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Abstract Increasingly government agencies are seeking to quantify the outcomes of proposed policy options in terms of ecosystem service benefits, yet conflicting definitions and ad hoc approaches to measuring ecosystem services have created confusion regarding how to rigorously link ecological change to changes in human well-being. Here, we describe a step-by-step framework for producing ecological models and metrics that can effectively serve an economic-benefits assessment of a proposed change in policy or management. A focus of the framework is developing comparable units of ecosystem goods and services to support decision-making, even if outcomes cannot be monetized. Because the challenges to translating ecological changes to outcomes appropriate for economic analyses are many, we discuss examples that demonstrate practical methods and approaches to overcoming data limitations. The numerous difficult decisions that government agencies must make to fairly use and allocate natural resources provides ample opportunity for interdisciplinary teams of natural and social scientists to improve methods for quantifying changes in ecosystem services and their effects on human well-being. This framework is offered with the intent of promoting the success of such teams as they support managers in evaluating the equivalency of ecosystem service offsets and trades, establishing restoration and preservation priorities, and more generally, in developing environmental policy that effectively balances multiple perspectives.

Keywords Benefit transfer · Cost–benefit analysis · Ecological-economic modeling · Ecological indicators · Ecosystem services · Ecosystem valuation · Environmental economics · Environmental policy and management · Spatial analysis

Introduction

What is a fair and efficient way to decide whether ecosystem services should be protected or restored? Should tax dollars be used to purchase conservation easements and to restore habitat? Should businesses and consumers pay, through new taxes or regulatory requirements, for more environmental protection? Or, would money be better spent on education, infrastructure, or any of number of competing needs? Policy makers struggle with these questions and seek analyses to inform equitable and efficient decisions. Because protection and restoration of public ecosystem goods and services often requires individuals to forego private benefits, economic methods have been developed to compare both private and public costs and benefits of alternative policies as one type of input into decision-making. Numerous questions remain about the cost–benefit approach (CBA) including concerns about whether the approach adequately represents society's collective welfare rather than narrow self-interest (Turner 2007), whether economic valuation methods can adequately capture the complexity of people's preferences (Sugden 2005; Hanley and Shogren 2005), and whether we are counting the right things, including social justice, when we consider what constitutes public benefits (Norgaard 2010). Although CBA

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has limitations, it can be useful for clarifying some trade-offs; for this reason, government agencies are increasingly seeking to use CBA to quantify the outcomes of proposed policy options in terms of how actions will alter ecosystem service benefits (Ruhl and others 2009; United States Environmental Protection Agency [USEPA] 2009a).

Ecosystem goods and services (hereafter referred to as "ecosystem services") are the outputs of natural systems from which humans may derive benefits (National Research Council [NRC] 2005; Boyd and Banzhaf 2007). By this definition, ecosystem services require use or appreciation by people, although not all changes in ecosystem services can be demonstrated to result in substantial benefits or harms to people. The magnitude of gains and losses, as accounted for in CBA, depends on how much people rely on or desire those services (in a particular place and time) and whether substitutes are affordable and available. An ecosystem-services framework, therefore, provides a means to identify and assess how policies change ecosystem processes and outputs so they can be analyzed for effects on social well-being, including financial impacts and a broad array of effects on health and happiness.

The challenges to implementing an ecosystem services framework are many. First and foremost, the science to quantify links between ecological change and human welfare, although continuously evolving, remains incomplete (Carpenter and others 2006). Well-defined approaches are lacking for linking structural and functional changes in ecosystems to outcomes important to human well-being, even though conceptual models of the links may be well-accepted. Furthermore, economic-valuation approaches are often stretched to their limits when applied to ecosystem services. Most ecosystem services depend on a complex array of spatially heterogeneous conditions that challenge people's ability to develop well-informed preferences, and preferences for certain services (e.g., many of the "cultural" services) tend to reflect altruistic or other ethical motivations, which may be more appropriately considered by way of collective or deliberative processes (Spash 2008; Turner 2007).

Ecologists and economists, in particular, have much work ahead if joint models are to become a routine part of decision-making. Although considerable work has gone into developing well-integrated ecological and economic models (e.g., Brookshire and others 2010; Barbier and others 2008; Bockstael and others 1995; Johnston and others 2002; Milon and Shogren 1995; NRC 2005; Polasky and Segerson 2009; Tschirhart 2009; Turner and others 2008), some models that aim to capture benefits of ecosystem services raise concerns of methodological integrity. Questions generally center on whether models are consistent with fundamental principles of either ecology or economics and whether they are appropriate for making trade-offs. By working together to ensure model integrity, teams of ecologists, economists, and others can develop models that are rigorous from all perspectives.

It has been our experience in working in interdisciplinary teams that effective collaboration entails some degree of stepping out of our respective comfort zones. Ecologists will likely need to move away from saying, "Here's my ecological metric (e.g., the index of biotic integrity). Can you put a value on it?," and economists will likely need to move away from asking for unrealistically deterministic measures, such as, "Exactly how many fish kills will be avoided in an estuary if impervious surfaces in the watershed are reduced by 20%?" Instead, to manage data gaps and scientific uncertainty, interdisciplinary teams may need to ask, "How can we develop tools to better manage risk given that our ideal scientific information may not be achievable?" Such collaborations may involve a range of activities, beyond this framework, to engage stakeholders in deliberative collective approaches to managing risks (Spash 2008; Wilson and Howarth 2002; Ostrom and others 1999).

Because of the ongoing need for scientific information to inform decisions, natural and social scientists have substantial opportunities to bring current research to bear on decisions that seek to balance protection of ecosystem services and competing activities. Although many frameworks have been proposed for assessing ecosystem services or calculating risks and benefits, our framework seeks to promote a focus on "translational ecology" in which scientists work in teams that convey information in ways that the public and policy makers can use to inform meaningful action (Schlesinger 2010) while retaining sufficient rigor to evaluate trade-offs. More specifically, our objective in this article is to present a framework to promote management-relevant interdisciplinary models so that the best science can be used to inform effective policy. Effective research teams will likely involve multiple disciplines, but here, as a starting point, we focus on improving collaborations between economists and ecologists by clearly describing the steps that are necessary for rigorously linking ecological change to economic benefits.

This article begins with elaboration of the conceptual underpinnings of a joint ecological and economic modeling framework and then proceeds to describe a series of functional relations that support quantification of the economic benefits of environmental management actions. Although, in practice, the lines between these functional relations may be blurred, we break the process down to make the data and analytical needs more transparent. For each relation we provide examples of methods that are in use and discuss whether these methods are amenable to reaching an end goal of quantification of social welfare changes. Because to date the implementation of all aspects of the framework in one application is rare, we carry a hypothetical example throughout the article to represent the ideal application but offer examples of practical approaches and simplifications that are commonly used to make the framework tractable. The framework further clarifies how intermediate steps in the analysis can provide the best evidence of social welfare effects, even when those effects cannot be monetized.

Background: What the Framework Must Accomplish

An analysis framework to link policies to ecological change and ecological change to social benefits requires several key components. First, because we aim to support policy decisions, the framework should link a factor that can be controlled or influenced by policy (e.g., nutrient discharge limit, carbon tax, fishing quota) to a change in an ecosystem stressor (e.g., total nutrient delivery, carbon emissions) or to a direct change in the ecosystem (e.g., fish productivity). Second, the change in a stressor or ecosystem function will need to be linked to an outcome that matters to people. Finally, the framework should address how this change in outcome affects people's well-being.

Before we can move forward with a framework to measure how ecosystem services changes affect human well-being, we need to define what we mean by "outcomes that matter to people." The merits and rationales for various definitions of ecosystem services and benefit metrics have been widely debated in the literature (e.g. Wallace 2007, 2008; Boyd and Banzhaf 2007; Brown and others 2007; Costanza 2008; Fisher and Turner 2008; Fisher and others 2009; Turner and others 2008; de Groot and others 2002; Millennium Ecosystem Assessment 2005; Wainger and others 2001). Our definition of ecosystem services is based on economic utility theory (Mansfield and Yohe 2000; NRC 2005; Blaug 1997) because it ensures that benefits and harms reflect outcomes that people use or value. A utilitarian framework includes both use and nonuse values, thereby encompassing a wide range of outcomes for which people can express preferences. Although use values require either direct or indirect interaction with the good or service, nonuse (or passive-use) values include preferences for preserving the existence of ecosystems, retaining the option to use them in the future, or holding them in trust for future generations (i.e., existence, option, and bequest values). Risk-reduction metrics, such as "improved resilience of a rare ecosystem," may be evaluated as nonuse values because people often are willing to pay to decrease the risk of a rare ecosystem disappearing.

In contrast to some other definitions of ecosystem services (Daily 1997; de Groot and others 2002; Ehrlich and Ehrlich 1981; Millennium Ecosystem Assessment 2005), we exclude metrics of basic ecological functions and processes (e.g., nutrient cycling) from our definition of ecosystem services because people do not have well-established preferences for these types of outcomes (Diamond and Hausman 1994). Without the ability to evaluate people's preferences, such metrics cannot be used to directly measure social benefits or make trade-offs among competing values. In addition, using ecological functions and processes to measure benefits can easily lead to double-counting of benefits if these functional outcomes are added to the goods and services that result from the functions. Ecological functions are, nonetheless, critical to understanding socially relevant outcomes.

We further distinguish between quantitative metrics of ecosystem goods and services and the benefits derived from those goods and services. According to economic welfare theory, we define benefits as the social welfare resulting from the use or enjoyment of ecosystem services by people, where social welfare is an aggregate measure of what people are willing to give up (i.e., willing to pay) in exchange for something they value. However, we recognize that in some contexts, social well-being is broadly defined to include financial impacts and a wide array of effects on health, happiness, and social justice. The primary difference between service and benefit metrics is that demonstrating a change in supply of an ecosystem good or service does not automatically imply a change in social benefits. Rather, to understand the potential for benefits or harm from a change in supply, the likely changes in demand must also be evaluated to estimate the number of people affected and their willingness and ability to substitute for losses or otherwise adapt to changes.

