# GENERAL VALUATION ISSUES AND APPROACHES FOR ADDRESSING THEM

(Text Drafted for Part 4 of C-VPESS Report)

## 1.1. Single vs. Multiple Metrics of Value

*(Comment from Kathy Segerson: Would this text be more appropriate earlier in the document?)*

Making decisions about which policy alternative is preferred requires that the alternatives under consideration be evaluated according to metrics that help to establish their overall value or benefit. The same is true of retrospective evaluation; here, a choice must be made regarding the extent to which the current state of a managed system (e.g., after a protective policy has been in place for some time) is, by some measure, better than it was in the past.

Thus, a central issue facing EPA when valuing the protection of ecological systems and services focuses on how to best express the value of the benefits derived from environmental protection—either during decision making or during retrospective evaluation. On the one hand, can—or should—the various dimensions of benefit derived from environmental protection be measured in terms of a common metric. In the case of dollars, for example, can EPA base the value of environmental protection, which includes moral, economic, aesthetic, and other dimensions, solely on monetary estimates obtained from established, inferred, or contingent markets? Or, should some attempt be made to express the value of environmental protection in non-monetized terms? For example, the value of protecting of endangered species or systems may be measured based on estimates of productivity or overall system diversity or resilience.

Alternatively, can—or again, should—the various dimensions of benefit derived from environmental protection be measured in terms multiple metrics? Under multiple metric approaches (e.g., multi-attribute utility theory), the value of environmental protection is measured simultaneously across multiple dimensions. No attempt is made to determine the value of the various individual dimensions—like production of commodities and endangered species protection—in common terms (e.g., dollars or units of ecological productivity). When making or evaluating decisions in this case, managers and analysts face choices among alternatives or past and present states that may perform well on some dimensions and poorly on others. In choosing which alternative is preferred or whether an achieved level of improvement is satisfactory, decision-makers must make tradeoffs across the different dimensions to establish the relative worth of each option being considered.

In practice at EPA, there isn't a clear contrast between these two approaches; most decision-making and evaluation processes involve elements of both approaches. It is frequently the case, for example, that multiple dimensions of value are reported to analysts and decision makers; 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 (Millennium Ecosystem 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) (need to cite CAFO report here). 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. As a result, these varied value inputs are left as-is.

However, there are cases where 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). Under these circumstances, EPA has 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 back-calculating travel costs based on visitation patterns at a given recreation area or inferring the economic benefit from an increase in the numbers of a marketable species).

There are many examples, at EPA and elsewhere, where these types of translate-and-aggregate operations have been used. As part of the analysis for the CAFO rule, 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 analysis for the CAFO rule 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 (need to cite CAFO report here; or there may be other, better examples that an economist could provide).

It is worth noting, however, 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.

Despite the fact that managers and analysts at EPA can and frequently 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 degree of both 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 who is 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. Even if analysts are able to do it "successfully", they is likely to face intense criticism for her efforts from many outside (and likely some inside) observers (e.g., Arrow et al. 1993) (also cite "Money" chapter provided by Paul Slovic?).

A second issue related to the defensibility with which even the same value inputs can be aggregated may also be raised. 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.

The role of the analyst in decisions about what can or should be combined is a third issue that deserves some attention here. The "can-do" approach of most economists in terms of their ability to identify values for almost all ecological systems and services serves as both a benefit and a hindrance. Clearly, the ability to quantify monetized benefits for a host of systems and services is of significant benefit to EPA, particular during the RIA process. However, the need to quantify monetized benefits can overshadow more fundamental questions that deal with what aspects of a system ought to be monetized. Simply put, would more credible and reliable values result if some aspects were not quantified, monetized, and then added to existing and often more defensible monetized inputs (i.e., if some attributes of a systems were kept separate for stakeholders, analysts, and decision makers)? It is questions like these that can become obscured by the confidence of an analyst with respect to their ability to generate estimates that attempt to describe the aggregated net benefits of ecological systems and services.

