must be modified in light of conversational conventions about word order. For example, in a list of terms, it is conventional to say the positive before the negative (e.g., "for or against," "support or oppose"; Cooper and Ross, 1975). Similarly, Guilford (1954) asserted that it is most natural and sensible to present evaluative response options on rating scales in order from positive to negative. Holbrook, Krosnick, Carson, and Mitchell (2000) showed that measurement validity is greater when the order of answer choices conforms to this convention.