Federal Resource Management and Ecosystem Services Guidebook

National Ecosystem Services Partnership
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Section 3—Ecosystem Service Assessment Methods

SECTION 3

ECOSYSTEM SERVICE ASSESSMENT METHODS

OVERVIEW AND BEST PRACTICES

HOW TO READ THIS SECTION OF THE GUIDEBOOK

The overview presents a bird's-eye view of ecosystem services assessment methods and how they can be put together and contains links to detailed descriptions of various steps. The most efficient way to browse this section is to use this overview to find additional sections of interest.

The original description of ecosystem service methods were developed through the efforts of two technical working groups hosted by the National Center for Ecological Analysis and Synthesis and the National Socio-Environmental Synthesis Center (SESYNC), each of which consisted of agency staff and academic experts. One group focused on ecological analysis relevant to ecosystem services. The other focused on social analysis relevant to ecosystem services. The updated version of the framework and methods, which is now available, is based on refinements developed by another SESYNC working group that are described in the companion report *Best Practices for Integrating Ecosystem Services into Federal Decision Making.*[1]

Brief descriptions of all guidebook sections (in boldface) follow.

Overview and Best Practices

- **How to Read**—introduces the assessment methods section.
- **Methods Overview**—reviews all of the ecosystem services assessment methods and how the pieces fit together.
- **Best Practices**—summarizes overarching best practices for ecosystem services assessment.

Scoping

- **Understanding Socio-Cultural Context**—explains how socio-cultural information can inform an assessment.
- **Stakeholder Engagement during Scoping**—describes how stakeholder engagement techniques can use ecosystem services concepts in planning and management.
- **Conceptual Diagrams**—provides a systematic way to connect ecological conditions and societal benefits.
- **Identifying Services**—points to conceptual diagrams and causal chains as approaches for identifying services.

Benefit-Relevant Indicators (BRIs)

- **What Are BRIs**—defines and describes benefit-relevant indicators (BRIs) and their relevance to ecosystem services assessments.

Analysis

Selecting Services and Causal Chains

- **Selecting Services**—provides a set of questions for narrowing the set of services that need to be quantified and analyzed in the assessment process.
- **Building Causal Chains**—describes how to develop causal chains showing how current conditions, desired

[1] Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

conditions, or a management action or policy will propagate through an ecosystem, affecting the provision of ecosystem services and benefits to people.

Quantifying BRIs

- **Quantifying BRIs**—describes beyond-narrative methods for quantifying production, provision, and other changes in ecosystem services and the parties affected by these changes and hence information on what is valued by whom.
- **Quantifying Social and Economic Context of BRIs**—informs the development of meaningful BRIs and benefits assessment by adding information on the social and economic conditions that impart value to goods and services.

Benefits Assessment

- **Overview of Benefits Assessment—**describes when an assessment of benefits is needed and compares monetary and non-monetary approaches.
- **Monetary Valuation**—how to properly use monetary valuation in ecosystem services assessments.
- **Non-Monetary Methods: Multi-Criteria Evaluation for Ecosystem Services**—how to use non-monetary methods to value ecosystem services
- **Other Methods**—alternative methods for assessing social impact.

The Decision Process

- **Using BRIs in Decision Making**—describes how BRIs can be used in alternative matrices, efficiency frontiers, and benefits assessments to inform decision making.
- **Displaying Assessment Results with Alternatives Matrices and Maps**—describes different ways that assessment results can be displayed.
- **Weighting and Aggregation—**describes how to weigh competing outcomes and aggregate values or ranks across services.

Stakeholder Engagement—describes how stakeholder engagement can be adapted for ecosystem services assessments.

Using Indicators Effectively—explains how to clearly define indicators, estimate quantitative indicators, and correctly use qualitative measures.

Scenario Analysis and Green Accounting—compares two uses of ecosystem services assessments.

METHODS OVERVIEW

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Summary

This section of the guidebook provides an overview of ecosystem services assessment methods that could be applied across geographies and decision contexts and shows how they fit together. It will help managers incorporate the effects on ecosystem services of different management alternatives or scenarios, policy options, and locations of actions or infrastructure. It will also help with the development of performance metrics and may suggest changes in data collection and monitoring. The overview contains links to more detailed discussions of assessment methods in other sections of the guidebook as well as to additional resources.

Takeaways

- Ecosystem services should be used if decision makers want to go beyond an assessment of ecological conditions alone to consider how changes in ecosystems affect people.
- The ecosystem services assessments combines insights and methods from the natural and the social, behavioral, and economic sciences.
- The methods described in this guidebook can be applied flexibly to meet decision-maker needs.

What are Ecosystem Services Assessments and Who Will Use Them?

Ecosystem services assessments can be used to describe how management choices affect the well-being of people, communities, and economies through their effect on natural systems. Ideally, such an assessment will help decision makers incorporate the less commonly quantified but no less important benefits of nature, along with more commonly considered benefits of management actions, into their decision making.

The ecosystem services assessment methods described in this guidebook apply primarily to regional, local, or project-scale decisions affecting actual locations. They can be applied in different geographies and with different levels of expertise and resources. They can also be adapted and used in a wide range of decision contexts, including

- Species or biodiversity management,
- Ecological restoration or conservation (approach or location),
- Risk management (reducing floor, fire, or storm surge risks),
- Infrastructure decisions and siting (roads, housing, trails, campsites, energy production, mining, pipelines, etc.),
- Selection of performance metrics, and
- Identification of data and monitoring needs.

This overview of methods is designed to help managers explore simpler and more accessible methods for ecosystem services analysis, building toward the use of more quantitative and in-depth approaches over time. Although the overview depicts a relatively linear process for putting these methods together, the actual process of incorporating ecosystem services considerations into decision making is likely to be iterative.

Best practices for using ecosystem services assessments can be found in this companion paper.[2]

How Does This Overview of Ecosystem Services Methods Fit in with Other Decision Support Guidance?

The ecosystem services assessment methods described here are generic and align with many other decision support frameworks under development or already in use by academics, NGOs, and consultants (e.g., The Natural Capital Project approach[3], and World Resources Institute's Ecosystem Services Guide for Decision Makers[4]). Several agencies have also produced or are working on ecosystem services frameworks or guidance that follow similar steps and use similar methods (e.g., the U.S. Army Corps of Engineers, the U.S. Bureau of Land Management, the U.S. Environmental Protection Agency, and the U.S. Forest Service.)[5]

By providing a relatively complete overview of common methods for incorporating ecosystem services considerations into decision making, this guidebook aims to improve consistency and validity in applying these methods across federal agencies and to help agencies customize ecosystem services analysis to meet their needs and decision contexts.

The FRMES ecosystem services assessment methods review

- Provides examples of integrating ecosystem services into existing federal planning processes from project scoping through monitoring of decision outcomes;

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[2] L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

[3] "Natural Capital Project," last accessed January 27, 2016, http://www.naturalcapitalproject.org/.

[4] J. Ranganathan, K. Bennett, C. Raudepp-Hearne, N. Lucas, F. Irwin, M. Zurek, N. Ash, and P. West, Ecosystem Services: A Guide for Decision Makers (Washington, D.C.: World Resources Institute, 2008), http://www.wri.org/publication/ecosystem-services/.

[5] **Army Corps of Engineers:** E. Murray, J. Cushing, L. Wainger, and D. Tazik, "Incorporating Ecosystem Goods and Services in Environmental Planning – Definitions, Classification and Operational Approaches," Technical Note, ERDC TN-EMRRP-ER-18, Ecosystem Management and Restoration Research Program, U.S. Army Corps of Engineers (2013).**U.S. Bureau of Land Management:** U.S. Bureau of Land Management, Guidance on Estimating Nonmarket Environmental Values, Instruction Memorandum No. 2013-131, Change 1 (2013). **U.S. Environmental Protection Agency:** U.S. Environmental Protection Agency, *A Framework for the Economic Assessment of Ecological Benefits* (Washington, D.C.: U.S. EPA, 2002). This framework is focused on integrating ecological risk assessment and economics. It covers some of the same ground as the framework presented in this guidebook but is focused on economic analysis and in some cases provides more depth on the analytical options. **U.S. Forest Service:** U.S. Forest Service, 2012 Planning Rule.

- Emphasizes how ecological changes result in changes in the provision of ecosystem services and benefits to people;
- Provides detailed information on methods to assess changes in ecosystem services, including guidance on quantification and valuation;
- Provides flexibility in application from quantifying the benefits (what is valued) to quantifying the preferences and values people have for the benefits; and
- Is consistent with other frameworks as well as existing software and downloadable tools (e.g., Miradi, InVEST, ARIES[6], Envision.[7]

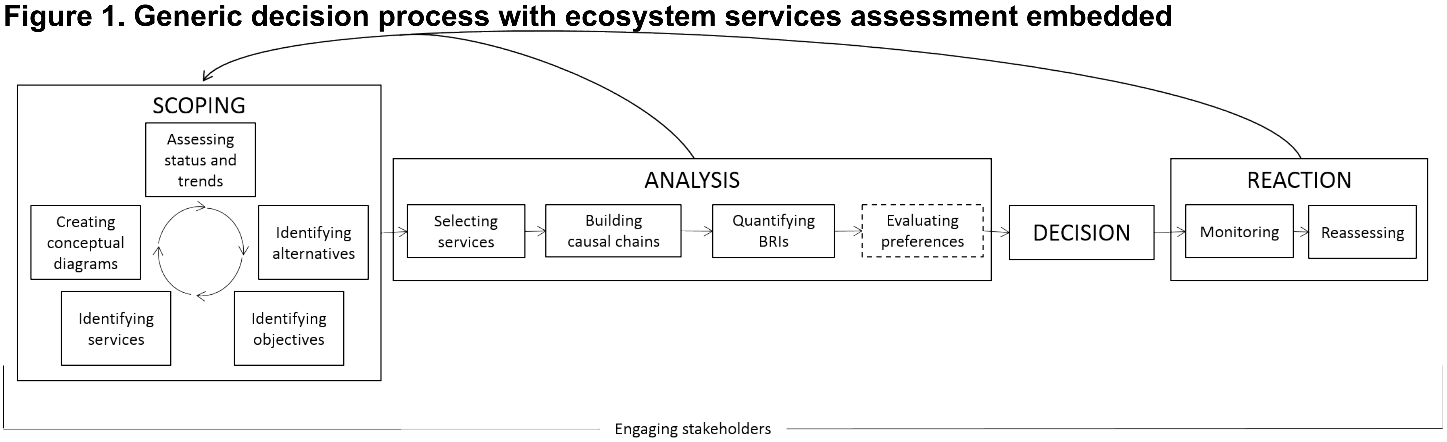
The methods presented in this guidebook are designed to be useful across geographies, natural resource priorities, and all types of ecosystem services and to be applicable to a range of decision contexts important to federal agencies. Although most obviously applicable to decisions that directly target natural resource issues, consideration of ecosystem services can be salient to any decision that directly or indirectly affects ecosystems, natural resources, or the environment. For example, ecosystem services might be useful in tracking performance of programs and projects to assess progress toward goals; in comparing outcomes to better allocate future funding; in considering the potential effects of proposed natural resource and infrastructure plans, siting decisions, or permit allocations for use of public resources; and in revising rules and regulations that drive implementation of laws such as the Clean Water Act and Clean Air Act.

This review of methods is intended to help managers across federal agencies design decision processes that incorporate ecosystem services, to help them fit the methods to their needs, and to understand the expertise needed to apply them.

Ecosystem Services Assessment in Decision Processes

This guidebook organizes the ecosystem services assessment methods around an example planning process using four general stages: (1) scoping, (2) analysis, (3) the decision, and (4) reaction (Figure 1). These stages also occur in other comprehensive decision-making approaches, such as economic valuation and structured decision making, elements of which are included in this framework.

Figure 1. Generic decision process with ecosystem services assessment embedded



Note: Integrating ecosystem services considerations into the decision making process—that is, translating ecological changes into implications for people—requires changes throughout the decision process, particularly in the scoping and assessment phases. Stakeholder engagement continues through the full decision process.

[6] "ARIES (ARtificial Intelligence for Ecosystem Services)," last accessed January 20, 2016, http://www.ariesonline.org/.

[7] "Envision Integrated Modeling Platform:A spatially explicit, multiparadigm modeling framework for analysis of coupled natural/human systems and alternative future scenarios," last accessed January 27, 2016, http://envision.bioe.orst.edu/.footnote])

This section of the guidebook emphasizes what is new about incorporating ecosystem services relative to standard ecological assessments. It describes how ecological information and stakeholder engagement are used together to inform ecosystem services priorities in the scoping process. It also describes methods for assessing how ecological changes matter to people, including approaches for integrating social preferences, and procedures for comparing options. The resulting information on ecosystem services feeds into the decision process.

Scoping

Scoping is the process of identifying an initial set of desired outcomes and management objectives based on an assessment of ecological and social data on past management outcomes, current conditions, and anticipated future needs. This stage might include problem definition and needs assessments. It may include a significant component of stakeholder engagement. In a federal agency context, Congress, the Office of Management and Budget, and departmental and agency policy generally set the sideboards for the set of desired outcomes and management options to be considered. Consideration of ecosystem services can be integrated into these assessments of problems and needs by connecting the ecological outcomes to people.

In a typical planning process, scoping would entail an analysis of ecological status and trends, which would include information on ecological conditions and potential threats and stressors over time for the area of interest. For example, such an analysis might include information from FIA and LANDFIRE databases status and trends of forestlands (e.g., forest growth and forest departure from natural conditions due to fire suppression). It might also use local monitoring data, research findings, or field evaluations from resource staff to assess ecological conditions. Ideally, scoping will also incorporate landscape-scale or regional information about larger-scale processes that will affect or be affected by the decisions at hand. **A focus on ecosystem services in this scoping phase would extend a typical status and trends assessment to an entire affected area (e.g., watershed, airshed, fishery) in order to encompass the flow of services to and from the resource management area to the people who are affected.**

Scoping often involves an assessment of user needs, perhaps involving meetings with stakeholder communities (within the bounds of regulatory limits, e.g., the Federal Advisory Committee Act). The result is a set of desired social outcomes that, when incorporating ecosystem services, can be described as ecosystem-service-derived benefits.

A focus on ecosystem services requires an explicit connection between the desired ecological conditions and how these conditions will affect people. For example, when assessing policies to address forest fire risk, ecological conditions might be described in terms of changes in fire frequency and resulting changes in forest structure and species composition. To make these changes more meaningful to people, the ecosystem services might be described in terms of enhanced populations of species of interest (e.g., bison on the prairie, birds to watch, fish to catch), increased opportunity to hike and camp in the forest, decreased incidence of smoke, and changes in the provision of wood. This is an example of a conceptual mapping exercise focused on ecosystem services. Such conceptual maps or diagrams will play a critical role in helping managers and assessors identify services to include and thus beneficiaries to consider or engage in an assessment process because they make explicit the connection between ecological conditions and benefits people derive from them.

When stakeholders are engaged directly, they can be asked to identify what they value in a given planning area. Such expressions provide initial qualitative evidence of the ecosystem-derived benefits (the ecosystem services) they would like to obtain or experience. The scoping process is likely to be iterative, linking desired ecological conditions and desired social outcomes. This process will help managers identify the management objectives and options that have the capacity to achieve both ecological and social objectives. The stakeholder engagement process is likely to capture a range of opinions about which outcomes are valuable. Reconciling these different opinions and evaluating tradeoffs are fundamental components of assessment and decision processes. Another key issue is balancing stakeholder preferences with the need to sustainably manage ecological processes over time. The stakeholder process complements program mandates and legal constraints, which also drive policies and options considered.

Once desired management objectives (ecological conditions and social outcomes) are determined, the next step is to identify a set of management alternatives, project options, or policy choices for achieving those outcomes and perhaps to identify performance metrics and measures. For example, within the context of forest fire management, managers might want to use prescribed fire and removal of invasive plants (management actions) to help reestablish healthy longleaf pine forest (desired conditions) in the southeast. Managers may need to decide *where* to use these actions to best enhance wildlife viewing and improve recreational opportunities while minimizing smoke exposure (desired social outcomes). Other decisions might involve choices between different management actions: understory thinning versus prescribed burning to alter forest structure and reduce fuel loads or a levee versus floodplain restoration to address flooding. In other policy contexts, decisions might involve policy choices such as changing water releases from dams or establishing incentives programs for biomass energy. These could be ecosystem service assessments so long as the policies can be articulated in terms of likely changes to both ecological conditions and ecosystem services. At this point conceptual maps and causal chains can be created to help connect agency actions to the ecosystem benefits identified as important.

Conceptual Diagrams

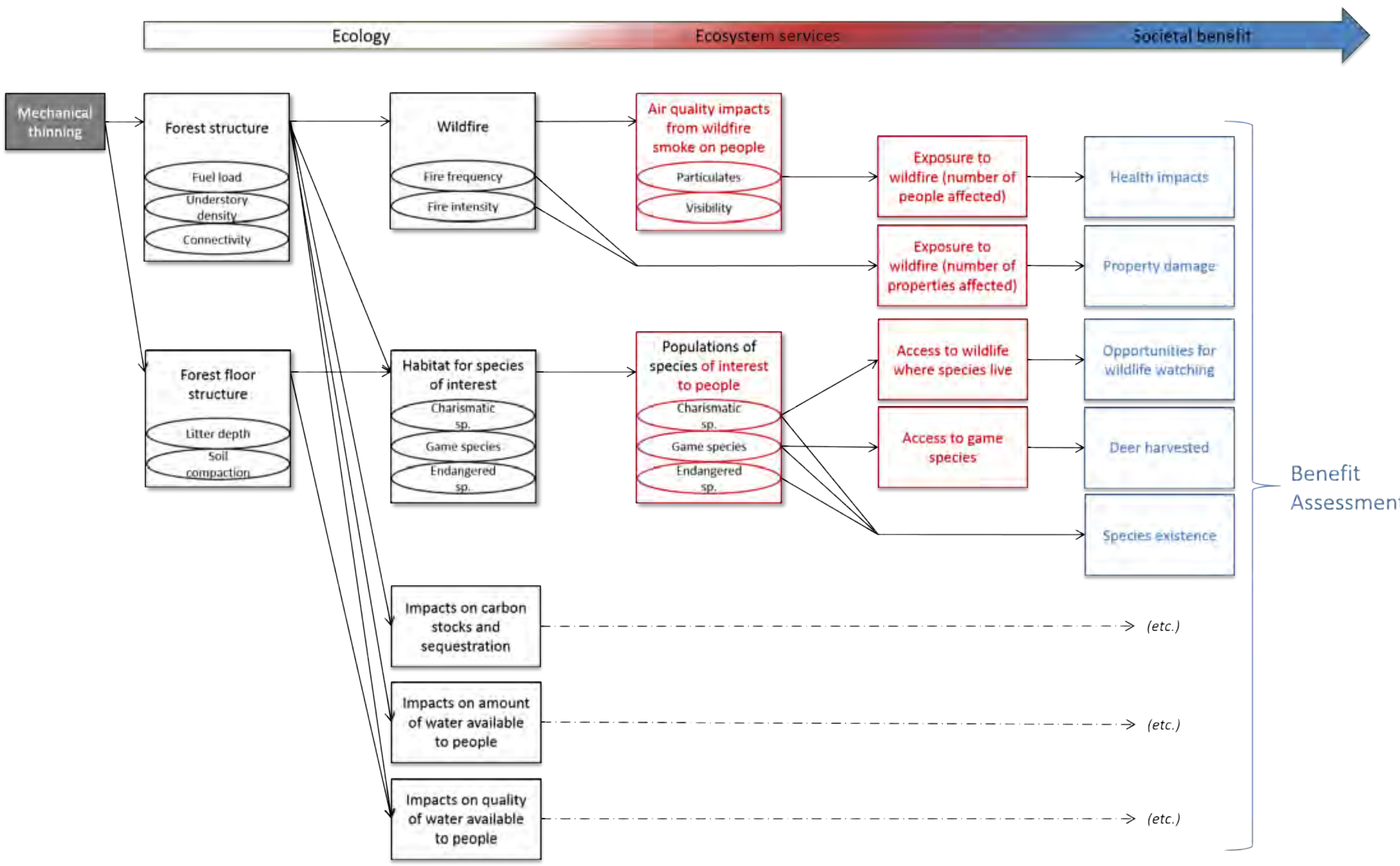
Many decisions require assessments or measures of current conditions as well as how a policy, project, or action is likely to affect an ecosystem. Conceptual diagrams, also known as means-ends diagrams, are tools that can facilitate this process. Conceptual diagrams are made up of causal chains, also known as path models, which are logical models that declare how a management action or policy is expected to propagate through natural and social systems to effect changes in the provision of ecosystem services and benefits to various segments of society (For example Figure 2). As part of scoping, the causal chains embedded in conceptual diagrams may be preliminary best guesses requiring relatively little effort yet providing a comprehensive overview of all potentially significant services. Incorporating ecosystem services into conceptual diagrams can improve how agencies define problems and formulate solutions by expanding the focus of the decision maker beyond ecological outcomes to social outcomes caused by the ecological changes.[footnote]See, e.g., K.J. Bagstad, D. Semmens, R. Winthrop, D. Jaworski, and J. Larson, "Ecosystem Services Valuation to Support Decisionmaking on Public Lands: A Case Study of the San Pedro River Watershed, Arizona," U.S. Geological Survey Scientific Investigations Report 2012–5251 (2012), http://pubs.usgs.gov/sir/2012/5251/; E. Nelson, G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D.R. Cameron, Kai MA Chan, G. C. Daily, J. Goldstein, P.M. Kareiva, E. Lonsdorf, R. Naidoo, T.H. Ricketts, and M.R. Shaw, "Modeling Multiple Ecosystem Services, Biodiversity Conservation, Commodity Production, and Tradeoffs at Landscape Scales,"*Frontiers in Ecology and the Environment* 7 (2009): 4–11, http://dx.doi.org/10.1890/080023. Developing conceptual diagrams is a critical step to ensure that ecosystem services assessments are comprehensive and transparent.[8]

Conceptual diagrams incorporate all potentially significant causal chains for affected services. In doing so, they identify how a policy or management action can affect multiple aspects of an ecosystem and how each of the impacts on an ecosystem can have multiple impacts on social benefits. Creation of these diagrams forces the users to recognize intermediate ecological structures or processes and connections to nontarget ecological and ecosystem service outcomes. This process often leads to a richer (if more complicated) view of the system as new branches and paths are elaborated in iterative discussions of experts and stakeholders. These diagrams can help identify potential tradeoffs, or unintended consequences. They can also be used to compare the outcomes of management in different places. Such spatial analysis can be incorporated into geographic information system models.[9] When assessing a policy or management action a conceptual diagram should be created for each distinct option (policy, management, or project alternative). This process may reveal that particular options are unlikely to impact services of interest. Those options could then be eliminated from further analysis. **Constructing conceptual diagrams forces managers to embrace the complexity of the systems they manage and to identify gaps in understanding how ecosystem components and environmental benefits are linked.**

[8] For more information on final ecosystem goods and services, see J. Boyd and S. Banzhaf, "What Are Ecosystem Services? The Need for Standardized Environmental Accounting Unites." *Ecological Economics* 63 (2007): 616-626. and D.H. Landers and A.M. Nahlik, "Final Ecosystem Goods and Services Classification System (FEGS-CS), "EPA/600/R-13/ORD-004914, Office of Research and Development, U.S. Environmental Protection Agency, Corvallis, Oregon (2013), http://gispub4.epa.gov/FEGS/FEGS-CS%20FINAL%20V.2.8a.pdf.

[9] See, e.g., K.J. Bagstad, D. Semmens, R. Winthrop, D. Jaworski, and J. Larson, "Ecosystem Services Valuation to Support Decisionmaking on Public Lands: A Case Study of the San Pedro River Watershed, Arizona," U.S. Geological Survey Scientific Investigations Report 2012–5251 (2012), http://pubs.usgs.gov/sir/2012/5251/; E. Nelson, G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D.R. Cameron, K.M.A. Chan, G.C. Daily, J. Goldstein, P.M. Kareiva, E. Lonsdorf, R. Naidoo, T.H. Ricketts, and M.R. Shaw, "Modeling Multiple Ecosystem Services, Biodiversity Conservation, Commodity Production, and Tradeoffs at Landscape Scales,"*Frontiers in Ecology and the Environment* 7 (2009): 4–11, http://dx.doi.org/10.1890/080023.

Figure 2. Conceptual diagram with causal chains for an ecosystem services assessment of a forest management decision



Note: This conceptual map of simplified causal chains shows possible outcomes from forest fire management activities like mechanical thinning. Black text indicates an ecological assessment and indicators, red text indicates extension to an ecosystem services assessment, and blue text indicates measures of social benefit and value. (See Figure A-1 in the appendix for an expanded version).

In considering possible impacts of management actions or policies on ecosystem services, it might be tempting to refer to a "master list" of services to help ensure that the conceptual diagram is complete. Such lists are often created in an effort to classify ecosystem services.[10] Generic lists of services can provide a useful starting point for considering which services and beneficiaries are relevant in a decision context, but given context-specific variation in services, generic classifications will almost always be insufficient and can often be misleading. Rather than or in addition to using classification systems, agencies should use causal chains and conceptual maps to identify ecosystem services and the groups potentially affected by agency actions for each decision.

Analysis

The analysis uses the outputs of the scoping processes, desired ecological conditions and social outcomes (i.e., ecosystem services), and the suite of management, project, or policy options that make it through the scoping process. Ecosystem services analysis estimates the effects of management or policy on the production of ecosystem services using measures of ecosystem changes that matter to people.

[10] Millennium Ecosystem Assessment, *Ecosystems and Human Well-Being: Synthesis* (Washington, DC: Island Press, 2005); Common International Classification of Ecosystem Services (2013), http://cices.eu/.; D.H. Landers and A.M. Nahlik, *Final Ecosystem Goods and Services Classification System (FEGS),* EPA/600/R-13/ORD-004914, 2013, http://ecosystemcommons.org/sites/default/files/fegs-cs_final_v_2_8a.pdf; P. Sinha and G. Van Houtven, *National Ecosystem Services Classification System (NESCS): Framework Design and Policy Application*, draft report prepared for the U.S. Environmental Protection Agency.

Selecting Services for Further Analysis

Developing an initial conceptual diagram is useful for considering all possible impacts to valued services. This process will likely identify too many services to be meaningfully quantified in any ecosystem services assessment. Thus, the quantitative assessment can focus on only those effects likely to be most important to the decision—often those expected to have the largest impacts on ecological processes and human welfare. Assessors can use a few key questions to determine which services should be included:

- Does the ecosystem service fall under the legal mandate of the assessor?
- Is the impact on the ecosystem service likely to be large and strongly driven by the proposed activity or decision?
- Will the expected changes to the ecosystem service matter to or affect the social welfare of many people or groups of special concern?

By proceeding in this manner, the agency acknowledges the full suite of affected ecosystem services and can be more transparent about the services that are (and are not) subsequently analyzed and the rationale for these decisions. A full empirical assessment cannot and likely should not be conducted for all services initially identified in a conceptual map. And indicators used as performance metrics likely cannot be monitored for all services.

Agencies may need to collaborate with one another to include ecosystem services outside their authorities in an empirical assessment or monitoring plan. This may be important if the service outside of agency authority experiences a significant change.

Causal Chains

Causal chains are logic models that describe how actions propagate through systems to effect changes in outcomes. When used in ecological assessments, causal chains often end with expected environmental changes and stop short of including impacts on benefits to society. In contrast, a causal chain in an ecosystem services assessment follows through to effects on social outcomes and human well-being (Figure 3a,b). In the analysis, the causal chains sketched out during the scoping process in conceptual diagrams that are selected as important are now more fully developed. Developing robust quantifiable causal chains is a critical step to ensure that ecosystem services assessments are comprehensive and transparent.[11] **Constructing causal chains forces managers to identify gaps in knowledge, data, and modeling capacity as well as in understanding how ecosystem components are linked.**

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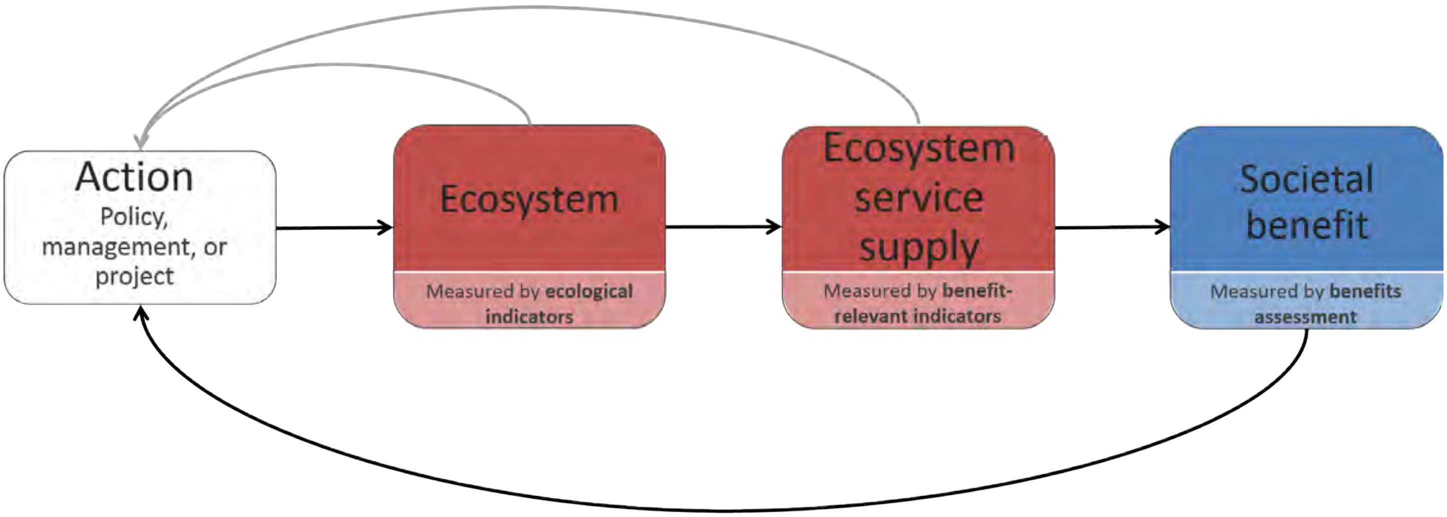
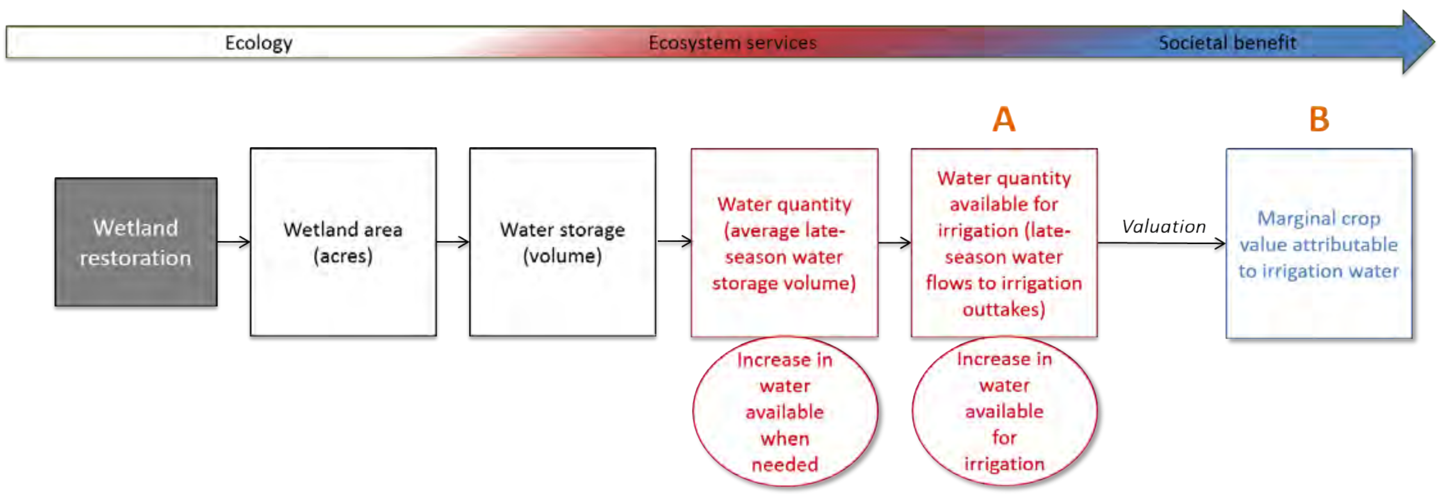
[11] National Ecosystem Services Partnership, *Federal Resource Management and Ecosystem Services Guidebook* (Durham: National Ecosystem Services Partnership, Duke University, 2014), https://nespguidebook.com.

Figure 3. Example causal chains

(a) Components of an ecosystem service causal chain



(b) Example of an ecosystem services causal chain for wetland restoration and water availability for crops



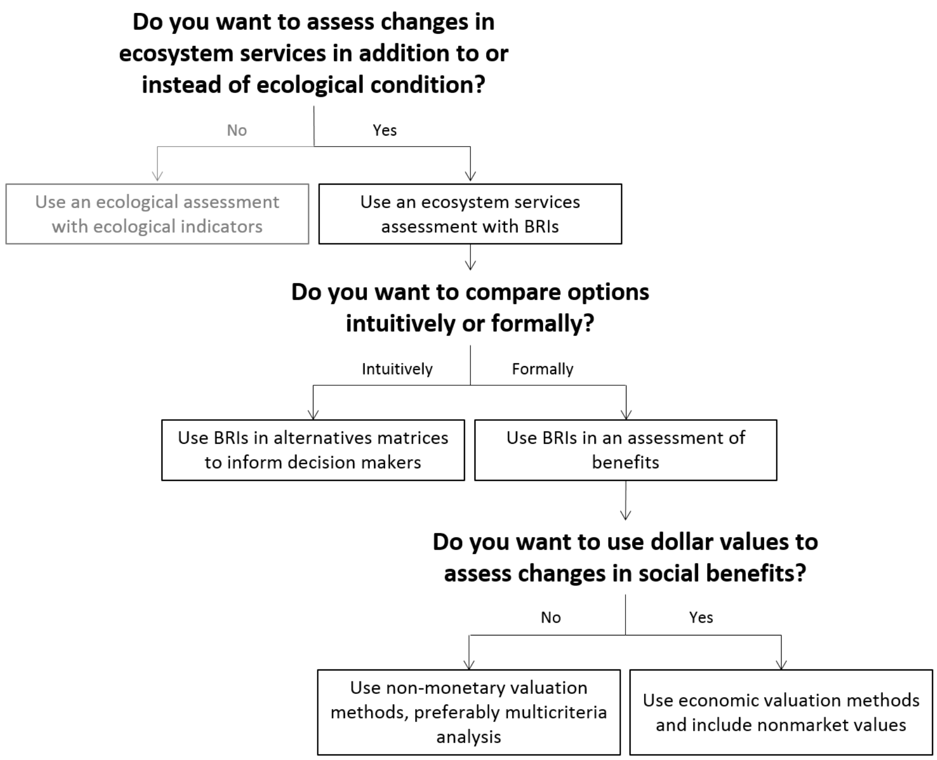
The process of creating causal chains and conceptual diagrams evolves during the analytical process. It may start conceptually, but then indicators are added to make the concepts measureable, followed by the insertion of data, and ultimately it can become the template for a data-driven model used to estimate changes in services expected from a policy or management action (Figure 3b). The indicators identified in the causal chains can be used to monitor changes or assess performance against an objective (e.g., performance metrics).

Benefit-Relevant Indicators

Ecosystem services assessment, at minimum, requires measures of ecosystem changes that matter to people. To quantify these changes we need to use what we call *benefit-relevant indicators* (BRIs). BRIs are measurable indicators that capture the connection between ecosystems and people by considering whether there is demand for an ecosystem service, how much it is used (for use values including services that reduce risks) or enjoyed (for nonuse values), and whether the site provides the access necessary for people to benefit from the service, among other considerations. Many indicators used to capture ecological changes are not easily appreciated by stakeholders as being of direct relevance to personal values or well-being. Chlorophyll or nitrogen concentrations might be good indicators of water quality, but most stakeholders would have little appreciation for or place any particular value on these indicators. Instead, they might simply want to know whether the water is fishable or swimmable.

The analysis phase of an ecosystem services assessment quantifies changes in ecosystem services using BRIs identified in causal chains. BRIs can be used directly in an intuitive decision-making process or as an input into a formal evaluation process in which preferences are quantified through monetary or nonmonetary methods (Figure 4).

Figure 4. Questions to guide an ecosystem services analysis using benefit-relevant indicators



Note: Intuitive comparisons require decision makers to use their knowledge of preferences (stakeholder or institutional) implicitly, rather than to assess them explicitly.

Agencies should use the information in the guidebook to select which of these methods fit their decision-making needs—for example using benefit-relevant indicators alone to estimate changes to the supply of services if monetary valuation or multicriteria decision analysis are not feasible**.** Choosing an assessment technique requires evaluating the type and accuracy of information required in a given policy context. For example, legal proceedings might require different standards from collaborative decision-making with stakeholders. Monetary valuation and multicriteria decision analysis can be used to characterize values and tradeoffs, whereas BRIs alone cannot. In addition, assessment techniques vary in terms of data requirements and the amount of time required, so the availability of funding and other resources will influence choice of methods. Fortunately, the development of BRIs can provide information that is directly relevant to any subsequent evaluation of social benefits, so that identification and potential quantification of BRIs is often useful whether or not there is a current plan to conduct monetary or nonmonetary valuation.

Quantifying BRIs Using Causal Chains

BRIs need to meet two criteria—(1) reflect changes that are relevant to beneficiaries and (2) where relevant capture some aspect of intensity of use and physical and institutional access. BRIs also need to be well-defined measures that are easily reproducible or testable; descriptive narratives alone are not sufficient and cannot be used in assessments of value.

To quantify the change in ecosystem service provision (the changes relevant to beneficiaries) using BRIs, causal chains must be converted into operational empirical models. There are several ways to estimate the relationship between an action (policy, project, management) and its effect on the production of services. One common example, ecological production functions, are ecological models that capture the responsiveness of ecosystem services provision to changes in the environment.

Identification and quantification of the people who could benefit from an ecosystem service—beneficiaries—and where needed, the constraints on access, involves defining the flows of services or servicesheds.[12] A serviceshed analysis is a spatial analysis that incorporates information on where services are produced (what ecosystem and location) and connects this to an assessment of where beneficiaries or risks are located (e.g., downstream, within the airshed, within a one-hour driving distance). For most ecosystem services decision makers need to know not only where these people are, but who they are, how many, and whether they are affected by potential changes in the provision of services (e.g., reduction in flood or fire frequency or intensity). Like other parts of an assessment process, identifying beneficiaries is an iterative process. Perhaps best guesses are used at the outset of the scoping process with more quantification during the analysis that feeds back to inform what stakeholders are engaged in the process.

As a general principle, the scale (geographic extent and time horizon) and types of beneficiaries in an assessment should be matched to the scale and the priority services affected by the decision or planning process. Decisions that affect very few services and only local services can be assessed on a local scale; plans that involve multiple services and geographically far-ranging beneficiaries must be assessed at more extensive spatial scales (and probably over a longer time). In the case of habitat supporting a rare species of mushroom, stakeholders might range from local mushroom collectors to people around the world for whom the mere existence of the species holds value. In many cases, beneficiaries are located beyond traditional boundaries and across jurisdictions. Information on the potential extent of service provision is relevant to an assessment of value and can also help identify beneficiaries, reveal gaps in service provision, or suggest locations where stakeholders might experience a degradation or loss of current services. This information could also highlight possible equity issues and further inform the decision process.

In selecting which BRIs to quantify, it is important to note that some BRIs are better at reflecting the most relevant information about an ecosystem service than others. **The best BRIs will indicate a highly certain link between the environment and a human benefit and will also indicate the intensity of human use, enjoyment, or impact.**

When directly assessing or monitoring the ecosystem service outcomes of an action (perhaps to track performance), a direct measure of a BRI can be used (e.g., number of fish caught by recreational anglers, abundance of iconic species, or number of hospitalizations from airborne particulates during wildfires). Monitoring and assessments need to use well-defined measures that are easily reproducible or testable; descriptive narratives alone are not sufficient.

If the analysis stops with estimated changes in BRIs, managers will have predicted percent or unit changes in measures that represent the outcomes most relevant to people (e.g., particulate levels in airshed and number of people exposed). Changes in these indicators can be incorporated into alternatives matrices to allow a side-by-side comparison of options and then integrated into the decision process (Table 1). While the insight that BRIs alone can provide into tradeoffs is limited, sometimes it is possible to see that one subset of alternatives results in more ecosystem service provision than others (assuming more is better) or that certain alternatives can maintain multiple services without significant losses.

The analysis process increases in complexity as analysts build a comprehensive picture of predicted changes in ecosystem services and social preferences, but it can also be simplified as alternatives are

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[12] Tallis, S. Polasky, J.S. Lozano, and S. Wolny, "Inclusive Wealth Accounting for Regulating Ecosystem Services," in *Inclusive Wealth Report 2012: Measuring Progress towards Sustainability* (Cambridge: Cambridge University Press, 2012).

dropped. Alternatives are dropped as it becomes apparent that they (1) fail to achieve objectives, (2) produce few desired outcomes, (3) create unacceptable tradeoffs, or (4) are too expensive. Some of this winnowing may also be accomplished during the scoping stage, either through expert opinion (e.g., what is technically feasible through management) or by stakeholder consensus (e.g., outcomes nobody cares about). The rest will happen in the analytical process.

Assessing Social Benefits

BRIs capture information on the intensity of stakeholder interaction with services and the number or groups of stakeholders affected by the service. But BRIs do not directly capture stakeholder preferences or values for different levels of performance or provision of service, nor do they capture stakeholders' relative preferences for different services or their willingness to trade one service off relative to another. Thus, formal evaluation of alternatives requires an assessment of benefits considering values or preferences via monetary or nonmonetary methods. An assessment of benefits is needed if (1) service provision varies substantially across different stakeholder populations (i.e., there are differences of opinion about the outcomes) or (2) changes in services in response to management or policy vary in direction (or magnitude) across services. In either case, tradeoffs will have to be made. **When a decision involves tradeoffs (e.g., alternative policies that provide more of some services and less of others), it is often critical to understand the relative value people place on the different services.** Otherwise, it is not possible to know which alternative policy option is preferable. Without an assessment of benefits, the analysis is left with conclusions regarding quantities of *what is valued* (e.g., irrigation water), without any information on *how much they are valued* (e.g., is more irrigation water worth the investment in wetland restoration?). In this guidebook, *assessment of benefits* refers to both economic valuation and nonmonetary multicriteria analysis. To clarify often-used terms, *value* is used in the economic sense to imply well-defined, generally monetary measures of value. *Preference(s)* is often used to reflect how individuals order outcomes on the basis of the relative satisfaction or enjoyment (i.e., utility) they provide rather than using monetary measures; outcomes that generate greater utility also generate greater value. If we align management with particular social values, we also need to consider the impacts of those choices on ecological integrity and the ability of the landscape to provide ecosystem services into the future.

Most regulatory impact analyses require economic valuation of some type, and many other types of federal decisions encourage or require some type of valuation. Office of Management and Budget guidance suggests that assessments of significant federal actions should monetize all primary effects that can be monetized.[13] Monetary expressions of value are often preferred in federal decisions. Expressing all benefits in a common monetary metric allows for analysis of tradeoffs among services and a clear bottom line in terms of net benefits. However, there are limitations to the use of monetary values to express the value of ecosystem services. In some federal decision contexts, the role of economic values is expressly limited.[14] In others, there is reluctance to monetize some kinds of ecosystem services, or the difficulty or expense of estimating monetary values may be large relative to agency resources.[15] Other limitations arise from cultural or religious prohibitions on monetizing some kinds of ecosystem services; cultural values to tribes of spiritual and religious artifacts and sites are a frequently cited example.[16] Nonmonetary methods can be used when dollar values are not desired and when understanding the differences among multiple stakeholder groups' preferences is preferable to the quantification of economic values.

BRIs are important and desirable inputs to assessments of benefits because the measure being valued needs to capture and be responsive to how stakeholders use, interact with, and appreciate ecological outcomes.[17] Hence, BRIs often serve as ideal inputs into models used for valuation.

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[13] Office of Management and Budget, *Guidance on Regulatory Impact Analysis*, 2003, http://www.whitehouse.gov/sites/default/files/omb/inforeg/regpol/circular-a-4_regulatory-impact-analysis-a-primer.pdf.

[14] Examples include the Clean Air Act, the Endangered Species Act, the Resource Conservation and Recovery Act, the Safe Drinking Water Act, and sections of the Fishery Conservation and Management Reauthorization Act, among many others. See K.J. Arrow, M.L. Cropper, G.C. Eads, R.W. Hahn, L.B. Lave, R.G. Noll, P.R. Portney, M. Russell, R. Schmalensee, V.K. Smith, and R.N. Stavins, "Is There a Role for Benefit Cost Analysis in Environmental, Health and Safety Regulation?," *Science* 272 (1996): 221–222. A detailed, illustrative discussion of this issue in the context of fish stock rebuilding is provided by Committee on Evaluating the Effectiveness of Stock Rebuilding Plans of the 2006 Fishery Conservation and Management Reauthorization Act, *Evaluating the Effectiveness of Fish Stock Rebuilding Plans in the United States* (Washington, DC: The National Academies Press, 2014).

[15] L. Wainger and M. Mazzotta, "Realizing the Potential of Ecosystem Services: A Framework for Relating Ecological Changes to Economic Benefits," *Environmental Management* 48 (2011): 710–733.

[16] R. Winthrop, "The Strange Case of Cultural Services: Limits of the Ecosystem Services Paradigm," *Ecological Economics* 108 (2014): 208–214.

Methods for economic valuation have been developed and evaluated over the past five decades and are well established in both the scientific literature and guidance documents.[18] Protocols and standards for these methods document the circumstances in which different types of valuation methods are appropriate. An economist trained in monetary valuation can help decision makers ensure that the translation from BRIs to values is based on the application of valid and reliable methods.

Similarly, methods for multicriteria decision analysis are described in a handbook developed by the London School of Economics to advise local governments on use of multicriteria analysis.[19] Several books address practical applications of these methods, and a primer[20] has been developed to accompany this guidebook.[21] In addition, the U.S. Geological Survey and other federal agencies have developed instructional materials on structured decision making that enable training of agency personnel.[22] People who have participated in this training can be called on by agencies that want to apply these tools for nonmonetary valuation of ecosystem services.

Other approaches to estimate stakeholder preferences include using qualitative estimates of preferences. “Popularity votes” by stakeholders solicited through meetings or electronic forums or by decision makers acting on behalf of stakeholders by proxy may be useful for engaging stakeholders in the scoping stage of an assessment. However, these methods are not viewed by experts as sufficient to determine values or preferences for a formal analysis.

It is important to keep in mind that values are context dependent, as are decisions.

The Decision Process

When comparing alternatives, measures of changes in ecosystem services—what is valued (BRIs) or how much they are valued (dollar values or relative satisfaction)—can be organized by ecosystem service and management into an alternative or decision matrix (Table 1). **The alternatives matrix is a summary of the state of knowledge for a management decision: what is known about the ecosystem and how it might respond to management, stakeholders’ preferences for those changes, and analysts’ confidence in that information.**

When BRIs (in a variety of different units) or multiple types of measures (BRIs and dollars) populate an alternatives matrix, it is difficult for managers to consider changes in services or benefits on an equal footing. This places additional burden on a decision maker especially if there are tradeoffs across services or values. Communication of impacts in a single, directly comparable metric (e.g., money in the case of effects valued using economic valuation) alleviates these “apples versus oranges” comparisons. If economic valuation or multicriteria evaluation is used to assess social impact, aggregated values for each alternative may be the primary outcome of interest, but it could be accompanied by an alternatives matrix to provide greater transparency.

