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1 INTRODUCTION

The Environmental Protection Agency's (EPA's) mission 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. In fact, EPA's Strategic Plan (U.S Environmental Protection Agency 2006b) explicitly identifies the need to ensure "healthy communities and ecosystems" as one of its five major goals. In addition, EPA's efforts in protecting ecological resources--and its authority for doing so--have been documented in Agency publications and independent historical sources (U.S. Environmental Protection Agency 1994;U.S. Environmental Protection Agency Risk Assessment Forum 2003; U.S. Environmental Protection Agency Science Advisory Board 2000; Hays 1989; Russell III 1993).

EPA's mission to protect the environment requires that the Agency understand and protect ecological systems. "Ecosystem" is the term used by ecologists to describe the dynamic complex of plant, animal, and microorganism communities and the non-living environment interacting as a system. For example, a forest ecosystem is comprised 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 (see, for example, Daily 1997). Ecosystems provide not only goods and services directly consumed by society such as food, timber, and water, but also services such as flood protection, disease regulation, pollination, and disease, pest, and climate control. In addition, there is increasing recognition of the impact of human activities on ecosystems (see, for example, Millennium Assessment). Examples of this impact include not only traditional air and water pollution (such as sulfur dioxide emissions, ground-level ozone, and eutrophication), but also land conversions that lead to deforestation or loss of wetlands and biodiversity; global warming; changes in the nitrogen cycle; invasive species; and aquifer depletion.

Given the vital role that ecosystems play in our lives, changes in the state of these systems or the flow of services they provide can have important implications. EPA actions (e.g., regulations, rules, programs, policy decisions) can be one source of these changes. Many EPA actions relating to air quality, water quality, and land use affect the condition of the environment and the flow of ecological services from it. These impacts can occur narrowly at a local scale or broadly at a national scale.

Despite their importance, the ecological impacts of EPA actions have received relatively limited consideration in EPA policy analyses. Rather, these analyses have tended to focus on a relatively narrow set of ecological endpoints, such as those identified by tests required for pesticide regulation (e.g., the effects on survival, growth, and reproduction of aquatic invertebrates, fish, birds, mammals, and both terrestrial and aquatic plants) or mortality to fish, birds, plants, and, animals, as required by provisions of several laws administered by the Agency[1] (U.S. Environmental Protection Agency Risk Assessment Forum 2003). Given EPA's responsibility to ensure healthy communities and ecosystems the Agency must consider the full range of impacts that its actions will have not only on human health but also on individual organisms and plant and animal populations, as well as on the key structural and functional characteristics of communities and ecosystems.

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

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 might 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), which was formed by EPA's Science Advisory Board (SAB) in 2003. 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. Toward this end, it formed C-VPESS,[2] a group of experts from the disciplines of decision science, ecology, economics, engineering, philosophy, and psychology, with an emphasis on ecosystem protection. The committee's charge was to undertake a project designed to improve the Agency's ability to value ecological systems and services.[3] The SAB set the following goals for this project: a) assessing Agency needs for valuation to support decision making; b) assessing the state of the art and science of valuing protection of ecological systems and services, and c) 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 identifying the key 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. It provides advice to the Administrator, EPA managers, EPA scientists and analysts, and EPA staff across the Agency concerned with ecological protection. It adopts a broad view of EPA's work, which it understands to encompass national rule making, regional decision making, and programs in general that protect ecological systems and services. It recommends that EPA expand its current approach in important ways.

This report appears at a time of lively interest internationally, nationally, and within EPA in the issue of 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 (Millennium Ecosystem Assessment Board 2003; Silva and Pagiola 2003; National Research Council 2004; Pagiola, von Ritter et al. 2004; Millennium Ecosystem Assessment 2005). In addition, the Agency itself has engaged in efforts to improve ecological valuation. The most recent product of these efforts is the *EBASP* report noted above (U.S. Environmental Protection Agency 2006a). This report examines EPA efforts to improve ecological valuation, which have been geared toward the use of economic valuation in benefit-cost analysis. EPA also has sought to strengthen the science supporting ecological valuation through the extramural Science to Achieve Results (STAR) grants program. STAR grants involving ecological valuation have primarily applied economic valuation methods to various ecosystem services.

The committee has both learned from and built upon these recent efforts. The C-VPESS distinguishes its work from the earlier efforts, however, in several key ways. First, the C-VPESS considers EPA the principal audience for its work. In particular, it 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 (for example, the Millennium Assessment and NRC report) do not consider the specific policy contexts or constraints faced by EPA. Second, most previous work has concentrated on economic valuation as the primary valuation method. C-VPESS, by contrast, is inter-disciplinary and does not focus solely on economic methods or values. The committee will offer advice on several approaches to characterizing or estimating values and in each case will emphasize issues relevant to EPA policy and decision-making.

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 the basic principles outlined in Chapter 2 in more detail, with a focus on implementation. Chapter 3 discusses the part of the implementation process related to prediction of changes in ecological systems and services that stem from EPA actions or decisions. Chapter 4 then discusses a variety of methods for valuing these changes. Cross-cutting issues relating to uncertainty and communication are discussed in Chapter 5 Recognizing that implementation of the process may vary depending on the decision context, Chapter 6of the report discusses implementation in three specific contexts where ecological valuation could play an important role in EPA analysis: national rulemaking, site-specific decisions (regarding, for example, cleanup and restoration), and regional partnerships. Finally, Chapter 7provides a summary of the report’s major conclusions and recommendations..

2 CONCEPTUAL FRAMEWORK

2.1. An Overview of Key Concepts

2.1.1. The Concept of Ecosystem Services

As noted above, the term ecosystem describes a dynamic complex of plant, animal, and microorganism communities and the 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.

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; however, they are not synonymous with ecosystem services. Ecosystem processes and functions describe biophysical relationships and exist whether or not humans benefit from them. These relationships only generate ecosystem services, though, if they contribute to human well-being. Thus, ecosystem services cannot be defined independently of human values.

The following categorization of ecosystem services has been used by the Millennium Ecosystem Assessment:

a) 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 services are traded in markets.

b) Regulating services - services received from 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.

c) 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.

d) 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.

This categorization suggests a very broad definition of services, limited only by the requirement of a contribution (direct or indirect) to human well-being. This broad approach reflects the recognition of the myriad ways in which ecosystems support human life and contribute to human well-being. Alternatively, Boyd and Banzhaf (2006) propose a definition that focuses on services as "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 services, since these are what affect people most directly and are consequently what they are most likely to understand. Under this definition, ecosystem functions and processes, such as nutrient recycling, are not considered services; while they contribute to the production of ecological end products or outputs, they are not outputs themselves.[5] Principles for defining ecosystem services are discussed in more detail in Chapter 3 of this report.

Regardless of the specific definition used, the general concept of ecosystem services plays a key role in evaluating policies related to ecological protection. Even without any subsequent valuation, explicitly listing the services derived from an ecosystem - and using the best available methods in the ecological, social, and behavioral sciences to develop that list - can help to ensure appropriate recognition of the full range of potential impacts of a given policy option. This can help make the analysis of ecological systems more transparent and accessible and can help inform decision makers of the full range of potential impacts stemming from different options before them.

The concept of ecosystem services provides an approach to evaluating the many ways in which ecological systems, and changes to those systems induced by human actions, affect human well-being. Ecosystems, however, can also be valued for reasons that are independent of effects on human well-being. As discussed in the following section, 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.

2.1.2. The Concepts of Value

The committee recognizes that there are many sources or types of value that are relevant when valuing the protection of ecosystems and their services. In considering concepts of value, a fundamental distinction can be made between those things that we value as ends or goals and those things that we value only as means. To value something as a means is to value it for its usefulness in helping to bring about an end or goal that is valued in its own right. Things or actions valued for their usefulness as means in this sense 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.[6]

The distinction between ends and means plays an important role when thinking about valuing ecological systems and 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 (e.g., ends vs. means). For example, some people claim that the very existence of a species or ecological system has value in itself in addition to any instrumental value derived from the usefulness of the services it provides. This claim can be interpreted in different ways. It could mean that the existence of a species or an ecological system is valuable because people derive satisfaction from its existence, independent of specific uses they may make of its services. Economists would interpret this type of value as "existence value", which is a form of instrumental value since it is based on the premise that the existence of the species or ecological system is one of many things that contribute to human well-being. Alternatively, the claim could be interpreted to mean that an ecological system is valuable as an end or goal for its own sake, implying that the reasons for this claim are independent of the contribution that the existence of the ecological system can make to human well-being. This interpretation of the claim is consistent with values in which the existence or well-being of other species or the state of ecosystems can be ends in themselves.

The committee recognizes that ecosystems can be valued both as independent ends or goals and as instrumental means to other ends or goals. To reflect this recognition, throughout this report, the term value is broadly used. It includes values that stem from contributions to human well-being as well as values that reflect other considerations, such as social and civil norms (including rights), and moral, religious, and spiritual beliefs and commitments.

Recognizing that values can be instrumental or intrinsic, this report next turns to how those values can be defined. A key implication of instrumental values is substitutability. Substitutability means that more of one thing can be traded off against less of something else as long as both contribute to achieving the same goal. Assuming there is more than one thing that contributes to the achievement of a goal and that alternative means are substitutable, the instrumental value of something can be defined as the amount of something else that would make an equivalent contribution to the goal and could replace the thing in question if it were to be lost. For an example taken from economic valuation methods, if the end goal is the maximization of human well-being and both the existence of a species and money contribute to that goal, then the value of the species can be defined as the amount of money that would be needed to offset the loss in human well-being that would result from loss of the species. Likewise, if the end goal is the provision of clean water to a given community and this goal can be achieved through either watershed protection or the construction of a water purification plant, then given that the clean water from either source is accepted as equivalent the value of watershed protection can be defined as the cost savings from not having to build the purification plant.

While the definition of instrumental value is clear, it is less clear how to define, measure, and ultimately quantify intrinsic value. When something is an end in itself, its value cannot be determined in terms of trade-offs since there are no substitutes for something that is an end in itself. For example, if ecosystems are viewed as ends in themselves or are valued for other than human utilitarian purposes (e.g., out of respect for, or acceptance of, ethical obligations toward nature), then a water purification plant cannot be substituted for watershed protection. Defining the value of items as their contribution toward achieving a goal (Costanza 2000) requires that the identification of the goal be separate from the item or good being valued (i.e., separate from the means for attaining the goal). With intrinsic values, this is not possible since the good being valued and the goal are not separate. In this sense, intrinsic values cannot be quantified or measured. Nonetheless, as envisioned by the committee in this report, identifying and providing information about intrinsic goals relating to ecosystem protection, including measures of how strongly people care about them (perhaps relative to other goals), is an important component of the assessment of ecosystem values and a legitimate consideration for Agency decision making.

This raises the question of how these intrinsic values can be compared to instrumental values when tradeoffs are, in fact, required. In other words, how does society balance these intrinsic values - moral, aesthetic, religious or spiritual goals - with its interests in instrumental contributions to human well-being, both as individuals and as a society? This cannot be done by a direct comparison of the associated values. Rather, if trade-offs are required, society must engage in political and deliberative discussion of alternative goals and visions for the future in order to balance intrinsic and instrumental values. This discussion should be an ongoing and vital part of any democratic society.

Although the concepts of instrumental and intrinsic value provide a broad categorization of values, other distinctions between different types or concepts of value can also be made and are important for understanding the values associated with ecological systems and services. For example, values can be classified as either *anthropocentric values* or *nonanthropocentric values*. Anthropocentric values are based on the contributions that ecological systems and services make to human well-being. Nonanthropocentric values are based on a variety of ethical and philosophical perspectives. This category includes biocentric and eco-centric values, which are based on an evaluation of ecological changes and their effects on ecosystems or nonhuman species, and values stemming from theories of value that are not based directly on human well-being. Note that the anthropocentric values derived from contributions of ecosystem services to human well-being are often referred to as the "benefits" from ecosystem services (see

Millennium Assessment). The term benefits, however, has a very precise meaning in the context of EPA regulatory impact analyses conducted under OMB guidance (see further discussion that follows). In that context, benefits are defined by the economic concept of willingness to pay for a good or service or willingness to accept compensation for it. Thus, the term "benefits" means different things in different contexts. For this reason, throughout this report the committee refers to the broader concept of anthropocentric values as contributions to human welfare, and uses the term "benefits" only when there is no potential for confusion about what it includes.

In addition to the distinction between anthropocentric and non-anthropocentric values, values can also be distinguished by whether they are *preference-based* or *bio-physical*. Values based directly on human preferences can be either instrumental or intrinsic values and can be either anthropocentric or non-anthropocentric. In contrast, bio-physical values do not directly reflect human preferences. However, they can still be either implicitly anthropocentric or non-anthropocentric. They are non-anthropocentric when they reflect intrinsic values unrelated to human well-being; and they are implicitly anthropocentric when they reflect a prior decision or commitment to a bio-physical goal that is deemed to be important for human welfare. For example, values based on contributions to a goal of preserving biodiversity can reflect either a belief that biodiversity preservation has intrinsic value (a non-anthropocentric approach), or a prior commitment to preserving biodiversity because of its importance for human welfare (an implicitly anthropocentric approach). In either case, the value of an ecosystem change is defined in terms of its contribution to the goal of preserving bio-diversity, which does not require direct information about people's preferences for that particular change. Similarly, if society has identified a goal of ensuring clean water to a community (an anthropocentric goal), then the contribution of watershed protection to that goal can be valued without direct information about human preferences.

The discussion above highlights the fact that there are many concepts of value and alternative ways to categorize them. Table 1 lists the various concepts of value that the committee has considered in its deliberations, categorized as preference-based versus bio-physical. While this is not the only way to categorize values, it is one that has proven useful for the committee. What follows is a brief description of the major features of each of these concepts of value. Note that these value concepts are not mutually exclusive. For example, values expressing attitudes or judgments can be based on the same self-interested utilitarian goals as those underlying the concept of economic values, or on the considerations that underlie social/civic values. Likewise, preferences that are constructed can relate to self-interested attitudes or judgments as well as expressed social/civic values.

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

<u>Preference-based Values</u>:

A. Economic Values

B. Constructed Preferences

C. Community-based or Social/Civic Values

D. Attitudes or Judgments

<u>Bio-physical Values</u>

A. Bio-ecological Values

B. Energy-based Values:

Economic values are based on individuals' preferences and assume that individuals are self-interested and that they should be allowed to value goods and services based on their judgment of the contribution those goods and services make to their own well-being or utility (the concept of consumer sovereignty, Freeman, 2003). People are assumed to be rational actors and have well-defined and stable preferences over alternative outcomes. In addition, the choice of one outcome over another is assumed to imply that the chosen outcome was judged to result in a higher level of well-being for the individual, consistent with the principle of consumer sovereignty. Economic values can include both use and nonuse values. They are based on a coherent theory of welfare economics and identify the tradeoffs that individuals are willing to make, given their income and the prices they face. They are normally expressed in monetary units and allow a comparison of the values of ecosystem services with the values of other services produced through environmental policy changes (for example, effects on human health) and with the costs of those policies.

In contrast to the assumption underlying economic values, some researchers have argued that, particularly when confronted with unfamiliar choice problems, individuals do not have well-formed preferences and hence values, implying that simple statements of preferences or willingness to pay are unreliable (Gregory and Slovic, 1997; Lichtenstein and Slovic, 2006). These authors have advocated using a structured or deliberative process as a way of assisting respondents in learning about the ecological 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 deliberative 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 this process are expected to be influenced by the process itself. Constructed values can include both individual values (reflecting self-interest) and community or social/civic values.

Community-based or social/civic values are based on the assumption that, when placed in a position of making choices about public goods (goods that when made available to one person are available to all), 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 other words, people base their choices on their conception of social preferences or community-based preferences rather than their own self-interest. In this case, individuals could place a positive value on a change that would reduce their own individual well-being (see, for example, Jacobs 1997, Costanza and Folke 1997, or Sagoff 1998).

Attitude or judgment-based values are based on empirically derived descriptive theories of human attitudes, preferences, and behavior. In contrast to economic values, preferences are not expressed in terms of willingness-to-pay (or accept) and they 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 ecosystems or services presented in the assessment. Preferences and judgments are often expressed through responses to surveys (e.g., choices, ratings or other indicators of importance). The basis for judgments may be individual self-interests, community well-being, or accepted civic, ethical, or moral obligations relevant to ecosystems and ecosystem services. Moreover, emotions and intuitions are accepted as having equal and often greater influence on value-relevant judgments and preferences than rational processes.

All of the above concepts of value are based directly on human preferences. Alternatively, bio-ecological values are defined in a way that does not depend directly on human preferences; rather, they reflect the contribution of a change to a pre-specified ecological or conservation goal (e.g., species or biodiversity preservation). As noted above, this goal can reflect intrinsic values (e.g., a biocentric or ecocentric view) or an underlying assumption or prior decision based on instrumental value (i.e., a belief that biodiversity is important for human well-being). Bioecological values are based on known or assumed relationships between targeted ecosystem conditions (e.g., biodiversity, biomass, energy transfer, and transformation) and ecosystem functions. For example, the value of changes in biodiversity could be defined in several different ways, including individual measures such as genetic distance or species richness, as well as more comprehensive measures that reflect multiple ecological considerations. What levels of bioecological measures are deemed better or worse in a given policy context may be determined solely on biological grounds (a biocentric approach) or on the basis of determined (or presumed) relationships to things people value.

Energy-based values, which reflect an energy theory of value, are based on the impact 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. While these values reflect human preferences indirectly, they were designed to provide an alternative way to define value independently of human preferences.

As noted above, the committee considered all of these various types of value in its deliberations. The committee's recommendations throughout this report reflect a recognition that not only different individuals, but also different disciplines (e.g., decision science, ecology, economics, philosophy, psychology), think of the concept of value in different ways. The committee believes that recognizing this is an important first step in valuing the protection of ecological systems and services.

2.1.3. The Concept of Valuation and Different Valuation Methods

The term "valuation" generally refers to the process of measuring either the value, or change in value, of an ecosystem, its components, or the services it provides. The committee believes that valuation should seek to characterize or measure the values actually generated by ecological systems, regardless of how well those values are currently perceived by the general population. This is a broader conception of valuation than one often used in practice, where assessments tend to focus on values currently perceived, and expressed by individuals in the general population. As discussed below, in some 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. Yet, their analysis showed that the general public had no understanding or appreciation of these services. They do have an appreciation of the higher level services or end-point services, such a clean water and aesthetics, and, of course, foods that could be derived from the system.

Regardless of the level of public understanding, valuation should seek to measure the value of the actual impact rather than simply the perceived impact. Thus, valuation can be viewed as providing a comparison of the predicted outcomes, based on the best available science, under two alternative scenarios: having a specific, proposed policy in place or maintaining the baseline or status quo. In valuing a change in ecosystem services, both the baseline before the policy change and the alternative world with the policy change must be specified. Similarly, when measuring the value of an ecosystem itself (rather than a *change* in that ecosystem), the baseline is the world without that ecosystem, a world which might be difficult to describe in any meaningful way. It is important to note, however, that 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. Basing valuation on the personal preferences of scientists rather than the general public would undermine the usual presumption that, in a democratic society, the values held individually and collectively within that society should be considered in public policy decisions, and that public involvement is central to democratic governance (e.g., Berelson, 1952).

Just as there are many types of values, there are a number of valuation methods that can be used for estimating or measuring values from ecosystems or services. Some of these methods are well developed while others are in need of further development and testing in the context of valuing the protection of ecosystems and services. Specific methods are discussed in detail in Chapter 4 and Appendix B of this report. A key tenet of valuation as defined in this report is the need to explicitly identify the type(s) of value to be measured and the appropriate method(s) for measuring those values. Methods differ on a number of dimensions, including the type(s) of value they attempt to measure (and hence their theoretical foundations and assumptions), the type(s) of metrics or outputs produced and whether they produce single or multiple metrics. These differences need to be explicitly recognized and considered as part of the valuation process.

As noted, different valuation methods express values in different ways, including monetary units, biophysical units, or indices. Economic valuation methods typically use metrics expressed in monetary units. Other social scientists and ecologists have developed measures or indices expressed in a variety of non-monetary units such as relative preference or importance ratings by samples of the general public or stakeholders, or biophysical indices calculated through expert analyses. When these measures or indices are used to make judgments about which outcomes are preferred, these measures are considered a form of non-monetary valuation. For example, bioecological valuation methods might be used to value alternative landscape management plans in terms of how well they do in conserving biodiversity, where landscape management alternatives that conserve more biodiversity are considered to be more valuable.

When multiple methods are used to capture different sources of value, the question of aggregation across methods arises. It is clear that 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. Because of their different assumptions, the 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 method and monetary estimates obtained from a citizen jury (or, alternatively, a deliberative process in which preferences are constructed) since the two are not based on the same underlying premises. Nonetheless, information about both estimates of value may be of interest to policy makers and play a key role in policy decisions. In such cases, EPA should report value estimates separately rather than seeking to aggregate across methods.

Aggregation issues also arise when considering alternative valuation methods. Some methods involve aggregation across components of value and yield a single metric of the value of a particular ecosystem or ecological change, while others yield multiple metrics of value. Valuation methods that seek to aggregate all components of value into a single metric, such as a formal economic willingness-to-pay or willingness–to-accept analysis, must weight various sources of value as part of the valuation process and report estimated aggregate values that reflect these weights. In contrast, other methods do not seek to aggregate sources of value as part of the assessment. Rather, they report the information about the various value dimensions separately and allow decision makers to weigh these components more or less formally in the process of coming to a decision. Methods that produce a single metric are not necessarily preferred to those that do not. Which approach is more appropriate or useful will in general depend on the decision context. For example, if the context requires a ranking or choice based on a single criterion (e.g., net economic analysis of benefits and costs), then a valuation approach that yields a single metric will be 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 will be preferred. Depending upon the context, this weighing and balancing might be done through political discourse or through a deliberative, decision-aiding process (see, for example, Clemen 1996; Arvai, Gregory, and McDaniels 2001; Arvai and Gregory 2003). It is important to note, however, that in either case a decision ultimately requires, explicitly or implicitly, weighing and making trade-offs among the multiple values. What varies among valuation approaches is where in the decision making process the weighing of values is done and by whom.

Finally, some valuation methods, such as economic methods and socio-psychological methods based on surveys, assume that (1) individuals know and can consistently express their preferences, and (2) individuals are well informed about alternatives, at least those they face in the assessment, and are aware of the potential consequences of the choices they make. These assumptions can be problematic when it comes to applying these valuation methods to ecosystems or services. First, for complex issues such as ecosystem protection, individuals are not likely to be aware of or fully appreciate all of the ecosystem's contributions. For example, 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.

As a result, the policy preferences or values they express through survey methods or through their behavior will reflect that incomplete information. For example, 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.

Second, as noted above, when people have limited information about and little experience with an ecosystem or service, their preferences may not be well-formed and may be subject to intentional or unintentional manipulation or bias through (e.g., as by changes in wording or framing in surveys or by labeling or placement of items in retail stores. The extent to which this is true is the subject of debate, and most likely varies with the context. (See a more detailed discussion in Appendix B.) If preferences and values regarding ecological systems and services are not well-formed, then they cannot be accurately measured or characterized by valuation methods that assume well-formed preferences. For example, individuals can have strongly held values that they find difficult, impossible or inappropriate to express in terms of monetary units. If this is so, requiring individuals to express such values in monetary equivalents (as is typical in economic valuation) may compel them to assume an individual consumer perspective that is unfamiliar or even offensive in that context. When preferences are not well-formed, survey-based methods, whether using willingness-to-pay or attitude ratings, may force the respondent to construct their preferences from more basic values in the context of the valuation itself, jeopardizing the validity of the values derived from those responses. Alternatively, and in many cases preferably, the construction of people's preferences can be made explicit and facilitated through use of a valuation method based on discourse and deliberation.

When considering alternative methods, policy makers should look for which of these methods, or what combinations, might give the best assessment of the values of ecosystems and services in particular policy contexts. In circumstances in which the individuals involved can be expected to be well informed and to have well-formed preferences for the policy options and outcomes in question, decision makers should put more weight on the stated and revealed preferences of stakeholders and the public as measured by appropriate economic and social-psychological methods. In circumstances in which individuals are likely to be ill-informed or to have ill-formed preferences, policy makers should seek to ensure that individuals expressing values have a sufficient understanding of the likely biophysical impacts of alternative policy options and their implications for ecosystems and the services they provide. For example, in specific policy contexts, using deliberative methods such as mediated modeling (see Appendix B) as part of the valuation process can help stakeholders better understand the ecological effects of alternative choices. More generally, public agencies have an obligation to aggressively pursue public education and involvement when a gap exists between public knowledge (and hence expressed preferences) and scientific understanding.

2.2. Ecological Valuation at EPA

As noted in the introduction, in contrast to previous studies, this report is focused specifically on ecological valuation within EPA. This necessitates consideration of some issues that might not be considered in more general discussions of ecological valuation. The committee recognizes that EPA operates in a variety of different decision contexts where valuation might be useful. While much of the interest in ecological valuation at EPA has focused on valuation needs in the context of national rule making, valuation can also be useful in other decision contexts as well. The need for valuation arises in different parts of the Agency for different purposes and for different audiences. Some of the needs present structured requirements for valuing protection of ecological systems and services, while needs in other contexts are less prescriptive. In addition, EPA faces institutional constraints that both influence and limit how it typically conducts valuation. This section first describes the committee's understanding of the Agency's needs and constraints related to ecological valuation. It then discusses the committee's view of how ecological valuation is typically done at EPA, using an illustrative example. The committee's observations from this section form the basis of its recommendations regarding use of an expanded and integrated approach to valuation discussed in sections 4 and 5 of this chapter.

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 represent the ecological benefits of EPA actions in analyses related to 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." A circular on "Regulatory Analysis" issued by the Office of Management and Budget (OMB) in September 2003, OMB Circular A-4, identified key elements of a regulatory analysis for "economically significant rules."[7] . In developing the *EBASP*, EPA identified the need for improved models and methods to help implement the requirements of this circular and provide better information on ecological effects that are currently not quantified or monetized.

Valuation can also be useful to EPA in a second decision context, decision-making for the remediation, restoration and redevelopment of contaminated sites. Decisions at clean-up sites, whether they involve the hazardous waste sites listed on the Superfund National Priority List that are eligible for federal cleanup funds under the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) or other clean-up 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), could be enhanced by ecological valuation that could demonstrate the potential impact of ecological services obtained from site redevelopment.

A third decision context for valuation relates to EPA's regional office partnerships with other governments and organizations where the contributions of ecological protection to human welfare are potentially important. In this context, regional offices may find valuation useful in priority setting, such as targeting projects for wetland restoration and enhancement or identifying critical ecosystems or ecological resources for attention. Regional partnership efforts may also involve assisting state and local governments, other federal agencies, and non-governmental organizations with protecting lands and land uses. In these contexts, assessment of the value of ecological protection options could aid in the decision making process and help partners communicate the value of the option chosen.

Although many of the issues and recommendations throughout this report apply across decision contexts, the committee recognizes that specific valuation needs and opportunities vary across these contexts. For this reason, Chapter 6 of this report is devoted to detailed discussions of the implementation of the report's general recommendations in these three specific decision contexts: national rule making, site-specific restoration or redevelopment, and regional partnerships. While the report discusses these three contexts explicitly, the committee also notes that ecological valuation may also be useful for EPA in other contexts and for other purposes as well. These include:

- Program assessments mandated by the Government Performance and Results Act (GPRA) of 1993;[8]
- Setting Supplemental Environmental Protection penalties for enforcement cases where those penalties involve protection of ecological systems and services;
- Choice of options for Superfund and Resource Conservation and Recovery Act cleanups that could take ecological valuation into account;
- Review of Environmental Impact Statements prepared by other federal agencies, to comply with the National Environmental Protection Act; and
- Executing ecological protection duties otherwise delegated to states, for those specific states that have not applied for or been approved to run programs on their own, such as issuing permits to protect water quality.

Although not discussed explicitly in this report, the committee believes that selected valuation methods and the approach described in this report can be useful in the above contexts as well.

2.2.2. Institutional and Other Issues Affecting Valuation at EPA

The committee recognizes that EPA must conduct ecological valuation within a set of institutional, legal, and practical constraints that affect what can be done to incorporate ecosystem values into policy evaluations. 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.[9] 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 OMB. The interviews proved equally beneficial in leading to a better understanding of strategic planning, performance reviews, regional analysis, and other situations where the Agency has the need 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. The Agency also develops rules to meet court-imposed deadlines. Despite the rigidity of this process, there is no single way in which Agency analysts assess the benefits of protecting ecosystems. Practices vary considerably across program offices, reflecting differences in mission, in-house expertise, and other factors. Program offices have different statutory and strategic missions. The organization, financing, and skills of the program offices differ enormously. The National Center for Environmental Economics (NCEE) is the Agency's centralized reviewer of economic analysis within the Agency.[10] Nonetheless, 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. This is related to court-imposed deadlines on the rule process, as well as to intervening requirements related to the collection and analysis of new data. The scientific community is accustomed to much longer time horizons for their analyses. Unfortunately, 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. This hurdle is required by the Paperwork Reduction Act and imposes the review responsibility on OMB, adding a significant amount of time to the assessment process. With a year or two at most to conduct a study, this kind of review significantly limits the scope of analysis the Agency can conduct. In particular, the Agency must by necessity rely heavily on transferring both ecological and social values information from previous studies to the new analysis.

Another issue is OMB's role in defining or directing ecosystem valuation exercises at EPA. Among its activities, OMB acts as an oversight body that reviews EPA's economic benefit analyses. EPA is required to provide sufficient justification for its claims regarding the economic benefits of its actions, including any analyses of willingness to pay or willingness to accept related to ecological protection. As noted above, EPA has been given explicit guidance by OMB in the Circular A-4. 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).[11] 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. Little guidance is provided, however, 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 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. The committee recognizes the importance of consistency in the methods used for valuation, but also sees the limitations resulting 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 (U.S. Environmental Protection Agency, 2003; also see U.S. Environmental Protection Agency 2006). This raises the question of how the Agency (and perhaps OMB) defines "novel." Moreover, the novelty standard ironically creates a clear incentive to avoid conducting innovative analyses since the fastest, cheapest option is to avoid review altogether.

Finally, the committee notes the importance of the organization of assessment science within the Agency. The Agency relies, to varying degrees, upon a variety of offices to develop assessments, including individual program offices and NCEE. It is not clear what form of organization is most effective. A further complication is the availability and location of data used to support ecological valuation. To resolve this issue, data that are housed within individual program offices should be made public and readily shared with other offices.

The *EBASP* 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 efforts. Nonetheless, 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, namely, the environmental and economic benefits analysis that EPA prepared in support of new regulations for Concentrated Animal Feeding Operations (CAFOs) (U.S. Environmental Protection Agency 2002a).[12,13] 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. The final rule was published in December 2003. The new CAFO regulations, which cover over 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” under Executive Order 12866, EPA was required to assess the economic costs and benefits of the rule.[14] An intra-agency team at EPA, including economists and environmental scientists, worked together with the U.S. Department of Agriculture on the economic benefit assessment. Prior to 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.[15] Using various economic valuation methods, EPA provided monetary quantifications in its CAFO report for seven benefit categories.[16] Approximately eighty-five percent of the monetary estimate of the benefits that were quantified by EPA was attributed to recreational benefits. According to Agency staff, EPA’s analysis was driven by what it 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. The table in the Executive Summary listed a variety of non-monetized contributions[17] but designated them only as “not monetized.” EPA did not try to 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" place-holder 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.

Although it involved considerable effort, the CAFO economic benefits assessment illustrates a number of limitations in the current state of ecosystem valuation at EPA. First, as noted above, in implementing the Executive Order, the CAFO analysis did not provide the full characterization of ecological contributions to human welfare using quantitative and qualitative information, as required by the OMB Circular A-4. 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 (transfer of economic benefits).[18] These benefits did not include all of the major ecological contributions to human welfare that the new CAFO rule would likely generate, nor all of the contributions that generated public support for the new rule.[19] The Circular requires that an assessment identify and characterize all of the important benefits of the proposed rule, not simply those that can be monetized. By focusing only on a narrow set of contributions that could be readily monetized, the CAFO analysis and report understate the total benefits of the rule change and distort the rationale supporting the final rule. An unfortunate effect of this presentation is to suggest to readers that the benefits that were monetized constitute the principal justification for the CAFO rule.[20] In this case the focus on monetized benefits did not affect the outcome of the regulatory review. It is certainly possible, however, that 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 emphasized 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 make it very difficult for the Agency to conduct new surveys or studies and virtually force 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 the underlying limitations due to uncertainty and data quality.

Third, EPA apparently did not embark on a comprehensive effort to model the rule's ecological impacts. The report presents merely 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 thorough analysis of the rule's ecological impacts. As a consequence the analysis was unduly directed by Agency presumptions (or discoveries) about the availability of relevant data and the likely opportunities to quantify effects precisely and to link and monetize associated economic benefits. This was undoubtedly driven in part again by the time pressures of putting together the regulatory impact analysis. Without a comprehensive modeling effort at the outset, however, EPA had insufficient insight into the potential economic benefits and other values that needed to be analyzed and estimated. Developing integrated models of relevant ecosystems from the start of a valuation project would also help in identifying important secondary effects, which frequently may be of even greater consequence or value than the primary effects.[23]

Fourth, the CAFO analysis clearly 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, both in terms of ecological impacts and associated values. It is, however, possible (and desirable) to make use of 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. 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.

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 impacted by the proposed regulations and to determine the regulatory effects that are likely to be of greatest value. Through this added effort, valuations would be more likely to include the most important impacts.

Sixth, EPA failed to follow its own advice regarding the use of outside peer-reviewed data, methods, and models. While the Agency appropriately emphasized peer review in its analysis and report, EPA did not seek peer review in deriving values for the CAFO rule. Once again, this shortcoming is undoubtedly a function of time and resource constraints. It should be noted, however, that peer review, especially early in the process, could help EPA staff identify relevant and available data, models, and methods to support its analysis. In addition, it could provide encouragement, direction, and sanction for more vigorous and effective pursuit of ecological and human well-being effects associated with the proposed rule. An effective method is to review not only individual components of the analysis (e.g., watershed modeling, air dispersal, human health, recreation, and aesthetics) but the overall analytic scheme as well.

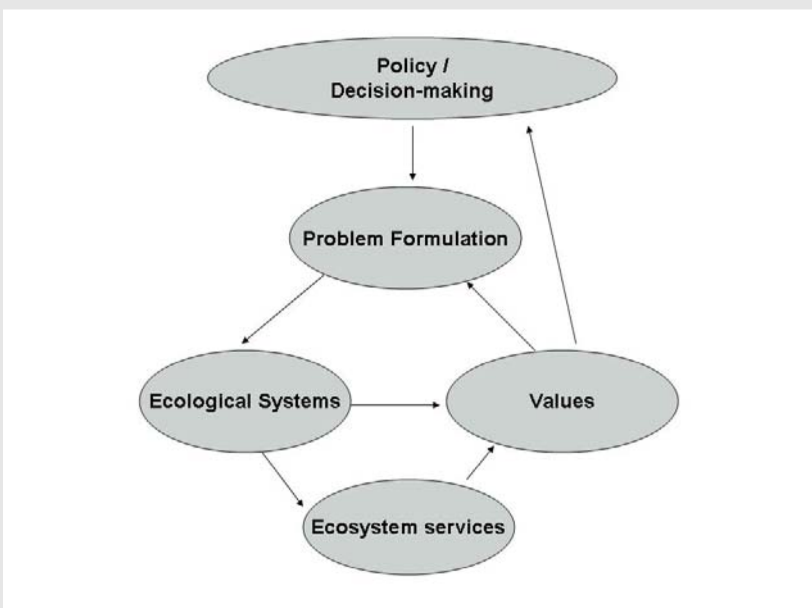
Finally, EPA's analysis and report closely adhered to the requirements of Executive Order 12866, which provided the proximate reason for preparing the analysis and report. Nevertheless, when EPA prepares a benefit assessment specifically to comply with Executive Order 12866, the Agency need not limit itself to the goals and requirements of the Executive Order, which directs EPA to conduct an "analysis" and "assessment" of the "benefits anticipated from the regulatory action" and, "to the extent feasible, a quantification of those benefits." The Executive Order, to be clear, does not preclude EPA from adopting broader goals. By adopting a narrow focus, the CAFO report failed to consider the broader purposes served by a benefit assessment. 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 have an incentive to consider it more regularly.

2.3. An Integrated and Expanded Approach to Ecosystem Valuation: Key Features

The CAFO example discussed in the previous section 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 impacts of EPA actions, focusing on the impacts of greatest concern to people and integrating ecological analysis with valuation. This section describes an 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 the provisions of Circular A-4, as well as in decisions regarding regional and local priorities and activities. A more detailed discussion of the implementation of the approach and the framework for specific decision contexts is provided in subsequent chapters of this report.

As noted, the committee focused on valuation in EPA contexts where there is an environmental protection decision to be made. The major components of the ecological valuation process proposed by the committee are depicted in Figure 1.

Figure 1: Components of Ecological Valuation



The committee's proposed approach for implementing the valuation process has three key, interrelated features: a) early consideration of effects that are socially important; (b) predicting ecological changes in value-relevant terms; and (c) drawing on a suite of methods for characterizing 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 impacts or contributions to human welfare that are likely to be most significant or of greatest importance to people, whether or not the impacts are easily measured, monetized or widely recognized by the public. These could include changes in the ecosystem itself that people value directly, or the resulting changes in the ecological services provided by a system. Information about the ecosystem services or characteristics that are of greatest concern needs to be obtained early in the valuation process so that efforts to quantify and characterize values can be focused on the related ecological changes. The importance of a given change will depend on the magnitude and bio-physical importance of the effect and on the resulting importance to society.

Identifying socially relevant effects requires a systematic consideration of the many possible sources of value from ecosystem protection and an identification of the types of values that provide the impetus for a particular policy change. This focus will likely lead to an expansion of the types of services to be characterized, quantified, or explicitly valued. For example, even in the context of national rule making, a specific contribution to human welfare should be included as part of an overall valuation whether or not it is possible to monetize that benefit in terms of willingness to pay or willingness to accept; if there is evidence that it is important to people, the benefit should be included as a key component of the total benefits, complete with a detailed and careful (even if not monetized) characterization of its importance. Previous assessments have often focused on what can be measured relatively easily rather than what is most important to society. This diminishes the relevance, usefulness, and impact of the assessment.

An obvious question is how to assess the likely importance of different ecological impacts prior to completion of the valuation process. In fact, a main purpose of conducting a thorough valuation study is to provide an assessment of this importance. Nonetheless, in the early stages of the process, preliminary indicators of likely importance can be used as screening devices to provide guidance on the types of impacts that are likely to be of greatest concern. Relevant information can be obtained in a variety of ways. Examples 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[25] or the use of focus groups or workshops, comprised of representative individuals from the affected population and relevant scientific experts, can help to identify ecological changes for the specific context of interest.

In eliciting information about what matters to people, it is important to bear in mind that people's preferences depend on their mental models (i.e., their understandings of causal processes and relations), the information that is at hand to influence their understanding, and how that influence occurs. As noted previously, expressions of what is important (e.g., in surveys) or of the tradeoffs people 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. The ecological valuation process can in fact be used as a mechanism for educating the public about the services provided by ecological systems and how those services are affected by EPA actions, thereby narrowing the gap between expert and public knowledge of ecological effects.

2.3.2. Predicting Ecological Changes in Value-relevant Terms

The second major component of the C-VPESS process is the need to predict ecological changes in terms that are relevant for valuation. This requires both the prediction of bio-physical impacts of EPA actions using ecological models and the mapping of those impacts into changes in ecosystem services or features that are of direct concern to people. Ideally, this would be done using an ecological production function that is specified and parameterized for the ecosystem and associated services of relevance in the assessment.

Numerous mathematical models of ecological processes and functions have been developed. These models cover the spectrum of biological organization and ecological hierarchy (e.g., individual level, the population level, the community level, the ecosystem level, and the level of the global biosphere). They can be used to predict ecological impacts associated with a given EPA action at different temporal and spatial levels. Some have been developed for specific contexts, such as particular species or geographic locations, while others are more general.

Ecological models provide a basis for estimation of 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 of these ecological models have been developed to satisfy research objectives and not EPA policy or regulatory objectives. This poses challenges when using these models to assess the contributions of EPA actions to human welfare.

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 impacts that would stem from 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.

Ecological models additionally need to be appropriately parameterized for use in policy analysis. Numerous detailed ecological studies have been conducted at various levels, for example, at Long-Term Ecological Research Sites (Farber et al. 2006). These could provide a starting point for parameterizing policy-relevant models. A key challenge is to determine whether (or 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 other words, to what extent are estimated parameters adaptable from one context to another in estimating the contributions to human welfare and values associated with EPA actions? In many cases, data do not currently exist to parameterize existing models so that they could be used 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, it would be extremely valuable to have a central depository that EPA could draw on for this information.

The final, but perhaps most important, challenge is translating the changes predicted by standard ecological models into changes 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, including 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). Most ecological models, however, are not currently designed with this objective in mind. In particular, they do not translate bio-physical impacts in ways that lay individuals can understand, or in ways that reflect how those changes are of value to them.

2.3.3. Drawing on Multiple Methods for Characterizing Values

Given predicted ecological changes, the value of these changes needs to be characterized and, when possible, measured or quantified. As noted above, a variety of valuation methods exist. The C-VPESS approach envisions drawing on a wider range of methods than EPA has typically utilized to capture a broader array of values. It recognizes that there are many sources and types of value and many valuation methods. In addition, 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. Given the array of values and methods, a key tenet of the valuation process proposed by the committee is that each valuation process should include a conscious choice regarding the type(s) of value to assess and the appropriate methods for assessing those values. However, this expanded approach should include only those methods that meet accepted scientific standards of precision and reliability, are appropriately responsive to relevant changes in ecosystems and their services, and are properly related conceptually and empirically to things people value. The suite of methods used could vary with the specific policy context, due to differences across contexts in information needs, legal and regulatory requirements, the underlying sources of value being captured, data availability, and methodological limitations.

Through expanded methodology EPA can better capture the full range of contributions stemming from ecosystem protection and the multiple sources of value derived from ecosystems. In addition, where resources allow, 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 those estimates. Certainly, the possibility exists that the application of multiple assessment methods to an environmental decision problem could suggest conflicting information about relative values. It then would be essential to try to ascertain the source of the differences. In some cases, they may be due to the application of methodologies (e.g., eliciting values from different population groups or samples), or study limitations (e.g., inappropriate application of techniques or interpretation of results), or the inherent uncertainty in estimating values that results from data limitations, theory limitations, and randomness (see related discussion in Chapter 5). In other cases the differences may reflect the fact that the alternative methods are capturing fundamentally different sources, components, or concepts of value. In any case, information regarding the similarities or differences of alternate assessment methods, including their conclusions about the value of an ecological change, would be an important input into a policy decision.

The committee evaluated a number of different methods for characterizing values (described in detail in Chapter 4 and Appendix B). These include not only economic valuation methods (the usual focus of EPA valuation) but other methods that could be used to value ecological changes as well. These include social and psychological methods, assessments based on voting and other group expressions of social or civic values, and assessment methods based on indicators or bio-physical rankings that are less directly dependent on human preferences and value judgments.

Underlying many valuation methods (including preference-based methods) are metrics that are primarily bio-physical or socio-economic indicators of impact. These include such indicators as acres of habitat restored, the number and characteristics of individuals or communities affected, the likely injuries avoided, and the duration of impact. These metrics can provide useful information in at least three ways. First, in some cases, they can be used directly in policy decisions. For example, decisions based on human impact criteria (e.g., protection of children's health) or environmental goals (such as promotion of biodiversity) may draw directly from these measures as indicators of the appropriate policy choice. Second, they might be used as a proxy for some component of the contributions of ecosystem protection to human welfare, when that component cannot be readily valued. As noted earlier, in contexts requiring benefit-cost analyses, the OMB Circular A-4 requires that benefits be quantified when they cannot be monetized; these metrics provide potentially useful forms of quantification in such circumstances. Finally, even when human impacts can be valued, these metrics provide information about human impacts that would presumably be relevant in the determination of the associated value of the ecological change. Thus, in all of these contexts, estimates of the impact of the ecosystem change on human populations are needed.

In contexts where monetary metrics are required or desired and the necessary data and methods exist, the impact of the ecological change on the provision of some services to human populations may be translated into a monetary equivalent of that change using standard economic valuation techniques. For some valuation contexts, economic methods for valuing changes are relatively well developed. As noted previously, existing EPA ecological valuation efforts, such as the *EBASP* and the Science to Achieve Results (STAR) Grant program, have focused on valuing changes using economic methods. These methods are designed to estimate the economic benefit or cost of a given ecological change using a willingness-to-pay or willingness-to-accept measure of the utility equivalent of that change. They have been applied to the valuation of ecosystem services in a number of studies that have produced results that are useful for policy evaluation and decision making.

As in the CAFO study, however, economic valuation methods have generally been applied to a relatively narrow set of services. In some cases, these services might not have been those that people are most concerned about protecting. While there are continuing discussions about the role of economic valuations in principle, it is unlikely as a practical matter that all of the important benefits (or costs) of a change in ecological conditions will be sufficiently captured by economic valuation methods. For this reason, the EBASP calls for exploring "supplemental" approaches to valuation.

The valuation approach proposed by this committee calls for a more prominent role to be played by a variety of methods for characterizing values than envisioned in the EBASP. This is a practical alternative for use when economic methods cannot fully capture contributions to human welfare because of limitations in data or other knowledge-based gauges. It is also a means of capturing many components of value that are not fully reflected in those value measures that are based solely on economic measures of willingness to pay or willingness to accept. Including other scientifically-based assessment approaches that can be applied along with, or in place of, economic assessments will allow EPA to more fully represent the contributions of ecosystems and their services to human well-being.

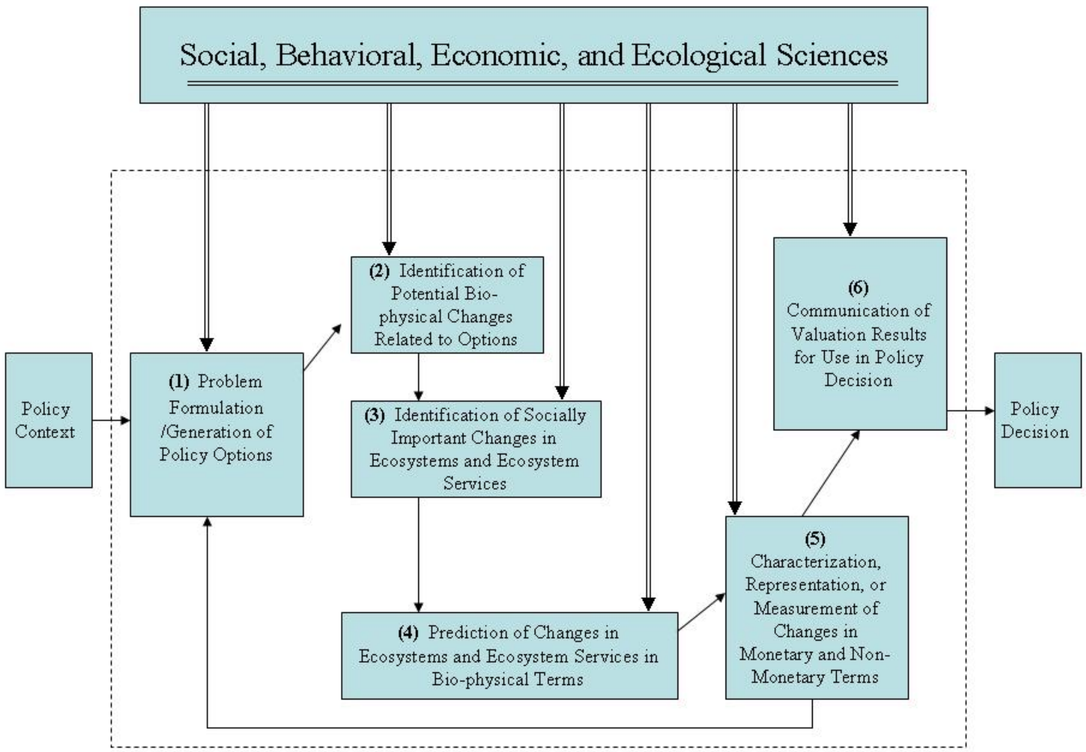
2.4. Steps in Implementing the Proposed Approach

The previous subsections provide 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 2:

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

Although Figure 2 depicts these steps as sequential, in practice interactions and iterations across steps are likely during the process. For example, information about the value of changes in ecosystem services stemming from 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 captured in initial public/stakeholder processes.

Figure 2: Process for Implementing an Expanded and Integrated Approach to Ecological Valuation



As depicted in Figure 2, the implementation of the approach is contingent upon the specific policy context. 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 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 is 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 in which the assessment is cast is therefore a key influence on the appropriateness of data, models and methods.

Figure 2 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 impacts that matter to the estimation of the value of those impacts. Thus, instead of having ecologists work independently from economists or other social scientists, this approach envisions collaborative work across disciplines. The result is an analysis that identifies the impacts that are of greatest concern to society in a manner that is informative for valuation. 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/bio-physical effects that are currently underrepresented in value assessments.

Figure 2 additionally suggests a structure that in many ways parallels the Agency's Framework for Ecological Risk Assessment (U.S. Environmental Protection Agency Risk Assessment Forum 1992; U.S. Environmental Protection Agency Risk Assessment Forum 1998). This framework underlies the ecological risk guidelines developed by EPA to support decision making that is intended to protect ecological resources (U.S. Environmental Protection Agency Risk Assessment Forum 1992). The committee views ecological valuation as a complement to ecological risk assessment. Both processes begin with an EPA decision or policy context requiring information about ecological effects. Following that is 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 effects of an EPA action or decision under consideration. They also both ultimately use this (and related) information in the evaluation of alternative decisions or policy options.

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, as depicted in both Figure 1 and Figure 2, ecological valuation seeks to characterize the importance to society of predicted ecological effects by providing information on the value that society places on either the ecological improvements or the loss it experiences from ecological degradation. By incorporating human values, ecological valuation is closer to risk characterization than risk assessment, and many of the principles that should govern risk characterization outlined in the 1996 NRC Report *Understanding Risk: Informing Decisions in a Democratic Society* would 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.

In contexts involving complex ecological impacts and tradeoffs, deliberative processes have been successfully used as a means of identifying stakeholder concerns, educating stakeholders about the ecological impacts of alternative policy choices, eliciting information about stakeholder values, and ultimately describing and possibly evaluating tradeoffs. Examples include the decision-aiding processes developed by decision scientists (refs) and mediated modeling, in which stakeholders participate in the development and interactive use of simulation models of complex ecological systems to compare and evaluate policy options (refs). The process in Figure 2 has a structure that parallels these deliberative processes and shares many of the same goals. In some contexts (e.g., site-specific and regional valuations), a single, holistic deliberative process could be applied in a very similar way to accomplish the entire valuation process. In other contexts, implementation of the valuation process could involve elements of a deliberative process at different points in the overall value assessment (e.g., early on when identifying impacts that are socially important or educating the stakeholders about potential impacts), coupled with the use of non-deliberative methods at other stages of the process. In either case, the goals and overall structure of this report's proposed valuation approach closely parallel those of the deliberative processes that have been developed and successfully used in a number of contexts.

2.5. Conclusions and Recommendations

Ecosystems play a crucial role in supporting life as we know it. They provide a wide array of services that directly or indirectly support or enhance human populations. In addition, they can be valued in their own right, for non-anthropocentric reasons stemming from ethical, religious, cultural or biocentric principles. Part of 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. However, to date ecosystem impacts have received relatively limited consideration in EPA policy analysis, which has typically focused on human health impacts. It is imperative that EPA improve its ability to value ecosystems and their services to ensure that ecological impacts are adequately considered in addition to human health impacts in the evaluation of EPA actions at the national, regional and local levels.

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 impact of EPA actions on ecological systems and the services derived from them and the difficulty of quantifying, measuring, or characterizing the resulting contributions and associated values. The presumption that contributions need to be monetized in order to be carefully characterized also restricts the range of ecological impacts that are typically considered in EPA analyses, particularly at the national level.

The committee views EPA's efforts to improve its ability to value ecological systems and services as very important and timely. The committee recommends that the Agency move toward covering an expanded range of important ecological effects and human considerations using an integrated approach. Such an approach would:

a) Expand the range of ecological changes that are valued, focusing on valuing the ecological changes in systems and services that are most important to people and recognizing the many sources of value, including both instrumental and intrinsic values;

b) Highlight the concept of ecosystem services and provide a mapping from changes in ecological systems to changes in services or ecosystem components that can be directly valued by the public; and

c) Utilize an expanded set of methods for identifying, characterizing, and measuring the values associated with these changes.

Such an approach would, from the beginning and throughout, involve an interdisciplinary collaboration among physical/biological and social scientists, as well as direct and early involvement and input from the public or representatives of individuals affected by the ecological changes. In implementing the approach, EPA should recognize the multi-dimensional nature of value and make a conscious choice regarding the type of value(s) it wants to assess and the appropriate methods for assessing those values. In addition, the Agency should be transparent about the reasons for choosing specific valuation methods and communicate clearly what the methods that it chooses measure and do not measure.

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. In addition, EPA can better use the ecological valuation process as a mechanism for educating the public about the role of ecosystems and the value of ecosystem protection. The remainder of this report develops the ideas embodied in the C-VPESS integrated value assessment system through a more detailed look at how the approach could be applied.

3 BUILDING A FOUNDATION FOR ECOLOGICAL VALUATION: PREDICTING EFFECTS ON ECOLOGICAL SYSTEMS AND SERVICES

Chapter 2 of this report presented an overview of an integrated and expanded approach to valuing ecological changes that result from EPA actions or decisions. The approach was described in general terms. This chapter focuses on one part of that approach, namely, predicting ecological changes in value-relevant terms. No matter what valuation method is used, the valuation process requires an assessment of the impact of a given EPA action on ecosystems and the services they provide. To conduct the assessment, a prediction of the bio-physical impacts is needed in terms that are relevant for ecological valuation. To the extent possible, this prediction should be quantitative. In the context of national rule making, quantification is necessary for values that will be monetized and is needed (as stated in Circular A-4 from the Office of Management and Budget), even for values that cannot be readily monetized. In every context where the need for valuation arises, information about the magnitude of effects will be a key component of value assessment.

This chapter begins with a discussion of the importance of developing an initial conceptual model of the relevant ecosystem and its services designed to guide the entire valuation process. It then turns to a discussion of how to operationalize the conceptual model, which will often involve the use of multiple specific ecological models. In this context, the key role played by the concept of an ecological production function is discussed. The discussion highlights the challenges that currently exist in trying to implement ecological production functions in specific valuation contexts. These include challenges associated with understanding and modeling the relevant ecology, clearly identifying the relevant ecosystem services, and mapping ecological changes into changes in the ecosystem services of interest. To a large extent, these challenges stem from the underlying complexity and site-specificity of ecosystems. The chapter then discusses some strategies for addressing these challenges and providing the ecological science necessary to support valuation. A final section summarizes conclusions and recommendations.

3.1. <u>The Road Map: A Conceptual Model</u>

Formulation of a conceptual model is a key first step in predicting the effects of EPA actions on ecological systems and services. The committee recommends that EPA start each ecological valuation by developing a conceptual model of the relevant ecosystem(s) and associated services. The conceptual model should be constructed at a general level to provide a road map to guide the valuation process. As a result, the model should be context-specific. More detailed analyses involving ecological production functions should follow to identify the key interactions, predict specific ecological impacts, and calculate the ecological values. This will often require the use of ecological or valuation-related models with a narrower focus (see section 3.3). The conceptual model's basic purpose is to guide the process by providing a framework for integrating these more specific analyses into the overall valuation exercise.

Key features of the conceptual model are a clear identification of the relevant functional levels of the ecosystem, the inter-relationships between ecosystem components, and how they contribute to the provision of ecosystem services, either directly or indirectly. An example illustrating some aspects of ecosystem services related to nutrient pollution is provided in Figure 3, adapted from Covich et al. (2004).

Figure 3: Illustration from Covich et al., 2004, Showing Relationships of Major Functional Types to Ecological Services

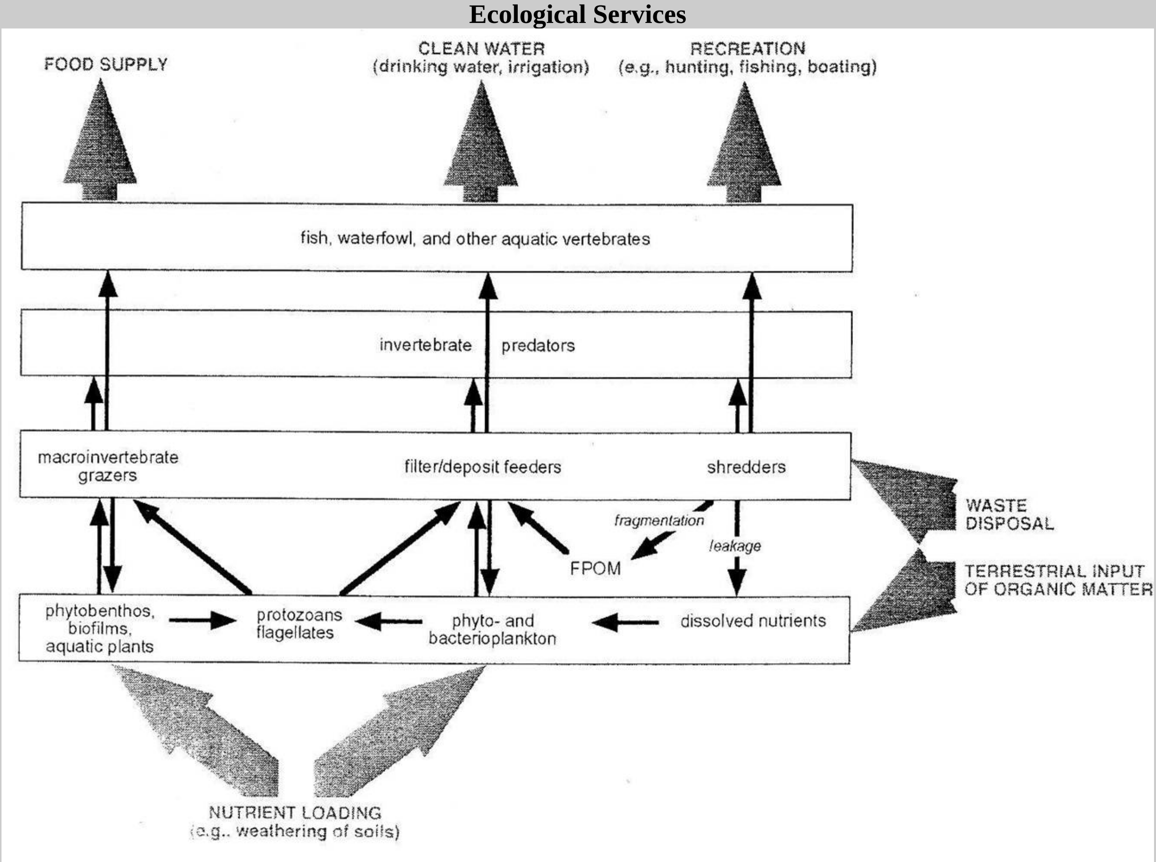


Figure 3 highlights the need for the conceptual model to include both information about the underlying ecology and a link to ecological services that are of importance to society. There is a need to include, for example, the impacts of environmental stressors, such as waste disposal on organisms at different trophic levels, the key interactions among species at different levels, and the changes at different levels that affect ecological services, such as the food supply, clean water, or recreation.

Ecologists, not surprisingly, often focus on the underlying ecological aspects (depicted in the lower part of Figure 3), while valuation experts tend to focus on the later, value-oriented stages of the process, starting with ecosystem services (i.e., starting at the top of Figure 3). A key principle of the C-VPESS approach 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 impacts, including modeling of ecosystem impacts, is insufficient unless those impacts are mapped to changes in ecological services or system components of importance to people. It is similarly insufficient to conduct valuation exercises that do not reflect the key ecological processes and functions affected by the decisions under consideration. Both steps are essential, and 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.

As envisioned here, the development of the conceptual model is a significant task that deserves the attention of EPA staff throughout the agency, experts in the relevant topics of consideration (from both the bio-physical and social sciences), and the public. Involving all constituents at this stage will enhance transparency, provide the opportunity for more input and better understanding, and ultimately give the process more legitimacy. Participatory methods such as mediated modeling (see Appendix B) can play a valuable role in the development of the conceptual model.

In addition, the process for development of the conceptual model should allow for iteration and possible model changes and refinement over time. For example, an action at a local site may initially be considered to have only local ecological effects, but, once the stressors are considered, it may become apparent that effects reach to more distant regions downstream or down wind, requiring a change in the conceptual model. Similarly, as the stressors are identified in the context of the relevant ecological system, the conceptual model may need to be modified to incorporate additional stressors. As an example, a relatively nontoxic 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 conceptual model, the process for developing and completing it, and the decisions that are embedded within it should all be a part of the formal record.

