SAB Committee on Valuing the Protection of Ecological Systems and Services (C-VPESS)

METHODS REPORT

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1 SCOPE AND BACKGROUND OF REPORT

1.1 Scope

This report focuses on the methods available to EPA for valuing the protection of ecological systems and services. It discusses a variety of methods and aims to convey the purpose and usefulness of each and the limitations and issues associated with their use. It is intended to enrich the information available to decision makers and analysts so they can make better informed decisions about the use of information and methods for valuing the protection of ecological systems and services. The committee recognizes that different information and methods will be important for different decisions contexts.

Charge to committee, summary of meetings

Discussion of how EBASP review, Document Zero's call for an expanded and integrated approach led to this document.

Box with key definitions (e.g., value, benefits, ecosystem, valuation).

1.2 Role of benefits assessments (especially viz. ecosystems/services) in Agency policy/decision making

Efforts to assess the benefits of ecosystems and ecosystems services are intended to support Agency policy and decision making with the ultimate goal of effectively and efficiently protecting and promoting the health and well-being of US citizens. In that context it is generally recognized that the health, integrity and sustainability of ecosystems and the continuing availability of the many direct and indirect services they provide to society are an essential foundation (see APPENDIX A: INSTITUTIONAL CONTEXT OF DECISION-MAKING LEGAL MANDATES AND EPA'S NEED FOR INFORMATION ON ECOLOGICAL BENEFITS AND A BRIEF HISTORY OF ECOLOGICAL PROTECTION AT EPA for EPA's legal mandates and need for information on ecological benefits and a brief history of ecological protection at EPA). In addition to these “instrumental” values of ecosystems/services, many advocate that ecosystems have “intrinsic” values (based on moral/ethical grounds) and should be protected regardless of any implications for human welfare. Moreover, surveys of public opinion have consistently identified a widely-shared concern about and strong support for the protection of ecosystems and “nature” among US citizens, even if the basis of this concern is not consistently or clearly articulated.

The extent to which the policies of a public/government agency should consider intrinsic values or be affected by public opinion is a matter of some controversy. However, given current and foreseeable levels of knowledge of ecosystem processes and how the services they support and/or provide may affect the well-being of current and future citizens, the question of whether Agency policy should be restricted to instrumental values or extended to encompass intrinsic values may be of little consequence. Whether the protection of ecosystems and ecosystem services is predicated on their instrumental value to society (given a reasonably enlightened view of social well-being over a reasonable period of time) or on some ethical or biocentric notion of intrinsic values or simply on the political force of current public opinion, the practical implications for environmental policy and decisions will often be the same. If instrumental, intrinsic and socially assigned values do conflict in some specific decision context, it will be important that Agency decision makers are informed of these conflicts and take them into consideration. The principal role for ecosystems/services benefits assessments is to identify and articulate the implications of Agency policies and decisions for the full array of economic, social and ecological values, whether instrumental or intrinsic.

1.3 Recognition that different institutional settings, ecosystem types and policy objectives will mean that different information and methods will be important in different decision contexts.

Contexts for EPA actions extend from national rule making to regional/local decisions to review and evaluation of existing programs at all of these levels. Actions may focus on particular ecosystems/types (e.g., wetlands, rivers, alpine, estuaries), on specific environmental components that transcend ecosystems (e.g., air and water quality) or on very specific pollutants/toxics (e.g., mercury, lead) wherever they may occur in the environment. Each of these contexts will require different levels and scales of bio-ecological information which may be more or less well-known with greater or lesser degrees of certainty. Similarly, the social consequences of projected bio-ecological changes may be more or less well-known and certain. Each of these contexts presents different demands and constraints for ecosystems/services benefits assessments. In some cases specific types of benefits assessments are stipulated by statute, regulation or executive order. For example, monetary assessments are needed to support required benefit-cost or cost-effectiveness analyses, human mortality and morbidity assessments support decisions based on human health criteria, and effects on quality of habitat support decisions that fall under threatened and endangered species statutes. However, these stipulations do not generally preclude other assessments and in most cases the relevant rules and guides require or strongly encourage a broad approach to benefit assessment to assure full consideration of the relevant bio-ecological, social and economic consequences of agency decisions and actions.

1.4 EPA valuation with respect to ecosystems and services

Figure 2-1 presents a generic schematic of the interrelationships among the key components of Agency valuation that supports policy and decision making relevant to ecosystems/services. All components are seen as interacting, with each component mutually affecting and being affected by the others.

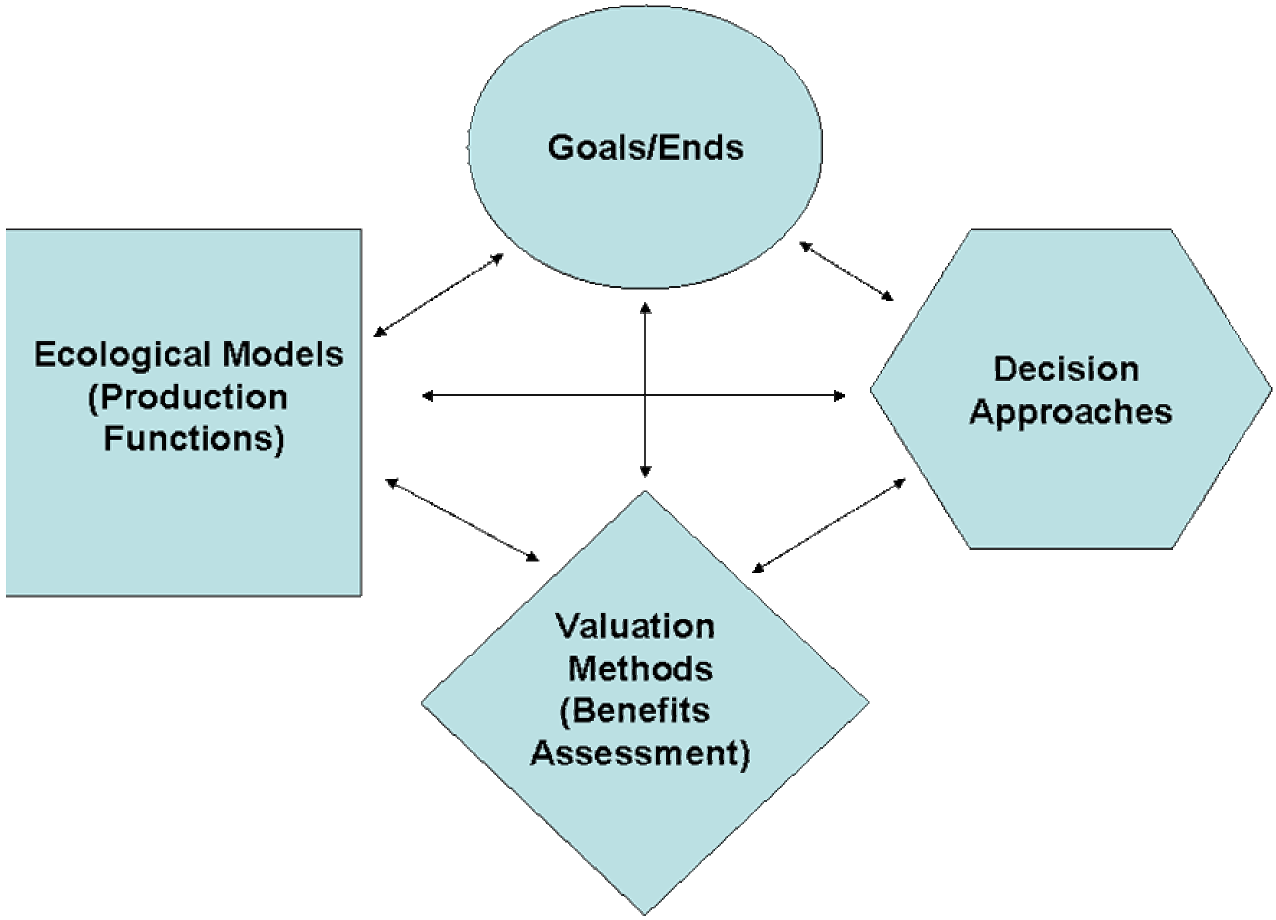

Figure 1-1: Conceptual Diagram of Major Components of Valuation.

The *Goals* for actions may be quite broad (protect human health and well-being) or very specific (reduce exposure to a specific toxin). Goals may be framed in terms of bio-ecological effects/conditions (minimum standards for surface water quality, biodiversity, or areas of wetlands) or in terms of human/social effects (minimum standards for drinking water quality, maximum tolerances for mortality/morbidity, protection or enhancement of opportunities for water-based recreation).

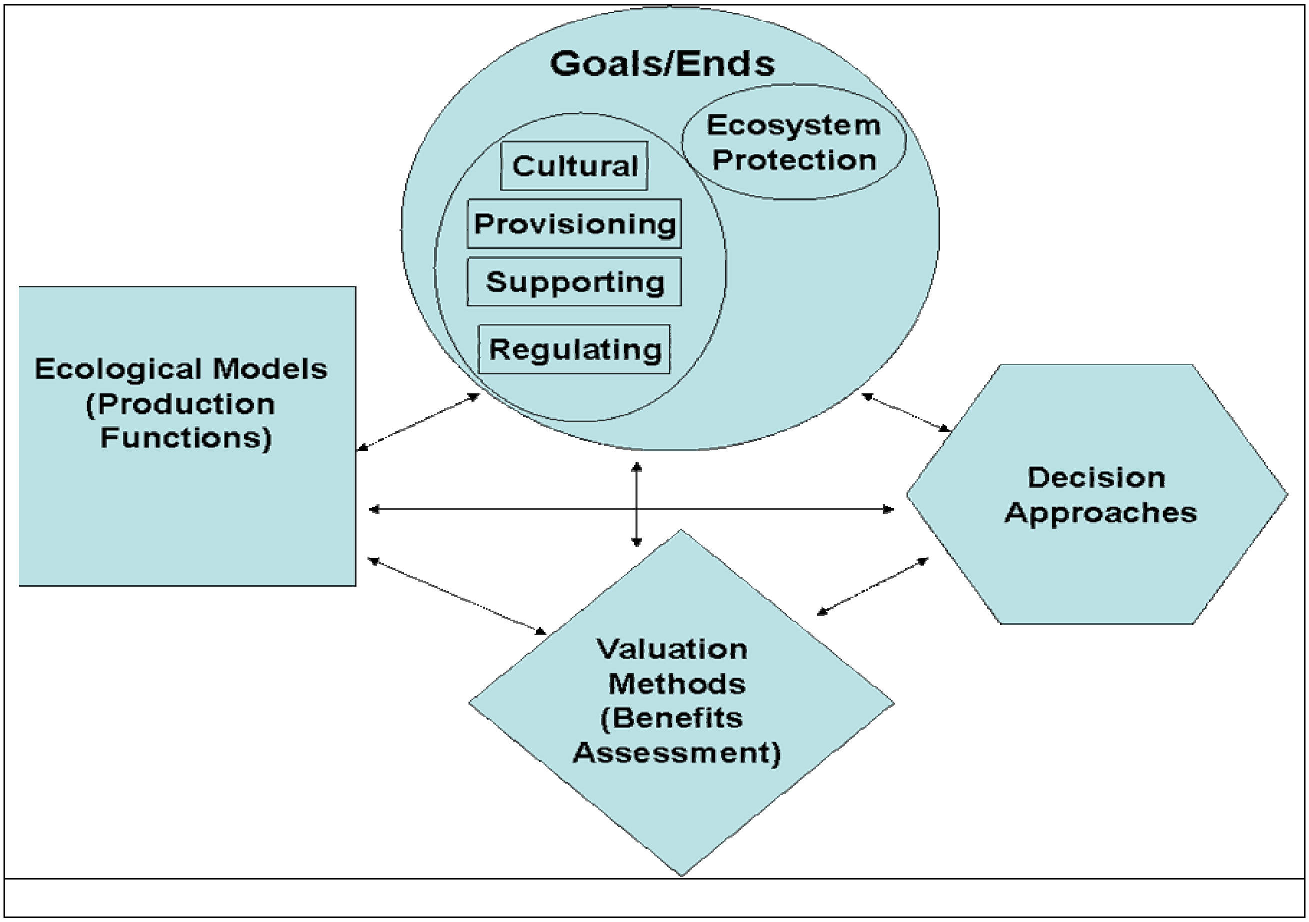

Figure 1-2: Conceptual Diagram of Major Components of Valuation: Choice of Goals or Ends

Goals are affected by and have implications for *Ecological* conditions and model-based projections of environmental changes with and without proposed Agency actions. In the *Valuation* component changes in relevant ecological conditions are translated into consequences for individuals, communities and society more generally. Interactions between the bio-ecological and social values components may be required to identify ecological endpoints that are important to specific social values, and/or to determine social values associated with ecological endpoints that are projected to change significantly. The assessed values (benefits and costs) of proposed actions and alternatives are passed to the *Decision* component, and the operable form of the decision making process may dictate a particular form for the value assessments. Depending on the results of the value assessments and the relevant decision rules, regulations are promulgated and/or actions are taken by the Agency to affect changes in human/institutional behavior to maintain or promote environmental conditions that are deemed effective for achieving the goal.

For any given policy/decision case, the interaction among system components may be played out several times at different levels of abstraction and at different levels of specificity. For example, at the highest level the goal of protecting human health and well-being may imply a general concern for the effects of pollutants on water quality in streams and lakes. Bio-ecological assessments and models may identify the effects of specific pollutants in specific water systems, some of which may be determined by preliminary benefits/risk assessments to have the potential of substantial effects on human health and social well-being. Legal/institutional considerations may prescribe a "safe minimum standards" approach for determining Agency response to the identified pollutants. This institutional choice feeds back to other components of the system resulting in the more precise goal of determining the projected deposition, dispersion and concentration of specific pollutants in specific water bodies, and the determination of the human-social consequences of targeted pollutant concentrations, including direct and indirect effects on human health, along with other ecological, social and economic benefits and costs. A later iteration of the system may focus on setting the minimum safe concentration of a specified pollutant, as measured by specified procedures, to be accomplished by emissions controls on designated industries and enforced by a particular schedule of penalties and/or incentives.

2 DECISION MAKING AND VALUATION

2.1 Introduction

The simplest policy decisions about which to provide information on the values of ecosystems and services are decisions with the following characteristics: a) there is relatively complete information about how ecosystems and services are affected by alternatives policy options, b) there is relatively complete information about how ecosystems and services translate into values, c) there is have a single metric by which all consequences of alternative decisions can be compared, and d) decision-making involves a one-time (static) choice. In this case, valuation can provide clear information on the net benefits of various policy options making for relatively easy comparisons across options. However, almost every important environmental policy decision is made without full knowledge about how policy decisions affect ecosystems and the provision of various ecosystem services, or about how these changes impact the values associated with ecosystems and services (uncertainty). Policy choices typically have consequences that affect not only the present but also the future and inevitably influence future policy choices (dynamics). Policy choices typically have multiple dimensions, many of which may be difficult to compare in the same metric (multiple objectives).

For example, EPA regulation of nitrogen oxides (NOx) from stationary or mobile sources will reduce NOx emissions, which will lead to improvement in ambient air quality and the reduce nitrogen deposition in terrestrial and aquatic ecosystems. To date, valuation of the benefits of reduced NOx emissions has focused on human health consequences while benefits in terms of ecosystems and services have received far less attention. Reduced NOx emissions may affect vegetation, with consequent changes in the structure and function of terrestrial systems. Nitrogen runoff from terrestrial systems and air deposition into freshwater and marine systems can cause significant changes in those systems as well. These effects may cause a change in an array of ecosystem services from crop and fisheries productivity to amenity and existence values. The effects of policies on ecosystems and services, and the effects of these changes on values may be largely uncertain. The values of these services may be difficult to evaluate or compare in common terms. Some of these impacts may be long-lasting or difficult to reverse.

The dynamic and stochastic nature of ecosystems requires valuation methods capable of handling both dynamics and uncertainty. Many existing valuation methods as currently practiced are not well suited to such situations. For example, economic valuation methods typically generate a probability distribution of value estimates rather than simply a point estimate so in this sense these methods incorporate uncertainty. However, many economic valuation techniques, as currently applied, are static in that they ask about values under a given set of conditions. For revealed preference methods, such as hedonics or the travel cost, estimates of value are based on current conditions and currently held expectations about future conditions. One could use the estimated equation to forecast values under new sets of conditions. However, major changes in conditions or expectations will change fundamental relationships, making forecasts based on the current estimated relationships invalid. On the other hand, stated preference methods can be designed to ask about a range of potential conditions, some of which could be quite different from current conditions. How well people are able to respond to valuation questions about unfamiliar conditions, however, is open to debate. A key issue is whether current valuation methods are capable of addressing valuation in the context of dynamic stochastic systems.

This section discusses uncertainty, dynamics and multiple objectives in the context of valuing ecosystems and services. Different methods for incorporating uncertainty, dynamics and multiple objectives are discussed as are the implication of these methods for valuation and decision-making.

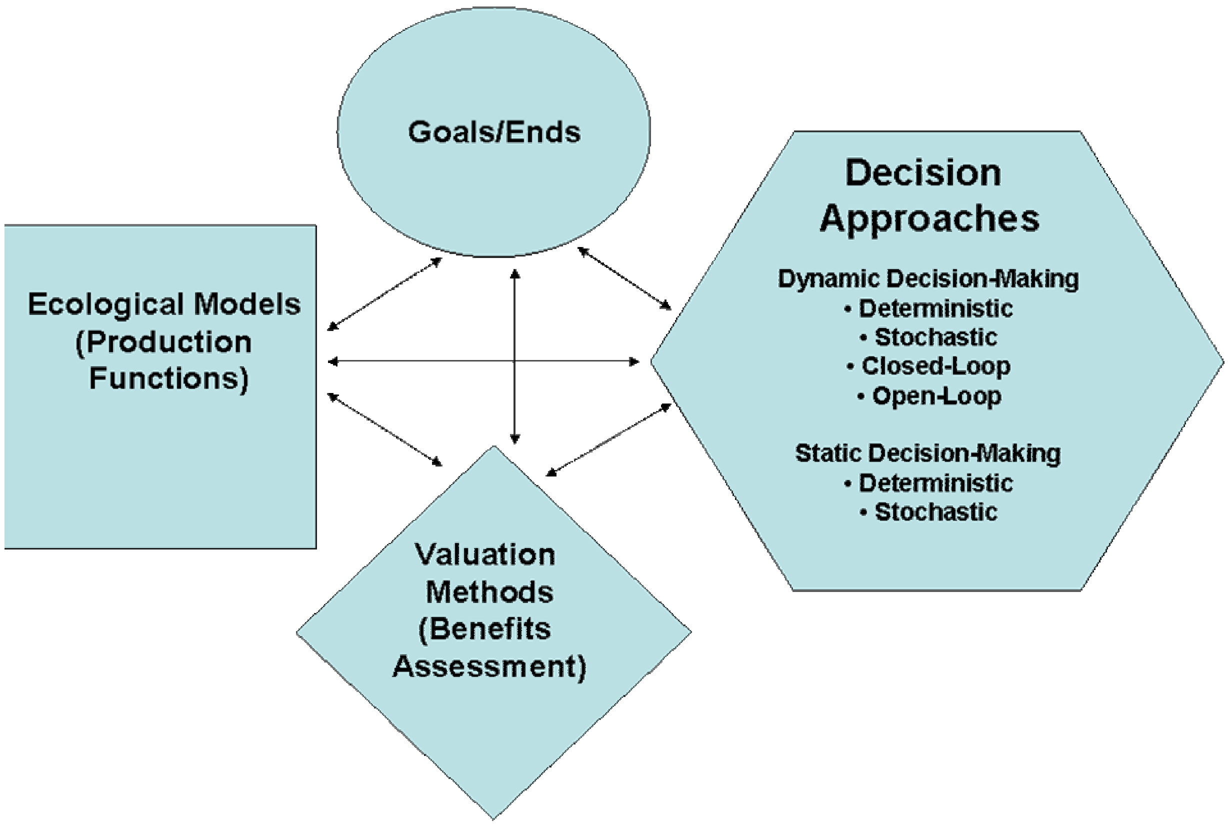

Figure 2-1: Conceptual Diagram of Major Components of Valuation: Decision Approaches

2.1.1 Valuing Ecosystems and Services: Single or Multiple Dimensions of Value?

A major issue in valuing ecosystems and services is whether all dimensions of ecosystems and services can be compared using a common metric. For example, can we compare the value of increased production of market commodities, typically measured in monetary terms, with protection of endangered species or increased ecosystem resilience? Benefit-cost analysis measures all consequences of policy alternatives in terms of a common metric (money). On the other hand, approaches such as multi-attribute analysis keep accounts in multiple dimensions without converting to a common metric.

Making decisions about which policy alternative is preferred requires comparing alternative policies according to some criterion of overall worth or value. In a fundamental sense then, decision-making would seem to require a common metric of value. But this does not necessarily mean that the valuation process requires analysts to choose methods that require a single metric of value. Whether to use a single metric or multiple metric valuation method is really a question about where in the decision-making process to make difficult value judgments. With single metric approaches like benefit-cost analysis, valuation exercises are conducted that attempt to measure all of the consequences of policy alternatives in terms of the common metric. All consequences of a policy decision, from effects on the production of commodities to effects on endangered species, are measured in dollar terms. The difficult value judgments on the relative worth of different dimensions are made in this valuation step. Once this is done, the decision-maker can then easily compare across policy alternatives to see which alternative generates the greatest net benefits. Under multiple metric approaches like multi-attribute analysis, the consequences of alternative policies are measured on multiple dimensions. There is no attempt to compare the values of disparate dimensions like production of commodities and endangered species protection in common terms. The decision-maker then faces decisions involving alternatives that may be good on some dimensions but poor on others. To decide which policy alternative is preferred, the decision-maker may need to make value judgments of relative worth of different dimensions. These two approaches emphasize different research strategies and different decision-making agendas. In the single metric approach, valuation is the crucial step while decision-making is relatively transparent. In the multi-metric approach, valuation can lay out tradeoffs but ultimately it is value judgments in the decision-making stage that are crucial.

In practice, there isn't as stark a contrast between the two approaches as described in the prior paragraph. Most real decision-making processes involve elements of both approaches. Most supporters of benefit-cost analysis readily admit that it is difficult to capture all relevant dimensions of net benefits in a formal analysis and inevitably there will be a role for judgment in decision-making (Arrow et al. 1996). On the other hand, there is typically a vast array of potential dimensions affected by policy alternatives so that some formal analysis is required to organize information and reduce the number of dimensions in laying out policy alternatives to decision-makers.

Section 2.3 discusses single metric valuation methods. The section briefly discusses methods for static decision-making contexts with little uncertainty and then extends this to briefly discuss methods that incorporate uncertainty. Next, methods that incorporate issues with on-going (dynamic) decisions are discussed. Multiple metric valuation methods are discussed in section 2.3.

2.2 Single Metric of Value

Economists use the notion of a utility function to compare the value to an individual of choosing different consumption bundles, each of which specifies the quantities of numerous different goods consumed. The utility function maps the multiple different dimensions of consumption, one for each good consumed, into a single dimension, utility. Consumption bundles can then be compared in terms of their utility, with the individual choosing the bundle that generates the highest utility from the set of available consumption bundles. Utility, however, cannot be used directly as a metric of value. Utility is subjective, not comparable across people, and not directly observable. However, choices that people make can be observed. These choices provide a window onto the individual's underlying utility function. By observing choices, economists can observe which alternatives are preferred by an individual and make inferences about their utility function. In particular, when individuals make choices in markets that involve trading money for goods or services, economists can infer an individual's willing-to-pay (or willingness-to-accept) for those goods and services. The willingness-to-pay of individuals can be summed across all affected individuals yielding an aggregate willingness-to-pay (or willingness-to-accept) measure of value denominated in money. This measure can then be used to compare policy alternatives.

The major difficulty with using observed choices to infer willingness-to-pay or willingness-to-accept for valuing ecosystem and services is that markets do not exist for ecosystems or for many services. Economists have developed a number of non-market valuation techniques to address the problem of estimating willingness-to-pay (or willingness-to-accept) when markets do not exist. These methods are described in greater detail in section 4.

For the rest of this section, some means of estimating willingness-to-pay or willingness-to-accept are taken as given and the focus is on how to utilize these estimates in guiding decision-making involving ecosystems and services. Assuming that a single metric of value is available, the question then becomes what information or considerations are important to include for decision-makers when values of ecosystems and services are at stake. In part, this depends on the circumstances in which values are to be considered. The discussion here proceeds from the simplest case involving a single decision with no uncertainty and progressively working to more complicated cases involving a series of decisions and uncertainty. Throughout this discussion, an aquatic ecosystem supporting various services including a fishery will be used as an illustrative example.

2.2.1 Static Decision-Making with Complete Information

When there is a one-time (static) policy decision, for which most important information is known, the context for providing valuation information to decision-makers is straight-forward. Analysts collect information on the effects of alternative policy choices on ecosystems and the production of ecosystem services. Analysts then apply valuation methods to convert changes in ecosystems and services into estimates of the monetary value of those changes. Adding up the benefits and costs under a particular policy yields the net benefits of the policy (relative to the status quo). Various policy alternatives can be compared on the basis on which alternative yields the highest net benefits.

What is necessary for decision-makers to have confidence in using such a simple benefit-cost approach is that ***all*** of the effects of policy alternatives are incorporated into the analysis and that all effects can be translated into monetary values. If some values are left out because they are difficult to estimate, then the net benefits calculation is incomplete and may provide misleading signals of the relative value of policy alternatives. Given the current state of the art, there are likely to be important changes in ecosystems and services that for which monetary estimates of value are unavailable. How to incorporate information about effects on ecosystems and services for which monetary estimates of value are unavailable is a topic of primary importance.

Even though this section is concerned with static decision-making, some of the effects of the decision may occur in the future. Aggregating benefits and costs across time requires some method of comparing the value of benefits and costs that accrue today with those that accrue in the near or far distant future. The standard approach to aggregating net benefits across time is to use discounting to calculate net present values. Critics of discounting content that it places too little weight on events that happen far in the future, making it inappropriate for use in climate change or species extinction for which consequences are long-lived. Alternatives to standard discounting involved hyperbolic discounting, in which the discount rate falls as the time period considered is lengthened, or some time of sustainability criterion that requires the future to be at least as well off as the present.

Of course, few if any real world policy decisions involving ecosystems and services will have a complete set of information about how policy links to changes in ecosystem conditions or provision of services, and how these changes link to changes in human well-being, for all important . For this reason, we now turn attention to consideration of decision-making with uncertainty.

2.2.2 Static Decision-Making With Uncertainty

When there is incomplete information about how policy will affect ecosystems and the provision of services or how such changes affect human well-being, decision-makers will face making choices under uncertainty. This section outlines several methods for incorporating uncertainty into decision-making.

One approach for decision-making under uncertainty is to assign probabilities for all possible outcomes and then to make decisions based on expected net benefits. For example, suppose the benefits of reducing air emissions depend on whether it will be sunny or cloudy. The probabilities of each event (sunny, cloudy) will be multiplied by the net benefits in each case to derive the expected net benefits. When there are both present and future benefits and costs the approach would be to calculate expected net present value. Maximizing expected (present value) net benefits has the advantage of being transparent and straight-forward. The disadvantages of this approach are that it requires probability assessments and assumes risk neutral behavior.

When faced with uncertain outcomes, many people would rather choose an option with a lower risk of bad outcomes even though the expected net benefits of this option are the same or lower than some other option. For example, in choosing between an option that gives $100 for sure versus a 75% chance of no return and a 25% change of $400, the vast majority of people choose the $100 option (Kahneman and Tversky 2000). Both options have an expected return of $100 but they differ in the amount of risk that a person faces. One could ask whether people would rather receive $90 or $80 versus the 75% chance of no return and a 25% change of $400. The difference between the sure thing and the expected value of the risky option that makes people indifferent between the two choices is called the risk premium. People willing-to-pay a risk premium are said to be risk averse. It may the case that people are risk loving rather than risk averse, as when they buy lottery tickets. When people are asked whether they would rather choose an option in which they lose $300 for sure or an option in which there is a 25% chance of losing nothing and a 75% chance of losing $400, the majority of people choose the risky option (Kahneman and Tversky 2000). Note that this is really the same gamble as the first one where this time it is framed in terms of losses rather than gains, showing the importance of framing of risk on decision-making.

Incorporating risk aversion (or risk loving) behavior into decision-making requires changing the objective from one of maximizing expected net benefits to maximizing expected utility, where the latter incorporates attitudes toward risk. In practice, it may be difficult to get an accurate assessment of attitudes toward risk making it difficult to make expected utility operational.

In many cases, it is not possible to establish objective probabilities based on prior experience or first-principles. For example, novel situations (designing a high-level nuclear waste repository) or situations in which there is a regime shift in system structure mean that past experience cannot be relied upon to generate objective probabilities. In such cases, probability assessments must be subjective. For individual decision-making, subjective probabilities are whatever the individual assesses the odds to be. For policy purposes, subjective probabilities will need to be set by some means, either through asking experts or surveying the affected public.

In instances of truly novel events, people may be unable or unwilling to assign even subjective probabilities. It may also be difficult to even know what outcomes might be possible. Some methods for decision-making under uncertainty do not rely on probability assessments. Rather than trying to maximize expected net benefits or expected utility, methods such as safe minimum standards or the maxi-min rule focus on minimizing the risk of a very bad outcome occurring. Under safe minimum standards decision-makers should avoid decisions that might push a system beyond a threshold that could lead to large negative consequences, unless the costs of doing so are intolerable. Under maxi-min rules, society should choose among alternatives based on the alternative that generates the best worst-case outcome. Maxi-min rules can be justified either on the basis of extreme risk-aversion or in cases where it is not possible to assign probabilities to outcomes. Maxi-min rules have been justified as being an appropriate strategy in cases where probabilities for events cannot be assigned (Arrow and Hurwicz 1972, Maskin 1979).