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[17] Evaluation sometimes refers only to economic or monetary valuation methods. In this case, it includes monetary and nonmonetary multicriteria methods. Both approaches include the preferences and values of people; they just use different units (e.g., dollars versus utilities) to do so.

[18] P.A. Champ, K.J. Boyle, and T.C. Brown, *A Primer on Nonmarket Valuation: The Economics of Non-Market Goods and Resources*(New York: Springer, 2003); A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd ed. (Washington, DC: RFF Press, 2014); D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar,*Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington, DC: RFF Press, 2010); National Ecosystem Services Partnership, *Federal Resource Management and Ecosystem Services Guidebook*(Durham: National Ecosystem Services Partnership, Duke University, 2014), https://nespguidebook.com; Office of Management and Budget, *Guidance on Regulatory Impact Analysis*, 2003, http://www.whitehouse.gov/sites/default/files/omb/inforeg/regpol/circular-a-4_regulatory-impact-analysis-a-primer.pdf; U.S. Environmental Protection Agency, Office of Policy, National Center for Environmental Economics, *Guidelines for Preparing Economic Analyses*, 2014, http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0568-50.pdf/$file/EE-0568-50.pdf; Department of Commerce, National Oceanic and Atmospheric Administration, and National Marine Fisheries Service, *Guidelines for Economic Review of National Marine Fisheries Service Regulation Actions*, 2007,http://www.nmfs.noaa.gov/sfa/domes_fish/EconomicGuidelines.pdf.

[19] Department of Communities and Local Government, *Multi-Criteria Analysis: A Manual*, 2009, www.communities.gov.uk. See appendices 1 to 8.

[20] “Ecosystem Services Working Group, Publications,” last modified August 1, 2013, https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

[21] R. Gregory, L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson, *Structured Decision Making: A Practical Guide to Environmental Management Choices* (Oxford: Wiley-Blackwell, 2012).

[22] http://nctc.fws.gov/courses/SDM/home.html.

Table 1. Example of an alternatives matrix

Ecosystem Services	Alternative Management Actions		
	Status quo	Mechanical thinning	Prescribed burning
Fire-risk reduction	*These cells are populated with some measure of the expected change in service provided; when possible, this measure is augmented with measures indicating benefit to people.*		
Timber available for mills			
Wildlife-related recreation			
Water yield			
Cost	*These cells are populated with the costs for each alternative.*		

Ideally an alternatives matrix will be accompanied by information about what was included and what was excluded from the assessment, analysis, or both as well as information about participants in each step, data used and data gaps, assumptions and uncertainties, and an explanation of discarded options. This information increases the transparency of the decision process and facilitates communication of information to stakeholders. Of course, other information will also inform the decision process, including agency mandates and legal authority, cost allocations, local economic effects, and distributional implications (equity). Because these considerations are not particular to the use of ecosystem services considerations in decision making, they are not elaborated on here. **The use of alternatives matrices can communicate information on uncertainty, support objective decision making, and increase the transparency of the decision process. This summary can be a powerful communications tool for managers and stakeholders.**

Providing information on the flow of ecosystem services benefits to stakeholders creates an opportunity for an analysis of the distributional implications of management or policy decisions on different demographic groups, communities, or generations. Such analysis can answer questions such as: Who benefits? Does the relative flow of services to different communities raise equity or environmental justice issues? Are the flows of services sustainable?

The result of a decision process is often that a particular management activity or set of activities is selected as the preferred option and implemented.

Evaluation and Reaction

The adaptive management cycle—plan, act, monitor, react—presumes that the consequences of management actions are monitored. This monitoring often addresses implementation itself: in prescribed fire, managers monitor how effectively the burn was administered; in wetland mitigation, managers monitor the success of restoration activities (e.g., the establishment and short-term survival of plantings). Less frequently, managers monitor the *ecological* effects or *outcomes* of these activities: whether the prescribed fire reduced the incidence of catastrophic fire (e.g., crowning of surface fire) or whether wetland mitigation activities had the desired effect on hydrology, biogeochemistry, or wetland species composition. In short, managers often monitor their management actions but rarely monitor actual desired ecological outcomes—much less ecosystem-derived benefits, which would require tracking BRIs.

On the other hand, social benefits can be monitored directly via proxies (e.g., by counting visitors), but assessing benefits often requires surveys especially for services with nonuse values, which may be difficult for federal agencies to implement.

However, it might be feasible to monitor the ecological indicators on which social benefits depend—the BRIs. An ecosystem services assessment process, with development of causal chains with BRIs, provides an opportunity for managers to identify important BRIs and incorporate them into monitoring programs. Whenever possible, such indicators should be supplemented with information on access and other factors that affect the benefits produced. Such indicators can be used as performance metrics that will assess how programs or projects are doing in terms of achieving goals on the provision or maintenance of particular services and benefits.

The result of this monitoring informs future actions in two ways. First, monitoring of actual management outcomes (whether social benefits, BRIs, or ecological outcomes) provides information on the effectiveness of those actions and therefore provides direct feedback to future management decisions and helps agencies develop more meaningful performance measures. It will also inform development of ecological models of management effects (ecological production functions) that are needed for quantitative ecological assessments. It also feeds back explicitly to the scoping stage of the next planning process, in which management alternatives are weighed in terms of desired outcomes, closing the adaptive management loop.

Monitoring indicators of outcomes rather than management actions can also help agencies update regional status and trends to reflect ecological conditions and flows that are important to people. This information, in turn, would revise the context for future management decisions.

Recommended Reading

U.S. Environmental Protection Agency. 2002. *A Framework for the Economic Assessment of Ecological Benefits.* http://www.epa.gov/stpc/pdfs/feaeb3.pdf.
This EPA framework integrates ecological risk assessment and economics. It covers in greater depth some of the methods presented in this guidebookbut is primarily focusedon economic analysis.

Boyd, J.W., and S. Banzhaf. 2007. "What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units." *Ecological Economics* 63: 616–626.
This article links the concept of final ecosystem goods and services to ecological and social impact analysis or accounting. This concept is used by many agencies and is consistent with the framework presented in this guidebook.

Gregory R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. *Structured Decision Making: A Practical Guide to Environmental Management Choices*. 1st ed. Chichester, UK: R. Blackwell Publishing.
This book describes the use of structured decision making for real-world environmental decision making, involving multiple stakeholders. It is not a "how-to" manual.

Office of Management and Budget. 2003. Circular A-4. http://www.whitehouse.gov/sites/default/files/omb/assets/regulatory_matters_pdf/a-4.pdf.
This memo provides guidance to federal agencies on standard ways to measure and report the benefits and costs of regulatory action. Although not specific to ecosystem services, the memo explains how valuation should be used and identifies other methods for contexts in which valuation is not appropriate.

Ranganathan, J., C. Raudsepp-Hearne, N. Lucas, F. Irwin, M. Zurek, K. Bennett, N. Ash, and P. West. 2008. "Ecosystem Services: A Guide for Decision Makers." Washington, D.C.: World Resources Institute. http://pdf.wri.org/ecosystem_services_guide_for_decisionmakers.pdf.
An introductory guide that discusses the links between development and ecosystem servicers. It includes a general list of ecosystem services on p. 23-24, but does not connect these to beneficiaries. It also provides a high level screening process for assessing risks and opportunities related to ecosystem services p30.

Ruckelshaus, M., E. McKenzie, H. Tallis, A. Guerry, G. Daily, P. Kareiva, S. Polasky, T. Ricketts, N. Baghabati, S. Wood, and J. Bernhardt. 2013. "Notes from the Field: Lessons Learned from Using Ecosystem Services to Inform Real-World Decisions." ***Ecological Economics*****. doi: 10.1016/ j.ecolecon.2013.07.009. http://www.sciencedirect.com/science/article/pii/S0921800913002498.**

This article reviews 20 ecosystem services applications to assess how they informed or influenced decisions. It provides information on when simple models and approaches can be most useful.

BEST PRACTICES

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Reviewers *- Greg Arthaud, U.S. Forest Service; Bruce Carlson, U.S. Army Corps of Engineers; Christine Davis, U.S. Environmental Protection Agency; Chris Hartley, USDA Office of Environmental Markets; Shawn Komlos, U.S. Army Corps of Engineers; Elizabeth Murray, U.S. Army Corps of Engineers; Lynn Scarlett, The Nature Conservancy; George Van Houtven, RTI International; Maria Wegner, U.S. Army Corps of Engineers*

This section is an excerpt from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making."

Three best practices can significantly improve and expand ecosystem service considerations in decision making:

- **Extend assessments beyond purely ecological measures that are not explicitly tied to people's values to measures of ecosystem services that are directly relevant to people.** This task can be accomplished by using ecosystem service values or preferences or by using measures referred to here as ***benefit-relevant indicators*** **(BRIs)**. BRIs reflect well-defined measures of "things valued" by people because they have a direct causal impact on human welfare.
- **Assess ecosystem services using well-defined measures that go beyond narrative description and that are appropriate to the analyses, even when data, time, or resources are limited.** A data-based approach greatly facilitates the use of formal methods for structured decision making and clear communication of the decision process. Various measurement scales can be developed for such analyses, including continuous, categorical, rank order, and interval scales. The key to such measures is that they can be used subsequently in more formal valuation or decision analysis methods. Narrative descriptions or ambiguously defined categories (e.g., high-medium-low, with no measurable criteria defining these categories) are not best practice.
- **Include all important services, even those difficult to quantify**. For federal agencies, "important services" may be defined by legal requirements or policy or by evaluating the magnitude of expected change from an action and the importance of that change to people. Some authorities may allow broad consideration of services across agencies and mandates, like the National Environmental Protection Act (NEPA).[23] However, other authorities and mandates are narrower and will not include all services. Although decision making is clearly better when all significantly affected services that matter to people are included, doing so may require increased coordination across authorities, agencies, and other affected entities to achieve this.

[23] D. Bear, "Integration of Ecosystem Services Valuation Analysis into National Environmental Policy Act Compliance: Legal and Policy Perspectives," in Federal Resource Management and Ecosystem Services Guidebook (Durham: National Ecosystem Services Partnership, Duke University, 2014), http://www.nespguidebook.com.

SCOPING

Understanding Socio-Cultural Context
Conceptual Diagrams
Identifying Services

UNDERSTANDING SOCIO-CULTURAL CONTEXT

Authors - *Rob Winthrop and Christy Ihlo*

Reviewers *- Kenneth J. Bagstad, U.S. Geological Survey; Jimmy Kagan, Institute for Natural Resources, Oregon State University and Portland State University*

The flow of ecosystem service benefits is always mediated by social systems. Every human use of nature has a socio-cultural context: relatively enduring relationships and understandings among individuals and groups that shape both the ends and means of actions affecting ecosystems. This context can determine the nature of received ecosystem benefits, their value, and their beneficiaries and non-beneficiaries.

Humans do not experience their environment as an external and objective reality. Rather, "nature is seen by humans through a screen of beliefs, knowledge, and purposes, and it is in terms of their images of nature, rather than of the actual structure of nature, that they act."[24] Such "images of nature" are not universal. Although many societies consider pigs a valuable source of meat, Islamic and Judaic communities consider them to be unclean animals, not to be eaten.[25] For these religious communities, pigs provide no provisioning service.

Socio-cultural shaping of the flow of ecosystem benefits can be seen in the social mapping of fuels within a landscape. In the Peruvian Andes, for example, rights to fuel wood are determined by multiple factors. For fuel from planted trees, these include "community residency, house and field ownership, and the degree of human labor in tree planting and harvest—a complex mix of ownership and usufruct [use] rights." In contrast, trees in the uncultivated monte are a common pool resource; rights to fuel are conveyed by membership in a nearby community.[26]

In the United States, access to lands and resources for non-commercial hunting, fishing, and gathering is shaped in complex ways by socio-cultural factors, involving both law and customary practice. Two centuries of American Indian treaties and case law have overlaid the territory of the United States with a grid of native resource rights. In the post-contact era, pre-contact tribal fishing practices at "usual and accustomed" places were recognized as rights retained by tribes, even when the use of such sites required access across private property.[27] Similarly, in rural America, it is customary to hunt on other people's land. Though the landowner's permission is required, the practice is sufficiently common to be recognized in state fish and game regulations.[28]

Human ecology mapping(HEM) is becoming popular as a tool to chart the complex connections between humans and landscapes. Human ecology mapping refers to a broad suite of techniques, including community values mapping, counter-mapping, cultural opportunities mapping, landscape values mapping, mental mapping, participatory mapping, place-based mapping, public-participation geographic information system mapping, and social values mapping.

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[24] Roy A. Rappaport, *Ecology, Meaning, and Religion* (Richmond, CA: North Atlantic Books, 1979), 97.

[25] K. Maxwell, "Beyond Verticality: Fuelscape Politics and Practices in the Andes," *Human Ecology* 39 (2011): 465–78.

[26] The key case is *United States v. Winans*, 198 U.S. 371 (1905). See V. Mulier, "Recognizing the Full Scope of the Right to Take Fish under the Stevens Treaties: The History of Fishing Rights Litigation in the Pacific Northwest," *American Indian Law Review* 31 (1) (2006): 46–50, doi: 10.2307/20070773.

[27] See, e.g., Minnesota Department of Natural Resources, *2014 Minnesota Hunting & Trapping Regulations Handbook*, MinnesotaDepartment of Natural Resources, http://files.dnr.state.mn.us/rlp/regulations/hunting/2014/full_regs.pdf.

[28] K. M. Chan, A.J. Goldstein, T. Satterfield, N. Hannahs, K. Kikiloi, R. Naidoo, N. Vadeboncoeur, and U. Woodside, "Cultural Services and Non-Use Values" in *Natural Capital: Theory and Practice,* edited by P. Kareiva, H. Tallis, T. H. Ricketts, G. C. Daily, and S. Polasky, 206–228 (New York, NY: Oxford University Press, 2011); B.C. Sherrouse, J.M. Clement, and D.J. Semmens, "A GIS Application for Assessing, Mapping, and Quantifying the Social Values of Ecosystem Services," *Applied Geography* 31 (2011):748–760; and T.C. Daniel, A. Muhar, A. Arnberger, O. Aznar, J.W. Boyd, K.M.A. Chan, R. Costanza, T. Elmqvist, C.G. Flint, P.H. Gobster, A. Gret-Regamey, R. Lave, S. Muhar, M. Penker, R.G. Ribe, T. Schauppenlehner, T. Sikor, I. Soloviy, M. Spierenburg, K. Taczanowska, J. Tam, and A. von der Dunk, "Contributions of Cultural Services to the Ecosystem Services Agenda," *Proceedings of the National Academy of Sciences of the United States of America* 109 (2012): 8812–8819.

Aside from helping analysts quantify (though not monetarily value) cultural ecosystem services that cannot be assessed using more traditional biophysical modeling and nonmarket valuation techniques, HEM offers a promising approach to mapping cultural ecosystem services.[29] It couples existing and emerging technologies to visually capture the interactions in socio-ecological systems, helping analysts to answer questions such as

- Where do conflicts arise over land rights, uses, and access?
- How and why does the spatial distribution of human activity vary temporally (seasonally, annually, and so on)?
- What values are associated with sites within the project area?

Therefore, HEM may be a valuable tool in understanding the socio-cultural context of ecosystem services.

Different situations will require greater or lesser attention to the socio-cultural context of ecosystem services provision and value. Understanding how social systems mediate the human experience of the environment and the consequences of environmental change is basic to several disciplines, including environmental anthropology, environmental sociology, and human geography. Ideally, understanding the social-cultural context for the natural resource management decision at hand will be one of the first steps in an analysis of that decision.

Recommended Reading

Allen, S.D., D.A. Wickwar, F.P. Clark, R. Potts, and S.A. Snyder. 2009. *Values, Beliefs, and Attitudes Technical Guide for Forest Service Land and Resource Management, Planning, and Decision-making*. Gen. Tech. Rep. PNW-GTR-788. Pacific Northwest Research Station, Forest Service, U.S. Department of Agriculture, Portland, OR. http://www.fs.fed.us/pnw/pubs/pnw_gtr788.pdf.
This Forest Service guide provides a conceptual overview defining values, beliefs, and attitudes and suggesting ways for incorporating them into decision processes.

McLain, R., M. Poe, K. Biedenweg, L. Cerveny, D. Besser, and D. Blahna. 2013. "Making Sense of Human Ecology Mapping: An Overview of Approaches to Integrating Socio-Spatial Data into Environmental Planning." *Human Ecology* 41 (5): 651–65. http://link.springer.com/article/10.1007%2Fs10745-013-9573-0.
This synthesis paper describes several techniques of human ecology mapping.

Vaccaro, I., E.A. Smith, and S.Aswani, eds. 2010. *Environmental Social Sciences: Methods and Research Design*. Cambridge, UK: Cambridge University Press. http://www.amazon.com/Environmental-Social-Sciences-Methods-Research/dp/052111084X.
This book summarizes methods and research strategies for social research focused on environmental issues.

Winthrop, R. H. 2014. "The Strange Case of Cultural Services: Limits of the Ecosystem Services Paradigm." Ecological Economics 108:208-214. http://www.sciencedirect.com/science/article/pii/S0921800914003152.
This article discusses the limitations of ecosystem services frameworks in incorporating cultural services using several examples from American Indian communities of the Pacific Northwest.

[29] R. McLain, M. Poe, K. Biedenweg, L. Cerveny, D. Bessert, and D. Blahna, "Making Sense of Human Ecology Mapping: An Overview of Approaches to Integrating Socio-Spatial Data into Environmental Planning," *Human Ecology* 41 (2013): 651–665.

CONCEPTUAL DIAGRAMS

Authors - *Lydia Olander, Christy Ihlo, Dean Urban, Robert J. Johnston, Heather Tallis, Jimmy Kagan, Lynn Maguire, Steve Polasky, James Boyd, Lisa Wainger, and Margaret Palmer*

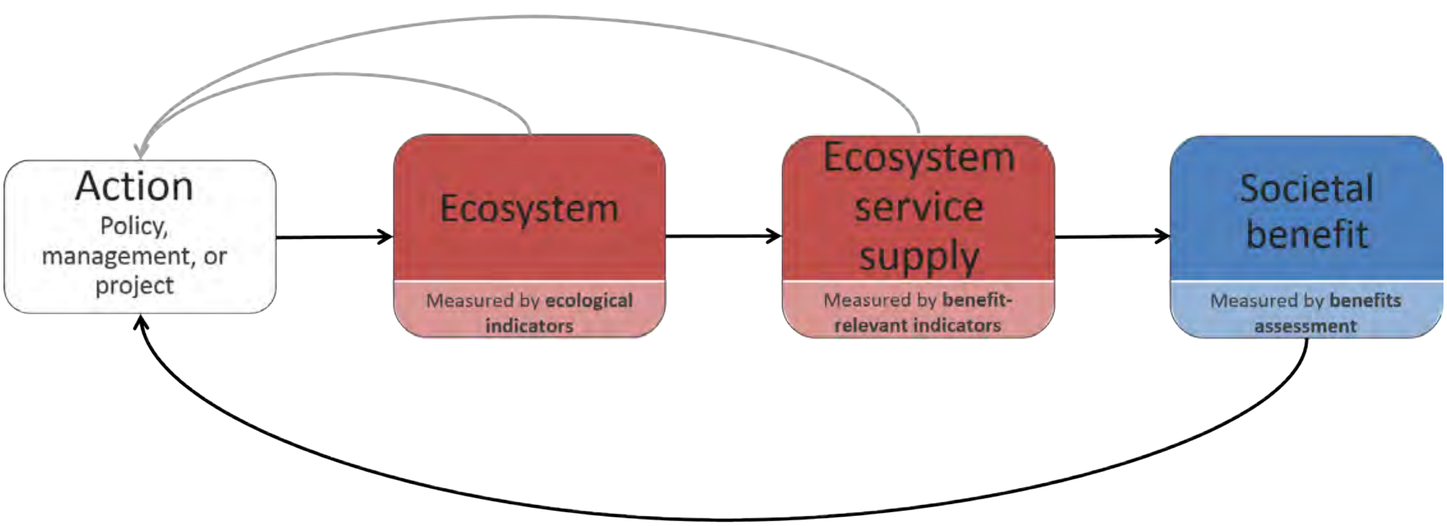
Reviewers - *Nikola Smith, David Yoskowitz*

This section includes excerpts from "Best Practices for Integrating Ecosystem Services into Federal Decision Making" and an earlier version of this guidebook as well as new content.[30]

In an ecosystem services assessment, conceptual diagrams—also known as means-ends diagrams—provide a systematic approach to connecting ecological conditions and societal benefits. They are composed of multiple causal chains, wherein each chain is a logical model that declares how ecological conditions (current, desired, or changed by a management action or policy) affect the provision of ecosystem services and benefits to various segments of society.

Causal chains are commonly used in ecological assessments but are slightly different when used in an ecosystem services assessment. As in an ecological assessment, they begin with either ecological conditions (e.g., current healthy wetland habitat) or the action or policy affecting those conditions (e.g., wetland restoration or invasive species removal). However, they end with effects on human well-being caused by changes in ecosystem services, rather than expected environmental changes or outcomes, which are common endpoints for ecological assessments. When connections to people are not made explicit, it is unclear whether and how each ecological change is related to changes in social benefits, and important changes to societal benefits may be left out of the analysis. For this reason, causal chains in an ecosystem services assessment extend to human well-being (Figure 5). As part of scoping, the causal chains embedded in conceptual diagrams may be preliminary best guesses requiring relatively little effort yet providing a comprehensive overview of all potentially significant services. More robust quantifiable causal chains are needed in the analysis.

Figure 5. Components of an ecosystem service causal chain



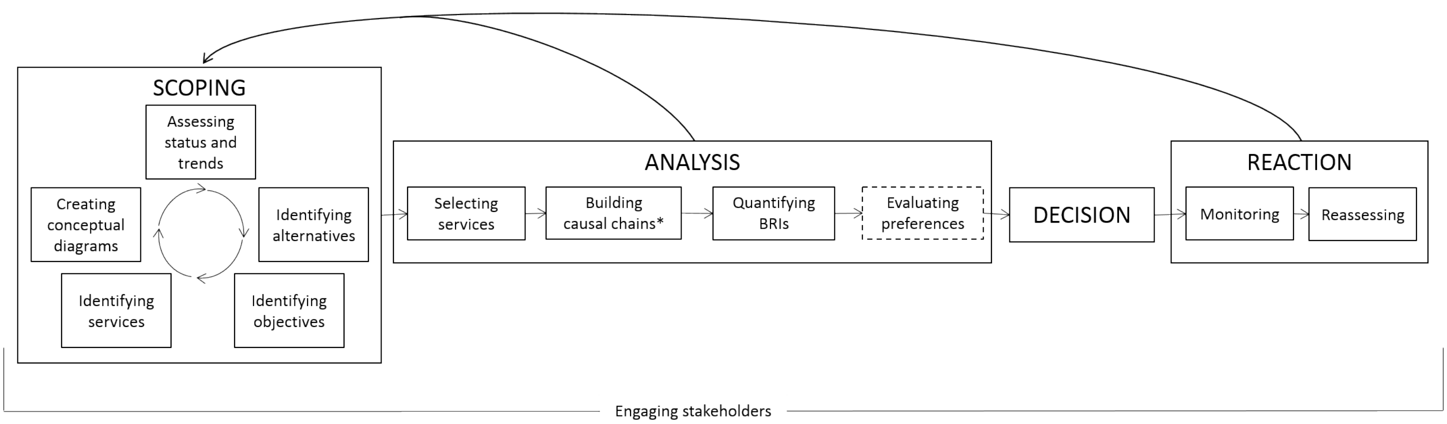
As mentioned above, causal chains used in conceptual diagrams can be used to evaluate current conditions, desired conditions, and/or changes caused by a management alternative. In an assessment of current or

30 L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

desired conditions, these diagrams reveal the services provided and help stakeholders and decision makers consider what services they may want to enhance or sustain when setting future objectives. In alternatives assessment, they display how management options will change the biophysical landscape (and related human behaviors) and show how those changes relate to the provision of ecosystem services and ultimately societal benefits. These alternatives can include individual changes in conditions at one site or comparisons of changes in condition for multiple sites or for combinations of actions that make up a management scenario or management plan. They also reflect changes in human behavior that may be expected as part of these ecosystem changes, such as increases in recreational visits (and hence benefits) that might be expected when relevant conditions improve at sites used for recreation. Causal chains can also be used to consider project options (e.g., funding wetlands restoration in Florida versus Minnesota) or policy choices, which are evaluated in terms of the actions they imply; for example, policies affecting incentive payments for protecting riparian zones would be evaluated in terms of the changes to those zones. Conceptual diagrams linked to alternatives will be the inputs for the analysis step.

Creating conceptual diagrams is one of many simultaneous and interactive scoping activities, the others being assessing current status and trends (both ecological and social) and identifying services, objectives, and alternatives (Figure 6). Stakeholders can be engaged in all these activities. For example, by identifying services they care about, stakeholders can provide information directly relevant to the development of causal chains and conceptual diagrams. Conceptual diagrams can also serve as communications tools with stakeholders. For example, resource managers often think about outcomes in terms of ecological conditions alone—healthy long-leaf pine forest, fire-resilient riparian ecosystems, or hydrologically functional wetlands. Unlike managers, stakeholders may be apt to think about concrete experiences and opportunities when thinking about changes to their surroundings: areas available for fishing, trails open for hiking, water available for irrigation, risk of flood or fire to personal property. In other words, stakeholders think in terms of *ecosystem services,* though they may not use that term, whereas resource managers may think in terms of the *conditions that lead to those services.*

Figure 6. Simultaneous and interactive scoping activities initiate the decision process

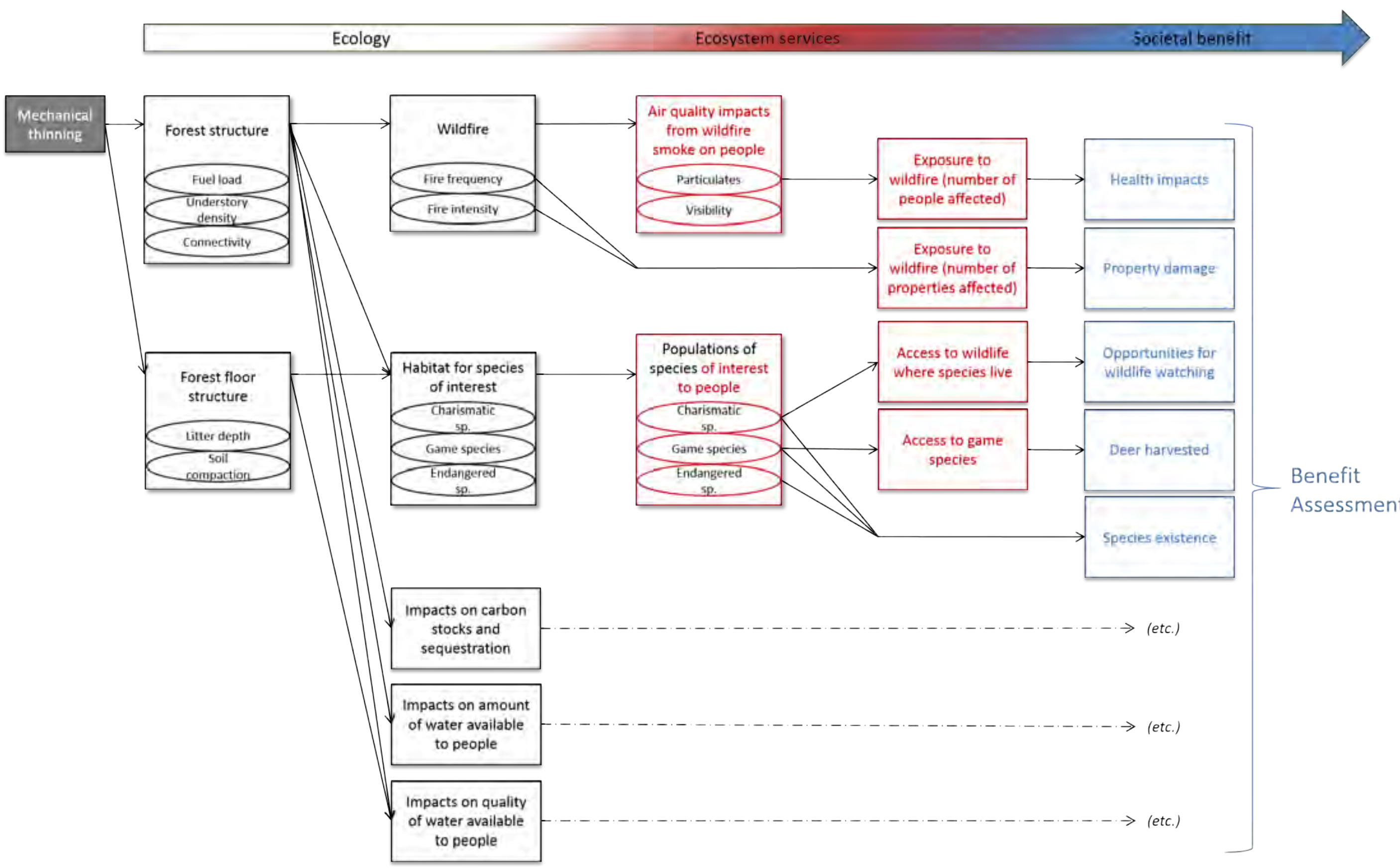


Note: While less formal, perhaps narrative, causal chains can be used in the scoping process as part of conceptual diagrams, it will be necessary to fully flesh out causal chains incorporating indicators and models for quantification during the analysis.

At the end of the scoping process, practitioners should have a conceptual diagram, built around causal chains, for current conditions and for each proposed action or option. These causal chains may include some draft indicators but can be made into measurement models with specific indicators and data later in the analysis process.

Developing conceptual diagrams is a critical step to ensure that ecosystem services assessments are relevant, comprehensive, and transparent. Conveniently, they do not require particular software or technological expertise to develop. Rather, practitioners should take adequate time to brainstorm with experts and stakeholders about possible interactions and impacts and then refine them over time to capture the suite of interactions and impacts as completely as possible. This diagramming process can help identify how a policy or management action can affect multiple aspects of an ecosystem and how each of the impacts on an ecosystem can have multiple impacts on social benefits. For example, mechanical thinning of forests is used to reduce the risk of catastrophic wildfire. Thinning affects forest structure, which changes not only the intensity of fires but also species habitat, risk of pest and pathogen outbreaks, and forest carbon storage. Each of these ecological changes can then be followed down individual causal chain branches of the conceptual diagram to one or more ecosystem services and anticipated human benefits. All possible impacts to valued services should be included in the diagram, even those likely difficult to measure or model or likely to have only minor effects on people. This transparency enables practitioners to explain to stakeholders why only select services are carried forward in further analysis and hopefully reduces accusations of forgetting or ignoring services. Because all impacts are included, this process will likely identify too many services to be meaningfully quantified in any ecosystem services assessment. Those effects likely to be most important to the decision—often those expected to have the largest impacts on human welfare—can be targeted for quantitative analysis (see “Selecting Ecosystem Services”). However, by identifying the full range of pathways through which actions can influence people regardless of jurisdiction, these diagrams can often provide important insights for decision making and present opportunities to identify necessary partners and stakeholders.

Figure 7. Conceptual diagram with causal chains for an ecosystem services assessment of a forest management alternative



Note: This conceptual map of simplified causal chains shows possible outcomes from forest fire management activities like mechanical thinning. Black text indicates an ecological assessment and indicators, red text indicates extension to an ecosystem services assessment, and blue text indicates measures of social benefit and value. (See Figure A-1 in the appendix for expanded version).

Once conceptual diagrams are developed, practitioners can select a suite of services for analysis and build quantifiable causal chains.

Best Practice Questions: Creating Conceptual Diagrams for Ecosystem Services Using Causal Chains

To follow best practice in considering current conditions and objectives, the assessor should be able to answer *yes* to this question:

- Have all services that people care about been included in the diagram (even if they will not all be included in the final analysis)?

To follow best practice in considering changes in conditions, the assessor should be able to answer *yes* to ALL of these questions:

- Have all effects of a policy, management decision, or program on ecological conditions been included?
- Have the changes in ecological conditions that lead to changes in the delivery of affected ecosystem services been included?
- Have the effects on individuals or groups from changes in the delivery of ecosystem services been included?
- Have all impacts that people care about been included in the diagram (even if they will not all be included in the final analysis)?

IDENTIFYING SERVICES

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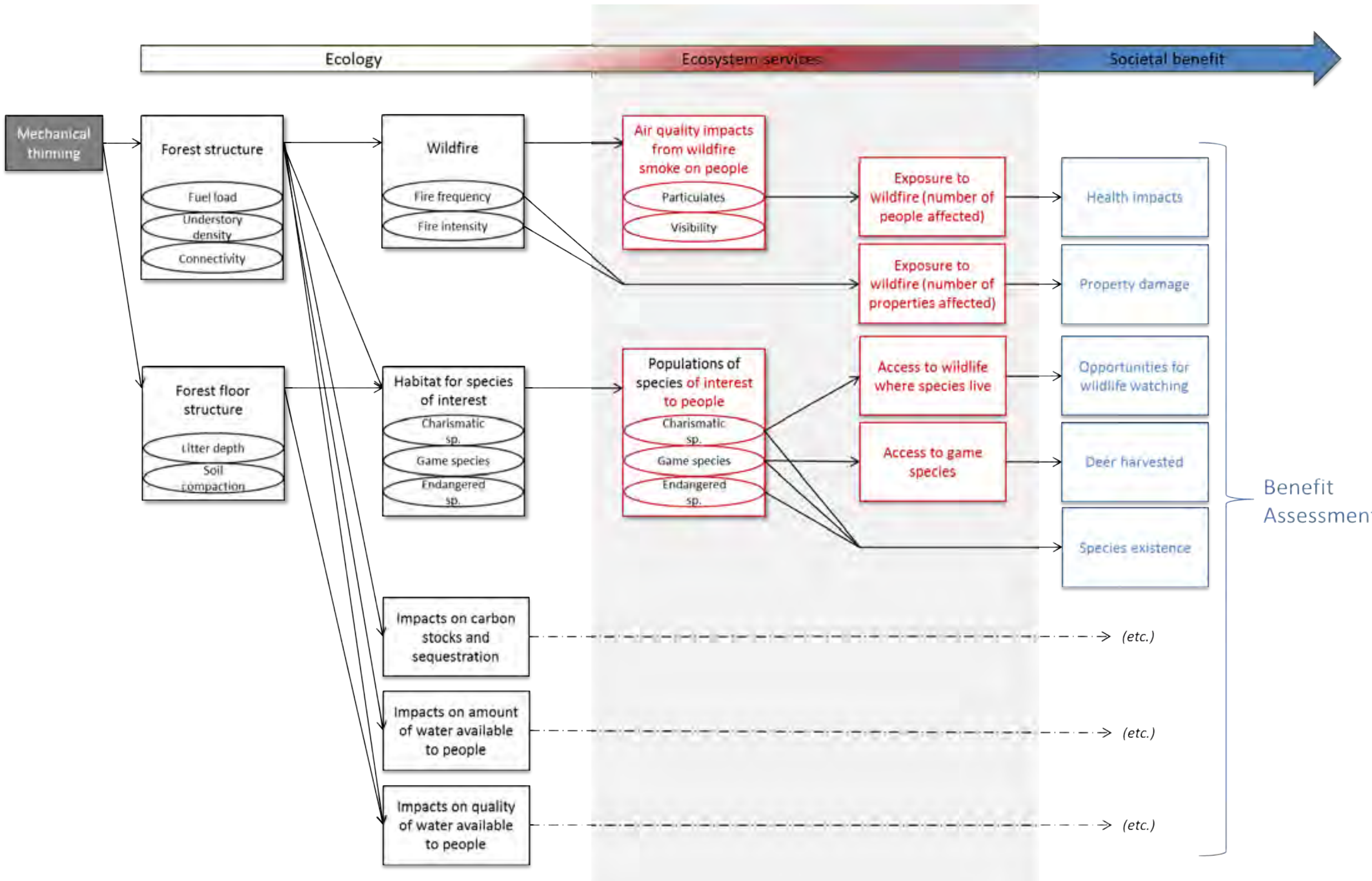
This section adapted from excerpts from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making." [31]

Ecosystem services important to a decision process are identified through the process of developing a conceptual diagram that connects policy or management alternatives to the affected ecosystem services and beneficiaries using causal chains. When analyzing changes in ecological conditions brought about by mechanical thinning in a forest, for example, managers can think through how those changes will affect the provision of services and benefits to people. This conceptual mapping process will identify the services important for the decision at hand.

[31] L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

Figure 8. Conceptual map of causal chains indicating possible outcomes of a forest-fire management activity



Note: Black text indicates an ecological assessment and indicators, red text indicates an extension of an ecosystem services assessment, and blue text indicates measures of social benefit and value. (See Figure A-2 in the appendix for expanded version).

When developing a conceptual diagram, it might be tempting to refer to a "master list" of services to help ensure no services are inadvertently excluded. Such lists are created in an effort to classify ecosystem services.[32] Generic lists of services can provide a useful starting point for considering which services and beneficiaries are relevant in a decision context, but given context-specific variation in services, generic classifications will almost always be insufficient and can often be misleading. The reason is the fundamental incapacity of *any* generic classification to capture context-specific variations that are critical to linkages between ecosystems and human value and that will occur no matter how much effort has been put into development of the classification system. Creating conceptual diagrams using causal chains will reveal location-specific considerations in ways that generic classification cannot.

One reason for development of classification systems is to avoid double counting; some classification systems may be useful in achieving this goal.[33] However, double counting can be avoided more effectively using context-specific causal chains (see Building Causal Chains for details).

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[32] Millennium Ecosystem Assessment, *Ecosystems and Human Well-Being: Synthesis* (Washington, D.C.: Island Press, 2005); European Environment Agency, Common International Classification of Ecosystem Services (2015), http://cices.eu/.; D.H. Landers and A.M. Nahlik, *Final Ecosystem Goods and Services Classification System (FEGS)*, EPA/600/R-13/ORD-004914, 2013, http://ecosystemcommons.org/sites/default/files/fegs-cs_final_v_2_8a.pdf; P. Sinha and G. Van Houtven, *National Ecosystem Services Classification System (NESCS): Framework Design and Policy Application*, draft report prepared for the U.S. Environmental Protection Agency.

[33] The Millennium Ecosystem Services Assessment provides a commonly used classification of services. This classification was not generated using a causal chain approach and can result in double counting.

Classification Systems

A number of different systems have been developed and are under development to classify ecosystem services into categories. These systems are intended to increase consistency in use. Decision-specific classification of services may be helpful in some contexts, though they are never essential. The Common International Classification of Ecosystem Services (CICES) is being designed to support incorporation of ecosystem services into national accounts.[a] National accounts have strict rules about double counting, but the inconsistent ways that ecosystem services are named and distinguished can make it difficult to avoid. The CICES classification, if well designed, will help ensure that the rules of the national accounting decision context are followed. The Environmental Protection Agency is developing two other services classification systems: the Final Ecosystem Goods and Services Classification System (FEGS-CS) and the National Ecosystem Services Classification (NESCS).[b] Both are intended to enhance consistency across decisions at different scales. Existing classifications should be used with caution and interpreted using context-specific causal chains.

Notes:

[a] S. Polasky, H. Tallis, and B. Reyers, "Setting the Bar: Standards for Ecosystem Services," *Proceedings of the National Academies of Science of the United States of America* 112(24)(2015): 7356–7361; S. Banzhaf and J. Boyd, "The Architecture and Measurement of an Ecosystem Services Index," *Sustainability* 4(4)(2012):430–461; J. Boyd, "The Nonmarket Benefits of Nature: What Should Be Counted in Green GDP?," *Ecological Economics* 61(4)(2006):716–723.

[b] European Environment Agency, Common International Classification of Ecosystem Services (2015), http://cices.eu/; D.H. Landers and A.M. Nahlik, Final Ecosystem Goods and Services Classification System (FEGS), EPA/600/R-13/ORD-004914, 2013, http://ecosystemcommons.org/sites/default/files/fegs-cs_final_v_2_8a.pdf; P. Sinha and G. Van Houtven, *National Ecosystem Services Classification System (NESCS): Framework Design and Policy Application,* draft report prepared for the U.S. Environmental Protection Agency, http://water.epa.gov/learn/confworkshop/upload/FINAL-Summ-WS2-NESCS.pdf.

BENEFIT-RELEVANT INDICATORS

What are BRIs?

WHAT ARE BENEFIT-RELEVANT INDICATORS?

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This section consists primarily of excerpts from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making," [34] *with a small addition from the previous version of the* Federal Resources Management and Ecosystem Services Guidebook *("Bridging Indicators" by James Boyd and Lisa Wainger).*

Ecological features and processes are essential for the provision of ecosystem services but are not the same as services.[35] Until there is some person somewhere who benefits from a given element or process of an ecosystem, that element or process is not a service. *Benefit-relevant indicators* (BRIs) are measurable indicators that capture this connection by considering whether there is demand for the service, how much it is used (for use values) or enjoyed/valued (for nonuse values), and whether the site provides the access necessary for people to benefit from the service, among other considerations.[36] An ecological measure can become a BRI if it is tied directly and causally to something important to people, e.g., the presence of bald eagles, which are clearly identified as important to the American public.

BRIs can also be measures of a disservice that result in lower rather than higher benefits. For example, wolves can create a disservice to ranchers who lose livestock to predation. In other cases, BRIs provide positive benefits up to a certain quantity, above which point benefits may become negative. For example, many wildlife species (for example, deer) are valued for recreational (e.g., hunting, viewing) and existence purposes up to a certain density, but at higher densities they are viewed as pests (e.g., due to damage caused to crops and landscaping). Hence, some BRIs will not have an unambiguously positive or negative impact on human welfare and may in fact simultaneously have positive impacts for some groups in society and negative impacts for other groups.

BRIs are not the only indicators that should be measured, nor do they represent the only things in nature that are valuable. They are simply indicators meant to improve the linkage between ecological analysis on the one hand and public policy, social evaluation, and lay communication on the other hand.

BRIs fulfill two important needs for ecosystem services assessments. The first is the need for indicators that are socially comprehensible, in that lay audiences can relate them clearly to their own well-being. Biophysical outcomes that are directly experienced, comprehended, and acted on are more amenable to social interpretation.

The second need is for indicators that enhance the accuracy of social evaluations. Just because lay audiences see a causal connection between an ecological outcome and their welfare doesn't mean they will see it accurately. Accordingly, it is desirable to choose indicators that reduce the need for lay audiences, social science researchers, or both to estimate or speculate about the relationship between the indicator and the production of services. For example, it is more appropriate to ask people to assess how much they value safe drinking water than it is to ask them to assess how much they value the nutrient cycling that contributes to the quality of the water. Outcomes nearer to human experience minimize lay audiences' need to speculate about ecological relationships and thereby lead to better, more accurate social evaluation.

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[34] L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

[35] H. Tallis, S.E. Lester, M. Ruckelshaus, M. Plummer, K. McLeod, A. Guerry, S. Andelman, M.R. Caldwell, M. Conte, S. Copps, D. Fox, R. Fuita, S.D. Gaines, G. Gelfenbaum, B. Gold, P. Kareiva, C. Kim, K. Lee, M. Papenfus, S. Redman, B. Silliman, L. Wainger, and C. White, "New Metrics for Managing and Sustaining the Ocean's Bounty," Marine Policy 36 (2011): 303–306.

[36] National Ecosystem Services Partnership, Federal Resource Management and Ecosystem Services Guidebook (Durham: National Ecosystem Services Partnership, Duke University, 2014), https://nespguidebook.com/assessment-framework/benefit-relevant-indicators/.

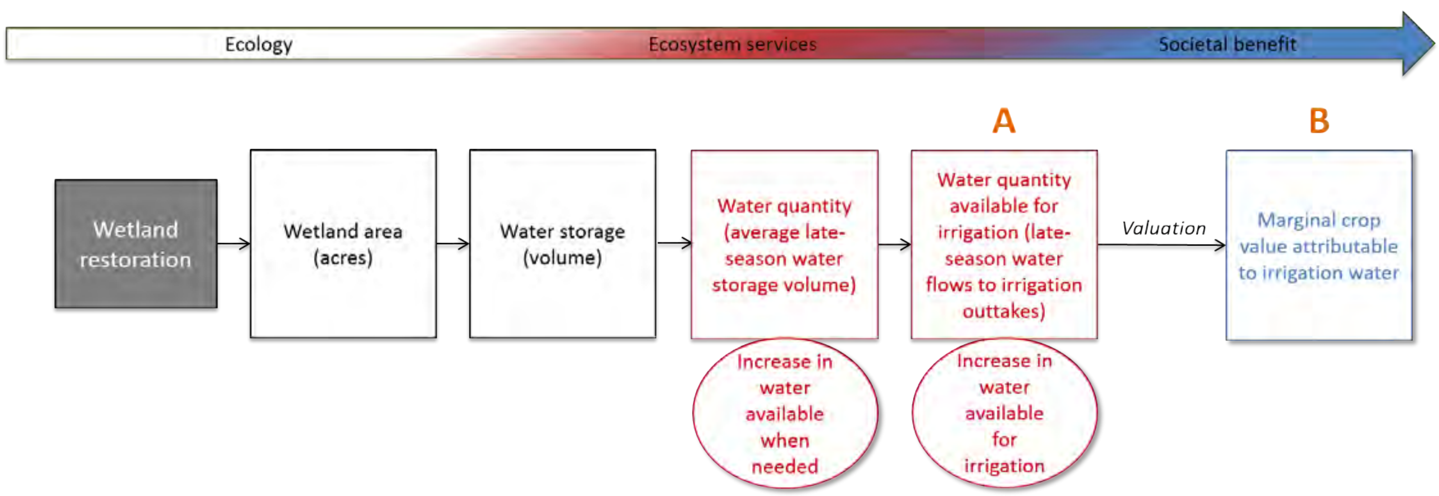
A few key questions can assess if an indicator is benefit relevant.

Does the indicator reflect changes in ecological condition that are relevant to the beneficiaries?

BRIs must reflect changes in ecological condition (they must be good indicators of the ecological changes), and the changes must be relevant to people. As an example of ecological condition, marsh, reef, or mangrove habitat are all known to dampen incoming waves and, in so doing, protect coastal areas from erosion and inundation. For this service, habitat area is not the most relevant ecological metric; multiple studies have shown that the leading offshore habitat edge plays a disproportionate role in dampening waves compared with more interior acres of habitat. In this case, contiguity of offshore habitat edge is the appropriate ecological indicator to reflect a step on the causal chain for a coastal protection ecosystem services assessment.

