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PART 1: OVERVIEW

1. INTRODUCTION

<u>EPA's Mission Regarding Ecosystem Protection.</u> The Environmental Protection Agency's (EPA's) mission is to protect human health and the environment. During its history, the EPA has focused decision making and much of its 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 the EPA's traditional expertise, the broad mission of the EPA goes beyond this. In fact, EPA's Strategic Plan explicitly identifies the need to ensure "healthy communities and ecosystems" as one of its five major goals (U.S Environmental Protection Agency 2006) and 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 attention to ecological systems to ensure the wise and thoughtful use and protection of our environment. An "ecosystem" is the term used by ecologists to describe living organisms plus their physical environment working together. For example, a forest ecosystem is comprised of the trees in the forest plus the birds, insects, soil micro-organisms, 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.

Given the important 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. Many EPA actions (e.g., regulations, rules, programs, policy decisions) affect the condition of the environment and the flow of ecological services from it. EPA actions can lead to improvement or deterioration of ecosystems or prevent degradation that would otherwise have occurred. These impacts can occur both at a relatively small, local scale as well as more broadly at a national scale.

Despite their importance, to date, ecological impacts have received relatively limited consideration in EPA policy analyses. EPA's ecological analysis has generally focused primarily on ecological endpoints such as those identified by tests required for pesticide regulation (e.g., effects on survival, growth, and reproduction of aquatic invertebrates, fish, birds, mammals, and both terrestrial and aquatic plants) or mortality to fish, birds, and plants and, more generally, animals, wildlife, aquatic life, as required by provisions of several laws[1] administered by the Agency (U.S. Environmental Protection Agency Risk Assessment Forum 2003). However, given EPA's responsibility to ensure healthy communities and ecosystems, the Agency's actions must encompass the key structural and functional characteristics of communities and ecosystems, not simply endpoints related to impacts on individual organisms or impacts on plant and animal populations. Failure to consider ecological impacts as fully as possible can lead to distorted policy decisions. This can occur, for example, when actions are evaluated based primarily on their impacts on human health, without adequate recognition of potentially important ecosystem impacts.

In addition to its mission to protect ecosystems, EPA also seeks to evaluate policy options and make policy decisions with a recognition of the tradeoffs that are inevitably involved. To promote good decision-making, policy makers require information about how ecosystems contribute to society's well-being and how those contributions are valued. 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 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, there is a gap between the need for protection of 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 for valuation of ecological systems and services can help the Agency describe and measure the value of protecting ecological systems and services and hence better meet 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). The SAB saw a need to complement the Agency's ongoing work in ecological science, ecological risk assessment, and ecological benefit assessment by offering advice on how EPA might better value the protection of ecological systems and services and how that information might better support decision making to protect ecological resources. Toward this end, it formed C-VPESS,[2] an interdisciplinary group of experts from the following areas: decision science, ecology, economics, engineering, philosophy, psychology, and social sciences with emphasis on ecosystem protection.[3] C-VPESS began its work in 2003 on a project designed to strengthen the Agency's analysis for protecting ecological resources. The purpose of science advice on ecological valuation is to strengthen the Agency's knowledge and set of analytical tools to help navigate difficult trade-offs that inevitably arise when regulatory or other decisions must be made to protect ecological resources. In this project the SAB set the goals of: a) assessing Agency needs and the state of the art and science of valuing protection of ecological systems and services and b) identifying key areas for improving knowledge, methodologies, practice, and research at EPA.

Scope of report and intended audience. This report provides advice for strengthening the Agency's approaches for valuing the protection of ecological systems and services, facilitating their use by decision makers, and identifying the key research areas needed to strengthen the science base.[4] It focuses on 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 rulemaking, regional decision making, and programs in general that protect ecological systems and services. It outlines a call for EPA to expand and integrate its approach in important ways.

This report appears at a time when there is lively interest internationally, nationally, and at EPA itself in the issue of valuing the protection of ecological systems and services. Since the establishment of the SAB C-VPESS major reports have been developed by others focusing on how 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 2006). This report discusses in length past and current EPA efforts to improve ecological valuation (see Appendix B), which have focused on economic valuation for use in benefit-cost analysis. EPA has also 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's work has benefited from and has built upon these recent efforts. The C-VPESS distinguishes its work from those efforts, however, in the following ways. First, the C-VPESS focuses on EPA as an audience for its work. In particular, it focuses on how 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 focused on economic valuation. In contrast, C-VPESS 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 and address how the Agency could better represent the value of ecological protection.

2. AN OVERVIEW OF KEY CONCEPTS

2.1. The Concept of Ecosystem Services

The term "ecosystem" describes a dynamic complex of plant, animal, and microorganism communities and the non-living environment, interacting as a unit. For example, ecosystems can describe organism-physical environment interactions in a woodlot, a watershed, or a larger landscape. Ecosystems encompass all organisms within the prescribed area, including humans, who are often the dominant element. 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 (U.S Environmental Protection Agency 2004). These 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., such as pollination, predation and parasitism). These processes in total describe the functioning of ecosystems.

"Ecosystem services" is an anthropocentric concept denoting the contributions that ecosystems make to people either directly or indirectly. In popular terminology these contributions are sometimes labeled the "benefits" that humans derive from ecosystems.[5] The following operational categorization of ecosystem services has recently been proposed by the Millennium Ecosystem Assessment:

a) Provisioning services (products obtained from ecosystems). These include food, fuelwood, fiber, biochemicals, genetic resources and fresh water. Generally these services are traded in the open marketplace.

b) Regulating services (benefits received from regulation of ecosystem processes). This category includes a host of pathways that stem from the presence and functioning of ecosystems and influence people in positive ways, both direct and indirect. These 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 and this value will increase for many of these services as the many dimensions of global change proceed.

c) Cultural services (the nonmaterial benefits people obtain from ecosystems). Ecosystems 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. These are the processes that maintain ecosystem functioning such as: soil formation, primary productivity, biogeochemistry, and provisioning of habitat. They all affect human well-being, but generally indirectly through their support of the provisioning, regulating and cultural service functions.

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 need to recognize the myriad ways in which ecosystems support human life and contribute to human well-being.[6] Alternatively, ecosystem services can be defined more narrowly to include only end-point services that contribute directly to well-being. (See Part 2 of this report for a more detailed discussion of the implications of using a broad vs. narrower definition.)

Whether broadly or narrowly defined, the concept of ecosystem services provides an approach to evaluating the many ways in which ecological changes induced by human actions affect human well-being. However, ecosystems can be valued for reasons that are independent of human well-being. As discussed below, the committee recognizes that ecosystems can be important not only because of the services they provide directly or indirectly but also for other non-anthropocentric reasons, including respect for nature based on moral, religious, or spiritual beliefs and commitments.

2.2. The Concept of Value

Because only people define values, all values are anthropogenic. However, there is disagreement about whether the ultimate goal of all values is only to promote human well-being, so there is disagreement about whether all values are anthropocentric. When people talk about environmental values, the values of nature, or the values of ecological systems and services, they may have different things in mind. People have moral, economic, 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.

The most basic philosophical distinction in values is the distinction between means and ends. To value something as a means is to value it for its usefulness in helping to realize or bring about some thing or state of affairs that is valued in its own right or as an end. Things valued for their usefulness as means in this sense are said to have instrumental value. Alternatively, something can be valued for its own sake or as an end. Things valued as ends are sometimes said to have intrinsic value. [7] If intrinsic value applies to things other than human beings or human experiences, then this conception of value is non-anthropocentric. Some people defend a non-anthropocentric conception of value or goodness (Goodpaster 1978; Taylor 1986; Rolston III 1991). However, others argue that only human beings or human experiences have intrinsic value, thereby defending an anthropocentric conception of intrinsic value (Sidgwick 1901; Glover 1984; Williams 1994).

Some people also claim that the very “existence” of a species or ecological system has value in addition to any instrumental value derived from the usefulness of the services it provides. This claim can mean several different things. If it means that the existence of an ecological system is valuable because people derive satisfaction from its existence independent of specific uses they may make of its services, then it has what economists call “existence value.” This concept is anthropocentric. In addition, this value can been viewed as a kind 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 generate human satisfaction, and that the various things that contribute to human satisfaction are potentially substitutable or commensurable. In contrast, some people claim that an ecological system may have intrinsic value of its own, and hence its existence is valuable for its own sake. If the explanation of this claim refers to reasons that are independent of the contribution that the existence of the ecological system can make to human well-being, then as noted above this is a claim that the ecosystem or one or more of its components has a non-anthropocentric intrinsic value.

This committee recognizes that there are many possible sources of value derived from ecosystems and the services they provide. To reflect this, throughout this report, the term "value" is used broadly to include 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. (See Table 1 for a summary of the usage of key terms throughout this report.)

As distinct from the broader concept of value, in this report we use the term "benefit" to refer more narrowly to the contribution of ecosystems and their services to human well-being. As such, benefits include only anthropocentric sources of value. However, as defined here, the term "benefits" includes both the economic/utilitarian concept of benefits based on individuals' preferences and benefits based on communitarian or constructed preferences. In addition, throughout the report benefits are defined relative to changes in the state of an ecosystem or the flow of services it provides stemming from an actual or proposed action by EPA. Thus, the term "ecosystem benefits" refers to the positive contribution to human well-being of a change in an ecological system and/or its services. A negative contribution, for example from damages to an ecosystem, can be viewed as a "negative benefit" or cost. Similarly, the term "valuation" will refer to the process of estimating or measuring either the value of, or the value of a change in, an ecosystem, its components, or the services it provides.

Finally, throughout the report the concept of value or benefit refers to social value or benefit of a given change, which could differ from the value or benefit to a private party such as a firm. For example, allowing a firm to emit one more unit of a pollutant can have a positive value or benefit for that firm (equal to the savings in abatement costs) but this private value is not the social value of that increase (which might very well be negative, i.e., entail a net social cost).

Table 1: Usage of Terms

For purposes of this report, the following terms are used as indicated:

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

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 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 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.

Valuation: The process of characterizing, estimating or measuring either the value of, or the value of a change in, an ecosystem, its components, or the services it provides.

Valuation Method: A methodology, based on theory and data, for estimating or measuring the value of, or the value of a change in, an ecosystem, its components or the services it provides.

Monetary Valuation: Valuation in which estimates are expressed in monetary units.

Non-monetary Valuation: Valuation in which estimates are expressed in non-monetary terms.

Benefits: The contribution of ecosystems and their services to human well-being and satisfaction.

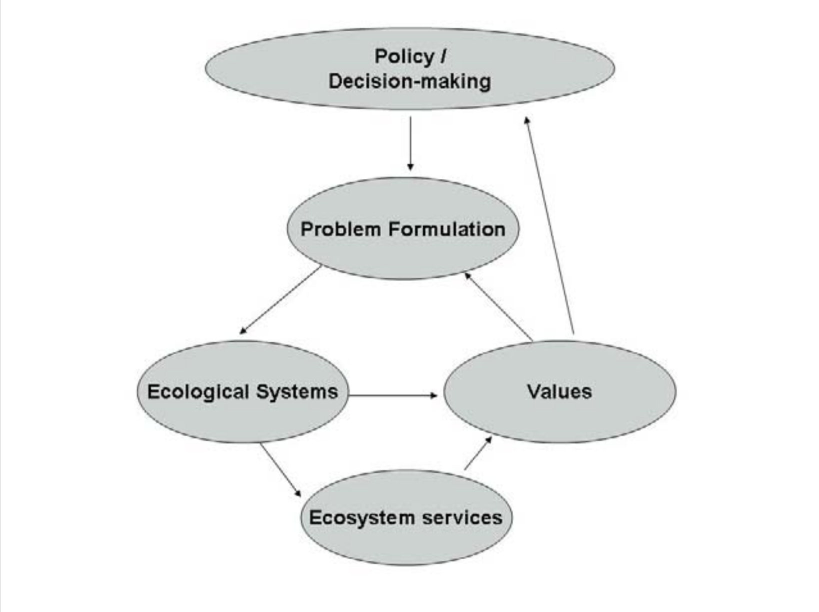
Economic Valuation Methods: Methods that estimate the tradeoffs individuals are willing to make for improvements in, or to avoid degradation of, an ecosystem, its components, or the services it provides. These approaches typically focus on the amount of money an individual is willing to forgo or pay to enjoy a particular positive change (willingness-to-pay) or the amount of monetary compensation a person would accept in lieu of receiving that change (willingness to accept). This includes benefits derived from both use and non-use values.

Social-Psychological Valuation Methods: Methods that focus on individuals' judgments of the relative importance of, acceptance of or preferences for changes in ecosystems, their components, or the services they provide, typically focusing 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, or civic or ethical/moral obligations relevant to ecosystems and ecosystem services.

2.3. The Concept of Ecological Valuation

"Valuation" is the process of estimating or measuring either the value of, or the value of a change in, an ecosystem, its components, or the services it provides. The committee is focusing on valuation in EPA contexts where there is an environmental protection decision to be made. The major components of ecological valuation are depicted in Figure 1.

Figure 1: Components of Ecological Valuation



Some of the components of ecological valuation parallel components of the ecological risk framework that underlies the ecological risk guidelines developed by EPA to support decision making to protect ecological resources (U.S. Environmental Protection Agency Risk Assessment Forum 1992; U.S. Environmental Protection Agency Risk Assessment Forum 1998). The committee views ecological valuation as a complement to ecological risk assessment. Both begin with an EPA decision or policy context for which information about ecological effects is needed. This leads to a formulation of the problem and identification of the purpose and objectives of the analysis and the policy options that will be considered. In addition, both ecological risk assessment and ecological valuation involve prediction and estimation of possible ecological effects of the EPA action or decision that is under consideration, and ultimately the use of this (and related) information in the evaluation of alternative decisions or policy options.

However, 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, locations, etc., but do not provide information about the societal importance or significance of these effects. In contrast, as depicted in Figure 1, ecological valuation takes the predicted ecological effects and seeks to characterize their importance to society by providing information on the value society places on the ecological improvements or the loss they experience from ecological degradation. As shown in Figure 1, these values can reflect either changes in the flow of services provided by the ecosystem or values that are attached directly to the ecosystem itself that are independent of its contribution to human well-being. 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.

There are a number of methods that can be used for estimating or measuring values. In some cases, these methods lead to estimates that are expressed in monetary units ("monetary valuation"), while in other cases estimates are expressed in non-monetary terms ("non-monetary valuation"). In addition, these methods differ in their focus and, in some cases, their underlying premises. For example, economic valuation measures benefits by estimating the tradeoffs individuals are willing to make for ecological improvements or to avoid ecological degradation. These approaches typically (but not always) estimate benefits, including existence values, in monetary units that measure the amount of money an individual is willing to pay to enjoy a particular positive change (willingness-to-pay) or the amount of monetary compensation a person would accept in lieu of receiving that change (willingness to accept) given the individual's current income and the current prices of other goods and services that can be purchased with that income.[8]. Alternatively, social-psychological methods for valuation focus on individuals' judgments of the relative importance of, acceptance of, or preferences for ecological changes. These approaches typically focus on choices or ratings among sets of alternative policies, and may include comparisons with potentially competing social and economic goals. Individuals making the judgments may respond on their own behalf or on behalf of others (society at large or specified sub-groups) and the basis for judgments may be changes in individual well-being, or civic or ethical/moral obligations relevant to ecosystems and ecosystem services. Similarly, assessment methods based on voting or other group expressions of social/civic values provide information about human values revealed through these processes.

In some cases, the output of a valuation process will be a single metric of the value of a particular ecosystem or ecological change, while in other cases the process will yield multiple metrics of value. As noted above, the estimated values may reflect multiple sources of value from multiple interested parties. Valuation methods that seek to aggregate all of these components of value into a single metric, such as economic valuation, weight these various sources of value as part of the valuation process and report estimated aggregate values that reflect these weights. In contrast, valuation processes based on multi-metric approaches, such as multi-attribute utility, do not seek to aggregate sources of value; rather, they report the information about the various components of value separately and allow decision-makers to supply the weights to be attached to these components. 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, then a valuation approach that yields a single metric will be needed. In contrast, in a decision context where the decision makers themselves are charged with appropriately balancing competing interests, a multi-metric approach will be required.

2.4. Some Caveats Regarding Valuation

A basic tenet of valuation as defined in this report is that it seeks information about the value of protecting ecological systems and services as expressed by lay individuals within society. The presumption is that, in a democratic society, the values held by individuals within that society should be considered in public policy decisions and that public involvement can aid a democratic government (e.g., Berelson, 1952). The involvement of citizens in decisions about their future environments and what would best serve their individual and collective well-being is presumably a basic tenant of democratic societies. In addition, while under this premise everyone's values count, values held, expressed and advocated by larger numbers of people should carry more weight in public policy.

However, some believe that, for complex problems such as ecosystem protection, majority values or values held by the public are not an appropriate basis for public policy decisions. Concerns about basing policy decisions on values expressed by individuals stem from at least two sources. The first is a view that the preferences that people express are not well-formed or stable and are easily subject to (intentional or unintentional) manipulation (see a detailed discussion in Appendix A). This suggests that some preferences are "constructive" and that expressed attitudes and preferences can be changed if the judgment an individual is asked to make is presented in a different way. For example, studies have shown that responses to surveys sometimes exhibit instability in the form of response choice order effects, question order effects, question framing effects and interviewer effects. Some believe that these effects provide evidence of fundamental changes in the preferences and values themselves resulting from the specific interview context. However, this does not necessarily imply that opinions expressed in surveys provide no information about people's values. Most of these observed effects, although statistically significant in some studies, have been quite small. In addition, many studies that seem to suggest that people's opinions are uncrystallized and easily manipulated have been shown to have had problems with the research design, such as problematic choice of the participant population involved or the setting in which the data were collected. Nonetheless, for contexts that are very unfamiliar, initial responses to survey questions regarding attitudes and values may not reflect well-formed preferences. For example, individuals can have strongly held values that are not coded mentally in terms of dollars. Asking them to express these values in monetary equivalents such as willingness to pay may be asking them to make connections that in their own minds are not clear or do not exist. Thus, if not carefully conducted, valuation studies based on techniques such as surveys could yield misleading representations of the public's preferences.

The second source of concern about the use of values from the public in policy choice relates to the quantity and quality of information individuals have when expressing their values. In principle, public policy decisions should consider all of the benefits associated with alternative options. However, for complex issues such as ecosystem protection, individuals may not be aware of or fully understand all of these benefits. For example, although the public might understand the recreational benefits associated with a given EPA action to limit nutrient pollution, they might not recognize or fully understand the associated nutrient cycling benefits. As a result, the values they express either through survey responses or through their behavior will reflect that incomplete information. Some argue that if members of the public are in fact not well informed, perhaps they cannot offer thoughtful opinions or make rational choices about related policy matters, so allowing public influence on decisions bearing on those issues would be irresponsible (de Tocqueville, 1835; Schumpeter, 1950).

Two possible responses to concerns about the use of public expressions of value exist. The first is to rely instead solely on the advice of experts (e.g., ecologists, biologists, toxicologists) when determining ecosystem and ecosystem service protection policies. In some cases, there may be high levels of agreement among experts about the bio-physical outcomes of proposed policies, and even about the implications of those outcomes for individual and social well-being. However, when expert judgments on these matters are incongruent with the beliefs, preferences or intentions of the public, it is not always clear what the necessary normative principles are for basing decisions on the views of experts, or who should decide among competing principles. An alternative approach is to determine what information, deliberations or other interventions might be needed to achieve better congruence of expert and public opinion and incorporate these into the valuation process. This would require that valuations that elicit public expressions of value present all relevant information to the public so that the expressed values can reflect the current state of scientific knowledge. Valuation methods that employ deliberative methods can address this concern by providing the relevant information throughout the deliberative process and ensuring that it is understood by those expressing values.

The discussion above suggests that policy makers using values expressed by the public as an input into decisionmaking should be cognizant of the above concerns and interpret the value estimates accordingly. *Ceteris paribus,* policy-makers should put more weight on measures of public preferences that are based on well-informed and thoughtful expressions of value. Additionally, EPA should consider taking direct steps to assess the level of understanding brought to issues that have complex policy and scientific implications and the implications for valuation, particularly where there are concerns about the public's understanding of the issues addressed by the Agency. [9]

3. ECOLOGICAL VALUATION AT EPA

There are several different contexts in which EPA policy decisions result in ecological impacts and hence in which the need for ecological valuation will arise. In addition, EPA operates within a set of institutional, legal, organizational and practical constraints that affect this process at the Agency. Thus, EPA has specific needs in this regard that must be recognized and addressed. These needs arise 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.

3.1. Policy Contexts at EPA Where Ecological Valuation Can be Important

There are at least three policy contexts in which information about the value of ecological systems and services could be very useful to EPA: a) national rule-making; b) regional decision-making; and c) local assessment and evaluation.

Benefit assessments are required for national rulemaking by two of EPA's governing statutes (the Toxic Substances Control Act and the Federal Insecticide, Fungicide and Rodenticide Act) and by Executive Orders 12866 and EO 13422 for "significant regulatory actions." The 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 such "economically significant rules." One of these elements is an evaluation of the benefits and costs of a proposed regulatory action and the main alternatives identified. The circular explicitly defines benefits using the economic/utilitarian concept of willingness to pay (or willingness to accept).[10] The circular contains general guidance on how to provide monetized, quantitative, and qualitative information to characterize benefits as fully as possible. EPA itself has developed broad guidance for ecological benefit assessment (U.S. Environmental Protection Agency 2000) and an *Ecological Benefits Assessment Strategic Plan* (EBASP) (U.S. Environmental Protection Agency 2006) with the goal "to help improve Agency decision-making by enhancing EPA's ability to identify, quantify, and estimate the value of the ecological benefits of existing and proposed policies." In developing the *EBASP*, EPA identified the need for improved models and methods to help implement the requirements of the circular as they relate to ecological valuation. The Agency identified the need both to expand methods and data for economic valuation and to explore other assessment methods to provide information on ecological effects that are currently not quantified or monetized and assigned an implicit value of $0. Managers seek approaches that are "sound, credible, and scientifically supportable" as well as flexible, affordable, and able to be implemented within the time constraints required by rulemaking (U.S. Environmental Protection Agency Science Advisory Board 2004 –check this quote…..).

EPA's regional offices, although generally not responsible for national rule-making, are responsible for several kinds of regional and local decisions and activities where the benefits of ecological protection are potentially important. These include:

- Priority setting for regional action, such as targeting projects for wetland restoration and enhancement or identifying critical ecosystems or ecological resources for regional attention;
- Setting Supplemental Environmental Protection (SEPs) 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 (RCRA) cleanups that could take ecological benefits into account;
- Review of Environmental Impact Statements prepared by other federal agencies to comply with the National Environmental Protection Act;
- Assisting state and local governments and other federal Agencies with protecting lands and land uses, where assessment of the value of protection options could help decision-makers make better-informed decisions, 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.

Regions also seek low-cost methods that can be implemented quickly to inform site-specific decisions. They seek methods that provide information on the value of ecological services; ecological diversity; conservation opportunities and threats; sustainability; and historical and cultural values associated with ecological systems or parts of ecosystems at the watershed or landscape scale. Regions experience the need to communicate the value of ecological protection as they collaborate with other federal agencies and with government partners at the local, state, and regional levels.

The need to assess the ecological benefits of policy options arises in most of the Agency's decisions, including the assessment of ecological protection programs. EPA's need to assess the value of its ecological protection programs has two dimensions: 1) a retrospective dimension, because assessments focus on the value of EPA's current and past protection efforts, and 2) a prospective dimension, because such assessments are meant to inform decisions about future EPA programs and priorities. Program assessments are mandated for EPA, as they are for all agencies of the executive branch, by the Government Performance and Results Act (GPRA) of 1993. As part of that assessment, 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 benefits 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 the ecological benefits associated with the goals in its strategic plan.

In addition, GPRA established requirements for assessing the effectiveness of federal programs. Part of that assessment involves assessing 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 benefits and to evaluate their effectiveness in delivering meaningful, ambitious program outcomes. Characterizing ecological benefits associated with EPA programs is a necessary part of the program assessment process.

Although ecological valuation can be an important part of program assessment, this report focuses on the use of valuation to inform policy decisions relating to national rule-making and regional and local priorities and activities. Nonetheless, the committee believes that the methods and issues discussed throughout the report can be used to improve the evaluation of EPA programs that protect ecosystems by demonstrating and, where possible, estimating the ecological benefits derived from those programs.

3.2. Institutional and Other Issues Affecting Valuation at EPA

The committee recognizes that ecological valuation at EPA must be conducted within a set of institutional, legal, organizational, and practical constraints that affect what is and can be done to incorporate ecosystem values into policy evaluations. These constraints include procedural requirements relating, for example, to timing and oversight, as well as the Agency's own resource constraints (both monetary and personnel). In an effort to better understand these issues and their implications for the committee's charge, the committee conducted a series of interviews with Agency staff. [11] The interviews were focused on the process of developing benefit analyses for Regulatory Impact Assessment (RIA) for rulemaking and the relationship between EPA and the Office of Management and Budget. However, many of the questions raised are equally applicable to strategic planning, performance reviews, regional analysis, and other situations in which the Agency is called upon to assess the value of ecosystems and the services they provide. Below are some key observations made by the committee based on those interviews.

EPA has a formal rule-development process with several stages, each of which imposes demands on the Agency, and the Agency also develops rules to meet court-imposed deadlines. However, despite the commonality of the underlying rule-development process, it is clear that there is no single way in which analysts within the Agency assess the tradeoffs that people would be willing to make to enhance ecosystems. Practices vary considerably across program offices, reflecting differences in mission, in-house expertise, etc. 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. [12] However, the primary expertise and development of the rules resides within the program offices.

Secondly, 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 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 (ICR), 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. The requirement can add a significant amount of time to the assessment process. With perhaps a year or two at most to conduct a study, this kind of review significantly limits the kind of analysis the Agency can conduct. In particular, it implies that the Agency must by necessity rely heavily on previous studies through the application of benefits transfer techniques.

A third issue is the role of the Office of Management and Budget (OMB) in defining or directing ecosystem valuation exercises at EPA. Among its activities, OMB acts as an oversight body that reviews EPA's benefit analyses. EPA is required to provide sufficient justification for its claims regarding the benefits of its actions, including any ecological benefits. As noted above, EPA has been given explicit guidance by OMB in the Circular A-4, which the committee views as a reasonable document on its own because of its call for a full characterization of the impacts of different policy options, including where possible a characterization of benefits that cannot be monetized or cannot be quantified (Office of Management and Budget 2003). For a benefit 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, if this is not possible either, to "describe the benefit or cost qualitatively" (add page number). [13] Thus, although Circular A-4 does not require that all benefits be monetized, it does require at a minimum a scientific characterization of those benefits. However, little guidance is provided on how this should be done. Instead, the Circular 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" (add page number).

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 value of consistency in the methods used for valuation, but also sees the limitations resulting from sole reliance on previously approved methods and the potential benefits from efforts to explore innovative or expanded approaches.

A related issue involves review of Regulatory Impact Assessments (RIAs) by external parties. The Agency does not take a standardized approach to RIA review. EPA staff and managers reported that peer review was focused only on "novel" elements of an analysis, meeting the requirements of EPA's peer review policy (EPA, 2003; also see EPA 2006). This raises the question of how the Agency (and perhaps OMB) defines "novel." Moreover, the novelty standard actually creates a clear incentive to avoid conducting novel analyses (however defined). It is clearly cheaper and quicker to avoid review altogether. This suggests a possible role for a standing expert body that can bring consistency to the review of analysis, avoid duplication of review, and be sensitive to timing and resource constraints.

Finally, the committee notes the importance of the organization of assessment science within the Agency. Currently, the Agency relies upon a variety of offices to develop assessments, with varying degrees of reliance on other offices (e.g., NCEE) or outside assistance. It is not clear which approach is most effective. In addition, the organization of assessment has implications for the availability and location of data to support ecological valuation. It is important that data that are housed within individual program offices 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 benefits assessment forum. The committee endorses these efforts. (KS: Do we?? I added this but it has not been discussed by the committee.) Nonetheless, the Agency will continue to face significant external constraints when conducting 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.

3.3. An Illustrative Example of Ecosystem Benefit Assessment at EPA

In an effort to better understand the current state of ecosystem valuation at EPA, the committee examined in detail one specific case where benefit assessment 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 2002). [14,15] The Agency indicated that this analysis was illustrative of other EPA regulatory analyses of ecological benefits in form and general content.

Because the proposed new CAFO rule constituted a “significant regulatory action” under Executive Order 12866, EPA was required to assess the costs and benefits of the rule. [16] EPA identified a wide variety of potential “use” and “non-use” benefits as part of its analysis. [17] Using various economic valuation methods, EPA provided monetary quantifications in its CAFO report for seven environmental benefits.[18] Approximately eighty-five percent of the monetary benefits quantified by EPA were attributed to recreational use and non-use of affected waterways. According to Agency staff, EPA’s analysis was driven by what it could monetize. EPA focused on those benefits for which data were known to be available for quantification of both the baseline condition and the likely changes from the proposed rule, and for translation of those changes into monetary equivalents. EPA’s final benefits assessment provides only a brief discussion of the benefits that it could not monetize. The benefits table in the Executive Summary listed a variety of non-monetized benefits[19] but designated them only as “not monetized.” EPA 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 gave a brief description of these "non-monetized" benefits, the remainder of the report devotes little attention to them.

Although considerable effort was invested in the CAFO benefits assessment, the 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 benefits using quantitative and qualitative information, as required by the OMB Circular A-4. Instead, the report focused on a limited set of environmental benefits, driven primarily by the ability to monetize these benefits using generally accepted models and existing value measures (benefit transfer).[20] These benefits did not include all of the major ecological benefits that the new CAFO rule would likely generate, nor all of the benefits that generated public support for the new rule. [21] The Circular requires that a benefit 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 benefits, the CAFO analysis and report understates the benefits of the rule change and distorts the rationale supporting the final rule. An unfortunate effect of this presentation is to suggest to readers that the monetized benefits constitute the principal justification for the CAFO rule.[22] Although in this case the focus on monetized benefits did not affect the outcome of the regulatory review, it is certainly possible that in a different context a benefits 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 benefits.

Second, the monetary values for many of the emphasized benefits were estimated through highly leveraged benefit transfers that were generally based on dated studies conducted in contexts quite different from the CAFO rule application.[23] 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 monetize benefits using existing value estimates. However, reliance on dated studies in quite different contexts raises questions about the credibility or validity of the monetary 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 engage in a sufficiently comprehensive effort at the outset to model the rule's ecological impacts. The report presents only 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.[24] 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 benefits. This was undoubtedly driven in part by the time pressures of putting together the regulatory impact analysis. However, without a comprehensive modeling effort at the outset, EPA had insufficient insight into the potential benefits that needed to be analyzed and valued. Developing integrated models of relevant ecosystems at the outset 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.[25]

Fourth, the CAFO analysis clearly demonstrates the challenges of conducting ecological benefit assessments at the national level.[26] National rule-makings inevitably require EPA to generalize away from geographic specifics, both in terms of ecological impacts and associated values. However, it is possible (and desirable) to make use of existing and on-going 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. However, both representative and non-representative case studies can provide useful information. Representative case studies offers more detailed data and models that could both fill in gaps in broad-scale national analyses and to check the validity of these analyses systematically. Systematically performing and documenting comparisons to intensive study sites could indicate the extent to which the national model needs to be adjusted for local/regional conditions and could provide data for estimating the range of error and uncertainty in the projected national-scale effects. As a complement, 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 EPA consulted with the public during its analysis to help it identify, assess, and prioritize the effects and values addressed in its analysis, nor is there discussion in the final CAFO analysis of any comments received on the draft CAFO analysis. Early public involvement could play a valuable role in helping the Agency both a) identify all of the systems and services impacted by the proposed regulations and b) determine the regulatory effects that are likely to be of greatest value. This would ensure that the benefits assessment includes the most important impacts.

Sixth, while EPA in its analysis and report appropriately emphasized the importance of using outside peer-reviewed data, methods, and models, EPA did not seek to peer review its application of them or its integration of these components in deriving benefit values for the CAFO rule. Once again, this is undoubtedly due in part to time and resource constraints. However, peer review, especially early in the process, would help EPA staff identify relevant and available data, models, and methods to support its analysis, and provide encouragement, direction, and sanction for more vigorous and effective pursuit of ecological and human wellbeing effects associated with the proposed rule. The general idea is to have individual components of the analysis (e.g., watershed modeling, air dispersal, human health, recreation, aesthetics) each reviewed, as well as a more general review of the overall analytic scheme.

Finally, EPA's analysis and report focused nearly exclusively on meeting the requirements as described in Executive Order 12866. This may not be surprising since the Executive Order provided the proximate reason for preparing the analysis and report. However, 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. The Executive Order does not preclude EPA from adopting broader goals. The Executive Order provides merely that EPA shall conduct an “analysis” and “assessment” of the “benefits anticipated from the regulatory action” and, “to the extent feasible, a quantification of those benefits.” By adopting a narrow focus, the report failed to consider or reflect the broader purposes that a benefit assessment can serve. Environmental benefit assessments such as the CAFO study can serve a variety of important purposes, including helping to educate policy-makers and the public more generally about the benefits that stem from EPA regulations, and it is important for EPA to recognize and have an incentive to consider this broader purpose.

