ANNUAL REVIEWS Further
Click here for quick links to Annual Reviews content online, including:
• Other articles in this volume
• Top cited articles
• Top downloaded articles
• Our comprehensive search

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Integrating Ecology and Economics in the Study of Ecosystem Services: Some Lessons Learned

Stephen Polasky

Department of Applied Economics and Department of Ecology, Evolution, and Behavior, University of Minnesota, St. Paul, Minnesota 55108; email: polasky@umn.edu

Kathleen Segerson*

Department of Economics, University of Connecticut, Storrs, Connecticut 06269; email: kathleen.segerson@uconn.edu

Annu. Rev. Resour. Econ. 2009. 1:409–34

The *Annual Review of Resource Economics* is online at resource.annualreviews.org

This article's doi:
10.1146/annurev.resource.050708.144110

Copyright © 2009 by Annual Reviews.
All rights reserved

1941-1340/09/1010-0409$20.00

*Corresponding author.

Key Words

ecosystem services, valuation, ecological modeling, Millennium Ecosystem Assessment

Abstract

This paper discusses both the opportunities for and the challenges associated with integrating economics and ecology in the study of ecosystem services. We distinguish between integration in positive versus normative analysis. There is rapid growth in positive research that combines the two disciplines to provide insight and better understanding of the bidirectional linkage between economic and ecological systems. This research is a crucial part of addressing growing large-scale environmental challenges. This integration is equally important, but potentially much more difficult, in normative analysis, especially when interdisciplinary groups include individuals with different views regarding appropriate normative criteria. In such cases, reaching consensus can be difficult and slow, even when the practical implications of the different perspectives (i.e., the general policy prescriptions they imply) are the same. We suggest an approach for increasing the scope for collaboration among economists and ecologists in normative analysis.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

1. INTRODUCTION

Although ecology and economics have a long history of sharing ideas and reciprocal influence, dating back to Malthus, Darwin, and Marshall (Rapport & Turner 1977), the past decade or so has seen a growing recognition of both the rewards and the challenges of conducting interdisciplinary research that draws from these two disciplines. Recent interest in integrating ecology and economics stems in large part from the growing appreciation of the important role that ecosystems play in providing goods and services that contribute to human welfare and from the growing recognition of the impact of human actions on ecosystems and on the flow of these services from very local to global scales (e.g., Daily 1997). These impacts include not only traditional air and water pollution (such as sulfur dioxide emissions, ground-level ozone, and eutrophication), but also climate change, global changes in the nitrogen cycle, deforestation, loss of wetlands, and reductions in biodiversity. The need to understand and address these problems has led to calls for more closely integrating natural and social sciences, particularly ecology and economics, as part of an effort to ensure that human actions do not damage ecological processes necessary to support the continued flow of ecosystem services on which the welfare of present and future generations depends (Millennium Ecosystem Assessment 2005).

Economists have long acknowledged and studied the goods and services provided by nature—or, more precisely, provided by combining inputs from nature with other inputs such as labor, leisure time, and capital. However, the focus has typically been on single resources or services stemming from the use of natural resource–based inputs (such as land, fish or forest stocks, minerals, and energy resources) in the production of commercial products, or on nonmarket goods and services provided by natural environments (such as clean air, clean water, aesthetics, and recreation). Only recently have economists begun to work with ecologists to study the broader set of services provided by ecosystems and their impacts on human welfare. This reflects an increased appreciation of the complexity and interconnectedness of nature (or "natural capital"), the pervasiveness of joint production in nature, and the crucial role of intermediate services in sustaining the provision of the final services or endpoints that have historically been the focus of economic research.

At the same time, viewing ecosystems services as valuable to humans raised the possibility of creating incentives to provide these services by, for example, creating markets or related mechanisms through which individuals, businesses, governments, or nonprofit organizations could pay for (effectively, purchase) these services. Although the United States and other countries have a long history of paying for conservation (see, for example, Claassen et al. 2008), the recent interest in creating markets for ecosystem services has given the notion of payments for ecosystem services a new appeal and prominence in policy debates (Bulte et al. 2008, Daily & Ellison 1999, Heal 2000, Jack et al. 2008). For example, in December 2008, the U.S. Department of Agriculture announced the formation of a new Office of Ecosystem Services and Markets. Designing and understanding markets for ecosystem services require integrated knowledge of the production of these services (from ecology), the values of these services, and the incentives created by alternative market designs (from economics).