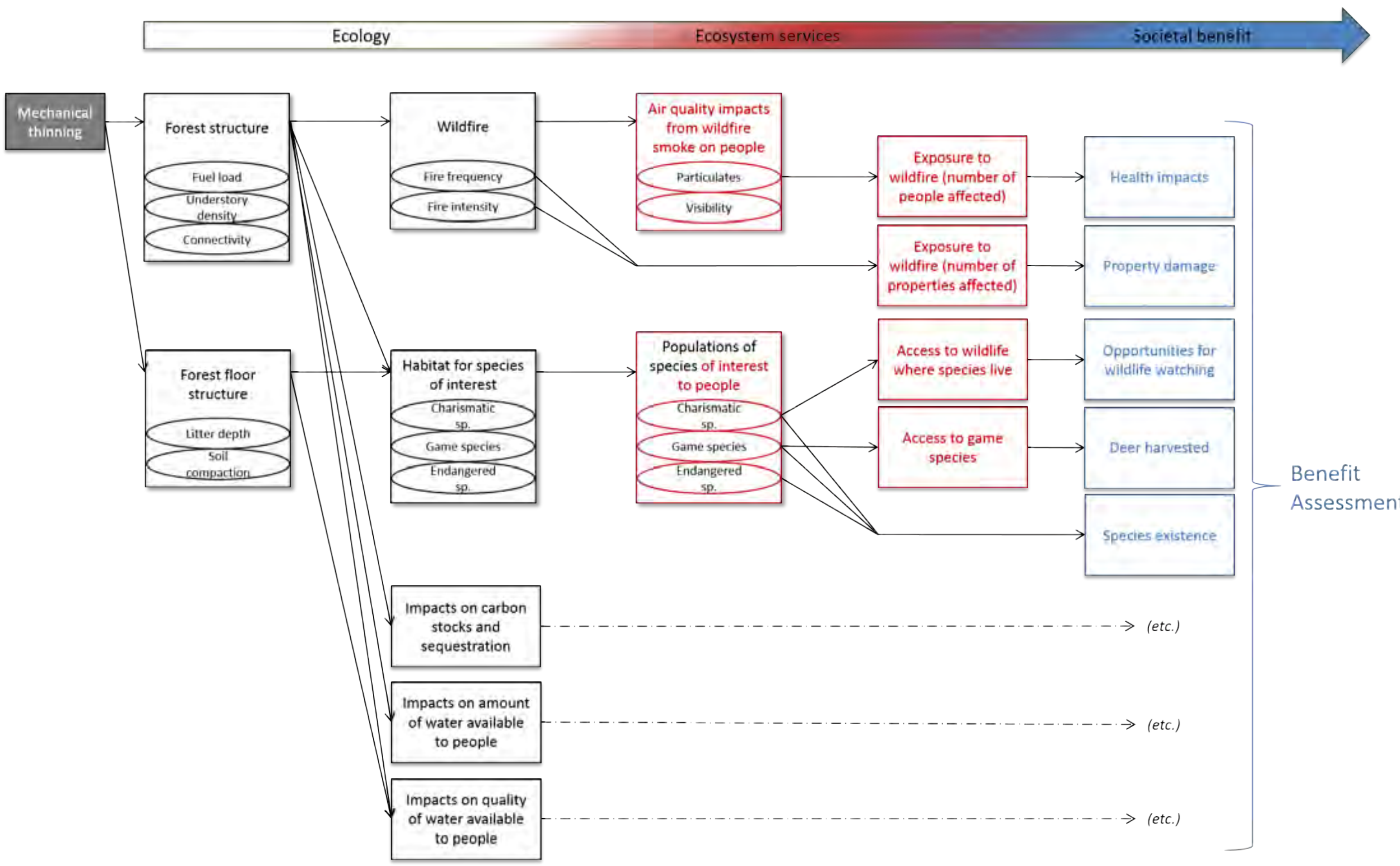
An indicator becomes benefit *relevant* when it is cast in units that resonate with stakeholders as something that affects their welfare proximally (Table 2). For example, "numbers of catchable fish" is more relevant to fishers than other measures such as dissolved oxygen content in the water or an index of biotic integrity—even though water quality might directly influence fish populations. Similarly, in the causal chain connecting a change in forest management to changes in the risks of wildfire, the BRI emerges when fire behavior is translated into units directly relevant to human health (Figure 9). Likewise, fire behavior might be translated into other BRIs connecting fire to other services of concern for various stakeholder groups such as hikers or homeowners. As a simple rule of thumb, if reasonably well-informed members of the beneficiary groups affected by an ecological change (e.g., those whose health is affected by airborne particulates) cannot easily understand why an indicator is relevant to their welfare, it is unlikely that the indicator is an effective BRI.

Figure 9. Differences between ecological and ecosystem services assessments and indicators



Note: Causal chains consider expected outcomes from forest fire management activities like mechanical thinning. Black text indicates an ecological assessment and indicators, red text indicates extension to an ecosystem service assessment, indicators within ovals illustrate BRIs, and blue text indicates measures of social benefit and value. The demarcation among ecology, ecosystem services, and social benefits is not absolute (the lines between categories are drawn differently by different people), as represented by the tri-colored arrow.

Figure 10. Conceptual diagram with causal chains for an ecosystem services assessment of a forest management activity



Note: This conceptual map of simplified causal chains shows possible outcomes from forest fire management activities like mechanical thinning. Black text indicates and ecological assessment and indicators, red text indicates extension to an ecosystem services assessment, and blue text indicates measures of social benefit and value. (See Figure A-1 in the appendix for expanded version).

Does the indicator capture relevant physical and institutional access constraints on the flow of the service?

Many ecological measures and indicators used in ecological assessments fulfill the first requirement of a BRI because extensive research has identified sensitive interactions in the environment of interest to people. However, a BRI must capture only those ecological components and processes that can be enjoyed or used by people for some benefit. Capturing these components and processes requires information on relevant physical or institutional limits on people's ability to access (physically or otherwise) a benefit. For example, for the service of timber production, tree density alone is not a sufficient BRI. Physical infrastructure such as roads or features such as terrain may limit tree harvests in some areas. Separately, legal restrictions may limit physical access to areas with trees (e.g., protected areas) or regulate harvest rates or areas (e.g., through riparian buffer restrictions). A BRI must reflect these constraints so that the flow of services is not overestimated. In this case, a BRI would be the density and size of harvestable trees accessible to forest managers.

BRIs are relevant to all ecosystem services, including those with nonuse values such as existence, educational, and spiritual values. When people value the existence of an old-growth forest, a historical or culturally important place, or particular species like bald eagles or endangered tortoises, BRIs need to represent the elements that impart value to people, including the presence, quantity, quality, and sustainability of these places, habitats, or species. When species or ecosystems are protected by law, BRIs are likely to consist of well-constructed ecological metrics because laws are evidence of public interest. When a species or ecosystem is not federally protected, agencies may use BRIs to represent other types of evidence of people's values such as conservation priorities developed by nongovernmental groups.

Alone, ecological measures may be insufficient to reflect an ecosystem service, but many are important components of causal chains that link agency actions to BRIs (see Table 2). An indicator becomes a BRI once it reflects the relevant links in a causal chain ending with the potential benefit of a service to an identifiable group of people. Choosing BRIs from links in established causal chains is critical for ensuring that a metric is specific enough to reflect the ecosystem condition causally tied to a human benefit.

Table 2. Examples of what would and would *not* qualify as a BRI

Ecosystem service	Not BRI	BRI
Existence or abundance of wolves	People donating to general conservation organizations[a]	Number of wolves x number of people holding existence value for wolves
Ecological production of commercially harvested fish	Fish abundance	Amount of fish landed commercially by Native Americans
Flood regulation	Flood frequency	Number of vulnerable people (elderly, ESL) in areas with flood risk reduced by management action
Water quality regulation	Nitrogen concentration (proxy measure)	Swimmable days x number of people with ready access to the swim sites

[a] Donating to general conservation organizations is not a BRI because (1) there is no direct link between conservation donations and wolf populations—individuals may donate for reasons other than value for wolves—and (2) wolf existence is a public good—each individual can in principle obtain this benefit without paying for it—so individuals will free ride on payments made by others, and free riders will thus not be accounted for by only considering donations.

Best Practice Questions: Determining If Indicators Are Benefit Relevant

To follow best practices, the assessor should be able to answer *yes* to BOTH of these questions:

- Does the indicator reflect the changes in ecological condition in units that are relevant to the benefit and beneficiaries of interest?
- Does the indicator capture relevant physical and institutional access constraints on the flow of the service?

Recommended Reading

Boyd, J.W., and S. Banzhaf. 2007. "What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units." *Ecological Economics* 63: 616–626.

This paper describes intermediate and final ecosystem goods and services as concepts that align well with the bridging indicator concept and means-ends method described in this guidebook.

Boyd, J.W., and A. Krupnick. 2013. "Using Ecological Production Theory to Define and Select Environmental Commodities for Nonmarket Valuation." *Agricultural and Resource Economics Review* 42 (1): 218–249.

This paper defines ecological endpoints (what this guidebook defines as bridging indicators) as ecological commodities for nonmarket valuation. It provides many examples and a theoretical structure for determining an ecological endpoint.

Haines-Young, R., and M. Potschin. 2009. "Methodologies for Defining and Assessing Ecosystem Services." Report No. 14, Center for Environmental Management, University of Nottingham, Nottingham, UK.

This paper reviews approaches to classifying ecosystem services. It discusses the Millennium Ecosystem Assessment (MA) approach relative to hierarchical approaches like those used in this guidebook and concludes that the latter are probably required to produce results useful for decision makers.

Johnston, R.J., G. Macnusson, M.J. Mazzotta, and J.J. Opaluch, 2002. "Combining Economic and Ecological Indicators to Prioritize Salt Marsh Restoration Actions." *American Journal of Agricultural Economics* 84(5): 1362–1370.

Although primarily an economic valuation study, this article also demonstrates methods for developing and using bridging indicators of ecological condition to assess human values.

Ringold, P., J.W. Boyd, D. Landers, and M. Weber. 2013. "What Data Should We Collect? A Framework for Identifying Indicators of Ecosystem Contributions to Human Wellbeing." *Frontiers in Ecology and the Environment* 11: 98–105.

This paper provides a detailed example of identifying final ecosystem goods and services (which this guidebook calls *bridging indicators*) and metrics for assessing them. It uses a process based on identifying beneficiaries of the goods and services.

Schiller, A., C. Hunsaker, M.A. Kane, A.K. Wolfe, V.H. Dale, G.W. Suter II, C.S. Russell, G. Pion, M.H. Jensen, and V.C. Konar. 2001. "Communicating Ecological Indicators to Decision Makers and the Public." *Conservation Ecology* 5(1): 19. http://www.consecol.org/vol5/iss1/art19/.

This paper describes efforts to translate indicators of ecological condition into language appropriate for communicating with the public.

Wainger, L.A., and J.W. Boyd. 2009. "Valuing Ecosystem Services." In *Ecosystem-Based Management for the Oceans*, edited by K. McLeod and H. Leslie, 92–111. Washington D.C.: Island Press.

This paper discusses alternative measures of the value of ecosystem services—ecological endpoints that capture ecosystem goods and services—that this guidebook terms *bridging indicators*.

ANALYSIS

SELECTING SERVICES: SCOPE AND SCALE OF ANALYSIS

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This section is made up of slightly adapted excerpts from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making." [37]

Conceptual diagrams can help identify how a policy or management action can affect multiple aspects of an ecosystem and how each of the impacts on an ecosystem can have multiple impacts on social benefits. They can be useful for exploring all possible impacts to valued services. However, conceptual diagrams will likely identify too many services to be meaningfully quantified in an ecosystem services assessment. Thus, the quantitative assessment can be focused on those effects likely to be most important to the decision—often those expected to have the largest impacts on human welfare. Assessors can use a few key questions to determine which services should be included.

Does the ecosystem service fall under the legal mandates or authorities of the assessor?

Many laws, and the rules that agencies have developed to implement them, mandate an analysis of specific environmental attributes as well as social impacts, economic impacts, or both. These laws include the National Environmental Policy Act, the Clean Water Act, the Endangered Species Act, the Forest Land Policy and Management Act, and the American Indian Religious Freedom Act, among others. Any ecosystem services assessment conducted under a specific agency mandate will need to include changes to ecosystem services derived from the attributes and impacts specified in that mandate. Other regulations may also require assessments to consider services outside the assessor's direct jurisdiction. For example, the National Oceanic and Atmospheric Association Fisheries has responsibility for managing anadromous fish, meaning that changes in ecosystem services associated with a river in which fish spawn before migrating offshore must be a consideration in decision making. In addition, these decisions may require consideration of many related services called for by other mandates from the Environmental Protection Agency, soil and water conservation districts, and water treatment facilities.

Agencies and other decision makers might not want to invest in analyzing changes to ecosystem services that are outside of their authorities. Yet, broad analysis (or at least a recognition of all affected ecosystem services, whether or not they are subsequently analyzed) can improve understanding of the potential benefits of activities and may provide an opportunity for improved collaboration across agencies and with other affected entities.

Is an impact on the ecosystem service likely to be large *and* strongly driven by the proposed activity?

If decision makers aim to comprehensively assess ecosystem services and potential benefits, an effect on services should be included in the assessment if the policy, decision, or action is likely to have a large impact on it, whether or not the service is the intended target of the action or required by a mandate. For example, the U.S. Forest Service broadly recognizes the importance of the national forest system in providing drinking water

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[37] L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

to communities and habitat for many aquatic and riparian species. Thus, forest plans should not only focus on direct services, such as wood production, forest species maintenance, and forest jobs, but also on the relationship between forest restoration or timber harvest actions and downstream water uses. Only if the impact of an action on a benefit-relevant indicator (BRI) is insignificant can it be safely excluded from further analysis.

When determining whether an impact on an ecosystem service is likely to be significant, the time frame of possible impacts should be matched to the time frame of the action. For example, if the decision is about placing a dam that will exist for 100 years, the magnitude of impacts on that river should be considered over the 100-year time frame. The most appropriate time scale should cover the likely impacts during the project and for the period during which effects will remain substantial.

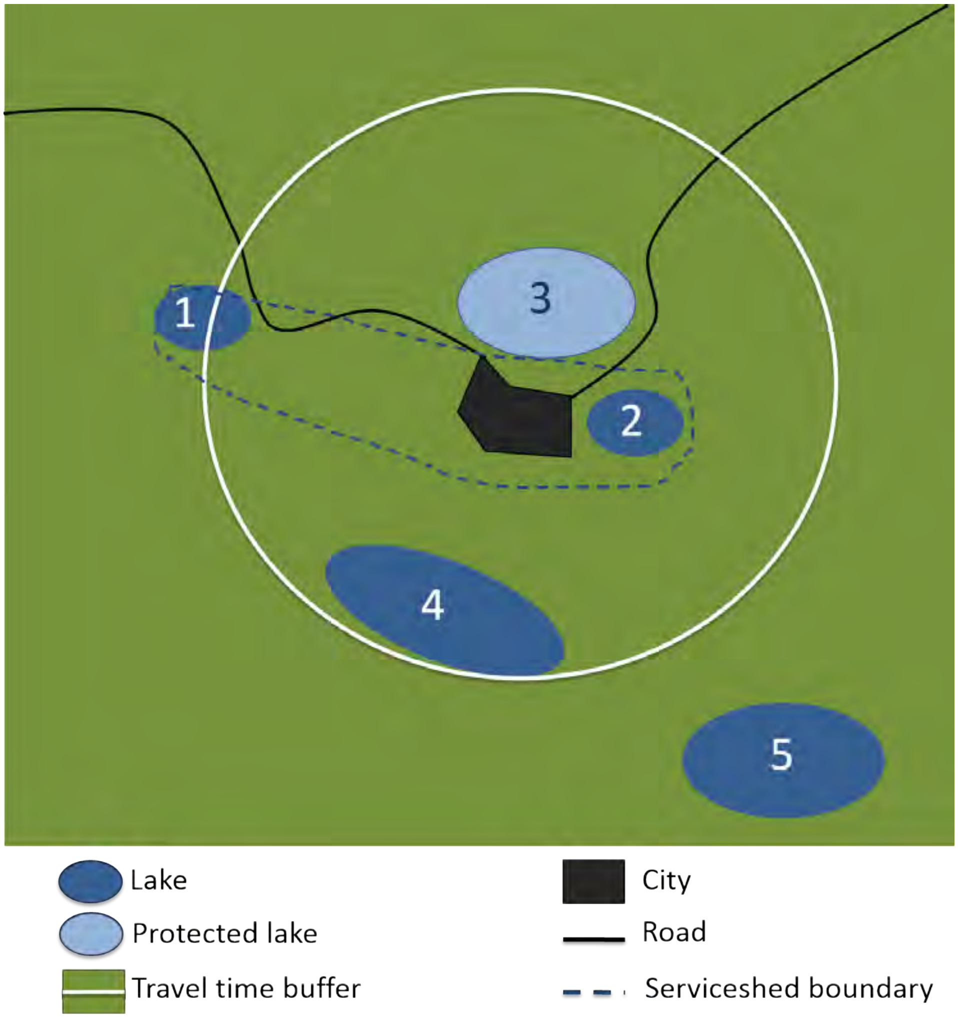
Will the expected changes to the ecosystem service matter to many people or to groups of special concern?

Answering this question means giving consideration to the "service areas" or "servicesheds" likely to be affected and to how many and which people will be affected by likely changes in a service. A serviceshed captures the area that provides a specific ecosystem service to a specific group of people (Figure 11). Serviceshed boundaries are defined by the area that supports the biophysical production of the service, by relevant access constraints (physical and institutional) to the service, and by demand for the service within that area.[38] For example, change in the water quality of a lake popular for recreation affects people who do or would potentially visit the lake, which may include people who live outside of the watershed of the lake. In some cases, there is biophysical supply of an ecosystem service but no realized benefit. Fish abundance for recreational fishing will generate no benefit in a water body where fishing is prohibited by law, or is otherwise inaccessible for recreation (Figure 11, lake 3). If, however, the existence of a place, habitat, or species is what people care about, its condition and continuance is what matters; physical and institutional constraints preventing access do not limit the benefits being realized. In addition, the serviceshed, including all those who value the particular service, can be national or even worldwide. Servicesheds for nonuse values in particular can often span very great distances.[39]

[38] H. Tallis and S. Polasky, "Mapping and Valuing Ecosystem Services as an Approach for Conservation and Natural-Resource Management," Annals of the New York Academy of Sciences 1162(2009):265–283, doi: 10.1111/j.1749-6632.2009.04152; J. Boyd and S. Banzhaf, "What Are Ecosystems?," Ecological Economics 63(2007):616–626.

[39] R.J. Johnston, D. Jarvis, K. Wallmo, and D. Lew, "Multi-Scale Spatial Pattern in Nonuse Willingness to Pay: Applications to Threatened and Endangered Marine Species," Land Economics 91 (4)(2015): 739–761.

Figure 11. Hypothetical serviceshed boundaries



Source: H. Tallis, C.M. Kennedy, M. Ruckelshaus, J. Goldstein, and J.M. Kiesecker, "Mitigation for One and All: An Integrated Framework for Mitigation of Development Impacts on Biodiversity and Ecosystem Services," Environmental Impact Assessment Review 55 (2015): 21–34.

Note: The serviceshed for recreational fisheries is determined by the accessible lakes (or rivers) with harvestable recreational fish species that are within an acceptable travel time of people. Lakes 4 and 5 are outside the example serviceshed because they lack physical access or are too far away, respectively. Lake 3 is within the potential serviceshed area but is protected and so lacks legal access.

A serviceshed captures the population that will be affected and can help decision makers consider where a change in provision of a service may have a large impact on vulnerable populations or other social groups of special concern. All services do not flow to all people equally, and some decision contexts present a requirement to consider those differences. For example, Native American groups have fishing and hunting rights on all federal lands, and a NEPA assessment on such lands should capture impacts to those groups distinctly. A general BRI for commercial fishing benefits would be abundance of fish landed commercially, whereas a group-specific BRI would be abundance of fish landed by Native American groups. When such interests exist, drawing an explicit causal chain for the group of interest can be a helpful way to understand key connections and identify a group-specific BRI.

By starting with the full conceptual diagram and then selecting the critical subset of services for further analysis, practitioners acknowledge the full suite of affected ecosystem services and can be more transparent about the services that are (and are not) subsequently analyzed and the rationale for these decisions. In the next step, individual causal chains for the selected services are expanded with additional details required for analysis.

Best Practice Questions: Selection of Ecosystem Services

To follow best practices, an assessor should include a service in an assessment if he or she answers *yes* to ANY of these questions:

- Does the ecosystem service fall under the legal mandate of the assessor?
- Is the impact on the ecosystem service likely to be large and strongly driven by the proposed activity?
- Will the expected changes to the ecosystem service matter to or affect the social welfare of many people or groups of special concern?

Agencies may need to collaborate with one another to include services outside their authorities.

SELECTING SERVICES AND CAUSAL CHAINS: BUILDING CAUSAL CHAINS

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This section is made up of adapted excerpts from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making." [40]

A causal chain—also known as a path model or means-end diagram—is a logical model that declares how current conditions, desired conditions, or a management action or policy is expected to propagate through the ecosystem to the provision of ecosystem services and benefits to various segments of society (Figure 12). An ecosystem services assessment requires a well-crafted causal chain whereby the indicators used to quantify the supply of services are defined as BRIs. The initial conceptual diagrams created in the scoping process consist of high-level (or preliminary) causal chains. Before analysis, each chain for a selected service must be elaborated and verified. Indicators are added or refined as needed to make the concepts more measurable in terms of societal benefits (e.g., hospital visits). Ultimately, the chains may be implemented as data-driven models that are used to estimate current provision of services or changes in services expected to result from management or policy actions.

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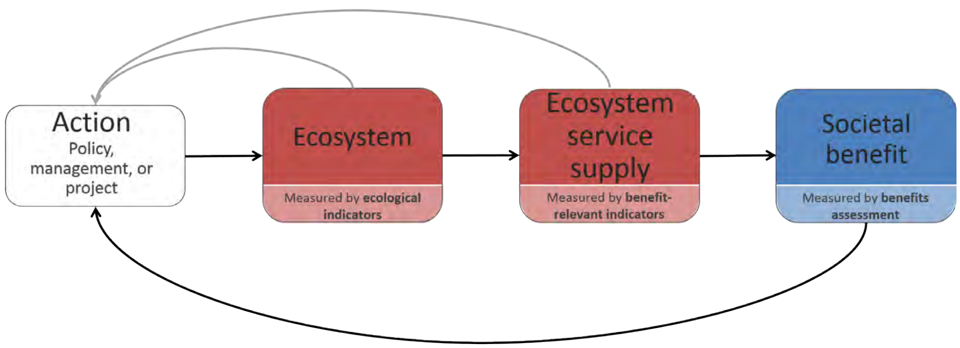
[40] L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

Answering the following sequential questions will help assessors build causal chains for decision making (Figure 12):

- How does a policy, management decision, or program action affect ecological conditions and related human behaviors?
- How do these changes affect the delivery of ecosystem services (defined as ecological changes that directly influence people)?
- How do those changes in the delivery of ecosystem services affect benefits or costs to individuals or groups?

Causal chains that connect current ecological conditions to current delivery of ecosystem services and societal benefits without any changes or alternatives introduced into the system would be represented by a three-box diagram without the initial action.

Figure 12. Components of an ecosystem service causal chain

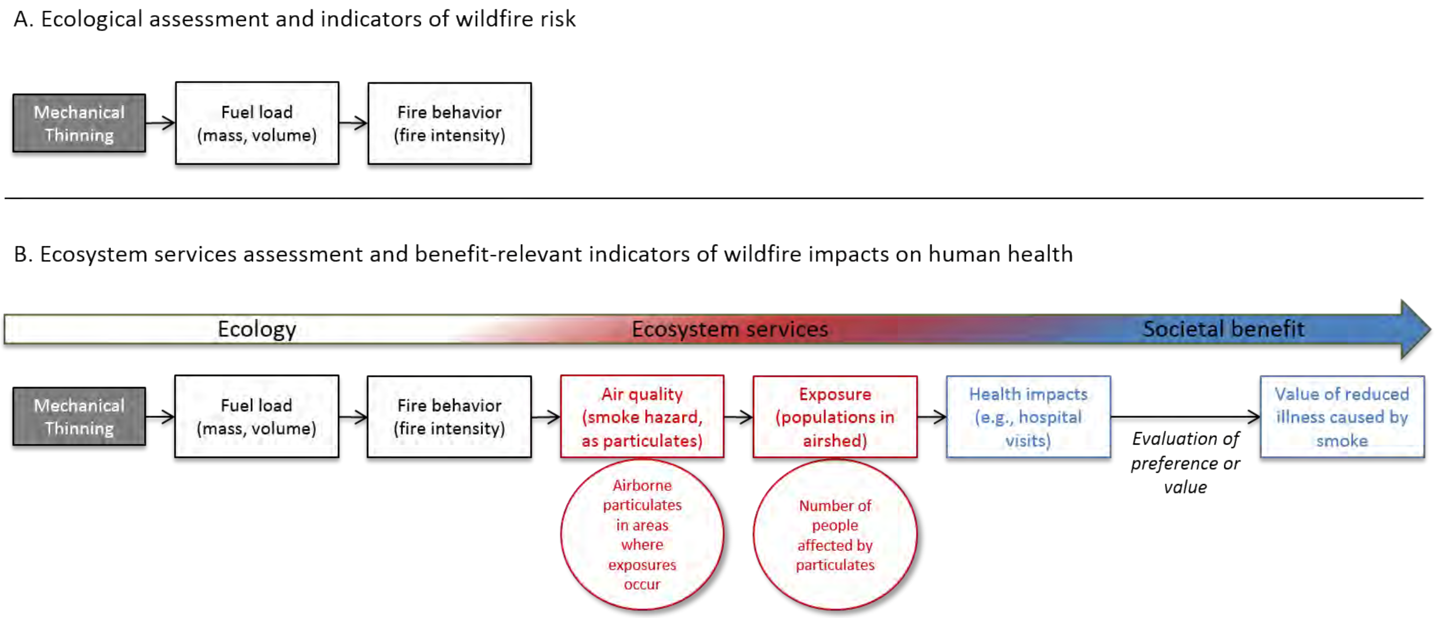


An ecosystem services assessment must consider how and which changes in the environment affect benefits to people. When causal connections to people are not made explicit, it is unclear whether and how each ecological change is related to changes in social benefits, and important changes to societal benefits may be left out of the analysis.

Figure 13 compares an ecological assessment with ecological indicators that are not explicitly linked to things people value, to an ecosystem services assessment using BRIs. In this example, resource managers are assessing mechanical thinning of forests to reduce the intensity of fire. An ecological assessment of this option might consider changes in the fuel load, which affects fire intensity (Figure 13a), along with a variety of other biophysical implications. An ecosystem services assessment, in contrast, would extend these causal chains to specific benefits to people that would result from mechanical thinning and the consequent management of fire risk (Figure 13b). There are many ways that people might be influenced by this action. For example, by reducing fire intensity the management action would reduce the incidence of smoke and the extent of poor air quality and exposure, reducing adverse health impacts from fire for nearby residents (e.g., as hospital visits, missed work days, or actual health care costs).[41] These considerations extend the ecological assessment to an ecosystem services assessment by including the interaction of people with the ecology (Figure 13b). Best practice for ecosystem services assessment will focus on estimation of changes in ecosystem service values or preferences (blue text in Figure 13b), but when time or resources are limited, the minimum standard for assessment is to focus on BRIs (red text in Figure 13b).

[41] For example, a study from Canada shows the costs of fire on human health: University of Alberta, "Forest Fires a Huge Cost to Health," Science Daily, August 10, 2006, http://www.sciencedaily.com/releases/2006/08/060810211036.htm.

Figure 13. Differences between ecological and ecosystem services assessments and indicators



Note: Causal chains consider expected outcomes from forest fire management activities like mechanical thinning. Black text indicates an ecological assessment and indicators, red text indicates extension to an ecosystem services assessment, indicators within ovals illustrate BRIs, and blue text indicates measures of social benefit and value. The demarcation among ecology, ecosystem services, and social benefits is not absolute (the lines between categories are drawn differently by different people), as represented by the tricolored arrow.

Adding these details (and subsequent quantification) requires expertise. Practitioners should engage experts from all relevant fields to ensure that the maps are as complete and accurate as possible. This group may include (but is not limited to) physical and biological scientists (hydrologists, wildlife biologists, botanists, ecologists, fisheries managers, ecological modeling experts, and foresters), social scientists, and economists.

Spatial context also plays an important role in many natural resource decision processes. Since the delivery of ecosystem services is not uniform across the landscape, the use of spatial information can be particularly important to ecosystem services assessments in addressing, for example, whether to conserve an upland rather than a lowland site or which management practices work best for reducing fire risk in an upland versus a lowland site.

One reason that causal chains are useful is that they can help to prevent double counting. Double counting occurs when an estimate of a value for an output (something further to the right on a causal chain) is added to the estimated value of an input along the same chain (something further to the left on the causal chain). A comprehensive value estimate for any element on a causal chain will capture some of the values associated with all of the elements to the right of it on the same causal chain—even if one ecological outcome leads to multiple social benefits. In principle, a causal chain diagram helps identify the logical endpoints of how the system responds to management, with each endpoint (on the right side of the diagram in Figure 13b) representing a single service or benefit that is meaningful to an identified beneficiary or stakeholder population. If a single indicator is selected to capture each meaningful endpoint to each affected beneficiary group, double counting is unlikely to occur. Hence, a properly constructed causal chain can be used to minimize double counting in an ecosystem services assessment because it clearly illustrates these input-output relationships.[42] However, no causal chain can eliminate all possibility of double counting. Hence, if the analysis is to proceed to values assessment, it is important to involve experts in monetary or nonmonetary valuation to ensure that double counting is eliminated or minimized and that all major sources of value are considered (i.e., to avoid under counting as well as double counting).

[42] More details on how to avoid double counting through use of causal chains and objectives hierarchies can be found in L. Maguire, Multi-criteria Evaluation for Ecosystem Services: A Brief Primer, 2014, https://sites.nicholasinstitute.duke.edu/ecosystemserFseevices/research/publications/.

Once causal chains are constructed, changes in the benefit-relevant indicators can be quantified, with or without an assessment of value, and used in a decision process.

QUANTIFYING BENEFIT-RELEVANT INDICATORS

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This section includes excerpts adapted from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making." [43]

How are Benefit-Relevant Indicators Quantified?

A large body of literature explains how to quantify changes in ecological conditions.[44] But these analyses alone are not sufficient for an ecosystem services assessment because they typically focus on ecosystem processes or features (e.g., net primary productivity) rather than on benefit-relevant endpoints. In fact, there is far more literature on ecological assessments than on changes in ecosystem services. Once agencies and other organizations start using ecosystem services assessment, the literature on such assessments should expand and mature.

Measuring Changes in Ecosystem Services

When assessing or monitoring the ecosystem service outcomes of an action, perhaps as a performance metric, a direct measure of a benefit-relevant indicator (BRI) can be used. In contrast, predicting changes in the provision of services resulting from management or policy actions (a necessary step for preference evaluation) involves converting the conceptual model depicted as a causal chain into an operational empirical model. There are several ways to measure the relationship between an action (policy, project, management) and its effect on the production of services. These methods differ in the time, resources, and capacity required. A narrative description of changes in ecosystem services could take the least time and resources, but it would not meet the minimum best practice requirement proposed for ecosystem services assessment because it is neither repeatable nor comparable, nor is it readily used in valuation or decision analysis methods.

Informal and formal methods of expert elicitation (e.g., Bayesian belief networks) can be used to generate quantifiable causal chains, including estimates of uncertainty. Alternatively empirical methods can be used. The empirical method that is likely to take the least time and resources and that meets the proposed best practice is to use existing models or to derive new models that use available data (collected by the agency or by others) or well-established relationships from the literature. For example, in the wetland restoration example (Figure 15), a study of fish mortality and reproduction that collected data on the effects of wetland restoration in

[43] L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

[44] See, for example, a recent assessment of progress on Mississippi water quality and hypoxia by the Mississippi River Gulf of Mexico Watershed Task Force, Reassessment 2013: Assessing Progress Made since 2008, 2013, http://water.epa.gov/type/watersheds/named/msbasin/upload/hypoxia_reassessment_508.pdf, a national wetlands assessment procedure by the U.S. Environmental Protection Agency, National Wetland Condition Assessment Fact Sheet, n.d., http://water.epa.gov/type/wetlands/assessment/survey/upload/Wetland-Survey-Fact-Sheetv7.pdf, and a number of USDA examples and guides at http://www.fs.fed.us/emc/rig/ecosystem_assessment.shtml. See also S.A. Bortone, ed., Estuarine Indicators (Boca Raton, FL: CRC Press, 2005).

a similar region could be used to estimate the proposed project's effect on services. Likewise, the health effects of smoke from fires might be estimated using a concatenation of several models (fire intensity from a fire behavior model, smoke production from fire intensity, a plume model for the airshed, and so on). Again, the overall uncertainty of the full model would reflect the concatenation of models (and error propagation) as well as the uncertainty arising because the models would likely not reflect conditions at the study site.

It will often be the case that an ecosystem services assessment will be based on models derived from secondary data because primary data collection is not always possible.[45] Clearly, these models could be improved if they were based on data collected in the study area. The gold standard for these assessments would be a model generated on-site or within the study region, based on manipulative experiments using the management actions being evaluated and explicitly measuring outcomes in terms of the BRI (and any intermediate variables needed to build the model). This approach works well with adaptive management, in which management treatments are implemented as experiments (with controls) and outcomes are monitored over time. In this case, the measured outcomes would empirically support the BRI, and the result would be a local model that explicitly translates the management action into its ecosystem services outcomes. Clearly, this approach is ambitious. But because adaptive management is a stated ambition of most, if not all, federal land management agencies, this aspiration is consistent with agency missions.

Moving Beyond Narrative Measures

Many federal decisions use descriptive or narrative information to describe changes in ecological conditions and ecosystem services resulting from possible actions (e.g., Environmental Assessments and Environmental Impact Statements for NEPA). Narrative information can provide context for creating well-defined measurement scales. But narrative information is difficult to evaluate and cannot be used in preference evaluation (e.g., economic valuation) or tradeoff analyses. Narrative information is also not easily reproducible or testable in the same ways as information expressed using a well-defined scale. Given these limitations, **descriptive narratives alone do not qualify as minimum best practice for ecosystem services assessments.**

In most cases, however, it is relatively easy to transform descriptive narrative data into well-defined categorical or quantitative data that can satisfy minimum best practice. For example, descriptive measures can be transformed into a binary measure of presence and absence, or a categorical measure, or a continuous quantitative measure. Quantitative and categorical measures of ecosystem services will make the services easier to evaluate intuitively and to incorporate into formal valuation or tradeoff analysis, making the services more likely to be fully considered in decisions.

For measurements to be effective, their scales must be defined clearly enough to be applied by different users and to different decision contexts with consistent results (e.g., they must be repeatable). Numerical measurement scales, whether continuous (e.g., board feet of merchantable timber available from a specified land parcel) or discrete (e.g., numbers of deer taken by recreational hunters during a specified period of time from a specified geographic region), are the most obvious scales, but some categorical measurement scales can also meet these standards.

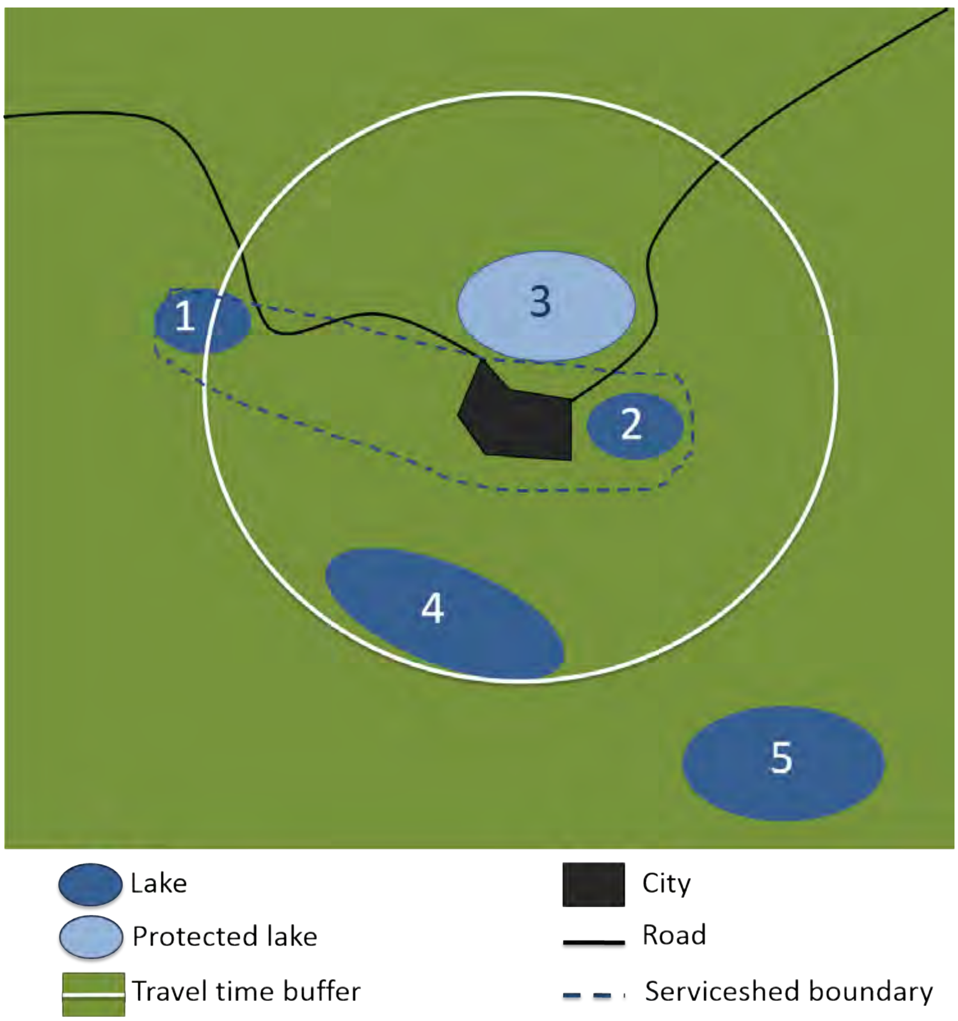
Categorical measurement scales can be used when numerical scales would be inappropriate or when estimation using numerical scales is too difficult or too expensive. An example is a scale describing degree of preservation of a tribal cultural site (e.g., "destroys a specified cultural site," "preserves the site but prevents access by tribal members," and "preserves the site and permits access on specified days"). In another example, categorical data may simply reflect presence or absence, such as the presence/absence of a particular listed species in a specific geographic area during a specific period of time, as determined by an agreed-on detection method. Other types of categories might reflect key thresholds or officially defined categories—for example, whether a population is considered endangered or threatened according to established guidelines. Thresholds between categories need to be defined clearly to provide reliable results. Scales such as "low," "medium," and "high" fail to meet this standard of clarity, unless such terms are clearly linked to well-defined thresholds. Categorical measures of BRIs must be defined using a scale that is unambiguous, measurable, and replicable to meet best practice guidelines.[46]

[45] Primary data are new data observed or collected directly at the site or region of interest for purposes of a particular study. Secondary data are data already collected for another purpose (e.g., a prior study in the literature), often but not always at other locations.

Identifying and Quantifying Who Is Affected

Identification and quantification of those people who could benefit from an ecosystem service—beneficiaries—involves defining the serviceshed and flows of services (Figure 14).[47] For a locally used service like municipal water supply, the serviceshed is easily drawn around those using water within the watershed downstream of the policy or project action. For a service used or appreciated by a broader or spatially distributed group of people, like recreational use or cultural appreciation of a particular location, the serviceshed would include the area providing the service and its connections to those using or appreciating the service even if they live scattered about the region. Decision makers need to know not only where these people are but who and how many they are and whether they are affected by potential changes in the provision of services (e.g., reduction in flood or fire frequency or intensity). In the absence of a primary study or other direct means to identify the distribution of affected individuals (e.g., a survey conducted using a random sample over the potentially relevant area), indirect methods may be used.

Figure 14. Hypothetical serviceshed boundaries



Source: Tallis, H., C.M. Kennedy, M. Ruckelshaus, J. Goldstein, J.M. Kiesecker. 2015. "Mitigation for One and All: An Integrated Framework for Mitigation of Development Impact of Biodiversity and Ecosystem Services." *Environmental Impact Assessment Review* 55: 21–34.

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[46] E.T. Schultz, R.J. Johnston, K. Segerson, and E.Y. Besedin, "Integrating Ecology and Economics for Restoration: Using Ecological Indicators in Valuation of Ecosystem Services," Restoration Ecology 20(3)(2012): 304–310.

[47] H. Tallis, S. Polasky, J.S. Lozano, and S. Wolny, "Inclusive Wealth Accounting for Regulating Ecosystem Services," in Inclusive Wealth Report 2012: Measuring Progress towards Sustainability (Cambridge: Cambridge University Press, 2012).

Note: The serviceshed for recreational fisheries is determine by the accessible lakes (or rivers) with harvestable recreational fish species that are within an acceptable travel time of people. Lakes 4 and 5 are outside the example serviceshed because they lack physical access or are too far away, respectively. Lake 3 is within the potential serviceshed area but is protected, so lacks legal access.

Although indirect methods are almost always less accurate than direct methods of identifying affected individuals, they can provide sufficient insight for many purposes, particularly when direct methods are infeasible. For example, data from the U.S. census or large-scale surveys like the National Survey on Recreation and the Environment, and perhaps information on what people purchase (e.g., fishing gear or bird identification guides), can help identify and quantify affected people.[48] Small reductions in nitrogen oxide and sulfur oxide pollution can have significant health benefits over large areas, which can be characterized from air movement patterns. Direct engagement and outreach with communities and community groups, along with social media and surveys, can also help identify and determine the size of affected communities. A considerable economic literature is devoted to determining the "extent of the market" for ecological benefits (or where benefits occur); this literature details approaches that can be used for various types of applications.[49]

Best Practice Questions: Measuring BRIs

To follow best practice, the assessor should be able to answer *yes* to ALL of these questions:

- Does the method for estimating the change in BRI capture the causal path from action to change in ecosystem service?
- Is the BRI in an appropriate (well-defined and repeatable) measurement scale and stakeholder-relevant unit?
- Does the method capture relevant changes in demand and access (e.g., intensity of use, number of people affected, and access)?

Choosing the Best BRI

Benefit-relevant indicators **(BRIs) need to meet two criteria—(1) reflect changes that are relevant to beneficiaries and (2) capture some aspect of intensity of use and physical and institutional access where relevant.** However, some BRIs are better at reflecting the most relevant information about an ecosystem service than others. **The best BRIs will indicate a highly certain link between the environment and a human benefit and will also indicate the intensity of human use or enjoyment.**

Where Does the BRI Fall in the Causal Chain?

BRIs that capture biophysical outcomes as close as possible to human use, enjoyment, or appreciation are preferred. As causal relationships are established between management or policy actions and various ecological outcomes, indicators along the chain of ecosystem services production can be distinguished on the basis of their distance or proximity to social outcomes.

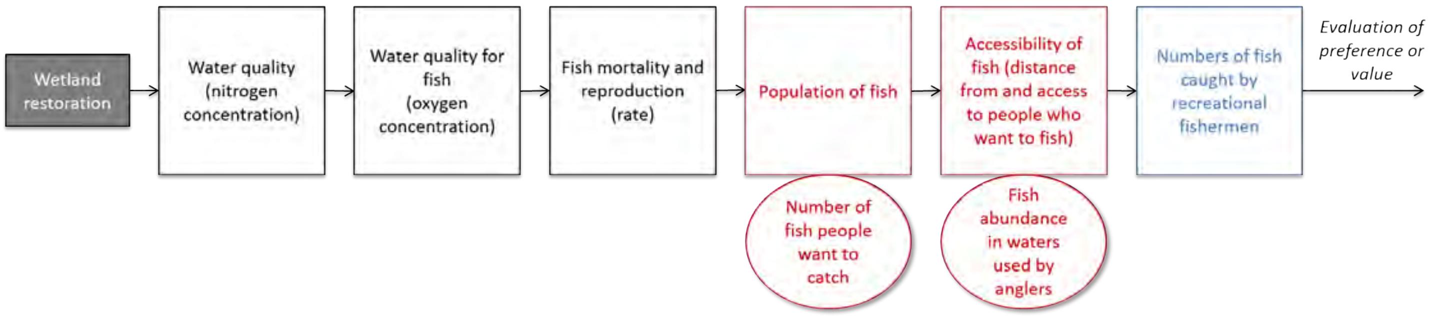
Consider the following causal chain arising from restoration of a wetland (the policy action) (Figure 15): (1) wetland restoration affects nitrogen levels in surrounding waters, (2) those nitrogen levels affect the water's oxygen levels through algal blooms, (3) oxygen levels affect fish mortality and reproduction, and (4) fish mortality and reproduction affect fish abundance in waters used by anglers. Measuring fish abundance in waters used by anglers is a BRI. Measuring wetland restoration is not, unless a tight relationship has already

[48] National Survey on Recreation and the Environment, http://www.srs.fs.usda.gov/trends/nsre-directory/.

[49] I.J. Bateman, B.H. Day, S. Georgiou, and I. Lake, "The Aggregation of Environmental Benefit Values: Welfare Measures, Distance Decay and Total WTP," Ecological Economics 60 (2006): 450–460; J. Loomis, "Vertically Summing Public Good Demand Curves: An Empirical Comparison of Economic versus Political Jurisdictions," Land Economics 76 (2000): 312–321; J.B. Loomis, "How Large Is the Extent of the Market for Public Goods: Evidence from a Nation Wide Contingent Valuation Survey," Applied Economics 28 (7)(1996): 779–782.

been firmly established between wetland restoration and fish abundance. BRIs that capture intermediate outcomes “earlier” in the causal chain are less desirable than BRIs that capture more final outcomes “later” in the causal chain because the earlier BRIs increase the number of links to be established to firmly anchor the measure to benefits. All else equal, therefore, it is preferable to develop indicators that capture “final” biophysical outcomes rather than “intermediate” outcomes.

Figure 15. Use of BRIs to assess the fishing benefits derived from wetland restoration



Note: Black text indicates an ecological assessment and indicators; red text indicates extension to an ecosystem services assessment and indicators, with ovals illustrating BRIs; and blue text indicates measures of social benefit and value.

The notion of final ecosystem goods and services (FEGS) is a concept used by a number of agencies. FEGS emphasizes the distinction between “final” and “intermediate” ecological goods and services.[50] Final ecosystem goods and services are “components of nature, directly enjoyed, consumed, or used to yield human well-being.”[51] Intermediate ecosystem goods and services are ecological processes, functions, structures, characteristics, and interactions that are essential to the existence of final ecosystem goods and services but are not directly enjoyed, used, or consumed by beneficiaries.[52]

In some cases, the links between intermediate ecosystem goods and services to final ecosystem goods and services are well established, and a measure of an intermediate ecosystem good or service can be used as a BRI. Carbon sequestration provides a good example. Carbon sequestration is not a final good or service because it is not directly linked to benefits. Rather, carbon sequestration is an input into climate regulation, which is linked to the severity of future climate change and its associated impacts. A large research effort has gone into establishing the causal links between atmospheric CO_2 concentrations and climate change and between climate change and potential future damages (from sea level rise, changes in precipitation patterns, and so on) to derive estimates of the social cost of carbon.[53] Insistence on measuring only final ecosystem goods and services would not allow measurement of carbon sequestration and the social cost of carbon as an approach. However, it is only because the work has been done to link carbon sequestration to benefits through the social cost of carbon that carbon sequestration is an acceptable BRI.

Summarizing the above, BRIs are related to but not the same as FEGS. One of the key features characterizing BRIs is that BRIs are clearly and measurably relevant to human welfare. Hence, all FEGS can and should be measured using BRIs. However, some things that are not FEGS may qualify as BRIs, if causal chains are sufficiently well developed to link those things clearly and measurably to welfare. An example is an intermediate service for which links to FEGS are well established. Hence, from a conceptual perspective, all FEGS are BRIs, but not all BRIs are FEGS.

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50 Boyd, J. 2007. “The Nonmarket Benefits of Nature: What Should Be Counted in Green GDP?” Ecological Economics 61(4): 716–723; J. Boyd and S. Banzhaf, “What Are Ecosystem Services?” Ecological Economics 63(2007): 616–626; R.J. Johnston and M. Russell. 2011. “An Operational Structure for Clarity in Ecosystem Service Values,” Ecological Economics 70:2243–2249; P. Ringold, J. Boyd, D. Landers, and M. Weber, “What Data Should We Collect? A Framework for Identifying Indicators of Ecosystem Contributions to Human Wellbeing,” Frontiers in Ecology and the Environment 11(2)(2013):98–105.