A related notion in environmental policy circles is the Precautionary Principle. The Precautionary Principle states that society should avoid actions that may result in large damages even though there is not conclusive scientific proof of cause-effect relationships. Precautionary Principle language is included in many international treaties including Agenda 21 from the 1992 United Nations Conference on Environment and Development

2.2.3 Dynamic Decision-Making with Uncertainty

Most policy decisions that affect ecosystems and services have potentially long-lasting effects, can be revisited in the future, and are subject to considerable uncertainty. For these reasons it is important that the value of protecting ecosystems and service incorporate both dynamics and uncertainty. However, doing so makes the valuation exercise considerably more difficult. It is not sufficient to do analysis about the current situation and do valuation studies about the current situation. Rather, what is needed is information about the likely state of the ecosystem and the provision of services through time and consequent associated values through time.

An approach to dynamic decision-making is to choose a series of decisions that maximize the present value of expected net benefits or expected utility. In principle, optimal decisions can be derived by applying optimal control theory or dynamic programming. When there is little opportunity or it quite costly to revisit decisions, the path of decisions may be chosen at the outset, what is called an open-loop strategy. Without uncertainty, open-loop strategies can be optimal. However, with uncertainty and learning through time, new information may reveal that a change in plan is needed. Closed-loop (feedback) strategies allow decisions through time based on the information available at that time. Closed-loop strategies can be ex-ante optimal, i.e., optimal given the information at the time of the decision, even with uncertainty. Stochastic dynamic programming is the mathematical tool that can be applied to finding optimal closed-loop strategies. A drawback to the practical application of stochastic dynamic programming is that the complexity of the analysis rises exponentially with time and potential alternative choices considered (the curse of dimensionality). This difficulty may require that analysts adopt rules of thumb or considerably simplify the problem in order to make headway.

An approach that tries to be both practical and to incorporate the ideas of stochastic dynamic programming is adaptive management. In adaptive management, current management actions are designed partly as experiments to reduce uncertainty and provide a broader base of knowledge that contribute to more effective future management decisions (Walters 1986). There are tradeoffs in experimentation: expected benefits in the near term may have to sacrificed in order to learn information that may be of value for future decisions. There may also be institutional impediments to using management decisions as experiments or in changing management decisions too often.

Special considerations arise when outcomes are irreversible (e.g., species extinctions), or reversible only at great cost (e.g., ecosystem restoration), and when there is uncertainty, particularly about how the future generations might value benefits. In such cases there is value to preserving flexibility and avoiding irreversible or difficult to reverse decisions until uncertainty is resolved. The value of avoiding irreversible outcomes is called option value, or in some literature quasi-option value (Arrow and Fisher 1974, Henry 1974). The importance of avoiding irreversible outcomes (or accounts that can be reversed only at some cost) when the passage of time reduces uncertainty can be illustrated with a simple example based on Arrow and Fisher (1974). They considered a two-period model in which there is a choice between developing or preserving land. If land is preserved in the first period, then there will again be a choice between developing or preserving land in the second period. Development, however, is assumed to be irreversible so that if land is developed in the first period there is no choice in the second period. Suppose that the benefits of development are 100 per period. Suppose that the current benefits of preservation are 90, while the second period benefits of preservation are uncertain with a 50% chance of being 160 and a 50% chance of being 20. For simplicity assume there is no discounting of future benefits. If one were making a one-time decision with a goal of maximizing expected present net benefits, then the choice would be to develop in the first period rather than preserve. Developing in the first period would give expected benefits of 100 + 100 = 200, versus expected benefits of preservation of 90 + (0.5x160 + 0.5x20) = 180. However, choosing to preserve in the first period allows the flexibility to making a choice over whether to preserve or develop after it is learned whether preservation is highly valued or not. If preservation value turns out to be high in the second period then the land can continue to be preserved. If preservation value turns out to be low in the second period then the land can be developed. In this case, the expected net benefits by choosing to preserve in the first period are 90 + (0.5x160 + 0.5x100) = 230. Taking account of the value of preserving options means that it is optimal to make the choice to preserve in the first period. This type of theory has been well developed in financial theory (see for example Dixit and Pindyck 1994).

Other notions of how to manage complex ecosystems over time focus on system properties such as stability or resilience of the system rather than attempting to optimize present value of expected utility derived from the system. Stability may be desirable because it is costly to adjust to variable flows of ecosystem services. Lack of stability might also cause fundamental shifts in ecosystem state to less desirable conditions (Carpenter et al. 1999). This latter notion is related to system resilience. Resilience can be defined as a measure of the ability of a self-organized system to absorb shocks and disturbances and remain in a desirable state. Management actions should be designed to increase system resilience, both biophysical and social, and build capacity for learning and adaptation. A second definition of resilience is the speed with which a system returns toward an equilibrium state.

2.3 Multiple Metrics of Value

2.3.1 Issues associated with analyzing multi-dimensional values using a single metric

It is frequently the case at EPA that multiple dimensions of value are reported to analysts and decision makers and emphasis is placed on monetized values as a single metric of analysis. For example, the ecological value associated with species diversity may be reported alongside income generated from the provisioning services of ecosystems (i.e., the products obtained from ecosystems such as food, fiber, biochemicals, genetic resources and fresh water, which are often are traded in the open marketplace (Assessment 2005)). Indeed, the agency's Environmental and Economic Benefits Analysis conducted in support of new regulations aimed at Concentrated Animal Feeding Operations (CAFOs) alluded to a diversity of potential "use" and "non-use" values worthy of consideration under the rule. These were associated with ecological systems and services and included commercial fisheries, navigation, recreation, non-contact recreation (e.g., camping), wildlife viewing, the provision of drinking water, irrigation, and a host of aesthetic and as yet unknown attributes (i.e., option values) (cite CAFO report). It goes without saying that analysts at EPA (as well as at other agencies) face significant challenges in integrating these varied inputs to create values that are expressed using a single metric.

However, if EPA desires—or is constrained by a requirement—to make or evaluate a given decision using a strict optimizing strategy—i.e., cases where one is required to maximize performance across a single, aggregated metric such as economic or ecological productivity—there is little choice but to simply translate as many of the inputs as possible into a single metric and, if necessary, isolate the others (e.g., as in the case of "+B", which was utilized during the CAFO analysis). These translations into monetary equivalents may take place during an elicitation itself (e.g., by asking for an individual's willingness to pay for improving recreation access, an objective that has both monetizable and non-monetizable attributes) or after the fact (e.g., by inferring travel costs from visitation patterns to a given recreation area).

There are many examples, at EPA and elsewhere, where these types of translate-and-aggregate operations have been used. As part of the CAFO analysis, for example, EPA computed the monetized benefits of the proposed rule by combining the results from surveys that elicited values (via a contingent valuation approach) associated with improvements in the context of recreation (e.g., boating, swimming and fishing) as well as water quality. Also included among the valued benefits during the CAFO analysis were those obtained using a benefits transfer approach. Among these were a national survey from 1983 that determined public willingness to pay for changes in surface water quality on water-based recreational activities, a series of verbal CV surveys from 1992, 1995, and 1997 of public willingness to pay for reduced contamination of drinking water supplies, and several studies—e.g., from 1988 and 1995—of recreational fishers' values for improved angling success related to a reduction in nitrate pollution levels in a North Carolina estuary (U.S. Environmental Protection Agency 2002).

It is worth noting that these single-metric approaches for use during in an optimization model for decision making or evaluation needn't focus strictly on values expressed in monetary terms. One could imagine cases where the value of a given suite of benefits is expressed using ecological units such as those for productivity (e.g., g•C•m-2). For example, improvements in water quality in a given estuary associated with a new regulation could be expressed in terms of an aggregate measure of pre-harvest primary and secondary productivity.

Although analysts can (and in many cases, do) integrate multiple, diverse value inputs to create single-metric outputs, several issues associated with this operation must be raised. Chief among these is the degree to which the aggregation of multiple value inputs can be undertaken with a requisite degree of validity and defensibility. For instance, an analyst can—with relative ease and high degrees of validity and defensibility—calculate (via fieldwork and modeling) the benefits of a particular decision in terms of the expected or actual net increase in productivity for all plant and animal species in a given grassland (e.g., in g•C•m-2).

Other cases where aggregation is desired may prove more difficult. As noted above, for example, there are significant challenges facing an analyst asked to aggregate value inputs associated with species diversity and those that describe the income in dollar terms generated from the provisioning services of ecosystems. In the case of monetized values, for example, one can question the product of an operation that combines dollar values obtained from an established market (e.g., the market value of total catch obtained in a commercial fishery) with those obtained from a hypothetical one (e.g., anglers willingness to pay for an X% increase in catch rates). Similarly, one can reasonably ask about the degree of validity and defensibility with which monetized inputs from an established market be combined with those obtained via a benefits transfer. These concerns, are not unique to monetized inputs. The same concerns can be raised about combining values presented on Likert Scales obtained from two focus groups that utilized different facilitators. Any analyst is likely to face intense criticism for efforts from many outside (and likely some inside) observers (Arrow, Solow et al. 1993)) (also cite "Money" chapter provided by Paul Slovic?).

2.3.2 Multi-Attribute Analysis for Non-Optimizing Decision Rules

In many policy and evaluation contexts (at EPA and elsewhere) it is not necessary to utilize a strict, optimizing decision rule. As a result, it not necessary in these cases to identify and use a single metric that attempts to capture the benefits of an ecological system or its suite of services. Under these "non-optimizing" decision rules, there is an explicit recognition of the multi-attribute nature of the values that can be used to describe ecological systems services (e.g., values that can be expressed in envirocentric, moral, economic, aesthetic, and other terms).

In these cases, the attributes of an environmental system for which values are estimated may come from multiple sources. These may include the concerns of stakeholders (e.g., aesthetics, recreation, community stability); aspects of a system that are identified by technical experts (e.g., services such as pollination and denitrification); and economic or commercial interests (e.g., the value of resources in established markets). Indeed, this multi-stakeholder, multi-input point of view is consistent with arguments that the estimated value of an ecological system or service reflects judgments from a variety of different actors during many stages of the valuation process (e.g., the identification of the system or service to be valued, choices about methods for analysis, and—perhaps most importantly—the selection of the attributes (monetary and non-monetary) that will be used to characterize value (Keeney, Winterfeldt et al. 1990); (Keeney and Gregory 2005). To be comprehensive and defensible, in other words, estimates of value must go beyond the judgments of the expert community to also reflect a careful and comprehensive assessment of key concerns obtained from the wide range of interested and affected stakeholders.

Given this diverse group of people from whom value inputs can be sought, analysts and decision makers within EPA must be sensitive to a wide variety of potential objectives and concerns that can potentially shape management decisions and evaluation processes. A typical consultation process aimed at meaningfully integrating these views during decision making and evaluation involves five steps (Gregory 2000; Hammond, Keeney et al. 1999):

a. defining the evaluation context or the decision that needs to be made,
b. identifying what attributes of an ecological system or its services matter in the context of an impending decision; these attributes are drawn from stakeholders' stated objectives,
c. in the case of decision making, creating a set of options that address these objectives; for evaluation, identifying the standard against which a current state of affairs will be compared,
d. employing the best available information or predictions to characterize (via appropriate valuation processes) the attributes of the options (for decision making) or current state and the comparative standard (for evaluation purposes);

this includes characterizing the degree of uncertainty associated with each attribute, and

e. carrying out an in-depth evaluation of the options (for decision making) or the current state and the comparative standard (in the case of evaluation) by addressing the tradeoffs ultimately selecting one option over the other entails.

As noted above, the attributes for which valuation processes are undertaken are identified based on the objectives that are defined for a given decision problem (Arvai and Gregory 2003; Gregory 2000; Hammond, Keeney et al. 1999). To illustrate this point, take for example the case of a simple management decision where the objectives are to maximize the returns associated with a given species of fish while allowing for a requisite level of hydroelectric generation on a river. Here, analysts would be required to estimate the values associated with each management option under consideration by estimating both the monetary value of electricity generated and the number of fish that would be allowed to return.

It is worth noting that unlike single-metric expressions of value, which tend to be meaningful in the absence of an explicit comparison (i.e., it is relatively easy for an economist to understand if a given monetized result—e.g., the value of Pacific Salmon per kilogram in an established market—is high or low; likewise an experienced ecologist can with relative ease determine if a productivity estimate for a given species is high or low), multi-attribute expressions of value tend to require that contrasts be made across options for which values have been estimated. This is the case because, when multiple metrics are used simultaneously, it is likely that some attributes of an ecological system or service will show improvements relative to a reference point whereas others may indicate a worsening in conditions. For example, it is frequently the case that improvements in the productivity of commercially valuable species as a result of environmental protections come at a monetary and social expense. As a result, the value of a given option and its suite of attributes is determined by the tradeoffs that people are willing to make across objectives that oftentimes conflict (Hammond, Keeney et al. 1999); making these tradeoffs therefore, requires an explicit framework for comparison either among competing options or with some established standard (such as the status quo).

As with the case of optimizing decision rules, several issues must be raised for these multi-metric and tradeoff-based approaches. First among these is the question of how much information—i.e., data from valuations carried out for the defined attributes—needs to be collected. On the one hand, information ought to be gathered for each of the attributes that are being used to characterize a given objective. In this sense, the objectives that are specified for a decision or for a program being evaluated ought to guide choices about both what and how much valuation data is collected. However, it is worth noting that as the amount of data collected increases, so to does the complexity of tradeoffs that will need to be made during decision making and valuation. While computerized decision aids can make some of these tradeoff operations seem relatively easier (such as @Risk and Logical Decisions), significant cognitive burdens associated may persist. As a result, one must take into account practical considerations when considering a large set of attributes to be used for decision making or evaluation.

A second issue relates to the question of how much information ought to come from experts (e.g., ecologists, toxicologists, economists, etc.) as compared to lay stakeholders. Once again, the answer to this question is guided by the objectives that are specified for a decision or for a program being evaluated. For example, information that describes attributes linked to objectives that are technical in nature (e.g., improved air quality as defined by a reduction in particulate matter or sulfur dioxide) ought to come from the expert community (e.g., toxicologists and atmospheric chemists in this case). Likewise, economists will need to be consulted if monetized inputs are needed to describe the change in value of a resource traded in an established marker or a contingent one. The same is true for ecologists who may be asked to estimate, and then make tradeoffs between, productivity estimates calculated for denitrifying bacteria in a wetland.

On the other hand, if an agreed-upon objective is to maintain or improve the aesthetic quality associated with a nature area, then both information about the current state and possible improvements, and the tradeoffs people are willing to make when moving from, for example, the current state to some proposed level of improvement, ought to come primarily from those who utilize the resource for its aesthetic qualities (namely visitors to the nature area, the majority of which tend to be lay stakeholders).

In either of these cases, it is important that the attributes that are used to characterize objectives utilize measures that are both clear in terms of context and direction. This is a relatively easy task when considering attributes provided by the expert community. For example, there is very little ambiguity associated with the economic returns associated a particular species or the change in productivity associated with a given system of interest; likewise, it is generally understood in these cases that $100 is better than $1 and 100 g•C•m-2 is higher that 1 g•C•m-2. However, attributes for which data is collected from non-experts may be more problematic. For example, qualitative descriptions of attributes that define objectives related aesthetic quality or community stability may often lack meaning in formal analyses (i.e., to what extent is something that is “good” better than something that is “fair”?). Even if quantified measures are possible (as is the case with numerical scales), it can be exceedingly difficult to ascertain the qualities of a system that interacts to yield a given score. In the case of aesthetic quality, for example, to what extent do elements such as scenic vistas, the Colorado River, wildlife viewing opportunities, and the ratio of exposed to “green” landscape play a role in a stakeholder giving the Grand Canyon a score of, for illustrative purposes, 8 on a 10-point scale (where 1 is “poor” and 10 is “excellent”)? Similar questions can be raised for a host of other objectives and attributes that typically fall within the purview of non-experts (e.g., community stability, the quality of recreation opportunities, etc.) and each must be addressed when conducting valuations in these contexts.

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3 ECOLOGICAL SYSTEMS AND SERVICES

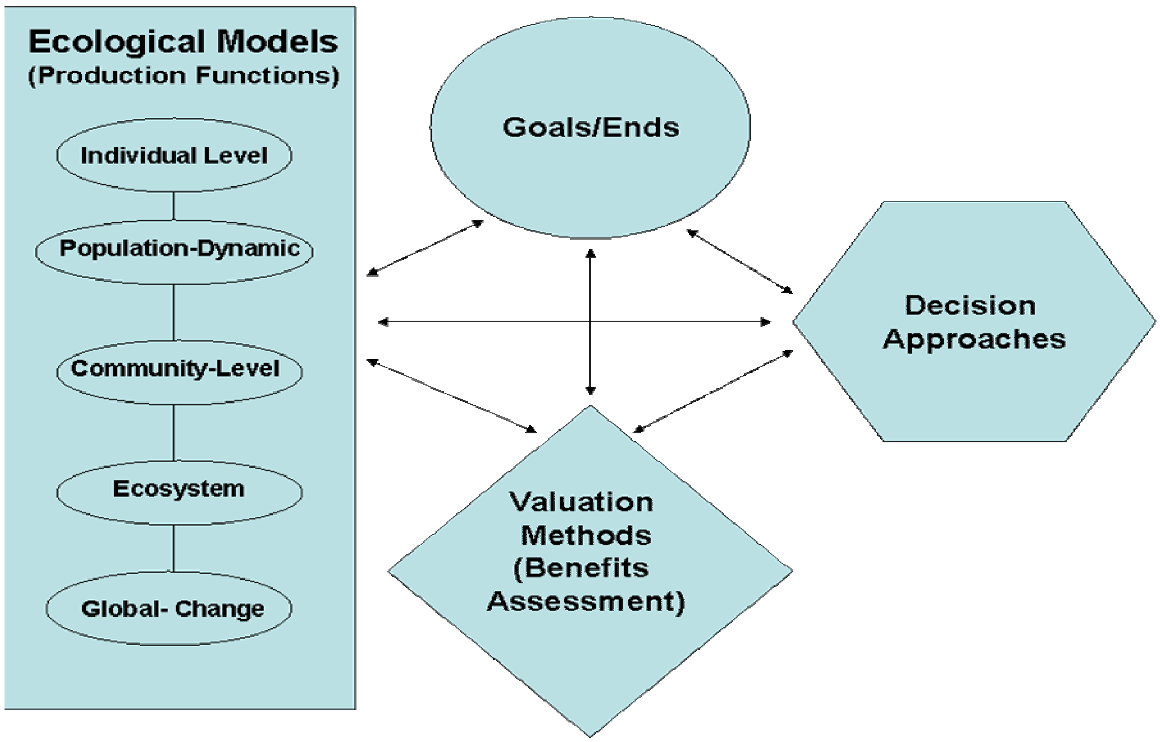

Figure 3-1: Conceptual Diagram of Major Components of Valuation: Levels of Ecological Organizational / Models

3.1 Ecological systems and services

To many it must seem bizarre to ask "what good is nature?" or in today's jargon, "what is the value to us of nature's goods and services?" Isn't the value of nature the value of life itself? The premise in a ``culture of life'' is that life itself is good. If so, nature, which we spring from and are part of, is as valuable as we ourselves are. Breaking nature into good and bad pieces seems sacrilegious, like breaking the Bible into chapters, valuing the services we get from each chapter, calculating what every verse yields for the weekly collection plate, and producing a Bible-lite with the most profitable verses. Still, that's where we are today, when moral precepts of environmentalism compete in the market with demand for sports-utility-vehicles. The following paragraphs sketch how to win for the environment in moral markets.

Living organisms supply goods and services at all levels of organization, from the individual to the ecosystem. Ecological science is organized according to scales. Ecology textbooks have chapters in the following sequence: ecology of individuals, ecology of populations, ecology of communities, ecology of ecosystems, and ecology of the biosphere. Rival textbooks begin small and end large, or begin large and end small. Either way, ecologists invariably view their subject as ordered along axes of scale, both in space and time.

3.1.1 Individual-Level Ecology

At the individual level, we might be interested in whether a parcel of land is habitable, much as we might be concerned with whether a cat, dog or parrot is comfortable---are they panting in heat or shivering in cold, is there food and water, or are they scrounging for crumbs and spills? The principles of physiological ecology and foraging ecology allow us to determine whether a parcel of land is habitable to the organisms living here, and why it is uninhabitable to those not there. Even more, these subjects in ecology allow one to compute the number of hours per day, for each day of the year, that the parcel is habitable, allowing us to determine the number of “animal-hours” for which the parcel is suitable for wildlife.

In this approach, an animal’s upper and lower tolerable body temperatures are measured empirically in the laboratory. Then in the field the fluxes of heat are measured---how many days are sunny, what the daytime air temperature is, and whether the wind blows hard or softly. From these data one computes the time during the day that the animal can exist within its upper and lower temperature limits. This “envelope” of tolerable conditions is called the “climate space”. If one is interested in maintaining a city park with birds, say for school children to see, then one can compute the interval of the day that the park is habitable for birds. If this is short, then the birds won’t be there very often, or may not bother to come at all. If a road is built nearby, leading to dryer soil and more reflected light and heat, the habitat becomes less livable for birds. Or if streams and water sources are provided, more birds will come.

Foraging too is well understood. Animals continually make decisions about what food to commit to eating and what to ignore. Animals usually sort into two types of foragers---sit and wait predators (also called central-place foragers) or searchers. The former, like a zoological Buddha, sit at a spot and wait for food to come to them. The latter, like a Buddhist monk, go from place to place looking for food. Either way, consuming food involves expending energy and time. The anticipated return for consuming any particular item must be better than bypassing it for the animal to decide to accept the item. By taking data on an animals’ energy expenditures while pursuing prey and waiting around for prey, together with data on how often and where prey appear throughout the foraging period (a period necessarily within the climate space), one can compute how many items of food an animal will actually consume. From this in turn, the health and reproductive potential of the animals can be calculated. From these ecological models, one can compute the implications for habitat suitability of policy recommendations that impact the parcel of land where animals are presently located.

The “service” being produced in this situation is simply having wild animals and plants nearby. One could potentially determine the economic value of each animal-livability-hour for visitors to the park. In this way, the usefulness of the park, as a park, can be ecologically engineered according to economic criteria.

3.1.2 Population Ecology

The ecological modeling best known to economists is population dynamics, equations that describe how a population changes in abundance through time because of the birth and death rates in it. These models have been used in economics texts for decades to describe how to harvest a population optimally.

A simple model in ecology that traces back to the early 1900's is called the logistic equation. It describes "sigmoid growth" as follows: when some animals are introduced into empty habitat, the population starts growing slowly, then picks up speed until resource limitation begins taking a big bite out of the potential reproductive rate. The population's growth then decelerates, and levels off at a point where resources are so limiting that each birth statistically matches each death, resulting in a constant population.

If one is harvesting a population, then one strategy is to thin the population to a point where its growth rate is fastest---a point with lots of animals to produce births and yet not too many animals to make resource limitation into a big factor. This point is called the point of maximum sustainable yield, and has often been used in fisheries management as the goal when setting harvest policies and quotas. Economists have modified the maximum-sustainable yield goal. They advocate a maximum-sustainable profit goal that also takes into account the commercial interest rate in the economy where the fishing industry is located. This modification leads typically to recommending maintaining fewer organisms in the population to produce the births, and accepting a somewhat lower yield as a result, because harvesting some extra animals and banking the earnings from selling them earns more interest than what has been sacrificed in settling for a somewhat lower harvest. Still, the goals of maximum-sustainable yield and maximum-sustainable profit are quite similar, and in my opinion, practically indistinguishable. Both may be ill conceived.

Both maximum sustainable yield and maximum sustainable profit fail to take into account the ability of the population to recover from environmental fluctuations and management errors. The capability to recover is called the population's "stability in response to perturbation". At either the traditional-fisheries goal or the modern-economists goal, the population is left without any stability and quickly falls to extinction even though it's being managed under the illusion that the harvesting policies are protecting against this eventuality.

All the managed fisheries around the world are suffering from depleted stocks. Fishery biologists and economists claim this represents political disdain for good management practice. Yet, passing the buck to politicians and greedy or ignorant fisherman is not accurate because the stocks have also collapsed in locales where the fishing industry has been conscientious and educated, such as Newfoundland. The problem lies with the theory that fisheries biologists and economists have been using, theory that does not value the population's stability at the harvested equilibrium.

Harvesting theory can be corrected to include the population's stability in designing an optimal harvesting policy. This corrected theory leads to the recommendation that more animals be maintained in the population than the traditional-fishery goal, not less as economists recommend, and that on the sigmoid growth curve, the number of animals should be set about halfway between the point where the population growth is fastest in an ideal word and the point where the population size levels off. This is a point where there is some resource limitation to stabilize the stock but not too much to choke off all net reproduction. At this point there are also lots of animals so that the stock can recover when some are lost through environmental catastrophe or management error.

The hot research topic in population dynamics over the last decade has been in spatially distributed populations, called “meta-populations”, i.e., a population of populations. For most of the last 70 years or so, population dynamics research has focused on populations located at one spot. Many, perhaps most, populations actually occur in pockets, with both a “local” dynamic and “regional” dynamic. The traditional models of population dynamics are assumed to apply at the local level, with the addition of “coupling” to adjacent local populations by migration back and forth. The whole species is thus a collection of coupled local populations---a meta-population. Managing a species therefore involves understanding the dynamics on this large scale, not only what happens in one’s backyard. Models are increasingly available for purpose.

3.1.3 Community Ecology

An ecological “community” is a set of interacting populations. An ecological assemblage is all the populations living in the same area, whether or not they interact. The main property of a community is its species diversity, how many species are in it. People like variety, it’s interesting and pretty, the more the merrier, and the better the chance that one of the species is especially valuable now or the future. The major issue in community ecology is to understand what sets the number of species in the community to understand how to conserve them, to prevent their going extinct, and possibly how to restore degraded sites to some semblance of their earlier state.

The problem with diversity is that the more there is, the harder it is to keep around. Plants and animals are always getting in one another way, they’re always tripping over each other, eating the food off each other’s plate, even eating each other. These processes make it hard to produce and maintain diversity. Moreover, what goes on in communities is imperfectly understood. The nature red-in-tooth-and-claw part is relatively well known: competition, predation, disease, and so forth, because ecologists like economists have focused on conflict, not love. But animals and plant species do help each other too, at least as much as they hurt each other, and this entire side of community ecology, the cooperative side, is very poorly understood relative to the competitive/predator-prey side.

Species diversity to some extent reflects species doing different things, staying out of each other’s way as much as possible. Models explaining diversity in these terms are called niche-theory models, and a tradition of equations about this approach traces to Robert MacArthur. Each species is assumed to have a “niche” unto itself and there are as many species as niches according to this approach. A niche, by the way, can be thought of as a species’ occupation, and a habitat as its address. So the economy of nature in the niche-theoretic approach boils down to an economy with many different industries, one industry per species. The problem is that there aren’t enough niches to account for all the species. A spot with fifty species of birds doesn’t have fifty different bird niches, and a spot with 300 plants doesn’t have 300 different plant niches.

So the alternative approach, identified with Steve Hubbell over the years, has been to assume that all the species are the same, no one has an advantage over any others, and so the collection of species lumbers along with the species diversity that results when the speciation rate (rate at which new species form) balances the extinction rate. This is the "neutral-theory" approach to species diversity. Well, all species are not the same of course, but perhaps they are within a category, say all trees are more or less the same, all vines the same, all annual grasses the same. A naturalist can always find species differences, some flower sooner than others, some have blue flowers and others red, and so forth, but according to the neutral-theory approach, these differences while real and perhaps fascinating, are nonetheless irrelevant to accounting for the forest's species diversity. The economy of nature in the neutral theory approach boils down to a few big industries with lots of interchangeable firms (species) within each industry.

Now what are the services that an ecological community provides, above and beyond the services supplied by each species considered in isolation? Well, here ecologists differ. To some, the provision of diversity itself is a service. These ecologists value diversity. To them, the fact that two species do successfully coexist in nature, say a bee with a flower, is an element of interest, significance, and worth itself. We know that an ecological community is hard to assemble. Throw a bunch of species in an aquarium, and all that remains after a week are dead fish and the stench of sulfur. So, when species are demonstrated to coexist, that fact is nontrivial and valuable. If one was stocking an aquarium for a child, one would pay more for two fish that coexist with each other, than the sum of what one would pay for each separately knowing that one would have to place the fish in different aquaria.

The other approach by ecologists to valuing the diversity of an ecological community has been to refer the community's diversity to ecosystem processes. But first, what is an ecosystem?

3.1.4 Ecosystem Ecology

An ecosystem is the union of biological populations with their surrounding physical environment. Thus, a stream ecosystem is the fish, plus the water, plus the dissolved organic carbon in the water, plus the air above the water and so forth. Because an ecosystem has both living and nonliving parts, the influence of each on the other is the main topic of interest. Biological populations trap energy from sunlight and take CO_2 from the air to produce carbohydrates in leaves. These in turn are eaten by caterpillars, or dropped to the ground and decomposed by fungi in the soil. The combustion of the CO_2 originally trapped by the plants releases the CO_2 back to the atmosphere, and if the release rate exceeds the trapping rate, then the overall level of CO_2 in the air increases through time---this is happening now, resulting in global warming with its melting of glaciers, sea level rise, and so forth.

Many models now exist for how ecosystems carry out their activities, models for forests, deserts, tropical islands, tundra, streams, lakes, oceans, and so forth. These models generally do not resolve their biological part into species, but instead deal with highly aggregated state variables, such as all the plants combined, all the animals combined, or major categories or plants or animals (e.g., grasses, evergreen trees, deciduous trees). Ecosystem models typically simulate the flux of elements or energy among different components of an ecosystem, both living and non-living. For example, a watershed model would track the inputs (in precipitation and deposition), outputs (in stream water or transpiration), and internal exchanges (e.g., uptake and release by plants) of water as well as key nutrients such as nitrogen or phosphorus within a spatially defined watershed. A terrestrial carbon cycle model would track the flux of carbon into an ecosystem by the process of photosynthesis, the cycling of carbon between components in the ecosystem, and the release by respiration or leaching. Most terrestrial ecosystem models only simulate key plant and microbial processes, but there is increasing effort to also incorporate the effects of important animal groups such as grazers. In contrast, aquatic ecosystem models have traditionally included both plants and animals.

So a major research issue in the last decade has been to see whether the species composition of the “plant pool” or “animal pool” matters, or whether the ecosystem activities are more or less invariant to the species composition within its major components. By the neutral-theory approach to community ecology, ecosystem activities are indifferent to interchanging plant species provided the major categories of plants remain, and by the niche-theory approach, ecosystem activities suffer with the loss of each and every species.