4. AN INTEGRATED AND EXPANDED APPROACH TO ECOSYSTEM VALUATION: KEY FEATURES

The CAFO example discussed above highlights a number of limitations to the current state of ecosystem valuation at EPA. The committee's analysis points to the need for a comprehensive, integrated approach to valuing the ecological impacts of EPA actions, one that focuses on the impacts of most concern to people and integrates ecological analysis with valuation. This section describes an approach to ecological valuation developed and endorsed by the committee, followed in Section 5 by an overview of implementation. A more detailed discussion of the use of the approach and the methods that could be used to implement it are discussed in subsequent parts of the report (Parts 2 and 3). 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.

The proposed approach has three key features, which are interrelated: a) a focus on impacts of most concern to people; b) an integration of ecological analysis and valuation; and c) inclusion of an expanded set of possible valuation methods.

The first feature reflects the committee's view that ecological valuation or benefit assessment should focus on the impacts or benefits that are likely to be most significant or of greatest concern to people, which might or might not be those that are most easily measured and monetized. This requires a systematic consideration of the many possible sources of value from ecosystem protection and an identification of the types of values that are providing the impetus for a particular policy change. Information about the ecosystem services or characteristics that are of greatest concern needs to be obtained early in the valuation process so that efforts at quantification and characterization of values can be focused on the related ecological changes. This requires a mapping of the effects of a given policy change on ecological assessment endpoints to the corresponding effects on ecosystems and ecological services. In addition, 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, the inability to monetize a specific benefit should not preclude its inclusion; if there is evidence that it is important to people, it should be included as a key component of total benefits and a detailed and careful (even if not monetized) characterization of that benefit should be provided.

The second key feature of the framework is the integration of ecological analysis with valuation. This implies a focus on predicting ecological impacts in terms that are relevant for valuation. In particular, it requires a translation of bio-physical impacts into changes in ecosystem components and services that can be understood by lay individuals and are closely linked to the values they hold. This translation requires collaboration across various disciplines, both at an early stage (in the identification of the impacts that matter) and at a later stage (when estimating the value of impacts). Thus, instead of having ecologists work independently initially to estimate ecological impacts in scientific terms and then "pass the baton" on to economists or other social scientists seeking to value those impacts, the approach envisions collaborative work across disciplines to ensure that the analysis focuses on the impacts that are of greatest concern to society and that the ways in which these impacts are defined and measured are 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.

Third, the approach draws on a variety of methods to characterize and measure the importance of changes in ecosystems, including economic methods, social/psychological assessments, and other methods based on bio-physical rankings or public or group expressions of value. It recognizes that different methods provide different ways of characterizing or providing information about values. Different methods could be used at different stages of the valuation process. For example, some methods might be well-suited to providing information that would be used early in the process to guide decisions about which ecological changes are likely to be most important to people, while other methods would be well-suited to quantifying or monetizing benefits that are specific to the EPA action. In addition, the suite of methods used could vary with the specific policy context, due to differences across contexts in: a) information needs; b) the underlying sources of value being captured; c) data availability; and d) methodological limitations.

5. IMPLEMENTING THE INTEGRATED AND EXPANDED APPROACH

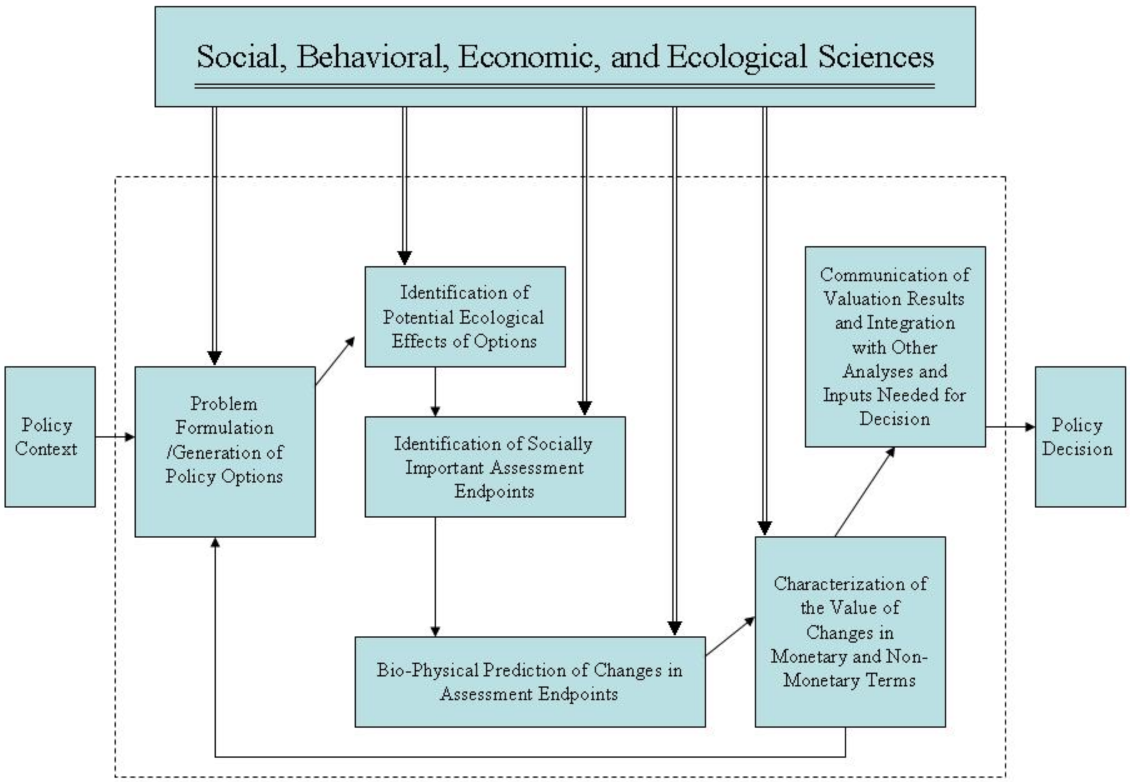
The previous section provides an overview of an integrated and expanded approach to ecological valuation proposed by the committee. This section discusses implementation of the approach in general terms. Part 2 of the report is devoted to a more detailed discussion of implementation, drawing on actual examples of decision contexts where ecological valuation could have played an important role.

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 ecological changes that could result under the different options;
3. identifying the ecological changes that are socially important, i.e., choosing the relevant assessment endpoints;
4. predicting the changes in these assessment endpoints in biophysical terms;
5. characterizing, representing, or measuring the value of changes in the assessment endpoints 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 it is likely that iteration will occur at some points of the process. For example, information about the value of changes in assessment endpoints stemming from a given set of policy options might cause a reformulation of the problem or identification of alternative policy options that could be considered.

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 where a benefit cost analysis is required, benefits need to be monetized whenever possible. In contrast, expressing benefits in monetary terms might be of little or no relevance to EPA analysts in other contexts, for example, when decisions are based on other criteria. Therefore the policy context in which the assessment is cast is 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. In addition, it suggests a structure that in many ways is parallel to the Agency's Framework for Ecological Risk Assessment (U.S. Environmental Protection Agency Risk Assessment Forum 1992). However, ecological valuation goes beyond the risk assessment framework by assessing the human values associated with predicted effects and hence provides a broader framework for assessment of ecological effects of EPA actions.

Several issues or considerations arise in implementing the steps of the process outlined above. A brief overview of these issues is provided here, as a prelude to the more detailed discussions that is included in Part 2.

5.1. Early Consideration of Effects that are Socially Important

A key component of the proposed approach is the identification and predictions of ecological changes that are important to people. These could include both changes in the ecosystem itself that people value directly, or the resulting changes in the ecological services provided by those systems. The importance of a given change will depend on both the magnitude and bio-physical importance of the effect and the resulting importance to society.

Although Figure 2 suggests a linear process, this part of the process will generally be somewhat iterative. The first step is to determine a preliminary list of potentially important ecological effects, based on both the magnitude and bio-physical importance of the effect. Development of this list would draw primarily on ecological science. However, it is important to identify early in the process what effects people are likely to be concerned about. Consideration of what seems to be important to people can lead to a subsequent refinement of the list of ecological effects that will be the focus of any valuation. For example, do individuals care mainly about the native-ness, the aesthetics, or the ecological functions of grasses in a marshland? Is animal waste disposal a concern to society primarily because of the recreational opportunities lost due to the resulting deterioration in water quality or is society primarily concerned about other impacts? The range of ecological changes that are the focus of the valuation study needs to include the changes people care most about. Previous benefit 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[27] or use of focus groups or workshops comprised of representative individuals from the affected population and relevant scientific experts can help to identify relevant or potentially important 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. Expressions of what is important (e.g., in surveys) or of the tradeoffs people are willing to make can change with the amount and kind of information provided, as well as how it is provided. Collaborative interaction between analysts and public representatives can ensure that respondents have sufficient information when expressing views and preferences.

5.2. 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 changes into changes in ecosystem services or features that are of direct concern to people.

The bio-physical impacts of a given EPA action can be identified at different temporal, spatial, and ecological levels. The latter include the individual level, the population level, the community level, the ecosystem level (union of biological populations with their surrounding physical environment), and the level of the global biosphere. Living organisms supply goods and services that differ across all levels of organization, from the individual to the ecosystem or global biosphere. For example, the service provided by an individual animal unit is different from the service provided by a given animal population.

Estimating bio-physical impacts requires information about relevant ecological production functions. These functions 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. In identifying and predicting ecological changes, it is important to consider their full range, including both primary and secondary effects, adequately accounting for uncertainty, stability of the system (including the effect of random shocks and management errors and the system's resilience), heterogeneity within a population or ecosystem, heterogeneity across populations or ecosystems, and dynamic changes in the ecosystem over time.

Numerous mathematical models of ecological production have been developed. These models cover the spectrum of biological organization and ecological hierarchy. Some have been developed for specific contexts (species, geographic locations, etc.) while others are more general. Primers on ecological theory and modeling (e.g., Roughgarden 1998) can provide a starting point for identifying available models.

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 ecological benefits of EPA actions. 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 above 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.

In addition, the ecological models need to be appropriately parameterized for use in policy analysis. Numerous detailed ecological studies have been conducted at various levels, at Long-Term Ecological Research Sites, for example (Farber et al. 2006), which could provide a starting point for parameterizing policy-driven models. A key challenge is to determine whether (or to what extent) parameters estimated from a given study site or population at a given point in time can be "transferred" for use in evaluating ecological changes in a different location or time or at a different scale. In other words, to what extent are estimated parameters adaptable to the context of interest in estimating the benefits and values associated with EPA actions? In many cases, data do not currently exist to parameterize existing models so they can 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 community structure that influence function, assessing the key environmental factors that influence the provision of services, and measuring the spatial and temporal scales over which services are provided (Kremen, 2005). However, most ecological models are not currently designed with this objective in mind. In particular, they do not translate bio-physical impacts into impacts or metrics that lay individuals can understand and reflect changes that are of direct value to them.

5.3. 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. There are a variety of methods that can be used to characterize values, and the C-VPESS approach envisions drawing on a wider range of methods than EPA has typically utilized in the past.

Some methods rely on metrics that are primarily bio-physical or socio-economic indicators of impact. These include indices or indicators such as acres or miles of habitat restores, the number or characteristics of communities or people affected, the likely symptoms or injuries avoided or reduced, the duration of impact. There are at least three ways in which these metrics can provide very useful information. First, in some cases, these metrics may be used directly in policy decisions. For example, decisions based on human impact criteria (e.g., protection of children's health) may look directly to these measures as indicators of the appropriate policy choice. Second, they might be used as a proxy for some component of the benefits of ecosystem protection when that component cannot be readily valued. For example, in contexts requiring benefit cost analyses, the OMB Circular A-4 requires that benefits that cannot be monetized be quantified to the extent possible, and these metrics provide potentially useful forms of quantification. 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, to date 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 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.

However, as in the CAFO study, economic valuation methods have generally been applied to a relatively narrow set of services. In some cases, these might not have been the services that people are most concerned about protecting. While there are continuing discussions about the role of economic valuations in principle, as a practical matter it is unlikely that all of the important benefits (or costs) of a change in ecological conditions can now, or soon 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, both as a practical alternative when economic methods cannot fully capture benefits because of data or other knowledge-based limitations and as a means of capturing the components of value that are not fully reflected in value measures based solely on economic measures of willingness to pay or willingness to accept. Expanding the methods "toolbox" to include other scientifically-based assessment approaches that can be applied along with or in place of economic assessments, where appropriate, will allow EPA to more fully represent the benefits of ecosystems and services. Of course, this toolbox should include only methods that meet accepted scientific standards of precision and reliability, are appropriately responsive to relevant changes in ecosystems/services, and are properly related conceptually and empirically to things people value. For all methods, appropriate application will depend on the underlying scientific basis as well as the specific policy context.

The committee evaluated a number of different methods for characterizing values (described in detail in Part 3). These include social/psychological methods, which have been successfully used to identify and to assess a wide range of values that people hold and that have been important considerations for environmental policy and decision making. Social/psychological methods bear close resemblances to economic methods, but they do not seek to attain a unidimensional monetary measure of benefit, allowing instead for multiple dimensions of value to be expressed and considered by decision makers. Other approaches include assessments based on voting and other group expressions of social/civic values, as well as assessment methods based on bio-physical rankings that are potentially less directly dependent on human preferences and value judgments (although these clearly enter indirectly).

An expanded toolbox of methods could allow EPA to capture more completely the full range of benefits stemming from ecosystem protection. In addition, where resources allow, use of multiple methods to characterize the same underlying value can in some cases provide evidence of conjoint validity and increase the confidence of policy/decision makers and the public. Of course, a significant risk of applying multiple assessment methods to an environmental decision problem, even when multiple valuation methods are permitted by law, is that in particular contexts the methods may suggest conflicting information about relative values. In this case, it would be essential to try to ascertain the source of the differences. In some cases, differences may be readily explained by differences in 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 simply the inherent uncertainty that exists in estimating values as a result of from data limitations, theory limitations, and randomness (see discussion in Part 2). In other cases the differences may reflect the fact that the alternative methods are capturing fundamentally different sources or components of value. In any case, information about the extent to which the different assessment methods yield similar or different conclusions about the value of an ecological change would be an important input into a policy decision.

5.4. Communicating Results

Information regarding the value of ecological changes stemming from EPA actions will only be useful in improving decision-making if it is communicated effectively to policymakers and integrated with other information used in policy decisions. In addition to policymakers, information about the value of ecological changes is likely to be of interest to community members and scientists alike.

Communicating the value of protecting ecological systems and services requires conveying not only value information, but also information about the nature and state of the ecological systems and services to which they apply and the ecological processes involved. Information can be and is often conveyed using mapped ecological information, other visualizations including photographs and graphs, ecological indicators and narratives. Integrated models with a geospatial interface, such as those developed by Costanza (Costanza and Farber 1986; Costanza, Sklar et al. 1990; Costanza 1993; Bockstael, Costanza et al. 1995; Fitz, DeBellevue et al. 1996; Cowling and Costanza 1997; Higgins, Turpie et al. 1997; Costanza 2002; Binder 2003; Costanza and Voinov 2003; Costanza 2004) are another approach to depicting the state of ecological systems and services. The SAB has proposed a framework for reporting on the condition of ecological resources (EPA, 2003). The EPA's draft Report on the Environment (U.S Environmental Protection Agency 2002) and reports of the Regional Environmental Monitoring and Assessment Program (REMAP) illustrate a range of approaches that can be used.

6. SUMMARY

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. It is imperative that EPA improve its ability to value ecosystems and their services to ensure that ecological impacts are adequately considered 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 ecological benefits. 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 benefits. The perception that benefits need to be monetized in order to be carefully characterized 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. As EPA continues these efforts, the committee encourages the Agency to 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 and solicit input from the public or representatives of individuals affected by the ecological changes.

Through the use of an expanded and integrated valuation framework of this type, EPA can move toward greater recognition and consideration of the effects that its actions have on ecosystems and the services they provide. In addition, it will allow EPA to improve environmental decision-making at the national, regional and local levels and contribute to EPA's overall mission regarding ecosystem protection. The remainder of this report develops the ideas embodied in this approach through a more detailed look at how the approach could be applied (Part 2) and the methods that might be used in implementing it (Part 3).

PART 2: APPLYING THE APPROACH

1. INTRODUCTION

Part 1 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. In this part of the report, we discuss implementation of the approach in greater detail. The purpose is to discuss in more detail a number of issues that arise in implementing the approach and demonstrate its application in different EPA decision contexts in which ecological valuation can contribute to improved policy analysis and decisions. As background for this, the committee examined a number of examples of specific valuation contexts and used these examples to inform its views about application of the proposed approach. The discussion throughout this part of the report reflects the general insights gained from examination of these source examples.

Part 2 begins by discussing of prediction of ecological effects, a key step in implementing the proposed approach. Part 2 next examines in more detail the implementation of the concept of ecosystem services specifically in the context of valuation. This discussion identifies a number of issues that arise in implementing this concept. (Editor's Note: These two sections were not both received in sufficient time to integrate and align their text for this draft.)

Part 2 then provides a brief overview of the expanded set of valuation methods that were examined by the committee. (These methods are discussed in more detail in Part 3 of this report.) This overview is intended to provide the reader with basic information regarding the different methods at a level sufficient to allow the reader to follow the context write-ups that follow. The context write-ups discuss application of the proposed valuation approach in three different contexts where ecological valuation can play an important role for EPA: a) national rulemaking, b) site-specific assessments, and c) regional partnerships. Finally, this Part discusses issues regarding the uncertainty associated with ecological valuation and the communication of valuation results to both policy makers and the public.

2. PREDICTION OF ECOLOGICAL EFFECTS

The valuation process requires more than simply listing the relevant ecosystem components or services that might be affected by a given EPA action; it also requires a prediction of how those services or the associated ecosystems are or would be affected. To some extent, a preliminary prediction of impacts should already have played a role in the identification of relevant services, mainly as a screening device. However, after that initial screening, a more detailed and rigorous prediction of bio-physical effects is needed. To the extent possible, this prediction should be quantitative, using metrics that are either indicators of value or can be used as a direct input into an assessment of the associated value. In the context of national rulemaking, this quantification is necessary for values that will be monetized, and is required (to the extent possible) by the OMB Circular A-4 even for those values that cannot be readily monetized. In contexts where monetization is not required or desired, information about the magnitude of effects will still be a key component of value assessment.

3.1 The Road Map: A Conceptual Model

Formulation of a conceptual model is a key first step in predicting the ecological effects of EPA actions. This conceptual model should be constructed at a general level to guide the process and to incorporate more detailed analyses that will subsequently be considered in identifying the key interactions, assessing the endpoints and calculating the ecological valuations. This approach includes the possibility of subsequently incorporating more detailed ecological models for defining production functions.

Development of the conceptual model is a significant task that deserves the attention of all the constituents of the process. These constituents include EPA staff from throughout the agency, experts in the relevant topics of consideration, and the public.. Although building the conceptual model is more complex than this linear sequence, in general the process involves these steps:

a) Building or importing a conceptual model of the ecological system(s), including preliminary decisions about the spatial and temporal boundaries. This step will be conducted largely by EPA staff and outside experts.

b) Inserting the likely stressors into the conceptual model; this may alter the initial and spatial boundaries. Most of this conceptualization will be accomplished by EPA staff and associated experts, but involving other constituents including the public at this stage will enhance transparency, provide the opportunity for more input and better understanding, and ultimately gives the process more legitimacy.

c) Identifying the significant ecological services requires the participation of all the constituencies as does anticipating ways in which the ecological services can be valued.

All three components (characterizing the ecological system, identifying the stressors, and identifying relevant endpoints and the significant ecological services) should be included in the beginning conceptual model. Building the conceptual model should be accomplished always with the recognition that one of the primary goals ultimately is to be able to value ecological services. This conceptual model, and the process for completing it and the embedded decisions within, should be a part of the formal record.

2.1.1 Characterizing the Relevant Ecosystem

Evaluating the ecological effects is an iterative rather than a linear process and EPA's process for ecological valuation should incorporate mechanisms for this iteration. This iterative process will identify both the geographic scale of the analyses as well as the ecosystems that should be included. As an example of how the iterative process might change the definition of the relevant ecosystem, an action at a local site may initially be considered to affect only nearby regions. However, once the stressors are considered, the reactions may involve consequences in distant downstream watersheds or airsheds. Ecological effects may involve different persistence times (e.g., of carbon dioxide in the atmosphere vs. acute toxic exposures to hazardous chemicals), affecting both the temporal and spatial scales of the relevant ecological system. There are numerous studies, including EPA's regional analyses, risk analyses and the EMAP program, that 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).

2.2. Identifying the Stressors

Although the identification of stressors to the relevant ecological system might be thought of as a relatively straightforward step, it too must be an iterative process. The important first step is a complete description of the project, whether it is a specific action on the ground or national rule promulgation. The description should include the quantity and quality of the anticipated stressors. 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. For example, a relatively non-toxic chemical effluent might be considered insignificant as a stressor, but might become significant if the conceptual model indicated low stream flows or intermittent streams that would increase the concentration of the chemical to toxic levels during some parts of the year. Identifying socially important stressors and ecological endpoints can be accomplished by involving both experts and the public via several techniques, such as surveys, public meetings, focus groups, content analysis of public comments, solicitation of expert opinion and testimony, and summaries of previous decisions in similar circumstances.

2.3. Identifying Relevant Assessment Endpoints

Because of their inherent complexity, ecological systems cannot be characterized in their entirety, nor can their responses to stressors (endpoints) be completely measured and predicted. Because of the complexity of ecosystems, they are often categorized not by species but by the abundance of the various functional groups present as exemplified by functional types of bacteria or guilds of birds that behave in a similar manner. bacteria types. There are a number of approaches to limiting the indicators to those that will provide the most direct information relevant to the services in question, for example, to focus on those functional groups that play a most prominent role in service (see previous discussion on ecosystem services).

An alternative approach to dealing with complexity is to rely on a set of generic assessment endpoints. EPA has developed a set of generic assessment endpoints that are based on environmental legislation and EPA's policies and precedents (Generic Ecological Assessment Endpoints, GEAE, 2003). These generic ecological assessment endpoints can be used as a starting point in the assessment of ecological effects. For example, Table 2 lists possible endpoints at the organism, population and community levels, including the policy relevance and the practicality of each endpoint.

Table 2: Table of Generic Assessment Endpoints Reproduced from U.S.EPA, 2003

Table 2-2. Generic ecological assessment endpoints (GEAEs): summary of the policy support for their use and their practicality[a]

GEAE #	Entity: attribute(s)	Policy support	Practicality
Organism-level endpoints			
1	Organisms: kills (mass mortality, conspicuous mortality)	Supported by many EPA programs; e.g., EPA has restricted the use of pesticides (e.g., diazinon and carbofuran) due to incidents of bird mortality.	Likelihood of kills from chemical pollutants can be estimated from toxicity testing. Incidents may be easy or difficult to observe, but when seen, they suggest a common mechanism or stressor exerting a strong effect.
2	Organisms: gross anomalies	Gross anomalies in birds, fish, shellfish, and other organisms are a cause for public concern and have been the basis for EPA regulatory action and guidance (e.g., assessed at Superfund sites, incorporated into biocriteria for water programs).	External gross anomalies are readily observed and are commonly included in survey protocols for fish and forests. They are also reported in toxicity tests of fish, birds, mammals, and plants.
3	Organisms: survival, fecundity, growth	Many EPA programs rely on organism-level attributes of survival, fecundity, and growth in assessing ecological risks (e.g., water quality criteria, pesticide and toxic chemical reviews, Superfund sites). Organism-level species protection is mandated by the Endangered Species Act, Marine Mammal Protection Act, Bald Eagle Protection Act, and Migratory Bird Treaty Act.	Results of toxicity tests of the survival, fecundity, and growth of organisms are abundant and often can be extrapolated to endangered species and other species of concern. Information on the ranges of listed endangered species is available through state and federal governments.
Population-level endpoints			
4	Assessment population: extirpation	EPA has taken action or provided guidance to prevent extirpation of local populations (e.g., assessment of likelihood of extirpation of fish populations due to acid rain). See also the description for Assessment population: abundance.	Extirpation can be predicted using population viability analysis. Demonstrating extirpation may be easy or difficult, depending on the conspicuousness of a species. See also the description for Assessment population: abundance.

Table 2-2. Generic ecological assessment endpoints (GEAEs): summary of the policy support for their use and their practicality[a] (continued)

GEAE #	Entity: attribute(s)	Policy support	Practicality
5	Assessment population: abundance	Major environmental statutes mandate protection of animals, plants, aquatic life, and living things generally, which can be inferred to entail protection of populations. EPA policies for pesticides, toxic chemicals, hazardous wastes, and air and water pollutants are intended to protect assessment populations of organisms. Mammals, birds, fish, aquatic invertebrates, and plants are typically assessed.	Changes in abundance may be predicted using conventional toxicity data with statistical extrapolation models and population models. OPPT evaluated a population model to explore effects of chloroparaffins on fish populations. Measurement of abundance in the field may be easy or difficult, depending on the species.
6	Assessment population: production	See description for Assessment population: abundance. Additionally, a number of laws are intended to maintain production of various economically valuable species. EPA water programs (e.g., National Estuary Program) and air programs (e.g., criteria pollutant standards) have involved protecting production of resource species populations.	Changes in production may be predicted using conventional toxicity data as well as population-based approaches. For resource species such as tree or fish species, production changes may be measurable in the field but may require long periods of observation.
Community and ecosystem-level endpoints			
7	Assessment communities, assemblages, and ecosystems: taxa richness	EPA water quality biocriteria frequently incorporate measures of community taxa richness. Additionally, EPA testing for pesticides, toxic chemicals, and water pollutants is intended to assess impacts to communities as well as populations and organisms. Fish, aquatic invertebrates, and aquatic plant assemblages are often assessed.	Changes in communities can be inferred or modeled from conventional toxicity data. Measuring taxa richness and abundance of aquatic communities, at least for fish and macroinvertebrate communities, is practical and well established. Ecosystem models that assess effects of toxicants on community properties are available and can use data acquired from organism-level laboratory testing, but they have not been routinely applied to date.
8	Assessment communities, assemblages, and ecosystems: abundance	As in the case of taxa richness, water quality biocriteria incorporate measures of community abundance, and EPA testing protocols are intended to assess impacts to communities.	See description above for taxa richness within assessment communities.

Tables such as this, expanded to include landscape-, regional- and global-level endpoints (see EPA GEAE, 2003, Table 4.1; Harwell, et al. 1999; Young and Sanzone, 2002) can be used as a first step in characterizing the relevant ecological system and quantifying the responses to stressors—the ecological effects. Considerations prompted by the table can be helpful in constructing and evaluating the initial conceptual model. In identifying and predicting ecological changes, it is important to consider their full range, including both primary and secondary effects, adequately accounting for uncertainty, stability of the system (including the effect of random shocks from external drivers, management errors and the system's resilience), heterogeneity within a population or ecosystem, heterogeneity across populations or ecosystems, and dynamic changes in the ecosystem over time (see Part 1 of this Report).

2.4. The Use of Ecological Models

While a conceptual model can provide a road map for predicting ecological effects, specific ecological models are needed to quantify effects and incorporate dynamic interactions among the ecosystem components, such as interactions among species, changing dynamics of population numbers with alterations in habitats, or accumulation of toxic materials in substrates with different absorption capacities. Because of the complexity of most ecosystems, models are used to organize information, elicit the interactions among the variables represented in the models, and when run under different sets of assumptions or driving variables, to predict possible outcomes. [28] Thus, statistical or simulation models become imperative to determine aspects of ecosystem structure that will influence future service production. The 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 question or hypothesis to be evaluated.

There are numerous ecological models that are used to describe ecological "systems" and various production functions, including scales from individual plants to regional characteristics such as crop productivity to continental migration of large animals. These models frequently focus on specific ecological characteristics, for example, populations of one or more species or the movement of nutrients through ecosystems. Models cover the spectrum of biological organization and ecological hierarchy. Primers on ecological theory and modeling such as Roughgarden 1998, Primer of Ecological Theory can provide a starting point for identifying available models. Some statistical models are relatively small, containing a few equations. Other ecological models are very large, involving hundreds of interacting calculations. Although many of these ecological models have been developed to satisfy research objectives and not agency policy or regulatory objectives, they are adaptable for use in estimating the benefits and values associated with agency actions. In fact, EPA currently employs a number of ecological models, ranging from fairly straightforward toxicity models to population model of fish and wildlife species to regional landscape models.

The primary focus of ecological models has been on understanding the dynamics in ecological systems, including for example, the role of abiotic driving variables on production, the interaction among species and the rate of carbon sequestration on continental scales. Although many of these models are well established and are used routinely for describing ecological systems, the results from all ecological models are approximations—they are estimates with known or unknown levels of statistical uncertainty—and no ecological model includes all the possible interactions. Because of the inherent complexity of ecological systems, most models are descriptive and predictions are more tentative, especially as ecological systems are frequently non-linear and subject to threshold changes. Although some ecological models explicitly or implicitly incorporate human dimensions, many of them focus primarily on ecological functions. 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.

Because so many ecological models exist, because none of them explicitly represents all the possible variables and their interactions, and because none is proven to be completely “accurate” under all defined circumstances, EPA is faced with deciding which models to employ at the site, regional and national scales. In theory, EPA could outline the types of ecological conditions under which it expected to consider risks and impacts, inventory the existing ecological models, conduct an assessment of their effectiveness and then offer a catalog of “approved” models with specifications and restrictions for their application. Although such an approach would have some appeal, it does not accommodate the dynamics of the scientific process, namely that existing models are always being modified on the basis of new understanding or additional data. Moreover, new models are continually being created and tested. Such a catalog of approved models would have some utility in the sense that use of these models would imply a level of credibility and acceptability that would not otherwise need to be reestablished with every new assessment. In addition, such a catalog of approved models would at some level create greater consistency among the methods used in the various EPA regions, presumably evolving toward a smaller number of models with greater validity.

The alternative approach recognizes the dynamics of evolving science, and specifies prerequisite characteristics of models rather than specifying particular models. Under this approach, models would be selected in the judgment of EPA that best address the particular issue.. For example, EPA could specify as a goal that models and data sets used to predict changes in ecological endpoints should meet the following seven conditions:

a) A beginning conceptual model that identifies, at least in a preliminary way, the state of the ecological system, the likely stressors and responses to those stressors and all the socially important anticipated interactions.

b) Utilization of databases that are in existence for the site, region or country that can provide, at a minimum a first approximation of the probable changes in endpoints. These more general data sets may need to be refined for the specific region or site depending on the project or the rule being considered, but initial assessments using these more generalized data sets will produce a range of likely outcomes which may be analyzed in more detail.

c) Adaptation of existing models should consider the congruent alignments among: (1) models; (2) ecological systems; (3) ecological services; (4) ecological service providers; (5) potential injuries; and (6) the stressors under EPA purview.

d) Models that are sufficiently comprehensive and have been used repeatedly so that there is a sufficient depth of understanding about their implicit assumptions, their reliability (robustness) and the reasonable range of applicability (space and time scales). These models would have been subjected to sensitivity analysis so there was a well-defined domain of outcomes from stochastic inputs.

e) Analytic output from the models should include a measure of variance that can be used to describe uncertainty in the predicted outcomes in a statistical distribution.

f) Results (production functions) from the analytical model should provide outcomes that are amenable to monetary and non-monetary valuation techniques.

g) Results from the models should provide guidance in a form that not only can be subjected to valuation techniques, but is readily usable by managers and rule- and policy-makers as well as by the interested public.

All of this emphasizes the importance of continued research aimed not only at improving understanding of ecological systems, but in particular at identifying the minimum information requirements for adequately describing and modeling the properties of ecological systems that result in important ecological services.

2.5. Gap between ecological models and the needs of ecological valuation

There is currently a gap between the outputs of most ecological models and the inputs required for valuation of ecological services. This gap arises for two general reasons. First, ecological models have largely focused on describing ecological systems in terms of ecological structure and function rather than in terms of social values. That is, the links between outputs of some ecological models and human uses of the ecosystem are not known or easily quantified. Many of these ecological models offer powerful comparisons among ecosystems as they are intrinsically different or respond differently to stressors or changes in driving variables. As such, outputs of these model may or may not be cast in terms of direct concern to people, and as such are not designed as inputs to valuation techniques. For example, evapotranspiration rates, rates of carbon turnover and changes in leaf area are important for ecological understanding, but are not easily translated in to values of direct human importance. Of course, there are some examples of models with outputs directly related to human values, such as those that predict fish and game populations or forest productivity. However, these represent a limited set of ecosystem services.