51 J. Boyd and S. Banzhaf, “What Are Ecosystem Services?” Ecological Economics 63 (2007): 616–626.

52 http://unstats.un.org/unsd/envaccounting/seeaRev/meeting2013/EG13-BG-3.pdf.

53 Interagency Working Group on Social Cost of Carbon, United States Government, Technical Update of the Social Cost of Carbon for Regulatory Impact Analysis Under Executive Order 12866, 2013, https://www.whitehouse.gov/sites/default/files/omb/assets/inforeg/technical-update-social-cost-of-carbon-for-regulator-impact-analysis.pdf.

How much uncertainty is there in the measure of the BRI?

Measurement of BRIs often involves considerable uncertainty. The complexity (length) of the causal chain amplifies uncertainty because information loss occurs at each link of the chain. For example, we will be more confident about the impact of restoration on nitrogen concentrations than we will be about oxygen content and about fish population demography, and still less confident about numbers of catchable fish. Similarly, we might expect some uncertainty about human health impacts of smoke from fires because of the propagation of model uncertainties about fire behavior, smoke production, plume dispersion in the airshed, and human response to smoke exposure. It is worth underscoring that one advantage of using causal chain diagrams is that they facilitate communication of these uncertainties.

Another source of uncertainty in BRIs arises from the difficulty of measuring impacts in relevant terms. For example, an ideal measure of “catchable fish” would be provided by an empirical stock assessment. In the absence of such information, for commercial fishery we might index “catchable fish” directly from commercial landings. This measure is confounded by fishing effort, and is hence not an ideal indicator, although related measures (e.g., catch per unit effort) can at least partially address this problem. In other (noncommercial) instances, we might have to be satisfied with estimates of fishing success derived from fishing permits, visitor days, or some other measure imperfectly related to actual numbers of fish caught. Use of proxies should be accompanied by an estimate of confidence in the accuracy of the proxy estimate, and recognition that some proxies are superior to others. When choosing the most suitable BRIs for any particular application, analysts may need to balance the direct proximity or relevance of the measure to benefits with the ability to obtain accurate information.

Does the BRI reflect the intensity of use or enjoyment by people?

Additional information about the importance of a service is added when information about the intensity of use or enjoyment of a service is available. For example, knowing whether the affected waters in the wetland example are the most popular fishing areas in the state given their accessibility (averaging 100 people per day during the season) or are highly prized for their beauty but somewhat isolated and used by fewer people (10 people per day during the season) would be helpful. Data on fish mortality and reproduction can be a sufficient BRI, but number of fish caught would provide some information about the intensity of fishing, making that measure a better BRI. This aspect of a BRI is nicely illustrated by the example of the health impacts of smoke from fire, by developing a BRI explicitly in terms of exposure (how many and which people?) and hazard (how bad is the air?).

Even “better” BRIs leave out information about people’s preferences. For example, people may value a service or good more if it is scarce, if it has no substitutes (other ways to gain goods or services that provide similar benefits), or if it does not require many other inputs (or complements) to produce benefit. Although BRIs often lack measures of these components of benefit, they represent a significant advance beyond ecological assessment alone. When feasible, benefit-relevant factors and benefits assessments can be used to capture these additional components of benefit. Indeed, BRIs might be viewed as nearly ideal inputs into more formal valuation methods because they are already in appropriate units and the relevant stakeholder populations are identified in at least a preliminary way.

QUANTIFYING SOCIAL AND ECONOMIC CONTEXT IN BENEFIT-RELEVANT INDICATORS

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Summary

This section describes conditions, like number and characteristics of users and reliability of future streams of services, that can provide context for determining the values or preferences held for different ecosystem services benefits. This context can inform development of meaningful benefit-relevant indicators (BRIs) and can therefore also inform an assessment of benefits (monetary and non-monetary valuation).

Takeaways:

- The conditions that tend to impart value to goods and services provide social and economic context for ecological changes helping to generate meaningful BRIs.
- The conditions that influence value or preference can be broken down into five categories: (1) quality of the service for its intended use, (2) availability of capital and labor that complement the ecological outputs in order to create goods and services, (3) number and characteristics of users or beneficiaries, (4) reliability of the future stream of services, and (5) scarcity and substitutability.
- Scarcity is the overarching concept that imparts value to an ecosystem good or service. In general, the scarcer a service is, the more an increase in its quantity is likely to be valued, all else equal.

To fully appreciate what ecological changes mean to people, those changes must be contextualized to reflect their social relevance using benefits assessment methods or other means to represent demand for or value of those changes. Although people may understand the relevance of a directly measurable biophysical quality (e.g., water clarity improvements) to their well-being, they may not be able to say whether they would be willing to give up some of that water clarity for a competing outcome (e.g., reduced risk of high-intensity fires), without having more context about the benefits and tradeoffs among services.[54] The context that determines willingness to give up some goods or services in order to get others requires that managers capture conditions such as the type and degree of the effect, the numbers and distribution of people affected, and people's preferences, concerns, and vulnerabilities.

[54] Tradeoffs occur when an increase in the provision of one service (1) comes at the expense of a decrease in the provision of other services or (2) changes the timing or spatial pattern of the delivery of other services, resulting in winners and losers.

Five Types of Conditions That Influence Values or Preferences

The conditions that tend to contribute to the value of an ecosystem good or service fall into several categories: (1) quality of the service for its intended use, (2) availability of capital and labor that complement the ecological outputs in order to create goods and services, (3) number and characteristics of users or beneficiaries, (4) reliability of the future stream of services, and (5) scarcity and substitutability.[55] Some of these conditions determine whether a good or service can be derived from the ecological feature or output that is represented by the ecological indicator, whereas others suggest the relative magnitude of benefits.

Service Quality

In the case of a service quality, managers might determine whether provision of a service is possible by asking, for example, "Does the water quality of the lake make it safe (or desirable) for swimming?" If the answer is yes, then the ecological feature of a lake with a certain water quality can provide a swimming opportunity. When assessing whether a change in water quality will create a benefit, managers might ask whether the change in water quality changes the type of use that is possible or markedly superior (e.g., does the lake become swimmable or much better for swimming?) or, whether the risk of contracting illness while swimming is reduced (e.g., do cases of skin rashes in swimmers decline?).

Other qualities might be useful for comparing sites to assess relative magnitude of benefit that would be provided due to an action. If a water body dries up during part of the year, it will not provide the same level of service for irrigation as a water body that maintains water availability year round. As a result, an investment that extends the duration of available water must be judged in terms of whether that change is sufficient to make the site more useful for irrigation or whether it remains insufficient. Thus, the *qualities* of the ecological structures and processes must be compared to the user needs or preferences to confirm that the outputs are beneficial and to compare benefits among sites.

Capital and Labor

Similarly, the availability of complementary labor and capital might determine whether or not a service can be provided at a site before and after a change. For example, pollinators from natural areas are only able to enhance crop yields if human labor has provided the crops that require pollination in range of the pollinators. In other cases, the availability of complements may not only enable use but also suggest higher use rates at a site. The availability of fishing piers and boat ramps, for example, might increase the use of a lake for recreational fishing. Greater use tends to imply greater benefits from ecological changes because, all else equal, the more site users, the more valuable the aggregate benefits provided by the site.

Number and Characteristics of Users

Population characteristics (size, demographics, and income levels) are also known to influence how much people value certain types of ecological conditions. For example, members of a tribal community value an intact natural area because it is integral to their cultural and spiritual practices. The use of a site by a tribal community imparts a value to the site that can be captured, to some extent, by documenting the number and demographics (age, gender, and so on) of the site users. More generally, surveys and interviews are used to relate people's values to their socio-demographic profile in order to project the number of site users. For example, national surveys are used to evaluate how participation rates in hunting, fishing, and wildlife watching vary by demographic characteristics and how demographics of nearby populations may translate into likely levels of site use.[56]

In addition to demographic characteristics, location or physical characteristics of a household or business can

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[55] These categories are based on Wainger et al. (2001, 2010) and were refined by the SESYNC working group in April 2014 (L. Wainger, D. King, J. Salzman, and J.W. Boyd, "Wetland Value Indicators for Scoring Mitigation Trades," Stanford Environmental Law Journal 20 (2001): 413–477; L.A. Wainger, D.M. King, R.N. Mack, E.W. Price, and T. Maslin, "Can the Concept of Ecosystem Services Be Practically Applied to Improve Natural Resource Management Decisions?" Ecological Economics 69 (2010): 978–987).

determine the number of likely users of an ecological feature, and a greater number of users can suggest greater value. For example, an aquifer's value as a source of drinking water increases with the number of households that have access to it, all else equal. Similarly, a park with 10,000 visitors a year is more recreationally valuable than a park with 100 visitors a year, all else equal. The aggregate social value of a change can be more sensitive to the size of the beneficiary pool than the magnitude of change to an individual.[57] Thus, the number of beneficiaries will be an important benefit-relevant metric.

Reliability

The benefits derived from ecosystems typically arrive as a stream of goods and services through time, and their value is the sum of the expected future stream of benefits. Market behavior suggests that people are often willing to pay more for reliable goods and services (i.e., low performance risk) than for unreliable ones, all else equal. Therefore, anything that affects system reliability, including both controllable and uncontrollable factors, can affect value. Controllable factors might include the types of human activities that are allowed on site and whether the site or surrounding land has been protected from conversion to incompatible uses through purchase of land or easements. Uncontrollable factors might be related to human activities throughout the watershed that affect the site (e.g., hydrologic modification) or outcomes of climate change.

Scarcity and Substitutability

Perhaps the most critical component of the socially relevant indicators list is the issue of scarcity and the related issue of substitutability. The idea that the scarcer an ecological feature or process is, the more valuable it tends to be, is well supported by economic theory and market evidence. However, people can adapt to scarcity by modifying their behavior or finding substitutes. If irrigation water becomes scarce, people can switch to more efficient technologies or non-irrigated crops, or they can stop farming and develop alternative businesses. Together, these actions might eliminate the former evidence of scarcity of irrigation water, if the system achieves a new equilibrium of supply and demand. However, the costs of adaptation or substitution would be considered losses during the time it takes to reach the new equilibrium. In addition, the new equilibrium could create continuing losses to producers, relative to the prior condition, if, for example, the producers were forced to grow a smaller quantity of a crop or to grow a lower-value crop on the same land and costs did not decrease proportionally (i.e., substitutes were imperfect).

Not all of this potential adaptation can be captured, but something that can be measured is whether substitutes are available, either in the immediate surroundings or at a scale that is meaningful for a given service.[58] Substitutes may be alternative locations that provide the same service (e.g., alternative fishing sites) or technical substitutes (e.g., levees for wetlands). Thus, scarcity can be evaluated by considering the total amount of a good or service (at the relevant scale), the number of substitute sites, and whether technical substitutes are either already in place or feasible. All else equal, the availability of abundant substitutes suggests that relatively few people will care about a change that affects only a small portion of the available options.

People's willingness to modify their behavior (rather than pay to restore an ecological condition) is one reason that measures of ecological change, even when modified to reflect the benefits they could provide, may not always reflect a significant willingness to pay (WTP) for a change. A robust valuation study includes information about whether people would be willing to pay for a change, rather than to adapt in some other way.

On the other hand, using BRIs that incorporate information on conditions that influence value can have some advantages over monetary values used alone. For one, they can be more sensitive to social equity concerns

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[56] J.M. Bowker, D. Murphy, H.K. Cordell, D.B.K. English, J.C. Bergstrom, C.M. Starbuck, C.J. Betz, and G.T. Green, "Wilderness and Primitive Area Recreation Participation and Consumption: An Examination of Demographic and Spatial Factors," Journal of Agricultural and Applied Economics 38 (2006):317–326; Mazzotta, M. J., L. A. Wainger, S. D. Sifleet, J. T. Petty, and B. Rashleigh, "Benefit Transfer with Limited Data: An Application to Recreational Fishing Losses from Surface Mining," Ecological Economics 119C:384–398.

[57] The beneficiary pool is the group of individuals that benefit from or are harmed by a gain or loss of a service.

[58] The issue of how people would alter their behavior is difficult to address because people innovate in numerous ways when goods and services become scarce.

because they are not dependent on people's ability to pay. For example, the number of households displaced by the loss of a service is a more egalitarian metric than the value of homes at risk. In a related example, if managers consider the proportion of at-risk households that are occupied by socio-economically vulnerable groups, they can demonstrate that changes that protect such groups have the benefit of protecting people with little ability to adapt (by modifying their homes) or substitute (by moving).

The population characteristics that suggest limited ability to substitute or otherwise adapt to loss of some types of services will depend on the service. For example, the presence of subsistence fishers at a site might suggest that the benefits of preventing increased toxics levels in fish will tend to be greater than in areas that the fishers do not use. This potential for greater benefits from avoiding toxic contamination is due to the likelihood that the subsistence fishers will continue fishing, even if toxin levels make the fish unsafe to eat. In comparison, economically secure populations are likely to already be substituting other food sources, if fish contain levels of toxins. Therefore, they will not benefit as much as subsistence fishers from preventing additional toxic contamination. Thus, establishing the presence and estimating the size of socially or economically vulnerable populations (e.g., subsistence fishers, households in poverty) or considering economic dependency (e.g., number and size of local businesses that depend on ecological conditions) can reveal when vulnerable groups may incur greater benefits from a change than non-vulnerable groups.

Services with Potential Non-Use Values

The types of factors that influence value will vary by the type of service. All of the categories described above are applicable to goods and services that are directly or indirectly used, but not all are appropriate for evaluating *non-use values*, which are values people hold for goods and services that they will not directly use. For example, a person may be willing to pay to preserve wild pandas even though she will never see or otherwise use them. When services are identified as *potentially* reflecting non-use values, managers recognize that some parties *may* hold economic value (willingness to pay) for their existence, regardless of use. However, not all such services actually possess a value that can be traced empirically and thus their non-use value potential may go unrecorded. Thus, conveying the importance of a biophysical change for non-use value requires characterizing it in a way that communicates the feature that people value, as described further below.

Many surveys have revealed that people value the integrity and long-term persistence of ecosystems and species populations.[59] Thus, one way to enhance an indicator that reflects additional acres of a rare ecosystem is to add information to the indicator that conveys capacity to support the ecosystem's integrity or persistence. For example, the metric might quantify that a restoration project will increase the total area of a priority ecosystem by 25%. Alternatively, a metric might suggest that an action increases long-term population viability as a result of enhancing and protecting the only remaining connection between two parts of an ecosystem that have otherwise become isolated by land conversion. Both metrics put the affected acres of ecosystem into a larger context of what remains and what might be necessary to promote the ecosystem's long-term existence.

For a species, an example of adding information regarding persistence to an indicator would be to convert the number of the species' breeding pairs to a change in the probability of the species population viability.[60] As in the example of the ecosystem, this change links the ecological indicator to the specific human desire that a species be allowed to persist and be available for future generations. Even if a species or ecosystem does not have its own constituency (i.e., is non-charismatic), it may still be deemed important through an analysis of the scarcity of the genetic or functional information represented by the ecosystem. This information may be made relevant to non-experts by identifying its significance, if any, for agricultural and pharmaceutical product development, for example. Although relevance indicators are currently difficult to assess, they are identified here to promote research and to provide examples of useful metrics in order to guide thinking about the types of available data that might enhance existing metrics.

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[59] See, e.g., L. Richardson and J. Loomis, "The Total Economic Value of Threatened, Endangered and Rare Species: An Updated Meta-analysis," Ecological Economics 68 (2009):1535–1548; K. Wallmo and D.K. Lew, "Public Willingness to Pay for Recovering and Downlisting Threatened and Endangered Marine Species," Conservation Biology 26 (2012):830–839.

[60] D.F. Staples, M.L. Taper, and B.B. Shepard, "Risk-Based Viable Population Monitoring," Conservation Biology 19 (2005):1908–1916.

Relevance evaluations are often the only enhancement that can be made to indicators used to suggest non-use values. Some might even argue that the enhancements discussed here are just flavors of ecological indicators. Such a distinction is not critical; what is critical is recognizing that, when managers are using ecological indicators to reflect non-use values, those indicators should reflect the aspects of the species or ecosystem that are of interest to people. This interest is typically translated into metrics capturing how the proposed action changes the scarcity of the ecological asset.

In these examples, managers rely on the well-established economic concept that goods and services that are scarce will be more valued than those that are abundant. However, this approach does not suggest that common species have no value because people also enjoy keeping abundant species (e.g., songbirds) abundant. However, the scarcer something is, the higher its value, all else equal.

Methods for Quantifying Social and Economic Context in BRIs

Ecological indicators can be more closely tied to what people value and prefer by incorporating information on the conditions described above. The methods for adding such information range from simple to complex. Consider the options available for turning biophysical metrics of wetland area and plant density into a metric that more clearly demonstrates the benefits of a wetland restoration project (Figure 16):

- *Option 1: Narrative alone (not a quantitative BRI)*—The restored wetlands are reducing risk of damage to a heavily used bridge.
- *Option 2: Simple geographic information system or GIS calculations (a weak BRI)*—The wetlands reduce risk of damage to the bridge by reducing maximum fetch (distance of open water in the direction of wind) from a maximum of 100 kilometers to 10 kilometers, and the bridge is used by 4,000 drivers per day.
- *Option 3: Complex calculations (a strong BRI)*—The wetlands are expected to prevent the closure of the bridge for an expected duration of 1 week every 10 years and, for each week of closure, they will prevent 25,000 additional hours of commuting time necessitated by use of alternate routes.

This list represents the options that analysts have to (1) simply identify the potential services on the basis of location, (2) create one or more distinct indicators that quantify site and location characteristics that suggest potential magnitude of benefits, or (3) combine the benefit indicators with models to more precisely quantify the potential magnitude of benefits (Figure 16). In both simple and complex calculations, the analysts conduct two types of modeling. First, they evaluate the effectiveness of the wetlands for preventing damage to the bridge (reflected in the fetch calculation in option 2 or the estimated days of closure in option 3).[61] Second, they evaluate how many people were affected. In option 2, they simply quantify the number of users. In option 3, they evaluate the number of users and the availability of one type of substitute by calculating the additional hours required to take alternative routes. By considering the ease of taking alternate routes, they more precisely show the level of harm, by considering the capacity of drivers to substitute one route for another.

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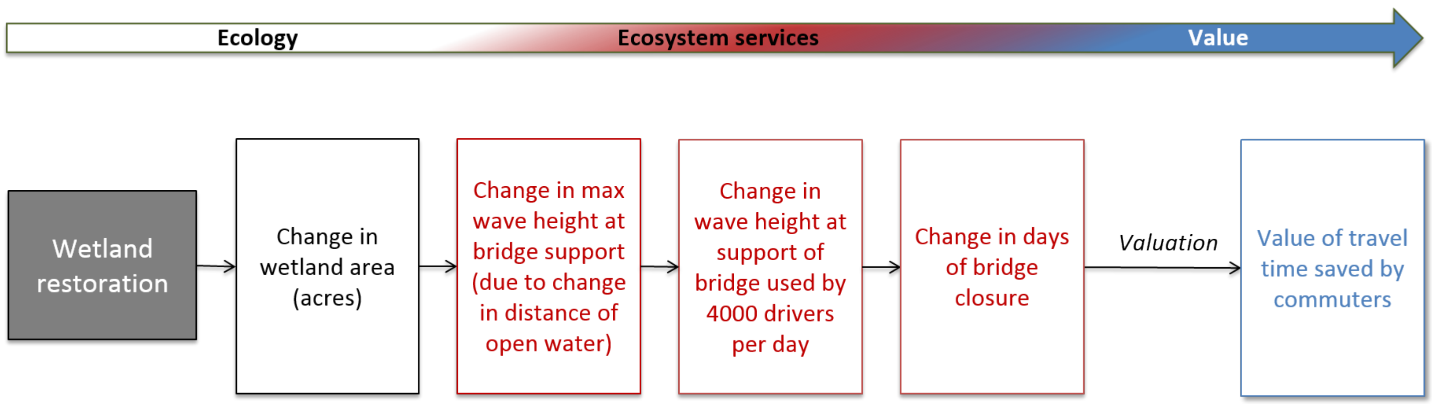
[61] Fetch calculation is only a benefit-relevant metric if it has already been established that waves can damage the bridge.

Figure 16. Causal chain representing the natural protection provided to a bridge by increase in wetlands



Thus, these examples reveal that two types of calculations are needed to move from ecological indicators to benefits: the *effectiveness* of the ecosystem change at creating benefits (or preventing harms) and the *number of people* who care or are affected. Measuring the effectiveness of the ecological change at producing the service (quantifying the change in the BRI) can require many types of expertise, including hydrology, engineering, restoration ecology, and anthropology. Ideally, the benefit-relevant factors will also suggest *how much* people care by evaluating the availability of substitutes for use services or by evaluating the scarcity of ecosystems or their elements for services with potential non-use values, which, in this example, is represented by how much commuters are collectively inconvenienced. However, still missing from this analysis for decision making is a comparison of the costs and benefits of restoring sufficient wetlands to prevent harm to the bridge versus enhancing the bridge to withstand storms.

The number of beneficiaries for many types of services can be estimated using everything from simple GIS analysis to sophisticated modeling. As one example, numbers of recreation users are well studied and can often be estimated directly from visitation data or by combining survey data on participation rates (e.g., 2011 National Survey of Fishing, Hunting, and Wildlife Associated Recreation[62]) with demographic data.[63] In other words, simple GIS models can be used to select likely participants within a site's user area. The user area might be defined by driving distance from a site (e.g., for recreation) or by hydrologic characteristics (for flood risk mitigation) or by other geographic characteristics, depending on the service. An estimate of use transforms a good BRI (e.g., game population increase) into a better BRI (e.g., change in annual provision of hunting user days) by showing the potential magnitude of welfare effects.[64]

Measuring substitutability requires making assumptions about what will change. If managers assume that land cover can be held constant, simple GIS screening rules can be used. For example, the number or areal extent of sites that have the same ecological characteristics and are in an appropriate location (e.g., within driving distance of the same population, or in the same floodplain) can be used to quantify potential substitutes for the site undergoing improvement or degradation through a management action. Screening for existing technical substitutes is also important. A wetland behind a dam may not be providing much in the way of flood control mitigation if the dam is already providing effective flood control.

Measuring reliability of a service requires making projections of expected future benefits given various sources of risk. Performance risk of restoration actions is often a key factor in determining whether future ecosystem services will be reliably produced. This risk can be calculated when data are available or can be derived from best professional judgment.[65] The future stream of benefits is also determined by the probability of the

[62] U.S. Fish and Wildlife Service. 2011. "2011 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation."

[63] J.M. Bowker, D. Murphy, H.K. Cordell, D.B.K. English, J.C. Bergstrom, C.M. Starbuck, C.J. Betz, and G.T. Green, "Wilderness and Primitive Area Recreation Participation and Consumption: An Examination of Demographic and Spatial Factors," Journal of Agricultural and Applied Economics 38 (2006):317–326.

[64] R.H. Von Haefen and D.J. Phaneuf, "Estimating Preferences for Outdoor Recreation: A Comparison of Continuous and Count Data Demand System Frameworks," Journal of Environmental Economics and Management 45 (2003):612–630.

[65] See, e.g, L.A. Wainger, D.M. King, R.N. Mack, E.W. Price, and T. Maslin, "Can the Concept of Ecosystem Services Be Practically Applied to Improve Natural Resource Management Decisions?" Ecological Economics 69 (2010):978–987.

best professional judgment.[65] The future stream of benefits is also determined by the probability of the ecosystem's persistence and desirable functioning. These factors can be predicted using land conversion models or assessed qualitatively by considering the land's protected status or zoning.[66] A variety of trend data might be brought to bear on explicit risk modeling or qualitative assessments, including use rates (e.g., well pumping effects on aquifer levels) and external drivers (e.g., sea level rise, fire risk). In all cases, managers use this approach to understand the probability that the site will stop functioning for the intended use, and they reduce the expected benefits in proportion to the level of risk. Thus, if the same benefits are measured at two sites but only one site has substantial risk of lost ecological function, the expected benefits of the high-risk site will be much lower than those of the site with minimal risk.

When economic valuation and multi-criteria decision analysis are not possible or appropriate, BRIs may be used to explore the magnitude of benefits, to evaluate how benefits vary across sites, or to compare potential benefits among sites. Carefully selected BRIs that include information, such as numbers of users and available substitutes, can provide additional information when used directly in decision making or when fed into analytic processes that quantitatively compare options in terms of one or more beneficial outcomes.

Recommended Reading

Boyd, J.W., and L. Wainger. 2002. "Landscape Indicators of Ecosystem Service Benefits." *American Journal of Agricultural Economics* 84(5): 1371–1378.

This article describes the rationale for and use of quantitative indicators to represent benefits and presents an example of their application in judging the adequacy of wetland mitigation.

McPhearson, T., P. Kremer, and Z.A. Hamstead. 2013. "Mapping Ecosystem Services in New York City: Applying a Social-Ecological Approach in Urban Vacant Land." *Ecosystem Services* 5: e11–e26.

This paper examines social and cultural considerations that are applicable to developing benefit-relevant indicators.

Naeem, S., J.E. Duffy, and E. Zavaleta. 2012. "The Functions of Biological Diversity in an Age of Extinction." *Science* 336: 1401–1406.

This paper describes dimensions of biodiversity characterization that may be useful for developing benefit-relevant indicators of non-use values.

Wainger, L.A., D.M. King, R.N. Mack, E.W. Price, and T. Maslin. 2010. "Can the Concept of Ecosystem Services Be Practically Applied to Improve Natural Resource Management Decisions?" *Ecological Economics* 69: 978–987.

This paper further develops some of the concepts presented in this guidebook and provides an example of the use of benefit indicators and risk assessment in optimization models designed to support decisions regarding invasive species management.

Wainger, L., and M. Mazzotta. 2011. "Realizing the Potential of Ecosystem Services: A Framework for Relating Ecological Changes to Economic Benefits." *Environmental Management* 48:710–733.

This article describes the rationale for benefit-relevant indicators and methods for managing or overcoming knowledge gaps during creation of indicators.

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[65] See, e.g, L.A. Wainger, D.M. King, R.N. Mack, E.W. Price, and T. Maslin, "Can the Concept of Ecosystem Services Be Practically Applied to Improve Natural Resource Management Decisions?" Ecological Economics 69 (2010):978–987.

[66] E.G. Irwin and J. Geoghegan, "Theory, Data, Methods: Developing Spatially Explicit Economic Models of Land Use Change," Agriculture, Ecosystems & Environment 85 (2001):7–24.

OVERVIEW OF BENEFITS ASSESSMENT

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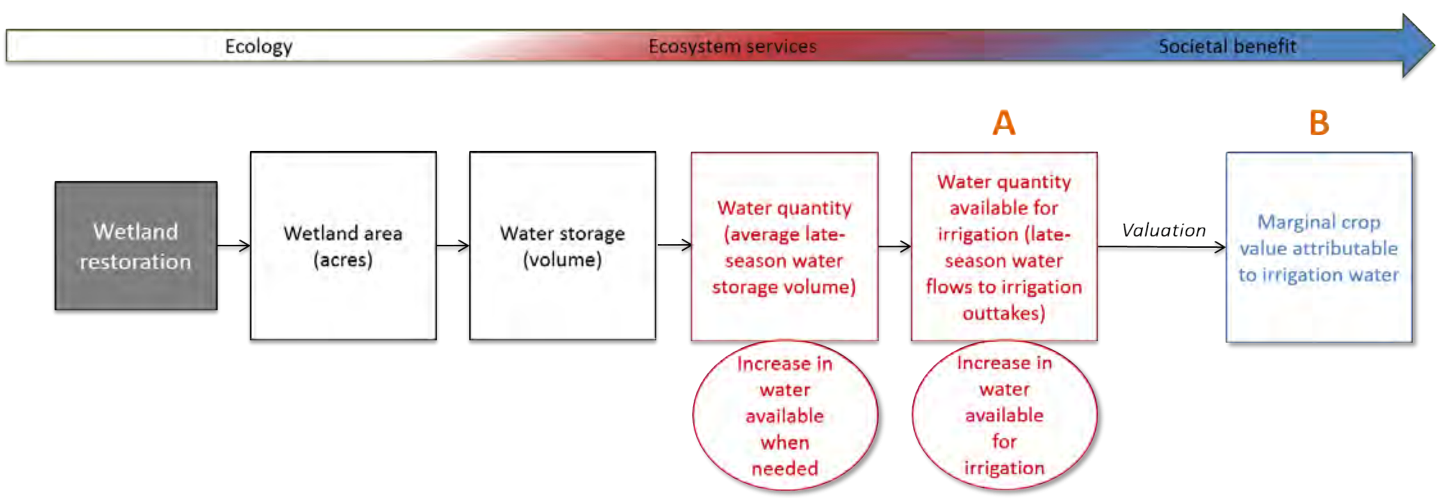
This section is adapted from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making" and combined with content from the first version of the FRMES online guidebook.

Information (for a formal analysis) or assumptions (for an intuitive decision) about social preferences or values are essential for decision makers to draw conclusions about how changes in the provision of ecosystem services will affect societal benefits. Even "more is better" conclusions require decision makers to assume a universally positive relationship between services and social welfare. When a decision involves tradeoffs (e.g., alternative policies that provide more of some benefit-relevant indicators (BRIs) and less of others), it is critical to understand the relative value people place on the different services. Otherwise, it is not possible to know which alternative policy option is preferable. Without benefits assessment, the analysis is left with conclusions regarding quantities of *what is valued* (e.g., irrigation water) (Figure 17, box A), without any information on *how much they are valued* (e.g., is more irrigation water worth the investment in wetland restoration?) (Figure 17, box B). Here, *benefits assessment* refers to a broad set of analytical methods, including both economic valuation and nonmonetary multicriteria analysis. While the term *value* can be used broadly, it is defined more narrowly in this framework to distinguish the methods commonly used. *Value* is used in the economic sense to imply well-defined, generally monetary measures of value. *Preference(s)* is used to reflect how individuals order outcomes on the basis of the relative satisfaction or enjoyment (i.e., utility) they provide; outcomes that generate greater utility also generate greater value.

Figure 17. Ecosystem services supply measured as a benefit-relevant indicator (BRI) (quantities of what is valued, A) and social benefit (how much the service is valued, B)



Note: The black text shows the ecological assessment and indicators; the red text shows the transition to ecosystem services assessment, with BRIs shown in the circles; and the blue text shows the final benefit as a value.

"Marginal" refers to a small additional change to an existing quantity. Consequently, marginal crop value would refer to the additional crop value provided by the action under study.

Although we consider quantification with BRIs to be minimum best practice, an ecosystem services analysis will always be more informative when rigorous information on values or preferences is included. A benefits assessment is helpful because a policy that influences a greater number of services is not necessarily superior to a policy that influences fewer. And more of a service is not always better. An increasing quantity of water in a river used for recreation will be a benefit up to a point but becomes a problem once the river begins to flood (i.e., value is not always a monotonic function of ecosystem service quantity). Benefits assessment can help ensure that assessors make appropriate inferences regarding the effect of changes in services on human wellbeing. One way of expressing people's preference for a given level of service, or for one service as compared to another, is in monetary terms (economic valuation); another is to form a unitless ranking (using nonmonetary methods).

Most regulatory impact analyses require economic valuation of some type, and many other types of federal decisions encourage or require some type of valuation. Office of Management and Budget guidance suggests that assessments of significant federal actions should monetize all primary effects that can be monetized.[67] Monetary expressions of value are often preferred in federal decisions. Expressing all benefits in a common monetary metric allows for analysis of tradeoffs among services and a clear bottom line in terms of net benefits. However, there are limitations to the use of monetary values to express the value of ecosystem services. In some federal decision contexts, the role of economic values is expressly limited.[68] In others, there is reluctance to monetize some kinds of ecosystem services, or the difficulty or expense of estimating monetary values may be large relative to agency resources.[69] Other limitations arise from cultural or religious prohibitions on monetizing some kinds of ecosystem services; the cultural value that tribes place on spiritual and religious artifacts and sites is a frequently cited example.[70] Nonmonetary methods can be used when dollar values are not desired and when an understanding of the differences among multiple stakeholder groups' values is preferable to the quantification of economic values.

Economic Valuation

Estimation of economic values, including both market and nonmarket values, enables an ecosystem services analysis to:

- Identify options that are socially beneficial (or optimal) from the broader set that are feasible,
- Provide a more informative analysis of tradeoffs, including benefits and costs realized by different affected groups (who gains and who loses and how much they gain or lose),
- Demonstrate whether the economic benefits of agency actions (including benefits realized outside of markets) outweigh the costs (benefit-cost analysis), and
- Inform the design of market-based programs to encourage provision of ecosystem services (payments for ecosystem services). For example, estimation of economic values can identify which types of users would be willing to pay to access or use ecosystem services of various types.

Methods for economic valuation have been developed and evaluated over the past five decades and are well established in both the scientific literature and guidance documents (see Monetary Valuation section).[71] Protocols and standards for these methods document the circumstances in which different types of valuation methods are appropriate. Valuation can be conducted at differing levels of accuracy, depending on

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[67] Office of Management and Budget, Guidance on Regulatory Impact Analysis, 2013, http://www.whitehouse.gov/sites/default/files/omb/inforeg/regpol/circular-a-4_regulatory-impact-analysis-a-primer.pdf.

[68] Examples include the Clean Air Act, the Endangered Species Act, the Resource Conservation and Recovery Act, the Safe Drinking Water Act, and sections of the Fishery Conservation and Management Reauthorization Act, among many others. See K.J. Arrow, M.L. Cropper, G.C. Eads, R.W. Hahn, L.B. Lave, R.G. Noll, P.R. Portney, M. Russell, R. Schmalensee, V.K. Smith, and R.N. Stavins, "Is There a Role for Benefit Cost Analysis in Environmental, Health and Safety Regulation?," Science 272 (1996): 221–222. A detailed, illustrative discussion of this issue in the context of fish stock rebuilding is provided by the Committee on Evaluating the Effectiveness of Stock Rebuilding Plans of the 2006 Fishery Conservation and Management Reauthorization Act, Evaluating the Effectiveness of Fish Stock Rebuilding Plans in the United States (Washington, DC: National Academies Press, 2014).

[69] L. Wainger and M. Mazzotta, "Realizing the Potential of Ecosystem Services: A Framework for Relating Ecological Changes to Economic Benefits," Environmental Management 48 (2011): 710–733.

[70] R. Winthrop, "The Strange Case of Cultural Services: Limits of the Ecosystem Services Paradigm," Ecological Economics 108 (2014): 208–214.

[71] P.A. Champ, K.J. Boyle, and T.C. Brown, A Primer on Nonmarket Valuation: The Economics of Non-Market Goods and Resources (New York: Springer, 2003); A.M. Freeman, J.A. Herriges, and C.L. Kling, The Measurement of Environmental and Resource Values: Theory and Methods, 3rd ed. (Washington, DC: RFF Press, 2014); D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments (Washington DC: RFF Press, 2010); National Ecosystem Services Partnership, Federal Resource Management and Ecosystem Services Guidebook (Durham, NC: National Ecosystem Services Partnership, Duke University, 2014),

the reasons for the analysis as well as data, time, and expertise available to conduct the analysis.[72] The level of accuracy may also be determined by required regulatory approvals (e.g., the Office of Management and Budget requires approvals for many types of data collection and regulatory analyses).[73] Economic valuation can be conducted alongside other forms of preference evaluation, to provide multiple perspectives on social value and preference.

An economist trained in monetary valuation can help decision makers ensure that the translation from BRIs to values is based on the application of valid and reliable methods. Valuation can be accomplished using a primary study at the site of interest (generating new valuation data and results) or with *benefit transfer*. Benefit transfer uses research results from preexisting primary valuation studies at one or more sites or policy contexts (often called study sites) to predict economic values at other, typically unstudied, sites or policy contexts (often called policy sites).[74] Primary data collection is more accurate but generally requires more time and resources to conduct.[75] In either case, specific methods are required to ensure that values meet minimum standards for validity and accuracy.[76] Only a subset of possible monetary measures associated with ecosystem services may be interpreted as measures of economic value. In the absence of the expertise required to conduct economic valuation methods, to evaluate them, or both, it is generally preferable to refrain from valuation (i.e., stop the analysis at the quantification of BRIs) rather than to generate values with unknown or questionable validity and accuracy.

Establishing the Link between BRIs and Economic Valuation

BRIs, if chosen appropriately, serve as the necessary ecological or biophysical inputs into economic valuation models; they are the measures that link biophysical measures or models to valuation estimates or models. That is, BRIs reflect the things that generate benefits or that are valued (directly or indirectly) within an economic valuation study. A carefully developed causal chain (or conceptual diagram) and a comprehensive set of BRIs associated with any policy action can also help analysts identify *all* the ways that the action might influence social value—whether through market or nonmarket channels. In this way, the use of BRIs can help ensure that values of nonmarket ecosystem services are appropriately recognized.

As noted in “Quantifying BRIs,” superior BRIs will be “closer” to the final services that provide value to people and hence better suited to valuation. That is, valuation models are more accurate and less subject to bias when the included variables are those that directly (rather than indirectly) influence values and behavior.[77] The economic literature provides guidance on the choice of specific BRIs for different types of revealed and stated preference valuation, although these works do not necessarily use the “BRI” terminology.[78] Different BRIs will

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https://nespguidebook.com; Office of Management and Budget, Guidance on Regulatory Impact Analysis, 2013, http://www.whitehouse.gov/sites/default/files/omb/inforeg/regpol/circular-a-4_regulatory-impact-analysis-a-primer.pdf; U.S. Environmental Protection Agency, Office of Policy, National Center for Environmental Economics, Guidelines for Preparing Economic Analyses, 2014, http://yosemite.epa.gov/ee/epa/eerm.nsf/vwAN/EE-0568-50.pdf/$file/EE-0568-50.pdf; Department of Commerce, National Oceanic and Atmospheric Administration, and National Marine Fisheries Service, Guidelines for Economic Review of National Marine Fisheries Service Regulation Actions, 2007, http://www.nmfs.noaa.gov/sfa/domes_fish/EconomicGuidelines.pdf.

[72] With respect to reason for the analysis, more accurate values are required for some types of applications. See J.D. Kline and M.J. Mazzotta, Evaluation of Tradeoffs among Ecosystem Services in the Management of Public Lands, U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, General Technical Report PNW-GTR-865, 2012, http://www.fs.fed.us/pnw/pubs/pnw_gtr865.pdf.

[73] R. Iovanna and C. Griffiths, “Clean Water, Ecological Benefits, and Benefits Transfer: A Work in Progress at the U.S. EPA,” Ecological Economics 60(2)(2006): 473–482.

[74] R.J. Johnston and R.S. Rosenberger, “Methods, Trends and Controversies in Contemporary Benefit Transfer,” Journal of Economic Surveys 24 (2010): 479–510; R.J. Johnston, J. Rolfe, R.S. Rosenberger, and R. Brouwer, Benefit Transfer of Environmental and Resource Values: A Guide for Researchers and Practitioners (Netherlands: Springer, 2015).

[75] B. Allen and J. Loomis, “The Decision to Use Benefit Transfer or Conduct Original Valuation Research for Benefit-Cost and Policy Analysis,” Contemporary Economic Policy 26(1)(2008): 1–12.

[76] R.J. Johnston, J. Rolfe, R.S. Rosenberger, and R. Brouwer, Benefit Transfer of Environmental and Resource Values: A Guide for Researchers and Practitioners (Netherlands: Springer, 2015); A.M. Freeman, J.A. Herriges, and C.L. Kling, The Measurement of Environmental and Resource Values: Theory and Methods, 3rd ed. (Washington, DC: RFF Press, 2014).

[77] J. Boyd and A. Krupnick, “Using Ecological Production Theory to Define and Select Environmental Commodities for Nonmarket Valuation,” Agricultural and Resource Economics Review 42 (2013): 1–32; R.J. Johnston and M. Russell, “An Operational Structure for Clarity in Ecosystem Service Values,” Ecological Economics 70(12)(2011): 2243–2249.

[78] R.J. Johnston and M. Russell, “An Operational Structure for Clarity in Ecosystem Service Values,” Ecological Economics 70(12)(2011): 2243–2249; R.J. Johnston, E.T. Schultz, K. Segerson, E.Y. Besedin, and M. Ramachandran, “Enhancing the Content Validity of Stated Preference Valuation: The Structure and Function of Ecological Indicators,” Land Economics 88(1)(2012): 102–120; J. Boyd and A. Krupnick, “Using Ecological Production Theory to Define and Select Environmental Commodities for Nonmarket Valuation,” Agricultural and Resource Economics Review 42 (2013): 1–32; P.L. Ringold, J. Boyd, D. Landers, and M. Weber, Report from the Workshop on Indicators of Final Ecosystem Services for Streams, EPA/600/R-09/137, 2009, http://www.epa.gov/nheerl/arm/streameco/docs/IndicatorsFinalWorkshopReportEPA600R09137.pdf; P.L. Ringold, A.M. Nahlik, J. Boyd, and D.

generally link to different values realized by different beneficiary groups. Hence, the most appropriate BRIs for use within any particular valuation model will depend on the type of value being estimated and the type of valuation model being used. This information is the same as that required to define any BRI, regardless of whether valuation will be conducted. For example, an analysis of recreational fishing values would require information on BRIs directly relevant to the behavior and values of recreational anglers such as changes in average or expected harvest rates of targeted species.

Accounting for Scope and Scale in Monetary Valuation

Economic values are meaningful only for a particular quantity of a market or nonmarket commodity, relative to a specific baseline. In other words, they are only meaningful when valuing a specific change in the provision of a service. If the change is large (i.e., nonmarginal), value estimation must account for the fact that per-unit values for any commodity generally diminish as more of that commodity is obtained (a phenomenon referred to as *diminishing marginal utility*, where utility is the amount of benefit obtained). For example, a recreational angler is generally willing to pay more per fish to increase her catch from 0 to 1 fish than from 99 to 100 fish; the value of a marginal fish depends on how many fish have already been caught.[79] In most cases, the change in a BRI cannot be multiplied by a simple “unit value” to arrive at a total value of the change (at least for nonmarginal changes); doing so would overlook the fact that marginal values tend to diminish as quantity or quality increases. Similarly, values per unit of area (e.g., per acre) generally cannot be calculated and multiplied by the total affected area.

Consequently, applying values determined for one scale of change to another scale of change is inappropriate, making it difficult to estimate regional or national values from local values. In a small number of cases, value scaling may be feasible. An example would be small-scale localized changes in a good valued due to its global consequences (or because it is sold on global markets), such as changes in local greenhouse gas emissions. An economist trained in valuation can help determine whether and how value scaling is (or is not) appropriate in particular valuation contexts.

Nonmonetary Multicriteria Analytical Methods

When monetization of all or some of the ecosystem services measures in an analysis is inappropriate or too difficult to do well, assessors can use a variety of analytical methods with both monetized and nonmonetized components to develop a ranking or rating of alternatives with respect to their contributions to stakeholder preferences for ecosystem services. Several of these methods are described in a handbook developed by the London School of Economics to advise local governments on use of multicriteria analysis.[80] Some of the methods (e.g., outranking procedures and the Analytical Hierarchy Process) are less demanding of information on both performance of management alternatives and expressions of preference than others (e.g., multiattribute utility analysis, or MAUA), but they are correspondingly less transparent and thus less informative for the kinds of multiparty deliberative decision making that often characterizes resource management.[81] Although MAUA has been criticized as too time-consuming and too dependent on expertise that agencies may not have, it has the advantage of obliging users to think carefully about all the elements of preference evaluation in a systematic way. There is value in using MAUA concepts to inform decision making when incorporating ecosystem services, even when a full quantification does not appear feasible.

Multicriteria analysis can be used for resource management problems such as impact assessment, in which one alternative must be selected; for resource allocation among potential activities; and for prioritization of

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Bernard, Report from the Workshop on Indicators of Final Ecosystem Goods and Services for Wetlands and Estuaries, U.S. Environmental Protection Agency, 2011; P.L. Ringold, J. Boyd, D. Landers, and M. Weber, “What Data Should We Collect? A Framework for Identifying Indicators of Ecosystem Contributions to Human Well-Being,” Frontiers in Ecology and the Environment 11(2013):98–105, http://dx.doi.org/10.1890/110156; E.T. Schultz, R.J. Johnston, K. Segerson, and E.Y. Besedin, “Integrating Ecology and Economics for Restoration: Using Ecological Indicators in Valuation of Ecosystem Services,” Restoration Ecology 20(4)(2012): 304–310; M. Zhao, R.J. Johnston, and E.T. Schultz, “What to Value and How? Ecological Indicator Choices in Stated Preference Valuation,” Environmental and Resource Economics 56(1)(2013): 3–25.

[79] R.J. Johnston, M.H. Ranson, E.Y. Besedin, and E.C. Helm, “What Determines Willingness to Pay per Fish? A Meta-Analysis of Recreational Fishing Values,” Marine Resource Economics 21(1)(2006): 1–32.

[80] Department of Communities and Local Government, Multi-Criteria Analysis: A Manual, 2009, www.communities.gov.uk. See appendices 1 to 8.

[81] Department of Communities and Local Government, Multi-Criteria Analysis: A Manual, 2009, www.communities.gov.uk. See outranking procedures in appendices 6 and 7, the analytical hierarchy process in appendix 5, and multiattribute utility analysis in appendices 3 and 4.

targets for action (such as candidates for ESA listing). The stakeholder group selected to assign relative utility to ecosystem services outcomes (or BRIs) will vary depending on whether the agency is aiming to assess general public preferences or to fulfill select mission goals. MAUA will only be considered representative of public preferences if representative members of the public are included. Processes that occur within agencies can only be said to represent agency goals.

Multicriteria analysis has been used in various federal decision contexts, such as in National Estuary Program planning in Oregon and remedial planning for contaminated sites.[82] It is useful for comparison of preferences among alternatives but not for estimations of value in any absolute sense. The components and results of analysis are tied to the decision context, including the items being evaluated (e.g., alternative management plans) and the range of performance of those alternatives for each ecosystem service. Multicriteria analysis can be used at any scale from local to international, but it cannot readily be scaled up or down without a great deal of additional work to establish that preference information is relevant to contexts other than those for which preferences were originally gathered. It is particularly useful for decisions affecting multiple stakeholders and for public decisions requiring a transparent decision process.

BRIs and associated measurement scales represent the ecosystem services being pursued in a particular decision context. Multicriteria analysis assigns relative preferences to different levels of a single BRI (and these preferences can differ among stakeholders and among decision contexts) and different weights/priorities among multiple BRIs in order to create a single combined metric of overall contribution to ecosystem services.

Multicriteria evaluation and the accompanying primer on multicriteria analysis[83] Several books also address practical applications of these methods.[84] In addition, the U.S. Geological Survey and other federal agencies have developed instructional materials on structured decision making that enable training of agency personnel.[85] People who have participated in this training can be called on by agencies that want to apply these tools for nonmonetary valuation of ecosystem services.

Context Dependency and Transferability of Stakeholder Values and Preferences

Both monetary valuation and multicriteria analysis are context dependent in ways that make it difficult to transfer values or preferences among case studies. In both, stakeholder preferences are expressed in terms that depend on the considered management alternatives relative to the stated or observed baselines. The stakeholders themselves are localized in space (people in different regions might have very different values) and in time (both at the time that the assessment is completed and to the extent that the planning horizon affects values). These situational particulars are, of course, precisely the particulars that should inform any decision. But they also are a drawback in that they imply that monetary valuation and multicriteria decision analyses must be conducted for every management decision—a substantial investment.

One response to the context dependency of value is to extrapolate the results of social impact or valuation studies from one or more study sites to unstudied sites that have sufficiently similar contexts. Within economic valuation, this approach is called *benefit transfer,* and it can be conducted using single values (e.g., average values) or transfer models that adjust values according to site differences. In all cases, analysts must match studies from which values or drawn or models are built as closely as possible to the ecological outcomes and decision contexts of interest. Once benefit transfer models are built, model variables are used to adjust values according to sociodemographic variables and degree of ecological change, among other variables. Best practices for conducting benefit transfer have been established and should be followed closely in all uses.