Now if ecosystem processes are of particular value, say the decontamination that natural soils are supposed to provide to polluted waters, then this valuable ecosystem function may impart value on the species within it. According to the niche-theory view of community structure, each and every species contributes to the ecosystem’s function, and so species diversity becomes valuable not only in its own right, but because of its implications for ecosystem function. According to the neutral-theory view of community structure, diversity of the major categories of species is valuable, but the particular species. So, if the neutral-theory view of community structure is correct, little value can be place on species diversity because of ecosystem function. In the neutral-theory view, any value for diversity has to stand on its own, and cannot be derived from ecosystem functions.

Empirically, data support both the niche-theory and neutral-theory views. Animals, especially vertebrates, definitely accord with the niche-theory view, according to a long line of workers tracing to MacArthur, and including Martin Cody and Jared Diamond with birds, Joan Roughgarden's work with lizards and Jim Brown’s with desert rodents. Plants in forests apparently accord with the neutral-theory view according to Steve Hubbell and his many collaborators. However, grasses and forbs in prairies accord with the niche-theory view according to Dave Tilman.

3.1.5 Global Change Ecology

Finally, the largest scale for which ecological models have been developed pertains to the entire biosphere. These models are surprisingly simple in spite of their grand scope. Older biosphere models simulated the entire biosphere as though it were a giant plant with a giant canopy rooted in a gigantic pot filled with giant amounts of soil. This plant photosynthesizes and respires---its giant metabolism waxes and wanes with the seasons. Its open stomates capture CO_2 for photosynthesis, and release some of it back from respiration. The fungi in the soil release still more CO_2, joined by the Herculean amount released from human fossil fuel combustion. The giant plant is losing the battle. It is taking in CO_2 more slowly that CO_2 is being released by the soil and fossil fuel combustion, resulting in the retreat of mighty glaciers, the flooding of low-lying countries, and the movement of species northward---all signatures that Earth is heating up.

In the past decade, far more sophisticated biosphere models have been developed, called dynamic global vegetation models (or DGVMs), that simulate the spatial and temporal dynamics of multiple "functional groups" of plants, as well as critical disturbance processes such as fire. DGVMs integrate processes of physiology and biogeochemistry, biophysics, plant geography, and vegetation dynamics, with the goal of simulating the response of the biosphere to global change. More recent versions include spatial representation of human activities (e.g., agriculture and forestry).

3.2 Ecological Services

Ecosystems consist of the biotic and abiotic realms interacting through the biotic processes of production and decomposition, which involve energy capture and transfer, water movement from the soil to the atmosphere mediated by vegetative cover, and the uptake and subsequent release of minerals from the soil through organisms and back to the soil. An ecosystem can be structured around a very few organisms in severe environments, such as Antarctica, or they can be based on the extraordinarily complex and rich biota of a tropical forest. All ecosystems, whether they are human managed or natural, or simple or complex, have these traits. They fix carbon from the atmosphere and build it into biomass and in the process exchange water and nutrients. Evolution is shaped by the diverse strategies utilized by organisms to get their share of energy, nutrients and water (resources) in a given environment.

The result that derives from these processes is a vast array of organisms with the capacities to acquire, store and protect or share these resources. Thus, a given piece of landscape will have an ensemble of organisms that are processing the natural resources available to them. As a result, through time, soil is formed by the results of mining of minerals by the roots of plants and the subsequent loss and decomposition of organic matter through the activities of microbes. The local climate is modified by the structure of the organisms themselves through interception of radiation and alteration of air flow patterns and by the movement of water from the deeper layers of soil out into the atmosphere.

Ecosystems, including its composition, structure and processes, have been viewed in an anthropogenic manner in relation to the benefits people derive from them which are termed ecosystem services. The Millennium Ecosystem Assessment (2003) classified these services into four categories. *Supporting Services*, those that result directly from ecosystem functioning—such as soil formation, nutrient cycling and primary production. All other ecosystem services derive from supporting services. *Provisioning Services*, the products obtained directly from ecosystems such food, fresh water, fuel wood, fiber, and biochemicals. These services are easily valued economically since they are part of the local, and even global, marketplace. *Regulating Services.* These are the benefits that derive from the regulation of ecosystem processes and include a host of benefits including climate regulation, water flow and purification regulation, erosion control, regulation of human diseases, biological control of pests and pollination services. These types of services can be valued economically, but they have not been done so to a great degree. They are often viewed as "free" services. Finally, *Cultural Services,* are the great number of nonmaterial benefits that people derive from ecosystems. Examples are spiritual and inspirational values, sense of place and cultural heritage values, and educational, recreational and ecotourism values. There have been a diversity of approaches to valuing these services including economic valuation for recreation and tourism as an example, however most of these are valued by non-economic approaches. Then, of course, there is life itself, in all of its variety that is the foundation of ecosystem functioning and which has intrinsic value.

3.3 Scarcity of Ecological Services

The biotic systems, and most physical processes of the earth, are driven by energy from the sun. The amount of energy received from the sun from year to year varies to such a relatively small degree that it is called the solar constant. However, the amount of solar radiation received on the earth's surface varies with latitude and season. This variability drives the striking global patterns of both physical and biological processes that affect our lives. Although the atmosphere is well mixed, driven by solar variability, most components of terrestrial ecosystems are localized with the exception of some highly mobile components such as migrating animals. Thus the natural resources locally available to humans vary greatly with geography and historically local resources have strongly controlled people's livelihoods. However, with globalization trends of increasing magnitude over the recent centuries, these natural resources have become globally available to those with the resources to acquire them. Natural systems, or ecosystems, provide more to society though than tradable goods. These can be viewed in an anthropogenic context as ecosystem services (see section 3.2 above). In many ways humans are still heavily dependent on local ecosystem services and in many areas of the globe, totally so, due to lack of capital to purchase globally traded goods or engineered substitutes, if they exist.

As humans have modified the face of the earth in many places they have depleted many local ecosystem services. They have converted forests to grazing lands or farm land. These conversions have amplified certain services locally available to people, such as food, but have resulted in carbon loss due to vegetation conversion, loss of soil and clean water, and in hilly terrain loss of localized flood control due to the reduction in tree cover. Human modification of landscapes has resulted in the diminution of many more ecosystem services than have been augmented. It for this reason that a move toward intensification of agriculture, that saves land, however with vast improvements in current agricultural practices, particularly nitrogen and water management, is an important trend.

Ecosystems are being impacted by other global changes. Historically, land use change has been the biggest driver of the changes in ecosystems, this now accompanied by increasing numbers of invasive species in modified landscapes. These impacts have been local. Since the industrial revolution other changes have occurred, principally modification of the atmosphere of the earth through energy consumption as well as impacts of the production of nitrate through industrial processes. These latter changes have operated globally, or regionally, since they are tied to atmospheric movement. These atmospheric changes have in turned modified the earth's energy budget, and along with changes in the land surface, have resulted in human-induced climate change with potentially major consequences on human well-being.

According to the results of the Millennium Ecosystem Assessment, globally we are seeing a reduction in such ecosystem services as water quality, erosion and flood control, loss of genetic resources, in addition to the loss of many cultural services derived from ecosystems. These losses, which are predicted to intensify without major changes in current policies, are impacting human well-being in rich as well poor nations as witnessed by the recent impacts of tsunamis and hurricanes which were augmented by the reduction of protective coastal ecosystems. At the same time those human populations which are inhabit arid regions that are at most risk of ecosystem degradation are those most directly dependent on natural ecosystems for their livelihood.

3.4 Ecological Risk Assessment at EPA--Its Relationship to Ecological Systems and Services

In the 1990's, EPA developed a structured, risk-based approach for assessing ecological information needed fort risk management decisions involving ecological systems and services (U.S. Environmental Protection Agency Risk Assessment Forum 1998; U.S. Environmental Protection Agency Risk Assessment Forum 1992). The Agency defined ecological risk assessment as "a process for evaluating the likelihood that adverse ecological effects may occur or are occurring as a result of exposure to one or more stressors" (U.S. Environmental Protection Agency Risk Assessment Forum 2003). Since the Agency introduced ecological risk assessment, such analysis has become the standard approach for evaluating chemical, biological, and physical stressors in the Agency

In 2003, EPA published *Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment*, which described the choice of ecological assessment endpoints as a "critical early step in conducting an ecological risk assessment…deciding which aspects of the environment will be selected for evaluation." The document reviews the components and attributes of ecological systems that are the legal focus of EPA's mission and that have been the focus of EPA decision-making and proposes that EPA build on these precedents as "generic ecological assessments endpoints" to consider when beginning an ecological risk assessment to support a particular risk management decision. EPA structured its analysis of generic endpoints in terms of ecological scale. Figure 3-2 Generic Ecological Assessment Endpoints as Described in U.S. Environmental Protection Agency (2003) excerpted directly from U.S. Environmental Protection Agency Risk Assessment Forum's 2003 document, highlights the levels of ecological systems and the attributes of those systems that EPA is mandated to address or where EPA has established policy and precedent.

Entity	Attribute	Identified EPA Precedents
Organism-level endpoints		
Organisms (In an assessment population or community	Kills (mass mortality, conspicuous mortality)	Vertebrates
	Gross anomalies	Vertebrates Shellfish Plants
	Survival, fecundity, growth	**Endangered species** **Migratory birds** **Marine mammals** **Bald and golden eagles** Vertebrates Invertebrates Plants
Population-level endpoints		
Assessment population	Extirpation	Vertebrates
	Abundance	Vertebrates Shellfish
	Production	Vertebrates (game/resource species) Plants (harvested species)
Community and ecosystem-level endpoints		
Assessment communities, assemblages, and ecosystems	Taxa richness	Aquatic communities **Coral reefs**
	Abundance	Aquatic communities
	Production	Plant assemblages
	Area	**Wetlands** **Coral reefs** Endangered/rare ecosystems
	Function	**Wetlands**
	Physical Structure	Aquatic ecosystems
Officially designated endpoints		
Critical Habitat for threatened or endangered species	Area Quality	
Special Places	Ecological properties that relate to the special or legally protected status	e.g. **National parks, national wildlife refuges, Great Lakes**

Figure 3-2 Generic Ecological Assessment Endpoints as Described in U.S. Environmental Protection Agency (2003) [1]

The document states three criteria for generic ecological assessment endpoints. Endpoints should be generally useful in EPA's decision-making processes; practical; and well-

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[1] **Generic ecological assessment endpoints for which EPA has identifies existing policies and precedents, in particular the specific entities listed in the third column.** Bold **indicates protection by federal statute. See Figure 3-1: Conceptual Diagram of Major Components of Valuation:**

Levels of Ecological Organizational / Models for additional endpoints that could be considered by EPA in the future.

defined so that they would be understandable by the public and decision makers without appearing ambiguous to environmental scientists. The document also details the "policy support" and EPA's view of the "practicality" (e.g., whether methods were available and reasonably practicable to estimate risks to the endpoint entity and attribute in various assessment contexts) of assessing risks by a particular endpoint and attribute. Many of the practicality considerations (See Table 2-2 in EPA's 2003 Document) reflect the current limitations of data available to the Agency for decision making.

EPA also presents a proposed short list of "potential" future generic ecological assessment endpoints (see Figure 3-3 Potential Generic Ecological Assessment Endpoints as Described in U.S. Environmental Protection Agency (2003), which excerpts Table 4-1 from EPA's 2003 document) and a recommendation that EPA develop and support a continual, open process for reviewing, amending and creating new generic endpoints.

Entity	Attribute
Organism-level endpoints	
Organisms (in an assessment population or community)	Physiological status (in addition to growth) Disease or debilitation (in additions to gross anomalies) Avoidance behavior Courtship behavior (e.g., birds) Migratory behavior (e.g., birds and salmonids) Nurturing and rearing behavior (e.g. nest abandonment)
Population-level endpoints	
Assessment population	Genetic diversity
Community and ecosystem-level endpoints	
Assessment communities, assemblages, and ecosystems	Trophic structure Energy flow Nutrient cycling (ecosystems in addition to wetlands) Nutrient retention Decomposition rates Sediment and material transport Area or function of estuaries and riparian ecosystems Resilience Vertical structure of plant communities Attributes that influence public health
Landscape-level endpoints	
Assessment landscapes (of multiple populations, communities, assemblages, and ecosystems)	Spatial pattern (random, clustered, or uniform; dominance; contagion; contiguity or fragmentation; juxtaposition)

Figure 3-3 Potential Generic Ecological Assessment Endpoints as Described in U.S. Environmental Protection Agency (2003)

In proposing formal recognition of a set of generic ecological assessment endpoints (or a process for developing future ones) along the lines described, the Agency did not factor the notion of "ecological services" or "value" into the choice. Indeed, the concept of "value" appears only in Appendix B of the document, which defines an assessment endpoint as "an explicit expression of the *environmental value* that is to be protected, operationally defined by an ecological entity and its attributes." The document does not discuss how the concept of "value" might enter as a factor in choosing generic endpoints or how the Agency might use a conceptual framework, such as categories used for identifying services in the Millennium Assessment) in linking ecological services to endpoints.

The Committee advises the Agency to consider the importance of how changes in a particular attribute of an ecological entity might affect ecological services and other important values. This consideration may be as important a consideration in choosing an ecological endpoint as policy, precedent support, and practicality. Factoring concerns about values and ecological services into ecological risk assessment will help focus risk assessments on endpoints relevant to decision making; will provide the inputs needed for ecological valuation; and work to enhance the management and communication of ecological risks.

4 APPROACHES AND METHODS FOR VALUATION

4.1 Proposal for revising Chapter 4

Note to Committee: *Rather than just presenting a "laundry list" of possible methods that can contribute to valuation, it seems that it would be better to have a structure into which these methods would fit, suggesting how each might contribute to an overall valuation process. What follows is a proposed structure. It envisions a valuation process with a number of components, as illustrated in Figure 4-1 Proposed Process for Valuation. (Note that the components in the figure are limited to the valuation process, i.e., they do not include all of the components that would be involved in the full decision making process.)*

We would then ask how the various methods contribute by providing information of various types for use in the different components of the valuation process.

Please find below, provided below for discussion:

- *A list of Possible Components of the valuation process*
- *A Figure showing how these methods fit into the valuation process, and*
- *A preliminary suggested mapping of methods listed in the current draft (Figure 4-2* Preliminary Suggested Mapping of Methods listed in this Current Draft to Possible Components of Valuation Process

Possible Components of Valuation Process

1A. Identify potentially important ecological changes.

1B. Identify what ecological changes people care about (e.g., because they lead to changes in ecological services, or because of their intrinsic importance to people)

2. Combine results of 1A and 1B to identify the ecological changes to focus on.

3. Characterize and, if possible, quantify the ecological changes of interest (if possible)

4. Characterize and, if possible, quantify the human consequences of these ecological changes (e.g., # of people affected).

5A. Characterize and /or quantify the value or importance of these changes in non-monetary terms.

5B. Characterize and/or quantify the value or importance of these changes in monetary terms.

6. Communicate information about values to decision makers and the public for use in policy decisions (current or future).

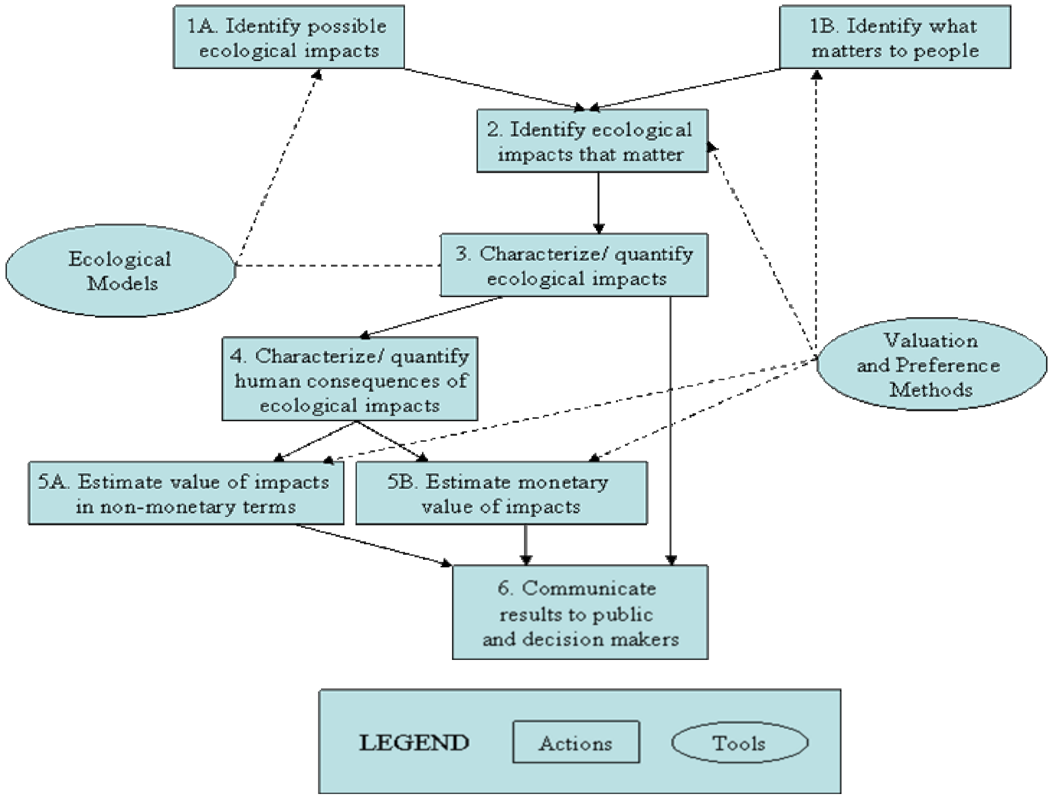

Figure 4-1 Proposed Process for Valuation.

	Components	***Methods***
1A.	*Identify potentially important ecological changes*	*Non-existent models discussion in Ch 3* *Non-existing ecological production function* *Energy and Material Flow (5.6.1)* *Biodiversity and Conservation Value (5.6.2)*
1B	*Identify what ecological changes people care about, i.e., what matters to people*	*Mental Models (5.2)* *Social Psychological Approaches (5.4)* *Mediated Modeling (5.5.2)* *Deliberative Value Elicitation (5.5.3)* *Citizen Jury (5.5.4)*
2.	*Identify ecological changes to focus on*	
3	*Characterize and, if possible, quantify ecological changes of interest*	*Non-existent models discussion in Ch 3* *Non-existing ecological production function* *Energy and Material Flow (5.6.1)* *Biodiversity and Conservation Value (5.6.2)*
4	*Characterize and, if possible, quantify the human consequences of these ecological changes*	*Environmental Benefit Indicators*
5A	*Characterize and, if possible, quantify the value or importance of these changes in non-monetary terms*	
5B	*Characterize and, if possible, quantify the value or importance of these changes in monetary terms*	*Economic Methods (5.3)* *Civil Court Jury Awards (5.5.5)* *Referenda (5.5.1)* *HEA (5.6.5)*
6	*Communicate information about values to decision makers and public for use in policy decisions*	*Deliberative Approaches (6.1.3)* *NEBA (6.3)*

Figure 4-2 Preliminary Suggested Mapping of Methods listed in this Current Draft to Possible Components of Valuation Process

4.2 Introduction to major approaches for valuation

The decision contexts and the nature of the actions/regulations under consideration have important implications for the types of benefits assessments that will be useful to decision makers. In this report we are concerned with the very substantial portion of Agency actions that have the potential of affecting ecosystems and the services that they provide. In these contexts it is important that the potential benefits and costs of contemplated changes in ecosystems and their services are identified and systematically assessed. Assessments that are useful and relevant to Agency policy and decision making in these contexts may be accomplished in a number of ways. Economic approaches offer one set of methods for estimating the value of changes in ecosystems/services. Monetary measures of the benefits of ecosystems and ecosystem services are necessary whenever formal Benefit Cost Analysis is specifically required. Because the vast majority of ecosystems and their services are not exchanged in markets, monetary valuations must be derived indirectly from peoples' expenditures for other marketed goods and services or from statements about their willingness-to-pay (or willingness-to-accept compensation) in presented hypothetical markets. While there are continuing arguments about the sufficiency 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 assessment methods. Thus, all regulations and guides relevant to ecosystems/service benefits assessments explicitly acknowledge the need for and encourage the development and application of additional assessment approaches and methods.

An important goal of this report is to improve the assessment of the benefits of ecosystems and their services to provide better support for Agency decision making. A key component of this effort is to improve and extend economic methods so that they achieve more useful assessments over a broader range of ecosystems and services. Of equal importance is the identification of other systematic assessment approaches that can be applied along with or in place of economic assessments to more fully represent the benefits of ecosystems and services. For example, this report will describe a number of social/psychological methods that 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. These 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 to be discussed include assessments based on voting and other group expressions of social/civic values. The report will also consider an ecological approach to benefits assessment, an approach which is potentially less directly dependent on human preferences and value judgments. The ecological assessment approach begins with the assumption that the healthy functioning and sustainability of ecosystems is fundamentally important to the well-being of human societies, and all living things. This assumption is widely accepted among the citizens of the US and it receives substantial support and detailed explication from relevant environmental and ecological science. By this ecological approach the benefits of any change in ecosystems is assessed in terms of the calculated effects on overall ecosystems health and sustainability, often represented by projected changes in indicators such as species biodiversity, bio-mass production, carbon sequestration or energy use/redistribution. This approach can provide a precise and reliable relative measure of ecological health benefits across a relevant range of potential alternative policies.

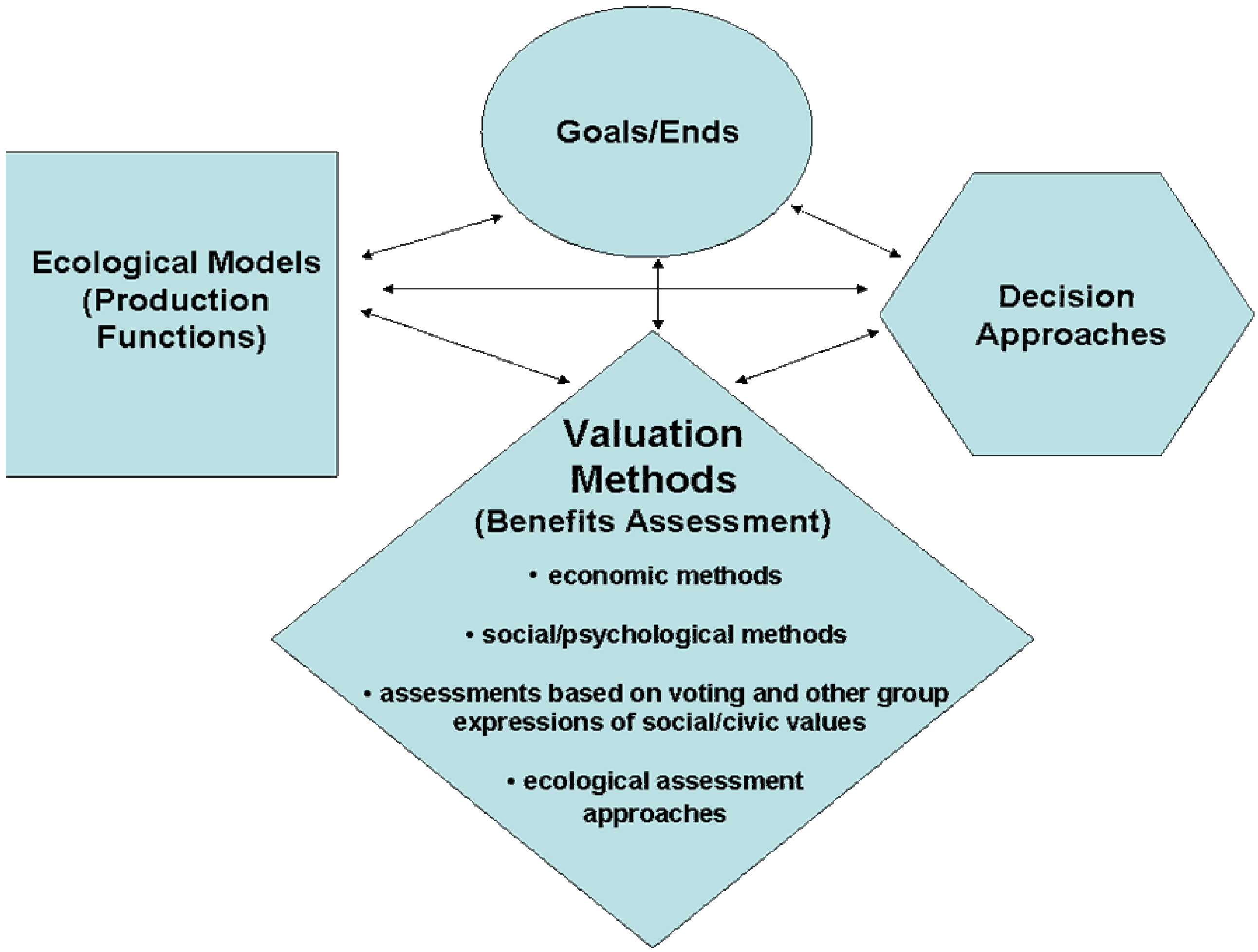

Figure 4-3: General Schematic of Agency Policy/ Decision Making: Valuation Approaches

A significant risk of applying multiple assessment methods to an environmental decision problem is that in particular contexts all methods may not agree on which alternative policy is best. In the happy circumstance where economic, social and ecological assessments converge, all methods accrue evidence of conjoint validity and the confidence of policy/decision makers and the public is increased. When the various assessments significantly diverge, the decision maker and the public may have to decide which assessment to trust or how much weight to place on each. Assuming that each method meets accepted scientific standards of precision and reliability, is appropriately responsive to relevant changes in ecosystems/services and is properly related conceptually and empirically to things people value, there is little basis for choosing among them. When assessments diverge, the sufficiency of all assessments should be suspect, and analyses should proceed with considerable caution. Further research and analysis may be needed to resolve the conflict between assessments, or at least to identify the source of the conflict (such as payment vehicle issues, method-caused misconstrues of the target, or more basic rejection of assessment scenarios). Some formal decision making schemes (maximize the minimum, precautionary principle, weight of evidence) might help to resolve such disagreements in assessments. In some cases benefits assessors may have to pass on the conflicting assessments to be resolved by decision makers, perhaps using a formal stakeholder and/or expert deliberation process.

The descriptions and assessments of each method include the following elements:

- General description
- EPA decision contexts where this method could be used
- Where this method fits into EPA's overall process for valuing the protection of ecological systems and services (e.g., Figure 6-1.)
- Considerations for the most appropriate use of the method at EPA.
- How the method deals with uncertainty.

The description and assessment of methods begins with Mental Models, a method that can be useful in the application of methods from different approaches.

4.3 When is a cost a suitable measure of benefits?

Section text to be added.

4.4

4.5 Ecological Modeling

4.5.1 Ecological Production Functions

Section text to be added.

4.5.2 Spatial Representation of Biodiversity and Conservation Value

Description of method: This method results in the spatial representation of the uniqueness and irreplaceability of biological and ecological diversity in a regional context. This is a scientifically based approach to assign a conservation value to select species and ecological systems that are representative of an ecological region.

The values are represented as a numeric representation of the uniqueness, irreplaceability and level of imperilment for plant and animal species, vegetation, habitats and ecological systems.

Key assumptions:

- Representative biological and ecological diversity can be elaborated spatially across any region.
- The conservation value (status and quality) of each occurrence can be ascribed to each element of biodiversity as a repeatable and consistent procedure.
- The cumulative biological and ecological diversity and conservation values can be practically applied to inform and direct critical resource management and conservation decisions.

Key steps in the method include:

a) Define the biological and ecological targets for valuation
b) Define occurrence standards for each target
c) Define standards for valuing the quality of each occurrence
d) Define standards for measuring range wide status of each target
e) Create a 'conservation value layer' for each target that represents values and goals of the stakeholder
f) Create 'conservation value summary' of all targets that represents values and goals of the stakeholder
g) Modify the conservation value through incorporation of threats and opportunities.

An extended description of this method can be found in Appendix B.

<u>Decision contexts where this method could be used</u>:

- 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 EIS 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.

The method can be applied to a broad range of local to regional to national scales. The types of data and the spatial representation of this data change relative to the questions that are being addressed.

<u>Resource inputs and limitations:</u>

- The assumption is that there is a sufficient coverage of standardized biodiversity data required to run these models. The standards have been developed, and the data required changes associated with the application questions. Where there is a paucity of required data, it is readily 'developable', but can require the resources complete the required databases to run the models. The method is useless without good appropriate data.
- 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.
- Lack of data, currency and confidence of data, and data sharing issues associated with 'sensitive' data, training, and tools are the most important obstacles to the use of this method. However, there are many ways to create surrogate datasets that will allow users to adapt to different types of 'barriers'.

<u>Uncertainty</u>. There are confidence measures built into the methodology that can be brought into the decision making process or displayed separately for analysis. 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 elements of biodiversity.

<u>Other important dimensions:</u>

- The method is adaptable: it 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.
- The results are repeatable, and the process and algorithms are very transparent.

4.5.3 Energy and Material Flow Analysis.

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. These methods are covered here because they typically reveal insights not available through other means. In addition, they hold the possibility of treating ecological and economic systems in the same conceptual framework. In emphasizing the biophysical basis of the economic process, energy and material flow analyses are only indirectly based on human preferences; this feature is viewed by some as a negative, by others as a positive feature of this approach.

The first law of thermodynamics says energy can neither be destroyed nor created, while the second law says that, in an isolated system (a closed system is open to energy but not matter, an open system is open to energy and matter), all work processes tend toward greater entropy (disorder, degradation) over time. In other words, you can't finish any real physical process with as much useful energy as you had to start with — some is always irreversibly lost. In order to maintain organized structures (like an economy), one must constantly add organized, low entropy energy from outside the system. The earth as a whole is such a system. In thermodynamic terms it is essentially “closed” (i.e. energy, but not matter cross the boundaries). Of course, some matter does cross the boundaries (i.e. meteorites and spaceships) so the earth system is at least slightly “open,” but these flows of matter are very small compared to the flow of solar energy in and heat energy out. This section provides general background followed by two particular examples: embodied energy and emergy.

Energy and material flows analyses typically use input-output analysis or flow accounting methods, both used to quantify the flows of material and energy in an economic or ecological system. These analyses produce estimates of the energy (or more correctly available energy or “exergy”) cost of goods and services. Cleveland (1987) traced the early roots of this work dating back to the Physiocrats, those 18th century French thinkers who believed that wealth is derived solely from agriculture.