3.2. Operationalizing the Conceptual Model: The Role of Ecological Production Functions

While the conceptual model serves as a guide for the overall valuation process in a specific context, 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 changes in ecosystem components or services that can then be valued. To operationalize the conceptual model, it is necessary to map or describe: a) how the EPA action will affect the ecosystem, b) how the change in the ecosystem will lead to a change in the provision of ecosystem services; and c) how people value that change in ecosystem services. For the first step, it is necessary to describe how change in stressor or in some other environmental factor that could be altered by the EPA action results in changes in important aspects of ecosystem structure or function. Does the change in stressor cause a species to disappear or change in abundance? Does it result in a change in biogeochemistry? For any important changes, a quantitative relationship must be determined.

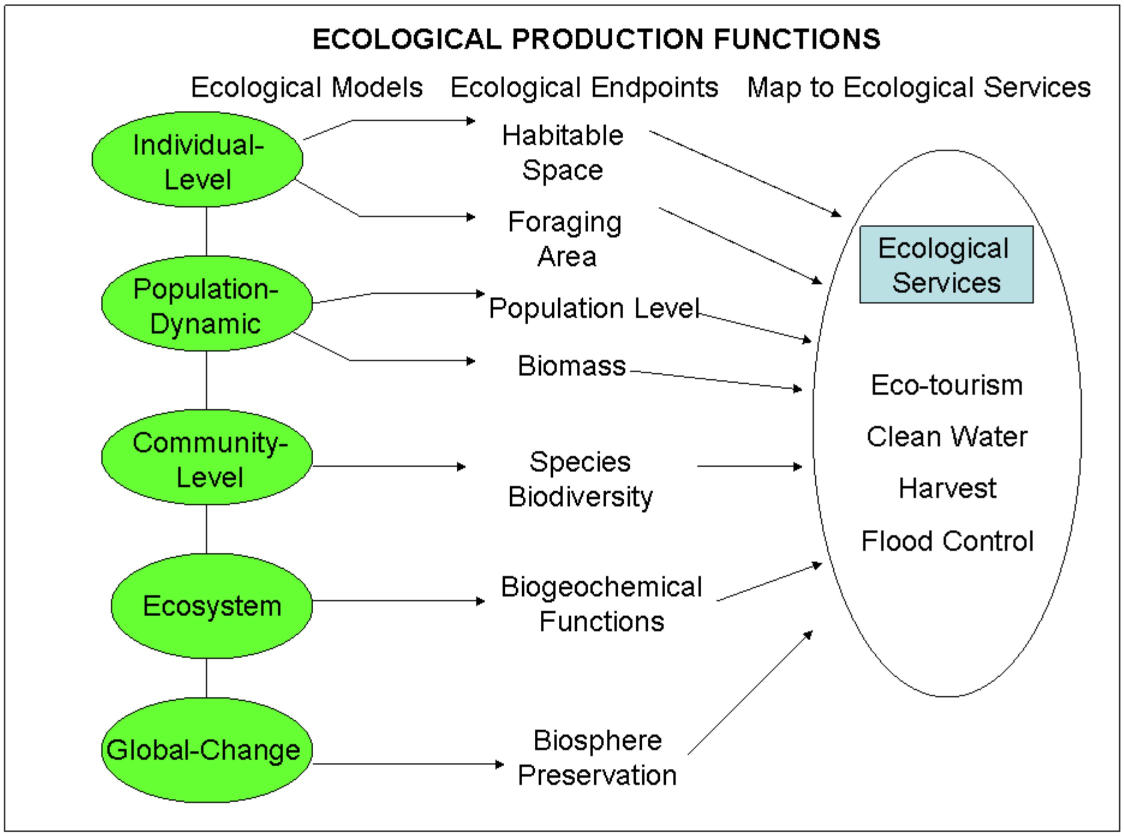
A fundamental concept for describing the second step in this mapping is the ecological production function. Ecological production functions are similar to the production functions used in economics to define the relationship between inputs (labor, capital equipment, raw materials) and outputs of goods and services. For example, a farmer uses inputs of seeds, fertilizer, labor, and equipment to produce outputs of agricultural crops.[26]

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. Expanding on the farming example, in addition to the inputs mentioned above, there are ecological inputs provided by ecosystems, such as soil nutrients, rainfall, and pollinators, that have a major impact of crop production. However, crop production is not the only ecosystem service provided by these inputs. Beyond crop production, additional important outputs (i.e., ecosystem services) provided by agriculture include the effect of farming operations on carbon sequestration, water quality, habitat of pollinators and other species. An ecological production function could be developed for each of these services separately. Alternatively, to the extent that some services are linked (e.g., produced jointly or in competition), a multiple-output function could be developed to capture these linkages.

Thus, ecological production functions generate an accounting of the relationship between a broad suite of inputs and a broad suite of goods and services from ecosystems. Coupled with information about how changes in stressors affect the ecological inputs, these functions can be used to predict the changes in ecosystem services that will result from alternative Agency actions or management scenarios. In addition, they allow answers to questions such as: How can forests be managed to reduce catastrophic damage from fire? What kinds of marine reserves lead to larger fish populations? How much more wetland is needed to recharge sub-surface aquifers used for irrigation?

Implementing the concept of an ecological production function requires: a) characterization of the ecology of the system, b) identification of the ecosystem services of interest; and c) development of a complete mapping from the structure and function of the ecological system to the provision of the relevant ecosystem services. Figure 4 provides a graphical representation of how this concept can be implemented. The left-hand side represents ecological models at various organizational levels that are used to predict ecological endpoints (see further discussion of endpoints below). While these are important components of an ecological production function, it is not the complete function. An ecological production function requires that the predictions regarding the levels or changes in these ecological endpoints be translated into corresponding predictions regarding ecosystem services.

Figure 4: Graphical Depiction of Ecological Production Functions



Rapid progress is being made in understanding ecological production functions for certain ecosystem services. One such service is pollination. Animal pollination is essential for the production 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). One application of such models is to link changes in ecosystem conditions to the level of pollination of agricultural crops and their yields. Empirical studies using this approach 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 where 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 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 quantify above ground carbon stores in various types of ecosystems such as forests fairly accurately (e.g., Birdsey 2006, Smith et al. 2006, U.S. Environmental Protection Agency Office of Atmospheric Programs 2005), while 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, U.S. Environmental Protection Agency Office of Atmospheric Programs 2005).

Despite this progress, our current understanding of ecological production functions for the complete range of services from ecosystems remains limited (Balmford et al. 2002, Millennium Ecosystem Assessment 2005, National Research Council 2004). Although many ecological models exist (see further discussion below), most of these are not designed to link changes in ecological inputs or endpoints to changes in ecosystem services. The following section discusses some of the challenges in developing ecological production function models for use in ecological valuation.

3.3. Challenges in Implementing Ecological Production Functions

3.3.1. Understanding and Modeling the Underlying Ecology

As noted above, operationalizing the conceptual model using ecological production functions requires a fundamental understanding of the components, processes, and functioning of the ecosystem(s) that underlie and generate the ecosystem services. In other words, analysts must have a strong understanding of the underlying ecology. While much is known about ecological systems, current knowledge is still very incomplete due in large part to the fact that 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, which in turn provide pathways for the movement of water and air. Other kinds of organisms shred the organic material into smaller units that are in turn utilized by microbes. These microbes then release nutrients in a form that can be utilized by higher plants for their growth or in a dissolved form that is hydrologically transported from the immediate site into the water table or stream. Other groups of specialized microbes may release various nitrogen gases directly to the atmosphere. Thus, the nature of the soil organisms and the products that they utilize, store, or release all help to regulate the biogeochemistry of the site as well as its hydrology, productivity, and carbon storage capacity. As Figure 4 suggests, these kinds of functions relate to the services that people more readily appreciate and value, such as the natural processing of wastes and the provision of clean water (Wall 2004). This evaluation requires an understanding of the complex ecological relationships that contribute to these services.

Complexity also stems from the fact that ecological effects may persist for different periods of time (e.g., carbon dioxide in the atmosphere vs. acute toxic exposures to hazardous chemicals), affecting both the temporal and spatial scales that are relevant for any analysis. Numerous studies including EPA's regional analyses, risk analyses and the Environmental Monitoring and Assessment Program (EMAP) provide guidance in identifying the proper boundaries and time scales for the ecological system under study as well as the ecosystem characteristics, stressors and endpoints (Harwell, et al., 1999, Young and Sanzone, 2002).

Because of the complexity of most ecosystems, models are used to organize information, elicit the interactions among the variables represented in the models, and reveal outcomes when run under different sets of assumptions or driving variables.[27] 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). As shown in Figure 4, these models frequently focus on specific ecological characteristics, such as populations of one or more species or the movement of nutrients through ecosystems, and can cover the 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 could predict effects on the levels of various chemical elements in soils, ground water, and surface waters. A terrestrial carbon cycle model could project changes in plant growth and in carbon sinks or sources. Population and community models would project changes in specific animal and plant populations that are of concern.

Primers on ecological theory and modeling such as Primer of Ecological Theory (Roughgarden 1998b) can provide a starting point for identifying available models. Some models are statistical, while 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.

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 as reliable as the data the models use. The dynamism of a system adds to the challenge of modeling, as does the non-linear responses of system components. The model outputs are estimates with known, or sometimes unknown, levels of statistical uncertainty. No ecological model can include all possible interactions. Some ecological models explicitly or implicitly incorporate human dimensions, but most focus primarily on ecological functions. Models additionally 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 can lead species to "reshuffle into novel ecosystems unknown today" for which there are no analog, current models will not predict theses impact (Fox 2007).

Finally, the applicability - and to some degree the formulation - of ecological models, is frequently constrained by the insufficiency of data to build and test the models. Even when a full theoretical model of the ecosystem exists, that model will need to be parameterized for the specific valuation context of interest. However, parameterization is generally difficult because of the complexity of ecological systems and their dependence on an array of site-specific variables. Many ecological models, as a result, are site specific. Moreover, the relatively large amounts of site-specific data required to build and parameterize models means that their transferability is limited, either because the model has been developed using spatially constrained data or because inadequate data are available at secondary sites with which to drive or parameterize the model. This site-specificity may significantly limit the models' applicability to the spatial and temporal complexities required in valuing ecological services, especially at regional and national scales.

Despite these caveats, utilizing ecological models provides a means of incorporating the best available scientific knowledge of how ecosystems will respond to a given perturbation and the sensitivity of various ecosystem components. Hence, they provide an essential way to represent and ecological production functions and allow them to be analyzed. Guided by the conceptual model, ecological models should be utilized to quantify the likely effects of an Agency action on the ecosystem and how this will result in changes in the provision of ecological services. The committee recommends that all ecological valuations conducted by EPA be sufficiently supported by ecological modeling and ecological data designed to provide insight into or estimates of the likely ecological impacts associated with major alternatives being considered by decision makers. The committee recognizes that EPA is 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 make effective ecological modeling one of its priorities.

Since many ecological models exist to choose among and for any particular valuation process a variety of ecological models might be utilized, the Agency will often be faced with selecting one or more predictive models for use in operationalizing the conceptual model. The appropriate choice of models, and the availability and appropriateness of supporting databases, will be different depending on the scale of analysis (e.g., local vs. national) and the precision of the analysis related to the relevant policy decision. The committee recommends that EPA identify clear criteria for selection of ecological models for use in ecological valuation and that the Agency apply these criteria in a consistent and transparent way.

Several available reports discuss the selection and use of models for environmental decision making, and the committee believes that these can provide valuable guidance to EPA regarding criteria for model selection. 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 how it should be revised (U.S. Environmental Protection Agency Science Advisory Board 2006). A final report is expected. Until the final guidance is published, the original draft guidance and SAB review can provide the EPA with valuable advice in the selection of models. Similarly, in 2007 the NRC Board on Environmental Studies and Toxicology published a report entitled "Models in Environmental Regulatory Decision Making." The EPA should utilize this NRC report as a primary guidance document in selecting appropriate ecological models for use in valuation exercises. Criteria such as these can guide the Agency both in selecting from among existing models and in setting priorities for future model development.

The reports discussed above address environmental modeling in general and do not focus on the use of ecological models for valuation purposes. Thus, in addition to the criteria discussed in these reports, at least one other criterion specific to the valuation context should be considered. The committee recommends that EPA selects predictive models for use in valuation, the Agency should choose models that generate outputs in terms of the important, highly valued ecological services identified in the conceptual model or outputs that are easily translatable into effects on such services. This will greatly facilitate the valuation of ecological effects. Thus, when using the reports referred to above, the EPA should keep in mind that the ultimate goal is to provide a measure of the value of the effects of an action on ecological services.

3.3.2. Identifying Ecosystem Services

Another challenge in implementing ecological production functions in a specific valuation context is identifying the relevant outputs, i.e., the ecosystem services. The discussion in the previous section relates primarily to using ecological science to model and understand the ecology underlying the ecosystems impacted by EPA actions and to predict ecological changes stemming from those actions. As illustrated in Figure 4, to be useful for valuation, these changes must ultimately be linked to changes in ecosystem services through an ecological production function. However, the relevant services must first be identified in a consistent and appropriate way.

Throughout this report, the committee uses the term "ecosystem services" to refer broadly to the ecological characteristics, functions, or processes that directly or indirectly contribute to the well-being of human populations (or have the potential to do so in the future). This definition includes the intermediate and end products that ecosystems provide. Regardless of how ecosystem services are defined, the key point is the identification of a set of changes to ecosystem components that will be valued in a way that is meaningful in the specific context of interest. For example, if a given ecological change reduces the population of bees, which in turn reduces pollination, then one would want to value the change in pollination by comparing or characterizing human well-being with and without the change. Similarly, if an ecological change increases habitat suitable for a particular species or activity, one would want to value the change in habitat by comparing human well-being with and without the change.

Identifying the relevant ecosystem services cannot be done deductively; it is dependent upon what is important to people, once they have been informed about potential ecological effects. The ultimate goal is to identify what matters in nature and to express this intuitively and in terms that can be commonly understood. Technical expressions or descriptions meaningful only to experts are not sufficient; similarly, the identification of relevant services must be informed by the underlying ecological science. Thus, the identification of relevant services requires a collaborative interaction between ecologists, social scientists, the public and stakeholders. Input from the public and from stakeholders can come from a variety of sources, such as the valuation methods described later in this report (e.g., surveys, individual narratives, mental model research, and focus groups) or from content analysis of public comments, solicitation of expert opinion and testimony, and summaries of previous decisions in similar circumstances. The Millennium Ecosystem Assessment (Millennium Ecosystem Assessment Board. 2003 provides a good starting point for this exercise by providing an extensive discussion and classification of ecosystem services.

The committee believes that moving toward defining ecological impacts in terms of changes in services or ecosystem components that are commonly understood is a key to success in valuing the protection of ecological systems and services, and urges the Agency to promote efforts to move in this direction. 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 then be valued. In order to value the health effects of air pollution, it was necessary to move from describing impacts in terms such as oxygen transfer rates in the lung to terms that were more easily understood and valued by the public, such as asthma attacks. The search for common health outcomes that can be used for valuation has been difficult. Nevertheless, the lesson is clear: if health and social scientists are to productively interact (e.g., to assess the economic value of improved air quality), measures of health outcomes that are understandable and meaningful to both are necessary. These outcomes are now understood by disciplines as divergent as pulmonary medicine and urban economics (U.S. Environmental Protection Agency Science Advisory Board, 2002). The search for common outcomes that can be valued will be especially 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 (e.g., including those that generate use value - such as harvested species and pollinator species – and those that generate existence values); land cover types (e.g., forests, wetlands, natural land covers and vistas, beaches, open land and wilderness); resource quantities (e.g., surface water and groundwater availability); resource quality (e.g., air quality, drinking water quality, soil quality); and biodiversity. These services play a role in a variety of contributions to human well-being provided by ecosystems.

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 only way to debate and convey a shared mind-set, and that it will concretely 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. While achieving agreement on a common list might be an important ultimate goal, 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.

The identification of relevant ecosystem services, either as a common list or for a specific problem, should follow some basic principles to ensure that the services identified capture socially important ecological changes. These principles include the following:

a) In identifying the relevant services to be valued, it is important to include all ecosystem services, but avoid double counting. Here the principle is to count all things that matter, but to count them only once. The conceptual model developed to guide the valuation process should be designed to ensure that this principle is followed.[28] In identifying and listing the ecosystem services to be valued, it is important to capture both intermediate and final services of importance, recognizing that ecological functions or processes are generally inputs into the production of another ecological good or service.

b) Ecological services should have concrete outcomes that can be clearly expressed in terms that lay publics can understand. In order to provide useful input into valuation, ecological outcomes must be described in terms that are meaningful and understandable to those whose values are to be assessed. Thus, ecosystem services need to be identified through interactions between technical experts and lay publics. This will involve input from both the scientific community and from a wide range of interested parties, as a means of validating the relevance of the services.

c) The delineation of services should reflect the basic principles of ecology. In particular, the delineation should reflect the role of spatial and temporal phenomena and the importance of place. In practice, the delineation means that they should be derived from processes that take place at large spatial and temporal scales, but they should be expressed in local terms at specific times. For example, the availability of water in a particular place at a particular time is what people care about, but landscape-level and inter-temporal analyses are necessary to predict changes in that specific service. Advances in information technology, mapping, and remote sensing technologies in particular will increasingly enable this kind of measurement.

d) The delineation of ecological services should reflect scarcity, and the availability of substitutes and complements. This is related to the need for spatially- and temporally-explicit services. The social value of ecological changes will often be related to the existence of substitutes and complements. Is this the only clean lake people can swim in or are there others nearby? If people want to hike in the woods, are there trails they can use? If people like to kayak in June, will there be adequate water volume? These are often key determinants of the value of a change. Services should be defined so as to allow a consideration of scarcity, substitutes, and complements in estimating or characterizing values.

Figure 4 distinguishes between ecological endpoints and the concept of ecosystem services, and highlights the fact that identifying ecological endpoints is not the same as identifying ecosystem services. EPA has several on-going initiatives related to ecological endpoints, but these fall short of identifying ecosystem services, mainly because they do not follow the basic principles outlined above.

One ecological endpoint initiative is the Environmental Monitoring and Assessment Program (EMAP), which the Agency created in the early 1990s. It was designed to be 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). Referring to EMAP, the EPA recently stated that, “A useful indicator must produce results that are clearly understood and accepted by scientists, policy makers, and the public.” (Jackson et al. 2000) While this goal is consistent with the goals underlying the identification of ecosystem services, the indicators developed in EMAP are not generally direct measures of ecosystem services. Authors have noted the need to translate EMAP indicators "into common language for communication with public and decision-making audiences" (Schiller et al. 2001.). In one analysis, focus groups were used to evaluate the indicators. In general, the study demonstrates the need "to develop language that simultaneously fit within both scientists' and nonscientists' different frames of reference, such that resulting indicators were at once technically accurate and understandable." The committee agrees with this conclusion, and urges EPA to move toward this goal.

EPA has also developed a set of Generic Ecological Assessment Endpoints (U.S. Environmental Protection Agency Risk Assessment Forum 2003) based on legislative, policy, and regulatory mandates. If expanded to include landscape-, regional-, and global-level endpoints (see U.S. Environmental Protection Agency Risk Assessment Forum 2003 Table 4.1, Harwell, et al. 1999; Young and Sanzone, 2002), the GEAEs can be used as a first step in characterizing the relevant ecological system and quantifying the responses to stressors. Thus, the committee views these initiatives as steps in the right direction.

While the GEAEs are a starting point, they also are an example of how far EPA must go in moving toward consideration of impacts on ecosystem services. First, the GEAEs are expressed in technical terms and do not generally describe concrete outcomes and are not expressed in terms that the lay public can understand. While these technical terms are certainly appropriate for some regulatory purposes, most of the public is not likely to be familiar with them. Hence, 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 were developed via explicit reference to policy and regulatory needs (U.S. Environmental Protection Agency 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, the lost value from a single dead or diseased fish depends not on the number of kills or anomalies but rather on how it affects the abundance of healthy fish in the landscape.

Finally, the GEAEs do not enable analysis of scarcity and the availability of substitutes or complements. This is related to the previous limitation. For example, if anglers care about fish populations because of their impact on catch rates, then the lost value from a single dead fish in a single lake will depend (among other things) on the scarcity of fish and availability of substitutes in the relevant vicinity.

The Agency is aware of these issues. The committee raises them primarily: a) to highlight the difference between the Agency's current approach to defining relevant ecological endpoints and the committee's vision of ecosystem services, and b) to encourage the Agency to move toward identification and development of measures of ecosystem services that are relevant and directly useful for valuation.

The identification of relevant ecosystem services will require increased interaction within the Agency between natural and social scientists. The committee urges the Agency to foster this interaction through a dialogue related to the identification and development of measures of ecosystem services. One vehicle for increased dialogue is through greater coordination among the Agency's research programs, especially between the Agency's extramural research programs in ecological research and in Decision-Making and Valuation for Environmental Policy. The committee believes that these two programs could and should be more closely linked. 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 Changes in Ecological Inputs to Changes in Ecological Services

Once the underlying ecology is understood and modeled and the relevant ecosystem services are identified, development of the corresponding ecological production functions will still require a correlation from the ecological inputs to the ecosystem services that those inputs produce. Although numerous ecological models exist for modeling ecological systems, as noted above, most of them fall short of what is needed to fully develop this relationship. Many of these models have been developed to satisfy research objectives, not Agency policy or regulatory objectives. In the past, outputs of these models have not generally been cast in terms of direct concern to people, and thus are not designed as inputs to valuation techniques. They have typically focused on understanding the dynamics in ecological systems, such as the effect of abiotic driving variables on production, the interaction among species, and the rate of carbon sequestration on continental scales. For example, evapotranspiration rates, rates of carbon turnover, and changes in leaf area are important for ecological understanding, but have not been translated into values of direct human importance. This reflects the fact that the links between outputs of some ecological models and human uses of ecosystems have only recently been a subject of research. Certainly, there exist some examples of models with outputs directly related to human values. These include those that predict fish and game populations or forest productivity. These examples, however, represent a limited set of ecosystem services.

3.4. Strategies to Provide the Ecological Science to Support Valuation

As noted above, the effect of changes in ecosystem structure and functions on the provision of ecosystem services should be represented by the relevant ecological production functions; however, implementation of this ideal faces numerous challenges at this time. Nonetheless, some promising developments suggest approaches that could be used to move the Agency toward this goal. These include the use of proxies based on functional groupings or indicators, and the use of meta-analyses. Proxies represent a form of simplification, while meta-analysis is based on data aggregation. In addition, opportunities exist for improving the availability of data for use in parameterizing models of ecological systems and the provision of ecosystem services. These approaches are described briefly below.

3.4.1. Use of Indicators

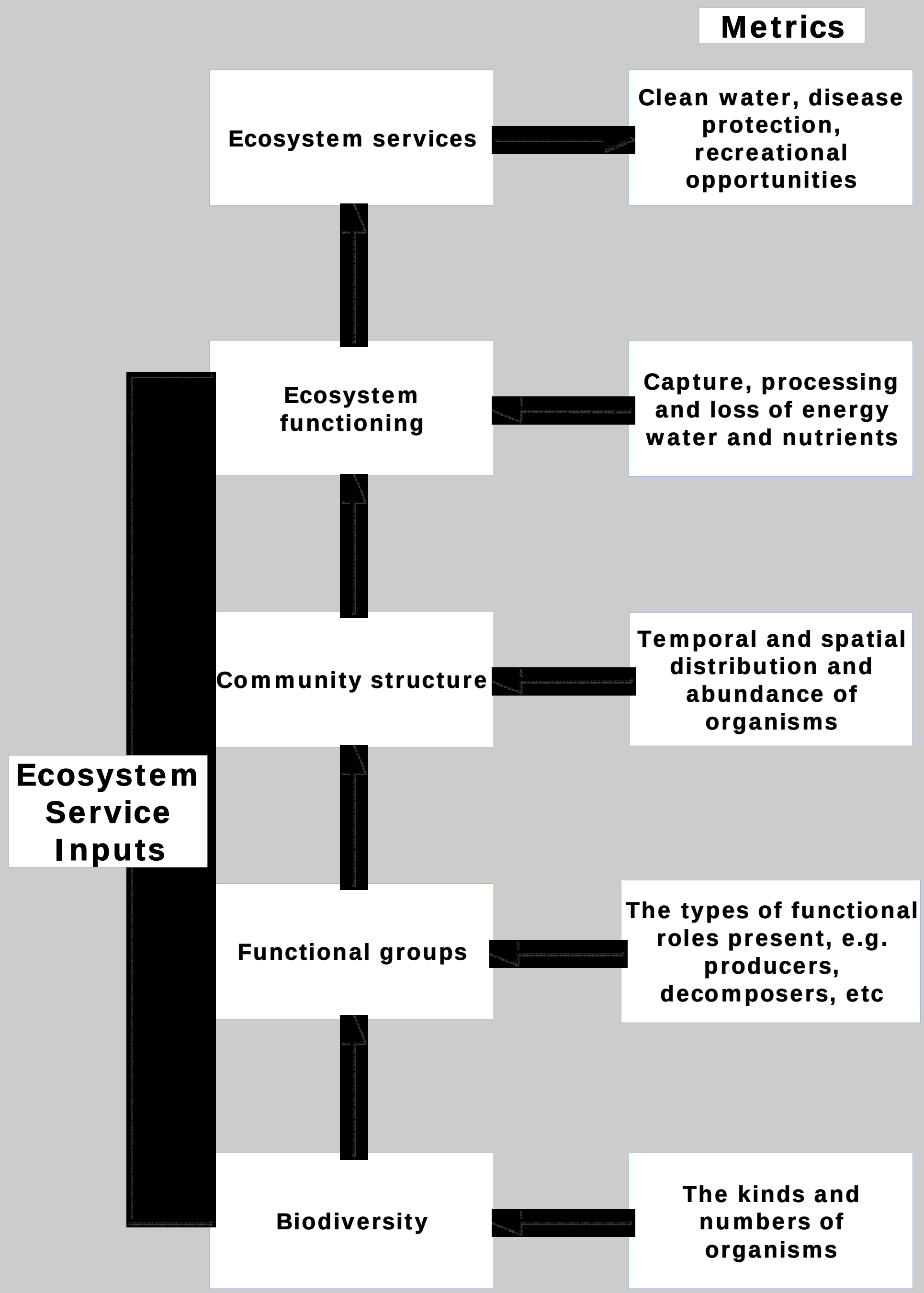
As noted above, an ecological production function describes ecological inputs and outputs (i.e., services), and the relationship between them. When a full characterization of this relationship is not available, some indication of the direction and possible magnitude of the changes in the provision of services that would result from an Agency action might still be obtained using indicators. Indicators are measures of key inputs whose changes are correlated with changes in ecosystem services. In general, an indicator approach involves selecting key predictive variables or indicators rather than attempting to measure and value all the possible significant outputs.

To the extent that the indicators used are grounded in ecological science but expressed in terms relevant for valuation, they can provide information about how ecological impacts might affect ecosystem services. If it is known that the indicator is positively or negatively correlated with a specific ecosystem service, then predicting the change in the indicator can provide at least a qualitative prediction of the change in the corresponding ecosystem service. In addition, the use of large, complex ecological models can be difficult pragmatically, especially because of the quantities of required data and the time to implement. As a result, making numerous or rapid evaluations is difficult (Hoagland and Jin 2006) and simplification would be far more practical. Thus, 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 easily can be linked to specific services. Many ecosystem indicators have been proposed (U.S. Environmental Protection Agency, 1996; National Research Council, 2000, U.S. Environmental Protection Agency, 2002b, U.S. Environmental Protection Agency 2007) and several states have sought to define a relatively small set of indicators of environmental quality to convey the value of ecological services. 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 particular decision contexts where valuation can be useful.

Figure 5 illustrates possible indicators or metrics at different levels of ecological organization. One type of indicator is provided by functional groupings. Because of their inherent complexity, ecological systems cannot be characterized in their entirety, nor can their responses to stressors be completely measured and predicted by single indicators. For example, because of the large number of species in most ecosystems, it is rarely possible to list, characterize, or model all of them when attempting to understand the services they provide. For this reason, ecologists often aggregate large numbers of species into functional groupings. All members of one functional group are similar in terms of the role that they play in the ecosystem. For instance, all deciduous tree species might comprise a single functional group, as might insect-eating birds, or nitrogen-fixing bacteria. The appeal of this approach is that within a given functional group there may be many different species that provide a given function even though one or more of the species of the group may not be present. Changes in the functional grouping can provide an indication of the likely changes in the associated services even when a precise estimate of the change in those services is not possible.

Figure 5: Indicators of Ecological Properties at Different Levels of Organization



Another approach to indicators is designed to incorporate multiple dimensions into a coherent presentation that describes the status of ecosystems within a region, especially as they relate to social values and ecosystem services. For example, the "ecosystem report card" in South Florida (Harwell, et al., 1999) is an example of an indicator based on particularly germane criteria, namely, that it: a) be understandable to multiple audiences; b) address differences in ecosystem responses across time; c) show the status of the ecosystem; d) characterize the selected endpoints, and e) transparently provide the scientific basis for the assigned grades on the report card. Through application of these criteria, the indicator is intended to provide information about the status and trends associated with the ecological services provided by the South Florida ecosystem. The report card identifies seven essential ecosystem characteristics that are thought to be important, i.e., habitat quality, integrity of the biotic community, ecological processes, water quality, hydrological system, disturbance regime (changes from natural variability), and sediment/soil quality, which were then related to the goals and objectives for the ecosystem integrity report card.[29] Related ecological outputs were selected based on both scientific issues and societal values. The outputs are not designed to be monetized, but rather are described by narratives or quantitative/qualitative grades that are scientifically credible and easily understood by the public. There are other examples of using report cards to characterize the status of a given ecosystem. The extension of this idea, of course, is to use changes in the grades as indicators of ecological effects of EPA actions. The report card approach is a possible method for characterizing contributions to human well-being for the purposes of Circular A-4 when economic benefits or ecological services cannot be readily monetized.

Functional groupings provide an examples of possible indicators. Many others exist. 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. Because of the complexity of the interactions between economic and ecological systems, economists frequently take a similar simplification approach that focuses on effects occurring only in the relevant markets, assuming that the effects on the broader market are negligible and can be ignored (Settle et al. 2002).

This simplification approach to ecological modeling will never satisfy those who will always want to identify all the possible consequences of EPA actions. For example, Barbier's (2001) study of the economics of species invasion involved a predator-prey model with inter-specific competition and dispersion. The model results demonstrated that the types of ecological interaction determined the extent to which the introduction and spread of invasive species reduced commercial fishing. He further argues that future models should consider more complex ecological interactions, habitat modification and non-market damages (Hoagland and Jin 2006). [Is the suggestion here that Barbier wants to identify all possible consequences? And is Hoagland and Jin the right reference? Is Barbier arguing this in a paper by Hoagland and Jin?? KS]

The challenge is the practicality of building ever more complex models that must address a wide array of issues over multiple spatial and temporal scales. It may well be that with accumulated experience, it will be shown to 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 rulemakings 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. In addition, use of the chosen indicators should be accompanied by an effective monitoring and reporting program.

3.4.2. Use of Meta-analysis.

A second promising approach to providing information about changes in 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 the associated services. For example, Worm et al. (2006) attempted to measure the impacts 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, it is impossible to separate correlation and causation, which is a severe limitation. But examining data from site-specific studies, coastal regional analyses and global catch databases will allow 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 to fish. This approach a) links fish, habitat, and chemistry data collected from hundreds of sites in Ohio streams; b) assesses the biological condition at each site; c) attributes impairment [e.g., loss of one or more of 117 fish species] to multiple probable causes; and d) provides the results of the analyses in simple-to-interpret pie charts. When data were aggregated from throughout Ohio, 50% of the biological effect was associated with unknown factors and model error; the remaining 50% was associated with alteration in stream chemistry and habitat. While the results do not fully explain the biological effect, , the point is that the technique combines multiple data sets and assessment tools (models) to arrive as estimates of loss of fish species based on broad patterns. Thus, like the previous 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, can be monetized.

3.4.3. Opportunities regarding ecological data

Although data availability is a serious problem in the development of ecological production functions, 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 forms in electronic format. Also, as governmental agencies are being held more accountable, data used in decision-making are expected to be made available to constituents.

Within the ecological research community, the National Science Foundation (NSF) Long-Term Ecological Research (LTER) program has had an emphasis on organizing and sharing data in easily accessible electronic datasets. Although these data were rarely collected for valuing ecological services, they are particularly valuable because they frequently measure long-term trends. As such, these data are useful in separating short-term fluctuations from longer-term patterns in ecological properties. Also, the LTER program recently has focused on regionalization, in which data are collected from sites surrounding the primary site, providing a regional context for site-based measurements and models. Planning for the forthcoming NSF National Ecological Observatory Network (NEON) includes a Networking Information and Baseline Design (NIBD) component, which connects the key scientific questions to the data required to answer the questions. The committee recommends that EPA effectively link into the NEON planning process, and expand its involvement with the NSF LTER program, which is undergoing a major refreshing of its research and data sharing protocols.

1.4.4 Transferring Ecological Information

Despite the increasing availability and organization of ecological data, there is rarely enough available information to support many of these analyses. In addition, the costs are too prohibitive to allow extensive data to be collected from all the sites on which EPA is considering action. From an ecological perspective, therefore, an issue arises regarding the reliability of transferring ecological information, whether from one site to another, or over different spatial or temporal scales. Information in this sense 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; the 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. Certain generalizations, however, are possible. Information is more likely to be transferable with greater similarity among 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, 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 could be transferable to other spatial or temporal scales if the dynamics over time and space scales 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, then 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 (local climate, soil type, land-use history), then 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 (Citations?). 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.

3.5. Directions for Ecological Research to Support Valuation

The committee is aware that EPA 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)]. Based on these preliminary briefings, the committee welcomes these efforts as a way to strengthen the foundation for ecological valuation, although the committee notes with concern the 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 cautions the Agency to design the research program in a focused way because the cost of implementing an ecological production function approach 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 the C-VPESS report..

3.6. Conclusions/Recommendations

Implementation of the C-VPESS valuation process requires prediction of the ecological impacts of EPA actions, identification of the relevant ecosystem components and services to be valued, and linking predicted ecological impacts to changes in those components and services. This is an essential part of valuation and must be done before the value of those changes can be assessed.

With regard to predicting ecological impacts and changes in services, the committee recommends the following:

- EPA should begin each valuation with a conceptual model designed to provide a road map to guide the process. A process for constructing the initial conceptual model should be formalized, recognizing that as an iterative process, it responds to the addition of new information and multiple points of view. The conceptual model and its documentation should clearly describe the reasons for decisions about the spatial and temporal scales of the target 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, always recognizing that the selected effects should relate to the valuation process. In constructing the conceptual model, participation should be required from staff throughout the EPA, outside experts from the bio-physical and social sciences, and members of the public who have a standing in the results of the outcomes
- EPA should move toward identification and development of measures of ecosystem services that are relevant and directly useful for valuation. This will require increased interaction within the Agency between natural and social scientists. The identification of services should satisfy the basic principles outlined above. a) counting all things that matter once and only once; b) expressing outcomes as services that are commonly understood; c) incorporating appropriate spatial and temporal considerations; and d) reflecting the role of relevant substitutes or complements, or both.
- EPA should seek to use ecological production functions wherever possible to describe how changes in the ecosystem (resulting from stressors created by different policies or management decisions) ultimately lead to changes in the provision of ecosystem services.
- To operationalize the modeling of the ecological processes that will produce the ecological services, EPA should use predictive ecological models. There are many ecological models out there. 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 apply these criteria in a consistent and transparent way.

- EPA should continue and accelerate research to develop key indicators for use in ecological valuation for key repeated rulemakings or other repeated decision contexts. Such indicators should meet ecological science 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 applied in ecological valuation.
- EPA should actively participate in the major efforts to organize ecological data (e.g., LTER, NEON) both in terms of providing data and in using the most applicable data sets in its assessments. EPA should 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 investments in ecological research to generate ecological production functions for valuation, including research funding investments in STAR research on ecological services and support for modeling and methods development. In addition, the EPA's National Center for Environmental Research's programs on evaluating ecosystem services and valuing ecosystem services should be more closely linked.

4 METHODS FOR ASSESSING VALUE

The process for implementing the C-VPESS approach requires the use of an expanded set of methods for characterizing the value of the predicted ecological effects of EPA actions. This chapter provides information about methods that the committee examined for possible use in implementing the integrated and expanded valuation process proposed in Chapter 2 including methods and approaches for transfer of valuation information.

4.1. An Expanded Set of Methods

As noted above, this section discusses methods that the committee examined for possible use in implementing the integrated and expanded valuation process proposed in Chapter 2. This list illustrates the variety of methods available and should not be viewed as exhaustive.

The methods discussed differ in a number of respects, 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), the extent to which they involve the public or stakeholders, the degree to which the method has been developed or utilized, the potential envisioned by the committee for future use at EPA, and the issues involved in implementing the approach.

While these methods are not easily categorized, the committee has organized the discussion of methods around groupings based on the premises that underlie the methods. In each case, the goal is to provide the reader with sufficient information about the methods to allow a preliminary assessment of the role that various methods could play in implementing the proposed valuation process (including strengths and possible weaknesses of different methods) and to direct the interested reader to the relevant scientific literature for further information. The intent is not to provide an exhaustive treatise on any given method.

Table 5 immediately below provides an introduction to these methods. General descriptions of the categories of methods follow. The concluding section summarizes the committee's assessments of methods and its recommendations for EPA. Detailed discussion of specific methods appear in Appendix B of this report. In addition, Appendix B provides detailed information about the use of survey methods for ecological valuation.

Table 2: Introduction to Methods Assessed by the Committee

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value		Reference to Discussions in C-VPESS Report
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?	
BIOPHYSICAL RANKING METHODS					
Conservation Value Method	Map of biodiversity, scarcity, and/or conservation values across landscape	Contribution to biodiversity	Measurements related to previously identified goal of biodiversity	Expert - ecologist or conservation biologist	p. 76 p. 200
Embodied Energy Analysis	Units of free or available energy from the sun (plus past solar energy stored as fossil fuels) per unit of production	Direct and indirect energy cost of goods and services	Measurements related to previously identified goal, reduction in energy depletion	Expert	p. 76-77 p. 210
Emergy	Units of solar energy used to produce one Joule of a service or product	Direct and indirect energy cost of goods and services	Measurements related to previously identified goal, reduction in energy depletion	Expert	p. 77 p. 213
Ecological Footprint	Area of ecosystems required to produce resources consumed and to assimilate waste produced	Biologically productive land area required (directly and indirectly) to meet consumption patterns	Measurements related to previously identified goal, reducing ecosystem services consumed per unit of land	Expert	p. 77 p. 212
ECOSYSTEM BENEFIT INDICATORS					
Ecosystem Benefit Indicators	Map of the supply of ecosystems/services showing quantities of expressed or estimated demand for those ecosystems/services across a landscape	Quantitative but not monetary approach to preference weighting for the ecological effects of policy options	Measurements related to demand variables that can be identified by experts or non-expert lay publics and supply variables as identified by experts.	Expert and selected non-expert lay public	p. 77 p. 215
MEASURES OF ATTITUDES, PREFERENCES, AND INTENTIONS					
Surveys Including Questions about Attitudes, Preferences, and Intentions	Attitude scales, preference rankings, behavioral intentions toward depicted environments/conditions	Public concerns, attitudes, values, beliefs, and behavioral intentions	Verbal reports, choices, rankings, ratings	sample from public	p. 78 p. 223-255

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value		Reference to Discussions in C-VPESS Report
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?	
Conjoint Attitude Survey Questions	Attitudes, preference rankings implied from expressed trade-off preferences	Public concerns, attitudes, values, beliefs, and behavioral intentions related to specific trade-offs	Verbal reports, choices, rankings, ratings	sample from public	p. 78 p. 223-255
Individual Narratives	Narrative summaries	Implied knowledge, belief and attitude structures	Verbal report from lay public	sample from public	p. 79 p. 223-255
Mental Models	Concepts/categorized 'events' in conceptual models	Causal beliefs and inferences	Observed decision making behavior, verbal reports	any individual (expert or non-expert)	p. 79 p. 223-255
Behavioral Observation/Trace	Observations of current or prior (trace) use of ecosystems/services	Responses to policies, outcomes, and consequences, in situ	Past behavior	sample from public	p. 78 p. 223-255
Interactive Environmental Stimulation Systems	Observations of behavior in simulated/game environment, implied preferences	Responses to investigator-controlled changes in environmental conditions	Behavior	sample from public	p. 223-255
ECONOMIC METHODS					
Market-Based Methods	Monetary unit: changes in consumer and producer surplus	Well-being of individuals in society, defined as the individuals' preferences and their willingness to pay for gains and compensate for losses	Behavior	participants in the market	p. 79-80 p. 256
Travel Cost	Monetary unit: WTP as revealed by responses to differences in travel cost		Behavior	sample from public	p. 80 p. 260
Hedonic pricing	Monetary unit: marginal WTP as revealed by responses to differences in characteristics and prices of different units of the product		Behavior	sample from public	p. 80 p. 263
Averting Behavior	Monetary unit: WTP as revealed by responses to opportunities to avoid or reduce damages through purchases of protective goods, substitutes, etc.		Behavior	sample from public	p. 80 p. 266

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value		Reference to Discussions in C-VPESS Report
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?	
Survey questions measuring stated preferences	Monetary Units: WTP, expressed purchase intentions or in the case of Conjoint Economic Surveys, Monetary Units, WTP implied from expressed trade-off preferences		Verbal Reports of WTP or responses to hypothetical choices.	sample from public	p. 80-81p. p. 269
GROUP AND PUBLIC EXPRESSIONS OF VALUES					
Focus Groups	Narrative summaries, frequency tallies, consensus	Full discovery and articulation of all the values that are relevant and exploration of agreements and conflicts among stakeholder constituencies	verbal reports	sample from public	p. 81 p. 283-284
Referenda and Initiatives	Historical monetary data on communities' choices regarding ecological impacts	What the body politic as a collectivity values in terms of policy outcomes	Behavior	Selected stakeholders	p. 81-82 p. 284
Citizen Valuation Juries	Qualitative summary of jury decisions which may include quantitative or monetary decisions	How a representative group views the social civil value of changes to ecological systems and services	Verbal reports	Selected stakeholders	p. 82 p. 296
DECISION-SCIENCE APPROACHES					
Decision-Science Approaches	Language to be added here	Language to be added here	Language to be added here	Language to be added here	page numbers to be added
METHODS USING COST AS A PROXY FOR VALUE					
Replacement Cost (also called Avoided Cost)	Monetary Units	Cost of replacing ecosystem services with human engineered services as an estimate of value.	Observed behavior	Experts in engineering	p. 83-84 p. 324
Tradable Permits	Monetary Units	Incremental willingness to pay for the reductions in emissions of specific pollutants covered by the permits	Observed behavior	Participants in the permit market	p. 327

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value		Reference to Discussions in C-VPESS Report
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?	
Habitat Equivalency Analysis	Units of habitat (e.g., equivalent acres of habitat)	Compensation for loss of ecological services resulting from injury to a natural resource over a specific interval of time	Measurements related to previously identified goal (e.g., units of habitat)	Experts in ecology	p. 83-84 p. 328

4.1.1. Biophysical Ranking Methods

In some contexts, policy makers or analysts are interested in values based on quantification of biophysical indicators. Possible indicators include species 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. 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 preferences. 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 overall ecosystems health and sustainability. Opinion is mixed on whether it is an asset or a drawback that these ranking methods are not tied directly to human preferences.

The committee evaluated two types of biophysical rankings. The first was a ranking method based on conservation value. This 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. The method provides a scientifically based approach to assigning conservation values that can used by policy makers or stakeholders to prioritize land for acquisition, conservation or other uses. Based on GIS technology, the ranking method has the capability to combine information about a variety of ecosystem characteristics and services across a given landscape, and to overlay ecological information with other spatial data. In addition, data layers can be used for multiple policy contexts. Conservation values have been used in various contexts by federal agencies (e.g., Forest Service, Fish and Wildlife, National Park Service, and Bureau of Land Management) as well as by non-governmental organizations (e.g., The Nature Conservancy, NatureServe) and regional and local planning agencies.

The second group of biophysical methods that the committee evaluated was based on energy and material flows. Energy and material flow analysis is the quantification of the flows of energy and materials through complex ecological or economic systems, or both.

These analyses are based on an application of the first (conservation of mass and energy) and second (entropy) laws of thermodynamics to ecological-economic systems. Examples include embodied energy, emergy (the available solar energy used up directly and indirectly to make a service or product), and ecological footprints. Of these three, embodied energy and ecological footprints are based on a consistent set of principles recognized by the committee as potentially useful for EPA, while for emergy, some members of the committee question whether a consistent set of principles appropriate for valuation are used. Embodied energy measures the (available) energy cost of goods and services using input-output analysis or flow accounting methods. Ecological footprint analysis also uses input-output analysis, but measures costs in land units (rather than energy units) based on the biologically productive land area (rather than the amount of energy) required to meet various consumption patterns. While such costs can be used to rank alternatives based, for example, on an energy theory of value, they will provide a proxy for preference-based values only under limited conditions.

4.1.2. Ecosystem Benefit Indicators

Ecosystem Benefit Indicators (EBIs) offer a quantitative way to illustrate ecological contributions to human well-being 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.

Ecosystem benefit indicators (EBIs) are quantitative inputs to valuation methods. They can serve as important inputs to valuation methods as diverse as citizen juries and econometric benefit transfer analysis, which is a monetary weighting technique. EBIs provide a way to illustrate 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. Existence values (where spatial location is irrelevant to both provision and value) are the only ecosystem benefit category where the method would be inapplicable.

4.1.3. Measures of Attitudes, Preferences, and Intentions

Social and psychological methods seek to characterize the values that are held, expressed, and advocated by people. They focus on individuals' judgments of the relative importance of, acceptance of, or preferences for ecological changes. Individuals making the judgments may respond on their own behalf or on behalf of others (society at large or specified subgroups). The basis for their judgments could be changes in individual well-being, or civic, ethical, or moral obligations relevant to ecosystems and ecosystem services. That is, people may hold, express, and advocate bioecological values or ethical values that are unrelated or even counter to their own wants and needs.

Social and psychological methods provide scientific means for determining people's 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. Social and psychological methods elicit information about preferences and values primarily through surveys, focus groups, and individual narratives. Experts in this field recently have been experimenting with eliciting this information through observations of behavioral responses by individuals interacting with either actual or computer simulated environments.

Surveys typically involve face-to-face, telephone, or mail interviews with large representative samples of respondents (see Appendix C for a more detailed description of survey methods). Survey questions are framed as choices (among two or more options), rankings, or ratings; responses are self-reported by individuals. Survey questions about social and psychological constructs may include assessments of attitudes, beliefs, and knowledge, as well as reports of past behaviors and future behavioral intentions. Variations on survey methods that may be especially useful in assessments of ecosystems and services values include 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 relative measures of differences in assessed values for the alternatives offered. Similarities and differences among segments of the public also can be identified and articulated. 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. Surveys to elicit value-related information have been used extensively by other federal agencies (see Appendix C for a representative list).

In contrast to surveys that are based on large representative samples, individual narrative methods - including mental models analyses, ethnographic, and other relatively unstructured individual interviews - generally employ small, specially selected samples of informants and analyze responses qualitatively. Rigorous qualitative analyses can expose subtle differences in individual beliefs and perspectives and the inferential bases of participant's value positions, as well as identify opportunities for achieving consensus. The broad class of studies that fall under the umbrella of individual narrative methods can be particularly useful in identifying unanticipated value perspectives, positions, and concerns that might be missed by other value-assessment methods.

4.1.4. Economic Methods

The economic approach to valuation is an anthropocentric approach based on utilitarian principles. It includes consideration of both instrumental values and intrinsic values, but only to the extent that preservation based on intrinsic value contributes to an individual's welfare. Because it is utilitarian-based, it assumes there is the potential for substitutability between the different sources of value that contribute to welfare. In addition, it assumes that individual preferences, which determine the degree of substitutability for that person, are well-formed. Most of EPA's work to date on ecological valuation has been based on the use of economic methods, and these methods are the focus of EPA's *Ecological Benefits Assessment Strategic Plan.*

The concept of value underlying economic valuation methods is based on substitutability, or, more specifically, on the tradeoffs individuals are willing to make for ecological improvements or to avoid ecological degradation. By itself, 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 (typically money) that the individual is willing to give up to enjoy that change (willingness to pay) or the amount of compensation (typically in money) that a person would accept in lieu of receiving that change (willingness to accept). The economic benefits captured by this concept of value can be derived not only from good and services for which there are markets but also from non-market goods and services. In addition, both use and non-use (e.g., existence) values are included. 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 deontological concept of intrinsic rights. In addition, both willingness-to-pay and willingness-to-accept measures depend on the individual's current income (as well as market prices), implying that individuals with higher incomes will typically have higher economic benefits. This is viewed by many as a drawback of this approach to defining value.

There are multiple economic valuation methods that can be used in principle to estimate willingness to pay. These include methods based on observed behavior (market-based and revealed preference methods) methods based on information elicited from responses to survey questions (e.g., stated preference methods). In contrast, in general measures of willingness to accept can only be obtained using stated preference methods.

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 change. If the change is large, then 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 are limited to valuing "provisioning" services supplied in well-functioning markets.

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). 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 (e.g., specific levels of ecosystem services) to infer how much they value changes in those attributes. Hedonic methods use 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. In contrast, averting behavior methods use observations on how much people spend to avoid adverse (environmental) effects to infer how much they value or are willing to pay for the improvements those expenditures yield.

In contrast to revealed preference methods, stated preference methods infer values or economic benefits in terms of willingness to pay or willingness to accept from responses to survey questions. In some cases, survey questions directly elicit information about willingness to pay (or accept), while under some survey designs (e.g., conjoint or contingent behavior designs) monetary measures of benefits are not revealed directly. Rather, some form of quantitative analysis is needed to derive economic benefit measures from responses to survey questions. Although the use of stated preference methods for environmental valuation has been controversial, there is considerable evidence that the hypothetical responses in these surveys provide useful evidence regarding values.

4.1.5. Group Expression of Values and Social/Civic Valuation

Several methods prove useful in eliciting expressions of values from groups. Focus group methods elicit information about values and preferences from small groups of relevant stakeholders engaging in group discussion led by a facilitator. Given the small number of participants, the goal of a focus group is rarely value assessment per se, but rather an articulation of all of the values that may be relevant. Use of focus groups early in the decision process can help in identifying ecosystem effects that might be particularly important to the public. Focus groups may also be used to develop measurement strategies for value assessment (e.g., to design a survey).

One type of method focuses on public and group expressions of public value, in contrast with traditional economic valuation methods that attempt to measure and aggregate the values that individuals place on changes in ecological systems and services based on their personal preferences. Using this alternative approach, known as social/civic valuation, researchers measure the values that groups place on changes in such systems and services explicitly in their role as citizens. This approach measures the monetary value that groups place on changes in the systems and services. The groups are asked to evaluate how much the public as a whole should pay for increases in such systems and services (public willingness to pay) or should accept in compensation for reductions in the systems and services (public willingness to accept). The value measurement purposefully seeks to assess the full "public regardedness" value, if any, that the group attaches to any increase in community well-being attributable to changes in the relevant systems and services.

Social/civic values, like values based on personal preferences, can be measured either through revealed behavior or through stated valuations. One principal source of revealed values for changes in ecological systems and services are votes on public referenda and initiatives involving environmental decisions. Other public decisions, however, also may provide measures of social/civil values, including official community decisions to accept compensation for permitting environmental damage, and jury awards in cases involving damage to natural resources. Where revealed values are difficult or impossible to obtain, social/civil values also can be measured by asking "citizen valuation juries" or other representative groups the value that they, as citizens, place on changes in particular ecological systems or services.

Analyses of the outcomes of referenda or initiatives (with or without a follow-up survey) seek to determine, for example, if the majority of the voting population feel that a given environmental improvement is worth what it will cost the relevant government body, given a particular means of financing the associated expenditure. 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 feel that the environmental services that would be lost are worth less than the contributions to well-being the community would realize in the form of tax revenues, jobs, monetary compensation, etc. These approaches provide information about the policy preferences of the median voter and, under certain conditions,can provide information about the mean valuations of those who participate in the voting process. The logic of using formal public outcomes to infer how much society values particular outcomes has been used previously to estimate the public's willingness to pay (in the form of a commitment of public expenditure) to reduce mortality rates from health and safety risks.

Like initiatives and referenda, citizen valuation juries provide information on social/civic values, but they measure stated rather than revealed value. They also incorporate elements of the deliberative valuation process. Essentially, the group is given extensive information and, after a lengthy discussion, is 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. A jury could also 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, how much it should be willing to accept for an environmental degradation. Experience with the use of citizen juries for ecological valuation is very limited to date.

4.1.6. Decision Science Methods

Text to be inserted on Decision Science Approaches for valuing changes in attributes not readily measured in dollar terms (e.g., they might instead be measured in physical terms, such as number of birds, or using constructed scales, such as a scale for aesthetics).

Description would show how Decision Science Approaches allow comparisons by providing ways to weight changes in attributes using such methods as swing weights, even swaps, and ratio methods.

4.1.7. Methods Using Cost as a Proxy for Value

A fundamental principle in economics is the distinction between economic benefits and costs. Economic benefits reflect what is gained by increasing the amount of a given good or service. Costs, on the other hand, reflect what must be given up in order to increase a given good or service. Nonetheless, several methods using the cost of producing equivalent substitutes for an ecosystem service have been used as proxies for value of that ecosystem service. Methods that use cost as a proxy for value include replacement cost, habitat equivalency analysis (HEA), and valuing pollution reduction by the price of tradable emissions permits. Cost methods have gained some popularity, especially in estimating the value of protecting ecosystems for provision of drinking water or habitat, because it is often easier to collect information on the cost of providing an equivalent substitute than it is to provide information on economic benefits. But because costs and economic benefits are two distinct notions, great care needs to be taken in the application of these methods and in the interpretation of results using these methods.

The cost of producing a good or service can provide information about the value of that production only under specific and limited conditions. First, there must be multiple ways to produce an equivalent amount and quality of ecosystem services. If so, then one could replace the loss of an ecosystem service via some other means. Second, the value of the ecosystem service must be greater than or equal to the cost of producing the service via this alternative means. If so, society would be better off paying for their replacement rather than choosing to forego the ecosystem services.

An example in which these two conditions may be met is the provision of clean drinking water for a metropolitan area. Protecting an ecosystem that serves as a watershed and building a filtration plant may be two ways of providing the same quantity and quality of drinking water to a city, in which case the loss of watershed protection could be replaced with a filtration plant. Further, the value of providing clean drinking water for a metropolitan area far exceeds the cost of a filtration plant to provide it. In this case, one could value the protection of an ecosystem for the purpose of providing clean drinking water as equal to the cost of building the filtration plant.

When these two conditions are met, it is valid to use the cost of providing the ecosystem services via an alternative means as the value of the loss of one means to produce ecosystem services. It is important to note that this value is not the value of the ecosystem services themselves, but only the value of losing one means to produce them. It is not valid to use cost as a proxy for value, even in this limited sense of value, when these conditions are not met.

The committee urges great caution in the adoption of methods using incremental cost as a proxy for value. It must be demonstrated that the conditions for valid use are satisfied and analyses of incremental costs should not be interpreted as incremental benefits unless these conditions are met.

4.1.8. Summary and Recommendations

The methods described in this section, and in more detail in Appendix B, were evaluated by the committee to help the Agency move toward valuations that include an expanded range of important ecological effects and human concerns. The committee observes and strongly reminds the Agency that no single method, metric, or index of value can be used to fully reflect important ecological effects and human concerns for decision-making, because value is such a complex concept.

The committee advises EPA to follow the "Process for Implementing an Expanded and Integrated Approach to Ecological Valuation" (Figure 2). High-quality valuations will follow that proposed process for a specific decision context, will involve a conscious choice about the types of values to be assessed, and will have transparent communication about the types of methods used and the uncertainties associated with methods used at different parts of the valuation process.

Different kinds of decision contexts might call for use of different kinds of methods. In some cases, the environmental values at stake may principally involve ecosystem services easily understood by the general public. Recreation services might be involved, for example, and survey methods or travel cost methods might be appropriate methods to choose. In other cases, the decision context may involve ecosystem services that are more complex or not commonly understood by the broader population (e.g., nutrient cycling or biodiversity). In those instances, decision makers may be interested in what experts value or they might choose to use mediated modeling efforts to bring experts and lay publics together. In addition, some types of decisions have different legal constraints affecting the type of valuation output sought (e.g., economic benefit-cost analyses associated with Regulatory Impact Assessments call for the use of methods that generate economic values wherever feasible) and some methods work better at certain geographic scales (e.g., Habitat Equivalency Analyses at a site-specific scale; Conservation Value Methods at a landscape or regional scale). The choice of method should be appropriate to the decision context and the geographic scale of use. Finally, EPA must consider the cost of using a state-of-the-art valuation method in terms of the information gained for decision making, while operating under Agency budget constraints. Table 3 below briefly summarizes the committee's conclusions regarding methods discussed in this report. It provides cross-references to sections of Appendix B that discuss methods in more detail.

Table 3: Table Summarizing Methods Discussed in this Report

	Degree to Which Method has Been Developed or Utilized	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation
Conservation Value Method	Components of approach used by • U.S. Department of Agriculture • U.S. Forest Service • U.S. Fish and Wildlife Service • National Park Service • Bureau of Land Management • IUCN • The Nature Conservancy • NatureServe	• Use to focus available conservation funds related to conservation goals • Use as a prediction of ecological impacts that would then be used as an input in an economic valuation study • Use in combination with other non-monetary value information (for example, from social-psychological surveys) to characterize preference-based values when monetization is not possible or desirable • Use as a means of quantifying biophysical impacts when they cannot be quantified (as required by the OMB Circular A-4)
Embodied Energy Analysis	• Has been used by some ecologists and physical scientists to implement an energy theory of value	• When costs can be used as a proxy for value, this method provides information about ecological values as defined by the energy theory of WHAT? • Can be used to rank options or assess impacts in biophysical terms based on required energy inputs • Does not provide an alternative means of monetizing economic values based on WTP
Emergy	• Has only been used by a small circle of researchers, some at EPA	• Substantial questions remain about the appropriateness and usefulness as a method for ecological valuation
Ecological Footprint	• Has been used extensively by ecologists to compare resource use by different populations	• Most useful as an index of the quantity of ecosystem services consumed • Can be used to rank options or assess impacts in biophysical terms based on relative resource use
Ecosystem Benefit Indicators	• The method is new and relatively undeveloped	• Input to a wide variety of trade-off analyses (for regulatory analyses or performance measures) • Use as part of public processes designed to communicate the implications of a change or policy across a variety of scales • Use as inputs to economic and econometric methods such as economic benefit transfer, or stated preference models • Use to systematize alternative choice scenarios in choice experiments and stated preference surveys
Surveys Including Questions about Attitudes,	• Survey questions measuring social-psychological constructs are the oldest and most frequently used methods for determining public beliefs, concerns, and preferences	• Can contribute to initial problem formulation by identifying ecological services and impacts that most concern citizens and/or identified stakeholders, as well as by uncovering assumptions, beliefs, and values that underlie that concern • Can help to determine socially important assessment endpoints • Can be used to assess relative public preferences among policy options

	Degree to Which Method has Been Developed or Utilized	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation
Preferences, and Intentions	• Survey questions have been and continue to be used effectively by all levels of government to measure citizen desires concerns and preferences	• Quantitative outcomes may be especially useful when the values at issue are difficult to express or to conceive in monetary terms or where monetary valuations are viewed as ethically inappropriate • Can be used to help inform and involve publics in decision-making where valuation has been involved
Conjoint Attitude Survey Questions		• May be especially well-suited for gauging public preferences across sets of complex multi-dimensional alternatives, likely involved in many EPA regulations and actions for ecosystems/services protection
Individual Narratives	• Provides qualitative information and generally no representative sampling but may have a role in earlier stages of valuation	• Can make important contributions to improving the design, development and pre-testing of more formal surveys that can provide reliable and valid quantitative assessments of public concerns and values
Mental Models	• Research has focused more on enabling and informing risk reduction, rather than motivating or understanding preferences and trade-offs per se	• Appropriate precursor (i.e., formative analysis) to any formal survey or preference elicitation method, to improve the validity and reliability of the method
Behavioral Observation/ Trace	• Relatively new and untested	• Might be used to attain quantitative measures of human use levels useful in conjunction with economic measures or as separate measures to be correlated with changes in ecological conditions
Interactive Environmental Simulation Systems	• Relatively new and untested	• Can engage and communicate with public audiences about what outcomes they prefer and policies required to achieve those outcomes • Respondents can learn through experience about how the ecosystem of interest responds to various policies or policy aspects and can progressively modify their expressed policy preferences
Market-Based Methods	• Are based on well-established economic principle and econometric practices • Have been used for more than 30 years to evaluate a variety of economic and environmental policies	• Provides estimate of willingness-to-pay measures of value for the economic valuation of environmental policies (benefit-cost analysis) that affect ecosystem services that support the provision of goods and services bought and sold in markets
Travel Cost	• Method is based on well-established economic principles and econometric practices • Has been extensive use of this method in analyzing the demand for recreation services and the value of attributes of	• Provides estimate of willingness-to-pay measures of value for the economic valuation of environmental policies (benefit-cost analysis) that affect ecosystem services that support the provision of recreation services

	Degree to Which Method has Been Developed or Utilized	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation
	recreation sites and activities	
Hedonic pricing	• Has been widely applied to estimate the values of site specific amenities and disamenities as reflected in the prices of houses	• Provides estimate of willingness-to-pay measures of value for the economic valuation of environmental policies (benefit-cost analysis) that affect ecosystem services that affect the market prices of houses
Averting Behavior	• Substantial literature on the theoretical dimensions of the method but relatively few convincing studies demonstrating that it yields valid estimates in practice	• Provides estimate of willingness-to-pay measures of value for the economic valuation of environmental policies (benefit-cost analysis) that affect ecosystem services for which there are substitute activities or goods
Survey Questions Measuring Stated Preferences	• Extensive literature covering principles and applications to valuing environmental changes extending over a 40-year period	• Provides estimate of willingness-to-pay measures of value for the economic valuation of environmental policies (benefit-cost analysis) that affect any type of ecosystem service • The only set of methods capable of capturing the economic concepts of non-use value and existence value
Focus Groups	• Not clear the extent to which focus groups are systematically used in EPA policy making • The OMB and other guidelines do not clearly specify the criteria for using focus groups	• Can be useful and important for designing and pre-testing more formal surveys • May also contribute to the design of more effective communications of Agency decisions
Referenda and Initiatives	• Logic has been used primarily in the literature on health and safety	• Can provide monetized values—of the community's formal decision and values, ceilings, or floors of the median voter's valuation • With follow-up surveys can provide information on beliefs, assumptions and motives regarding the ecosystem preservation issues that voters perceive are at stake • Any EPA decision context calling for monetized valuation could employ these variants, either singly or as cross-checks with conventional revealed preference or stated preference approaches
Citizen Valuation Juries	• Experimental method in the context of ecological valuation • Used primarily to help governments rank options for achieving particular goals • Only a few efforts have been made to date to use citizen juries to generate monetary or other estimates of the social/civic value of	• Potentially useful both to identify socially important assessment endpoints and to attach a value, monetary or socio-psychological, to changes in the assessment endpoints • Can expand the role that the public plays in valuations of changes in ecological systems and service

	Degree to Which Method has Been Developed or Utilized	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation
	environmental changes	
Decision Science Approaches	• Language to be added here	• Language to be added here
Replacement Cost (also called Avoided Cost)	• The method has been used to provide estimates of the value of protecting watersheds for the purpose of providing clean drinking water	• There is great potential for abuse in using replacement costs to estimate the value of ecosystem services and it should be used with care
Tradable Permits	• With the development of tradable permits for non-market environmental goods, it has been suggested that the price of a tradable permit is a proxy for the economic value of provision of environmental quality or conservation	• There are no conditions under which the cost of permits could be used as a proxy for economic value
Habitat Equivalency Analysis	• Originally developed in 1992 to quantify damages associated with contaminated wetlands and has since been applied to cover injuries due to chronic contamination, spills, and vessel groundings in a variety of habitats • Currently used in Natural Resource Damages Assessment (NRDA) under Oil Pollution Action (OPA) And CERCLA (Superfund)	• Provides a framing for characterizing bio-physical change • Could be used ex ante to compare alternative future actions to identify the action with the least impact and to compare alternative actions to identify which will yield the most service or equal service in the shortest time frame

4.2. Value Transfer

4.2.1. Transfer of Economic Benefits

Economists often use information that allows the measurement of economic benefits for hypothesized changes in the amount, terms of availability, or quality of resources that can be derived from a previously conducted valuation study to assign values to policy-induced 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 have been used to estimate the marginal value of small improvements in air quality in other cities, such as Cleveland, New York City, or Los Angeles (the policy site)[30]. The adjustments that are necessary to use benefit information from a previous study in a new context depend on a number of factors, including the needs of each proposed policy application, the available information about the policy site, and the added assumptions each analyst is prepared to make.