Our framework has the primary goal of promoting the quantification of social benefits through rigorous interdisciplinary research and a secondary goal of enhancing the information content of outcome measures used when benefits cannot be fully quantified or monetized. For example, instead of simply measuring the nitrogen concentration in an estuary (low information content for decisions), we would prefer to know what percent of the estuary no longer supports a desirable species, such as submerged aquatic vegetation (SAV), as a result of excess nitrogen that limits habitat quality (intermediate information content for decisions). To move even closer to a measure that an economist can use to demonstrate welfare impacts, we would need to relate a change in nitrogen concentration and/or SAV distribution to a change in something that would be valuable to a range of people, such as abundance of a recreational or commercial fish species (e.g., crabs) or shoreline erosion control (high information content). We recognize that ecologists use environmental and ecological metrics to measure a wide range of system conditions, processes, and functions; here we stress the need to produce a *subset* of measures that represent meaningful outcomes, which we define as those outcomes that directly affect people's well-being, and thereby communicate the social welfare impacts from changes in natural systems.

The Benefits Assessment Framework

From this point forward, we describe the specific components of a framework that links ecological and economic models to evaluate the benefits of management options. The framework reflects well-accepted concepts and practices used in economics and risk assessment (e.g., Brown and others 2007; NRC 2005; Harwell and others 1992) and therefore is not intended to provide a completely new framework. Rather, the intent is to use this framework and illustrative examples to clarify some concepts, in particular those that are used inconsistently in the ecosystem services literature, to enhance the ability of interdisciplinary teams to measure social welfare using widely accepted and time-tested principles. By clearly communicating concepts from economics that have been developed during a half century of work on natural resource issues, and illustrating both ideal application and the types of real-world simplifications that preserve the intent of a scientifically sound analysis of trade-offs, we hope to provide a common language for teams of biophysical and economic researchers.

The metrics and functions that link a change in human actions (i.e., an outcome of a management action) to a resulting change in social welfare are shown in Fig. 1. The boxes contain metrics that progress from biophysical measures that capture significant ecological outcomes (top box) to metrics that *suggest* social benefits (middle boxes) and finally to metrics that *explicitly measure* social benefits (bottom box). The arrows represent the four main functions or models needed to produce one endpoint from another: (1) impact function (IF), (2) response function (RF), (3) ecoservice production function (EPF), and (4) benefits/damage function. IFs connect human actions to increases or decreases in stressors, whereas RFs are used to show how changes in stressors result in meaningful ecological changes. Next, EPFs translate ecological changes into outcomes that people use or value. Finally, the benefit function (BF) demonstrates what people would be willing to pay to achieve a gain or avoid a loss in an ecosystem service or suggests a relative magnitude of social value when willingness-to-pay is not measurable (see Fig. 1 caption for an example).

The key distinction between the ecological metrics (in the third box from the top) and ecosystem service metrics

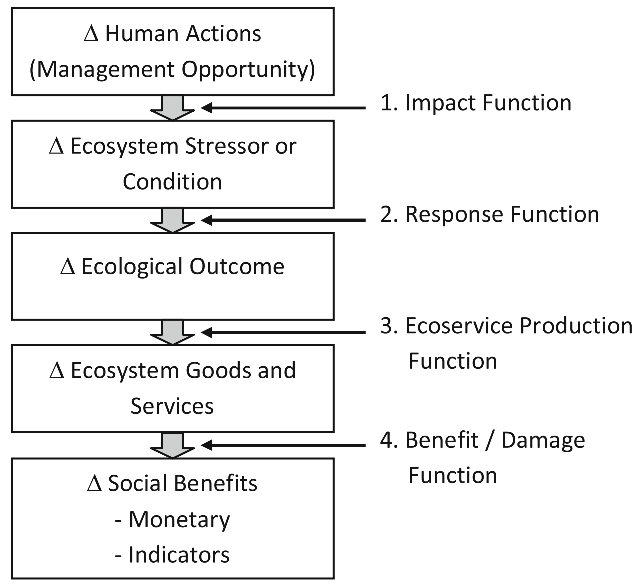

Fig. 1 Framework to estimate economic benefits of a management change. To link a change in behavior to the loss or gain in social welfare derived from ecosystem services requires multiple steps. First, the change in behavior may need to be translated into a change in an ecosystem stressor. For example, the IF may represent how a change in lawn-care practices translates into a change in nutrient runoff to streams. Second, the positive or negative change in a potential ecosystem stressor (e.g., nutrient runoff) is then related to a change in an ecological outcome of interest (e.g., HAB extent and frequency) by means of an RF. Third, the ecological outcome (HAB extent and frequency) is related to an impact on the ecosystem services that people value (e.g., safe swimming opportunities or shellfish harvesting) through an EPF. The EPF models services by location, such as swimming opportunities, by evaluating whether people are using the site for swimming. Finally, the change in an ecosystem service (swimming opportunities) is quantified in terms of lost social welfare (i.e., value) through a benefit/damage function that considers how many people are affected by the ecological changes and how their well-being changes when swimming opportunities change

(in the fourth box from the top) is that the ecological metrics are chosen to indicate a potential change in human welfare (e.g., change in water-bacteria levels), whereas ecosystem service metrics represent a change in the quality or quantity of an end use of the system (e.g., recreational swimming) after likely use of the service in a given location or time frame has been established. Note that for some services, particularly nonuse services (e.g., existence of polar bears), ecological and ecosystem service measures may be the same.

Readers familiar with the literature on ecosystem services will see that we have substituted ecoservice production function for the more commonly used term "ecological production function" (Roughgarden 1997; Polasky 2008; Nelson and others 2009). The ecological production function usually defines ecosystem services in terms of biophysical measures only, whereas the EPF aims to establish that complementary inputs, such as public access, are present to allow use in a given location. The potential for a service to be used (or evidence that a service is appreciated in the case of nonuse services) is what creates an ecosystem service in our definition because it affects whether people will value a change in that service in a given location. In addition, the EPF establishes whether ecological quality is sufficient to support particular uses of a good or service, such as whether water is safe for swimming. In sum, the EPF highlights the nonecological nature of some necessary inputs and establishes qualities needed to measure ecosystem services as we define them here.

IF: Defines the Expected Effect of a Human Behavior on Ecosystem Stressors

The IF describes any increase or decrease in ecosystem stressors in response to a change in human behavior and includes relevant temporal and spatial variability. Stressors are changes in biological or physical structures, features, flows, or processes that can decrease performance of the natural system, such as increased nutrient or sediment loads to surface waters, increases in invasive species cover, changes in hydrologic regime, or loss of tree canopy. Although the IF can model either beneficial or detrimental changes, we characterize its outputs as "stressors" because environmental policy and management is typically aimed at ameliorating human stresses on the natural environment. Behaviors that may change include fertilizer application rates, tilling practices, road width, house and lot sizes, vegetation plantings, siting of hiking trails, water withdrawal rates, and fish-harvesting rates.

Directly incorporating a potential behavior change or "management opportunity" into the system of linked equations or models is a critical step in implementing the framework because it allows the system of models to demonstrate what types and levels of behavior change may be needed to achieve a desired outcome. Consider a hypothetical case study in which the goal is to enhance social welfare derived from an estuarine system by improving the safety of water contact and fish consumption. The IF is the first step in relating something a manager can control to these desirable outcomes. If the safety of water contact and fish consumption is being degraded by an excess of nutrients or contaminants, the IF might address the question: How will a change in farm-tillage practices change the nutrient and contaminant loads to streams? Such questions can depend on many social, technical, and biophysical factors, such as whether farmers follow technical recommendations, type of fertilizer used, soils, geology, hydrology, amount of agricultural land, or presence of farm animals, reservoirs, or steep slopes within the basin of interest. Therefore, the IF may need to incorporate many factors to suggest likely outcomes of a given action.

In at least two types of cases, the IF is not needed. The first is where there is no proposed change but the system continues to change through natural processes. For example, allowing an invasive plant to spread unimpeded would not be modeled as an IF but instead as the stressor-response relation between invasive species cover and the ecological endpoint of interest, such as loss of rare plants. Such "no-action" analyses are often needed to establish the baseline case for examining the benefits of a management action. The second case in which the IF is not needed is when human actions directly impact the ecological endpoint of interest, such as when people harvest fish or control water availability using dams.

Depending on data availability and the needs of decision makers, an expected change in a system stressor as a result of human behavior may be represented by a simple relation (e.g., an average nutrient removal efficiency of an installed best management practice) or as a complex bioeconomic model, such as one that evaluates the effect of land-development choices on changes in nutrient and sediment fluxes while also considering the synergistic or antagonistic effects of myriad other diffuse activities or stressors within a watershed (e.g., Cerco and Noel 2004). In between these two ends of the complexity spectrum are many options of intermediate complexity, such as empirical models or weighted indices, that can be used to estimate changes in stressors or conditions with varying degrees of specificity (see Fig. 2).

The modeling spectrum exists because model development can be limited by availability of resources or by availability of data and scientific understanding. In some cases, conceptual models (e.g., more energy use tends to degrade multiple environmental outcomes) may be sufficient to motivate action, particularly if the adoption costs are low or negative (i.e., the actions save money). When quantitative models are desirable but analysis resources are limited, pre-existing generalizable models are a potentially attractive option. Such off-the-shelf models are an inexpensive approach to estimating a coarse level of system response. They generally consist of one or more equations that represent general biophysical conditions and processes and can be parameterized for different systems. Although such models will always be desirable for their cost-effectiveness, their rigor and applicability is likely to vary dramatically by setting and analysis question. Specifically, off-the-shelf models may not be sensitive to the proposed management action or system change and may not adequately represent local conditions. For example, Boomer and others (2008) found that accuracy of the Revised Universal Soil Loss Equation, which is widely used to estimate soil erosion in response to farm management, was poor in some areas of the mid-Atlantic.