This paper draws on our recent experiences serving on various interdisciplinary committees charged with understanding ecosystem services. We discuss both the opportunities for and the challenges associated with integrating economics and ecology to address environmental issues. We distinguish between integration in positive analysis (understanding and predicting behavior and outcomes) versus normative analysis (evaluating and ranking outcomes), because the challenges of integration vary in these different types of analyses. In the context of ecosystem services, positive analyses include research characterizing the linkages between incentives and human decisions (e.g., responses to incentives created by markets or policies), the effect of such decisions on ecosystems, and the development of ecological production functions that can map ecosystem structure and function to the flow of services. In addition, it includes generating information about the trade-offs people are willing to make to protect ecosystems and the flow of services from them. This information can then be used in normative analyses, which involve evaluating policy or management options. One approach to normative analysis is to use integrated ecological-economic models to estimate the net benefits associated with different options. Under a strict benefit-cost decision rule, the efficiency rankings that these analyses generate would determine policy prescriptions. More generally, this analysis would provide input into policy decisions that would reflect not only economic efficiency but other considerations such as intra- and intertemporal equity (Arrow et al. 1996). However, as we note below, even this more general approach to normative analysis is not universally accepted.

Our primary message can be summarized as follows: Given the inextricable linkages between humans and ecosystems, much can be learned from interdisciplinary analyses that bring insights from ecology and economics to the study of pressing environmental problems. In addition, ecology and economics share many similarities that make their integration natural and allow useful crossfertilization of ideas, concepts, and results. Nonetheless, integrating the two disciplines poses challenges. In some cases, the difficulties stem primarily from limitations in current ecological and economic methods, data, and models that inhibit effective joint understanding and integration. These limitations will lessen as the amount of joint research between ecologists and economists increases.

More profound difficulties arise in normative analysis involving policy prescriptions. In our view the primary source of these difficulties is twofold: (*a*) differences in views on the sources or nature of value and (*b*) differences in views on the social choice rule that should be used to rank policy options or outcomes. Economists define value in terms of trade-offs that individuals are willing to make, but this definition is not universally accepted outside of economics. For example, an alternative approach based in environmental ethics recognizes an intrinsic value of nature that is not defined in terms of trade-offs and is separate from concerns about human welfare. These different views regarding the source(s) and nature of value generally give rise to varying views about social choice rules. In some cases these differences will be irreconcilable. For example, collaboration in normative analysis is nearly impossible between an ecologist who believes policy options should be ranked solely on the basis of intrinsic rights or biophysical impacts and an economist who believes in the importance of evaluating trade-offs based on preferences of individuals in society. The ecologist's view would render the assessment of trade-offs irrelevant, whereas the economist's view would not grant legitimacy to a policy ranking based solely on the beliefs or implicit preferences of the ecologist.

Nonetheless, even in policy evaluation contexts without complete agreement, we suggest that substantial progress can be made in integrating ecology and economics in the normative analysis of ecosystem services. The essential component necessary for collaboration in these contexts is agreement among researchers on the relevance of evaluating the trade-offs individuals are willing to make to protect ecosystems or to secure ecosystem

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

services, as reflected in their revealed and stated choices, and an agreement to focus the collaborative effort on assessing those trade-offs. Importantly, such an approach does not require agreement on (*a*) the motivations that underlie the trade-offs individuals are willing to make (i.e., the reasons that people want to protect ecosystems) or (*b*) what considerations beyond an assessment of trade-offs should enter into public policy decisions. By recasting the discussion to focus on the relevance and assessment of trade-offs, we believe that economists and ecologists will be able to find greater common ground for collaboration in normative analysis, even if they hold very disparate views on these other issues, and as a result will be able to make greater progress in efforts to evaluate policies designed to protect ecosystems and the services they provide.