In light of constraints imposed by 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 RIAs and policy evaluations rely on adaptation of information from the existing literature. EPA's *Economic and Benefits Analysis for the Final Section 316(b) Phase III Existing Facilities Rule June 1, 2006* (U.S. Environmental Protection Agency 2006), EPA's *Final Report to Congress on Benefits and Costs of the Clean Air Act, 1990 to 2010*. (U.S. Environmental Protection Agency 1999), and the economic benefit-cost analysis of the CAFO regulations offer recent examples of policy evaluations that used benefits transfer methods. While benefits transfer has been used extensively by EPA for economic values, parallel approaches can and have been used to transfer other information relevant to ecological valuation (such as information about biophysical relationships). This section focuses on issues related to economic benefit transfer, but the committee notes that similar issues are relevant to the transfer of other types of information from one application or site to another.

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 extended to new applications? Inappropriate benefits transfer often is a weak link in valuation studies. Prior to 2000, the challenges and limitations of benefits transfer received little attention. This relative lack of attention is surprising, given the prevalence of benefits transfer in practical valuation efforts, particularly at EPA. Since 2000, however, a number of environmental economists and other policy analysts have devoted considerable attention to the issue of benefits transfer, including an entire 2002 special issue of the journal *Ecological Economics* (the Wilson and Hoehn [2006] editorial provides a good overview).

The evaluations of benefits transfer in the literature are uniformly negative. 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 "national benefit transfer seems to be possible if margins of error around 50% are deemed to be acceptable" (p. 334). However, these evaluations do not do justice to the potential for careful economic benefits transfer, since they typically adopt a mechanical process to mimic the steps in an economic benefits transfer. Because benefits transfer is 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, which must consider the similarities and dissimilarities between the study site and the policy site(s). By this criterion, some applications of benefit transfer are valid while others are not. For this reason, overall the committee believes that general conclusions regarding the validity of the application of these methods are not possible.

4.2.2. Transfer Methods

As noted above, benefits transfer refers to a collection of methods rather than a single approach. For example, values derived from one or more study sites can be transferred to a policy site in three alternative ways. The first is the transfer of a "unit value." A unit value transfer usually interprets an estimate of the trade-off 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 would be used to estimate an adjusted measure for the unit value based on the specific conditions in the policy area (see Brouwer and Bateman 2005 for another example in the health context). As noted above, the required adjustments will depend on a number of factors (see further discussion below).

The 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 (i.e. a minor symptom day such as a day with mildly red watering itchy eyes; a runny nose with sneezing spells; or a work-loss day described as one day of persistent nausea and headache with occasional vomiting).[31] A value function in this context would relate the 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 could then be 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 would be transferred and then used to estimate economic benefits at the policy site. In this case, the function being transferred would be an estimated behavioral model rather than a value function. Meta-analyses, which statistically combine results from numerous studies, 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. One area with a large number of applications is water quality relevant to recreation [see Johnston et al. (2003) as an example of meta-analyses for water quality; Smith and Kaoru (1990a, 1990b) for other recreation-based meta-analyses]. This approach was used recently in EPA's assessment for the Phase III component of the 316(b) rules.[32]

The third approach to benefits transfer is the "preference calibration" approach. 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. 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. With a specified algebraic function describing a preference relationship, along with information about the factors constraining an individual's choice in the study application and in the policy application, the preference calibration approach considers whether there is sufficient information in existing estimates to calculate or infer the relevant parameters of the preference relationship. When the parameters can be calibrated or estimated from the existing literature, the transfer involves using the calibrated preference function, together with the conditions at the policy site, to measure the trade-off for the change associated with the policy application The task does not require that the parameters required for all possible trade-offs (i.e., the complete preference relationship) be calculated, but only those parameters that are needed to construct a set of trade-offs associated with the economic benefit measures that are necessary for the policy analysis.[33] This technique imposes specific requirements on the information from existing studies. As a rule, these information needs are defined by the trade-off concepts measured in the literature (see Smith et al. [2002] for an example).

4.2.3. Guidance Regarding Benefits Transfer

Regardless of the type of transfer method used, in general economic benefits or economic value functions derived from a particular ecosystem study site should not necessarily be expected to be relevant for a particular policy site. Differences in both biophysical characteristics and human values dictate that great care must be taken in deciding whether the valuation of economic benefits in one context can be validly used in another context.

The challenge of transferring benefit estimates is exacerbated by the fact that often few economic benefit studies are available for a given ecosystem, thereby limiting the set of comparable cases. One consequence is that analysts sometimes rely on estimates that are too old to be reliable for new applications. For example, the RIA conducted for the CAFO rule based its willingness-to-pay estimates for improved water quality on indices taken from a contingent valuation study (Mitchell-Carson) that was more than 20 years old. In addition, due to lack of suitable previous studies, analysts might inappropriately use values or functions derived from studies designed for purposes other than those of the policy site. For example, the Mitchell-Carson study used in the CAFO RIA 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 the 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. As the example in Text Box 1 shows, several different metrics of value are possible (e.g., the number of fish anglers catch per outing; the number of fish caught per hour), and the different metrics will have very different implications for the valuation at the policy site. The choice of unit values has to be appropriate to the scale and context as well. For example, the 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 much more existing wilderness area.

Text Box 1: 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 one 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 the quality. It is assumed to be constant for each proportionate improvement. Another approach would take the same information on average trade-offs and calculate a unit value using the level of the quality variable –in this case a catch rate which itself embeds another economic decision variable –the effort a recreational fisher devotes to fishing. For the 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 is caught during a trip. These additional data of the features of the trips might be used to imply that the improvement made 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 trade-off estimates. However, the same conclusions do <u>not</u> 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 was known from technical analysis that this regulation would lead to 5% improvement in fishing success along rivers affected by a rule reducing fish entrainment. If these areas have 2,000 fishers, each taking about 3 trips per season and currently they catch 1 fish per hour, the alternative unit value transfers would be:

Table 4: 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	$5.00 per trip	improved fishing trips	$5 * 3 * 2000 = $30,000

Clearly these examples deliberately leave out some important information. Trips may be different – longer, require more travel time, or involve different features such as different species or related activities. These added features were aspects that were omitted in the example. These estimates also do not allow for the possibility that fishing success induces existing recreationists to take more trips and 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 differ by a factor of four.

Two approaches can address the challenges of determining whether and how a benefits transfer should be conducted.: a) developing *guidance* for the analyst to determine whether a value derived from a previous analysis ought to be transferred; and b) creating *procedures* to ensure the appropriateness of the choice of study site(s), the assumptions underlying the methods used, and the resulting values.

The broad categories for evaluating the appropriateness of economic benefits transfer arise from understanding that how people value the preservation or alteration of an ecosystem depends on two dimensions: (a) their preferences and (b) the nature of the biophysical system. The similarities or differences expected in preferences are likely to depend on how close the stakeholders in the two cases are along social and economic dimensions that influence the marginal willingness to pay (MWTP). 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 salmon more highly, and are more likely to value the water quality attributes regarded as important for preserving the salmon stock.

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 the 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 effectively 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), socio-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 socio-economic characteristics of the populations, major biophysical differences will affect the value even if every individual in the study case were matched by one in the policy case (e.g., the value of improving the water quality of one small lake in Minnesota compared to Texas). Therefore, the capacity to adjust for biophysical differences is typically more limited.

4.2.4. Screening Process

This procedural approach is based on the premise that a deliberate effort to examine the similarities and differences between study sites and the policy site, by both EPA analysts and those providing oversight of their work, will help to 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 presumably will 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. For example, Desvousges, Johnson and Banzhaf (1998) relied on expert judgments to convert estimates of trade-offs to avoid health-related symptoms into the implied trade-offs expressed in terms of changes in an index of the quality of life (i.e. the quality of well-being). 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 part of the regular routine of the in-house review of EPA analyses using benefits transfer. Such oversight would require the analysts to clarify the assumptions, purposes, and units of the study-site analysis so that the in-house reviewers could judge the appropriateness of the transfer. Analysts must also be fully transparent regarding the origin and 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 an additional rationale for developing databases of valuation studies. The establishment 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. While some limited efforts along these lines are currently underway (see, for example, the database being developed for recreational use values -- http://www.cof.orst.edu/cof/fr/research/ruvd/Recreation_Letter.html), a systematic effort across a wide range of ecosystems services is needed.

4.2.5. Recommendations

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 develop a Web-based catalogue 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.

5 CROSS-CUTTING ISSUES

This chapter addresses two topics important to ecological valuation in all decision contexts contexts: analysis and communication of ecological valuation information. The sections below describe special issues related to ecological valuation and committee recommendations about how they can best be addressed by EPA.

5.1. Analysis and Representation of Uncertainties in Ecological Valuation

5.1.1. Introduction

Ecosystem valuation efforts are subject to uncertainty, regardless of the method used. Assessments of uncertainty allow more informed evaluations of proposed policies and comparisons among alternative policy options. Because any given policy may result in a range of different outcomes, decision makers must be provided with sufficient information regarding what is known about the distribution of possible outcomes so that uncertainty can be taken into account when they make their policy choices. By identifying key uncertainties, it is also possible to develop potentially important insights regarding the design of research strategies, thus reducing uncertainty in future analyses.

When reflecting on the role of uncertainty in ecological valuation, three key questions arise. First, what are the major sources of uncertainty? More specifically, what types of uncertainty are likely to arise when using alternative valuation methods for specific applications? Second, what methods are available to characterize and communicate uncertainty in the results of ecological valuations? A third and final key question is associated with the types of research - data collection, improvements in measurement, theory building, theory validation, and others - that can be pursued to reduce uncertainty for particular sources in specific applications. Section 2 briefly describes the major sources of uncertainty in ecosystem and ecosystem services valuation. (See Appendix B for more specific discussions of the uncertainty arising from the use of specific valuation methods.) Section 3 then discusses two approaches to characterizing or communicating uncertainty regarding ecological values, namely, Monte Carlo analysis and expert elicitation. Finally, Section 4 discusses how uncertainty analysis can be used to set research priorities.

5.1.2. Sources of Uncertainty in Ecological Valuations

Valuation of the contributions to human well-being of proposed public policies entails three analytic steps, each potentially subject to uncertainty: predicting biophysical outcomes; predicting behavioral reactions to these outcomes; and valuing the consequences of all of these changes. It might be tempting to limit attention to uncertainty in the third step where values are ultimately estimated, but the 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 a priori. Rather, the relative magnitude of the uncertainty involved in each step in the valuation process is fundamentally an empirical question.

At each stage, uncertainty can arise from several sources. 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.[34] Finally, there are uncertainties involved in the statistical estimation of the parameters of the models used in the analysis.

For example, at the biophysical level, 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). In addition, our knowledge of these systems is incomplete and uncertain. Errors in projections of future states of ecosystems are thus unavoidable, and constitute a significant and fundamental source of uncertainty in any ecological valuation.

Every social, economic, or political forecast is also based on implicit or explicit theory 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 per se are also subject to data and theory limitations. They unavoidably rely on assumptions that introduce uncertainty. The uncertainties that arise with various methods are discussed in Appendix B. In addition, analysts are often required to apply estimated values to contexts that differ from those in which they were developed. The possibility that appropriate adjustments are not made in transferring estimates to different contexts introduces another source of uncertainty. .

In order to identify the types of uncertainty most likely to be at issue for individual valuation approaches in specific contexts, two issues are relevant: the sensitivity of an 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. <u>Approaches to assessing uncertainty</u>

Probabilistic uncertainty analysis, by its very nature, is complex, particularly in the context of ecological valuation. The simplest and probably most common approach to representing 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.

While sensitivity analysis may be sufficient for some simple problems, when used in the context of ecological valuation it 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 be able to account for the implications of all the sources of uncertainty. In addition, this approach becomes unwieldy when the outcomes relevant to the value assessment are themselves composed of multiple interrelated variables. For example, at the biophysical level it is extremely difficult 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, which are below, will generally 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 that a model be developed to predict the system's outputs from information about inputs (including parameter values). In addition, the underlying inputs into the system that are uncertain are assigned probability distributions. A computer algorithm is 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, conditional on the distributions used for the inputs.

Developments in computer performance and software over the years have substantially reduced the amount of 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, with minimal additional effort relative to that needed to produce point estimates. 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 data necessary for such characterizations already will have been collected for the development of point estimates (although characterizing other sources of uncertainty in inputs can require more effort).

Over the years, the use of Monte Carlo analysis has been shown to provide a more reliable and rich characterization of the implications of uncertainty than simple sensitivity analysis. In contrast to sensitivity analysis, Monte Carlo analysis provides information on the likelihood of particular outcomes within a range. Indeed, an understanding of the likelihood of values within a range is essential to any meaningful interpretation of that range. Without such an understanding, inappropriate conclusions may be drawn from the presentation of a range of possible outcomes. For example, when a range of possible ecological values is provided, some may assume that all values within that 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, too, often may not be 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 also been found to be useful in certain policy contexts. In particular, EPA recognized as early as 1997 that it can be an important element of risk assessments (U.S. Environmental Protection Agency 1997). However, efforts to formally quantify uncertainties rarely have been undertaken in the context of ecological valuations. More often, uncertainty has been addressed qualitatively or through sensitivity analysis.

Despite its advantages, it is unlikely that a Monte Carlo analysis will comprehensively address all sources of uncertainty in the estimation of ecological values. Thus, the results of such an analysis will likely understate the range of possible outcomes that could result from a related public policy. Nonetheless, the ranges produced by such an analysis would still provide more reliable information about the implications of known uncertainties than simple sensitivity analysis. In turn, these ranges can better inform judgments by policymakers as to the overall implications of uncertainty for their decisions. Thus, the committee urges EPA to move toward greater use of Monte Carlo analysis as a means of characterizing the uncertainties associated with estimating the value of ecological protection.

A host of “expert elicitation” methods can provide indications of the amount and nature of uncertainty associated with estimates of specific values or predictions regarding impacts of a given activity or change. (See, for example, Morgan and Henrion (1990) or Cleaves (1994).) In its very 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, such that no judgment can be discarded in favor of another, the range of disagreement reflects uncertainty. That is, if top scientists express strong divergences in their estimates, forecasts, or valuations, the existence of a high level of uncertainty is irrefutable. This relationship, however, is asymmetrical, in that narrow disagreement does not necessarily reflect certainty. In other words, 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 & Overholt, 1984: 86-87). When experts interact prior to providing their final conclusions (e.g., by exchanging estimates and adapting them in reaction to what they learn from one another), the errors due to incompleteness can be reduced. For example, biologists may benefit from the kind of information that can be provided by atmospheric chemists, 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 are sufficient to result in a table with the range of estimates. They are unable, however, to convey the degree of uncertainty that each expert would attribute to his or her estimate. This information can be conveyed, however, when the compilation of estimates also includes confidence intervals. The committee believes that expert elicitation should be used to characterize uncertainty when more formal uncertainty analysis (e.g., using Monte Carlo methods) 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. 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 due to the exclusion of detailed analysis of fault-line dynamics; bringing fault-line behavior into the analysis may lead to reductions in such uncertainty (Budnitz et al. 1997).

In addition, 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 to guide the decision on how to allocate scarce resources for research, or whether further research is worth pursuing. Some data needs are simply 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, since it always will.

5.2. Communication of Ecological Valuation Information

Nearly all of this report focuses on a new conceptual approach to ecological valuation and the methods and processes for implementing it. Much of the success of the multi-disciplinary approach described in Chapter 2, however, depends on how EPA communicates ecological valuation information as it conducts its valuation process. Although the committee has not devoted extended discussions to the particular communication challenges presented by ecological valuation, it believes that generally accepted practices for communication of technical information apply. The committee also makes several recommendations to help EPA address some of the special communication challenges that arise for ecological valuation.

Three essential functions of communication in the context of valuing the protection of ecological systems and services are: a) communication among and between technical experts and publics within the valuation process itself; b) analysts' communication of valuation analyses to decision makers; and c) EPA's communication of the results of the valuation and decision making processes to interested and affected publics. While at first glance, these communication functions may appear to be separate steps, they overlap. Success of the overall valuation process and any communication step within it depends on understanding how decision makers use valuation information. Communication spokesmen must understand how different publics and experts have framed valuation issues before they can communicate the results of a formal valuation analysis effectively.

5.2.1. Applying General Communication Principles to Ecological Valuation

Any effective communication strategy requires interactive deliberation and iteration (National Research Council 1996). Effective communication of valuation information on implementing the conceptual approach to valuation described in this report where technical experts and interested and affected publics interact to clarify the values to be represented in the analysis. The potential pool of interested parties for ecological values include interested and affected publics and scientists, especially economists, social scientists, and environmental policy scientists. There is likely a broad public audience interested in better understanding the value of protecting ecological systems and services, but also an intermediate group of those who would use data and models, who through their analyses and activities serve as important mediators for this kind of information. They will need to access technical details and models, as well as resulting value estimates.

Effective values communication requires systematically supporting interactions with interested parties, the character of which will differ depending on the technical expertise and focus of the interested parties. In general, interactive processes are critical for improving understanding, although messages or reports (such as EPA's *Draft Report on the Environment*) are also important, especially in the context of assessment. The committee recommends that EPA develop an empirical analysis of the users of valuation and adapt valuation communications to their needs.

Fundamental guidelines for risk and technical communication are generally applicable to communicating ecological values. To support decisions effectively, communications must be designed to address the recipient's goals and prior knowledge and beliefs, taking into account the effects of context and presentation (Morgan et al. 2002). For example, linear graphs 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 doesn't (Shriver, 1989).

Two examples of risk and technical communication guidelines are the communication principles from EPA's *Risk Characterization Handbook* (U.S. EPA Science Policy Committee 2000) and Guidelines for effective Web sites (Spyridakis 2000). The *Risk Characterization Handbook* principles include transparency, clarity, consistency, and reasonableness. Spyridakis provides guidance in five categories: content, organization, style, credibility, and communicating with international audiences. She provides a concise table worth consulting for those providing information via websites and provides generally accepted guidance useful for communication of valuation information, such as:

- 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 effects of communication efforts. Good communications practice requires formative evaluation of communications as part of the design process. Summative evaluation after the fact will enable assessments of effectiveness, ultimately leading to continued improvement in communications (e.g. Scriven, 1967; Rossi et al., 2003). The committee recommends that EPA evaluate ecological valuation communications to assess their effects and to learn how to improve upon them.

5.2.2. Special Communication Challenges Related to Ecological Valuation

Although application of these general communication principles will improve communications relating to ecological valuation, special challenges arise in this context. As discussed in this report ecological values can be defined qualitatively or quantitatively and they can be communicated in a wide variety of ways. Several critical design choices can influence the communication of: a) the ecological functions, systems, and services to which the valuation pertains; b) the values analyzed - whether to use a quantitative or qualitative representation and how to accommodate multiple metrics; and c) how to communicate uncertainty.

Communicating the value of protecting ecological systems and services requires conveying not only value information (in terms of such metrics as monetized values, rating scales, or the results of decision-aiding processes, for example) 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* (U.S. 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 (e.g., ecological services).

The latter can be, and often is, 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 made interactive, they will facilitate sensitivity analysis that can address audience questions about scale and aggregation and may be more effective as communication tools. The U.S. EPA Science Advisory Board has proposed this kind of framework for reporting on the condition of ecological resources (U.S. 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.[35]

The communication of ecological values is complicated by the many uses and definitions of the term value. The broad usage of the term in this report includes all the concepts of value described in Table 1 of this report (A Classification of Concepts of Value as Applied to Ecological Systems and Their Services). A corollary is that people communicate – and elicit – different kinds values in very different ways, as discussed earlier in this report. In addition, context and framing can influence strongly 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 policy contexts, ranging from national rulemakings to site-specific decision making and prioritization of environmental actions and educational outreach in the context of regional partnerships. In some cases monetization is required, whereas in others (e.g., educational outreach by regional partnerships), narratives and visual representations of values appear to play an important role. Little direct evidence exists about how such value measures are perceived, although users of cost-benefit analyses appear not to have fully considered non-monetized quantitative measures or qualitative assessments. In contrast, participative decision making exercises have used ecological indicators (quantified but not monetized) as a basis for valuation.

While there is little direct evidence on the perception of different kinds of ecological value measures *per se*, other research on perception suggests conclusions relevant for effective communication of ecological values. Response scales tend to promote responses congruent with their structure. So, for example, 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 – affect, and is unlikely to influence respondents' estimates of the value of the stimulus much (e.g. Dunn and Ashton-James, 2007, On emotional innumeracy), as is also demonstrated by studies on scope insensitivity. Visual information often dominates other representations. Taken together, this evidence suggests that quantitative cost benefit analyses will inevitably be more strongly influenced by monetized values than qualitative or non-monetized quantitative information that is not readily included in a cost benefit calculus. It also suggests that attitudes, opinions, and values elicited based on qualitative and visual stimuli will dominate those elicited based on numbers alone.

One mechanism for mitigating these disconnects related to ecological values reported in different metrics is to employ an iterative, interactive approach to eliciting, studying, and communicating values and tradeoffs, in which values are represented in multiple ways. To exemplify the potential pitfalls: verbal quantifiers (e.g., "many" or "very likely") are often proposed as a way of making 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).

Interactive communication of ecological valuation information is likely to be more effective in many circumstances 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, because of differences in educational or cultural background, and different intellectual abilities, people's understandings of presented content vary. Interactive exploration tools give the audience a chance to investigate freely the part that they are either interested in or about which they still have questions.

As argued by Strecher, Greenwood, Wang, & Dumont (1999), the advantage of interactivity lies in: a) allowance for active, instead of passive, participation of the audience; b) the ability to tailor information for individual users; c) the ability to assist the assessment process; and d) 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. Interactivity is necessitated and challenged (by the sheer computational power required) at the same time by 3-D visualization, which has become increasingly popular in visualization practice (Encarnacao et al. 1994).

In order to assess how much confidence to attribute to the projections involved in the valuation, decision makers must be informed about the analyst's own judgment of the uncertainty of the valuation and its prior steps, and the assumptions underlying the valuation analysis. Making decision makers aware of these assumptions is important because decision makers often have to explain and justify their decisions by clarifying the assumptions driving the analysis.

5.2.3. Communicating Uncertainties and Ecological Valuation

Finally, because ecological valuation involves multiple kinds of uncertainty, effective communication regarding ecological valuation involves effective communication of uncertainties both to decision makers and to the public. In order to convey to decision makers the degree of uncertainty in an ecological valuation, the simplest expressions - whether quantitative (measures of dispersion, such as variance) or qualitative (such terms as "likely," "very likely," etc.) - are typically inadequate. Analysts can specify the central tendency of an estimate (mean or median value, as appropriate) plus a confidence interval (for example, the 95% confidence interval around the mean value, or the range of estimated values), but in some cases this may require possibly arbitrary judgments on the part of the analyst (Moss & Schneider 2000). Furthermore, providing decision makers with such ranges of results can be highly misleading, because those without training in probability and statistics might be led to faulty assumptions, such as that the probability distribution of values between the end-points is uniform. Sensitivity analysis can help in this regard, although what is really needed is a description - verbal or pictorial - of the full probability distribution.

Institutional obstacles to conveying uncertainty may be related to the understandable reluctance of analysts to expose themselves and their work to the risk of appearing to be lacking in rigor. Analysts may thus have an unfortunate incentive to exclude or otherwise downplay components of their analyses that they fear may jeopardize the credibility of their overall effort. Suppressing less certain information runs counter to the need for transparency and the reality that all estimates have some degree of uncertainty (Arrow et al. 1996).

Historically, efforts to address uncertainty in ecological valuations - and more broadly, in economic benefit assessments that are part of Regulatory Impact Analyses - have been limited. But guidance set forth in the U.S. OMB Circular A-4 on Regulatory Analysis in 2003 has the potential to enhance the information provided in RIAs regarding uncertainty.

In the past, point estimates have been given far greater prominence in public documents such as RIAs and other government valuations than discussions of 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 is perhaps 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 provides a means of summarizing uncertainty that can be communicated nearly as concisely as point estimates. The need for and means of communicating uncertainty in such a fashion has been addressed in the existing literature. If a summary of uncertainty in an estimate is not given prominence relative to the estimate itself, context for interpreting the estimate and opportunities to learn from uncertainty associated with it may be lost.

Some resistance to the use of formal uncertainty assessments such as through Monte Carlo analysis, and prominent presentation of the results may be due to the perception that such analysis requires more 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 would incorrectly lead readers to conclude that the results of an ecological valuation are more certain than they actually are. Both concerns seem to be unfounded. First, as described above, developing characterizations of uncertainty (such as for inputs in a Monte Carlo analysis) often simply involves making explicit and transparent those expert judgments that necessarily already must be made to develop point estimates for those inputs. Moreover, to the extent that an uncertainty analysis is thought to be incomplete in its characterization of uncertainty, that fact can surely be communicated qualitatively. Finally, MacEachren et al. (2005) suggest animation as an effective technique for conveying uncertainties in space-time processes, which can help viewers distinguish between spatial and temporal uncertainties. It's important to communicate uncertainty appropriately in all contexts, regardless of the difficulty of doing so.

5.2.4. Recommendations

In conclusion, the committee provides the following preliminary recommendations to assist EPA in strengthening the communication of ecological valuation information.

- use the iterative approach described in this report where technical experts and interested and affected publics interact to clarify the values to be represented in the analysis to provide a foundation for effective communications
- develop an empirical analysis of the users of valuation and adapt valuation communication to their needs
- follow demonstrably effective basic practices for risk and technical communication

- evaluate communications to assess their effects and to learn how to improve upon them.
- use GIS and interactive geospatial information systems integrated with other ecological models where feasible, to represent the state of ecological systems and services. Use best cartographic principles and practices
- use interactive tools for communications, where feasible.

6 APPLYING THE APPROACH IN THREE EPA DECISION CONTEXTS

This chapter discusses a number of important issues that arise in implementing the ecological valuation approach as they arise 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 in the sections below elaborate on the three key features of the valuation approach advocated in this report as they relate to the specific decision contexts: early identification of and focus on impacts that are likely to be most important to people, predicting ecological changes in value relevant terms, and the use of multiple methods in the valuation process. The discussions are meant to be illustrative rather than comprehensive. For example, 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 B for descriptions of individual methods).

6.1. VALUATION FOR NATIONAL RULEMAKING

6.1.1. Introduction

The objective of this section is to examine the valuation of ecosystem services by the Agency with an emphasis on the monetary valuation of the economic benefits and costs of national rules promulgated by the Agency and to make recommendations as to how the C-VPESS valuation framework could be implemented in this context.

Most of the environmental laws administered by the Agency require that regulations such as environmental quality standards and emissions standards be based on a set of criteria other than economic benefits and costs. Indeed, in some cases the legislation explicitly precludes consideration of costs or economic benefits in the standard setting process. For example, in the case of the Clean Air Act, rules to establish primary ambient air quality standards for criteria air pollutants are to be set to protect human health with an adequate margin of safety. Even in those cases where the law allows consideration of the economic benefits and costs, such as the Safe Drinking Water Act, adherence to a strict "benefits must exceed costs" criterion is not required.

Nonetheless, an assessment of the economic benefits and costs of EPA actions plays an important role in the context of national rule making for a number of reasons. First, analyses of major Agency rule makings are required under the terms of Executive Order 12866 (as amended by Executive Order 13422), which states, "Each Agency shall 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 or 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, an assessment of economic benefits and costs can be mandated by law. For example, the prospective analysis of the economic benefits of the Clean Air Act Amendments of 1990 was mandated by Section 812 of the Amendments, which requires the Agency to develop periodic reports to Congress that estimate the economic benefits and costs of various provisions of the Clean Air Act. Finally, the economic benefit and cost estimates developed in national rule making may later be taken into account by executive branch officials and legislators in formulating and proposing new national rules or for other purposes. Therefore, a complete, accurate, and credible analysis of the economic benefits and costs of a given rule can have broad impacts even if the analysis does not determine whether the current rule is enacted.

Circular A-4 from the Office of Management and Budget (OMB, 2003) makes it clear that what is intended by Executive Order 12866 is an economic analysis of the benefits and costs of the proposed rules conducted in accordance with the methods and procedures of standard welfare economics. Thus, in the context of national rule making, the terms "benefit" and “cost” have specific meanings. To the extent possible, the economic benefits associated with changes in goods and services or prices due to the rule are to be measured by the sum of the individuals’ willingness to pay for them. 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. Thus, the analysis begins by specifically describing environmental conditions in affected areas, both with and without the rule. These changes are then evaluated based on individual willingness to pay and to accept compensation and aggregated over the people (or households) experiencing them. Circular

A-4 includes recognition that it might 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 still be a qualitative description of the benefits and costs (OMB, 2003, p. 10). While Circular A-4 is clear about what should be included in regulatory analyses, it does not preclude the inclusion of information drawn from non-economic valuation methods. We believe that including this information along with economic estimates of benefits and costs can prove useful to decision makers in many circumstances.

This section considers ecological valuation in the context of national rule making governed by Executive Order 12866, as amended, and OMB's Circular A-4. It focuses on the use of economic valuation methods that seek to monetize economic benefits based on the concept of willingness to pay (or accept compensation), recognizing that when monetization is not possible, the Agency should seek to quantify impacts in biophysical terms or provide a science-based, qualitative description as required by Circular A-4. As background for this discussion, the committee examined three specific examples of previous Agency economic benefits assessments: a) the Agency's benefit assessment for the final effluent guidelines for the aquaculture or the concentrated aquatic animal production industry (US EPA 2004); b) its assessment for the recent rule making regarding concentrated animal feeding operations (CAFOs) (US EPA 2002; see also the discussion in Chapter 2 of this report); and c) the prospective analysis of the benefits of the Clean Air Act Amendments of 1990 (US EPA 1999).[36] Brief descriptions of the three benefit analyses are presented in separate text boxes. These examples provide insights that are reflected in the discussion and recommendations throughout this section.

6.1.2. Implementing the Proposed Approach

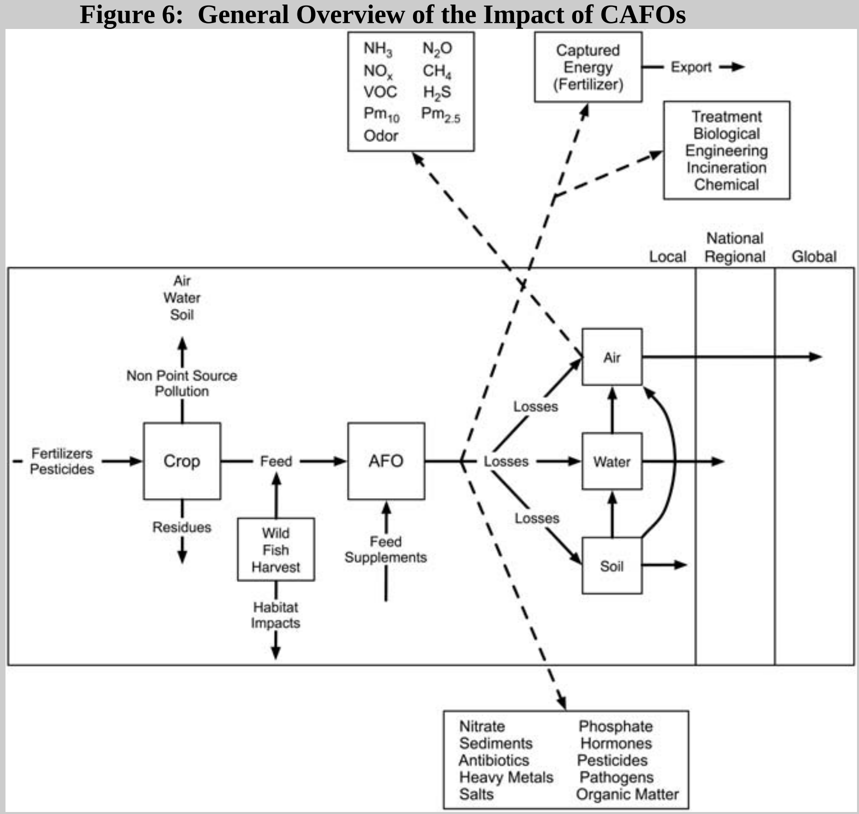
This section describes how EPA could implement the integrated and expanded approach to ecological valuation proposed in this report in the context of national rule making and RIAs. It illustrates how the three key features of the C-VPESS approach could be implemented in this context. .

6.1.2.1 Early identification and focus on socially important impacts

Identification of the socially important impacts of a given rule requires information about both the potential biophysical effects of the Agency's actions and the ecological services that matter to people. To guide the collection of this information, the Agency should develop a conceptual model of the ecological and economic system being analyzed. Conceptual models can allow the Agency to take a broad view of the complexities involved in addressing ecological changes (see discussion in 6.1.2.2). It should be standard practice for the Agency to develop such a conceptual model before other analytical work begins on an economic benefit assessment or RIA. The analytical blueprint required as part of EPA's process for developing rules should call for development of a conceptual model for ecological valuation and specify the interdisciplinary team to be involved in developing it.

Determination of the relevant ecological effects to include in the conceptual model could draw on technical studies of impacts and their magnitudes, as well as solicitation of expert opinion regarding the nature of physical and biological effects of a regulatory change. As an example, Figure 6 gives a general overview of the ecological impacts of CAFOs, which enables a comprehensive evaluation of what is happening to the environment and where the levers are for improving environmental performance. Although the CAFO rule was adopted pursuant to the Clean Water Act, RIAs do not need to restrict the benefit measure to the direct focus of the authorizing statute. As illustrated in Figure 6, the environmental impacts of CAFOs extend beyond simply the water quality impacts. For example, CAFOs are the source of interactive pollutants that impact the air as well as the water. Further, the feed supply chain providing inputs to CAFOs involves many adverse environmental impacts that need to be considered. A comprehensive overview such as this could be used to develop a conceptual model that identifies potential ecological services that might be affected by CAFO regulation.

Figure 6: General Overview of the Impact of CAFOs



The conceptual model should reflect not only the ecological science but should also be based on information about the changes that are likely to be of greatest importance to people. In the past, the Agency has generally chosen to focus on impacts that can be monetized with readily available techniques or estimates from the existing literature, or both. All three of the rule making benefit assessments that the committee reviewed provide evidence of this practice. For example, for both the CAFO and the aquaculture rules, the focus of the assessments appears to have been driven largely by the ability to use existing estimates of willingness to pay for water quality improvements taken from Carson and Mitchell contingent valuation study that had been used in previous EPA rule makings (Carson & Mitchell 1993). Rather than choosing the focus based on ability to monetize, the Agency should seek to identify those impacts that are likely to be of greatest importance to society.

The committee believes that identification of socially important impacts reflecting public preferences cannot be done deductively. Rather, it requires an examination of the evidence regarding public preferences, i.e., what matters to people. This can be gleaned from a variety of research approaches. In considering alternative approaches, it is important to distinguish the research approaches that can provide valuable information about the goods and services that are important to people from the approaches to be used to evaluate contributions to human well-being and costs. Where the analysis is being conducted to meet a mandate for economic benefit-cost analysis (as is the case for RIAs), the computation of economic benefits and costs must be consistent with the methodological requirements of the benefit-cost framework. However, the process of identifying early on the public concerns associated with a given rule can be undertaken with a variety of methods.

The suite of methods that can be used to identify the socially important ecological changes includes surveys, public meetings, focus groups, and content analysis of public comments. More specifically, possible approaches for obtaining information about public preferences and concerns include:

- Inventory of the reasons invoked in similar rule making processes in other jurisdictions (e.g., state and local).
- Inventory of the concerns expressed in public hearings at various governmental levels (perhaps with weightings based on the frequency of concerns raised). For example, local vs. national concerns can be quantified through content analysis of transcripts.
- Focus groups and surveys of concerns (can be lists of concerns, or quantified by ranking priorities).

Relevant initiatives, referenda, or community decisions might also be available in some jurisdictions to get a more robust indication of the preferences for various types of ecosystem services or the avoidance of the 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. Many ecosystem services, while well known to the scientific community, are little known or misunderstood by the general public (Weslawski, et al. 2004). For example, the full chain of connections in the production of animals in CAFOs, as described in Figure 6, is not generally understood or appreciated by the public. Similarly, the public does not generally understand the organisms and processes involved in breaking down waste products, or the services provided by those processes. For example, certain groups of soil organisms maintain soil structure by their burrowing activities, while others facilitate the use of nutrients by higher plants for growth. This problem of lack of public understanding might be exacerbated in national level analyses where ecological impacts and vulnerabilities can vary substantially across locations. 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, when expressing preferences, the public understood these contributions sufficiently well to provide informed responses.

The above discussion envisions a process under which the analysts conducting the valuation would develop the conceptual model, drawing on information provided by other experts and the public. Alternatively, the conceptual model could be developed through a more participatory process, such as mediated modeling (see Appendix B). Participatory processes can be particularly useful when the services generated by an ecosystem are not well-understood by the public and hence information about public preferences expressed through non-participatory methods may be misleading. While time and resource constraints may preclude use of such a process in many contexts, the committee suggests that EPA experiment with participatory processes (for example, holding open meetings for the public and Agency staff) to aid in the development of the conceptual model for a particular rule making. Such an approach would provide an interactive forum for determining the ecological changes that are important both biophysically and socially.

Regardless of the specific process used to develop the conceptual model and identify the ecological impacts that will be the focus of the valuation exercise, in order to increase transparency the Agency should document in its economic benefit assessments and RIAs how the decisions underlying the conceptual model were made. It should clearly identify the criteria for including effects within the core analysis and how these criteria were applied to those analytical choices. In addition, EPA should specifically document in final economic benefit assessments and RIAs how the Agency incorporated relevant input on ecological values related to the rule from public meetings on the proposed rule. It would also be helpful to provide a specific section in RIAs and economic benefit assessments describing how the Agency addressed the most significant public comments regarding ecological values and valuation. Finally, the final conceptual model that was used to guide the analysis should be part of the public record for every rule making and be available online.

6.1.2.2 Predicting biophysical changes in value-relevant terms

As discussed in Chapter 3, the C-VPESS approach calls for the use of ecological production functions to operationalize the conceptual model. This requires first a prediction of the change in relevant stressors resulting from an EPA action, and then a prediction of how that change will affect the ecosystem and ultimately the provision of ecosystem services.

In some cases, the links between stressors and ecosystem services are well understood and relatively easily quantified. Examples include the movement of phosphorus and nitrogen from manure into surrounding waters. Phosphorus in particular has been studied intensively and, importantly, its impact has been well demonstrated by whole ecosystem experiments for fresh water.[37] Similarly, species that the public or experts particularly value have been studied in sufficient detail that there are process models of production and interaction with other species. Scientists can specify an ecological production function for these organisms and use that function to predict the impact of changes in stressors.

However, for many services, developing the relevant ecological production functions is much more difficult, particularly in the context of national rule making, for a number of reasons. First, in many rule making contexts, predicting the changes in stressors is difficult. As illustrated in Figure 6, CAFO operations involve many stressors with complex interactions, which greatly complicate the development of quantitative estimates of changes in stressors. In addition, changes must be defined relative to a baseline, which might not be known. 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. In other cases the rule only required best management practices rather than quantitative maximum discharge levels, and it was difficult to predict how the adoption of best management practices would affect the stressors.

Second, many of the links between stressors and ecosystem services are not fully understood by scientists. For example, one of the important ecosystem services affected by the CAFO rule is the support of populations of fish species that are targets of recreational angling. To predict the effects of the rule on ecosystem services, one would need to know how populations of these species change and how population changes affect anglers' success rates. These links are not well understood at the level required for a comprehensive national analysis. Scientific knowledge is especially lacking in understanding the ecological impacts of substances such as heavy metals, hormones, antibiotics, and pesticides. Yet these substances can have important and far-ranging impacts that could be significant at the national level. For example, arsenic in poultry manure moves into local environments as well as through different pathways to places more distant, either through the sale of incinerator ash for fertilizer or by the use of dried and pelletized manure (Nachman, et al., 2005).

Finally, both the nature and magnitude of impacts can have substantial variation across regions of the country, implying the need for a more comprehensive analysis. Yet comprehensive analysis is particularly difficult precisely because of this scale and the associated complexity. For example, the committee's review of the CAFO rulemaking noted the following issues that stem from the varied and complex environmental consequences of CAFOs (see Figure 6):

- Multimedia effects, i.e., interrelated impacts on both water and air quality;[38]
- Impacts across multiple geographical scales (e.g., local, regional, global);[39]
- Differences in the time persistence of pollutants (e.g., days vs. decades);[40]
- Geographical clustering and the need for site-specific analysis due to uniqueness of site characteristics associated with impacts;[41] and
- Ecological impacts through supply-chain effects that are geographically dispersed.[42]

Thus, the combination of variation, complexity, and gaps in information and understanding make it difficult for the Agency to assess the ecological impacts of its actions, particularly at the national scale. Yet, this is an essential component of benefit assessment for national rule making, as laid out in Circular A-4. As noted previously, Circular A-4 requires the Agency to monetize impacts that can be monetized, quantify those that cannot be monetized but can be quantified, and describe qualitatively (based on scientifically-credible theories or evidence) impacts that cannot be quantified. Despite the difficulties described above, the approaches to predicting ecological impacts discussed in Chapter 3 can help the Agency meet the requirements of the Circular, through providing scientifically-based qualitative descriptions of ecological impacts and then, where possible, quantifying those impacts for use either as a means of quantitatively describing effects that cannot be monetized or as an input into monetization of the associated values.

As noted above, characterization of ecological impacts requires a conceptual model (see detailed discussion in Chapter 3, Section 3). Such a model would link the various levels of organizations of ecosystems that are involved in the provision of ecosystem services, as illustrated in Figure 6. A carefully developed and scientifically-based conceptual model can be used as the basis for a qualitative but detailed description of the ecological impacts of a given change. A listing that simply summarizes possible impacts, however, is insufficient. Such a summary should be accompanied by justification based on the conceptual model and the associated theoretical and empirical scientific literature. To the extent possible, the existing literature should be used to draw inferences about the likely magnitude or importance of different effects, even if only qualitatively (e.g., high, medium, low).

To move from a qualitative to a quantitative prediction of impacts, the impact of changes in stressors on the ecological system must be estimated, and the predicted changes in the ecological system must then be used to quantify predicted changes in the provision of ecosystem services. 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. The objective is to use the models to generate metrics to compare biological conditions with and without the rule to see the potential effect of the rule on the delivery of ecosystem services. Since there may be a long chain of ecological interactions between the stressors and the ecosystem services of interest, the use of quantitative models of the various components of the system will often be required to determine the net effect of these interactions on the levels of ecosystem services of concern. Outputs from these models give quantitative values of the stressor impacts even though all cannot be monetized.

Ecological models are currently utilized in rule making. However, sometimes their complexity, cost, and time constraints encourage the use of the simplest modeling approaches available that can be tailored to economic valuation. In addition, as noted previously, in the past the Agency has generally chosen to focus on impacts that can be monetized with readily available techniques and chosen ecological models based on this rather than on the important links identified in the conceptual model. For example, for the aquaculture rule, the Agency used the QUAL2E model to predict ecological impacts. While 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. The choice of QUAL2E appears to have been driven largely by the ability to link its outputs with existing estimates of willingness to pay for water quality improvements taken from Carson and Mitchell contingent valuation study that had been used in previous EPA rule makings (Carson & Mitchell 1993).

Chapter 3 Section 2 discussed the need to develop criteria for choosing among alternative models. In general, rather than basing this choice on the ability to link to existing value estimates, the Agency should use the conceptual model of the relevant ecosystem(s) to guide the selection of ecological models and seek to predict the impacts of changes in stressors on a broader set of potentially important ecosystem services.

Chapter 3 also discussed the use of indicators when available ecological models do not provide a full characterization of the relationship between ecosystem structure and function and the provision of ecosystem services. For example, fully tested techniques are available for evaluating different functional groups, and, in theory, metrics related to these groups could be used to quantify the ecological impacts of a given rule (see Figure 5). Specifically, the abundance of these groupings can be readily quantified in any before-and-after rule condition. For example, at the base of the ecosystem is its potential and realized biological diversity. Thus, metrics that look at the impact of a rule on species richness and various diversity indices can quantify potential and realized biological diversity Such metrics, however, cannot be tied directly to the provided ecosystem services without embedding this information into an ecosystem model that reveals ecological functions and related services. The key, though, is to identify those components of each of the functional levels that are most directly related to the services of interest and thus provide ecological indicators of the state of the system in relation to the change in stress level. A number of approaches are able to limit the indicators to those that will provide the most direct information relevant to the services in question. One approach is to focus on those functional groups that play a most prominent role in service provision as noted above.

Finally, the site-specific nature of many ecological impacts makes national level benefit assessments difficult. This difficulty has been noted and discussed by the SAB in previous advisories, including the *Advisory on EPA's Superfund Benefits Analysis* (2006d). Rather than conducting a "top-down" analysis at the national level, to address variability across sites the Agency should explore the use of a "bottom-up" approach. Under this approach, a number of case studies that reflect different types of ecosystems could be conducted. If information about the distribution of impacted ecosystem types is available, these case studies could in principle be aggregated to provide national level estimates of impacts. Even without full information about the distribution of ecosystem types affected by the rule, the individual 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 for which it was conducted, but also in guiding future research. In particular, it could suggest key ecosystem characteristics that would be useful in categorizing ecosystems for future valuation analyses, and for which additional information about their distribution is needed.

In summary, the initial conceptual model of a system, which provides the big picture of the possible environmental impacts of the rule, can provide a detailed and scientifically-based way of qualitatively characterizing the ecological impacts of a rule. Even if some of the identified effects cannot be quantified, this detailed characterization will provide valuable information regarding the impact of the rule. Ecological models can then be used to operationalize the conceptual model and quantify impacts, where possible. The choice of models should be guided by the conceptual model rather than by the ability to easily monetize the model's outputs. The quantification should consider not only changes in a set of final ecosystem services (e.g., clean water), but also changes in intermediate services when the contribution of those services is not fully captured by the final services included in the assessment. Even when changes in ecosystem services cannot be quantified explicitly, metrics can be used that would indicate the success of rule making in providing better ecosystem services to society. These can provide a means of quantifying impacts that cannot be monetized, and, where feasible, serve as an input into monetizing or otherwise characterizing the value of the changes in ecosystem services. In addition, site-specific variability can be addressed by including in the benefit assessment case studies for important ecosystem types, with the possibility of aggregating across these case studies if information about the distribution of ecosystem types is available.

6.1.2.3 Monetary Measures of Value

To comply with the requirements of Executive Order 12866, as amended, Circular A-4 calls for the monetization of economic benefits whenever possible. Although a variety of methods can be used to determine values for purposes of identifying socially relevant ecosystem characteristics and services (see discussion in Chapter 6 Section 1.2.1) and for value assessments in other contexts (see Chapter 6, Sections 2 and 3), in the context of economic benefit-cost analysis the only approach to monetization consistent with the premises underlying this analysis is the use of economic valuation methods. Monetizing values using other methods and then aggregating the resulting estimates is problematic because it implies adding together numbers that are based on quite different methods and underlying premises. Thus, for both theoretical and empirical consistency, the monetization of benefits in a benefit-cost analysis should be based on economic valuation.

Economic valuation methods are well-developed and there is a large literature demonstrating their application. (See Chapter 4 and Appendix B for descriptions of economic valuation methods.) Nonetheless, applying these methods to national-level analyses of the ecological benefits of a rule is difficult. As with the prediction of ecological impacts, the value of ecological impacts is likely to be site-specific, depending on local conditions and the characteristics of the affected population. As a result, generalizing to the national level is difficult. In principle, this variability across affected sites could be addressed by conducting case studies and aggregating the results across the sites affected by the rule. However, time and resource constraints may preclude doing this kind of original economic benefits research. As a result, the Agency will generally need to rely on benefits transfer instead. Although the existing economic valuation literature is extensive, most of the previous ecological valuation studies that might serve as study sites are not national in scope. Rather, they involve valuing relatively localized changes affecting a local or regional population. In addition, these studies have generally focused on a limited number of ecosystem characteristics or services (primarily related to recreation). Few studies provide national level value estimates for a range of services that could be readily used in a national level benefit assessment. [is this an accurate statement? I think so, but need to check – KS]

The Agency needs to ensure that the call for monetization, coupled with the need to use benefits transfer and 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. As noted above, in the past Agency decisions regarding the focus of ecological benefit assessments have been driven to a large extent by the objective of monetizing the value of impacts at the national level using benefits transfer. This applies both to the types of ecosystem services included in the detailed assessments and to the choice of the ecological models used to predict biophysical impacts. For example, the Agency's assessment of the CAFO rule focused primarily on valuing recreational impacts, driven to a large extent by the ability to link the QUAL2E model with off-the-shelf monetary estimates of willingness to pay for changes in water quality indices taken from the Carson-Mitchell contingent valuation (CV) study. The principal advantage of this approach is that it utilizes a study designed to be national in scope and has a simple willingness-to-pay relationship that allows the analysis to be done relatively quickly, without new research and the associated significant expenditures on research resources. In addition, it can be applied using a straightforward conceptual logic that is easy to understand. However, use of the Carson-Mitchell estimates has a number of limitations that raise concerns about the resulting economic benefit estimates. Most notably, the study was conducted more than 20 years ago. In addition, it was designed for a different purpose and was not intended to apply to specific rivers or lakes. The water quality index was highly simplified and was never designed to reflect ecological services related to water quality (other than those related to fish). Thus, in an effort to focus on effects that could be readily monetized at the national level using benefits transfer, the Agency appears to have limited both the types of services considered and the ecological and economic models used to estimate the impacts of the rule on those services.

Since the Agency will inevitably need to rely on benefits transfer for many, if not most, RIAs, it must take care to ensure that the transfer of economic benefit estimates is appropriate. Chapter 4 discusses issues that arise in transferring economic benefit estimates and provides suggestions for ensuring that the transfer is appropriate, given both the biophysical and the socio-economic characteristics of the study and policy sites. The use of the Carson-Mitchell study to estimate the benefits associated with the proposed CAFO rule provides an example where the transfer of benefits was problematic. However, in other cases EPA has appropriately used benefit transfer. For example, EPA estimated the recreational benefits of reducing acid deposition in Adirondack's lakes by transferring benefit estimates from a fairly recent published study of recreational angling choices of households in New York, New Hampshire, Maine, and Vermont (Montgomery and Needelman 1997; for more detail see Text Box 4: The Prospective Economic Benefits of the Clean Air Act Amendments). This study explicitly compared populations of target species and pH levels at the source and target sites. If the socio-economic characteristics of the population of these four New England states match those of the Adirondacks region of New York State, this study is a good source for economic benefits transfer.

The above example illustrates a benefits transfer based on an individual RUM study. As discussed in Chapter 4, benefits transfer can also be based on meta-analyses, which combine information about values from multiple studies. For example, several studies have used random utility models to link physical descriptors of water quality to recreation behavior to estimate the willingness to pay or willingness to accept per recreational trip for a change in water quality (e.g., water quality that improves to boatable or fishable status), had it been experienced in each of the areas. These estimates could be used in a summary or meta function describing how the local choice set of recreation sites and economic characteristics of the recreationists, as well as the character of the changes from existing baseline conditions, influenced the estimates of unit economic benefits. If the changes considered in these studies are comparable to what would have been experienced under the proposed rule, then the meta function could be used to estimate values at sites affected by the rule. (references)

Alternatively, the models could be adapted to be directly applied to choice sets composed for affected areas. In this case the recreation behavior necessary to operationalize the model could be extracted for some of the areas from EPA's National Survey on Recreation and the Environment (NSRE) for 2000 and 2004. The logic involved has two key steps: a) translation of the effect of the rule for a set of local water quality conditions that is matched to some set of economic behavior for that area that is influenced by the water quality; and b) adaptation of an economic model of trade-offs people would be willing to make to improve one or more aspects of the water quality for the area so that economic and ecological factors affecting the trade-offs are represented in the summary function. There is precedent in the literature on economic benefits transfer for these types of analyses (see Rosenberger and Loomis 2003 and Navrud (in press), for examples of how this logic might be used in benefits transfer). [I don't understand the idea behind this second approach from the description here. What is the key distinction? I think it would be helpful to have some clarification, but I can't revise this to be clearer without more info. KS]

A second class of studies for transferring benefits using meta analyses are the stated preference and stated choice studies (such as Carson and Mitchell) that highlight water quality attributes. While the record here is not as extensive as it is for the revealed preference random utility studies (RUMs), there are several candidate studies (references??). These analyses are based on surveys that require respondents to choose from among a set of options, such as plans for reducing effluents or improving water quality. The logic is comparable to that described for the RUM. The effects of the rule need to be adapted to the features of each of the models, and the projected unit economic benefits must be derived. Then the factors affecting the economic benefit measure for each are modeled in a summary analysis that can be applied to other areas that are affected by the rule. [It is not clear to me whether this paragraph was intended to refer to meta-analyses based on stated preference, or to the transfer of benefit estimates from individual SP studies (like Carson and Mitchell). It looks like the former, but I'm not sure. We should try to clarify this. KS]

As noted above, the existing literature on economic valuation of ecosystem services has focused to a large extent on estimating the value of recreational impacts. In addition to recreational impacts, some ecological services affect the well-being of homeowners living near the ecological systems providing these services. Examples include water regulation and flood control, and the amenities associated with healthy populations of plants and animals. The willingness of residents to pay for these services can be capitalized into housing prices. The hedonic property value method can be used to obtain estimates of the values of these services. 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 studies such as these could be candidates for use in an economic benefit transfer. However, as with the recreation studies, these studies tend to be local rather than national in scope, which makes extrapolation to national level benefit assessment difficult.

The above discussion suggests that, to improve the Agency's ability to value the ecological impacts of national rules using economic valuation methods, additional research is needed to a) generate national-level value estimates that can be used in benefits transfer, particularly for recurring rulemakings, b) generate information about the distribution of ecosystem and population characteristics across local or regional sites that could be used to aggregate localized case studies in a "bottom-up" approach to national-level analysis, and c) expand the range of ecosystem services valued using economic valuation methods so that benefits transfers can incorporate a wider range of services.

6.1.2.4 The Role of Other Valuation Methods

Although Circular A-4 calls for the use of economic valuation methods to monetize benefits, other valuation methods can also play an important role in RIAs. The valuation approach proposed in this report envisions three possible roles for other valuation methods in the context of national rule making.

First, as already discussed, other methods can be used to identify early in the process those ecosystem characteristics or services that are likely to be socially important and hence should be a focus of the analysis. For example, focus groups, participatory/interactive processes, surveys of attitudes and judgments, analyses of public views regarding related ecological impacts (expressed through hearings, public comments, citizen juries, etc.) and other similar methods can provide valuable public input into the development of a conceptual model that captures the most important ecosystem services.

Second, although in principle economic valuation methods can fully capture the relevant population's willingness to pay for changes in ecological systems and services, as discussed above, in practice there are significant limitations that can make this very difficult, particularly at the national level. When benefits cannot be monetized, Circular A-4 calls for them to be quantified, or at least qualitatively described, to the extent possible, using scientifically-based analysis. When estimates of willingness to pay cannot be generated for the full range of important ecosystem services, it may be possible to use other methods as proxies for, or indicators of, willingness to pay. To the extent that other methods generate non-monetary measures that are likely to be correlated with willingness to pay, they can provide useful information about likely changes in willingness to pay when direct monetary measures of those changes are not available. For example, economic benefit indicators (see Chapter 4 and Appendix B) can be viewed as non-monetary measures of impacts that are likely to be correlated with willingness to pay; ceteris *paribus*, the more people living within the vicinity of an impacted ecosystem, the higher is the willingness to pay to protect that system likely to be. Similarly, *ceteris paribus*, the more people who judge the protection of a given ecosystem service to be "somewhat important" or "very important" in a survey of attitudes and judgments, the higher is the willingness to protect that service likely to be. While use of these proxies would not provide monetary estimates of benefits that could be compared to cost, they can provide important information about possible benefits. Care must be taken, however, to avoid misinterpretation of these proxies. For example, just because a large population lives in the vicinity of an impacted 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 would need information not only about the number of people affected but also about the importance that individuals attach to the service, as revealed for example through surveys or other methods.

Finally, although benefit-cost analysis requires the use of economic valuation to estimate benefits, RIAs need not be limited to information generated for use in a strict comparison of benefits and costs. Information about other sources of value that do not fit within the theoretical framework underlying benefit-cost analysis can still be useful to policy makers when making decisions related to ecological protection. For example, the religious, spiritual, or cultural value of some ecosystems and their related services may be an important consideration not adequately captured by standard measures of willingness to pay. Valuation methods other than economic valuation can provide information about these other sources of value. However, as noted previously, even when other methods yield value estimates measured in monetary units (dollars), these values should not be added to monetary estimates derived from economic valuation methods, since they are not based on the same underlying assumptions and principles.

6.1.2.5 Reporting Value Estimates

To assess and report on changes in service flows, economic benefit assessments and RIAs should feature prominent discussions of ecological services that describe how these services were identified and analytical choices were made. In addition, consistent with the guidance in Circular A-4, they should clearly identify the values that were a) monetized using economic valuation methods; b) quantified (but not monetized); and c) described qualitatively. 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 they provide proxies for, or indicators of, willingness to pay (or accept). If methods other than economic valuation are used to capture sources of value other than those typically reflected in willingness to pay, the methods used and the results should be described in a separate section of the RIA as supplemental information.

Rather than simply designating some impacts as “non-monetized,” as in the CAFO benefit assessment, the committee recommends that the quantified but non-monetized impacts be reported explicitly (in conjunction with the monetized economic benefits). For those described only in biophysical terms, they should also be measured in the units that make sense from a biological perspective, and the non-quantifiable impacts should be described in as much detail as is feasible. Furthermore, any summary listing of the economic benefits and costs should include all three types of contributions to human welfare with the monetized and quantified values measured in the appropriate units (dollars or biophysical units). When monetized economic benefits are aggregated, the resulting sum should always be described as the “Total Economic Monetized Benefits” rather than the “Total Benefits.”