The second reason for the gap between ecological models and valuation needs relates to the complexity of ecological systems and their dependence on an array of site-specific driving variables. Because of this, many ecological models 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, certainly at regional and national scales. The absence of key data was identified in an analysis of the Millennium Assessment (Carpenter, et al. 2006): *serious constraints in the Millennium Assessment included systematic information on: stocks, flows, and economic values of many ecosystem services (e.g., freshwater fisheries, natural hazard regulation, groundwater, and pollination); knowledge of trends in human reliance on ecosystem services, particularly services without market value (e.g., domestic fuel wood and fodder; systematic local and regional assessments of the value of ecosystem services; and connections between data on human systems and ecosystems*.

Generic economic models and ecological models are useful, but in particular places (specific social values, specific ecological systems) either or both may fail to a degree because of the lack of specific information and the cost to obtain the information necessary for one or both models. There are numerous inter-twined challenges in the development of both economic and ecological models, including incorporating the required range of primary and secondary effects, characterizing the stability and dynamics of the systems, and dealing with the multiple-dimensioned heterogeneity of human populations and ecosystems.

2.6. Closing the Gap

Predictions of ecological effects of EPA actions are used for two fundamental purposes: as direct input into valuation methods (socio-psychological, economic, mediated modeling, etc.), and to quantify impacts when they cannot be monetized (in accordance with guidance in OMB Circular A-4) or when monetization is not necessary or desired. To date, direct measurements of ecological services and models that predict ecological services have been primarily aimed at describing and comparing the characteristics of ecological systems, not providing inputs to various methods of quantitative or qualitative valuations. However, there are now several promising directions for closing the gap between ecological measurements and model outputs and valuation inputs. These advances are in three areas:

- Organization of ecological data sets
- Evaluation of benefits transfer of ecological data
- Advances in modeling ecological services

Organization of Ecological Data:. Data on the structure and function of ecological systems are becoming much more available and better organized across the country. Part of the increased availability is simply that web-based publication now enables authors to easily post data and further analysis in electronic forms, available to other researchers. 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 the purpose of 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 more recently has focused on “regionalization” in which data from sites surrounding the primary site are collected, thus 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. It will be important for EPA to have effective links into the NEON planning process, and to expand its involvement with the NSF LTER program, which is now undergoing a major refreshing of its research and data sharing protocols.

Evaluation of benefits transfer of ecological data. Despite the increasing availability and organization of ecological data, 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, the issue is the reliability of transferring ecological information from one site or study to other sites 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 benefits derived from an ecosystem.

There are no hard and fast rules for when ecological information can be transferred, and the confidence with which information can be transferred depends on the type of information and the system in question. Given the complexity, richness of interactions and the propensity for non-linearity, extrapolation of ecological information requires caution. However, certain generalizations are possible. Information is more likely to be transferable with greater similarity between and among contexts, e.g., ecosystems. Also, aggregate information, such as data on ecosystem properties are 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 U.S. 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 ecosystem. For instance, if data are available on how the characteristics of an oak-hickory forest change as it develops or through cycles of disturbance, then it should be possible to transfer data from one point in time to another. 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 it should be possible to extend information from one spatial context to another. EPA and other national and international agencies have sponsored extensive research on "scaling up" of data from particular sites to regions, and 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.

The applicability of transferring benefits depends on characteristics of related resources and conditions, and also, on the reasonableness of using a static definition of an economic trade-off in a dynamic ecological system. Farber, et al. (2006) have attempted to classify the benefits transfer of ecosystem services from one context to another (see Table 3 below). In some cases, e.g., carbon sequestration (gas regulation) the transfer is appropriate at large spatial scales; in other cases, the processes operate at small scales but the processes are so general that they can be transferred with high confidence (e.g., value of game harvest). Some characteristics, such as genetic biodiversity (genetic resources) or spiritual values are very site-specific and thus the benefits cannot be transferred with confidence.

Table 3: Farber et al., 2006, Classification of Benefits Transfer of Ecosystem Services from One Context to Another

Ecological Service	Transferability
Gas regulation	High
Climate regulation	High
Disturbance regulation	Medium
Biological regulation	High
Water regulation	Medium
Soil retention	Medium
Waste regulation	Medium/high
Nutrient regulation	Medium
Water supply	Medium
Food	High
Raw materials	High
Genetic resources	Low
Medicinal resources	High

Ornamental resources	Medium
Recreation	Low
Aesthetics	Low
Science and education	High
Spiritual and historical	Low

Advances in modeling ecological services.

As noted above, EPA currently relies on a number of reasonably well-established models to predict ecological effects, recognizing that these models do not include all the possible considerations, especially in complicated issues or those that prevail over extended space and time scales. In some cases, the Agency has coupled these predicted effects with estimates of willingness to pay to generate monetized values of ecological changes. However, the application of this approach has been limited to a fairly small number of ecosystem services (see discussion of the CAFO example in Part 1 of this report). With the acquisition of increasing amounts of data, greater confidence in transferring data benefits and with greater model maturity derived from more experience, the process will allow for broader application of this approach. This approach, based on the use of existing ecological models, has the advantage that it fits many of the seven criteria listed above. In addition, it has been tested or used in a number of circumstances so its reliability can be assessed.

Although reliance on a single existing ecosystem model is one approach for predicting ecological effects of EPA actions, there are two other approaches that EPA should consider. The first is based on the use of indicators (a form of "simplification") while the second is based on meta-analysis (a form of "data aggregation"). Each is briefly discussed below.

The first approach involves selecting key predictive variables or indicators rather than attempting to measure and value all the possible significant outputs. These indicator variables have been established for specific ecosystems such as streams (e.g. Karr, 1993) and for entire countries (e.g. Heinz Foundation, State of the Nation's Ecosystems, 2002). Trends in ecosystem services are often most effectively communicated through indicators that simplify and synthesize underlying complexity. In addition, the use of large, complexity 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 can have advantages in terms of both generating and conveying information about ecological effects.

Many ecosystem indicators have been proposed (EPA/EC, 1996; National Research Council, 2000) and several states have sought to define a relatively small set of indicators of environmental quality to convey the value of ecological services. There currently is 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 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, 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 extent to which commercial fishing was reduced by the introduction and spread of invasive species was determined by the types of ecological interaction. He further argues that future models should consider more complex ecological interactions, habitat modification and non-market damages (Hoagland and Jin 2006) The question, of course, 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, the simplified model of selecting a few key indicators or ecological processes that can be valued may prove to be the most practical approach.

Similarly, there are ecological frameworks 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. For example, the "ecosystem report card" in South Florida (Harwell, et al., 1999) is based on particularly germane criteria:

- be understandable to multiple audiences,
- address differences in ecosystem responses across time, '
- show the status of the ecosystem
- characterize the selected endpoints, and
- transparently provide the scientific basis for the assigned grades on the report card.

The report card identifies seven essential ecosystem characteristics that are thought to beimportant, 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 that were then related to the goals and objectives for the ecosystem integrity report card.[29] Related ecological endpoints 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 a possible method for characterizing ecological benefits for the purposes of Circular A-4 when these benefits or ecological services cannot be readily monetized.

A third alternative, the use of meta-analysis or data-aggregation, involves collecting data from multiple sources and attempting to draw out consistent patterns and relationships. 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. On the other hand, by examining data from site-specific studies, coastal regional analyses and global catch databases, at least correlative relationships could be drawn 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 assemblages. 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 are not perfect, 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 which can be easily interpreted and at least in the case of game fish species, can be monetized.

In summary, EPA can continue to refine the models it uses, paying particular attention to the seven principles described above as a screen for this model selection process. The success of this approach will require continued expansion of data collection and increasingly demanding data management systems. In addition, EPA can explore the possibility of selecting key variables or indicators that are highly correlated with other ecological services. Finally, EPA can also focus on various levels of data aggregation that enable meta-analyses to identify broad relationships that obviate the need for ever more detailed data collection and model construction.

3. IMPLEMENTING THE CONCEPT OF ECOSYSTEM SERVICES

3.1. Implementation Issues

Part 1 of this report includes an overview of the concept of ecosystem services, which forms a key element in the C-VPESS valuation approach. While the general idea of ecosystem services is fairly clear, operationalizing this concept in a specific valuation context is often complicated and requires very careful consideration. This section discusses some issues that arise in applying the concept of ecosystem services in a valuation context.

An important step in implementation of the C-VPESS valuation approach is a listing of the relevant ecosystem services. Such a listing raises the question of what to "count" or include in the definition of services. As noted previously, the Millennium Assessment used a very broad definition of ecosystem services that included both indirect and direct contributions of ecosystems to human well-being (ref). An advantage of this broad definition is that it recognizes the many different ways in which ecosystems contribute to life as we know it, and hence can lead to greater appreciation of the service role that various classes of biota play in providing services. An analysis by Weslawski, et al. (2004) for example, 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 the general public had no understanding or appreciation of these services in his analysis. They do have an appreciation of the higher level services or what could be termed "end-point services," such a clean water and aesthetics, and of course foods that could be derived from the system.

For some valuation purposes, it might be preferable to adopt a definition of ecosystem services that is narrower than the definition proposed by the Millennium Assessment and focuses instead on these end-point services. For example, Boyd and Banzhaf (2006) proposes 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 stresses the need to distinguish between (ecological) *inputs* (or intermediate products) and (ecological) *outputs* (end or final products), and include only outputs in the definition of services. 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. 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).

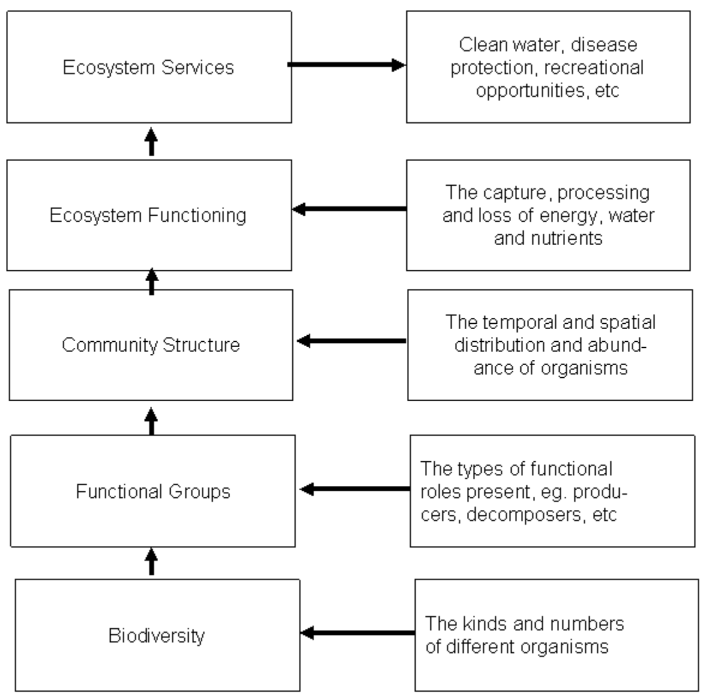
An advantage of defining ecosystem services in terms of end products is that, by highlighting the important distinction between inputs and outputs, it avoids the potential for double-counting. For example, for an ecological change that increased pollination, it would be double-counting to value both the improved pollination process (as an improved ecosystem service) and the increased agricultural output that results from it (as a separate service). Similarly, for a habitat improvement that leads to an increase in a bird population, it would be double-counting to value both the increased habitat (as an improved ecosystem service) and the resulting increase in the bird population (as a separate service). Doing so would be akin to valuing both the parts that went into production of an automobile and the final product, the automobile, itself. In principle, 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. Thus, in identifying and listing the ecosystem services to be valued, it is important to identify mutually exclusive services and to distinguish functions or processes that are inputs into the production of another ecological service so as to avoid double counting of the value of both the intermediate service (the input) and the final service (the output).

A careful delineation of ecosystem services in a specific context requires not only a determination of those ecological outputs that contribute to human well-being but also a determination of the source of that contribution. This cannot be done deductively, but rather depends on what people value and why. For example, a list of ecosystem services by Wilson (2004) includes maintenance of riparian vegetation habitat for mammals, birds, amphibians, and insects. Even if the ecological role of this habitat is clear, the service that it provides to humans depends not only on its ecological role but also on what people care about. Do individuals care about the habitat per se (e.g., for aesthetic or moral reasons), or do they care about the populations it supports (e.g., the insect population)? If the value derives from the insect population, do individuals care about the insects for their own sake, or because they are a food source for fish that people care about? In the first case, the insect population itself provides a service (produced using habitat as an input) and should be valued directly. In the latter case, the insect population should not be treated as an end product or service; rather, the fish population provides the service (produced using the habitat, including the insect population, as an input), and we should value the change in fish resulting from the change in the insect population instead. Of course, a full delineation of services would also require information about why the fish population is valued, e.g., for its own sake or as an input into production of another good or service such as recreation or bald eagles.

Even with a clear delineation or listing of ecosystem services based on the concept of end products, ecological valuation requires an understanding of the functions, processes and components of the ecosystem that underlie and generate these services, which are inherently complex. Consider, for example, the ecological services associated with the activities of soil organisms that might be affected by disposal of waste on that soil. These organisms make their living from organic matter that is in, or added to, the soil. In the process of breaking it down for use certain groups maintain soil structure by 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 that release nutrients in a form that can be utilized by higher plants for their growth, for example, or in dissolved form that enters into the water stream that leaves the immediate site into the water table or stream. Other groups of often-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 regulates the biogeochemistry of the site as well as its hydrology and productivity and carbon storage capacity. These functions can be evaluated in general terms and related to the services that people more readily appreciate and value such as the capacity of the soil biota to process wastes and provide clean water (Wall, 2004).

Even when defined as end products, all of the links in the various levels of organizations of ecosystems are involved in the provisioning of ecosystem services, as indicated in Figure 3.

Figure 3: Illustration of How Abundance of Functional Groups Can Characterize the Complexity of an Ecosystem

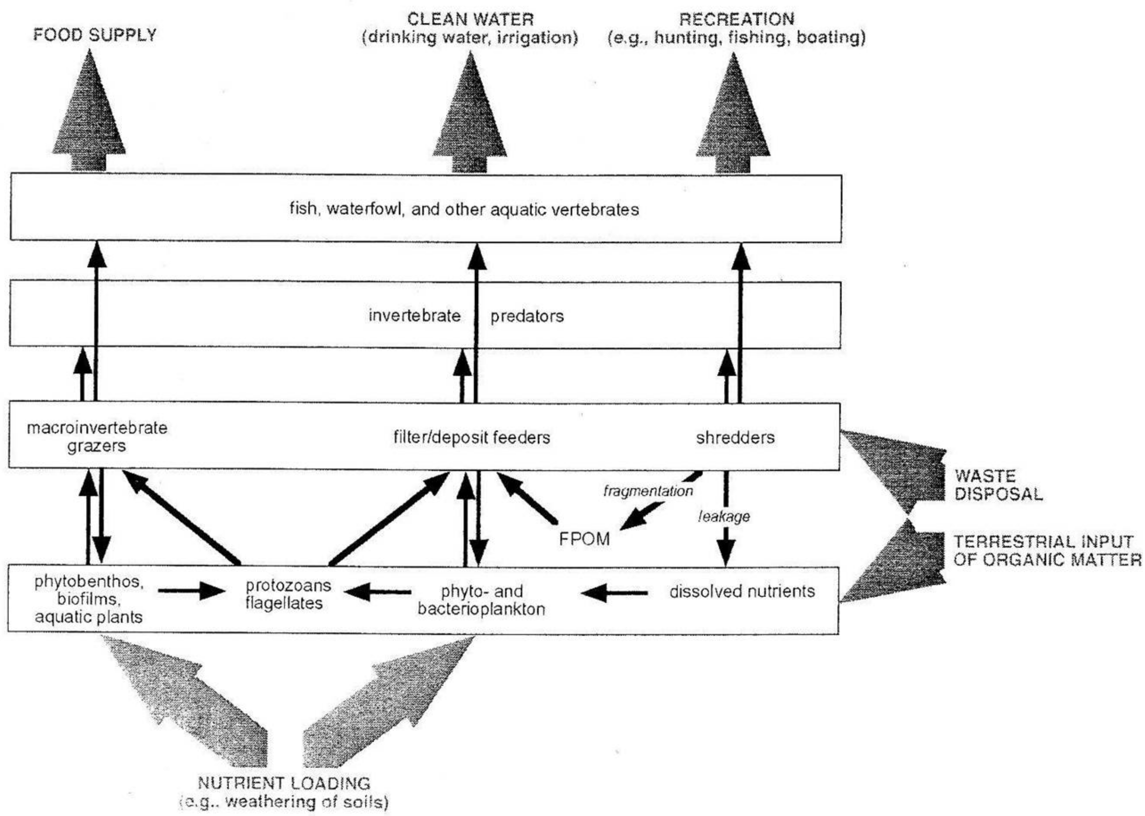


(Is the committee comfortable with use of this figure to illustrate the concept of functional groups? KS)

For a given ecosystem, the basic structure in Figure 3 can be used to develop a conceptual model of the functional levels of the ecosystem and how they contribute to the provision of ecosystem services. An example is illustrated in Figure 4 (taken from

Covich, et al., 2004). The important point of the figure is the emphasis on the major functional types and the role they play in providing the end point services. The key is identifying those components of each of the functional levels that are most directly related to the services of interest. Ecologists are at an early stage of linking ecosystem services with ecosystem functioning. Understanding these linkages better is an important research agenda.

Figure 4: Illustration from Covich et al., 2004, Showing Relationships of Major Functional Types to Ecological Services



Short of a full characterization of all of the linkages, it is possible to focus on groupings of organisms directly involved in the biological chain that affect the services of interest. This provides information about inputs as a proxy for the outputs. Because of the complexity of ecosystems they are often categorized not by species but by the abundance of the various functional groups present, e.g. decomposers of various kinds, as exemplified by the array of nematode types mentioned earlier, or bacteria that are specific to producing a particular breakdown product, such as the methane producers, those that produce nitrous oxides, and so forth. 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. In general, it is the functioning of a system that is of principal interest in terms of service provision, not "what species does the job." However, this principle does not hold at the recreational service level where a particular species, such as a given fish species, is the target of interest and the metric of concern.

The abundance of the groupings depicted in Figures 3 and 4 can be readily quantified. (References?) There are readily available and fully tested techniques for evaluating all of the components in this chain. (References?) For example, at the base of the ecosystem is its potential and realized biological diversity. Thus metrics that look at species richness and various diversity indices get at this directly. Through an analysis of the structures of the systems that are impacted, it should be possible to focus on functional types that are directly involved in providing the services of interest. For example, Weslawski et al. analyzed the services provided by various functional groups in estuaries and near-shelf sediment systems, providing a good starting point for relating functions to services. Ultimately, though, a better understanding is needed of how the various functional groups are affected by EPA actions and how these impacts in turn affect ecosystem services. Some taxonomic groups with wide functional diversity that are important in decomposition, such as the ubiquitous nematodes, offer promise in this regard for indicator purposes and have been so used in the past (Bongers and Ferris, 1999).

3.2. Common Endpoints are the Key to Progress

One of the Committee's fundamental conclusions – and one commonly voiced elsewhere -- is that the coordination or full integration of ecological and social analysis is necessary. As the Committee notes as an important recommendation in Part 1, it is necessary to "involve from the beginning and interdisciplinary collaboration among physical/biological and social scientists." The methods and examples described in this report do not themselves always live up to this standard. In fact, the organization of this report is an example of the distinctions drawn between biophysical and social analysis since much of the structure of Part 3 of this Report mirrors the separation between different disciplines .

A specific need that deserves much more attention by the Agency is the development of *ecological endpoints* for social science analysis. While the committee has not delivered a coherent, practical set of such ecological endpoints, it is optimistic about their development (Boyd, 2007). Further, the committee urges the development of such endpoints as the next logical step for the Agency to take as it pursues "methods for the evaluation and protection of ecosystem services."

> *Ecological endpoints are concrete statements, intuitively expressed and commonly understood, about what matters in nature.*

> *Technical expressions or descriptions meaningful only to experts are not ecological endpoints.*

The pursuit of common ecological endpoints will concretely foster the integration of biophysical and social approaches. In fact, common endpoints are the only way to debate and convey a shared mindset. They will lead to coordination, scientific advance, greater legitimacy in the halls of public debate, and clearer public communication about what in nature is being gained and lost.

The relative success of EPA efforts to translate air quality problems into human health-related social effects is due in part to extensive, ongoing debate over the definition of *health endpoints*. These endpoints are a lingua franca understood by disciplines as different as pulmonary medicine and urban economics. (EPA SAB, 2002)

The search for common health endpoints 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) connective endpoints are necessary. This will be even truer in the ecological realm, where biophysical processes and outcomes are even more varied and complex than in the human body.

Endpoints require the translation of technical outcomes into more intuitive, tangible outcomes – asthma attacks rather than oxygen transfer rates in the lung.

"Common person" descriptions of outcomes are usually a prerequisite to social science. The social sciences tend to rely on the assumption that people are reasonably well informed when they make choices. How can people be well informed if outcomes are not described in terms that are meaningful to average people?

It is important to emphasize that economists (or other social scientists) are not authorized to define endpoints in isolation. Rather, the natural and social sciences—with the imprimatur of both science and government—should collectively debate and define these endpoints. Step 1: acknowledge that common endpoints are important to public policy. Step 2: create endpoints informed by discussion *between* ecology and economics.

<u>3.2.1</u> <u>Ecological Endpoints and This Report</u>

The analytical challenge facing this committee is, first, the translation of Agency actions and decisions into biophysical outcomes. What is happening in nature? Then a second translation must occur: from biophysical outcomes to social. How does society value what is happening in nature? Endpoints lie between these two activities.

The first translation. The first translation identifies changes in the natural world resulting from natural changes and human activities, such as environmental protection actions including regulations and environmental programs, and adaptive management. Part 2, Section 2 of this report discusses current capabilities to predict ecological effects and the potential of ecological models to provide ecological production functions. Some have called the science that provides these predictions "conservation science," and they note that it includes many one disciplines: ecology, biology, hydrology, and atmospheric science, as well as the science of environmental management, which looks at how society affects nature and how human actions can improve or preserve natural resources.

The biophysical sciences often depict nature as a collection of inter-related processes and functions; examples include sequestration, predation, and nutrient cycling. To be clear, processes and functions are not endpoints.

Endpoints are the biophysical end-results of natural and social processes and functions. Nutrient loads in a particular body of water are an endpoint. Landscape, social, and ecological processes – the "production function" – determine and can be used to predict how endpoints change. How can we manage forests to prevent fire damage? What kinds of marine reserves lead to larger fish populations? How many more wetlands do we need to recharge sub-surface aquifers used for irrigation? These are questions relating to the production function of things we care about – reduced fire damage, larger fish populations, and plentiful water. [cite production function sections]

Understanding ecology as process and function is what allows us to test and depict causality in nature. Ecological process and function are necessary to predict changes in nature, particularly changes in endpoints. Biophysical production functions are the foundation of all environmental valuation. Only the science of biophysical production (conservation science) can deliver understanding of biophysical end-results.

The second translation. Once biophysical conditions are described or predicted, the second translation occurs: from biophysical conditions to the valuation of those conditions. "Valuation science" is used to set priorities, assess tradeoffs, and in some cases place monetary value on biophysical outcomes. Valuation science does not produce endpoints. Rather, valuation science requires endpoints as the basic input to valuation analysis.

Endpoints, or changes in endpoints, are the things in nature to which social science can attach value. Thus, endpoints are the baton that is passed from natural science to social science.

Part 3 of this report discusses a set of valuation methods. Much of the debate in this committee, and in society in general, is over the best ways to conduct valuation. The range of methods described in Part 3 represent the variety of approaches that can be used to *judge the value of endpoints or changes in endpoints.* There is ongoing debate on the merits of using different methods. For example, the most intellectually and emotionally potent of these debates is over whether dollars should be used to express the value of different outcomes in nature.

Nevertheless, debate over these methods should not obscure a fundamental conclusion: that all can – and perhaps should – rely on the *same* biophysical endpoints as inputs to the valuation process.

Consensus on biophysical endpoints, if it can be developed, will be desirable for two reasons. First, the more consensus regarding endpoints, the more productive will be the interaction between natural and social science. Conservation science will be able to smoothly hand the baton off to valuation science. Second, consistent endpoints will allow social scientists to more productively debate the merits of alternative valuation approaches. With a common set of endpoints, the strengths and weaknesses of the alternative methods will be easier to test and debate.

3.2.2 The Development of Endpoints – Principles.

Useful endpoints have several characteristics. As noted elsewhere in the report, endpoints are "ecological changes that are socially important." This statement, however, begs the question of what is "socially important." The committee believes several core principles can help refine the search for ecological endpoints.

The common person standard. First, endpoints should be concrete outcomes that can be intuitively expressed and commonly understood outside of the biophysical sciences. Endpoints should be as directly relevant to human experience as possible. Conservation science's contribution to social policy is limited if it cannot describe end-results in understandable, meaningful terms.

If human life itself depends on nature and if nature is an integrated whole, aren't all things in nature "socially important?" From a philosophical and ethical perspective, the answer is yes. From a measurement perspective, however, the answer is no. Consider all the things that can be counted in nature: the number of things and qualities is almost infinite. Focusing on those that are "directly relevant to human well-being" is, first, a way to make the problem manageable. Second, direct features can be thought of as nature's *end products*. [Cite Boyd& Banzhaf: which citation?] Their value will embody all of the indirect products necessary to them. Here the principle is count everything that matters, but only count them only once.

Clearly, the biophysical sciences should and will continue to explore all of nature. But if the goal is *integrated* natural and social science, EPA cannot focus on *all* aspects of nature. Rather, the focus should be on natural outcomes with the most direct relevance to human welfare.

This committee does not presume to understand what in nature is directly meaningful to people. This represents a scientific effort in itself. A first step, of course, is asking people what matters to them. The social sciences already do this to some degree, but more extensive empirical social science research is called for. Endpoints should be tested with real people, real decision-makers, and real communities to validate their relevance.

Endpoints should be purely biophysical. Second, endpoints should be purely biophysical characteristics or qualities. Biophysical outcomes will usually be the result of both social and natural processes. But what is needed specifically are the endpoints reflecting changes in nature.

What is meant by "purely" biophysical? If you catch a fish, isn't the fish "purely biophysical?" No, a fish in the hand is different from a fish in the lake. A fish in the hand is the result of several things, not all of them biophysical: in particular, the rod and reel, the skill, and the time provided by the angler. A fish in the hand is a combination of biophysical and social factors. The ecological endpoint—the thing that is purely biophysical here—is the fish population in the particular lake.

Another important clarification is that "purely biophysical" does not mean "untouched by human hands." Most things in nature are touched in some way by human action. In this case, the fish population may be reduced by harvests or improved by stocking. Human influence does not rule something out as an ecological endpoint.

Endpoints should be place- and time-specific. Third, endpoints should reflect the basic principles of conservation science: namely, the role of spatial and temporal phenomena and the importance of place. In practice, this means that endpoints 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 (the endpoint). But landscape-level and inter-temporal analysis is necessary to predict changes in that specific endpoint.

Endpoints should allow for the analysis of scarcity, substitutability, and complements. Fourth, endpoints should empower social analysis by allowing for analysis of scarcity, substitutes, and complements. This is related to the need for spatially- and temporally-explicit endpoints. The social value of ecological endpoints will often be related to the existence of substitutes and complements. Is this the only clean lake I can swim in or are there others within an hour's drive? What about streams? If I want to hike in the woods, is there a trail I can use? If I want to kayak in June will there be adequate water volume? Valuation science must look at these kinds of questions. Endpoints that allow valuation scientists to evaluate scarcity, substitutes, and complements, greatly enhance the power of valuation science.

Use proxies but relate them to real endpoints. Fifth, the endpoints that are most desirable in theory may be difficult to measure in practice. For example, it is relatively easy to measure atmospheric carbon. This is not a theoretically ideal endpoint because the implications of atmospheric carbon levels to average people are not intuitive or directly meaningful. Desirable climate-related endpoints include location specific temperature, species abundance, water availability, avoided coastal damages, etc. Society uses atmospheric carbon as a rough proxy for these kinds of outcomes, however. This is as it should be. Proxies are important because they economize on information costs and can act as a signal of a range of natural outcomes.

But the availability of proxies should not distract from the basic point: what society cares about is not atmospheric carbon, but rather the *end-results of* atmospheric carbon in intuitive terms in specific places at specific times.

We also note that statutes, regulations, and existing management practices, may mandate the use of particular proxy-like endpoints that do not conform to the ideal. Here again, the point is not to jettison such proxy endpoints but to develop understanding of how those proxies relate to what society really cares about.

3.2.3 Existing Endpoint Initiatives

The Committee is aware of existing activities with EPA to develop endpoints. For example, in the early 1990s the Agency created an Environmental Monitoring and Assessment Program (EMAP) designed to b 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: 4).

The C-VPESS is also aware that the Agency has developed so-called Generic Ecological Assessment Endpoints (GEAE, 2003) based on legislative, policy, and regulatory mandates.

The committee views these initiatives as steps in the right direction. However, in both cases the endpoints identified are not ideal. Regarding the EMAP effort, authors have noted the need to translate EMAP indicators "into common language for communication with public and decision-making audiences." (Schiller et all 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." This committee – interdisciplinary as it is – underscores this conclusion as it reflects our collective experience.

As for the GEAE, the committee begins by noting that these endpoints were developed via explicit reference to policy and regulatory needs ("Criteria used for selecting the GEAEs were that they must be useful in the EPA's decision-making process, practical, and well defined. Utility was based on policy support including citation in statutes, treaties, regulations, or Agency guidance and on precedents."). A set of these endpoints is reproduced in Table 2: Table of Generic Assessment Endpoints Reproduced from U.S.EPA, 2003.

The GEAE's are a starting point but are also an example of how far EPA must go in the development of ecological endpoints. First, by design, they depict a narrow range of ecological outcomes: confined to organism, population, and community/ecosystem effects. They do not relate to water availability, aesthetics, air quality, etc. Second, their technical nature – entirely appropriate for some regulatory purposes – does not appear to satisfy the "common person" standard described above. Endpoints relate to kills, gross anomalies, survival, fecundity, and growth, extirpation, abundance, production, and taxa richness. These are clearly relevant to biological assessment.

However, the connection of these endpoints to what is "socially important" is less clear. The easiest endpoint to interpret is abundance. Our guess is that people care about species abundance when they angle or hunt or when they are worried about the existence of a threatened species. The presence, density and population of a given species are clearly directly relevant to people.

However, the relevance of data related to production, taxa richness, gross anomalies, and kills is less clear. These things are important because they influence abundance – what people really care about – but does society care about these things as ends in themselves.

Consider "kills" and "gross anomalies." Hydropower facilities kill salmon. Toxic waste leads to tumors in fish. People may directly care about these organism-level problems. But EPA should also be clear that these data are being used as proxies for abundance. The more kills and tumors, the lower the abundance presumably. Except that this need not be true. Kills may be offset by greater production the community elsewhere. Use of proxies in this case should not district from the fact that what people really care about is place-specific abundance.

Another problem with these proxies is that they do not enable analysis of scarcity. Anglers care about the abundance of healthy fish in a particular location at a particular time. But kills and anomalies tell us little about that. They therefore do not enable valuation science. What is the social cost of a single dead or diseased fish? To answer that question you cannot rely on these proxies. Rather you need to know how those proxies relate to what people really care about: the abundance of healthy fish in the landscape.

The Agency is aware of these issues. The committee raises them only to motivate more extensive and comprehensive development of ecological endpoints.

3.2.4 Examples

Endpoints should be developed via collaborative discussions between natural scientists, social scientists, decision-makers, and the public. The following table provides only an illustration of what the committee means by an extensive and comprehensive list of endpoints.

Table 4: Table Illustrating an Extensive and Comprehensive List of Endpoints

Source: Boyd, James and Spencer Banzhaf, "What are Ecosystem Services," Ecological Economics, 2007.

In all cases, these endpoints should be measured with both spatial and temporal specificity. The committee notes that advances in information technology, mapping and remote sensing technologies in particular, will increasingly enable this kind of measurement.

The above list is in no way complete. The committee emphasizes that the development of endpoints must include a range of parties, including the public.

3.2.5 The Development of Endpoints – Process

Endpoints are a common language used to connect disparate academic disciplines and communicate to decision-makers and the public. How is a common language developed? Only through a process that brings these parties together.

As a result of its fact-finding, this committee has concluded that not enough interaction currently exists within the Agency between natural and social scientists. A reinvigorated endpoint initiative is a natural and place for more interaction to occur. The committee urges the Agency to initiate such a process.

Also, the C-VPESS is aware of opportunities for greater coordination between the Agency's research programs. The good news is that robust research programs on ecosystem issues already exist. For example, ORD's NCER has an established program on the ecological evaluation of ecosystem services. The stated mission of this program is – in part – to forecast, quantify, and map the production of ecosystem services. (see briefing by Ms. Iris Goodman included in EPA SAB 2006)]

NCER also has a grant program (though it is smaller than the ecological program) to look at the valuation of ecosystem services. Our fact-finding suggests that these two programs could and should be more closely linked. A joint research initiative focused on the development of ecological indicators will not only address a critical policy need, it is also a way for the Agency to concretely integrate its ecological and economic expertise.