The energy and environmental events of the 1960’s and 1970’s prompted a number of economists, ecologists and physicists to examine the energy and material flows underlying the economic process. Economists such as Boulding (1966) and Georgescu-Roegen (1971, 1973) demonstrated the environmental and economic implications of the mass and energy balance principle. Ecologists such as Lotka (1922) and Odum (Odum and Pinkerton 1955, Odum 1971) pointed out the importance of energy in the structure and evolutionary dynamics of ecological and economic systems. And physicists such as the Nobel Laureate Prigogine contributed substantially to our understanding of the irreversible nature of thermodynamic processes.

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 first law of thermodynamics (conservation of mass and energy) has formed the basis for a number of important contributions. The integration of this law 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 volume of waste deposited in the environment.

Ayres (1978) described some of the important implications of the laws of thermodynamics for the production process, including the limits they place on the substitution of human-made capital for natural capital. Indeed a primary insight of this work is that 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). This insight has implications for the interpretation of traditional economic production functions.

Furthermore, work based on the laws of thermodynamics shows the limits of technological progress. In sectors that are largely concerned with processing and/or fabricating materials, technical change is subject to diminishing returns as it approaches a thermodynamic minimum (Ayres 1978), i.e. the minimum amount of energy and materials required to produce a unit of output. Ruth (1995) used equilibrium and non-equilibrium thermodynamics to describe the materials-energy-information relationship in the biosphere and in economic systems. In addition to illuminating the boundaries for material and energy conversions in economic systems, thermodynamic assessments can provide insights about depletion and degradation of the natural world, phenomena that often escape traditional economic analyses based on supply and demand curves.

Biophysical foundations have been incorporated into models of natural resource supply and of the relationship between energy use and economic performance. Cleveland and Kaufmann (1991) developed econometric models that explicitly represent and integrate the geologic, economic, and political forces that determine the supply of oil in the United States.

For explaining the historical record, these models are superior to any from a single discipline. Larsson et al. (1994) also 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.

One important advance generated by this work is recognition of the economic importance of energy quality, namely, that a kcal of primary electricity can produce more output than a kcal of oil, a kcal of oil can produce more output than a kcal of coal, and so on. Solar energy, although ubiquitous, requires man-made energy and materials to capture and upgrade it to more usable forms. Hence, the relative abundance and universality of solar energy does not translate into usable quality and abundant availability in the right forms at the right locations. An analogy can be made with water and water quality: "Water, water, everywhere, but not a drop to drink." Odum (1971) describes how energy use in ecological and economic hierarchies tends to increase the quality of energy, and that 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. Cleveland et al. (1984) and Kaufmann (1992) show that much of the decline in the energy/real GDP ratio in industrial nations is due to the shift from coal to petroleum and primary electricity. Their results show that autonomous energy-saving technical change has had little, if any, effect on the energy/real GDP ratio. Stern (1993) finds that accounting for fuel quality produces an unambiguous causal connection between energy use and economic growth in the United States, confirming the unique, critical role that energy plays in the production of wealth.

The analysis of energy flows has also been used to illuminate the structure of ecosystems (e.g. Odum, 1957). Hannon (1973) applied input-output analysis (originally developed to study interdependence in economies) to the analysis of energy flow in ecosystems. This approach quantifies the direct plus indirect energy that connects 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 using this framework to estimate "shadow prices" of ecosystem goods and services (Hannon et al. 1986, 1991, Costanza and Hannon 1989). Summarized below are three different approaches to analyzing energy flows: embodied energy and value, emergy, and ecological footprint analysis.

Embodied Energy and Value. Ecologists and physical scientists have also 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.

This theory of value represents a return to the classical ideas of Ricardo and more recently Sraffa (1960), but with some important distinctions. The classical economists recognized that if they could identify a "primary" input to the production process then they could 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.

Costanza (1980) has shown that at the global scale, the traditional "primary" factors are really "intermediate" factors of production. 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. Costanza noted that available energy has those characteristics which satisfy the criteria for a “primary” input: it is ubiquitous and essential to all production processes and cannot be replaced by something else (although substitution may occur between types of energy). 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 inputs.

Estimating total “energy” consumption for an economy is not a straightforward matter because (as noted above) not all fuels are of the same quality--i.e., they vary in their available energy, degree of organization, or ability to do work. Electricity, for example, is more versatile and cleaner in end use than petroleum, and it also costs more energy to produce. In an oil-fired power plant it takes from 3-5 kcal of oil to produce each kcal of electricity. Thus, adding up the raw heat equivalents of the various forms of fuel consumed by an economy without accounting for fuel quality can distort the picture, especially if the mix of fuel types is changing over time.

There have been a few attempts to empirically test this theory using both time series data and cross sectional data. Studies that have tried to adjust for fuel quality have shown a very close relationship between “available energy” consumption and economic output. Cleveland et al (1984) and Kaufmann (1992) have shown that almost all of the changes in E/GNP (or E/GDP) ratios in the US and OECD countries can be explained by changes in fuel quality and the percent of personal consumption expenditures (PCE) spent directly on fuel. The latter effect is due to the fact that PCE is a component of GNP and spending more on fuel directly will raise GNP without changing real economic output. Much of the apparent gain in energy efficiency (decreasing E/GNP ratio) is due to shifts to higher quality fuels (like natural gas and primary electricity) from lower quality ones (like coal). Renewable energy sources are generally lower quality and shifts to them may cause significant increases in the E/GNP ratio.

Another way of looking at the relationship between available energy and economic output uses cross-sectional rather than time-series data. This avoids some of the problems associated with changes in fuel mix and distortions in GNP. For example, Costanza (1980) and Costanza and Herendeen (1984) used an 87-sector input-output model of the US economy for 1963, 1967, and 1973, modified to include households and government as endogenous sectors (to include labor and government energy costs) to investigate the relationship between direct and indirect energy consumption (embodied energy) and dollar value of output. They found that dollar value of sector output was highly correlated (R^2 = .85 to .98) with embodied energy when this was calculated including the energy costs of labor, government, and environmental inputs (though not with direct energy consumption or with embodied energy calculated excluding labor and government energy costs). Thus, if one makes some necessary adjustments to estimates of energy consumption in order to better assess “available energy”, it appears that the empirical link between available energy cost and economic value is rather strong.

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 cost and price will, in general, come to equilibrium. This is the essence of the basic ideas of supply and demand. If a commodity has a much higher value than its cost of production, profits will be high and more of the commodity will be produced (with increasing marginal cost) until cost just equals price and profits are 0 for the last unit of production. Likewise, if cost is much higher than price, less will be produced until they are again equal. Therefore, a method which estimates costs, while not technically estimating price or value, should, in fact, be a fairly good approximation to price and value in those cases where markets have reached an approximate equilibrium.

Since markets are in constant flux and are far from perfect, we would expect there to be some divergence between cost and price in real systems. The question then becomes: "are the commodities overpriced or underpriced?" Should the energy costs be taken as the standard and market exchange values be adjusted or should the energy costs be ignored and the market prices taken as the standard? Given, on the one hand the enormous data requirements to calculate energy costs accurately, and on the other hand the pervasive market imperfections complicating market prices, there is no unambiguous right answer to this question. But one can learn much by looking at energy costs and market prices and comparing their degree of correspondence. Costanza (1980) did just that at the aggregate level of an 87 sector Input-Output model and found a fairly high degree of correspondence for those sectors where markets were fairly functional and lacking severe externalities, and low correspondence for those sectors where they were not (basically the extractive sectors at this level of aggregation). Since the method is based on the measurement of biophysical variables, it is less prone to bias caused by subjective or "framing" issues. However, given the large data requirements and the need to aggregate sectors in some inherently subjective way, caveats and caution must accompany any conclusions.

As described above, the embodied energy method assesses the direct and indirect energy costs of economic and ecological goods and services. These methods are most applicable on a broad policy scale such as energy policy and questions of sustainability. In fact, if the United States were ever to undertake serious moves toward ecological tax reform, embodied energy calculations would be extremely useful for taxing products based on their energy content. 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, 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. 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 quite useful, given the large uncertainties all around (see Costanza et al. 1989 for an example of wetlands valuation using this pluralistic approach).

Emergy, specifically Solar Emergy, is "the available solar energy used up directly and indirectly to make a service or product" (Odum, 1996). Emergy analysis considers all systems to be networks of energy flow and determines the emergy value of the streams and systems involved. Odum et. al. (2000) consider Emergy to be a common basis to measure the real wealth of the work of nature and society made on a common basis" (Odum et al., 2000). Emergy analysis presents an energetic basis for quantification or valuation of ecosystems goods and services.

Valuation methods in environmental and ecological economics estimate the value of ecosystem inputs in terms that have been defined narrowly and anthropocentrically, while emergy tries to capture the ecocentric value. It attempts to assign the "correct" value to ecological and economic products and services based on a theory of energy flow in systems ecology and its relation to systems survival.

A fundamental principle of emergy analysis is the Maximum Empower Principle. It states that "systems that will prevail in competition with others, develop the most useful work with inflowing emergy sources by reinforcing productive processes and overcoming limitations through system organization" (Brown and Herendeen, 1996). Odum (1996) states that this principle determines which systems, ecological and economic, will survive over time and hence contribute to future systems.

Since the early 1980s, emergy and emergy analysis have been used widely to analyze systems as diverse as ecological, industrial, economic and astronomical (Odum, 1996; Odum, 1995; Brown and Ulgiati, 1997; Lagerberg and Brown, 1999; Brown and Ulgiati, 2002).

Emergy is measured in solar embodied joules, abbreviated sej. Emergy analysis characterizes all products and services in equivalents of solar energy, that is, how much energy would be needed to do a particular task if solar radiation were the only input. It considers the Earth to be a closed system with solar energy, deep earth heat and tidal energy as major constant energy inputs and that all living systems sustain one another by participating in a network of energy flow by converting lower quality energy into both higher quality energy and degraded heat energy.

Since solar energy is the main energy input to the Earth, all other energies are scaled to solar equivalents to give common units. Other kinds of energy existing on the Earth can be derived from these three main sources, through energy transformations. Even the economy can be incorporated to this energy flow network as, "wealth directly and indirectly comes from environmental resources measured by emergy" (Odum, 1996). Examples are elevated and purified water, timber and oil. Therefore, the circulation of money is related to the flow of emergy.

An important concept in emergy analysis is Solar Transformity, defined as " the solar emergy required to make one Joule of a service or product" (Odum, 1996). Solar transformity is measured in sej/J. The solar transformity of a product is its solar emergy divided by its available energy, that is

$$M \; t \times B \quad (1)$$

where M is emergy, t is transformity and B is available energy. Since solar energy is the baseline of all emergy calculations, transformity of solar energy is unity. From a practical point of view, transformity is useful as a convenient way of determining the emergy of commonly used resources and commodities. Most case studies in the literature rely on the transformities calculated by Odum and co-workers to calculate the emergy of their inputs. Most transformities are calculated from the yearly emergy flow to the Earth (Odum, 2000).

The total emergy input to the Earth is the sum of the emergy of solar insolation, deep earth heat and tidal energy. However, even these inputs are not added directly due to their different abilities to do work. The emergy of deep earth heat and tidal energy are calculated by comparing their energy quality to that of solar insolation. The detailed calculations are based on energy balance equations for the earth, and are described by Odum (2000). These global emergy inputs are the driving force for all planetary activities. Determining their contribution to ecological goods and services is essential for further analysis.

Odum and coworkers have determined the emergy of the earth' s main processes such as, the total surface wind, rain water in streams, Earth 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 because they are considered co-products of the global geological cycle and cannot be produced independently with less amount of the total emergy.

At this point there is no indication that this has been applied in a specific EPA decision context. It is noted that efforts to evaluate the potential for application of Emergy are underway in ORD at their Atlantic Ecology Division Laboratory,

Clearly Emergy is a valuation technique that relies heavily on ecological modeling. Hau and Bakshi (2004) state that Emergy analysis overcomes the inability of many existing approaches to adequately consider the contribution of ecological processes to human progress and wealth. A large range of ecological products and services do not receive any value from conventional economic approaches despite the fact that they are used and spent for the making of economically valuable products, or indeed may be essential for life.

Hau and Baksi (2004) note that emergy has encountered a lot of resistance and criticism, particularly from economists, physicists, and engineers. Consequently, it has not been used much outside a small circle of researchers. They state that limited use of emergy analysis despite its broad relevance may be due to inadequate attention to details, poor communication of its potential importance, and lack of clear links with related concepts in other disciplines. They also suggest that the publication of Odum's " how to" book (Odum, 1996) and the more recent emergy folios (Odum et al., 2000; Odum, 2000; Brown and Bardi, 2001; Brandt-Williams, 2001) are important steps in making emergy more accessible. However, much more work is needed to connect emergy with concepts in other disciplines and to overcome a preconceived negative notion of emergy that is prevalent among many researchers outside of systems ecology.

Emergy theory has been characterized as simplistic, contradictory, misleading and inaccurate (Ayres, 1998; Cleveland et al., 2000; Mansson and McGlade, 1993; Spreng, 1988). Rebuttals to many critiques have also been published (Patten, 1993; Odum, 1995a). The persistent skepticism seems to stem from the difficulty in obtaining details about the underlying computations, and a lack of formal links with related concepts in other disciplines.

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 energy and material flow analysis section) are used to estimate direct and indirect land requirements. The EF is an effective heuristic and pedagogic device for presenting current total human resource use in a way that communicates easily to a broad range of people. Although there are ongoing debates about specific methods for calculating the EF (cf. Costanza 2000, Herendeen 2000; Simmons et al. 2000) using land area as a numeraire seems to communicate well to those who have trouble with money or energy as a numeraire.

4.5.4 Habitat Equivalency Analysis (HEA)

Habitat Equivalency Analysis (HEA) is a valuation method, which was developed to calculate compensation for interim lost service. The method calculates the amount of Habitat/resource to be created or enhanced to provide the same level of services over time as were lost due to the injury. The method simultaneously quantifies injury and scales the size of restoration. An implicit assumption is that the value of a unit of lost service and that of the replacement service are comparable. The primary steps in performing a HEA are as follows:

a) Document and quantify injury
b) Identify and evaluate replacement project options
c) Scale the replacement project to compensate for injury
d) Determine the appropriate means of compensation
    - Cost selected restoration options
    - Set performance standards for compensatory restoration projects

The HEA approach focuses on scaling value on a service to service or resource to resource approach. Therefore in expressing values HEA relies on biophysical units such as acres of habitat service and services projected over time in service acre years. Since this is a restoration/compensation method that is projected into the future the final unit is a Net present Value (NPV) measure of the service into the future sated as Discounted Service Acre Years (DSAYs).

The HEA methodology like the NEBA framework in which it is sometimes applied does relies on structural or spatial measures of ecological components such as acres of habitat. Specific service types such as provisioning, regulating, cultural and supporting services as expresses in the Millennium Assessment (MA) framework are not identified or expressed but would be considered to be present and operating. This is not problematic because of the simplifying assumption that the lost acre equals the replacement acre and therefore the service is present in both.

HEA was developed for use 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 to regulatory frameworks there is a different focus on compensation. Under Superfund actions compensation for damages is focused on monetary compensation which requires value be scale to cost while under OPA the focus is on resource compensation. Under OPA the question is how much public resources the public requires to be made whole for their loss, so therefore value is scale from resource to resource.

With regard to where to place HEA in Figure 4-1 Proposed Process for Valuation., it would seem to bridge a number the process elements. Because HEA simultaneously quantifies the injury and scales the value of restoration it would appear to be both a valuation and impact assessment tool. As well, it would seem to be flexible enough to work both as a tool for estimating non-monetary values under OPA and monetary valuation under CERCLA. The later being accomplished by including the costs to restore a DSAY as the measure of value.

The spatial scale at which HEA has typically operated has been at the level of local to regional impacts. HEA also operates over a time scale in that it may have to compensate for injury over time as well as allow for time for restoration projects to mature to full service value.

Uncertainty in a resource to resource approach, such as HEA, would primarily be associated with gaps in knowledge associated with

a) How much area was injured?
b) To what degree was the resource injured
c) How long has the injury been in place?
d) How long will it take to replace the injury?
e) Is the replaced service equal to loss?

To some degree, uncertainty is dampened (or ignored) by the principal operating assumption that the habitat lost and the habitat created are equal or eventually will be equal. In practice such an assumption is tracked by monitoring the performance of projects in the field to see if they do or don't yield adequate replacement service.

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 has to be selected for the NPV calculations.

4.5.5 Ecosystem Benefit Indicators

Ecosystem benefit indicators are a quantitative and visual, but not monetary, approach to the assessment of habitats and land uses. Like ecological indicators they summarize and quantify complex information. Like monetary assessment they employ the principles of economic analysis.

Environmental benefit indicators (EBIs) are a way to illustrate ecological benefits in a specific setting. An individual EBI might be the presence of invasive species or the number of acres under active cultivation. A collection of indicators about a given area can portray the complex relationships among habitats, species, land uses, and human activities. 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.

Benefit indicators can capture the landscape, or spatial, factors that contribute to social well-being. This is in fact a virtue of indicator methods. Indicators can be derived from and mapped within a GIS context. Spatial analysis is important because the ecological production function is a function of spatial interdependencies. From an economic standpoint, the social determinants of service benefits depend upon the landscape context in which those services arise. The consumption of services often occurs over a wide scale. 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.

Benefits are expressed as bundles of indicators. Some indicators are biophysical, others relate to the socio-economic environment. The indicators are calculated and can be depicted spatially.

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 elicit public preferences over environmental and economic options – as in mediated modeling exercises or more informal political derivations. 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.

Decision Contexts where this Method can be used: At the national level, in the evaluation of new rules as part of the RIA process, government performance reviews, strategic planning, budget justification, and priority setting. They are also applicable at more local scales as a tool to improve regional and local planning, such as watershed planning in the context of TMDLs.

Advantages/ Disadvantages: The method is applicable to the full range of ecological services. In practice, applicability may be limited by data gaps.

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

EBIs strip environmental valuation of much of its technical content, but do so to reach and convert a much wider audience to economic reasoning applied to ecological issues and impacts. Their simplicity, and transparency, is a benefit. They can be used to communicate and educate. By avoiding monetary estimation of benefits they are also a way for the agency to overcome resistance to economic assessments of the natural world – while still conveying economic principles and the dependence of human well-being on natural assets.

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 service flows 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.

4.6 Socio-Psychological Approaches

4.6.1 Conceptual Foundation

Both economic and attitude surveys assume that all value is anthropogenic, so that the necessary basis for the valuation of ecosystems and ecosystem services must be the judgments/preferences of people. This conceptualization clearly includes anthropocentric utilitarian or aesthetic values, but it can also encompass the assignment of values by humans to biological/ecological objects and systems based on non-use and/or ethical/moral grounds. In principle, both means and ends values can be assessed. Economic and social-psychological theories differ on the nature of the formal logic postulated to underlie preference expressions, and on the extent to which human valuations are expected universally to adhere to those rules. Economic methods rely on the rational theory of welfare economics, while social-psychological methods more often rely on less formal empirically-based theories that emphasize the contextual and emotional basis of preferences. An extended description of this method can be found in Appendix B.

4.6.2 Surveys

Surveys encompass a broad range of methods for systematically asking people questions and recording and analyzing their answers. The prototypical survey is a standard list of questions to which sampled individuals provide responses in the form of choices, rankings, ratings or magnitude estimates. 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. Ends, means and tradeoffs among multiple benefits and costs have all been addressed with some degree of success. Surveys may be conducted face-to-face, by mail, telephone or more recently over the internet. Respondents may be selected by a formal random sampling process, by convenience or based on some individual or group-membership criteria assumed a priori to be relevant to the targeted issues. Survey methods provide a flexible and efficient approach to finding out what people value about ecosystems and ecosystem services, what their preferences are for alternative protection measures, and why. Multiple-item verbal surveys are the most common form, but other formats include conjoint analysis, where options are presented as multi-dimensional packages (rather than separate items), perceptual surveys, which feature direct experience, visualizations or other representations of attitude targets (instead of verbal descriptions/labels), and interactive games, where the respondent converges on a preferred option by learning through virtual experience with modeled/simulated alternatives. Surveys by sociologists and psychologists are often termed *attitude surveys*, but they are conceptually and operationally very similar to the stated preference methods used by economists, such as contingent valuation surveys. The principal difference is that economic surveys seek a unidimensional and universal monetary index of value.

Attitude surveys typically employ choices, rankings or ratings of *preference*, *liking*, *importance* or *acceptability* as indicators of attitude (subjective utility). The indices produced are usually claimed only to be ordinal (ranks) or roughly interval scale measures (equal differences on the measurement scale indicate approximately equal differences on the underlying value being measured, but the origin of the scale is arbitrary). Magnitude estimation procedures can support a claim of ratio scale indices, but these methods are rarely used in attitude surveys. Separate attitude indices are typically developed for multiple value scales (e.g., utilitarian, ethical, anthropocentric, biocentric), and value indices along these scales are assumed to be strongly constrained by the particular contents and contexts of the survey with which they were collected. That is, attitudes are assumed to be at least in part created in the context of the survey, rather than pre-existing transituational states that are uncovered.

<u>EPA decision contexts where this method could be used.</u> Attitude survey methods could make positive contributions throughout the EPA decision and rule making process. Before any specific rule or regulation is considered, broadly-based surveys could ascertain general public wishes and concerns relevant to EPA's mission. Surveys could be helpful in determining public understanding of what ecosystems and ecosystem services are, and for determining the existing levels of awareness and appreciation for the kinds of benefits ecosystems provide directly and indirectly for individuals and for society more generally. The results of such surveys could help EPA to establish ecosystem protection priorities and/or to identify important public information

and education needs. Where policy options can be specified in more detail, attitude surveys could be used to rank or more precisely scale alternatives in terms of public preferences. At the regional levels, where options are being considered for a specific site or sites, surveys might ascertain public preferences or support for, and/or likely responses to, the alternative actions under consideration. Perceptual survey components might be added in the form of inspection of example sites exhibiting comparable ecological conditions and/or by presentation of computer projections and visualizations of expected outcomes for alternatives. In these more specific contexts, conjoint survey methods might be especially useful for assessing tradeoffs among the multiple ecological, utilitarian, aesthetic and other values represented by the range of feasible alternative policies.

<u>Where this method fits into the overall process for valuing the protection of ecological systems and services</u>. In the context of Figure 4-1 Proposed Process for Valuation., survey methods have a useful role to play in the identifying what matters to people.. Attitude surveys could also be a regular member of the "Valuation and Preferences Methods" that contribute to the identification of relevant ecological impacts and could be combined with biological and economic indicators to estimate the relative value of (public preferences for) outcomes across the range of action alternatives under consideration. As the conceptual model implies, attitude surveys could also help to translate various ecological endpoints into socially relevant effects that are more suitable for economic evaluation and monetization, where appropriate (as in formal benefit-cost analysis). Surveys could make an additional contribution not represented in the conceptual model diagram. The value of ecosystems/services will inevitably be represented by multiple economic/monetary, bio-ecological and social/attitudinal indicators, leaving EPA administrators with the task of combining these diverse and potentially conflicting measures to make and/or to rationalize management decisions. Properly structured attitude surveys could effectively involve citizen stakeholders in this value consolidation process, providing an additional relevant input to the decision maker, and adding to the political validity and social acceptability of the final action.

<u>Considerations for the most appropriate use of the method</u>. Attitude surveys do not claim to achieve a unidimensional, transituational measure of value, and thus would not meet the requirements of conventional economic cost-benefit or cost-effectiveness analyses. However, given the identification of a feasible set of alternative regulatory/protection actions, attitude survey methods would be appropriate for assessing public preferences among these options and for calculating relationships among the multiple component attributes (costs and benefits, biological and social, means and ends) of those actions. Conjoint methods are especially well-suited for gauging public preferences across sets of complex multi-dimensional alternatives (e.g., alternative EPA regulations for ecosystems/services protection). Respondents can be required to choose among (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. Expressed preferences among a carefully constructed array of such alternatives can provide the basis for regression analyses to determine the relative contributions of each component attribute/dimension. Following the simple example where alternative conjoint policies are represented by different combinations of water and air quality and job opportunities, preferences for options might be represented by

$$\textit{Preference for option } j = w_1(\mathrm{WQ}_j) + w_2(\mathrm{AQ}_j) + w_3(\mathrm{Jobs}_j)$$

where option j is a particular policy that produces specific levels of water and air quality (WQ_j and AQ_j) and jobs ($Jobs_j$) and the preference for this option is found by multiplying the measure of each attribute by its respective derived coefficient/weight (w_i) and summing the products. This equation can 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 presented 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.

In principle, attitude surveys could be applied across all of the temporal and geographic scales relevant to EPA actions. That is, people can be asked to express preferences for alternative states of the world from microscopic to universal and from momentary to epochal. On a more practical level, even when there is high confidence in the projections of the relevant bio-ecological conditions and expressed preferences for those conditions are highly internally consistent (reliable) there is no way to ascertain whether the valuations that they imply are valid over those geographic/temporal scales. The effective scale of any valuation method is limited by current science and experience, and by the capacities and frailties of the human mind. There is no assurance that current conceptualizations of value are valid at scales that exceed contemporary human's experiences. Indeed, the whole enterprise of managing and protecting ecosystems and ecosystem services necessarily proceeds on an article of faith (or a multitude of articles) not subject to scientific validation or logical proof.

Theoretically, both economic and attitude surveys must assume that their representations of the ecological and social outcomes of alternative policies are correct and are adequately understood by respondents. It must also be assumed that respondents can determine the implications of those outcomes for their own (or other designated constituents) welfare and can form and express meaningful and relatively stable preferences for those outcomes. The more familiar economic valuation model further assumes that all objects (states of the world, events, processes, etc) can be mapped to a single subjective utility scale (usually denominated in dollars) and that individuals will be equally satisfied (happy, well-off) with any two objects with the same measure (dollar value) on that scale (the commensurability and substitutability assumptions). Thus, dollars paid (or accepted) in one place and time for changes in ecosystem protections or services can meaningfully be added to or subtracted from (and multiplied or divided by) dollars paid for rent, groceries, military hardware or social welfare programs at other places and times (as in a benefit-cost analysis). Moreover, monetary indices based on individual judgments (actions) may be summed to achieve a global social scale of economic value, usually assuming equal weight per individual (substitutability of people, perhaps adjusted for income)

In contrast, attitude surveys assume multiple value dimensions and expect a high level of situational/contextual constraint on their value metrics. It is generally not appropriate to combine (add) indices across different value dimensions, and generalizations to other value contexts (e.g., other people, places and times) are usually only qualitative and require considerable caution. With regard to combining individual's attitudes, the emphasis is more on classifying respondents by socio-demographic, personality or value-profile categories than on aggregating to global, social-scale indices. Mean indices are typically reported for separate subgroups of respondents based on a priori policy relevance, or on discovered patterns of preferences across the multiple value dimensions addressed in the survey.

In practical use, the human resources required to implement attitude 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. The mail and/or telephone (or computer) resources required for distribution can be significant in large surveys. All of these costs are usually quite low relative to the physical, biological and/or ecological science and field study required to create adequate representations/projections of value-relevant outcomes for a suitable range of alternative regulatory/protection actions. In many ways, the quality of social evaluations of ecosystem and ecosystem service effects is dependent upon the quality of the relevant projections and specifications of ecological endpoints and social consequences. In some cases considerable resources may have to be devoted to translating targeted ecological outcomes into understandable representations of socially relevant effects.

The largest barriers to greater use of attitude survey methods in the EPA are institutional. First, while the agency 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 attitude surveys, 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 Benefit-Cost Analyses (which also apply to other agencies), but none of the currently applicable laws preclude using valuations based on social/attitude survey 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 (economic and attitude) are discouraged by federal rules nominally designed to protect the public's privacy and tranquility. Over the past several decades it has been very difficult or impossible for federal agencies to attain required clearances (e.g., from the OMB) for surveying the public. This institutional barrier is formidable, but some significant surveys continue to be conducted (more or less openly). Even when this institutional barrier is hurdled, the practical problems of selecting and contacting a proper sample for a significant survey are substantial, and low "response rates" are a continuing, and by some accounts increasing problem. Irrespective of protection from federal government surveys, the public is inundated with surveys and pseudo-surveys from political, commercial and charitable entities, so that there is very considerable reluctance to participate in any survey.

In terms of adaptability, attitude surveys are quite flexible. However, the utility of the results of a survey applied in the context of assessing public preferences for alternative ecosystems/services protections depend strongly on how well the alternative biological outcomes and associated social effects have been determined, and how well they have been represented to the respondents. If the ecological data and projections of alternative conditions are vague and uncertain, and/or the social effects of those outcomes are poorly specified or too complex for respondents to understand, survey results can be meaningless. In the context of EPA's need to assess the benefits of ecosystems and ecosystem services, complexity of outcomes is likely to be a significant obstacle for attitude surveys. When the valuation target is a complex of multiple biological and social attributes that interact and change over time in ways that are uncertain, it will be very difficult or impossible to represent policy-outcome alternatives to public respondents in ways that can assure proper understanding and responses. Certainly terse telephone surveys are unlikely to be helpful in these circumstances. It may be better to accept a more restricted sample of respondents who are allowed more time and opportunity to learn about the options under consideration through more interactive, deliberative survey procedures.

Uncertainty. There are two broad levels of uncertainty in any evaluation of changes in ecosystems and ecosystem services. 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. At the social level existing and projected ecological conditions and their socially relevant consequences must be represented to people so that they can express their preferences for current and/or alternative future conditions, providing the basis for analysts to derive valid measures of the value of the targeted changes. Within the social level, survey methods specifically must address sampling errors (e.g., representative sampling, non-response), specification errors (e.g., adequate description/representation 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.

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 bio-physical level it is extremely difficult or completely unclear how to calculate the uncertainty (error) in the projection of even a single outcome (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. While highly trained and experienced experts may find ways to calculate a relevant measure of uncertainty for some complex ecological outcomes, it is problematic how to meaningfully communicate this level of uncertainty to concerned lay citizens in a survey. Yet such communication can be crucial, as often the level of uncertainty in outcomes is a key factor affecting the preferences for the alternative policies under consideration.

Accepted methods are available and are commonly used for calculating confidence intervals or complete probability distributions for individual survey responses (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 items, 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, and of the classifications of sub-groups of respondents, based on patterns of individual's responses to multiple items in the survey.