In the past, EPA has sometimes reflected the non-monetized economic benefits in aggregate measures of benefits by including an entry in the summary table of economic benefits (and costs) such as +X or +B to indicate the unknown monetary value that should be added to economic benefits if the value could be determined. While such an approach indicates that a measure of monetary economic benefits (and costs, too, if appropriate) is incomplete, the +X or +B designation provides insufficient information and can be easily overlooked when the results of the economic benefit assessment are used. Always designating the sum as "Total Monetized Economic Benefits" provides a continual reminder of what is (or is not) included in this measure. By also including key quantified but non-monetized impacts that are measured in biophysical units, the Agency will be providing a more accurate and complete indication of total benefits, as called for by Circular A-4.

Because of the difficulties in estimating biophysical impacts of an EPA rule and the associated economic benefits or costs that result from that rule, the Agency must characterize the uncertainty associated with its assessment. EPA should include a separate chapter on "Uncertainty Characterization" in each economic benefit assessment and RIA. This chapter should discuss the scope of the economic 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, information similar to that reported in the Agency's prospective analysis of the economic benefits and costs of the Clean Air Act Amendments (US EPA, 1999) should be reported and a summary of this information should appear in the executive summary of the RIA or economic benefit assessment. Specifically, EPA should report: a) potential source of error; b) the direction of potential bias for overall monetary economic benefits estimate; and c) the likely significance relative to key uncertainties in the overall monetary estimate.

6.1.3. Conclusions

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 could be applied in the national rulemaking context. This would entail challenges, but important opportunities for improvement as well.

To ensure that the benefit assessment considers all socially important impacts, the Agency should develop a conceptual model of the ecological and economic system being analyzed to serve as a guide or road map for the benefit assessment. Development of the model requires input from ecologists, social scientists and the public. The conceptual model should adopt a multimedia (air and water) perspective, since focusing on a single media (such as water quality) can miss major interactions among media that impact ecosystem services. In developing the conceptual model, the Agency should draw from research based on a variety of different methods to determine early on in the process which of the possible ecological impacts are likely to be of greatest importance to people and hence should be the focus of the assessment. The committee recommends that the Agency consider use of an open, interactive public forum for identifying issues of concern. In addition, it should document in the RIA the process used to identify those ecosystem characteristics and services that were included in the assessment, as well as those that were excluded.

Given a conceptual model, a significant challenge to ecological benefits assessment for national rules is predicting how the levels of ecosystem services would be affected by the rule, particularly at the national level. The combination of variation, complexity, and gaps in information and understanding make it difficult for the Agency to assess the ecological impacts of its actions, particularly at the national scale. Reasons for this include:

- In some cases (e.g., requirements for best management practices, absence of baseline data), the changes in the levels of ecological stressors are not known.
- The models used in the analysis often do not predict changes in the relevant ecosystem services. For example, the links between outputs of some ecological models and human uses of the ecosystem are not known (e.g., the relationship between changes in fish populations and changes in recreational angling).
- The needed ecological data are often not available.

The Agency should take steps to improve its capacity for predicting the ecological consequences of Agency policies and regulations at the national level. Possible steps include developing better quantitative ecosystem models for predicting the consequences of changes in ecological stressors on the production of ecosystem services and developing better baseline data on ecological stressors and ecosystem service flows. In addition, site-specific variability can be addressed by including in the benefit assessment case studies for important ecosystem types, with the possibility of aggregating across these case studies if information about the distribution of ecosystem types is available. This bottom-up approach would proceed by establishing separate estimates for each regional grouping of similar facilities and then adding them together to obtain the national estimate.

Circular A-4, which serves as the Agency's guide for preparing RIAs, requires that benefits be monetized, if possible, using economic values. Methods exist for estimating economic values for at least some ecosystem services; these methods have been used to estimate values in a number of cases. However, applying these methods to new cases (including an expanded range of ecosystem services) to analyze proposed regulations at the national level could require original research that is costly and time consuming. As a consequence, the Agency will often have to resort to economic benefits transfers to estimate ecosystem values for rule making. Since economic values are context dependent, steps must be taken to ensure that the transfer of economic benefits information is appropriate. This will very likely require a much larger set of value estimates than is currently available. The Agency should continue to support research to build an improved database for economic benefits transfer for ecosystem service valuation.

In the past, the Agency has selected the ecosystem services to include in its assessment as well as the ecological models to use in quantifying impacts based on the objective of monetizing benefits at the national level using off-the-shelf value estimates (benefits transfer). This can lead to benefit estimates that are not scientifically sound. Instead, the Agency should use the conceptual model to drive the choices about which services to include, even if that choice implies an inability to monetize the associated values at the national level. In cases where benefits cannot be monetized, Circular A-4 requires that the impacts be quantified, if feasible, or qualitatively characterized. The conceptual model can provide a detailed and scientifically-based way of qualitatively characterizing the ecological impacts of a rule. Ecological models can then be used to quantify impacts, where possible. The choice of models should be guided by the conceptual model rather than by the ability to easily monetize the model's outputs. It might also be possible to use other non-monetary valuation methods to develop metrics that would likely be strongly correlated with willingness to pay and hence serve as a proxy or indicator measure when monetized values are not available.

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. In addition, aggregate monetized economic benefits should be labeled as "Total Monetized Economic Benefits" rather than "Total Benefits." In addition, EPA should include a separate chapter on "Uncertainty Characterization" in each economic benefit assessment and RIA.

Methods also exist for estimating non-economic values for at least some ecosystem services. While these methods do not properly fit within a formal economic benefit-cost analysis, they can provide important additional information to support decision making. When value estimates from these methods are included in RIAs, the RIA should clearly both the method and the results in a separate section.

In general, EPA should seek to build additional capacity, externally and in-house, specifically designed to facilitate ecological valuation for national rulemaking, particularly for recurring rule makings. The committee advises the Agency to develop an extramural grant program focused on method development specifically for recurring rule makings (e.g., for rule making associated with programs like EPA's National Ambient Air Quality Standards or Effluent Guideline programs). Such a focused effort could help develop methods for expanded applications of monetary and non-monetary methods for valuing ecological effects, which will assist Agency regulatory programs addressing ecological protection issues.

The committee also advises the Agency to host annual Agency-wide meetings to discuss methods used in regulatory impact analyses and economic benefits assessments, and methods needed for full characterization of the effects addressed by the regulatory actions associated with those efforts. One objective of this effort should be to build an improved database for economic benefits transfer for ecosystem service valuation.

Text Box 2: The Aquaculture Effluent Guidelines

Title III of the Clean Water Act (CWA) 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 would require that all applicable facilities prevent discharge of drugs and pesticides that have been spilled and 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 (U.S. EPA, 2004).

For most of these requirements, it is not possible to specify the change in the levels of environmental stressors since the rule called for adoption of "best management practices" rather than imposing specific quantitative maximum discharge levels. In addition, for most of these stressors, baseline data on discharges in the absence of the rule were not available.

The Agency identified the following potential ecological stressors: 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 (for example, drugs and pathogens) were thought by the Agency to be very small in magnitude and not requiring further analysis. To this list, C-VPESS added habitat alteration from changes in water flows.

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

- The Agency lacked data on baseline stressor levels and how regulation would change these levels.
- The Agency did not use a model capable of characterizing a wide range of ecological effects. The Agency used the 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 described below.

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 (1993). 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. This contingent valuation survey was conducted in 1982-83 and was not intended to apply to specific rivers or lakes.

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.

Text Box 3: The CAFO Effluent Guidelines

Context:

In recent years there has been substantial growth of the livestock industry in the United States as well as in many other parts of the world. This growth has been characterized by a dramatic reduction in the number of farm operations producing livestock, and a big increase in the number of animals per farm unit. Finally, there has been a geographic concentration of these intensive units, particularly in the Southeast and mid-Atlantic states. Manure production in these intensive facilities simply exceeds the capacity of nearby farmland to utilize it in plant production, resulting in a major disposal issue and hence a threat to ground and surface waters as well as a problem with local air pollution.

These structural changes in the industry led to the present CAFO rule that was issued in December of 2002. This rule focused on the largest operations that represent the greatest environmental threats. These units are required to implement comprehensive nutrient management plans and to submit annual reports summarizing their operations.

What are the environmental issues?

The manure from livestock operations produces a variety of potential pollutants which 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). Further, CAFO facilities 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, which are covered in the CAFO rule, excess nutrients can cause direct impacts on human water supply through excess nitrates, impacts on agriculture through excess salts in irrigation waters, as well as 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, for example, crops and shellfish. The potential human health impacts of antibiotics and hormones in wastes have not been well identified but are of concern.

How were the environmental impacts quantified?

Of all of 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 quantifying the economic benefits that would accrue from the manure management requirements of the CAFO rule. To do so they utilized the GLEAMS model (Groundwater Loading Effects of Agricultural Management Systems) which uses natural inputs of precipitation, radiation, temperature, and soil type and management inputs of irrigation, crop type, tillage, fertilizer and pesticides. 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 calculated for large- and medium-sized CAFOS that would result from the application of the rule due to nutrient management plans.

How were the economic benefits valued?

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 & Mitchell study described in Text Box 2: The Aquaculture Effluent Guidelines above. 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 whole series of potential impacts were not included in the economic benefits analysis that would relate to 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. Other ecosystem impacts that were not considered include the potential changes to aquatic ecosystem functioning that relate to their capacity to produce goods of value to society.

Text Box 4: The Prospective Economic Benefits of the Clean Air Act Amendments

The first Prospective Benefit-Cost Analysis mandated by the 1990 Clean Air Act (CAA) Amendments included estimates of the economic benefits of protecting ecosystems related to reductions in air pollutants to be expected from the amendments (U.S. EPA, 1999). The Agency included qualitative discussions of the following potential ecological effects of atmospheric pollutants based on a review of the peer-reviewed literature (US EPA, 1999, Chapter 7, and pp. E-2-E-9):

Table 5: Table of Qualitative Discussions of Potential Ecological Effects of Atmospheric Pollutants Discussed in the First Prospective Benefit Cost Analysis (1999)

Pollutant	Acute Effects	Long-term Effects
Acidic deposition	Direct toxic effects to plant leaves and aquatic organisms	• Progressive deterioration of soil quality • Chronic acidification of surface waters
Nitrogen deposition		• Saturation of terrestrial ecosystems with nitrogen • Progressive enrichment of coastal estuaries
Mercury, dioxins	Direct toxic effects to animals	• Persistence in biogeochemical cycles • Accumulation in the food chain
Ozone	Direct toxic effects to plant leaves.	• Alterations of ecosystem wide patterns of energy flow and nutrient cycling

The Agency used two criteria to narrow the scope of work for quantification of impacts:

- The endpoint must be an identifiable service flow
- 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:

- Economic benefits to commercial agriculture associated with reductions in ozone,
- Economic benefits to commercial forestry associated with reductions in ozone,
- Economic benefits to recreational anglers in the Adirondacks lakes region due to reductions in acidic deposition.

For agriculture, the Agency used crop yield loss functions from the National Crop Loss Assessment Network to estimate changes in yields. These yield effects were then fed into a model of national markets for agricultural crops (AGSIM) to estimate changes in consumers' and producers' surplus. The Agency did not quantify or monetize effects on ornamental plantings, nurseries, or flower growers.

For commercial forestry, the PnET-II model was used to estimate the effects of elevated ambient ozone on timber growth. The PnET-II model is a monthly time step canopy to stand level model of forest carbon and water balances based on maximum net photosynthesis as a function of foliar nitrogen content. The 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 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 economic surplus (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.

For estimating 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). This was a random utility model of site choice. Measured pHof 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 (see Phaneuf and Smith [2005] and Freeman [1995] for reviews). To the extent that populations of target species are correlated with pH levels, pH will be 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.

Modeled reductions in acidification were used as an input to the Montgomery-Needelman (1997) site choice model to simulate the effect of reduced acidification on angler choice and angler welfare. This simulation requires access to the data used to estimate the model because the economic benefit measures to anglers depend on individual anglers' travel costs and site alternatives.

The Agency also presented an estimate of the economic benefits of reducing nitrogen deposition in coastal estuaries along the east coast of the US. In order to estimate the economic benefits of reduced nitrogen deposition in coastal estuaries, it would be necessary to carry out the following steps:

1. Estimate the changes in nitrogen deposition. The Agency was able to do this for the three estuaries covered in the Prospective Analysis.
2. Use appropriate ecological models to estimate the changes in the populations of species of concern to people. These species include fish and shellfish species that are targets of commercial exploitation, fish species that are targets of recreational anglers, and perhaps other species that are of concern to people such as birds and marine mammals. Decreasing atmospheric deposition of nitrogen was expected to reduce the deterioration of breeding grounds for fisheries and reduce the habitat loss for aquatic and avian biota. It might be

necessary not only to estimate population changes for species that are resident in and exploited within the estuaries but also for species that use the estuaries for reproduction and shelter of young or that are dependent on species from these estuaries as a food source at some stage in their life cycle.

3. Estimate people's willingness to pay for increases in the services provided by these species. There are models that can be used to do this for commercial and recreational fisheries. But there is very little data on willingness to pay for other types of services such as bird watching and whale watching.

The Agency was unable to establish the necessary ecological linkages to quantify these recreational and commercial fishery effects. Hence it resorted to an avoided cost or replacement cost measure of economic benefits. Reductions in nitrogen deposition reaching Long Island Sound, Chesapeake Bay, and Tampa Bay were estimated. 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, that is, that the alternative's costs would actually be avoided. Since 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.

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6.2. VALUATION FOR SITE-SPECIFIC DECISIONS

6.2.1. Introduction

The Environmental Protection Agency makes many decisions at the local level, including 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, these 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.

The U.S. EPA Science Advisory Board staff, with assistance from the Agency's National Regional Science Council, surveyed the regional offices to assess their needs for valuation information related to Agency regulatory programs. 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 (U.S. EPA Science Advisory Board Staff 2004). The committee's goal is to help direct the Agency in building the capacity to satisfy that need. Thus, 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 discussion of 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). The discussion that follows is applicable to any remediation and redevelopment processes for contaminated properties that contain the following basic and common elements:

a) Site identification identification, selection, and prioritization of sites
b) Site characterization - establish site condition
c) Site assessment - evaluation of risks and impacts
d) Selection of remedial and redevelopment approaches
e) Performance assessment - clean up and redevelopment
f) Public communication - assessment results; proposed actions and outcomes

The goal in this section is to explore how the use of valuation methods can positively influence individual steps in a remediation and redevelopment process and lead to a better outcome. As appropriate, individual valuation approaches or methods relevant to specific steps are identified and discussed. This section of the report aligns its analysis with a white paper funded by EPA's Superfund Program (Wilson 2005) to evaluate the potential of valuation for redevelopment of contaminated sites. The white paper provides an assessment of the improvement in ecosystem service and implied ecological value from the remediation and redevelopment of Superfund sites. Although the Wilson paper doesn't actually perform a formal valuation for any individual redeveloped property, it does provide a useful starting point for further exploration of the utility of valuation methods in the remediation and redevelopment process. In preparation for his analysis, Wilson (2005) reviewed approximately 40 superfund cases before selecting three case studies that represent urban (Charles George Landfill), suburban (Avtex Fibers), and exurban (Leviathan Mine) environments. The committee has chosen to analyze and rely on these same three cases, as well as an additional urban example, the DuPage Landfill, because it provides a useful counterpoint to the Charles George Landfill example. The DuPage example shows how an early focus on ecosystem services can more completely identify potential ecosystem services that can be targeted during the remediation and restoration phases. A brief overview of each of these cases is provided in Text Box 6 through Text Box 9 below.

6.2.2. <u>Opportunities for using valuation to inform remediation and redevelopment decision.</u>

The Superfund process and its individual steps or stages are well defined (U.S. 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 (U.S. Environmental Protection Agency, 2004) has integrated consideration of an upstream redevelopment focus into the remediation process. The Agency built the Reuse Assessment tool (Davis, 2001) to integrate a focus on land use into the Superfund process. Integrating remediation and redevelopment makes evident the need to bring value concepts and considerations to the beginning of the process and carry them through the individual steps or stages of the process. Net Environmental Benefit Assessment (NEBA) (Efroymson et al. 2004, see Text Box 5) is a recent advance in thinking that provides a framework for using valuation tools to inform the comparison of alternative remedial strategies based on net impacts on ecological services. Similar efforts are needed for other steps in the remediation and development process.

Text Box 5: 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 7, 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."

Since NEBA is a framework, the 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 (i.e. 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 7: Framework for Net Environmental Benefit Analysis (from Efoymson et al., 2003)

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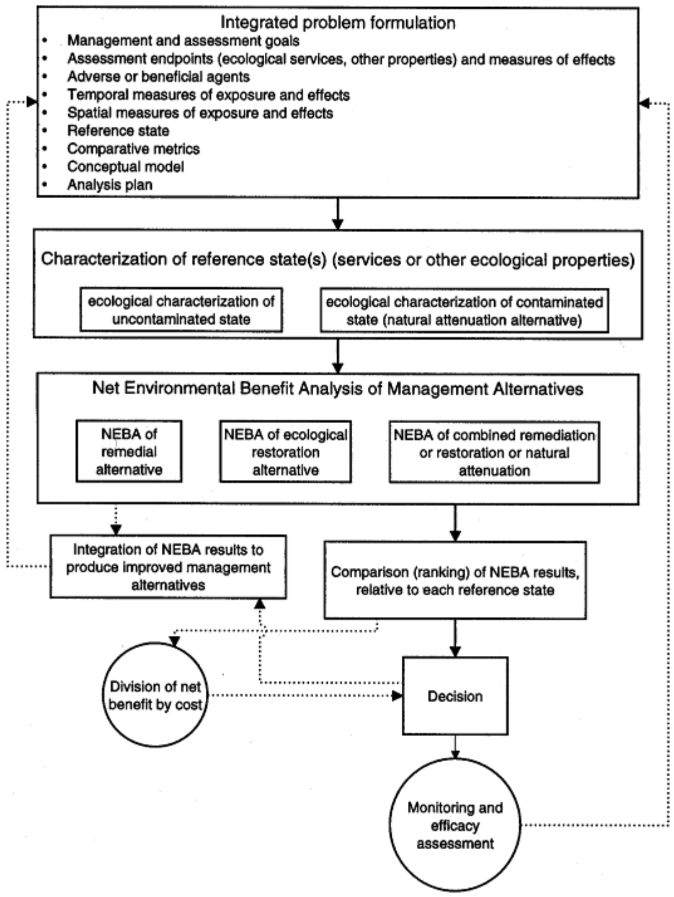
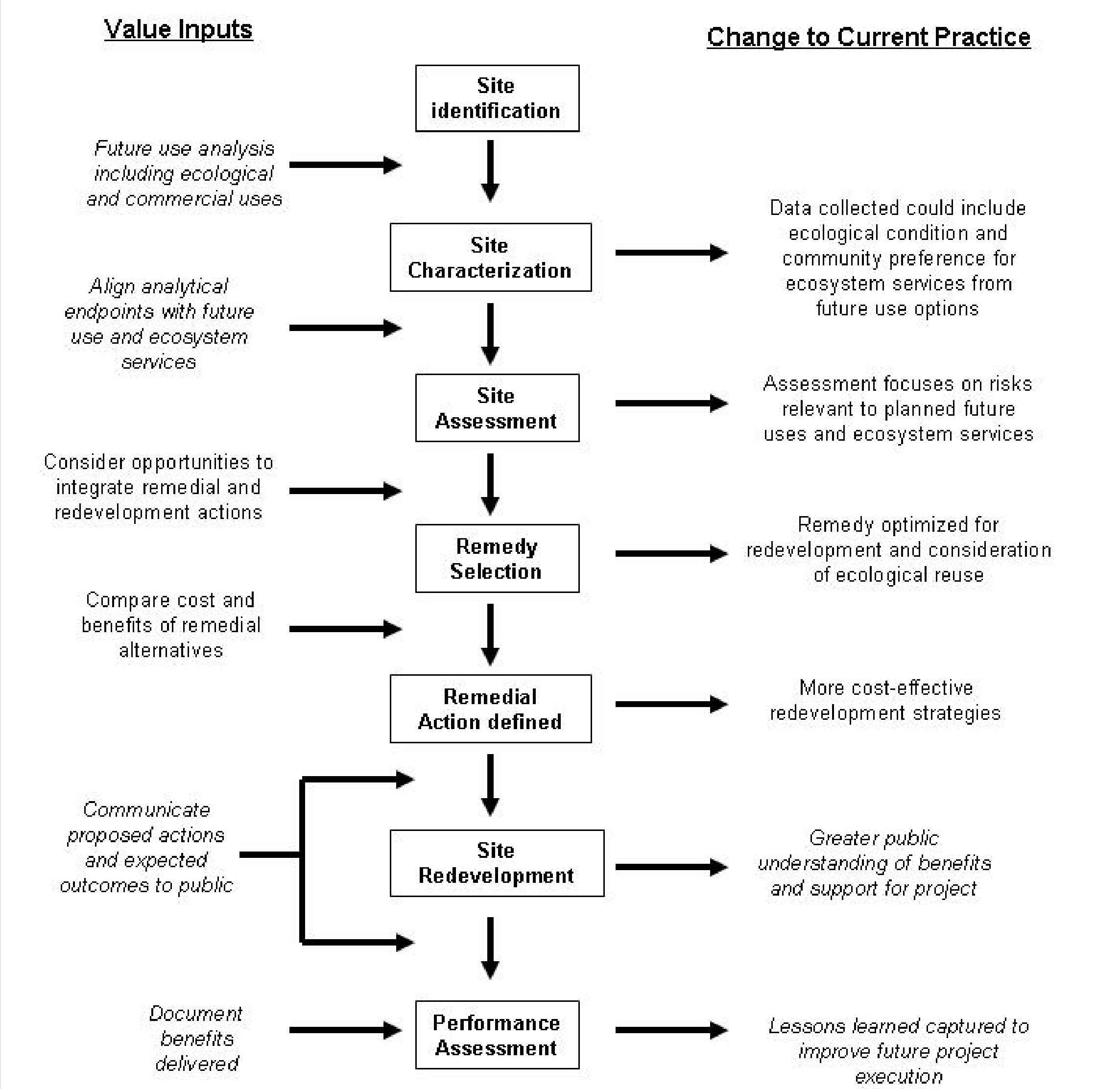


As noted, a generic process that encompasses remediation and redevelopment would include a series of steps or discrete activities that involve ecological valuation. Figure 8 represents a generic remedial process in which opportunities to include valuation concepts and assessment methods have been identified. As is clearly shown, early recognition of future uses and ecosystem services that matter to people carries through to inform assessment of the site and the ultimate selection of remedial actions and redevelopment options. Optimally, expressing expected or actual contributions to human well-being will lead to more effective communication with concerned publics. The following sections discuss opportunities and utility of adapting valuation methods to this new merged process.

Figure 8: Integration of Valuation information with traditional process to achieve improved performance.



Valuation methodologies can be most useful for identifying how a site and the current or potential ecosystem services matter to the surrounding community. Such methods should be focused on determining what contributions to human well-being have been derived, or can be derived, from the site and how potential effects on ecological components diminish those contributions. When the ecosystem services that matter to people are well-defined and when the assessments of ecological production and risk can be coupled with these specific services, then the outcome is likely to be a remediation and redevelopment plan that is targeted on what really matters to the local community. A key recommendation, therefore, is that consideration of ecosystem services and their contributions to human well-being and other forms of value be considered from the earliest stages of addressing contaminated properties.

Even as early in the management process as site selection or prioritization, tools that allow for comparison among sites for their potential to provide ecosystem services could be informative. Assessment of the contribution of ecosystems or ecological protection to human well-being should be considered in the design of any site characterization plan. While a typical site characterization is focused 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 service flows would better align ecological risk and economic benefit assessments, as well as other kinds of assessments of 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 assure that the remedial actions will address the restoration of the contributions to human well-being derived from any important ecosystem service flows that have been diminished or disrupted. Aligning risk assessment endpoints with ecosystem services should result in multiple benefits, such as: a) improved alignment with community goals; b) improved ability to perform meaningful assessments of economic benefits and other assessments of contributions to human well-being; c) improved ability to communicate proposed actions; and d) improved ability to monitor and demonstrate performance.

The success of remediation and redevelopment of contaminated sites depends in great part on the degree to which the ecosystem services and associated contributions to human well-being important to the community are either protected or restored. If, as recommended, values have been broadly explored and effectively integrated into the site assessment and remedy selection processes, then measures of performance will be apparent. Ecological measures of productivity or the aerial extent of conditions directly linked in an understandable manner to valued ecosystem service flows will be useful in tracking the performance of remediation and redevelopment processes. Advancing the Agency's capability to do performance evaluation 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 biological, chemical, and physical sciences and engineering principles 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 the expected or actual outcomes will facilitate effective communication.

6.2.3. Recommendations and discussion of valuation through illustrative site-specific examples

Chapter 2, Section 6 of this report included high-level recommendations. The committee recommended that ecological values and contributions to human well-being derived from ecosystem services be considered from the outset when framing any analytical process to support Agency decisions and associated actions. The recommendations direct the Agency to broaden its consideration of the types of ecological values and align them with what matters most to the people involved in or affected by the decision.

In the following text, the general recommendations of Chapter 2 are applied to valuation at 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. Text Box 6 and Text Box 7 provide background on the urban landfill cases. Text Box 8 and Text Box 9 provide background on the suburban and ex-urban cases respectively.

Text Box 6: Charles George Landfill

From the late 1950s until 1967, the Charles-George Reclamation Trust Landfill, located 1 mile southwest of Tyngsborough and 4 miles south of Nashua, N.H., 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, the EPA listed the site on the 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 live within a mile of the site in this residential/rural area; 2,100 people live within 3 miles of the site. The nearest residents are located 100 feet away. Benzene, tetrahydrofuran, arsenic, 1,4-dioxane, and 2-butanone, among others, had been detected in the groundwater. Sediments have been shown to contain low levels of benzo(a)pyrene. People face a potential health threat by ingesting contaminated groundwater. Flint Pond Marsh, Flint Pond, Dunstable Brook, and nearby wetlands are threatened by contamination migrating from the site.

EPA's involvement at the site began with groundwater testing conducted by EPA contractor Ecology and Environment, Inc. during 1981 and 1982. The site was proposed for the National Priorities List (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 Department of Environmental Quality Engineering's 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 (RI/FS) was also begun in September 1983. The basis for the removal action was documented in the first Record of Decision (ROD) issued on December 29, 1983.

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

Text Box 7: DuPage County Landfill

The 40-acre tract of land that is now 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. The landfill was designed to be 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 are 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; and metals such as lead, chromium, manganese, magnesium, and mercury. VOCs and agricultural pesticides have also been detected in private wells down gradient of the site but at low levels. Some metals (manganese and iron) have 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)

Text Box 8: Avtex Fibers Site

The Avtex Superfund site consists of 440 acres located on the bank of the Shenandoah River within the municipal boundaries of Front Royal, VA. The site is bordered on the east by a military prep school (grades 5 -12), on the south by a residential neighborhood, and on the west by the Shenandoah River. From 1940 to closure in 1989, industrial plants on the site manufactured rayon and other synthetics. Tons of manufacturing wastes and by-products 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 and secured with a protective material where needed, and a thick soil cover and vegetation known as a cap.

Front Royal is located in close proximity to the Appalachian Trail, the Shenandoah National Park and George Washington National Forest, 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 group process, divides the site into three areas: a) a 240-acre River Conservancy Park along the Shenandoah River combining ecological restoration and conservation of native habitats; b) a 25-acre Active Recreation Park with boat landings, picnic shelters, and a developed recreation area including a visitor center and soccer fields; and c) 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\

Text Box 9: Leviathan Mine Superfund Site

In May of 2000, the EPA added the Leviathan Mine site in California to the National Priority List (NPL) 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 located in a rural setting near the Nevada border, 24 miles southeast of Lake Tahoe. The physical disturbance from the mine itself is 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 impacted 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 for the site 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 Natural Resource Damage Assessment (NRDA) regulation under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), 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 requires a preliminary assessment of the likelihood of significant ecological or other impacts from the contamination (corresponding to Step 2 in Figure 2 of this report). The decision was made at that time (July 1998) to move forward with a Type B NRDA, which in principle is a decision to move forward with an assessment of the value of the ecosystem services that have been lost as a result of the site contamination. A Type B assessment involves three phases: a) injury determination to document whether ecological damages have occurred, b) quantification phase to quantify the injury and reduction in services (corresponding to Step 4 of Figure 2), and c) damage determination phase to calculate the monetary compensation that would be required (corresponding to Step 5 of Figure 2). In the Leviathan mine case, the Trustees proposed using resource equivalency analysis (REA) based on a replacement cost estimate of the lost years of natural resource services to determine damages for all impacted 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. Finally, in the decision by EPA about whether to list the site on the NPL and 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 are often 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, prepared by the Leviathan Mine National Resource Damage Trustees (Washoe Tribe of Nevada and California U.S. Bureau of Indian Affairs U.S. Fish and Wildlife Service U.S. Forest Service California Department of Fish and Game and Nevada Division of Environmental Protection with Stratus Consulting)
http://www.fws.gov/sacramento/ec/Leviathan%20NRDA%20Plan%20Final.pdf

6.2.3.1 At the beginning of the remediation and redevelopment process, define the ecosystem services and values important to the community and key stakeholders related to the site.

The urban examples of the Charles George landfill and DuPage County landfill show the difference in outcome that can be produced by engaging with the community at an early stage to focus on the ecosystem services of importance to them. Although there was no evidence of formal valuation methods at the onset in either example, the focus on ecosystem services and the inclusion of additional experts (i.e., forestry experts) led to a more positive outcome for the DuPage County community.

At the Charles George landfill, ecological values or future uses were not considered at the start. The human health risks at this site were so salient at the time they were discovered that they were the focus of the subsequent decisions. When the landfill site was capped and the water system from the city of Lowell, Massachusetts, was extended to the affected community, the health and safety concerns were addressed. Although the Record of Decision was published over 20 years ago, the potential for ecosystem services remains untapped.

By contrast, the remediation and redevelopment of the DuPage County landfill site, now known as the Blackwell Forest Preserve, 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, sledding). The remediation effort succeeded. 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."

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, even 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 had a sense of the ecological potential of the area, particularly for hawks, and a sense that, where hawks abound, birders will come to watch them. In this case, the difference was not one of methodology as much as conception. Once planners understand an area has ecological potential, it may be fairly easy to utilize qualitative differences to show likely quantifiable consequences

The Avtex Fibers case provides an example of the importance of engaging key stakeholders. At the Avtex Fibers site, the public complained about offensive sights and smells and contamination of drinking water wells. Over several decades, local government and environmental protection agencies conducted tests, filed thousands of complaints, and took various regulatory actions that ultimately resulted in the location's listing and designation as a Superfund site. Once the site was listed and a management process established, a clear effort was undertaken to engage stakeholders through a multi-stakeholder process in the development of the Master Plan. Although there was some consideration of ecosystem services, it is unclear whether there was any systematic means of assessing the ecological services that people cared the most about.

For situations like the Avtex Fibers site, deliberative group processes involving stakeholders and relevant experts (including historians) could provide an effective approach to identify and document ecosystem service values of most concern to stakeholders. In framing the dialogue with stakeholders, methods such as Ecosystem Benefits Indicators or the Conservation Value Method might have helped EPA's site managers understand the potential ecosystem service potential from 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 methods).

Defining the ecosystem services that matter to people requires a carefully constructed and systematically implemented program that integrates the use of multiple methods to fairly and faithfully reflect the perspectives of multiple stakeholders. There is no simple recipe for accomplishing this task and no simple algorithm for calculating values and summing them up to make a decision.

The Leviathan Mine is a good example of how EPA is often faced with the need to consider a complex array of competing interests. In this case the Agency is faced with a clear dichotomy between the ecosystem services valued by the full-time resident population of American Indians and the community of occasional recreational users. Recreational users would gain from the cultural services associated with hiking, fishing, and camping. However, the Washoe tribe that lives in the area year-round values the ecosystem as a provisioning service for food as well as for its spiritual and cultural services.

The Leviathan Mine case study additionally highlights the need to consider the existence or intrinsic values of the ecosystem. The ecosystem near the Leviathan mine site provides a habitat for threatened species such as the Lahontan cutthroat trout and bald eagle, which many individuals might value. In considering site restoration or remediation, or 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 these other sources of value as well.

For the Leviathan example, information about the impacts of greatest concern to affected individuals might be obtained in at least three ways. The first would be to gather information from them about the relative importance of the various services 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 some basic information that could be used to judge the importance of different services. This might be of the type used to construct Ecosystem Benefit Indicators, such as: water use data for the Washoe tribe and others in the vicinity of the site (e.g., sources, quantities, purposes); harvesting information for the Washoe (e.g., what percent of their harvesting of nuts, fish, etc., comes from the area impacted by the site); recreational use data (number of people visiting the area of the national forest impacted by the site for hiking, camping, fishing, wildlife viewing); data on flooding potential and what is at risk in the vicinity of the site; data on spiritual/cultural land-use practices by the Washoe. It is unclear whether some of the other data exist or would have to be collected. The third approach would be to review related literature and previous studies to learn about impacts of concern in other similar contexts. For example, previous social/psychological surveys (not specific to this site) or other expressions of environmental preferences (e.g., outcomes of referenda, civil court jury awards, etc.) 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 (e.g., if studies show that existence value is large). 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.

Analysis of the values of disparate users for a site is needed to identify the aspects of the site contamination of greatest concern to people and the related ecosystem services. It may be a significant challenge to identify and address the interests of different groups in restoration and redevelopment. In the Leviathan Mine case, it is likely that this would have to be considered both for tribal and non-tribal individuals, since the sources of value are likely to be different for these two groups. .

6.2.3.2 Involve the mix of interdisciplinary experts appropriate for valuation at different sites.

Interactions among experts and the affected publics form a key component of any hazardous site assessment, planning, and implementation program. Ideally, collaboration among all relevant experts [physical, chemical, biological scientists (ecology, toxicology etc.), and social scientists (economists, social psychologists, anthropologists, etc.)] and communication with affected stakeholders begin very early in the planning stages of remediation and redevelopment and remain throughout implementation and post-project monitoring and evaluation. A key point for collaboration among expert disciplines is the development of alternative management scenarios, particularly translating physical and biological conditions and changes at the site into value-relevant outcomes that can be communicated to stakeholders.

The Leviathan mine case provides examples of the need for collaboration among disciplines to understand how the human population's values are affected. Because of the unique cultural and spiritual values associated with ecosystem services, anthropologists could play an important role in characterizing the value of the ecosystem services to the Washoe

Tribe. Similarly, economists or others seeking to estimate existence value for an impacted species (e.g., fish) would need to work closely with ecologists to determine the likely impact of any change (or proposed project) on that species (e.g., effect on fish population) so that the change could be valued.

6.2.3.3 Construct conceptual models that include ecosystem services.

Ecological assessments associated with the remediation and redevelopment of contaminated property will be most meaningful for decision making if they incorporate ecological production functions that link remediation and redevelopment actions to ecosystem services. Historically, such assessments were not conducted at the four sites chosen by the committee. The examples did, however, provide illustrations of how assessments using ecological production functions could have influenced the site-specific results in a positive manner.

While it is now standard practice to develop a conceptual model in performing ecological risk assessments for contaminated site evaluations, EPA analyses of adverse impact have generally not been linked to ecosystem services in ways that enabled alignment of ecological risk assessments with economic benefits or other assessments of existing or foregone ecosystem services. The primary focus of the Agency's remediation efforts is to control anthropogenic sources of chemical, biological, and physical stress that could lead to adverse impacts to human health or the environment. Developing a conceptual model that incorporates the linkage between ecological endpoints and community-identified services can help guide valuation of ecological protection, leading to practical information for site remediation and redevelopment.

The Avtex Fiber case highlights what EPA could gain from developing the capacity to use conceptual models that integrate ecological and social value attributes of a site. A noteworthy feature of the Avtex Fiber process was the development of a Master Plan, which provided evidence that some ecosystem services were considered but no evidence that ecosystem services were broadly considered. For example, early concerns about contamination of groundwater and 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 providing 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 consideration of 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, but no evidence suggests that a comprehensive ecological model identifying ecosystem services guided redevelopment at the site. As a consequence, it is unclear whether the particular pattern of restored forests and wetlands, developed recreation areas, and industrial parks produced the best possible outcomes for protecting ecosystems and ecosystems services. Different siting and design of the soccer fields, for example, might have returned the same recreational value while achieving greater ecosystem services in the form of wildlife habitat, water quality, or aesthetic values for visitors, nearby residents or both. The declared ecological, "green" focus of the industrial park as a component of the Master Plan implies 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 left 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.

6.2.3.4 Adapt current ecological risk assessment practices to ecological production to predict relevant ecosystem services

As discussed in Chapter 3 of this report, development of a conceptual model should be followed with predictive analyses of effects of EPA's actions on ecological services. To some degree, EPA's Ecological Risk Assessment Guidelines and Framework (U.S. Environmental Protection Agency risk Assessment Forum 1992 and 1998) have endorsed the concept that ecological risk assessments need to be built on a conceptual model linked to one or more assessment end points. Expanding ecological risk assessments to include assessments of the services that matter to people may present technical challenges, given that current ecological risk assessments are often dominated by the available 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 for contaminated sites assessments. 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.

EPA's assessments are important not only for EPA decisions related to site remediation and development but also for decisions by other federal agencies. Although other trustee agencies, such as the National Oceanic and Atmospheric Administration (NOAA) and the U.S. Fish and Wildlife Service (USF&WS), are the regulatory leads for Natural Resource Damage Assessment (NRDA), the ecological risk assessments and conceptual models produced by EPA in the remediation process are often the basis for damage assessment. The extrapolation from risk to injury to damages is often a controversial aspect of the dialogue between the Agency, its trustee partners, and the parties responsible for the damages. The estimate of risk and the estimates of uncertainties associated with chemical exposure and toxic response introduces controversy because these data are often used as a surrogate for injury to the environment. The related damage claim, an expression of the restitution for lost or forgone use of ecosystem services, is likely to be challenged. Predictive ecological production functions play a critical role in such decisions.

The Leviathan mine case illustrates how the concept of ecosystem services has been used and can be used in damage assessment and restoration, as well as some of the issues associated with delineating ecological services using ecological production functions to predict impacts on them. If EPA could effectively conduct assessments that incorporate ecological production functions to predict impacts on ecological services identified in conceptual models, those assessments would enhance the ability of resource trustees to appropriately assess injury, define restoration goals, and calculate damages.

For Natural Resource Damage Assessments, impact or injury occurs when some standard (e.g., water quality or drinking water standards in the Leviathon Mine, for example) is exceeded. Impact or injury also could occur when toxic substances are present in a concentration or duration sufficient to cause a loss of services to the general public or a loss of services unique to the Washoe Tribe. Thus, the concept of ecosystem services plays a key role in defining or focusing categories of possible injuries to further evaluate.

Similarly, the concept of ecosystem services underlies the use of Habitat Equivalency Analysis (HEA) or the related Resource Equivalency Analysis (REA) to determine compensation for damages. In principle, application of HEA requires a determination of the flow of ecosystem services that would have been provided by a given site had it not been contaminated. This flow is then compared with the ecosystem services flow resulting from a restored site or a site providing equivalent services. Ideally, the value of the ecosystems services under the two would be equal. In order to apply this concept, it is necessary to delineate and value the service flows.

How can EPA estimate the impact of relevant ecosystem services? The Leviathan Mine Natural Resource Damage Assessment Plan (NRDAP) 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 (since 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 approaches can be a direct demonstration of injury, which can only be truly 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 translate these impacts into predicted changes in the flows of services. Estimations of the site's impact on the fish population in the nearby water body would need to be considered to determine if recreational fishing is likely to be significantly impacted,. Such an analysis requires estimation of the impacts of the 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 are known to exist in insects and fish tissue, EPA must 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 will not 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 described in section 3.31 of this report for in selecting from among existing models. In the absence of such a site-specific model, how should EPA proceed in looking at the impact on ecosystem resources or services? At this stage, EPA might look to the scientific literature for guidance on how sensitive the insects and fish species of concern are to these types of stressors. It could then ask expert ecologists to judge the likely magnitude of the impacts in this specific case. This would be akin to an ecological impact transfer, which is similar to the notion of economic benefits transfer. The Leviathan Mine NRDAP suggests this approach. As for any issue involving transfer of information related to valuation, 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 information.

In addition, the Leviathan Mine NRDAP suggests studying the fish population downstream of the mine and comparing it to the population in a reference location, assuming a realistic reference site can identified. More generally, it also 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 aid in framing the types of damages resulting from the mining activity (which is most useful in an NRDA policy frame). Since reference sites and exposed sites may differ for a number of reasons not related to the contamination, such a comparison may not directly predict the injury and will not take into consideration the impact of proposed remedial actions on ecosystem services. Decisions about remediation and restoration require analysis of proposed actions and it may not be reasonable to assume that remedial actions will be 100% effective in restoring the ecosystem services to their original level (presumed to be the level at the reference site). Comparative analyses using ecological production functions are needed and can be facilitated through ongoing use of comparative tools such as Net Environmental Benefit Analysis (Efroymson et. al., 2004).

6.2.3.5 Define ecosystem services carefully and develop a standard approach for cataloging and accounting for ecosystem services for site remediation and redevelopment.

There is a need for accounting rules to recognize and avoid double-counting or under-counting the contributions to human well-being from ecological service flows. Ecosystems and their numerous components are linked in an intricate and complex network of biological, chemical, and energy flows. By looking at isolated impacts to individual organisms or components and their associated services, the potential arises for double counting or undercounting contributions to human well-being generated by Agency actions addressing contaminated sites.

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 very useful for sorting out the different things to be valued. It fails to identify mutually exclusive services and seems to present a high likelihood of double counting. It also does not seem to adequately distinguish between inputs and outputs. The significance of protecting habitat or riparian vegetation, for example, is not clearly addressed. Is it because society cares about the populations it supports? Or is it because these populations are an input into something else of value, such as recreation? Consider the example of insect populations. If society cares about the insects for their own sake, then this should be included as an existence or intrinsic value. If they are valued because they are a food source for fish, and society cares about fish, then there is value in the change in fish brought about by the change in insects. But in the latter case, they should not be valued separately. EPA should view both clean water and insects as inputs into the production of more fish and value either the inputs or the outputs. In order to determine how to measure value, it first must be known why society values insects or fish.

Similarly, the listing of services by Wilson (2004) shown in Table 6, based on the U.N. Ecosystem Millennium Assessment (2005) definitions of ecosystem services, is not very useful for valuation purposes and could create confusion in valuation. For example, it is unclear how or where the use of surface water or groundwater for drinking would fit in Wilson's list. Is the service "Freshwater Regulation" intended to include drinking water or is it intended as an input into aquatic and other habitat-related services?

Table 6: Ecosystem Service Matrix for Leviathon Mine (from Wilson, 2004)

Ecosystem Function	**Ecosystem Service**
Regulating	Disturbance Moderation • Flood prevention from on-site evaporation ponds • Regulation of surface water runoff and river discharge during snowmelt and heavy rain events
	Freshwater Regulation • Restoration of groundwater discharge beneath the pit and waste-ore piles • Non-hazardous surface water drainage into Leviathan Creek, Bryant Creek and East Fork River
	Wildlife Habitat

	• Nursery, feeding, and breeding ground for indigenous fish species including the threatened Lahontan cutthroat trout • Restoration of habitat and feeding habitat for the threatened Bald Eagle • Maintenance of riparian vegetation habitat for mammals, birds, amphibians, and insects
Supporting	Soil Formation • Restoration of productive floodplain soils in the Leviathan-Bryant Creek watershed and the East Fork of the Carson River
Provisioning	Food and Raw Materials • Edible freshwater fish • Pine nut harvesting by Washoe tribe
	Ornamental Resources • Raw material for traditional Washoe Tribal crafts
Cultural	Recreation and Amenity • Improved hiking and camping opportunities • Recreational fishing
	Inspirational and historic • Washoe Tribal heritage site • Spiritual and ritual uses such as spiritual bathing, and cleaning religious implements

Perhaps a better delineation of ecosystem services for use in ecological production functions would involve, as discussed in section 3.3.2. of this report, the identification of directly experienced, measurable, and spatially and temporally explicit measures of services. Such a list of ecosystem services might consist of the following elements:

a) Water used by Washoe Tribe members and others for washing and drinking
b) Non-consumptive use values of wildlife (e.g., people like to view bald eagles and other species)
c) Harvesting (hunting, nuts, fish) by Washoe tribal members
d) Cultural, spiritual and ceremonial value of land used by Washoe tribal members
e) Flood control (e.g., reduction in flooding from snowmelt or runoff)
f) Recreational services (e.g., fishing, hiking, camping)

Values that are broader and do not meet the principles described in section 3.3.2. (e.g., "existence" or intrinsic values (broadly defined, based on moral or other principles) from threatened and other species (e.g., cutthroat trout, bald eagles, and other impacted species of concern); the value of the natural process leading to ecosystem outputs, beyond the value of the outputs themselves (e.g., preference for natural processes over man-made ones, or native species over introduced species) could be discussed qualitatively as considerations to supplement the quantitative ecological production function analyses.

6.2.3.6 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.

Chapter 4 of this report provides an overview of a broad range of methods that could be explored for assessing ecosystem services lost or gained from current conditions or proposed Agency action in monetary or non-monetary terms. Currently, without such valuation of options, the typical comparison of remedial alternative strategies includes two tests: a) whether a remediation action controls risk to an acceptable level; and if so, then b) whether it is cost effective. Under this scheme, if a proposed remediation action is adequate with regard to risk reduction, the least costly is the obvious choice. Such an approach decouples remediation and development, leading to a delayed development process, possibly off-mark from what matters to key stakeholders.

If remediation and redevelopment alternatives are to be compared based on an analysis of their contributions to human well-being, a number of methods can be used. As mentioned previously, NEBA (Text Box 5) offers a conceptual framework for comparing remedial and redevelopment alternatives on a basis of contributions to human well-being, whether monetized or non-monetized. For example, the contributions to human well being associated with different remedial and redevelopment alternatives could be derived through methodologies such as Habitat Equivalency Analysis (HEA) or Resource Equivalency Analysis (REA) that report results in ecological units over time (e.g., discounted service acres years). The cost of creation or replacement of those ecological units can be estimated in monetary terms (i.e. replacement cost). This approach does not provide a direct measure of the value of ecosystem services, but it does support a comparison of the services provided under different options. Alternatively, impacts of alternatives could be compared purely in ecological terms, such as through use of Biodiversity and Conservation Values approach or energy-based approaches.

Comparison of remediation and redevelopment alternatives using economic valuation methods might include hedonic pricing studies to determine the economic impacts of the identified cleanup and redevelopment options on adjacent residential property values. New contingent valuation studies or studies of the value of recreational fishing (e.g., from travel cost models) could be useful in capturing in monetary terms some of the values lost or gained related to options being considered. Models might be used to compare expected gains to the local economy across the feasible set of redevelopment scenarios. Monetary/economic assessments and models might also be used to estimate the expected long-term contributions to the local economy from industrial development versus recreation or tourism-focused use options. For the Leviathon Mine, Ecosystem Benefit Indicators, as discussed above, might also be used to evaluate the impacts of different mediation or redevelopment options.

If stakeholders are involved in testing remediation and redevelopment alternatives, their preferences for or weighting of alternatives could be assessed directly through decision-aiding processes and information about ecosystem services derived. This would allow non-monetary methods such as biophysical ranking methods to be used as to compare changes in biodiversity, habitat quality, energy flow, and other indicators of identified and accepted bio-ecological goals, expressed in their own biophysical terms, across the cleanup and restoration and redevelopment alternatives. Formal social-psychological surveys of potential recreational users, visitors, and tourists could measure the relative preferences (importance, acceptance) across the restoration/redevelopment plans under consideration from the perspectives of these important groups. Parallel economic or monetary assessments, perhaps using contingent valuation or travel cost methods or both, could extend and cross-validate survey results. Decision-aiding methods could provide dollar-denominated value indices to facilitate analyses of trade-offs with development costs and between recreation, tourism, and industrial development emphases at a site.

6.2.3.7 Communicate information about ecosystem services in discussing options for remediation and redevelopment of sites

The committee advises EPA to explicitly address issues regarding ecosystem services in communications about site remediation and redevelopment. Because non-technical audiences often find scientific information obscure, information about ecosystem services might be communicated effectively through the use of visual communication techniques. EPA might make effective use of 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. For example, the restoration plan for the Avtex site 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 be very useful for communicating the implications of alternative plans to stakeholders. Achieving and effectively using such visualizations would first require interactions between forest ecologists and visualization experts (such as some landscape architects). These interactions 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, or other relevant social or decision scientists might create appropriate vehicles and contexts for presenting the visualizations to relevant audiences. Technical computer graphics expertise might also be useful in this context. 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 (decades in the case of Superfund projects). With proportional resource allocations, this level of effort is likely appropriate.

If valuation concepts and techniques are incorporated early and often throughout the contaminated property redevelopment process, the Agency should be prepared to communicate with interested publics more effectively. Managers will be able to communicate the reasoning behind their selection of preferred options if analyses effectively integrate consideration of ecosystem services and their derived contributions to human well-being into the selection of the remedial and redevelopment actions,. Demonstrating to the public that there has been a focus on ecosystem services that matter to them, and the ability to communicate in terms of those contributions as they relate to proposed actions, should lead to greater public understanding of options and acceptance of the proposed plan. Projected contributions to human well-being should make the selection of performance measures relatively straightforward. Communicating the progress and challenges of the redevelopment process should be facilitated by using performance measures defined in terms of contributions to well-being that the interested public understands and accepts as important.

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

The committee recommends that EPA should actively pursue the broad and rapid transfer of experience with integrating valuation concepts and techniques into the redevelopment of contaminated sites. The Agency will build its capacity to utilize valuation to inform its local decisions through a systematic exchange of information about site-specific valuations. The lessons learned from these trial efforts, whether successes or failures, need to be shared widely across the Agency with the regions, program offices, and the tool builders in the research organizations. The Agency can catalogue and share such experiences in a number of ways, such as reports, databases or BestNets (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 how it is done, the information should be shared broadly.

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

The committee advises EPA to pursue opportunities for ecological valuation to support decisions about site remediation and redevelopment. To effectively value the protection of ecological systems and services in this context, the committee recommends that EPA:

- Define the ecosystem services and values important to the community and key stakeholders related to the site 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 ecological production to predict 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 options for remediation and redevelopment of sites
- Create formal systems and processes to foster a information-sharing about ecological valuations at different sites.

6.3. VALUATION IN REGIONAL PARTNERSHIPS

6.3.1. EPA Role in Regional-scale Value Assessment

Many important ecological processes take place at a landscape scale, making regional analysis an appropriate scale at which to analyze the value of ecosystems and services. For example, understanding habitat connectivity on landscapes, water and nutrient flows through watersheds, or patterns of exposure and deposition from air pollution in an airshed pose issues larger than a particular site. Rather, they are specific to a regional area and thus require regional-scale analysis. 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 similarly expanded. The increase in data and methods has opened up new frontiers for regional-scale analysis of ecosystems and their services. An active EPA extramural program in ecological research is under way 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, Oregon; decision support tools to meet human and ecological needs in rivers in New England; and the provision of multiple services from agricultural landscapes in the upper Midwest. Region 4 has developed a tool for regional ecological assessment (discussed in Section 3.3.2). Other regions have undertaken assessments of ecosystem services as well. Great potential exists, 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. 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 offer the potential for expanding local, state, and EPA capacity to value 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 local governments, other federal agencies, and the private sector, EPA can enhance its environmental protection activities by engaging important local stakeholders, gaining access to regional expertise, and gaining access to decision-making on important regional-scale environmental decisions. Local public and private 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 analysis is often constrained by specific mandates, the regional level enjoys great latitude 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. EPA, for example, can use regional-level partnerships as a mechanism for testing and improving various valuation methods that might ultimately be used at the national level. There is also a downside of not having legal or statutory requirements for EPA to undertake valuation of ecosystems or services at the regional scale. EPA regional offices with limited resources and a long list of mandated activities may have little time or ability to undertake such activities with local partners. In addition, there may be limited expertise in regional offices for undertaking at least some of the crucial steps that the committee recommends in carrying out valuation of ecosystems or services. For example, few regional offices have economists on staff who can work on valuation exercises. Partly for these reasons, many of the potential advantages of regional partnerships for valuing ecosystems or their services at a regional level have not been realized to date.

In analyzing regional opportunities for partnerships, this section explores several case studies that illustrate some potential approaches to regional partnerships and regional-scale analysis of ecosystems and services, including cases from Chicago; Portland, Oregon; and the Southeast Region. Case studies illustrate several general lessons about regional-scale analysis of the value of ecosystems and services and the potential usefulness of regional partnerships.

6.3.2. Case Study: Chicago Wilderness

Chicago Wilderness is an alliance of more than 180 public and private organizations. It represents a bottom-up organization that reflects the views of its member organizations to protect the environment in and around Chicago. No single decision maker or agency controls or guides Chicago Wilderness. It pursues objectives, as defined by its members, through consensus. The overall goal within Chicago Wilderness, as stated in Page 7 of the Executive Summary of 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 pursues its goals by attempting to create 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 the Chicago Wilderness, EPA Region 5 provides 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 stresses affect ecosystems and the provision of services. Chicago Wilderness has produced several reports, including a *Biodiversity Recovery Plan* and a green infrastructure map for the region.[43] The Chicago Wilderness Web site (http://www.chicagowilderness.org/) contains a complete chronology and links to many relevant documents, including the *Biodiversity Recovery Plan.*

Chicago Wilderness is interested in the valuation of ecosystems and services, but is only beginning to explore the opportunities for valuation in its activities. Members of Chicago Wilderness enjoy only limited technical expertise and practical experience in valuing the protection of ecological systems and services. EPA Region 5 also has limited capacity to undertake economic analysis of the value of ecosystem services. No specific legal authority mandates valuation of ecosystems or services as part of the work of Chicago Wilderness. Though not required, quantifying values associated with the conservation of green space and biodiversity could be helpful for Chicago Wilderness in meeting its own stated objectives and in communicating its analysis to other groups and the general public. The possible uses of additional valuation tools identified by Chicago Wilderness members include the following options:

- To inform decisions on the establishment of green infrastructure, including priorities for acquisition of land by, for example, forest preserve districts or soil conservation districts;
- To assess the value of preserving ground water and other ecosystem services related to clean water;

- To assess the relative value of investing in different research projects to establish priorities for funding decisions;
- To assess the relative value of conventional versus alternative development efforts and to demonstrate conditions in which development decisions that have positive impacts on the environment might be in the financial interest of the developer;
- To communicate effectively with residents of the Chicago region regarding the value of green infrastructure and biodiversity and how these are related to quality of life for area residents.

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

6.3.2.1 An Example of How Valuation Could Support Regional Decision-Making: Open-Space Preservation in the Chicago Metropolitan Area

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

Voters in four counties in northeastern Illinois have passed referenda authorizing bonds for land purchase for open space preservation or watershed protection. In November 1997, voters in DuPage County passed an open space bond for $70 million. In November 1999, voters in Kane County and Will Counties passed bond issues of $70 million for open space acquisition or improvement. In 2001, the voters in McHenry County passed a $68.5 million bond for watershed protection. While these multi-million dollar bond proposals have put a substantial amount of money into efforts to preserve open space and ecological processes in the region, they are insufficient to provide adequate protection for all worthwhile open space or watershed protection projects. Given this shortfall, input about what lands should be purchased, or what management actions should be undertaken to maintain or restore natural communities would help to ensure that counties invest these funds wisely.

This section of the report therefore looks at how valuation could help inform conservation investments under the local county bonds. For this example, three types of values derived from protecting natural systems will be examined: a) species and ecological systems conservation; b) water quality and quantity; and c) recreation and amenities. The water quality and quantity discussion will focus on McHenry County because the bond issue there related directly to watershed protection. The example follows the process outlined in Chapter 2 of this report. The following sections describe: a) the process of stakeholder involvement and input into defining values of ecosystems and services of interest; b) predicting ecological impacts in terms of changes in ecosystem services; and c) using methods to assess and characterize the values of ecosystems and services.

6.3.2.2 Process of Stakeholder Involvement, Scientific and Technical Input, and Public Participation

The planning documents and activities of Chicago Wilderness reflect several of the themes from Chapter 2 of this report, including interdisciplinary collaboration and broad involvement. The Chicago Wilderness Biodiversity Recovery Plan (1999) discusses specific roles for private property owners; local, state, and regional governments; intergovernmental agencies; and federal agencies. The document also highlights the actions of EPA that affect biodiversity and EPA's role in Chicago Wilderness.

Chicago Wilderness provides an excellent example of an organization that has made extensive efforts to engage the local community in determining the most important features of ecosystems and services in the region. Two of the great strengths of Chicago Wilderness are the broad range of groups included and the commitment to open processes. This inclusion allows the participants themselves to define the objectives, goals, and priorities of the organization. The open and democratic process and the extensive efforts to include multiple views and voices results in the group's goals and objectives being largely reflective of what people in the region view as important to conserve in their region. 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 endorsed by Chicago Wilderness includes 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, and assessment of alternative actions, to adopting a plan).

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 included non-monetary valuation exercises with qualitative rankings of importance. The *Biodiversity Recovery Plan* also references other measures, such as polls and The Nature Conservancy's global rarity index. 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).

Chicago Wilderness also conducted eight workshops to assess the status and conservation needs with regard to natural communities in the area: four species workshops addressing birds, mammals, reptiles and amphibians, and invertebrates; and four consensus-building workshops on natural communities addressing forest, savanna, prairie, and wetland. The natural communities workshops developed overall relative rankings based on the amount of area remaining, the amount protected, and the quality of remaining areas that incorporate fragmentation and current management. The workshops also 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), and identified visions of what the areas should look like in 50 years. The workshop participants judged the data as insufficient to allow quantitative assessment of natural communities.

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

Fostering public support through education and outreach is also an explicit goal of Chicago Wilderness. The group emphasizes working with schools (including universities); it also identifies individuals, agencies and organizations as targets for outreach and involvement.

Chicago Wilderness' strengths in engaging local communities, however, also highlight some of the difficulties involved in doing so. Different individuals and different member groups define value differently. Some groups care more about restoring pre-settlement ecosystem conditions. Issues of open space and recreation are the primary motivation for others. 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 trade-offs 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 goal. The inability to make trade-offs among objectives 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. Chicago Wilderness is not well placed to make rapid analyses or provide feedback on decisions that occur over a short time period.

6.3.2.3 Predicting ecological impacts in terms of changes in ecosystem services:

Since Chicago Wilderness is committed to the value of protecting biodiversity, it is interested in predicting impacts related to species conservation and conservation of ecological systems at the landscape scale. Chicago Wilderness 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 principles of conservation science can be used for planning and how a transparent approach to mapping conservation goals can be useful in a regional partnership. Chicago Wilderness' *Biodiversity Recovery Plan* describes in detail the organization's conservation goals.

Water quality and quantity figure prominently in many ecological processes and in the provision of many ecosystem services. Text Box 10 describes some effects on the provision of ecosystem services that may result from the protection or restoration of watersheds. In some instances, Chicago Wilderness and its member organization have conducted prior studies making it possible to identify site-specific ecological characteristics important to considerations of ecosystems and services.

Text Box 10: Possible Ecological Impacts and Provision of Services from the Protection or Restoration of Watersheds Based on the Work of Chicago Wilderness

<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—will be increased because percolation and subsurface recharge will be enhance 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

<u>Biological systems that depend upon water quantity</u>

- Special status species—increased persistence of those habitats that depend on increased quantities of water in the watershed and containing protected species
- 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

<u>Effect on water quality</u>

- 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

To illustrate how Chicago Wilderness might characterize impacts on water quality and quantity in McHenry County this report supposes that stakeholders and experts together decided that the most important ecological services for comparing watersheds within the county are: a) minimizing flooding; b) maintaining or increasing groundwater recharge; and c) maintaining or increasing wetland communities. In reality, the most important ecological services related to water would be determined by the stakeholder involvement and input process discussed in Section 6.3.2.2.

To predict impacts related to flooding, Chicago Wilderness could make use of the GIS database it collected, which includes layers depicting rivers, streams, wetlands, forest lands, and floodplains. As a first approximation, historical records of flooding in McHenry

County watersheds could be examined and watersheds with the greatest flooding could be identified. The analysis could then evaluate the potential for restoring floodplain forests and wetlands for mitigating flooding. To address whether groundwater resources would be adequately maintained or increased by a development option, Chicago Wilderness 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. The analysis 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, Chicago Wilderness could use topographic maps and GIS data on rivers, streams, floodplains, forests, wetlands, and land cover to rank watersheds within McHenry County in terms of potential wetlands minus current wetlands. The areas within watersheds with the potential for expanding existing wetlands or restoring wetlands could be measured.

A number of GIS data files are available from McHenry County that 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 can be used to predict how provision of ecosystem services will change with alterations in management or policy. It may 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, then 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).