Finally, the committee reiterates that research, guidance documents, and program-level efforts to develop ecological endpoints should not take place in a public vacuum. Endpoints can easily be viewed by the Agency as something to satisfy its relatively narrow reporting and assessment mandates. The committee advocates a more ambitious agenda: the development of endpoints that speak to public and political concerns and to all levels of government. Endpoints should be developed according to what the public wants, needs, and can understand.

4. INTRODUCTION TO DIFFERENT TYPES OF METHODS

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. In this section, we provide a brief overview of the categories of methods that the committee evaluated. The purpose is to provide the reader with enough information to follow the subsequent discussions about applying the approach in different policy contexts. A detailed discussion of specific methods is included in Part 3 of this report. In addition, Appendix A provides detailed information about survey methods.

The methods that we discuss differ in a number of ways. They capture different components of value and elicit values in different ways. Methods can elicit value information from individuals, groups or experts. Some methods are interactive or deliberative, while others are not. Some produce single metrics, while others produce multiple metrics. Some are well-developed, while others are still experimental or exploratory. Some elicit values based on self-interest, while others seek to elicit civic values or values based on bio-physical criteria. The methods can also vary in the geographic scale at which they can most appropriately be applied. Lastly, but perhaps most importantly, the different methods can play different roles in ecological valuation, depending on the specific decision context. After the brief overview of methods in this section, we turn in Sections 5, 6, and 7 to a discussion of three specific decision contexts to illustrate the implementation of the C-VPESS valuation process in those contexts. The role for different methods in these different contexts is a key part of that discussion.

4.1. Biophysical Ranking Methods

In some contexts, policymakers or analysts define values based on quantification of bio-physical indicators. Possible indicators include species biodiversity, bio-mass production, carbon sequestration or energy and materials use/redistribution/flows. Quantification of ecological changes in bio-physical 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 varying human values based on human preferences. Rather, it reflects either an alternative 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 benefits of any change in ecosystems can be assessed in terms of the calculated effects on overall ecosystems health and sustainability. Some people view the fact that these ranking methods are not directly tied to human preferences as a drawback, while others view it as a positive feature of these methods.

The committee evaluated two types of bio-physical 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 their contribution to sustained ecosystem diversity and integrity. The method provides a scientifically-based approach to assigning conservation values that can used by policymakers or stakeholders to prioritize land for, for example, acquisition and conservation. Based on GIS technology, it 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 bio-physical methods that the committee evaluated were based on energy and material flows. Energy and material flow analysis is the quantification of the flows of energy and materials through complex ecological and/or economic systems. 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, and ecological footprints. Of these three, embodied energy and ecological footprints are based on a consistent set of principles, while emergy is not (and is hence not scientifically sound). 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. These techniques have been used to estimate implicit costs or "shadow prices" of providing ecosystem goods and services, measured in physical rather than monetary units. 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 (see Part 3, section 7).

4.2. Ecosystem Benefit Indicators

Ecosystem Benefit Indicators (EBIs) offer a quantitative way to illustrate ecological benefits 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 indicators include percentage of a watershed in a particular land use or of a particular land type, number of users of a service (e.g., water or recreation) within a given area, and distance to nearest vulnerable community.

4.3. Social-Psychological Methods

Social/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 sub-groups) and the basis for judgments may be changes in individual well-being, or civic or ethical/moral obligations relevant to ecosystems and ecosystem services. That is, people may hold, express and advocate bio-ecological values or ethical values that are unrelated or even counter to their own wants and needs.

Social/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-psychological methods elicit information about preferences and values primarily through surveys, focus groups, and individual narratives. However, recently experts in this field are also experimenting with eliciting this information through observations of behavioral responses by individuals interacting with either actual or computer simulated environments.

Attitude survey questions are typically framed as choices (among two or more options), rankings, or ratings. Survey methods that may be especially important in the context of ecological valuation include perceptual surveys (e.g., assessment of ecosystem attributes) and conjoint survey methods (e.g., choice among different combinations of ecosystem attributes). Quantitative analysis of responses are usually interpreted as ordinal rankings or rough interval scale measures that provide relative measures of differences in assessed values. Similarities and differences among different segments of the public can also be identified and articulated. Surveys 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, including the U.S. Department of Agriculture's Forest Service.

In contrast to surveys, which are based on large samples, individual narratives are subjected to qualitative analyses to identify and possibly to ascertain levels of consensus on relevant issues, perspectives, and positions represented by participants. Individual narratives can help in identifying ecosystem effects that might be particularly important to the public, although individual participants typically come from a fairly narrowly-defined target group.

4.4. 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 the recently released *EBASP*.

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 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 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 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) and methods based on information elicited from surveys (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 benefits from survey responses. 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 benefit measures from responses to 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.

4.5. Group and Public Expressions of Values

There are methods to elicit expressions of values from groups. Focus group methods elicit information about values and preferences from small groups of relevant stakeholders engaging in group discussion lead 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.

Most of the valuation methods described in the sections above involve a process under which information flows primarily in one direction (from the person whose values are being elicited to the analyst who is seeking to measure those values). In contrast, in some cases valuation is part of a deliberative process or facilitated interaction among decision makers, analysts, and stakeholder representatives or other interested parties that occurs over a more extended period of time (e.g., days or weeks). In such cases, the process itself becomes an important component for understanding and conveying information about values. These processes seek to elicit or incorporate information about all possible sources of value. In addition, they involve directly confronting tradeoffs that inevitably arise. Two examples of deliberative processes are decision-aiding processes and mediated modeling.

Decision aiding processes have been developed by decision scientists and applied in a number of contexts, including contexts involving environmental choices. From the perspective of decision science, valuation is not a separate exercise that then feeds into a decision made by others. Rather, it is part of a process designed both to discover values and, in many cases, ultimately to make policy decisions. This is based on the premise that people's preferences and values for complex, unfamiliar goods (such as many ecosystem services) are often constructed during the process of elicitation and are multi-dimensional. This premise is in contrast to the premise underlying some of the methods discussed previously, most notably economic valuation methods, which assume that preferences are given and that values or benefits can be measured using a single metric such as willingness to pay.

Decision aiding processes can be applied either in a decision making context or an evaluative context. In either case, they involve a number of steps, including identification of the objectives of the process, definition of the attributes that will be used to judge progress toward the objectives, specification of the set of management options, measurement of changes in relevant attributes that would be realized under alternative management options, etc. These steps draw on inputs from a variety of disciplines, such as economics, ecology, psychology, and sociology. The final output is either the selection or identification of a preferred management option (if the context is decision making) or a judgment about the current state of the system relative to a previous state (if the context is evaluative). Generally, the objectives that guide these final outputs are diverse and often multi-dimensional. Examples include maintaining some requisite level of ecological services, protecting endangered or threatened species, production of outputs such as resource extraction, tourism, and recreation opportunities, and supplying a sense of pride or awe (Gregory et al. 2001).

With mediated modeling the deliberative process focuses on stakeholder interactive development of a model representing a particular environmental system of interest. Stakeholder participation in model development occurs at all stages of the modeling process, from initial problem scoping to model development, implementation, and use. The models that are developed can be at any geographical scale, from watersheds or specific ecosystems to large regions or even the globe. The output of the process is a model that can be used to evaluate alternative scenarios or options of interest to those stakeholders. Most importantly, the model and the results derived from it have stakeholder buy-in and reflect group consensus. If the model is used to consider tradeoffs, then values must be explicitly incorporated into the model through the specification of related parameter values. These values can be drawn from other valuation exercises or based on other information that relates to value (e.g., use data).

There also are a group of methods that focus on public and group expressions of public value, in contrast, for example, 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 as consumers of those systems and services. By contrast, 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 referendum and initiative 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 benefits 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, and they incorporate elements of the "deliberative valuation" processes. The group is given extensive information and, after extensive 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. However, 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.

4.6. Methods Using Cost as a Proxy for Value

A fundamental principle in economics is the distinction between benefits and costs. Benefits reflect what is gained by increasing the amount of a given good or service. The value of goods and services is synonymous with the benefits. 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 purposes of providing 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 benefits. However, because costs and 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 where these two conditions may be met is the provision of clean drinking water for a metropolitan area. Protection of 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 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 cost as a proxy for value. It must be demonstrated that the conditions for valid use are satisfied and results should not be interpreted as the value of ecosystem services themselves but only the value of having one means to provide them.

4.7. Summary of Methods Available for Implementing the Integrated and Expanded Approach

(Draft text to be developed for May 1-2, 2006 meeting that fills in the blank table that follows and provides context discussion)

Table 5: Table Summarizing Methods Discussed in this Report

	Form of output/units?	What is method or measurement approach intended to measure?	Source of Information About Value		Status of Method
			Does method measure observed behavior, verbal or written expressions, or progress related to previously identified goal?	Who expresses value?	
Conservation Value Method					▪
Embodied Energy Analysis					▪
Emergy					▪
Ecological Footprint					▪
Ecosystem Benefit Indicators					▪
Surveys of beliefs, attitudes and intentions					
Conjoint attitude surveys					▪
Individual Narratives					
Mental Models					
Behavioral Observation/Trace					
Interactive Environmental Stimulation Systems					
Market-Based Methods					▪
Travel Cost					▪
Hedonic pricing					▪
Averting Behavior					▪
Stated Preference Economic Surveys					▪
Focus Groups					▪
Decision-Aiding/Structured Decision Making					▪
Mediated Modeling					
Referenda and Initiatives					▪
Citizen Valuation Juries					▪
Replacement Cost (also called “Avoided Cost”)					▪
Tradable Permits					▪
Habitat Equivalency Analysis					▪

5. ECONOMIC VALUATION FOR NATIONAL RULEMAKING

5.1. Introduction

The objective of this section is to examine the valuation of ecosystem services by the Agency in the context of the evaluation of the 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 by a set of criteria other than benefits and costs. Indeed in some cases the legislation explicitly precludes consideration of costs or 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 benefits and costs, e.g., Safe Drinking Water Act, adherence to a strict "benefits must exceed costs" criterion is not required.

Nonetheless, an assessment of the 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 Agency Rulemakings 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 benefits and costs can be mandated by law. For example, the prospective analysis of the benefits of the Clean Air Act Amendments of 1990 was mandated by Section 812 of the Clean Air Act Amendments of 1990, which requires the Agency to develop periodic Reports to Congress that estimate the benefits and costs of various provisions of the Clean Air Act. Finally, the benefit and cost estimates developed in national rulemaking 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 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 rulemaking, the terms "benefit" and “cost” have a specific meanings. To the extent possible the 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 with a specification of what environmental conditions would be throughout the areas affected by the rule with and without it. 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).

This section considers ecological valuation in the context of national rulemaking governed by Executive Order 12866 as amended and OMB's Circular A-4. It thus focuses on the use of economic valuation methods that seek to monetize 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 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 rulemaking regarding concentrated animal feeding operations (CAFOs) (US EPA 2002); and c) the prospective analysis of the benefits of the Clean Air Act Amendments of 1990 (US EPA 1999).[30]

Brief descriptions of the three benefit analyses are presented in separate text boxes. These examples provided insights that are reflected in the discussion and recommendations throughout this section.

5.2. 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 rulemaking and RIAs. It illustrates how the three major recommendations in Part 1 of this report (see Part 1 section 6) could be implemented in this context. These recommendations relate to: a) early identification of the ecological changes that are most important so that these can be the focus of the valuation; b) prediction bio-physical changes in assessment endpoints in terms that can be directly valued by the public; and c) characterization of the value of changes in monetary and non-monetary terms. Each is discussed in turn. For each, we also provide specific recommendations for implementing the general recommendations in this specific context.

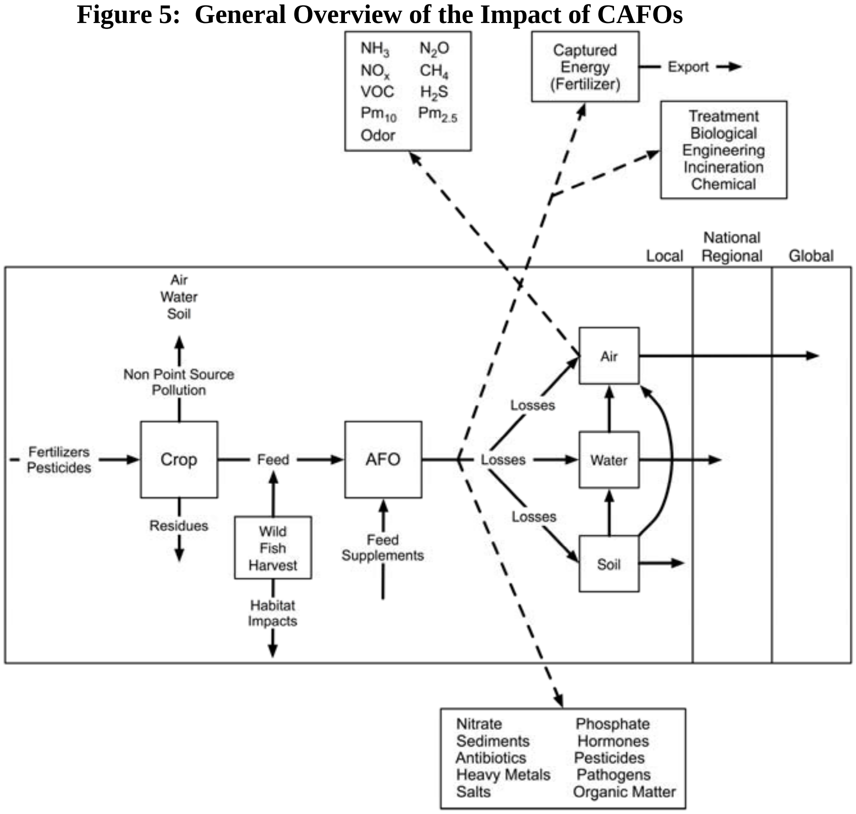
<u>5.2.1 Early identification of socially important assessment endpoints</u>

Identification of socially important assessment endpoints requires information about both the potential biophysical effects of the Agency's action and the ecological services that matter to people.

Recommendation: 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 sections 2 and 3 above). Determination of the important ecological effects 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 5 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. This overview could be used to develop a conceptual model that identifies potential ecological services that might be affected by CAFO regulation. It should be standard practice for the Agency to develop such a conceptual model before other analytical work begins on a 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.

Figure 5: General Overview of the Impact of CAFOs



Recommendation: 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 concern to people.

As noted, the conceptual model should include information about the changes that are likely to be of greatest concern to people. The committee believes that identification of what matters to people cannot be done deductively. Rather, it requires an examination of the evidence gleaned from a variety of research approaches. It is important to distinguish the processes used to enumerate the goods and services that are important to people from the process used to evaluate benefits and costs. Where the analysis is being conducted to meet a mandate for benefit-cost analysis (as is the case for RIAs), the computation of 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 assess public concerns includes surveys, public meetings, focus groups, content analysis of public comments, and so forth. 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 and/or the avoidance of the various risks. More specifically, possible approaches for obtaining information about public concerns include:

- Inventory of the reasons invoked in similar rulemaking processes in other jurisdictions (e.g., state and local).
- Inventory of the concerns expressed in public hearings (perhaps with weightings based on the frequency of concerns raised). For example, local vs. national concerns can be quantified through content analysis of transcripts. Where local debates over allowing fish farming have occurred, the discourse could reveal what people care about.
- Focus groups and surveys of concerns (can be lists of concerns, or quantified by ranking priorities).

Recommendation: Consider use of an open, interactive public forum for identifying issues of concern.

The committee suggests that EPA experiment with holding open meetings for the public and Agency staff to aid in the development of the conceptual model for a particular rulemaking. Such an approach would provide an interactive forum for determining the ecological changes that are important both biophysically and socially.

Recommendation: Use a transparent, documented process for identifying the ecological changes that will be the focus of the valuation.

Whatever methods are chosen to increase transparency related to the analytical process for developing rulemaking documents, the Agency should document in its benefit assessments and RIAs how "socially important assessment endpoints" were identified for the analysis. 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 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 benefit assessments describing how the Agency addressed the most significant comments regarding ecological values and valuation. Finally, the analytical blueprints and conceptual model that was used to guide the analysis should be part of the public record for every rulemaking and available on-line.

5.2.2 Bio-physical prediction of changes in assessment endpoints

Recommendation: Utilize, or develop, quantitative ecosystem models to identify the consequences of stressors on the production of the services of concern.

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. As noted below, such models are now utilized in rule making but sometimes their complexity, cost, and time constraints, promote the use of the simplest modeling approaches available that can be tailored to economic valuation. Short cuts can be taken if only a single service is considered and the chain is simple. For example, in an analysis of the regulating service of human lyme disease control, the ecosystem service provider was identified as the numbers and abundance of the alternative vertebrate hosts, and from this the production function of disease dilution rate could be calculated (Kremen, 2005). However, as illustrated in Figure 5, there are many stressors involved in CAFO operations and they have complex interactions which only can be revealed by a fuller consideration of ecosystem dynamics. Further, outputs from these models give quantitative values of the stressor impacts even though all of these cannot be monetized.

In many rulemaking contexts, it is difficult to predict even the changes in stressors, let alone the resulting impact on endpoints. 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 and in other cases the rule only required "best management practices" rather than quantitative maximum discharge levels. In addition, in the past the Agency has generally chosen to focus on stressors whose impacts can be monetized with readily available techniques and/or estimates from the existing literature. All three of the rulemaking benefit assessments that the committee reviewed provide evidence of this. 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, organics, etc., 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. Rather than choosing stressors based on the ability to readily monetize their impacts, the Agency should use the conceptual model (see discussion above) to guide the selection of stressors, and then seek to use a suite of ecological models that can predict the impacts of changes in these stressors on a broader set of the relevant assessment endpoints.

Quantifying changes in assessment endpoints is particularly challenging in national rulemaking contexts, and there are many issues that need to be addressed in order to establish a convincing analysis of the benefits of a national rule. 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 5):

- Multi-media effects, i.e., interrelated impacts on both water and air quality;[31]
- Impacts across multiple geographical scales (e.g., local, regional, global);[32]
- Differences in the time persistence of pollutants (e.g., days vs. decades);[33]
- Clustering and the need for site-specific analysis due to uniqueness of site characteristics associated with impacts;[34] and
- Ecological impacts through supply-chain effects that are geographically dispersed.[35]

Some of the links between stressors and endpoints 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.[36] 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 a production function for these organisms and use that function to predict the impact of changes in stressors.

However, many of the links between stressors and assessment endpoints are: a) not fully understood scientifically, and/or b) not fully appreciated by the public. 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 sales of incinerator ash for fertilizers from poultry-waste fueled generators, or directly by the use of dried and pelletized manure in places distant from the source (Nachman, et al., 2005).

There are many things that are well known scientifically, yet the general public is not fully aware of and hence has no appreciation of or informed opinion about them. For example, the full chain of connections in the production of animals in CAFOs as described in Figure 5 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 and the resulting services provided. For example, certain groups of soil organisms maintain soil structure by their burrowing activities, while other kinds of organisms shred the organic material into smaller units that are in turn utilized by microbes that release nutrients in a form that can be utilized by higher plants for their growth. However, the general public has little appreciation for the "services" these organisms provide (e.g., Weslawski, et al. 2004). Again, this problem of lack of public understanding might be exacerbated in national level analyses where ecological impacts and vulnerabilities can vary substantially across locations.

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. As noted above, 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. The actual process for implementing the Circular, however, requires a reversal of this order, namely, first impacts should be described or characterized qualitatively, followed by quantification and ultimately monetization where possible.

As noted above, characterization of ecological impacts requires a conceptual model (see detailed discussion in Part 2 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 Figures 3 and 4. This model can be used as the basis for a qualitative but detailed description of the ecological impacts of a given change. However, just a listing that summarizes possible impacts is not sufficient. 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 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. Criteria for choosing among alternative models were discussed in Section 2. 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.

There are readily available and fully tested techniques 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. 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 the rule on species richness and various diversity indices achieve this. However such metrics cannot be tied directly to the ecosystems services provided without embedding this information into an ecosystem model that reveals functioning which in turn can be related to 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. There are a number of approaches to limiting the indicators to those that will provide the most direct information relevant to the services in question. One is to focus on those functional groups that play a most prominent role in service provision as noted above.

In summary, the initial conceptual model of a system provides the big picture of the possible environmental impacts of the rule. Then, when focusing on just the outputs from specific facilities such as CAFOs or aquaculture facilities that are covered in a rule, there is a large array of potential metrics that would indicate the success of rulemaking in providing better ecosystem services to society. In addition to looking at end point services only, it is important to look at the ecosystem service providers, even though they cannot be directly monetized. The suggestion here is through an analysis of the structures of the systems that are impacted it should be possible to focus on functional types that are most directly involved in providing the services in question. There are ample tools available for making these measurements.

Recommendation: Start building toward a more holistic approach to rule making.

From the information embedded in Figure 5 it can be readily appreciated that focusing on the *outputs* from CAFOs only, and further, only on those outputs that impact water quality there is an inadequate attention to the full environmental impacts of CAFOs. Outputs contain pollutants that impact not only the water but the air, and these outputs are interactive. Further, the feed supply change providing inputs to CAFOs involve many adverse environmental impacts that are not considered if only outputs from CAFOs are analyzed. Of course there are presently regulatory restrictions that do not allow such a complex undertaking but nonetheless the reality is there and needs to be addressed and not hidden.

5.2.3 Monetary Measures of Value

Circular A-4 calls for the monetization of benefits whenever possible. Although there are a variety of methods that can be used to determine values for purposes of identifying socially relevant assessment endpoints (see discussion above) and for value assessments in other contexts (see Sections 6 and 7), in the context of benefit-cost analysis the only approach to monetization consistent with the premises underlying this analysis is the use of economic valuation methods. The inclusion of measures of values based on other methods such as those mentioned above, even if measured in dollar terms, is problematic because it implies adding together numbers that are based on quite different methods, assumptions, and underlying premises. Thus, for both theoretical and empirical consistency, the measure of benefits in a benefit-cost analysis should be based on economic valuation.

There is a large and growing theoretical and empirical literature within economics on methods for assigning monetary values to environmental changes. These methods use either observed behavior (revealed preference) or responses to surveys (stated preference or contingent valuation/choice) to estimate willingness to pay (or accept compensation) for these changes. While there have been controversies surrounding the use of these methods, particularly the stated preference methods, existing research supports the view that, when appropriately used, these methods can provide informative and useful estimates of willingness to pay (see related discussions in Part 3, section 5.4 and Appendix A).

Recommendation: The Agency should make a greater effort to select endpoints for valuation based on its assessment of the social importance of the of the ecosystem service rather than to allow the choice of endpoints to be dictated by the available models and data.

The Agency needs to ensure that the call for monetization does not unduly restrict the types of ecosystem impacts considered in the benefit assessment, or lead to inappropriate application of economic valuation methods (including benefits transfer). As noted above, the call for monetization has often driven Agency decisions regarding the focus of ecological benefit assessments. This applies not only to the types of ecosystem services included in the detailed assessments but also the ecological models used to predict biophysical impacts. For example, the Agency's assessment of the CAFO rule focused primarily on recreational impacts and its assessment of the aquaculture rule focused almost exclusively on recreational impacts and used the QUAL2E water quality model to predict the changes in several water quality indices that would result from implementation of the rule. The choice of QUAL2E appears to have been driven largely by the ability to link its outputs with readily available, 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. Also, 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 benefit estimates. Most notably, the study was conducted more than 20 years ago, it was designed for a different purpose and was not intended to apply to specific rivers or lakes, the water quality index used was highly simplified, and the index it used 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, 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.

The previous section discusses the need to consider a broad range of ecosystem services when assessing the benefits of national rules, even if the benefits associated with changes in those services cannot be monetized. However, even when the benefits can be monetized, the above example highlights the need for appropriate application of monetization techniques to ensure scientifically-credible benefit estimates. In many cases, time and resource constraints will necessitate use of benefits transfer. However, care must be taken to ensure that the benefit estimates that are used are .

There have been individual studies of recreational angling in specific areas relating the choice of recreation site to measures of travel cost and proxy measures for the availability of specific fish populations that could serve as a basis for benefit transfer. Most of this work has focused at marine fishing. However, some studies have been undertaken for freshwater systems. [37] And in at least one case, EPA has used such a study in a benefit-cost assessment. For example, when estimating the recreational benefits of reducing acid deposition in Adirondack's lakes, the Agency used a fairly recent 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 pH of lakes was used as an indicator of the level of ecological services from each lake. The literature on the economics of recreational angling shows that likelihood of success as measured by numbers of fish caught is a major determinant of demand for recreational angling (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. And to the extent that 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 benefits transfer.

There are several types of alternative models that can also be used that would allow direct use of the outputs of some type of ecological model for ecological impacts.

One possibility would be to use an existing model linking the physical descriptors of water quality to recreation behavior to estimate the benefits per trip for a change in water quality conditions comparable to the rule's effect, had it been experienced in each of the areas. These estimates could then 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 benefits. Such a meta function could then be considered for other areas. (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 tradeoffs 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 tradeoffs are represented in the summary function. There is precedent in the literature on 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).

A second class of models for evaluating stressors affected by the rule are the stated preference and stated choice models that highlight water quality attributes. While the record here is not as extensive as it is for the revealed preference random utility (RUM) models, there are several candidate studies (references??). These analyses are based on surveys that elicit respondent choices among a set of options, plans for reducing effluents or for improving water quality defined in terms of pollutants and or characteristics of ecosystems. 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 projected unit benefits derived. Then the factors affecting the benefit measure for each are used with a model in a summary analysis that can facilitate transfer to areas that do not have such models but are affected by the rule.

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. Residents' willingnesses 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. For examples, see Leggett and Bockstael (2000), Mahan, et al. (2000), Netusil (2005), and Poulos, et al. (2002). These estimates could then be candidates for use in a benefits transfer.

A preferable approach for estimating values based on recreation activities would be to do site-specific revealed preference (travel cost or random utility model) or stated preference analyses for a set of representative sites and to aggregate the results of these models to the sites affected by the rule. The difficulty in undertaking such an analysis stems from the limited regional character of the available applications. Often the affected areas represent very idiosyncratic local conditions and are not nationally generalizable. And time and resource constraints may preclude doing this kind of original benefits research.

Recommendation: To the extent possible, non-monetized ecological effects should be reported in appropriate units in conjunction with monetized benefits. In addition, aggregate monetized benefits should be labeled as "Total Monetized Benefits" rather than "Total Benefits."

Benefit assessments and RIAs should feature prominent discussions of ecological services that describe how ecological services were identified and analytical choices were made to assess and report on changes in service flows. In addition, they should clearly identify the values that were a) monetized using economic valuation methods, b) quantified (but not monetized), and c) described qualitatively. However, rather than simply designating them as "non-monetized", as for example in the CAFO benefit assessment, we recommend that the non-monetized but quantified impacts be reported explicitly (in conjunction with the monetized benefits) measured in the units that make sense from a biological perspective, and that the non-quantifiable impacts be described in as much detail as is feasible. Furthermore, any summary listing of the benefits and costs should include all three types of benefits, with the monetized and quantified benefits measured in the appropriate units (dollars or biophysical units). When monetized benefits are aggregated, the resulting sum should always be described as the "Total Monetized Benefits" rather than the "Total Benefits." In the past, EPA has sometimes reflected the non-monetized benefits in aggregate measures of benefits by including an entry in the summary table of benefits (and costs) such as +X or +B to indicate the unknown monetary value that should be added to benefits if the value could be determined. While such an approach indicates that the measured monetary benefits (and costs, too, if appropriate) is not a complete measure of benefits, the +X or +B provides little information about the extent or nature of the under-estimation and can be easily over-looked when the results of the benefit assessment are used. Always designating the sum as "Total Monetized Benefits" provides a continual reminder of what is (or is not) included in this measure. In addition, always reporting total monetized benefits together with key quantified but non-monetized impacts measured in biophysical units provides a more accurate and complete indication of total benefits than a simply designating total benefits by the sum of the monetary estimates plus an unknown factor X or B.

Recommendation: EPA should seek to build additional capacity, externally and in-house, specifically designed to facilitate ecological valuation for recurring rulemakings.

The committee advises the Agency to develop an extramural grant program focused on method development specifically for recurring rulemakings (e.g., for rulemaking 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 that will have foreseeable benefits for 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 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 data base for benefits transfer for ecosystem service valuation.

5.2.4 Uncertainty Analysis

Because of the difficulties in both estimating biophysical impacts of an EPA rule and the associated benefits or costs, it is important that EPA characterize the uncertainty associated with its benefit assessment.

Recommendation: EPA should include a separate chapter on "Uncertainty Characterization" in each benefit assessment and RIA.

The chapter should discuss the scope of the benefit assessment, the different sources of uncertainty [e.g., Biophysical Changes and their Impacts; social information about endpoints, valuation methods (including use of "benefit transfer")], and report on methods used to evaluate uncertainty. Within the section on "scope," the Agency should discuss the types of "socially important" values related to the issue that were included in the assessment and those that were excluded because they were not conceptually appropriate for the benefit assessment or RIA. 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 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 Benefit Assessment. Specifically, EPA should report: a) potential source of error; b) the direction of potential bias for overall monetary benefits estimate; and c) the likely significance relative to key uncertainties in the overall monetary benefit estimate. More generally, benefit assessments and RIAs should highlight in quantitative and qualitative terms any "socially important assessment endpoints" identified as appropriate for the analysis that were not monetized.

Recommendation: EPA should supplement RIAs with sensitivity analyses based on alternative models and methods for estimating economic values.

To stimulate the exploration and development of methods needed to enhance EPA's capacity for ecological valuation, EPA should seek, for each rulemaking, to conduct a sensitivity analysis using different methods from the core analysis, and preferably appropriate innovative methods, for one or more components of the core analysis. Such a sensitivity analysis would serve to develop experience with innovative methods and to test the results of findings in the core analysis. The plan for the sensitivity analysis should be discussed in the analytical blueprint for the benefit assessment or RIA or the rationale for not including the sensitivity analysis should be discussed in this document, which would be part of the public record for the rulemaking and available on line.

5.3. Conclusions

a) A significant barrier to any kind of valuation is the lack of information on how the levels of ecosystem services would be affected by the rule. 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 were not known.
   - The models 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 were not known (e.g., the relationship between changes in fish populations and changes in recreational angling).
   - The lack of site specific ecological data.

b) Methods exist for estimating economic values for at least some ecosystem services. But applying these methods in the context of new regulations could require original research that is costly and time consuming.

c) The Executive Order that mandates a benefit-cost analysis for major ruls adopts a national perspective. Thus analysts undertaking the research needed to prepare benefit-cost analyses have tended to favor models and or estimates that also have a national perspective. This so-called "top down approach has caused them to overlook the possibility of adapting a set of regional studies more closely aligned to the changes in the ecological effects so that these studies could meet the goals of a national analysis. This alternative "bottom-up" approach would proceed by establishing separate estimates for each regional grouping or group of similar facilities and then adding them together to obtain the national estimate.

d) Since economic values are context dependent, benefits transfer very likely requires a much larger set of value estimates than is currently available.

e) Methods exist for estimating non-economic values for at least some ecosystem services. But these methods do not have a role in a benefit cost analysis.

f) It is important to involve both ecologists and economists at the earliest stages in the development of an analytical plan for ecological benefits assessment.

g) There needs to be better communication between the program offices and the Agency's Office of Research and Development (ORD) concerning the research needs of the program offices and the resources available from ORD.

Text Box 1: 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 uneaten food and feces; 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 make possible 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 2: 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 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 that are 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 as well as 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.

<u>How were the environmental impacts quantified?</u>

Of all of the potential environmental impacts, the CAFO benefits analysis focused to a large extent on the nutrient runoff from land where manure has been applied and quantifying the 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.

<u>How were the benefits valued?</u>

Seven categories of 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 shell fish harvests were valued using estimated changes in consumer surplus. Water-based recreation was valued using the Carson & Mitchell study described in Text Box 1 above. Ground water contamination was valued using benefits transfer based on a set of stated preference studies. There was no national estimate of the benefits of reduced eutrophication of estuaries; but there was a case study on one estuary focusing on recreational fishing and using benefits transfer based on revealed preference random utility models.

There are a whole series of potential impacts that were not included in the 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 both a lack of models and data to quantify the impacts and, in some cases, the lack of methods to perform the monetization. Then there are a whole series of ecosystem impacts that were not considered—e.g. the potential changes to aquatic ecosystem functioning that relate to their capacity to produce goods of value to society.

Text Box 3: The Prospective 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 ecological benefits of reductions in air pollutants to be expected from the 1990 Clean Air Act Amendments (US 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):

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 and 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 ecological benefits based on standard economic models and methods:

- benefits to commercial agricultural associated with reductions in ozone,
- benefits to commercial forestry associated with reductions in ozone,
- 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 than 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 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 benefit measures to anglers depend on individual anglers' travel costs and site alternatives.

The Agency also presented an estimate of the benefits of reducing nitrogen deposition in coastal estuaries along the east coast of the US. In order to estimate the 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 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 Part 3 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 benefits, but only to show them as an illustrative calculation.