<u>Other important dimensions</u>. In principle, survey methods are very flexible, and assuming that relevant ecological endpoints and effects can be provided for suitable time steps, preferences could be assessed for those steps and the patterns of values over time could be inspected. However, following the reasoning in item 2.e. above, there is reason to be cautious about human judgments/preferences extended over long time periods, especially into the future. There is no clear way to validate people's preferences for ecological conditions and effects beyond their own expected life span, and there is no generally accepted calculus for aggregating preferences at separate time steps over such intervals. Some important ecological changes occur very gradually (at least up to some threshold), and while these changes can be speeded up for presentation in surveys (as by time-lapse visualizations, for example) there is no guarantee that responses to these accelerated presentations are valid indicators of responses to their referent changes in the real world. Moreover, the required assumption of static human preferences into the future, especially between generations, is almost surely wrong. Still, there is no viable option but to rely on the judgments of contemporary humans for valuing future states of the environment.

Attitude surveys typically assume multiple value dimensions. Thus, the value of changes in ecosystem protections and ecosystem services would typically be assessed and reported separately from the value of changes in human health, jobs, community stability and/or regional economic product. There is no generally accepted calculus, like benefit-cost analysis, for combining the multiple values assessed in attitude surveys into a single benefit number. Multi-attribute utility methods can be applied to aggregate across and to assess tradeoffs among multiple value dimensions, and conflict negotiation, consensus building and decision aiding techniques can help participants appreciate and resolve such tradeoffs. Conjoint methods can be used to assess directly preferences for the "bundles of goods" represented by alternative policies, but there are practical constraints on the number of attributes/dimensions and the number of levels on each dimension that can effectively be evaluated with this method.

Like economic valuation methods, attitude survey methods are constrained by and pose some constraints on ecological models. In particular, endpoints of ecological modeling systems must be related to interpretable and socially relevant biological indices and effects if attitude surveys are to be used to determine public preferences among alternative outcomes. In some cases, social-psychological survey methods could be instrumental in translating ecological endpoints into social preference indicators that would be more compatible with economic valuation methods. For example, in the context of the visibility components of the Clean Air Act, projected concentrations and dispersions of chemical air pollutants were translated into realistic visualizations of the implied changes in park and wilderness area vistas. Perceptual surveys with these visualizations were used to develop psychophysical functions quantifying the relationships between changes in air quality parameters and public perceptions of visibility and scenic beauty. Travel cost and other methods were applied to assess the economic contributions of changes in perceived visibility and scenic beauty to the economic value of recreational visits to areas proposed for protection under the act.

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., the sampling 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 how these changes affect evaluation outcomes.

Internal consistency is one of the key technical criteria for attitude surveys, and high levels of reliability are generally achieved. Where the representativeness of the respondent sample is important, surveyors typically take great pains to confirm the fit of their samples to relevant population parameters, and to confirm the generality of their results by explicit tests, as by extra assessments directed at non-responders or underrepresented groups.

Determining the external validity of survey results is considerably more problematic. It is impossible to confirm that the results of an attitude survey have successfully captured the "true value" of targeted changes in ecosystems/services. In the end the validity of any valuation method is a theoretical matter--assessments are valid to the extent that the values produced show appropriate relationships, positive, negative and null, with a large network of other variables that are deemed by theory to be relevant. At a more practical level, high positive correlations between independent assessments of the same alternatives using different methods are taken as a measure of conjoint validity for both methods. For example, the results of economic surveys and attitude surveys for the same objects by the same respondent populations have sometimes been found to be strongly positively correlated, implying some validity for both methods. However, low or even negative correlations between economic and attitude measures can occur, even for the same respondent within the same survey. In these cases the apparent value of an object/event depends significantly upon the method used to assess it. Such differences between economic and attitude measures are most likely to occur when the objects being evaluated are not usually (or ever) bought and sold in markets, or where respondents find dollar valuations inappropriate or unethical. A substantial share of ecosystem and ecosystem service valuations are likely to fall in this category.

4.6.3 Mental Models

Survey research assumes that survey recipients understand the questions they receive in the way intended by the survey designer, and that their responses can be interpreted accordingly. Economists who use survey research methods can benefit from approaches to improving this exchange of information.

One widely used approach is to use exploratory, open-ended research methods to understand better the target population's conceptual landscape of the survey topic, before designing specific survey items and response scales. Focus groups are perhaps the best known of these kinds of exploratory approaches. The group dynamics of focus groups may reveal, but can also obscure, specific conceptual issues, wording choices, and individual differences in understanding of a topic or domain. Interviews designed to probe individuals' mental models of the topic are a useful complement or alternative to focus groups. 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 studies aim at eliciting people's understandings 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.

Key steps in mental models research as applied to risk management

- Develop expert-informed decision model to address the targeted risk management decision(s).
- Elicit stakeholder mental models of the underlying hazardous process and related decisions, using a semi-structured interview protocol (initial questions are open-ended).
- Analyze results with an open-ended, exploratory coding scheme, to allow comparisons with expert decision model as well as qualitative characterization of beliefs that do not map onto the expert model.
- Develop survey instrument to check reliability of findings from mental models interviews. Survey representative stakeholder population, using both closed-ended and some open-ended survey items.

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.

In mental models research, values are expressed qualitatively, sometimes ordinally (comparative statements), and sometimes using quantitative scales. This approach is designed to explore the conceptual landscape for risks, 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 have to be used in conjunction with another method in order to obtain benefits numbers. 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.

The method is 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). The method requires qualitative analysis of results, in order to provide effective input to survey instrument design.

Issues in implementing mental models research. Mental models research is resource-intensive, if carried out carefully, but can be used effectively as a starting point for any survey or broader scale research on values. The method assumes that a fairly small sample will characterize the distribution of basic beliefs about the hazard/risk to be found in the population of interest, and that a larger representative sample can be drawn and will respond to surveys. A follow-up survey is generally necessary before drawing conclusions about the distribution of particular beliefs and mental models.

As with most methods, to some extent the effort invested will correspond to the quality of the product the method produces. A casual application of the method could be carried out by a single researcher, with sufficient time and training in decision analysis, interview and survey research methods, and the focal domain. A team of 3-4 researchers working together fulltime could probably carry out an entire mental models study in a month, if they had access to domain experts and members of the target population to interview/survey.

Potential obstacles to the effective use of the method include the following: institutional review board clearances, Information Collection Request Clearances under the Paperwork Reduction Act, lack of training in interview and survey research, and qualitative research methods more generally, difficulty obtaining responses from randomly sampled members of the population, and lack of familiarity with decision analysis are probably the largest obstacles to effective use of the method.

Mental models research assumes some homogeneity in how people conceptualize the world, and requires an underlying theory of culture and meaning (e.g., Romney, Weller et al. 1986, Romney, Moore et al. 2000) on the theory of culture as consensus), but no more so – and possibly less so - than other survey or interview research. Variability in beliefs is captured, as well as qualitative statements of certainty or uncertainty. The method produces neither point estimates nor probability distributions, per se. (Add comment on stability of beliefs)

The method could be adapted to assess beliefs about system dynamics, although in studies to date the temporal dimension of underlying processes has not been probed much.

The output of mental models studies is generally easy to communicate, understandable, and of interest to intended audiences. Even simple analyses of the data, including frequencies of beliefs and co-occurrences of beliefs, can go a long way toward clarifying how people respond to messages/statements/questions about the focal topic.

In those few comparisons that have been made to date (e.g., mental models of global climate change), results from a mental models approach have been consistent with results from other exploratory analyses and cognitive maps (e.g., studies by cognitive anthropologists, such as Kempton – add ref), and results from the surveys have been consistent with the interview results, within the method. A possible point of sensitivity is the choice of expert decision model(s) to be used as the basis for the coding of the interviews. [Add further discussion of convergent validity and reliability].

4.6.4 Open-response qualitative analysis formats

Section to be added.
- Focus groups
- Open-end questions in interviews and surveys
- Narratives

4.6.5 Behavioral observation

Section to be added.
- Visitor counts
  - Intercepts
  - Car/parking lot surveys
- Visitor flow tracking
  - Travel logs, trip diaries, registration boxes, entry permits
  - Direct observation (trail/road intercepts, campsite census)
  - Video/photo monitors
  - Automated sensors/counters

4.6.6 Behavior traces

Section to be added.

- Registration records (facilities, hotels, etc)
- Ancillary purchases of equipment and supplies
- Trail erosion, vegetation disturbance, road wear
- Trash distribution
- Document review and analysis (news papers, tax records, etc)

4.7 Assessments Based on Voting and Other Group Expressions of Social/Civic Values

4.7.1 Referenda and public decisions

Referendum votes and other formal public decisions provide the basis for a set of valuation approaches that can provide 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), initiatives (ballot measures proposed by citizens), or other official public decisions 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.

Referenda and initiatives are formal solicitations to the public to determine the public's willingness to pay. In a referendum or initiative, officials or policy activists present voting choices that formally specify environmental objectives, such as reducing air pollution, establishing a wildlife preserve, or building a storm run-off system. In some cases, these objectives are clearly specified in quantitative terms: number of tons of sulfur dioxide expected to be removed, number of acres of reserve, or reduction of the area subject to flooding. The costs of achieving these objectives are specified in various ways, ranging from the financial costs in taxes or bonds, to the restrictions that would be expected to impose opportunity costs such as reduced employment opportunities or restricted resource extraction.

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 and 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 and Shapiro 1975 and (Shabman and Stephenson 1996).

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 and Matsusaka, List and Gallet 2001, Murphy, Allen et al. 2003, Polasky, Gainutdinova et al. 1996, Schläpfer, Roschewitz, & Hanley 2004, Vossler and Kerkvliet 2003). As Arrow, Solow 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. T he 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)

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 intrinsic validity within the paradigm that gives standing to the community votes as reflecting the policies that the public prefers.

Direct Referendum/Initiative Analysis. The valuation analyst can chose to take the referendum choices as they are formally specified, in which case a winning proposal can be interpreted as having standing as the electorate's choice. For example, a municipal government may propose a referendum measure to purchase and maintain 500 acres of currently unused land as a forest reserve costing $1,000,000 annually for a community of 10,000 households. Assume that the measure is not significantly entangled in controversies over how it will be financed (e.g., there is no opposition that a bond measure would simply saddle future generations). The measure passes by 51%. The value can be metricized in various ways; e.g., as

- $1,000,000 per annum for the 500 acres for the community
- $2,000 per annum per acre for the community
- $100 per annum for the 500 acres per household
- $.20 per annum per acre per household.

If the initiative or referendum passes by a slim majority, this valuation can be considered to be quite close to the "community's" valuation. If the vote is more strongly in favor, then the valuation represents a floor on the community's value of the eco-system benefits. If the initiative or referendum loses by a slim majority, then (more arguably) one could assert that the community's valuation is also close to the value implied by the proposed measure.

If the outcome is not close (e.g., the initiative or referendum passes by 70%), the inferred value is a floor on the community's value. This is because a higher cost may have still gained a majority, albeit probably a narrower one.

However, the fact that a referendum or initiative fails to pass does not necessarily mean that the inferred value is a ceiling on the community's value, because other issues, such as how the measure is to be financed, may lead to the rejection of a measure that otherwise would have been accepted. 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.

Note that the approach does not primarily address the mean value of the ecosystem improvement or protection. This is because the electorate's choice is not the conventional utilitarian notion of the total value summed across all individuals who vote. It is possible to determine a very modest floor on this aggregate value (and therefore on the mean value) by attributing to the "yes" voters the value of the benefit-cost ratio specified by the proposal, and a value of zero to all voters who opposed the proposal. For example, in the case of the forest reserve proposal described above, if the proposal had received a 70% "yes" vote, the minimum mean value would be $1,400 per annum per acre for the community (i.e., .7 x $2,000 + .3 x 0).

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.

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 eco-systems. 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.

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. 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. 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 sometimes difficult for voters to assess the actual stakes involved. The benefits will often have to be predicted (e.g., how much biodiversity will be reserve really safeguard; how much less 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.

Referendum/Initiative Analysis Followed by a Survey. Therefore another variant that relies on referendum and 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 stakes. 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. As with Variant 1, if the initiative or referendum passes by a slim majority, this valuation can be considered to be quite close to the median voter's valuation. If the initiative or referendum loses by a slim majority, then (more arguably) one could assert that the median voter's valuation is also close to the value implied by the proposed measure. (Note: again, a losing initiative or referendum does not necessarily mean that the inferred value is a ceiling on the median voter's value, because other issues may lead to the rejection of a measure that otherwise would have been accepted.) As with Variant 1, 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.

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.

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, it does represent what the voters have chosen. Different logics underlie their standing.

<u>Direct Analysis of Public Decisions to Accept Pollution or Resource Depletion.</u> While the approaches outlined above provide information about willingness to pay, there are some public decisions that 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 eco-system. Assuming that 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 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.

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 clear and transactions are low, the ratio of the perceived benefits and costs 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 survey questions 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.

Uses and Limitations of All Four Variants. All of these approaches attempt to measure the sum total of values of improving or protecting eco-systems 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 eco-system improvement or protection. The variants are flexible in terms of levels of data, detail and scope, inasmuch as initiatives, referenda and other public 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, referenda, and other public decisions 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 any of these variants, either singly or as cross-checks with conventional revealed preference or sated 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.

The first two variants, in analyzing referenda and initiatives, 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.). The third and fourth variants 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).

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

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.

These approaches would work best when:

a) applied to the same jurisdiction (e.g., if Portland is considering another storm control issue, the analysis of the Portland 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.

The resources needed to implement the variants would depend on the applications. If the purpose is to compile a set of initiative and referendum results, this could be done for the first approach by a) assigning an EPA economist to oversee the effort (perhaps 10% effort over a year); b) assigning an intern to compile as many U.S. municipal, county and state initiatives and referenda related to environmental and conservation held over the past half-decade. (perhaps 50% effort over a year). The analysis to generate valuations would require 10% of the time of a two-person team of EPA economists, perhaps one being a consultant. For the second variant, more effort is required for each survey: two EPA analysts (or consultants) each devoting one month to develop, administer, and analyze the survey results.

The major obstacle to the effective use of these approaches may be the lack of familiarity within government of the approach of drawing inferences from public decisions, although the method has had a respectable history of use in estimating the value of a “statistical life.” It is striking that despite the multiple studies of how conventional valuation methods such as contingent valuation compare to initiatives or referenda outcomes, there is apparently no literature that takes the outcomes of the initiatives or referenda per se as valuations, except to study why different subunits [e.g., counties within California (Kahn & Matsusaka, 1997)] yield different outcomes. Perhaps it is just too simple a finding—that a particular initiative or referendum that devotes X dollars to gain Y enhancement or protection of the eco-system—to warrant publication. Nevertheless, the paucity of literature may be an obstacle to adopting this approach.

<u>Uncertainty</u>. The uncertainties involved in the variants (first and third) that focus on benefits and costs specified in the proposals lie in the estimates of actual benefits and costs entailed in the proposals. They should be analyzed with the standard methods of projecting consequences, and conveyed through probability distributions and confidence intervals. The uncertainties involved in the approaches that rely on surveys lie in the potential for biased sampling in the selection of survey respondents, as well as poor memory and response set (e.g., respondents may report that they voted). These can be reduced through careful random sampling and cross-checks within the questionnaires.

4.7.2 Mediated Modeling

<u>Brief Description of Method.</u> 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 the involvement of stakeholders (parties interested in or affected by the decisions the model addresses) as active participants in all stages of the modeling process, from initial problem scoping to model development, implementation and use (Costanza and Matthias 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 et al. 1995). An extended description of this method can be found in Appendix. B.

<u>Mediated Modeling and Value.</u> 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 get 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. (Higgins, Turpie 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 et al. 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 which 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 the goals, an 'atelier' 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 affairs which 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) plant; 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.

<u>Decision contexts where this method can be used.</u> As described above, the method is fairly general and could be used to assess any value (means toward and ends) 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 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.

<u>Resource Inputs/Limitations</u>. 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; 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. The final outcome of this process cannot be predetermined.

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.

Other important dimensions:

- 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. 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. No research has yet been done on whether application of the process to exactly the same problem by two 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.

4.7.3 Deliberative value elicitation

The objective of contingent valuation (CV) is to elicit a respondent's true (or one might suggest, pre-existing, based on the utility associated with the good) value as it relates to a given good or, in other words, the maximum amount the good is judged to be worth to a respondent before they would prefer not have it (Mitchell and Carson 1995). In the case of ecological systems and service, this value is intended to reflect the value of system itself (e.g., the aesthetic value of a watershed) or the value associated with a change in the level of service associated with an ecosystem (e.g., a reduction or improvement in the ability of the watershed to reduce eutrophication).

Various CV approaches have been used in an attempt to obtain the most accurate and precise estimates of an individual's willingness-to-pay (WTP); e.g., open-ended protocols, dichotomous choice, etc. All of these approaches have been the focus of intense scrutiny and criticism ((Arrow, Solow et al. 1993; Fischhoff 1997; Portney 1994). One of the most intense criticisms of contingent valuation is the role that preference construction plays in the formation of an individual's values.

Rather than approaching decision problems with stable and thoughtful preferences that are merely revealed during decision making, the theory of preference construction states that people instead construct their preferences "on the spot" in response to cues that are available during the elicitation process (Payne, Bettman et al. 1993, Slovic 1995). In this sense, elicitation procedures, whether experimental or practical, have the de facto purpose of serving as architects of decision making rather than simply revealing, as would an archaeologist, a person's pre-existing preferences (Gregory, Lichtenstein et al. 1993). Applied to the CV approach, the theory of preference construction asserts that for most goods—and certainly those that are unfamiliar to a respondent—people don't have well formed, pre-existing, or otherwise "true" judgments about their value. Instead, they construct their judgments about the good's worth in response to a variety of cues present during the elicitation process. As a result of the implied plasticity of these judgments, many critics deem the values obtained via the CV approach to be invalid.

In response to these concerns, some researchers and practitioners have begun to explore modified CV approaches that help respondents to go beyond the simple statement a value for a given good. Specifically, these modified CV approaches are designed to help respondents to construct (and then state) a more thoughtful and, some would argue, more defensible value for a good or service.

One approach is to design and implement a "deliberative" CV methodology that uses both small-group discussion along with an established decision making structure to foster more informed and defensible valuation estimates. The reasoning behind this approach is twofold: First, small-group discussion will lead to a more in-depth consideration by respondents of both contextual information about a valuation problem (i.e., technical information provided by an agency) and how the objectives of various stakeholders may be affected by a change in the level of service. Second, it is thought that the use of decision structuring tools alongside deliberation will lead to higher quality preference construction.

To illustrate these points, a recent studies (McDaniels, Gregory et al. 2003) sought to design a "deliberative" CV approach that helps to overcome the embedding effect in environmental valuation. This study based its modified CV approach on the emerging literature on structured decision making (SDM) approaches designed to help people construct more thoughtful and defensible preferences (e.g., see Arvai and Gregory 2003; Arvai, Gregory et al. 2001; Arvai, McDaniels et al. 2002; Gregory, Arvai et al. 2001a; Gregory, McDaniels et al. 2001b; (McDaniels, Gregory et al. 1999). Structured decision-making approaches, typically applied in small-group settings, provide respondents with both sufficient contextual information about the problem at hand and a series of steps that comprise a more complete basis for the decisions. These steps, based on insights from decision analysis and behavioral decision research include helping decision makers to (Hammond, Keeney et al. 1999):

- define, in a comprehensive fashion, the specific decision to be made,
- identify "what matters" in the form expressed objectives in the context of the impending decision,
- use these objectives as the basis for alternatives,
- employ the relevant technical information to characterize the consequences of each identified alternatives, and
- carry out an in-depth evaluation of the tradeoffs that choosing one alternative over another entails.

McDaniels et al. (McDaniels, Gregory et al. 2003) relied upon these steps applied in a deliberative, small-group setting (which was managed by a trained facilitator) to help respondents gain a better understanding of both the technical context of the valuation decision they were making (one that dealt with obtaining respondents' willingness-to-pay for fisheries improvements on 1 and then 10 rivers; the order of the valuation questions was reversed in a second treatment of the study) and the variety of objectives that deserved consideration when making a decision of this type. Both the decision structuring steps and the deliberation periods took place prior to eliciting WTP judgments from respondents. In the end, McDaniels et al. (McDaniels, Gregory et al. 2003) demonstrated that the implementation of this deliberative CV approach yielded a significant reduction in embedding, which was viewed as an improvement in the quality of the preference judgments compared with a standard contingent valuation (CV) approach.

A similar study was the focus of a recent doctoral dissertation completed at The Ohio State University (Kruse 2005). Here a deliberative CV approach (similar to the one used by McDaniels et al. (McDaniels, Gregory et al. 2003) was compared with a more tradition mailed CV survey. The context for this study focused on respondents' willingness-to-pay for dam removal and its associated benefits in Ohio. The goal of this study was to determine if the deliberative, small-group CV format (again led by a trained facilitator) would help to alleviate some of the common biases associated with a mailed CV survey, namely scenario misspecification (Mitchell & Carson 1995) and tradeoff avoidance (Luce 1998). Scenario misspecification describes the situation where a respondent does not interpret a given valuation scenario in the way intended; this, in turn, leads to off-target WTP responses. The consequences of tradeoff avoidance are equally problematic in that they often lead to the selection of options that reflect the status-quo or the statement of "protest" values (e.g., a WTP of $0 or a refusal to respond).

The Kruse (Kruse 2005) study involved adult respondents who either received a paper-and-pencil survey through the mail or took part in a small-group deliberation that ended with the elicitation of individual willingness-to-pay values. While this study revealed several benefits of a deliberative CV approach (e.g., the male/female ratio of the structured elicitation groups was more representative of the population than the mail survey), the two approaches yielded wildly different WTP estimates. This result was attributed to a strong self-selection bias that was present in the deliberative CV treatment; most subjects who took part in the small-group sessions were steadfast in their opposition to dam removal, which led to a negative WTP. As a result, Kruse (Kruse 2005) deemed the small-group results to be invalid. A follow-up study involving a hypothetical valuation context and student subjects provided equally dubious results; once again, however, sampling issues accounted for many of the problems encountered.

Despite the positive results obtained by McDaniels et al. ((McDaniels, Gregory et al. 2003)), this study as well as the research conducted by Kruse (Kruse 2005) point to some significant challenges associated with the development and implementation of a deliberative CV methodology. As Kruse (Kruse 2005) notes, simply inviting stakeholders to deliberate and then respond to valuation questions is not a precursor to the elicitation of defensible WTP estimates. Indeed, there is the strong likelihood that those individuals that choose to take part in a deliberative elicitation format will come to the table with strong self-interests in mind. Similarly, changes in small-group composition (and therefore, interests) from one meeting to the next are bound to have potentially significant implications on the resulting valuation judgments.

Accounting for these biases during the recruitment of respondents and the organization of deliberations must, therefore, be addressed by researchers and practitioners alike.

In addition to sampling concerns, neither study was able to account for differences in either facilitation style that may result from the use of multiple facilitators, or subtle (or overt) differences from one deliberative session to the next (assuming a single facilitator). Clearly, the role (i.e., active vs. passive, aggressive vs. diffident, etc.) played by the facilitator(s) during a small-group meeting will have an influence on valuation judgments. However, the magnitude of any differences from session to session or facilitator to facilitator cannot be ascertained without further research.

Finally, the valuation judgments obtained via any small-group CV approach will be influenced by the structure of the deliberations. Both McDaniels et al. (McDaniels, Gregory et al. 2003) and Kruse et al. (Kruse 2005) relied upon an extensively studied and practically applied SDM approach. However, the role of alternative decision or deliberative structures (e.g., consensus-based models such as alternative dispute resolution) in the formation of valuation responses can only be determined with further study.

4.7.4 Citizen Jury

Section text to be added.

4.7.5 Civil court jury awards

Section text to be added.

4.8 Economic approaches

4.8.1 Conceptual Foundation

More than a century ago, economists developed the basic notion of "utility," meaning the subjective sense of satisfaction, usefulness or enjoyment derived from consuming goods. Utility is one analytical construct used in economics to describe how choices that people make can be associated with economic tradeoffs.

In characterizing the tradeoff a person would make to obtain a change that improves some feature of the environment (or the compensation required to be willing to accept some deterioration), economists usually express these measures in monetary terms. In the early economic literature these methods were explained using a framework that assumes individuals are attempting to maximize their well-being (or utility) and face financial and time constraints. In this context their choices were described as motivated by the differences in utility they could realize at the margin with the alternatives available given their constraints. Thus for economists, the term "value" is interpreted as the monetary worth of something. More specifically, economic values summarize tradeoffs implied by people's choices. Popular definitions sometimes refer to economic value as "a fair or proper equivalent in money" (Merriam Webster, 2005). The formal details associated with establishing this equivalency for services that are not priced requires a number of assumptions. Whether an individual is "willing to pay" a certain amount for a good is to ask whether that individual would be better off with the good minus the required monetary payment, or not. Human preferences or human utility is the basis for economic measures of value; and prices for marketed goods are taken to measure the amount that an individual would be willing to give up for one more unit of that good. In efforts to measure these tradeoffs , economists gather information from the prices and quantities of goods purchased in markets because the prices that people pay reveal something about marginal evaluations people make as they choose less of one good and more of another good (Freeman, 2003). Using this information to describe tradeoffs at an aggregate level requires that public and private goods be distinguished. In the case of public goods a decision to provide more implies everyone has access to the increase and can consume it without interfering with what is available for anyone else. These features of public goods imply the community level tradeoff is the sum of the marginal values of all its members. For private goods the situation is different and measurement of the aggregate tradeoff requires a specification of the conditions describing how people can get access to the good. Descriptions of community level tradeoffs also can require summaries for situations where some members of the community gain from the proposed action to increase a good and others lose. Adding to the coastal wetlands that are protected requires that some new areas that are not be protected be given this status. As a result the individuals who own them will not be able to use the areas for development. They loose an opportunity that may well have provided appreciable income. Society as a whole gains by providing protection from coastal flooding, habitat, etc. Developing an aggregate measure of the tradeoff implied by this choice requires some principles for dealing with gains and losses.

Environmental economics recognizes that each individual's welfare depends not only on that individual's consumption of private goods, but also on the quantities and qualities of nonmarket goods and service flows from the natural environment. Accordingly, economic methods of valuing environmental good and services can be broadly grouped into two categories: market and non-market choices. Both are used to measure the value of a given change in the environment.

Policies that influence market choices can be expected to influence market prices. These situations relate to those environmental policies that directly affect products or services are bought and sold in commercial markets. In most instances, however, environmental policies have effects on decisions that are made outside markets hence non-market choices must be used as well.

The principal distinction among non-market methods for valuing changes in environmental goods is based on the source of the data (Freeman, 2003). The data can come either from observations of people's real choices (recreation, housing, etc.) or in the form of people's responses to survey questions. Revealed preference and stated preference are the most common terms used to distinguish between these two types of methods. Figure 4-4 Classification of Economic Valuation Approaches below classifies economic valuation methods under these two broad categories.

Revealed Preferences	Stated Preferences
Market Demand and Supply Models for Private goods Productivity Methods	
Non-market Hedonic Housing Price and Wage Models Travel Cost Household production Averting Behavior models	Contingent valuation Conjoint analysis (attribute based)
Benefits Transfer	

Figure 4-4 Classification of Economic Valuation Approaches

4.8.2 Market Based Valuation

Market based valuation can be used to estimate the gains and losses to producers and consumers when ecosystem changes directly affect commercial activities. Fish and timber are two examples of traded market goods whose prices and quantities might be affected by environmental policies. For goods and services purchased in competitive markets, the price of a good reflects the economy's valuation of an extra unit of that good or service. As an example, if a proposed environmental policy has an estimated effect on fish populations, information on the prices and quantities in fish markets can be extracted to calculate an economic value for a policy change.

For small changes, market prices can be used as a measure of value. 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. 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. 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. If the empirical problems are solved, however, the market based valuation can provide an important set of information to environmental decision-making.

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. Another limitation of this method is that market imperfections can confound the measurement of demand and supply and can 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. {Include example from Freeman 1991 on open access fisheries influencing estimates of value}

There are many non-environmental factors that can affect demand and supply relationships that are also important to assuring that policies are authentically reflected in the models. Seasonal variations in use or availability of goods and services related to environmental policies can affect prices, and this needs to be sorted out. 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 effect (or effects) that are hypothesized to reflect how environmental policy influences behavior?

<u>Productivity Methods</u>. Productivity methods, also called production function models, are used to estimate the economic values of ecosystem services used as an input to commercially marketed goods. Wetlands, for example, often contribute as a breeding ground for fisheries and are thus an input to commercial and recreational fishing. The economic benefits of protecting wetlands can then be estimated by their contribution to commercial and recreational fish markets. Similarly, when a river is used as a source of water for a municipal drinking water plant, the river's water quality directly contributes to the production of drinking water. Change in water quality can be assessed for their potential effects on the production and cost of drinking water.

A more recent example of the productivity method can be found in Opaluch (1999). In Recreational and Resource Economic Values for the Peconic Estuary System, Opaluch, et. al. estimated the contribution that the Peconic Estuary System makes to the production of commercial and recreational harvests of fin fish and shell fish, and of birds and other wildlife used for viewing and for hunting. Economic values were assigned to the biological contribution from restoring or protecting increments of each wetland type (eelgrass, saltmarsh, and intertidal mud flats).

4.8.3 Non-Market Methods: Revealed Preference

When market prices and quantities are not directly affected by environmental policies, 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 environmental goods to reveal underlying values. This section covers three revealed preference methods: travel cost, hedonics and household production function models.

Travel Cost. 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 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 opportunity cost of travel time and the out-of-pockets costs of travel are usually labeled the travel costs. They provide a measure of the implicit price for each trip taken by a person who uses a recreation site.

Up until the middle 1990s, most travel cost studies estimated travel costs for the simple case of a new site or loss of 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 recreational resource. In practice most economic models for recreation now use random utility models (RUM) which describe the decision process associated with each individual deciding which recreation site among a number of alternatives to visit picking a site. 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 probable given a person's constraints. The result can be expressed as a function of travel costs, site characteristics, facilities to support specific activities (e.g. boat ramps, ski lifts etc), and users' attributes.