The third set of values included in this example are recreational and amenity values. Recreation covers a broad set of potential activities, from walking in the park to large game hunting. Community input is required to establish what are important recreational activities in the area. Access to parks and open space is a primary concern in many urban and suburban communities. A study conducted in the Chicago Metropolitan Area found a tradeoff between trying to locate open space close to people to provide access and locating open space to conserve species (Ruliffson et al. 2003). Some recreational activities (e.g., fishing, hunting, bird watching) require input from ecological models, while others (e.g., walking in the park) may be more a function of location. Similar comments apply to amenity values where community input is important in determining what factors most contribute to amenities in the eyes of the community.

Chicago Wilderness has done an admirable job of collecting spatially explicit information relevant to land use, open space, recreation, biodiversity conservation, and water quality and quantity issues. For this information to be relevant to decisions that affect ecosystems, however, Chicago Wilderness needs cause-and-effect relationships that can predict how policy choices would affect ecosystems and the provision of services. Chicago Wilderness does not have the kind of information at its disposal that would allow it to estimate ecological production functions. Chicago Wilderness can be quite effective in providing descriptive information, particularly in the form of maps, but will be limited in its ability to analyze alternative policies and make recommendations about which alternatives are preferable. For example, it will be limited in providing analysis to a decision maker in McHenry County concerning how to invest the $50 million approved by voters for watershed protection in a way that will maximize the value of ecosystems and services, because it will not be able to martial information about how particular actions affect systems and services identified as important.

Gathering the necessary technical and scientific expertise to predict how policy choices will affect ecosystems and the provision of services is a difficult task that introduces another potential problem. The experts best placed to provide evidence may be tempted to substitute their values on what is important for those of the stakeholders and community that ideally set the objectives for the organization. For example, defining the levels at which biodiversity targets can be considered as being met involves judgment. Different judgments used in models may give rise to different sets of 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 faithfully.

6.3.2.4 Valuation of Changes in Ecosystems and Services in Monetary and Non-Monetary Terms

When there are trade-offs among different services, (habitat protection versus improvements in water quality, for example), information about the value of various aspects of ecosystems and services is necessary to inform decision makers about what alternatives are more beneficial for the community. This requires information about relative values that goes beyond understanding the ecological impacts of management and policy alternatives.

As noted in other parts of this report, the valuation of ecosystems and their services can be conducted in numerous ways. This section begins with a discussion of the potential contributions that valuation could make for Chicago Wilderness and briefly describes 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. Chicago Wilderness has conducted very few valuation studies to date and largely lacks the resources and the expertise to conduct valuations.

The overall goal of Chicago Wilderness 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." This goal was derived with active input from member organizations and represents a consensus view of their values. In some sense, the important valuation exercise for Chicago Wilderness was carried out at the first stage in which Chicago Wilderness engaged the community 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.

Given this understanding and the clear statement of the overall goal of the organization, 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. The Conservation Value Methods could be used for identification and prioritization of conservation actions that would contribute to this goal, through spatial representation and analysis of biodiversity and conservation values. Chicago Wilderness has devoted most of its attention to stakeholder involvement and 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, such as "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. Information about how various potential strategies contribute to the protection and restoration of natural communities along with information about the cost of these strategies is the main information necessary for cost-effectiveness analysis. Cost-effectiveness analysis addresses the issue of how best to pursue an objective given a budget constraint. There is no need to estimate the value of protecting natural communities or of ecosystem services.

Of course, things are rarely so clear. Even with a single overall goal, there are often multiple dimensions and trade-offs among those dimensions that require the analyst to go beyond cost-effectiveness analysis. For example, in protecting natural communities, there may be trade-offs between protecting more of one type of natural community versus another. When there are multiple natural communities of interest, or multiple ecosystem services of interest, it becomes important to address questions of value, a practical matter when investment of bond monies are at stake. Is it more valuable to allocate more resources to restoring upland forest or wetlands? Is it more valuable to mitigate flood risk or improve water quality? Such questions can only be addressed 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 trade-offs among different facets of biodiversity conservation or among protection of different natural community types, is ultimately the same question as how one makes trade-offs among multiple objectives. Again, establishing prices on different components of biodiversity or on different natural communities allows for analysis of trade-offs 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 more readily understood by the public.

Non-monetary valuation can also be used. If trade-offs among different natural communities or among different services are needed, surveys containing attitude questions may be helpful. It may be easier for people to answer attitude survey questions about whether they think it more important to provide additional protection of forests versus wetlands, as compared to responding to questions about monetary valuation of forest protection versus wetland protection.

Protecting natural communities may be done for reasons related to the provision of ecosystem services or it may be done because people value intact natural communities (e.g., because they hold existence values or intrinsic values). 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: contingent valuation and conjoint analysis. In trying to estimate of the value of protecting species and ecological systems, Chicago Wilderness could survey respondents in the Chicago area using contingent valuation or conjoint analysis. Alternatively, Chicago Wilderness could attempt to use an economic benefits transfer approach 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 using contingent valuation or conjoint analysis, Chicago Wilderness could not include non-use values and would be able to estimate a partial economic value for each strategy.

Any effort to place a monetary value on non-use values through stated preference methods 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 public attitudes regarding biodiversity involving 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.

Deliberative valuation exercises using citizen juries or other small focal groups might be a particularly useful means of evaluating trade-offs among potential strategies to protect natural communities in the Chicago Wilderness context. Under deliberative valuation, experts would 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. Deliberative valuation might enable participants to develop more thoughtful and informed valuations, to better trade off among multiple factors, and to engage in a more public-based consideration of values. Experts could use deliberative valuation either to try to come up with monetary comparisons of the values of the alternative properties or with weights that could be used to aggregate multiple layers of data.

Monetary values derived through deliberative valuations may differ considerably from traditional private values, both because of the consent-based choice rules that deliberative valuation employs 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 cost-benefit analysis (Howarth & Wilson, 2006). Monetary values from deliberate processes do not necessarily yield economic benefit measures.

As mentioned above, protecting natural communities may be done because people value provision of ecosystem services (e.g., water quantity and water quality, recreation and amenity values), as well as because they hold existence values or intrinsic values. Changes in water quantity can be valued either because there is too much (flood control) or too little water (water scarcity). One approach to measuring the value of flood control is to measure avoided damages with reduction in probabilities of flooding. 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 later study found estimated flood control benefits of $52,340 per acre from forest preserves. For water quality, an important ecosystem service in many metropolitan areas is the provision of clean drinking water. Protection of ecosystems may help reduce the fluctuation of water availability by storing water during wet periods and gradually releasing it during dry periods. Ecosystems protection may also be beneficial in providing relatively clean water for municipal supply. There is also value of surface recharge of aquifers (NRC 1997). The value of providing clean drinking water to the public is extremely high, far exceeding the costs of supplying it whether by natural or human-engineered means. Because it is a question of how – not whether - to supply clean drinking water, replacement cost (for example, the cost of building a filtration system to replace lost water purification services provided by wetlands) can be used as a method 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 forms of recreational opportunities and amenities created by the natural environment. Typical methods used by economists to estimate the monetary value of recreation and amenities include hedonic property price analysis, travel cost, and stated preference. In addition, a smaller literature uses evidence from referenda voting to infer values for open space and other environmental amenities.

Applications of the hedonic property price model are 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. The hedonic property price model has been applied 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). Given the large number of residential property sales in the Chicago area and existing spatially explicit databases on many environmental attributes, there is great potential for Chicago Wilderness to utilize such studies to estimate values of various environmental amenities. This method has not been used by Chicago Wilderness to date.

A large literature also exists on the value of recreation sites using the travel cost method. With the large number of visitors to Lake Michigan beaches, forest preserves, and parks in the Chicago metropolitan area, great potential exists for Chicago Wilderness to apply travel cost to estimate the value of recreational activities. There have been several applications of travel cost studies in urban areas (e.g., Binkley and Hannemann 1978, Lockwood and Tracy 1995, Fleischer and Tsur 2003). To date, these methods have not been applied by Chicago Wilderness.

Stated preference methods can also be used to estimate the value of recreational opportunities and environmental amenities. One such study has been done for Chicago Wilderness. Kosobud (1998) used a contingent valuation survey to estimate the 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. Extrapolated over the number of households in the region, this would generate about $50 million in benefits from expansion of natural areas in the region per year.

Finally, there is a small but growing literature that analyzes the results of voting behavior in referenda involving environmental issues to estimate values. 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, several counties in the Chicago metropolitan area have passed referenda authorizing bonds to purchase open space or for watershed protection. Though the number of referenda is relatively small, making it difficult to generalize or make comprehensive statements about values, analysis of the results of these referenda could provide insights into the values of different segments of the public for various environmental amenities.

Application of valuation methods would generate quantitative estimates of the value of the protection of ecosystems and the provision of various ecosystem services. This information could be of great use to decision makers in evaluating alternative strategies to protect natural communities. Valuation studies could also be quite useful in communicating consequences of various alternatives to the public. Chicago Wilderness could usefully apply a number of valuation methods for these purposes.

To date, however, Chicago Wilderness has initiated very little valuation research. Despite some attempts to collect information about the value of protecting natural communities and ecosystem services (e.g., Kosobud 1998), this effort has not been comprehensive or systematic. This contrasts with the major efforts undertaken to garner stakeholder involvement and input into setting the goals for the organization, and the large-scale effort collecting technical and scientific knowledge to characterize the status of ecosystems and species. In part, the lack of valuation activity is 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 choice made by the organization about the set of activities most important to it (which is a different sort of revealed preference). Interest exists within Chicago Wilderness to include economic and other social science approaches to study the value of protecting natural communities, but the right mix of available expertise and circumstances has been unavailable to make this a reality.

6.3.3. Other Case Studies: Portland, Ore.; and the Southeast Region

6.3.3.1 Portland, Ore., Assessment of the Value of Improved Watershed Management

The city of Portland, Oregon, facing potentially major expenses from meeting its obligations under the Clean Water Act, Superfund, and the Endangered Species Act, decided to invest in an analysis of ecosystem impacts and the value from ecosystem services that would result from improved watershed management. By taking a systems approach and considering the multiple economic benefits of actions, Portland officials hoped to find more effective watershed management that would both save the city money and improve the welfare of its citizens. Of primary interest were impacts on flood abatement, water quality, aquatic species (salmon in particular), human health, air quality, and recreation. The City of Portland's Watershed Management Program requested David Evans & Associates and ECONorthwest to undertake the study, which they completed in June 2004 (David Evans & Associates and ECONorthwest, 2004). Though not an example of a regional partnership with EPA, the project provides one of the best current examples of the kind of landscape-scale analysis of the value of ecosystems and services and exemplifies many of the recommendations this report.

Portland city officials realized that they only understood a portion of the contributions to well-being from improved watershed management. To be able to make intelligent decisions about watershed management, these officials wished to have a more complete accounting, which required applying methods that could quantify a range of ecosystem values that are normally not quantified. The project aimed to expand the range of ecological changes that are valued, focusing on those changes in ecosystems and their services that are likely to be of greatest concern to people. From the beginning, the effort attempted to solicit input from the public and important stakeholder groups about significant ecological impacts. In addition to the value of direct flood-abatement impacts, the study monetized the economic benefits of biodiversity maintenance, as represented by improvement of avian and salmon habitat, air quality improvement, water quality improvement, by reduction of water temperature, and “cultural services”, which the study defined as including the creation of recreational opportunities and the increase of property values.

In order to carry out the project, both biophysical and economics analyses were commissioned. The biophysical analyses included studies of hydrology and flooding potential, water quality, water temperature, habitat analysis for salmon and other aquatic species, habitat analysis for birds and other terrestrial species along riparian buffers, and air quality impacts (ozone, sulfur dioxide, carbon monoxide, carbon, particulates). The economic analyses included studies of the impact of ecosystem changes on property values (including public infrastructure, residential and commercial property), the value of flood risk reduction, the value of recreation, and the value of impacts on 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 analyzed for their effect on various ecosystem services. Finally, the economic analysis attempted to value the changes in various ecosystem services. While the ecological and economic analyses were largely conducted by separate teams, the project was designed to provide a close linkage between ecological results and economic valuation.

Of particular note in this study was the emphasis on focusing the analysis to estimate 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, the approach taken in Portland tried to estimate cause-and-effect relationships that would allow the systematic appraisal of the set of consequences 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 case provides a good example of the potential advantages of integrated regional level analysis. The project undertook an integrated approach capable of analyzing the impact of alternative management actions on ecological systems and the consequent changes in the value of ecosystem services. Attempts were made to solicit input from the public in the design of the project so that it captured the impacts of greatest interest to the public. Results of the project were presented 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, but 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 in this type of analysis there are gaps in data and understanding. Gaps in understanding include how ecological systems will be affected by changes in management actions, and how this will affect the provision of ecosystem services and the consequent value of those services. For example, how will songbird populations change in response to changes in the amount and degree of fragmentation of habitat? What is the value to residents of Portland of changes in songbird populations? The study often had to use economic benefits transfer methods from cases quite different from the Portland context to generate estimates of values.

Though the project was commissioned by the City of Portland and 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 advocated by this report. In particular, the project aptly illustrates the sequence of steps, from input from stakeholders, through characterizing change in ecosystem functions under various alternative policy and management options, to valuation of services under alternatives. The project shows great potential for this type of analysis to provide important and useful information to decision makers.

6.3.3.2 Southeast Ecological Framework Project (EPA Region 4)

The Southeast Ecological Framework (SEF) project represents a regional GIS approach for the identification of important ecological resources to conserve across the southeastern United States. This region is one of the fastest growing regions in the country. Even so, it still harbors a significant amount of globally important biodiversity and other natural resources. The SEF is designed to meet EPA's goals of gathering and disseminating information pertinent to the ecological condition of a region. The SEF project's goal is to enhance regional planning across political jurisdictions and to 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 of 2001.

This framework has been developed for the eight southeastern states in EPA Region 4 (Alabama, Florida, Georgia, Kentucky, Mississippi, North Carolina, South Carolina, and Tennessee). This project has created a new regional map of priority natural areas and connecting corridors, along with geographic information system (GIS) tools and spatial datasets. The framework identified 43% of the land that should be protected and managed for specific contributions to human well-being. Two additional applications of the SEF were developed to demonstrate its utility for conservation planning at the sub-regional and local scales. This approach is now being evaluated for utility in other regions and nationally.

The SEF differs from the prior two case studies (Chicago Wilderness and City of Portland) because it focuses on a broad regional analysis of eight states, rather than a single metropolitan area or watersheds within a metropolitan area. The SEF also differs in that it focuses almost exclusively on habitat conservation rather than a broad suite of ecosystem services. The SEF did not undertake an extensive stakeholder involvement process to set its objective; it started with the focus on habitat conservation. It also does not attempt to combine economic analysis with ecological analysis to value the protection of ecosystems or services in monetary terms. Discussion of values focuses on conservation value, which is the ability to sustain species and ecological processes. In this regard, the SEF is a good tool to carry out regional analysis of ecological components, particularly habitat conservation. Because of its focus, the level of scientific knowledge underpinning the SEF is in general far higher than in the other case studies. An important challenge facing regional analysis 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, at a broad regional scale like the eight-state Southeast region.

6.3.4. Summary and Recommendations

Regional-scale analysis has 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 expansion 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. The 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 or 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 offices have pursued some innovative and promising directions despite limited budgets and lack of specific legal 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 makes the following set of recommendations:

- 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 approaches to valuing 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.
- Integrated research teams: 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.

- Community input and involvement: Gathering extensive stakeholder input is of great importance to establish the set of ecological consequences of greatest importance to the community at large. All regional-scale analyses of the value of ecosystems and services need to include stakeholder involvement 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. As the Chicago Wilderness example illustrates, different individuals and different groups see ecosystems in different lights and have different objectives. A good rule of thumb is to go bottom-up instead of top-down. In other words, it is important to understand what various communities view as being valuable rather than asserting what is valuable. An important question that should be addressed by EPA regional offices is how to develop effective stakeholder involvement at broader regional scales.
- Misapplication of valuation: Some EPA staff have expressed a desire to be given 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 typically uninformed by local social, economic, and ecological conditions and often generate results that are not meaningful. This approach to valuation should be avoided.
- Information exchange: Regional staffs need to be able to learn effectively from efforts to value the protection of ecosystems and services being undertaken by other regional offices and extramural research. 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 be encouraged to publish their studies.
- Extramural research: Future calls for proposals for extramural research should incorporate the research needs of regional offices for systematic valuation studies. Doing so will maximize opportunities that future grant funding will be useful for EPA's regional offices.

- Regional partnerships: Regional staff should be encouraged to 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 protect environmental quality.

7 SUMMARY OF MAJOR RECOMMENDATIONS AND CONCLUSIONS

Among the most important benefits that EPA provides to society are the safeguarding and enhancement of ecological systems and services. Any effort to value particular EPA actions, whether in the form of a national regulation, local cleanup action, or regional conservation partnership, that does not account as fully as possible for these ecological benefits risks substantially understating the value of the actions. To date, however, most EPA valuation efforts have provided only limited information about the ecological effects and their value, focusing instead on the human health benefits of EPA's actions.

To ensure complete and accurate valuation of its actions, EPA must therefore take a more comprehensive approach to assessing, valuing, and reporting on the ecological effects of its actions. A more comprehensive approach can benefit EPA and the nation in multiple ways. First, valuations that do a more complete and accurate job of incorporating changes in ecological systems and services can help decision makers make better, more informed decisions. Second, ecological valuation information can help EPA, other governmental agencies, stakeholders, and the public as a whole to be more proactive in taking actions that protect ecological systems and services. Third, a more comprehensive approach can help EPA educate the public and stakeholders about ecological systems and services and their importance to society. Finally, by providing for greater public involvement in valuation efforts and communicating more clearly about the methods used, a more comprehensive approach can build trust in Agency science and decision-making.

In valuing the ecological effects of an action or proposed action, EPA should identify all of the affected services that are important to the public and then provide as much information as practical about both the impacts of the action or proposed action on these services and the value of these impacts. EPA in the past has sometimes worked in the reverse direction – first identifying those services that it believed it could monetarily value or that it has valued in previous assessments, and then focusing its valuation efforts on these services. Such an approach risks ignoring those services that are of greatest societal importance and under-representing the value of EPA's action to decision makers and the public. Even where full monetary valuation of a specific service is impossible, EPA often can provide other information of use to decision makers such as the likely biophysical changes in the services and the relative importance of the services to the public. **EPA, in short, should start any valuation effort by deciding what it *should* value, not what it can *most easily* value.**

The critical first step in ensuring that EPA's valuation efforts address all relevant ecological effects is to begin all valuation assessments by developing a conceptual model of the potential ecological changes. This model, which should be constructed at a general level, can serve as a roadmap to guide both the identification of relevant ecological effects and the development of more specific ecological analyses. The model should include information and linkages for all relevant levels of the ecological system, including information about both the important underlying ecology and the linkages between the ecological outputs and the ecosystem services of importance to society. Peer review of the conceptual model may be helpful in ensuring that the model is sufficiently comprehensive to serve as the foundation for the valuation assessment.

Guided by the conceptual model, valuation assessments should involve four key steps. First, experts must predict the effect of policy-induced changes on basic characteristics of the relevant ecosystems, using ecological models that are scaled and parameterized to the ecosystems. Second, EPA must identify the ecosystem services that are of public importance. These are the services that should be the focus of the valuation assessment. Third, experts must map the predicted ecological changes to changes in these ecosystem services. Finally, experts must quantify or characterize the value of the changes in the ecological systems and services to the extent possible.

This multi-step process requires greater collaboration among ecologists, economists, and other experts than has historically been found at EPA – as well as greater participation by the public. In developing the initial conceptual model of the potential ecological changes, for example, EPA should involve experts in both relevant biophysical aspects of the modeling and social scientists. Through the use of mediated modeling and similar techniques, EPA should also involve the public, and incorporate public views and understandings, in the development of the conceptual model. The identification and development of effective measures of ecosystem services also requires input from ecologists, who know what biophysical changes can be measured; social scientists, who know what can be valued; and the public, who can help in the identification of those services of social importance.

One of the principal questions that EPA faces in the first step of predicting ecological effects is which models to employ. **Because models are continuously being modified, EPA should develop a set of clear criteria for the selection of appropriate models in each valuation exercise rather than dictating the use of particular models.** These criteria should draw on prior work by EPA, the National Research Council, and other experts to identify relevant factors to consider in choosing among alternative models.

Both the public and experts, as noted above, have a significant role to play in the second step – determining which ecological services are important to the public. The relevant question is what is of importance to the public, not to experts. Public surveys, individual narratives, mental model research, focus groups, content analysis of public comments, and similar approaches can help identify relevant public attitudes. **The public, however, often may not fully understand the nature or potential importance of particular effects. Experts therefore have an important role to play in helping to educate the public about the science of ecological systems and services, the role that particular services play in advancing societal interests and goals, and the likely impact of particular changes in those services on the public.**

One of the critical gaps today in many valuation assessments comes in the third step – identifying how the biophysical effects of an action on an *ecosystem* will in turn impact the *ecosystem services* of importance to the public. Even where ecologists are able to assess the likely ecosystem effects, ecological production functions often do not exist for determining the quantitative relationship between changes in an ecosystem and changes in the services that the ecosystem supports. **A major priority in EPA's research therefore should be the development of ecological production functions that can be effectively applied in valuation assessments to predict changes in ecosystem services based on changes in the underlying ecosystem.** A number of research groups are currently working to develop a first generation of models for measuring and mapping ecosystem services and changes to those services under various scenarios. **The committee believes that EPA can significantly contribute to defining what types of ecological production functions and models would be useful to its work and then supporting the further development of these production functions and models.**

As discussed in Chapter 4and Appendix B of this report, multiple methods exist not only for identifying potentially important services but also for the final step in the assessment process – valuing changes in ecological systems and services. EPA has historically focused on using economic methods to assess the value of such changes, and the Office of Management and Budget requires that economic methods be used, where possible, in valuations for Regulatory Impact Assessments of major rules and regulations. Economic methods have the advantage in cost-benefit assessments of measuring the value of all costs and benefits in a common metric that permits ready comparison of the costs and benefits of particular actions. Economic methods also rest on a well-developed and consistent theoretical framework, and significant economic data has been collected that may be of use in developing monetary valuations in specific cases.

A number of additional methods identified in Chapter 4 also exist for assessing the value of changes in ecological systems and services and have been used far less in actual valuation efforts by EPA. Some of these methods (such as psycho-social survey of public preferences) are grounded in substantial research and experience and are usable by EPA today. Other methods (such as citizen juries) need additional research and development before EPA considers using them as part of formal value assessment processes with significant legal or regulatory consequences. **The detailed descriptions of the various methods contained in Chapter 4 provide guidance on the committee's views of the strengths and weaknesses of each method, which methods are currently ready for EPA use, and the research needed to strengthen and improve each method.**

The committee believes that EPA can and should, where and to the extent permitted by law, make greater and more sophisticated use of those methods that have already been validated by substantial research and experience (including the survey techniques discussed in Appendix C). Such methods can serve a variety of roles. Where current economic methods cannot provide an accurate assessment of the economic value of a particular change in ecological systems and services, psycho-social surveys and other proven methods may provide decision makers with important and useful information on the value that the public attaches to the actions that they are considering. In some situations (e.g., where decision makers are attempting to maximize a particular end such as biodiversity), these methods may provide information of more direct relevance to decision makers or the public in the context of the particular action or actions being contemplated. Providing multiple measures of value also can be important in many settings. Because different methods measure different aspects or concepts of value, the use of multiple methods can provide decision makers with a more comprehensive and robust understanding of the value of pursuing a particular action and thereby help them to make more informed decisions. In some cases, multiple measures of value may reinforce the societal importance of a specific action; in other cases, the multiple measures may push in different directions, requiring decision makers to weigh or balance the various types of value reflected by the measures. In all cases, however, multiple measures will increase the information available to decision makers.

EPA also should be proactive in identifying potential opportunities for testing, using, and further developing new methods of valuation such as citizen juries or ecosystem benefit indicators. Regional and local decisions, in particular, may present settings where new methods can be appropriately tested and refined. In these settings, legal mandates may not constrain the specific valuation methods that can be used, and the decision making setting might be particularly suited to a new method. By seeking out opportunities to use and test new methods, EPA can advance the understanding of these methods and ultimately expand the set of established methods that it has available to use in all settings. EPA also can help advance new methods by developing an extramural grant program focused explicitly on this task.

In choosing which valuation methods to use in any particular setting, EPA should recognize and take into account that different methods rest on different assumptions and concepts of value. The economic methods that EPA has traditionally used, for example, assume that the key values of importance to decision makers are the monetary values that individual members of the public attach to particular ecosystem services based on their role as consumers of such services. Several other methods of measuring public value (e.g., measurements based on the results of initiative or referenda, and citizen juries), by contrast, assume that members of the public attach different values when placed in the role of citizen rather than the role of consumer. Various deliberative and assisted methods assume that many people do not have well formed monetary values for ecosystem services and that accurate valuation requires experts to actively assist people in constructing and determining the value. **EPA should be conscious of the different concepts of value underlying various valuation methods and choose methods for particular assessments based in part on which concepts of value are important or relevant to decision makers in that context. As noted earlier, decision makers may often benefit from the development of multiple measures of value.**

In assessing and reporting value, EPA should also be as transparent and explicit as possible as to what methods it has used, why it chose the methods that it has used, the

assumptions underlying the methods, and the limits of the methods. One goal should be to provide decision makers and the public with information about the assessment, the choices underlying it, and the limitations to it that they need in effectively evaluating the underlying action. A second goal should be to help decision makers and the public understand how EPA derived the values embodied in the assessment report. From the perspective of decision makers and the public, valuation assessments today can frequently be black boxes that yield estimates of benefits and costs but little insight into the makeup of the underlying estimates. Providing more information about the assessment methods can again help decision makers and the public understand the relevance and credibility of the valuation assessment.

Full quantitative valuation, whether in monetary or other metrics, of particular changes in ecological systems and services of importance to the public is not always possible**. Where a full quantitative valuation is not possible, EPA should provide decision makers and the public with as much biophysical information as possible about the change and with other available information that can help decision makers evaluate relevant actions and tradeoffs.** Not surprisingly, OMB's Circular A-4 calls for exactly this type of information when fully monetized valuation is impossible in a regulatory impact assessment.

In these settings, EPA should pursue a hierarchy of information. Where a full quantitative valuation is impossible, EPA should attempt to provide whatever quantitative and qualitative information is available regarding the value that the public attaches to the estimated ecological changes (e.g., information regarding the general importance that the public attaches to biodiversity). Where no valuation information is available, EPA should try to provide information on estimated change in ecosystem services and the reasons that the services are of importance to the public. Where even this information is not available, EPA should provide information on estimated changes in the underlying ecosystem (e.g., in functional groupings of organisms) and how those changes may affect connected ecosystem services. In this regard, EPA should develop key ecological indicators that can be used in multiple contexts to characterize likely changes in ecosystem services of public importance.

One of the most important but difficult issues involved in many major valuation assessments, no matter what valuation method is used, is benefit transfer. Particularly in the case of national assessments, information may not be available to directly value changes in ecological systems and services at all relevant locations. To assess value, experts therefore often seek to use the results of benefit studies done in one location to estimate the benefits of similar ecological changes in other locations and settings. **In conducting such benefit transfers, EPA must be aware of the limitations and risks of such transfers. Inappropriate benefit transfers are often a very weak link in valuation studies. Resource and information limits often make benefit transfer imperative. The committee's review of various uses of benefit transfer, however, indicates that the drive for numerical valuations may present the temptation to push benefit transfers farther than they legitimately can be pushed (e.g., by using valuation estimates that are too old to be reliable). EPA should consider both developing criteria and guidance for the use of benefit transfer and establishing procedures (e.g., expert and in-house reviews) for assessing and determining whether a benefit transfer is appropriate in a particular situation. EPA should also support research that is likely to lead to a larger set of value estimates of likely use in future benefit assessments.**

Similar issues arise in the transfer of ecological information from one context to another. Such extrapolation of ecological information requires caution, and agency experts should carefully evaluate each proposed transfer to determine its appropriateness. Transfers of ecological information may be more acceptable in some contexts than others. Information is more likely to be transferable, for example, where there is significant similarity between contexts or where the information is aggregate.

Valuation exercises are inevitably subject to a variety of uncertainties, and valuation reports should include assessments of that uncertainty. Such assessments are essential if decision makers are to make informed evaluations of proposed policies and alternative policy approaches. Assessments of uncertainty also can help EPA to develop research priorities for the improvement of valuation methods. **Uncertainty assessments should at a minimum report ranges of values and statistical information about the nature of uncertainty for which data exist. Where possible, uncertainty assessments should use formal quantitative methods, such as the Monte Carlo approach, which provide a more reliable and rich characterization of the uncertainty.**

How ecological benefits are communicated is also important, and EPA should focus more attention on communication issues. One area where the Agency can immediately improve its communication is in the characterizing of non-monetized ecological effects in regulatory impact assessments. EPA, for example, should label total monetized benefits as "Total Monetized Benefits" rather than "Total Benefits" to clarify that they do not include all ecological effects. EPA, moreover, should report the non-monetized ecological benefits explicitly and, where possible, in units that are biologically appropriate and socially relevant.

The resources needed to complete an accurate assessment of the value of changes in ecological systems and services should not be understated. The committee encourages EPA to develop a complete vision of what they would like to be able to achieve in valuation assessments and then to seek the resources needed to do so. Regional offices, in particular, need additional resources with which to produce ecological benefit assessments of importance to local and regional decision making either within or outside the Agency.

EPA can maximize the use of the experts and the resources that it currently enjoys by providing for increased and improved information sharing about valuation methods and the results of prior valuation studies and assessments. A number of EPA regions are experimenting with valuation efforts in different settings, and the Agency as a whole would benefit by creating a mechanism for widely sharing the lessons of these efforts. Data compiled in one assessment process, moreover, may prove of value in another setting. **EPA therefore could also benefit by creating a mechanism for identifying and sharing data – not only within the Agency but also from sources outside EPA.** As part of this effort, EPA should establish links with various efforts to collect relevant data, including the NEON planning process and the NSF LTER program.

In conclusion, the committee recommends that EPA "think big" in valuing the changes in ecological systems and services that flow from its actions. Too often in the past, benefit assessments have under-accounted for such ecological changes. In order to ensure that decision makers fully appreciate the benefits of EPA's actions, the Agency must try to understand and assess such ecological changes as completely as possible and in terms that matter to the public. This will require the use of a broader set of tools and a more comprehensive, integrated approach than the Agency has typically utilized in the past.

APPENDIX A: SPECIAL TERMS AND THEIR USE IN THIS REPORT

Ecosystem: A dynamic complex of plant, animal, and microorganism communities and the non-living environment, interacting as a system.

Ecosystem functions or processes: 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 (e.g., pollination, predation, and parasitism).

Ecosystem Services: Those ecological characteristics, functions, or processes that directly or indirectly contribute to the well-being of human populations or have the potential to do so in the future.

Value: This term is used broadly to include contributions to human well-being and goals or ends, such as social and civil norms (including rights) and moral, religious, and spiritual beliefs and commitments.

Valuation: The process of measuring the value or the change in value in terms of the contribution to a specified goal (e.g., human well-being, biodiversity conservation).

Valuation Method: An approach based on theory and data, for measuring the value or change in value in terms of the contribution to a specified goal.

Monetary Valuation: Valuation in which the measurement is done in dollars or some other financial unit.

Willingness-to-Pay (WTP) or Willingness-to-Accept (WTA) Valuation Methods: Methods that estimate the trade-offs individuals are willing to make, expressed in monetary terms. These approaches typically focus on the amount of money an individual is willing to forgo to enjoy a positive change in terms of availability, quantity, or quality of the good or service (willingness to pay). Alternatively, willingness to accept is the amount of monetary compensation a person would accept in lieu of receiving that change.

Social-Psychological Valuation Methods: Methods that focus on individuals' or groups' judgments of the relative importance of, acceptance of, or preferences for changes in ecosystems, their components, or the services they provide. These methods typically focus on choices or ratings among alternatives. Individuals making the judgments may respond on their own behalf or on behalf of others (society at-large or specified sub-groups) and the basis for judgments may be changes in individual welfare, changes in group welfare, or civic, ethical, or moral obligations relevant to ecosystems and ecosystem services.

APPENDIX B: DISCUSSION OF METHODS

BIOPHYSICAL RANKING METHODS

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value	
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?
Conservation Value Method	Map of biodiversity, scarcity and/or conservation values across landscape	Contribution to biodiversity	Measurements related to previously identified goal of biodiversity	Expert - ecologist or conservation biologist

	Degree to which Method Has Been Developed or Utilized	Recommendations for Research to Strengthen Use of Method	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation	Issues Involved in Implementation
Conservation Value Method	Components of approach used by • U.S. Department of Agriculture, • U.S. Forest Service, • U.S. Fish and Wildlife Service, • National Park Service, • Bureau of Land Management, • IUCN, • The Nature Conservancy, • NatureServe	• Integration of stakeholder elicitation approaches (e.g. social scientific surveys) with ecological condition mapping • R&D to show how GIS-based systems could be designed to integrate monetized and other quantitative valuation approaches on a common spatial and temporal GIS background • Where sufficient data does not yet exist, additional resources will need to develop this information in order to complete the methodology	• Use to focus available conservation funds related to conservation goals • Use as a prediction of ecological impacts that would then be used as an input in an economic valuation study • Use in combination with other non-monetary value information (for example, from social-psychological surveys) to characterize preference-based values when monetization is not possible or desirable • Use as a means of quantifying biophysical impacts when they cannot be quantified (as required by the OMB Circular A-4)	• Issues with the lack of data • Currency and confidence in available data • Access to 'sensitive' data represent potential obstacles for the application of this method

Conservation Value Method

Overview In many contexts, decision makers need to know the conservation values for specific biophysical characteristics across different geographies, and the distribution of these values across the landscape. Examples requiring the use of these values include the need to know what sites are important for the conservation of biological diversity, and numerous decisions regarding the protection of wetlands and mitigation of wetland impacts. Every landscape can be characterized by a suite of ecological properties that form the basis for environmental, social, and economic values. The Conservation Value Method is a scientific process to map these values across the landscape for use in decision making. Conservation value can be defined as a measure of the contribution of a landscape unit to the conservation of species diversity, as defined or estimated by relevant experts.

This method also allows the incorporation of social preferences through the development of preferred conservation goals for different biophysical and ecological properties. More than one set of goals can be developed to represent the interests and objectives of different stakeholders. The conservation values are used as the basis for the evaluation of alternative actions in contributing to the social goals that are being addressed. If the social goal is biodiversity conservation, for example, the evaluation of any action is a measure of the contribution of this action to sustained ecosystem diversity and integrity.

This method assigns a value to each individual land area within a given region based on its contribution to a conservation-based goal. This application of scientific information and methodology results in the mapping and valuation of biological and ecological features in a regional context. This provides spatial value attributes for the representative biological and ecological characteristics and features of that area. These can include both biotic factors (e.g., distribution and abundance of plant and animal species) and abiotic factors (e.g., soils, hydrology, climate) that are spatially distributed across the landscape. Some of these features in turn provide information about the ecosystem services provided by the land. This method can be completed with current Geographic Information System-based technologies.

Because each land area has multiple ecological dimensions, the values associated with the contributions of these different dimensions are often weighted and aggregated, with the weights determined by the relevant stakeholders in a given decision context. Different stakeholders will apply different weights, depending on the objective of their analysis (e.g., biodiversity vs. wetlands protection). In addition, spatial information about ecological characteristics can be overlain with other spatial data of interest to these stakeholders.

This process of weighting and mapping the resources that represent what people want to preserve is sometimes referred to as "green printing." For example, groups such as Trust for Public Lands use this phrase when working with Watershed Stakeholder groups to get them focused on steps to implement conservation. It allows for an effective approach with multiple stakeholders to prioritize parcels in the landscape for acquisition and conservation.

Brief description of the method The Conservation Value Method, as detailed by Grossman and Comer (1994), was developed as a general approach to create biodiversity-based conservation values. It represents a structured set of steps for constructing those values, and is built to incorporate the input of stakeholders at multiple points in the process. These values are generated from system attributes for uniqueness, irreplaceability, level of imperilment, and ecological services.

The method begins with an identification of the species, ecosystems, and associated ecological services – and an assessment of their status and condition across the landscape of concern. The evaluation is based on characteristics such as rarity, representation, threat, landscape integrity, and other relevant factors. There are several national databases that can provide much of the baseline information. The network of state Heritage Programs develop and maintain status and distribution information about thousands of plants and animals, along with different vegetation and ecosystem types. The Integrated Taxonomic Information System (ITIS) maintains a standardized list of species names for use by scientists and federal agencies. The U.S. Fish and Wildlife Service maintains information about endangered species and wetlands, the U.S. Geological Survey manages databases characterizing ecosystem characteristics and integrity, and the Department of Transportation manages information on the density and location of roads and infrastructure across the country. The standardized integration of these datasets within the Conservation Value methodology provides a robust foundation for decision making.

The places where a given element of conservation interest is found (termed an "occurrence") is assigned a quality and viability score based on attributes of size, condition, and landscape integrity. The trends and condition for each conservation element are presented in a summary status attribute, a conservation rank (reference NatureServe, IUCN). The global assessment and the quality information about individual occurrences are then used to develop a spatial "ecological value layer," which portrays a spatial distribution of the conservation value along with metadata regarding the quality and confidence of each occurrence. This layer can reflect the specific conservation goals of the stakeholders, as they can alter the relative importance of different conservation elements based on their management or conservation objectives. To the extent that stakeholders are interested in multiple ecological features (e.g., multiple species), the information for each ecological value layer is aggregated to create an overall "conservation value summary." This summary value layer provides a spatially aggregated representation of the biodiversity and conservation values that represent the values of the conservation or management stakeholders. The final (aggregate) conservation values are used to support decision making, e.g., to prioritize preservation-based land acquisitions, mitigate wetland loss, direct point and non-point source permits, etc. These spatial conservation values can also be integrated with socio-economic and other spatial data to integrate those data into the decision-making process.

The Conservation Value Method was developed primarily to identify priority areas and activities that would sustain or improve the condition of biodiversity and ecosystem health. This GIS based methodology can support different types of decisions by adding different data and values to the model. For example, one could quantify Bureau of Land Management land for its value as recreational use, natural resource extraction (timber, mineral, oil and gas), and water quality (denitrification, water purification) and quantity (flood control, snow pack).

This method is often used to evaluate the impact of a proposed action on current conditions. This requires the development of future scenario maps that can reflect a new policy, a development action, modeled population growth, a natural disaster, or any number of different change scenarios. The intersection of the change scenario with the conservation value model allows for clear reporting on the changes to either the composite conservation value or the individual conservation values. This is often used to choose between change scenarios (e.g., road placement, point source licenses), and to protect against potential threat (toxic transport, oil line placement).

The Conservation Value Method can contribute to EPA decision making in a number of ways. First, in contexts where the Agency's goals are defined in terms of conservation objectives or requirements, such as under the Endangered Species Act, the method could provide a means of making decisions about where to focus available conservation funds. In addition to contributing to decision making focused on specific conservation goals, the outputs from the conservation method could play a key role in EPA decision making (and the C-VPESS valuation framework) in the following ways: a) it could be used as a prediction of ecological impacts that would then be used as an input in an economic valuation study; b) it could be combined with other non-monetary value information (for example, from social-psychological surveys) to characterize preference-based values when monetization is not possible or desirable; and c) it could be used as a means of quantifying biophysical impacts when they cannot be quantified (as required by the OMB Circular A-4).

<u>Status as a method</u> The Conservation Value Method approach represents a sequence of iterative steps that have been developed by the scientific community over the past thirty years. (References?)The components that have been aggregated into this emerging methodology include ecological classification and mapping standards, conservation ranking standards, conservation planning methodology, and occurrence mapping standards. There is widespread use of various components of these methods across U.S. federal agencies, though the utility use of the comprehensive integrated methodology has only recently become accessible and manageable for the non-specialist. The ranking methodologies for conservation elements (plant, animals, and ecosystems) has been documented in the scientific literature over many years and is in common use by numerous federal agencies (e.g., U.S. Department of Agriculture, U.S. Forest Service, U.S. Fish and Wildlife Service, National Park Service, and Bureau of Land Management). (References?) The viability and quality ranking criteria for the occurrences of conservation elements has been the topic of widespread

analysis by IUCN, The Nature Conservancy, NatureServe and others. The conservation planning methods have emerged from Australian natural resource agencies (e.g., CSIRO) and are well published in the conservation science literature. (References?) EPA has used different components of this methodology to identify and prioritize rare and threatened species that need protection (e.g., working with the pesticide industry to protect biological diversity) and to characterize different wetland ecosystems to prioritize protection activities. (References?)

This methodology is increasingly being used by the larger planning community for different purposes at multiple scales. The examples listed below will illustrate the breadth of these applications. The Land Trust of Napa County has used the methodology to identify priority conservation acquisitions for the next ten years. The U.S. Forest Service is testing its use for the development and monitoring of National Forest plans. The Conservation Trust of Puerto Rico has applied these methods to clarify conservation and development priorities and options across the island. The state of Mata Grosso in Brazil is using this approach to integrate a conservation reserve program into private landholdings.

Decision contexts where this method could be used by EPA include:

- Enumeration of biodiversity protection implications that result from policy changes (i.e., change of protection status for isolated wetlands)
- Identification of critical riparian habitat
- Prioritization of remediation action on superfund sites
- Due diligence reviews and Environmental Impact Statements as a prerequisite for permitting
- Identification of reference conditions for establishment of baseline quality metrics for wetland and aquatic habitats
- Assessment of the status of target species and ecosystems
- Analysis of mitigation equivalencies and priorities
- Baseline information for ecosystem integrity and environmental impact monitoring

Strengths/Limitations

Conceptual Strengths/Limitations The Conservation Value Method will create a quantitative spatial representation of ecological and biological values within a regional context. The spatial range of these analyses can vary from local to regional scales. This data provides a baseline for a broad range of natural resource assessment and management decisions, and can be integrated with spatial monetary valuations to inform cost-effective land management and regulatory decisions. The specific decisions will determine that types of data and analyses that are required to address the question.

The Method's Strengths

- The method is adaptable to address different questions.
- The method can be run repeatedly to represent temporal change or different landscape scenarios.
- Results are commonly aggregated to derive a single benefits number, but all of the native data is constantly maintained in the system and can be presented separately.
- The output is both understandable and communicable to the interested audience and other stakeholders. Provides the opportunity for visualization of outcomes that many other methods lack.
- The results are repeatable, and the process and algorithms are very transparent.

The method's weaknesses Issues with the lack of data, the currency and confidence in available data, along with access to 'sensitive' data represent potential obstacles for the application of this method. There are many ways to create surrogate datasets that will allow users to adapt to different types of barriers. Some training and tools are also required to use this method.

Practical Strengths/Limitations

The assumption is that there is sufficient coverage of standardized biodiversity data required to implement these methods. The standards for each step of the method have been developed, and the data that is required will be dependent upon the specific application questions. Where sufficient data does not yet exist, additional resources will need to develop this information in order to complete the methodology. In some cases, surrogate information and models are required to incorporate the spatial representation of poorly inventoried conservation targets across the landscape.

This method requires local scientific data, knowledgeable scientific interpretation and conservation planning expertise. The magnitude of the need is contingent upon the application and the current state of data and knowledge. There are many sources available from which to obtain this knowledge.

<u>Treatment of Uncertainty</u> There are confidence measures built into the methodology that can be integrated into the decision-making analysis or displayed independently for consideration. The most significant sources of uncertainty in the use of this method include:

- The variability in the quantity and quality of the data
- The limitations of scientific understanding of distribution and quality criteria for some ecological factors
- The level of stakeholder understanding of the linkages between ecological components and the services they value

<u>Research needs</u> There is both a need and an opportunity to actively explore integration of stakeholder elicitation approaches (e.g., social scientific surveys) with ecological condition mapping. Additional R&D to show how GIS-based systems could be designed to integrate monetized and other quantitative valuation approaches on a common spatial and temporal GIS background could yield significant benefits.

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Rankings Based on Energy and Material Flows

Introduction

Energy and material flow analysis is the quantification of the flows of energy and materials through complex ecological or economic systems, or both. These analyses are based on an application of the first (conservation of mass and energy) and second (entropy) laws of thermodynamics to ecological-economic systems. A recent report by the National Research Council (NRC) covers the basic elements and need for such analyses (Committee on Materials Flows Accounting of Natural Resources, Products and Residuals 2004). The NRC report concludes that information about material flows can be a very useful input for policy decisions. It can be used to identify potential environmental concerns and key sources of pollution and to develop strategies for preventing environmental releases.

This section provides general background on energy and material flow analysis as a means of identifying and quantifying important relationships within ecological and economic systems. It then discusses two methods that translate the physical energy and material flows into measures that could be used in the context of ecological valuation. The first is embodied energy analysis, which estimates the direct and indirect energy (or more correctly, available energy or "exergy") cost of goods and services. The second is ecological footprint analysis, which estimates the biologically productive land or water areas required (directly or indirectly) to meet various consumption patterns. We also briefly discuss the use of the concept of "emergy" for estimating energy costs and valuation.

Energy and Material Flows Analysis

Energy from the sun drives plant productivity as well as climate and hydrologic cycles, nutrient cycles, ocean currents, weathering and soil formation. Thus a study of energy and material flows in ecosystems relates very directly to the production of ecosystem services. Ecologists have long utilized studies of the flow of energy and materials (e.g., nitrogen, phosphorus) through ecosystems as a way of describing key relationships and understanding the functioning of those ecosystems. Early studies of energy flow in aquatic (e.g., Lindeman 1941) and terrestrial (e.g., Golley 1960) systems illustrated how energy moved through food chains. Ground-breaking analyses of the cycling of critical nutrients in lakes (e.g., Hutchinson 1947) and forests (e.g., Likens and Borman 1977) set the stage for many subsequent analyses and established the field of biogeochemistry. Studies of energy and materials flows can be especially useful for understanding how changes to an ecosystem, such as an increased or decreased level of pollution, may alter the system and the services it provides. For instance, increases or decreases in the inputs of nitrogen to a forest from acidic deposition may impact forest productivity, species composition, and nitrogen runoff in streams and rivers (e.g., Johnson and Lindberg 1991). Larsson, et al. (1994) used energy and material flows to demonstrate the dependence of a renewable resource such as commercial shrimp farming on the services generated by marine and agricultural ecosystems. The committee seconds the view expressed by the NRC (2004) that analyses such as these can provide very valuable information about ecological services and how the flow of services might change in response to specific stressors.

The energy and environmental events of the 1960s and 1970s prompted a number of economists, ecologists, and physicists to examine the energy and material flows underlying the economic process (Boulding 1966, Georgescu-Roegen 1971, 1973). Ecologists noted the importance of energy in the structure and evolutionary dynamics of ecological and economic systems (Lotka 1922, Odum and Pinkerton 1955, Odum 1971). The integration of the first law of thermodynamics with the economic system was first made explicit in the context of an economic general equilibrium model by Ayres and Kneese (1969) and subsequently by Mäler (1974). It is also a feature of a series of linear models developed after 1966 (Cumberland 1966, Victor 1972, Lipnowski 1976). All reflect the recognition that the earth is a thermodynamically closed (but not isolated) system, with energy from the sun crossing the boundaries and maintaining the structure and function of the earth system. A closed system must satisfy the conservation of mass condition. Ayres (1978) described some of the important implications of the laws of thermodynamics for the economic production process, noting that both manufactured and human capital require materials and energy for their own production and maintenance (Costanza 1980).

A key feature of energy flow analysis is the recognition of the importance of energy *quality*, namely, that a kcal of one energy form (e.g., electricity) may produce more useful work than a kcal of another (e.g., oil). Estimating total energy consumption for an economy is therefore not a straightforward matter because not all fuels are of the same quality, that is, they vary in their available energy, degree of organization, or ability to do work. This effort to incorporate energy quality is often referred to as “second law analysis.”

Embodied Energy Analysis

As noted, methods have been developed that seek to use energy and material flows information to determine values associated with different systems or changes in those systems. One such method is embodied energy analysis. The embodied energy method assesses the direct and indirect energy costs of economic and ecological goods and services. It uses input-output tables to determine the direct and indirect energy inputs used to produce these goods and services. Although there is no stated Agency policy to use or develop supplemental valuation methodologies in this area, there is substantial Agency interest in how Energy and Material Flow methods might aid decision making. Recent efforts to explore the utility of such methods, mostly at the regional or local level, are underway (Bastianoni et al., 2005, Campbell 2001, 2004, Lu, et al. 2006).

Some ecologists and physical scientists have used estimates of embodied energy to implement an energy theory of value either to complement or replace the standard neoclassical theory of subjective utility-based value (Soddy 1922, Odum 1971, 1983, Slesser 1973, Gilliland 1975, Costanza 1980, Cleveland, et al. 1984, Hall, et al. 1992). The energy theory of value is based on thermodynamic principles, where solar energy is recognized to be the only primary or external input to the thermodynamically closed global ecosystem. At the global scale, the traditional primary factors of production (labor, manufactured capital, and natural capital) are viewed as intermediate factors (Costanza 1980).

There has been ongoing debate about the validity of an energy theory of value (Brown and Herendeen 1996). Some believe that it is the only reasonably successful attempt to operationalize a general biophysical theory of value that does not hinge completely on consumer preferences (see also Patterson 2002). Neoclassical economists, on the other hand, have criticized the energy theory of value as an attempt to define a concept of value that does not directly reflect consumer preferences regarding the good being valued (see Heuttner 1976). This criticism is, on the one hand, axiomatic, since a major purpose of an energy theory of value is to establish a theory of value not completely determined by individual preferences. On the other hand, techniques for calculating embodied energy utilize economic input-output tables. These tables summarize production interdependencies, but they are not completely independent of consumer preferences, which helped to structure the production interdependencies over time. Neoclassical economists also question the primary status of energy, because in any concrete, short-term situation the scarcity and prices of the conventionally-defined inputs of manufactured capital, labor, and technology are also important. While not denying the importance of these short-term considerations, energy theorists take a broader, more evolutionary perspective, recognizing that these factors are intermediate and that production relationships adapt over time.

As noted, the energy theory of value (like the labor theory of value developed by classical economists) is inherently based on relative production costs, i.e., it yields a measure of (direct plus indirect) energy cost. The question arises as to when these energy-based production cost estimates can provide a measure of value. This is similar (but not identical) to the question that arises in the context of replacement costs based on the standard economic concept of opportunity cost (see Chapter 4.1.7 and more detailed discussion of replacement costs in the Appendix below). In economic systems, marginal cost and price will be equal in a perfectly competitive equilibrium. This means that, in the absence of other market distortions, an estimate of marginal cost can provide a proxy for the value of an additional unit of production. Similarly, an estimate of production cost can provide a proxy for value, but only under certain circumstances (see discussion in section on replacement cost). For example, the aggregate individuals must be willing to incur these costs rather than forego the good or service. One difference between replacement and production costs is that while replacement costs are hypothetical, production costs have already been incurred, implying that aggregate individuals *were* willing to incur the costs, thus satisfying this condition. To the extent that the necessary conditions are met, energy costs can provide information about the value of the associated goods or services as defined by the energy theory of value.

Costanza, et al. (1989) provide an example of wetlands valuation that uses both a conventional WTP approach and a simplified energy analysis approach based on the gross primary productivity (GPP) of coastal wetlands in Louisiana. The energy analysis valuation technique compared total biological productivity of a wetland versus an adjacent open water ecosystem. Primary plant production, which supports the production of economically valuable products such as fish and wildlife, was converted to a monetary value based on the cost to society to replace this energy source with fossil fuel as measured by the overall energy efficiency of economic production. While the results of the WTP- and GPP-based methods were fairly consistent, the authors note that the GPP approach probably represented an upper bound and “may overestimate their value if some of the wetland products and services are not useful (directly or indirectly) to society" (Costanza, et al. 1989, p 341). However, it should be noted that the basic assumptions underlying an energy theory of value imply that there is no reason to expect measures based on energy cost to be the same as preference or WTP-based measures of value.

Ecological Footprint Analysis

The ecological footprint (EF) method is a variation of energy and material flow analysis that converts the impacts to units of land or water rather than energy or dollars. The EF for a particular population is defined as the total "area of productive land and water ecosystems required to produce the resources that the population consumes and assimilate the wastes that the population produces, wherever on Earth that land and water may be located" (Rees 2000). While usually discussed in the context of the footprint of specific human populations, this concept can also be applied to non-human populations. For example, a portion of the southern Chesapeake Bay has been set aside as a blue crab sanctuary since large numbers of the organisms spawn in this area relative to elsewhere (Virginia Marine Resources Commission, Newport News, VA). In the context of human societies, input-output methods (see previous discussion) are used to estimate direct and indirect land requirements.

Although there are ongoing debates about specific methods for calculating the ecological footprint (Costanza 2000, Herendeen 2000, Simmons, et al. 2000), the ecological footprint is an effective device for presenting current total human resource use in a way that communicates easily to a broad range of people (http://www.footprintnetwork.org/). In terms of valuing ecosystem services, the ecological footprint concept is most useful as an index of the quantity of ecosystem services consumed (expressed in units of a standardized land area) for various consumption patterns. This measurement, however, does not directly convert to a monetary measure of the value of ecological services. It does, however, allow a relative comparison of one footprint to another based on areas or sizes involved. Under this approach, *ceteris paribus*, a population that has a smaller footprint is viewed as more sustainable. On the other hand, a larger footprint implies a larger biocapacity supporting a given population and a larger required contribution of ecosystem services to maintain that population in its current state.

Emergy Analysis

Emergy analysis shares many of the same goals and assumptions as embodied energy analysis. For example, solar emergy is defined as "the available solar energy used up directly and indirectly to make a service or product" (Odum 1996). Emergy analysis differs from embodied energy analysis and ecological footprint analysis in terms of the method used to estimate the energy required. While embodied energy and footprint analysis use methods based on input-output (a well-developed set of methods for this type of accounting), emergy analysis uses different methods (See recent work by Ukidwe and Bakshi, in press).

Emergy analysis starts with the creation of an energy flow diagram. The "Solar Transformity" is then defined as "the solar emergy required to make one Joule of a service or product" (Odum 1996). This is calculated by dividing any flow in the diagram by the total solar energy input. Odum and coworkers have thus calculated the emergy of the earth's main processes, such as the total surface wind, rain water in streams, the sedimentary cycle, and waves absorbed on shore, to be that of the total emergy input to the earth (Odum 1996). Each of these processes is assigned the total value of incoming sunlight because they are considered co-products of the global geological cycle and cannot be produced independently with less amount of the total emergy.

However, emergy has encountered considerable resistance and criticism, particularly from economists, physicists, and engineers (Hau and Baksi 2004, Ayres 1998, Cleveland, et al. 2000, Mansson and McGlade 1993, Spreng 1988). Consequently, the emergy approach has only been used by a small circle of researchers, although some work at EPA is ongoing (U.S. Environmental Protection Agency 2005). Emergy's accounting method does not produce an estimate of the energy cost of goods and services, but rather "the relative equivalence between energies of different kinds in terms of a universal quality factor." This concept is difficult to understand and to apply in a standard accounting framework. Although the committee as a whole did not study the debate over emergy in detail, the committee believes that substantial questions exist regarding the appropriateness and usefulness of emergy as a method for valuing ecological systems and services.

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ECOSYSTEM BENEFIT INDICATORS

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value	
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?
Ecosystem Benefit Indicators	Map of the supply of ecosystems/services showing quantities of expressed or estimated demand for those ecosystems/services across a landscape	Quantitative but not monetary approach to preference weighting for the ecological effects of policy options	Measurements related to demand variables that can be identified by experts or non-expert lay publics and supply variables as identified by experts.	Expert and selected non-expert lay public

	Degree to which Method Has Been Developed or Utilized	Recommendations for Research to Strengthen Use of Method	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation	Issues Involved in Implementation
Ecosystem Benefit Indicators	The method is new and relatively undeveloped	• Integration of EBIs with biophysical endpoints • Integration of EBIs with econometric valuation methods (benefit function transfer, stated preference and choice modeling) • Suitability for group decision techniques, such as mediated modeling • Practical application to illustrate data needs and measurement issues	• Input to a wide variety of trade-off analyses (for regulatory analyses or performance measures) • Use as part of public processes designed to communicate the implications of a change or policy across a variety of scales • Use as inputs to economic and econometric methods such as benefit transfer, or stated preference models • Use to systematize alternative choice scenarios in choice experiments and stated preference surveys	• Do not directly yield dollar-based ecological benefit estimates • Do not in themselves weight or estimate the trade-offs associated with different factors relating to benefits • Uncertainty with regard to how indicators are perceived, particularly when presented visually should be acknowledged

Introduction

This report describes a range of valuation methods. The choice of method will depend on the environmental question at hand, the political and regulatory process involved, and differing philosophical perspectives on the nature of value and how it is to be determined by society. All of these methods, however, require the analyst or decision maker to be informed.

Two basic forms of information are required: the knowledge of what is at stake in nature and the ability to determine how ecological endpoints change as a result of management or regulation. The first piece of information comes under the realm of biophysical production function analysis. If the agency can achieve clear, actual or predicted production function-based outcomes, that would be a great advance over current practice.

Assuming these kinds of information and analysis are available, social scientists are then called upon to weight, prioritize, or value different outcomes in nature. What kind of information should be relied upon for weighting, priority-setting, and valuation of ecological changes?

Recommendation: The committee advocates the Agency more broadly collect and communicate ecosystem benefit indicators (EBIs) to inform the social weighting and valuation of ecosystem services. EBIs are not themselves a valuation method. Rather they are an inventory of data and set of principles that should be used to inform the public or analyst as part of any valuation exercise.

Elsewhere in this report the committee has emphasized the importance of ecosystem *services'* spatial and landscape context. *Where* services arise is very important, both ecologically and socially. From a social science standpoint, the determinants of value depend upon the landscape context in which ecosystem services arise. Habitat support for recreational and commercial species, water purification, flood damage reduction, crop pollination, and aesthetic enjoyment are all enjoyed in a larger area surrounding the ecosystem in question. EBIs allow for spatial representations (both geo-coded data and corresponding visual depictions) of social and biophysical features that enhance or decrease the benefits of a particular ecosystem services in particular places.

Regulatory and ecological ecosystem assessments, including many of those reviewed by this committee, often ignore information that is fundamental to valuation – however valuation is defined. For example, how many people benefit from a particular ecological function or service? The number of people who can enjoy the service in a given location is an example of an important EBI.

- The committee also found scant evidence that the Agency analyzes the scarcity of particular ecosystem services, the presence of substitutes for those services, or the dependence of environmental benefits on the presence of complementary goods and services. EBIs are a way to relatively quickly and cheaply address this information gap.
- EBIs are of practical use to the Agency because the cost of collecting them is relatively low. EBIs are generated from GIS data and can be quickly assembled, usually using existing data sets employed by federal, state, and local governments.
- EBIs can and should be used to educate decision makers and stakeholders about the underlying complexity of ecological and economic relationships. They are not a way to simplify the decision maker's problem. Rather, they provide basic information that informs the decision process about the trade-offs arising from a particular decision.

Examples

To illustrate the use and benefits of EBIs, consider the following example: wetlands can improve overall water quality by removing pollutants from ground and surface water. This ecological function is valuable, but just how valuable? To answer this question one can count a variety of things, such as the number of people who drink from wells attached to the same aquifer as the wetland. The more people who drink the water protected by the wetland, the greater its value.

But other things matter as well. For example, is the wetland the only one providing this service or are others contributing to the aquifer's quality? The more scarce the wetland, the more valuable it will tend to be. There may also be substitutes for wetland water-quality services provided by other land-cover types such as forests. Mapping and counting the presence of these other features can further refine an understanding of the benefits being provided by a particular wetland.

Many ecosystem benefits arise only in the presence of complementary features. Recreation typically requires access to natural areas. Road, trail, dock or other forms of access are thus important to the analysis of benefits. In some cases, if there is no access, there can be no benefit.

Consider another type of environmental benefit: aesthetic value arising from natural viewscapes. Here, relevant stakeholders and decision makers would benefit from the following kinds of EBI: population in viewshed of the natural area (primary demand); percent of that population's viewshed that is natural (scarcity); the number and extent of substitute viewsheds for this population (substitutes); the presence of roads, trails, boatable surface waters, public lands, and access points that allow the natural area to be viewed (complements).

In general, EBIs should be specific to the ecosystem service and benefit in question. Consider two different ecosystem benefits: recreational angling and provision of clean drinking water. The EBIs relevant to these two benefits will be different. In both cases, the number of people benefiting is relevant, but the populations are different. Demand for recreational angling would involve assessment of the number of potential anglers. This population is different from the population benefiting from a given aquifer's water quality. The determination of scarcity and substitutes is very different as well. All of these examples of EBIs can be mapped and counted using geo-coded social (e.g., census) and biophysical data.

<u>Brief Description</u>

EBIs are countable landscape features that tell us about demand for, scarcity of, and complements to particular ecosystem services. Ecosystem benefit indicators (EBIs) are quantitative inputs to valuation methods. They can serve as important inputs to valuation methods as diverse as citizen juries and econometric benefit transfer analysis, which is a monetary weighting technique. EBIs provide a way to illustrate ecological benefits in a specific setting. For example, if water is available at a particular place and time, how many water users (e.g., recreationists, farms) are present to enjoy that service? What other sources of water are available to those same users? These questions are central to economic valuation of the resource.