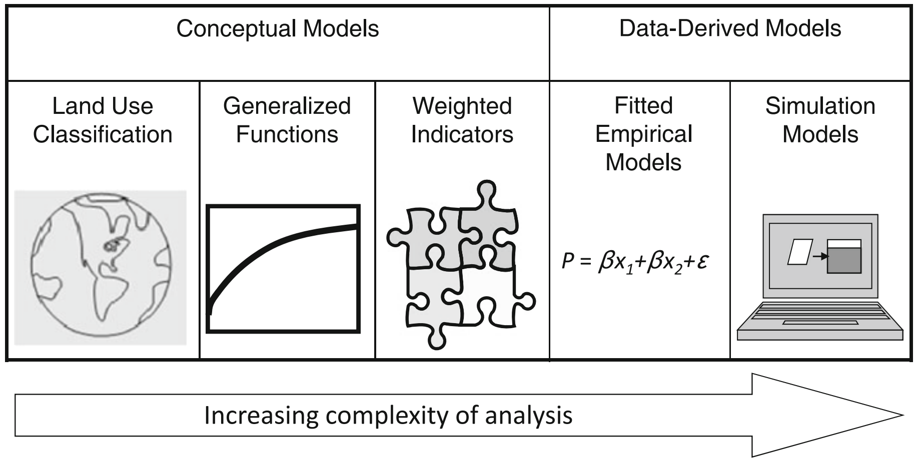

Fig. 2 Modeling complexity spectrum. A variety of techniques that rely on conceptual models or data analysis are used to generate the functional relations used in the framework. Approaches in use span a complexity spectrum from simple classification schemes that associate land-cover types with impacts or ecological outcomes to complex simulation models that incorporate multiple system dynamics to project system changes. In between these end-members are approaches that use general functions or theoretical models, such as individual indicators, weighted indices, and fitted empirical models, to suggest likely changes

Socioeconomic Studies may Inform the IF

As an aside, we note that the question of how to achieve behavior changes through various actions, such as regulatory or voluntary approaches, is an area of economics and other social sciences that is often overlooked when discussing ecosystem services and will often be a preliminary step to applying this framework. Socioeconomic models aimed at projecting how people might respond to a regulatory or policy change (e.g., use prohibitions, taxes or subsidies, or establishment of caps and trading markets) can identify policies that are most likely to be effective. For example, economic models can suggest how high a fertilizer tax must be before decreases in fertilizer use are likely to be noticeable. Such models might examine the profit margins of farmers, the change in yield with a change in fertilizer application, and the nature of global commodity markets to understand how a tax may change production costs and therefore encourage decreased application rates. Similarly, social scientists may be able to suggest the relative effectiveness of attempts to modify behavior through an advertising campaign versus peer-to-peer social marketing. Such models can both identify opportunities and show the potential for unintended negative consequences that may arise from a policy (e.g., Heberling and others 2010).

Examples of IFs from the Literature

In our estuarine-restoration case study, it is reasonable to assume that agricultural producers will be asked to contribute to decreasing nutrient and contaminant delivery to the estuary to promote ecosystem services derived from that system. However, farmers may be reluctant to engage in some practices that decrease nutrients for fear that crop yields or farm-management costs will be adversely affected. Therefore, to understand management effectiveness, both the ecological and the agricultural production impacts must be considered.

Two studies demonstrate the potentially competing effects from adopting a management practice that could decrease nutrient loads to an estuary. First, Tonitto and others (2006) developed a meta-analysis that suggested that adoption of cover crops substantially decreases nitrogen (N) leaching to groundwater, while either maintaining available soil N or decreasing it somewhat, depending on cover crop and fertilization choices. They therefore created two important impact relations: one between the management action of adopting cover crops and the relevant outcome of N leaching, which has the potential to contribute to degradation of the estuary, and the other between cover crops and soil N availability, which is an input to crop production. Their study carried the soil N results through to the crop production RF to demonstrate that yields were either unaffected (nonlegume cover crop with recommended fertilizer application rate) or were decreased by 10% on average (legume cover crop used as "green manure" to fertilize crop). Further work would be needed to create a more generalizable IF to relate different acreages of cover crop adoption in a watershed to N flux in surface and subsurface waters to the estuary, but a summary of plot-scale nutrient leaching effects is a step toward that goal.

Another study developed two IFs that addressed competing effects of land-cover management by comparing a variety of land-management practices aimed at restoring natural vegetation in agriculturally dominated landscapes. Using field results, Freemark and others (2002) demonstrated a relation between increased landscape complexity (i.e., converting farmland to natural cover) and average number of weedy plant species within a landscape, an impact measure related to potential crop productivity losses. A second function was used to demonstrate a relation between landscape complexity and native plant species richness, an impact relevant to a variety of habitat-related ecological outcomes. They found that as landscape complexity increased (i.e., with a wider variety of natural land types and larger patches), the average number of weedy plant species initially increased but then leveled off. In contrast, the average number of native plant species increased steadily as naturalness and size of vegetated patches increased. Because the rate of increase of weedy species decreased while the rate of increase of native species remained constant, their results suggest that the positive effects on native species richness could potentially be greater than harms from weedy species once landscape complexity is sufficiently high. However, to confirm this suggested effect requires moving to the next part of the analysis, i.e., the RF, to quantify both the impacts of weedy species on crops, and therefore profits, and the benefits generated by the increase in native species richness.

RF: Estimate Expected Changes in Ecological Outcomes When Stressors Change

In its simplest form, the RF relates a stressor change that is an output from an IF (or the result of a direct activity, such as water withdrawals) to a change in the quality and/or quantity of an ecological outcome of interest. As with the IF, models can range from the simple to complex (Fig. 2) but will ideally include some common elements, including thoughtful selection of stressor and response variables and functional forms (Table 1). The ecological outcome measure is defined as a biophysical metric that can be conceptually related to the provision of one or more ecosystem goods or services. The choice of this metric will determine the usefulness of the RF for later benefits assessment.

The RF, as defined here, captures both the ecosystem degradation and improvement that may occur as a result of an increase or decrease in a stressor. However, in many degraded systems, it is thought that the stressor-recovery relation diverges significantly from the stressor-response relation (Kelly and Harwell 1990; Niemi and others 1990; Jackson and Hobbs 2009) because the amount of stress that must be removed to get back to a desirable system state differs from the amount of stress that was tolerated before degradation. In these cases, the empirical modeler must choose whether a stressor-response or a stressor-recovery relation is most useful and which type of function is better supported by available data.

Not all ecosystem services depend on having ecosystems that function close to their natural condition. For example, the enjoyment of open-space amenities (e.g., by joggers, dog walkers, and bicyclists) may not depend on many aspects of ecosystem condition, such as native plant diversity. However, for services that do depend on ecological quality, ecological changes that affect quantity or quality of functions must be captured in the RF model in order for there to be any chance of capturing the value of these changes in the benefits assessment.

Table 1 Summary of steps, with examples, in developing the stressor-response function

Steps	Description	Examples
1	Identify the ecological outcome metric that affects the quality or quantity of ecosystem goods or services (i.e., the public-friendly dependent variable that will respond to a change in stress)	1. Extent and frequency of harmful algal blooms (related to services of protecting human health and fish harvesting) 2. Groundwater storage (in an arid climate, related to water supply for residential or industrial use)
2	Identify stressor variables that are either a direct management opportunity or provide a link to one by way of the IF (i.e., independent variables that will change when human behavior changes)	1. Output from IF: nutrient flux to streams 2. Direct management opportunity: groundwater pumping rates
3	Consider whether the system can be robustly represented with a simple model or whether a complex multistressor framework is needed to represent necessary system drivers (i.e., choose the type and complexity of the RF model)	1. Empirical relation with a limited set of variables 2. A complex system of equations that is solved using simulation methods
4	If using models that incorporate best professional judgment (conceptual models, simulation models), consider the potential for nonlinear relations in the function over different levels of degradation	1. Relation is linear regardless of average in-stream nutrient concentrations 2. Relation is logistic showing greatest slope (and response to changing conditions) at an intermediate level of average nutrient concentration

Choosing the Outcome to Measure in the RF

Much has been written about criteria for choosing ecological indicators that reflect key ecosystem properties and dynamics (Palmer and others 2005; Niemi and McDonald 2004; Dale and Beyeler 2001). Here, we emphasize that choosing appropriate ecological or biophysical outcome metrics (hereafter described as "ecological outcome metrics") for the RF means identifying the response variables that will support the next steps of quantifying ecosystem services and benefits in units that can be compared across ecosystem types and locations.

Ideal ecological outcome metrics reflect quantity or quality changes that users of the resulting ecosystem services appreciate and, as a result, will immediately communicate why an ecological change is important. Returning to our hypothetical case study example in which the goal is to improve the safety of water contact and fish consumption in an estuary, we might consider an ecological outcome metric of N concentration in the estuary because we know conceptually that excess N increases the risk of harmful algal blooms (HABs) and can degrade habitat for some species of fish by decreasing dissolved oxygen and killing sea grasses that serve as nurseries. However, as the length of the last sentence indicates, N concentration is not readily understood by most people as an important outcome because it needs further translation to link it to potential ecosystem services.

Instead, the ecological outcome metrics used in RFs will ideally go beyond such basic biophysical outputs and directly capture changes in outcomes of direct relevance. In our example, the HAB risk endpoint is more useful as a response metric than nutrient concentration because it has direct relevance to ecosystem services (e.g., recreational fishing and swimming, aesthetic benefits) that are only produced by water bodies that are safe for human contact and/or are free of other unpleasant effects associated with fish kills or contamination of fish and shellfish (i.e., disamenities). However, this RF will only be useful if it relates HABs to nutrient concentrations in the estuary and, in turn, if an IF relates nutrient concentrations to sources of nutrient loads from the watershed, to maintain the connection between HABs and the management opportunity.

In contrast to this ideal of a response metric that can be immediately perceived as important by the public, many ecological metrics are designed to measure either deviation from a reference condition or degree of stress (e.g., percent cover of invasive species). A "public-friendly" outcome metric will not only require minimal explanation but will also identify what is important about the ecological change given how people use or value the ecosystem in *a particular location*. For example, understanding that the function of groundwater storage is important in an area where water is scarce can help researchers generate a better metric of harm than percent cover of a nonnative plant. If a nonnative invasive plant has greater evapotranspiration rates than native vegetation, it will be meaningful to represent the effect of the invasion in terms of loss of water supply (e.g., Le Maitre and others 2002; Zavaleta 2000).