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[82] J. Arvai and R. Gregory, "Testing Alternative Decision Approaches for Identifying Cleanup Priorities at Contaminated Sites," Environmental Science Technology 37(8)(2003): 1469–1476; R. Gregory and K. Wellman, "Bringing Stakeholder Values into Environmental Policy Choices: A Community-Based Estuary Case Study," Ecological Economics 39 (2001): 37–52; I. Linkov, A. Varghese, S. Jamil, T.P. Seager, G. Kiker, and T. Bridges, "Multi-Criteria Decision Analysis: Framework for Applications in Remedial Planning for Contaminated Sites," in Comparative Risk Assessment and Environmental Decision Making, ed. I. Linkov and A. Ramadan, 15–54 (Amsterdam: Kluwer, 2004).

[83] Lynn A. Maguire, *Multicriteria Evaluation for Ecosystem Services* (Durham, NC: Nicholas School of the Environment, Duke University, 2014), http://sites.nicholasinstitute.duke.edu/ecosystemservices/files/2014/08/MCDA-Primer-Maguire.pdf.

[84] R. Gregory, L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson, Structured Decision Making: A Practical Guide to Environmental Management Choices (Oxford: Wiley-Blackwell, 2012).

[85] http://nctc.fws.gov/courses/SDM/home.html.

With multicriteria decision frameworks, the aim of any given project is to capture the richness of context dependencies as faithfully as possible. This aim does not lend itself readily to the transfer of results among studies. Extrapolation of study results has not been an objective of practitioners of decision analysis, and transfer methods are not well developed. Practitioners of both economic valuation and multicriteria analysis would consider generally transfer of results from one study to another as inappropriate when the decision context and site details cannot be closely matched, unless formal, quantitative models can establish that values from dissimilar sites and contexts are expected to be similar.[86]

MONETARY VALUATION

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Summary

This section provides basic guidance for using and conducting economic valuation, including criteria for judging whether valuation is appropriate for supporting decisions. It provides an introduction to the economic techniques used to measure changes in social welfare and describes which methods may be most appropriate for use in valuing particular ecosystem services. Rather than providing comprehensive valuation instructions,it directs readers to additional resources.More generally, it establishes that the valuation of ecosystem services is grounded in a long history of non-market valuation and discusses how ecosystem services valuation can be conducted within established economic theory and techniques.

Takeaways

- Ecosystem services valuation provides information unavailable through other approaches and is the only means to compare the net social benefits (benefits – costs) associated with policies and actions that affect ecosystem services.
- The validity, consistency, and comparability of ecosystem service values require that managers use methods that consistently measure values as the tradeoffs that individuals are (or would be) willing to make.
- Ecosystem services values are meaningful only for quantified changes in ecosystem services compared to a baseline; they are not meaningful when summed over entire ecosystems.
- It is challenging to design and conduct valuation studies that overcome knowledge gaps and avoid the common pitfalls of ecosystem services valuation.
- Engaging economists with expertise in non-market valuation, along with ecologists and other relevant scientists, will promote sound economic analysis.

[86] Moeltner, K. and R.S. Rosenberger. 2014. Cross-Context Benefit Transfer: A Bayesian Search for Information Pools. American Journal of Agricultural Economics 96(2): 469-488. Johnston, R.J., and K. Moeltner. 2014. Meta-Modeling and Benefit Transfer: The Empirical Relevance of Source-Consistency in Welfare Measures. Environmental and Resource Economics 59(3): 337-361.

Introduction to Monetary Valuation

Economic valuation of ecosystem services can provide decision makers with evidence of the social benefits provided by, and tradeoffs among,regulatory alternatives and other ecosystem management or policy actions.Valuation implies a systematic quantification of benefits and costs realized by society in commensurable (typically monetary) units, using methods grounded in economic theory.[87] **The strengths of monetary valuation, including aggregation of benefits and comparison of benefits and costs, have led to its widespread use and acceptance by decision makers.**

A basis in economic welfare theory is one of the primary distinguishing features of economic valuation. This underlying theory provides a formal structure necessary to link estimated monetary values with changes in net well-being or social welfare. Without this link, there is no guarantee that a monetary measure will have any correspondence to social welfare.

Economic value measures can also be aggregated over affected populations, typically by multiplying the number of affected parties by the magnitude of benefit or harm to each. **When monetary measures of ecosystem services have been estimated using methods that share this theoretical grounding, they are internally consistent and directly comparable**, regardless of the source or cause of the benefit or cost. This internal consistency and comparability, together with more than five decades of methodological development, are some of the main reasons that monetary values are the most commonly used quantitative measures of ecosystem services value within policy analysis.

The *FRMES Guidebook* describes a broad suite of approaches for assessing ecosystem services benefits using monetary values and non-monetary benefit indicators. This section focuses on valuation and, reflecting U.S. Office of Management and Budget (OMB) guidance for benefits analysis, defines *valuation* as economic valuation or monetization.[88] Other approaches are referred to as benefit assessments and include the use of benefit indicators that are weighted using risk ranking, multi-attribute utility theory, and multi-criteria decision analysis to reflect public preferences, agency goals, or both and are covered in more detail in the Non-Monetary Valuation Methods section of this guidebook.

When choosing among the benefit assessment techniques presented in this guidebook, consider that economic valuation is rarely the sole input to decision making. In most cases, a variety of other considerations will enter into the decision process that either cannot or should not be monetized. When valuation cannot provide an adequate range of quantitative information required for policy or program analysis, policymakers can supplement or supplant economic valuation with other metrics or approaches. For example, non-monetary benefit metrics or benefit-relevant indicators can be a cost-effective method for comparing alternatives that are difficult to distinguish with monetization.

Agencies can review the Ecosystem Services Assessment Framework Overview to determine which benefit assessment approach best fits their decision-making needs. Choosing a benefit assessment technique(monetary valuation, multi-criteria decision analysis, or quantification of benefit-relevant indicators) requires evaluating the type and accuracy of information needed in a given policy context. For example, legal proceedings (e.g., natural resource damage litigation) may require different standards than collaborative decision making with stakeholders. Some techniques might characterize the effects and tradeoffs of greatest concern better than others. In addition, valuation and non-monetary benefit assessment techniques vary in terms of data and time requirements, so the availability of funding and other resources will influence decisions.

As this guidebook's ecosystem services assessment framework makes clear, accurate valuation, or any benefit assessment, requires analysts to be able to estimate biophysical cause-and-effect relationships, effects on ecosystem services, and the welfare effects of those biophysical changes. Economic theory imposes specific guidelines on how empirical data can be used to quantify and combine values to prevent the misuse and misinterpretation of data and results.[89] Consequently, successful valuation almost always requires the

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[87] A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,*3rd ed. (Washington, D.C.: RFF Press, 2014); Just, R.E., D.L. Hueth, and A. Schmitz, *The Welfare Economics of Public Policy: A Practical Approach to Project and Policy Evaluation* (Cheltenham, UK: Edward Elgar Publishing, 2004).

[88] U.S. Office of Management and Budget, "Regulatory Analysis," M-03-21, OMB Circular No. A-4 (2003), http://www.whitehouse.gov/omb/memoranda_m03-21.

[89] N.E. Bockstael, A.M. Freeman, R.J. Kopp, P.R. Portney, and V.K. Smith, "On Measuring Economic Values for Nature," *Environmental Science and*

collaboration of ecologists and economists (and possibly others) from the beginning of the process to ensure that the biophysical assessment generates metrics (i.e. benefit relevant indicators) that can be used within economic analysis and to avoid misuse of data when these biophysical changes are translated to measures of economic value. After the metrics have been generated, economists tailor the economic analysis to the ecosystem, social setting, and decision context to ensure that values are estimated from the biophysical changes and that metrics are valid, representative, and useful.

The intent of this section is to provide basic guidance for economic valuation, not to provide comprehensive instructions for all valuation methods. A great deal of research has explained and demonstrated methods and best practices for broadly valuing ecosystem services and non-market resources. This section presents concepts and considerations applicable to choosing an approach to benefits assessment and describes valuation approaches that are most relevant to ecosystem service analysis. Best practices for applications of economic valuation can also be found in the NESP report "Best Practices for Integrating Ecosystem Services into Federal Decision Making."[90]

Defining Economic Value

Economic values or benefits measure gains in social welfare (or well-being). The economic value of a good or service to an individual or group reflects the increase in well-being that the good or service generates. In the context of ecosystem services, economic values measure gains in social welfare resulting from changes in outputs of natural ecosystems. Because it is impossible to directly observe changes in social welfare, economists measure values by observing the tradeoffs that individuals are (or would be) willing to make. That is, they measure economic value as the amount of one good or service that a person would be willing to exchange for a specific quantity of another good or service, rather than go without the service. When monetary units are used for quantifying a change in welfare, the result is a measure of monetary value. Monetary units are a convenient unit of measure for assessing value, but other metrics can, under certain conditions, be used to represent value.

Valuation often quantifies economic value using individuals' or groups' *willingness to pay* (WTP) for an additional unit of ecosystem goods or services, given the quality of the good or service, the availability of substitutes, and other context variables that affect demand relative to supply. This definition also implies that economic values are based on subjective assessments of individuals regarding their own welfare and the types of tradeoffs that would enhance that welfare. Although the vast majority of economists accept this approach, some competing schools of economic thought propose that individuals may not have or be able to generate preferences that reflect their well-being for all types of goods and services.[91] These competing, minority perspectives argue that it may sometimes be appropriate to use group, expert,or political judgment for valuation rather than relying solely on actions or responses of individuals. For example, public spending to enhance wetlands might be used to value a bundle of ecosystem services that the wetland generates—a practice known as public pricing—under the assumption that the government has the information necessary to act in the public interest or the assumption that regulations and government policies reflect public preferences because the public votes for the legislators making the regulations. However, because this is a minority view, valuation methods that rely on such judgments may not be acceptable to all types of decision makers, nor are they approved for use by most government agencies to measure economic benefits.[92]

Drawing from the above definition of value, WTP measures the amount of money or some other commodity that an individual or group would be willing to give up to obtain a specified quantity of an ecosystem service, compared to a given baseline quantity. Value may also be quantified in terms of *willingness to accept* (WTA),

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Technology 34 (2000):1384–1389; A.M. Freeman, J.A.Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd ed. (Washington, D.C.: RFF Press, 2014); R.E. Just, M.A. Wilson, and J.P. Hoehn, "Valuing Environmental Goods and Services Using Benefit Transfer: The State-of-the Art and Science," *Ecological Economics* 60 (2006):335–342; D.M. Bauer and R.J. Johnston, "The Economics of Rural and Agricultural Ecosystem Services: Purism versus Practicality," *Agricultural and Resource Economics Review* 42 (2013):iii–xv; and R.E. Just, D.L. Hueth, and A. Schmitz, *The Welfare Economics of Public Policy: A Practical Approach to Project and Policy Evaluation* (Cheltenham, UK: Edward Elgar Publishing, 2004).

[90] L. Olander, R.J. Johnson, H. Tallis, J. Kagan, L. Maguire, S. Polasky, D. Urban, J. Boyd, L. Wainger, and M. Palmer, *Best Practices for Integrating Ecosystem Services into Federal Decision Making* (Durham: National Ecosystem Services Partnership, Duke University), accessed January 27, 2016, https://nicholasinstitute.duke.edu/ecosystem/publications/best-practices-integrating-ecosystem-services-federal-decision-making/.

[91] D.W. Pearce and R.K. Turner, *Economics of Natural Resources and the Environment* (London: JHU Press, 1990).

[92] See, e.g., U.S. Environmental Protection Agency, "Guidelines for Preparing Economic Analyses," EPA 240-R-10-001. Washington, D.C.: U.S. EPA Office of the Administrator (2010).

defined as the minimum amount that a person or group would be willing to be compensated in order to give up a specified quantity of an ecosystem service that they already have or would otherwise get in the future under business as usual. Given these definitions, **ecosystem services values are only meaningful for quantified changes in ecosystem services compared to a baseline, and not when summed over entire ecosystems.**[93]

Valuation must account for many variables that can influence the benefits provided by ecosystems, but an issue of particular relevance to ecosystem services is that the value of a change in a system depends on the initial state of the ecosystem or resource. System state (i.e., level of degradation) affects both the magnitude of system response and the significance of that response to users or beneficiaries. For example, a pollution reduction that allows game fish to re-establish in a stream can generate a more substantial increase in recreational fishing opportunities compared to a nutrient reduction in a moderately degraded stream that already contains abundant game fish. In addition, an angler may be willing to pay more per fish to increase her catch from 0 to 1 fish than from 49 to 50 fish, reflecting diminishing marginal returns.

Thus, meaningful economic valuation requires information on the magnitude of the change in ecosystem services from current (and future baseline) conditions and the context-specific value of that change to beneficiaries. The size of the change alone does not dictate value. Rather, **the relative scarcity of the good or service (demand relative to supply) determines the value of a change in that good or service**. The supply of an ecosystem service is determined by natural ecological processes (often subject to external human influences or stressors). Demand for an ecosystem service is influenced by many factors, including human (subjective) preferences, income, and people's willingness or ability to substitute other goods and services for the good or service in question.

WTP and WTA are alternative inputs into the calculation of benefits.These distinct ways to measure economic value are appropriate to use in different circumstances because they make different assumptions about who has the right to a good or service.[94] WTP and WTA measures may be calculated in a variety of ways, depending on the beneficiary group. For example, one of the most common metrics used to quantify economic benefits or value for individuals is *consumer surplus*. Consumer surplus is interpreted as the difference between what an individual (or group of individuals) would be willing to pay for a given level of a good or service and what is actually paid, summed over all units of the good or service that are consumed (or used). The change in consumer surplus across all affected individuals may be aggregated to estimate the value of a change in an ecosystem good or service. A parallel measure for producers is *producer surplus*, which is conceptually similar to economic profit.[95]

Other metrics that are frequently used as proxies for welfare changes, such as prices, avoided costs, or replacement costs have weaker (or no) ties to social welfare within economic theory. These metrics can still prove useful,because they are relatively easy to measure and can sometimes provide information similar to economic benefit metrics.[96] However, the use of these metrics, which do not directly measure economic benefits, must be applied with caution.[97]

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[93] N.E. Bockstael, A.M. Freeman, R.J. Kopp, P.R. Portney, and V.K. Smith, "On Measuring Economic Values for Nature," *Environmental Science & Technology* 34 (2000):1384–1389; M.L. Plummer, "Assessing Benefit Transfer for the Valuation of Ecosystem Services,"*Frontiers in Ecology and the Environment* 7 (2009):38–45.

[94] W.M. Hanemann, "Willingness to Pay and Willingness to Accept: How Much Can They Differ?" *The American Economic Review* 81 (1991):635–647.

[95] K.E. McConnell and N.E. Bockstael, "Valuing the Environment as a Factor of Production," in *Handbook of Environmental Economics,* ed. by K.G. Mler and J.R. Vincent (Elsevier: 2005), 621–669; D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010); A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd ed. (Washington, D.C.: RFF Press, 2014). For a technical discussion of consumer surplus, producer surplus, and other similar welfare measures used by economists, see R.E. Just, D.L. Hueth, and A. Schmitz, *The Welfare Economics of Public Policy: A Practical Approach to Project and Policy Evaluation* (Cheltenham, UK: Edward Elgar Publishing, 2004).

[96] An example of the use of property damages in an economic valuation framework of wetlands can be found in E.B.Barbier, I.Y. Georgiou, B. Enchelmeyer, and D.J. Reed, "The Value of Wetlands in Protecting Southeast Louisiana from Hurricane Storm Surges," *PLoS ONE* 8 (2013):e58715.

[97] National Research Council, *Valuing Ecosystem Services: Toward Better Environmental Decision-Making* (Washington D.C.: National Academies Press, 2005), http://www.nap.edu/catalog.php?record_id=11139; and U.S. Environmental Protection Agency, *Valuing the Protection of Ecological Systems and Services: A Report of the EPA Science Advisory Board*(Washington, D.C.: U.S. EPA Science Advisory Board Committee on Valuing the Protection of Ecological Systems and Services, 2009).

To understand the concerns raised by the use of prices to estimate values, consider a scenario in which a person pays $8 for a bottle of maple syrup. The person loves maple syrup and would have been willing to pay $20 for the bottle. Now suppose someone uses the $8-per-bottle market price and market size to estimate willingness to pay to protect a grove of maple trees.That person will have underestimated willingness to pay to protect the trees for maple syrup production, not to mention all the other values people might have for protecting the trees.

On the other hand, the market value of property saved due to a given flood control investment might be considered acceptable for comparing alternative investments in flood control, as long as decision makers are aware of the pros and cons of this measure. Furthermore, they should be aware of the assumptions that are embedded in use of this measure to reflect social welfare, namely that the value of the property is equivalent to lost welfare. Decision makers should be aware that price and cost data are considered proxies, often for only one service, rather than direct measures of social welfare. As such, they do not provide reasonable or informative metrics of value in all circumstances. An economist trained in valuation can help determine whether and how these proxies may be useful for approximating economic values.

Total Economic Value

Decision makers often seek the most comprehensive assessment possible in order to promote understanding of how ecosystems provide everything from basic life support to financial and social well-being. Although thoroughness is desirable, complete measurement of all direct and indirect ecosystem services benefits due to a change is rarely possible. Moreover, attempting to be comprehensive increases the risk that benefits will be double counted. **The difficulty of putting a dollar value on all ecosystem services is well understood among practitioners and is the reason that OMB guidance suggests that assessments should monetize what is possible to monetize, quantify what cannot be monetized, and describe what can be neither monetized nor quantified** for regulatory rule making.[98]

In an effort to be comprehensive about assessing ecosystem service changes, practitioners have attempted to quantify the total economic value of entire landscapes, biomes, or other very large systems. As noted above, **economic value measures are only meaningful for changes in ecosystem services from a known baseline**. Moreover, estimated values for small changes cannot generally be scaled up (at least to any significant degree) to calculate values for large changes in systems, such as the systems' complete loss. Careful selection of ecosystem services for valuation and calculation of values only for changes in ecosystem services rather than entire landscapes can avoid some of these common pitfalls.

Sometimes those seeking to calculate a total value for all or some services provided by a large ecosystem have confused the resulting (generally invalid) estimates with valid measures of *total economic value* (TEV). This confusion reflects a misunderstanding of the concept. Economists use the concept of TEV to reflect the fact that changes in ecosystem services can simultaneously affect many different types of values, including both *use* and *non-use* (also known as *passive use*) *values*.[99] That is, in the context of ecosystem service valuation, TEV is used by economists to emphasize that a total or complete value measure must incorporate the full range of the *types* of value and people, including user and nonuser groups, affected by a change. Overlooking one or more of these distinct types of value can lead to large errors and sub-optimal decisions even if many ecosystem services have been valued.The concept of TEV does *not* imply that one is measuring the total value of an entire landscape or ecosystem. Like all economic values, TEV is only meaningful when well-defined changes from a known baseline are considered.

A typology of ecosystem services values can be used to define TEV (Figure 18). Use values are those that result from direct use (e.g., bird-watching or hunting on site) or indirect use (e.g., flood risk mitigation from proximal wetlands) of ecosystem services or related resources. *Non-use values*, in contrast, are values that do not require observable use or consumption of the service.[100] More broadly, non-use values for ecosystem

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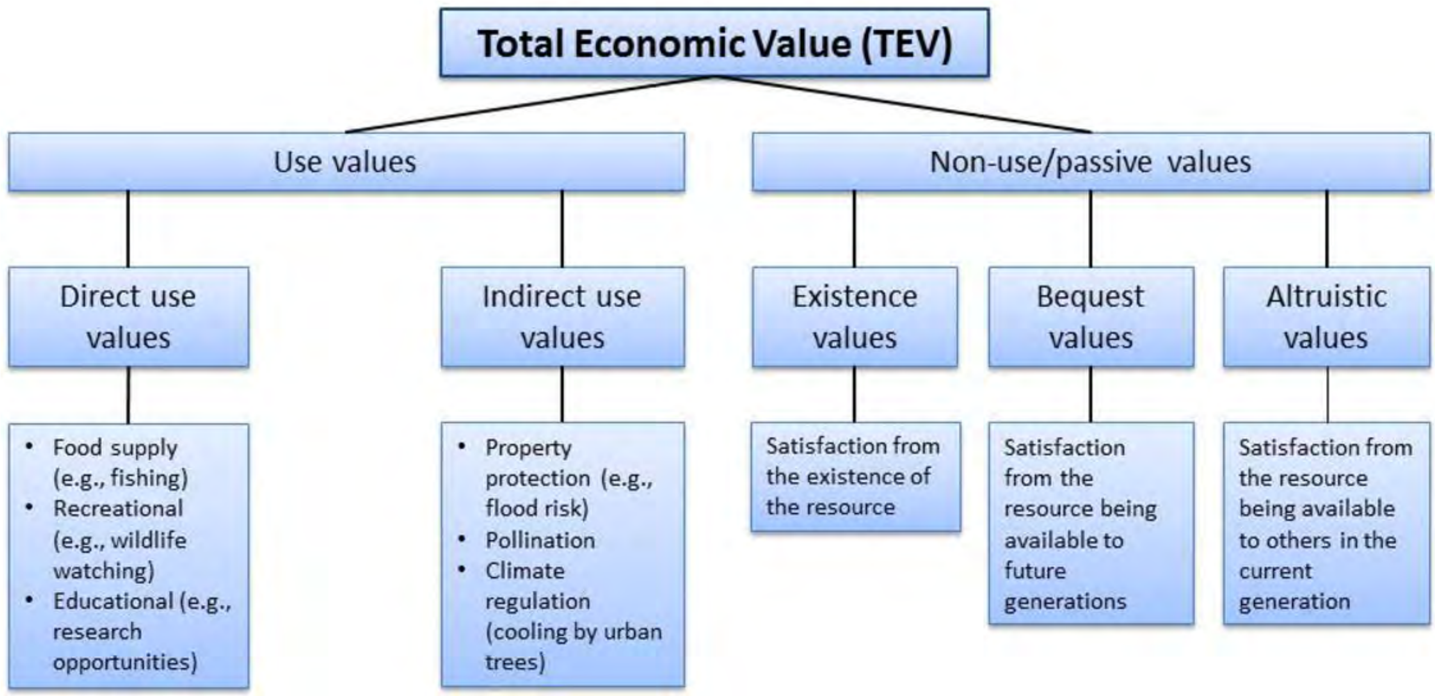
[98] U.S. Office of Management and Budget, "Regulatory Analysis," M-03-21, OMB Circular No. A-4 (2003), http://www.whitehouse.gov/omb/memoranda_m03-21.

[99] A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd ed. (Washington, D.C.: RFF Press, 2014), and D.W. Pearce, A. Markandya, and E. Barbier, *Blueprint for a Green Economy* (London: Earthscan, 1989).

[100] A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,*3rd ed. (Washington, D.C.: RFF Press, 2014), and D.W. Pearce, A. Markandya, and E. Barbier, *Blueprint for a Green Economy* (London: Earthscan, 1989).

services are associated with protecting natural assets (e.g., species or ecosystems) because people value the pure existence of these assets, want to pass these assets along to future generations, or think that these assets ought to be protected regardless of human use.[101] *Option value*, which reflects the value of preserving the option to use a resource in the future, had been considered a non-use value. But recent research suggests that option value is more appropriately considered an implicit component of use services, rather than a theoretically distinct and separable component of use value.[102]

Figure 18. Components of total economic value and relevant valuation methods



So, for example, the TEV of a marginal change in waterfowl abundance at a wetland site might include use values realized by waterfowl hunters and birders, along with non-use values realized by those who simply value the existence of these birds. In concept, these and other values are simply summed to calculate the total value generated by the marginal change. Summing must avoid any overlaps in these value estimates that can lead to double counting.

Applying Monetary Benefit Assessment to Decision Making

Many economic frameworks are available to evaluate policy choices with and without monetization of ecosystem services values.The most common are cost-benefit analysis (CBA) and cost-effectiveness analysis (CEA).

CBA is the most comprehensive and is specifically designed to quantify total net effects on human welfare. Monetized ecosystem services values are a key component of a CBA, for decisions that affect ecosystems. Although CBA provides more explicit guidance than CEA about whether an action is socially desirable, it also requires that the outcomes expected to generate the major benefits and the costs be monetized.

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101 National Research Council, *Perspectives on Biodiversity: Valuing Its Role in an Everchanging World* (Washington D.C.: National Academies Press, 1999).

102 N. Hanley, J.F. Shogren, and B. White, *Introduction to Environmental Economics* (Oxford, UK: Oxford University Press, 2001) and A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,*3rd ed. (Washington, D.C.: RFF Press, 2014).

CEA uses non-monetary metrics of beneficial outcomes as proxies for social welfare (e.g., lives saved, species extinctions prevented) and evaluates the costs of achieving changes in these metrics (e.g., dollars per extinction prevented). CEA is used to identify the options that achieve desired outcomes at the lowest cost, but it does not quantify benefits in monetary terms.

Which approach—CBA or CEA—is most appropriate will depend on the type of decision being made as well as the state of the ecological and social science.

Cost-Benefit Analysis

Cost-benefit analysis (CBA) is often required for federal rule making and is the most well-accepted method for communicating the economic desirability and importance of an action to government officials and the public.[103] However,it is not without its critics who question its capacity to provide a full accounting of benefits and to help decision makers weigh distributional or equity issues.[104]

CBA includes a systematic quantification and comparison of well-defined economic benefits and costs resulting from a particular set of actions (e.g., a policy change or management action). It is designed to help society make decisions that increase net economic benefits, considering both present and future outcomes. To that end, CBA requires that key benefits be monetized so that they can be directly compared to costs. For example, CBA may be used to demonstrate that the public welfare is served by a decision (e.g., enacting a regulation or policy) because, when the major benefits and costs are compared, benefits exceed costs.

One of the primary strengths of CBA is that it enables diverse benefits to be evaluated with a common unit (dollars) that directly and explicitly measures social welfare. It is also the only broadly accepted economic approach that is designed to estimate the full range of economic costs and benefits, or effects on human welfare, associated with management or policy actions.

CBA has a long history in economics, and there are now many textbooks and instructional manuals describing its use, along with an established research literature. Individual government agencies have also provided guidance for the use of CBA within policy analysis.[105]

Cost-Effectiveness Analysis

Cost-effectiveness analysis (CEA) is used to identify the most efficient means of achieving a goal that has been established by legislation, group consensus, technical evaluation, resource management plan, or other means.[106] Within CEA, policy benefits are frequently quantified in terms of biophysical indicators that are conceptually linked to human welfare.[107] For example, a biophysical change (i.e., a benefit relevant indicator) such as increased groundwater recharge can affect welfare when it reduces pumping costs or enables water use restrictions to be relaxed. However, rather than value groundwater recharge directly, CEA would evaluate the costs of obtaining different possible levels of these indicators (i.e., dollars per million gallons of groundwater recharge).

[103] A.E. Boardman, D. Greenberg, A.R. Vining, and D.L. Weimer, *Cost-Benefit Analysis: Concepts and Practice,* 3rd ed. (Upper Saddle River, NJ: Prentice Hall, 2006), http://www.alibris.com/Cost-Benefit-Analysis-Concepts-and-Practice-Anthony-Boardman/book/1356068.

[104] M.D. Adler and E.A. Posner, “Rethinking Cost-Benefit Analysis,” *Yale Law Journal* 109 (1999):165–247; F. Ackerman and L. Heinzerling, “Pricing the Priceless: Cost-Benefit Analysis of Environmental Protection,” *U. Pa. L. Rev.* 150 (2001):1553; and R. Turner, “Limits to CBA in UK and European Environmental Policy: Retrospects and Future Prospects,” *Environmental and Resource Economics*37 (2007):253–269.

[105] See, e.g., U.S. Environmental Protection Agency, “Guidelines for Preparing Economic Analyses,” EPA 240-R-10-001, Washington, D.C.: U.S. EPA Office of the Administrator (2010).

[106] D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010).

[107] C.L. Spash and A. Vatn, “Transferring Environmental Value Estimates: Issues and Alternatives,” *Ecological Economics* 60 (2006):379–388 and L.A. Wainger, and J.W. Boyd, “Valuing Ecosystem Services,” in *Ecosystem-Based Management for the Oceans,* ed. by K. McLeod and H. Leslie (Washington D.C.: Island Press, 2009), 92–111.

CEA can be particularly useful for comparing similar alternatives when the outcomes have been linked to a benefit but either cannot be readily monetized or monetization cannot differentiate among sites with variable qualities (i.e., when the effects of an ecosystem services change on human welfare are too subtle or indirect to be measured by valuation studies). CEA is also useful when management or policy outcomes have been predetermined (e.g., to increase groundwater recharge by 20%), and policymakers and resource managers wish to determine the most efficient (or lowest-cost) means of achieving that goal.

Like CBA, CEA has a long history of use throughout federal agencies and other institutions.[108] The simplicity of using ecological outcome metrics instead of values can be an attractive option for suggesting benefits, yet it presents unique challenges. For results to be most indicative of social benefits derived from ecosystems, practitioners must develop a process for selecting relevant outcomes and metrics and for ensuring that key tradeoffs of decisions are represented. In addition, if multiple benefit metrics will be aggregated into an index, appropriate methods are needed to maintain the relative social importance of metric changes as they are combined. More broadly, policies or projects chosen on the basis of cost-effectiveness may still generate a net loss of social benefits—nothing in CEA guarantees that benefits will exceed costs.

Economic Impact Analysis

Another economic tool that is commonly used to evaluate policy effects is economic impact analysis (EIA). This technique is used to estimate the impacts of a given investment (e.g., building a bridge) in terms of changes in economic activity as measured by jobs, economic output, and other metrics. As applied to ecosystem services, EIA is typically used to suggest how investments in ecosystem restoration will generate or maintain economic activity or to show levels of economic dependencies on a natural resource.

Although economic activity outcomes may be desired by policymakers and stakeholders, they are not, strictly speaking, measures of social welfare and are not acceptable for use in CBA or other approaches that evaluate net benefits. To understand why economic activity is not a social welfare metric, consider that spending on any investment comes at the expense of another type investment. Thus, spending will create jobs and economic activity regardless of the investment, although the number of jobs or the level of economic output will vary by the type of investment. Moreover, actions that reduce net social welfare can increase economic activity, particularly in the short run (e.g., cleanup of an oil spill).[109]

Commonly Used Valuation Techniques

An extensive economic literature addresses theoretical and empirical approaches to valuation of benefits derived from ecosystems.[110] The major primary approaches are summarized below using a common typology and selected examples (Table 3).

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[108] See, e.g., R. Robinson, W. Hansen, and K. Orth, *Evaluation of Environmental Investments Procedures Manual Interim: Cost Effectiveness and Incremental Cost Analyses,* IWR Report 95-R-1, USACE Institute for Water Resources, Ft. Belvoir, VA (1995).

[109] For additional discussion of the role and limitations of economic impact analysis applied to ecosystem services, seeD.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010).

[110] P.A. Champ, K.J. Boyle, and T.C. Brown, *A Primer on Nonmarket Valuation: The Economics of Non-Market Goods and Resources* (New York, NY: Springer, 2013); A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,*3rd ed. (Washington, D.C.: RFF Press, 2014); andD.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010).

Table 3. Primary valuation methods applied to ecosystem services

	Valuation Method	Description	Examples of Ecosystem Services Valued
Market Valuation[a]	Market Analysis and Transactions	Derives value from household's or firm's inverse demand function based on observations of use	Fish Timber Water Other raw goods
	Production Function	Derives value based on the contribution of an ecosystem to the production of marketed goods	Crop production (contributions from pollination, natural pest control) Fish production (contributions from wetlands, seagrass, coral)
Revealed Preferences	Hedonic Price Method	Derives an implicit value for an ecosystem services from market prices of goods	Aesthetics (from air and water quality, natural lands) Health benefits (from air quality)
	Recreation Demand Methods	Derives an implicit value of an on-site activity based on observed travel behavior	Recreation value (contributions from: Water quality and quantity Fish and bird communities Landscape configuration Air quality)
Defensive and Damage Costs Avoided[b]	Damage Costs Avoided	Value is inferred from the direct and indirect expenses incurred as a result of damage to the built environment or to people.	Flood protection (costs of rebuilding homes) Health and safety benefits (treatment costs)
	Averting Behavior / Defensive Expenditures	Value is inferred from costs and expenditures incurred in mitigating or avoiding damages	Health and safety benefits (e.g., cost of an installed air filtration system suggests a minimum willingness-to-pay to avoid discomfort or illness from polluted air)
	Replacement / Restoration Cost	Value is inferred from potential expenditures incurred from replacing or restoring an ecosystem services.	Drinking water quality (treatment costs avoided) Fire management
	Public Pricing	Public investment serves as a surrogate for market transactions (e.g., government money spent on purchasing easements).	Non-use values (species and ecosystem protection) Open space Recreation
Stated Preference	Contingent Valuation (open-ended and discrete choice)	Creates a hypothetical market by asking survey respondents to state their willingness-to-pay or willingness-to-accept payment for an outcome (open-ended), or by asking them whether they would vote for or choose particular actions or policies with given outcomes and costs (discrete choice).	Non-use values (species and ecosystem protection), Recreation Aesthetics
	Choice Modeling / Experiments	Creates a hypothetical market by asking survey respondents to choose among multi-attribute bundles of goods with associated costs and derives value using statistical models.	Non-use values (species and ecosystem protection), Recreation Aesthetics

Valuation methods fall into two main categories: primary valuation and benefit transfer. Primary methods include market value approaches, non-market revealed preference approaches, and non-market stated preference approaches, including contingent valuation and choice experiments. Defensive and avoided damage costs are discussed as a separate category, even though they include revealed preference approaches, because they are more weakly grounded in economic welfare theory and include techniques that do not necessarily derive values solely from individual preferences, as in the case of public pricing that was discussed above. Benefit transfer approximates economic values for one or more policy sites (where proposed changes will occur) using data or models generated and published for other, similar locations with similar goods (referred to as study sites).

Primary Methods

The type of primary valuation that is most applicable and appropriate depends on which component of TEV is being valued (Figure 18).

- **Market value approaches** measure use values by analyzing market behavior and outcomes (e.g., prices and quantities) for ecosystem services goods and services that are sold directly or for goods that rely on ecosystem services as direct inputs.[111] For example, fish sold in markets depends on multiple ecosystem processes. Thus, the benefits of an ecosystem restoration that increased fish abundance could be measured in terms of consumer surplus and producer surplus realized through fish sold in markets.[112]
- **Revealed preference methods** are used to value non-market use goods and services on the basis of observed behavior, including some types of market transactions.[113] A common approach is the use of housing markets to demonstrate WTP for different levels of nearby natural amenities (e.g., higher versus lower air quality). The WTP for changes in natural systems can be revealed by comparing the market value of homes with different levels of associated natural amenities, holding all else constant, a method known as hedonic.[114] Parallel approaches can be used to estimate values based on observed recreational behavior.
- **Defensive and avoided *damage costs* methods** can, in some cases, provide a practical approach to approximating values. These methods calculate proxies for value based on consumer or government defensive behavior or by estimating the cost of restoring or replacing ecosystem services that are damaged or lost. For example, avoided loss in the value of agricultural harvest has been used to measure the benefits of preserving pollination functions of natural areas.[115] Although useful for demonstrating the existence of an ecosystem service, this estimate is not considered an accurate measure of economic value because the calculations do not generally consider how profits might be maximized through alternative adaptations. For example, if pollinators decline, farmers might find it more profitable to switch to alternative crops that do not require insect pollination rather than to invest in natural land preservation.
- **Stated preference methods** use sophisticated survey techniques to elicit values for goods and services and individual's willingness to make tradeoffs among these services.They are the most flexible approaches to valuation and are the only ones capable of measuring non-use values, in addition to use values.[116] Common stated preference approaches include contingent valuation and choice experiments.

Further explanation of these methods is provided in Table 3. General summaries of different types of valuation methods and discussions of ecosystem services applications are found in many works.[117]

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[111] A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd ed. (Washington, D.C.: RFF Press, 2014); A.E. Boardman, D. Greenberg, A.R. Vining, and D.L. Weimer, *Cost-Benefit Analysis: Concepts and Practice,* 3rd edition (Upper Saddle River, NJ: Prentice Hall, 2006), http://www.alibris.com/Cost-Benefit-Analysis-Concepts-and-Practice-Anthony-Boardman/book/1356068; N. Hanley and E. Barbier, *Pricing Nature: Cost-Benefit Analysis and Environmental Policy* (Cheltenham, UK: Edward Elgar Publishing, 2009); and D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010).

[112] E.B. Barbier, "Ecosystems as Natural Assets," *Foundations and Trends in Microeconomics* 4 (2008):611–681.

[113] N.E. Bockstael and K.E. McConnell. *Environmental Resource Valuation with Revealed Preferences: A Theoretical Guide to Empirical Models*. (Dordrecht, The Netherlands: Springer, 2010).

[114] C. Won Kim, T.T. Phipps, and L. Anselin, "Measuring the Benefits of Air Quality Improvement: A Spatial Hedonic Approach," *Journal of Environmental Economics and Management* 45 (2003):24–39 andA.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd ed. (Washington, D.C.: RFF Press, 2014).

[115] T.H. Ricketts, J. Regetz, I. Steffan-Dewenter, S.A. Cunningham, C. Kremen, A. Bogdanski, B. Gemmill-Herren, S.S. Greenleaf, A.M. Klein, M.M. Mayfield, L.A. Morandin, A. Ochieng, and B.F. Viana, "Landscape Effects on Crop Pollination Services: Are There General Patterns?" Ecology Letters 11 (2008):499–515.

[116] I.J. Bateman, R. T. Carson, B. Day, M. Hanemann, N. Hanley, T. Hett, M. Jones-Lee, G. Loomes, S. Mourato, E. Özdemirog?lu, D. W. Pearce, R. Sugden, and J. Swanson, *Economic Valuation with Stated Preference Techniques: A Manual* (Cheltenham, UK: Edward Elgar Publishing, 2002).

[117] See, e.g., R.J. Johnston, T.A. Grigalunas, J.J. Opaluch, M. Mazzotta, and J. Diamantedes, "Valuing Estuarine Resource Services Using Economic and Ecological Models: The Peconic Estuary System Study," *Coastal Management*30 (2002):47–65; P.A. Champ, K.J. Boyle, and T.C. Brown, *A Primer on Nonmarket Valuation: The Economics of Non-Market Goods and Resources* (New York, NY: Springer, 2013); A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd

Both revealed and stated preference methods have been used extensively since the mid-1980s to estimate values associated with natural resources and ecosystem services. They have been extensively tested and validated and are grounded in large scientific literature.[118] However, not all techniques presented here are viewed as equally reliable or appropriate for all decision contexts. In particular, the conditions under which defensive and damage costs provide accurate measures of ecosystem service value are restricted to cases in which substitutability or adaptation is low.[119] When defensive and damage costs are appropriate, they are often underestimates of value.

Stated preference methods can be controversial due to their reliance on survey data and hypothetical behavior rather than on observed behavior.[120] Because of these concerns, some federal agencies allow the use of stated preference techniques or data only with special permission. Many economists, however, support these methods as a means to quantify values that would otherwise be assumed to be zero or infinite and have countered some criticisms to suggest that CV methods are worth continued use and investigation.[121] These methods are considered a valid approach to valuation by many federal agencies and have a long history of use in some agencies such as NOAA National Marine Fisheries Service.[122] The well-known NOAA Blue Ribbon Panel also concluded that stated preference methods—when appropriately used—provide information of comparable reliability to many other methods used to support public decisions.[123] Balancing these two perspectives, a recent evaluation concludes "that the last 20 years of research have shown that some carefully constructed number based on stated preference analysis is now likely to be more useful than no number [for] cost-benefit analysis."[124] Furthermore, it is almost universally acknowledged that in the absence of these methods, non-use values for ecosystem services cannot be monetized.

Benefit Transfer

Benefit transfer is the use of information from primary studies at one or more study sites to estimate welfare estimates such as WTP or related information at unstudied policy sites. Although the use of primary research to estimate values is generally preferred, the realities of the policy process often dictate that benefit transfer is the only feasible option for estimating ecosystem services values. Benefit transfer is most

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ed. (Washington, D.C.: RFF Press, 2014); N. Hanley and E. Barbier, *Pricing Nature: Cost-Benefit Analysis and Environmental Policy* (Cheltenham, UK: Edward Elgar Publishing, 2009); U.S. Environmental Protection Agency, *Valuing the Protection of Ecological Systems and Services: A Report of the EPA Science Advisory Board,* U.S. EPA Science Advisory Board Committee on Valuing the Protection of Ecological Systems and Services, Washington, D.C. (2009); N.E. Bockstael and K.E. McConnell, *Environmental Resource Valuation with Revealed Preferences: A Theoretical Guide to Empirical Models* (Dordrecht, The Netherlands: Springer, 2010); D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010); and I.J. Bateman, R. Brouwer, S. Ferrini, M. Schaafsma, D.N. Barton, A. Dubgaard, B. Hasler, S. Hime, I. Liekens, and S. Navrud, "Making Benefit Transfers Work: Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-Market Benefits of Water Quality Improvements Across Europe," *Environmental and Resource Economics* 50 (2011):365–387.

[118] P.A. Champ, K.J. Boyle, and T.C. Brown, *A Primer on Nonmarket Valuation: The Economics of Non-Market Goods and Resources* (New York, NY: Springer, 2013);A.M. Freeman, J.A. Herriges, and C.L. Kling, *The Measurement of Environmental and Resource Values: Theory and Methods,* 3rd ed. (Washington, D.C.: RFF Press, 2014);N. Hanley and E. Barbier, *Pricing Nature: Cost-Benefit Analysis and Environmental Policy* (Cheltenham, UK: Edward Elgar Publishing, 2009);N.E. Bockstael and K.E. McConnell, *Environmental Resource Valuation with Revealed Preferences: A Theoretical Guide to Empirical Models* (Dordrecht, The Netherlands: Springer, 2010); and D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010).

[119] P.A. Champ, K.J. Boyle, and T.C. Brown, *A Primer on Nonmarket Valuation: The Economics of Non-Market Goods and Resources* (New York, NY: Springer, 2013) and U.S. Environmental Protection Agency, *Valuing the Protection of Ecological Systems and Services: A Report of the EPA Science Advisory Board,* U.S. EPA Science Advisory Board Committee on Valuing the Protection of Ecological Systems and Services, Washington, D.C. (2009); and J. Hausman, "Contingent Valuation: From Dubious to Hopeless," *Journal of Economic Perspectives* 26 (2012):43–56.

[120] K. Arrow, R. Solow, P.R. Portney, E.E. Leamer, R. Radner, and H. Schuman, *Report of the NOAA Panel on Contingent Valuation* (1993); W.H. Desvousges, F.R. Johnson, and H.S. Banzhaf, *Environmental Policy Analysis with Limited Information: Principles and Applications of the Transfer Method* (Cheltenham, UK: Edward Elgar Publishing, 1998).

[121] R.T. Carson, "Contingent Valuation: A Practical Alternative When Prices Aren't Available," *Journal of Economic Perspectives* 26 (2012):27–42 and T.C. Haab, M.G. Interis, D.R. Petrolia, and J.C. Whitehead, "From Hopeless to Curious? Thoughts on Hausman's 'Dubious to Hopeless' Critique of Contingent Valuation," *Applied Economic Perspectives and Policy* 35 (2013):593–612.

[122] R.L. Hicks, *Stated Preference Methods for Environmental Management: Recreational Summer Flounder Angling in the Northeastern United States,* Department of Coastal and Ocean Policy, Virginia Institute of Marine Science, College of William and Mary, Gloucester Point, VA (2002) and K. Wallmo and D.K. Lew "Valuing Improvements to Threatened and Endangered Marine Species: An Application of Stated Preference Choice Experiments," *Journal of Environmental Management* 92 (2011):1793–1801.

[123] K. Arrow, R. Solow, P.R. Portney, E.E. Leamer, R. Radner, and H. Schuman, *Report of the NOAA Panel on Contingent Valuation* (1993).

[124] C.L. Kling, D.J. Phaneuf, and J. Zhao, "From Exxon to BP: Has Some Number Become Better Than No Number?" *Journal of Economic Perspectives* 26 (2012):3–26.

often used when time, funding, or other constraints prevent primary research and is the most commonly applied valuation technique.[125] When used properly, benefit transfer is a useful tool for policy analysis and can be conducted in a manner that minimizes sources of error. But even in the best of circumstances, **benefit transfer can be limited by the availability and quality of primary studies on the ecosystem services of interest.**[126]

Benefit transfer involves trading off empirical accuracy for speed and pragmatism (Iovanna and Griffiths 2006). Although benefit transfer enables many ecosystem services benefits to be valued for many sites, the value estimates generated by this method are subject to errors not present in primary value estimates. The size of these errors depends on many factors, including the type of transfer methods applied and the similarity of study and policy sites.[127] A significant literature on benefit transfer methods describes methods, limitations, and adaptations.[128]

Benefit transfer methods are generally grouped into unit value transfers and benefit function transfers. Unit value transfers include the transfer of a single value, for example an average value across multiple studies (e.g., average consumer surplus per bird-watching trip). Benefit function transfers, in contrast, calculate values using an estimated function from empirical research that allows multiple factors (e.g., socio-demographic variables) to be used to adjust the study site value to the policy site. Those applying benefit transfers (and particularly unit value transfers) should be aware of the significant transfer errors that can result, particularly when using unit value transfers across dissimilar biophysical, social, and economic contexts.[129] The literature generally finds that more sophisticated benefit function transfers outperform unit value transfers, although unit value transfers can perform satisfactorily if the study and policy contexts (e.g., social factors, geographic and time scales, degree of resource scarcity) are very similar.[130]

Assessing the Value of a Stream of Future Benefits

It is standard practice to apply a *discount rate* to understand the *present value* of a future stream of ecosystem goods and service benefits. This practice enables current and future values to be compared and aggregated in consistent terms. The discount rate is a function of multiple factors and can be calculated in multiple ways, as described elsewhere.[131] As a simplification of this complex topic, one can think of the discount rate as reflecting both the change in the value of money (or a valued commodity) over time and the willingness of current generations to forgo current consumption in order to obtain future consumption.

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[125] R.J. Johnston and R.S. Rosenberger, "Methods, Trends and Controversies in Contemporary Benefit Transfer," *Journal of Economic Surveys* 24 (2010):479–510.

[126] See the discussion in J.B. Loomis and R.S. Rosenberger, "Reducing Barriers in Future Benefit Transfers: Needed Improvements in Primary Study Design and Reporting," *Ecological Economics* 60 (2006):343–350.

[127] R.S. Rosenberger and T.D. Stanley, "Measurement, Generalization, and Publication: Sources of Error in Benefit Transfers and Their Management," *Ecological Economics* 60 (2006):372–378 and R.J. Johnston and R.S. Rosenberger, "Methods, Trends and Controversies in Contemporary Benefit Transfer," *Journal of Economic Surveys* 24 (2010):479–510.

[128] R. Ready and S. Navrud, "Benefit Transfer: The Quick, the Dirty, and the Ugly?" *Choices: The Magazine of Food, Farm and Resource Issues* 20 (2005):195–199; K.J. Boyle, N.V. Kuminoff, C.F. Parmeter, and J.C. Pope, "The Benefit-Transfer Challenges," *Annual Review of Resource Economics* 2 (2010):161–182; R.J. Johnston and R.S. Rosenberger, "Methods, Trends and Controversies in Contemporary Benefit Transfer," *Journal of Economic Surveys* 24 (2010):479–510. For a discussion of benefit transfer in the context of ecosystem services valuation, see I.J. Bateman, R. Brouwer, S. Ferrini, M. Schaafsma, D.N. Barton, A. Dubgaard, B. Hasler, S. Hime, I. Liekens, and S. Navrud, "Making Benefit Transfers Work: Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-Market Benefits of Water Quality Improvements Across Europe," *Environmental and Resource Economics* 50 (2011):365–387. For a discussion of benefit transfer within the context of federal agency decision making related to ecosystem services, see Appendix C of K.J. Bagstad, D. Semmens, R. Winthrop, D. Jaworski, and J. Larson, *Ecosystem Services Valuation to Support Decisionmaking on Public Lands: A Case Study of the San Pedro River Watershed, Arizona,* Scientific Investigations Report 2012-5251, U.S. Geological Survey, Reston, VA (2012).