<u>Key inputs -</u> EBIs are drawn mainly from geospatial data, including satellite imagery. Data can come from state, county, and regional growth, land-use, or transportation plans; federal and state environmental agencies; private conservancies and nonprofits; and the U.S. Census.

Key outputs - Spatially specific measures (both geo-coded data and corresponding visual depictions) of social and biophysical features that enhance or decrease the desirability of particular ecosystem services.

Scale - The method is entirely scalable. One strength, however, is the ability to relate ecological and economic features in a specific landscape context. For example, the method can be applied to individual projects, investments, or decisions made in a particular watershed. They can also be expressed as local, regional, state, or national aggregates.

Example of How the Method Could be Used as Part of the C-VPESS Framework

The method relates to framework item (4): "Characterization of the Value of Changes in Monetary and Non-Monetary Terms." Benefit indicators are countable features of the physical and social landscape. More specifically, they are features that influence – positively or negatively – ecosystem services' contributions to human well-being. The consumption of services often occurs over a wide scale. For example, habitat support for recreational and commercial species, water purification, flood damage reduction, crop pollination, and aesthetic enjoyment are all services typically enjoyed in a larger area surrounding the ecosystem in question. EBIs help people understand the larger social and physical landscape so that they can better assess the relative importance of particular services in particular places at particular times.

The value of ecosystem services is likely to be affected by the following factors: the ecosystem feature's scarcity, natural and built substitutes, complementary inputs, and the number of people in proximity to it. For a given ecosystem service scarcity, substitutes, complements, and demand can be related to landscape characteristics. Landscape features that relate to human well-being can be systematically counted and mapped, and then aggregated into bundles of indicators (an index). Some indicators are biophysical, others relate to the socio-economic environment.

Benefit indicators are an input to a wide variety of trade-off analysis approaches, but do not independently make or calculate the results of such trade-offs. First, they can be used as ends in themselves as regulatory or planning performance measures. Second, they can be used as part of public processes designed to communicate the implications of a change or policy across a variety of scales. Indicators or an index based on them can then be used to elicit public preferences over environmental and economic options – as in mediated modeling exercises or more informal political derivations. In this way, benefit indicators are a potentially powerful complement to group decision processes. Third, they can be used as *inputs* to economic and econometric methods such as benefit transfer, or stated preference models. This is an area where research is needed. Economic methods must be developed to link indicator outcomes to dollar-based valuation in a way that is both statistically and theoretically sound. In principle, benefit indicators could be used to calibrate the transfer function in benefit transfers. They could also be used to systematize alternative choice scenarios in choice experiments and stated preference surveys.

As a method to inform the weighting of ecosystem services in a social decision context, the benefit indicators method requires information provided by the biophysical sciences. The method requires spatially depicted biophysical endpoints. EBIs are then related to those endpoints.

The method can be applied to any ecosystem service benefit where benefits are related to the spatial delivery of services and social landscape in which the benefit is enjoyed. Existence benefits (where spatial location is irrelevant to both provision and value) are the only ecosystem benefit category where the method would be inapplicable.

The data used in EBI analysis is well-suited to delivery via a national data bank.

<u>Status as a Method</u>

The method is new and thus relatively undeveloped. EPA has funded a small amount of research on the topic. For citations to peer reviewed research, see below.

<u>Strengths/Limitations</u>

EBIs are designed to be a relatively non-technical way to express the factors that contribute to conventional economic measures of benefits provided by ecosystem services. Their simplicity, and transparency, is an advantage. They can be used to communicate and educate. By stopping short of monetary estimation of benefits (unless integrated in a benefit function transfer method) they are also a way for the agency to overcome resistance to economic assessments of the natural world – while still conveying outcomes in a way designed to be consistent with economic principles and the dependence of human well-being on natural assets.

The principle disadvantage is that they do not directly yield dollar-based ecological benefit estimates. They also do not in themselves weight or estimate the trade-offs associated with different factors relating to benefits (though as noted previously they can be married to more formal methods designed to do such weighting).

Because indicators can be cheaper to generate than econometric value estimates they better allow for landscape assessment of multiple services at large scales.

Treatment of Uncertainty

A core rationale for the use of a benefit indicator approach is to explicitly convey the sources of complexity – and hence uncertainty – characterizing biophysical systems and the benefits arising from them. The visual depiction of benefit indicators, for example, can mimic sensitivity analysis by presenting a range of benefit scenarios in GIS form. However, the visual depiction of quantitative information introduces uncertainties of its own. In particular, visual depictions can strongly influence perceptions. Uncertainty with regard to how indicators are perceived, particularly when presented visually, should be acknowledged.

Research Needs

- Integration of EBIs with biophysical endpoints
- Integration of EBIs with econometric valuation methods (benefit function transfer, stated preference and choice modeling)
- Suitability for group decision techniques, such as mediated modeling
- Practical application to illustrate data needs and measurement issues

Satisfying these needs would be a significant undertaking in terms of expertise, financial resources, and coordination within the agency.

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MEASURES OF ATTITUDES, PREFERENCES, AND INTENTIONS

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value	
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?
Survey questions measuring social-psychological value constructs	Quantitative indices of attitudes , preference rankings, or behavioral intentions toward depicted environments or conditions	Public concerns, attitudes, values, beliefs, and behavioral intentions	Verbal reports, choices, rankings, ratings	representative sample from public
Conjoint attitude survey questions	Indices of expressed attitudes or preferences for multi-attribute alternatives and implied trade-off weights for composite attributes	Public concerns, attitudes, values, beliefs, and behavioral intentions related to specific trade-offs among attributes	Choices, rankings, ratings	representative sample from public
Individual Narratives	Summaries of individual's value-relevant narratives	Implied knowledge, beliefs, attitudes and concerns	Verbal reports from individual stakeholders and lay publics	select sample from public
Mental Models	Systematic, structured models of beliefs and assumptions underlying value positions	Value-relevant knowledge, beliefs and assumptions and their interrelationships	Verbal narratives from individual stakeholders, lay publics or experts	select sample from public
Behavioral Observation/Trace	Observations of current or prior (trace) use associated with ecosystems/services	Responses to policies, outcomes, and consequences, in situ	Current or past behavior associated with changes in ecosystems/services	representative sample of current or past visitors/users
Interactive Environmental Simulation Systems	Observations of behavior in simulated/game environments, implied preferences	Responses to investigator-controlled changes in simulated (virtual) environments	Interactive behavior in response to changes in simulated environments	representative sample from public

	Degree to which Method Has Been Developed or Utilized	Recommendations for Research to Strengthen Use of Method	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation	Issues Involved in Implementation
Surveys Including Attitude, Preference and Behavioral Intention Questions	• Survey questions measuring social-psychological constructs are the oldest and most frequently used methods for determining public beliefs, concerns, and preferences. • Survey questions have been used, and continue to be used, effectively by all levels of government to measure citizen desires concerns and preferences.	• How can attitude/preference surveys best be used in EPA policy and decision making, including how decision makers can and should use the relative quantitative (non-monetary) value indices provided? • How can attitude/preference indices be used to cross-validate decisions implied by estimates of monetary values (e.g., from CBA) for alternative policies/outcomes? • How should value indices from attitude/preference surveys be integrated with bio-ecological and economic value indices to strengthen support for ecosystems/services protection policies and decisions?	• Can contribute to initial problem formulation by identifying ecological services and impacts that most concern citizens and/or identified stakeholders, as well as by uncovering assumptions, beliefs, and values that underlie that concern • Can help to determine and quantify socially important assessment endpoints • Can be used to assess relative public preferences among policy options and their attributes • Quantitative attitude/preference indices may be especially useful when the values at issue are difficult to express or to conceive in monetary terms or where monetary valuations are viewed as ethically inappropriate • Can be used to help inform and to systematically involve publics in the balancing of multiple values for ecosystems/services protection decisions	• Institutional barrier of the Paperwork Reduction Act • Responding public may not have adequate knowledge and understanding of complex ecosystem processes, or well-formed opinions and preferences for protection options • Designing and implementing a well-designed survey requires expertise that may not be sufficiently represented within the Agency (see Appendix C
Conjoint Attitude Survey Questions	Relatively new variation on survey methods used sparingly in environmental valuation context, but increasingly being used in business/marketing, health care, tourism-recreation, and other value assessment applications	How do the overall values for multi-attribute conjoint policy options and the individual attribute weights inferred from choices/ratings relate to separately assessed values for the same policies and attributes? What are the specific advantages and disadvantages of conjoint methods for ecosystems/services value assessments?	• May be especially well-suited for gauging public preferences across sets of complex multi-dimensional policy alternatives, as are likely to be involved in many EPA regulations and actions for ecosystems and services protection	Same as for surveys in general, plus special concerns about publics' ability to understand and respond appropriately to complex, multi-attribute alternatives Special experimental designs and data analyses can be complex and are unfamiliar to many researchers and analysts
Individual	• Infrequently applied in	• What productive roles can	• Can make important contributions to	The selection of participants can

	Degree to which Method Has Been Developed or Utilized	Recommendations for Research to Strengthen Use of Method	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation	Issues Involved in Implementation
Narratives	EPA valuation contexts, but increasingly used to address social, psychological, and anthropological questions related to values, attitudes, and behavior intentions in other environmental management contexts	individual interviews and other qualitative methods play in Agency policy and decision making? • How should the results of qualitative analyses best be integrated with quantitative assessments (bio-ecological, attitude/preference, and economic) to strengthen support for policy/decision making?	improving the design, development, and pre-testing of more formal surveys that can provide reliable and valid quantitative assessments of public concerns and values • Can assist in identifying and articulating the conceptual basis of public values and concerns, especially in early stages of problem formulation and value assessment	have very important effects on outcomes—formal representative (probability) sampling is not typically used and no scientifically accepted alternative selection method has been developed. Rigorous qualitative analysis methods have been developed but are rarely used and qualitative methods in general have not been adequately tested in ecosystems/services valuation contexts.
Mental Models	• A relatively new variation on individual narrative procedures which can use rigorous analytic methods to extract and structure participant's knowledge, beliefs, and assumptions into a coherent logical structure.	• How might mental models be effectively used to design appropriate value elicitation and assessment methods for ecosystems and services? • How might mental model structures best be integrated with the results of other methods to provide deeper insights into value assessment to support policy/decision making?	• Appropriate precursor (i.e., formative analysis) to any formal survey or preference elicitation method, to improve the validity and reliability of the method. • May be especially useful for exploring the bases of value conflicts between segments of the public, or between publics and expert/scientific opinions.	Research and development is needed to secure a consistent and rigorous set of methods for qualitative analysis and mental model construction.
Behavioral Observation/ Trace	• Relatively new and untested in value assessment and policy formulation contexts, but research and trial applications are increasing	• How might the development of emerging behavior observation and behavior trace methods be shaped to effectively contribute to Agency policy and decision making needs?	• Might be used to attain quantitative measures of human use levels useful in conjunction with economic measures or as separate measures to be correlated with changes in ecological conditions	In-situ field observations are constrained to existing conditions and are subject to the effects of uncontrolled variables.

	Degree to which Method Has Been Developed or Utilized	Recommendations for Research to Strengthen Use of Method	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation	Issues Involved in Implementation
Interactive Environmental Stimulation Systems	• Relatively new and untested in value assessment and policy formulation contexts, but research and trial applications are increasing	• How might the development of emerging interactive computer simulation and game methods be shaped to effectively contribute to Agency policy and decision making needs?	• Can engage and communicate with public audiences about what outcomes they prefer and policies required to achieve those outcomes • Respondents can learn through experience about how the ecosystems/services of interest respond to various policies or policy aspects and can progressively modify and test their expressed policy preferences • Provides opportunities to introduce and experimentally control policy relevant (and confounding) variables in evaluated policies	Technological demands can be high, but off-the-shelf environmental simulation and VR systems are increasingly available and affordable. As simulations approach VR standards, demand for detailed specification of environmental conditions and processes increases, possibly exceeding current bio-ecological knowledge in some cases

EPA has a number of laws, regulations and guides to assure that "the Agency considers public concerns, values, and preferences when making decisions" (EPA 2003, p. 1). The social-psychological methods described in this section are consistent with that goal and can also contribute to systematic quantitative assessments of the values of protecting ecosystems and ecosystem services. Survey methods are the most frequently used means for identifying public values and concerns ("what people care about") and for measuring the degree of public preference, acceptance and support for alternative environmental outcomes and associated social consequences (see Appendix C for a detailed discussion of survey methodology). Surveys are also used to predict how various segments of the public are likely to respond to projected changes in environmental conditions and to alternative management means for affecting those changes. Additional methods, such as individual narrative interviews, can support agency decision making by elaborating and enriching understanding of the different perspectives of various stakeholders and concerned citizens.

EPA's charge to protect ecosystems and ecosystems services is consistent with widely shared public concerns and values (e.g., Dunlap, et al. 2000). However, the formulation and implementation of specific ecological protection policies will often involve scientific and technical considerations that the lay public cannot be expected to fully understand and appreciate. Surveys and the other methods described in this section have proven effective in uncovering assumptions, knowledge, beliefs, and feelings that underlie expressed preferences and concerns so that decision makers can better understand and address conflicts between various publics, and between public preferences and ecological science. Moreover, there are a number of methods for introducing relevant information into or prior to a systematic survey that can help to assure that respondents have an adequate and appropriate foundation for expressing requested preferences and other judgments (see Appendix C).

While public opinion is sometimes directly used to make policy decisions (see Chapter 4 and Appendix B sections on Referenda and Initiatives and on Citizen Valuation Juries), social-psychological assessment methods more typically are intended for decision support. These methods address the psychological foundations for subsequent actions toward the measured alternatives, including political support, direct, indirect, or hypothetical monetary payments, and acceptance of and compliance with relevant regulatory mandates. Typically, separate measures are reported for several different value dimensions (e.g., aesthetic, ethical, personal-utilitarian, civic) across designated sets of policy alternatives or for specific features of those alternatives. Consistent with a multi-attribute value framework, there has been little emphasis on mapping all expressed concerns and preferences onto a single, universal value scale (as required for economic cost-benefit analysis methods, for example). Differences between different value dimensions or between various subsets of the public are not typically resolved through aggregation algorithms or other calculation devices within the assessment process. Rather, resolution of such differences is more typically deferred to later stages of the decision making process, where information integration, deliberation, and negotiation is left to authorized decision makers or is addressed in more or less formal interactions between stakeholders/publics and decision makers.

The social-psychological approach to assessing the value of ecosystems and ecosystem services enlists both quantitative and qualitative methods. Formal surveys and questionnaires typically rely on standardized descriptions of alternative objects/states (e.g., alternative environmental conditions, management policies, socially-relevant outcomes), with respondents recording explicit choices, rankings, or ratings that are analyzed to develop appropriate quantitative metrics (e.g., preference, importance, or acceptance indices). Individual narrative interview methods typically employ less restrictive representations of options, are frequently directed at specific local cases that are familiar to respondents, and collect open narrative responses that are subjected to more or less rigorous qualitative analyses. These methods have often been used to support the design and pre-testing of subsequent quantitative surveys, but they are increasingly being offered as stand-alone assessments. In addition to the more established methods, some emerging methods base assessments on more direct observations of behaviors in the environments at issue. Behavioral observation and behavior trace methods have been developed and evaluated, especially in the context of the assessment of recreation and tourism values (e.g., Daniel & Gimblett 2000, Gimblett, et al. 2001). Computer simulation ("virtual reality") and interactive game methods are also being developed, but have mostly been applied in research settings (Bishop, et al. 2001a, 2001b). These emerging methods may not yet be sufficiently proven for application in EPA policy-making contexts, but they do show considerable promise for applications in circumstances where the validity of verbal expressions of preferences and concerns in response to described hypothetical conditions may be suspect. They will only be briefly described in this section and are offered primarily as potential targets for future research and development.

Brief description of the Methods

Surveys Including Attitude Survey Questions

Attitude surveys encompass a broad range of methods for systematically asking people questions and recording and analyzing their answers (e.g., Dillman 1991, Krosnick 1999, Schaeffer and Presser 2003, Appendix C to this report). Questions may assess knowledge, beliefs, desires, or behavioral intentions about a virtually unlimited range of objects, processes, or states of the person, society, or the world. Multiple questions/issues are typically presented and responses are reported as choices (among two or more options), rankings, or ratings. The most popular survey formats have involved face-to-face, mail, or telephone contacts with individually sampled respondents. Web/Internet media are increasingly being used and are rapidly becoming more sophisticated, but representative sampling issues require special attention. Open-ended response formats are less often used, and may pose special problems for quantitative analysis.

Social-psychological surveys have been extensively used to assess preferences, attitudes, importance, and acceptability of presented policies, actions, outcomes or the expected personal or social consequences thereof (see the lists in Appendix C). An example is the extensive national survey conducted to support the USDA Forest Service GIPRA process (Sheilds, et al. 2002), which is discussed in Text Box 11. Multiple value dimensions (e.g., utilitarian, aesthetic, ethical) may be addressed within and between different surveys, and surveys may specify individual/personal, household/family or social/civic constituencies. The indices produced by application of appropriate quantitative analyses of recorded responses usually claim to be only ordinal (ranks) or roughly interval scale, relative measures of differences in assessed values among offered alternatives. Moreover, expressed preferences or other value judgments are assumed to be at least in part created in the context of the survey (Schaeffer and Presser 2003). Thus, generalization of obtained values measures (e.g., “values transfer”) beyond the objects specifically assessed within a given survey must be approached with caution.

Text Box 11: National Telephone Survey

A nationwide telephone survey was conducted to provide support to the USDA Forest Service Strategic Plan for 2000 required by the Government Performance and Results Act. The survey randomly sampled over 7000 U.S. citizens to determine held *values* relevant to public lands, preferred *objectives* for management of public forests and grasslands, *beliefs* about what the role of the Forest Service should be with regard to these objectives, and public *attitudes* about the job the Forest Service is doing toward fulfilling the desired objectives. The items for this “VOBA” survey were developed and pre-tested through more than 80 focus groups conducted across the country. Individual survey respondents were presented with only a subset of the 115 items/questions developed for the survey. Each respondent assigned ratings to the items presented on five-point scales, with *objective* statements (30 items) rated on an *importance* scale, beliefs (30 items) and values (25 items) rated on a *disagree-agree* scale and attitudes (25 items) rated on an *unfavorable-favorable* scale.

Some example items from the survey and their mean ratings over the full national sample are presented in Table 7 below. Items are selected for potential relevance to C-VPESS interests and they are grouped to display the observed discrimination in responses. Many of the same items were rephrased and repeated in several of the values, objectives, beliefs, and attitudes categories (across but not within respondents). Only the values and objectives category formats and mean ratings (agreement and importance, respectively) are presented here, as the beliefs and attitudes items were specific to the Forest Service. Some items may be reversed from the original presentation so that higher means always indicate higher agreement/importance ratings.

Table 7: Example Items from Survey Supporting USDA Forest Service Strategic Plan for 2000 required by the Government Performance and Results Act

	Values	**Objectives**
Item Examples	***Mean Agreement***	***Mean Importance***
Wildlife, plants, and humans have equal rights to live and grow.	4.28	
Future generations should be as important as the current one in the decisions about public lands.	4.52	
We should actively harvest more trees to meet the needs of a much larger human population.	2.88	
The decision to develop resources should be made mostly on economic grounds.	2.92	
Protecting ecosystems and wildlife habitat		4.58
Conserving and protecting forests and grasslands that are the source of our water resources, such as streams, lakes, and watershed areas		4.73
Expanding access for motorized off-highway vehicles on forests and grasslands (for example, snowmobiling or 4-wheel driving)		2.41
Designating more wilderness areas on public land that stops access for development and motorized uses		3.84
Developing new paved roads on forests and grasslands for access for cars and recreational vehicles		2.62
I am glad there are National Forests even if I never see them	4.66	
I would be willing to pay five dollars more each time I use public lands for recreational purposes (for example, hiking, camping, hunting).	3.49	

Individual item standard deviations ranged from 0.75 to 1.50. Sample sizes were not reported per item, but would be large (at least hundreds of respondents each) so that standard errors of the reported means would be very small.

Respondents also answered a number of demographic questions and provided information about their use of public forests and their knowledge of and association with the forest service. These items were used to identify several sub-groups that produced different patterns of response to the items in the survey. For example, the authors report, “Metropolitan residents in both the East and West see the objective of protecting ecosystems and wildlife habitat as more important than do those in non-metropolitan areas. Within non-metropolitan areas, those in the East are more in favor of such programs than are westerners." p 11.

Similar surveys could obviously be designed to address items relevant to EPA efforts to protect ecosystems and services. The example Forest Service survey was targeted on broad national strategic goals and issues, but surveys may be even more effective in assessing beliefs, preferences, and attitudes about more specific management alternatives and outcomes. In some cases, where the relevant dimensions of outcomes may be subtle and difficult to describe in words, visualizations and other perceptual representations may be more effective in eliciting public preferences (as illustrated in Text Box 12, Perceptual Surveys, later in this section).

Shields, D. J., Martin, I.M., Martin, W.E., Haefele, M.A. (2002) Survey results of the American public's values, objectives, beliefs, and attitudes regarding forests and grasslands: A technical document supporting the 2000 USDA Forest Service RPA Assessment. General Technical Report, RMRS-GTR-95. Fort Collins, CO: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station. 111 p

Surveys have become ubiquitous in modern society, with uses ranging from assessments of diners' satisfaction with the service at a restaurant to citizens' support for major national policies (Dillman 2002). Surveys are now frequently directed by computer programs that can select and order questions individually for each respondent, sometimes based on responses to prior questions. Increasingly, surveys are fully implemented by computer, allowing the respondent to control (with more or less restriction) the pace of questions and to record their responses directly into a computer database by key presses, clicks, or voice commands (Tourangeau 2004). Internet-based methods, whose use is increasing, offer extended possibilities for contacting respondents, presenting questions, and recording responses. However, Web surveys may raise representative-sampling and other issues that require special attention (e.g., Couper 2001, Tourangeau 2004, and Appendix C to this report).

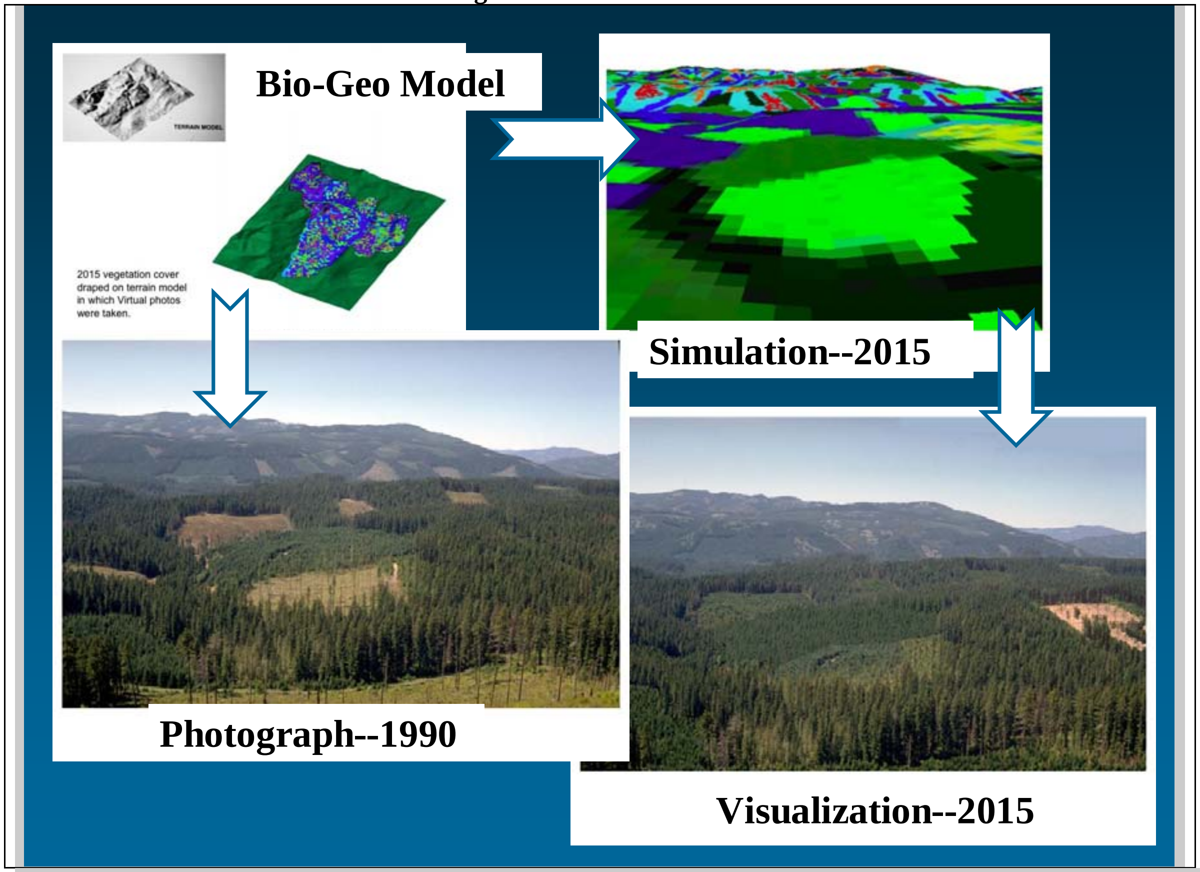
Variations on survey research methods that may be especially appropriate for assessments of ecosystems and services include perceptual and conjoint representations of assessment targets. In perceptual surveys, assessment targets (e.g., existing environmental conditions and/or projected policy outcomes) are represented by photographs, videos, computer visualizations, audio recordings, or even chemical samples representing different smells. As for verbal surveys, responses are typically choices, rankings, or ratings of the offered alternatives. Perceptual surveys may be seen as extensions of traditional psychophysical research methods that have long been applied to assess qualities and preferences for foods and other products that are difficult or impossible to describe effectively with words (Daniel 1990). Relevant examples include assessments of the visual aesthetic effects of alternative forest management policies in the northwestern United States (Ribe, et al. 2002, Ribe 2006), of in-stream flow levels on scenic and recreational values (e.g., Heatherington, et al. 1993), of visibility-reducing air pollution on visitor experience in National Parks (e.g., Malm, et al.1981), and assessment of the annoyance produced by aircraft over-flight noise in the Grand Canyon (Mace, et al.1999). An illustration of perceptual survey methods based on Ribe, et al. (2002) is presented in Text Box 12.

Text Box 12: Perceptual Surveys

A study by Ribe et al. (2002) provides a good illustration of a perceptual survey employing computer visualization technology. The focus of this study was on the aesthetic effects of the shift to more ecologically motivated forest management in the Northwest United States. The survey sought to determine how the Northwest Forest Plan (NFP, arising out of the spotted owl controversy) would affect the perceived scenic beauty of affected landscapes in public forests. Another study objective was to investigate the possible contributions of landscape design principles contained in the U.S. Forest Service Scenery Management System for shaping NFP harvest prescriptions to provide better aesthetic results. The description here will focus only on the visualization and perceptual survey components, and how these methods were used to attain quantitative measures of the aesthetic affects of shifting the emphasis in forest management from economic to ecological goals.

The basic strategy of this assessment was to first select a representative set of forest areas where the NFP prescribed changes to forest management. From within these areas, 15 forest landscape scenes (“vistas”) were selected to represent a range of forest conditions consistent with pre-NFP management practices. Geographic information system technology was used to delineate and to create 3-D terrain models and detailed maps of the existing vegetation cover in the visible area of each scene (from a designated viewpoint). GIS perspective view techniques were used to create a “virtual photograph” of the scene so that color-coded vegetation features (e.g., existing forest, clearcuts of various sizes, and stages of re-growth) could be accurately located within the view. An actual photograph was also taken from the viewpoint and was compared with the virtual view to assure accuracy. Forest harvest and growth models and expert judgments of trained foresters working in the study area were then combined to develop detailed forest management plans for the area within each selected view, following NFP prescriptions, and to project changes in forest vegetation (removal and re-growth) over 20 years following the implementation of the NFP. A virtual photograph was again created to represent the projected changes in the visible landscape. Finally, digital montage methods were used to map appropriate video textures (e.g., five-year re-grown clearcut, undisturbed mature forest, etc.) onto the scene to create a biologically accurate but photographically realistic visualization of the future forest conditions. Figure 9 below illustrates some of the key steps in this visualization process.

Figure 9: Visualization of Forest Conditions and Actual Photos from Ribe et al. (2002)



The visualizations of future forest conditions and the actual photos for each of 15 selected study scenes were rendered to color slides. Study slides were intermixed with 90 additional scenes representing a wide range of forest conditions in the area and presented in a perceptual survey. The 608 respondents were sampled (not randomly) from 31 stakeholder groups in the Cascade region affected by the NFP. Respondents recorded their judgments of the scenic beauty of each scene independently on an 11-point scale ranging from "extremely ugly" (-5) to "extremely beautiful" (+5). Because all scenes were rated by the same groups of respondents in the same context, simple mean ratings were judged an appropriate index of the relative scenic beauty of the scenes. Because the study scenes were specifically selected to represent particular forest management-by-view parameters (not random samples) comparisons were restricted to the pre-NFP versus post-NFP pairs for the same base scenes.

The mean differences between pre- and post-NFP pairs for the 15 forest scenes ranged from -3.05 (favoring the pre-NFP prescription) for a close-up view of a recent harvest to +2.39 favoring the NFP prescription in a larger scale vista with numerous harvest sites in the visible area. For six of the eight scenes selected to have large to medium-sized view areas, scenic beauty ratings were significantly higher for the post-NFP scene. Regression analyses determined that the key, objectively measured variables affecting scenic beauty differences between pairs of scenes were the percent of the visible area covered by fresh, high-contrast clearcuts in the middle distance and in the far distance of the view (both with negative coefficients).

The NFP management prescriptions were primarily driven by ecological considerations, but in the most conspicuous cases (the larger views) these ecological prescriptions also produced significant improvements in scenic beauty as perceived by the most likely visitors to and viewers of those and similar sites. While this study did not directly address the question, a similar perceptual survey, along with standard forest vegetation cover and harvest data could be used to measure and map trade-offs among economic, ecological and aesthetic values for forest management alternatives (including NFP and other approaches) based on a systematic sample of viewpoints/scenes across a landscape of interest. Such trade-off assessments and regression-based models can be used by forest planners to develop detailed harvest prescriptions and schedules for specific sites allowing NFP ecological guidelines to be met while maintaining or enhancing economic and aesthetic goals for the public landscape. In EPA contexts, similar perceptual survey methods might be applied to assess aesthetic and other visual impacts at contaminated sites, and to assess the relative merits of restoration and reuse options.

Ribe, R.G., Armstrong, E.T.,& Gobster, P.H. (2002) Scenic vistas and the changing policy landscape: visualizing and testing the role of visual resources in ecosystem management. Landscape Journal, 21: 42–66.

Surveys most often present the individual attributes of assessment targets separately. For example, a survey to assess the effects of a proposed environmental policy might present separate questions to determine respondent’s judgments about effects on air quality, water quality, and local employment. Conjoint survey questions (e.g., Adamowicz, et al.1998, Boxall, et al. 1996) instead present options as multidimensional composites or scenarios presenting integrated/conjoined combinations of different attributes (e.g., different levels of air quality, water quality, and local employment). Combinations generally reflect actual or projected variations in the attributes (e.g., different levels of air and water quality and local employment opportunities). In the more sophisticated conjoint surveys, the particular combinations of attributes represented are specified by an experimental design that allows estimates of the separate and interacting effects of component attributes (Louviere 1988). Multiple regression (or similar) analyses are used to estimate the relative contributions of individual components (attributes) to the expressed preferences (or other judgments) for the conjoint alternatives.

Conjoint survey questions can provide relatively direct estimates of the value trade-offs people make when choosing among outcomes composed of multiple attributes that naturally covary and whose values potentially conflict and compete. When at least one of the attributes that forms the conjoint alternatives is (or can be) valued in monetary terms, the regression equation based on expressed preferences among the conjoint alternatives can be translated so that coefficients for all attributes are expressed as monetary values (see the following Appendix B Section on Economic Methods). An illustration of conjoint survey methods is presented in Text Box 13: Conjoint Surveys.

Text Box 13: Conjoint Surveys

Conjoint methods may be especially well-suited for gauging public preferences across sets of complex multi-dimensional alternatives, such as alternative EPA regulations or management options for ecosystems/services protection. Respondents choose among (or rank or rate) multi-dimensional "conjoint" alternatives that present specific packages of desired and less-desired attributes. Analyses of the patterns of preferences values (e.g., probability or percent choice or mean rating) among the conjoint alternatives can be used to estimate the contribution (e.g., regression coefficients) of each of the separate attributes.

Chattopadhyay, Braden, and Patunru (2005) used a conjoint survey method to assess the effects on residents' home preferences of various cleanup options for the Waukegan Harbor Superfund site in Wisconsin. This study also employed and compared results of a hedonic pricing method (see the following Appendix B Section, Economic Methods, for a description), but the monetary estimates of willingness to pay for the cleanup options evaluated were based on stated preferences in a conjoint survey, which is the subject of this illustration. Adjustments for differences in respondents' incomes, annual costs for current housing and for the hypothetical housing options offered (based on real estate market data) and a number of composite and interaction terms involving economic variables were introduced to conform to assumptions of relevant economic theory and practices. However, the basic data are simply respondent's choices (expressed preferences) among alternative hypothetical conjunctions of housing and environmental-condition attributes, so the core features of the study nicely illustrate an application of a conjoint choice survey that could just as easily, or more easily, be used to obtain an interval scale measure of the effects of cleanup options on housing preferences.

Housing market data for 47,100 transactions (1996-2001) for Waukegan and 12 similar nearby cities along with focus group sessions with homeowners were used to determine the six housing/environmental attributes that were conjoined to describe the hypothetical housing options and to describe the respondents' own current home/environment. Housing attributes were *lot size, house size*, and *house price*. Environmental attributes were *elementary school class size, public areas near the harbor,* and *extent of changes proposed in the harbor-area pollution*. Each of the 6 housing/environmental attributes was represented by four levels, so that in principle there could be $4^6 = 4096$ distinct conjoint options. A fractional factorial experimental design (with a "fold-over" to allow estimation of two-way interaction terms) was used to determine the 64 x 2 = 128 conjoint options that were actually assessed in the survey. The details and rationale for this complex design is beyond the scope of this illustration, but the key point is that the selected alternatives allow for statistical estimates of the separate effects of each of the housing/environmental attributes on overall preferences (or overall w-t-p estimates in the present study) across all of the options. All 128 selected options were assessed in the study, but each of the 954 respondents (from 2339 surveys mailed to the 13 targeted communities) only responded to a random subset of 16 options.

In a typical conjoint choice study, respondents would see pairs of the conjoint house/environment options and be required to choose between them. Chattopadhyay et al. instead chose to reduce the length and complexity of the task by comparing each hypothetical alternative to a standard—the respondent's current home/environmental conditions. The difference on each of the six attributes between the current home and each hypothetical option was expressed as a percentage. For example, the *house size* attribute could be 15% smaller, unchanged, 15% larger or 25% larger than the respondent's current home, and the *harbor area environmental condition* could be additional pollution, no change (from current conditions), partial cleanup, or full cleanup. A facsimile of an illustrative choice question in the survey is presented in Table 8 below.

Table 8: Facsimile of Illustrative Choice Questions from Chattopadhyay et al. (2005)

Home #1: Imagine your home modified to fit this description						
	Lot size	House size	School class size	Public/natural areas in harbor area	Harbor area environmental condition	House price
Compared to your current home:	Smaller by 15%	Smaller by 15%	Smaller by 2 students	Smaller by 20%	Additional pollution	Less expensive by 10%

The core data for the conjoint choice study is the observed probability of choice for each of the 128 hypothetical house/environment options over the current home. These probabilities can be used to derive more sophisticated quantitative value scales, but basically the worst options (least preferred) would be chosen less often and the best would be chosen more often. In conjoint studies, choices for the hypothetical multi-attribute options is usually of less interest than are the estimates of the contributions of the respective house/environment attributes to those expressed preferences. There are numerous methods for attaining these estimates, most based on multiple regression analyses of one kind or another. In the Chattopadhyay, et al. study, a multinomial/conditional logit model was used. The details of this analysis are not relevant to this illustration, but the basic outcome of such a conjoint choice study can adequately be portrayed as a regression equation of the following form:

$$P_i = w_1(A1_i) + w_2(A2_i) + w_3(A3_i) + w_4(A4_i) + w_5(A5_i) + w_6(A6_i)$$

where

Pi is probability of choice (versus current home) of conjoint alternative i
w_1 is the regression coefficient for house/environment attribute 1 (e.g., lot size)
$A1_i$ is the level for attribute 1 for alternative i (e.g., 15% smaller)
and so on for each of the other 5 house/environment attributes.

Chattopadhyay, et al. scaled the weights in a much more complex equation (including derived economic terms and interactions) to attain monetary benefit estimates on the basis of which they offered conclusions such as:

> …the significant coefficient for the interaction variable *full*highinc* indicates that high-income residents prefer full cleanup more than other categories, while the insignificant coefficients on *addpol*highinc* and *part*highinc* indicate that high-income residents are no different from others (income levels) with respect to their dislike for additional pollution and their preference for partial cleanup. p 367

The authors went on to estimate aggregate monetary benefits of partial and full clean up of the Waukegan Harbor Superfund site ($249 million and $535 million, respectively). The validity of these monetary estimates, of course, depends upon a complex set of assumptions required by general economic theory and by specific features of the present study. These assumptions would not be required for the more basic analysis of expressed preferences suggested in this illustration. The attribute weights (regression coefficients) in the suggested simple preference equation could, however, safely be interpreted as relative (interval scale) measures of the trade-offs the sampled respondents made between the offered changes in harbor environment cleanup (from additional pollution to full cleanup) and the other house/environmental attributes represented by the options in the study.

Once determined, the preference-based regression equation could also be used to estimate preferences for new policy alternatives based on their respective projected changes in environmental conditions, so long as those options fit sufficiently within the range of the attributes and levels assessed and the constraints imposed by the context of the survey in which the house/environmental condition options were offered and judged. Optimization or less formal heuristics might be applied to create additional policy options for consideration and/or for direct evaluation in subsequent conjoint surveys.

Chattopadhyay, S., Braden, J. B., & Patunru A. (2005) Benefits of hazardous waste cleanup: new evidence from survey- and market-based property value approaches. *Contemporary Economic Policy*, 23, 3: 357-375.

Individual Narratives

Researchers using individual narrative methods contact individual respondents, who participate alone, without interaction or discussion with experts, facilitators, or other respondents. Individuals nominally representing possible stakeholder perspectives are contacted and asked to comment on relatively broadly defined topics with relatively little direction from the interviewer/assessor (e.g., Brandenburg & Carroll 1995). Respondents are not typically selected by a random, probability sampling process. Instead, particular individuals are specifically targeted because of their known or assumed nominal group membership or personal relationship to the problem/policy/outcome at issue. The sample may be extended by having prior respondents refer others, as in the “snowball” technique. The number of individuals to be included is quite variable, and in a relatively few cases has been determined by some formal process based on a rolling analysis of collected narratives (e.g., using a criterion of diminishing new perspectives/positions being discovered). Collected narratives are subjected to more or less rigorous qualitative analyses, (essentially similar to the analysis of focus group responses, see Appendix B section on Focus Groups) to explore and articulate the breadth and depth of expressed understandings and concerns relevant to the assessment target. Included in this category are various ethnographic methods and mental modeling procedures.

A mental models approach can inform debate about the best ways to elicit values, and how people use and understand different qualitative and quantitative expressions of value, response scales and response modes. People use their prior (pre-existing) mental models to interpret survey questions and other preference-elicitation probes. People make inferences not only about texts in surveys, but also about values and risks in the actual environment and hence their mental models and mental representations of causal processes underlie all decisions. Mental models methods aim at eliciting people’s understanding of causal processes associated with the events, processes, and actions that are projected to result from specific decisions. As applied to understanding hazardous processes, the method has been used to characterize people's understanding of how risks arise and can be mitigated, and entails a mixture of decision modeling, semi-structured interviews (ethnographic in nature), survey research, comparisons between these, and both qualitative and quantitative modeling of the results. To date, this research has focused more on enabling and informing risk reduction, rather than motivating or understanding preferences and trade-offs per se.

Mental models research would be an appropriate precursor (i.e., formative analysis) to any formal survey or preference elicitation method, to improve the validity and reliability of the method. Values are typically expressed qualitatively, sometimes in ordinal terms (e.g., lexicographic scales or comparative statements) and sometimes using quantitative scales. The approach is designed to explore the conceptual landscape for risks and benefits, including underlying causal beliefs, specific terminology/wording, and the scope and focus of mental models in the decision domain of interest. The approach is principally qualitative, designed to elicit how an individual conceptualizes and categorizes a process, such as protecting an ecological service, and how that individual would make inferences about and decisions to influence that process.

Emerging Methods

The assessment methods described in this section are relatively new and untested. They are characterized by more direct observation of responses to policies, outcomes and consequences in situ, avoiding problems of relying on hypothetical responses to described conditions. In that context, these methods parallel the revealed preference methods used in economic value assessments (Appendix B, Economic Methods). Observed environmental behavior is often not consistent with what people say they would do in the specified circumstances (Cole and Daniel 2004) and people are often incorrect at identifying, or are unaware of the environmental factors that affect their behavior (e.g., Nesbitt and Wilson 1977, Wilson 2002). In the context of ecosystems and services, *behavioral observation* methods monitor the activities of people in a particular environmental context and observe changes in behavior as relevant conditions change over time within a site or over sites with differing characteristics. *Behavior trace* methods are based on indirect evidence of people's behavior in specific environmental contexts. For example, the number of visitors to recreation sites might be estimated by counting the number of autos parked at access points, by the number of passers-by recorded by automated trail counters, by the number of fire rings in dispersed camping areas or by the amount of trampling and disturbance of vegetation along trails and at destination points. Direct observations or traces of visitors' activities can be correlated geographically with relevant environmental/ecological conditions or monitored over time as changes in conditions occur at the same sites, revealing the effects of these changes on environmental preferences and reactions (e.g., Gimblett, et al. 2001, Wang, et al. 2001, Zacharias 2006).

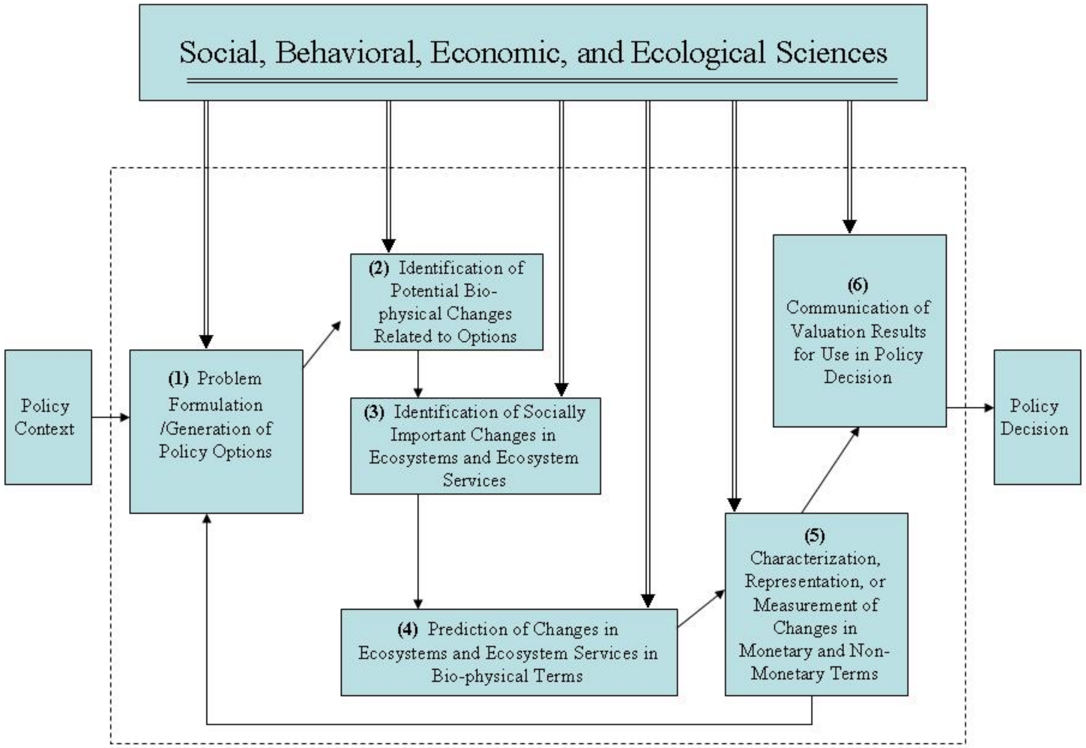
These methods do not seem to have been applied in the context of assessments of the effects of changes in ecosystems and services. However, changes in human use of rivers, lakes, and estuaries are often important indicators of the need for and the value of EPA interventions to protect water quality and associated aquatic systems, and the travel cost methods employed by economists in these contexts is fundamentally similar. Behavioral observation and trace methods might be effectively employed to attain quantitative measures of human use levels that could be used in conjunction with economic measures or as separate measures to be correlated with changes in ecological conditions. Numbers and durations of users, their geographic distribution and the activities that they engage in might be correlated with relevant bio-physical measures of ecological conditions to develop useful assessments of the effects of ecological degradation or the effectiveness of ecological protection efforts.

Interactive environmental simulation systems provide means to overcome some of the limitations and difficulties of conducting direct behavioral observations or interpreting behavior traces. Direct observation methods are necessarily limited to existing conditions and are potentially confounded by uncontrolled or unrecognized irrelevant variables. Most policy decisions hinge on people's responses to specific changes to not-yet-existing, projected environmental conditions. Rapidly advancing computer technology has enabled effective and economical simulation of complex dynamic environments at high levels of realism (e.g., Bishop and Rohrmann 2003, Bishop, et al. 2001a, 2001b). The emphasis has been on visual presentations, but the technology can readily include auditory features and in some systems tactile, proprioceptive, olfactory, and other senses can also be effectively simulated to achieve very compelling, immersive environmental experiences. Moreover, expanding response options, ranging from the computer mouse to video-game controllers to gloves to full-body movement enable increasingly natural interactions with simulated environments. In the context of assessing the effects of changes in ecosystems and services, interactive computer simulation systems offer the opportunity to conduct virtual in situ experiments to determine how persons respond to specific investigator-controlled changes in environmental conditions. Thus the effects of manipulated conditions on environmental preferences and other reactions can be revealed in a context closely approximating “real world” circumstances.

Interactive computer simulation systems may be viewed as games, in which human respondents attempt to (virtually) navigate through and perhaps alter virtual environments to accomplish desired goals. There may be no particular outcome that can be defined as “winning” such a game, but the behavior of the player and the outcome on which s/he settles can reveal the values that motivate and guide the player’s responses. *Interactive games* can be informative in this regard, even if they are played in substantially less than virtual environments. Indeed, more limited and/or more abstract games may have important advantages in some circumstances. For example, it may not be possible to project the explicit and detailed outcomes of a proposed policy that are required for a realistic environmental simulation, and the specific implications of particular responses to changing environmental conditions may not be known. In many situations only changes in some particular ecological component may be known and relevant (e.g., a reduction in a particular contaminant or an increase in survival rates of a particular wildlife or plant species). Still, a game-like context may be an effective and engaging way to communicate with public audiences about what outcomes they would prefer, and what policies are required to achieve those outcomes. A major advantage of games over surveys, for example, is the opportunity for respondents to learn through experience about how the ecosystem of interest responds to various policies or policy aspects and to progressively modify their expressed policy preferences to converge on some acceptable balance among desired and undesired outcomes.

Relation of Methods to the C-VPESS Expanded and Integrated Assessment Framework

Survey and individual narrative methods have useful roles to play throughout the valuation process envisioned by C-VPESS. For example, representative surveys and selected individual interviews could contribute to initial problem formulation by identifying ecological services and impacts that most concern citizens and/or identified stakeholders, as well as by uncovering assumptions, beliefs, and values that underlie that concern. Similarities and differences in assessed concerns, attitudes, and beliefs toward proposed policies among different segments of the public can also be identified and articulated. Once relevant ecological endpoints have been identified, surveys could be very useful for determining the personal and social consequences of those outcomes, and for exploring public understanding of the links between chains of ecological effects and the policy options under consideration (Box 3 in Figure 2, reproduced below). Given a set of potential policy options, with their respective ecological endpoints (from Box 4 in Figure 2, surveys could be used to assess relative public preferences [and/or other judgments, such as importance or acceptability]) for those options (Box 4 in Figure 2). Quantitative indices of public/stakeholder preferences (or judgments of importance or acceptability) from surveys could be combined with bio-ecological and economic/monetary measures of the value of the same alternatives to provide cross validation for all measures and to strengthen the foundation for policy decisions. Surveys may be especially useful when the values at issue are difficult to express or to conceive in monetary terms or where monetary expressions/valuations are viewed as ethically inappropriate. In those cases survey questions could provide reliable quantitative measures of public preferences among the policy alternatives or ecological endpoints that are under consideration, improving the basis for Agency decision making.



Attitude survey questions could make an additional contribution after Box 5 in the C-VPESS model. The values of ecosystems/services coming out of Box 5 must inevitably be represented by multiple economic/monetary, bio-ecological and social-psychological indicators. EPA administrators can be left with the difficult task of integrating these diverse and potentially conflicting measures, along with legal, budgetary and other constraints to make and rationalize policy decisions. Properly structured attitude survey questions, perhaps including material to inform respondents about relevant ecological and social effects and other considerations affecting the policy/decision at issue, could effectively involve citizen stakeholders in this value integration and trade-off process, providing an additional relevant input to the policy decision, and adding to the political validity and social acceptability of the final action.

Individual narrative methods, such as the mental models method, would be most appropriate and most useful at the earliest and latest stages of the decision making process. While individual interview methods do not generally provide quantitative assessments for alternative policies or outcomes, they can make important contributions to improving the design, development and pre-testing of more formal surveys that can provide reliable and valid quantitative assessments of public concerns and values. Mental models methods are appropriate for use in all identification stages (ecological modeling; what matters; ecological impacts that matter), with the possible exception of identifying EPA's objective(s). Genuine probing interactions with individuals or groups representing key stakeholders and including divergent views and concerns should be a central part of problem definition and identification of significant ecological and associated social effects components of the process. Such interactions with key stakeholders and with citizens could also inform the values integration and negotiation in the final decision process and guide and pre-test the communication of that decision.

Status of Methods

Survey questions measuring social-psychological constructs are the oldest and most frequently used methods for determining public beliefs, concerns, and preferences. Survey questions have been used and continue to be used effectively by all levels of government to measure citizen desires, concerns, and preferences. Economists have lately adapted survey methods to measure stated willingness to pay for non-market goods and services, and surveys are often relied upon to collect the data needed to exercise other economic valuation efforts, such as travel cost and hedonic pricing methods (see Appendix B, Economic Methods). Environmental management agencies have made use of surveys, either directly or indirectly, in setting policy and in making and monitoring the effects of management decisions (e.g., Shields, et al. 2002, illustrated in Text Box 11 and the many surveys listed in Appendix C to this report).

It is not clear the extent to which individual narrative interviews are systematically used in EPA policy making, nor do the OMB and other guidelines clearly specify the criteria for using these methods. While no specific evidence has been found either way, it seems reasonable to assume that individual narrative interviews have not been important components of formal EPA decision making processes. Certainly the qualitative nature of the information provided by both focus groups and individual interviews, and the general disinterest in representative sampling makes them poor candidates for formal policy evaluation exercises, but that does not preclude their having a role in earlier stages of the decision making process as envisioned by the C-VPESS. Mental models research could in theory be applied as a first step to investigate either "means" or "ends" values. This method would be an appropriate precursor (i.e., formative analysis) to any formal survey or preference elicitation method, to improve the validity and reliability of the method.

Limitations

The largest barriers to greater use of survey methods in ecosystems and services valuation and decision making by the EPA are institutional. First, while the EPA seems to have embraced economic surveys (e.g., CVM, or at least "transfers" from prior CVM surveys) as a valuation method, there is a noticeable reluctance to use the larger class of systematic surveys using attitude, preference, and intention questions, relative to the practices of other federal agencies with similar environmental protection mandates and valuation needs. This predisposition may in part be due to specific legal requirements for formal monetary benefit-cost analyses (which also apply to other agencies), but none of the currently applicable laws preclude using a fuller range of value measures and methods, and the most prominent laws and guides explicitly urge a broadly based evaluation effort not limited to monetary measures. Aside from this agency-level barrier, survey methods in general are discouraged by federal rules implementing the Paperwork Reduction Act. Over the past several decades it has been difficult for federal agencies to attain required clearances (e.g., from the OMB) for surveying the public in a manner and in a time frame that effectively addresses policy evaluation needs. This institutional barrier is formidable, and the proliferation of surveys and pseudo-surveys for commercial and political purposes has dampened citizen's willingness to participate, but many significant surveys continue to be conducted by a number of government agencies (see Appendix C for further discussion).

When used, survey questions have proven effective for measuring public knowledge, beliefs, attitudes, and intentions. However, especially in the context of the complex processes of selecting alternative policies and actions to protect ecosystems and services it is important to recognize that the responding public may not a priori have a great deal of information or knowledge about the issues or policies about which they are asked. First, limitations on length and complexity of content (especially for telephone surveys) make it unlikely that the full complexity, including uncertainties of policies and their outcomes can be effectively communicated to respondents within the survey. Second, the general public is unlikely to have the breadth and depth of ecological knowledge that is often required to understand and evaluate a given policy, its bio-physical outcomes or the implications of outcomes for the respondent or for society more generally. Finally, even when the respondent fully understands these aspects of a proposed policy he/she may still be uncertain (or incorrect) about his/her projection of how well (or badly) the respondent will feel about the outcomes/implications when they are actually encountered (Wilson, et al. 1989). Some approaches to addressing these problems in surveys are presented and discussed in Appendix C to this report.

The technical issues that have been of the greatest concern to users of survey information, to quality control agents (e.g., OMB), and to survey researchers have been associated with the sampling of respondents. The results of a survey are typically intended to be generalized to some specified population (e.g., adult citizens of the United States) that includes many members that will not be included in the sample of individuals who actually respond to the survey (the respondents). The integrity of generalizations to the population of interest is assured if the respondents are a formal representative sample ("probability sample") of the population. However, recent research shows that departures from strict sampling rules, such as the loss of intended participants by non-response or failed contacts, may not have as strong an effect on the representativeness of survey outcomes as some have thought. More difficult and potentially more potent errors are in survey design, including the crafting, selection, and ordering of questions/items to be included in the survey, the form of the response options offered (e.g., the type of ratings scales), and uncontrolled events that occur during the time of survey implementation (see Krosnick 1999 and Appendix C to this report).

Social-psychological surveys do not meet the requirements of economic cost-benefit or cost-effectiveness analyses because they do not achieve a unidimensional, transituational measure of value. That is, the scale values computed for the ecosystem and service options addressed in a survey cannot be directly compared to (may not be commensurate with) values for extra-survey options, or to values and costs in other domains of the respondents' lives. It is arguable whether any value assessment method fully meets this requirement. However, given a feasible set of alternative regulatory/protection actions and outcomes in a specified environmental-social context, surveys of public attitudes, preferences, and intentions would be appropriate for quantitatively measuring public preferences among offered sets of policy/outcome options, for estimating the relative importance to people of the multiple attributes of those policies and outcomes, and for gauging the acceptability of management means for achieving them. Properly designed conjoint methods may be especially well-suited for gauging public preferences across sets of complex multi-dimensional alternatives, such as will likely be involved in many EPA regulations and actions for ecosystems/services protection.

In practical use, the human resources required to implement surveys range from a sufficient cadre of technically competent survey designers and analysts to temporary hourly wage employees to perform the mailing, phoning, or interviewing tasks. Material needs may be very low ("paper and pencils") or quite high, as when sophisticated computer simulations/visualizations or interactive response formats are employed. Face-to-face surveys, where trained interviewers are required and participant-contact costs may be high, are generally the most expensive, but costs for mail, telephone and/or computer resources can also be significant in large surveys using those formats. All of these costs are usually quite low relative to the physical, biological and/or ecological science and field study required to create adequate projections and credible characterizations of value-relevant means and outcomes for a suitable range of alternative regulatory or protection actions. In many ways, the quality of evaluations of ecosystems and ecosystem services protections most depends upon the quality of the relevant projections and specifications of ecological endpoints and their social consequences. In some cases considerable resources may have to be devoted to translating targeted ecological outcomes into understandable representations of socially relevant effects. Once these essential factors have been accomplished, the cost of a systematic public value assessment survey can be comparatively quite small.

Individual interviews can have important and useful roles to play in Agency policy and decision making. However, their emphasis on qualitative analyses and their typical disregard for representative sampling can make them less useful for formal evaluations or comparisons of alternative policies and outcomes. These methods can be very useful and important for designing and pre-testing more structured surveys that do provide quantitative assessments of values for alternative policies and outcomes. Qualitative methods may also contribute to the design of more effective communications and rationalizations of Agency decisions to stakeholders and to the general public. In mental models research, values may be expressed qualitatively, sometimes in ordinal terms (e.g., lexicographic or comparative statements), and sometimes using quantitative scales. The approach is designed to explore the conceptual landscape for risks and benefits, including underlying causal beliefs, specific terminology/wording, and the scope and focus of mental models in the decision domain of interest. A mental models approach would best be used in conjunction with another method in order to obtain quantitative measures of values. The approach is qualitative, designed to elicit how an individual conceptualizes and categorizes a process, such as protecting an ecological service, and how that individual would make inferences about that process, as well as any decisions to influence it.

Treatment of Uncertainty

Survey methods specifically address the uncertainty introduced by sampling errors (e.g., representative sampling, non-response), specification errors (e.g., adequate descriptions or representations of alternatives, clear and understandable response system), and the effects of a variety of contextual and external factors that may affect (bias) participant responses. Methods for reducing and quantifying the magnitude of most of these sources of uncertainty and error in surveys are part of the well-documented technology and the accumulated lore of survey research (e.g., Dillman 1991, Krosnick 1999, Tourangeau 2004, and Appendix C to this report).

Accepted methods are available and are commonly used for calculating confidence intervals or complete probability distributions for individual survey responses over respondents (e.g., the importance ratings assigned to a particular item). The internal reliability and cohesiveness of survey responses can be calculated per individual respondent, but more often the focus is on the mean response of homogeneous groups of respondents. Multiple items are frequently combined, as by cluster or factor analysis, into latent variables (factors) implied by the inter-correlations among individual-item responses, and there are several conventional statistical indices of the internal consistency and coherence of those derived factors. More complete analyses calculate and quantitatively assess the internal consistency and distinctiveness of latent variables, based on the patterns of responses across the multiple respondents, as well as classifying sub-groups of respondents, based on patterns of individual's responses to the multiple items in the survey.

The detailed results of a survey of a representative sample of a population are unlikely to be fully appreciated by anyone without relevant training and experience. On the other hand, results can be, and routinely are, simplified for communication to lay audiences. Most people would find reports such as "alternative A was preferred over all others offered in the survey by 75% of respondents" to be clear and intuitively understandable. A table or graph showing mean preference ratings on a 10-point scale for all alternatives evaluated would be clear to many members of the public, as well as to experts from other scientific and managerial disciplines that are involved in EPA rule and decision making. Some of the uncertainty associated with these indices (e.g., sampling and measurement error) could be displayed by conventional confidence intervals or error bars. The potential effects of more complex sources of uncertainty might be revealed by bracketing mean estimates for each alternative assessed with 25th and 75th percentile estimates derived from sensitivity analyses exercised over the entire biological-social evaluation system. The most sophisticated communication devices might be based on interactive game systems, where the audience is allowed to alter input variables and assumptions about functional relations and stochastic events and observe and learn for themselves how these changes affect projected evaluation outcomes.

Research needs

Issues that should be addressed in future research relevant to social-psychological value assessment methods include:

- How can structured surveys of public/stakeholder attitudes, preferences, and intentions best be used in EPA policy and decision making, including how decision makers can and should use the relative quantitative (non-monetary) value indices provided?
- How can social-psychological value indices best be used to cross-validate estimates of monetary values (e.g., from CBA) and ecological indices (e.g., biodiversity, energy flow) and strengthen the basis for Agency decisions about alternative ecosystems/services policies?
- How, and when in the decision process, can social-psychological, economic, and bio-ecological evaluations of changes in ecosystems and ecosystems services most effectively be integrated to support Agency policy and decision making?
- What productive roles can individual interviews and other qualitative methods play in Agency policy and decision making?

- How might the development of emerging methods (behavior observation, behavior trace, interactive computer simulations, and games) be shaped to effectively contribute to Agency policy and decision making needs?

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ECONOMIC METHODS

Overview

Brief Description of Methods

The economic concept of value is based on two fundamental premises of neoclassical welfare economics: that the purpose of economic activity is to increase the well-being of the individuals in the society, and that individuals are the best judges of how well off they are in any given situation and of what changes would enhance that well-being.