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6. VALUATION FOR SITE-SPECIFIC DECISIONS

6.1. Introduction

Among the numerous environmental management processes and related decisions that face the U.S. Environmental Protection Agency (the Agency), many are related to specific operating or formerly operated industrial and or municipal sites. The social and ecological implications of such decisions generally are local in nature and affect society at the level of towns, townships and counties rather then at the level of states or regional geographies. Therefore, the goals and performance objectives for these decision processes and their specific decisions need to rely on valuation approaches that are geared to similar levels of spatial sensitivities and are robust enough to adapt to the range of local stakeholder interests that may come to focus through the decision process.

In general, the types of regulatory processes that occur at this geographic level under the Agency's or its delegated authority's (i.e. individual states) include: a) permits (air, water and waste); b) policies that influence the boundaries for establishing permits (e.g. impaired water bodies designations); and c) administrative orders related to environmental contamination linked to recent non-permitted releases or historical practices prior to current regulatory standards.

In this section we have focused on the regulatory processes associated with the remediation and redevelopment of historically contaminated sites. In particular, we focused on the Superfund program and its efforts to assess the benefits to ecosystem services from site redevelopment efforts (Davis, 2001; Wilson, 2004). But ultimately the discussion that follows is generally applicable to any remediation and redevelopment processes for contaminated properties that contain the following basic and common elements:

a) Site Selection - Identification, selection and prioritization of sites
b) Site Characterization - Establish site condition
c) Site assessment – Evaluation of risks and impacts
d) Remedy Selection - Remedial and redevelopment
e) Performance Assessment - Clean-up and redevelopment
f) Public Communication: Assessment results; proposed actions and outcomes

Our goal in this exercise was 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 which could be relevant to specific steps are identified and discussed briefly. In no way is this an exhaustive list of what could be done and the exclusion of a particular method is not implied to mean it is not appropriate for any of the steps discussed.

To explore the opportunities for valuation we have selected to align our analysis with the recent efforts by the Agency's Superfund Program. As noted above, Wilson (2005) 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 of the individual redeveloped properties, it does provide a useful platform from which we can further explore the utility of valuation methods in the remediation and redevelopment process. In preparation for his analysis Wilson (2005) reviewed ~ 40 superfund cases before selecting three case studies which represent urban (Charles George Landfill); suburban (Avtex Fibers) and exurban (Leviathan Mine) environments. We have chosen to analyze and rely on these same three cases to illustrate our discussions about the utility of valuation in the various stages of the remediation and redevelopment process. In addition we have introduced an additional urban example, the Dupage Landfill because it provides a useful counterpoint to the Charles George Landfill example. The Dupage example (http://www.epa.gov/superfund/programs/recycle/impacts/pdfs/dupage.pdf) 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 boxes 4-7 integrated in the following text.

6.2. Opportunities for using valuation to inform contaminated property decisions

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 need for and/or use of valuation information related to Agency regulatory programs. For waste management and remediation programs (Superfund/RCRA/Brownfield/UST) seven of the eight regions responding indicated that information to help value the protection of ecosystems was needed. Our goal is to help direct the Agency in building the capacity to satisfy that stated need.

The Superfund process and its individual steps or stages are well defined (U.S. EPA CERCLA Education Center, 2005). The steps in the process are provided in Table 6: Steps in the Superfund Process.

Table 6: Steps in the Superfund Process

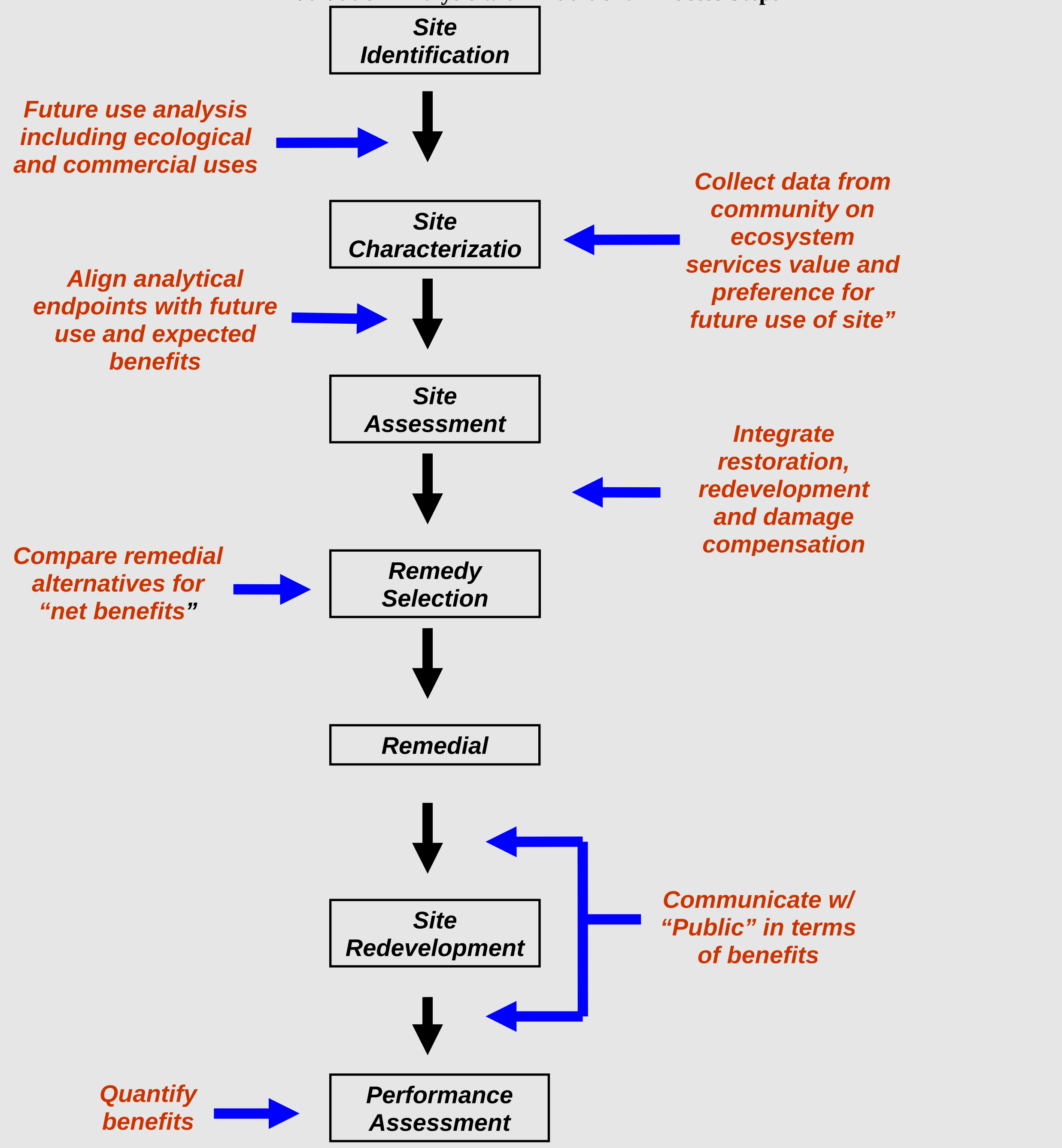
1	Discovery and Notification	Initiation: The Superfund cleanup process begins with site discovery or notification to EPA of possible releases of hazardous substances. Sites are discovered by various parties, including citizens, State agencies, and EPA Regional offices. Once discovered, sites are entered into the Comprehensive Environmental Response, Compensation, and Liability Information System (CERCLIS), EPA's computerized inventory of potential hazardous substance release sites (view CERCLIS Hazardous Waste Sites). EPA then evaluates the potential for a release of hazardous substances from the site through these steps in the Superfund cleanup process:
2	Assessment	Preliminary Assessment/Site Inspection (PA/SI) — investigations of site conditions
3		Hazard Ranking System Scoring — screening mechanism used to place sites on the National Priorities List
4		NPL Site Listing Process — list of the most serious sites identified for possible long-term cleanup
5		Remedial Investigation/Feasibility Study (RI/FS) — determines the nature and extent of contamination
6	Decision	Records of Decision (ROD) — explains which cleanup alternatives will be used at NPL sites
7		Remedial Design/Remedial Action (RD/RA) — preparation and implementation of plans and specifications for applying site remedies
8	Cleanup	
9	Closeout	Construction Completion — identifies completion of cleanup activities
10		Post Construction Completion — ensures that Superfund response actions provide for the long-term protection of human health and the environment. Included here are Long-Term Response Actions (LTRA), Operation and Maintenance, Institutional Controls, Five-Year Reviews, Remedy Optimization, and NPL Deletion

More generally, 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. M ore recently the evolution of Brownfield initiatives (insert a citation or two) has advanced the integration of a redevelopment focus upstream in the remediation process.

In response, the Agency built the Reuse Assessment tool (Davis, 2001) to integrate a focus on Land-Use into CERCLA/Superfund process. By driving remediation and redevelopment closer as a unified process the need to bring value concepts and considerations to the front-end of the process and carry them through the individual steps or stages of the process becomes evident. Net Environmental Benefit Assessment (Efroymson et al. 2004) is a recent advance in thinking that provides a framework for using valuation tools to inform the comparison of alternative remedial strategies. Similar efforts are needed for other steps in the Remediation and Development process.

As noted above a generic process that encompasses the remediation and redevelopment would include a series of steps or discrete activities. Figure 6 represents a generic remedial process on 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 will carry through to inform assessment of the site and the ultimate selection of remedial actions and redevelopment options. Optimally, by expressing expected and/or capture benefits will lead to more effective communication with concerned publics. The opportunities and utility of such adaptation of valuation methods to this new merged process is discuss in the following sections.

Figure 6: Changing Focus from Remediation to Redevelopment Would Benefit from Increased Integration of Valuation Analysis with Traditional Process Steps



In general, valuation methodologies should be most useful for identifying how a site and the current or potential ecosystem service flows matter to the surrounding community. Such methods should be focused on determining what benefits can be or have been derived from the site and how any potential effects on the ecological components diminish those benefits. When the ecosystem services that matter to people are well defined and when the assessments of ecological production and risk can be coupled to 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. Therefore a key recommendation is that consideration of ecosystem services and their benefits to human well-being and other forms of value need to be considered from the earliest stages of addressing contaminated properties.

Even as early in the management process as site selection or prioritization, tools which allow for comparison among sites for their ecosystem service s potential could be informative. Additionally, valuation can be used to capture the benefits linked to site ecological attributes and identified ecosystem service s to the surrounding community. Data that supports or aids in the design of benefits assessment 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 ecosystem service s flows and how they matter would lead to a better alignment of ecological risk and economic benefit assessments. Aligning risk and benefits assessment should be a critical objective for the Agency as it will assure that the remedial actions selected for consideration will address the restoration of the benefits derived from any important ecosystem service flows that have been diminished or disrupted. As well, aligning risk assessment endpoints with ecosystem service s and the derived benefits from those services should lead to improved a) alignment with community goals; b) ability to better perform meaningful benefits assessment and c) ability to communicate proposed actions and d) ability to monitor and demonstrate performance

As has been pointed out through the introduction of Net Environmental Benefit Assessment (Efryomson et. al., 2004) valuation can be a useful approach to aid selection among remedial technology options by weighing and comparing the benefits among the options. Incorporating valuation methods into the NEBA framework would provide the basis for balancing trade-offs between risks and benefits of the ultimate remedial design. Additionally it will aid in keeping the set of ecosystem service s preferred by the community as driving function in the prioritization and selection of remedial and redevelopment actions.

Ultimately, the test of the process is to what degrees were the ecosystem service s and associated benefits of importance to community either protected or restored. If as originally recommended, values have been broadly explored and effectively highlighted and integrated into the site assessment and remedy selection processes then measures of performance will be apparent. Ecological measures of productivity or aerial extent of condition which are 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 in the future the overall performance of remediation and redevelopment of contaminated sites.

Finally, the remediation and redevelopment of a property is really an exercise in social engineering that encompasses more then just the biological, chemical and physical sciences and engineering principles that historically have underpinned the remediation process. Therefore, effective communication with stakeholders, those actively participating in the management process and the general public is a critical element to success of the management process. Both of these audiences will be bringing a value set to the table when they are assessing any proposed actions or evaluating the results of any action taken. Therefore having a strong alignment between the ecosystem service s valued by these audiences and the expected or actual outcomes will facilitate effective communication.

6.3. Use of source examples to illustrate recommendations

In Part 1, Section 6 of this report, a series of high-level recommendations were provided. In essence, it was recommended that ecological values and benefits derived from ecosystem services should 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 to align them with what matters most to the people involved or affected by the decision. This does not direct the agency to ignore important ecosystem services whose value is not recognized by any community but to more broadly consider stakeholder preferences in their planning and analysis. To the degree there is a conflict of values a facilitated process to educate all parties could be useful. Additionally the Agency is encouraged to explore expanded use of socio-economic and ecological models to characterize and measure the values associated with environmental change.

In order to facilitate the charge to expand its focus on values, it is recommended that from the outset that expertise and opinions be brought to the process by integrating technical disciplines and engaging interested and affected stakeholders. Ultimately by aligning those ecosystem services that benefits people the most with ecological production functions that drive their availability, the Agency will be able to focus its actions to produce maximum protection or in the case of contaminated site maximize restoration of benefits.

In the following text (sec. 6.4) we have taken and adapted those general recommendations to the site-specific application context. The recommendations are presented in Table 7: Recommendations for Ecological Valuation for Site-specific Decisions. In addition we have supported these site-specific recommendations with lessons gleaned from a series of Superfund examples at the Urban (Charles George and Dupage Landfills), Suburban (Avtex Fibers) and Exurban (Leviathan Mine) demographies. Text boxes 4 and 5 provide background on the urban landfill cases. Text box 6 and 7 provide background on the suburban and exurban cases respectively.

Table 7: Recommendations for Ecological Valuation for Site-specific Decisions

	Recommendations	**Supporting Actions**
1	At the beginning of the process, broadly define the range of ecological services and associated value(s) recognized as important by key stakeholders and the community at large as attributable to the site or locale. To achieve this objective:	• Explore the utility of a variety of group process (e.g. Deliberative facilitated) and survey methods (e.g. Social-Psychological or “attitude”) to engage stakeholders in this process from the outset. • Consider the many sources of ecological value including both instrumental and intrinsic. • Consider not only current or diminished ecological services, but also the potential for developing or enhancing ecological services not presently utilized.
2	Appropriately involve the right mix of interdisciplinary collaboration from physical, chemical, biological (ecology, toxicology etc.) and social scientists (economists, social psychologists, anthropologists, etc.) in line with site-specific considerations and conditions and the specific step in the process	
3	Clearly demonstrate the alignment between ecological services the ecological functions that produce those services and potential positive or negative effects from current conditions or proposed agency actions. To achieve this objective:	• Develop the capacity to utilize an ecological – economic conceptual model to inform the site assessment design. • Develop the “accounting rules” to recognize and avoid double- counting or under-counting the benefits from ecological service flows. A consistent focus on production function will aid this objective. • Develop approaches to sort, weight or otherwise prioritize ecological services for primacy for actions and also to weigh benefits derived.
4	Expand the variety of methods in the Agency’s arsenal to quantify the ecological service, to describe ecological production functions and to capture in monetary and non-monetary terms the value lost or gained from current conditions or some proposed agency action.	• Explore the current state and extent of ecological production function models • Develop a strategy for adapting existing general models to site-specific applications
5	Develop the capability to utilize valuation techniques to provide a basis to demonstrate Agency performance and communicate the expected or actual outcome from Agency actions.	
6	Create formal systems and processes to foster an information sharing environment.	• Actively document lessons-learned from applications of valuation methods and share broadly among program and project managers.

6.4. Source Example Analysis

Recommendation #1. *At the beginning of the process, broadly define the range of ecosystem services and associated socio-economic value(s) recognized as important by key stakeholders and the community at large as attributable to the site or local.*

Broadly define ecosystem services early in process. The urban examples of the Charles George (See text box 4) and the Dupage County landfills (See text box 5) strongly show the difference in outcome that can be produced by engaging with community to focus on the ecosystem services of importance to them. Although there was no evidence of formal valuation methods at the onset of either example, the focus on how the site will provide future benefits and the inclusion of additional disciplines form lead 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 that they were discovered that they controlled the focus of the subsequent decisions. When the landfill site was capped and the water system from the city of Lowell, MA was extended to the affected community, the health and safety concerns were addressed. Although an effort to make the site work environmentally has now begun (insert URL for restoration plan), still some 20 years later, 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 value (rare birds; e.g., hawks) and recreational (e.g., hiking, bird watching, boating, camping, picnicking, sledding, etc.) benefits. 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.”
http://www.epa.gov/superfund/programs/recycle/impacts/pdfs/dupage.pdf

Engage key stakeholders. Public input for the Axtex Fibers case was an evolving process of growing complaints about offensive sights and smells and about contamination of drinking water wells. Over several decades, local government and environmental protection agencies made tests, filed thousands of complaints and took various regulatory actions that ultimately resulted in the listing and designation as a Super Fund Site. Once the site was listed and a management process established there was a clear effort to engage stakeholders through the

Multi-stakeholder process for development of the master plan. But as in the case of the Dupage landfill, although ecosystem services or at least ecological components were considered, it is not clear there was any systematic assessment of "what people cared about" regarding the Avtex site. Whether a more formal assessment of values would have reached a different or clearer description of community values is an open question. In any case, the commissioned Master Plan (insert reference or URL) that was developed in interaction with a "Multi-Stakeholder Group" implies that ecological restoration and ecosystem services (especially relevant to water quality) were important considerations for cleanup and redevelopment of the site. A substantial part of the site plan is devoted to restoration of forests consistent with natural conditions at the site, and waste pits are being redeveloped as ponds and meadow/wetland areas to provide important runoff control, water purification and wildlife habitat services. Much of the redevelopment of the site is directed at enhancing aesthetic values by restoring naturalistic landscapes to be enjoyed by recreational users, nearby residents and passing tourists.

<u>Define the ecosystem services that matter to people</u>, Determining what people care about requires a carefully constructed and systematically implemented program integrating assessments of multiple values using multiple methods to fairly and faithfully reflect the perspectives of multiple stakeholders. There is no simple recipe for accomplishing this, and no simple algorithm for calculating values and summing them up to make a decision. Value assessments serve to support decisions that must in the end be based on the judgment of administrators charged by society with that responsibility.

The Leviathan mine is a good example of how the Agency 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 native people and the community of occasional recreational user. The recreational users would gain from the cultural services associated with hiking, fishing and camping. However the Washoe tribe which lives in the area year round would benefit from the resource both as a provisioning service for food but also from the spiritual and cultural services.

Additionally the Leviathan mine case study highlights the need to consider the existence or intrinsic value of the ecosystem. For example, the ecosystem near the Leviathan mine site provides habitat for threatened species such as the Lahontan cutthroat trout and bald eagle, which many tribal and non-tribal individuals might value even though they provide no direct instrumental or use value. In considering site restoration or remediation, or measuring damages from contamination at the sight, the Agency would be missing the primary sources of value if it limited consideration to standard types of use value and did not consider these other sources of value as well.

Finding effective ways to both understand the values of disparate users for the same resource and to effectively weigh their interests in restoration and redevelopment of the site is not a small challenge. To include the relevant sources of value in an assessment, the Agency has to determine what aspects of the ecosystem generate these values, i.e., what aspects of the site contamination are of greatest concern to people. For the Leviathan Mine case, it is likely that this would have to be considered separately for tribal and non-tribal individuals, since the sources of value are likely to be different for these two groups.

<u>*Recommendation #2.*</u> *Appropriately involve the right mix of interdisciplinary collaboration from physical, chemical, biological (ecology, toxicology etc.) and social scientists (economists, social psychologists, anthropologists, etc.) in line with site-specific considerations and conditions and the specific step in the process.*

<u>*Integrate disciplines*</u>. Interrelationships among experts and between experts and the affected publics form a key component of any hazardous site assessment, planning and implementation program. Ideally, collaborations among all relevant experts and communications with affected publics/stakeholders begins very early in planning and decision making and remains active throughout implementation and post-project monitoring and evaluation. A key point for collaboration among expert disciplines is in 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 another instance of the need for integrating unique or non-traditional disciplines into efforts to understand what affected human population's value. Because of the unique cultural and spiritual values associated with ecosystem services, anthropologists could be involved in understanding and quantifying or characterizing the value of the ecosystem services to the Washoe Tribe. Likewise, in order for economists or others to try to estimate existence value for an impacted species (e.g., fish), it is necessary for them 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 this change can be valued.

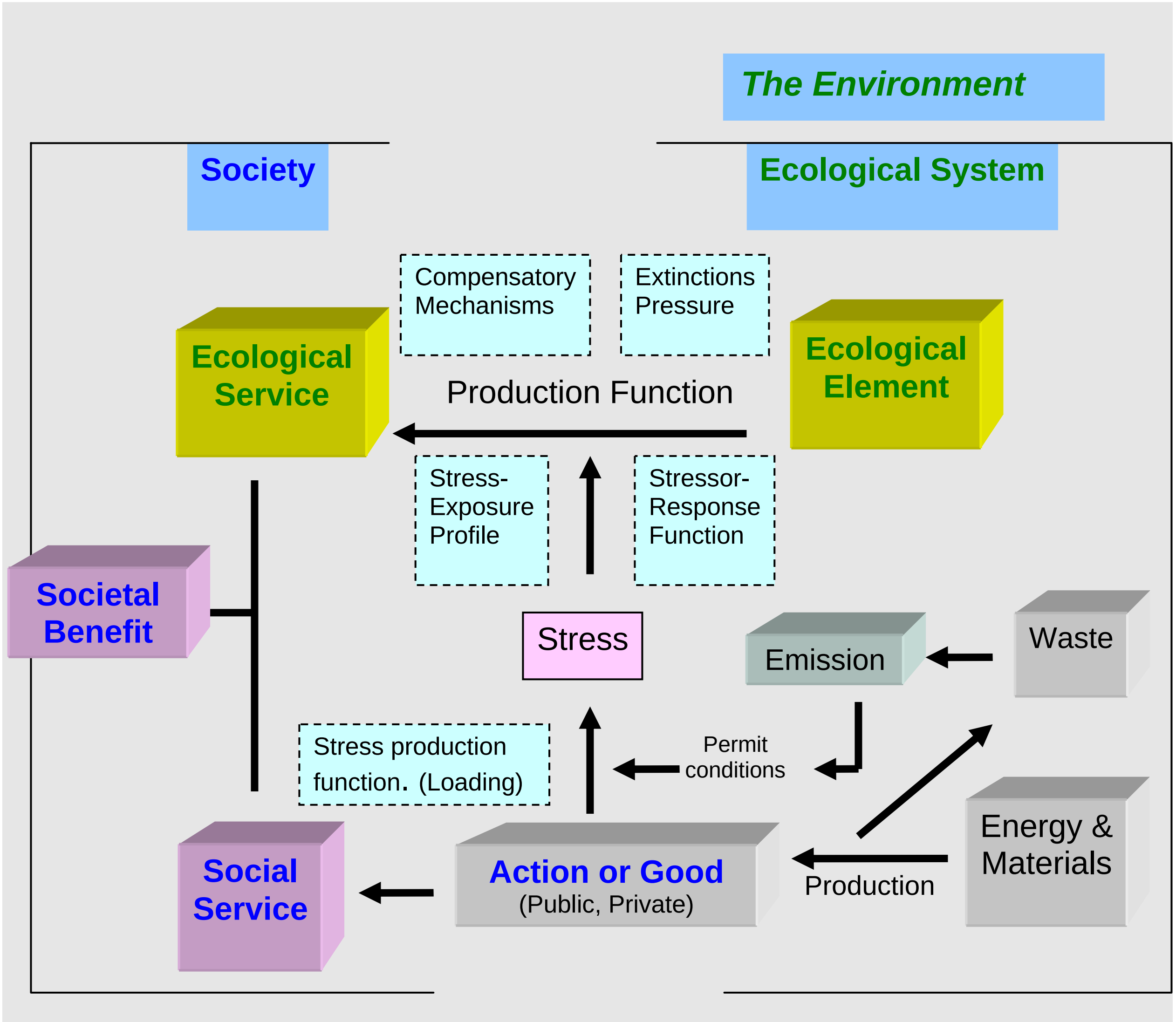
Recommendation #3. Clearly demonstrate the alignment between ecosystem services, the ecological functions that produce them and potential positive or negative effects from current conditions or proposed agency actions.

The call for alignment between ecosystem services, production and risks in recommendation # 3 is at the technical core of performing any of the risk or benefit assessments associated with the remediation and redevelopment of contaminated property. Unfortunately none of the source examples chosen for this example provide a demonstration of active intention to create such alignment. For the most part the best we can do is use the examples to illustrate where for those cases we believe such alignment would have influenced the results in a positive manner.

Utilize an ecological- social value conceptual model. Developing a conceptual model is an expected and standard practice in performing ecological risk assessments for contaminated site evaluations. A conceptual model which integrates and aligns the ecological aspects of risk with economic benefits from existing or foregone ecosystem services would facilitate better alignment between remediation and redevelopment. The primary focus of the Agencies efforts is to control anthropogenic sources of chemical, biological and physical stress which could lead to adverse impacts to human health and or the environment. Traditionally, the Agency relies on a combination of technology-based and risk-based approaches to establish acceptable or permitted levels of stress. In general Agency approaches to characterize potential exposures and the possible effects to those levels of stress are not linked to the ecological production functions which drive ecosystem services generation.

Figure 7 attempts to provides a generic and simplified representation of the linkage between the environmental stress generated by public and private efforts to generate societal services or manufacture goods, and its potential (i.e. risk) for affecting ecological production and their associated ecosystem services to society . It is at the interface between our ability to estimate risk and our lack of knowledge of what the real consequences are to ecosystem service production, if that risk goes unchecked, that the Agency needs to focus its efforts to advance its capabilities.

Figure 7: Ecosystem Services Linkages Conceptual Model



The potential benefit to the Agency from developing the capacity to use conceptual models that integrated ecological and social value attributes of the site is highlighted by the Avtex Fiber case. Health threats to workers and to nearby residents were highly salient concerns and strongly guided initial management plans and actions at the Avtex site, potentially reducing opportunities to recognize and address important ecosystem risks and associated ecosystem services. Technical risk assessors and public observers/participants would have benefited from a clear and comprehensive model of the ecological roles being played, and that potentially could be played by the Avtex site. Early concerns about contamination of groundwater and discharge of toxic substances into the Shenandoah River focused attention on water quality.

A noteworthy feature of the Avtex Fiber process was the development of a Master Plan. There is evidence form that plan that ecosystem services were considered. For example, aquatic basins constructed to contain contaminants on site were also designed to restore important ecosystem services as well, including providing safe habitat for water fowl, runoff control and water purification services. In this regard some at least rudimentary ecological production function is implied by the plan although not documented or for that matter their benefit quantified. It is not clear that other aspects of the Avtex Fibers site Master Plan were as effective at addressing ecological risks or at capitalizing on opportunities to enhance ecosystem services in the redevelopment of the site.

The development of an ecological-social value conceptual model would have systematically informed greater integration of building ecosystem service into remedial design and future uses. To that point, recreational and aesthetic services were clearly important considerations for many features of the plan, but it is not clear whether the specific allocation of facilities and uses to spaces within the site was guided by any comprehensive ecological model. For example, it is not clear whether the particular pattern of restored forests and wetlands, developed recreation areas and industrial park produces the best possible outcomes for protecting ecosystems and ecosystems services. It is possible that different siting and design of the soccer fields, for an example, might have returned the same recreational benefits while achieving greater ecosystem services in the form of wildlife habitat, water quality or aesthetic values for visitors and/or nearby residents. The declared ecological, "green" focus of the industrial park component of the master plan implies that ecological concerns will be paramount 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 is presented. This leaves open the prospect that future industrial, recreational and tourist developments and uses at the Avtex site might simply substitute one set damages to ecosystems and ecosystem services for another.

<u>Need for "accounting" rules to count benefits</u>. Ecosystems and their numerous components are linked in an intricate and complex network of biological, chemical and energy flows. By looking at impacts to individual organisms or components and their associated services in isolation, there is a serious opportunity for double counting service losses and or benefits generated by Agency actions.

For example, the listing of services (aquatic biota and habitat, riparian vegetation, terrestrial wildlife, recreational uses, and tribal uses) in the Leviathan Mine case does not seem to be very useful for sorting out the different things to be valued. It does not identify mutually exclusive services and seems to have a high likelihood of double counting. It also does not seem to adequately distinguish between "inputs" and "outputs." As well, the question of why we care about protecting habitat or riparian vegetation is not clearly addressed. Is it because we care about the populations it supports for their own sake, or because these populations are an input into something else we value, such as recreation? Take insect populations, if we care about the insects for their own sake, then maybe this should be included as an existence or intrinsic value. If we care about them because they are a food source for fish and we care about fish, then we should value the change in fish brought about by the change in insects but not value both separately, i.e., we should view both clean water and insects as inputs into the production of more fish, and value either the inputs or the output. Of course, then there is the question of why we value the fish because of their existence, their recreational use, or their cultural significance to the Washoe tribe. Perhaps part of this whole exercise is to first try to answer the question of why we value the insects or fish. It seems we need to know this before we can figure out how to measure how much we value them.

Similarly, the listing of services by Wilson (2004) shown in Table 8: Ecosystem Service Matrix for Leviathon Mine (from Wilson, 2004), based on the U.N. Ecosystem Millennium Assessment (2005) definitions of ecosystem services is not very useful for valuation purposes, and in some cases we believe it could create confusion in valuation. For example, it is not clear how or where the use of surface water or groundwater for drinking would fit in Wilson's list. Is the service from "Freshwater Regulation" intended to include drinking water or is it intended as an input into aquatic and other habitat-related services? The valuation approach used is likely to be different depending on which of these services freshwater regulation is intended to reflect.

Table 8: 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 beneth 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 speies 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 services (defined as outputs rather than inputs) would be the following:

a) Water used by Washoe Tribe members and others for washing and drinking
b) "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)
c) Non-consumptive use values of wildlife (e.g., people like to view bald eagles and other species)
d) Harvesting (hunting, nuts, fish) by Washoe tribal members
e) Cultural/spiritual and ceremonial value of land used by Washoe tribal members

f) Water flow regulation (e.g., reduction in flooding from snowmelt or runoff)

g) Non-tribal recreational services (e.g., fishing, hiking, camping)

h) Value of the natural process leading to ecosystem outputs, above and beyond the value of the outputs themselves (e.g., preference for natural processes over man-made ones, or native species over introduced species)

In any case, it is clear that there is a need to establish some accounting guidance for working with complex social and ecological situations.

<u>Align ecosystem services with ecological production functions and impacts/risks</u>. To achieve that objective of alignment the Agency will need to bring forward in the planning process for site remediation and revitalization a robust discussion of what are the ecosystem services and to what degree they matter to the affected local community or the ability of the environment to sustain its integrity. To some degree the Agency has already settled on the concept that ecological risk assessments need to be built on an ecological construct and a conceptual model that is linked to an assessment endpoint. The gap in practice maybe as simple as doing a more thorough analysis of the breadth of ecosystem services and how they matter to people. This will present technical challenges as today the design of ecological risk assessments are dominated by what toxicological data we have in the literature for a limited range of species. It is very likely that the species data we have will not link well to the ecosystem services that matter. This may require the agency to revisit its assessment approach for chemical exposures from an ecosystem services perspective rather than toxic response of individuals. In the mean time, more attention to creating the alignment between ecosystem services, the assessment and measurement endpoints used in the risk assessment and the ability of economists and other social scientists to estimate value will likely lead to significant improved outcomes in efforts to revitalize land. In addition, a significant Agency effort to estimate the population or community level consequences of chemical exposures on ecosystem service flows will advance this objective greatly. To do that the Agency will need to develop the capacity to adapt and apply ecological production models in its contaminated sites assessment processes. 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 as well as the remedial and restoration alternatives.

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 the Agency in the remediation process are often the basis for damage assessment. That extrapolation from risk to injury and then onto damages is often a significant point of departure in the dialogue for the Agency and their trustee partners with the parties responsible for the damages. The uncertainty in estimates of chemical exposure, toxic response and therefore the estimate of risk makes using these data as a surrogate for injury to the environment controversial and therefore the resultant damage claim for reduction in human use or ecosystem services is likely to be challenged. Damages are an expression of the needed restitution for lost or forgone use of ecosystem services. To link risk or potential for injury with actual loss of service and the estimate of the values of that service (i.e. damages), will require linking ecosystem services with the environmental components producing those services and then defining the risk to or likely response of those ecological components to chemical exposures.