Of course, 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 in this report that the estimated value of an ecological system or service reflect 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 et al. 1990; Keeney & Gregory 2005). To be comprehensive and defensible, in other words, estimates of value must go beyond—but not necessarily discount—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 process aimed at meaningfully integrating these views during decision making and evaluation involves five steps (Gregory 2000; Hammond 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 & Gregory 2003; Gregory 2000; Hammond 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—as noted above—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 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).

There are many established methods for addressing these tradeoffs, or in other words, making use of multiple value inputs during decision making. Simple methods based on the rules of rationality (Keeney 1992; Simon 1956) focus on eliminating dominated or practically dominated alternatives (i.e., when an alternative performs better than another across all of the critical objectives or which values have been obtained). More complex methods exist in the form of specific tradeoff approaches from decision analysis (Clemen 1996) such as swing-weighting and analytic hierarchy processes (Keeney & Raiffa 1993).

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 interact 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.

## *1.2.* <u>Data and model bank</u>

Needs for a data and model bank

- Right now valuation is decentralized within EPA…don't get feeling that Agency is building on its own experience
- Need to set standards, encourage institutional learning, information sharing across the Agency so that (i) resources are used efficiently, and (ii) there is more consistency across offices and applications
- Need for outside researchers to validate, build upon Agency-funded research and data
- Needs for quality information for "benefit transfer" "valuation transfer" information

Precedents

- Learning laboratory recommendation in 2004 council report ( EPA-SAB-COUNCIL-ADV-04-004)
- IRIS (Integrated Risk Information precedent) helped to standardize Agency's approach to human health risk assessment
- Council for Regulatory Environmental Modeling calls for modeling platform; SAB supports that (see: EPA-SAB-06-009, *Review of Agency Draft Guidance on the Development, Evaluation, and Application of Regulatory Environmental Models and Models Knowledge Base by the Regulatory Environmental Modeling Guidance Review Panel of the EPA Science Advisory Board)*
- Labor and health economists have access to research platforms for sharing information
- LTER, NEON, Agency's own geo-spatial data work (http://metacat.lternet.edu/knb/index.jsp , http://www.neoninc.org/ )

- Environmental Valuation Reference Inventory (http://www.evri.ca/english/default.htm)

What would make a good data and model bank? What makes for good data and models within the bank (i.e., good content)?

- Audience/Users? Would serve needs for decision-making at national, regional, local level
- Attributes that ensure it is user-friendly?
- Structure: How entries should be organized
    - Geo-spatial units?
    - Ecological Scale
    - Political scale
- Content Information that should be included:
    - Time dimension of processes studied
    - Include variety of valuation information (ecological, economic, social, demographic, cultural information)
    - Ecological impact (like ecological risk information)
- Processes for QA and QC/ Peer Review metadata

Who the Agency should work with

### *1.3.* <u>**Transfer of valuation-related information**</u>

Benefits transfer methods adapt existing estimates of the tradeoffs people make for changes in ecological services so benefit measures can be used in other contexts or locations. These methods are frequently classified into three categories:

- Unit value transfers—interprets an estimate for the tradeoff people make for a change in ecological services as locally constant per unit change.
- Function transfers—replaces the unit value with a summary function that includes other values or a statistical summary of existing research.
- Preference calibration—begins by identifying the parameters of a preference relationship required to measure the tradeoff for a policy application

<u>Transfer of ecological information</u>

From an ecological perspective, the issue is the reliability of transferring one ecological value to other sites or over different spatial and time scales. This applicability of transferring benefits depends on characteristics of related resources and conditions, and on the reasonableness of using a static definition of an economic trade-off in a dynamic ecological system. Thus, there are significant uncertainties within the assumptions used in benefits transfer,