The concept of value underlying economic valuation methods is based on substitutability, or, more specifically, on the trade-offs individuals are willing to make for ecological improvements or to avoid ecological degradation. These trade-offs provide an indication of changes in well-being that result from increases and decreases in goods and services people value. By itself, an ecological change that an individual values will increase that person's utility. The value or benefit of that change can be defined in two ways. The first is the amount of another good that the individual is willing to give up to enjoy that change (his "willingness to pay" or WTP). The second is the amount of compensation that a person would accept in lieu of receiving that change (his "willingness to accept" or WTA). These trade-offs are typically defined in terms of the amount of money an individual is willing to pay or willing to accept and hence benefits are measured in monetary terms. In this case, WTP is the amount of money that would make the individual indifferent between paying for and having the improvement and foregoing the improvement, while keeping the money to spend on other things. Likewise, WTA is the amount of money that would generate an increase in utility equivalent to that realized from the improvement in the environmental amenity.

However, it is important to note that the concept of benefit does not hinge on the use of monetary units. In principle, benefits could be defined in terms of changes in any other good or service that the individual would willingly agree to in exchange for the environmental change (e.g., food). The use of money as the basis for exchange is simply a convenience. In particular, use of a common money metric allows all benefit measures to be easily aggregated and compared with monetary measures of cost.

The benefits captured by the concepts of WTP or WTA can be derived not only from goods and services for which there are markets (e.g., forest products) but also from goods and services for which markets might not exist (such as clean air and clean water). In addition, they include values derived from use of the environment (e.g., hiking in the woods) as well as those derived from the "existence" of a valued species or condition. 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 deontological concept of intrinsic rights.

All economic measures of value based on willingness to pay are limited by the fact that the maximum amount a person could pay for anything is constrained by that person's ability to pay, which is indicated by the individual's wealth. Thus the value estimates derived from economic valuation methods are conditional on the existing distribution of income and prices. As a result, acceptance of these benefit estimates implies acceptance of the underlying distribution of wealth. One way to incorporate concern for equity in the distribution of well-being, with roots going back to Bergson (1938), is to weight the measures of economic value or welfare change for each individual by that person's relative degree of "deservingness"; that is, to attach a higher weight to benefits going to those judged to be more deserving because of some attribute such as their lower level of income. However, there is no clear way to determine the appropriate weights. In practice, analysts typically use the value measures derived from the mean individual in the sample that is providing data for the valuation model in use. If value or willingness to pay is an increasing function of income, the analyst is implicitly underestimating the values of the highest income individuals and overestimating the values of the lowest income individuals. The result, in a crude qualitative sense at least, is equivalent to assigning more weight to the values of low income than high income individuals.

The key input for all of the economic methods is data on the choices that people have made or indicate they would make about the things that contribute to their economic well-being. These choices are made in several contexts. The first is choices about quantities demanded and supplied in markets at alternative prices, e.g., the amount of commercial fish that are harvested and sold at various prices. These choices generate demand and supply functions that can be estimated with the information on the amounts purchased at different prices using statistical (i.e., econometric) methods. Changes in these demand and supply functions in response to changes in the levels of ecosystem services (e.g., a change in water quality) can be analyzed to obtain market-based estimates of the values of the changes in these services.

Second, choices can involve the selection of quantities of goods and services (or responses to changes in the availability of goods and services) that are not sold in markets, such as many ecosystem services. Non-market revealed preference methods can be used to obtain estimates of the values of changes in these goods and services. Third, hypothetical choices made in response to survey questions can be analyzed with one of the several stated preference methods for valuation to provide information on trade-offs people would be willing to make. The specific methods that employ these three different types of choice data to value ecological changes are discussed in more detail in the following sections.

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Market-Based Methods

Brief Description of Method

The market-based approaches to economic valuation are used to estimate the economic values of ecosystem services that are an input into the production of a good or service that can be bought and sold in a market at an observable price. For private goods and services purchased in competitive markets, the price of a good reflects the valuation of an extra unit of that good or service by the set of participants in that market. For small changes, market prices can be used as a measure of economic value of each unit of the goods involved. For larger changes, however, marginal willingness to pay (demand) and marginal cost (supply) are unlikely to remain constant, requiring estimation of changes in consumer and producer surplus. [44]

This approach can be applied in a variety of contexts. For example, wetlands often serve as nurseries for fish species that are harvested for commercial markets. They are thus an input to commercial fishing, and their services affect the supply and market price of harvested fish. The economic benefits of protecting wetlands can then be estimated by their contribution to the market value of the output of the commercial fishery. For relatively small changes, the additional output of the fishery can be valued simply by multiplying the change in output by the market price of the fish. Similarly, when a river is used as a source of irrigation water for agriculture, both the water quantity and quality directly contribute to the production of food. The economic benefit of an improvement in either water quantity or quality can be estimated by its contribution to the market value of food production. Again, for small changes, the market price of the agricultural product multiplied by the resulting change in output provides a measure of the value of the water quality or quantity change.

Status as a Method

Market-based methods are based on well-established economic principles and econometric practices (Boardman, et al. 2006, McConnell and Bockstael 2005). They have been used for more than 30 years to evaluate a variety of economic policies (Hufbauer and Elliott 1994, Winston 1993). Applications to the valuation of ecosystem services include Barbier and Strand (1998) and Barbier, Strand, and

Sathirathai (2002). EPA has used these methods to value ecosystem service benefits from air pollution control in the markets for agricultural products and for timber products (US EPA 1999).

Limitations

Estimating both consumer and producer surplus requires the development of empirical models for the demand and supply relationships describing market outcomes. Depending on each application this can be difficult due to lack of data at the level of resolution required to describe how economic policies affect each of these relationships.

The majority of environmental policies do not directly impact the prices and quantities of goods and services traded in markets, so this method is only available in a limited subset of cases. In addition, it will only capture the benefits of a change that are manifested in marketed outputs. For example, a wetland may contribute not only to commercial fishery production but also to flood control, water purification, wildlife habitat, etc. These other benefits would not be captured by a market-based approach. Another limitation of this method is that, if there are market imperfections stemming for example from market power, this can confound the measurement of demand and supply and distort the relationship between prices and the marginal value and marginal cost of providing a private good. As a result, this distortion will carry over into any estimation of economic values based on market prices.

Many non-environmental factors can affect demand and supply relationships that are also important. Seasonal variations in use or availability of goods and services related to environmental policies can affect prices, and this needs to be considered. The modeling and estimation of demand and supply functions can be complicated. Ultimately, what can be learned about the influence of environmental or any other policy is limited by the available data. These limitations are best described as an identification problem – do we have sufficient information to identify the effects that are hypothesized to reflect how environmental policy influences market supply and demand?

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Non-market Methods – Revealed Preference

When environmental changes affect goods and services that are not traded in markets, non-market valuation, using either revealed preference or stated preference, becomes necessary. Revealed preference methods look at people's behavior in markets that are related to ecological services to reveal underlying values. For example, someone's decision about which of two houses to purchase might reveal information about how they value air quality or a scenic view if the two houses vary with regard to that environmentally-related attribute. Because the revealed preference methods for measuring values use data on observed behavior, some theoretical framework must be developed to model this behavior and to relate the behavior to the desired monetary measures of value and welfare change. A key element in the theoretical framework is the model of the optimizing behavior of an economic agent (individual or firm) that relates the agent's choices to the relevant prices and constraints, including the level of ecological services being provided. If a behavioral relationship between observable choice variables and the ecosystem service can be specified and estimated, this relationship can be used to calculate the economic value of changes in these service flows. For example, one well-established behavioral relationship is that between the costs to individuals of visiting a recreation site and the numbers of visits make to the site. See the discussion of the travel cost method that follows. If the numbers of visits also varies systematically with the level of an ecosystem service provided by the site, then the value of the ecological service can be inferred from these relationships.

The degree to which inferences about the value of a change in ecosystem services can be drawn from market observations, and the appropriate techniques to be used in drawing these inferences, both depend on the way in which the ecosystem service enters individual utility functions. The exploitation of possible relationships between environmental goods and private goods leads to several empirical techniques for estimating environmental and resource values. This section covers three revealed preference methods: travel cost, hedonics, and averting or mitigating behavior models.

Travel cost

Brief description of the method

The travel cost method accepts as a maintained hypothesis that people have economic demand functions for the services of environmental resources that are associated with observable choices they make to travel to a particular location. While in principle this method could be applied to travel for a variety of purposes, in practice it is applied in the context of travel associated with outdoor recreation. Lakes, rivers, forests, and beaches are examples of the types of resources involved. The essence of the method is recognition that users pay an implicit price by giving up time and money to take trips to these areas for recreation. This recognition is important because most of the public facilities for recreation in the United States do not have market determined fees for that use. The cost of a visit to a site is the out-of-pocket costs of travel including any site admission fees, opportunity cost of travel time, and the opportunity cost of time on site.[45]

The values of ecosystem services are captured by the method to the extent they can be represented as factors that influence a person's decision about where or how often to travel. For example, a measure of the availability of fish in a lake used for fishing would presumably influence (along with other factors) a person's decisions about whether or how often to visit the site for fishing.

Until about the middle 1990's, the travel cost literature estimated travel costs for the simple case of a new site or loss of site. The loss of an area (due to activities that eliminate its recreational value) is represented as "equivalent to" a price or travel cost change that is large enough to cause all existing users to no longer take trips to the site. To use the travel cost method for more sophisticated environmental policy choices, i.e., those that change the quality of recreational opportunities, analysts need to know how those quality attributes influence the demand function for recreation. In practice, most economic models for recreation now use random utility models (RUM), which describe the decision process associated with each individual selecting which recreation site among a number of alternatives to visit. A RUM framework describes these choices as the result of a constrained optimization process: selecting the site that yields the maximum level of utility (or well-being) that is possible given a person's constraints. The result can be expressed as a function of travel costs, site characteristics such as the level of ecosystem services and the facilities to support specific activities (e.g., boat ramps, ski lifts, etc.), and users' attributes.

Status as method

The travel cost methodology is based on well-established economic principles. There has been extensive use of this method in peer-reviewed literature, dating to 1947 when Harold Hotelling first proposed it. There is less experience with using the method to estimate trade-offs for a wide range of attributes of recreation sites. Assumptions are understood and documented. Meta analyses – Smith and Kaoru (1990), Walsh, Johnson and McKean (1992), Rosenberger and Loomis (2000), Johnston, et al. (2003) and Johnston, et al. (2005) have documented the performance of the model in different circumstances.

Measures of the economic value have been used in EPA's RIA analyses for regulations affecting recreation resources. A recent example is the Phase III component of the 316B rule. The rule seeks to reduce impingement and entrainment of fish and other organisms through power facilities' uptake of cooling water.

Strengths and Limitations

The primary data requirements of the travel cost methodology are as follows: data on people's usage of recreation sites; measures of individuals' values of time and time constraints; information that allows measures of the environmental attributes of the resources used for recreation to be linked to those resources; and information that describes the relationship between technical indexes of the attributes of recreation sites and measures that users can be expected to understand and know.

The analysis requires technical training in micro-economic modeling of demand and extensive experience with micro-econometrics to estimate recreation demand models. Less experience is required to use existing models to estimate economic values for changes in factors hypothesized to affect people's recreation behavior.

Uncertainties

One important source of uncertainty in the travel cost model is the value of recreationists' time as a component of the cost of a recreation trip. Randall has argued that for several reasons "travel cost is inherently unobservable" (1994, p. 88). The role of time in explaining recreation demand and in valuing recreation visits and sites raises some thorny issues for both the standard travel cost and RUM approaches of analysis. Clearly, time is an important variable in the analysis of recreation demand and value. However, numerical estimates of demand and value require either that the numerical value of the shadow price of time be known or that it be estimated from a model of the choices made regarding the uses of time. A variety of models of choice and time are available in the literature. However, as yet, different model structures yield quite different estimates of the shadow price of time, and there is no clear basis for preferring one model and its value over other models. Until these issues can be resolved, estimates of recreation values should be presented as conditional upon a specific value of the shadow price of time or a specific modeling approach regarding the role of time, and the uncertainty in the estimates that this implies should be acknowledged. For more on this issue, see Freeman (2003, Ch. 13).

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Hedonics

Brief description of the method

Hedonic methods seek to exploit possible relationships between demands for private goods and their associated bundle of characteristics, including environmental characteristics. For example, the demand for a house depends not only on its physical attributes (e.g., total size, the number of bedrooms, etc.) but also on the surrounding environmental characteristics (e.g., air quality, proximity to beach, etc.) When people select from among the set of available goods (e.g., available houses), the hedonic model assumes that they will choose the one that is their most preferred given its price and attributes. In equilibrium, the set of prices for these differentiated goods will be structured so there is no incentive for anyone to change their choices. The hedonic price function relating prices to characteristics is a reduced form description of this equilibrium condition. The primary applications of this logic in the field of environmental economics involve housing prices and the wage rates for jobs

Assuming that the price of a house reflects the attributes of that house, its property, neighborhood, and facilities that are “near” it, then the hedonic price function reflects a buyer’s marginal willingness to pay (WTP) for small changes in one of these attributes. This measure is a single point estimate of the marginal value. The method does not provide the basis for measuring, without additional assumptions, any economic benefits that are associated with a large change in one or more of these attributes. These attributes can include the structural features of the house, its lot, and the characteristics that are conveyed to those living in the home because of its location. For example, if a house is on the coast, residents can experience the coastal views, any beach related amenities, as well as any greater risk of damage that might arise from coastal hazards. If that feature is some aspect of an ecological service available to an individual because she lives in the house, the model allows that incremental value of a change in that service to be estimated.

If the attribute measures a characteristic that can be related to a policy, e.g., proximity to a Superfund site before and after clean up, then it is possible to describe a buyer's willingness to make trade-offs for small changes in that attribute. There are important qualifications that must be considered in evaluating the results from these models. For example, to the extent the prices for homes near wetlands or in flood zones are found to be related to (i.e., have a statistically significant association with) the measures that are used to isolate these features, then there is indirect evidence that these features are recognized by buyers and sellers. This result follows because they contribute to the observed equilibrium prices for the homes represented by the hedonic function. Relating such a recognition to a measure of the incremental value for the change in services requires assumptions describing how changes in the variable that can be measured and included in the price function relate to changes in the service of interest.

Extensive data are needed to estimate a statistical function that relates housing prices to housing characteristics that include environmental attributes so that small changes in the quality or quantity of that environmental attribute can be related to small changes in housing prices.

Status as a Method

The hedonic method has been widely used to evaluate site-specific amenities and disamenities. Examples of applications involve: air pollution, noise pollution, proximity to water bodies, wetlands, coastal areas, and location of homes in hazardous areas such as earthquake or flood zones. See Palmquist (2005) for a general overview of the literature and Smith and Huang (1995) for a meta-analysis of the studies of air pollution and property values. This and other meta-analyses indicate clear support for the methods for applications where we can expect buyers and sellers to have knowledge of the amenities.

Applications involving site attributes that might be more closely aligned with services of ecosystems are much more limited. Several studies have investigated the effects of proximity to wetlands of different types as well as for distance to open space. Examples include

Mahan, et al. (2000), Netusil (2005), and Smith, et al. (2002). An important difficulty in using these results arises in converting the incremental value estimated for a change in distance to a measure more directly related to changes in ecosystem service.

Strengths and Limitations:

Hedonic methods are familiar to most people who have purchased or sold a house because realtors do an informal hedonic type analysis comparing homes described as “comparables” to price a proposed new listing.

The main strength of the hedonic housing method is that it is based on people’s actual choices. However, all hedonic methods face significant econometric hurdles and are subject to the standard criticism of statistical relationships that they reveal correlation but fall short of revealing causation. Hedonic estimates can be sensitive to the choice of model specification (see, for example, Cropper, Deck and McConnell 1988). Moreover, relating housing prices to many ecosystem services remains elusive. Finally, hedonic methods can only capture the value of environmental changes that individual homeowners recognize. The method is best suited for local housing markets. While several studies have estimated national hedonic property value models, it is generally agreed that it is unreasonable to assume that there is a single national market for housing with an equilibrium that adequately describes the trade-offs among housing attributes in very different locations.

To implement the method for estimating the hedonic price function, it is important to have access to a real estate transaction database with sales prices, housing characteristics, and the latitude/longitude coordinates for each property. These data can then be merged to GIS files describing access to various spatially delineated environmental resources such as air quality as well as to ecosystem services.

Uncertainty

The primary sources of uncertainty with the hedonic model for policy applications arise with the measurement of attributes that are assumed to represent the environmental services available to people due to living in the house. Further research on how people learn about these aspects of a location and what they consider to be conveyed by a location would help to address this issue.

In addition, simulation analysis evaluating the performance of hedonic price functions as approximations to an equilibrium matching process would also contribute to our understanding of the sensitivity of the method to assumptions about model structure and functional form. See, for example, Cropper, Deck and McConnell (1988).

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Averting behavior models

Brief Description of the Method

Averting or mitigating behavior models simulate consumer behavior and rely on the existence of an activity that substitutes for the services provided by an environmental resource. The averting behavior method infers values from defensive, mitigating, or averting expenditures, i.e., those actions taken to prevent or counteract the adverse effects of environmental degradation. For example, an individual might purchase a water filter to avoid the health risks associated with drinking unfiltered water. By analyzing the expenditures associated with these defensive purchases, researchers impute a value that individuals place on small changes in environmental or health risks. In effect, a defensive expenditure is spending on a good that is a substitute for health protection or an environmental quality or service. Because the method is based on an estimation of the marginal rate of technical substitution between the environmental service and a market good or service with a known market price, it is capable of producing monetary estimates of the value of the environmental service. What is required is an understanding of the technical relationships underlying the ability of the environmental service and its market good substitute to enhance human well-being.

Status of the Method

There is a substantial literature on the theoretical dimensions of the method (for example, Freeman 2003, Dickie 2003, Smith 1991) but relatively few convincing studies demonstrating it will work in practice. Examples of defensive expenditures include the choice of automobile type (as it relates to fatality risk), safety helmets, fire alarms, and water filters. However, since these expenditures only capture a portion of an individual's willingness to pay (WTP) for these protections, averting behavior results are sometimes interpreted as a lower bound on willingness to pay to avoid a particular harm. The most common application of averting behavior models has been the estimation of values for morbidity (illness) risk.

Limitations

Averting behavior studies rarely provide economic values for ecosystem services. Even for those averting behavior studies for water quality, the motivation for the averting behavior is usually to protect health or life.

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Non-market Methods – Stated Preference

Brief Description of the Method

Stated preference methods rely on survey questions that ask individuals to make a choice, describe a behavior, or state directly what they would be willing to pay for specified changes in environmental services not traded in markets. The various stated preference techniques are distinguished by how the information is presented, what questions are asked, and how their responses are formatted. It is important to acknowledge that the choices, stated values, or revised patterns of use are derived from answers to questions that ask respondents what they would do, or how much they would pay for, or how they would alter their choices in response to changes in the amount of a non-market good or service in a specified hypothetical setting. This is in contrast to Revealed Preference Methods, which are based on observing the actual choices made by people facing real constraints on income, etc. Stated preference methods offer the opportunity to measure trade-offs for anything that can be presented as a credible and consequential choice. Hence, their primary advantage is their ability to, in principle, measure a wider set of values. In particular, they are the only economic methods that can measure non-use values.

Although not all authors use the same terminology, the term stated preference methods generally include any survey questions in which respondents are asked hypothetical questions designed to reveal information about their preferences or values. The term encompasses three broad types of questions. The first type involves questions that ask directly about monetary values for a specified commodity or environmental change. These are usually called contingent valuation method questions (CVM). In the past, the most commonly used CVM questions simply asked people what value they place on a specified change in an environmental amenity or the maximum amount they would be willing to pay to have it occur. These are usually open-ended in that the individual has to state a number rather than respond to a number offered by the researcher. The responses to these questions, if truthful, are direct expressions of value. The other major type of CVM question asks for a yes or no answer to the question, “Would you be willing to pay $X ...?" Each individual's response reveals only an upper bound (for a no) or a lower bound (for a yes) on the relevant welfare measure. Questions of this sort are termed discrete choice questions. Responses to discrete choice questions can be used to estimate willingness-to-pay functions or indirect utility functions.

The second and third major types of Stated Preference methods do not reveal monetary measures directly. Rather, they require some form of analytical model to derive welfare measures from responses to questions. The second type of question is called variously "choice experiment," "conjoint analysis," or sometimes an "attributes based method" (Holmes and Adamowicz 2003). In this approach to questioning, respondents are given a set of hypothetical alternatives, each depicting a different bundle of environmental attributes. Respondents are asked to choose the most preferred alternative, to rank the alternatives in order of preference, or to rate them on some scale. Responses to these questions can then be analyzed to determine, in effect, the marginal rates of substitution between any pair of attributes that differentiate the alternatives. If one of the other characteristics has a monetary price, then it is possible to compute the respondent's willingness to pay for the attribute on the basis of the responses.

In the third type of SP question, individuals are asked how they would change the level of some activity in response to a change in an environmental amenity. If the activity can be interpreted in the context of some behavioral model such as an averting behavior model or a recreation travel cost demand model, the appropriate indirect valuation method can be used to obtain a measure of willingness to pay. These are known as contingent behavior or sometimes contingent activity questions.

Status of the Method

The method has an extensive literature of principles and applications extending over a forty-year period. Mitchell and Carson’s (1989) pioneering treatise is still the primary reference on CVM, especially for design and implementation questions. See also Carson (1991). Two new works that focus on best practice and empirical estimation for CVM and stated choice studies are Boyle (2003) and Holmes and Adamowicz (2003), respectively. The so-called NOAA Blue Ribbon Panel (U.S. National Oceanic and Atmospheric Administration 1993) reviewed CVM in the context of assessing damages to natural resources in support of litigation and provided its guidelines for best practice. Other important references are: Bjornstad and Kahn (1996) for a review of theoretical and empirical issues that includes assessments by both proponents and critics of stated preference methods; Kopp, et al. (1997); Bateman and Willis (1999); Bateman, et al. (2002); and Smith (2004, 2007).

Use of the stated preference methods for environmental valuation has been controversial. A major issue concerning the status of stated preference methods is the validity of the resulting value estimates. There are several concepts of validity and various approaches to assessing the validity of responses. A commonly cited issue related to validity is the existence of what is known as hypothetical bias. The argument is that the hypothetical nature of stated preference questions results in the overstatement of economic values, or what is known as hypothetical bias. However, the evidence regarding the extent of this bias is mixed (see Murphy, et al. 2005 for a recent discussion). The controversy surrounding stated preference methods had the salutary effect of stimulating a substantial body of new research on both practice and on the credibility or validity of stated preference estimates of value. A good overview of the issues raised in this controversy is contained in the three essays published as a symposium in the Journal of Economic Perspectives (Portney 1994, Hanemann 1994, and Diamond and Hausman 1994). See also, Hausman (1993) and Freeman (2003) and references therein for further discussion.

Strengths and Limitations

Strengths include the accumulated experience of forty years of practice and research. Also in principle, stated preference methods are the only set of methods capable of capturing so-called nonuse values, since without use there is no behavior that can reveal values through application of revealed preference methods.

In addition to the controversy stemming from the hypothetical nature of the questions noted above, some people question whether surveys are capable of providing useful information about preferences. One issue is whether preferences regarding unfamiliar environmental goods are well-formed and stable (see Part 1, section 2.4). In addition, since responses to questions must reflect in some sense the knowledge that individuals have about the thing being valued as well as respondents' preferences, the methods cannot be used to value ecosystem services about which people are ignorant. For example, if respondents were asked questions concerning phytoplankton but were ignorant of the role of phytoplankton in supporting the aquatic food chain and higher order species that they might value, their responses might be interpreted as placing no value on phytoplankton. In such a case, stated preference methods will not generally be useful for valuing changes in supporting ecosystem services (see Part 1, section 2.1) since most lay individuals are not aware of the crucial role of these services. One solution to this problem is to use the survey instrument to convey information to respondents about the role of the ecosystem service being valued and the potential consequences of changes in the level of this service. See, for example, Banzhaf, et al. (2004). Then, of course, the question becomes one of the validity of the information provided to respondents and the potential for biasing responses by providing biased information.

Finally, even if preferences are well-formed and individuals are aware of the role of the relevant environmental attributes, the survey might not provide incentives for respondents to reveal their preferences accurately. This depends, among other things, on the degree of incentive compatibility of the various questioning formats and the set of methods as a whole. Carson, et al. (2000), reasoning from first principles about what is in the best interest of respondents faced with a scenario, payment vehicle, and elicitation question, have established under what conditions stated preference questions give people incentives to reveal their true values. The first two conditions are that the survey question be about something that matters to the respondent and that the respondent believes that his/her response might affect the outcome of the policy issue that is the subject of the survey. If both conditions hold, then the survey question is termed "consequential" to respondents. For consequential questions, it is possible to reason from an assumption of acting on rational self interest to predict whether responses will be truthful and if not, then at least in some cases what the direction of bias will be.

For consequential questions, the only question format that can in principle be incentive compatible is the single discrete choice question. In addition, this form requires the further condition that the government agency is perceived as being able to compel payment of some amount from the respondent if the good is provided. For example, questions that ask about the willingness to make a voluntary contribution to support some government action fail this condition and provide incentives to respond "yes" even when the requested contribution is greater than the respondent's WTP.

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Combining Revealed and Stated Preference Methods

It is possible to combine revealed and stated preference methods to estimate what both types of choices imply for characterizing an individual's willingness to pay for changes in environmental services. Cameron (1992) was the first to propose this idea for environmental applications. To be informative, this strategy must be based on an analysis of the revealed and stated behaviors to establish that the empirical models used to describe these outcomes share at least one parameter. That is, they must each be capable of identifying at least one common parameter. Ideally there would be more parameters shared between the models. Most applications collect the two types of data (i.e., revealed and stated preference) from the same respondents. This requirement is not essential. It would be possible in principle to combine samples with different respondents providing the revealed and stated components of the analysis. A key issue in applying these methods to the task of valuing ecosystem services is the need to have measures for the quality and amount of ecosystem services that are compatible with models and data typically available for revealed and stated preference models.

See Adamowicz, et al. (1994), Earnhart (2001, 2002), and McConnell, et al. (1999) for more recent applications.

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GROUP EXPRESSION OF VALUES AND SOCIAL/CIVIC VALUATION

Valuation of ecological systems can also involve expressions of group or public value, rather than elicitations of the values of individuals or biophysical rankings according to a previously agreed-upon scale. Group or public expressions of ecological value have attracted attention for at least two reasons. First, some experts believe that group discussions and deliberations can help people form clearer understanding of values. Second, a number of experts believe that group expressions of the "public good" in general, and of ecological value in particular, may be distinct from the aggregation of individuals' reports of their private welfare because they explicitly reflect public regardedness.

Although many reports briefly discuss the potential role of deliberative processes in helping to develop more informed valuation (National Research Council 2004, Millennium Ecosystem Assessment Board 2003, Science Advisory Board 2000), the reports do not evaluate or recommend any specific method or approach. The committee notes parallels between group and public expressions of value for ecological valuation and the deliberative-analytic process recommended for risk characterization by the National Research Council (1996). The National Research Council report, however, did not address in any detail how deliberative approaches might be implemented or assessed or how they might be transferred to ecological valuation.

Traditional economic valuation methods attempt to measure and aggregate the values that individuals place on changes in ecological systems and services based on their personal preferences as consumers of those systems and services. An alternative approach is to try to measure the values that groups of individuals place on changes in such systems and services explicitly in their role as citizens – social/civic valuation. This approach measures the monetary value that groups place on changes in the systems and services when asked to evaluate how much the public as a whole should pay for increases in such systems and services (public willingness to pay) or should accept in compensation for reductions in the systems and services (public willingness to accept). The value measurement purposefully seeks to assess the full "public regardedness" value, if any, that the group attaches to any increase in community well-being attributable to changes in the relevant systems and services.

Social/civic values, like values based on personal preferences, can be measured either through revealed behavior or through stated valuations. One principal source of revealed values for changes in ecological systems and services are votes on public referenda and initiatives involving environmental decisions. Other public decisions also may provide measures of social/civic values, including official community decisions to accept compensation for permitting environmental damage and jury awards in cases involving damage to natural resources. Because all research on sources of revealed public value have focused on referenda and initiatives, however, this section discusses only the use of referenda and initiatives as a source of revealed value. Other public decisions raise unique issues as sources of revealed value. The committee does not recommend that EPA currently pursue their development. Where revealed values are difficult or impossible to obtain from referenda or initiatives, social/civic values may be measured by asking "citizen valuation juries" or other representative groups the value that they, as citizens, place on changes in particular ecological systems or services.

This section discusses several approaches to forming, eliciting and considering group or public values. Some of the methods are designed to help elicit clearer understandings of value, while others focus on identifying group expressions of public valuation. The committee recommends each method be considered for its merits at different stages in the ecological valuation process and in difference decision-contexts relevant to EPA

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value	
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?
Focus Groups	Narrative summaries, frequency tallies, consensus	Full discovery and articulation of all the values that are relevant and exploration of agreements and conflicts among stakeholder constituencies	verbal reports	sample from public
Referenda and Initiatives	Historical monetary data on communities' choices regarding ecological impacts	What the body politic as a collectivity values in terms of policy outcomes	Behavior	Selected stakeholders
Citizen Valuation	Qualitative summary of jury	How a representative group views the	Verbal reports	Selected

Method	Form of output/units?	What is method intended to measure?	Source of Information About Value	
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?
Juries	decisions which may include quantitative or monetary decisions	social civil value of changes to ecological systems and services		stakeholders

	Degree to which Method Has Been Developed or Utilized	Recommendations for Research to Strengthen Use of Method	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation	Issues Involved in Implementation
Focus Groups	• Not clear the extent to which focus groups are systematically used in EPA policy making • The OMB and other guidelines do not clearly specify the criteria for using focus groups		• Can be useful and important for designing and pre-testing more formal surveys • May also contribute to the design of more effective communications of Agency decisions	
Referenda and Initiatives	• Logic has been used primarily in the literature on health and safety	• The research needed to make the results of public decisions through referenda and initiatives most useful for inferring values would consist of the creation of a data bank of referenda and initiative outcomes, optimally screening out those involving multiple, confounding elements.	• Can provide monetized values—of the community's formal decision and values, ceilings, or floors of the median voter's valuation • With follow-up surveys can provide information on beliefs, assumptions and motives regarding the ecosystem preservation issues that voters perceive are at stake • Any EPA decision context calling for monetized valuation could employ these variants, either singly or as cross-checks with conventional revealed	• Analysis meets the criteria for when method "works best"

	Degree to which Method Has Been Developed or Utilized	Recommendations for Research to Strengthen Use of Method	Potential for Future Use by EPA in an Integrated and Expanded Approach for Valuation	Issues Involved in Implementation
			preference or stated preference approaches.	
Citizen Valuation Juries	• Experimental method in the context of ecological valuation • Used primarily to help governments rank options for achieving particular goals. Only a few efforts have been made to date to use citizen juries to generate monetary or other estimates of the social/civic value of environmental changes.	• Do citizen valuation juries arrive at different valuations than individual respondents to CV surveys? If so, how and why do the valuations differ? • How stable are valuations provided by citizen juries? How much variation exists among the valuations produced by different citizen juries? • How do jury selection processes affect the valuations of the jury? What methods exist to overcome the inevitable bias arising from the small size of citizen juries? • How should information be provided to citizen valuation juries? • How do decision making rules (e.g., consensus versus unanimity) affect valuations? What are relevant considerations in choosing among the different decision making rules?	• Potentially useful both to identify socially important assessment endpoints and to attach a value, monetary or socio-psychological, to changes in the assessment endpoints • Can expand the role that the public plays in valuations of changes in ecological systems and service	• Hypothetical character of all stated preference valuations • Issues of group dynamics • Choice of jurors

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Focus Groups

Brief description

Focus group methods engage small groups of relevant stakeholders in facilitated discussion and deliberation on selected/focused topics relevant to the assessment of the effects of a policy, or alternative policies, outcomes, and/or consequences. Typically, experts and/or trained facilitators present the context, motivation and goals for the group and open-ended narratives are collected from the participants, usually in the context of discussion and deliberation with other members of the group and the experts/facilitators. Collected narratives are subjected to qualitative analyses to identify and possibly to ascertain levels of consensus on relevant issues, perspectives, and positions represented by the participants. Reports of focus group results typically include numerous quotations of collected comments, along with the investigators' interpretations of the implications for the problems/policies/outcomes being addressed (e.g., Winter and Fried 2000). Less often collected narratives are subjected to more rigorous analyses based on formal logic models or discourse analysis systems (Abell 2004, Bennett and Elman 2006).

Relative to formal surveys, focus groups use small numbers of respondents and do not typically attempt formal probability sampling to select participants. Emphasis is instead on assuring that at least one representative from the full range of interests and perspectives relevant to the policies or outcomes at issue are included. The goal of a focus group is rarely value assessment per se, but a full discovery and articulation of all of the values that are relevant, and exploration of agreements and conflicts among the stakeholder constituencies represented by participants. Thus, focus groups are often employed early in policy and decision making, including the identification of the problems to be addressed and the formulation of alternative policies to address those problems. It is common for focus groups to be used in the process of designing and pre-testing more formal surveys. For example, Shields, et al. (2002) reported that 80 focus groups distributed across the nation were used in developing the USDA Forest Service survey illustrated in Text Box 12.

Relation of Method to the C-VPESS Expanded and Integrated Assessment Framework

Focus groups would be most appropriate and most useful at the earliest and latest stages of the decision-making process. While focus groups do not generally provide quantitative assessments for alternative policies or outcomes, they can make important contributions to improving the design, development, and pre-testing of more formal surveys that can provide reliable and valid quantitative assessments of public concerns and values. Genuine probing interactions with individuals or groups representing key stakeholders and including divergent views and concerns should be a central part of problem definition and identification of significant ecological and associated social effects components of the process. Such interactions with key stakeholders and with citizens could also inform the values integration and negotiation in the final decision process and guide and pre-test the communication of that decision.

<u>Status of Method</u>

It is not clear the extent to which focus groups are systematically used in EPA policy making, nor do the OMB and other guidelines clearly specify the criteria for using these methods. Focus groups are widely used in marketing and political polling contexts and the U.S. Forest Service national survey by Shields, et al. (2002) described above reported that "over 80 focus groups conducted around the continental United States" (p. 1) were used in the design and development of the survey, as well as to support the interpretations and conclusions from the survey. Public meetings and on-site demonstrations are frequently cited as playing a public involvement role in EPA policy decisions, and a formal "Multi-Stakeholder Group" was assembled and used in the Avtex Fibers Superfund Site decision and implementation process, but it is not clear whether any of these activities can be construed as using a focus group, nor is it clear how often such methods have been used to systematically compare alternative policies/actions.

The use of focus groups would seem to be completely consistent with previous advice of the EPA Science Advisory Board (US EPA 2001) recommending increased use of "stakeholder processes" in Agency decision making. Stakeholder processes were defined as "…group processes in which the participants include non-expert and semi-expert citizens, and/or representatives of environmental non-governmental organizations, corporations and other private parties in which the group is asked to work together to: define or frame a problem; develop feedback in order to better inform decision makers about proposed alternative courses of action; develop and elaborate a range of options and/or criteria for good decision-making which a decision-maker might employ; or, either explicitly or implicitly, actually make environmental decisions." (p 8) Still, the term "focus group" was not used anywhere in this document. While no specific evidence has been found either way, it seems reasonable to assume that individual narrative interviews have not been important components of EPA decision-making processes. Certainly the qualitative nature of the information provided by both focus groups and individual interviews, and the general disinterest in representative sampling makes them poor candidates for formal policy evaluation exercises, but that does not preclude their having a role in earlier stages of the decision-making process as envisioned by the C-VPESS.

Focus groups can have important and useful roles to play in Agency policy and decision making. However, their emphasis on qualitative analyses and their typical disregard for representative sampling can make them less useful for systematic evaluations or comparisons of alternative policies and outcomes. The method can be very useful and important for designing and pre-testing more formal surveys that do provide quantitative assessments of values for alternative policies and outcomes. Qualitative methods may also contribute to the design of more effective communications and rationalizations of Agency decisions to stakeholders and to the general public.

Referenda and Initiatives

Brief description of the method

Referendum and initiative votes provide the basis for a set of valuation approaches that can yield monetized values, but use somewhat different logic than that of the conventional individually based revealed-preference and stated-preference methods. The outcomes of referenda (measures placed on the ballot by a legislative body) and initiatives (ballot measures proposed by citizens) directly express what the body politic collectivity values in terms of policy outcomes. These expressions may or may not correspond closely to the aggregated values of the individuals in the community in terms of outcomes. Referenda approaches (not to be confused with the "referendum format" often used for posing questions to solicit contingent valuation responses) provide information about the policy preferences of the median voter; under certain circumstances this information can tell us about the median voter's valuation of specific environmental amenities, and can even provide information, albeit weaker, about mean valuations of those who participate in the voting process. They can also be useful for cross-validating any other valuation approach that permits a prediction as to the outcome of a referendum or initiative. When a referendum or initiative is followed by a survey to determine what voters believed the financial burden to be, the approach can also elicit relevant beliefs and motives to reinforce the specific willingness-to-pay or willingness-to-accept information.

There are four variants for analyzing referenda and initiatives:

- Referendum/initiative analysis
- Analysis of public decisions to accept pollution or resource depletion
- Referendum/initiative analysis followed by a survey
- Analysis of public decisions to accept pollution or resource depletion followed by a survey

Direct referendum/initiative analysis, with or without a follow-up survey, can evaluate trade-offs between community and/or household costs (higher taxes, possibly job losses) and eco-system improvements (establishment or improvement of air, water, biodiversity protection, etc.). Direct analysis of public decisions to accept pollution or resource depletion, with or without a survey, can evaluate trade-offs between community and/or household benefits (increase in tax base, job creation, infrastructure improvements, etc.) and eco-system deterioration (greater pollution, amenity reductions).

Text Box 14: Direct Analysis of Public Decisions to Accept Pollution or Resource Depletion

Some public votes can provide inferences for willingness-to-accept decisions. These decisions involve a community's vote as to whether to permit the entry of a new firm or a new (or increased) economic activity despite the expectation that such permission will degrade the ecosystem. The payment represents the ceiling on the community's valuation of the environmental amenities that are being relinquished. It is a ceiling because of the possibility that the community would have accepted a lower level of compensation, and if the community valued the forgone eco-system services more than the compensation, then presumably it would not have accepted the compensation. However, if there is a vote and the outcome is close, the calculated valuation can be considered to be close to the community's valuation.

The estimation task involves assessing the amount of environmental damage in physical terms and the amount of compensation in monetary terms. Typically this compensation will come in the form of additional sources of taxes, the value of infrastructure that the new entrants provide for the community, additional income earned by community members, etc. The per-household as well as per-community compensation would be relevant. For example, the entry of an air-polluting factory may be accepted only after the factory's owner commits to a certain number of jobs for the community, building a park, upgrading roads, contributing to the community's vocational program.

Obviously many "community decisions" to permit the entry of polluters or other activities that degrade the ecosystem are not amenable to this approach, because community leaders negotiate the level of benefits that the community will receive without a vote being taken, or the benefits or costs are difficult to estimate.

Text Box 15: Referendum/Initiative Analysis Followed by a Survey

The alternative to relying solely on the referendum or initiative outcomes to make willingness-to-pay estimates consists of combining the voting outcome with a follow-up survey to determine the perceptions of the stakeholders. This variant amounts to a hybrid of the first variant and the "referendum format" contingent valuation approach. The floor of the willingness-to-pay value of the proposed eco-system improvements is estimated by determining the voters' perceptions of the eco-system improvements and costs proposed by a recent referendum or initiative. The respondents are asked whether they voted, how they voted, and what they believed the benefits and costs of the proposal were. The quantitative analysis of results of the referendum/initiative is the same as direct analysis without a survey, but using the perceived rather than actual stakes.

If, in addition to asking how respondents voted and their perceptions of the benefits and costs of the proposal, the randomly-sampled respondents who opposed the proposal are asked what (lower) cost would have induced them to vote for the proposal, and those who supported the proposal are asked how much more they would have been willing to pay, this approach also permits an estimate of aggregate and mean values, just as a standard contingent valuation study would, with less potential distortion arising from respondents' desire to be regarded in a favorable light. Thus the survey following a referendum or initiative can provide an internal cross-check of how much correspondence there is between the stated-preference approaches and the referendum or initiative findings (Schläpfer, Roschewitz, & Hanley 2004, Vossler and Kerkvliet 2003). In fact, the voting results can serve as a cross-check for any of the survey or other individual or group assessment methods.

It should be noted that in focusing on the benefits and costs that respondents report, rather than the actual benefits and costs that the referendum or initiative proposal specifies, the results do not reflect the community's formal decision. This is a significant difference in the philosophy underlying the standing of the results. That is, the first variant, even if it does not necessarily reflect the values that voters perceive, does represent what the voters have chosen. On the other hand, without the survey, the analyst cannot be certain what financial impact the voter believes is at stake, inasmuch as many initiatives and referenda do not explicitly specify the voter's financial burden. Different logics underlie their standing.

Text Box 16: Public Decisions to Accept Pollution or Resource Depletion Followed by a Survey

Just as the analysis of referendum and initiative outcomes can be augmented by determining voters' perceptions of the stakes, the ceiling of the willingness-to-accept value of eco-system deterioration can be estimated by determining the benefits perceived by voters who supported the arrangement accepting the entry of a polluting or depleting operation into the community, and their perceptions of the damage that would be done. Like the direct analysis of willingness-to-accept votes, if the arrangement was approved by the electorate, and the property rights are clear and transactions are low, the ratio of the perceived costs and compensation represents the ceiling of the median voter's valuation. The survey, best administered as soon as possible after the actual vote, would reveal what the community members interpreted the benefits and costs to be, thus bringing the valuation closer to individual values. But again, there exists a trade-off that the results would not have standing as the "community's choice." If the survey includes the questions of the conventional contingent valuation regarding how much each respondent would have been willing to accept, then the results would be even more robust in finding mean and aggregate valuations as well as median valuations.

How the method could be used as part of the C-VPESS expanded and integrated framework

These public decision approaches can provide monetized values—of the community's formal decision and values, ceilings, or floors of the median voter's valuation. In addition, with the follow-up surveys they can provide information on beliefs, assumptions, and motives regarding the ecosystem preservation issues that the voters perceive are at stake. Because the approaches focus on the content of proposals before the voting public, they do not directly identify ecosystem service impacts as a natural scientist or engineer would, but they will reflect voters' assessments of ecosystem service impacts. The approaches focusing exclusively on the decision outcomes do not directly identify changes in ecosystems and ecosystem services that are of greatest concern to people, although the survey variants can include questions to elicit this information. The approaches do address ecological impacts that other monetized approaches may underestimate, in that participation in citizenship, in contrast with the private-utility decisions reflected in the standard revealed-preferences approaches, can reflect concern for community well-being ("public regardedness") insofar as voters hold such regard. The approaches do not involve inter-disciplinary collaboration among physical/biological and social scientists or ecologists. There is a very strong potential that a data bank of inferred values from fairly large numbers of referenda and initiatives would assist EPA in presenting ranges of value for benefit transfers.

Status as a method

The logic of using formal public outcomes to infer how much society values particular outcomes has been used primarily in the literature on health and safety. For example, the value of a "statistical life" has been estimated by calculating how much public policies commit to spend in order to reduce mortality rates from health or safety risks, or, conversely, how much economic gain is associated with public decisions that reduce safety (e.g., by examining official decisions of U.S. states to raise or lower speed limits, Ashenfelter & Greenstone [2004] estimated the market value of the time saved by getting to the destination more quickly, and from that estimated the value of the additional expected traffic fatalities). The logic of making valuation inferences from referenda and initiatives has been addressed in a few publications, most directly in Deacon & Shapiro (1975) and Shabman & Stephenson (1996).

In comparing the valuations yielded by stated-preference approaches with those derived from public decisions, the studies typically show the inferences from public decisions to yield lower values—not surprising in light of the absence of the hypothetical element in the public-decision results. Although systematic comparisons with conventional revealed preference approaches are lacking, it is likely that the valuations of eco-system components calculated from public decisions would be higher, because public decisions do capture whatever elements of public-regardedness are present among the voters. The valuations based on public decisions have relevance within the paradigm that gives standing to the community votes as reflecting the policies that the public prefers. Even when a referendum or initiative passes by a wide margin, which reduces the precision of estimating the value held by the median voter, these outcomes provide strong input to decision makers regarding publicly held values.

Strengths/Limitations

Willingness to pay (WTP): The results will be most easily interpreted if the initiatives or referenda are: a) as focused as possible on a single dimension of environmental protection or amenity; b) free of ideological debate; c) confined to easily identifiable government costs rather than diffused and uncertain costs such as job losses; and d) the wording of the referendum or initiative is both unambiguous and clarifies the costs to the voters if the measure passes.

Willingness to accept (WTA): The results will be most meaningful if: a) the vote is explicit; b) the expected damage is well specified: c) property rights are clearly held by the community (i.e., the community has the right to refuse entry) d) the community's gains can be easily estimated; and e) the transaction costs are low.

The most useful referenda or initiatives would propose direct costs to the voters, typically in the form of taxes, fees, or bonds to finance actions designed to improve or protect ecosystems. Referenda or initiatives that entail restrictions on development (such as more stringent emissions or effluent standards) are less useful, because of the uncertainty of the level and incidence of the economic impacts. Similarly, in order to isolate the values attributed to particular ecosystem benefits, referenda and initiatives that address only one objective, such as preserving habitats or reducing air pollution air pollution, are more useful. With multiple objectives, the analysis cannot assign the willingness to pay to each component. Similarly, if it is clear that a referendum or initiative entails additional partisan political stakes (e.g., if it is widely viewed as a political test of a government official), the results are less illuminating in terms of the ecosystem values that the voters hold. The criterion of unambiguous wording is important in light of the findings that the wording of the questions can make a significant difference in the responses (Cronin 1989, Magleby 1984). However, the problem of misleading wording has been addressed in many jurisdictions, where election commissions have to approve the wording of both referenda and initiatives. Moreover, the fact that specific wording can influence responses is obviously not unique to the actual referendum and initiative situations; stated preference approaches, and surveys in general, face the same wording challenge.

Valuation based on initiative or referendum results would work best when:

- applied to the same jurisdiction (e.g., if a city is considering another storm control issue, the analysis of that city's referendum would be most appropriate), but can still be used via benefits transfer;
- a unitary conservation or environmental benefit is involved;
- the initiative or referendum outcome was a close vote (this yields stronger inferences about the actual valuation, rather than floors or ceilings);
- extraneous issues (such as whether the vote is a "political test" on particular politicians, or the mode of financing is controversial) are unimportant; and
- surveys can be accomplished soon after the actual vote.

These approaches attempt to measure the sum total of values of improving or protecting ecosystems and eco-system services; therefore both means and ends (instrumental and intrinsic) values can be involved. All variants in principle could measure the values attributed to all types of services, expressed in terms of monetary values per unit of ecosystem improvement or protection. The variants are flexible in terms of levels of data, detail, and scope, inasmuch as initiatives and referenda decisions have been made at all sub-national levels.

The valuations can be aggregated across benefits and with other methods, as long as the scale and magnitude of benefits are roughly the same. While highly complex initiatives and referenda are not good candidates for estimating value, the valuations generated from simpler cases can be used as inputs for complex applications.

Any EPA decision context calling for monetized valuation could employ these variants, either singly or as cross-checks with conventional revealed preference or stated preference approaches. Benefit transfer applications will be limited to cases of similar magnitudes of benefits, because of the likelihood that community decisions are highly sensitive to such magnitudes.

In uses that apply valuations directly to the jurisdiction previously experiencing the initiative or referendum, the scale would be the same municipality, county, or state. For benefits transfer, the scale should also be the same, given the need for similar magnitude of benefits and costs mentioned above.

Making valuation estimates directly from referendum or initiative outcomes has two advantages over conventional valuation methods. Unlike the standard revealed-preference approaches, such as hedonic pricing or the travel-cost method, voting on referenda or initiatives will reflect as much (or as little) public-regardedness as the voters actually hold toward the objectives involved. Standard revealed-preference approaches reflect the private utility-maximizing decisions of individuals who purchase homes, spend money to visit parks, etc.; these decisions do not reflect what individuals want for their communities. Voting affirmatively for referendum- or initiative-proposed public expenditures does elicit valuing on behalf of the community, insofar as the voters are so disposed. Of course, a voter may vote for or against a referendum or initiative proposal strictly out of concerns for herself and/or her family, but the outcome does not exclude the existence value component should it exist.

Unlike the conventional stated preference approaches such as contingent valuation, the analysis based on referendum or initiative outcomes is not subject to the possible distortions of hypothetically-posed choices. If a voter supports the referendum or initiative proposal, the vote contributes to the likelihood that the expenditures will actually occur and the costs will actually be borne. Some might argue that the chance that any one vote will decide the outcome of the referendum or initiative is remote, and therefore the vote is more of a symbolic act than a trade-off choice. However, there are two important responses to this point. First, whatever the mix of motives of the voters, the outcome is the community's decision, and therefore has standing in and of itself. This is the same logic by which we accept elected officials as legitimate even if we are dubious about the motives or rationality of the voters. Second, even if a voter believes that the chances that his or her vote will make the difference are negligible, the vote is still an expression of support or opposition to the proposal. There is little reason to believe that a "yes" vote would reflect just the gratification of voting "yes" (especially in secret balloting) rather than a belief that the proposal merits support.

Another concern that some would level against inferences based on referenda or initiatives is that these votes are often subject to intense efforts by interest groups, advocacy groups, and even governments to manipulate public perceptions (Butler & Ranney 1978, Cronin 1989, Magelby 1984). This concern has two aspects: whether the information on which voters base their decisions has been distorted, and whether the votes are swayed by appeals on one side or the other, especially by the side with the greatest resources (Hadwiger 1992, Lupia 1992, Owens & Wade 1986). The first aspect is more compelling: we certainly would be less willing to accept the validity of an estimate derived from voting decisions driven by serious misconceptions of the proposed benefits and/or costs. The outcome is still the official decision of that community, but the justification for using the result as the basis of benefits transfer to other communities would be very weak. On the other hand, the fact that referenda and initiatives are often subject to intensive campaigns of persuasion may be considered a virtue rather than a drawback, insofar as it would provide more information on both sides. In addition, the fact that individuals are exposed to efforts at persuasion is by no means confined to referenda and initiative contests: respondents to contingent valuation surveys have of course been subjected to many years of promotional activities by environmental groups; people who travel farther to a particularly popular national park such as Yosemite have been influenced by all sorts of communications extolling its virtues. In short, efforts at value persuasion are pervasive, and in any event should not be a basis for rejecting the significance of decisions of individuals exposed to those efforts. The philosophical basis underlying the use of referenda or initiatives, namely that the public's preferences are legitimately shaped by the political process, and that the public's policy preferences are important beyond how the public values the outcomes that these policies may produce, is quite different from the "progressivist" position that individuals' values should be determined in isolation of politics (Sagoff 2004: 177-178).

Another difference in philosophical basis is that the referendum and initiative results reflect intensity of attention to the issue, at least insofar as those who do not care enough to vote are excluded from the analysis. From the progressivist, technocratic perspective, everyone's values ought to be incorporated, because the policies ought to maximize utility (i.e., the consequences of public decisions) regardless of whether specific individuals are mobilized to take action. On the other hand, prominent strains of pluralist democratic theory regard intensity as a fully legitimate factor in determining policy outcomes (Lowi 1964).

One limitation of estimating values from referendum or initiative outcomes is that it is often difficult for voters to assess the actual stakes involved. The benefits will often have to be predicted (e.g., how much biodiversity will a reserve really safeguard; how much flooding will the flood-control system actually prevent?), entailing an amount of uncertainty. The benefits that do occur will often be community-wide, with some uncertainty as to how much an individual or particular household can take advantage of the benefits. On the cost side, the burden of a tax increase or bond measure on household expenditures may be very difficult for the typical voter to estimate, and the impacts of development restrictions may be even more difficult in light of the uncertainty as to which families would ultimately be affected. Insofar as the costs specified by the referendum or initiative are not easily translatable into household budget terms, the outcome, though it is still "the community's decision," is less revealing about the values held by the voters.

The outputs of these approaches should be easy to understand and to communicate to the public. It is a significant advantage to be able to say that the valuation of an ecosystem component has been estimated on the basis of how communities have decided what these components are worth.

Text Box 17 Referenda and Initiatives Used to Validate Contingent Valuation

In addition to taking the valuation derived from the analysis of public decisions as an input in itself, the analysis of public decisions, particularly referenda and initiatives, can be used to validate the results of other valuation methods. Several studies have compiled the results of initiatives and/or referenda in order to try to validate more conventional valuation techniques, especially contingent valuation (Kahn &

Matsusaka (1997), List & Shogren (2002), Murphy, et al. (2003), Schläpfer, Roschewitz, & Hanley (2004), Vossler & Kerkvliet (2003), Vossler, Kerkvliet, Polasky & Gainutdinova (2003). As Arrow, et al. (1993) recommend:

The referendum format offers one further advantage for CV. As we have argued, external validation of elicited lost passive use values is usually impossible. There are however real-life referenda. Some of them, at least, are decisions to purchase specific public goods with defined payment mechanisms, e.g., an increase in property taxes. The analogy with willingness to pay for avoidance or repair of environmental damage is far from perfect but close enough that the ability of CV-like studies to predict the outcomes of real-world referenda would be useful evidence on the validity of the CV method in general. The test we envision is not an election poll of the usual type. Instead, using the referendum format and providing the usual information to the respondents, a study should ask whether they are willing to pay the average amount implied by the actual referendum. The outcome of the CV-like study should be compared with that of the actual referendum. The Panel thinks that studies of this kind should be pursued as a method of validating and perhaps even calibrating applications of the CV method

<u>Does this method incorporate any specific ways of treating uncertainty? Is there any approach unique to this method?</u>

There are two distinct sources of uncertainty involved with this approach, depending on which variant is employed and how the outcomes are interpreted. If the referendum or initiative results are used without a follow-up survey, and the results are interpreted as indicating the aggregation of individual valuations, then there is uncertainty as to whether the voters understood the benefits and the payments accurately. If the results are interpreted as the community's preference per se, then the result is accurate in itself, as long as vote miscounting is not an issue.

The follow-up survey provides a way to determine whether voters understood the benefits and payments accurately. However, like any survey it also has its own sources of uncertainty: biases in which voters agree to respond to the survey, and untruthfulness in the individual responses. An additional source of potential uncertainty would arise if non-voters are asked to respond to the survey because of error on the part of the survey team. Despite these potential pitfalls, the follow-up survey (equivalent to a contingent valuation study) would serve as a cross-check on the referendum or initiative results.

Another source of uncertainty in undertaking a benefits transfer of valuation based on referenda or initiatives is that communities where these efforts are tried may be atypical; for example, it is possible that referenda and initiatives are more likely to be launched in communities with a stronger commitment to conservation. However, if enough straightforward referenda and initiatives are analyzed and put into comparable terms, including those that failed to pass, the range of results would provide more robust information than any single result.

Research needs

The research needed to make the results of public decisions through referenda and initiatives most useful for inferring values would consist of the creation of a data bank of referenda and initiative outcomes, optimally screening out those involving multiple, confounding elements. Because more than 1,100 referenda on open space issues alone were conducted in the United States between 1997 and 2004 (Banzaf, et al. 2006), the chances are good that a sizable number of referenda will meet the criteria. A preliminary analysis is needed to determine whether the communities that hold referendum votes are atypical of communities in general (i.e., is there a selection bias among the referendum-holding communities that would make their valuations atypical of the entire set of communities?). Thus a group of researchers at Resources for the Future is conducting in-depth analysis of 15 county-level, open-land referenda in Colorado, and also assessing the other open-land referenda in the rest of the United States (Banzaf, et al. 2006), to determine what kinds of communities hold referenda and what explains why the majority of referenda pass. The analysis of the valuation of benefits or damage would be straightforward calculation of the ratios of benefits or costs to the per-household costs, when such ratios can be deduced from simple referendum or initiative choices. The survey variants would involve considerably more effort of developing the questionnaire, administering it immediately after a referendum or initiative, and analyzing the additional information, yet the results would provide information on both median and mean valuation. Once model surveys are developed, they could be used with minor adaptations in different settings. In terms of resources required to make progress, roughly three researcher-years could produce a credible data base and systematically distill the information from the voting results that would be useful for policymakers. Using initiative or referendum voting results to cross-validate other valuation methods can be done at relatively low cost, although the follow-up survey options entail more effort, depending of course on how elaborate they are.

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Citizen Valuation Juries

Description of the Method

Another potential process for attempting to measure the social/civic value of changes to ecological systems and services is to assemble and query a representative group of citizens (a "citizen jury"). The major use of citizen juries to date in environmental decision making has been to help governments rank options for achieving particular goals, e.g., reducing traffic in an urban area (Kenyon, et al. 2001). Citizen juries also can be used to measure the value of changes to ecological systems and services along a variety of different metrics. Information obtained during ranking deliberations, for example, can provide valuable insights for other valuation exercises (Aldred & Jacobs 2000). Citizen juries also have been combined with choice modeling to determine paired rankings of various ecological characteristics (Alvarez-Farizo & Hanley 2006).

Although citizen juries have generally been used to rank governmental options rather than to determine monetary values, citizen juries can also be asked to determine either a social/civic willingness to pay ("public WTP") or a social/civic willingness to accept ("public WTA") for any particular ecological change (Blamey, et al. 2000). For public WTP values, citizen valuation juries can be asked to determine the highest levy, tax, or other form of payment that the government should pay to obtain a particular ecological benefit. For public WTA values, citizen valuation juries can be asked to determine the highest monetary sum that the government should accept to avoid a particular ecological loss.

When asked to determine public WTP or public WTA, citizen juries bear both similarities to and differences from initiatives and referenda and contingent valuations. Like initiatives and referenda, citizen juries provide information on social/civic values, but they measure stated rather than revealed value, and they incorporate elements of the "deliberative valuation" processes described earlier in this section. Citizen valuation juries are also similar to contingent valuation surveys except that: a) juries are asked to determine how much the public should pay or accept in compensation for a specified ecological change (rather than being asked how much they would pay or accept as individuals); b) valuation juries are often asked to agree on a common value for the ecological change (rather than being asked for individual values that the expert then aggregates or otherwise combines); c) juries deliberate together as a group before determining value; and d) juries are provided with more extensive information about the ecological change and can be aided in their deliberations.

Although there is little experience using citizen juries to determine public WTP or public WTA, a number of governmental and academic experiments have examined the appropriate use of citizen juries to inform various governmental choices more generally. The process of forming and utilizing citizen juries has varied widely. In the typical situation, a small group of citizens, typically ranging from a cross-section of 12 to 20 persons, has been drawn from the relevant population. Approaches have differed as to how best to choose the jurors. Given the small size of citizen juries, there is an inevitable tension between choosing jurors to reflect the demographic characteristics of the relevant population as a whole and choosing jurors that represent the interests of major stakeholders. Although larger juries would reduce some of the tensions involved in juror selection, larger juries are likely to find it more difficult to reach agreement within a realistic time frame. Most citizen juries to date have been chosen using random sampling or stratified random sampling (Blamey, et al. 2000).

Once a citizen jury is chosen, the jury then meets and deliberates over a multi-day period, during which it hears and questions expert witnesses, deliberates in small and large groups, and agrees on a final recommendation to the sponsoring governmental body. These group deliberations allow jurors to hear alternative perspectives, test ideas, and carefully work through the valuation exercise. Several different techniques are used to provide information to the jurors. In some cases, the government or an expert facilitator chooses what information to provide to jurors, while in other cases, relevant interest groups make individual presentations to the jury. Jurors also can be permitted to request information and pose questions directly to expert witnesses (Blamey, et al. 2000). Two factors should guide choices among the processes for providing information to the jurors: a) ensuring that jurors have all the information that they believe is valuable to their valuation exercise; and b) ensuring that the information is balanced and not biased toward any particular result. Another important choice in designing a citizen jury is the process by which the jury will make decisions. In most cases, juries are asked to arrive at a group decision. Decision making rules in this context include a simple majority vote of the jury, consensus (where a majority favors the valuation and no juror opposes it), and unanimous agreement. Citizen juries also do not need to produce a collective value. In some experiments, for example, juries deliberate as a group, but members of the jury then report their valuations on an individual basis (Alvarez-Farizo & Hanley 2006). Researchers can then combine individual valuations into an overall evaluation. Measures of central tendency (means or mediums of the valuations provided by the individual jurors) can be used to develop a valuation measure in this context.

Experiments indicate that citizen juries often produce significantly different valuation results from economic or socio-psychological surveys. The additional information available to jury members, the opportunity to spend time thinking about the appropriate valuation, and the stress on collective rather than individual values all appear to generate significant changes in valuation (Alvarez-Farizo & Hanley 2006). The jury’s valuation of particular ecological improvements, however, can either increase or decrease compared to the results obtained through economic surveys (Alvarez-Farizo & Hanley 2006).

Because contingent valuation methodology and other traditional economic measurement approaches seek a very different valuation than citizen valuation juries, juries should not be seen as a substitute for the traditional approaches. Governmental agencies should employ citizen valuation juries as a supplement to traditional economic valuation approaches. When deciding whether to pursue particular regulations or other governmental actions, agencies should consider estimates of both private and public value, along with the strengths and weaknesses of each approach.