[129] R.S. Rosenberger and T.D. Stanley, "Measurement, Generalization, and Publication: Sources of Error in Benefit Transfers and Their Management," *Ecological Economics* 60 (2006):372–378.

[130] R.J. Johnston and R.S. Rosenberger, "Methods, Trends and Controversies in Contemporary Benefit Transfer," *Journal of Economic Surveys* 24 (2010):479–510 and I.J. Bateman, R. Brouwer, S. Ferrini, M. Schaafsma, D.N. Barton, A. Dubgaard, B. Hasler, S. Hime, I. Liekens, and S. Navrud, "Making Benefit Transfers Work: Deriving and Testing Principles for Value Transfers for Similar and Dissimilar Sites Using a Case Study of the Non-Market Benefits of Water Quality Improvements Across Europe," *Environmental and Resource Economics* 50 (2011):365–387.

[131] M.L. Cropper, "How Should Benefits and Costs Be Discounted in an Intergenerational Context?" *RFF Resources* 183 (2013); Kenneth J. Arrow, Maureen L. Cropper, Christian Gollier, Ben Groom, Geoffrey M. Heal, Richard G. Newell, William D. Nordhaus, Robert S. Pindyck, William A. Pizer, Paul R. Portney, Thomas Sterner, Richard S. J. Tol, and Martin L. Weitzman, "How Should Benefits and Costs Be Discounted in an Intergenerational Context? The Views of an Expert Panel," RFF Discussion Paper, Department of Economics, University of Sussex, Washington, D.C. (2013).

The choice of discount rate can have a large effect on the outcome of cost-benefit analysis, particularly when evaluating policies with benefits that do not accrue for many years, or that accrue for a very long period of time. The higher the discount rate, the smaller future benefits appear in present value. This issue has become particularly relevant in discussion of appropriate responses to climate change, since the major benefits accrue so far in the future that the present value of those benefits does not compare favorably to the present value of near-term costs, when a typical discount rate is used. The climate change case brings into focus how the choice of the discount rate implies a judgment about the relative importance of benefits received by future generations.

Accounting for intergenerational effects in policy analysis can be difficult and controversial because of competing perspectives on how to set the discount rate. For example, some analysts argue that the discount rate should be low (even zero or negative) in order to promote *intergenerational equity*. Others argue that using a positive discount rate in policy analysis enables capital investments today that improve the welfare of future generations to a much greater extent than if those investments had not been made.[132] The topic remains controversial and as a result analysts often rely on guidance provided by the OMB for regulatory decision making.[133] However, that guidance leaves open the possibility that alternative discount rates may be justified under some circumstances. A recent workshop to evaluate the issue found consensus among economists that it may be appropriate to use a declining discount rate for projects with long-term effects, although they did not agree on how to set the discount rate.[134]

Caveats for Use of Valuation

Economic valuation, like any tool, has both appropriate uses and limitations. Valuation is most often criticized for what it omits. However, valuation and CBA are not intended to be the only inputs into decision making and,therefore, they should be judged in terms of whether they can be useful analytic components, even if their results are incomplete.

Among the potential limitations of economic valuation is its anthropocentric (also called utilitarian) perspective, which is often perceived to exclude the *intrinsic value of nature*.[135] More broadly, if an ecosystem service change does not benefit people either directly or indirectly, it has no value within an economic framework.This perspective has led to unease among some, who argue that there is an ethical dimension to ecosystem preservation that is not captured by economic value estimates and that intrinsic values are distinct from other values in that they cannot be traded for other goods and services, as is suggested by a cost-benefit framework. Although an economic framework cannot include all human concerns and, in particular, cannot address the question of whether intrinsic value *should* be traded off for other uses or values, the TEV framework incorporates such values to the extent that it represents human concerns for protecting nature, independent of human use (i.e., non-use value). That is, because agencies cannot ask nature what it wants but, instead, must ask people to interpret this idea, intrinsic values become existence, bequest, or altruistic values in this framework.

From a practical perspective,many see a lack of diversity in the types of ecosystem services that have been valued. **A lack of data or required ecological or other scientific models is often the source of difficulty in estimating defensible economic values.**[136]Hence, even if one supports economic valuation in principle, empirical applications to ecosystem services can be challenging.

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132 C.R. Sunstein and A. Rowell, "On Discounting Regulatory Benefits: Risk, Money, and Intergenerational Equity," *The University of Chicago Law Review* 74 (2007):171–208.

133 U.S. Office of Management and Budget, Circular A-4. Subject: Regulatory Analysis (2003). Retrieved from http://www.whitehouse.gov/omb/memoranda_m03-21.

134 Kenneth J. Arrow, Maureen L. Cropper, Christian Gollier, Ben Groom, Geoffrey M. Heal, Richard G. Newell, William D. Nordhaus, Robert S. Pindyck, William A. Pizer, Paul R. Portney, Thomas Sterner, Richard S. J. Tol, and Martin L. Weitzman, "How Should Benefits and Costs Be Discounted in an Intergenerational Context? The Views of an Expert Panel." RFF Discussion Paper, Department of Economics, University of Sussex, Washington, D.C. (2013).

135 J.B. Callicott, *In Defense of the Land Ethic: Essays in Environmental Philosophy* (Albany, NY: SUNY Press, 1989).

136 L. Wainger and M. Mazzotta, "Realizing the Potential of Ecosystem Services: A Framework for Relating Ecological Changes to Economic Benefits," *Environmental Management* 48 (2011):710–733.

Another concern is that valuation, when used in CBA, can implicitly promote policies that exacerbate social inequities, because when benefits are aggregated, the values of those with higher ability to pay can exceed the values of the poor.[137] For example, consider that a choice to build flood protection based on the total value of property protected tends to distribute more resources to wealthy communities than to poor communities. Although ecosystem services value estimates can be extended to quantify effects on different groups or to address equity concerns, most often they do not.

Other concerns can arise from the misuse of valuation tools. Many of the transfers applied in past ecosystem services literature (e.g., particularly in non-economics journals) and in ecosystem services valuation tools have applied methods that would be expected to generate large errors or invalid estimates, particularly due to incorrect aggregation of marginal values, failure to account for spatial connections between ecosystems and their human beneficiaries and their change over time, and other generalization errors.[138] For example, benefit transfer studies that purport to value ecosystem services on a biome or worldwide scale are widely considered to be invalid by economists and of little practical value for decision making.[139] Similarly, ecosystem services valuation tools that simply multiply a unit value by the area of an ecosystem do not reflect changes in ecosystem services value that will occur with changes in the number of users or beneficiaries, nor do they reflect how values change as resources become scarcer.[140] Finally, benefit transfer tools that include values for all ecosystem services may aggregate benefits that should not be aggregated because they compete partially or wholly with one another, meaning that they cannot be simultaneously provided on the same parcel of land (e.g., timber and habitat-related benefits). When tools sum competing services, they double count benefits.

Challenges arising from misuse of tools can be addressed through careful framing of the analysis, and they speak to the need for teaming trained economists with other natural and social scientists throughout all phases of ecosystem services valuation.Many of the shortcomings in prior ecosystem services valuation efforts have been due to the minimal participation of economists in the early stages of the research, when key questions are established, conceptual models are framed, and research methods are determined.

Conclusions

Given its grounding in a long history of nonmarket valuation, ecosystem services valuation should not be viewed as a "new" type of analysis or valuation, but merely as an evolution and reframing of long-established theory and techniques. Ecosystem services valuation provides a systematic means to compare a broad range of social welfare effects using a common (monetary) unit. Furthermore, it applies an internally consistent framework that promotes equivalence among those units. In other words, dollars measure a comparable gain or loss, even if they measure different ecosystem services benefits delivered to different stakeholders. Without this consistent framework, measured units (e.g., dollars) can take on different meanings and fail to represent people's willingness to trade off goods and services.

Ecosystem services valuation provides information unavailable through other approaches, and is the only means to compare the net social benefits (benefits – costs) associated with policies and actions that affect ecosystem services. At the same time, it is a challenge to design and conduct valuation studies that avoid the common pitfalls of ecosystem services valuation. **Consulting an economist with expertise in nonmarket valuation can not only save time and financial resources for the practitioner, but can also ensure that appropriate valuation methods are used and that the resulting data are defensible.** When the use of

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137 M.D. Adler and E.A. Posner, "Rethinking Cost-Benefit Analysis," *Yale Law Journal* 109 (1999):165–247 and A.E. Boardman, D. Greenberg, A.R. Vining, and D.L. Weimer, *Cost-Benefit Analysis: Concepts and Practice,* 3rd edition (Upper Saddle River, NJ: Prentice Hall, 2006), http://www.alibris.com/Cost-Benefit-Analysis-Concepts-and-Practice-Anthony-Boardman/book/1356068.

138 N.E. Bockstael, A.M. Freeman, R.J. Kopp, P.R. Portney, and V.K. Smith, "On Measuring Economic Values for Nature,"*Environmental Science & Technology* 34 (2000):1384–1389.

139 See M. Toman, "Special Section: Forum on Valuation of Ecosystem Services: Why Not to Calculate the Value of the World's Ecosystem Services and Natural Capital," *Ecological Economics* 25 (1998):57–60; N.E. Bockstael, A.M. Freeman, R.J. Kopp, P.R. Portney, and V.K. Smith, "On Measuring Economic Values for Nature," *Environmental Science and Technology* 34 (2000):1384–1389; and D.S. Holland, J.N. Sanchirico, R.J. Johnston, and D. Joglekar, *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments* (Washington D.C.: RFF Press, 2010).

140 M.L. Plummer, "Assessing Benefit Transfer for the Valuation of Ecosystem Services," *Frontiers in Ecology and the Environment* 7 (2009):38–45.

prepackaged (e.g. software) tools is considered, users must understand their limits and verify that the underlying methods conform to economic theory.Although ecosystem services valuation may not represent all philosophical perspectives, well-designed economic valuation studies can support decision making by creating new understanding of the degree to which people benefit from natural systems and by facilitating clear and consistent communication of those benefits.

Recommended Reading

Arrow, K.J., M.L. Cropper, C. Gollier, B. Groom, G.M. Heal, R.G. Newell, W.D. Nordhaus, R.S. Pindyck, W.A. Pizer, P.R. Portney, T. Sterner, R.S.J. Tol, and M.L. Weitzman. 2013. "How Should Benefits and Costs Be Discounted in an Intergenerational Context? The Views of an Expert Panel." RFF DP 12-53, Washington, D.C.: Resources for the Future. http://www.rff.org/RFF/Documents/RFF-DP-12-53.pdf.
This paper provides a good reference for the handling of discounting in the case of ecosystem services.

Bateman, I.J., R.T. Carson, B. Day, M. Hanemann, N. Hanley, T. Hett, M. Jones-Lee, G. Loomes, S. Mourato, E. Özdemirog?lu, D.W. Pearce, R. Sugden, and J. Swanson. 2002. *Economic Valuation with Stated Preference Techniques: A Manual*. Cheltenham, UK: Edward Elgar Publishing.
This book provides a detailed explanation of the use of stated preference techniques for economic valuation, including the application of these techniques for non-market goods and services such as water or air quality or cultural assets.

Boyle, K.J., N.V. Kuminoff, C.F. Parmeter, and J.C. Pope. 2010. "The Benefit-Transfer Challenges." *Annual Review of Resource Economics* 2:161–182. http://www.annualreviews.org/doi/full/10.1146/annurev.resource.012809.103933.
This paper presents a review of benefit transfer literature and a conceptual framework to aid development of unified guidelines for the application of benefit transfer techniques in federal policies.

Champ, P.A., K.J. Boyle, and T.C. Brown. 2003. *A Primer on Nonmarket Valuation: The Economics of Non-Market Goods and Resources*. New York: Springer.
This book provides clear descriptions of the most commonly used nonmarket valuation techniques and their implementation.

Freeman, A.M., J.A. Herriges, and C.L. Kling. 2014. *The Measurement of Environmental and Resource Values: Theory and Methods*, Third edition. Washington, D.C.: RFF Press.
This book provides comprehensive coverage of the theory and methods involved in estimating environmental benefits.

Hanley, N., and E. Barbier. 2009. *Pricing Nature: Cost-Benefit Analysis and Environmental Policy*. Cheltenham, UK: Edward Elgar Publishing.
This book presents a cost-benefit analysis as an economic tool in environmental policy, highlighting special issues posed by environmental management such as valuing ecosystem services.

Holland, D.S., J.N. Sanchirico, R.J. Johnston, and D. Joglekar. 2010. *Economic Analysis for Ecosystem-Based Management: Applications to Marine and Coastal Environments*. Washington D.C.: RFF Press.
This book discusses the ways that tools of economic analysis inform ecosystem-based management, including applications of ecosystem service valuation.

Johnston, R.J., and R.S. Rosenberger. 2010. "Methods, Trends and Controversies in Contemporary Benefit Transfer." *Journal of Economic Surveys* 24:479–510. http://onlinelibrary.wiley.com/doi/10.1111/j.1467-6419.2009.00592.x/full.
This paper synthesizes the literature on benefits-transfer, highlighting methods, trends, and controversies in research and identifying challenges for practitioners.

Kling, C.L., D.J. Phaneuf, and J. Zhao. 2012. "From Exxon to BP: Has Some Number Become Better Than No Number?" *The Journal of Economic Perspectives* 26:3–26.
This article discusses the controversies surrounding the use of stated preference methods as well as recent advances in stated preference techniques.

McConnell, K.E., and N.E. Bockstael. 2005. "Valuing the Environment as a Factor of Production." In *Handbook of Environmental Economics*, edited by K. G. Mler and J. R. Vincent, 621–669. Elsevier. http://www.sciencedirect.com/science/article/pii/S1574009905020140.
This chapter discusses measuring the economic costs and benefits of the environmental changes that influence production.

National Research Council. 2005. V*aluing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington D.C.: National Academies Press. http://www.nap.edu/catalog.php?record_id=11139.
This report identifies methods for valuing ecosystem services (including non-use values) in the hope of increasing their use in environmental decision-making. It includes case studies and a discussion of uncertainty in valuation.

Ninan, K. N., ed. 2014. *Valuing Ecosystem Services: Methodological Issues and Case Studies*. Northampton, Massachusetts: Edward Elgar Publishing, Inc.
This book provides examples of different applications of valuation methods for use and non-use values and discusses methods such as benefit-transfer.

U.S. Environmental Protection Agency. 2010. "Guidelines for Preparing Economic Analyses." EPA 240-R-10-001, U.S. EPA Office of the Administrator, Washington, D.C. http://www.sra.org/sites/default/files/u32/EPA_Guidelines%20_2010.pdf.
This report is one example of agency guidance given for economic analyses.

U.S. Environmental Protection Agency, Science Advisory Board. 2009. "Valuing the Protection of Ecological Systems and Services: A Report of the EPA Science Advisory Board." Science Advisory Board Committee on Valuing the Protection of Ecological Systems and Services. http://yosemite.epa.gov/sab%5CSABPRODUCT.NSF/F3DB1F5C6EF90EE1852575C500589157/$File/EPA-SAB-09-012-unsigned.pdf.
This report summarizes ecological valuation practices and methodologies, identifies research needs, and recommends next steps for improving the valuation of ecosystem services.

U.S. Office of Management and Budget. 2003. “Regulatory Analysis.” M-03-21, OMB Circular No. A-4. http://www.whitehouse.gov/omb/memoranda_m03-21.
This guidebook details methods commonly applied by agencies in conducting cost-benefit analysis. Much of the guidance is directly applicable to the valuation of ecosystem service changes.

NON-MONETARY METHODS (MULTI-CRITERIA EVALUATION)

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Summary

This section describes concepts and stages of structured decision making, suggests a few simple ways to implement common elements of decision analysis, and notes when assistance from others with specialized training would be useful, or even necessary.

Takeaways:

- Multi-criteria analysis helps managers make environmental management decisions requiring tradeoffs among many desired outcomes of management action.
- Hierarchically organized objectives and clearly defined criteria (especially when using qualitative measures) are the starting points for transparent decision modeling.
- The “facts” modeling side of decision analysis describes the effects of management activities on features of the environment that decision makers and stakeholders care about.

Primary source:

Much of the material in this section is taken from “Multi-criteria Evaluation for Ecosystem Services: A Brief Primer” by Lynn Maguire.[141]

Other resources:

Many texts present multi-criteria analysis at various levels (see Recommended Reading at the end of this section).Training in structured decision making is available for federal agency land managers and staff analysts.[142] An increasing number of federal agency consultants have such training.

Introduction

This section suggests how and where structured methods of decision making, specifically multi-criteria evaluation, might help agencies integrate ecosystem services into their decision processes. It can be used as a

[141] L. Maguire, “Multi-criteria Evaluation for Ecosystem Services: A Brief Primer” (2014), https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

[142] (e.g., http://nctc.fws.gov/courses/programs/decision-analysis/index.html)

comprehensive model addressing all planning stages up to the decision process, or it can provide insight on specific stages, such as problem scoping. It can be carried out in great depth and with full quantification, or it can be used as a conceptual framework in qualitative form. It is appropriately described as a non-monetary approach for comparing stakeholder preferences and analyzing tradeoffs. It can be an alternative to monetary valuation, or it can be used in combination with monetary valuation. Although the focus here is on ecosystem services, the methods are useful generally for decision problems in which tradeoffs must be made among multiple resource management objectives.

This section is based on multi-attribute utility analysis (MAUA).[143]Because much of the material presented here also applies to other varieties of multi-criteria analysis, the section refers to the MAUA approach more generally as multi-criteria decision analysis (MCDA). And because different MCDA methods, different agency planning processes, and different ecosystem services approaches use different terminology for similar concepts and procedures, this guidebook calls attention, where possible, to corresponding terms from different practice domains.

Elements of this guidebook's MCDA framework explicitly link to all decision analysis steps, from initial scoping to the actual decision. The key elements are

1. constructing an objectives hierarchy that identifies the target services (outcomes) of concern to decision makers and stakeholders and specifying empirical indicators for those outcomes,
2. creating a conceptual diagram (or means-end model) that links management actions to their likely impacts on the provision of services as captured in the indicators and forming a matrix showing how management alternatives will affect outcome indicators, and
3. characterizing stakeholder preferences for varying levels of changes in the provision of the target services and stakeholder priorities among target services to support a calculation of overall value for each management alternative.

The MCDA framework can be used as a comprehensive and explicit analysis that covers all stages of the decision process, or it can be used to inform specific stages of the decision process, with other stages carried out intuitively, and often implicitly. The description that follows points out specific MCDA products short of a complete analysis that might inform the decision process.

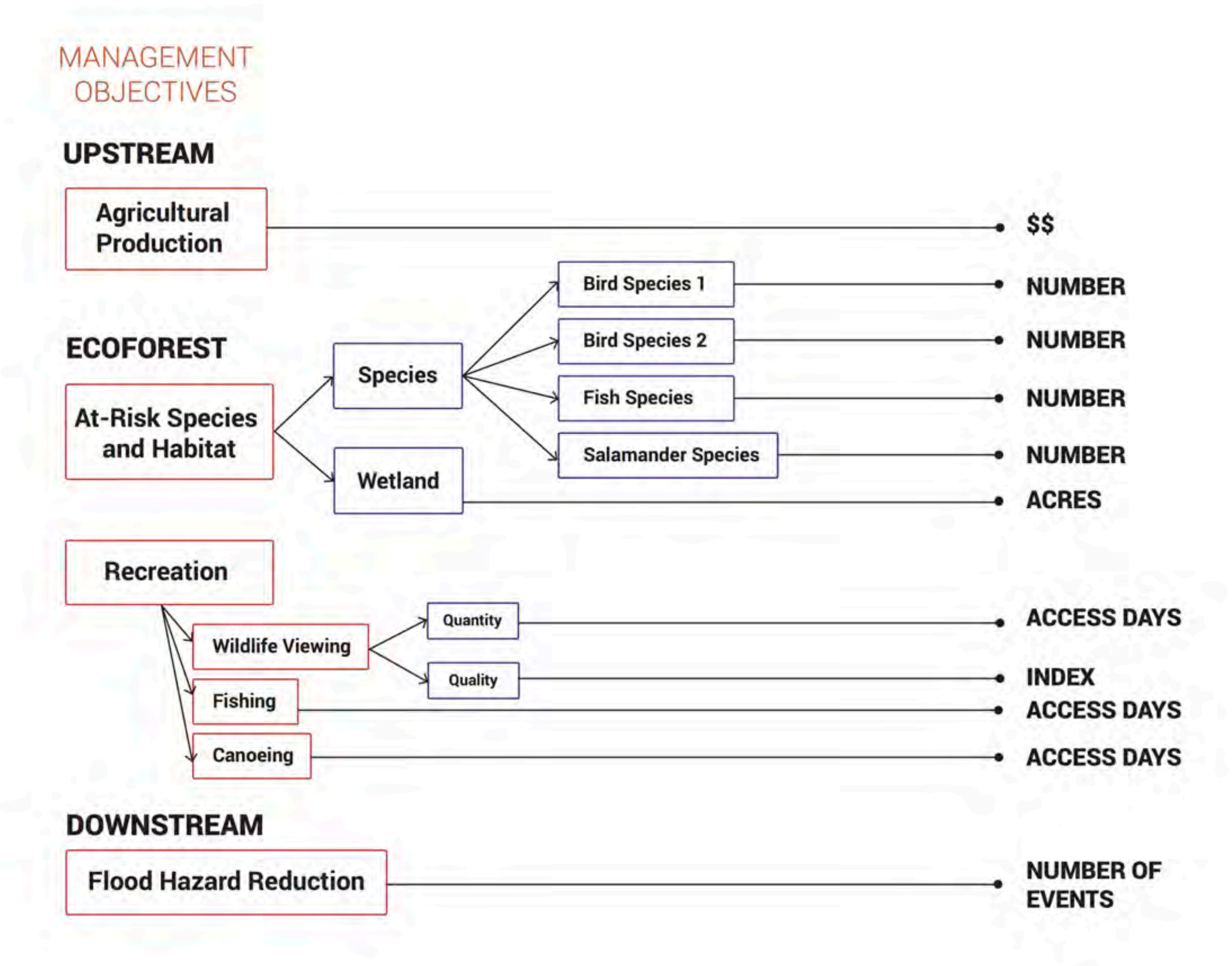
The MCDA approach is illustrated with a hypothetical decision about management of forested wetlands on a national forest. Briefly, a national forest wants to enhance forested wetlands for their own sake and as essential habitat for several at-risk species (two birds, a salamander, and a fish). Land managers are considering two alternatives to status quo management: (1) modifying water releases from upstream reservoirs to increase drought-year flows to benefit forested wetlands on the national forest, and (2) restoring a former wetland on the national forest through damming and dredging. The first alternative would affect upstream agricultural interests; the second would benefit downstream landowners whose lands are subject to flooding, as well as improve forest conditions for recreationists who fish, watch birds, and canoe.

Constructing an Objectives Hierarchy

One product of the scoping stage of a planning process is a list of objectives (e.g., desired conditions or outcomes) that resource managers want to influence by taking action. By organizing these objectives hierarchically (Figure 19), managers can link the major categories of objectives (e.g., agricultural production, at-risk species and their habitats) to more specific aspects of those categories that are important (e.g., the types of at-risk species that might be affected by management—two bird species, a fish, and a salamander). Constructing an effective objectives hierarchy is both a science and an art.

[143] L. Maguire, "Multi-criteria Evaluation for Ecosystem Services: A Brief Primer" (2014), https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

Figure 19. A possible hierarchy of objectives and measures



Consulting the full range of potential users of forest goods and services is essential to making sure the objectives hierarchy is complete. In the ecosystem services literature, these users are often termed *beneficiaries* (or *stakeholders*). Different beneficiary groups (e.g., hunters, birdwatchers) may emphasize different goods and services and may assign different values to receipt of the same service (e.g., an increase in numbers of an at-risk bird species).

One sometimes controversial point in articulating ecosystem services as objectives of land management is deciding which services have value in themselves and which have value only for their contributions to the production of another ecosystem service. It is generally obvious that ecosystem services with use value, such as recreation or agricultural production, have value in themselves (*final ecosystem services*) and therefore belong in the objectives hierarchy.[144] Whether or not non-use values, such as the value associated with the existence of at-risk species or their habitats, belong in the hierarchy depends on whose perspective is used to frame the decision context. Some stakeholders may value at-risk species as ends in themselves; others may

[144] The term *final ecosystem services* is often used to convey a similar concept. See J. Boyd and S. Banzhaf, "What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units." *Ecological Economics* 63 (2007): 616–626. doi: 10.1016/j.ecolecon.2007.01.002.

not. Some stakeholders (beneficiaries) may value habitats for at-risk species as ends in themselves (and thus belonging in the objectives hierarchy); others may regard them as *intermediate ecosystem services*, important only for their contribution to final ecosystem services (and thus belonging not in the objectives hierarchy but in a means-ends network). The hierarchy in Figure 19 includes both use values (e.g., fishing, wildlife viewing) and non-use values (e.g., at-risk species, wetlands) among management objectives.

The objectives hierarchy specifies empirical measures that will be used as indicators for the more general objectives identified by stakeholders. As indicators, these measures are subject to a variety of considerations that apply to all ecological indicators. These considerations include clarity and precision, repeatability, potential bias (e.g., as applied by different evaluators), and use of proxy measures.

If an objectives hierarchy is going to be used as the starting point for a quantitative analysis of tradeoffs among conflicting objectives, it should be reviewed by a specialized consultant to ensure that the objectives structure accords with the assumptions necessary for such an evaluation.

Comprehensive, Not Redundant

Organizing ecosystem services objectives hierarchically can help ensure that the ecosystem goods and services reflect the needs and desires of the full suite of stakeholders. For example, if improved recreation is a general objective, the hierarchy might specify the needs and desires of fishers and canoeists. An individual who fishes and also canoes is not double-counted in this scheme.

A hierarchical organization helps managers detect omissions (e.g., other at-risk species or habitats that should be included) as well as redundancy, which can lead to double-counting in evaluation of alternatives. Figure 19 includes both at-risk species and their habitats, because wetland habitat provides value in its own right—i.e., value beyond its role in supporting at-risk species.

Inclusivity

The objectives hierarchy in Figure 19 includes objectives important to stakeholders beyond the boundaries of federal land area addressed by the planning process. These stakeholders include (1) upstream farmers whose production may be affected by changes in reservoir management, (2) downstream landowners whose risk of flooding may be affected by reservoir management upstream and by wetlands restoration on the national forest land, and (3) members of the public who might never visit the forest, but who derive satisfaction from its existence. Making the objectives hierarchy comprehensive by including objectives important to this wider set of stakeholders facilitates creation of management alternatives that can garner widespread support. Iterations of the hierarchy during agency discussions and stakeholder engagement may provide opportunities to identify or remove services that are not critically affected by the decision and to add other key services.

Values, Not Actions

One principle of constructing objectives hierarchies like the one in Figure 19 is to include only services with value in themselves(i.e., final services) and to exclude services that are valuable only because of their contribution to final services (i.e., the production of intermediate services). The subset of services or objectives that have value in themselves are often monitored or measured with biophysical indicators (such as area of wetland habitat). A common mistake in creating objectives hierarchies is to include management activities that might be taken to achieve underlying objectives (e.g.,restore wetlands). Such activities don't belong in the objectives hierarchy, because they have value only in terms of their effects on the features of the environment that are of fundamental interest (e.g., at-risk species dependent on wetlands). As described below, these management activities belong in means-ends models (conceptual diagrams)—i.e., graphical, mental, or mathematical models used to depict relationships between actions taken and objectives achieved. These means-ends models show how a single action might affect many of the indicators that measure accomplishment of underlying objectives, whereas an objectives hierarchy divides overarching objectives into their components.

Objectives hierarchies, including the suite of indicators used to evaluate success in achieving objectives, can be the starting point for many subsequent analyses of management alternatives, including multivariate statistical analysis, multi-objective optimization, and multivariate simulation modeling, in addition to the form of multi-criteria analysis described here.

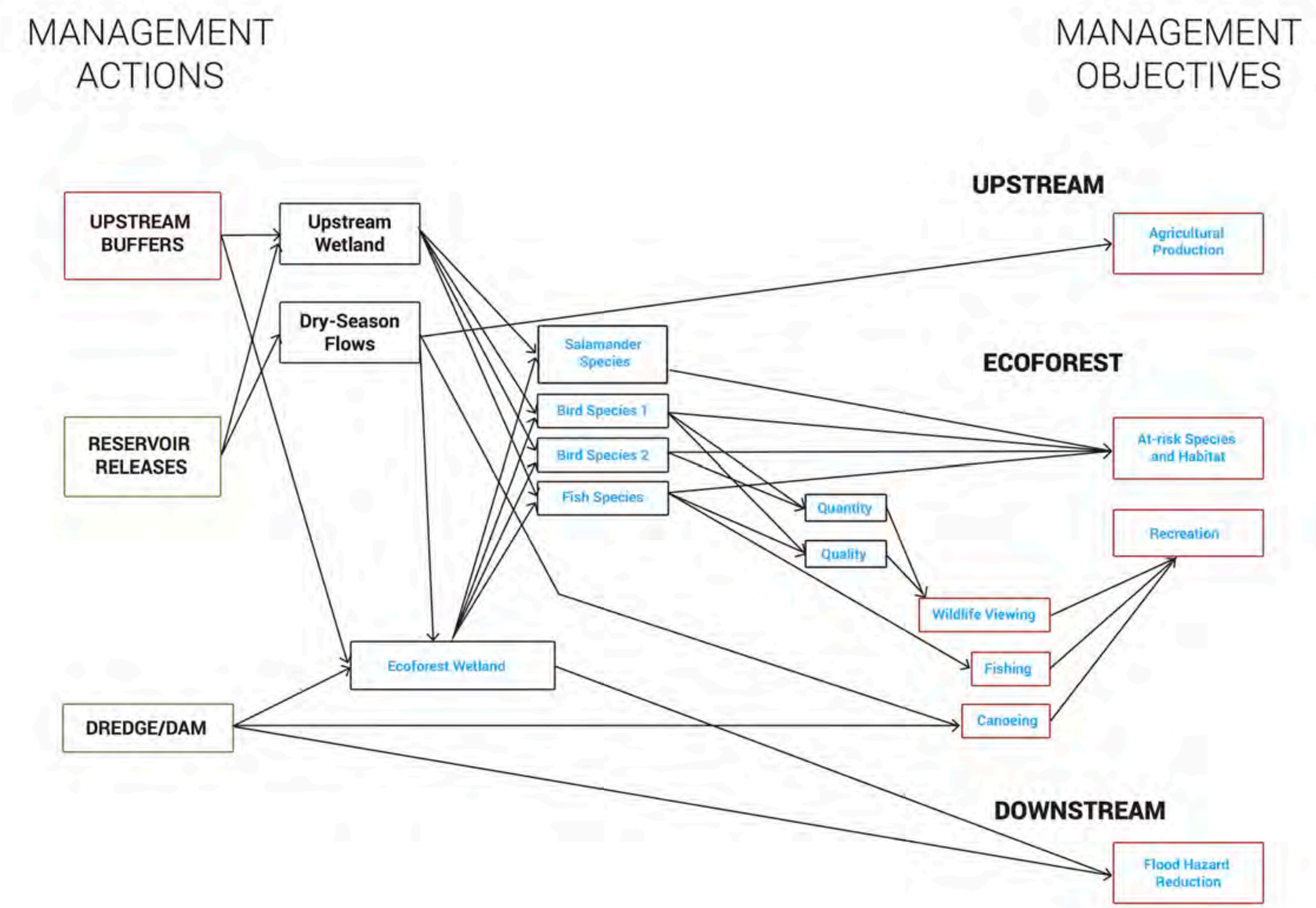
Creating a Means-Ends Model

A means-ends network (also known as a conceptual diagram, path model or influence diagram) illustrates how management activities propagate through the ecosystem to effect changes in the objectives identified in the objectives hierarchy (Figure 19)—that is, it is a model that links the objectives—the ends—to the means for achieving those ends (Figure 20). Unlike the objectives hierarchy, which is a static measurement model, a means-end network is an action-oriented process model.

A means-ends model can become a tangled web of arrows, showing that an action (e.g., modification of the schedule of reservoir releases) can affect many ecosystem elements and processes, both proximate and distant in space and time (e.g., reservoir releases affect dry-season flows upstream of national forest wetlands, changing their extent and status, which, in turn, affects the health and numbers of wetland-dependent species, on which both passive and active recreation may depend). The objectives are represented as endpoints in the means-ends model. As a general principle, intermediate structures and processes should be included in a means-end model only if they lead directly or indirectly to impacts on other target services. The construction of means-ends models (conceptual diagrams) is elaborated in Means-Ends Diagrams as a Tool for Incorporating Ecosystem Services into the Planning Process.

The complexity of the means/end model(s) can be reduced by focusing on key ecosystem service objectives that are most likely to be affected by the decision, that are most likely to be valued by beneficiaries, or both.

Figure 20. A possible means-ends network



Evaluating Performance of Alternatives

The next step in MCDA is to use the means-end model to distinguish the varying levels of services provided by different management alternatives. At this stage of MCDA, evaluation of management alternatives uses the measures (indicators) specified in the objectives hierarchy. These indicators might be purely biophysical indicators (e.g., area of wetland habitat), but it is better if they are benefit relevant indicators that describe how an ecological resource meets a social need (e.g., miles of stream accessible to fishing).

It is common to present the anticipated performance of several management alternatives in a matrix with alternatives as row or column headings and measures of performance (indicators) on the other dimension of the matrix. Table 4 shows such a matrix for the wetland example. It is simplified to include only three alternatives and four measures, including implementation cost over a 10-year period calculated as net present value (NPV).

Table 4. Matrix showing the performance of three alternatives for restoring ecoforest wetlands in terms of three ecological measurement scales.

ALTERNATIVE ACTIONS			
Measures	Status quo	Downstream dam	Upstream release
Number of bird 1 (breeding pairs on forest)	200	220	205
Wildlife viewing at walkway site (qualitative scale)	One iconic sp < 5	One iconis sp < 5, one > 5	Both > 5
Flood Events (annual average)	0.2	0.15	0.2
Implementation cost ($MM NPV)	0.1	1.0	0.8

Note: Each measurement scale represents an objective in Figure 19. The wildlife viewing measure refers to opportunities to view individuals of one or both of two bird species especially associated with the wetlands in question.

A matrix like this can make it easy to see if there is one alternative that is better (or no worse) on all measures than all other alternatives (and can be chosen without further analysis) or one that is worse (or no better) on all measures (and can be discarded without further analysis). There are no such clear winners or clear losers here.

Alternatives matrices are not solely the province of MCDA; they are applied in other decision frameworks, including conjoint analysis and choice experiments for monetary valuation. General guidelines for their development are detailed in Displaying Assessment Results: Alternatives Matrices and Other Tools.

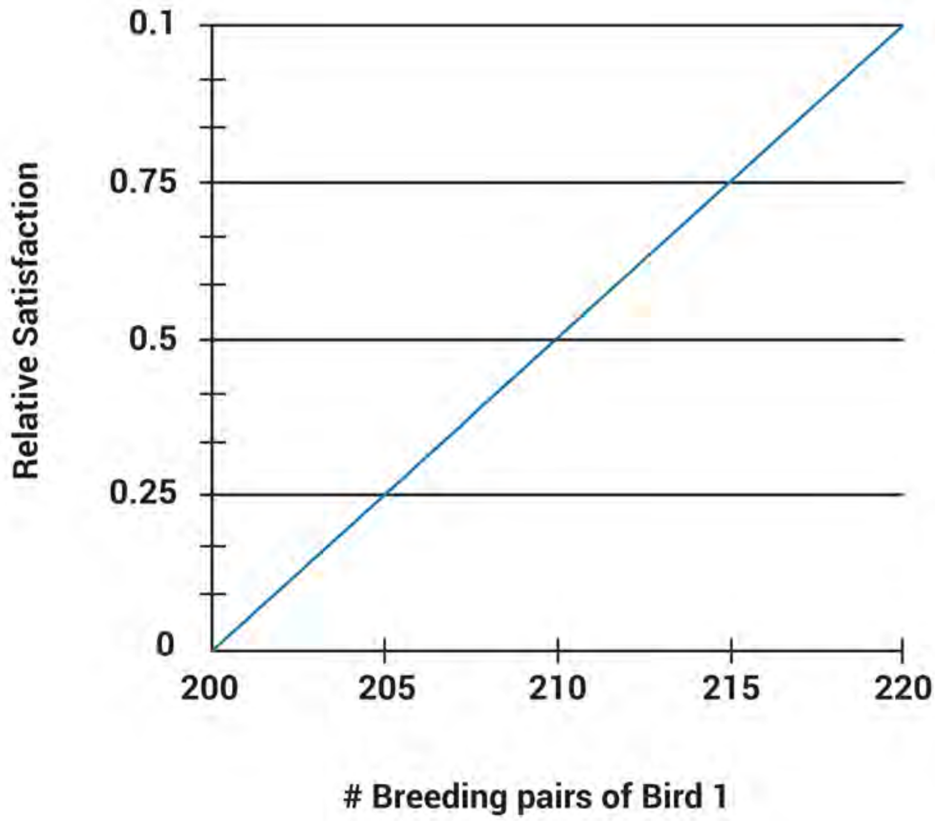
Expressing Relative Satisfaction with Performance on Individual Measures

Sometimes creating the alternatives/measures matrix is the final step of formal analysis, and expressions of relative satisfaction for different levels of performance and any tradeoffs among objectives are made intuitively, and often implicitly, during the decision process. A more formal consideration of relative satisfaction and tradeoffs requires establishment of a relationship between performance on each measure and a unitless scale (usually 0-1, sometimes 0-100) that describes how relative satisfaction changes over the range of performance levels encountered in an analysis of particular alternatives. These relationships, which are often called value functions or utility functions, serve the dual purposes of (1) putting unlike measures on a common scale so that they can be combined and (2) expressing relative satisfaction with different levels of performance for a single measure.

Relative Satisfaction (Value/Utility) Functions

It is common (but not always warranted) to simply assume a linear relationship between relative satisfaction and performance level, as in Figure 21 for number of breeding pairs of bird 1. A linear relationship is more likely to reflect relative satisfaction accurately when the management alternatives change performance relatively little compared to the status quo, as is the case for breeding pairs of bird 1, which varies only 10% from the status quo for any of the new management alternatives.

Figure 21. A linear relationship showing that the increase in relative satisfaction for each additional breeding pair of Bird 1 is the same over the range of 200 to 220 breeding pairs.



When performance levels vary more widely, as they do in this example for costs and flood events (Table 4), assuming a linear relationship may not be adequate. The shape of the relationship between relative satisfaction and performance is tied to the range of performance levels encountered in a particular problem. Therefore, the shape for numbers of breeding pairs ranging from 2 to 2000 may differ from than the shape for numbers of breeding pairs ranging from 200 to 220 or for numbers ranging from 2 to 22. This is one reason among many that a relationship derived for one set of alternatives in one context may not be meaningful for another context.

A common nonlinear relationship is diminishing marginal increases in relative satisfaction as the level of performance increases—i.e., the increment in satisfaction from an additional breeding pair is larger for small numbers of breeding pairs than it is for larger numbers of breeding pairs. The shapes of value or utility functions can be different for different stakeholders. Downstream landowners, who most immediately feel the pain of flood events, might experience a larger boost in relative satisfaction than upstream landowners from decreasing flood event frequency (Figure 22). Utility curves can also express more complicated issues in stakeholder preferences, such as levels of risk-aversion; such nuances are illustrated in Maguire (2014).[145]

Characterizing these relationships can be daunting. There are structured methods for eliciting value and utility functions (see Clemen 2001), and it is best to use the services of specialized consultants to implement them. Information can be collected through face-to-face interaction with decision makers or stakeholder representatives or remotely through surveys or social media.

A stopgap approach is to simply draw shapes on a graph that appear to capture the way relative satisfaction increases or decreases with performance level and read off the relative satisfaction that corresponds to a particular level of performance. It will often be the case that the choice of a particular alternative is not highly sensitive to the exact form of the relationship between satisfaction and performance.

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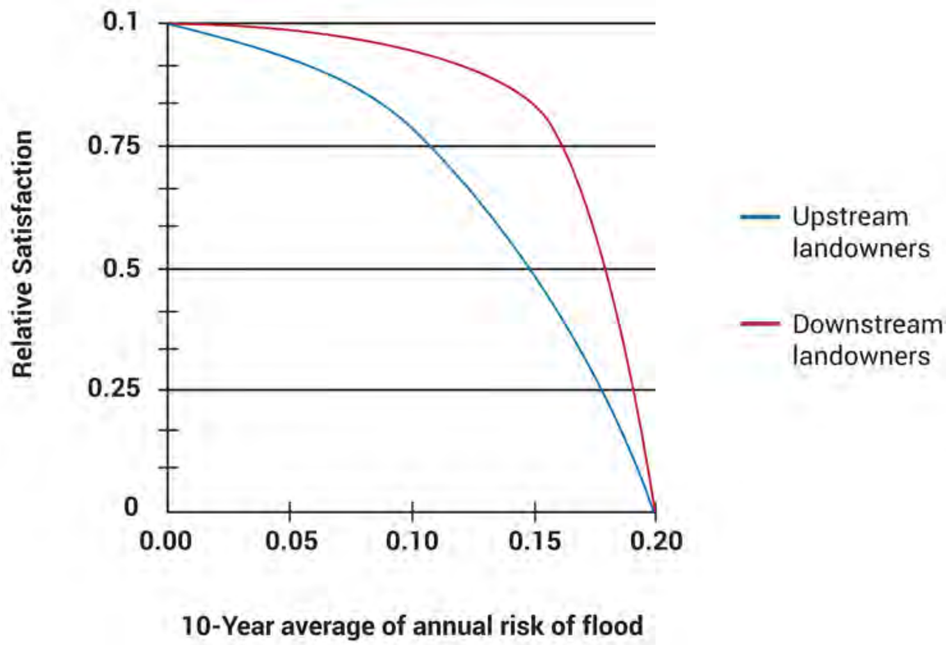
[145] L. Maguire, "Multi-criteria Evaluation for Ecosystem Services: A Brief Primer" (2014), https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

Figure 22. Curves representing an increasing increment in relative satisfaction for upstream and downstream landowners as the probability of flooding begins to decrease from the status quo of 0.2



Relative Preference for Qualitative Measures

Levels of quality of wildlife viewing (e.g., more than five of both iconic species, Table 4) are not given numerical labels because these labels are often misinterpreted as indicators of relative satisfaction. To express relative satisfaction for use in a more formal analysis of alternatives, numerical values between 0 and 1 must be assigned to the levels of the qualitative scale. The first step is to order the verbal categories from worst to best. This order might differ for different users, although the worst and best categories are likely to be obvious. For this scale, seeing none of either iconic species will receive a numerical value of 0 (worst).Seeing both in numbers greater than five will receive a numerical value of 1 (best). As mentioned above, the order of the four intermediate categories isn't entirely obvious because one stakeholder group might prefer seeing both species in smaller numbers to seeing only one species but in larger numbers, and another stakeholder group might prefer the opposite.

There are a number of techniques for obtaining expressions of relative preference for qualitative measures from stakeholders and decision makers. One of these, the ratio method, is discussed in Maguire (2014).[146]The hypothetical output of using the ratio method is shown in parentheses below the descriptions of levels of the wildlife viewing measure in Table 5.

[146] L. Maguire, "Multi-criteria Evaluation for Ecosystem Services: A Brief Primer" (2014), https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

Table 5. Performance evaluations and corresponding relative satisfaction for the three alternatives for restoring forested wetlands.

Measures	Status quo	Downstream dam	Upstream release
Number of bird 1 (breeding pairs on forest)	200 (0)	220 (1)	205 (0.25)
Wildlife viewing at walkway site (qualitative scale)	One iconic sp < 5 (0.14)	One iconis sp < 5, one > 5 (0.86)	Both > 5 (1)
Flood Events (annual average)	0.2 (0)	0.15 (0.8)	0.2 (0)
Cost ($MM NPV)	0.1 (1)	1.0 (0)	0.8 (0.6)

Note: Performance levels for each of the four alternatives are translated to a 0–1 scale (in parentheses) expressing relative satisfaction using Figure 21 for number of breeding pairs and Figure 22 (downstream) for flood-risk reduction. The methods for determining relative satisfaction values for wildlife viewing and cost can be found in Maguire (2014).[147]

The alternatives matrix in Table 5 captures stakeholder satisfaction with the varying levels of performance for each of the target services. Comparing the alternatives requires that preferences be integrated over the different services.

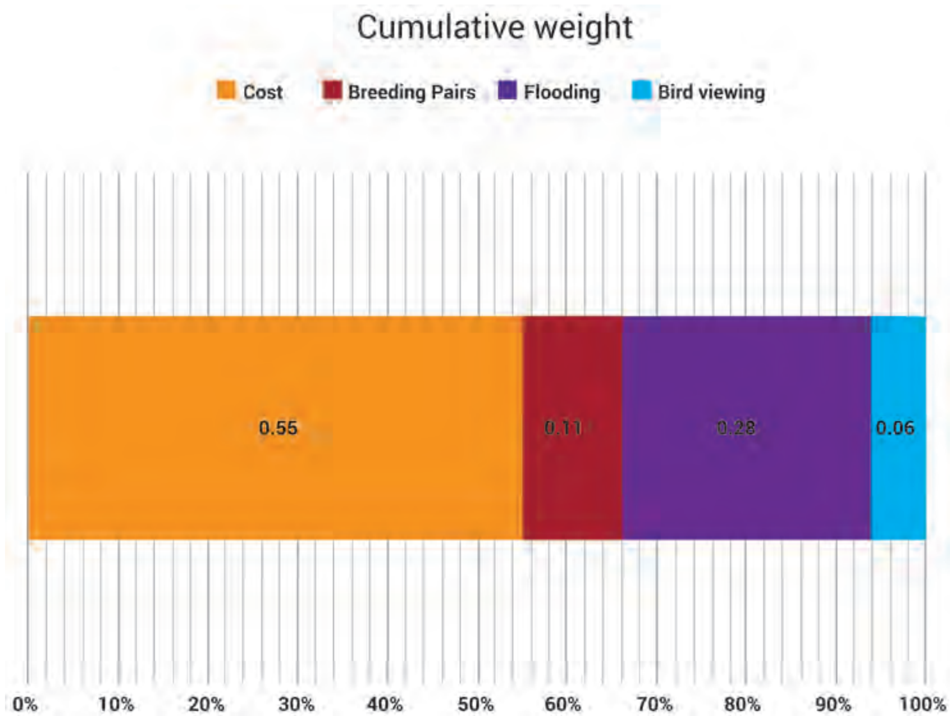
Using Weights to Express Tradeoffs among Objectives

In addition to numerical expressions of relative satisfaction for the levels of individual measures, a formal evaluation of tradeoffs among multiple measures requires some expression of the priority accorded each measure. A common expression of priorities among a suite of measures is a set of fractional weights that add up to 1. These weights reflect willingness to accept worse performance on one measure in order to secure better performance on another, i.e., willingness to make tradeoffs among conflicting objectives. In a multi-criteria analysis, weights can help address the concern that gains in one ecosystem service might be accompanied by losses in another service (or in another valued objective).