Farber, et al, (2006) have attempted to classify the benefits transfer of ecosystem services from one context to another (see below). In some cases, e.g., carbon sequestration (gas regulation) the transfer is appropriate at large spatial scales; in other cases, the processes operate at small scales but the processes are so general that they can be transferred with high confidence (e.g., value of game harvest). Some characteristics, such as genetic biodiversity (genetic resources) or spiritual values are very site-specific and thus the benefits cannot be transferred with confidence.

| | | | |
|---|---|---|---|
| Gas regulation | High | Food | High |
| Climate regulation | High | Raw materials | High |
| Disturbance regulation | Medium | Genetic resources | Low |
| Biological regulation | High | Medicinal resources | High |
| Water regulation | Medium | Ornamental resources | Medium |
| Soil retention | Medium | Recreation | Low |
| Waste regulation | Medium/high | Aesthetics | Low |
| Nutrient regulation | Medium | Science and education | High |
| Water supply | Medium | Spiritual and historical | Low |

A Potential Example:

For ecological impacts of interest to Chicago Wilderness, the approach could begin with the above general summary. Second, the ecological services of most significance could be identified and assessments made of the potential for transferring the benefits. Initially, this assessment could be organized by ecosystem type X stressor. For example, conversion of upland forests represents a loss of timber production which is generally transferable throughout the region. However, if the woodlands contain embedded flatwoods (fluctuating available soil water because of a perched water table or sandy soils) or are in low-lying topographic positions (more prone to loss), the benefits transfer would be reduced. Or, wildlife production from savannas is generally transferable across the region, but the biodiversity value from fine-texture soil savannas would be greater than savannas on sandy soils because more of the former have been lost to development. By aggregating these analyses, the Chicago Wilderness could create a summary table that could be the basis for assessing the economic benefits transfer.

Transfer of socio-psychological information

(Proposal: Add text here)

Benefits transfer

In economic terms, 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 and either revealed preference or stated preferences estimates can be transferred.

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, Table 8: 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[1] | added fish caught | $25 * .05 * 1 * 3 * 2000 = $7,500 |

**Table 1: Examples of Unit Value Transfer**

[1] This was computed assuming $5 for 0.2 fish added caught per hour of effort or $\$5/0.2 = \$25$ per fish per hour.

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 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, Besedin, and Wordwell 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 and Pattanayak 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.

## 1.4. Uncertainty

Introduction

Ecosystem valuation efforts are inevitably subject to uncertainty, which 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. Assessments of uncertainty allow more informed evaluations of proposed policies and comparisons among alternative policy instruments, and unless uncertainty is taken into account and thoughtfully conveyed to decision makers, the ultimate usefulness of respective assessments may be compromised. Because any given policy may result in a range of different outcomes, decision makers must be provided with sufficient information about 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 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.

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 4.2 describes the major sources of uncertainty in ecosystem and ecosystem services valuation. Section 4.3 examines the potential for uncertainty assessment of ecological values, describing both the merits of formal quantitative uncertainty assessments and the additional efforts that would be required for government agencies to carry out such assessments. Section 4.4 focuses on issues associated with communicating uncertainty to policy makers (and other audiences) in ecological valuations. Section 4.5 assesses the potential value of uncertainty assessments to the research agenda of the U.S. Environmental Protection Agency and other researchers.

<u>Sources of Uncertainty in Ecological Valuations</u>

Valuation of the benefits of proposed public policies entails three analytic tasks, each potentially subject to uncertainty: forecasting biophysical outcomes, forecasting socio-economic reactions to these outcomes, and valuing the consequences of all of these changes. In addition, the application of valuation methods per se also introduces uncertainty, in terms of the appropriateness of the assumptions, the comprehensiveness of information captured by the method, the representativeness of the sample, etc. It might be tempting to limit attention to the uncertainty of valuation per se, but all of these 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 essential steps in the valuation process is fundamentally an empirical question.

<u>Uncertainty of Biophysical Changes and their Impacts.</u> The first stage of valuation for policy analysis – 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 lack of information and 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).