EPA might also consider using some elements of the citizen jury approach to improve other valuation methods. Some researchers have investigated other group-based approaches out of concern, for example, about whether contingent valuation surveys provide sufficient time and information for survey respondents to generate reliable estimates of the value of often complex ecological changes. Under the "Market Stall" ("MS") approach, for instance, researchers meet with survey subjects in two one-hour meetings, separated by a week, and encourage the participants to discuss their valuations with household members and friends between the two sessions. Unlike citizen valuation juries, the MS approach asks survey subjects for their personal valuations, based on individual preferences and incomes, rather than social/civic valuation. Respondents are asked for their personal valuations in a confidential written survey at the end of the second meeting. In Macmillan, et al. (2002), the WTP measures obtained through the MS approach were significantly lower than the WTP measures generated from CV interviews, which is consistent with other studies that show a decline in WTP when survey subjects are provided additional time to consider their answers (Whittington, et al. 1992).

Text Box 18: A Valuation Exercise Illustrating Use of Citizen Juries

In one experiment, a citizen jury was used to examine the economic value of the control of a particular exotic weed, Bitou Bush (Chrysanthemoides monilifera L. Norl. ssp. rotundata), in an Australian national park (James & Blamey 2000). A jury of 14 was selected, using a two-phase telephone survey, in order to be representative of the New South Wales population on the basis of gender, age, place of residence, rating of the environment in relation to other social issues, occupation, income, income source, and education. The jury met for three days during which they heard and questioned seven expert witnesses. Prior to the hearings, jurors received training in note taking and questioning of witnesses, in order to maximize their ability to use the information provided.

In one of the charges, the jury was given two options: (Option #1) the then-current situation in which weeds were controlled on 3000 hectares per year, and (Option #4) an alternative management regime in which weed control would be expanded to 9600 hectares per year. The jury was then given the following charge: "How high would a park management levy have to be, before the jury would recommend Option 1 rather than Option 4 …? In other words, how high would the levy have to be before the … public would be no better off under Option 4 than Option 1?" The jury first decided that a progressive levy, calculated as a percent of gross income, was most appropriate. After discussing two proposed levies (0.1% and 0.25%) , the jury voted eight to two in favor of a levy of 0.1%. In a survey following the jury exercise, jurors reported that they found the valuation exercise to be both interesting and worthwhile.

Relation of Method to the C-VPESS Expanded and Integrated Framework

Citizen juries are potentially useful both to identify socially important assessment endpoints and to attach a value, monetary or socio-psychological, to changes in the assessment endpoints. Use of this method relates to steps 3 and 5 of the C-VPESS proposed valuation process (Figure 2).

Because citizen juries consist of representative members of the public, citizen juries also expand the role that the public plays in valuations of changes in ecological systems and services. Members of citizen juries actively evaluate information regarding changes, are permitted to ask questions of experts, and consciously deliberate over the appropriate social/civic value of the change.

Status as a Method

As discussed earlier, citizen juries have been used primarily to help governments rank options for achieving particular goals. Only a few efforts have been made to date to use citizen juries to generate monetary or other estimates of the social/civic value of environmental changes. Use of citizen juries for direct valuation of changes to ecological systems and services, therefore, should be considered experimental for the moment and should not be used to make significant governmental decisions until further research has been conducted on both the efficacy of the process and the appropriate jury processes. Given the potential use of citizen juries to evaluate social/civic values, however, this is an area in which research can be valuably focused. EPA may wish to use citizen juries on an experimental basis, moreover, to provide a comparison to valuations obtained through traditional economic valuation methods.

Strengths/Limitations

One of the major strengths of a citizen valuation jury is that, like referenda and initiatives, the citizen valuation jury incorporates public-regardedness. Jurors are asked to provide a valuation based on the perceived impact of an ecological change on the entire community rather than on his or her individual preferences alone. Citizen valuation juries thus incorporate a broader concept of value than standard contingent valuation approaches and place the jurors in a position similar to that of the governmental decision makers who are being advised.

Citizen valuation juries avoid a number of potential concerns regarding referenda and initiatives as a source of social/civic valuation information. First, the jury process ensures that juries receive more information regarding the ecological change than most voters receive prior to voting on an initiative or referendum. Second, because the jury evaluation process can be carefully structured, citizen evaluation juries are less subject to undue influence from political interest groups than are votes on referenda and initiatives. Finally, there are a limited number of referenda and initiatives from which valuations can be derived, while citizen valuation juries can be asked to assess a valuation for any ecological change. Unlike referenda and initiatives, however, citizen juries do not have standing as actual, official decision making bodies for their communities.

Citizen valuation juries build on a well-established legal institution in the United States – the criminal and civil jury system. The legal system uses juries to decide whether to initiate criminal prosecutions, determine guilt and innocence in criminal cases, decide between life and death in capital cases, and assess damages in often complex civil cases. Most adult members of the public have served as jurors, understand the importance of the role they assume, and act deliberately and responsibly.

Citizen valuation juries suffer from the hypothetical character of all stated-value methods of valuation. Because the juries do not themselves determine governmental policy, the juries may not reveal what they actually believe to be the social/civic value of an ecological change. The hypothetical character of jury valuations could be eliminated by providing that the valuations will directly determine whether particular governmental actions will be taken, but the government is unlikely to want to (or be legally able to) delegate its decision making powers to citizen juries. Despite concerns over hypothetical inquiries, experiments with citizen juries indicate that jurors approach their valuation task in a responsible fashion and reach well-thought-out conclusions (Aldred & Jacobs 2000).

Citizen juries also raise a number of other unique concerns. Some economists, for example, have worried that group dynamics and “norms” might reduce the reliability of jury decisions. Some jurors, for example, might not wish to be perceived as disagreeing with others, while some jurors may be able to dominate the discussion and result. Some jury experiments, however, have suggested that the design of the jury process can avoid such jury dynamics (Macmillan, et al. 2002). Trained facilitators may be able to overcome any structural pathology that might otherwise arise and should be involved in any valuation exercise involving citizen juries.

As discussed earlier, the choice of jurors also poses difficulties. Because of the small size of typical citizen juries, a demographic cross-section of the public may not adequately represent all interest groups. Choosing representatives of different interest groups to serve on citizen juries, however, may yield a jury that does not adequately represent demographics. Small citizen juries, moreover, will inevitably fail to fully represent the public as a whole. In order to ensure that jurors are other-regarded, experiments suggest that the government should choose a jury that is as demographically representative as possible (typically through stratified random sampling), so that the jury is at least symbolically representative, and then instruct the jury to adopt an impartial stance in its deliberations (Brown, et al. 1995, Blamey, et al. 2000).

Treatment of Uncertainty

The use of citizen juries to value changes in ecological systems and services raises many of the same uncertainties as traditional methods of economic or socio-psychological valuation. The small size of citizen juries, however, raises an additional uncertainty factor.

Research Needs

Because there is little experience with the use of citizen juries to directly value changes in ecological systems and services, further research is needed on a variety of topics before EPA should consider adopting the approach to develop social/civic valuations for decision making purposes on other than an experimental basis. Key questions include:

- Do citizen valuation juries arrive at different valuations than individual respondents to CV surveys? If so, how and why do the valuations differ?
- How stable are valuations provided by citizen juries? How much variation exists among the valuations produced by different citizen juries?

- How do jury selection processes affect the valuations of the jury? What methods exist to overcome the inevitable bias arising from the small size of citizen juries?
- How should information be provided to citizen valuation juries? What are the advantages and disadvantages of highly structuring the information that is provided to a jury, versus permitting the jury to determine the information that it receives?
- How do decision making rules (e.g., consensus versus unanimity) affect valuations? What are relevant considerations in choosing among the different decision making rules?

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DELIBERATIVE PROCESSES

Mediated Modeling

Brief description of the method

Computer models of complex systems are frequently used to support decisions concerning environmental problems. To effectively use these models, (i.e., to foster consensus about the appropriateness of their assumptions and results, and thus to promote a high degree of compliance with the policies derived from the models) it is not enough for groups of academic experts to build and run the models. What is required is a different role for modeling - as a tool in building a broad consensus across academic disciplines as well as between science and policy.

Mediated modeling is a process of involving stakeholders (parties interested in or affected by the decisions the model addresses) as active participants in all stages of the modeling, from initial problem scoping to model development, implementation, and use (Costanza and Ruth 1998, van den Belt 2004). Integrated modeling of large systems, from individual companies to industries to entire economies or from watersheds to continental scale systems and ultimately to the global scale, requires input from a very broad range of people. We need to see the modeling process as one that involves not only the technical aspects, but also the sociological aspects involved with using the process to help build consensus about the way the system works and which management options are most effective. This consensus needs to extend across the relevant academic disciplines, the science and policy communities, and the public. Appropriately designed and appropriately used mediated modeling exercises can help to bring these communities together. The process of mediated modeling can help to build mutual understanding, solicit input from a broad range of stakeholder groups, and maintain a substantive dialogue between members of these groups. Mediated modeling and consensus building are also essential components in the process of adaptive management (Gunderson, Holling, and Light 1995, van den Belt 2004).

Example of how the method could be used as part of the C-VPESS expanded and integrated framework

As described, the method is fairly general and could be used to assess any value that a group of stakeholders could identify and build into a model. Any decision context that requires the estimation of the values of ecosystem goods or services could employ this method, although to the committee's knowledge no EPA decisions have as yet employed this technique. The method covers all elements of the diagram representing the C-VPESS framework for valuation after the initial identification of EPA needs, and could be used in conjunction with the full range of decision models. Prior applications have been at a broad range of scales, from watersheds or specific ecosystems to large regions and the global scale. The method is in principle broadly applicable to the full range of time and space scales.

- The method is inherently dynamic.
- The results can be aggregated to get a single benefits number as needed.
- Participants in the mediated modeling process gain deep understanding of the process and products, if the process is done well. Those who have not participated can easily view and understand the results if they invest the effort. Usually the results can (with some additional effort) be made accessible to a broad audience.
- Since the method explicitly discusses and incorporates subjective or "framing" issues, it is at least open and transparent to users.

<u>Status as a method</u>

As mentioned above, mediated models can contain explicit valuation components. In fact, if the goal of the modeling exercise is to consider trade-offs, then valuation of some kind becomes an essential ingredient. How these trade-offs and valuations are incorporated into the model varies, of course, from exercise to exercise. Perhaps the best way to describe this process is with an example. The South African fynbos ecological/economic model described by Higgins, et al. (1997) is an illustrative example.

The area of study for this example was the Cape Floristic Region—one of the world's smallest and, for its size, richest floral kingdoms. This tiny area, occupying a mere 90,000 km2, supports 8,500 plant species, 68 percent of which are endemic (193 endemic genera and six endemic families [Bond and Goldblatt 1984]). Because of the many threats to this region's spectacular flora, it has earned the distinction of being the world's "hottest" hot-spot of biodiversity (Myers 1990).

The predominant vegetation in the Cape Floristic Region is fynbos, a hard-leafed and fire-prone shrubland which grows on the highly infertile soils associated with the ancient, quartzitic mountains (mountain fynbos) and the wind-blown sands of the coastal margin (lowland fynbos) (Cowling 1992). Owing to the prevalent climate of cool, wet winters and warm, dry summers, fynbos is superficially similar to California chaparral and other Mediterranean climate shrublands of the world (Hobbs, Richardson, and Davis 1995). Fynbos landscapes are extremely rich in plant species (the Cape Peninsula has 2,554 species in 470 km2) and plant species endemism ranks amongst the highest in the world (Cowling 1992).

In order to adequately manage these ecosystems, several questions had to be answered including: what services do these species-rich fynbos ecosystems provide and what is their value to society? A two-week workshop was held at the University of Cape Town (UCT) with a group of faculty and students from different disciplines along with parks managers, business people, and environmentalists. The primary goal of the workshop was to produce a series of consensus-based research papers that critically assessed the practical and theoretical issues surrounding ecosystem valuation as well as assessing the value of services derived by local and regional communities from fynbos systems.

To achieve these goals, an 'atelier' (or combined workshop/short course) approach was used to form multidisciplinary, multicultural teams, breaking down the traditional hierarchical approach to problem solving. Open space (Rao 1994) techniques were used to identify critical questions and allow participants to form working groups to tackle those questions. Open space meetings are loosely organized efforts that give all participants an opportunity to raise issues and participate in finding solutions.

The working groups of this workshop met several times during the first week of the course and almost continuously during the second week. The groups convened together periodically to hear updates of group projects and to offer feedback to other groups. Some group members floated to other groups at times to offer specific knowledge or technical advice.

Despite some initial misgivings on the part of the group, the structure of the course was remarkably successful, and by the end of the two weeks, seven working groups had worked feverishly to draft papers. These papers were eventually published as a special issue of

Ecological Economics (Cowling and Costanza 1997). One group focused on producing an initial scoping (or mediated) model of the fynbos. This modeling group produced perhaps the most developed and easiest-to-implement product from the workshop: a general dynamic model integrating ecological and economic processes in fynbos ecosystems (Higgins, et al. 1997). The model was developed in STELLA and designed to assess potential values of ecosystem services given ecosystem controls, management options, and feedbacks within and between the ecosystem and human sectors. The model helped to address questions about how the ecosystem services provided by the fynbos ecosystem at both a local and international scale are influenced by alien invasion and management strategies. The model consists of five interactive sub-models: a) hydrology; b) fire; c) plants; d) management; and (e) economic valuation. Parameter estimates for each sub-model were either derived from the published literature or established by workshop participants and consultants (they are described in detail in Higgins, et al. 1997). The plant sub-model included both native and alien plants. Simulation of the model produced a realistic description of alien plant invasions and their impacts on river flow and runoff.

This model drew in part on the findings of the other working groups and incorporates a broad range of research by workshop participants. Benefits and costs of management scenarios were addressed by estimating values for harvested products, tourism, water yield, and biodiversity. Costs included direct management costs and indirect costs. The model showed that the ecosystem services derived from the Western Cape Mountains are far more valuable when vegetated by fynbos than by alien trees (a result consistent with other studies in North America and the Canary Islands). The difference in water production alone was sufficient to favor spending significant amounts of money to maintain fynbos in mountain catchments.

The model was designed to be user-friendly and interactive, allowing the user to set such features as area of alien clearing, fire management strategy, levels of wildflower harvesting, and park visitation rates. The model has proven to be a valuable tool in demonstrating to decision makers the benefits of investing now in tackling the alien plant problem, since delays have serious cost implications. Parks managers have implemented many of the recommendations flowing from the model.

There are several other case studies in the literature of various applications of mediated modeling to environmental decision making, including valuation. Van den Belt (2004) is the best recent summary and synthesis. Some additional examples of mediated modeling projects where ecosystem service values were integrated are:

- Participatory Energy Planning in Vermont, Department of Public Service in Vermont, http://www.publicservice.vermont.gov/planning/mediatedmodeling.html
- Mediated Modeling of the impacts of Enhanced UV-B Radiation on Ecosystem Services (van den Belt, et al. 2006)
- Ria Formosa Coastal Wetlands (a case study in van den Belt 2004)
- Upper Fox River Basin (a case study in van den Belt 2004)
- A consensus-based simulation model for management of the Patagonian coastal zone (van den Belt, et al. 1998)

Models can be downloaded from: www.mediated-modeling.com

Strengths/Limitations

Resources needed to implement the method vary from application to application. The method can deal with a broad range of available data and resources, probably better that most other methods, since the model can adapt to the resources available across different levels of data, detail, scope and complexity. As a rule of thumb, one can produce a credible mediated model in 30-40 hours of workshops, requiring about 300-400 hours of organizing/modeling. Cost: about $40,000 - $100,000 depending on side activities.

The most serious obstacle seems to be the fact that this method is very different from the top-down approach most frequently used in government. It requires that consensus-building be put at the center of the process, which can be very scary for institutions accustomed to controlling the outcome of decision processes. An institutional mandate is important, however, to motivate various stakeholders to volunteer their time, knowledge, and energy to a mediated modeling process. The final outcome of this process cannot be predetermined.

Treatment of Uncertainty

In terms of uncertainty, there are all the usual sources, but the difference is that the stakeholders are exposed to these sources as they go, and learn to understand and accommodate them as part of the process. The method is compatible with formal or informal characterizing of uncertainty, producing probability distributions in addition to point estimates.

Research needs

No research has yet been done on whether application of the process to exactly the same problem by multiple independent groups would yield "consistent and invariant" results. One would expect general consistency, but some variation between applications. This is an area for further research.

To evaluate the impact of a mediated modeling process, surveys have been used before and after a process in the past and this research would deepen the understanding about exactly what elements of a mediated modeling process contribute to the success or failure of these processes.

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Valuation by Decision Aiding

Decision aiding approaches provide a method for valuing protection of ecological systems and services in terms of multiple attributes. These approaches are deliberative in nature, rely upon insights drawn from the discipline of decision analysis, and are based on research and practical findings from applications of decision-aiding approaches (Arvai & Gregory 2003a, Arvai, et al. 2001, Gregory, et al. 2001a, Gregory, et al. 2001b). Decision-aiding approaches consider value to be a product of a two-step process.

The first part of the process assists people in determining value based on a careful and comprehensive analysis of the suite of attributes that characterize ecological systems and services. For example, people may determine the value of an estuary based on multiple, ecologically-based attributes such as the degree to which it provides nutrient exchange, the re-supply of dissolved oxygen to near-shore habitat, or nursery habitat for anadromous fish species. Similarly, the value of the estuary will also be affected by a wide range of attributes that reflect economic or social interests, such as the degree to which it provides access to commercially important species, opportunities for recreation, and lanes for shipping traffic. Decision-aiding approaches consider both types of attributes.

The second aspect of these decision-aiding approaches focuses on helping people form judgments about the value of ecological systems and services by way of a comparative framework. From a prospective standpoint, decision-aiding approaches help people to evaluate competing alternatives, determining, for example, which option in a range of environmental, risk, or resource management options is most likely to lead to a preferred suite of outcomes. In other words, this approach helps people determine which option, in a set, is most valuable (i.e., is Option A in a set of alternatives better or more valuable to decision makers than Option B?). The value of ecological systems and services can also be determined retrospectively by comparing attributes associated with ecosystem health or the provision of ecological services that have been realized today with those that were realized at some point in the past (i.e., is the system being evaluated “better off”—or more valuable—today, at Time 2, than it was in the past, at Time 1?). Alternatively, value can be determined in a spatial comparison by evaluating the attributes associated with ecosystem health or the provision of ecological services in an area of interest relative to those that have been realized elsewhere (i.e., is System A more valuable than System B?).

It is important to note that valuation by decision aiding does not provide an estimate of how valuable ecological systems and services are. For example, this method cannot provide a specific estimate, which would state that a system today is X times more valuable than it was in the past, or that System A is Y times more valuable than System B. The concept, which is adapted from a framework for making choices among options, is ideally suited to providing a relative ranking of value or importance such as when EPA may wish to prioritize systems for management action.

In the important first step of valuation by decision-aiding process, one or more analysts facilitate the characterization of the ecological system (or systems) that is to be the focus of analysis. This step in this process entails identifying the relevant attributes of the ecological system, that is, all aspects of a system that are of interest or concern to people. The goal at this stage is to develop an explicit, comprehensive picture of all factors that contribute significantly to the overall value of the system in question. Diverse groups of stakeholders and relevant experts should be consulted to identify the attributes that will ultimately guide the analysis. These stakeholders are defined in an operational sense as groups of people who, for any reason—e.g., place of residence, occupation, favored activities—have legitimate concerns or opinions regarding the health of an environmental system. Careful selection of stakeholder groups ensures that the full range of views is adequately covered. For example, the representatives of an environmental advocacy organization might be expected to present a somewhat different list of attributes than would representatives of industry or government, but the views of each group are likely to encompass those of many other citizens.

In addition to consulting the broad spectrum of interested or affected stakeholders, an analyst should also consult with technical experts (e.g., ecologists, toxicologists, economists, behavioral scientists, etc.) as part of an interdisciplinary, analytic-deliberative process (Environmental Protection Agency 2000, National Research Council 1996) designed to identify both the relevant attributes of the system in question as well as the specific means by which each attribute can be measured (see Text Box 19: Types of Attributes).

Text Box 19: **Types of Attributes**

Previous work (Keeney 1992, Keeney & Gregory 2005) has led to an operational typology of attribute to inform their selection in a given valuation context. Generally speaking, attributes that help to define the different aspects of a system fall into one of three categories:

- Natural attributes – these are direct measure conditions that exist in a system. For example, if one attribute of an environmental system being evaluated is the economic value of a commercially important species (e.g., fish or trees), then the specific value of this attribute can be expressed directly in dollars. Likewise, if an attribute of a system is the number of individuals of a key indicator species living in it, then a straightforward count of these individuals represents another direct measure of value.
- Proxy attributes - these, by contrast, are used when it is not possible to directly measure an attribute of interest. For example, if one attribute of an environmental system is the recreational opportunities that it provides to tourists, economists may—by proxy—estimate, using the travel cost method, the recreational value of the resource. Similarly, a particular mudflat may be valued from an ecological standpoint because of the migratory shorebirds that it attracts. However, it is frequently the case that accurate, direct counts of shorebirds, which would a be natural attribute, are impossible to achieve. In these cases, an analyst may rely upon the amount of habitat that is available as a reasonable proxy for the number of shorebirds that may use the mudflat over the course of a season.
- Constructed attributes – these are most often used when neither a direct, natural attribute nor a reasonable proxy attribute exists. Proxy attributes are typically used to operationalize objectives that are psychophysical in nature (e.g., the objective to improve the aesthetic quality of a shoreline). Scales that may be administered during surveys often need to be constructed—e.g., by psychologists or sociologists—as a means of characterizing these attributes.

In the second step of this process, data or information about each of the identified attributes must be collected by those familiar with how to conduct the individual valuation methods (e.g., ecological, economic, psychosocial, etc.) discussed elsewhere in this report. This information must be collected at the site of primary interest as well as at other sites that will provide the basis for comparison. Alternatively, contemporary data at a site of interest must be collected and compared with archived information about previous conditions described by the same attributes at the site.

All this information, which describes both the attributes of an ecological system and specific information to be used as the basis for making comparisons (e.g., data describing conditions at another site or the same site at an earlier time), can be displayed visually in a matrix (Table 9). It is unlikely, except in very rare circumstances, that comparisons made apparent by this matrix will reveal improvements (or, on the other hand, declines) in the values associated with all of the attributes; in most cases, the comparison will reveal that improvements have been realized across some attributes while declines have occurred across others. In the hypothetical estuary described on the previous page, for example, it is not uncommon for improvements in the system's capacity for nutrient exchange to come at the expense of opportunities for recreation or industry.

	Option			Site			Time	
	A	B		A	B		1	2
Attribute 1								
Attribute 2			or			or		
Attribute 3								
Attribute n								

Table 9: Comparative Matrices of Attributes for Three Hypothetical Decision-Aiding Valuation Scenarios

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These differences necessitate the need for trade-offs—the third step in a valuation by decision-aiding process—across the attributes to determine if, on aggregate (1) a site, System A, is more valuable than another, System B, or (2) the system being evaluated, again System A, is more valuable today than it was in the past (Table 1). A detailed overview of specific methods for addressing these trade-offs, such as swing-weights (e.g., see Clemen 1996) or even swaps (e.g., see Keeney 1992), are beyond the scope of this discussion. However, these and other methods can be used by individuals or in deliberating groups to place weights on the various attributes, and in turn, to use these weights to develop an understanding of the overall, multi-attribute value associated with an environmental system of interest. In other words, despite the fact that conditions described by certain attributes may have improved while others may have declined, formal trade-off analysis across these attributes can help individuals or groups decide if conditions on the whole at a site are better or worse—i.e., have higher or lower value—relative to the reference condition.

Thus far, this discussion has not focused on the situation where people may wish to establish the multi-attribute value of an environmental system absent a comparative framework for trade-off analysis. Carrying out this kind of assessment is possible and requires that, in lieu of a comparison, individuals or deliberative groups translate the information obtained for each attribute (e.g., inputs in dollars for attributes that require monetization, constructed scales for attributes measured using psychosocial methods, etc.) into common terms.

Suppose, for example, that EPA wished to construct a value for the damage resulting from a specific pollutant accidentally spilled into a waterway. Technical experts working alongside stakeholders could be engaged in a process to both identify the relevant attributes of the system and provide information describing the conditions in the waterway as they relate to these attributes both before and after the insult to the system. For example, the physical event of the death of a large number of fish might imply not only an ecological loss, but also aesthetic (e.g., when the dead fish wash up on shore) and economic (e.g., the loss of commercial fishing jobs and profits) losses. Clearly, a host of other attributes would also need to be considered.

After the attributes have been identified and the quantitative information that describes them collected, deliberation and argument can be organized with the intent of deriving a single metric (e.g., dollars or units of ecological productivity) that can be used to capture information about all of the attributes. For example, the techniques of multi-attribute utility theory (Keeney & Raiffa 1993) can be used to construct a single "value" that encompasses the diverse array of attributes (Gregory, et al. 1993). EPA could then conclude that the value of the system in question is X. However, EPA may be required to repeat this procedure at other sites to determine, in relative terms, how significant this value (of X) is.

<u>Status of the Method</u>

Past studies and applications of this approach have focused primarily on group decision-making contexts where there is a need to evaluate a range of management options and select the one that seems like it will perform the best across the attributes judged by decision makers to be most important. The method has been applied in experimental studies in which people have been asked to evaluate its effectiveness across a range of criteria that include the self-ratings of decision makers and measures of internal consistency (i.e., the degree to which the approach helps people make choices that reflect their weighting of attributes) in choice (Arvai & Gregory 2003a, Arvai, et al. 2001). The method has also been applied in a variety of practical contexts, including the setting of a national energy policy in Germany (Keeney, et al.

1990), provincial water use planning in Canada (McDaniels, et al. 1999), and the management of a protected estuary (Gregory & Wellman 2001).

The goal of this discussion, however, is not to provide guidance about how EPA should make decisions; such advice falls outside the charge of this committee. Instead, the goal is to highlight how these methods, which decompose complex decision problems and help people carefully evaluate an option or range of options, may also be used for valuing the benefits of ecological systems and services. Because decision-aiding methods are designed to help people to evaluate and then rank options, they may also be used to evaluate an environmental system across a range of attributes and make judgments about its value relative to other systems, or indeed the same system at a previous point in time. The method may also be combined with insights from multi-attribute utility theory to construct a single, uni-metric "value" that encompasses the diverse array of attributes.

Strengths/Limitations

The strength of this method rests in its ability to not only integrate multiple attributes value, but also engage a broad spectrum of stakeholders, holders of traditional ecological or cultural knowledge, and technical experts in the valuation process. In doing so, the method has a high potential for identifying changes in ecosystems and their services that are likely to be of greatest concern to people. Moreover, by engaging this broad spectrum of people, there is a greater likelihood that the valuation process will include attributes that wouldn't normally be included by EPA, as well as those that may not easily be addressed by more traditional valuation approaches. Thus, this method may potentially overcome (primarily) public or stakeholder objections to other approaches that are not perceived to adequately include moral and other non-monetary aspects of value.

It is important to note, however, that the trade-offs, which are an important part of this process, are typically not easy to make. But, because they are not holistic judgments that require the simultaneous integration of the various attributes, the likelihood that people will fail to consider important attributes is low. Moreover, despite the effort that is required from those who use these methods, past experience suggests that the outcomes are both more easily understood by people, and met with higher levels of support and ratings of defensibility when compared with unstructured or unimetric approaches (Arvai 2003, Arvai & Gregory 2003b, Arvai, et al. 2001).

As with many of the methods discussed in this report, this one requires that resources—time and expertise—be devoted to implementing it. Engaging with stakeholders and technical experts to identify attributes that will be the focus of analysis, collecting data that characterizes these attributes, and the process of making trade-offs all will require effort on the part of EPA.

Research Needs

As the primary focus of this method has been on providing decision support, its usefulness—particularly to potential users of the method—as a complement to other valuation methods is unclear. For example, one wonders about its usefulness, in the context of many EPA applications such as benefits assessment as mandated by OMB. Other questions can be raised about the effect of facilitation on the process as one cannot guarantee that repeated applications of the process will produce the same outcomes. This question is not unique to decision aiding, however, as a variety of factors (e.g., contextual, temporal, and spatial differences) may adversely affect other valuation methods as well.

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METHODS USING COST AS A PROXY FOR VALUE

Cost as a proxy for value, including replacement cost, tradable emissions permits, and habitat equivalency analysis (HEA), are a distinct category of methods that use information about the cost of alternative means of providing the same quantity and quality of ecosystem services to infer the value of protecting one particular means of providing the ecosystem services. However, because costs and values are two distinct notions, great care needs to be taken in the application of these methods and in the interpretation of results using these methods.

Replacement Costs

Brief description of the method

This method, also called avoided cost, uses the cost of replacing ecosystem services with a human-engineered system as an estimate of the value of providing ecosystem services via protection of an ecosystem. For example, an estimate of the value of conserving an ecosystem that serves as a watershed that naturally provides clean drinking water could be derived by estimating the cost of building a water filtration plant that would provide the same quantity and quality of water. Replacement cost is exactly what it says: the cost of replacing an ecosystem service via some other means. Replacement cost is not a measure of the value of the ecosystem services themselves. Rather, it is the value of having one particular means of providing ecosystem services, and therefore not having to pay to replace services via some other means. Also, the replacement cost method should not be confused with applications of "averting behavior" based upon observed voluntary behavior on individuals (see revealed preference methods).

Status as a method

The method has been used to provide estimates of the value of protecting watersheds for the purpose of providing clean drinking water (NRC 2004). The most famous of such cases, and the example of valuing ecosystem services that is cited probably more than any other, is the case of protecting the Catskills watersheds that provide drinking water for New York City (Chichilnisky and Heal 1998, NRC 2000, 2004). New York City, faced with the possibility of being required by EPA to build a water filtration plant for water from the Catskills, opted to invest in greater watershed protection in the Catskills. New York City and EPA signed a Watershed Memorandum Agreement in 1997 that allowed New York City to pursue a watershed protection plan in lieu of building filtration. While commonly cited as a classic case of the value of protecting ecosystems, this case is not without controversy. It is not clear that protecting watersheds will ultimately be successful in maintaining drinking water quality, or that the protection of watersheds versus building a filtration plant will provide equivalent water quality in all dimensions (NRC 2004). Further, some analysts have suggested that the threat of building the filtration plant had more to do with government regulations than with real water quality issues (Sagoff 2005).

Another example using a replacement cost approach is the avoided cost of illness approach that EPA has used successfully to account for certain human health benefits of environmental regulations.

Strengths/Limitations

Replacement cost can be a valid measure of value if three conditions are met: a) the human-engineered system provides services of equivalent quality and magnitude; b) the human-engineered system is the least costly alternative; and c) individuals in aggregate would be willing to incur these costs rather than forego the service (Bockstael, et al. 2000, Shabman and Batie 1978). If these conditions are not met, then use of replacement cost is invalid. Even when these conditions are met, replacement cost, rather than being a value of ecosystem services themselves, is the value of having a means to produce the service via an ecosystem instead of through an alternative human-engineered system.

All valuation methods can be applied incorrectly and misinterpreted, but the replacement cost method requires special caution. Because there is great potential for abuse in using replacement costs to estimate the value of ecosystem services, it should be used with care. The loss of an ecosystem service does not necessarily mean that the public would be willing to pay for the least cost alternative. Similarly, a regulatory constraint requiring replacement in the event of loss of ecosystem service also does not guarantee that the public would be willing to pay to replace the service. If the value of the service does not exceed the cost of alternative means of providing the equivalent set of services, then use of replacement cost is invalid. Even when the benefits of the service exceed the least cost method of providing the service, replacement cost does not measure the willingness to pay for an environmental improvement or the avoidance of harm. It merely represents the value (avoided cost) of not having to provide the service via human engineered approaches. Still, if there are alternative ways of producing the same service, and if that service would be demanded if provided at the least cost human-engineered alternative method, then replacement cost is a valid measure of the change in value from loss of the service provided by the ecosystem.

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Tradable Permits

In the case of tradable permits, there are no conditions under which the cost of permits could be used as a proxy for economic value.

Emissions permit trading has been allowed under the Clean Air Act since the 1990 Amendments. Under a cap-and-trade system, such as that used by EPA to reduce sulfur dioxide emissions, the regulatory body determines the total number of permits available and some means of allocating permits among regulated sources. A regulated source must ensure that it has sufficient permits to cover its activities, or it will face penalties. In the example of tradable emissions permits, a regulated source can take actions to reduce its own emissions and/or purchase permits from other sources. For those firms with higher marginal cost of pollution control, cost savings can occur if they purchase emissions-reduction credits from firms with lower pollution control costs. Similarly, firms with relatively low pollution control costs can profit by undertaking greater abatement and selling extra permits. In so doing, trading can reduce overall costs of compliance. Tradable permits schemes have been proposed in fisheries management in the form of individual transferable quotas (ITQs), and in land conservation in the form of transferable development rights (TDRs).

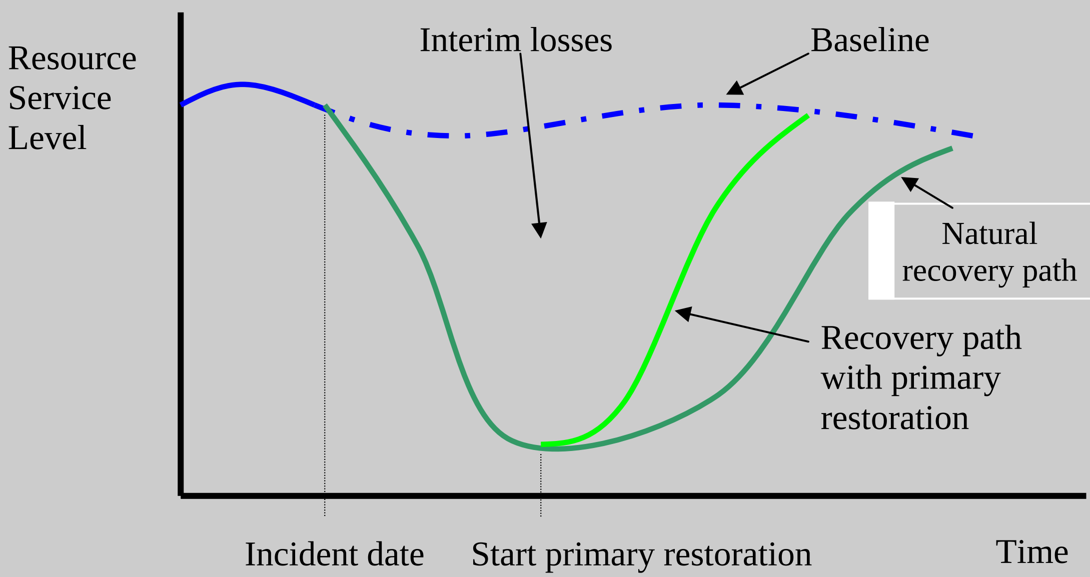
It has been suggested that the price of a tradable permit is a proxy for the economic value of provision of environmental quality or conservation. However, this confuses the notion of costs and benefits. In market equilibrium, the price of a tradable permit is equal to the marginal cost of supplying a unit of environmental quality or conservation covered by the permit. Permit price need not bear any relation to benefit of environmental quality or conservation. If there are a large number of permits issued relative to demand for permits then permit price will be low; with few permits, price will be high. This does not necessarily mean that the value of environmental quality or conservation is low (or high). Permit price only reflects value if price equals the marginal benefit of environmental improvement or conservation, which occurs only if the number of permits issued is such that marginal costs and marginal benefits are equal. But issuing the right number of permits to get marginal cost equal to marginal benefit requires knowing marginal benefit in the first place. There is no way to be confident that tradable permit prices reflect value without already knowing value. In other words, tradable permit prices do not constitute a valuation methodology capable of generating information about values.

Habitat Equivalency Analysis

Brief description of the method

Habitat Equivalency Analysis (HEA) is an analytical framework originally developed to calculate compensation for loss of ecological services resulting from injury to a natural resource over a specific interval of time (King and Adler 1991, NOAA 1995). Figure 10 provides a graphic representation of the relationship between the interim lost from an environmental incident or activity and the recovery of the environment over time both due to natural mechanisms and from primary restoration actions.

Figure 10: Graphical Representation of Ecosystem Service Loss and Recovery through Natural and Active Restoration Over Time



Essentially, HEA calculates the amount (e.g. acres, hectares) of habitat to be created or enhanced to replace an equivalent level of ecological services over time as were lost due to the injury. The basic HEA formula is shown in Text Box 20. Ultimately the HEA approach is not a valuation method but rather more appropriately defined as a "cost-replacement" method. Yet it is important to recognize that an implicit operational assumption for an HEA is that the quantity of ecological service flows, and their as yet undefined value, associated with any given unit of lost or injured habitat are equivalent (same type and comparative value) to a unit of the proposed replacement habitat.

Text Box 20: Equation for Habitat Equivalency Analysis

$$\underbrace{\sum_{t=t_0}^{t_l} L_t (1+i)^{(P-t)}}_{\text{“Debit: PDV Loss”}} = \underbrace{\sum_{S=S_0}^{s_l} R_s (1+i)^{(P-s)}}_{\text{“Credit: PDV Gain”}}$$

where

L_t =	lost services at time t
R_s	replacement services at time s
t_0	t m when lost services are first
t_l	time when lost services are last
s_0	time when replacement services are first
s_l	time when replacement services are last
P	present time when the natural resource damage claim is
i	periodic discount

There are two main steps in a HEA which are accomplished simultaneously: a) quantifying the injury, and b) scaling the size of restoration to compensate for the lost service over time due to that injury. To be clear, injury is not determined in a HEA, but such a determination of injury is a necessary pre-step to provide the input for scaling the restoration to match the degree of injury. The HEA approach focuses on scaling replacement costs on a service-to-service basis. Therefore in quantitative expressions HEA relies on biophysical units such as acres of habitat as a surrogate of service, and calculates the increase in habitat over time in service acre years. A similar methodology, Resource Equivalency Analysis (REA) focuses on scaling replacement costs on a resource-to-resource approach. In this context, resources are generally defined in terms of biotic type and mass (e.g., kilograms of fish) for the quantification of injury, but often ultimately revert back to an estimate of habitat required to replace or generate those lost resources in estimating the size and type of replacement actions required to restore the environment. HEA can also handle injuries to biotic resources but needs to equate those resource losses to the unit of habitat it would take to create or support that mass of birds, fish, and invertebrates in the first place. Those performing an HEA will thus need to be careful in this translation to avoid the potential for double counting if they are estimating habitat needs for species which are supported by a common habitat such as coastal wetlands.

Temporal assumptions are very important in working with HEA, especially in a damage assessment. Questions such as the following need to be answered or estimated:

- How long has the injury or lost service been in place?
- How much time is required to implement the restoration project?
- How long will the restoration project take before it reaches full replacement service?

Obviously, the answers to these questions can have a significant impact on the estimated compensatory value required to offset the injury. In HEA, a discount rate must be selected for the Net Present Value calculations.

There are some crucial assumptions associated with the HEA method. It can be used only when values per unit of replacement services and lost services are comparable, when it is possible to use a common metric to define an injury and the value of replacement services, and when replacement of ecological services is feasible and measurable.

Since HEA is a restoration/compensation method that is projected into the future, the final unit is a Net Present Value (NPV) measure of the services in the future stated in discounted terms (e.g., Discounted Service-Acre Years or DSAYs). Discounting or scaling of the equivalency of any given sets of injured or restored habitat is required since the resource types that are being addressed are not static over time (NOAA 1999). Injured resources can recover to baseline conditions on their own and planted habitat takes time to develop to full maturity. So factors such as baseline conditions and recovery times become key opportunities for uncertainty in any HEA. Additionally for HEA to operate effectively it must fully explore and determine that capacity of any project or suite of projects to achieve the required level of restoration. To accomplish this assurance step, in advance of an HEA, a process referred to as C.O.P.E. was developed (King 1997). The acronym C.O.P.E. stands for the attributes desired in the HEA, which are: a) Capacity to provide service; b) Opportunity for project(s) in the correct location; c) Payoff of comparable services; and d) Equity to provide service to people in the location that suffered the injury. Each restoration project must satisfy the presumptions of C.O.P.E. to be worth further quantification via HEA as a contribution to satisfy the needed service years equivalent to the lost interim service.

Example of how the method could be used as part of the C-VPESS expanded & integrated framework

The spatial scale at which HEA has typically operated has been at the level of local to regional decisions. Therefore it is not reasonable in its current state of development for HEA to be considered as a tool useful for creating input to national rule-making. HEA also operates over past and future time scales, involving compensation for injury or estimate service produced by past action, as well as allowing time for restoration projects to mature to full ecosystem service capacity.

With regard to where to place HEA in the C-VPESS integrated framework, it would seem to bridge a number of the process elements. Although it would not be fair to say that it is currently applied in a manner that would be classed as characterizing value, it does provide a framing for characterizing bio-physical change. The HEA methodology relies on structural or spatial measures of ecological components such as acres of habitat. Specific service categories such as provisioning, regulating, cultural, and supporting services as expressed in the Millennium Ecosystem Assessment framework (2005) are not identified or expressed but would be considered to be present and operating But if the type of habitat or resources can, with further research, be equated to a unitized measure of values or service flows, either monetary or otherwise, then HEA could be used to scale that associated value over time and across alternative actions. If, through research and development, service flows and associated values can be quantified for given habitat categories (e.g., an acre of coastal wetlands in Louisiana), then there is some hope that HEA may evolve to be a support for valuation.

Additionally, although HEA and REA are currently used in the post-hoc context of injury, damages, and compensation, there is no reason that these methods are constrained to managing adverse outcomes after the fact. They could just as easily be used ex ante to compare alternative future actions to identify the action with the least impact and to compare alternative actions to identify which will yield the most service or equal service in the shortest time frame. These methods or variations could be a fruitful avenue for the Agency to explore through its research and development activities.

As noted, HEA is a tool that has application constraints. Typically, the HEA is applied to support local decisions by scientific experts to evaluate project alternatives for achieving restoration objectives. Such analyses allow those experts to arrive at convincing trades among restoration options. Although there is not much evidence to indicate the use of HEA in support of a facilitated or mediated process that includes the general public, there do not appear to be any technical reasons why this could not be a useful application of HEA to project the services provided by possible alternative future scenarios resulting from a suite of restorations actions. Such engagement of the public in the identification of restoration projects and desired services is likely to leadto more widely accepted restoration decisions.

Status as a method

The HEA approach was originally developed in 1992 to quantify damages associated with contaminated wetlands (King and Adler 1991, Malcolm v. National Gypsum 1993 as referenced in Unsworth and Bishop 1993) and has since been applied to cover injuries due to chronic contamination, spills, and vessel groundings in a variety of habitats (Chapman, et al. 1998, Fonseca, et al. 2000, Milon and Dodge 2001, NOAA 2001). HEA is currently used in Natural Resource Damages Assessment (NRDA) under Oil Pollution Action (OPA) and CERCLA (Superfund). The purpose of NRD actions is to make the public's interest whole for injuries to natural resources that result from the release of hazardous substances or oil. It is important to note that restoration for damages is distinct from remediation activities.

Interestingly, under these two regulatory frameworks there is a different focus on compensation. Under Superfund actions, compensation for damages is focused on monetary compensation, which requires restoration of service ultimately to be converted to replacement costs in dollars. Under OPA, the focus is on replacement of resources to achieve compensation. The question is how much in the way of new public resources does the public require to be made whole for their loss. Therefore, value is scaled from resource or habitat lost to resource or habitat replaced. As noted previously, there are no barriers to applying these methods in proactive support of decisions. Therefore the Agency should explore such proactive applications of HEA and REA in other regulatory contexts and especially in collaborative partnerships with conservation as a focus.

Strengths/Limitations

The HEA method can be used as a way to scale surrogates measures (e.g. acres of Habitat or mass of fish) of non-market services often overlooked by other valuation methods when the specific assumptions associated with HEA can be met. The method is not complicated mathematically. It is by nature inter-disciplinary because determination of comparability per unit of replacement services and lost services requires collaboration between ecologists and economists.

Since HEA and REA are currently applied to support regulatory actions which link to a litigation process, to define compensation the analysis and supporting data need to be legally defensible with regards to analytical quality. The chief analytical difficulty is to determine defensible input parameters, especially an appropriate metric for lost and restored services and related time functions for recovery and development to maturity.

The HEA method is not appropriate for standard benefit-cost analysis, where the goal is to determine optimal (efficient) allocation of scarce resources. The cost of compensatory restoration projects should not be communicated as the benefit of the resources to the public.

Treatment of Uncertainty

Uncertainty can be, and should be, directly incorporated into any HEA analysis. Addressing uncertainty in inputs (e.g., percent service lost per unit of habitat and recovery time) can be effectively done. Tracking the effects of uncertainty on HEA outputs can be easily performed. One of the benefits of HEA is the transparency of the method. Sensitivity and uncertainty analysis can be directly incorporated into a HEA evaluation and the resulting change can be tracked in outputs (see NOAA 1999 for more details)

Research needs

There are a number of key areas for research and development that the Agency should explore in connection with HEA.

The Agency should look at HEA for its applications in contexts other than Natural Resource Damage Assessment. In particular, it should consider its utility tandem with Net Environmental Benefit Analysis (Efroymson, et al. 2004) in the selection of best alternatives for project investment.

The Agency should consider research to develop a more complete understanding of the service flows and the associated values of goods and services derived from those flows in specific important habitat types (e.g., coastal wetlands, bottomland hardwood forests). Such value definitions for ecosystem service could then be coupled to HEA to estimate values associated with a project or restoration action.

EPA should consider developing operating principles for considering on-site, in-kind changes in resources and ecological services, as compared with off-site and out-of-kind resources. In support of this objective, methods to assess and compare ecological capacity and the opportunity and payoff for restoration in the evaluation and design of restoration projects will also strengthen the method to assess comparability of ecological resources.

Finally, this method will be strengthened if the Agency develops guidance on the appropriate aggregation and accounting of services related to biotic resources and their supporting habitats in order to advance the utility of HEA to support local and regional valuation efforts.

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APPENDIX C: 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 December 13-15, 2005 Workshop, "Science for Valuation of EPA's Ecological Protection Decisions and Programs." 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.

Defining Survey Research

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).[46] Conducting a survey involves (1) drawing a sample from a population, (2) collecting data from the elements in that sample, and (3) 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.[47] Surveys are usually done for one of three reasons: (1) to document the prevalence of some characteristic in a population, (2) to compare the prevalence of some characteristic across subgroups in a population, and/or (3) 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.

Designs of Surveys

Surveys can take on a variety of designs, which are suitable for addressing different 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, although 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 & Kenny, 1986; James & 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 & 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 & Alwin, 1989). Panel surveys can also be used to test causal hypotheses, 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 & Greenberg, 1981, on the methods; see Rahn, et al., 1994, for an example).

Panel surveys also face a number of challenges. Respondent attrition (or "panel mortality") occurs when some of the people who provide data during the first wave of interviewing are unreachable or refuse 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 on panel attrition suggests that panel attrition minimally affects sample composition (Becketti, et al., 1988; Clinton, 2001; Falaris & Peters, 1998; Fitzgerald, et al., 1998a; 1998b; Price & Zaller, 1993; Rahn, et al., 1994; Traugott, 1990; Zabel, 1998; Zagorsky & Rhoton, 1999; and Ziliak & 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 on these issues is reassuring for the most part. A number of studies have found either no evidence of panel conditioning effects or very small effects (Clinton, 2001; Cordell & Rahmel, 1962; Himmelfarb & Norris, 1987; Sobol,1959; Willson & 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 & Holmberg, 1992; Kraut & McConahay, 1973; Willson & 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 & 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. If, for example, a researcher is interested in conducting a 2-wave panel survey but is concerned about conditioning effects, she could also administer the wave 2 panel questionnaire to an independent cross-sectional sample drawn from the same population at the time of the second wave. Differences between the data collected from the two wave 2 samples would suggest that carry-over effects were, in fact, 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

Introduction. High-quality, scientific surveys typically provide respondents with several key pieces of information when introducing the survey, whether it is through an introductory mailed 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, and it may help to increase the perceived validity of the survey and, as a result, respondent participation. This information includes information about the sponsor of the survey, a brief description of the topic of the survey, and how the data from the survey will be used. The introduction should also include a reassurance to respondents that their survey responses will be kept confidential and a description of 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 provide 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, as longer introductions place a greater burden on respondents and may also 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. A large body of relevant scientific studies about the questionnaire design decisions faced by researchers has now accumulated, and when taken together, their findings clearly suggest how to design 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 & Fabrigar, forthcoming). For bipolar scales, which 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 & 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 5 points long (e.g., Wikman & Warneryd, 1990).[48]

A number of studies show that data quality is better when all points on a rating scale are labeled with words than when only some are labeled thusly (e.g., Krosnick & Berent, 1993). Researchers should try to select labels that have meanings that divide up the continuum into approximately equal units (e.g., Klockars & 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 & Warner, 1968).[49]

Researchers must then decide how to order the response alternatives, and people's answers to rating scale questions are sometimes influenced by this order. After reading the stem of most rating scale questions, respondents are likely to begin to formulate a judgment. For example, the question, "How effective do you think the clean-up 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, thus resulting 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 therefore usually best to rotate the order of response choices across respondents and to statistically control for that rotation when analyzing the data.[50]

Pretesting. 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 & 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).

Mode of Data Collection

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 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), which does not yield results that can be generalized to the population of interest (Malhotra & 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 & 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 & 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, it is necessary to assess accuracy (or conversely error) in survey data. 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. According to the "total survey error" approach, a research can think about survey design issues within a cost-benefit framework geared toward helping researchers make design decisions that maximize data quality within budget constraints (cf. Dillman, 1978; Fowler, 1988; Groves, 1989; Hansen & 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* have previously been described. *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 ever home) 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 & 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 & 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 potential respondents' answering machines, 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% 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% and 65%) 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 non-response 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 which half of 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 form 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 a researcher spends to reduce one type of error, the less money he or she has 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 biggest bang for the buck spent.

Challenges in Using Surveys For Ecosystem Protection Valuation

Introduction. One application of the survey method is for 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 cost-benefit analyses by EPA in the policy-making process. When monetary values are not required or 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 ecosystems/services conditions with which they may not be familiar prior to the survey, but this does not mean that respondents lack a value for the ecosystem in question. Respondents' experiences have cumulated into beliefs and attitudes stored in long term memory that are the ingredients of their orientations toward objects they will encounter in the future. 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. It is important that the survey provides all of 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 begin by conducting 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 great deal of information can also be presented to respondents in a single visual display that a respondent can read or an interviewer can explain to the respondent. Information can also be presented in the narrative form of a story, for example, by telling respondents that they'll be told about:a) the state of an ecosystem as it used to exist 50 years ago; b) changes that have occurred to the ecosystem in the intervening years; c) the causes of those changes; d) what could be done to reverse those changes; and e) how this could be implemented. Rather than lecturing respondents for a long time period, a questionnaire can maintain respondent engagement by presenting information in small chunks, separated by questions allowing respondents to react briefly to the information they've been given (e.g., "Had you ever heard of the Golden River before today?"). Respondents can also be asked periodically to verbalize any information that they'd like to have as the story progresses, to allow them to express their cognitive responses to the presentation.

The choice of survey mode also impacts the presentation of information about an ecosystem. Face-to-face interviewing is optimal because it allows visual displays of any type 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 aural 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 large set of pages of paper may be intimidating or dispiriting, thus, minimizing respondent motivation and provoking superficial processing of the information. The self-administered mode may be the least desirable for this reason. For all modes, it is important to pretest the final instrument to be sure it's 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 8.1), 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 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% 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% probability of protecting the ecosystem). There is substantial evidence that people have difficulty the last type of evidence accurately (e.g., Tversky & 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 impact 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 impact 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 impact 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 transformations 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 literatures 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 these literatures. 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: a) representative sampling of the population of interest with minimal non-coverage error; b) a very high response rate or a plan to assess the presence of non-response bias; c) a measuring instrument that has been developed according to optimal design and pretesting practices; and d) 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 & 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: a) exploring the impact of response rates on data accuracy, and b) 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. So 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. Yet NOAA's Blue Ribbon Panel concluded that CV is a viable method of valuation. 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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1 **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 called 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 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 rulemakings 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 26-26, 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 also committee 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 public teleconferences on(insert additional dates).

[5] Likewise, this definition would not include goods or services like recreation that are produced by combining ecological inputs or outputs with conventional inputs (such as labor, capital, or time). In addition, Boyd and Banzhaf 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).

[6]There is controversy over the meaning of intrinsic value (Korsgaard, C. (1996). Two Distinctions in Goodness. Creating the Kingdom of Ends. C. Korsgaard. Cambridge, Cambridge University Press. 1996: 249-74. 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, J. (1985). Values and Secondary Qualities. Morality and Objectivity. T. Honderich, Routledge and Kegan Paul: 110-29., Sturgeon, N. (1985). Moral Explanations. Morality, Reason, and Truth. D. C. a. D. Zimmerman, Rowman and Allenheld: 49-78; Sayre-McCord, G. (1988). The Many Moral Realisms. Essays on Moral Realism. G. Sayre-McCord. Ithaca, Cornell University Press: 1-26;, Brink, D. O. (1989). Moral Realism and the Foundation of Ethics. Cambridge, Cambridge University Press.

[7] 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.

[8] Under GPRA, the Office of Management and Budget (OMB) 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 (U.S. Environmental Protection Agency 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.

[9] 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.

[10] NCEE is typically brought in by the program offices to both help 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.

[11] 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" (add page number).

[12] 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 2002) and a briefing provided by the EPA Office of Water staff. During the June meeting, members of the Committee divided into two workgroups. The workgroups each worked independently and reported their findings to the combined Committee. The leaders of the two working groups then prepared a consolidated summary of comments from the two workgroups.

[13] 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.

[14] 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.

[15] 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.

[16] 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.

[17] 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".

[18] 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.

[19] 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.

[20] In the case of this CAFO rule, 97% of the monetized benefits arise from recreation (boating, swimming and fishing) and from private well owners' willingness to pay for water quality, estimated using contingent valuation or travel cost methods.

[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] This could include either a robust public involvement process following Administrative Procedures Act requirements (e.g., FR publication), or some other public involvement

process [see EPA's public involvement policy, U.S. Environmental Protection Agency Office of Policy, E. a. I. (2003). *Public Involvement Policy of the U.S. Environmental Protection Agency*. EPA 233-B-03-002.; the SAB report on science and stakeholder involvement U.S. Environmental Protection Agency Science Advisory Board (2001). *Improved Science-Based Environmental Stakeholder Processes: An EPA Science Advisory Board Commentary*. EPA-SAB-EC-COM-001-006.

[26] Typically production functions in economics have been studied in the context of businesses that purchase inputs and sell outputs in markets.

[27] Models may be valuable in many of the steps of assessing ecological value including: estimating stress loading; estimating the exposure pattern of stress – especially spatial and temporal implication; identifying ecological element(s) receiving exposure; estimating exposure - response function of ecological elements; estimating the reduction or prevention of increased stress from agency action; estimating the response of service production or function to change in stress; valuating the ecological service associated with that change in production; linking to economic or socio-political for further valuation in dollars or other metrics

[28] 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, since 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.

[29] 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 (Young and Sanzone 2002).

[30] One issue relates to the assumption regarding functional form used in the original analysis. To illustrate the role of functional form, these estimates are interpreted as measuring the marginal willingness to pay for small improvements in air quality in Chicago. In these contexts, examples of economic benefits transfer would involve adapting the estimated marginal willingness to pay (MWTP) for air quality in Chicago so it could be used for another city, such as Cleveland, New York City, or Los Angeles. If the hedonic price function used in the Chicago study were linear, the estimated coefficient for the measure of air quality would be the estimate of the MWTP, which would be constrained to be constant by the use of the linear price function. In this case, the only adjustment that would be needed would be for the year of the Chicago study in relationship to the year the analysis sought to measure the MWTP. Alternatively, if the Chicago analysis used a nonlinear price function, the MWTP would not be constant and could not be determined solely by the estimated coefficient of the hedonic price function; rather, the MWTP estimate is itself a function of variables in the hedonic price function that might be assumed to influence how changes in air quality affect housing prices. In this case, the adjustment that is needed to conduct an economic benefit transfer might involve using different values for air quality and other

determinants of the MWTP that would be associated with the city being studied. It is important to note that this adjustment is based on an estimate of the MWTP at a particular point. In particular, it does not assume that the original study estimated a complete relationship between all air quality levels and housing prices (i.e., a complete marginal willingness-to-pay function). Estimation of the complete relationship requires added information. (For a discussion of the distinction between an estimate of MWTP that varies with other factors versus an estimate of the MWTP function, see Palmquist, 2005).

[31] These examples are taken from Ready et al. (2004).

[32] An unpublished analysis and peer review of the methods has been developed as part of the rulemaking process.

[33] This complex question reverses the logic used in the conventional analytical framework used to define an economic benefit measure. An economic benefit measure specifies something an individual would give up to obtain more of something else. In most applications, income is the commodity exchanged for a change in some other factor that is constraining an individual's choices. To assure the definition is complete requires specifying values for all the other factors that constrain the individual's choices as well as the level of income and the level of the factor to be relaxed prior to any change.

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

[35] The discussion of value in the National Research Council report (2001) and SAB review of the EPA's Draft *Report on the Environment* (US 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 (U.S. EPA SAB 2005) emphasize the importance of using regional and local indicators. Oover-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 (U.S. EPA SAB 2003)

[36] 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. In the prospective analyses the sequence of increasingly stringent rules called for under the 1990 Clean Air Act Amendments are compared with a situation where the rules were held constant at their 1990 levels (e.g. with the regulatory regime prior to the amendments).

[37] A syndrome has been identified that involves: increased biomass of phytoplankton, shifts in phytoplankton to bloom-forming species that may be toxic, in marine environments, increases of gelatinous zooplankton, increases in biomass of benthic and epiphytic algae, changes in macrophytic species composition, decreases in water transparency, oxygen depletion, increased incidence of fish kills, and loss of desirable fish species (Carpenter et al., 1998). There are a number of important features of this syndrome. It is easily recognized, it

is reversible, and there are some features that show up early and hence provide indicators of ecosystem disruption and early opportunities for mitigation. Clean water and recreational opportunities have been extensively treated in valuation projects. The impacts on the biological nature of a system may not be readily appreciated or valued by the public, but it certainly provides an indicator that the things they do value are in trouble. The power of public involvement in understanding, valuing and responding to eutrophication is shown by the classic example from Lake Washington (Smith, 1998). The understanding part took considerable efforts in educating the public by those few scientists who understood what was happening.

[38] A number of the gasses emitted from CAFOs have adverse air quality impacts that are interrelated with the water quality impacts.

[39] The pollutants that result from CAFOs have environmental effects that are local, regional and global. For example, in terms of emitted gases, methane and N20 are major greenhouse gasses of global concern; ammonia and nitrogen oxides have important regional impacts on air quality and nitrogen deposition; and odor and suspended particulate matter have important local or on-site impacts (NRC,2003)

[40] In the case of air, nitrous oxide has a lifetime of 100 years in the planetary boundary layer, whereas hydrogen sulfide has a lifetime of only about a day. These spatial and temporal dimensions of dispersion and lifetime of effects also apply to many of the water pollutants although the spatial dimensions do not extend to the global.

[41] CAFOs are not uniformly distributed in the country or even within a state. For various reasons they often are clustered. Each of these concentration areas has unique climatic, soil and topographic features that influence waste dispersion. Further, manure type, in addition to soil characteristics, has a differential impact on soil microbial populations and hence on decomposition rates (Larkin 2006).

[42] The animal feed used at CAFOs no longer comes from local surroundings but may be produced in areas remote from the sink facilities, including foreign sources. The production of these grain feeds results in non-point pollution in the production regions. Further, fish meal is an important feed supplement for pigs and chickens with the fish generally being harvested from coastal and marine ecosystems, often from places far distant from the United States, with consequences for local food chains.

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Table summarizing 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 infrastructure valuation calculator	Copyright 2004-2007	http://greenvalues.cnt.org/calculator

[44] Consumer surplus measures the excess of the sum of the marginal values over the expenditures that must be made to obtain the good at a fixed price. Thus, consumer surplus sums up the differences between the maximum a consumer would be willing to pay for a good minus the amount actually paid (price) for each unit consumed. Similarly, producer surplus measures the excess of receipts for the good over the sum of the marginal costs to provide each unit. Producer surplus is then a comparable concept. It aggregates the difference between what producers are willing to sell a product for (supply) and what they actually receive (price) for each unit they provide. Adding together changes in consumer surplus and producer surplus generates the change in total economic benefit.

[45] The last component of these costs, the cost of time on site per visit, is difficult to include because it is reasonable to assume it is jointly determined with decisions about the location to visit and the number of trips to take in a season . It is also related to measures of the amount of the site's services that are consumed. Most studies acknowledge these costs as an issue but don't include them in the analysis as a result of these difficulties. As a rule the time on site per trip is assumed to be held constant.

[46] 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 Table 1 for examples of federal funded surveys).

Table 1: 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	

[47] 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).

[48] Presenting a 7-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 7-point scale, and measurement reliability and validity are higher as well (Krosnick & Berent, 1993).

[49] 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 & 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, 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 answering a comparable rating question in one's mind first. For example, respondents 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 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 & Ross, 1991; Ruch & DeGraff, 1926; Wesman, 1946).

[50] 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 & 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.