Hedonics. Hedonic methods seek to exploit possible relationships between demands for private goods and their associated bundle of characteristics. The logic for the hedonic method stems from a recognition that prices of a set of closely related but heterogeneous market goods – types of a car or computer -- reflect the characteristics of each type of the product that describe how it is different. When people select from among the set available the hedonic model implies they will choose the type that is their most preferred given its price and attributes. In equilibrium with prices and amounts set recognizing this process the set of prices for these differentiated goods will be structured so there is no incentive for anyone to change their choices. Hedonic price function relating prices to characteristics are reduced form descriptions of this equilibrium condition. The primary applications of this logic in the field of environmental economics involve housing and jobs.

Housing Prices. Assuming that the price of a house reflects the attributes of that house, its property, neighborhood and facilities that are "near" it, then hedonic price function can reflect 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.

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 relates 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. Hedonics has been applied to ascertain the effects of changes in both air quality and water quality on local housing prices.

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 . For example, reduced air pollution may be correlated with higher housing prices but may not be driving a portion of the rise in housing prices. Moreover, relating housing prices to many ecosystem services remains elusive. Finally, hedonic methods assume that consumers operate with full information about local environmental attributes, but this assumption may be unrealistic. Two reviews (the earlier one a meta analysis) indicate clear support for the methods for applications where we can expect buyers and sellers to have knowledge of the local amenities (Smith and Huang, 1995 and Palmquist, 2005).

<u>Wages</u>. One of the most widely used estimates derived from a hedonic framework is the value of a statistical life (VSL). In this case, the hedonic market is the labor market and the matching underlying the price or wage equation is between workers and firms. Attributes of the workers, the jobs, and the locations of the jobs should in principle affect the wages paid. The VSL relies on the recognition by workers that their jobs involve risks of serious injury and/or accidental death. The framework assumes workers must be compensated to work under these conditions and, as a result, the hedonic wage function measures the incremental wage that must be paid in compensation for a worker to be willing to continue to work with a small added risk. The VSL is a shorthand measure for this marginal willingness to accept higher risk – or equivalently the marginal willingness to pay (in the form of reduced compensation) to have an incrementally lower risk on the job.

When calculating VSL, the concern is not with the value of a particular individual's life, but with the value of reducing the statistical probability of one additional premature death. To date, dozens of studies have been published which use labor market contracts to impute VSL. Reported estimates of VSL vary substantially, from $100,000 to $25 million. The EPA calculated a "best estimate" of $6 million for VSL from 26 studies, 21 of which were hedonic labor market studies (EPA, 2000).

In deriving VSL, the key to using hedonic wage studies is to separate the portion of wages associated with occupational health risks from other job characteristics. Since wage rates are affected by numerous factors (including the workers' age, education, experience, etc.), this can be a difficult -- and some say, insurmountable -- econometric challenge. To isolate values for mortality risks, researchers must be able to control for non-fatal risks and a host of other factors. Moreover, the assumptions behind hedonic wage studies can seem quite unrealistic. The assumption that workers are highly mobile ignores practical barriers to moving around the country. Hedonic wage studies further assume that workers make choices with full and accurate information on job risks when, in fact, it is likely that less than 100% of workers are so well informed. A key question for the use of these models is how much mobility and what fraction of the workforce needs to be fully informed for the model to provide a reasonable estimate of the marginal tradeoffs.

Key issues that arise in using hedonic wage models for site specific amenities concern the potential for joint effects in wage and housing price markets and the geographic extent of the area likely to be reflected in the was models.

<u>Averting Behavior Models</u>. Averting behavior models, also called household production models, simulate consumer behavior and rely on the existence of a process that substitutes for the services provided by an environmental resource. The first application of these methods assumed perfect substitution. Subsequent research by Feenberg and Mills (1980) suggested that imperfect substitution might also offer the prospects for revealing measures of tradeoffs people would make for changes in environmental services. To date there have been no applications of their proposed approach. The averting behavior method infers values from “defensive” or “averting” expenditures, i.e. those actions taken to prevent or counteract the adverse effects of environmental degradation. 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 (Freeman, 2003).

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. Averting behavior studies rarely provide economic values for ecosystem services.

Examples of averting behavior studies applied to ecological systems are services are thus far rare. Even for those averting behavior studies for water quality, the motivation for the averting behavior is usually to protect health or life. Similarly, Jenkins, Owens, and Wiggins (Jenkins, Owens et al. 2001) calculated the VSL implied by use of bicycle helmets and find it to be approximately $4.3 million in 2000 dollars for adults who purchase and wear the helmets. They consider their estimate to be a lower bound because buyers (and presumably users) find it worth at least as much as the cost to gain the added protection. Including time and disutility costs would increase the implied value and reinforce the claim that the estimate is a lower bound if only money costs are relevant to the use decision.

Uncertainties associated with averting behavior studies are not limited to the usual statistical caveats but instead have to do with the plausibility of the underlying assumptions.

4.8.4 Non-Market Methods: Stated Preference

Stated preference methods rely on surveys which 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.

Stated preference methods offer the opportunity to measure tradeoffs for anything that can be presented as a credible and consequential choice and hence their primary advantage is flexibility. The most well-known of these methods is contingent valuation.

<u>Contingent valuation</u>. The contingent valuation (CV) method uses hypothetical questions to ask people for their choices involving policies that are described as leading to for an improvement in environmental resources. Alternatively, they can be asked to state their willingness to pay (or their willingness to accept (WTA) compensation) for avoiding a loss (or for experiencing the loss) of an environmental good or service. This method is called "contingent" because people are answering these types of questions contingent on a hypothetical scenario and description of the environmental change. Economic values are defined by the tradeoffs people reveal they are willing to make through their choices. In the case of actual choices, these tradeoffs describe how much of one item a person gives up to obtain a change in something else. Revealed preference approaches for measuring these economic tradeoffs for services that are not exchanged on markets generally rely on some assumptions relating each service to another good, service, or activity that is available on markets and that the analyst can observe people consuming. As a rule, these related services usually have market prices. The nature of the relationship between the market and non-market goods determines how the choices of each market good can be re-interpreted to infer an implicit tradeoff for a change in the related non-market service.

Stated preference methods replace these assumptions with questions that elicit choices from people. The questions are especially important. The framing (or wording) used to present a baseline set of conditions describing the resource of interest, together with the proposed change from that situation replace the analytical assumptions of revealed preference methods.

Estimates for willingness to pay for morbidity related effects of air pollution from stated preference studies have been used in analysis of air quality regulations. The Carson-Mitchell (Carson and Mitchell 1993) analysis of the benefits from contingent valuation from water quality improvements has been the primary basis for all of EPA's benefit costs studies of regulations involving policies to control waterborne effluents. The Chestnut-Rowe (Chestnut and Rowe 1990) analysis of the economic tradeoffs people would make for enhanced visibility at national parks has also factored into EPA analyses.

Concerns about the reliability of values produced by CV studies have been vigorously voiced in the literature. Polasky, et. Al (2005) reviewed CV studies used to estimate values for species conservation and noted that the estimates of willingness-to-pay tended to be higher for more charismatic species and for situations with greater increases in population sizes. Others have noted the lack of distinction between willingness to pay for a single species and willingness to pay for a collection of species. Desvouges et al. (Desvouges, Johnson et al. 1993) found similar estimates for willingness-to-pay for preventing 2000, 20,000 and 200,000 bird deaths. Andreoni (Andreoni 1989; Andreoni 1990)) noted that survey responses may reflect the value of protecting an ecosystem (e.g., the value of old growth forests rather than the value of spotted owls), the environmental more generally, or the "warm-glow" of contributing to a worthy cause.

Other economists have considered the ethical objections to contingent valuation, the belief that environmental protection is a moral duty. Gelso and Peterson (2005) demonstrated the empirical significance of "lexicographic preferences," i.e. cases in which individuals were unwilling to trade off environmental protection for income, and demonstrated a kinked recreation demand function, exhibiting perfectly inelastic behavior over some range of income. Such an approach extended Edwards' recommendation (1989, 1992) for CV researchers to uncover the ethical motives of respondents.

The hypothetical nature of CV has led critics to argue that results are biased and unreliable. Carson, Groves, and Machina (Carson, Groves et al. 1999) have demonstrated that a belief (on the part of respondents) that the proposed change was feasible, along with the recognition that their responses are consequential, is sufficient to assure discrete response questions are incentive compatible.

In response to some of these controversies surrounding CV, the National Oceanic and Atmospheric Administration (NOAA) convened a panel of Nobel laureates and other distinguished social scientists who issued the NOAA Guidelines, a set of recommendations for "best practices" for CV (NOAA, 1993). The NOAA Panel suggested the method requires extensive effort in design of questionnaires – focus groups, cognitive interviews, pre-tests, pilot studies, and surveys. The Panel distinguished a subset of items from their guidelines for special emphasis and described them as burden of proof requirements. In describing the elements with this special focus, the Panel listed particular conditions under which a CV survey would be judged 'unreliable':

- a high non-response rate to the entire survey or to the valuation question;
- inadequate responsiveness to the scope of the environmental insult;
- lack of understanding of the task by the respondents;
- lack of belief in the full restoration scenario;
- 'yes' or 'no' votes on the hypothetical referendums that are not followed up or explained by making reference to the cost and/or the value of the program.'

The disciplines of psychology and survey research have also contributed enhancements to questionnaires that have improved the reliability of CV studies.

The low response rates to mail and telephone surveys have raised costs of data collection. Recent innovations have used paid internet based panels available through commercial firms such as Knowledge Networks and Harris Interactive. However, there is little experience in evaluating the correspondence between responses by "trained" respondents (i.e. the members of these panels) and actual random samples of individuals for the same questions.

Another important additional limitation arises through the stringent restrictions imposed by the Office of Management and Budget (OMB) on the development of new surveys to collect stated choice (or any other information) about household or firm behavior. Under the Paperwork Reduction Act, EPA is charged with limiting these burdens imposed on households and firms. As a result, there can be long delays and/or rejections of requests to conduct surveys. Another important limitation to the method is the time required to design, collect and analyze data, especially given the delays associated with OMB reviews of questionnaires and survey designs. This process usually means new stated preference surveys cannot be considered as approaches for addressing current regulatory needs.

Conjoint Analysis. Conjoint analysis (CA) is distinguished from contingent valuation because the presentation of choices relies on multiple attributes of scenarios. The response is a choice or a ranking from a set of alternatives with each alternative having multiple characteristics or attributes. Respondents are asked to make tradeoffs between prices and other features of commodities to that their marginal tradeoffs can be evaluated.

Conjoint analysis is discussed in further detail on page 53 of this report.

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 that implies 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.

4.8.5 Benefits Transfer

Benefits transfer refers to a class of methods that adapt existing estimates of the tradeoffs people make for changes in environmental resources from one context to another. A benefits transfer is not a new set of estimates for non-market tradeoffs. All benefits transfer methods simply transform existing results. Because either revealed preference or stated preferences estimates can be transferred, benefits transfer is listed in Table _-_ under both of these categories.

As an example, a hedonic property value study based on primary data associated with the sales of residential homes in Chicago can be used to estimate the incremental change in housing prices could be used for another city such as Cleveland, New York City, or Los Angeles.

The particular form of benefits transfer will be determined by the needs of each proposed application. The set of features describing the context for where an estimate is needed is usually described as the policy site. The set of conditions describing the context for the measured tradeoff available from past research is referred to as the study site. Baseline levels of the air pollutants (or more generally environmental quality or services of ecosystems) associated with air quality conditions, the character of the housing (e.g. square feet of interior space, lot size, style, age, etc.), and characteristics of the households may be different. As a result there are implicit and explicit assumptions associated with how the existing research is used to transfer the MWTP to the new city.

Interest in benefit transfer arose from a dearth of information available on the proverbial "research shelf." In addition, EPA and other agencies face significant time and research constraints when preparing economic analyses. It is rarely possible for analysts to conduct original primary research; hence estimates from the existing literature are adapted from one context to another.

The methods currently used in benefits transfer fall in three broad categories – unit value transfers, function transfers, and preference calibration. A unit value transfer usually interprets an estimate for the tradeoff people make for a change in environmental services as locally constant per unit of the change. To illustrate what is involved, suppose the literature has evidence that the average value of the willingness to pay to improve the catch rate (i.e. fish caught per unit of effort) for a sport fishing trip was estimated to be $5 per trip for a 10 percent improvement in this catch rate. One approach for developing a unit value transfer would divide $5 by 10 percent and assume the appropriate value for improvements in catch rate would be $0.50 for each one percent improvement. Another approach would take the same information on average tradeoffs and recognize that the number of fish caught in the study providing the estimated benefit with an hour of effort averaged (before the improvement) as 2. Thus a ten percent improvement implied the typical recreationist would catch 0.2 fish more with an hour's effort. After five hours effort, this change would mean one more fish would be caught on average. Suppose the average recreational trip is a day with about an hour and a half travel time each way. Under these circumstances the improvement implies an average of one more fish is caught during a trip (i.e. assuming 5 hours of "effort" available). These added data of the features of the trips might be used to imply the improvement made "typical" trips yield added incremental benefits of $5.* Alternatively, the conclusion could be made that added fish caught during a typical trip would be worth $5. For the study site all three interpretations are simply arithmetic transformations of the data describing the context for the choices that yield the tradeoff estimates.

These same conclusions do not hold when they are transferred to a different situation. Suppose the policy site concerns the entrainment of fish in the cooling towers of power plants. Assume further it was known from technical analysis that this regulation would lead to 5 percent improvement in fishing success along rivers affected by a rule reducing fish entrainment. If these areas have 2,000 fishers, each taking about 3 trips per season and currently they catch 1 fish per hour, Figure 4-5 Examples of Unit Value Transfer displays the alternative unit value transfers.

Assumption	Unit Value	Interpretation of Policy	Aggregate Value
Constant Unit value for a percentage improvement	$0.50	5% improvement per trip	$2.50 * 3 * 2000 = $15,000
Constant Value for an "improved" trip	$5.00	improved fishing trips	$5 * 3 * 2000 = $30,000
Constant Value for an extra fish caught per hour of effort	$25[2]	added fish caught	$25 * .05 * 1 * 3 * 2000 = $7,500

Figure 4-5 Examples of Unit Value Transfer

Clearly these examples are simplifications of the real world. Trips may be different – longer, require more travel time, or involve different features such as different species or related activities. Neither is it feasible to assume that fishing success induces existing recreationists to take more trips. The sources for error in the transfer compound under these possible outcomes. Even without such complications, these simple examples illustrate how the aggregate benefit measures differ by a factor of four. Moreover none of these adjustments take account of any behavioral changes that might be expected in response to the example policy (e.g. the people taking more trips or more people participating in fishing).

The function transfer replaces the unit value with a summary function describing the results of a single study or a set of studies. For example, a primary analysis of the value of air quality improvements might be based on a contingent valuation survey of individuals' willingness to pay to avoid specific episodes of ill health (i.e. a minor symptom day such as a day with mildly red watering itchy eyes; a runny nose with sneezing spells; or a work-loss day described as one day of persistent nausea and headache with occasional vomiting).* A value function in this context would relate the responses to these questions to the sample respondent's income, health status, demographic attributes, and other features describing factors that might influence their responses such as health insurance. For the policy site the relevant (and available) values for these factors would be used to estimate an "adjusted" measure based on the specific conditions in the policy area (see Brouwer and Bateman 2005) for another example in the health context).

Function transfer approaches can also transfer estimated behavior models. This is needed when for random utility models describing revealed preference choices. The demand model or

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[2] This was computed assuming $5 for 0.2 fish added caught per hour of effort or ${}^{\$5}\!/_{0.2} = \$25$ per fish per hour.

random utility model description of choices would be transferred and then used to estimate benefit measures.

A different type of function transfer involves the use of statistical summaries of existing research. These meta-analyses can be undertaken where there is accumulated evidence on measures of economic tradeoffs for a common set of changes in resources or amenities. One area with a large number of applications is water quality relevant to recreation (see Johnston et al. 2003 as an example of meta-analyses for water quality and Smith and Kaoru 1990a, 1990b for other recreation-based meta-analyses). This strategy was used recently in EPA's assessment for the Phase III component of the 316B rules.

The preference calibration approach assumes that the objective of a tradeoff should be to first identify the parameters of a preference relationship required to measure the tradeoff required for a policy application. In this context, benefit transfer becomes an identification problem. That is, the first step is to ask whether with a specified preference relationship there is sufficient information in existing estimates to isolate measures for the parameters required to estimate the tradeoffs associated with the required benefit measures. This complex question reverses the logic used in analytical defining a benefit measure. This technique imposes specific requirements on the information from existing studies. As a rule, these information needs are defined by the tradeoff concepts measured in the literature (see Smith et al. 2002]for an example). When the parameters can be calibrated or estimated from the existing literature, the transfer involves using the calibrated preference function together with the conditions at the policy site to measure the tradeoff for the change associated with the policy application.

The evaluations of benefit transfer in the literature on the economic measures of the benefits estimated for changes in environmental resources are uniformly negative, however these opinions may not fully appreciate the diversity of benefit transfer studies and the constraints under which government agencies operate. A realistic assessment would require case-by-case evaluations of the assumptions and steps used in the transfer.

4.8.6 Other: Replacement Costs, Tradable Permits

All valuation methods can be misconstrued, applied incorrectly and misinterpreted, however two approaches require particular caution: replacement costs and the use of certain environmental markets, e. g. tradable permits. In the case of replacement costs, there are particular conditions, explained below, under which replacement cost can approximate the economic value of environmental protection. 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.

<u>Replacement Costs.</u> The replacement cost method does not fall neatly into either revealed preference or stated preference. 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 those ecosystem services. For example, an estimate of the value of conserving 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 service. The replacement cost approach should not be confused with applications of "averting behavior" methods of estimating values which are

based upon observed voluntary behavior.

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. If so, 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 economic value from loss of the ecosystem service.

In sum, replacement cost can be a valid measure of economic value if three conditions are met: the human-engineered system must provide services of equivalent quality and magnitude, the human-engineered system must be the least costly alternative, and individuals in aggregate must be willing to incur these costs if the natural service were not available (Shabman and Batie, 1978, Bockstael, Freeman et al. 2000). Examples of use of replacement cost include estimates of the value of protecting watersheds for municipal drinking water supply, the most famous of which is the Catskills watersheds (see NRC chap 5 for references). Avoided cost of illness is an approach EPA has used successfully to account for certain human health benefits of environmental regulations.

Tradable Permits. 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 SO2 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.

4.9 Decision-Making and Communication Approaches

4.9.1 Conceptual Framework for the Decision Science Approach to Values

The decision science perspective on valuing the protection of ecological systems and services is, at its core, relativist. Form this perspective, the "value" surrounding ecological systems and services is not an absolute concept, despite the fact that numerical and narrative descriptions of individual components of it (absent a comparison) may be obtained using a variety of economic and non-economic (e.g., psychological, biophysical, etc.) methods. Instead, the decision sciences take the view that that the overall value that is ascribed to the environment and its services can only be fully understood in a comparative context; in other words, we can only say that a system—or indeed the suite of services provided by that system—has a high or low value in the context of:

a. retrospective evaluations undertaken by analyzing the degree of change experienced by the system relative to some previous or unaltered state (i.e., a system is either more or less valuable because it performs either better or worse than it did before), or
b. decision making for management undertaken by comparing predictions about how a system or its suite of services might behave—again better or worse relative to its current condition—after it has been subjected to one or more possible management or regulatory options.

The attributes across which these changes are accounted for are defined by the objectives of a given decision context. These objectives tend to be diverse and simultaneously incorporate inputs from a wide variety of disciplines. It is not atypical, for example, to ascribe an overall relative value to an ecological system or service based on the extent to which it maintains some requisite level of ecological function and productivity, provides security for endangered or threatened species, facilitates the maintenance of key services such as nutrient cycling or decomposition, yields economic outputs in the form of resource extraction and tourism, lends itself to desired recreation opportunities, and supplies a sense of pride or awe (Gregory 2000). In this sense, the decision sciences straddle the line between economic and non-economic approaches to valuation in that inputs for a formal comparison of options in the case of management decisions, and current and previous conditions in the case of evaluation, are required from fields such as economics, ecology, psychology, and sociology. However, absent an explicit framework for comparison across attributes, and options or alternative states, individual inputs from these sources have very little meaning in their own right.

Thus, a decision science approach to valuing the protection of ecological systems and services is explicitly multiattribute in nature. Absent this multiattribute view of value—with the various attributes of value tied to the concerns stated by stated by technical experts and other key stakeholders—the relative values obtained often fall short of providing the requisite guidance for decision making and evaluation, and run the risk of not meeting or surpassing the threshold of relevancy (Keeney & Raiffa 1993)—defined chiefly by those who will hold decision makers and agencies accountable. Of course, a multiattribute and comparative view of value presents challenges to decision makers and evaluators. For example, those who undertake valuations geared toward the decision sciences must be prepared to work with multiple and diverse stakeholders sometimes over extended temporal periods, conduct additional decision-specific technical analyses that are linked to stated objectives, and address complex and often contentious tradeoffs (Arvai et al. 2001; Gregory et al. 2001; Hammond et al. 1999; Keeney & Gregory 2005).

4.9.2 Deliberative approaches (e.g., mediated modeling, decision-aiding approaches): how they can integrate different kinds of information

Significant interest has been devoted to multi-stakeholder, deliberative processes for environmental decision making both at EPA (e.g., (Agency 2000; EPA 2000) and elsewhere (e.g., (Beierle and Cayford 2002),(Beierle 2002). Much of this interest has focused on deliberative processes as a means of legitimizing resulting policy decisions. To this end, there have been several examples of both research and practice where deliberative approaches to decision making have resulted in a high degree of participant satisfaction in a variety of different management contexts (McDaniels, Gregory et al. 1999), (Arvai 2003)). Results from these studies, and others (e.g., (Kraft 1988), (NRC 1989),(Heiman 1990) (Vari, Mumpower et al. 1993)), argue that people are more likely to accept outcomes that result from decision making processes that seem fair, reasonable, and amenable to allowing the public and other stakeholders an opportunity to voice their feelings and concerns.

This argument is also in line with writing on “procedural justice”, which suggests that a higher degree of acceptance is be expected for decisions that seem fair to the affected parties from the point of view of both the decision outcome and the process that resulted in it (Lind and Tyler 1988), (Kraft and Scheberle 1995). In other words, people whose individual interests are adversely affected by an outcome may be more willing to accept decisions because they perceive that they have been dealt with fairly, they understand the other participants’ positions, and they have had the opportunity—even if comes indirectly—to contribute to the debate (Syme, Macpherson et al. 1991, Hillier 1998)).

Why does this positive relationship between deliberative processes and support for resulting decisions exist? Some have suggested greater stakeholder satisfaction results from a frame shift during decision making from one that is imposed to one that is voluntary ((Slovic 1987)). Others have suggested that greater stakeholder satisfaction with decisions that are the product of deliberative approaches is simply the manifestation of a halo effect ((Thorndike 1920)). In this case, people tend to judge multiple dimensions of a stimulus in much the same was as they judge the most salient dimension. In other words, when one judges a decision to be “good” in one dimension (i.e., because it was made in a deliberative fashion), they are also likely to judge the same decision to be good in other dimensions (i.e., the outcomes of that decision). Beyond these “stakeholder relations” benefits, there are other reasons—reasons that are of greater interest to this committee—for advocating the use of deliberative approaches for valuation and decision making. Foremost among these is the fact that these approaches work to foster the inclusion of differently formulated objectives, concerns, and arguments in the valuation and decision making process (NRC 1996, Chess and Purcell 1999, Renn 1999, Gregory 2000)).

Indeed, EPA itself has acknowledged this point, stating in the past that the American people are the agency's primary "customer" and to this end issued the following policy statement (EPA 2000, p. 1): "We are committed to providing the best customer service possible. We aim to achieve this through increased public participation, increased access to information, and more effectively responding to customer needs." This is a sweeping statement that applies to a wide variety of valuation contexts, including both those that involve single valuation metrics (e.g., dollar responses obtained via contingent valuation) and multiattribute inputs obtained via multi-stakeholder approaches (e.g., such as mediated modeling and structured decision approaches).

For example, in the context of contingent valuation, a commitment to deliberative approaches implies that EPA will seek input from stakeholders regarding such things as:

- the ecological systems or services that will be the subject of valuations,
- the aspects of these ecological systems or services to be valued (e.g., the attributes by which an object such as aesthetic quality might be defined), and appropriate ways to frame and implement CV questions.

Likewise, in the context of multiattribute approaches, this commitment guides EPA to seek input regarding:

- problem identification and framing,
- stakeholders' objectives as they relate to a given decision or evaluation context,
- the range of options that may be considered as part of a management decision,
- valuation inputs to consider during decision making or evaluation; these include results from valuation processes that include, but are not limited to CV, deliberative value elicitations, and the results from (non-monetized) surveys, and information about the tradeoffs that exist when selecting one option over another.

4.9.3 Net Environmental Benefit Analysis Framework (NEBA)

The net environmental benefit analysis 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 ecological services 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 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.

4.9.4 Communicating the Results

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 represent the value of protecting them. Numerical information is meaningless without context and framing (reference Paul's discussion of framing here?).

Communicating the value of protecting ecological systems and services) is a fundamental part of EPA's mission. To inform policy choices, justify budget expenditures, and educate people about the environment requires that EPA do so.

Accordingly, the responsibility for communicating the value of protecting ecological systems and services is distributed throughout the EPA. Like other federal agencies, EPA has for several years been required to report on its programmatic performance, first through Government Performance and Results Act and more recently through the Program Assessment Rating Tool (PART). [Add discussion of reports to OMB and how values have been communicated in those – or would this duplicate other parts of the VPESS report?] In addition, EPA recently released its first Report on the Environment (ROE). More generally, results of research and program investments by the U.S. EPA are communicated through its Office of Public Affairs (OPA), which resides in the Administrator's office, the Office of Environmental Information, and individual scientific reports and publications. The Office of Public Affairs includes several subsidiary offices and a National Press Secretary. Each regional EPA office also has its own newsroom.

The Assistant Administrator for Environmental Information heads the Office of Environmental Information (OEI), located in the Administrator's office. The Office of Environmental Information: "enhances environmental data and information collection; manages EPA's information technology (IT) policy, infrastructure and oversight of Federal and Agency information technology (IT) Statutes, Regulations, and Standards; develops and implements policies for improving public access to environmental information; and oversees EPA's quality-related procedures and policies for environmental programs."

Both OPA and OEI might appropriately take the lead responsibility for communicating the VPESS. Values in this context often have a technical component, and so fall into the category of environmental data and information collection. In this regard, improving communication of VPESS fits within the scope of OEI efforts to improve public access to environmental information. The National Center for Environmental Economics could be a critical partner in this effort, given their relevant technical expertise. Values associated with protecting ecological systems and services are also of general interest. Given that they reflect on performance of the Agency as a whole, they should be communicated by the Office of Public Affairs, as well as individual programs within the Agency. Communicating such values is critical to how effectively the Agency carries out its mission, as well as to how it is assessed.

The potential interested parties for values associated with protecting ecological systems and services 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. (refer to the assessment of audiences for values elsewhere in this report, if there is one?) They will need to access technical details and models, as well as resulting value estimates, and so would be better served by OEI.

Effective values communication requires that systematically support interactions with interested parties and their participation, the character of which interactions 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 the ROE) are also important, especially in the context of assessment. Even for reports and indicators, having an empirical, end-user-informed basis for the design is critical. End-user engagement is itself an example of a participative process, requiring due consideration of such issues as sampling and representation.

The focus of the value discussion in the NRC report (NRC 2001) and SAB review of the ROE (EPA 2003) and related literatures (e.g., (Failing and Gregory 2003)) is not on dollars per se, but on ends and decision/management objectives, that is, qualitative expressions, or a wider variety of expressions of value - not just monetary expressions of value. Value elicitation is not restricted to contingent valuation, but includes qualitative expressions, participative discussions and narrative expressions of value, defined by the identification of associated ends, and the means to achieve those ends. Simple summary indicators are recommended by some (e.g. (Schiller, Hunsaker et al. 2001); (Failing and Gregory 2003)); others emphasize disaggregating indicators (US EPA SAB 2003) where necessary.

Communicating the value of protecting ecological systems and services requires conveying not only value information, but also about the nature and state of the ecological systems and services to which they apply. 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, for example those by Costanza et al (add refs), 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 Report on the Environment and REMAP reports illustrate a range of representational approaches.

It's critical to communicate ecological processes as well as static information or states; the SAB review of the ROE and several of the articles/authors cited below (e.g., (Schiller, Hunsaker et al. 2001); (Carpenter and Hanson 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).

Issues of scale and aggregation are also important. Both the NRC report (NRC 2001) and the SAB review of the ROE (EPA 2003) emphasize the importance of using regional and local indicators – of not aggregating information data to the point where it obscures critical ecological threats/problems.

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 represent the value of protecting them. Numerical information is meaningless without context and framing (reference Paul's discussion of framing here?).

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5 COMMON METHODOLOGICAL ISSUES FOR BENEFITS ASSESSMENTS

5.1 Criteria for scientific measurement systems

All methods for assessing the benefits of changes in ecosystems/services must address certain fundamental measurement issues, as well as issues that are specific to the measurement of values. Measurement is the process of assigning meaningful numbers to the properties of things (e.g., heights of people, sweetness of soft drinks, popularity of sporting events, biomass production of ecosystems). The numbers assigned may only indicate relative amounts of the targeted property (i.e., an ordinal scale of measurement) or they may provide more precise information about the magnitudes of the property that are held by the objects being measured. Some measurement methods allow quantitative differences in the measures to be interpreted as indicating equivalent (proportional) differences in the property being assessed (an equal interval scale of measurement). At the highest level of measurement ratios of the measures can be taken as indicating equivalent ratios in the property being measured. That is, it is appropriate to conclude that an object with twice the measure of another has twice as much of the property. In addition to meeting the equal interval/differences requirement of the interval scale, ratio scales must have an absolute zero, i.e., a measure of 0 must indicate that the object measured has absolutely none of the property being measured. Measures of length and weight and Kelvin measures of heat (but not Celsius) are classic examples of ratio scales. Ratio scales are rare in social sciences, which more often must settle for “roughly” interval scales of measurement. There is controversy about the extent to which measures of value can achieve a ratio scale.