The Leviathan mine case illustrates both how the concept of ecosystem services has and can be used in damage assessment and restoration, as well as some of the challenges associates with delineating services in a way that is useful for valuation. One could suggest that if the agency can achieve the recommendation to align ecosystem services, their production functions and risk profiles then it would also benefit the ability of resource trustees to appropriately assess injury, define restoration goals and calculate damages

In the Leviathan Mine example, impact or injury is defined not only as exceeding of some standard (e.g., water quality or drinking water standards) but also as concentration or duration sufficient to cause a loss of services provided by the resources to the general public in addition to unique service losses 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; also know as Resource Equivalency Analysis or REA) to determine compensation for damages. In principle, application of the HEA concept requires a determination of the flow of ecosystem services that would have been provided by a given site “but for” the contamination and a comparison of this flow with the flow of ecosystem services provided as a result of a restoration or other project designed to generate an equivalent service flow. 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 implicitly value the service flows.

How can the impact of the site on these services be estimated? The Leviathan Mine Natural Resource Damage Assessment Plan (NRDAP) gives detailed information on concentrations of key pollutants (in particular, 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 are compared to levels at reference sites (since historical information is not available) to illustrate impact. In addition, they are compared to baselines determined by water quality criteria (under the CWA), drinking water standards (under SDWA), etc. to quantify the magnitude of the impact/injury. In general, unacceptable risks are defined based on toxicity thresholds or other concentration criteria, as well as on the extent to which species impacts based on comparable concentration levels are documented in the literature.

Once the impacts on water quality, sediments, etc., have been determined, they need to be translated into predicted changes in the flows of the services listed above. In principle, it requires the estimation of an ecological production function. For example, to see if recreational fishing is likely to be significantly impacted, we would need to estimate the impact of the site on the fish population in the nearby water body. This requires estimation of the impacts of the changes in things like water quality, streambed, bank sediments and riparian vegetation, on fish population, both directly and indirectly through their impact on the insect population. For example, if we know that there are elevated levels of arsenic, copper, zinc, cadmium, etc., in insects and fish tissue, how do we use this information to predict an overall impact on the fish population? In most cases, an ecological model for doing this at a particular site such as the Leviathan mine will not exist.

How then should EPA proceed in trying to look at not only the impact on ecosystem resources but also (or instead) the impact on ecosystem services? At this stage, EPA might instead look at the scientific literature to see what it says about how sensitive the insects and fish species of concern here are to these types of stressors and then ask expert ecologists to provide some expert judgment on the likely magnitude of the impacts in this specific case. (This would be akin to an "ecological impact transfer", similar to the notion of benefits transfer.) In fact, the Leviathan Mine NRDAP suggests this.

In addition, the Leviathan Mine NRDAP suggests looking at, for example, the fish population downstream of the mine and comparing it to the population in a reference location. More generally, it suggests comparing not only fish populations but riparian vegetation, the composition of the benthic community, wildlife populations, etc. near the mine and at a reference site. Such a comparison can determine the damages resulting from the mining activity (which is most useful in an NRDA policy frame), but it cannot directly predict the impact of proposed remedial actions on ecosystem services (which is needed for a policy frame relating to evaluation of remedial actions), unless we make the assumption that the remedial actions will be 100% effective in restoring the ecosystem services to their original level (presumed to be the level at the reference site). Short of this, we would again need to predict the impact of the remedial actions on the ecosystem resources and then translate those into predicted changes in ecosystem services using an ecological production function. Such a balancing act could be assisted through the use of comparative tools such as Net Environmental Benefit Analysis (Efroymson et. al., 2004).

<u>Recommendation #4</u>. Expand the variety of methods in the Agency's arsenal to quantify the ecosystem service that matter to people and to capture in monetary and non-monetary terms the value lost or gained from current conditions or some proposed agency action.

<u>Expand methodological capacity</u>. Part 3 of this report provides an overview of a broad range of methods that could be explored for the integration of valuation into the typical contaminated property redevelopment. For any of the source examples selected to highlight local decisions, their decision making processes could have benefited from the application of a number of these methods. The Agency should be exploring the use and/or adaptation of many of the techniques listed in the methods section of this report to: a) engage stakeholders to define what they value; b) help align the sites risk assessments with expected benefits; c) test alternatives strategies for redevelopment to achieve those benefits and d) improve communications of proposed actions and their performance.

<u>Future uses that matter to stakeholders</u>. Determining local stakeholder interests with regards to preferred future property uses and the ecosystem services derived from that redevelopment scenario is an important starting point. Survey methods or facilitated dialogues

would be useful methods to achieve this objective. In helping to frame the dialogue with stakeholders, methods such as the Environmental Benefits indicators (Boyd, 2004; Boyd and Banzhaf, 2006) or the Biodiversity Indicators (Grossman, 2004; Stoms et. al., 2005) may be very suitable for helping Agency's site managers understand the ecosystem service potential from future uses and provide the basis for valuation by decision-aiding processes (see Part 3, section 6.3) or mediated modeling (Part 3 section 6.2) exercise.

The counterpoint represented by 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, it seems that only a qualitative focus on the utility of ecosystem services lead them to recognize 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 where hawks abound, so will birders to watch them. In this case, the difference is not one of methodology so much as conception – once planners "see" an area as having ecological potential, it may be a fairly easy matter to point to qualitative differences to show, by way of analogy and example, likely quantifiable or monetizable consequences. It might be a valuable learning useful exercise for the Agency to go back to a case like the Dupage or the Charles George Landfills and develop a valuation assessment plan. Such a plan would create a map of possible methods the Agency and responsible parties could use to integrate valuation into the decision process

For the Avtex Fibers site, deliberative group processes involving stakeholders and relevant experts (including historians) would have provided an effective approach to identifying the ecosystem and ecosystem service values of most concern to stakeholders. Systematic assessments of ecological values and of historic and sense-of-place values (assuming these were identified as important) are not well developed. Stated-preference monetary assessment methods, such as contingent valuation surveys (methods citation or reference to sec #) might be applied. People, however, have been shown to have difficulty expressing consistent willingness-to-pay estimates for such non-commodity outcomes and some people find assignments of dollar values to be ethically offensive in this context. Social-psychological "attitude" survey methods (See Part 3 section 4) could provide relative measures of preferences (importance, acceptance) for any defined population of stakeholders for the array of ecological, historic and sense-of-place outcomes across a defined set of cleanup and restoration/redevelopment options for the site.

For the Leviathan example there are at least three ways that information about the impacts of greatest concern to affected individuals might be obtained. The first would be to gather information about the relative importance of the various services in this particular context through focus groups, mental models, mediated modeling, deliberative processes and other similar methods. (editors Note: Suggested that this terminology was too general to be useful guidance and more specific methods might be an improvement) Similarly, for services relating to Tribal uses, anthropological or ethnographic studies based on detailed interviews can be used to determine the ecosystem features of most importance and the characteristics necessary for suitable restoration or replacement.

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 environmental benefit indicators. Examples would be: 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. The information regarding the Washoe could be collected through interviews. It is not clear whether some of the other data exist or would have to be collected.

The third approach would be a review of related literature and previous studies to draw from what has been learned in other contexts. For example, previous Social Psychological surveys (not specific to this site) or other expressions of environmental preferences/views (e.g., outcomes of referenda, civil court jury awards, citizen juries, 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 they 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 the use data above to provide an initial indication of the importance of the impact on recreational fishing.

Aligning ecosystem services with risk assessment. There is not a single method that could be identified which is focused on mapping prediction of ecological risk with Production functions and the services that derive from those structural or functional ecological units. Qualitatively or visually the linkages can be represented by the creation of an ecological – social value conceptual model as discussed previously in this section. Once a visual representation of the relationship between a stressor impacting ecological production and the change in an ecosystem service has been mapped, the agency is still left with the challenge of quantification.

The Agency has already approached the development of complex ecological risk assessment modeling tools (TRIM, EXAMS, AQUATOX) to estimate the fate and effects of chemical stresses on the environment and has even coupled such exposure-effects models with ecological production models to estimate population level effects. Although there not many examples of such integration it would not be impossible for the Agency to focus on expanding such capability by exploring the world of existing ecological production and ecosystem level models that exist in the literature (Roughgaren, Joan 1998a and 1998b; Roughgarden, Jonathan, 2001).

A major gap in the current ecological modeling capability is coupling the aforementioned modeling systems with models (or modules) that link ecological production models with explicit ecosystem services which can be quantified. This is very important because such a tool could be used not only to assess impacts and their acceptability but also as the quantitative basis for looking at the benefits derived by investments in alternative remedial and redevelopment strategies. Without this capability the agency is left with the narrow ability to look at risk reduction as the primary ecological benefit from any action.

Testing remedial and redevelopment alternatives. Currently the typical comparison of remedial alternative strategies includes two tests. The first test being does the action control risk to an acceptable level. All of those technologies that pass that minimal benchmark then go through a second test for cost-effectiveness. So if all technologies are adequate with regard to risk reduction then the least costly is the obvious choice. What such an approach does is decouple remediation and development, which leads to a delayed development process possibly off mark from what matters to key stakeholders.

If alternatives can be compared based on benefits generated then it opens up a number of methods that could be used to compare alternatives with or without stakeholder direct involvement. As mentioned previously, Net Environmental Benefit Analysis (Efroymson, 2004) is a framework for comparing remedial/redevelopment alternatives on a basis of benefits generated. Obviously the units of those benefits could be in either monetized or non-monetized units. For example in superfund sites the value of an action could be expressed through methodologies such as Habitat Equivalency Analysis (HEA) or Resource Equivalency Analysis (REA) (See Part e Section 7). Although HEA and REA generally produce 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. For the most part, we are looking to achieve a reasonably precise and representative measure of relative benefits for comparing alternatives. So to the degree that other methods which measure outcomes purely in ecological terms, such as the Biodiversity and Conservation Values approach (Grossman, 2004, Stoms et. al., 2005) provide a useful basis for comparison among options they might also be useful in conjunction with NEBA.

Comparison of alternatives via monetary/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. As well, input-output models (Editors note: comment was received that we should verify if this is correct use of term) 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/tourism-focused use options.

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

<u>Balancing tradeoffs</u>. Because the measures provided by the most common social-psychological survey methods are only relative (across the range of alternatives assessed) this would leave the difficult task of resolving tradeoffs among ecological, historic and sense-of-place values, and between these values and other values and costs, up to the decision maker. Given some consensus that improved biodiversity, habitat quality, energy flow, and/or other biological outcomes were desirable and important to stakeholders, ecological value assessment methods might provide effective and suitable quantitative indices for making comparisons among identified management alternatives in these terms. As with the social-psychological scales, however, ecological assessments would again leave the multi-attribute tradeoff questions to be resolved by the decision maker. This can be an appropriate allocation of decision making responsibilities in many policy contexts, but more sophisticated survey approaches could help to overcome some of the limitations of having only relative measures for multiple value dimensions (attributes), including protection of ecosystems and ecosystem services. Conjoint survey methods (see Text Box 12: What are conjoint surveys of attitudes?) require respondents to explicitly make tradeoffs among multiple value dimensions (attributes), thus revealing the relative contribution of each attribute (in the form of regression coefficients) to relative preferences among the cleanup and restoration/redevelopment plans under consideration.

Managing a site like Avtex Fibers is very complex, with many interrelated and interacting effects for ecosystems and for human society. Thus, a conjoint survey such as that proposed above would most effectively be conducted in the context of an informed, deliberative process, providing a limited set of motivated respondents with expert analyses and information about the inter-relationships among the many potentially competing values at play. For example, respondents would likely require more extensive instruction in the meaning of ecological measures (e.g., biodiversity, energy flow) and how they related to aspects of the actions and outcomes of the alternative management plans than is typically possible in any mass survey approach. In addition, in this context it could be useful for respondents to receive some expert feedback about the possible implications of their expressed preferences for management plans, as effects of environmental changes concatenate through ecosystems and social systems on-site and off, and changing over time. It is important in this context to recognize that the preferences that are derived (constructed) though such an informed deliberative process would not be representative of the reactions of the broader populations of stakeholders. For this reason it may be important to also conduct less intensive survey procedures with larger samples to better predict public response to the plans under consideration and to identify specific public information/education needs that should be addressed in communicating, justifying and implementing the decision.

Communicating outcomes. Additionally, the Agency should advance their capacity to communicate alternative futures and their associated benefits to stakeholders. Representation of scientific information is often obscure to lay audiences. Communicating in terms of the benefits the stakeholders can expect from proposed actions will help focus their interest. Additionally if there are visual ways techniques to represent alternative futures based on different actions it will help stakeholders understand the alternatives from an outcome basis. For example, both monetary/economic and social-psychological assessment methods 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/redevelopment alternatives under consideration. In any case the Agency can only benefit from developing communication tools that engage and satisfy the local community's concerns and demonstrates recognition of their preferred outcomes.

Recommendation #5. Develop the capability to utilize valuation techniques to provide a basis to demonstrate performance and communicate the expected or actual outcome from Agency actions.

If valuation concepts and techniques are incorporated early and often throughout the contaminated property redevelopment process then as is suggested in Figure 6: Changing Focus from Remediation to Redevelopment Would Benefit from Increased Integration of Valuation Analysis with Traditional Process Steps, the Agency should be in a position to communicate with interested publics. The expectation is that by effectively integrating consideration of ecosystem services and their derived benefits into the selection of the remedial and redevelopment actions, managers will be able to communicate "why" they selected the preferred options. 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 the benefits they will derive from the proposed actions, should lead to greater public acceptance of the proposed plan forward.

Additionally, the presence a clearly defined sets of aligned actions and projected benefits should make the selection of performance measures relatively straightforward. Communicating the progress or challenges to such progress as the redevelopment proceeds should be facilitated by having and using performance measures defined in terms of benefits that the interested public understands and accepts as important.

Additionally, the Agency should advance their capacity to communicate alternative futures and their associated benefits. For example, the restoration plan for the Avtex site included replanting and/or 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 (only one plan was actually proposed for the Avtex site), whether in an information context or for systematic value assessments. Achieving and effectively using such visualizations would first require interactions between foresters/forest ecologists and visualization experts (such as some landscape architects) to create 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 then be involved in creating 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. All of this could be a prelude and a perceptual component of a conjoint value assessment survey. The above example may seem a rather intricate process which will require significant time and resources, but keeping in mind that many contaminated properties are under redevelopment for years and in case of Superfund projects decades with proportional resource allocations, this level of effort seems appropriate.

Recommendation #6. Create formal systems and processes to foster an information sharing environment.

Actively document lessons-learned from applications of valuation methods and share broadly among program and project managers. Broad and rapid transfer of experience with integrating valuation concepts and techniques into the process of contaminated site redevelopment should be a lead objective for the Agency. In many ways no two local management situations are exactly alike, so the agency will ultimately build its capacity to utilize valuation to inform its local decisions through a systematic approach of local case-specific demonstrations. The lessons learned from these trial efforts, whether they be successes or failures need to be shared widely across the agency with the regions, program offices and the tool builders in the research organizations. There are a number of ways in which the agency could catalogue and share such experiences, such as reports, databases or BestNets (computer-based networks of users sharing best practices). Obviously the Agency is in the best position to know how to build off their existing information exchange systems, but however it is done the information should be shared broadly.

Text Box 4: 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, New Hampshire, was a small municipal dump. A new owner expanded it to its present size of approximately 55 acres and accepted both household and industrial wastes from 1967 to 1976. The facility had a license to accept hazardous waste from 1973 to 1976 and primarily accepted drummed and bulk chemicals containing volatile organic compounds (VOCs) and toxic metal sludges. Records show that over 1,000 pounds of mercury were disposed of 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 DEQE'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) were also begun in September 1983. The basis for the removal action was documented in the first ROD issued on December 29, 1983.

Text Box 5: 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 downgradient 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 residents 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.

Text Box 6: 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/or escaped into the Shenandoah River. The Avtex Fibers site was proposed to the National Priorities List on October 15, 1984, and the site was formally added to the list June 10, 1986. EPA began removal activities at the Site in 1989 to address various threats to human health and the environment. The cleanup/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 (cap).

Front Royal and 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 and 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: 1) a 240-acre River Conservancy Park along the Shenandoah River combining ecological restoration and conservation of native habitats; 2) a 25-acre Active Recreation Park with boat landings, picnic shelters, and a developed recreational area including a visitor center and soccer fields; and 3) a 165-acre Eco-Business Park, featuring the refurbished historic former Avtex administration building. Clean up of the Axtex site is on-going, and the redevelopment plan is being actively pursued by local government agencies and private industry groups.

Text Box 7: 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 of California, 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 1950's. 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 1950's 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 1 in the process diagram, 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 3 of the process diagram), and c) damage determination phase to calculate the monetary compensation that would be required (corresponding to step 4 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 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 (ROD) 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.

Editor's note: During committee discussions it was left open as to how to highlight the utility of NEBA for integration of ecosystem service valuation into environmental decisions. The options where 1) to provide a text box in this section or 2) develop a separate methodolological write-up for the methods section or 3) incorporate in the decision making methods section. Until that decision is made this is here as a placeholder. In an earlier version of the draft methods report we provided the following information on NEBA. This is clearly too much for a text box, so will need Discussion

Text Box 8: Net Environmental Benefit Analysis

The net environmental benefit analysis (NEBA) framework shares the same theoretical foundation as benefit-cost analysis. An important distinction is that, in NEBA only environmental effects of an action are considered. The NEBA approach identifies and values the primary environmental services that an area or portfolio of holdings may provide given different land uses and actions (e.g., wildlife management, building roads and infrastructure, siting facilities, discharging effluent, restoring stream habitat, etc.). The type, quantity, and quality of environmental services provided by an area or waterway are determined, in part, by the surrounding geographic landscape (i.e., land uses). The NEBA approach uses the recent emphasis (e.g., NOAA, DOI, USFWS) in the ecological sciences to consider environmental services within a landscape context. Proposed actions will affect the quality and quantity of ecosystem service s produced at the site or parcel differently. Some services may be improved, some may not be affected, and some may be harmed. A systematic evaluation of these changes in service flows is needed to make consistent comparisons across alternatives and to optimize the achievement of environmental objectives at least cost.

NEBA is a method comprised of a set of agency approved and litigation tested techniques and tools for quantifying the benefits of alternative land uses (e.g., restoration alternatives, land reuse designs) or actions (e.g., remedial alternatives) that affect the environment. The NEBA approach and quantification tools can be used to:

a) Estimate value of environmentally sensitive areas;
b) Develop and evaluate a suite of alternatives;
c) Provide a basis for balancing economic, human, and natural resource drivers affecting proposed alternatives;
d) Support measures to weigh and rank alternatives that meet cost effective objectives;
e) Provide a means to expand the range of potentially acceptable alternatives;
f) Provide documentation that provides a defensible alternative analysis and selection;
g) Provide basis for establishing appropriate mitigation measures; and
h) Provide performance-based measures that can be used to conduct monitoring and adaptive management activities.

When properly planned and implemented, the NEBA approach provides a systematic, consistent, and defensible process that can significantly enhance stakeholder support for selected environmental and land use planning decisions. This process also promotes the selection of decisions that demonstrate a balanced win for the environment and the stakeholders.

Since NEBA is a framework the resources, data inputs and limitations are principally going to be associated with whatever ecological models and an valuations tools that 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). Spatial or temporal scale does not seem to be an intrinsic limitation of NEBA rather more an indication of the experience in its application to date. As a framework NEBA should be highly adaptable to different levels of data, detail, scope and complexity.

Obstacles to its application would likely be more legal or regulatory rather than data or information. As some regulations may exclude or not implicitly include a benefits test then there may be organizational impedance to adding any additional steps. With regards to limitations associated with adequate data or information, those limitations would or should be controlled by the tools selected to support the NEBA process.

Uncertainty under NEBA would be controlled by the methods or tools selected to support the process. Therefore whether the uncertainty associated with the output from a NEBA evaluation was a formal or an informal characterization would likely vary from application to application.

Citations

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7. VALUATION IN REGIONAL PARTNERSHIPS

Editor's Note: Additional text on recommendations to be provided for this section.

7.1. EPA Role in Regional-scale Analysis of the Value of Ecosystems and Services

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, require regional-scale analysis. There has been a vast increase in publicly available spatially-explicit data on environmental, economic and social variables. There has been a parallel expansion in the ability to display data visually in maps, and to analyze spatially-explicit data using a variety of analytical models and statistical methods. The increase in data and methods has opened up new frontiers for regional-scale analysis of ecosystem and services. There is an active EPA extra-mural research program under way for regional-scale analysis of ecosystems and services. For example, EPA has funded research on restoring water infiltration in urbanizing watersheds in Madison, Wisconsin, restoring multiple ecosystem functions for the Willamette River, Oregon, decision support tools to meet human and ecological needs in rivers in New England, and research examining multiple services from agricultural landscapes in the upper Midwest. Great potential exists, largely untapped to date, to use this type of analysis to aid regional decision-making.

Many important decisions affecting ecosystems and the provision of ecosystem services are taken at a regional scale by municipal, county, regional and state governments. Examples of important regional-scale decisions affecting ecosystems and ecosystem services 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 services, or to incorporate the value of ecosystems or services into their decision-making processes.

Regional partnerships offer the potential for expanding local, state and EPA capacity to value ecosystems and services. EPA regional offices have many opportunities to partner at a regional scale with local and state governments, regional offices of other federal agencies, environmental non-governmental organizations and private industry. By partnering with local government, other federal agencies, and the private sector, EPA benefits 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 benefit from access to EPA technical expertise and resources. Such partnerships can improve the knowledge-base for decision-making and improve the analysis of the value of ecosystems and services.

Unlike national rulemaking, where analysis is often constrained by specific mandates, there is great latitude available at the regional level to experiment with novel approaches to valuing ecosystems and services. Such experimentation may lead to improved methods and practices with potential benefits well beyond the region in which they are pioneered. The downside of not having legal or statutory requirements for EPA to engage in regional partnerships or to undertake valuation of ecosystems or services at the regional scale, is that EPA regional offices with limited resources and with a long list of mandated activities, may have little time or resources 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 that can work on valuation exercises. Many of the potential benefits of regional partnerships for valuing ecosystems or services at a regional level have not been realized to date.

In analyzing the opportunities for regional partnerships, a C-VPESS subcommittee found it useful to explore 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. The subcommittee studied the example of Chicago Wilderness, a regional partnership involving EPA Region 5 and numerous local public and private partners, in greater depth. The subcommittee met at EPA Region 5 Headquarters in Chicago on April 28, 2006 with members of the partnership. The case studies included in this section are not meant to be a comprehensive summary of the many regional-scale analyses undertaken by regional office of EPA that relate to the value of ecosystems and services. Rather, they provide specific examples of approaches and issues likely to occur in doing regional-scale analysis. In what follows, details about the case studies are used to illustrate several general lessons about regional-scale analysis of the value of ecosystems and services and the potential benefits of regional partnerships.

7.2. Case Studies: Chicago Wilderness

Chicago Wilderness is an alliance of more than 180 public and private organizations. Chicago Wilderness represents a bottom-up organization that reflects the views of its member organizations. No single decision-maker or agency controls or guides Chicago Wilderness. Chicago Wilderness pursues objectives, as defined by its members, through consensus. .The member organizations Chicago Wilderness are brought together by a common interest in the environment of the Chicago metropolitan area. They have agreed to have as their common goal within Chicago Wilderness “to restore the region's natural communities to long term viability, enrich local residents' quality of life, and contribute to the preservation of global biodiversity.” Chicago Wilderness is pursuing its goals by attempting to create “green infrastructure” that will support biodiversity, and 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. The partnership has produced several reports, including its Biodiversity Recovery Plan and a green infrastructure map for the region. It has an active website for ongoing outreach activities (see Table 7 for references and full listing).

Table 9: Status of Valuation Work for Chicago Wilderness 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 (See attachment 1 for Introduction)	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	2006 (?)	http://greenvalues.cnt.org/calculator

The web page for the Chicago Wilderness (http://www.chicagowilderness.org/) contains a more complete chronology and links to many of these relevant documents, including the Biodiversity Recovery Plan.

Technical expertise and practical experience in valuing the protection of ecological systems and services is limited among members of Chicago Wilderness. There is also limited capacity in Region 5 to undertake economic analysis of the value ecosystem services. There is no specific legal authority that mandates that certain analyses related to valuing ecosystems or services be undertaken as part of the work of Chicago Wilderness. Though not required, quantifying values associated with the conservation of greenspace and biodiversity could be helpful for Chicago Wilderness in meeting its own stated objectives and in communicating its analysis with other groups and the general public. Chicago Wilderness is interested in the valuation of ecosystems and services, but has only begun to explore the opportunities for carrying out and incorporating such valuation in its activities. Among the possible uses of additional valuation tools identified by Chicago Wilderness members, including EPA Region 5, are:

- To inform decisions on where to establish green infrastructure and establish priorities for acquisition of land, for example by forest preserve districts and 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 where development decisions that have positive impacts on the environment might be in the financial interest of the developer;
- To effectively communicate with residents of the Chicago region 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 benefit from the ability to analyze the value of ecosystems and services, but is constrained by lack of expertise and resources in doing so.

<u>7.2.1 An Example of How Valuation Could Support Regional Decision-Making: Open-Space Preservation in the Chicago Metropolitan Area</u>

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

Voters in four counties in northeastern Illinois 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 in each county for open space acquisition or improvement. The voters in McHenry County passed a $50 million bond for watershed protection. While these multi-million dollar bond proposals 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, 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 these funds were invested wisely.

For purposes of this exercise, three types of values from protecting natural systems potentially relevant to the open-space and watershed protection 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 was explicitly directed towards watershed protection. We follow the process outline in Part 1 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.

<u>7.2.2 Process of Stakeholder Involvement, Scientific and Technical Input, and Public Participation</u>

Several of the themes from Part 1 of this report are reflected in the planning documents and activities of the Chicago Wilderness, including interdisciplinary collaboration, broad involvement. Chicago Wilderness consists of over 180 members, including local, state and regional governments. Partnership and participation are included as goals and operating principles. The Chicago Wilderness Biodiversity Recovery Plan (BRP) (see Table 9) discusses specific roles for private property owners, local, state and regional governments, intergovernmental agencies, and federal agencies. Actions of EPA that affect biodiversity and its role in Chicago Wilderness are also highlighted in this document. 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, 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 broadly construed. For priority-setting, several of the workshops included non-monetary valuation exercises with qualitative rankings of importance. The BRP also references other measures, for example the Nature Conservancy's global rarity index, and polls (e.g., "According to a 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 Russonello1996)." BRP, p. 117). Chicago Wilderness also carried out eight workshops to assess the status and conservation needs with regard to natural communities in the area: four species 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 incorporated 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 conservatism, and presence of important ecological functions (such as the role of wetlands in improving water quality in adjacent open waters)" (BRP 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. Streams were assigned recovery goals (protection, restoration, rehabilitation, and enhancement) or and lakes assigned priorities (exceptional, important, restorable, and other; based on Garrison 1994-95) in this effort. Streams were assessed using the index of biotic integrity (IBI), species or features of concern, the Macroinvertebrate Biotic Index (MBI), 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. Working with schools (including universities) is emphasized, but Chicago Wilderness also identifies individuals, agencies and organizations as targets for outreach and involvement.

Chicago Wilderness provides an excellent example of an organization that has made extensive efforts to engage the local community in figuring out what are the most important features of ecosystems and services in the region, according to people who live there. Two of the great strengths of Chicago Wilderness are the broad range of groups included and the commitment to open processes that allow community input and involvement. This process allows the participants themselves to define the objectives, goals and priorities of the organization. As a result of the open and democratic process and the extensive efforts to include multiple views and voices, its goals and objectives are largely reflective of what people in the region view as important to conserve in their region. The strengths, however, also highlight some of the difficulties involved. Different individuals and different member groups define value differently. Some groups care more about restoring pre-settlement ecosystem conditions, others are primarily motivated by issues of open space and recreation, while the primary objective of others is to maintain water quality or conserve the region's biodiversity. Because Chicago Wilderness is an organization based on consensus, they often cannot make choices involving tradeoffs between worthwhile objectives. It is easy to say that protecting biodiversity, protecting water quality, and providing open space and recreational opportunities are all good things. It is hard to say how to choose when doing more of conflicts with getting more of another goal. The inability to make tradeoffs among objectives limits their ability to make policy recommendations or have an influence on decision-making. In addition, the process of involvement and input is time consuming so that Chicago Wilderness is not well-placed to make rapid analyses or provide feedback on decisions that occur over a short time period.

7.2.3 Landscape Level Analysis of Ecosystems and Services

Species Conservation and Ecological Systems Conservation. Methods developed by NatureServe for identification and prioritization of conservation actions through spatial representation and analysis of biodiversity and conservation values have been applied across multiple scales and geographies. The application of the method results in spatial representation of the uniqueness and irreplaceability of biological and ecological diversity in a regional context. The methods support planning efforts to sustain biodiversity, ecological integrity and ecological services to identify best opportunities to meet stakeholder goals. The approach is based on principles of conservation science, strives for complete transparency, and can provide solutions that reflect different stakeholder values.

The key steps in applying the method are as follows:

a) Involve stakeholder to identify the biological, ecological and ecosystem service targets of interest
b) Define standards that represent a viable occurrence for each target, and for valuing the relative quality of each of these occurrences.
c) Define standards for measuring the conservation status of each target.
d) Create a “conservation value layer” for each target that represents the conservation status of the element and the viability/service value of each occurrence.
e). Create a “conservation value summary” that represents the composite values of all conservation targets.
f) Map current land uses, policies, threats, economic values, and compatibilities across the project landscape.
g) Analyze spatial solutions that address stakeholder goals and provide a clear delineation of priority actions.

Chicago Wilderness has generally followed the approach described above to identify biodiversity and conservation values. The conservation targets that the Chicago Wilderness has identified are described in detail in its Biodiversity Recovery Plan.

Water Quality and Quantity. Water quality and quantity figure prominently in many ecological processes and in the provision of many ecosystem services. Text Box 8 describes possible ecological impacts and impacts on the provision of ecosystem services that are possible 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 9: Possible Ecological Impacts and Provision of Services from the Protection or Restoration of Watersheds Based on the Work of Chicago Wilderness

Surface water

- 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 in reduced because of the retention capabilities of the intact watershed

Subsurface water

- 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

Biological systems that depend upon water quantity

- 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

For purposes of the following discussion, suppose that both stakeholders and experts decided that the most important ecological services to be used in comparing watersheds within the county were: 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 above in section 1.2.2.

Minimize flooding: The GIS database collected by Chicago Wilderness 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. Those watersheds with the greatest flooding could be identified. The analysis could then evaluate the potential for restoring floodplain forests and wetlands for mitigating flooding.

Maintain or increase groundwater recharge: The GIS database includes maps of aquifers and soils maps that described 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).

Maintain or increase wetland communities: Using topographic maps and GIS data on rivers, streams, floodplains, forests, wetlands and land cover, watersheds within McHenry County could be ranked 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.

There are a number of GIS data files available from McHenry County that can assist understanding how protecting 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 an ecosystem services will change with changes 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 including game fish production, on wetland and their consequent production functions such as waterfowl, fisheries, wildlife viewing, etc. (Kremen, 2005).

Recreation and amenities. The third set of values that we include in this example are recreational and amenity values. Unlike biodiversity conservation and water quality and quantity issues, recreation and amenities do not have a large technical or natural science component to them. The most important steps for recreation and amenities come at the first stage, getting community input on what is important, and the next stage on attempts to measure values.

Summary. 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. However, for this information to be relevant to decisions that affect ecosystem, cause-and-effect relationships that can predict how policies choices would affect ecosystems and the provision of services are needed. Chicago Wilderness often has fallen short on this score. For example, to invest the $50 million approved by voters for watershed protection in McHenry County in a way that will maximize the value of ecosystems and services, a decision-maker needs to know more than just how protecting a given part of a watershed or landscape contributes to ecosystem processes and services that people have 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 and one that introduces another danger. 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 targets can be considered as being met for conserving biodiversity involves expert judgment, which may be influenced by the values of the expert. Different judgments used in models may give rise to different sets of recommendations. Combining community values with expert knowledge requires honest communication as well as commitment on the part of experts to faithfully carry out the stated desires of the community.

When there are tradeoffs among different services, habitat protection versus improvements in water quality for example, then information about the value of flood control versus the value of improved water quality versus the value of biodiversity protection is necessary to know whether certain tradeoffs are worthwhile or not. This requires information beyond just understanding the ecological impacts of management and policy alternatives.

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

The Role of Valuation. The primary goal of Chicago Wilderness "is to protect the natural communities of the Chicago region and to restore them to long-term viability." Given this goal, it may be argued that monetary valuation is of secondary importance and of primary importance is to understand how various potential strategies contribute to the protection and restoration of natural communities, or to the provision of ecosystem services. In some sense, the important valuation exercises for Chicago Wilderness were carried out at the first stage where Chicago Wilderness engaged the community and gathered feedback on what it felt was important. Chicago Wilderness has, in fact, devoted most of its attention to stakeholder involvement and assessing biophysical measures of the status of natural communities and much less attention to quantitative measures of value, monetary or otherwise.

With a clearly stated single biological objective, such as "to protect natural communities," economic analysis may be restricted to estimating the cost of various potential strategies to achieve that objective. Combining 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 all the information necessary for cost-effectiveness analysis. Cost-effectiveness analysis addresses the issue of how best to pursue an objective given a budget constraint. In cost-effectiveness analysis, there is no need to estimate the value of protecting natural communities or of ecosystem services. Of course, with a goal such as "to protect natural communities," there will be tradeoffs between protecting one type of natural community versus another. Going beyond cost-effectiveness analysis may therefore be necessary.