2. THE CONCEPT OF ECOSYSTEM SERVICES

In the simplest economic terms, ecosystem services are analogous to other goods and services within the economy, all of which are produced through a combination of inputs and directly or indirectly generate utility. The production of ecosystem services can be represented by an "ecological production function," which is conceptually analogous to the standard production function used in economics to describe how inputs are combined to produce intermediate or final outputs. Although conceptually simple, the implementation of this concept, and what is included in it, often requires detailed and integrated understanding of ecology and economics.

We adopt a broad definition of the term ecosystem services that includes both intermediate and final services. Thus, following the Millennium Ecosystem Assessment (2005), we include not only food, fuelwood, fiber, biochemicals, genetic resources and fresh water (provisioning services), but also flood protection, climate regulation, human disease regulation, water purification, air quality maintenance, pollination, and pest control (regulating services); aesthetic, spiritual, cultural, educational, and recreational values of nature (cultural services); and the underlying services that support them, such as soil formation, nutrient cycling, and primary productivity (supporting services). Provisioning and cultural services are typically, though not always, outputs that directly affect human well-being. In contrast, regulating services are typically intermediate services, which are in turn inputs into other production processes. For example, pollination and pest-control services are inputs into the production of agricultural outputs. Supporting services, in economic terms, are akin to the infrastructure that provides the necessary conditions under which inputs can be usefully combined to provide intermediate and final goods and services of value to society. The Millennium Ecosystem Assessment definition of ecosystem services is useful in that it highlights the many ways in which ecosystems support human well-being.

3. INTERDISCIPLINARY EFFORTS TO STUDY ECOSYSTEM SERVICES

Although the term ecosystem services is relatively new, interest in combining economics and biological sciences to manage human interactions with nature has an extensive history. Combining biological growth models with an economic framework to define optimal use of natural resources dates back (at least) to Faustmann in 1849, who solved for the optimal forest rotation. The modern bioeconomic models of fisheries arose in the 1950s with seminal contributions from Gordon (1954), Scott (1955), and Schaefer (1957). There has been long-standing interest by economists in using insights from biological science, particularly the theory of evolution, in economics (e.g., Alchian 1950). Similarly, there has also been a long-standing interest by ecologists in using insights from economics to study biology and ecology. For example, microeconomic tools have been used to study optimal foraging and competition for scarce resources (e.g., Tilman et al. 2005, Vermeij 2004).

Despite this long history, the current interest in combining ecology and economics reflects a new level of concern, prompted by increasing recognition of the scale of the impact of human systems on the natural environment and the services it provides. With a global population approaching 7 billion people and the prospect of reaching 9 to 10 billion 50 years hence, as well as a large and growing global economy dependent on the use of fossil fuels, human activity is causing environmental changes not only locally but also globally (Millennium Ecosystem Assessment 2005). In addition, awareness of the critical importance of ecosystem services for human welfare has increased. The publication of the book *Nature's Services: Societal Dependence on Natural Ecosystems* (Daily 1997) and an article in *Nature* on the value of global ecosystem services (Costanza et al. 1997) did much to raise the profile of ecosystem services. A further boost was given by the Millennium Ecosystem Assessment (2005), which focused on the link between ecosystems and human well-being. Many ecologists saw work on ecosystem services as a means of putting ecosystem protection on a more equal footing with other (mainly commercial) interests and hence embraced it as a means of justifying ecosystem protection not just for its own sake but also for its contributions to human welfare.

Much of the work on valuing ecosystem services over the past several decades, especially at large regional or global scales, has been led by ecologists, and some of this work has been inconsistent with fundamental economic principles. Perhaps the most notable example of this is the paper by Costanza et al. (1997), which applied estimates of per hectare value derived from local-scale studies to all hectares of a given habitat type to generate an estimate of total economic value at a global scale. Summing up estimates across all habitat types, Costanza et al. (1997) estimated that global ecosystem services were worth \$33 trillion annually, far more than global GNP (\$18 trillion at that time). Bockstael et al. (2000), Pearce (1998), and Toman (1998), among others, have pointed out serious problems with this approach. Estimates from studies conducted at a local scale measure willingness to pay conditional on the premise that habitat in other places remains unchanged. When these estimates are scaled up to a regional or global scale, this condition is violated. In addition, if accurately conceived and measured, willingness to pay cannot exceed ability to pay (income or GNP) estimates. These flaws in early efforts to put a dollar value on global ecosystem services highlight the critical need for collaboration between ecologists and economists to ensure that assessments of ecosystem services and their value are scientifically sound.