Implicit in this discussion of outcome metrics is the notion that the selection of which metrics to use and how to calculate them is not "value-free" and that some subjective judgments are not only appropriate but necessary. Consider that greater biodiversity is often assumed to be more desirable to people. However, what if the greater biodiversity of a site is produced due to the presence of numerous nonnative invasive species? In some cases, number or richness of *desirable* species, rather than *all* species, may be a preferred measure of what people value (e.g., "biological distinctiveness" as described in Ricketts and others 1999). The outcome metrics are thus public friendly when they are compatible with measures of social value.

Clearly, the desire for public-friendly response metrics in the RF must sometimes be weighed against the need for scientific rigor. A response metric will not be useful if its relation to stressors is not supported by empirical data or strong conceptual models, although the decision context will usually dictate how strong the science must be (e.g., court cases vs. voluntary programs). In general, ecological metrics will be more useful for decision-making if they demonstrate that desired outcomes are produced as opposed to implying that they are produced. For example, data that demonstrate habitat use or breeding success by a species of interest, such as Bonter and others' (2009) use of radar data to track which wetlands were most used by migrating birds, better support evaluation of nonuse or recreational services than habitat suitability inferred from plant condition. That is not to say that more cost-effective indicators cannot be used, only that they should be empirically demonstrated to represent relevant outcomes (Bockstaller and Girardin [2003] discuss methods for validating ecological indicators) to clearly demonstrate cause and effect.

Narrowing the Set of Outcomes to Measure

Because it is seldom possible to model all of the potential changes to all ecosystem components, a useful strategy for narrowing the set of metrics is to focus on the outcomes or services that define the key trade-offs of a management action. For example, if an action, such as a fertilizer tax, decreases fertilizer application on farms, some portion of crop yields might be forfeited to decrease the risk of HABs in a receiving waterbody. These outcomes of crop yields and decreased HABs are so-called "joint products" because they both respond to changes in nutrients. They also partially compete with each other because they respond inversely to fertilizer-application rates. Building RFs for these two outcomes will promote the ability to evaluate a socially optimal level of fertilizer application. (See Freeman [2003], Randall [2002], and Wossink and Swinton [2007] for further discussions of joint production as well as Nelson and others [2008] for an example of using RFs to evaluate optimal production of competing services.)

When additional services need to be included to represent key stakeholder concerns, the complete list of outcomes of interest may be narrowed by considering what is jointly produced and complementary. Complementarity implies that increasing production of one outcome increases production of another. For example, planting riparian buffers in natural vegetation may jointly enhance habitat for songbirds, herpetofauna, and fish. Thus, these outcomes all respond positively to the input of increased natural vegetation in riparian zones. As a result, a subset of complementary services, particularly those that resonate with the greatest number of people or that can be shown to have the highest monetary values, may be sufficient to represent a suite of co-occurring services.

Stakeholder input will be invaluable for choosing outcomes that democratically represent competing interests (Dietz and Stern 2008). Methods to engage stakeholders in defining priorities or ranking risk include focus groups or surveys (mail, phone, and Web), whereas other participatory activities, such as advisory boards and group-modeling exercises, may be used to engage stakeholders throughout a management decision process (e.g., Miller and others 2010; Tidwell and Van Den Brink 2008; van den Belt and others 1998). Different approaches are applicable at different points in the analysis framework and for different types of management processes (Randhir and Shriver 2009; Grimble and Wellard 1997; Reed 2008; Morgan and others 2000).

Creating RFs Using Expert Judgment

When empirical data are not available to create the RF, it may be desirable to use expert judgment to build models to inform decisions, particularly if techniques can be applied to minimize bias in responses. For example, when Johnston and others (2002) sought to support restoration decisions by demonstrating the benefits of coastal wetland restoration in terms of habitat benefits for fish and birds, they found that restoration-recovery functions were not available. The investigators turned to a survey approach in which they used accepted survey techniques to first select site and landscape variables for testing and then to design and implement a survey to elicit the best professional judgment of a group of wetland scientists and practitioners. Using survey responses, they were able to combine the knowledge of many scientists to create empirical models relating combinations of site and landscape characteristics in salt marshes (e.g., vegetation type and composition, water features, upland land uses, and other nearby features) to habitat potential for several categories of bird and fish species. The drawback of this approach is that eliciting expert judgment in a manner that minimizes bias takes significant time and effort. Nevertheless, the approach can be worth the effort when targeted to fill key data and knowledge gaps, although the acceptability of using expert judgment will vary by the decision context.

Benefit Assessments will be Highly Sensitive to Shape (Linear vs. Nonlinear) of RFs

The effectiveness and benefits of management actions will be highly sensitive to the magnitude of response due to a change. If we consider an empirical example of the RF, this can be thought of as: The steeper the slope of the relation between the stressors and response over the range of the proposed intervention, the more responsive the system will be in terms of estimated beneficial outcomes. Most importantly, nonlinearity in this relation can mean that there are increasing or decreasing returns to management effort over different ranges of stress. Discontinuities, caused by thresholds or tipping points, can suggest that restoration will be ineffective under some conditions or that constraints (e.g., safe minimum standards) are needed to avoid crossing potential thresholds.

We highlight these issues because we have seen how common assumptions for the RF can lead to recommendations for management actions that produce unintended results. A linear RF assumes that the same beneficial change occurs for every increment of stress reduction regardless of system condition. In contrast, a nonlinear RF can capture either (1) a situation where, if a system is profoundly degraded, removing a given level of stress may have little functional effect or (2) a situation in a less degraded system, where removing the same stress may move the system to a state where it provides substantially more ecological function. Figure 3 represents the case of a nonlinear response relation.

Returning to our hypothetical case study, the curve in Fig. 3 may represent the relationship between reduced nutrient delivery (increased restoration) and the ecological outcome of reduced HAB extent and frequency (increased ecological function) for multiple estuaries with different total annual nutrient loads (or one estuary at different points in time as land is converted). The curve shows that the same decrease in nutrient load (stressor) into one water body might have little effect on HAB frequency (ecological

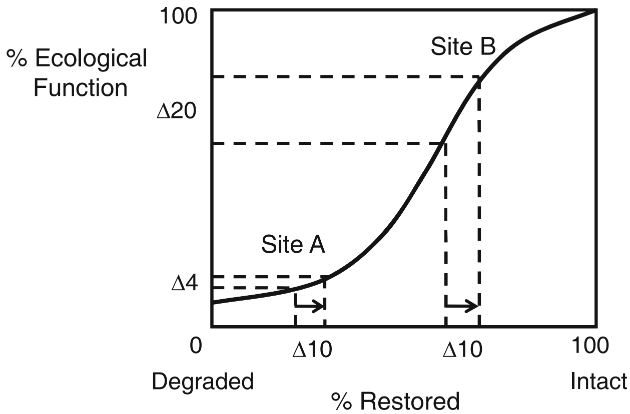

Fig. 3 The effect of slope in the RF on estimated response to stress removal. A variable slope in the RF will identify ranges of degradation over which effort will be most effective at restoring ecological function. For example, a 10% decrease in stress at *site A*, in the more degraded range, results in a 4% increase in ecological function. In contrast, a 10% decrease in stress at *site B* results in a 20% increase in function. An assumption of linearity in the RF, when inappropriate, will fail to identify opportunities for greater returns to effort

response at site A), whereas in another water body it may have a pronounced effect (site B). In this case, assuming a linear relation between nutrient loads and HAB frequency might provide highly misleading information for comparing potential outcomes of decreasing nutrients across locations.

Of course, researchers will aim to provide the best representation of the system possible, but because linear assumptions are a common default assumption in much of applied ecology, it is important to understand the ramifications of such choices. Practically, this means that all RFs should include realistic expectations of likely system response (shape of the function and potential discontinuities) to ensure that either (1) the appropriate empirical model is fit to the data or (2) appropriate conceptual models and assumptions are used in simulation or other types of models.

Examples of RFs

An example of an RF that has been applied to decision-making is the system for wetland evaluation developed in conjunction with the hydrogeomorphic (HGM) classification system (Smith and others 1995; Brinson and Rheinhardt 1996). In this system, the quantity and quality of ecological outcomes produced by wetlands within a particular HGM class and region (e.g., surface-water storage) are modeled as outputs of observable wetland characteristics (e.g., water dynamics and vegetation density) and evaluated relative to a reference (ideally undisturbed) wetland. The value of the HGM system lies in its ability to deconstruct wetland ecosystems into their component ecological RFs, relate those functions to wetland characteristics that are likely to respond to human actions, and then use those functions to compare impacts of alternative actions and wetland-mitigation requirements. Although the system has its critics (e.g., Hruby 2001; Stander and Ehrenfeld 2009) and is limited for economic benefits analysis because the outcome metrics are not directly related to risks and benefits and because the system does not readily allow for comparisons across HGM classes, it, nonetheless, demonstrates the types of model components that are needed to relate changes in wetland characteristics to ecological outcomes.

Models similar to HGM functional models are part of a growing library of ecological functional models that are used to represent what we might call "inventories" of potential ecosystem services but that are not necessarily useful RFs. For example, United States Geological Survey (USGS) researchers developed models to quantify ecological outcome metrics (or "services" in their terminology), including potential water storage (for flood mitigation), carbon sequestration, and wildlife-habitat suitability as functions of site and landscape variables (Gleason and others 2008). Although these models may be reasonable for estimating functional capacity of an ecosystem, they offer some challenges for a benefits assessment because of the endpoints and explanatory variables chosen.