Any science-based measurement method is expected to be based on and to be testable by observations that can be reliably repeated, both within and between different applications of the method. Counterbalancing the reliability criterion is the need for the measurement system to be sufficiently precise to serve the purposes for which it is intended. Precision and repeatability are typically assessed by standard reliability or agreement statistics. In addition measurement methods should be valid—that is they should provide measures of the property that they purport to measure. There is no one test that can affirm the validity of a measurement method. Rather incremental evidence of validity is accrued by demonstrations that objects with known (or assumed) levels of the property of interest are assigned appropriate measures (e.g., “hot” places get higher measures of heat, “smart” people get higher measures of intelligence), measures for a given set of objects by the measurement system properly correspond (correlate) with other independent measures of (nominally) the same property for those same objects (conjoint validity) and finally measures should correspond as expected (based on theory and prior empirical experience) with measures of other properties (construct validity). By some accounts the ultimate criterion for measurement systems is utility—the extent to which the measures produced effectively guide decisions about and actions toward the measured objects. Precision, reliability and validity criteria can be seen as instrumental to achieving utility.

5.2 Special criteria for value measurement systems

The measurement of the "value" of objects, events or states of the world must meet the basic standards for any scientific measurement, but there are some additional issues that must be addressed as well. First, the value of an object is not a property of the object, except perhaps in the more extreme notions of intrinsic value. Even when the value of an object is in principle taken to be intrinsic or an end in itself, as a practical matter that value must be assigned (and perhaps advocated) by some person or persons; the value is not a property of the object per se (see previous C-VPESS report, *Valuing the Protection of Ecological Systems and Services: An Expanded and Integrated Approach*). Most often the value of an object is based on the contribution that the properties of the object (e.g., its nutritional, structural, or energy producing properties) make (or are expected to make) toward meeting some individually or socially defined goal. Value is determined outside of the object and assigned to it based on moral, aesthetic or utilitarian considerations. Thus, the assessment of the value of an object is not directed at the properties of the object itself, but at the relationship of those properties to some goal, desire or need that is external to the object.[3]

5.3 Projected effects and anticipated consequences

Value measurement requires a determination of the relationship between changes in the relevant properties of the target object and their effects on some individual or social goal(s). In the context of EPA's assessments of the benefits of changes in ecosystems/services this requires both determining the relevant bio-physical changes in the ecological systems and how those changes can be expected to affect things that matter to people. Thus, some characterization of the existing and projected states of the targeted ecosystems/services is fundamental to all benefits assessments. It is important then that bio-ecological changes, the projected ecological endpoints, be characterized as accurately as possible and in such a way that they can be effectively linked to effects on things that people care about. Following the mutual interaction model for EPA decision making described above (Figure 2-1: Conceptual Diagram of Major Components of Valuation: Decision Approaches) benefits assessments may have to be extended to address important bio-ecological effects for which the links to human values are not well understood. 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.

[3] Of course, post-modern philosophers would point out that even the most basic properties (length, weight, etc) cannot be measured or known in any direct way. Indeed the notion of "objects" existing at all and being composed of or analyzable into distinct properties is a human construction imposed on a world that does not necessarily have either objects or properties. We acknowledge, but certainly will not attempt here to resolve the positivist-constructivist philosophical controversy. Rather we will assume, as a practical matter in the context of EPA environmental policy making, that objects and the environments and ecosystems they compose do exist, that their properties (however well or poorly these are known) do change in response to our actions and inactions (however uncertain we may be about those effects), and that it is meaningful and useful to measure both their basic properties and their values to the best of our ability.

In addition to the uncertainty introduced by bio-ecological projections, there are also uncertainties in the individual and social consequences of ecological changes. Part of this uncertainty arises from the fact that we are rarely able to determine with precision and certainty what the individual or social consequences of a given change in ecosystems/services will be. In addition, all assessments of expected consequences are about anticipated, not experienced satisfaction those consequences might bring. For 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).[4] 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.

5.4 Whose values, judged by whom?

For any anthropogenic values it is necessary to determine who is appropriate to assign those values. In most cases relevant to EPA decision making, values are assigned by, or on behalf of the citizens of the US. Often the relevant value assignments will be made through representative legislative or legal processes. In some cases experts may derive value estimates through analysis of relationships of projected changes in ecosystems/services to known or assumed social goals, such as protection and promotion of human health, economic prosperity, natural resources conservation or ecological sustainability. In other cases it may be appropriate to base benefits assessments more directly on the judgments of a sample of citizens. Depending on the nature of the ecological changes and agency actions under consideration, the relevant sample might represent a relatively narrow population of individuals, local governments or industries that are most affected. For actions that are national in scope a sample representative of all US citizens might be most appropriate.

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[4] This account is consistent with a rational model of decision making. Other models are possible, and indeed research in psychology and decision sciences indicates that very often valuations and choices are driven by rather automatic emotional responses, and by moods, habits and features of the immediate context that have little to do with the characteristics of the alternatives nominally under consideration or any careful or wide-ranging appraisal of their anticipated consequences.

In addition to determining who is to make value assignments to projected changes in ecosystems/services and/or their expected social consequences, it is also necessary to specify on whose behalf these judgments are to be offered. It is typical in assessments of the value of goods and services that are consumed by individuals that value judgments represent only the judge—that is, a self constituency is designated. In other cases the judge may speak for his/her household. For many of EPA's assessments of the benefits of changes in ecosystems/services the more appropriate constituency may be a larger social unit, such as current citizens of the US or both current and future citizens. This would add another element of uncertainty to the valuations, as the judge must try to estimate how others would feel about and respond to the projected ecological changes and their social consequences. [5]

5.5 Defining benefits assessment targets

At one level of analysis the EPA may wish to assess the value of a particular policy (or a set of alternative policies) for protecting some designated ecosystems/services. Alternatively, the target of the assessment might better be described as the set of bio-ecological outcomes/endpoints of the implementation of that policy or policies. From the public/stakeholder perspective, however, it may be more appropriate to define the assessment target in terms of the personal and/or social consequences of the biological changes projected to result from the policy/policies. In the latter case, it may still be necessary to include the means by which given outcomes are to be achieved (e.g., through voluntary compliance versus legal mandates, by private agencies versus government, by artificial contrivances versus alteration/protection of natural ecosystem processes). Additional considerations in defining the assessment target include whether and how uncertainties in policy implementation, ecological outcomes and/or social effects should be included, and whether/how places, times and people affected should be specified.

5.6 Representing targeted changes in ecosystems and services

Whatever changes in ecosystems/services are at issue in EPA decision making, they are rarely available for direct observation or experience, and very often they are only (uncertain) projections of possible future ecological effects and their expected social consequences. In this context, the target for benefit assessments may only be expressed in relatively abstract scientific and statistical terms. Such representations are rarely appropriate or sufficient. The most common means of representing targets for benefits assessments is some verbally narrative, perhaps supported by quantitative tables or graphics. A key concern here is whether the verbal descriptions adequately and accurately convey the critical, value- and policy-relevant aspects of the target they are intended to represent. A balancing concern is that the description is not so long and complex as to require extensive effort to understand it, or to be altogether incomprehensible to those who are being asked to make the value judgments. Balancing detail,

[5] Interestingly, when the constituency for a value assessment is not human, as when an ecosystem or some plant or animal species is assigned value for its own sake, a judgment must be made about what is good for that entity, further extending the distance between the judge and that for which the value is assumed to apply.

completeness and accuracy of representations with comprehension and efficiency of communication is particularly difficult when the bio-ecological effects/endpoints are unfamiliar and the relationships to relevant social consequences are remote and very complex. In some circumstances it may be possible and advantageous to create visual or multimedia simulations of targeted ecosystems/services changes. These representations too are subject to concerns about accuracy and sufficiency. Ecosystems and the services they provide frequently involve multiple biological and social variables interacting over a wide range of time and geographic scales. Adequately representing such complex relationships, with all of their associated uncertainties, to untrained audiences presents a daunting task for any benefits assessment method.

5.7 Selecting appropriate expressions of judged value

Concurrent with determining the means for representing the assessment targets, a suitable method must be devised for the judge to express his/her value judgment. In the case of economic assessment methods, for example, values are typically expressed in the form of a monetary payment in the context of some market. Changes in ecosystems/services are rarely if ever directly traded in any market, so it is usually necessary to infer willingness to pay indirectly, as from purchases of commodities that are somehow related to the targeted ecosystems/services, or to solicit expressed intentions to pay based on some contrived hypothetical market. Social/psychological assessment methods typically solicit “attitude” ratings (such as *preference*, *liking*, *importance*, *satisfaction*, or *acceptance*) or forced choices among multiple offered options (with the basis of choice specified as *preference*, *liking*, etc). In theory, value judgments should not be affected by the method of expressing that value, and a high level of agreement should be attained between assessments based on payments (direct, indirect or hypothetical) and assessments based on ratings or choices. In practice, the results of economic and attitude assessments for the same objects by the same respondent populations have sometimes been found to be strongly positively correlated, implying some validity for both methods. However, low or even negative correlations between economic and attitude measures can occur, even for the same respondent within the same survey. In these cases the assessed value of an object/event appears to depend significantly upon the method used. Such differences are most likely to occur when the objects being evaluated are not usually (or ever) bought and sold in markets, or where respondents find dollar valuations inappropriate or unethical.

5.8 Bringing valuer and to-be-valued together

When assessments of the benefits of changes in ecosystems/services are to be based on the judgments of a sample of some population of people, representations of the assessment targets must be brought together with the sample. Assessment targets, if they yet exist at all, are rarely directly accessible to all relevant members of the population of judges. As noted above, ecosystems/services are rarely or never traded in any market where values might be more or less directly revealed. Thus, economic assessments must generally rely on inferring values indirectly based on presumed relationships to purchases of other goods or services in some other market. In many cases, economic values will be derived from expressed intentions to pay in hypothetical markets, a survey procedure that is essentially identical to social/psychological attitude assessments. Typically, individual respondents for economic or social/psychological surveys are contacted by telephone, mail, internet or directly intercepted at their home, work or other locations, where representations of the assessment targets are presented and expressed values are recorded. The above steps of defining the relevant population of judges/respondents, developing appropriate representations of assessment targets and determining how values are to be expressed interact together and with the means of bringing valuer and valued together. Very often options are constrained by the available budget and the time within which the assessment must be completed. In practice options have most often been limited to telephone or mail, which in turn has limited the means of representing the assessment targets and recording responses—usually to short verbal descriptions on the one hand and expressed willingness to pay (bids) or ratings or choices on the other.

5.9 Spatial Data Issues (Jim Boyd)

5.10 Issue of consistency between stated preference and revealed preference

When the results of revealed-preference and stated-preference approaches are compared, the stated-preference approaches tend to yield higher valuations of eco-system elements. These comparisons have led to criticisms of the stated-preference approaches as being systematically biased in over-estimating environmental value. This raises three questions: 1) Do stated-preference results over-estimate environmental values? 2) Do biases occur with sufficient consistency to permit a general adjustment factor? 3) Can the biases, presumably toward over-estimation, be eliminated or at least reduced?

There are at least two reasons to expect that stated-preference approaches will exaggerate eco-system values. First, pro-environment respondents may believe that exaggerating their expressions of willingness to pay would increase the valuation and increase the government's commitment to the environment. Second, people may simply over-state their hypothetical willingness to pay in order to gain the esteem of the interviewer.

Determining whether stated-preference approaches over-state or under-state "true" values requires a benchmark of correct, or at least more correct, values. This has been approached in two ways: by comparing actual willingness to pay (often from small-stakes experiments), and, more rarely, by determining whether the predictions from stated-preference willingness-to-pay results are consistent with actual referendum or initiative results.

Meta-analyses do show that stated-preference approaches yield higher estimates of eco-system values. However, the magnitude of bias is not consistent. One meta-analysis found a positive bias of more than 300% in 34 of 39 stated preference results ((Harrison and Rutström 1999)); another found an average bias of only 35% (Murphy, Allen et al. 2003). This variation may be due to the fact that different framing of stated preference questions can elicit different responses. In addition, some stated-preference results reflect the efforts to reduce the most potentially distorting features of stated-preference surveys, for example by providing more balanced information or by reducing deliberate bias (Loomis, Brown et al. 1996), while others do not; we should not be surprised that the biases vary greatly.

These variations in magnitude of bias make it infeasible to apply an automatic adjustment to offset this bias. It has been suggested that the issue of exaggeration can be addressed by using a given adjustment to estimate the unbiased value; (Arrow, Solow et al. 1993) suggest dividing hypothetical results by two. However, the assessments mentioned above show no consistency that would justify a consistent adjustment factor.

It is also possible that the discrepancies between hypothetical stated preferences, and "real" preferences as revealed by market prices, do not reflect "errors" in the stated preference results, but rather that they are measuring a different—but arguably equally legitimate—sense of value. The market prices generally reflect decisions to maximize private utility of individuals and their families, whereas the stated preference elicitation may be measuring the public-regardedness aspects as well.

Additional Key Citations:

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6 Uncertainty

6.1 Uncertainty

6.1.1 Introduction

Ecosystem valuation efforts are inevitably subject to uncertainty, and unless such uncertainty is taken into account and thoughtfully conveyed to decision makers, the ultimate usefulness of respective assessments may be compromised. 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. Because any given policy may result in a range of different outcomes, decision makers must be provided with sufficient information about the distribution of probabilities 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, they have to incorporate the uncertainty that policy choices always entail.

In order to assess how much confidence to attribute to the projections involved in the valuation, decision makers must also be informed about 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.

Uncertainty is by no means limited to a single method or group of methods of ecological valuation. Rather, uncertainty is the rule, not the exception in this domain. For example, in general, revealed-preference methods of ecological valuation - such as random utility models - may involve smaller degrees of uncertainty than stated-preference methods, such as contingent valuation, because the former are based upon observations of actual market behavior while the latter are based upon responses to hypothetical questions. But revealed-preference approaches are themselves subject to uncertainty from various sources; indeed their statistical nature makes the expression of this uncertainty explicit. Furthermore, estimates in the bio-physical domain - which are at the heart of any ecological valuation - are themselves subject to tremendous uncertainty, which may be greater than or less than uncertainty associated with valuation per se.

It is important to distinguish between precision and accuracy, and likewise between imprecision (uncertainty) and inaccuracy. In a quantitative, statistical context, the contrast is captured, respectively, by measures of variance and bias. Our focus is exclusively on uncertainty, that is, precision or the lack thereof. Some existing and potential methods of ecological valuation may involve systematic bias, but that is not the subject of this essay. An exception is that we note that individual methods of ecological valuation may display inherent tradeoffs between precision and accuracy. For example, an attempt to make questions (and the implied budget constraint) in a contingent valuation analysis more realistic and credible can have the desirable effect of increasing precision (reducing uncertainty and variance in responses) while also having the unfortunate effect of bringing about truthful revelation of preference problems, that is, introducing strategic bias.

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. Related to this, how can uncertainty in ecological valuations be incorporated in various decision-analytic frameworks, such as benefit-cost analysis, cost-effectiveness analysis, and multi-criteria analysis? This raises the question of whether specific institutional obstacles to conveying uncertainty currently exist (and, if so, how they can be overcome), and likewise whether some institutional attributes currently favor the analysis and communication of uncertainty in ecological valuations. A third and final key question is associated with the types of research - data collection, improvements in measurement, theory-building, theory validation, and others - that can be pursued to reduce uncertainty for particular sources in specific applications.

Section 6.10.2 describes the major sources of uncertainty in ecological valuations. Section 6.10.3 examines the potential for formal 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 6.10.4 focuses on issues associated with communicating uncertainty to policy makers (and other audiences) in ecological valuations. Section 6.10.5 assesses the potential value of uncertainty assessments to the U.S. Environmental Protection Agency.

6.1.2 Sources of Uncertainty in Ecological Valuations

Valuation of the benefits of proposed public policies entails two analytic tasks, each potentially subject to uncertainty: forecasting biophysical outcomes and valuing those outcomes. It might be tempting to limit attention to the uncertainty of valuation per se, but both sources of uncertainty 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 two essential steps in the valuation process is fundamentally an empirical question.

The first stage - predicting the biophysical impacts of some public policy (relative to a predicted business-as-usual biophysical baseline) - typically involves three conceptually distinct but interrelated sources of uncertainty: (1) limitations due to data quality (lack of data, faulty data, or data of variable quality in particular contexts); (2) stochastic (random) variation, also known as within-model uncertainty (i.e., variation beyond analysis, such as temperature fluctuations caused by solar flares); and (3) theory limitations (across model uncertainty).[6] The second stage - valuation of biophysical impacts - is likewise subject to the same three sources of

[6] In the measurement of a relationship between two phenomena, one example of such uncertainty is the possibility that the measurement did not account for a third, confounding factor that is responsible for some of the perceived relationship between the two phenomena. A measurement that incorporates that third factor may result in a different estimate of the relationship

uncertainty. [2][7] 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.[3][8] The consequences of stochastic variation can be assessed in simple models by observations of measures of dispersion (for example, variance of estimates), whereas in more complex models, stochastic simulations - Monte Carlo analysis - can be employed (Jaffe and Stavins 2004).

6.1.3 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, which can be incorporated in a Monte Carlo analysis. Therefore over the years, the use of formal quantitative uncertainty assessment, and in particular Monte Carlo analysis, have been shown to provide reliable and rich characterizations of the implications of uncertainty, and therefore have become common in a variety of fields, including engineering, finance, and a number of scientific disciplines.[4][9] 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.

Monte Carlo analysis can be implemented with relative ease. The first step is the development of probability distributions of uncertain inputs (to an ecological valuation or other analysis), where the probability distributions reflect the implications of uncertainty regarding respective inputs for the range of its possible values and the likelihood that each value is the true value. Once probability distributions of inputs to an ecological valuation are established, a Monte Carlo analysis determines the resulting probability distribution of the valuation by carrying out the calculation thousands, or even millions, of times. With each iteration of the

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[7] For a discussion of methods of characterizing these uncertainties, see Morgan and Henrion (1990).
[8] Weyant *et al.* (1996) use a similar approach to assess the divergence among integrated assessment models of global change.
[9] This section of the paper draws, in part, on: Jaffe and Stavins 2004

calculations, new values are randomly drawn from each input's probability distribution and used in the calculation of the ultimate ecological valuation. Over the course of these iterations, the frequency with which any given value is drawn for a particular input is governed by that input's probability distribution. If a sufficient number of iterations are performed, the range of resulting valuation estimates and the frequency of particular estimates within that range can be used to determine the probability distribution of values arising from those input uncertainties that have been characterized in the analysis.

As it is unlikely that a Monte Carlo analysis will comprehensively address all sources of uncertainty in the estimation of ecological values, even 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.

Typical methods of addressing uncertainty, such as less systematic sensitivity analysis, often cannot provide meaningful guidance as to the likelihood that ecological values will exceed or fall below certain values.[5][10] In fact, such analyses can sometimes inadvertently provide misleading suggestions as to the likelihood of certain outcomes. These analyses can indicate the extent to which uncertainty in particular inputs contributes to overall ecological values, but the implications of uncertainty in one input cannot be put in context without the use of a formal quantitative assessment of uncertainty to characterize that overall uncertainty. Absent such an assessment, there is a risk that results may be perceived incorrectly as providing information regarding overall uncertainty in ecological values.

In some circumstances uncertainty assessments can change point estimates. Monte Carlo analysis can reveal when uncertainties in inputs to an ecological valuation cause the expected value to differ from what would be suggested by a deterministic analysis. Less systematic approaches may explore the implications of uncertainties that can bring about such results, but cannot address all possible outcomes resulting from those uncertainties or indicate the probability associated with any one outcome.

Implementation of a Monte Carlo analysis imposes two requirements that are not strictly necessary to develop point estimates. First, instead of requiring a single point estimate for each input, Monte Carlo analysis requires the development of probability distributions for important,

[10] There are circumstances in which sensitivity analysis may provide insights of this type, particularly when there are very few uncertain inputs and the sensitivity analysis explores the implications of uncertainties in all inputs simultaneously.

uncertain inputs. Second, numerous repetitions of the calculations of the ecological value must be performed. These requirements may appear burdensome, but to a large degree, they can entail little additional effort, relative to what is already expended in many ecological valuations. Furthermore, as with any ecological valuation, a Monte Carlo analysis does not need to be exhaustive to offer valuable insights.

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. For example, expert elicitation methods can be employed. These methods are formal, highly structured, and well documented procedures whereby expert judgments are obtained (Morgan and Henrion 1990; Cleaves 1994).

The amount of additional effort necessary to develop a Monte Carlo analysis can be minimized through careful consideration of which input uncertainties are worthwhile addressing in the analysis, since valuable insights can be gained even if the uncertainties in only a few inputs are characterized. Evaluation of how an input factors into an analysis and a preliminary assessment of uncertainty may make clear that efforts to characterize uncertainty in the input would have little affect on the findings of a Monte Carlo analysis. Thus, significant efforts to characterize uncertainty in that input would not be warranted. Such an assessment also could lead to the opposite conclusion, thereby justifying additional effort. And some inputs may be significant elements of numerous ecological valuations, providing additional justification for efforts to develop more complete characterizations of uncertainty in their values.

While a Monte Carlo analysis can require additional effort to characterize uncertainty in inputs to an ecological valuation, that effort often may be warranted even in the absence of the needs of a Monte Carlo analysis. This is because such characterizations of uncertainty may be necessary just to develop an accurate point estimate for an input. If a point estimate represents an input's expected value, the development of that point estimate requires an implicit judgment about that input's probability distribution. Characterizations of uncertainty required in a Monte Carlo analysis simply make those implicit judgments explicit. Therefore, in addition to making possible quantification of uncertainty in the results of an ecological valuation, these characterizations can improve the empirical basis for, and quality of point estimates used as inputs to the analysis.

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

6.1.4 Communicating Uncertainty in Ecological Valuations

[11] Examples of such software include Crystal Ball® and @Risk.

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 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.[12]

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.[13] 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

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[12] Circular A-4 requires the development of a formal quantitative assessment of uncertainty regarding a regulation's economic impact if either annual benefits or costs are expected to reach $1 billion (U.S. Office of Management and Budget 2003).

[13] See, for example: Morgan and Henrion 1990; U.S. Environmental Protection Agency 1997; and National Research Council 2002.

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.

6.1.5 Potential Value of Uncertainty Assessments for EPA

Assessments of uncertainty in ecological valuations can lead to both short-term and long-term gains for policymaking. In the short-term, such assessments allow more informed evaluations of proposed policies and comparisons among alternative policy instruments. In the long-term, such assessments can help establish research priorities.

Assessments of uncertainty can provide valuable information for policymakers in at least two respects. First, findings regarding uncertainty in an estimate can provide valuable context for interpreting that estimate. Second, consideration of uncertainties in underlying inputs, and how those uncertainties interact, can lead to different point estimates of values than would a purely deterministic analysis that relies on single values for each input.

Uncertainty assessments provide insights regarding the distribution of possible ecological values associated with a policy. Similar point estimates can be associated with vastly different distributions of possible outcomes around those estimates. The distribution of possible outcomes can significantly affect perceptions of an estimate of a policy's consequences. There may even be occasions where the distribution of possible outcomes associated with a policy should be given equal or greater consideration than the most likely or average outcome.

In addition to providing important context for interpreting point estimates in an ecological valuation, there are circumstances in which consideration of uncertainty can lead to different point estimates than would be developed in a deterministic analysis that does not account for uncertainty. First, careful consideration of uncertainty in inputs to an evaluation, which is necessary in the context of Monte Carlo analysis, can lead to different point estimates for those inputs than may be used in deterministic analyses. For example, the point estimate of an input used in a deterministic analysis may represent that input's expected value using one particular estimation model. But, consideration of model uncertainty may reveal that other plausible models would yield significantly different point estimates. Therefore, a point estimate of the input's expected value that accounts for the distribution of possible values resulting from model uncertainty would differ from an estimate based on just one model, which might be used in a deterministic analysis.

Second, an input may factor into an analysis in a manner in which the ultimate ecological value will differ depending on whether only a point estimate is used for the input, or the distribution of that input's possible values is explicitly incorporated in the analysis. This is because for certain mathematical functions applied to uncertain inputs, an estimate of the expected value of the function that accounts for the distribution of the input's possible values will differ from an estimate resulting from applying the function to a single value equal to the input's expected value.

Third, correlations among uncertain inputs can cause the expected value resulting from those uncertain inputs to differ from an estimate developed by using point estimates for the inputs. For example, if the values of two uncertain inputs are positively correlated, then the expected value of their product - accounting for uncertainties and this correlation - would be greater than the product of point estimates equal to each input's expected value. Thus, if that product were a component of ecological value, an analysis that does not account for uncertainties in the inputs and correlations between them would underestimate the expected ecological value.

Uncertainty assessments also can help identify the most significant determinants of uncertainty, and thereby the potential for future research to reduce that uncertainty. In the short-term, this information can be used to determine whether there may be reasons to accelerate or delay a policy action. In the long-term, the cumulative information gained from uncertainty assessments can be used to identify sources of uncertainty that have the greatest effect on overall estimates of ecological values. This knowledge can be used to help establish research priorities.

If the uncertainty arises from data quality problems due to lack of data, investments in data collection can be considered, while seeking to balance the benefits of greater potential certainty with the costs of additional data collection. On the other hand, if data quality problems arise from deficient measurement approaches, further research on measurement may be warranted, with the same benefit-cost caveat. If an important source of uncertainty is theory and modeling, then theory and model development may be indicated, and if stochastic (or random) variation is an important source of uncertainty, it can sometimes be reduced by research that focuses on the factors that have been beyond the scope of previous analysis.[14]

6.2 Expert elicitation (e.g., Delphi processes)

Many approaches have been developed to structure the interactions among experts in order to integrate their perspectives and to determine whether their forecasts and valuations will converge toward a consensus (Cooke 1991). These methods are also useful for gauging the degree of disagreement among experts, which is one indicator of uncertainty.

These methods work best when experts with different sources of expertise are involved. The purpose of the interaction, whether face-to-face or through remote contact, is to bring multiple perspectives into constructive interaction. Simpler approaches, such as simply compiling recent expert judgments, can also be useful. Any compilation will provide information about the degree of disagreement among experts, and it is a simple arithmetic fact that the error of averaged estimates may be smaller, and cannot exceed, the average error of individual estimates. However, interaction among the experts provides the additional benefit of broadening the perspectives of all involved.

The Delphi technique developed at the Rand Corporation in the 1960s typifies these

[14] For example, an earthquake-risk model based on historical frequency will have considerable random variation due to the fact that detailed analysis of fault-line dynamics are excluded from the analysis. Research into fault-line behavior may lead to reductions in such uncertainty (Budnitz *et al.* 1997).

approaches, and is the most thoroughly assessed of the multiple-expert elicitation techniques ((Helmer 1967)a; 1967b; (Sackman 1974)). The Delphi begins by soliciting judgments from a range of experts. In the classical Delphi, these experts do not have face-to-face interaction, in order to avoid intimidation, "group think," or other social dynamics that might contaminate the expression of each expert's input. Separate solicitation of judgments also permits the Delphi to be conducted through mailed (or e-mailed) questionnaires, without having to spend money on meetings.

After the first round of solicitation of judgments from a group of experts, the Delphi moderators report back the range of judgments, typically emphasizing the median and inter-quartile ranges of the judgments for each question. For example, if the experts are asked to predict how much damage a 3 degree Celsius increase in annual temperature would create for a specific low-lying area, the estimates may range from $1,000,000 to $10,000,000, but the inter-quartile range (i.e., the bounds of the middle 50% of the estimates) might be $3,000,000 to $8,000,000.

In the second round, the experts are asked whether they wish to change their estimates, having received the feedback from the first round, and if their estimates were outside of the inter-quartile range, they are also invited to provide a rationale for why their estimates are more likely than those within the inter-quartile range. In the third round, the experts receive the adjusted estimates from the second round, and the arguments for higher or lower estimates; they have another opportunity to adjust their estimates in light of this information. The Delphi may then terminate, or go through another round of providing arguments and adjustment.

Typically, the Delphi procedure results in some degree of convergence of estimates. This is to be expected if the experts have different bases of knowledge, and therefore each expert may be introducing considerations that the other experts may not have considered. For example, a fishery expert may have knowledge that a given rise in sea level would destroy a fishery worth $8,000,000 annually; in the initial round, but that expert may not have understand that the 3 degree increase would lead to this rise; whereas the oceanographer may not have realized that the sea-level rise would have that consequence on the fishery. Both may increase their estimates once they learn of the explanations of the other.

If the range of estimates for the same question remains broad, it is clear that there is considerable uncertainty. It reflects the fact that different models held implicitly or explicitly by the experts yield different predictions or estimates. However, the inverse does not hold: just because a narrow convergence may emerge, it does not necessarily follow that uncertainty is low. There are many instances in which most experts are in error in the same direction.

A very important by-product of the Delphi exercise, or other group methods that elicit the reasons for expert judgments, is the collation of the arguments that the experts have invoked to argue for higher or lower judgments. In clarifying the considerations that the experts believe drive the outcomes, and in identifying core assumptions, these arguments can be as useful as the quantitative results.

6.3 Conclusion

Uncertainty is inevitable in estimates of ecological values. Assessments of the extent and nature of such uncertainty can provide important information for policymakers evaluating existing and proposed public policies. Such information offers a context for interpreting valuation estimates, and can lead to point estimates that differ from what would be produced by purely deterministic analyses. In addition, these assessments can provide information that can help establish priorities for research.