Interest in the role and value of ecosystem services spurred several important national or international efforts to understand better the linkages between human systems and ecosystems. An example of such an effort is the Millennium Ecosystem Assessment, which was designed to "assess the consequences of ecosystem change for human well-being and to establish the scientific basis for actions needed to enhance the conservation and sustainable use of ecosystems and their contributions to human well-being" (Millennium Ecosystem Assessment 2005, p. v). This effort pointed to several key information/knowledge gaps, including the need for better information on the value of ecosystem services and greater integration of ecological and economic studies (Carpenter et al. 2006).

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

In addition, in 2002, the National Academy of Sciences (NAS) convened a panel composed primarily of economists and ecologists to evaluate economic methods for assessing the value of aquatic and related terrestrial ecosystems. The panel's report (National Research Council 2005) highlights the importance of valuing ecosystem services and the need for an integration of ecology and economics for that purpose. It concludes that the ability to value changes in ecosystem services varies significantly across contexts, for at least two reasons. First, the link between ecosystem structure and functions and the resulting provision of ecosystem services is better understood in some contexts than others. Second, in practice, some sources of value are easier to estimate than others. Valuation is especially challenging in contexts where there are multiple, interrelated services affected by a particular action or policy and where nonuse values are particularly important.

Following the establishment of the NAS panel, in 2003 the U.S. Environmental Protection Agency's Science Advisory Board initiated a study on Valuing the Protection of Ecological Systems and Services. The composition of the Science Advisory Board committee (known as C-VPESS) was broader than that of the NAS committee and included experts in economics and ecology as well as in decision science, engineering, law, philosophy, political science, and psychology. The committee's charge was also broader. C-VPESS set out to assess the state of the art and science in valuing the protection of ecological systems as well as ecosystem services, and unlike the NAS panel, it was not limited to considering valuation solely from an economic perspective. In addition, it was charged with specifically addressing ecological valuation needs and opportunities at the EPA. Because of differing views on the sources and nature of value, as well as the appropriate role of different valuation methods, the committee took five years to complete its work and to come to a reasonable consensus or, on some issues, a reasonable compromise regarding a proposed approach to ecological valuation. In its final report (U.S. Environmental Protection Agency 2009), the committee outlined an approach to valuation that included not only the use of economic valuation methods but also a role for other noneconomic approaches to valuation.

The Natural Capital Project, a joint venture among Stanford University, The Nature Conservancy, and the World Wildlife Fund, was launched in 2006. Unlike the NAS and C-VPESS committees, which focused on assessing the state of the art, the Natural Capital Project had a goal of "mainstreaming" ecosystem services into everyday decisions. A major thrust of the project has been to develop an integrated dynamic landscape model capable of predicting how various decisions will affect the joint provision of ecosystem services and species conservation (see Daily et al. 2009).

Our experiences with the Millennium Ecosystem Assessment, NAS panel, C-VPESS, and the Natural Capital Project have shaped our perspective on the opportunities for and challenges associated with combining economics and ecology in efforts to understand and address links between human and natural systems. The following sections draw from our collective experience and provide our perspective on these opportunities and challenges.