Consider that in the USGS model, the outcome indicator representing flood control is total volume of water held in wetlands. This may be an indicator of benefits, but it is not directly relatable to flood risk because it does not describe how much of that water-holding capacity is typically available for flood risk reduction, a question that would require an evaluation of hydrological and weather dynamics. Furthermore, the model that relates landscape condition to water-holding capacity uses predominantly geophysical variables, such as topography, to quantify this outcome instead of variables that are controllable by people, such as amount of impervious surface or tile drainage within watersheds. Therefore, the model cannot demonstrate changes in water-storage capacity as a result of human activities unless activities, such as wetland filling, drainage, and land conversion, can be translated into topographic changes. Thus, although the USGS models provide information useful to establishing potential ecosystem services, they are incomplete functions for relating stressors to outcomes that might be used in benefits assessment.

These examples from the ecosystem services literature demonstrate some of the challenges to creating ideal response relations. Modelers are challenged to develop practical models that incorporate public-friendly response metrics, controllable drivers of change, site and surrounding landscape variables, nonlinearities, and temporal dynamics when variability may be more important than average condition. The example models succeed in one important regard: They describe natural systems in terms of functional outputs that can be related, with additional work, to human well-being. However, as a first step, both sets of models could be improved by more directly linking measured response variables to public-friendly outcomes and by directly incorporating the controllable stressors that moderate outcomes (i.e., management opportunities).

In contrast to the ecosystem services literature aimed at inventorying goods and services, the environmental economics, risk assessment, and environmental management literature has a long history of developing RFs that evaluate how specific management actions affect ecosystem service benefits. The published models are too numerous and varied to thoroughly summarize here; however, they commonly evaluate proposed regulations or management of direct stressors and thus provide alternative approaches to building RFs. For example, many models have been developed to relate air or water quality to human health and aesthetics or to species impacts that are relevant to fishing, hunting, or nonuse values, which are all public-friendly outcomes (e.g., Calow 1998; USEPA 1999; Egan and others 2009; NRC 2005).

The models used in ecological risk assessment and economic valuation often rely on statistical approaches that have the advantage of well-constrained model error but have sometimes been critiqued for failing to adequately consider system complexity and dynamics (Munns 2006; Knowler 2002). The ability to readily explain inputs and quantify error in such models tends to engender trust among those who must use the ecological outputs and can help to build stakeholder support. However, empirical models may not be appropriate for projections under major changes in system state (e.g., under climate change), suggesting the need to match modeling approaches to questions and stakeholder needs. Some examples of models that incorporate greater system complexity and dynamics are those developed for coastal-system management that link nutrient and sediment inputs to ecological outcomes, such as fish habitat quality, using complex process-based models (Breitburg and others 1999; Rashleigh and others 2009). These models may be better able to incorporate system feedbacks but also have the disadvantage that the error associated with model projections is usually difficult to estimate. The importance of incorporating dynamics versus being able to quantify error is yet another trade-off that must be considered, particularly if decision makers find empirical models easier to understand or more acceptable (Dietz and others 2004; Cockerill and others 2004).

The EPF: Determine Whether Services are Produced at a Given Time and Location

Although the RF establishes that the necessary natural conditions exist to produce ecosystem services, the EPF establishes whether services are actually produced, through interactions of people with the ecosystem. The ecological outcomes of the RF describe the quality and quantity of functional outputs of an ecosystem but do not describe whether those outputs are sufficient to provide services to people. Therefore, what distinguishes the EPF from the ecological RF is the addition of factors that humans find necessary or desirable to derive benefit from an ecological outcome.

The EPF may be a relatively simple component of the analysis compared with the RF because the presence of some easily measured characteristics may be sufficient to demonstrate whether services can be realized. The EPF incorporates (1) whether the potential service (as indicated by the ecological outcome) is likely to be used in a given location and time frame and (2) whether sufficient quantity, quality, and reliability of ecological outputs are provided. The EPF may not be a continuous function but rather a set of conditions that support the creation of the ecosystem good or service. Because the EPF must consider end users and their preferences, developing this set of conditions will require collaboration between natural and social scientists to ensure that ecosystem opportunities to supply services are evaluated in the context of location-specific social and economic factors.

In many natural resource and environmental valuation studies, the EPF is subsumed into an overall benefits model and may not be presented as a separate model. Here, we present the EPF as a separate function because of the current interest in mapping and quantifying ecosystem services as a separate exercise from valuing social benefits. In many cases, the EPF may provide sufficient information for evaluating policies or making management decisions about how to minimize harm or maximize benefits (e.g., as part of a cost-effectiveness analysis). However, the EPF does not need to establish the frequency of use or total level of demand because this analysis belongs more appropriately under the benefits evaluation, where models can be developed to project the change in demand for a service when its quantity or quality changes.

Establishing Demand for Services Based on the Presence of Complementary Inputs

The factors used to evaluate demand in the EPF will differ by whether a service is provided on-site, in proximity to a site, or whether it is provided regardless of proximity. Some services may be provided both on- and off-site, requiring consideration of multiple sets of users, such as when bird-nesting habitat on a site provides bird-watching opportunities both at the nesting site and at nearby sites where birds visit. For our estuarine example, we are implicitly relating HAB frequency and extent to safety of water contact and fish consumption and therefore must seek evidence of use for these activities.

On-Site Services On-site services are by definition related to direct uses of the ecosystem. These may be consumptive uses (e.g., hunting or fishing) or nonconsumptive uses (e.g., hiking or wildlife viewing). Whether a service is offered on a site depends on whether users have access and whether any necessary complementary inputs are present. Therefore, certain characteristics, such as built infrastructure (e.g., roads, boat ramps), can be used to identify locations where a potential ecosystem service suggested by the RF is realized (note that lack of infrastructure does not preclude an area from providing the potential for future use, which is a separate analysis). The infrastructure or human activity that creates ecosystem services may include various biophysical, built, sociodemographic, or cultural conditions that are either colocated with the function or are within an appropriate distance of the ecosystem.

For our estuary example, both services are on-site services. The complementary inputs for services that involve water contact or fish consumption (swimming, boating, water skiing, commercial and recreational fishing) could be measured through direct observations of use, such as visitor surveys or fishing-vessel trip logs, or through indirect indicators of use, such as population centers within driving distance, beaches with public or private access, marinas, boat ramps, and commercial fishing ports. For other types of on-site services, complementary inputs include land-management activities other than built infrastructure. For example, the presence of *crops that are pollinated by bees* provides the complementary input to the ecological outcome of *bee pollination* to produce an ecosystem service of *enhancing crop yields*.

Off-Site Services For services that do not require access to a site but instead are used by people off-site, the indirect spatial connection between areas that produce ecological outcomes and off-site users may be suggested by the presence of off-site structural or cultural elements. For example, the presence of buildings, farms, and infrastructure in a flood plain (which are vulnerable to flooding because the area lacks built flood-protection infrastructure) is a fairly clear indication that the service of flood-risk mitigation provided by upstream natural areas would be important. Note that we mention evaluating whether built structures, such as levees, are present because if they have already been built, the natural flood control may not provide additional benefit. Alternatively, if the substitutes for natural flood control are unavailable or costly, then the flood mitigation services will be more desirable.

For a few use services, colocation of users is not necessary to create a potential use of the service. The primary example is carbon sequestration, which is used as a proxy for the ecosystem service of climate regulation to decrease hazards, such as storm damage. The benefits of hazard decrease will vary spatially because some populations will be more vulnerable to sea-level rise or other outcomes of climate change. However, because everyone shares one atmosphere, carbon sequestration contributes to ameliorating effects of greenhouse gas buildup regardless of location.

Nonuse Services For nonuse services, which provide existence, option, or bequest values, physical complementary inputs, such as infrastructure for access, are generally not required and may actually decrease value. The complementary inputs for these services are more often cultural conditions, such as a "sense of place" or preferences related to historic use or existence of specific ecosystem outputs. Generally, evidence from surveys, interviews, or focus groups is needed to demonstrate that preferences exist for preserving or restoring a nonuse service in a given location.

Necessary or Sufficient Quality, Quantity, and Reliability of the Ecological Outcomes

A major role of the EPF is to establish whether the quality, quantity, and reliability of the ecological outcomes are sufficient for potential services to be realized. In general, economics, human health and safety, sociological studies, market analysis, and opinion surveys will be used to suggest which ecological qualities are important for supporting an ecosystem service. Alternatively, user groups can be directly engaged to elicit preferences. For our HAB example, historical case studies demonstrate that people will stop using affected services when HABs are present (and for some time afterward), and surveys might be used to suggest what level of HAB frequency would more permanently change use of an ecosystem. When using the literature, studies that demonstrate willingness-to-pay for environmental quality improvements can suggest which site conditions are more or less supportive of particular services. Similarly, the engineering literature provides a wealth of evidence to support selection of biophysical characteristics that are needed for technical use, e.g., if wetlands are used to treat wastewater.

A basic empirical approach for determining how site-specific qualities are related to services is to identify locations used by people and then, through statistical models, relate use to observable characteristics of sites. Such studies are often informed by interviews and surveys with service users that explicitly test ecological and environmental quality variables, such as presence and abundance of species of interest (e.g., huntable, fishable, or watchable species) and variables related to attractiveness (visual, aural, and olfactory) (e.g., Garber-Yonts 2005; Cho and others 2009; Brown and Reeder 2007; Kinnell and others 2006; Smith and Desvousges 1985; Bockstael and others 1989; Lipton and Hicks 2003; Massey and others 2006). These types of studies provide the strongest evidence of the relation between ecological quality and use because use has been demonstrated. Once preferences have been established for a given service through these types of efforts, those seeking to map ecosystem services can identify spatial data that capture these preferences (e.g., Kliskey 2000).

Studies that rely only on expert judgment rather than user surveys to determine potential use may create biased results because scientists may be tempted to use metrics they consider important rather than objectively evaluating public wants. As an example of the potential divergence between expert and public preferences, Nassauer (2004) used surveys to compare characteristics that the public found desirable in urban wetlands to characteristics that ecologists found desirable. A key result was that a metric widely used by ecologists to assess function, plant-species richness, was not significantly related to perceived wetland attractiveness in the survey group. However, her results did not preclude using plant richness as a proxy for another desirable outcome, namely bird-species richness, which people were shown to value, if data support a relation between bird and plant species richness.