When there are multiple natural communities of interest, or multiple ecosystem services of interest, it becomes important to address questions of value. Is it more valuable to allocate more of budget 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 of different natural communities or services.

Monetary valuation of the protection of natural communities may be important for Chicago Wilderness, and more broadly to society 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 on the basis of the overall value generated. Second, some biological concepts such as "biodiversity" are multi-faceted. How one makes tradeoffs between different facets of biodiversity conservation, or among protection of different natural community types, is the ultimately the same question as how one makes tradeoffs among multiple objectives. Again, establishing prices on different components of biodiversity or on different natural communities allows for analysis of tradeoffs between 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 general public.

Value may also be addressed using non-monetary valuation. If what is needed is to assess tradeoffs between protection of different natural communities or among different services, this may be done most directly by making such comparisons without the additional complication of trying to convert these values into monetary terms. In other words, it may be far easier for people to answer questions about whether they think it more important to provide additional protection of forests versus wetlands, as compared to asking about the monetary valuation of forest protection and the monetary valuation of wetland protection.

Valuation of Species Conservation and Ecological Systems Conservation. Protecting natural communities may be done for reasons related to the provision of ecosystem services, or it may be done just because people value intact natural communities (e.g., existence value or intrinsic value). 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 (CVM) 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 CVM or conjoint analysis. Alternatively, Chicago Wilderness could attempt to use a 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 CVM or conjoint analysis is that it would allow Chicago Wilderness to calculate a total economic value for alternative strategies. Without using CVM 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.) CVM 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 tradeoffs 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 tradeoff 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.)

<u>Valuation of Water Quality and Quantity</u>. Changes in water quantity can be valued either because there is too much (flood control) or too little water (water availability).

Flood control: approach is to measure avoided damages with reduction in probabilities of flooding. Studies of the value of preserving wetlands for flood control have been undertaken in Illinois: Salt Creek Greenway in Illinois (Illinois Department of Conservation, 1993; USACE, 1978) and in Cook County where the estimated value of regional floodwater storage was $52,340 per acre (Forest Preserve District of Cook County Illinois, 1988).

Water availability: another important ecosystem service in many metropolitan areas is to provide clean drinking water. One of the more famous examples of the value of ecosystem services is the case of the provision of clean drinking water from watersheds in the Catskills for New York City (NRC 2000, 2004, Chichilnisky and Heal 1998). There is also value of surface recharge of aquifers (NRC 1997).

<u>Valuation of Recreation and Amenities</u>. 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, there is a smaller literature that uses evidence from referenda voting to infer values for open space and other environmental amenities.

There is a large empirical literature that estimates the value of environmental amenities on the value of residential property value using the hedonic property price model. 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 in any given time period, and large data bases on attributes of the property, 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. Given the large number of visitors to Lake Michigan beaches, forest preserves, and parks in the Chicago metropolitan area, there is great potential for Chicago Wilderness to apply travel cost to estimate the value of recreational activities. To date, these methods have not been applied by Chicago Wilderness. {Provide references on appropriate travel cost studies in an urban setting}

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) estimated the willingness-to-pay for "wilderness recovery and extension activities" in Chicago region. {Provide short summary of results}

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 Lissero 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 above, 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..

7.3. Other Case Studies: Portland, OR and the Southeast Region

7.3.1 Portland, Oregon Assessment of the Value of Improved Watershed Management

The city of Portland, Oregon undertook an analysis of ecosystem impacts and the value ecosystem services that would result from improved watershed management. 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 was completed in June 2004 (David Evans & Associates and ECONorthwest, 2004). The C-VPESS received a briefing on the project on September 13, 2005. Though the project was not an example of a regional partnership with EPA, the Committee was impressed with the analysis and results of the project and thought that it provides a good example of the kind of regional scale analysis of the value of ecosystems and services that could be undertaken in the future.

Portland city officials realized that they only understood a portion of the benefits of improved watershed management. The primary motivation for the analysis was to quantify a range of normally un-quantified ecosystem benefits. 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. In addition to the value of direct flood-abatement impacts, the study monetized the benefits of air quality, amenity, and recreational improvements. From the beginning, the effort many an attempt to solicit input from the public and important stakeholder groups about important ecological impacts.

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 and residential and commercial property, the value of flood risk reduction, the values of amenity and recreation, and the value of impact of changes on human health.

The project used a “system dynamics” approach that most closely resembles what the C-VPESS refers to as production function analysis. The approach linked management changes, such as flood project alternatives, to a range of ecological changes. These ecological changes were analyzed for the effect on various ecosystem services. Finally, the economic analysis attempted to value the changes in various ecosystem services. The ecological and economic analyses were largely conducted by separate teams. However, the project was designed to provide a close linkage between ecological results and economic valuation.

This example provides a good example of potential benefits 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 about which the public had the greatest interest. 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 potential problems in undertaking regional scale analysis. Inevitably in this type of analysis there are data gaps and gaps in understanding of ecological systems and how they will be affected by changes in management actions. {Provide example} {Possibly draw from material provided by City of Portland regarding how the results of the study have been used – or not used }

7.3.2 Southeast Ecological Framework Project (EPA Region 4)

The Southeast Ecological Framework (SEF) project represents a unique regional 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 US. Despite this, 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 ultimate SEF project goal is for the project results to enhance regional planning across political jurisdictions and to help focus federal resources to support state and local protection of ecologically important lands. The work was completed by the Planning and Analysis Branch of EPA Region 4 and the University of Florida in December of 2001.

The SEF applied a regional landscape analysis approach that represents conservation priorities and threats across the region in order to sustain critical ecological and biological values in the region, This approach builds from existing conservation areas and adds additional conservation areas and connecting corridors in order to secure and sustain the protection of critical native biodiversity and landscape functions. The conservation significance is determined from variables that characterize habitat type, protected areas and presence of rare species. The methodology is designed to meet standards of transparency and repeatability, and can be updated with new data. The GIS decision support approach provides a means to integrate complex data at a landscape scale to aid decision-making.

This framework has been developed for the eight southeastern states in EPA Region 4 (FL, GA, SC, NC, AL, MS, TN, and KY). 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 are appropriate managed for specific societal benefits. 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 Portland) because it focuses on a broad regional analysis, 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. 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.

7.4. Summary and Lessons Learned

A number of methods exist that could be applied by Chicago Wilderness to assess the relative value of alternative strategies to protect ecosystems and services. Application of these methods would generate information that could be of great use to decision-makers in evaluating alternative strategies to protect natural communities that would be most beneficial for the public at large. To date, however, Chicago Wilderness has focused almost exclusively on biophysical measures that assess the extent and current condition of natural communities. There have been some attempts to collect information about the value of protecting natural communities and ecosystem services (e.g., Kosobud 1998), but this effort has not been comprehensive or systematic. This is mostly due to the mix of expertise of members organizations that make up Chicago Wilderness. Interest exists to include economic and other social science approaches to study the value of protecting natural communities, but there has not been the right mix of available expertise and circumstances to make this a reality.

Regional-scale analysis has great potential to inform decision-makers and the general public about the value of protecting ecosystems and 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 local/state partners. Such partnerships offer great potential for improving science and management for protecting ecosystems and enhancing the provision of ecosystem services. At present, however, this potential is largely unrealized. To take advantage of this potential, EPA would need to increase the capacity of regional offices in both economic and ecological analysis. EPA would need to devote resources to make the study of the value of protecting ecosystems and services a high priority. Making this a high priority is hampered by the lack of specific legal mandates or authority to study these values. Given tight agency budgets, the valuation of ecosystems and services at present appears to be more of an unaffordable luxury rather than a necessity.

A review of several regional analyses of ecosystems and services yields the following general lessons:

- Important ecological processes take place at a regional scale, making it perhaps the most appropriate scale at which to analyze the value of 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 ecosystems and services.
- Many important decisions affecting ecosystems and the provision of ecosystem services are taken at a regional scale by municipal, county, regional and state governments 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, or to

incorporate the value of ecosystems or services into their decision-making processes.

- Many regional-scale analyses to date have greater ability to characterize current extent and condition of natural habitat types but much more limited ability to analyze likely consequences of changes in policy or management, and very limited ability to measure impacts on the value of protecting ecosystems or services.
- In addition, there is a great need to increase the ability of natural scientists to collaborate with economists and other social scientists in doing integrated research at a regional scale.

8. ANALYSIS AND REPRESENTATION OF UNCERTAINTIES IN ECOLOGICAL VALUATION AND COMMUNICATION OF ECOLOGICAL VALUATION INFORMATION

8.1. Uncertainty

8.1.1 Introduction

Ecosystem valuation efforts are inevitably subject to a variety of uncertainties, regardless of the method used. Assessments of uncertainty allow more informed evaluations of proposed policies and comparisons among alternative policy instruments. And unless uncertainty is taken into account and thoughtfully conveyed to decision makers, the ultimate usefulness of assessments may be compromised. Because any given policy may result in a range of different outcomes, decision makers must be provided with sufficient information about what is known about the distribution of possible outcomes so that they can take uncertainty into account in their policy choices. Whether decision makers wish to adhere to maximizing expected utility, avoiding major risks through a "maxi-min strategy," or some other decision principle such as the Precautionary Principle, they have to consider the uncertainty that policy choices always entail. The way in which uncertainties are represented should be consistent with the decision principle being utilized. In addition, if the sources of key uncertainties are not identified, an opportunity is lost to develop potentially important insights regarding the design of research strategies to reduce uncertainty in future analyses.

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 be most important with alternative valuation methods for specific applications? Second, what methods are available to characterize and communicate uncertainty in the results of ecological valuations? Here we are interested not only in the formats that can be employed – such as confidence intervals, probability distributions, and pictorial representations – but also the types of interactions between analysts and policymakers that can be employed to convey uncertainty most effectively. 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 8.1.2 describes the major sources of uncertainty in ecosystem and ecosystem services valuation. Section 8.1.3 examines the potential for uncertainty assessment of ecological values, describing both the merits of formal quantitative uncertainty assessments and the additional efforts that would be required for government agencies to carry out such assessments. Section 8.1.4 assesses the potential value of uncertainty assessments to the research agenda of the U.S. Environmental Protection Agency and other researchers.

8.1.2 Sources of Uncertainty in Ecological Valuations

Valuation of the benefits of proposed public policies entails three analytic tasks, each potentially subject to uncertainty: predicting biophysical outcomes, predicting socio-economic reactions to these outcomes, and valuing the consequences of all of these changes. It might be tempting to limit attention to the uncertainty of valuation per se, but the uncertainties in each of these stages of the analysis are of potential importance, and there is no reason – on the basis of theory alone – to judge one more important than the other a priori. Rather, the relative magnitude of the uncertainty involved in these essential steps in the valuation process is fundamentally an empirical question.

At each of these stages, uncertainty can arise from several sources. First, there are uncertainties involved in the statistical estimation of the parameters of the models used in the analysis. Second, some of the physical processes might be inherently random or stochastic. And finally, there can be uncertainty about which of several alternative models of the process best captures the essential features of the process.[38]

Uncertainty of Biophysical Changes and their Impacts. At the bio-physical level any characterization of current (or past) ecological conditions will have numerous interrelated uncertainties, and these uncertainties will be magnified and added to by any effort to project future conditions, with or without some postulated management action. Ecosystems are complex, dynamic over space and time, subject to the effects of stochastic events (such as weather disturbances, drought, insect outbreaks, fires, etc). And 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 assessment of ecosystems/services benefits.

While the currently available methods for dealing with uncertainty may be sufficient for some simple evaluation problems, the valuation of changes in ecosystems and ecosystem services raises issues not well addressed by any existing methods. For example, at the biophysical level it is extremely difficult or completely unclear how to calculate the uncertainty in the projection of even a single outcome or endpoint from a complex ecological system composed of multiple interacting variables that may be separately non-linear and collectively subject to the influence of external stochastic events. Modeling methods, such as sensitivity analyses, may be used to estimate the range of possible outcomes (or at least best-case, worst case extremes) for a single endpoint, but even this approach becomes unwieldy when the outcomes relevant to the value assessment are themselves composed of multiple interrelated variables.

Uncertainty of Socio-economic Reactions and their Impacts. The second stage of valuation – predicting the socio-economic reactions to biophysical impacts and the consequences of these reactions – is subject to the same three sources of uncertainty. Regarding theory limitations, every social, economic or political forecast is based on implicit or explicit theory of how the world works, represented either by the "mental models" in the minds of the forecasters or in the formal and explicit methods used in econometric modeling, systems dynamics modeling, etc. Theories and their expressions as models are unavoidably incomplete, and of course may simply be incorrect in their assumptions and specifications.

Uncertainty Arising from the Application of Valuation Methods. Valuation methods per se are also subject to data and theory limitations. They unavoidably rely on assumptions that introduce uncertainty. For example, as noted in Part 2 Section 4 above, different valuation methods are based on different premises about the nature and sources of value and/or assumptions about the behaviors of people. Thus, in principle, there is no one single correct measure of value.

In addition, all assessments of expected consequences are about anticipated, not experienced satisfaction those consequences might bring. To take a simple example, the choice of a vanilla ice cream cone over chocolate is based on the anticipation that consuming the vanilla will bring greater pleasure/satisfaction than the chocolate (and perhaps even further that a pleasant gustatory experience will contribute toward a more ultimate goal of improved well-being, happiness in life or self actualization). In fact research has shown that even in relatively simple and familiar situations people err considerably in their anticipation of the satisfaction they will attain from a given outcome. When the values and choices at issue are about imperfectly projected changes in ecosystems/services, where previous experience is limited and where the time horizons are much greater, there is even less certainty in the accuracy of anticipated satisfaction. These anticipation errors become even more problematic in the typical circumstances of an environmental management decision, where the goals and the intended beneficiaries are some loosely defined society, some members of which may not yet exist, and only a small number of whom are involved in any direct way in the consideration and decision making process. In such contexts any notion of a final and accurate assessment of the true value of some change in ecosystems/services must be illusory. Still, people and agencies must continue to evaluate alternatives and make decisions based on their best estimate of what consequences will follow and how they will contribute to proximate and ultimate goals.

Uncertainty in Benefits Transfer. In addition, even if existing estimates are developed using an appropriate model, analysts are often required to apply them 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 sensitivity analysis to determine 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 sensitivity analysis with alternative models.

8.1.3 Approaches to assessing uncertainty.

The simplest and probably most common approach to representing uncertainties is some form of sensitivity analysis in which, typically, one parameter or model assumption is varied at a time and point estimates are calculated. The results are considered "high" or upper bound and "low" or lower bound estimates of the "true" value. No effort is made to estimates the probabilities attached to the calculated values or the shape of the distribution of values within the range. At best sensitivity analyses give only an incomplete and potentially misleading picture of the true uncertainty of an analysis. So other approaches should be considered.

Under the various forms of probabilistic uncertainty analysis that are increasingly in favor in policy analysis, the tasks of assessing the uncertainty of the elements that go into a valuation involve estimating a distribution of values arising from the combined uncertainties of the elements of the analysis (rather than a single point estimate), and a diagnosis of the elements that are contributing most heavily to spreading this distribution. Given the multiple levels of elements that can add to uncertainty, the most complete approaches will be unavoidably complex themselves.

Monte Carlo Analysis as an Approach to the Formal Uncertainty Assessment of Ecological Values. Due to the number of sources of uncertainty in many ecological valuations and the complexity of their interactions, assessments of the extent of uncertainty that are conducted without formal quantitative analyses are unlikely to represent accurately the true extent of uncertainty. No sensitivity analysis or expert judgment is likely to be able to account for the implications of all the sources of uncertainty in inputs. Therefore over the years, the use of formal quantitative uncertainty assessment, and in particular Monte Carlo analysis, has been shown to provide a more reliable and rich characterization of the implications of uncertainty, and therefore has become common in a variety of fields, including engineering, finance, and a number of scientific disciplines.

Monte Carlo analysis has also been found to be useful in certain policy contexts. In particular, the U.S. Environmental Protection Agency (EPA) recognized as early as 1997 that it can be an important element of risk assessments (U.S. Environmental

Protection Agency 1997). But efforts to formally quantify uncertainties rarely have been made in the context of ecological valuations. More often, uncertainty has been addressed qualitatively or through sensitivity analysis.

As it is unlikely that a Monte Carlo analysis will comprehensively address all sources of uncertainty in the estimation of ecological values, the results of such an analysis will likely understate the range of possible outcomes that could result from a related public policy. Yet the ranges produced by such an analysis would still provide more reliable information about the implications of known uncertainties. In turn, these ranges can better inform judgments by policymakers as to the overall implications of uncertainty for their decisions.

Monte Carlo analysis also 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. But this is rarely the case. Others may assume that the distribution of possible values is symmetric. This, too, often may not be the case.

In developing probability distributions for uncertain inputs, uncertainty from statistical variation can 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. Characterizing other sources of uncertainty in inputs can require more effort.

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. Also, modern programming techniques allow the writing of Monte Carlo computer programs with minimal additional effort, relative to that needed to produce point estimates.

Expert Elicitation for Gauging and Conveying Uncertainty. A host of "expert elicitation" methods can provide indications of uncertainty as well as estimates and forecasts by the experts involved. See, for example, Morgan and Henrion (1990) or Cleaves (1994). In its very simplest form, a single expert's assessment of the uncertainty of his or her estimate, forecast, or valuation can be provided, whether it is based on implicit judgment or a more explicit approach like the Monte Carlo technique. Policymakers can elicit more information from the expert, such as the assumptions underlying his or her analysis or the bases for uncertainty, in order to get a deeper understanding of the reliability of the expert's input and the nature of the uncertainty. However, the bulk of expert elicitation methods involve multiple experts, who may or may not be brought into interaction with one another. Because eliciting the input from multiple experts permits compiling and comparing their judgments, expert elicitation can be used to assess the disagreement among experts. If the experts are of equal credibility, such that none of the judgments can be discarded in favor of others, 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. However, this is an asymmetrical relationship, in that narrow disagreement does not necessarily reflect justified certainty—the experts may all be wrong in the same direction, which is not uncommon in light of the fact that experts are often paying attention to the same information and operate within the same paradigm for any given issue (Ascher & Overholt, 1984: 86-87). When experts are brought into some form of interaction 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 be unaware of atmospheric trends that information from atmospheric chemists could redress. However, such interactions run the risk of "groupthink" – 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. For example, simple compilations of estimates (e.g., contemporaneous estimates of species populations) from different experts will provide a table with the range of estimates, but will not convey the degree of uncertainty that each expert would attribute to his or her estimate. And the compilation in itself cannot generate this information. In contrast, a compilation of estimates that come with confidence intervals could provide this information.

8.1.4 Contributions of Uncertainty Assessment in Guiding Research Initiatives

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 is in 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. 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).

Using uncertainty analysis to guide research priorities requires, of course, sensitivity to the feasibility of filling the gaps. Some data needs are simply too expensive to fulfill, and some methods have intrinsic limitations that no amount of refinement will fully overcome.

Uncertainty assessment can also provide insight into whether near-term progress in reducing uncertainty is likely, based on its sources and the feasibility of addressing these limitations promptly. However, it is important to avoid the pitfall of delaying actions to address problems simply because some uncertainty remains – it always will.

8.2. Communication and valuation

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

a) communication within the valuation process itself;

b) communication of resulting values to inform decision-making; and

c) communication of the results of the valuation and decision-making processes to stakeholders and others.

Understanding how information about values will and should be used by decision-makers is crucial for understanding how the valuation analysis should be conducted and its results conveyed, including how uncertainty should be conveyed.

Within the valuation process itself, how decision objectives, decision attributes, and specific measures of values are communicated can determine the outcome of the process. Good communication practices include the use of an analytic-deliberative process, in which analysis and deliberation occur iteratively and interactively (NRC, 1996). The valuation process (see Figure 2) includes iterative problem definition and description by stakeholders, to clarify what and whose values will be represented in the valuation process. Communication of resulting values to inform decision making is simplified to the extent that decision makers or their representatives are involved in the process.

Recommendation: As resources permit, analytic-deliberative process, involving iterative problem definition and description by stakeholders, should be engaged, as it will increase the transparency, credibility and usefulness of valuation exercises.

Values, decision objectives, and decision attributes can each be defined either qualitatively or quantitatively, and represented in a wide variety of ways. Several critical design choices are likely to influence the effects of communicating values to parties not involved in the valuation process, either to inform decision-making or to share results with other interested parties. In communicating VPESS, key choices include a) how to describe the ecological functions, systems and services to which the valuation pertains; b) how to express values most meaningfully - whether to quantify or use non-numerical representation of values, use of visual and narrative strategies for each, and interactivity and related choices regarding the medium and mode of information presentation; and c) how to communicate uncertainty. Those choices will in turn either facilitate or hinder specific kinds of deliberations and analyses. Finally, evaluating communications is critical to understanding their effects and improving them.

Recommendation: insert text here?

Decision making in public policy often requires translation and/or aggregation, from one specific context to another, or from one level of decision making (e.g., local) to another (e.g. regional), and inevitably involves value trade-offs. Specific choices of how to represent or communicate values will influence the ease and transparency with values can be translated or aggregated, and with which trade-offs can be made. Values that are quantified (e.g., monetized) may be easier to aggregate or compare than those represented qualitatively. Use of multiple metrics is likely to complicate aggregation and comparison.

8.2.1 Describing ecological functions, systems and services

The focus of the value discussion 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 literatures (e.g., Failing and Gregory, 2003) is not on dollars per se, but on ends and decision or management objectives - that is, qualitative expressions, and a wider variety of expressions of value - not just monetary expressions of value. In other words, the more prevalent mode of communicating values in these studies is through narrative and non-monetized description of attainment of management objectives.

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 and state of the ecological systems and services to which the value information applies. The latter can be and is often conveyed using mapped ecological information, other visualizations including photographs and graphs, ecological indicators, and narratives. Integrated models with a geospatial interface are another approach to depicting the state of ecological systems and services. To the extent that these can be made interactive, they will allow sensitivity analysis and may be more effective as communication tools. The US EPA Science Advisory Board has proposed this kind of framework for reporting on the condition of ecological resources (US 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.

Recommendation: Use GIS and interactive geospatial information systems integrated with other ecological models where feasible, to represent the state of ecological systems and services. Consider best cartographic principles and practices (Brenner, ????; MacEachren, 1995).

It is critical to communicate ecological processes as well as static information or states. The EPA Science Advisory Board review of EPA's Draft Report on the Environment (US EPA SAB 2005) and several other reports (e.g., Schiller et al., 2001; Carpenter et al., 1999; Janssen and Carpenter, 1999) make the point that people need to understand the underlying causal processes, to understand how ecological changes affect things they value (e.g., ecological services).

Related issues of scale and aggregation are also important. Both the NRC report (2001) and the SAB review of the EPA's Draft Report on the Environment (US EPA SAB 2005) emphasize the importance of using regional and local indicators - of not aggregating information data to the point where it obscures 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. The SAB states that "some environmental changes are best understood by considering regional impacts" (EPA SAB, 2003). Further, while some authors recommend simple summary indicators (e.g., Schiller et al., 2001; Failing and Gregory, 2003); others emphasize disaggregating indicators (US EPA SAB 2003).

Reporting on the nature, state of and changes in ecological systems and services is a key component of value elicitation and communication, but needs to be married with equal consideration of how to convey the value of protecting them.

8.2.2 Communicating values meaningfully

Communicating values is complicated by the likely disconnect between popular and other uses of the term "values" and what economists in particular mean by the term. Variously described by sociologists and social psychologists as beliefs, goals, or even cultural imperatives, stable sets of values (e.g., benevolence, self-direction, security, hedonism and others) have been identified across cultures, although values vary with history and culture (Hitlin and Piliavin, 2004). Conservation versus openness to change, and self-enhancement versus self-transcendence are two dimensions identified as underlying values (Schwartz, 1994). Values are sometimes conflated with attitudes (which are positive or negative evaluations of an object), traits (which are enduring attributes of personality), norms (which are situation specific) or needs (which are biophysical influences on behavior) (Hitlin and Piliavin, 2004). All of these concepts are embraced by Table 1, but differ from the sense in which economists use the term value, although attitude appears to come closest.

Value elicitation includes contingent valuation and attitude judgments (generally on rating scales, but also using ranking tasks). It also includes qualitative expressions and narrative expressions of value, defined by the identification of associated ends, and the means to achieve those ends.

As discussed in Appendix A, context and framing can have some influence on how people rank, rate and estimate values (Hitlin and Piliavin, 2004; Horowitz and McConnell, 2002), as well as the interpretation of all kinds of value-related information (add Slovic and other references). Decision makers and others come with their own prior beliefs and attitudes, of which communicators should be aware (Morgan et al, 2002). To support decisions effectively, it is critical that communications be designed to address the recipient's goals and prior beliefs, taking into account likely context and presentation effects. 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).

<u>What we know about perception and use of value measures.</u> As summarized elsewhere in the report, value measures are required or useful in a variety of regulatory and non-regulatory policy contexts, ranging from local government assessment and prioritization of environmental actions, to educational outreach, to federal assessment of agency programs. 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. There is little direct evidence about how such value measures are perceived, although there is considerable indirect evidence regarding their use. For example, measures that are not quantified and monetized in regulatory impact analyses appear unlikely to be fully considered or used in cost-benefit judgments. In contrast, participative decision making exercises can and do use ecological indicators as a basis for prioritizing and trading off actions to protection ecological systems and services, without monetization as has been done by NatureServe. (ref from Denny, recent state exercises).

Table 10: Table of Examples of Measures from Different Ecological Valuations Discussed in this Report
[To be completed with values from current draft]

VALUE	MEASURE	Characteristics	Context/Use	Reference	Communication
Avoided decrease in crop harvest	Avoided 7.5% decrease in crop harvest from UV-b radiation by 2075	Quantified	Context/Use: Regulatory Impact Analysis: Protection of Stratospheric Ozone Reference:	Table 7-9, Quantified and Unquantified Ecological and Welfare Effects of Title VI Provisions, page 96 of http://www.epa.gov/air/sect812/1990-2010/fullrept.pdf	Structured narrative
Unquantified ecological benefits	[List of benefits:] . recreational fishing . forests . marine ecosystem and fish harvests . avoided sea level rise, including avoided beach erosion, loss of coastal wetlands, salinity of estuaries and aquifers . other crops . other plant species . fish harvests	Unquantified measure, descriptive	Regulatory Impact Analysis: Protection of Stratospheric Ozone	Table 7-9, Quantified and Unquantified Ecological and Welfare Effects of Title VI Provisions, page 96 of http://www.epa.gov/air/sect812/1990-2010/fullrept.pdf	Unstructured list/narrative
Freshwater acidification from wulfur and nitrogen oxides regionally, in the Adirondacks	(in millions of 1990$) range of $12 to $88 for 2010; central estimate for 2010 is $50; $260 cumulative estimate 1990-2010.	Monetized ecological benefit. Captures only recreational fishing impact regionally (incomplete geographic coverage), based on an economic model of recreational fishing behavior.	Regulatory Impact Analysis	Tables 7-8 and 7-10, pp 91-92 and 97 in http://www.epa.gov/air/sect812/1990-2010/fullrept.pdf	Dollars, used in calculations of benefits

8.2.3 Guidelines for design choices: audience assessment, user needs, and visual and interactive communication strategies

The potential interested parties for values (VPESS) include community members, policy makers, and scientists, especially 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 (participative) 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.

Recommendation: EPA should develop an empirical analysis of the users of valuation and adapt valuation communications to their needs.

End-user engagement is itself an example of a participative process, in that it involves stakeholders in the valuation enterprise. End-user engagement requires due consideration of such issues as sampling and representation. Stakeholders are likely to vary considerably in their interests, abilities, and resources such as time or access to experts who can answer technical questions. While verbal quantifiers (e.g., “many” or “very likely”) are often proposed as a way of making technical information more accessible, the wide variability with which these 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. MacEachren (1995) emphasizes the function of visualization in facilitating viewers’ new and surprising insights.

Interactive communications are likely to be more effective in many circumstances than static displays. They allow users to manipulate the data or representations of the data – e.g., 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 presentation 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 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 users could be overwhelmed by the complexity of the visualization (Cliburn, Feddema, Miller, & Slocum, 2002). However, interactive visualization poses challenges as well. Interactivity is necessitated and challenged at the same time by 3-D visualization, which has become increasingly popular in visualization practice (Encarnacao et al. 1994),.

Recommendation: Support interactive exploration tools in valuation representations and communications, where feasible.

Finally, fundamental guidelines for risk and technical communication are generally applicable to values communication. Two examples of such guidelines are the communication principles from EPA's *Risk Characterization Handbook* (2000) and Guidelines for effective websites from Spyridakis (2000). The Risk Characterization handbook principles include transparency, clarity, consistency and reasonableness.

Recommendation: Follow demonstrably effective basic practices for risk and technical communication.

8.2.4 Communicating Uncertainty in Ecological Valuations

In order to assess how much confidence to attribute to the projections involved in the valuation, decision makers must also 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 also important because decision makers often have to explain and justify their decisions by clarifying the assumptions driving the analysis.

In order to convey to policy 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), but in some cases this may require possibly arbitrary judgments on the part of the analyst (Moss & Schneider 2000). Furthermore, providing policy makers with such ranges of results can be highly misleading, because those without training in probability and statistics may be likely to assume - in effect - that the probability distribution of values between the end-points is uniform, which is rarely, if ever, the case. 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 benefit assessments that are part of Regulatory Impact Analyses (RIAs) - have been limited. But guidance set forth in the U.S. Office of Management and Budget'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 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 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 makes the results presented 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 results of an ecological valuation are more certain than they 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 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.

8.2.5 Evaluation

In general, it is difficult to predict the effects of communications. Good communications practice requires formative evaluation of communications as part of the design process. Summative evaluation after the fact will enable assessments of effectiveness, and continued improvement (e.g. Scriven, 1967; Rossi et al., 2003) and other refs)

Recommendation: Evaluate communications, to assess the effects of the communication and how to improve them.

9. RECOMMENDATIONS

Text to be added after teleconference discussions

9.1. Research

9.2. Guidance documents

9.3. Institutional Recommendations

PART 3: Methods for Implementing Approach

1. INTRODUCTION

This part of the report provides more detailed information about the methods that the committee examined for possible use in implementing the integrated and expanded valuation process proposed in Part 1. These methods differ in a number of respects, including the underlying premises and 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 role that they might play in ecological valuation in different decision contexts, and the extent to which they have previously been used (by EPA or others) for this purpose. While there is no perfect way to categorize or group these methods, the Committee has organized the discussion of methods around groupings based primarily on the basic premises that underlie the different methods. In each case, the goal is not to provide an exhaustive treatise on a method; rather, it 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.

2. BIO-PHYSICAL RANKING METHODS

2.1. 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.

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.

In some parlances this process of weighting and mapping the resources that represent what people want to preserve is referred to as "green printing." For example, groups such as Trust for Public Lands use this 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.

<u>Brief description of the method</u>. 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 withing 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 other ways as well. For example, (1) it could be used as a prediction of ecological impacts that would then be used as an input in an economic valuation study; (2) 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 (3) it could be used as a means of quantifying bio-physical impacts when they cannot be quantified (as required by the OMB Circular A-4).

Status as a method. 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, occurrence mapping standards, and others There is widespread use of various components of these methods across US 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.

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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.

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Treatment of Uncertainty: 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.

Research needs: 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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2.2. Rankings Based on Energy and Material Flows

<u>Introduction</u>

Energy and material flow analysis is the quantification of the flows of energy and materials through complex ecological and/or economic systems. A recent National Academy report covers the basic elements and need for such analyses (Committee on Materials Flows Accounting of Natural Resources, Products and Residuals, 2004). 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. Using energy and materials as common currencies offers the possibility of treating ecological and economic systems conceptually with the same methodology. In theory, economic values could be assigned, with the advantage that these valuations are based on energy and/or material flows that are characteristics of both economic and ecological systems.

This section provides general background on energy and material flow accounting followed by three specific applications that use this method for ranking alternatives: embodied energy, emergy, and ecological footprint.

Both the embodied energy and ecological footprint methods use input-output analysis or flow accounting methods. Embodied energy analysis estimates the direct and indirect energy (or more correctly available energy or "exergy") cost of goods and services. Ecological footprint analysis estimates the biologically productive land area required (directly and indirectly) to meet various consumption patterns. Emergy analysis shares many of the goals of embodied energy analysis, but different methods are used in the calculations (Collins and Odum 2000).

Ecologists and physical scientists have proposed 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). It is based on thermodynamic principles where solar energy is recognized to be the only "primary" input to the global ecosystem. Classical economists recognized that if they could identify a "primary" input to the production process, they could then explain exchange values based on production relationships. The problem was that neither labor nor any other single commodity was really "primary," since they all require each other for their production.