There are a variety of structured ways to assess weights (see Clemen 2001); using a specialized consultant to implement these methods is a good idea. If that is not possible, a stopgap approach is to use a visual representation of weights, such as a bar with segment lengths proportional to the weight on each measure (Figure 23). The fact that the length of the whole bar is fixed at 1 requires that any increase in weight on one measure be compensated by decreases in the weight on one or more of the other measures.

[147] L. Maguire, "Multi-criteria Evaluation for Ecosystem Services: A Brief Primer" (2014), https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

Figure 23. A visualization of the relative weights assigned to the four measures of performance in the wetland restoration example



Beyond the algebraic constraint that weights over all services must sum to 1.0, three nuances warrant particular emphasis. First, weights depend on the ranges of performance. Second, weights might not be transferable from decision context to another. Third, weights may differ among stakeholder groups.

Weights Depend on the Ranges of Performance

As with the relationships that express relative satisfaction, willingness to make tradeoffs is tied to the range of possible performance levels that might be encountered when evaluating a particular set of alternatives. It is easy to see that this is so by imagining that the range of costs in Table 5 is $500,000 to $550,000 instead of $100,000 to $1 million. If the ranges for the other three performance measures (breeding pairs, wildlife viewing, and flood events) remain the same, the impact that different levels of cost have in determining overall satisfaction with each alternative will be far lower when the range of costs is narrow than when the range is wide. The weight on cost will be lower in the former case than in the latter. (And, because the weights must add up to 1, the weights on the other three measures will be correspondingly larger in the former case.)

Weights Might Not Be Transferable from One Decision Context to Another

Unlike relationships that express relative satisfaction, wherein the range of performance levels for a single measure affects only the pattern of relative satisfaction for that measure independent of the performance levels on other measures, the weights for a set of measures have meaning only in relation to each other and only in relation to the ranges of performance on all measures for a particular problem. It is not credible to elicit weights for one set of performance levels in one decision context and then apply those weights in another context without first verifying that the performance levels and the particulars of the decision context are similar enough to justify such a transfer (and see Further Discussion, below).

Weights Differ among Stakeholder Groups

For most contentious decisions, the fundamental disagreements among stakeholder or user groups are about the priorities placed on different objectives, as expressed by weights. Capturing these differences by eliciting separate sets of weights for different users is very helpful to both decision makers and user groups. Attempting to gloss over differences in priorities by eliciting weights from only one or a few perspectives, or by averaging weights across user groups, is a recipe for continued contention.Eliciting weightsfrom all user groups can sometimes suggest where compromises that satisfy some of the needs of each group can be found. It also can help address concerns about distributional equity by identifying the values and preferences of groups defined by ethnicity or other cultural or socioeconomic markers.

Combining Value for Multiple Services

Articulating the combined value of a suite of ecosystem services has been the target of much research and the expressed desire of federal regulatory and budgetary organizations. The type of multi-criteria analysis described here offers one way of meeting that need by yielding numerical values that describe the relative capacities of a set of management alternatives (usually including the status quo) to produce desired ecosystem services. These numerical expressions of relative merit are tied to a particular decision context, a particular set of alternatives, and particular characterizations of relative satisfaction with performance and priorities among conflicting objectives. This type of analysis easily blends measures that are typically monetized (e.g., financial costs of implementing management actions) with those that are not easily monetized (e.g., the experience of viewing iconic wildlife species).

Estimating utility and weights for multiple services is a process subject to uncertainty from various sources (e.g., imprecision in ecological indicators, choice of stakeholder subjects). Maguire (2014) discusses possible methods, such as sensitivity analysis, for dealing with these uncertainties.[148]

Sometimes decision makers do not want to create numerical values that express the relative merits of each alternative but instead prefer to intuitively integrate in the decision process information about performance, relative satisfaction,and weights. When a combined score for each alternative is wanted, a commonly used method is to calculate an overall value on a 0–1 scale by adding up, for all measures, the weight on each measure multiplied by the relative satisfaction associated with performance on that measure. Table 6 includes the weights shown in Figure 23 as well as the performance and corresponding relative satisfaction values in Table 5 and reports the overall value for each of the three analyzed alternatives (e.g., (0.11)(0) + (0.06)(0.14) + (0.28)(0) + (0.55)(1) = 0.56 for the status quo).

[148] L. Maguire, “Multi-criteria Evaluation for Ecosystem Services: A Brief Primer” (2014), https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

Table 6. A full representation of a multi-criteria analysis of tradeoffs in performance for three wetland restoration alternatives evaluated with four performance measures.

	ALTERNATIVES		
Measures (weights)	**Status quo**	**Downstream dam**	**Upstream release**
Number of bird 1 (breeding pairs on forest) (w = 0.11)	200 (0)	220 (1)	205 (0.25)
Wildlife viewing at walkway site (qualitative scale) (w = 0.06)	One iconic sp < 5 (0.14)	One iconis sp < 5, one > 5 (0.86)	Both > 5 (1)
Flood Events (annual average) (w = 0.28)	0.2	(0) 0.15 (0.18)	0.2 (0)
Cost ($MM NPV) (w = 0.55)	0.1 (1)	1.0 (0)	0.8 (0.6)
Overall value	0.56	0.39	0.42

Note: The weights from Figure 23 have been added to Table 5, and overall values for each alternative have been calculated by summing the weight times the relative satisfaction value associated with performance across the four measures.

As discussed in the overall framework, the decision process can play out at varying levels (from fully participatory to top-down authority), and other factors that were not included in formal analysis may enter into the decision (agency mandate, equity issues, jobs, costs). Information provided by multi-criteria analysis is advisory to the decision but does not necessarily dictate the decision itself.

Further Discussion

Advantages and Disadvantages

The characteristics of structured decision making that are likely to mean the most to agency practitioners are ease of use, transparency, and minimization of the potential for misleading results. From a practitioner's point of view, the type of MCDA presented here—multi-attribute utility analysis—may appear(1)too hard to implement, too much work to implement, or both; (2) limited by the decision context for which the analysis was made; and (3) useful only for comparison, with calculated values having no absolute meaning. Disadvantages 2 and 3 apply equally to other kinds of multi-criteria decision analysis and, therefore, are not a disadvantage of MAUA in particular.

Monetary valuation methods, to be used in benefit-cost analysis or some other type of economic analysis, are sometimes promoted as solutions to disadvantages 2 and 3. However, for the reasons described in the Monetary Valuation, and Benefits Assessments sections, values may be neither as transferable to other contexts nor as absolute as many assume they are.

The remaining disadvantage, that MAUA is too much work to implement or is too hard for non-specialists to implement competently, has some merit, as described in Going It Alone versus Engaging Specialized Consultants below. Some alternative types of multi-criteria analysis referenced in Maguire (2014) attempt to reduce the data required to implement an analysis and to simplify, or even automate, the judgments that must be elicited from decision makers or stakeholders (e.g., by requiring only pairwise comparisons of alternatives instead of numerically expressed evaluations or by applying a set of rules for reducing the dimensions of a decision problem).[149] The availability of user-friendly commercial software for implementing some of these methods has undoubtedly enhanced their use in environmental applications. However, some of these alternate methods have structural flaws that can lead to results that do not accord with common sense, and many of them incorporate assumptions that are not wholly transparent to users. Thus these methods can fall short in terms of lacking transparency and producing potentially misleading results.

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[149] L. Maguire, "Multi-criteria Evaluation for Ecosystem Services: A Brief Primer" (2014), https://sites.nicholasinstitute.duke.edu/ecosystemservices/research/publications/.

MAUA has several advantages as a tool for multi-criteria decision analysis: (1) It might be difficult to implement, but the problems being addressed are genuinely difficult for a host of reasons (e.g., disputes among parties, technical disagreements, limited information, conflicting goals and mandates). MAUA helps to identify and articulate these difficulties. (2) Going through the steps of MAUA (i.e., stating objectives, developing measurement criteria, evaluating performance, assessing relative satisfaction and weights), obliges decision makers and stakeholders to address all these sources of potential difficulty explicitly, even when they choose to do so only qualitatively. (3) Addressing all these stages of analysis explicitly enhances transparency, an especially important characteristic for public decision making. (4) The dependence of analytical results on decision context, and the inherently relative nature of those results, is real. That MAUA makes these limitations more obvious than some other types of analysis is to its credit rather than to its detriment.

Comparison of MCDA to Other Valuation Methods

In this guidebook, MCDA is presented as one of three ways of assessing the social impact of changes in the provision of ecosystem services. In the simplest alternative, methods based on socially relevant indicators, the decisionmaker implicitly invokes stakeholder satisfaction by accounting, as comprehensively as possible, for issues related to stakeholder access, stakeholder numbers or demographics, rarity and substitutability, and so on, but the stakeholders themselves are not directly involved in this process. In terms of how decision maker and stakeholder preferences are made explicit in a decision process, the essential choice is between monetary valuation and MCDA (or, as mentioned above, a combination of the two). The primary distinction between valuation and MCDA is the replacement of dollar values (or other currency recognized by stakeholders for trade or barter) with a unitless measure of relative satisfaction or preference (utility).

Expressions of Relative Satisfaction
Monetary valuation of non-market goods or services, such as wildlife viewing quality, is an alternative way of expressing relative satisfaction with different levels of performance. Monetization allows market and non-market goods and services to be compared on a common scale, but it is sometimes difficult, or perceived as inappropriate, to come up with monetary values. For example, many Native American tribes are culturally reluctant to assign monetary value to the gifts of nature. In such cases, the relative satisfaction expressed as utility offers an alternative approach.

Cost-effectiveness and Cost-utility Analysis
In valuation, an explicit aim is often to compare management alternatives in terms of their cost-effectiveness or to perform a full benefit-cost analysis. Cost-effectiveness analysis (CEA) reports how much it costs to increase performance of a particular measure by one unit (e.g., acres, number of animals). Because only one measure can be evaluated at a time, tradeoffs among non-monetized measures cannot be expressed.

Cost-utility analysis (CUA) involves much the same calculation as CEA, but it uses a multi-attribute function (weight times value or utility of each measure) to aggregate all measures except cost into a unitless metric on a 0–1 scale, as in the MCDA presented above. A CUA combines only the first three measures in Table 6 into overall value (on a unitless 0–1 scale), leaving the fourth measure, cost, in dollars. Overall value of a particular alternative divided by its dollar cost would then be used to compare the cost efficiency of different alternatives in enhancing the combined utility of the non-cost measures. (Note that this task would require re-estimation of the weights for the three measures other than cost.)

Capturing Diversity in Stakeholder Preferences
MCDA is often used to emphasize heterogeneity in preferences among different groups of stakeholders. Therefore, multiple alternative matrices may be created to show the range and diversity of perspectives. In contrast, often only one alternative matrix is created for economic valuation, which is commonly used to generate a single aggregated value across stakeholders for analyses like benefit-cost analysis. Such an aggregated value would incorporate the size of each stakeholder group population and their relative preferences for predicted changes in services. There is no real reason that preferences elicited using MCDA could not be aggregated, nor that economic valuation instruments could not be used to capture within-group heterogeneity; practitioners of the two approaches have simply pursued different applications with different aims.

Context Dependency of Social Impacts

As noted above, the relative satisfaction or preferences (utilities) elicited from stakeholders in MCDA are context-dependent in that the values depend on several factors: (1) which stakeholders are being engaged to inform expressions of preference (value or utility functions and weights), (2) the type and magnitude of ecological services that are produced by the management alternatives being evaluated, and (3) other particulars of the decision context (e.g., geographic, temporal). Because of this contextdependency, expressions of stakeholder preferences cannot easily be transferred to other decision contexts. The same caveats apply to utilities derived from broad surveys, because these utilities reflect the performance levels and management alternatives offered for evaluation. Moreover, the same caveats apply to monetary values estimated through nonmarket valuation methods, because these are estimated relative to or contingent on a specific decision context.

Transferring Expressions of Relative Satisfaction

Under a monetary valuation approach, benefits-transfer methods are sometimes used to transfer dollar values (or functional relationships between dollars and levels of a performance measure) to different user groups and different decision contexts. As noted above, such transfers are significantly limited. Recent methodological developments in benefits-transfer analysis and meta-analysis are promising in that they suggest relatively robust approaches to benefits-transfer modeling. In principle, similar approaches might be developed for MCDA. Devising robust methods for transferring estimates of social impacts remains a key challenge in generalizing the ecosystem services approach so that it can be implemented in different decision contexts without requiring new monetary or non-monetary valuations.

Going It Alone versus Engaging Specialized Consultants

Non-specialists can employ to good effect many of the elements of structured decision making: articulating objectives, scrutinizing objectives for completeness and redundancy, defining measures clearly, scrutinizing proposed measures for implicit inclusion of relative satisfaction (where it does not belong), and recognizing instances in which different decision makers or user groups may have different beliefs or different preferences that impinge on the decision structure (e.g., differing assessments of performance, differing relationships of relative satisfaction to performance, differing priorities among objectives).

Other elements of structured decisionmaking benefit greatly from the experience and judgment of specialized consultants in decision making. These elements include (1) scrutinizing objectives hierarchies to make sure that they accord with the assumptions for independence of different parts of the hierarchy, which are necessary to support subsequent stages of the analysis (i.e., eliciting relative satisfaction with performance levels, eliciting weights to express priorities, and forming a combined overall value of alternatives by summing the products of relative satisfaction and weight for each measure) and (2) identifying where and how to use sensitivity analysis to illuminate essential tradeoffs and establish how robust the results of an analysis are to changes in the ingredients used to compose it. (For example, sensitivity analysis can help reveal if different curves relating relative satisfaction to performance are likely to change the ranking of management alternatives).

Some elements of MAUA use complex procedures to elicit numerical representations of beliefs and preferences (e.g., eliciting expert opinion to fill in gaps in performance predictions, eliciting relationships for relative satisfaction with different levels of performance, eliciting weights to express priorities among objectives). Although some simplified methods of performing these tasks are offered here, non-specialists will face many challenges and therefore will benefit from the services of specialized consultants for these portions of an analysis. They can also take advantage of the expertise of the growing number of agency consultants trained in structured decision making.

Anyone at any level can clarify complex decision problems by taking a structured approach such as that recommended here. Even the most limited use of MAUA concepts in a purely qualitative fashion can help managers make decisions in a consistent and transparent way. Effort invested in articulating objectives and organizing them hierarchically and then defining transparent measurement scales is especially likely to improve any type of analysis that may follow. Even if they are not brought together in a full MAUA, any one or more of the stages of structured decision making (e.g., creating a performance matrix, defining relationships between satisfaction and performance for individual criteria, assessing priorities among criteria) can help inform decisions. But, where the stakes are high, engaging the help of consultants versed in structured decisionmaking is wise. Not doing so is tantamount to attempting an economic analysis of benefits and costs without engaging the help of economists.

Conclusions

MCDA offers a structured framework for estimating the relative preferences of stakeholders for changes in the provision of ecosystem services as affected by management alternatives. In this framework, decision makers are encouraged to embrace the complexity of the decision in terms of stakeholders, specific objectives, and the web of ecological interactions through which management actions can propagate and affect human systems. The approach can be applied to goods and services that are difficult to monetize, and it can be implemented with a variety of data, including expert opinion and qualitative metrics. There are intricacies and complications to the approach, but MCDA can be a useful tool in assessing the social impacts of managing ecosystem services.

Best practice questions for the use of multi-attribute utility analysis:

To follow best practices the assessor should be able to answer *yes* to ALL of these questions:

- Is an expert trained in multi-criteria analysis methods involved?
- Are the measures of preference tied to the decision context for which the preference evaluation input was obtained? Is a quantified difference in the provision of a service being evaluated?
- Are the preferences of all parties/stakeholders affected by a decision being assessed to ensure a transparent process? (If the assessment involves services and interests outside an agency's authorities, collaboration may be necessary.)
- Are different preferences being assessed to reflect different marginal changes if the scale or other elements of the analysis are changing?

Recommended Reading

Clemen, R.T. (with T. Reilly). 2001. *Making Hard Decisions*. 2nd ed. revised. Pacific Grove, CA: Duxbury Press.
This comprehensive reference for the technical aspects of decision analysis covers elicitation of expert opinion, assessment of value and utility functions, and assessment of weights (and many other topics).

Department of Communities and Local Government. 2009. *Multi-Criteria Analysis: A Manual*. www.communities.gov.uk.
This manual aimed at non-specialist readers presents a helpful overview of the steps in multi-criteria analysis. Chapter 5 discusses decision context, objectives and measures, and evaluations of performance. Chapter 6 covers expression of relative satisfaction, determination of weights, and calculation of overall value using purchase of a toaster as an example. Chapter 7 presents some more complex examples that will resonate with environmental managers.

Gregory, R., L. Failing, M. Harstone, G. Long, T. McDaniels, and D. Ohlson. 2012. *Structured Decision Making: A Practical Guide to Environmental Management Choices*. Oxford, UK: Wiley Blackwell.
This book describes the use of multi-criteria decision analysis for real-world environmental decision making involving multiple stakeholders. It is not a how-to manual.

Hammond, J.S., R.L. Keeney, and H. Raiffa. 1999. *Smart Choices*. Cambridge, MA: Harvard Business School Press.
This non-mathematical presentation of the concepts of decision analysis is aimed at the general public and uses everyday decisions (such as buying a house). It presents some simplified tools for expressing relative satisfaction and weights and for determining overall value.

Thompson, M.P., Marcot, B.G., Thompson, F.R., McNulty, S., Fisher, L.A., Runge, M.A., Cleaves, D., and M. Tomosy. 2013. *The Science of Decisionmaking: Applications for Sustainable Forest and Grassland Management in the National Forest System*. General Technical Report WO-88, U.S. Department of Agriculture, http://www.fs.fed.us/rm/pubs_other/rmrs_2013_thompson_m004.pdf.
This Forest Service Technical Report synthesizes key points from the body of work on structured decisionmaking and illustrates how it can be relevant for land management planning in National Forests and Grasslands.

OTHER METHODS

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Management outcomes can be understood and compared with many approaches outside of a monetization or multi-criteria decision analysis (MCDA) framework. These approaches include highly structured approaches, such as cost-effectiveness analysis (CEA), and less structured approaches, such as those in which subsets of stakeholders choose indicators for their relevance, weight those outcomes, and generate aggregate scores for alternatives.

A key distinction between these approaches and traditional benefit assessment methods (i.e., monetization and MCDA) are that these approaches are either less formal approaches to comparing benefits or are formal approaches that cannot give managers a quantitative understanding of social welfare effects. They are not full representations of benefits, either because they focus on maximizing one outcome (e.g., habitat acres) or because they have not used a democratic process to consider public preferences. It is perfectly valid to use these approaches to represent projects' capacity to achieve an agency mission or meet selected stakeholder goals, but that objective is not equivalent to quantifying improvements in social welfare. (For a more in-depth explanation, see Monetary Valuation.

A key distinction among methods is whether they represent one type of benefit or many. When one type of benefit is being represented, analysts can often use straight-forward empirical approaches to model the relationship between a variable and suggested benefit. For example, a specific improvement in an indicator (e.g., exceeding a quality threshold), can be combined with increased use (e.g., visitor days) to suggest a higher value for changes that are appreciated by more people. Either type of indicator might be used in CEA to compare projects.

When a few benefits are being analyzed, more complex models may be needed to compare tradeoffs among alternatives. Bioeconomic or other types of models have been used to optimize the production of multiple benefits. These models often use monetized benefits, but they may also aggregate benefits using benefit weightings. In this way, these models build on CEA by creating a system in which decisionmakers can evaluate maximization of multiple services per dollar spent.

Informal methods for analyzing multiple kinds of benefits often include many implicit assumptions about how benefits accrue (e.g., linearly with increases in benefit relevant indicators) and how benefits combine (e.g., all are equal and additive). Different units in the indicators used to represent different benefits are typically made comparable by normalized or standardized scores. These approaches can create biased results and often

double count benefits. However, some techniques attempt to overcome some of these biases.
Informal methods of assigning weights to benefits or ranking projects include a wide range of approaches, from simple weighted sums to complex aggregations. Visualizations rather than quantitative scores may be used to present information in easily digestible formats—for example, color-coded graphics that allow the user to make judgments about the importance of different indictors.

Although such approaches are easy and popular, they have also been criticized for failing to consider how the aggregation method may bias the interpretation of results. For example, assignment of scores to “high,” “medium,” and “low” categories can be arbitrary and lead to unintended consequences when data are used to make decisions. Similarly, standardizing scores can lead to very small physical changes being given the same weight as very large physical changes. Perhaps, most importantly, many techniques do not adequately consider thresholds or the context of the indicator changes. A large change in the concentration of a toxic chemical in a stream may not have any social welfare effects if the toxin is still too high to support fish or revive the ecosystem.

Recommended Reading

Ferraro, P.J. 2004. “Targeting Conservation Investments in Heterogeneous Landscapes: A Distance-Function Approach and Application to Watershed Management.” *American Journal of Agricultural Economics* 86: 905–918.
This article uses bridging indicators to support decision making by applying a multivariate statistical approach (data envelopment analysis) that has been grounded in an economic framework.

Kienast, F., J. Bolliger, M. Potschin, R.S. de Groot, P.H. Verburg, I. Heller, D. Wascher, and R. Haines-Young. 2009. “Assessing Landscape Functions with Broad-Scale Environmental Data: Insights Gained from a Prototype Development for Europe.” *Environmental Management* 44: 1099–1120.
This article’s example of ecosystem-services based indicator development also demonstrates an approach to aggregating indicators on the basis of the amount of change occurring in land use scenarios.

Mace, G.M., and J.E.M. Baillie. 2007. “The 2010 Biodiversity Indicators: Challenges for Science and Policy.” *Conservation Biology* 21: 1406–1413.
This article details considerations for selecting policy-relevant indicators for biodiversity conservation.

Zhao, M., R.J. Johnston, and E.T. Schultz. 2013. “What to Value and How? Ecological Indicator Choices in Stated Preference Valuation.” *Environmental and Resource Economics* 56: 3–25.
This article discusses characteristics that make ecological indicators useful for understanding and measuring welfare effects.

THE DECISION PROCESS

Using BRIs in Decision Making
Displaying Assessment Results with Alternative Matrices and Maps
Combining Results: Weighting and Aggregation

USING BENEFIT-RELEVANT INDICATORS IN DECISION MAKING

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This section is an excerpt from the paper "Best Practices for Integrating Ecosystem Services into Federal Decision Making."

Benefit-relevant indicators (BRIs) represent the minimum requirement for measurement that links policy options to ecosystem services analysis. BRIs can be used in intuitive decision making and tradeoff evaluation and as inputs to benefits assessment (monetary and nonmonetary approaches). BRIs, when used with nonmonetary methods like multicriteria analysis, can reveal options that produce the highest ecosystem services benefits for a given amount of spending, even when benefits cannot be monetized.[150]

BRIs in Intuitive Decision Making

Can BRIs stand alone as an input to decision making? Absolutely. By design, BRIs are more informative and intuitive inputs to ecosystem services analysis and stakeholder deliberations than purely biophysical measures or biophysical measures that are less directly relevant to social welfare. When decision makers prefer to form their own judgments, resolve their own tradeoffs, and set their own priorities (or if they lack the time or money to engage in preference evaluation methods), BRIs represent a more precise and transparent alternative to purely narrative claims of ecosystem services production. When a decision is being taken using BRIs, a basic, helpful step can be to construct an "alternatives matrix" that depicts each policy option's associated (measured or modeled) BRI outcomes (Table 7).

Table 7. Alternatives matrix for considering ecosystem services in intuitive decision making

Policy or Management Alternative			Option A	Option B	Option C
Ecosystem Service Benefit-Relevant Indicator	BRI 1	Vegetation density in areas upstream of flood-prone area with people or property of interest			
	BRI 2	Aquifer volume accessible by households			
	BRI 3	Amount of fish landed commercially			
	BRI 4	Acres of wetland habitat supporting recreationally important bird or fish species			

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[150] An application of BRIs to support decisions about where to manage invasive species revealed that BRIs with multicriteria decision support and an optimization framework can identify restoration sites that would generate the largest bundle of ecosystem services for a given level of spending and improve on current decision making. L.A. Wainger, D.M. King, R.N. Mack, E.W. Price, and T. Maslin, "Can the Concept of Ecosystem Services Be Practically Applied to Improve Natural Resource Management Decisions?" Ecological Economics 69 (2010): 978–987.

Source: National Ecosystem Services Partnership, *Federal Resource Management and Ecosystem Services Guidebook* (Durham: National Ecosystem Services Partnership, Duke University, 2014), https://nespguidebook.com/assessment-framework/alternative-matrices-and-maps/.

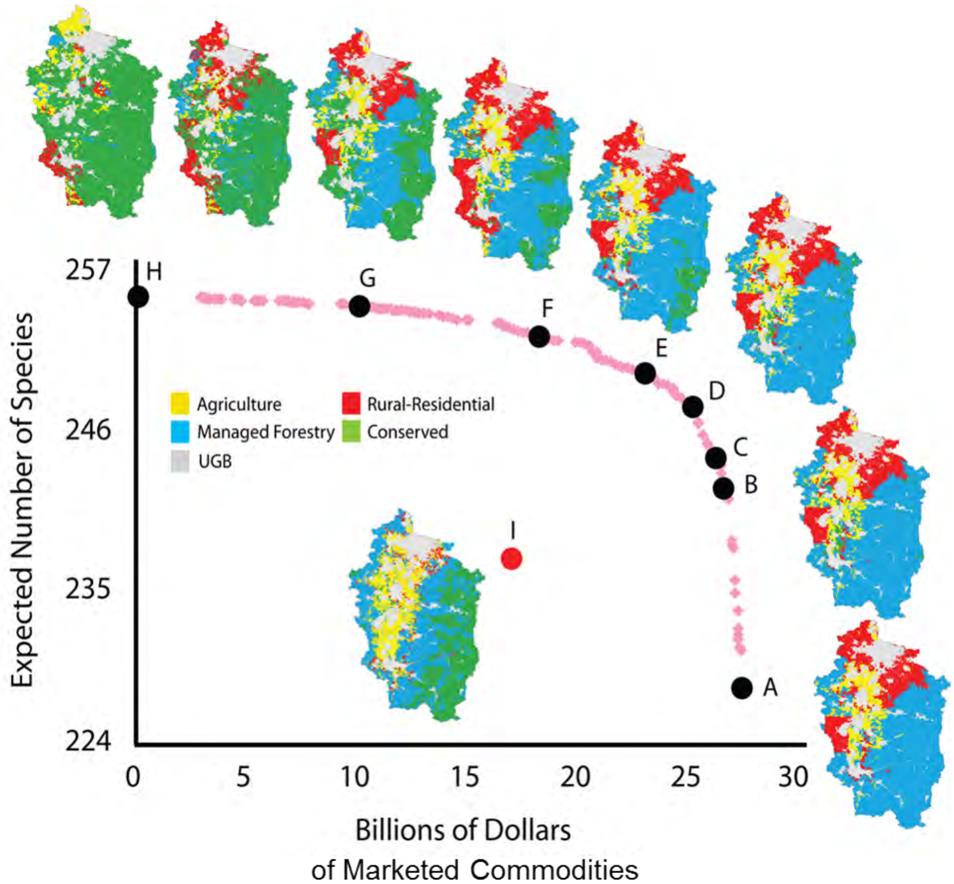
Evaluating Tradeoffs with BRIs

BRIs alone do not depict the importance, weight, or value attached to ecosystem services outcomes. Nevertheless, they can sometimes be useful in evaluating tradeoffs among options. However, the insight that BRIs alone can provide into tradeoffs is limited. Consider policies that incentivize different types of land use among agriculture, timber, housing, and conservation areas that affect the value of marketed commodities—agricultural crops, timber harvests, and housing values (measured in monetary value). These policies may also affect the persistence of terrestrial vertebrate species (measured in number of species expected to persist in the basin), and so tradeoffs in ecosystem services are inherent (Figure 24). It is assumed that species have existence value to the extent that people perceive benefits from the survival of a species, though putting that value in monetary or even nonmonetary terms is difficult.

Tradeoffs can be considered using this approach even when some services are reflected in value terms and others in BRIs, as in Figure 24. Clearly, points B, C, D, E, and F are superior to point I, which represents the current land use pattern, because they generate both higher conservation benefits in terms of more species and higher value of marketed commodities. But whether C is preferred to D or vice versa (or to any other two points on the efficiency frontier) depends on a value judgment about the relative importance of species conservation versus value of marketed goods. Is greater conservation or greater value of commodities preferred? In this case, BRIs help assessors consider the options in intuitive and socially relevant terms, but they do not identify a single best option without further analysis.

An action with positive effects on a greater number of BRIs will not necessarily have greater social value than an action that affects fewer BRIs. In general, the assessor cannot simply count (positively) affected BRIs provided by a system as a proxy for social value. Effects on social welfare depend not only on how many BRIs are affected but also on the degree of change in each BRI and the relative value of each BRI to all beneficiary groups. **Most decision contexts and policy options (environmental or not) involve tradeoffs that, if they are to be evaluated formally rather than intuitively, require application of benefits assessment methods.**

Figure 24. Schematic plot ("efficiency frontier") showing how marketed commodities and number of species can be used to assess tradeoffs between land use policies and species persistence



Source: S. Polasky, et al. "Where to Put Things? Spatial Land Management to Sustain Biodiversity and Economic Returns." *Biological Conservation* 141 (2008): 1516.

Using BRIs in Assessments of Benefits

An evaluation of preferences is needed if (1) service provision varies substantially across different stakeholder populations, i.e., there are differences of opinion about the outcomes, or (2) changes in services in response to management or policy vary in direction (or magnitude) across services. In either case, tradeoffs will have to be made, and that means valuation of some kind. BRIs are important and desirable inputs to *benefits assessment*, a broad term incorporating both economic (monetary) and nonmonetary valuation methods.[151]

[151] Evaluation sometimes refers only to economic or monetary valuation methods. In this case, it includes monetary and nonmonetary multicriteria methods. Both approaches include the preferences and values of people but use different units (e.g., dollars versus utilities) to do so.

THE DECISION PROCESS: DISPLAYING ASSESSMENT RESULTS WITH ALTERNATIVES MATRICES AND MAPS

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Two tools that can facilitate side-by-side comparison of management, project, or policy alternatives are alternatives matrices and maps. Alternatives matrices can incorporate information about estimated ecological changes and their effect on the provision of services, uncertainty about these changes, and the social importance of every service for each alternative. Maps can visually display changes in the provision of services by each alternative.

Alternative Matrices

Measures of *benefit relevant indicators* (BRIs), relative value (dollars or utils), or both can be organized by ecosystem service and management alternative and incorporated into alternatives or decision matrices. The columns of the table list management alternatives (do this, do that, or continue business as usual), and the rows list ecosystem services affected by these alternatives (Table 8). The matrices can be populated with ecological information (in which case they might also be referred to as performance matrices or alternatives/attributes matrices), or they can include information on stakeholder preferences for different levels of performance for each service.

Table 8. An example of an alternatives matrix for three management alternatives affecting four services

	Status Quo/BAU	Alternative A	Alternative B
Service 1			
Service 2			
Service 3			
Service 4			

Alternatives matrices can distill results from much of the assessment process documented in this guidebook, summarizing the state of knowledge for a management decision: what is known about how the ecosystem of interest will respond to management, stakeholders' preferences for the changes resulting from management, and confidence in that information. The measures and units that populate the matrix will vary (dollar value, percent change, or BRI), depending on the methods used for the assessment. Consequently, these matrices can be a powerful communications tool for managers and stakeholders. Ideally, they will be accompanied by a description of assumptions and discarded options.

In principle, alternatives matrices should include measures of all the important services significantly affected by management actions, even services about which there are uncertainties. Explicit recognition of uncertainties can point to needed research or collaboration. It also can suggest a conservative approach to decision making until knowledge gaps are filled.

Maps

Spatial variability in management and in the provision of services can often be visualized by complementing the alternatives matrix with spatially explicit maps of service provision. Geospatial tools can estimate the production and value of services for in a spatially explicit manner (e.g., InVEST, ARIES[152]).[153]Maps are most often used to show the supply (location and amount) of a service, but as described in this guidebook's discussion of benefit-relevant indicators (BRIs), that information alone is insufficient for analysts to determine whether the service provides a benefit to people (meets demand). In the case of a use service, the service (e.g., water yield) needs to reach the user, the user needs to be able to access the service (e.g., by boat), and the service needs to be sufficient for use (e.g., meet water quality thresholds necessary for swimming). In the case of a preventative service, the service needs to reduce risk to the user (e.g., the flood plain or coastal barrier system needs to be in the right place to provide protection).[154]Thus it is critical to have information about the supply of services (location and amount), the beneficiaries of services (location and amount of demand), and the characteristics of possible losses along the way. Mapping the flow of some services to beneficiaries (e.g., water-based services within watersheds) is straightforward, and mapping the flow of some other services (e.g., mapping the existence value of biodiversity) can be challenging.

When supply and demand information is integrated into an ecosystem services assessment through valuation methods, the resulting measure of ecosystem services benefits can be shown on a map, allowing analysts to explore spatial variability in services values. Figure 25 shows use of the InVEST model to estimate the dollar values of services affected by land use change scenarios.[155] In this example, as in many such examples, the map depicts the value of services at the location of production (the source), not at the location of the actual beneficiaries (though in this case the value is estimated from the beneficiaries' perspective).

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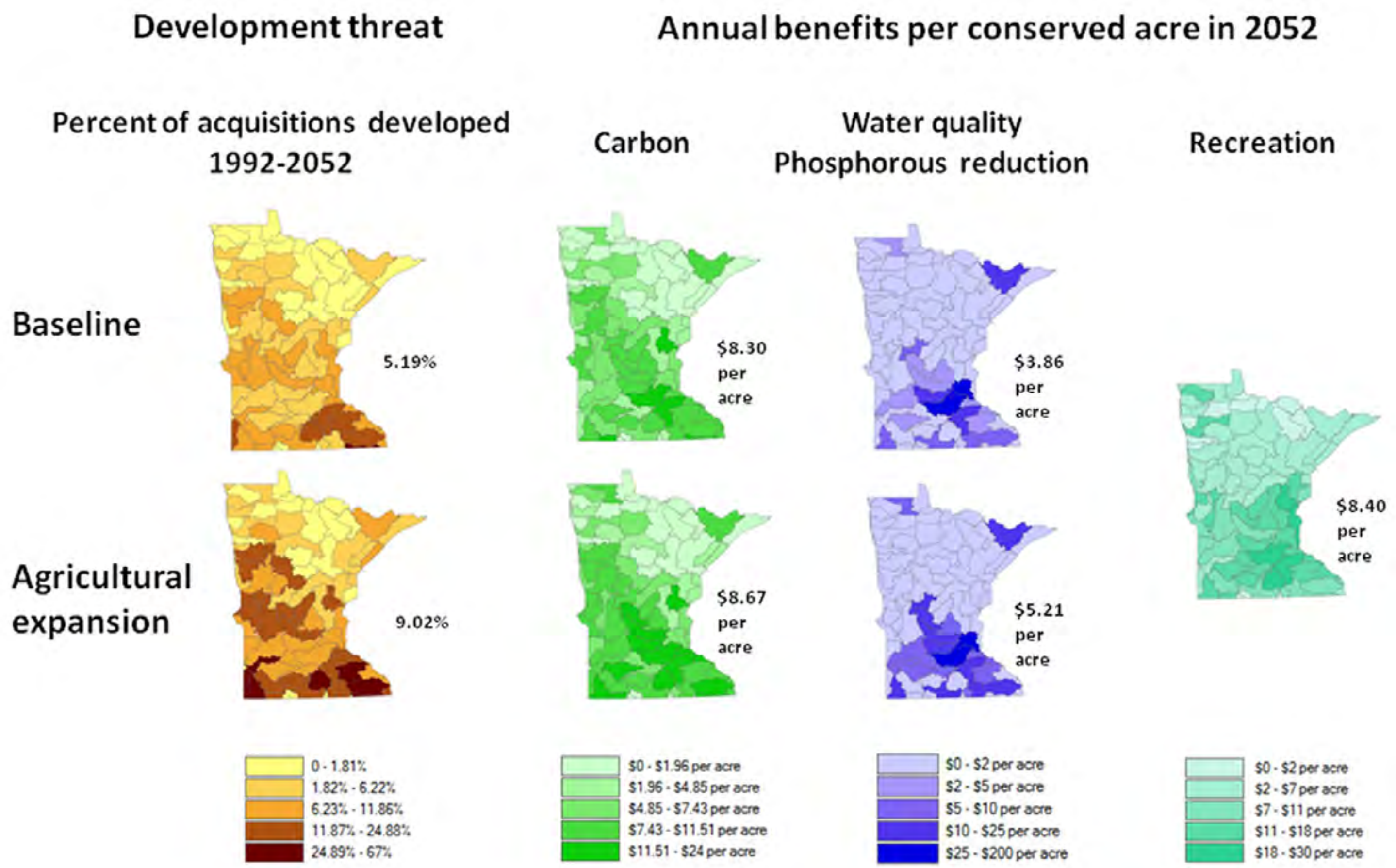
[152] ARIES (ARtificial Intelligence for Ecosystem Services), accessed January 20, 2016, http://www.ariesonline.org/.

[153] See, e.g., Figure 1 in E. Nelson, G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. R. Cameron, K.M.A. Chan, G.C. Daily, J. Goldstein, P.M. Kareiva, E. Lonsdorf, R. Naidoo, T.H. Ricketts and M. Rebecca Shaw, "Modeling Multiple Ecosystem Services, Biodiversity Conservation, Commodity Production,and Tradeoffs at Landscape Scales," *Frontiers in Ecology and the Environment* 7(1)(2009): 4–11.

[154] The supply of a service is different from actual benefits, depending on the spatial relationships among the source, intervening sinks (benefit-absorbing sites), and the location of the beneficiaries. For a depiction of this concept, see Figure 25 in F. Villa, B. Voigt, and J.D. Erikson, "New Perspectives in Ecosystem Services as Instruments to Understand Environmental Securities," *Philosophical Transactions of the Royal Society B*. 369(2014):20120286, http://dx.doi.org/101098/rstb.2012.0286. The figure shows a hypothetical map illustrating how the supply of a service is different from the actual benefits received, depending on the spatial relationships among the source, intervening sinks (benefit-absorbing sites), and beneficiaries.

[155] "Natural Capital Project, InVEST: Integrated Valuation of Ecosystem Services and Tradeoffs," last accessed January 27, 2016, http://www.naturalcapitalproject.org/invest/.

Figure 25. InVEST model estimations of the values of services affected by land use change scenarios



Source: K.Kovacs, S. Polasky, E. Nelson, B.L. Keeler, and D. Pennington, "Evaluating the Return in Ecosystem Services from Investment in Public Land Acquisitions," PLoS ONE 8(6)(2013): e62202, doi:10.1371/journal.pone.0062202.

COMBINING RESULTS: WEIGHTING AND AGGREGATION

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Many assessments will result in an alternatives matrix with multiple measures of services. These different measures make it difficult for analysts to consider changes in services or benefits on equal footing. If the goal is to combine results to produce one score or value for each alternative, only one method should be used to populate the matrix. Units must be reconciled to allow direct comparisons of options. If the units can be reconciled, an alternatives matrix can display the results for each option as well as an aggregation of the overall change in services and, perhaps, a dollar value for the expected change in benefits. Producing a valid aggregation (avoiding double counting, biases, and so on) is challenging and will require experts; therefore, in many cases it is an aspirational task. It can be accomplished in different ways given the methods used to populate the alternatives matrix.

Aggregating Benefit-Relevant Indicators

When estimated changes in ecosystem services production are provided as ecological measures or benefit-relevant indicators (BRIs) (e.g., expected floods per year or numbers of desired bird species in viewing areas), they are often aggregated across services and options by measuring change relative to the status quo or baseline scenario (e.g., percent change). This approach does not make the units commensurate (fractions of flood risk are not comparable to fractions of bird species), and worse, it implicitly assigns a measure of relative importance to the services whereby the largest ecological change will drive the decision outcome even if it is not the most important change. Moreover, this approach reflects neither stakeholder preferences for the different levels of ecological change (small or big relative to the baseline) nor preferences for one service over another. This process can be improved by allowing decision makers and stakeholders to weigh in on each service separately and explicitly; this approach can be incorporated into monetary and nonmonetary valuation methods.

Aggregating Monetary Valuation Measures

Monetary values for the estimated changes in ecosystem services production can be easily combined across services and options. It is important to ensure that the monetary values used are appropriate for the geography and context in which they are applied. Values for services are context dependent. It is also important to include all significantly affected services. If it is difficult to monetize all relevant services it will be important to find a way to fully include nonmonetized services in the decision process. One way to do this is to use nonmonetary (multicriteria decision analysis) methods that can combine monetary and nonmonetary methods.

Aggregating Nonmonetary Valuation Measures

In nonmonetary multicriteria decision analysis, methods are used to estimate stakeholder preferences for various levels of ecological performance (e.g., two or three more fish) as well as willingness to make tradeoffs among services (e.g., fish versus birds versus flood risk). These preferences can be combined with unitless measures of satisfaction for each service to obtain an overall measure of value for each alternative. These measures can then be combined across services and options to do a full comparison.

All aggregation methods have strengths and weaknesses that analysts should consider before and after their application to ensure that they are not biasing results or misrepresenting outcomes. Experts will be needed for this process.

STAKEHOLDER ENGAGEMENT

STAKEHOLDER ENGAGEMENT

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Summary

Incorporating ecosystem services into decision making can expand the stakeholder pool and the types of ecological outcomes that are important to discuss. Many existing forms of stakeholder engagement can be adapted or expanded to accommodate these and other changes suggested by an ecosystem services approach. This section discusses how stakeholder engagement processes may differ when ecosystem services considerations are incorporated into the decision process. It identifies discussion topics, communication tools, and the information to be gathered from stakeholders in each phase of the decision-making process.

Takeaways

- An ecosystem services perspective can result in engagement of stakeholders who care less about traditionally recognized services (e.g., harvests, recreation) and more about less recognized services (e.g., maintaining species habitat for future generations) as well as stakeholders who are located outside traditional jurisdictional (i.e., geographic), legislative (i.e., mandates), or temporal (i.e., present versus future) management boundaries.
- Ideally, ecosystem services enter the conversation when managers, communities, and experts begin to assess problems and objectives.
- Engagement of stakeholders can vary in intensity throughout an ecosystem services assessment.
- The level of technical expertise needed to effectively communicate with stakeholders may increase as an ecosystem services assessment progresses.

Introduction

Stakeholder engagement is not new to agency planning processes and is required by various laws, rules, and policies (e.g., 2012 Forest Service Planning Rule[156], National Environmental Protection Act).[157]In the federal agency context, Congress, the Office of Management and Budget, and departmental and agency policy generally set the sideboards for stakeholder engagement. Although in some cases such engagement might be limited or considered infeasible, unnecessary, or undesirable (e.g., because an agency has a well-defined mission goal in the federal interest and does not require public input), in most cases it is important to planning processes.

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[156] Department of Agriculture Forest Service, "National Forest System Land Management Planning," *Federal Register* 77(68)(2012):21162–21276, http://www.fs.usda.gov/Internet/FSE_DOCUMENTS/stelprdb5362536.pdf.

[157] National Environmental Policy Act, 42 U.S.C. § 4331 (1969)

Stakeholder engagement can take many forms, including face-to-face meetings or presentations, web-based materials, radio/TV/newspaper coverage, or written materials sent with surveys. An ecosystem services approach to planning does not require adoption of new forms of engagement. It does suggest ways to adapt or extend these forms to capture the diverse stakeholder pool made apparent by that approach.

Identifying Stakeholders

Within this guidebook, the term stakeholder refers to any person or party interested in or affected by a decision process. Stakeholders can include those who will be materially affected by a decision, those who need or want to take action to secure a flow of ecosystem services, those who might take action that would impede the flow of ecosystem services, and those who are not aware they are benefiting from or impeding flows.Ideally, all of these parties will be engaged as stakeholders.

Ecosystem services assessments expand or refine the breadth of stakeholders typically included in a decision process for two reasons.First, they reveal benefits (and thus beneficiaries)that are underappreciated within traditional decision processes—benefits like the existence of and maintenance for future generations of viewsheds, sacred places, old-growth redwood forests, or endangered species. By focusing on services that can be directly used (e.g. harvests, recreation), past decision processes may have unintentionally left out people who highly valued the less tangible cultural, spiritual, and existence values of ecosystems and species.

Second, ecosystem services assessments help define potential locations and numbers of positively and potentially negatively affected parties by extending traditional jurisdictional (i.e., geographic), legislative (i.e., mandates), or even temporal (i.e., present versus future) management boundaries. Ecosystem services assessments encourage managers to broaden how they define stakeholders, capturing not only local affected beneficiaries of services, but also non-local and even future beneficiaries.They would, for example, take into account that habitat restoration targeted to benefit a particular species at a national wildlife refuge might also affect services such as water quantity, water quality, flood risk, and fire risk for both current and future affected parties outside the refuge's geographic and legislative boundaries. This consideration could improve engagement strategies associated with the restoration effort.

Determining How Stakeholders Should Be Engaged

Although ecosystem services assessments enable managers to identify all stakeholder groups that may be affected by a decision,not all groups will require engagement in the same manner or with the same level of intensity. Some groups may need only to be informed of an assessment; others may need to participate in it. Whatever the case, managers will have to design an engagement process that meets laws and regulations regarding engagement and that realistically reflects time and funds constraints,but also sufficiently includes all interested or significantly affected parties.[158]An effective and efficient engagement strategy hinges on identifying the type of engagement (e.g., participatory discussions, online forums, formal surveys) and timing of engagement needed to obtain the desired information—whether general information about important services or the values that each stakeholder group attaches to those services.

Communicating about Ecosystem Services

Understanding how to communicate about ecosystem services to stakeholders is a critical element of an ecosystem services assessment. Successful public engagement, particularly at the outset of an ecosystem services assessment,will often feature intuitive, commonly used, and concrete language about specific resources, such as "abundant fish populations," "water suitable for swimming," "viewpoints over undeveloped landscapes," or "reduced flood risks."[159]

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[158] These laws include the National Environmental Policy Act and the Federal Advisory Committee Act; the regulations include the National Forest System Land Management Planning Rule (2012) and the rules in the Forest Service Environmental Policy and Procedures Handbook (FSH 1909).

Resource managers can tailor the information provided to stakeholders to the task at hand.The amount of technical terminology, level of detail, and number of experts needed to facilitate conversations with stakeholders will likely increase as the assessment progresses from defining values during the scoping phase to ranking or prioritizing outcomes that will affect those values during the assessment and analysis phase.

Stakeholder Engagement within the Ecosystem Services Assessment Framework

In the assessment framework outlined in this guidebook, stakeholders are engaged at the start of and, to the extent possible, throughout the planning process.The following sections present some of the key elements of stakeholder engagement that are particular to ecosystem services assessments. Because these assessments are unlikely to be truly linear in practice, there may be considerable overlap of key elements between steps.

Engagement During Scoping

In the ecosystem services assessment framework proposed in the guidebook, scoping involves two parallel and interactive processes: an ecological scoping process to identify status and trends in the condition of resources and a social scoping process to identify how stakeholders use the resources and what they value about them.
During scoping, stakeholder engagement activities may include

- Identifying beneficiaries and stakeholders to engage in the assessment,
- Determining effective ways to communicate about ecosystem services with stakeholders,
- Identifying key services for analysis (in particular, identifying the services that are important to stakeholders), and
- Clarifying the role stakeholders will play in the decision process.

Initially, agencies may rely on a pre-process assessment (either through internal means or through a facilitator) to identify stakeholders, taking care to comply with public comment and regulatory requirements for engagement.[160] The person conducting the pre-process assessment should not rely solely on historical key contacts, but instead should fully explore the universe of individuals and groups potentially affected by activities related to the decision process. This task can be facilitated by linking desired ecological conditions to possible beneficiaries and services. Understanding the demographic and cultural characteristics (socio-cultural context) of the affected area and how people are using the affected services may also help to ensure that all relevant stakeholder groups are identified. Importantly, additional stakeholders may be identified as the assessment progresses.