For nonuse services, it is likely to be more difficult to determine what ecological qualities people value. Stakeholders can be engaged to describe location-specific preferences through focus groups, structured interviews, or surveys. However, consulting the public to identify site conditions that produce nonuse services derived from preserving ecosystems or species may not always be appropriate because respondents may be unable to link specific conditions to the outcomes they value. In such cases, experts should seek to make their ecological metrics sensitive to the specific outcomes that people value but may choose to use ecological data and understanding to develop models that project where nonuse services are provided.

An example of making ecological metrics sensitive to preferences is establishing a reasonable likelihood that a nonuse service will persist well into the future to represent the values implied by bequest and option values. Tools, such as species population viability analysis, other species-population projection models, or threat assessments, might be used to suggest whether a population or ecosystem in a particular location appears sustainable into the future (Carroll and others 2004; Beissinger and McCullough 2002; Gotelli and Ellison 2006). Walker and others (2008) created a relatively simple "risk of biodiversity loss index" by first correlating species biodiversity to area of native plant cover using a nonlinear species–area relation. This empirical RF became an EPF when they added in land-protection status as an additional risk factor in the index to reflect long-term capacity of the landscape to support species.

For all services, reliability can affect people's ability to use or appreciate that service. In our framework, reliability is defined as the level of variability or duration of biophysical conditions over time that is acceptable for supporting human uses or a future stream of benefits. For example, if stream depth is not sufficient for recreational rafting during 11 out of 12 months of the year, then the site will be less likely to support commercial rafting businesses relative to a site where depth is more consistently appropriate for rafting, all else being equal. Or, if sea-level rise is expected to inundate a wetland, it cannot be considered to provide storm-surge protection during the long-term. Temporal consistency and probability of future function are thus factors in understanding whether a service is provided at a given location and during a given time frame.

Additional Examples of EPFs

For some use services, the built infrastructure and/or economic activities may be even more important than ecological qualities for establishing whether services are provided. For example, a coastal marsh that is close to urban areas and easily accessed by road can provide many users with aesthetic, safety, and recreational services, including photographic opportunities, storm-surge protection, bird watching, and kayaking. In contrast, a marsh that is isolated from human activities may provide important nonuse services but will not provide as large a range of services as a site that can be directly accessed by people. Thus, the type of services and the demand for some services is affected by whether people can readily access a site.

A simple illustration of this point is found in Boyd and Wainger (2003). A framework was developed to judge equivalency of wetland trades, in terms of types of services provided and potential benefits, by applying a set of indicators to reflect site and landscape conditions that reflected aspects of service quality, demand, and level of use. In one part of that analysis, a straightforward indicator—population density of the census block that contains the wetland—was used to demonstrate that in all but four cases, the mitigation site had lower population density than the impact site (Fig. 4). This result suggested that, overall, the mitigation program tended to shift wetlands to areas where potential for delivery of use services was lower. This example demonstrates that even isolated metrics can suggest a change in the types of services that are likely to be delivered, and reflect performance aspects of a policy, even although they do not fully characterize changes in benefits (see Boyd and Wainger [2002] for complete framework description).

An example of EPFs that apply quality standards to identify where ecosystem services are likely to be produced is the "water quality ladder," which relates a range of biophysical water-quality standards, based on health risks, to a range of "designated uses" which are similar to services. For example, water quality that ranks low on the ladder may support boating, but a greater ranking is needed to support swimming. Early studies (Vaughan 1986; Mitchell and Carson 1989; Carson and Mitchell 1993) did not explicitly link the services included in the ladder to specific water-quality conditions. However, regulators, in implementing the Clean Water Act, have built on the ladder concept to create standards that maintain designated uses. Recent work has sought to expand the list of uses and improve relations between conditions and outcomes (e.g., Hime and others 2009).

Such water-quality standards are relevant to our estuary example because they demonstrate the need to look at all factors that may limit provision of a service. Our example RF focused on the potential to decrease risk from HABs by decreasing nutrient loads. However, if water is also contaminated by toxins or excessive bacteria that make contact dangerous, then decreasing the risk of HABs will not change whether a service is realized. Similarly, fish-consumption advisories based on methylmercury or polychlorinated biphenyl levels may be a greater factor for safety of fish consumption than HABs.

The BF: Determine Social Welfare Impacts

The BF evaluates net gains or losses in social welfare resulting from changes in ecosystem service outcomes. Social welfare, the sum of individual welfare changes over all affected people, changes when an ecosystem service directly or indirectly changes the satisfaction (i.e., utility) that people derive from natural systems (Freeman 2003). Thus, the BF typically evaluates two things: (1) the number of individuals affected by a change in a service and (2) the degree to which each user values a change in a service. The BF may be developed by gathering detailed site-specific information from a study area or by using benefit transfer to relate information collected at another site or sites to the study site. A benefits assessment may also be based on factors other than the sum of individual benefits that can be readily monetized. Because society values protecting vulnerable groups despite their lack of ability to pay, public investments and policy choices may be based on broad consideration of the magnitude of potential harm to different groups regardless of the values that can be directly monetized (USEPA 2000). In general, such ethical preferences, which may not be easily expressed in dollars, may be more validly captured using discourse-based approaches to valuation rather than cost–benefit analysis (Spash 2008).

When quantitatively comparing benefits, both monetary and nonmonetary metrics may be used; however, when properly measured, monetary values are generally considered most robust for comparing outcomes in terms of social welfare (USEPA 2009b; United States Office of Management and Budget 2003). Being able to monetize changes for multiple services allows the total change in welfare to be summed across all affected individuals and multiple services. In particular, for purposes of conducting cost–benefit analysis, all effects must be monetized to evaluate whether the benefits for the winners (those whose welfare

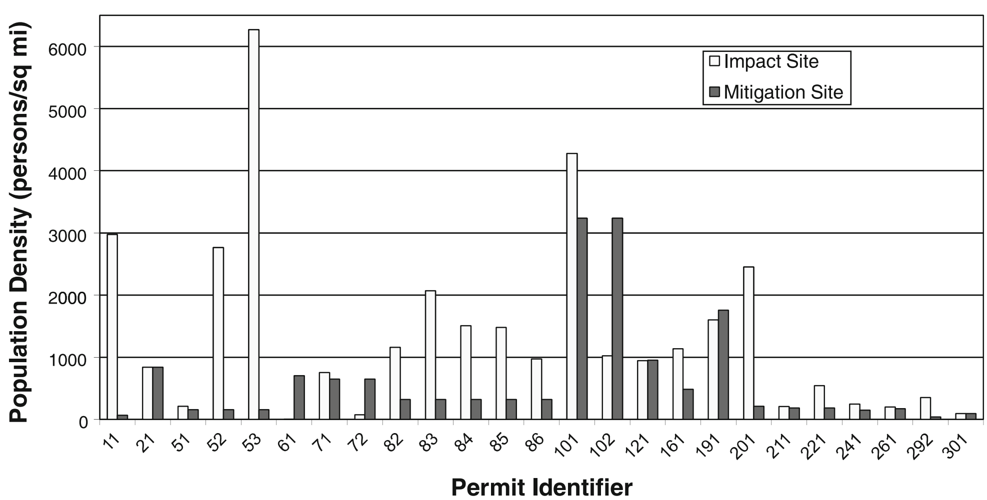

Fig. 4 Comparison of population density scores for matched wetland impact and mitigation sites. This Maryland case study showed that in all but four cases, the nontidal wetland mitigation site had lower population density than the impact site, suggesting that mitigation led to a decrease in the number of people served by wetlands use services and a likely change in the type of services from use to nonuse (from Boyd and Wainger 2003, Chap. 8)

increases) could theoretically compensate losers (those whose welfare decreases) to create a socially efficient solution under a cost-benefit criterion.

In some cases, economic analyses are cost-prohibitive or unnecessary, and nonmonetary benefit metrics are used to weight and rank decisions and illustrate the benefits of spending (Ribaudo and others 2001; USEPA 2009b). Nonmonetary metrics are not typically direct measures of social welfare, but due to the difficulty of monetizing changes in social welfare, they are often considered acceptable proxies for understanding relative changes in social welfare across management options (further described in Wainger and Boyd [2009]). Nonmonetary metrics are most useful for decision-making when they reflect the same concepts of well-being and willingness to make trade-offs among services compared with those from a study that monetizes benefits. Because the units of nonmonetary metrics may differ by service, multicriteria techniques are used to aggregate impacts across services, using people's relative preferences for different outcomes (e.g., Romero and others 1998; Kiker and others 2005).

Regardless of whether monetary or nonmonetary metrics are used, ecosystem service benefits are summed over the multiple competing and complementary services included in the analysis to estimate total benefits of a management option. The correct measure of total benefits is generally referred to as total economic value (TEV) and is the sum of all types of value, including both use and nonuse values (NRC 2005). Because aggregating values across a suite of ecosystem services creates the potential for double-counting of benefits, which can distort benefit estimation, the TEV typology is used to ensure the inclusion of all components of value without double-counting (NRC 2005).

Double-counting is often an issue when several methods are used to value different components of the full suite of changes in services, and it can be minimized by considering whether the same people's values, or the same source of value, are counted in multiple estimates. However, double-counting can be difficult to avoid completely due to a variety of methodological issues with value estimates (Turner and others 2008), including a well-recognized phenomenon in which participants in economic studies may not have independent preferences for each of a set of related outcomes (Hanley and Shogren 2005). Sometimes, TEV has been misinterpreted as meaning the value of a total ecosystem rather than the sum of all human values for a change in an ecosystem (NRC 2005). Thus, it is important to note that this framework, in contrast to some other systems for evaluating ecosystem service benefits, promotes evaluation of benefits associated with *changes* in ecosystem services that result from changes in human activities rather than values of total stocks or inventories of ecosystem goods and services.

A full discussion of economic valuation techniques is beyond the scope of this article. For further detail, see King and Mazzotta (2000), NRC (2005), Champ and others (2003), and Bockstael and McConnell (2006). However, we describe some concepts important to developing both monetary and nonmonetary metrics of social welfare, both for the main purpose of clarifying the inputs needed from the RF and EPF and to demonstrate how a benefits evaluation is distinct from simply measuring changes in ecosystem services.