At the bio-physical level any characterization of current (or past) ecological conditions will have numerous interrelated uncertainties, and these uncertainties will be magnified and added to by any effort to project future conditions, with or without some postulated management action. Ecosystems are complex, dynamic over space and time, subject to the effects of stochastic events (such as weather disturbances, drought, insect outbreaks, fires, etc) and our knowledge of these systems is incomplete and uncertain. Errors in projections of future states of ecosystems are thus unavoidable, and constitute a significant and fundamental source of uncertainty in any assessment of ecosystems/services benefits.

While the currently available methods for dealing with uncertainty may be sufficient for some simple evaluation problems, the valuation of changes in ecosystems and ecosystem services raises issues not well addressed by any existing methods. For example, at the biophysical level it is extremely difficult or completely unclear how to calculate the uncertainty (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.

<u>Uncertainty of Socio-economic Reactions and their Impacts.</u> In addition to the uncertainty introduced by biophysical projections, uncertainties in the individual and social consequences of ecological changes arise from the fact that we are rarely able to determine with certainty the individual or social consequences of a given change in ecosystems/services. Thus the second stage of valuation – predicting the socio-economic reactions to biophysical impacts and the consequences of these reactions – is subject to the same three sources of uncertainty. Regarding information and data quality every measure of economic activity, every inventory of crops and herds, and every survey is subject to some degree of error. Regarding random variation, socio-economic reactions are sensitive to the confluence of unrelated events whose co-incidence is intrinsically unpredictable. For example, a key vote on an important piece of environmental legislation may be scheduled for a date that turns out to follow a highly-publicized oil spill. Or a community's decision to relocate housing away from an increasingly flood-threatened area may be derailed by a national economic downturn that reduces matching funds for the relocation program. In addition, significant policy decisions are often made by single individuals or very small sets of individuals, whose personal idiosyncrasies create uncertainty in prediction quite parallel to the solar flares. Regarding theory limitations, every social, economic or political forecast is based on implicit or explicit

theory of how the world works, represented either by the "mental models" in the minds of the forecasters or in the formal and explicit methods used in econometric modeling, systems dynamics modeling, etc. Theories and their expressions as models are unavoidably incomplete, and of course may simply be incorrect in their assumptions and specifications.

Uncertainty Arising from the Application of Valuation Methods. Valuation methods per se are also subject to data and theory limitations. They unavoidably rely on assumptions that introduce uncertainty. Revealed-preference approaches, which seem more reliable than stated-preference approaches because of the hypothetical nature of the questions that the latter employ, are prone to some degree of model mis-specification and incompleteness. Consider the hedonic pricing method and its typical reliance on regression analysis that requires assumptions of either linear or specified curvilinear relationships. The choice of the assumption may be the best among the alternatives, but whichever assumption is chosen will not fit the data with total accuracy. The price data for the analysis is also subject to some degree of measurement error, and, insofar as hedonic pricing only reveals choices to secure private utility, its capture of existence values and the utility that comes from "public regard" is incomplete. The results of stated-preference approaches, in addition to the problem of the hypothetical nature of the questions, are also subject to uncertainty as to the impact of variations in how questions are posed, how they are interpreted by different respondents, and how reactions are influenced by differences in the information that is presented.

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

<u>Uncertainty in Benefits Transfer</u>. 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. 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).

Approaches to Assessing Uncertainty

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

Monte Carlo Analysis as an Approach to the Formal Uncertainty Assessment of Ecological Values. Due to the number of sources of uncertainty in many ecological valuations and the complexity of their interactions, assessments of the extent of uncertainty that are conducted without formal quantitative analyses are unlikely to represent accurately the true extent of uncertainty. No sensitivity analysis or expert judgment is likely to be able to account for the implications of all the sources of uncertainty in inputs, 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.