At the global scale, the traditional "primary" factors are really "intermediate" factors of production (Costanza 1980). Available energy or exergy is the only "basic" commodity and is ultimately the only "scarce" factor of production, thereby satisfying the criteria for a production-based theory that can explain exchange values. An energy theory of value thus posits that, at least at the global scale, free or available energy from the sun (plus past solar energy stored as fossil fuels and residual heat from the earth's core) is the only "primary" input to economic production. Labor, manufactured capital and natural capital are "intermediate inputs." Thus, one could base a theory of value on the use in production of available energy that avoids the problems the classical economists encountered when trying to explain exchange values in economic systems using only labor.

In thermodynamics terminology there are three categories of systems: open, closed, and isolated. Open systems allow matter and energy to cross the boundaries. Closed systems allow only energy to cross the boundaries (i.e. closed to matter, but not energy). Isolated systems allow nothing to cross the boundaries. The second law (that entropy always increases) applies only to isolated systems. The earth is (for the most part) a closed system (not an isolated system) with lots of energy crossing the boundaries, which is why it will not run down (or at least it can compensate for the running down with new energy inputs).

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 pointed out 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), but 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 a closed physical system must satisfy the conservation of mass condition, and hence that economic growth necessarily increases both the extraction of environmental resources and the mass of waste deposited in the environment. Ayres (1978) described some of the important implications of the laws of thermodynamics for the production process--natural capital and human-made capital ultimately are *complements* because both manufactured and human capital require materials and energy for their own production and maintenance (Costanza 1980).

The analysis of energy flows has been used to illuminate the structure of ecosystems (e.g., Odum, 1957). Hannon (1973) applied input-output analysis to the analysis of energy flow in ecosystems, quantifying the direct plus indirect energy flows that connect an ecosystem component to the remainder of the ecosystem. Hannon demonstrates this methodology using energy flow data from the classic study of the Silver Springs, Florida food web (Odum, 1957) and uses this framework to estimate "shadow prices" of ecosystem goods and services (Hannon et al. 1986, 1991, Costanza and Hannon 1989). 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. Of particular importance is the recognition of the economic importance of energy quality, namely, that a kcal of one energy form (e.g., electricity) can produce more output than a kcal of another (e.g. oil). Estimating total "energy" consumption for an economy is 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. Energy use in ecological and economic hierarchies tends to increase the quality of energy, and significant amounts of energy are dissipated to produce higher quality forms that perform critical control and feedback functions which enhance the survival of the system.

For regulatory decisions that fall within EPA's discretion, energy and material flow analyses may be useful in helping to value ecosystem services, especially those services that are far removed from consumer preferences. This includes services like nutrient cycling, waste treatment, and erosion control—services that revealed or stated preference methods may not be able to adequately address because consumers are not informed about the contribution of these services to their welfare or they have simply never "constructed" preferences for these services. As pointed out by Costanza et al. (1989, p 339): "The point that must be stressed is that the economic value of ecosystems is connected to their physical, chemical, and biological role in the overall system, *whether* the *public fully recognizes that role or not.* Standard economics has too often operated on the assumption that the only appropriate measures of value are the current public's subjective preferences. This yields appropriate values only if the current public is *fully informed* (among a host of other provisos). The public is most likely far from being fully informed about the ecosystem's true contribution to their own well being, and they may therefore be unable to directly value the ecosystem's services. However, scientists may be able to derive estimates of the values that a fully informed public would produce by analyzing the structure and function of ecosystems."

In general, a "pluralistic" approach is needed to valuing ecosystem services. Comparing and contrasting the results from conventional approaches and energy-based approaches may prove useful, given the large uncertainties all around.

Costanza et al. (1989) provide an example of wetlands valuation using this pluralistic approach. They used both a conventional WTP approach and a simplified energy analysis approach based on the gross primary productivity (GPP) of coastal wetlands in Louisiana. The method was described as: "The energy analysis valuation technique looks at the total biological productivity of wetland vs. adjacent open water ecosystems as a measure of their total contributory value. Primary plant production is the basis for the food chain which supports the production of economically valuable products such as fish and wildlife. It is converted to an equivalent economic 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. This technique is comprehensive and does not require a detailed listing of all the specific benefits of wetlands, but it 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). The results are summarized in the following table from their paper:

Table 11: Table Summarizing Wetlands Benefits from Costanza et al. (1989) Comparing a Conventional WTP Approach and a Simplified Energy Analysis

TABLE 7

Summary of wetland value estimates (1983 dollars)

Method	Per-acre present value ($) at specified discount rate	
	8%	3%
WTP based		
commercial fishery	317	846
trapping	151	401
recreation	46	181
storm protection	1915	7549
Total	2429	8977
Option and existence values	?	?
EA-based GPP conversion	6400–10600	17000–28200
'Best estimate'	2429– 6400	8977–17000

They conclude that: "The EA estimate may be an upper bound on the total value of wetlands. In practice there is enough imprecision in the data and uncertainty in the methods to make it difficult to tell whether the actual numbers are over or underestimates of the true value. It was encouraging that the EA based estimate was higher than the total WTP based estimate by an amount that seemed reasonable, given the known omissions from the WTP estimate" (Costanza et al. 1989. p 355).

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).

Embodied Energy Analysis

The embodied energy method assesses the direct and indirect energy costs of economic and ecological goods and services, based on empirical work showing that available energy cost is closely linked to the value of economic output. Some neoclassical economists have criticized the energy theory of value as an attempt to define value independent of consumer preferences (see Heuttner, 1976). This criticism is, on the one hand, axiomatic,

since a major purpose of an energy theory of value was 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.
In summary, the energy theory of value overcomes some of the problems with earlier production-based theories of value encountered by the classical economists and does a fairly good job of explaining exchange values empirically in the few cases where it has been tested.

Despite the controversy and ongoing debate about the validity of an energy theory of value (Brown and Herendeen, 1996), it seems to be the only reasonably successful attempt to operationalize a general biophysical theory of value (see also Patterson 2002). Energy (and earlier labor) theories of value are inherently based on relative production costs. Thus it is more accurate to speak of energy cost or labor cost and not energy value or labor value. However, in economic systems it is well known that in perfectly competitive markets marginal cost and price will, in general, be equal in 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. To the extent that energy costs approximate the marginal cost of production, then under these conditions they would provide a proxy for the preference-based value of a one-unit increase in output

In terms of valuing ecosystem services, at least one study has compared the embodied energy analysis approach with conventional preference-based approaches to valuation for the case of coastal wetlands in Louisiana (Costanza et al. 1989). There was reasonable agreement between the two approaches and significant benefit from performing both kinds of analyses simultaneously. Costanza et al. (1997) also included (in the supplementary information) several embodied energy analyses based methods in their synthesis of the value of global ecosystem services. While these estimates were not included in the numerical totals or averages, the energy analysis-based estimates “showed fairly close agreement” with the preference-based methods.

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 input-output based methods (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) and has been characterized as simplistic, contradictory, misleading and inaccurate (Ayres, 1998; Cleveland et al., 2000; Mansson and McGlade, 1993; Spreng, 1988). Consequently, the emergy approach has not been used much outside a small circle of researchers, including some at EPA (U.S. Environmental Protection Agency, 2005). The major reason for the general lack of use in the academic or policy community, is that 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", something that is difficult understand and to apply in a standard accounting framework.

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 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). Input-output analysis methods (see above) are used to estimate direct and indirect land requirements.

Although there are ongoing debates about specific methods for calculating the ecological footprint (cf. Costanza 2000, Herendeen 2000; Simmons et al. 2000), the measurement 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 (c.f. 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 measure of the value of ecological services.

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3. ECOSYSTEM BENEFIT INDICATORS

3.1. Introduction

This report describes a range of valuation methods, from econometric analysis to citizen juries to mediated modeling. 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: first, knowledge of what is at stake in nature. This is the realm of biophysical production function analysis and determinations of how ecological endpoints change as a result of management or regulation. 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 tradeoffs arising from a particular decision.

3.2. ***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 to stakeholders and decisionmakers would benefit from the following kinds of EBI: population in viewshed of the natural area (primary demand); p[ercent 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.

3.3. Brief Description

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., recreators, farms) are present to enjoy that service? What other sources of water are available to those same user? These questions are central to economic valuation of the resource.

Key inputs. 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. A 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.

3.4. 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 wellbeing. 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 are likely to be affected by the following: 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 wellbeing can be systematically counted and mapped, 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 tradeoff analysis approaches, but do not themselves make or calculate the results of such tradeoffs. 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.

3.5. <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>3.5.1</u> <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 tradeoffs associated with different factors relating to benefits (though as noted above 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.

3.5.2 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.

3.5.3 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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4. MEASURES OF ATTITUDES, PREFERENCES, AND INTENTIONS

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 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 A 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 responding to 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 which the lay public can not 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 underlying 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 to assure that respondents have an adequate and appropriate foundation for expressing requested preferences and other judgments (see Appendix A).

While public opinion is sometimes directly used to make policy decisions, as in elections and referenda (see Part 3 section 6.4), social-psychological assessment methods more typically are intended for decision support. Thus, there has been little emphasis on mapping all expressed concerns and preferences onto a single value scale (as required for economic cost-benefit analysis methods, for example). More often separate measures are reported for several different value dimensions (e.g., aesthetic, ethical, utilitarian, personal, civic) across designated sets of policy alternatives or for specific features of those alternatives. These measures provide the psychological foundation for subsequent actions toward the measured alternatives, including political support, direct or indirect monetary payments, and acceptance of and compliance with relevant regulatory mandates. 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. 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 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 or management policies), 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., Gimblett et al. 200X). Computer simulation (“virtual reality”) and interactive game methods are also being developed, but have mostly been applied in research settings (Bishop and Lange 200X). 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.

4.1. Brief description of the Methods

4.1.1 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 A to this report). Questions may assess knowledge, beliefs, desires and/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 and/or the expected personal or social consequences thereof (see the lists in Appendix A). An example is the extensive national survey conducted to support the USDA Forest Service GIPRA process (Sheilds et al. 2002), which is illustrated in Box XXX. 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 10: Text box with illustrations from Shields et al.

][Pre-draft material for box
For example, the USDA Forest Service used a nationwide telephone survey to inform the Forest Service Strategic Plan, 2000 Revision, as required by the Government Performance and Results Act (Shields et al. 2002). By the authors' description, "approximately 7000 randomly selected members of the American public were asked about their *values* with respect to public lands, *objectives* for the management of forests and grasslands, *beliefs* about the role the USDA Forest Service should play in fulfilling those *objectives*, and *attitudes* about the job the USDA Forest Service has been doing in fulfilling their *objectives*" (p 1). This survey provided useful information about public values and concerns relevant to Forest Service management mandates, as well as quantitative measures of the relative importance to the public of particular policies (e.g., roadless areas, wilderness, timber harvesting, recreation opportunities, ecosystem health). Results were reported collectively and separately for different regions of the country, different demographic groups and for groups evidencing in one section of the survey different levels of familiarity with the Forest Service and its management mandates.

The US 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.]

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 2003). 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 offer extended possibilities for contacting respondents, presenting questions, and recording responses and their use is increasing. However, web surveys may raise representative-sampling and other issues that require special attention (Couper 2001; Krosnik XXXX; Tourangeau 2004 and Appendix A 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 US (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 11: Ribe et al. 2002 visual simulations of NW forest management options.

Text Box 11: Ribe et al. 2002 visual simulations of NW forest management options
[Text box to be added--Ribe et al. 2002 visual simulations of NW forest management options, aesthetic value and acceptability survey results]

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 combinations of different attributes (e.g., different levels of air quality, water quality and local employment). Combinations generally reflect the actual or projected correlations among the combined attributes (e.g., air and water quality may be positively correlated, and both might be negatively correlated with 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 (Louiere 1988). Multiple regression 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 tradeoffs 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 discussion under economic assessment methods in Section XXX of this report). An illustration of conjoint survey methods based on is presented in Text Box 12: What are conjoint surveys of attitudes?

Text Box 12: What are conjoint surveys of attitudes?

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 can be required to choose among (or rank or rate) compound alternatives that present specific packages of desired and less-desired attributes. For example, a policy that produces cleaner air and water in a region, but constrains employment opportunities in local communities might be pitted against alternatives that allow various levels of degradation in air and water quality, coupled with different levels of expanded employment opportunities. A simplified example of alternatives that might be presented to a respondent in a conjoint survey might be:

Which option do you think would be the best policy for public agencies in your area

Policy A: Resulting in a 10% improvement (from current conditions) in air quality, a 15% improvement in water quality, and a 15% decrease in local employment opportunities;

or

Policy B: Resulting in a 5% improvement (from current conditions) in air quality, a 10% improvement in water quality, and a 10% decrease in local employment opportunities.

Choices (or rankings or ratings) among a carefully constructed array of such alternatives can provide quantitative measures of relative public preferences for each policy option compared, as well as provide estimates of the contributions of each individual component or attribute to the conjoint preferences expressed. Following the simple example above, preferences for conjoint options might be represented by

Preference for option j = w1(WQj) + w2(AQj) + w3(Jobsj),

where option j is a particular policy that produces specific changes in the levels of water and air quality (WQj and AQj) and jobs (Jobsj). The relative contribution of each component/attribute is estimated by the derived coefficients (the wi) in the multiple regression equation for preferences among conjoint alternatives. Once determined, the regression equation can also be used to estimate preferences for new policy alternatives (based on their respective projected measures of water and air quality and jobs), so long as those options fit within the range of the attributes assessed and the constraints imposed by the context of the survey in which the policy options were offered and judged. Optimization or less formal heuristics may be applied to create additional policy options for consideration and/or for direct evaluation in subsequent conjoint surveys.

4.1.2 Individual Narratives

Researchers using the individual narrative 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) to explore and articulate the breadth and depth of expressed understandings and concerns relevant to the assessment target. Included in this category of methods are various ethnographic and the 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, but also about risks and other processes - and hence decisions, based on their mental models and mental representations of causal processes. Mental models methods aim at eliciting people’s understanding of causal processes associated with the consequences from specific decisions or actions. 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 tradeoffs 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.

4.1.3 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 (see Part 3 section 5.3). 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, 197X, XXX). 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).

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. 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 the engage in might be correlated with relevant bio-physical measures of ecological conditions to develop useful assessments of the effects ecological degradation or the effectiveness of ecological protection efforts.

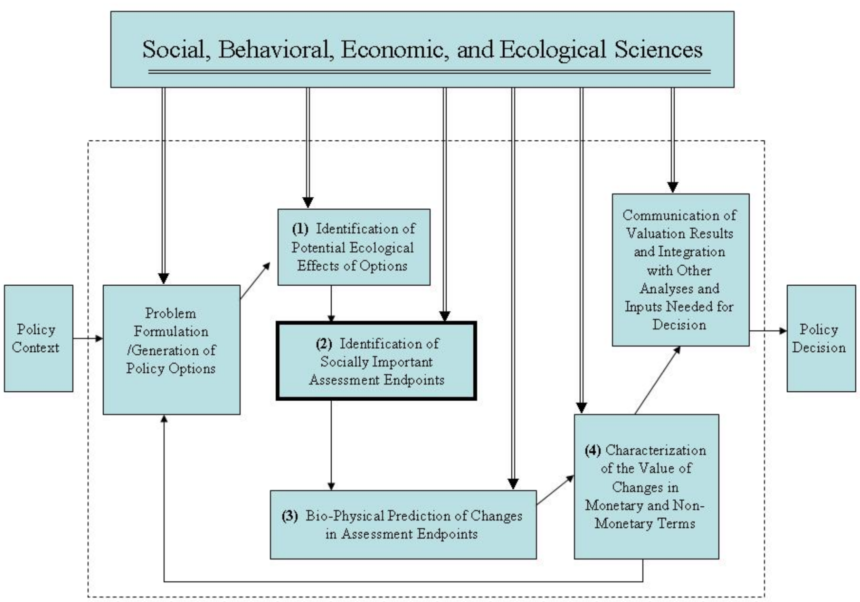
Interactive environmental simulation systems (sometimes approaching "virtual reality") provide means to overcome some of the limitations and difficulties of conducting direct behavioral observations or interpreting behavior traces. Direct observation methods are necessarily limited to existing conditions 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. XXX). 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, emersive 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.

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

Attitude survey questions have useful roles to play throughout the valuation process envisioned by C-VPESS. Attitude survey questions 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. Importantly, similarities and differences in concerns 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 policy outcomes, and for exploring public understanding of the links between chains of ecological effects and the policy options under consideration (Box 2 in Figure 2, reproduced below). Given a set of potential policy options, with their respective ecological endpoints (from Box 3 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 of all measures, or to identify possible limitations of either set of measures. 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 social-psychological surveys could provide 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 4 in the C-VPESS model. The values of ecosystems/services coming out of Box 4 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 tradeoff 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.

4.3. Status of Methods

Social-psychological surveys are the longest and most frequently used methods for determining public beliefs, concerns and preferences. Attitude survey questions have been and continue to be used effectively by all levels of government to ascertain citizen desires, concerns and preferences, by commercial marketers to determine the attractiveness of a wide array of goods and services, and by social and political scientists to measure and monitor shifting values and desires in the electorate. Economists have lately adapted survey methods to develop stated preference methods for estimating monetary values 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 Part 3 section 5.3). 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, and the many surveys listed in appendix A 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 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.

4.4. Limitations

The largest barriers to greater use of survey methods in 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 survey 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 has dampened citizen's willingness to participate, but many significant surveys continue to be conducted by a number of government agencies (see Appendix A for further discussion).

When attitude survey questions have been used they have proven effective for determining 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 have a sufficient basis for the opinions and preferences offered in a general population survey. 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 in his/her projection) regarding how well (or badly) the respondent will feel about the outcomes/implications when they are actually encountered (Wilson et al. 200X). Some approaches to addressing these problems in surveys are presented and discussed in Appendix A 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 US) that includes many members that will not be included in the sample of individuals who actually respond to the survey (i.e., the respondents). The integrity of generalizations to the population of interest is assured if the respondents are a formal “representative sample” of the population. 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 A 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 can not 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 the identification of a feasible set of alternative regulatory/protection actions, social-psychological survey methods would be appropriate for quantitatively measuring public preferences among offered sets of policy/outcome options, for estimating the relative importance to people of various attributes of alternative policies and outcomes, and for gauging the acceptability of various policies and management approaches. 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 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 systematic evaluations or comparisons of alternative policies and outcomes. These methods can 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. 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 and decisions to influence that process.

4.5. 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, Tourtangau 2004, and Appendix A 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 complex attitude survey 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.

4.6. Research needs

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

- How can social-psychological 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 social-psychological value indices be used to cross-validate estimates of monetary values (e.g., from CBA) for alternative policies/outcomes?
- How, and when in the decision process, can social-psychological, economic and bio-ecological evaluations of changes in ecosystems and ecosystems

services for alternative policies/outcomes 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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5. ECONOMIC METHODS

5.1. Overview

<u>Brief Description of Methods</u>: 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 what changes would enhance that well being.

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. These tradeoffs 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 tradeoffs 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. Nonmarket 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 tradeoffs 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 below.

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5.2. Market-Based Methods

<ins>Brief Description of Method</ins>: 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. [39]

There is a variety of contexts where this approach can be applied. 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.

<u>Status as a Method</u>: 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).

<u>Limitations:</u> 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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5.3. 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 below. 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.

5.3.1 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, beaches, etc. 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 U.S. 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.[40]

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 decision about whether and/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; that is, 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 tradeoffs 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 are: 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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5.3.2 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 tradeoffs 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.

<u>Status as a Method</u>: 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 tradeoffs 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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5.3.3 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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5.4. Non-market Methods – Stated Preference

Brief Description of the Method: Stated preference methods rely on surveys that ask individuals to make a choice, describe a behavior, or to 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 tradeoffs 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 has come to refer to any survey-based study in which respondents are asked hypothetical questions that are 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 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.

<u>Strengths and Limitations</u>: 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 discussions in Part 1, section 2.4, and Appendix A). 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 or 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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5.4.1 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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6. GROUP AND PUBLIC EXPRESSIONS OF VALUES

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/civil 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/civil 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.

6.1. Focus Groups

<u>Brief description</u>. 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 surfeys. 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 Box XXX.

<u>Relation of Method to the C-VPESS Expanded and Integrated Assessment Framework.</u> 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 US 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 (cite), 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 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.

6.2. 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 not only across academic disciplines, but also between science and policy.

Mediated modeling is 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 both across the gulf separating the relevant academic disciplines and across the even broader gulf separating the science and policy communities, and the public. Appropriately designed and appropriately used mediated modeling exercises can help to bridge these gulfs. 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 above, 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 – that is what it does best
- 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 of which 68% 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 implementable 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 of failure of these processes.

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6.3. Valuation by Decision Aiding

Decision aiding approaches provide a method for valuing protection of ecological systems and services in multiattribute terms. 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 a 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 to form judgments about the value of ecological systems and services by way of a comparative framework. Decision aiding approaches help people to, from a prospective standpoint, 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 to determine which, in a set of options is most valuable (i.e., *is Option A in a set of alternatives better—i.e., 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, an analyst (or analysts) facilitates 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 13: Types of Attributes).

Text Box 13: 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:

1. *Natural attributes* are direct measures 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.
2. *Proxy attributes*, 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 migratory shorebirds that it attracts. However, it is frequently the case that accurate direct counts of shorebirds, which would 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.
3. Constructed *attributes* 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, sociologists, etc.—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 12). 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 above, 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 12: Comparative Matrices of Attributes for Three Hypothetical Decision-Aiding Valuation Scenarios
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These differences necessitate the need for tradeoffs—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 tradeoffs, 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 tradeoff analysis across these attributes can help individuals or groups to 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 multiattribute value of an environmental system *absent* a comparative framework for tradeoff 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 multiattribute 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 $\mathcal{X}$. However, EPA may be required to repeat this procedure at other sites to determine, in relative terms, how significant this value (of $\mathcal{X}$) 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 which include the self-ratings of decision makers and measures of internal consistency (i.e., the degree to which the approach helps people to makes 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 multiattribute utility theory to construct a single, uni-metric "value" that encompasses the diverse array of attributes.

<u>Strengths/Limitations</u>

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 tradeoffs, 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 tradeoffs all will require effort on the part of EPA.

<u>Research Needs</u>

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 guaranteed 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, spatial, etc. differences) may adversely affect other valuation methods as well.

<u>Examples of Applications</u>

Arvai, J., and R. Gregory. 2003a. A decision focused approach for identifying cleanup priorities at contaminated sites. Environmental Science & Technology **37**:1469-1476.

Arvai, J. L. 2003. Using risk communication to disclose the outcome of a participatory decision making process: Effects on the perceived acceptability of risk-policy decisions. Risk Analysis **23**:281-289.

Arvai, J. L., and R. Gregory. 2003b. Testing alternative decision approaches for identifying cleanup priorities at contaminated sites. Environmental Science & Technology **37**:1469-1476.

Arvai, J. L., R. Gregory, and T. McDaniels. 2001. Testing a structured decision approach: Value-focused thinking for deliberative risk communication. Risk Analysis **21**:1065-1076.

Gregory, R., J. L. Arvai, and T. McDaniels. 2001a. Value-focused thinking for environmental risk consultations. Research in Social Problems and Public Policy **9**:249-275.

Gregory, R., T. McDaniels, and D. Fields. 2001b. Decision aiding, not dispute resolution: Creating insights through structured environmental decisions. Journal of Policy Analysis and Management **20**:415-432.

Gregory, R., and K. Wellman. 2001. Bringing stakeholder values into environmental policy choices: A community-based estuary case study. Ecological Economics **39**:37-52.

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Keeney, R. L., and H. Raiffa 1993. Decisions with multiple objectives: Preferences and value tradeoffs. Cambridge University Press, Cambridge, UK.

McDaniels, T., R. Gregory, and D. Fields. 1999. Democratizing risk management: Successful public involvement in local water management decisions. Risk Analysis **19**:497-510.

6.4. 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 as a 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 tradeoffs 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 tradeoffs 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, with the tradeoff 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.

<u>Strengths/Limitations:</u> 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; 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 transactions 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. 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:

a) 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;

b) a unitary conservation or environmental benefit is involved;

c) the initiative or referendum outcome was a close vote (this yields stronger inferences about the actual valuation, rather than floors or ceilings);

d) extraneous issues (such as whether the vote is a "political test" on particular politicians, or the mode of financing is controversial) are unimportant;

e) 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 do 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 if it exists.

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 tradeoff 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 so-called "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 a certain 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 eco-system 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.*(emphasis added)

<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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6.5. Citizen Valuation Juries

Description of the Method. Another potential process for attempting to measure the social/civil 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 (discussed in Part 3 section 6.4) and contingent valuations (discussed in Part 3 section 5.4). 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 valuations 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, have 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 and check on traditional economic valuation approaches. Decisions whether to pursue particular regulations or other governmental actions 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. Concern whether contingent valuation surveys provide sufficient time and information for survey respondents to generate reliable estimates of the value of often complex ecological changes, for example, has led some researchers to investigate other group-based approaches to valuation. Under the “Market Stall” (“MS”) approach, for example, 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.

<u>Relation of Method to the C-VPESS Expanded and Integrated Framework</u>.

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 2 and 4 of the C-VPESS proposed valuation process.

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.

<u>Status as a Method.</u> 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).

<u>Treatment of Uncertainty.</u> 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.

<u>Research Needs</u>. 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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7. 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.

7.1. 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: 1) the human-engineered system provides services of equivalent quality and magnitude, 2) the human-engineered system is the least costly alternative, and 3) 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 is a value not of ecosystem services themselves, but is the value of having a means to produce the service via an ecosystem rather than via an alternative human-engineered system.

All valuation methods can be misconstrued applied incorrectly and misinterpreted, however the replacement cost method require special caution. There is great potential for abuse in using replacement costs to estimate the value of ecosystem services and 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 constraints 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 that service would be demanded if provided at the least cost human-engineered alternative way of providing the service, 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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7.2. Tradeable 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.

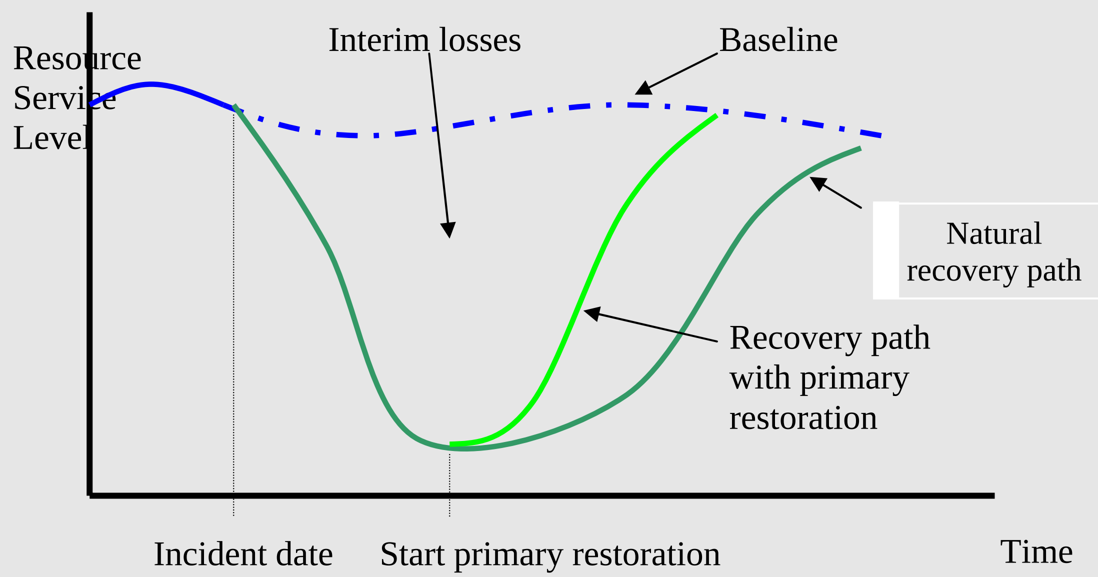
Emissions permit trading has been allowed under the Clean Air Act since the 1990 Amendments. Under a cap-and-trade system, such as that used by EPA to reduce sulfur dioxide emissions, the regulatory body determines the total number of permits available and some means of allocating permits among regulated sources. A regulated source must ensure that it has sufficient permits to cover its activities or 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 reflect 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 equal. But issuing the right number of permits to get marginal cost equal to marginal benefits 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.

7.3. 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 8) 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 8: 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 19: Equation for Habitat Equivalency Analysis. 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 19: Equation for Habitat Equivalency Analysis

$$\overbrace{\sum_{t=t_0}^{t_l} L_t (1+i)^{(P-t)}}^{\text{“Debit: PDV Loss”}} = \overbrace{\sum_{S=S_0}^{s_l} R_s (1+i)^{(P-s)}}^{\text{“Credit: PDV Gain”}}$$

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 require 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 NPV 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 the 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 both past and future time scales in that it involves compensation for injury or estimate service produced by past action, as well as, allows 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 expresses 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 uses 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 to 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. These methods 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 their 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 does 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 at in the identification of restoration projects and desired services to more widely accepted restoration decisions.

<u>Status as a method.</u> 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 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, while under OPA the focus is on replacement of resources to achieve compensation. Under OPA the question is how much new public resources the public requires to be made whole for their loss, so 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 or adaptations of them 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.

<u>Strengths/Limitations.</u> 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 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 in outputs be tracked (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 other contexts then Natural resource Damage Assessment. In particular they 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 derived form specific important habitat types (e.g. coastal wetlands, bottomland hardwood forest. etc). Such value definitions for ecosystem service could then be couple 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 A: SURVEY ISSUES FOR ECOLOGICAL VALUATION: CURRENT BEST PRACTICES AND RECOMMENDATIONS FOR RESEARCH

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

The C-VPESS recognizes, however, that issues related to survey methods are important to some methods of ecological valuation and learned they were of particular concern to EPA representatives participating in the SAB's 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).[41] 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.[42] 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).[43]

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).[44]

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.[45]

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 EPA 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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Editor's note: the references presented in three separate lists on pp. 342 – 383 will be merged.

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ENDNOTES

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

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

[3] The SAB Staff Office published a Federal Register Notice on March 7, 2003 (68 FR 11082-11084) announcing the project and 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 **(insert additional dates).**

[5] It is important to acknowledge that they are many different specific definitions for the term "benefits."
When used in technical discussions, different definitions make different assumptions that can be important to how the concept is interpreted. This implies that not all technical concepts for benefits are equivalent. Throughout this report, we use the term "benefits" to refer to the general concept of contributions to people. Economists generally use the term to refer more specifically to compensating or equivalent variations as measures of changes in the well-being of individuals (see Freeman, 2003). Despite these differences, the term services and the listing of types of services here refers exclusively to the contributions of ecosystems to human well being. As such, it is based on an anthropocentric concept of the benefits provided by ecosystems.

[6] 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 help develop that list, can ensure appropriate recognition of the full range of

potential impacts of a given policy option. This can help make the analysis of ecological science more transparent and accessible and can help inform decision makers of the relative merits of different options before them.

[7] 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. .

[8] A large literature exists on the use of economic valuation methods to estimate the value of changes in environmental quality. For a comprehensive description of these methods, see Freeman (2003).

[9] In order to address these concerns, one or both of two steps can be taken in measuring public opinion. First, the EPA can conduct surveys that educate survey respondents about a problem or policy before asking for their preferences. Alternatively, in addition to preferences, the EPA can routinely measure the degree of knowledge and thought that people bring to a survey. This latter approach could be used to determine whether the preferences of members of the public who are knowledgeable and thoughtful are substantially different from those who are not. If this is the case, the EPA could conduct statistical simulations to predict what preferences people would express if everyone were fully informed and deeply thoughtful.

[10] In particular, the circular states that "'Opportunity cost' is the appropriate concept for valuing both benefits and costs. The principle of 'willingness-to-pay' (WTP) captures the notion of opportunity by measuring what individuals are willing to forgo to enjoy a particular benefit. In general, economists tend to view WTP as the most appropriate measure of opportunity cost, but an individual's 'willingness-to-accept' (WTA) compensation for not receiving the improvement can also provide a valid measure of opportunity cost" (OMB, p. 18).

[11] 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.

[12] 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.

[13] In addition, the Circular states (p.27 – check page number) "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).

[14] 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.

[15] 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.

[16] 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.

[17] 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.

[18] 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.

[19] 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".

[20] 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.

[21] 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.

[22] 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.

[23] 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).

[24] 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.

[25] 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).

[26] 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.

[27] 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.

[28] 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

[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 Sanzome, 2002

[30] 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).

[31] A number of the gasses emitted from CAFOs have adverse air quality impacts that are interrelated with the water quality impacts.

[32] 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)

[33] 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.

[34] 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).

[35] 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.

[36] 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.

[37] See Phaneuf and Smith(2005) for a review of the literature and Phaneuf (2002, Phaneuf, Palmquist and Smith (2006), (Egan (2004), von Haefen (1999), and Egan and Herriges (2006) for examples of applications involving freshwater recreation sites in different regions.

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

[39] 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.

[40] 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.

[41] 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	

[42] 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).

[43] 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).

[44] 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).

[45] 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.