Quantifying the Number of Individuals Affected

The number of individuals affected by a change in an ecosystem service is typically measured by considering the total number of people using a service and the frequency of use (e.g., user days). In many cases, data are available to account for current use, especially for recreational uses, where visitor statistics or participation surveys are often collected (e.g., United States Fish and Wildlife Service 2006). However, when use statistics are not available, and for all nonuse services, it may not be possible to precisely estimate the number of people who benefit from a service. In these cases, it may be necessary to estimate use based on proxy measures that estimate the size of the population that could potentially benefit from a service, such as using the number of registered boats in counties that border the estuary to estimate the number of boaters or anglers who might benefit from improved water quality or fish safety.

For nonuse services, the extent of the affected population is not always geographically defined because nonuse services do not typically depend on proximity. In some cases, nonuse values can extend over large areas, such as those estimated for losses from the Exxon Valdez oil spill, where studies found that people throughout the United States expressed a loss of nonuse values (Carson and others 2003). In contrast, the loss of a local wetland might affect nonuse values within a county or state, but it might not extend beyond that area because people in other locations may not have been aware of that wetland and may have similar sites available in their region. Thus, the extent of nonuse benefits is context-specific and depends on awareness and significance of the service (Freeman 2003).

Current use statistics or proxy measures only establish the current population of users. To understand the likely number of users after a management change, models are often developed to predict changes in total users and to redistribute users among substitute sites based on projected quality changes (Whitehead and others 2000; Garber-Yonts 2005). For our estuary example, a thorough model would evaluate whether many new swimmers, boaters, and anglers would be likely to start using the estuary if risks were decreased.

Quantifying Magnitude of Loss or Benefit

Welfare effects are measured through a variety of techniques that in general capture the difference between the maximum someone would be willing to pay to prevent or create a change and what they actually spend. The concept, known as "consumer surplus," may sound arcane to noneconomists, yet it is a fundamental component of social welfare theory. Its usefulness is most obvious when considering ecosystem services that are free or inexpensive because it allows economists to capture values for services, such as safe drinking water, that may be provided inexpensively, despite high willingness-to-pay and benefits (Freeman 2003). Another way of thinking about consumer surplus is that it captures the magnitude of harm resulting from loss of an ecosystem service based on people's willingness and/or ability to adapt to that loss.

Primary Valuation Studies Consumer surplus is measured through two classes of techniques: those based on observed behavior, or revealed preference methods, and those based on asking people questions about their values, or stated preference methods (King and Mazzotta 2000; Champ and others 2003). Because revealed preference studies represent what people have actually spent, as opposed to what they only say they will spend, they are generally considered more robust measures of willingness-to-pay than stated preference studies (Diamond and Hausman 1994). However, for both types of studies, the accuracy of valuation often depends on how well-informed people are about the good or service. Researchers have found that the more familiar people are with the outcome being valued, the more confidence they can have in the estimated values (Hoehn and others 2003). Therefore, studies that elicit values for familiar goods, such as a recreational fishing day, generally provide more robust results than studies that elicit values for unfamiliar goods, such as protecting obscure endangered species.

A quality-conscious consumer of economic valuation studies should know that a thoughtfully designed stated preference study will consider and attempt to address several important issues. Often, respondents do not have fully developed preferences for goods they do not normally buy or may not use often, if at all. People may have trouble understanding complex connections between one good and another (e.g., to decrease the risk of HABs, prices of farmed products may increase); sometimes people may not have experience with conditions they may be asked to value, such as the total loss of a service throughout a region (Johnston and Duke 2009). To address these challenges, a robust stated preference survey will include an explanation of the importance of the service in terms that respondents understand and will use a series of interviews, focus groups, and pretests to ensure that other sources of bias and confusion in the survey instrument are minimized.

For revealed preference studies, a key component in quantifying benefits is assessing how easily people may adapt to the loss of a given service by considering the cost and availability of substitutes. For example, in the case of decreasing HAB risk in an estuary, the magnitude of the welfare increase that might result will be influenced by whether people have found acceptable substitutes to using the estuary for recreation that involves water contact. For example, people may have access to safer fishing, swimming, and boating opportunities in a nearby reservoir. The loss of the original ecosystem services (safe estuary boating and fishing) does not necessarily imply a complete loss of the welfare associated with those services. Rather, the loss of welfare is the change in consumer or producer surplus that resulted from having to make the switch. If people must drive farther to the reservoir, and if they value the fishing experience in the reservoir less than that in the estuary, then the loss of consumer surplus resulting from HABs in the estuary will be captured by the difference in costs of access and willingness-to-pay between the estuary and the reservoir.

A thorough benefits assessment will therefore examine the potential to substitute sites, technologies, or other natural services before estimating potential harms or gains. Ecosystem services vary greatly in their substitutability, and many ecosystem services are considered irreplaceable, and thus more valuable, because they have no technical substitutes or close natural substitutes (e.g., a rare charismatic species) (Bulte and van Kooten 1999). Substitutes may not exist for other ecosystem services because they are location-specific or not easily transportable (e.g., a scenic view). Therefore, to fully characterize benefits, losses, and useful trade-offs, ecologists, economists, engineers, and others must work together to identify whether potential substitutes exist as well as their cost, availability, and desirability relative to the service at risk.

Benefit Transfer Many government agencies want to avoid the time and expense of conducting primary economic valuation studies or may find that institutional constraints, such as the requirements of the Paperwork Reduction Act, make it effectively infeasible to carry out survey research and therefore rely on benefit-transfer techniques to estimate values for goods and services (Iovanna and Griffiths 2006). Benefit transfer is conducted by either taking average values from existing valuation studies or by using a transfer function to transfer values from primary studies (study sites) to new locations (policy sites) (e.g., Loomis and Rosenberger 2006; Wilson and Hoehn 2006; Rosenberger and Loomis 2001). A transfer function is often developed through meta-analysis, which is a statistical (usually regression) technique to model how values vary among primary valuation studies based on community demographics and other factors (Bergstrom and Taylor 2006).

Benefit-transfer techniques have promise if performed well, but they also can have significant limitations (Ready and Navrud 2006; Spash and Vatn 2006). The keys to using benefit transfer successfully are to apply generally accepted methods to (1) find robust values that can be appropriately transferred to the ecological and sociodemographic conditions present at a site and (2) identify how those values change with a change in quality or quantity of the service (Hoehn 2006; Loomis and Rosenberger 2006; Feather and Hellerstein 1997; Smith and others 2002). However, it may be difficult to find studies that demonstrate how values are sensitive to changes in ecological qualities or quantities. For our estuarine case study, we would seek studies that relate changes in consumer surplus associated with boating, fishing or swimming to changes in HAB extent and frequency. Or, if such studies were unavailable, we might seek to transfer studies that measured welfare effects of changes in boating, fishing, and swimming due to any kind of change in estuarine water safety.

A confounding effect when evaluating sensitivity to ecological change is that people's values are almost always location and context-specific in ways that have not been explicitly measured in the original studies. For example, Johnston and Duke (2009) demonstrated that willingness-to-pay for preserving natural land diminished with increasing area of preserved land. However, variables such as the percentage of land that respondents know has been preserved, are not typically measured by surveys and thus cannot be captured in transfer functions. A growing body of work seeks to capture effects of ecological characteristics on willingness-to-pay (e.g., Johnston and others 2002; Bark and others 2009; Weber and Stewart 2009; Egan and others 2009) to better demonstrate when ecological degradation corresponds to welfare loss, but conditions of regional abundance and substitutability that can strongly influence value remain problematic to incorporate.

Spatial Benefit Transfer and Mapping Ecosystem Services To better characterize how values vary with changing landscape conditions, researchers are increasingly seeking to apply spatial benefit transfer, in which they attempt to map values onto small areas (i.e., map pixels or land parcels) based on ecological and socioeconomic conditions in that small area. Using spatially detailed data and geographic information system (GIS) tools, researchers demonstrate how ecosystem service demand, reliability, or complementary inputs vary across regions (Bateman and others 2006; Naidoo and Ricketts 2006; Boyd and Wainger 2002; Wainger and others 2010; Willemen and others 2010; Natural Capital Project 2010; USEPA 2010). Many GIS approaches to ecosystem service mapping only evaluate the potential supply of ecosystem services based on ecological conditions or, in our terminology, outputs of the RF. However, it is becoming more widely recognized that mapping economic value requires considering (1) where use can be demonstrated and (2) where a change in services would create the greatest harms or benefits.

This spatial approach to benefit transfer differs from the more traditional benefit transfer functions previously described in that there is usually a greater attempt to capture the effects of heterogeneity of biophysical conditions on benefits. For example, in a recent study, Baerenklau and others (2010) used surveys to create a spatial-benefit transfer function that incorporated which locations within a park were preferred by recreators to spatially allocate values and use such maps to better characterize potential impacts of a management change. Similarly, for our HAB example, we might map preferred swimming, fishing, and boating sites to examine whether the areas of greatest use coincided with the greatest risk for HABs. Such analyses could be used to show whether alternative unaffected sites were accessible to user populations to improve understanding of the relative welfare impacts.

For benefit maps to be useful for assessing the benefits of investments in restoration or preservation, it is helpful if they measure values at a scale that corresponds to decisions (e.g., owned parcels or resource boundaries) and capture relative scarcity of ecosystem services (demand in excess of supply) to reflect the vulnerabilities to service losses. A recent example of scarcity mapping estimated which areas of the contiguous United States were at risk of experiencing water shortages based on expected supply and demand under changing climate conditions (Natural Resources Defense Council 2010). The study demonstrated the spatial variability of water stress based on supply and demand conditions, which, although it was not an explicit measure of benefits, was useful for understanding risk spatially.