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

<u>Expert elicitation for Gauging and Conveying Uncertainty.</u> A host of "expert elicitation" methods, described in Section Y.Z of this report, can provide indications of uncertainty as well as estimates and forecasts by the experts involved. In its very simplest form, a single expert's assessment of the uncertainty of his or her estimate, forecast, or valuation can be provided, whether it is based on implicit judgment or a more explicit approach like the Monte Carlo technique. Policymakers can elicit more information from the expert, such as the assumptions underlying his or her analysis or the bases for uncertainty, in order to get a deeper understanding of the reliability of the expert's input and the nature of the uncertainty. However, the bulk of expert elicitation methods involve multiple experts, who may or may not be brought into interaction with one another. Because eliciting the input from multiple experts permits compiling and comparing their judgments, expert elicitation can be used to assess the disagreement among experts. If the experts are of equal credibility, such that none of the judgments can be discarded in favor of others, the range of disagreement reflects uncertainty. That is, if top scientists express strong divergences in their estimates, forecasts, or valuations, the existence of a high level of uncertainty is irrefutable. However, this is an asymmetrical relationship, in that narrow disagreement does not necessarily reflect justified certainty—the experts may all be wrong in the same direction, which is not uncommon in light of the fact that experts are often paying attention to the same information and operate within the same paradigm for any given issue (Ascher & Overholt, 1984: 86-87). When experts are brought into some form of interaction prior to providing their final conclusions (e.g., by exchanging estimates and adapting them in reaction to what they learn from one another), the errors due to incompleteness (e.g.,

biologists may be unaware of atmospheric trends that information from atmospheric chemists could redress) can be reduced. However, such interactions run the risk of "groupthink" – unjustified convergence of estimates due to psychological or social pressures to come closer to agreement (Janis, 1982).

For many expert elicitation methods, translation into probabilities is difficult. For example, simple compilations of estimates (e.g., contemporaneous estimates of species populations) from different experts will provide a table with the range of estimates, but will not convey the degree of uncertainty that each expert would attribute to his or her estimate, and the compilation in itself cannot generate this information. In contrast, a compilation of estimates that come with confidence intervals could provide this information. To take another example, the well-known Delphi technique (described in some detail in Section Y.Z) typically provides the inter-quartile range (i.e., the middle 50% of estimates) following several rounds of exchanges of estimates, but does not provide probabilities reflecting the uncertainties of each estimate or of the set of estimates.

Communicating Uncertainty in Ecological Valuations

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

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

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

Historically, efforts to address uncertainty in ecological valuations - and more broadly, in Regulatory Impact Analyses (RIAs) - have been limited. But guidance set forth in the U.S. Office of Management and Budget's (OMB) Circular A-4 on Regulatory Analysis in 2003 has the potential to enhance the information provided in RIAs regarding uncertainty.

In the past, point estimates have been given far greater prominence in RIAs and other government valuations than discussions of uncertainty associated with them. Uncertainty assessments are often relegated to appendices and discussed in a manner that makes it difficult for readers to discern their significance. This is perhaps inevitable given that single point estimates can be communicated more easily than lengthy qualitative assessments of uncertainty or a series of sensitivity analyses. The ability of Monte Carlo analysis to produce quantitative probability distributions provides a means of summarizing uncertainty that can be communicated nearly as concisely as point estimates. The need for and means of communicating uncertainty in such a fashion has been addressed in the existing literature. If a summary of uncertainty in an estimate is not given prominence relative to the estimate itself, context for interpreting the estimate and opportunities to learn from uncertainty associated with it may be lost.

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

<u>Decision-Making with Uncertainty</u>

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

<u>Static Decision-Making with Uncertainty.</u> 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.

Dynamic Decision-Making under 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 be 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 and preserving land. If land is preserved in the first period, then there will again be a choice between developing and 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.