4. ECOLOGICAL AND ECONOMIC SYSTEMS

On the surface, ecology and economics appear to be quite different because of the difference in subject matter to which they are typically applied. Ecology studies the interactions of organisms with their (natural) environment. Economics studies the interaction of indi-

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

viduals, firms, and government in the (human) economy. However, the formal structures of the two disciplines are quite similar. Ecosystems and markets are composed of interacting (largely self-interested) agents whose actions jointly determine system outcomes, which evolve over time through conscious search for better opportunities or unconscious natural selection (Tilman et al. 2005). In **Table 1**, we provide a translation of parallel concepts and terms in ecology and economics. Similarities between the basic structures of the two fields make it relatively easy for an economist to understand ecology, and vice-versa. However, just as in learning a foreign language, learning to translate terms from one field to the other requires some time and effort. Translating between ecology and economics, however, is like translating between two languages from the same language family that share a common structure (e.g., French and Spanish) rather than translating between two languages with fundamentally different structures (e.g., English and Chinese).

The similarities between economic and ecological systems have led to productive cross-fertilization. For example, the basic principles of competition in the presence of resource scarcity developed in economics have been applied to the study of the foraging behavior of species. Likewise, principles from ecology relating to evolution and survival of species have been applied to economic systems where firms seek to adapt and survive in the long run (Tilman et al. 2005). These models do not seek to integrate economics and ecology; rather, they seek to apply concepts/insights from one discipline to the other.

The one area without a fairly direct translation between ecology and economics is normative analysis. Ecology is a positive science and does not have an analog to the normative framework of welfare economics. Nonetheless, ecologists sometimes advocate policy positions based on their personal views. These views then become implicit normative criteria for evaluating policy options. Other environmental disciplines have explicit normative criteria. For example, the mission of conservation biologists is to conserve the Earth's biological diversity. Policy outcomes can be ranked on the basis of how well they achieve this goal. Although the preferences of individuals or objectives such as conserving biodiversity might ultimately point in the same direction as an analysis based on an evaluation of social trade-offs, i.e., toward greater ecosystem protection, the degree of the prescribed protection will generally be different. For example, the potential threats to

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Table 1 Comparing ecology and economics

Feature	Ecology	Economics
Subject matter focus	Nature's economy	Human economy
Organizing systems	Ecosystems	Markets
Agents	Organisms	Individuals and firms
Forms of interactions among agents	Competitive and cooperative interactions in producer-consumer networks	Competitive and cooperative interactions among producers and consumers
Typical behavioral assumption	Maximize fitness/survival	Maximize utility/profit
Dynamics	Mutation and natural selection causing evolution	Innovation and entry/exit
Normative framework	None	Welfare economics

ecological systems posed by global climate change have led some natural scientists to advocate establishing stricter limits on atmospheric concentrations of greenhouse gases than those implied by most economic analyses (e.g., Nordhaus & Boyer 2000, Tol 2005; but see Stern 2007 for a different view).

Although it is useful to compare and contrast concepts from economics and ecology to highlight the potential for crossfertilization, the study of ecosystem services requires an explicit integration of the two disciplines. The remainder of this article discusses this integration.

5. INTEGRATING ECOLOGY AND ECONOMICS: AN OVERVIEW

A major impetus to integrate ecology and economics stems from the recognition of the important role that ecosystems play in providing goods and services that contribute to human welfare and of the impact that human actions have on ecosystems and the flow of services from very local to global scales. **Figure 1**, which builds on a figure developed in the Natural Capital Project (Daily et al. 2009), depicts a framework for integrating economics and ecology in the study of ecosystem services.

Starting from the top left oval, **Figure 1** highlights the fact that public policy decisions create incentives that affect the private decisions by firms and individuals (Link 1), which in turn result in actions that affect ecosystems (Link 2). These impacts include the effects of land clearing and habitat modification, changes in species populations from harvesting activities (hunting and fishing), changes in nutrient flows from fertilizer application and runoff, changes in the hydrological cycle from water withdrawals and operation of dams, changes in local air and water quality from discharge of pollutants, and changes in global climate from emissions of greenhouse gases. An understanding of the link between human decisions and ecosystem structure and function requires positive analysis integrating