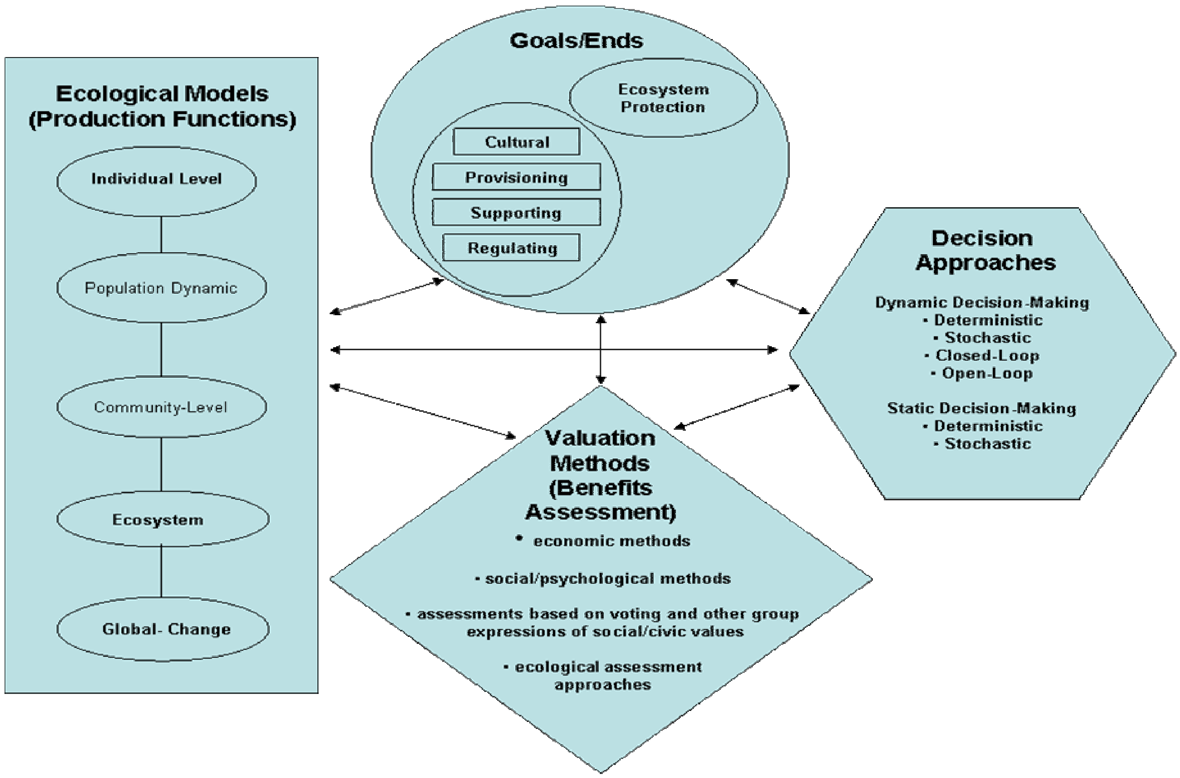

Figure 6-1. Diagram Showing the Complexity of the Decision-Making Process

Due to the complexity of interactions among uncertainties in inputs to ecological valuations, it is often the case that an accurate assessment of uncertainty can be gained only through the use of formal quantitative methods, such as Monte Carlo analysis. Such analysis involves relatively straightforward extensions to existing analytical approaches, and can offer significant insights, although additional effort is required.

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7 TABLE OF ACRONYMS

AQ: air quality

CA: conjoint analysis

CAFO: Concentrated Animal Feeding Operations

CERCLA: Superfund

CES: constant elasticity of substitution

CV (CVM): contingent valuation (method)

DGVM: dynamic global vegetation models

DSAY: discounted service acre year

EBASP: ecological benefits assessment strategic plan

EBI: environmental benefit indicators

E/GNP (E/GDP): energy gross national (domestic) product

HEA: Habitat Equivalency Analysis

IT: information technology

ITQ: individual transferable quota

MA: Millennium Assessment

NEBA: Net Environmental Benefit Analysis Framework

NOAA: National Oceanic and Atmospheric Administration

NPP: net primary production

NPV: net present value

NRDA: Natural Resource Damages Assessment

OEI: Office of Environmental Information

OMB: Office of Management and Budget

OPA: Oil Pollution Action

OPA: Office of Public Affairs

PART: Program Assessment Rating Tool

PCE: personal consumption expenditures

RIA: Regulatory Impact Assessment

ROE: Report on the Environment

RUM: random utility model

SDM: structured decision making

STELLA

TDR: transferable development rights

TMDL

UCT: University of Cape Town

VSL: value of a statistical life

WQ: water quality

WTA: willingness to accept

WTP: willingness to pay

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9 APPENDIX A: INSTITUTIONAL CONTEXT OF DECISION-MAKING LEGAL MANDATES AND EPA'S NEED FOR INFORMATION ON ECOLOGICAL BENEFITS AND A BRIEF HISTORY OF ECOLOGICAL PROTECTION AT EPA

9.1 Institutional context of decision-making

9.1.1 Legal Mandates and EPA's Need for Information on Ecological Benefits

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Ecological protection is part of EPA's mission to protect human health and the environment. To implement this mission, EPA administers many different statutes that require the Agency to protect ecological systems or services. Table 1 identifies EPA's statutory mandates and those statutes that specify protection of ecological systems and services of some kind. It also identifies statutes that call for benefit analysis.

In addition, the Agency also provides information and facilitates voluntary and partnership programs aimed at stewardship of natural resources that are distinct from those governed by the aforementioned legal statutes. These programs grew out of awareness that the complexity and scale of environmental concerns would require responses beyond the scope of traditional regulatory policy. Most of these programs are voluntary partnership efforts with states, tribes, and industries. They promote independent stewardship initiatives that complement, and at times exceed, government regulations, as well as several competitions for grants and awards, ecological awareness campaigns, and infrastructure support systems.(U.S Environmental Protection Agency 2004)

Provisions for Specific Ecol. Concerns 9-1

Major Environmental Laws Forming the Legal Basis For EPA Programs; Statutory Provisions that Authorize EPA to Consider Specific Ecological Concerns

Major Environmental Laws[15]	Statutory Provisions that Authorize EPA to Consider Specific Ecological Concerns, as listed in Appendix F of U.S. EPA's document "Priorities for Ecological Protection." (EPA/600/S-97/002, Jan. 1997)[16]
• National Environmental Policy Act of 1969 (NEPA); 42 U.S.C. 4321-4347	• § 2, 101, 102, 201, 204
• Chemical Safety Information, Site Security and Fuels Regulatory Relief Act Public Law 106-40, Jan. 6, 1999; 42 U.S.C. 7412(r)	•
• Amendment to Section 112(r) of the Clean Air Act	•
• The Clean Air Act (CAA); 42 U.S.C. s/s 7401 et seq. (1970)	• § 103, 108, 109, 111, 112(a)(1), 112(b)(1), 112(m), 160, 162, 164, 165(d), 173(a)(5), 302, 309, 401, 401 note
• The Clean Water Act (CWA); 33 U.S.C. ss/1251 et seq. (1977)	• § 101, 102, 104 , 117, 118, 119, 120, 302, 303, 304(a), 307(c), 311, 314, 320, 402, 404
• Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA or Superfund) 42 U.S.C. s/s 9601 et seq. (1980)	• § 101, 102, 107, 121, 301 •
• The Emergency Planning & Community Right-To-Know Act (EPCRA); 42 U.S.C. 11011 et seq. (1986)	•
• The Endangered Species Act (ESA); 7 U.S.C. 136;16 U.S.C. 460 et seq. (1973)	• § 2, 3(16), 7(a)(1), 7(a)(2), 9
• **Federal Insecticide, Fungicide and Rodenticide Act (FIFRA); 7 U.S.C. s/s 135 et seq. (1972)**	• § 2(j), 2(bb), 3(c)(5), 4, 10, 20
• Federal Food, Drug, and Cosmetic Act (FFDCA) 21 U.S.C. 301 et seq.	•
• Food Quality Protection Act (FQPA) Public Law 104-170, Aug. 3, 1996	•
• The Freedom of Information Act (FOIA); U.S.C. s/s 552 (1966)	•
• Marine Protection, Research, and Sanctuaries Act (MPRSA)	• § 2, 102(a), 102©, 103

[15] Statutes in bold require Benefit Analysis
[16] For a detailed explanation of each law's provisions, see Appendix F, table F-1.

• The Occupational Safety and Health Act (OSHA); 29 U.S.C. 651 et seq. (1970)	•
• The Oil Pollution Act of 1990 (OPA); 33 U.S.C. 2702 to 2761	•
• The Pollution Prevention Act (PPA); 42 U.S.C. 13101 and 13102, s/s et seq. (1990)	•
• The Resource Conservation and Recovery Act (RCRA); 42 U.S.C. s/s 321 et seq. (1976)	•
• The Safe Drinking Water Act (SDWA); 42 U.S.C. s/s 300f et seq. (1974)	•
• The Solid Waste Disposal Act (SWDA)	• § 1003, 3002, 3003, 3004, 9003
• The Superfund Amendments and Reauthorization Act (SARA); 42 U.S.C.9601 et seq. (1986)	•
• **The Toxic Substances Control Act (TSCA); 15 U.S.C. s/s 2601 et seq. (1976)**	• § 3, 4, 6, 8

As Table 1 illustrates, some statutes administered by EPA explicitly call for benefit analyses. Other EPA programs governed by statute also must conduct benefit analyses for Executive Order 12866 for "significant regulatory actions." The Office of Management and Budget has identified key elements of a regulatory analysis for "economically significant rules."(Office of Management and Budget 2003) One of the three elements is an evaluation of the benefits and costs of a proposed action and the main alternatives identified. Circular A-4 provides guidance on how to provide monetized, quantitative, and qualitative information to fully characterize benefits.

Beyond these specific legal mandates, weighing costs and benefits of different policy alternatives is part of EPA decision making that occurs across programs at the regional and national levels.(U.S. Environmental Protection Agency Science Advisory Board 2003) Decision makers factor such information into their decisions and also draw on benefit information to communicate the rationale for Agency decisions to the public.

There are also current requirements imposed on the Agency to assess the benefits of its current and past programs. The Office of Management and Budget has required that EPA include both social costs and budget costs of attaining the goals of its Strategic Plan. EPA's FY 2002 Strategic Plan described the current social costs and benefits of EPA's programs and policies under each strategic goal area for the year 2002. A repeated theme in this analysis is that ecological benefits have not been quantified. In addition, the Government Performance Results Act of 1993 established requirements for assessing the effectiveness of federal programs. Part of that assessment involves assessing the outcomes of programs intended to protect ecological resources.

9.1.2 Ecological Protection at EPA

EPA's efforts in protecting ecological systems and services have been documented in Agency publications and independent historical sources (Russell III 1993; U.S. Environmental Protection Agency 1994; U.S. Environmental Protection Agency Risk Assessment Forum 2003; U.S. Environmental Protection Agency Science Advisory Board 2003; Hays 1989). These sources document where the Agency has focused ecological protection and mention how many Agency decisions have focused on the benefits of protecting human health, rather than ecological systems and services. Several major reports focused on the need for EPA to strengthen its efforts to address ecological risks and protect healthy ecosystems because of the significance of the threat to ecological resources (U.S. Environmental Protection Agency 1987; U.S. Environmental Protection Agency Science Advisory Board 1990; U.S. Environmental Protection Agency Science Advisory Board 2000). In response, the Agency developed an ecological risk framework and ecological risk guidelines 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) and has developed a Agency-wide draft *Ecological Benefit Strategic Plan* to guide assessment of ecological benefits. (United States Environmental Protection Agency 2005).

Ongoing assessment of environmental programs provides important information for improving management and administration of public programs, whether they are regulatory, voluntary, or partnership programs. To make such assessments meaningful, EPA must be able to assess the value of programs that protect both human health and ecological systems and services.

10 APPENDIX B: DETAILED DESCRIPTION OF SELECTED METHODS

10.1 Attitude Survey: Detailed Description

Key assumptions. Perhaps the most fundamental assumption underlying attitude surveys (shared with economic valuation methods) is that the value of protecting ecosystems/services must be based on human preferences. A corollary is that public ("stakeholder") preferences represent a legitimate and important input into EPA policy and decision making. Economic and psychological/sociological surveys typically depart on how best to ascertain and represent public preferences in the policy/decision making process. Attitude surveys typically assume that value is multidimensional, and thus do not generally attempt to force preference expressions onto a single value scale. Separate indices are frequently reported for several utilitarian, aesthetic, ethical and ecological values/value dimensions. These are often reported separately to the decision maker/system, without commitment to any particular combination or resolution process. Separate attitude dimensions may be combined statistically (e.g., multiple regression, factor analysis), in more or less formal multi-attribute utility analyses or through some political negotiation or conflict resolution processes. In any event, the individual value indices and any combination or other relationships among them are assumed to depend substantially upon the specific context in which they were solicited. There can be no straightforward algorithms akin to benefit-cost analysis for comparing or generalizing among attitude indices derived from different surveys, or from non-survey methods (e.g., voting behavior, market or pseudo-market economic behaviors).

Key steps in the method. All value assessment surveys must present some representation of one or more target objects (e.g., some ecological protection policy and/or the biological and social consequences thereof) to a sample of individuals and record and interpret expressions of preferences for those objects. The key steps in implementing a survey are:

Define the target—At one level of analysis the EPA may wish to assess the value of a particular policy (or a set of alternative policies) for protecting some designated ecosystems/services. Alternatively, the target of the assessment might better be described as the set of bio-ecological outcomes/endpoints of the implementation of that policy or policies. From the public/stakeholder perspective, however, it may be more appropriate to define the assessment target in terms of the personal and/or social consequences/effects of the biological changes projected to result from the policy/policies. In the latter case, it may still be necessary to include the means by which given outcomes are to be achieved (e.g., through voluntary compliance versus legal mandates, by private agencies versus government, by artificial contrivances versus alteration/protection of natural ecosystem processes). Additional considerations in defining the assessment target include whether and how uncertainties in policy implementation, ecological outcomes and/or social effects should be included, and whether/how places, times and people affected should be specified.

Define the constituents—Surveys are directed at people who are deemed to be relevant to the assessment. The relevant population for assessing a given EPA rule/action might be representatives of affected industries, local government officials, individual property owners or resource users in a particular geographic area. Sometimes a representative national sample of the general public may be most relevant. For defining the constituents that are to be represented by a value assessment survey, errors of omission are generally more critical to public agencies than are errors of commission. That is, it is most important that all relevant stakeholders are included in the population addressed by an assessment survey. The related issues of representative sampling from the relevant population/s can be highly technical, but the general rule is that every member of any designated population must have an equal chance of being included in the sample that actually participates in the survey. This requirement is virtually never strictly met in actual surveys, but considerable effort is typically directed at attaining the best possible sample, at avoiding any systematic bias in the sample, and at confirming by statistical and empirical means what effects sampling errors may have on inferences about the population nominally represented.

Represent the targets—Assessment targets can rarely be presented directly to survey respondents, especially if the population of interest is widely dispersed and at great distances from where a proposed action is to take place. The most common means of representing targets for surveys is to verbally describe them, or to refer to them by a presumably recognized verbal label or phrase. A key concern here is whether the words used adequately and accurately convey the critical, value- and policy-relevant aspects of the target they are intended to represent. A balancing concern is that the description is not so long and complex as to require extensive effort for the respondent to understand it, or to be altogether incomprehensible to the relevant audience. Balancing detail, completeness and accuracy of representations with comprehension and efficiency of communication may be particularly difficult for ecosystems/services assessment targets. The social effects of ecosystem services that are most relevant to people may be substantially removed from and very complexly related to the policy actions and bio-ecological outcomes under consideration.

Ecosystems and the services they provide involve multiple biological and social variables interacting over a wide range of time and geographic scales. Adequately representing such complex relationships, with all of their associated uncertainties, to untrained audiences is a daunting task even for highly skilled survey researchers. Two approaches to reducing this complexity, without sacrificing too much accuracy and completeness, are to break the complex outcomes and effects into simpler parts (represented by multiple individual questions) or to conjoin the most important outcomes and effects into a limited set of concrete multidimensional scenarios. In the multiple items approach individual biological outcomes (e.g., changes in sedimentation in streams, regional air quality or biodiversity) and social effects (e.g., changes in municipal water quality, emission standards for personal automobiles or water-based recreational opportunities) might be represented and responded to separately (via *importance* ratings, for example). Possible disadvantages of the multiple items approach include the assumption, or even the imposition of independence among component outcomes/effects which may not be appropriate (i.e., more often such outcomes/effects will be interacting) and the related problem of allowing respondents to "have their cake and eat it too" (i.e., respondents may rate both improving municipal water quality and relaxing auto emissions standards as *very important*, even though these goals may in fact be in direct conflict). In the conjoint analysis approach the multiple outcomes and effects are combined into scenarios, ideally reflecting realistic combinations and interactions among the components, and survey participants respond to these multi-dimensional scenarios individually (e.g., via importance ratings) or by choosing between pairs of scenarios. While the conjoint approach allows relevant conflicts and interactions among component outcomes/effects to be presented and responded to, there are substantial practical constraints on the number of component outcomes/effects that can be meaningfully accommodated within each scenario, and on the number of scenarios that can be presented to any one respondent. Some of these limitations can be overcome by using sophisticated experimental designs to compose and present conjoint scenarios, but this can considerably increase the level of technical and statistical/analytical skills required of the surveyors.

Define response modes—Concurrent with determining the means for representing the assessment targets, surveyors must select or develop a suitable mechanism for the respondent to record his/her response to the target. The most popular options for attitude surveys are ratings (of *preference*, *liking*, *importance* or *acceptance*) or forced choices among multiple offered options (with the basis of choice specified as *preference*, *liking*, etc). Open response formats, where respondents provide narrative discourses in response to represented targets have often been used in informal interviews or focus group formats in the initial stages of developing a closed-response survey, but there is increasing use of open response formats for the final survey. Advocates for the use of open response formats cite the richer and less surveyor-determined information that can result, but these benefits are achieved at the expense of the quantitative data and analytic methods with specifiable levels of reliability and repeatability that have been developed for rating scales and choice responses. Within the closed response rating scale methods, options may include the number of levels in the scale (e.g., 5-, 7- or 10-point scales), bi-polar versus uni-polar scales (e.g., a scale extending from -5 *extremely dislike* to +5 *extremely like* versus 1 *not at all liked* to 10 *very much liked*), and whether to include (and how to interpret) a neutral point (e.g., -5 to 0 to + 5). For choice responses options include the number of alternatives offered for each choice (pairs, or multiple alternatives) and whether a "choose neither" option is allowed. In a few cases choices between pairs of alternatives may be followed by some indication of the magnitude of the perceived difference between the options, such as a rating of confidence in the choice made or an allocation of "points" (e.g., 60/40, 80/20) between the chosen and the rejected alternative. For each response format option there is an appropriate set of analytic methods that when properly applied are expected to produce equivalent (correlated) measures of the relative preferability of the targets presented for evaluation.

Contact respondents—A survey requires that representations of the assessment targets be presented to a sample of the relevant population for recording their preferences. After an appropriate sample of has been identified, individual respondents may be contacted by telephone, mail, internet or directly intercepted at their home, work or other locations. The above steps of defining the population of interest, developing representations of the assessment targets and determining response formats interact together and with the means of making contact. Ideally the primary determiners of the form of a survey will be the assessment target and the relevant constituent population, but often representation and contact options are constrained by the available budget and the time within which the survey must be completed. In practice contact options have most often been limited to telephone or mail, which in turn has limited the means of representing the assessment targets and recording responses—usually to short verbal descriptions and ratings or choices.

Analyze and interpret responses—

Points to cover (eventually)

Statistical reliability and “significance” of results
Developing and testing models (simple regression, least squares, logit, causal models)
Multiple-item versus conjoint analysis methods
Issues with analysis of open ended responses (content analysis, subjectivity, reliability?)

10.2 Mediated Modeling Method: Detailed Description

Various forms of computer models for scoping and consensus building have been developed for business management applications (Roberts 1978), (Lyneis 1980), (Westenholme 1990), (Westenholme 1994), (Morecroft, Lane et al. 1991), (Vennix and Gubbels 1994), 96 (Vennix 1996), (Morecroft and Sterman 1994), (Senge 1990). Recent trends emphasize problem structuring methods and group decision support (Checkland 1989), (Rosenhead 1989), (Phillips 1990). The use of computers to structure problems and provide group decision support has been spurred by the recognition that in complex decision settings bounds on human rationality can create persistent judgmental biases and systematic errors (Simon 1956), (Simon 1979), (Kahnemann and Tversky 1974), (Kahnemann, Slovic et al. 1982), Hogarth 1987). To identify relevant information sources, assess relationships among decisions, actions and results, and hence to facilitate learning requires that cause and effect are closely related in space and time. Dynamic modeling is one such tool that helps us close spatial and temporal gaps between decisions, actions and results.

Dynamic modeling has increasingly become a part of executive debate and dialog to help avoid judgmental biases and systematic errors in business management decision-making (Senge 1990), Morecroft 1994). It has also penetrated, albeit to a lesser extent, the assessment of environmental investments and problems (Costanza and Matthias 1998), (van den Belt 2004). Both areas of application of dynamic modeling have significantly benefited from the use of graphical programming languages. One of the main strengths of these programming languages is to enable scientists and decision makers to focus and clarify the mental model they have of a particular phenomenon, to augment this model, elaborate it and then to do something they cannot otherwise do: to run the model and let it yield the inevitable dynamic consequences hidden in their assumptions and their understanding of a system. With their relative ease of use, these graphical programming languages offer powerful tools for intellectual inquiry into the workings of complex ecological-economic systems (Hannon and Ruth 1994), (Hannon and Ruth 1997).

To model and better understand nonlinear dynamic systems requires that we describe the main system components and their interactions. System components can be described by a set of state variables—or stocks—such as the capital stock in an economy or the amount of sediment accumulated on a landscape. These state variables are influenced by controls—or flows, such as annual investment in new capital or seasonal sediment fluxes. The extent of the controls—the size of the flows—in turn may depend on the stocks themselves and other parameters of the system.

There are various graphical programming languages available that are specifically designed to facilitate modeling of nonlinear, dynamic systems. Among the easiest to learn of these languages is the graphical programming language STELLA (Costanza 1987), (Richmond and Peterson 1994), (Hannon and Ruth 1994), (Ford 1999). STELLA runs in both the Macintosh and Windows environments. A STELLA dynamic systems model consists of three communicating layers that contain progressively more detailed information on the structure and functioning of the model. The high-level mapping and input-output layer provides tools to lay out the structure of the model and to enable non-modelers to easily grasp that structure, to interactively run the model and to view and interpret its results. The ease of use of the model at this aggregate level of detail thus enables individuals to become intellectually and emotionally involved with the model (Peterson 1994).

Models are constructed in the next lower layer. Here, the symbols for stocks, flows and parameters are chosen and connected with each other. STELLA represents stocks, flows and parameters, respectively, with the following three symbols:

Icons can be selected and placed on the computer screen to define the main building blocks of the computer model. The structure of the model is established by connecting these symbols through “information arrows”

Once the structure of the model is laid out on the screen, initial conditions, parameter values and functional relationships can be specified by simply clicking on the respective icons. Dialog boxes appear that ask for the input of data or the specification of graphically or mathematically defined functions.

Equally easy is the generation of model output in tabular or graphical form through the choice of icons. With the use of sliders, a user can also immediately respond to the model output by choosing alternative parameter values as the model runs. Subsequent runs under alternative parameter settings and with different responses to model output can be plotted in the same graph or table to investigate the implications of alternative assumptions. Thus, the modeling approach is not only dynamic with respect to the behavior of the system itself but also with respect to the learning process that is initiated among decision makers as they observe the system's dynamics unfold. The process of learning by doing experiments on the computer rather than the real-world system gives model users the opportunity to investigate the implications of their assumptions for the system's dynamics and to assess their ability to make the "right" decision under alternative assumptions.

The lowest layer of the STELLA modeling environment contains a listing of the graphically or algebraically defined relationships among the system components together with initial conditions and parameter values. These equations are solved in STELLA with numerical techniques. The equations, initial conditions and parameter values can also be exported and compiled to conduct more sophisticated statistical analyses and parameter tests (Oster 1996) and to run the models in spatially explicit formats (Costanza, Sklar et al. 1990), (Costanza and Voinov 2003).

<u>10.3 Spatial Representation of Biodiversity and Conservation Value: Detailed Description</u>

10.3.1 Define the biological and ecological targets for valuation

Biological diversity is often characterized by different levels of biological and ecological organization, from genes to populations to species to natural communities to ecosystems and sometimes to ecoregions and biogeographic provinces. All of these levels can be used for characterization and valuation, but certain levels are most appropriate to address specific types of assessments. For regional scale valuation, species, natural communities and ecosystems are generally used for purposes of conservation assessment and biodiversity valuation.

Within these categories, it is helpful to use the concept of coarse filter and fine filter conservation elements. The fine filter elements are important biodiversity resources that often are sparsely distributed across the landscape. These would include imperiled, declining, endemic, vulnerable, "umbrella" species and subspecies, as well as Focal Communities such as unique environments, rare plant communities, rare aquatic habitats, vulnerable species aggregations, migratory stopover points, and others. These fine filter elements represent those components of biodiversity that can become extinct due to lack of knowledge or attention. The coarse filter elements are comprised of the broad vegetation types, habitats and ecological systems that represent aggregations of communities and natural landscape patterns and processes at scales useful for management and monitoring. It is by looking at the combination of these fine and coarse filter element that one can portray the biological and ecological valuation of the landscape based on well developed and applied standards.

The valuation of fine and coarse filter elements across the landscape required a defined level for the currency and level of standardization of the knowledge. For example, there needs to be a defined taxonomy for all species and standard classification approach for all ecological units. NatureServe and the network of state Heritage Program currently maintain this level of currency and standardization for over 30,000 animal taxa, 56,000 plant taxa, 7,000 vegetation types and 1,500 ecological systems.

10.3.2 Define occurrence standards for each target

This methodology then applies the concept of recognizing an area of land and/or water in which a species or natural community is, or was present. These Element Occurrences (EOs) have practical conservation value for the Element as evidenced by potential continued presence and/or regular recurrence at a given location. Biologists and ecologists have developed criteria and have been conducting inventories for many decades to document the best occurrences of these elements of conservation across the landscape. NatureServe databases alone manage and distribute information on the occurrences of over 500,000 imperiled species across the United States. This number grows dramatically when adding freshwater and coastal habitats, vegetation types and ecological systems.

10.3.3 Define standards for valuing the quality of each occurrence

Each of the element occurrences defined above must be given a relative quality rank to allow planners, managers and conservations to prioritize their actions relative to management of the landscape. Biologists and ecologists have developed an approach to designate A, B, C, and D quality ranks to these fine and coarse filter occurrences of conservation elements.

These methods incorporate factors of occurrence size, condition and landscape integrity. Size factors that are used in this assessment include a quantitative measure of area of occupancy, population abundance, population density, and population fluctuation. Condition looks at biotic/abiotic factors, structures, processes within the occurrence as measured by population reproduction and health, development and maturity, ecological processes, species composition and biological structure, along with abiotic physical and chemical factors. Landscape integrity compiles a qualitative measure of biotic factors, abiotic factors, and processes surrounding the EO. These factors include landscape structure and extent, community development and maturity, intactness of ecological processes, species composition and biological structure, and additional abiotic physical and chemical factors.

Many of coarse and fine filter occurrence quality metrics have been developed and used to provide a quality/integrity attribute to all occurrences. The quality ranks portray what experts determine to be within acceptable ranges of variation. These ranges are developed through the characterization of multiple, apparently undisturbed examples, examination of impact and response to human-induced alterations, review of literature and historical records, and the development and testing of ecological simulation models. “A” ranked occurrences are within the preferred ecological integrity threshold. “B” ranked occurrences have one key factor within the acceptable range of variation. “C” ranked occurrences do not have any key factors with their acceptable range of variation, but they are still considered to be 'restorable'. "D" ranked occurrences are no longer restorable. In some cases these factors can be directly measured, while in other they may be inferred/ estimated indirectly.

10.3.4 Define standards for measuring range-wide status of each target

The next step in this approach is to assign a range-wide conservation status rank to each of the conservation elements. This is primarily completed and is most useful as an element attribute at the global scale, but the standards can also be applied at the national, sub-national and local scales. The conservation rank factors differ as they are applied to species as compared to ecological communities and habitats.

For **species**, the factors that are considered in assessing conservation status include total number and condition of occurrences (e.g., populations); population size; range extent and area of occupancy; short- and long-term trends in the above factors; scope, severity, and immediacy of threats; number of protected and managed occurrences; intrinsic vulnerability and environmental specificity.

For **ecological communities**, there are primary and secondary factors used in assessing conservation status. The primary factors for assessing community status are the total number of occurrences (e.g., forest stands) and the total acreage occupied by the community. The secondary factors for assessing community status are the geographic range over which the community occurs, long-term trends across this range, short-term trend (i.e., threats), degree of site/environmental specificity exhibited by the community, and the imperilment or rarity across the range as indicated by sub-national ranks assigned by local natural heritage programs.

The definitions for each of the Global (G) Ranks are:

- **G1 – Critically imperiled:** At very high risk of extinction due to extreme rarity (often 5 or fewer populations), very steep declines, or other factors.
- **G2 – Imperiled:** At high risk of extinction due to very restricted range, very few populations (often 20 or fewer), steep declines, or other factors
- **G3 – Vulnerable:** At moderate risk of extinction due to a restricted range, relatively few populations (often 80 or fewer), recent and widespread declines, or other factors
- **G4 – Uncommon but apparently secure:** Uncommon but not rare; some cause for long-term concern due to declines or other factors
- **G5 – Widespread, abundant and secure**: Common; widespread and abundant

All fine and coarse filter conservation elements across North America have been evaluated and given a conservation status rank.

10.3.5 Create a 'conservation value layer' for each target that represents values and goals of the stakeholder

The biodiversity value attributes that have been created for the global range-wide conservation status and the quality of viable occurrences now allows the development of a conservation value surface layer for each individual conservation element. The creation of this layer requires the ability to spatially portray each of the occurrences as well as the quality and confidence of each occurrence. The spatial portrayal of element occurrences is derived from imagery, maps and field points, along with modeled distributions of specific elements. The element quality attributes are imported directly as available, and generated from landscape integrity models when necessary.

10.3.6 Create 'conservation value summary' of all targets that represents values and goals of the stakeholder

The combination of 'conservation value layers' for selected elements across a planning or assessment jurisdiction creates an aggregated 'conservation value summary' that provides a spatially explicit representation of the biodiversity and conservation values that are important to the conservation and resource management community. Different user groups can select the types of elements that there need to assess across the jurisdiction, and they can also modify the relative conservation weight of each fine and coarse filter conservation element. This will provide a customized conservation surface that portrays the values that they will need to incorporate into their planning and assessment work. This also becomes a baseline for monitoring the effects of their programs to manage for biodiversity value over time.

10.3.7 Modify the conservation value through incorporation of threats and opportunities in order to prioritize conservation and resource management activities.

The conservation values that are generated through processes 1-6 can be modified to reflect values that are relevant to a specific assessment. Zoning policies, growth models, economic values, ecological services and other values help to identify the effect of different or future scenarios relative to the current or desired future condition of the landscape.