Tools for Communicating with Stakeholders
One suite of tools that may help facilitate discussions about nature's value to people is human ecology mapping (HEM). HEM is becoming increasingly popular as a tool to show the complex connections between humans and landscapes,answering questions such as "Where do conflicts arise over land rights, uses, and access?" "How and why does the spatial distribution of human activity vary temporally (seasonally, annually)?" and "What values or meanings are associated with sites within the project area?"[161]Narrative mapping tools may also be helpful in communicating connections between ecology and ecosystem services, which stakeholders may find less obvious than resource managers. These types of tools can visually show how ecosystems contribute to the services that stakeholders value.

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[159] The Nature Conservancy conducted a survey regarding the terminology of ecosystem services. More information about that study can be found at https://www.conservationgateway.org/ConservationPractices/EcosystemServices/CommunicatingEcosystemServices/Pages/communicating-ecosystem-s.aspx.

[160] See, e.g., the National Environmental Policy Act, the Federal Advisory Committee Act, the National Forest System Land Management Planning Rule (2012), and the Forest Service Environmental Policy and Procedures Handbook (FSH 1909).

[161] R. McLain, M. Poe, K. Biedenweg, L. Cerveny, D. Bessert, and D. Blahna, "Making Sense of Human Ecology Mapping: An Overview of Approaches to Integrating Socio-Spatial Data into Environmental Planning," *Human Ecology* 41(2013): 651-665, http://link.springer.com/article/10.1007%2Fs10745-013-9573-0; Raymond, C.M., B.A. Bryan, D.H. MacDonald, A. Cast, S. Strathearn, A. Grandgirard, and T. Kalivas, "Mapping Community Values for Natural Capital and Ecosystem Services," *Ecological Economics* 68 (2009): 1301–1315,

Topics of Discussion
To identify what people care about and why, managers may focus discussions with stakeholders on the idea of value. What do people value about a particular ecosystem or resource? Who is benefiting from this resource? What benefits are they afraid of losing? These conversations can rely on non-technical language.

A key piece of information to communicate at the outset of an ecosystem services assessment is the role of stakeholders in the decision process. This information will help stakeholders set reasonable expectations about how their values and opinions will be incorporated into the final decision.

Information Gathered from Stakeholders
Conversations with stakeholders during scoping provide important context for the actual ecosystem services assessment and analysis. Stakeholders will help to identify the desired outcomes, described here in terms of the ecosystem services or benefits of nature that are important to them. These desired outcomes imply a suite of management, project, or policy alternatives. These alternatives become the starting point and the services become the endpoint of ecological assessment, which indicates how different alternatives will change the production of services. Understanding which benefits of nature people consider important is also critical information for social impact analysis.

Engagement During Assessment and Analysis

The goal of ecosystem services assessment is to evaluate management, project, or policy alternatives in terms of their capacity to yield the desired outcomes (both ecological conditions and ecosystem services) identified during scoping. This assessment includes an ecological analysis and may include a social impact analysis. Stakeholder engagement activities may include

- Refining the ecological analysis and identifying additional stakeholders,
- Clarifying characteristics of ecosystem services that make them more or less valuable to stakeholders (these benefit-relevant indicators may also be discussed during the scoping step), and
- Prioritizing stakeholder preferences for outcomes.

Topics of Discussion
The ecological analysis (charted in means-ends diagrams) can be shared with stakeholders to ensure that results resonate with them and that no critical values and concerns have been overlooked. This analysis may identify additional stakeholders, resulting in an increasingly iterative process. For example, if an analysis of the ecological impacts of tree thinning in a national forest finds that the thinning will increase sedimentation in a stream, affecting its water quality beyond the forest's boundaries, resource managers may need to consider who uses the stream outside those boundaries, when they use the stream, and for what purpose. These people should be considered stakeholders because they will be affected by a decision to thin trees.

In addition to identifying any new stakeholders, discussions may also explore clarifying information about changes in services that will make those changes more or less valuable to stakeholders. These benefit-relevant factors, possibly identified during scoping,can be explored in greater depth. Stakeholders may be asked to provide information that can help to assess factors such as accessibility or substitutability of resources. For example, a management option that increases fish populations in a pond may not have much value to local fishermen if fishing opportunities in the area are already plentiful. However, birdwatchers may highly value opportunities to view the bird species that feed on those fish, especially if those opportunities do not already exist.

Discussions may involve the following considerations:

- How the benefits of particular services vary by user group (the benefits and key features of a water source for irrigation to a farmer are different than the benefits and key features of that same water sources for a recreational fisherman),

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http://www.sciencedirect.com/science/article/pii/S0921800908005326; Sherrouse, B.C., J.M. Clement, and D.J. Semmens, "A GIS Application for Assessing, Mapping, and Quantifying the Social Values of Ecosystem Services,"*Applied Geography* 31 (2011): 748–760, http://www.sciencedirect.com/science/article/pii/S0143622810000858; Brown, G.G., and P. Reed, "Public Participation GIS: A New Method for Use in National Forest Planning," *Forest Science* 55(2009): 166–182, http://www.landscapemap2.org/PublicParticipationGIS_ForestScience.pdf.

- How different alternatives may involve tradeoffs among services (increasing fishing opportunities may also increase the number of boats on a lake, a development that lakeside homeowners who value lack of noise may view negatively),
- How easily substitutes for lost services might be found (if bird watching decreases at one site, do nearby sites provide the same experience?),
- How services can be delivered over varying temporal and spatial scales (the amount of water available and needed for irrigation may vary seasonally or annually), and
- How uncertain delivery of ecosystem services may be(a severe drought may stress the system, or extreme amounts of rain may overcome the benefits of a reconnected floodplain).

Another possible topic for discussion is stakeholder preferences for the outcomes generated by each management, project, or policy alternative. These preferences can be formally incorporated into monetary or non-monetary valuation methods. Preferences can be elicited through paper surveys or online questionnaires as well as through one-on-one conversations and focus groups in which stakeholders can assign weights or values to various outcomes, which help resource managers to assess tradeoffs.

For most contentious decisions, the fundamental disagreements among stakeholder or user groups are about the priorities placed on different alternatives as expressed by assigned weights. Capturing these differences by eliciting separate sets of weights for different users is helpful to both decision makers and user groups. The results show why different groups prefer different alternatives and sometimes suggest where compromises that satisfy some of the needs of each group can be found. Attempting to gloss over differences in priorities by eliciting weights from only one or a few perspectives, or by averaging weights across user groups, is unhelpful. More information about weighting preferences can be found in Non-Monetary Methods: Multi-criteria Evaluation for Ecosystem Services.

Communicating with Stakeholders

Assessment and analysis will likely involve more in-depth and potentially, more technical discussions than those occurring during scoping. Experts may be needed to help convey the expected outcomes of different management, project, or policy alternatives on the production of services and the ecosystem service benefits received by various groups. They can share means-ends diagrams generated by the ecological analysis. These diagrams visually map how alternatives may affect the services identified as important by stakeholders and thus help stakeholders understand those relationships and clearly see where tradeoffs occur. Experts can also be particularly helpful in explaining the ranking, weighting, or prioritizing of values or outcomes.

Information Gathered from Stakeholders

Analysts can use the information gained through stakeholder engagement during assessment and analysis to understand how people use and benefit from ecosystem services. With that information, they can refine estimates of changes in the provision of ecosystem services and benefits. If stakeholder preferences for management, project, or policy alternatives are obtained, analysts can conduct a monetary or non-monetary evaluation to assess the social welfare effects of different alternatives. Both refined estimates of changes in the provision of ecosystem services and benefits and understanding of social welfare effects will add valuable information to the decision process.

Engagement During Decision Making

Once analysts compile all information pertaining to the decision (e.g., estimated ecosystem services outcomes for each alternative, stakeholder preferences, institutional mandates or priorities, financial constraints), managers are in a position to make the decision. If appropriate, stakeholders may participate in the decision process; however, responsibility for final decisions involving federally managed lands often rests with the administrative federal agency. Information gleaned from stakeholder engagement does not supersede other constraints on the decision process (e.g., institutional mandates, financial resources). Ultimately, managers will determine the level of priority given to stakeholder preferences for ecosystem services outcomes in the final decision.

Managers may be interested in engaging stakeholders or beneficiaries in the decision and implementation process if beneficial opportunities to partner with these parties arise from the assessment of services. For example, they may realize that external beneficiaries are willing to help support restoration on federal lands if it provides important benefits to their communities. Or perhaps agencies will find opportunities to solve management challenges by partnering with neighboring landowners (public or private). In those cases, stakeholders should be part of the decision process and the decision's implementation.

A key element of stakeholder engagement at this stage is communication of the decision process. Managers should clearly indicate what information contributed to the final decision, what did not contribute, and why. An alternatives matrix can be a helpful tool for such communication, but it should be presented alongside additional information about the decision process.

Engagement Following Decision Implementation (Evaluation and Reaction)

Following implementation of a decision, managers can obtain stakeholder's perspectives on how well the desired outcomes were achieved or collaborate with stakeholders on the development and implementation of monitoring plans, data from which will inform future planning processes.

Conclusion

In an ecosystem services assessment, stakeholder engagement seeks to identify what people value about an ecosystem and to what extent. Early engagement ensures that relevant ecosystem services are included in the assessment, particularly services that fall outside the agency's normal management boundaries. Sustained engagement provides valuable information about stakeholder preferences for different outcomes. Incorporating information about values and preferences into the decision process may lead to decisions supported by stakeholders.

Recommended Reading

Bright, A. D., H. K. Cordell, A. P. Hoover, and M. A. Tarrant. 2003. *A Human Dimensions Framework: Guidelines for Conducting Social Assessments.* General Technical Report SRS-65. USDA Forest Service Southern Research Station, Asheville NC. http://www.srs.fs.usda.gov/pubs/gtr/gtr_srs065.pdf.
This Forest Service Technical report provides a framework and guidelines for incorporating human dimension information into forest planning.

McLain, R., M. Poe, K. Biedenweg, L. Cerveny, D. Bessert, and D. Blahna. 2013. "Making Sense of Human Ecology Mapping: An Overview of Approaches to Integrating Socio-Spatial Data into Environmental Planning." *Human Ecology* 41: 651-665. http://link.springer.com/article/10.1007%2Fs10745-013-9573-0.
This synthesis paper describes several human ecology mapping techniques.

National Oceanic and Atmospheric Administration. 2007. *Introduction to Stakeholder Participation*. NOAA Coastal Services, Charleston SC. http://coast.noaa.gov/digitalcoast/sites/default/files/files/1366311008/stakeholder_participation.pdf.
This agency document provides technical guidance on stakeholder engagement.

Reed, M. S. 2008. "Stakeholder Participation for Environmental Management: A Literature Review." *Biological Conservation* 141:2417–2431. http://www.sciencedirect.com/science/article/pii/S0006320708002693.
This paper summarizes the development of participatory approaches in environmental management and discusses the potential benefits, limitations, and drawbacks of derived from such participation.

U.S. Bureau of Land Management. 2009. "Collaborative Stakeholder Engagement and Appropriate

Dispute Resolution." Collaborative Stakeholder Engagement and Appropriate Dispute Resolution Program. U.S. Bureau of Land Management. http://www.blm.gov/pgdata/etc/medialib/blm/wo/Planning_and_Renewable_Resources/adr_conflict_prevention.Par.44228.File.dat/ADR.pdf.
This agency document provides guidance, suggested strategies, and best practices for stakeholder engagement and dispute resolution processes.

USING INDICATORS EFFECTIVELY

Defining Indicators Clearly
Estimating Quantitative Indicators
Using Qualitative Measures Correctly

USING INDICATORS EFFECTIVELY

Authors *- Lynn Maguire and Dean Urban*

Throughout this assessment framework, ecosystem services are measured using benefit-relevant indicators. These indicators play an explicit role in conceptual diagrams and means-ends diagrams, multicriteria decision models, valuation techniques such as benefits transfer models, and various versions of matrices displaying the performance of alternative management options. Among the issues associated with development of indicators that are clearly defined and appropriate to the task at hand are (1) use of expert judgment, (2) narrative versus quantitative measures, and (3) incorporation of uncertainty in estimates.

Defining Indicators Clearly

The vast amount of practical guidance on developing environmental indicators suggests that good indicators should

- capture the intended ecological and social attribute as directly and precisely as possible,
- be quantifiable using efficient and cost-effective measures rather than expensive postprocessing in the lab or expensive field equipment,
- be free of observer bias (i.e., the indicator should be the same regardless of who estimates it),
- be repeatable over time, allowing monitoring to capture temporal trends, and
- be sensitive to changing conditions

Describing What Is to Be Measured, When and How

To evaluate management alternatives, each objective must be represented by a measurable quantity or quality that can be observed or predicted for each alternative—for example, the dollar value of agricultural products to represent the financial consequences of a wetlands management alternative. Agencies must take care to define measures clearly: Over what area? What kinds of agricultural products? What period of time? What price-reporting service?

Similarly, to express flood frequency clearly, the agency must answer these questions: What water level measured where and when constitutes a flood event? Over what time period will the agency express frequency (monthly, annually)? Over what period might it average the number of flood events (one year, 10 years)?

Answers to questions like these depend on the decision context: Whose concerns will be included? What measures are meaningful to them?

Using Proxy Measures

When alternatives' performance in achieving a particular objective might be difficult or expensive to measure—for example, in assessing the size of populations of at-risk species—proxy measures can be used instead. These measures should be relatively easy to observe and should correlate well with the measure that is really of interest. For example, if bird species diversity is a desired condition (ecological outcome), managers might index this diversity directly with a tally of species known to actually occur at a site. Alternatively, they might index the site in terms of (modeled) habitat suitability of the species of interest—whether those species occur on-site or not. The former is a direct measure but logistically expensive to collect over large regions, whereas the latter is a proxy measure but easier to estimate because the models can be implemented in a geographic information system.

The choice between direct and proxy measures is often dictated by logistics, but using proxy measures can have substantive consequences. For example, in the case of rare species, sites of known occurrences might be much more compelling than sites with potential habitat suitability, but the known occurrences might not be on sites with high predicted suitability. Choosing species occurrence as a direct measure of the existence of biodiversity while choosing suitable, but possibly unoccupied, habitat as a proxy measure could permit substitution of an unoccupied, but otherwise highly suitable, site for one that is currently occupied.

Acres of high-quality wetland habitat might be a proxy for populations of wetland-dependent species as well as a direct measure of wetland habitat. In this case, agencies should ensure that the weight put on acres of wetland relative to other measures reflects the fact that this proxy is doing double (or perhaps triple or quadruple) duty by standing in for other measures. Ways and reasons to weight measures are discussed below and in Maguire 2014.

Using Synthetic Indices

Synthetic indicators are increasingly popular as measures of ecological conditions that are naturally multivariate. Biotic integrity indices (IBIs) based on multiple species have been developed for aquatic macroinvertebrates, birds, wetland plants, and many other taxa. These indices are appealing because they can be weighted to emphasize particular ecological conditions (e.g., water quality) or species traits (e.g., rarity). In many cases, the synthetic indices are indirect or proxy measures (see above); for example, the aquatic macroinvertebrate IBI based on taxa sensitive to water quality is used because direct measures of water quality require lab analyses, making them difficult and expensive to collect.

Although appealing for many applications, synthetic indices can be problematic for management applications because it can be difficult to know how a specific management action might affect the index. For example, a simple response to management would require that all species used to compute an index would respond similarly to the management action. But changes in forest structure might improve habitat for some species while degrading habitat suitability for other species—and this information would be lost in a synthetic index. It is best to use synthetic indices as measures only if they can be interpreted unambiguously.

Quantitative Indicators and Their Estimation

An increasing array of formal models is being developed to predict production of ecosystem services in a variety of landscapes and to capture stakeholder preferences for varying levels of service provision. In this framework, the aim is to develop quantitative estimates for benefit-relevant indicators that capture both ecological and social information. Where formal models are not available, judicious use of expert opinion can help fill in gaps, thus avoiding omission of what is important but hard to observe or predict.

Using Expert Opinion

When quantitative models are not available, it can be tempting to simply omit an objective and its measure, but leaving out an important but troublesome measure is tantamount to declaring it to have no importance. A better tactic is to make use of expert opinion to fill in blanks in the matrix of alternatives and performance ratings. There are established procedures for choosing experts and eliciting their opinions (see references in Gregory et al. 2012). Implementing these procedures is exacting, and use of a consultant experienced in them is advised. Clearly defined measures and alternatives are necessary precursors to reliable use of expert opinion.

Expressing Uncertainty

Imprecise estimates of performance for one or more measures are typical. A common but undesirable way of dealing with uncertainty about performance is to create measurement scales that lump quantitative results into “bins,” such as 0–10 breeding pairs of a particular bird species, 11–20 breeding pairs, and so on. The problem with this tactic is that to unambiguously assign a particular result to the correct bin, i.e, to know that it belongs to the 0–10 bin and not to the 11–20 bin—the evaluator must know whether the number of breeding pairs is 10 or 11.

A better way to handle uncertainty about the number of breeding pairs is to express performance as a range of values in cells of the alternatives/attribute matrix. Instead of describing performance as falling into a predefined “bin,” such as 0–10, express performance as a range considered likely to encompass the true performance (e.g., 5–8 breeding pairs) or as a probability distribution (e.g., a mean of 6.5 with a standard deviation of 2). Then carry out the rest of the analysis by using the extremes of the range (or by sampling from the probability distribution) to see if that uncertainty affects the overall rating of alternatives.

A formal uncertainty analysis might be conducted by using a model in which a parameter (here, an indicator) is varied according to its error of estimation (e.g., plus and minus 1 standard error of the estimate). If the output of this model varies enough to alter the rank order of management alternatives, effort should be invested in refining the estimate to reduce uncertainty in the parameter estimate. Conducted for a set of indicators, this analysis would focus further effort on those parameters whose imprecision most affects the agency’s ability to discriminate reliably among management alternatives.

Using Categorical Measures Correctly

Some important features of an ecosystem are hard to express quantitatively. One solution is to use proxy measures, such as numbers of waterfowl seen in a day, to express the quality of a wildlife viewing experience. Another solution is to verbally define categories of performance, such as characterizing the viewing experience in terms of the observed species’ rarity or iconic stature for a particular region.

Transparently Defining Verbal Categories

Verbal categories must be defined clearly enough that they can be used consistently by different evaluators. Terms such as “high, medium, or low” or “good, fair, or poor” are commonly used to define performance but, without direction, different evaluators are likely to use different terms to describe the same conditions. To be fully transparent and unambiguous, measures need to have (a) a category for every condition likely to result from the management alternatives (so that evaluators will always be able to find a category for any observed or predicted result), (b) nonoverlapping categories (so that evaluators will have only one category for each possible result), and (c) sufficiently clear category definitions (so that any evaluator with access to the same information as other evaluators would use the same category to describe a given result). For quality of wildlife viewing in a region where there are two iconic species of waterfowl, a categorical measure could include these categories:

- no iconic species seen in a day’s viewing,
- one but not both iconic species seen in numbers fewer than 5,
- one but not both iconic species seen in numbers greater than or equal to 5,
- both iconic species seen in numbers fewer than 5,
- one iconic species seen in numbers fewer than 5 and the other in numbers greater than or equal to 5, and
- both iconic species seen in numbers greater than or equal to 5.

Any number of either iconic species seen in a day will fall into one and only one of these six categories, and any evaluator knowing the number of each species seen will assign a category consistently.

Note that these categories treat both species equally—they do not distinguish between numbers of species A seen and numbers of species B seen. Additional verbal categories would be needed to treat the species differently.

Distinguishing Ratings of Performance from Expressions of Relative Satisfaction with Different Levels of Performance

The verbal categories used to define categorical measures should not embed expressions of relative satisfaction such as “worst,” “medium,” and “best.” These categories are ambiguous: where does “worst” stop and “medium” begin? They also assume an order of preference that may not suit all users.

In the iconic species measure above, the six verbal categories are purposely not numbered because later in the analysis such numbers are likely to be misused as expressions of relative satisfaction (e.g., the category labeled “six” is assumed to be six times as desirable as the category labeled “one”). Without scrutiny, the agency cannot know whether that expression of satisfaction is warranted and whether it suits all users. Without further investigation it does not even know how to order the six viewing categories: is it better to see both species in numbers fewer than one or only one species but in numbers greater than five? Different users might answer that question differently. Agencies must be wary of categorical scales represented by numbers or ranks. It is better to create a shorthand code for lengthy verbal descriptions of categories by using a word or a letter, rather than a number.

All of these principles for developing good measurement scales to evaluate performance of alternative management actions apply equally to measures of ecosystem services (benefit-relevant indicators). Properly developed categorical measurement scales may be especially important when evaluating the production of intangible ecosystem goods and services, such as the existence value of unique species or landscapes.

Recommended Reading

Keeney, R.L., and R.S. Gregory. 2005. “Selecting Attributes the Measure the Achievement of Objectives.” *Operations Research* 53(1): 1–11.
This paper outlines theory and guidelines for selecting appropriate attributes to clarify objectives in a decision process.

SCENARIO ANALYSIS AND GREEN ACCOUNTING

DIFFERENT USES OF THE ECOSYSTEM SERVICES FRAMEWORK: SCENARIO ANALYSIS AND "GREEN ACCOUNTING"

Authors *- Brian Murray, Jeffrey Vincent, and George Van Houtven*

Reviewers *- James Boyd, Resources for the Future, and Glenn-Marie Lange, World Bank*

Ecosystem services (ES) can be incorporated into two distinct forms of assessment:

Scenario analysis (SA), or estimating *change* in ecosystem service values due to *change* in management, policy, natural resource condition, or project implementation.

Green accounting (GA), or developing jurisdictional (national, state, local) accounts that record the economic contributions of ecosystem services over time, through adjusted versions of macroeconomic measures (e.g., “comprehensive” or “inclusive” wealth, “adjusted” or “genuine” savings, and “green GDP”).

Scenario analysis is more straightforward conceptually and what the ES paradigm is most commonly used for in practice. It is also more directly relevant for land and resource managers. For this reason, the SA use of the ES framework has been the primary focus of this guidebook. However, there is considerable interest in green accounting, especially on the part of parties interested in broad-scale issues of placing ecosystem value “on an even footing” with traditional measures of economic performance such as gross domestic product (GDP) or national wealth accounts. Green accounting can provide more inclusive measures of national (or other jurisdictional) economic variables as they are tracked over time. These more inclusive measures can be used to inform how policy and other factors are affecting economic outcomes, once the value of natural capital and ecosystem services are taken into account.

Differences in Intent and Types of Decisions Informed

Both forms of ES assessment are intended to inform decisions, but they address different decisions and answer different questions, as illustrated in Table 9.

Table 9. Examples of questions addressed and decisions informed by form of ecosystem service assessment

SCENARIO ANALYSIS	
Questions:	**Decisions Informed:**
How would adopting an alternative management plan for Forest X affect the value of ecosystem services generated?	Which forest management plan to adopt
Which projects are most justified given a fixed budget, taking net ecosystem benefits into consideration?	Resource allocation decisions in implementing management plans under an agency budget constraint
How do ecosystem service losses contribute to economic damages from an oil spill?	Compensation decisions under natural resource damage assessments

GREEN ACCOUNTING	
Questions:	**Decisions Informed:**
How much does consideration of the economic value of ES change aggregate measures of an economy's performance, in particular its sustainability?	Economic and environmental policy design at broad (macro) levels, such as saving, investment, and industrial policies and climate, energy, and conservation policies
What portion of total GDP is attributable to ecosystem services that are hidden in income accounts as conventionally constructed?	Resource allocation decisions at the federal or other jurisdictional level; identification of potential opportunities for payment for ES programs
Is Country (state) A's forestry policy leading to an increase or decrease in the aggregate value of nonmarket ecosystem goods and services, in addition to timber production and consumption?	National (state-level) overall forest policy decisions, balancing commodity production versus other nonmarket ecosystem servicesPrivate decisions on commodity and ES investments

Scenario analysis is primarily focused on the consequences of geographically specific discrete actions such as management plans, narrowly targeted policies, projects, or natural resource damage assessment. The ES framework provides the theoretical and empirical foundation for determining the effect of these actions on human welfare. As discussed below, these assessments can be conducted *ex ante* to assess options or *ex post* to evaluate performance.

The idea of incorporating ecosystem services into jurisdictional accounts through green accounting, however, has different objectives. Jurisdictional accounting, such as national income and wealth accounting, and the aggregate measures that are based on it, such as gross domestic product (GDP), are developed to provide a barometer of the state of the economy rather than to assess the consequences of specific actions. Nevertheless, measures such as GDP are used to inform public policy and private sector decisions. For instance, if GDP growth is slow, policy makers may opt to take action to stimulate the economy. Investors may adjust their stock valuations to reflect any revised expectations of future economic growth. More inclusive measures, enabled by green accounting to consider aggregate measures of natural wealth (capital) and the value of ecosystem services produced, can alter these perceptions and strategies. Stimulative policies, for instance, might also include those that spur investment in natural capital.

Differences in Analytical Approach

Monetized or Non-Monetized Ecosystem Services?

The analytical approach to ES assessment covered in this guidebook focuses on scenario analysis to inform management decisions. The assessment framework presents a structured decision-making approach that incorporates the preferences and values of people in framing the assessment. When possible, the assessment also incorporates these preferences and values in comparisons of the ecological outcomes of alternative scenarios. Although the basic intent is to connect ecological outcomes to human preferences, the reported outcomes may or may not be monetized.

Green accounting generally seeks to incorporate ecosystem services into a monetary measure of production or consumption that acts as a signal of an economy's growth and health. This task ultimately involves the compilation of biophysical measures (either stocks or flows) in aggregate form for the jurisdiction of interest and, correspondingly, placement of a price on them to provide a monetary measure of the jurisdiction's natural wealth (stocks) or production (flows). This task is complicated by the fact that many natural stocks and flows do not have market prices, therefore proxies are often developed through nonmarket valuation approaches. In some cases, these proxies are the biophysical measures themselves.

Ex Ante or Ex Post?

The SA approach is often used to assess management options before a decision is made (*ex ante*), but it can be modified to examine events after they occur (*ex post*). In either case, analysts must define a baseline that reflects expectations of what would likely occur *ex ante* if the option is not chosen or what likely would have occurred *ex post* if the event of interest had not happened.

By comparison, green accounting is more typically used to track the status of natural capital stocks and flows from the past to the present (*ex post*) to reveal the status of and trends for the given jurisdiction. However, this does not mean that GA analysis is always backward looking. Ideally, the monetary values assigned to biophysical measures of natural capital stocks incorporate people's current perceptions about future demands, scarcity, and substitutes for the resource, whether they are revealed by market prices or by nonmarket valuation methods. Moreover, analysts can develop *ex ante* simulations of future natural capital stocks and flows using GA principles embedded in large integrated assessment models (e.g., computable general equilibrium models of the economy with biophysical production elements). These models can then be used to assess the future consequences of changes in policy regime, technology, environmental conditions such as climate change, or other large-scale factors.

Finally, if collected over a sufficiently long period and with sufficient geographical detail, the biophysical time series data used to track non-market production in green accounting can be used to empirically test the effect of policies on biophysical outcomes—a capability that can greatly improve forward-looking scenario analysis.

Green Accounting: Wealth versus Product Accounts

In the last few decades, the United Nations Statistics Division, the World Bank, the United Nations Environment Programme, and various other organizations have focused attention on the failure of conventional income and wealth accounts to incorporate the effects of natural resource depletion and environmental degradation.[162] Most of the attention has focused on adjustments to the wealth accounts—the "balance sheets"—to measure the value of natural capital ("natural capital accounting") as a component of a country's total national wealth. The underlying idea is that a sustained increase in a country's standard of living requires a sustained increase in its national wealth. If natural capital is depleted, other forms of capital need to be built up to offset its loss. For example, countries that draw down their mineral wealth could eventually suffer economic decline—a phenomenon known as the natural resource curse—if they do not invest the proceeds from the extraction and sale of minerals in other forms of capital (produced, human, or natural).[163] This idea draws on a sometimes-controversial (especially outside of economics) notion of sustainability, which allows for the substitution of natural capital for other forms of physical capital. This interpretation of sustainability has been articulated by Nobel laureate Robert Solow, drawing from others, including John Hartwick.[164]

Over time there has been an increase in estimated and published national wealth accounts. The World Bank's online World Development Indicators database provides estimates of annual changes in different forms of capital for most of the world's countries from the early 1970s to the present; these estimates were originally called "genuine savings," but more recently have been called "adjusted net savings."[165] In the case of natural capital, the Bank's estimates refer mainly to changes in the value of stocks of priced natural resources, such as

[162] For a technical review of fundamental economic concepts, see Weitzman, Martin L., *Income, Capital, and the Maximum Principle*(Cambridge, MA: Harvard University Press, 2003).

[163] Auty, R.M., *Sustaining Development in Mineral Economies: The Resource Curse Thesis* (London: Routledge, 1993).

[164] Solow, R.M, "Sustainability: An Economist's Perspective," *National Geographic Research and Exploration* 8 (1992): 10–21; Hartwick, J.M., "Intergenerational Equity and the Investing of Rents from Exhaustible Resources," *American Economic Review* 67(5)(1977): 972–74.

fossil fuel deposits, minerals, and timber: what the Millennium Ecosystem Assessment labels "provisioning resources."[166] The World Bank has also published periodic national estimates of "comprehensive wealth," snapshots of the total value of countries' wealth stocks, not just changes in those stocks (e.g., World Bank 2011);[167]UNEP has published an independent set of estimates, which it labels the "Inclusive Wealth Index".[168] The United Nation's System of Environmental Economic Accounting-Central Framework (SEEA-CF), which was issued in 2014, provides guidelines for wealth accounting related to these types of natural resources.[169]

Accounting for unpriced ecosystem services is at an earlier stage; appropriately, the UN's report on them is titled "Experimental Ecosystem Accounts."[170]The World Bank's Wealth Accounting and Valuation of Ecosystem Services (WAVES) program is sponsoring projects in selected countries to learn more about the practical challenges of designing and implementing ecosystem accounts.

The focus of official GA initiatives on wealth accounts instead of the better-known product accounts—which are the source of GDP estimates—reflects national accountants' reluctance to extend the boundary of product accounts beyond the market economy to areas in which prices and quantities are less easily measured. The nonmarket nature of most ecosystem services thus creates a roadblock for including them in national product accounts. National accountants and economists alike have long recognized that GDP is an incomplete measure of economic well-being. Economists have pointed out that GDP excludes many components that contribute to well-being (utility) such as goods and services that are not traded in markets, including uncompensated household work, leisure, and environmental public goods. National accountants accept this view but argue that the measurement of well-being is not the purpose of the accounts; instead, it is the measurement of the size of the exchange economy. Consequently, although an environmental economist would argue that a more complete measure of national economic well-being should include existence values provided by a country's protected area system, a national accountant would object to their inclusion because these values cannot be readily imputed from market exchanges between suppliers and users. Moreover, the omission of many ecosystem services from product accounts is not unique in making these accounts imperfect measures of economic welfare. As such, focusing on the inclusion of natural capital in wealth accounts may be a more feasible initial step toward green accounting than redefining national income to be a more complete measure of national well-being.

Product Accounting Issue: Whether to Count Both Intermediate and Final Products

On the product accounts side, green accounting faces certain challenges in developing measures that are theoretically consistent with national income accounting, the presumed benchmark. Boyd and Banzhaf (2007) take up several of the issues associated with GA in the product account.[171]They argue that symmetry with national income accounting requires separate accounting of ES quantities and prices. "Prices" need not be market prices—they often are not with ecosystem services—but consistency requires that they be some measure of marginal willingness to pay. They also argue that, for adjustments that affect the level of GDP, ES quantities should be final consumption quantities and not intermediate goods.[172]Although this is true, national accountants will be less opposed to accounting for ecosystem services that can be interpreted as intermediate goods, as such accounting does not require changing the production boundary of the accounts. The economic

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165 "The World Bank World Development Indicators," last accessed December 22, 2015, http://data.worldbank.org/data-catalog/world-development-indicators.

166 Millennium Ecosystem Assessment, *Ecosystems and Human Well-Being: Synthesis* (Washington, D.C.: Island Press, 2005), http://www.millenniumassessment.org/en/index.html.

167 World Bank, *The Changing Wealth of Nations* (Washington, D.C.: World Bank, 2011).

168 United Nations Environment Programme, "A New Balance Sheet for Nations: Launch of Sustainability Index that Looks Beyond GDP," June 17, 2012, UNEP News Centre, http://www.unep.org/newscentre/default.aspx?DocumentID=2688&ArticleID=9174.

169 United Nations, Statistics Division, *System of Environmental-Economic Accounting 2012: Central Framework* (New York: United Nations, 2014), http://unstats.un.org/unsd/envaccounting/pubs.asp.

170 United Nations, Statistics Division, *System of Environmental-Economic Accounting 2012: Experimental Ecosystem Accounting* (New York: United Nations, 2014),

171 Boyd, J., and S. Banzhaf, "What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units," *Ecological Economics* 63(203)(2007): 616–626.

impacts of many important ecosystem services, especially regulating services, are already reflected in market outcomes. For example, a loss of water purification services due to deforestation affects the water utility industry's treatment costs (which increase) and operating surpluses (which decrease). Green accounting can be used to make hidden flows of ecosystem services between industries and sectors in the product accounts explicit, without changing the overall level of GDP.

GA Aggregation Challenges

Another challenge is actually measuring ES (either intermediate or final) quantities and prices and aggregating them to the jurisdictional level of the accounting system. The challenges there are practical. Not all ecosystem services have been measured, and some that have been measured have not been priced. Moreover, as described throughout this guidebook, ecosystem services are spatially differentiated in that ES quantities are distributed across the landscape. Because they are generally not fungible with each other and have different values (prices) by location, aggregation is difficult, but not impossible. Where are missing, proxies may be needed. As Boyd and Banzhaf point out, these aggregation challenges are not too different than some of the other more conventional elements of national accounts such as restaurants, banking, and other services and are all practical problems under the development of economic indexes.[173]They have been solved for other goods and services, and presumably can be solved for ecosystem services if sufficient investment is made in the required data-collection processes.

The Need for a Common Classification Scheme

An ideal classification scheme for defining and tracking natural capital and ecosystem services would

- Distinguish between natural capital stocks (wealth) and flows (production),
- Distinguish between intermediate and final ecosystem services,
- Accommodate meaningful aggregation of natural capital stocks and ecosystem service flows (i.e., is hierarchical, comprehensive, and non-overlapping), and
- Is applicable to both scenario analysis and green accounting efforts.

The reasoning for the first three characteristics are provided in the previous section. The fourth characteristic—applicability of the classification scheme to both project-scale scenario analysis and jurisdictional GA measures—would be useful in ensuring consistency across decisions at different scales of operation, much like evaluation of individual projects for conventional investment decisions follows generally accepted accounting standards established at the jurisdictional level. A conceptual model for a broadly applicable ecosystem service classification system could draw from systems like the North American Industrial Classification System (NAICS)/North American Product Code System (NAPCS). These economic classification systems organize sectors and products logically from large industrial categories all the way down to detailed product codes within each sector. The model might also draw from existing environmental classification schemes, such as water quality classification schemes (e.g., waterways that are boatable, fishable, or swimmable).

Currently, no generally agreed-on classification scheme exists for natural capital and ecosystem services. Boyd and Banzhaf argue that initial efforts to classify ecosystem services, such as Daily (1997) and the Millennium Ecosystem Assessment (2005), although useful for describing the connections between ecosystems and human well-being, are limited as formal classification systems.[174]They certainly do not address all of four characteristics listed above. For instance, they do not attempt to distinguish final and intermediate ecosystem services.

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[172] To clarify the definition of an ES final good, Boyd and Banzhaf (2007) use the example of recreational fishing, which requires both a water body and a fish population. It also requires a boat and the fisher's time. One would count the water body itself and the fish population as the final ES. One would not count the fish caught as the final product, as that is produced by combining the ES with physical capital (boat) and human capital (labor and knowledge). One would also not include water quality or the underlying food web in the water body as the final ES as each of those contribute to the size of the fish population and thus are captured in that variable.

[173] Boyd, J., and S. Banzhaf, "What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units," *Ecological Economics* 63(203)(2007): 616–626.

Efforts are under way internationally to develop more formal ES classification systems. For example, the European Environment Agency is developing the Common International Classification of Ecosystem Services.[175] This system builds on the basic Millennium Ecosystem Assessment classification structure, which distinguishes among provisioning, cultural, regulating, and supporting service categories.[176] It is primarily being designed to support GA efforts, in particular the revision of the System of Environmental-Economic Accounting.

In the United States, the U.S. Environmental Protection Agency is developing ES classification systems. These systems include the Final Ecosystem Goods and Services Classification System (FEGS-CS) and the National Ecosystem Services Classification System (NESCS).[177] Given its mission as a regulatory agency, the EPA's main motivation for developing these classification systems is to support SA applications; however, the systems may also prove useful for GA applications. Both systems are focused on identifying and classifying final ecosystem services by linking specific environmental classes with specific categories of human benefits and uses. FEGS-CS approaches the human dimension through a detailed classification of final ES "beneficiaries," whereas NESCS includes classifications for both human "uses" and human "users" of ecosystem end-products.

Further development, application, and evaluation of these ES classification systems will be needed to determine whether a single unified system that addresses both SA and GA applications is feasible and appropriate.

[174] Boyd, J., and S. Banzhaf, "What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units," *Ecological Economics* 63(203)(2007): 616–626; Daily, G., *Nature's Services: Societal Dependence on Natural Ecosystems* (Washington, D.C.: Island Press: 1997); and Millennium Ecosystem Assessment, *Ecosystems and Human Well-Being: Synthesis* (Washington, D.C.: Island Press, 2005).

[175] Common International Classification of Ecosystem Services (CICES) 2013, http://cices.eu/.

[176] Millennium Ecosystem Assessment, *Ecosystems and Human Well-Being: Synthesis* (Washington, D.C.: Island Press, 2005), http://www.millenniumassessment.org/en/index.html.

[177] Landers, D.H., and A.M. Nahlik*, Final Ecosystem Goods and Services Classification System (FEGS)*, EPA/600/R-13/ORD-004914, 2013, http://ecosystemcommons.org/sites/default/files/fegs-cs_final_v_2_8a.pdf; Sinha, P., and G. Van Houtven, *National Ecosystem Services Classification System (NESCS): Framework Design and Policy Application*, draft report prepared for the U.S. Environmental Protection Agency, 2014

APPENDIX

**These figures are larger versions of two figures found in Section 3—Figure A-1 is the same as Figures 2, 7, and 10, and Figure A-2 is the same as Figure 8. These figures differ only slightly from one another. In Figure A-2, the boxes within the grey area contain ecosystem services identified in the conceptual mapping process.*

Figure A-1: Conceptual diagram with casual chains for an ecosystem services assessment of a forest management activity

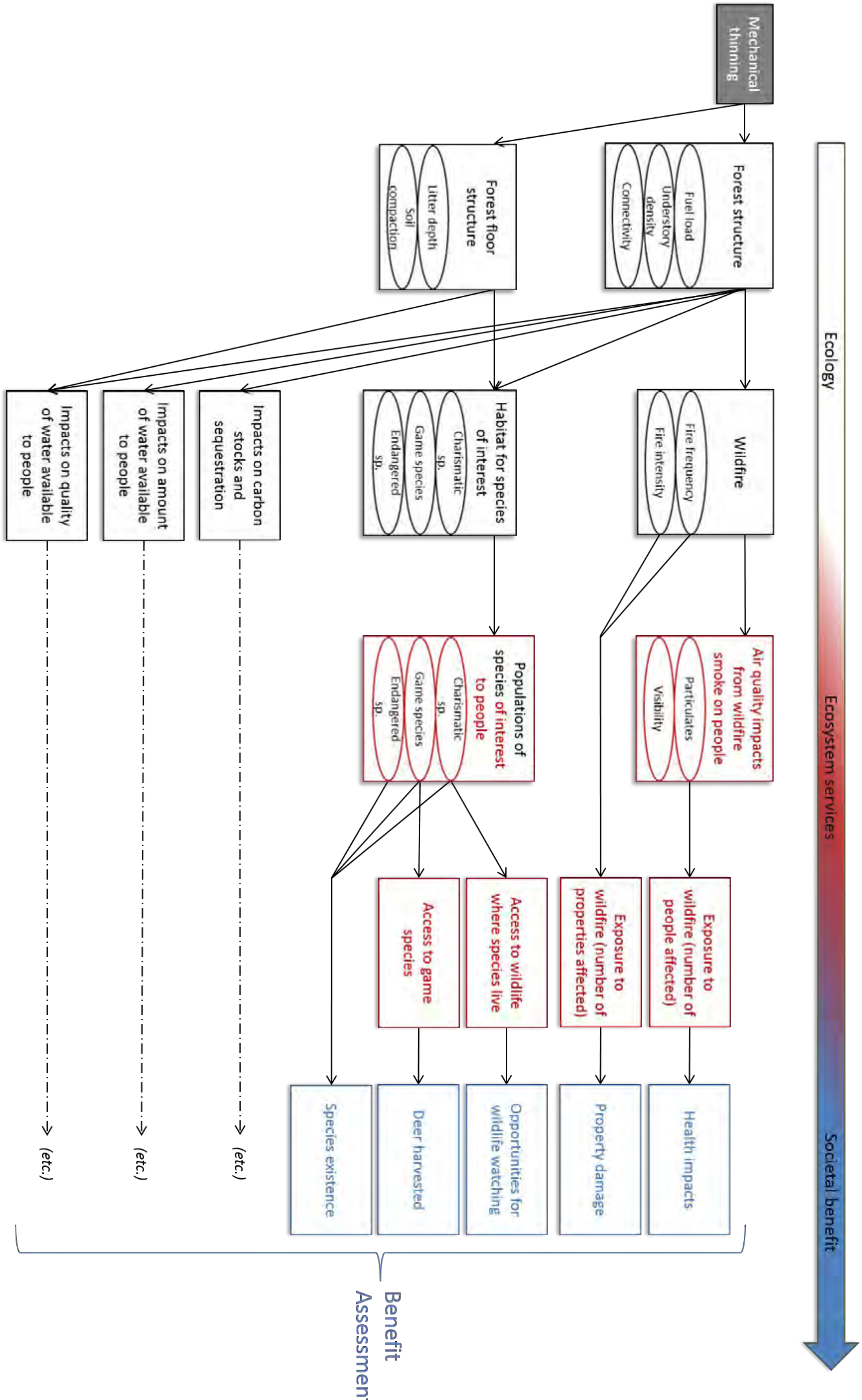
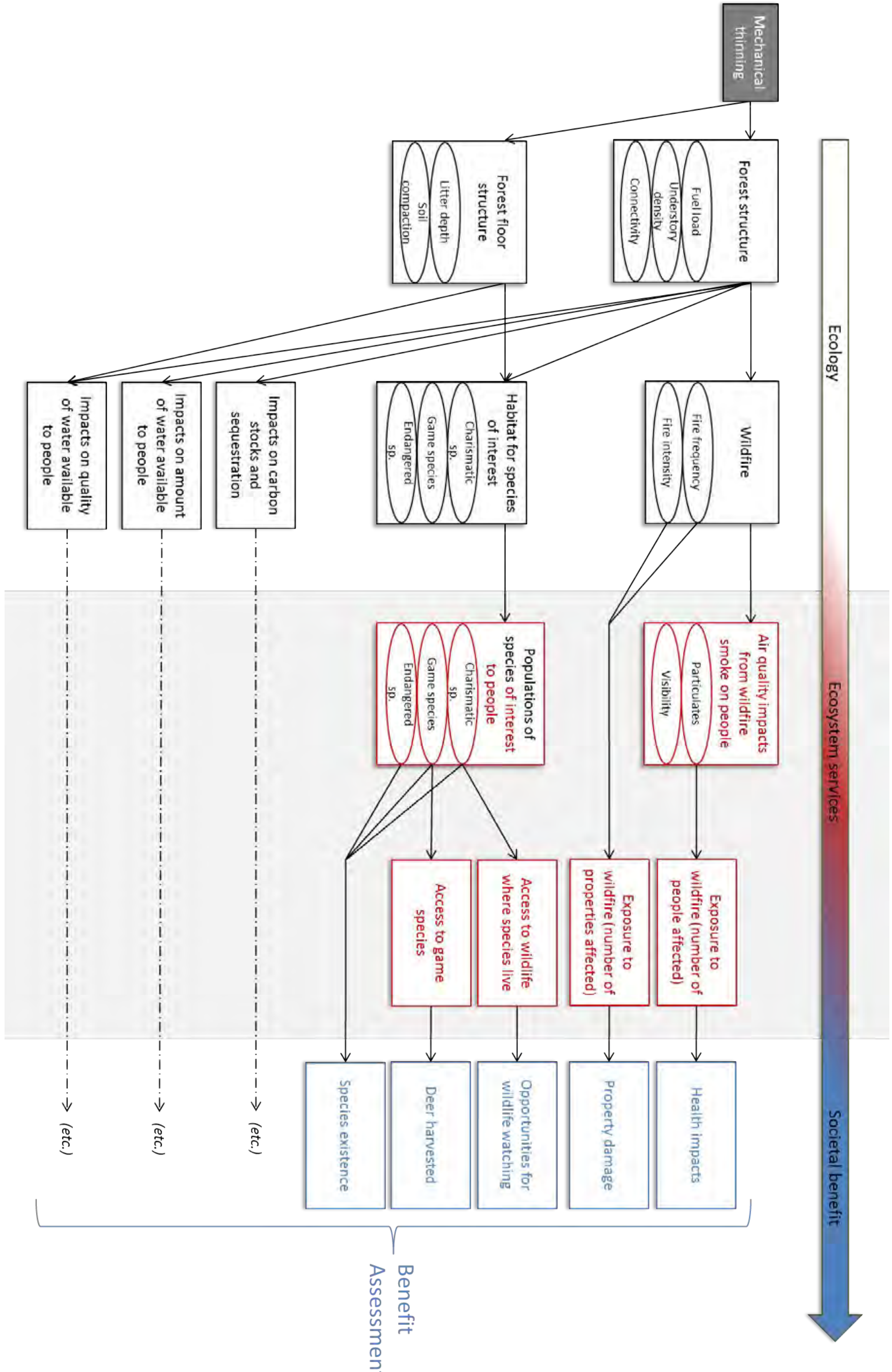


Figure A-2: Conceptual map of casual chains indicating possible outcomes of a forest fire activity



About the National Ecosystem Services Partnership

The National Ecosystem Services Partnership (NESP) engages both public and private individuals and organizations to enhance collaboration within the ecosystem services community and to strengthen coordination of policy and market implementation and research at the national level. The partnership is an initiative of Duke University's Nicholas Institute for Environmental Policy Solutions and was developed with support from the U.S. Environmental Protection Agency and with donations of expertise and time from many public and private institutions. The partnership is led by Lydia Olander, director of the Ecosystem Services Program at the Nicholas Institute, and draws on the expertise of federal agency staff, academics, NGO leaders, and ecosystem services management practitioners.

About the Nicholas Institute for Environmental Policy Solutions

Established in 2015, the Nicholas Institute for Environmental Policy Solutions at Duke University improves environmental policymaking worldwide through objective, fact-based reasearch in the areas of climate change, economics of limiting carbon pollution, emerging environmental markets, oceans governance and coastal management, and freshwater management. The Nicholas Institute is part of Duke University and its wider community of world-class scholars. This unique resource allows the Nicholas Institute's team of economists, scientists, lawyers, and policy experts not only to deliver timely, credible analyses to a wide variety of decision makers, but also to convene decision makers to reach a shared understanding of the century's most pressing environmental problems.

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