<u>Contributions of Uncertainty Assessment in Guiding Research Initiatives</u>

Assessments of the magnitude and sources of uncertainty can help establish research priorities and whether policy changes should be delayed until research reduces the degree of uncertainty associated with possible changes. Determining whether the major source of uncertainty is in weak data, weak theory, randomness, or inadequate methods can help to guide the decision on how to allocate scarce resources for research, or whether further research is worth pursuing. Even stochastic uncertainty can sometimes be addressed by initiating research that focuses on factors previously treated as exogenous to the theories and models. For example, an earthquake-risk model based on historical frequency will have considerable random variation due to the exclusion of detailed analysis of fault-line dynamics; bringing fault-line behavior into the analysis may lead to reductions in such uncertainty (Budnitz et al. 1997).

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

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

## 1.5. Communication and Valuation

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

(1) communication within the valuation process itself;
(2) communication of the resulting values, to inform decision-making; and
(3) communication of the results of the valuation and decision-making processes to stakeholders and others.

This discussion of communication and valuation will elaborate on points 2 and 3. As discussed elsewhere in the report, within the valuation process itself, how decision objectives, decision attributes, and specific measures of values are communicated can determine the outcome of the process. Good communication practices include the use of an analytic-deliberative process, in which analysis and deliberation occur iteratively and interactively. Values, decision objectives, and decision attributes can each be defined either qualitatively or quantitatively, and represented in a wide variety of ways. Those choices will in turn either facilitate or hinder specific kinds of deliberations and analyses. Decision making in public policy often requires translation and/or aggregation, from one specific context to another, or from one level of decision making (e.g., local) to another (e.g. regional), and inevitably involves trade-offs. Specific choices of how to represent or communicate values will influence the ease and transparency with values can be translated or aggregated, and with which trade-offs can be made.

The valuation process (see diagram) includes iterative problem definition and description by stakeholders, to clarify what and whose values will be represented by the process. It's critical that the process include explicit:

- discussion and description of the nature and state of the ecological systems and services to which they apply, and (potential) changes in these;[2] as well as

[2] 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.

- discussion and description of ecological (causal) functions/processes.[3]

1.5.1. How are value measures being used now? What do we know about their perception and use?

a. Table summarizing measures from different case studies in the report

b. Summary of other approaches (e.g., Net Environmental Benefit Approach, Appendix A of HM Treasury (2004). Managing risks to the public: appraisal guidance)

c.. Analysis of table addressing communications and psychological literature on empirical evidence regarding effects of these different kinds of measures on decision making (e.g., example – national rulemaking, discuss Buzz's concern that congress only attends to $$ values, not to biophysical indicators, how do you get them to pay attention to values that are not monetized as well?) Discussion of persuasion and presentation issues

1.5.2. Guidelines for representation and analysis of value (VPESS) measures

a. Assessment of audiences and uses for valuation information (e.g., to evaluate use of technical jargon, need for details): communication to inform decision making, based on value inputs (decision analytic perspective - include here specific discussions of the decision contexts that we focus on in CVPESS: local, regional, regulatory/national).
b. Focus on empirically tested approaches (reference section 2 above)
    i. quantitative as well as qualitative communications (based on user needs, availability of info)

[3] 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 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 Slovic's discussion of framing here?).

  ii. Appropriate use of graphical and visual approaches, GIS
  iii. Interactive communications likely to be more effective in many circumstances (ability to manipulate the data or representations of the data, tailor for different analyses and audiences)

c. Communicate about valuation process as well as the result of the valuation
d. Fundamental guidelines for risk and technical communication generally applicable (provide examples: e.g., Transparency, Clarity, Consistency and Reasonableness principles from Risk Characterization Handbook; Guidelines for effective websites from Spyridakis - Spyridakis, J.H. Guidelines for Authoring Comprehensible Web Pages and Evaluating Their Success. *Technical Communication*, *47*, 3, 301-310, 2000, accessible at http://www.uwtc.washington.edu/people/faculty/jspyridakis.php)
e. Evaluate effects of communications (as resources permit)