Implementing the Entire Framework

Our hypothetical case study, which linked a change in tillage practices to a change in ecosystem service benefits demonstrated that multiple functional relations are needed to apply this or similar frameworks. The IF captured the effectiveness of a management action by relating a tillage change to changes in stressors: nutrient and contaminant runoff. The RF evaluated how changes in nutrients affected an outcome that could clearly be linked to values: extent and frequency of HABs. The EPF explicitly linked the HAB outcome to estuarine ecosystem services that are used and/or appreciated: safe water-contact recreation and fish consumption. And, in the final step, changes in the ecosystem services were evaluated in terms of their potential welfare impacts by considering the number of users affected and their ability or willingness to adapt to that change. Many of the relations used in the framework are purely in the domain of biophysical sciences, but they are nonetheless informed by a goal of benefits assessment.

The framework is primarily aimed at informing a cost-benefit or cost-effectiveness analysis, but the processes of developing the models and the outcomes of each framework step can also support a discourse-based process to manage a resource according to community values, which may be particularly important if substantial benefits cannot be monetized. To apply the framework results in decision-making may require yet another step of integrating the results into a system for comparing multiple management alternatives or optimizing outcomes. The process of linking cause and effect of a management choice may need to be repeated for numerous pathways to create an integrated ecological and economic modeling framework that offers the capacity to examine effects of multiple stressors on multiple services. Although a comprehensive model may be intractable, modeling to support decisions should, at a minimum, include a sufficient representation of competing services to capture important stressors, outcomes, and trade-offs. It is worth noting that some researchers have successfully built the teams, databases, and modeling tools necessary to carry out major parts of the framework, despite the many challenges (Brookshire and others 2010; Nelson and others 2008; Barbier 2007; Murdoch and others 2007; Johnston and others 2002).

To ease the burden of developing all necessary relations to handle messy environmental problems, many researchers are creating libraries of response relations to inform ecosystem service evaluation and valuation. Some of these approaches integrate decision support and optimization tools to explore trade-offs between multiple systems outcomes and suggest optimal solutions or summaries of welfare impacts due to alternative scenarios. Two of the best known are perhaps the InVEST suite (© 2011, The Natural Capital Project, Stanford, CA) of ecosystem service models (Kareiva and others 2011; Nelson and others 2009) and Marxan (© 2009 The University of Queensland, Australia), which is intended to optimize nature reserve selection (Ball and Possingham 2000). In addition, a host of software tools are cataloged by the Ecosystem-Based Management Tools Network (2011) (EBM Tools, © 2010 NatureServe, Arlington, VA, USA). As with any off-the-shelf product, these tools can ease the computational burden faced by a researcher, but if they have not been demonstrated to be accurate and representative for a given system and for a given spatial scale, they may be unacceptable in some decision-making contexts. In addition, complex models that are developed outside of an open and democratic process may incorporate unexplored biases of the researchers, e.g., by excluding or devaluing goods and services that compete with ecological outcomes (e.g., food production). Therefore, models for assessing social welfare are most appropriate when they include a broad range of demonstrated social preferences and clearly show trade-offs.

Conclusion: Fundamentals of Ecosystem Service Measurement

The ecosystem service analysis framework that we present here will be recognized by many economists, risk assessors, and decision analysts as necessary for estimating benefits derived from a change in ecosystem services due to a management action. Yet, it is relatively rare to find case studies that meet all of the information requirements and include all of the necessary quantitative relations to calculate social benefits from a management change. Many more studies implement some parts of the framework, which can be appropriate if the decision-making context does not require quantification of each relationship. However, a lack of available rigorous quantitative information forces many agencies to justify management actions based on conceptual models and associated ecological metrics (Ribaudo and others 2001) or best professional judgment (Roman and others 2008), which have not all been tested for rigor and which may not fairly represent trade-offs.

It is widely recognized that ecological metrics or benefit indicators can serve as useful decision tools when this complete framework cannot be implemented. Nevertheless, any progress toward strengthening the functional or conceptual relations between human actions and meaningful ecological outcomes will improve our ability to make appropriate trade-offs between different types of benefits. Therefore, interdisciplinary scientists should not be discouraged by the many impediments to fully implementing such a framework because contributing to any part of the framework is likely to have useful management implications. Even improving the conceptual models that relate stressors to valued outcomes (e.g., recognizing when nonlinearities or thresholds are probable) can improve decisions by highlighting where actions are likely to generate the best returns.

In applying the framework, some points emerge as fundamental to a robust analysis.

1. *Include a management opportunity when evaluating ecological condition.*

In other words, link the ecological outcome to a management choice. Do not model changes in the ecological outcome only as a function of uncontrollable natural conditions. Rather, seek a connection to something that a manager can change, such as the proportion of impervious surfaces, presence of stream buffers, or proportions and spatial arrangement of land uses.

2. *Choose public-friendly endpoints for the RF.*

The response variable will be most useful for managers if it communicates outcomes that are readily understood as important (Boyd 2007). Useful response metrics are those that describe species that are directly appreciated by people (usually birds or fish instead of invertebrates) or that effectively communicate tangible risk (e.g., probability of a harmful algal bloom instead of annual average nutrient concentration). Developing appropriate outcome metrics promotes the ability to compare ecosystem qualities using common units, which in turn promotes sound decision-making regarding the prioritization of use and protection of natural systems. Without such tools, we will not have the ability to judge when a wetland that is no more than an irrigation ditch would be better to sacrifice than a forest patch that is providing high carbon storage, water purification, and habitat functions. Ideally, economists and other social scientists should work collaboratively with ecologists to choose metrics that will best inform benefits assessment.

3. *Work to overcome lack of data by developing models that synthesize existing high-quality information and improve the knowledge base.*

Ecologists seeking to develop the relations between management actions and stressors, or the relations between stressors and system responses, are challenged by limited data availability and understanding. Lack of appropriate data or scientific understanding can lead researchers to conclude that robust models cannot be built. Yet, managers cannot wait for perfect information; therefore, researchers can aim to synthesize the best information available for critical ecosystem trade-offs and use expert judgment in ways that minimize bias to fill gaps. Many modelers accept the edict “all models are wrong, but some are useful” (Box and Draper 1987), as a way to make progress in developing models that are appropriate to answer specific management-related questions, despite the many uncertainties. Imperfect models or expert judgment, if reasonably robust, are necessary interim products to support decisions that may avert harm to ecosystem services and social welfare in a timely way.

Researchers who must confront the lack of existing models have the option to collect new data or create new models from existing data. For the latter case, literature reviews, expert elicitation, meta-analysis, and combinations of all three approaches have been successfully used to build management-relevant models (Niemi and others 1990; Johnston and others 2002; Tonitto and others 2006; McKinney and Wigand 2006). Although it is likely we will never have perfect understanding of systems, models that synthesize existing research provide a means to identify the most important data gaps and inform the new research programs that are seeking to support decisions.

4. *Consider supply and demand conditions to understand where and when changes in ecological conditions and processes generate benefits.*

Understanding the quality of ecological conditions and the supply of ecosystem services is only part of the information needed for making trade-offs among goods and services to be produced in an area or in a given time frame. An examination of the presence of complementary technical, social, or economic inputs is necessary to identify how people use and benefit from the system. Because people can adapt and make substitutions for ecosystem services, an evaluation of reliability and substitutability, or other aspects of service scarcity, are needed to understand the magnitude of benefits from protecting or restoring ecosystem services.

5. *Monetary valuation has its limitations, and other “democratic” approaches to decision-making are often used to make trade-offs.*

Many of those who implement the ecosystem services framework seek to monetize the benefits of nature’s services because they suspect that unless benefits can be monetized, they will be ignored. They may be right, particularly in certain circumstances, but it is important to understand the limitations of monetary estimates of non-market goods before relying too heavily on them. Valuation of ecosystem services is limited by two main issues: inability to capture robust values for certain types of services (especially nonuse services), and lack of information necessary to transfer monetary estimates of benefits across sites.

Even if we assume that a given monetary estimate of ecosystem service value is accurate where it is measured, many problems arise in attempting to transfer that value to another area. Important issues, such as regional scarcity, will influence willingness-to-pay and benefits, but rarely is scarcity measured in valuation studies. And if a variable is not measured, it cannot be used to transfer values. As a result, monetary values (both original study values and transferred values) are model-based estimates with many sources of error, just as ecological endpoints are model-based estimates with many sources of error. As such, the appropriateness of monetary values for use in any particular case study should be thoroughly scrutinized.

Monetary valuation is just one way to capture preferences of multiple groups to fairly consider competing needs and priorities. Other "democratic" approaches, such as risk ranking (Morgan and others 2000), multicriteria decision analysis (Clemen 1997), and community self-organization (Ostrom and others 1999) can be and are used to demonstrate what people value or would be willing to trade off. We did not review these approaches here but mention them to illustrate how ecological endpoints or ecosystem service outcomes can be readily incorporated into decision analysis when monetization is not possible or not necessary.

6. *Interdisciplinary teams are needed to robustly measure and effectively communicate the potential costs and benefits of a management action.*

The history of environmental economics is one in which economists have often used stylized models of ecosystems to support management decisions (e.g., maximum sustainable yield for fisheries as in Clark [1990] and as discussed in Eppink and van den Bergh [2007]). In recent years, ecologists have similarly used stylized models of economics (e.g., Costanza and others 1997). Although these models have their uses, they can also lead to inefficient decisions or unintended consequences or be unacceptable to decision-makers because of their simplifications. Cross-disciplinary work can be time-consuming and frustrating, yet it is the best way to ensure that a simplified version of reality (i.e., a model) includes the key components and appropriate levels of complexity required by each discipline for supporting decisions. Future success will depend on sincere communication and collaboration across disciplines and between researchers and decision-makers.

In summary, perhaps the most important thing for interdisciplinary teams to understand is that, even if economic benefits cannot be monetized by implementing most or all of this framework, well thought-out indicators of ecosystem services and their benefits can still improve the completeness and representativeness of outcomes used in the decision-making process. We recognize the challenges in implementing the framework laid out here. They include lack of data and understanding, lack of time to develop appropriate models, and impediments to interdisciplinary research. However, given that many government agencies are seeking a rigorous foundation for evaluating a broad range of social-welfare impacts that result from environmental change, the opportunities are many for interdisciplinary teams to better quantify how ecosystems deliver outcomes that affect human well-being and to support the difficult decisions regarding how we will manage our use of nature's goods and services.

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