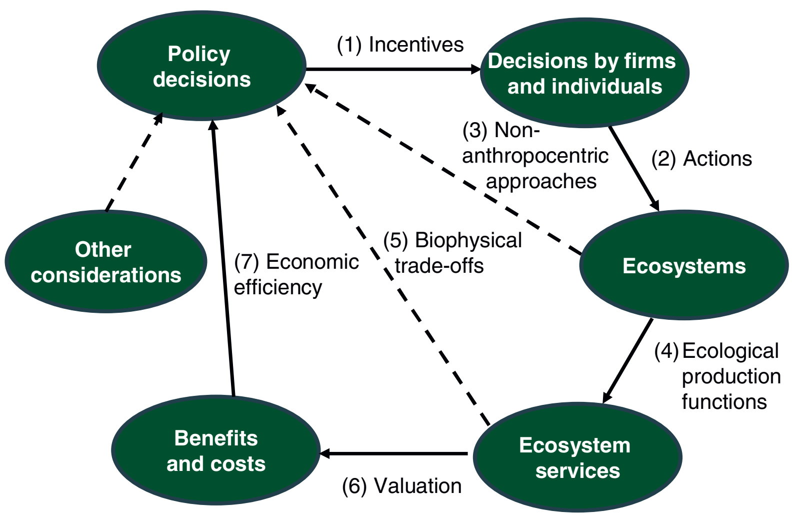

Figure 1

Framework for integrating economics and ecology in the study of ecosystem services. Solid lines indicate the links where the integration of the two disciplines can play a critical role.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

economics and ecology. Under a social choice rule where policy rankings are based solely on impacts on the ecosystem (as, for example, under a nonanthropocentric approach—see Section 8), this analysis would form the basis for policy evaluation and social choice (Link 3). Thus, for ecologists who advocate a social choice rule of this type, consideration of the effect of these ecosystem changes on humans is irrelevant. As a result, the potential for collaboration between economists and ecologists who hold this view is limited to positive impact analyses (Links 1 and 2), without any role for collaborative normative analysis based on a consideration of ecosystem services or trade-offs.

Of course, a fundamental element of the ecosystem services paradigm is the recognition that changes in ecosystem structure or function, in turn, influence the provision of ecosystem services enjoyed by humans (Link 4). Ecological production functions can be used to understand how various ecosystem services are produced and how changes in ecosystem conditions affect the provision of these services. The basic understanding required for developing ecological production functions comes from ecology and other natural sciences. The framework and endpoints, however, come from economics. The cooperation between ecologists and economists to develop ecological production functions is analogous to the cooperation between engineers and economists to define production functions in industrial processes. Ecological production functions can in turn be used to identify biophysical trade-offs that are analogous to the marginal rates of transformation embodied in production possibility frontiers. There is considerable scope for collaborative research between economists and ecologists in identifying relevant biophysical trade-offs. Some analysts may prefer to base policy decisions on consideration of impacts on the flow of various ecosystem services, recognizing the potential for trade-offs in those flows (Link 5) rather than assessing them in terms of the public's preferences. As with ecologists who adhere to a nonanthropocentric approach, there is little hope of collaboration on normative analysis between economists and ecologists who hold this view.

Link 6 in **Figure 1** captures the contribution of ecosystem services to human welfare. A fundamental principle of economics is that these contributions can be represented as the benefits of an increase in the flow of ecosystem services or the cost of a decrease in flows, where benefits and costs reflect the preferences of the individuals affected by the change. The value of the change in the flow of an ecosystem service, as defined in economics, is measured in terms of the trade-offs that those individuals are willing to make, regardless of their underlying motivations. Both market and nonmarket valuation methods can be used to estimate these trade-offs (see further discussion below). Information about the benefits and costs of changes in the flow of ecosystem services can then be used to assess the net benefits associated with alternative policy options or outcomes. Although few economists believe that information about the net benefits of alternatives should be the sole basis for social choice, nearly all believe that it should be an important consideration in public policy decisions (Arrow et al. 1996). The potential for collaborative work by ecologists and economists in normative analysis based on aggregate net benefits hinges on acceptance of this premise. Ecologists who believe that trade-offs are important, and that they should be assessed based on the public's preferences, can work with economists both on estimating the trade-offs individuals are willing to make (i.e., valuation—Link 6) and on net benefit analyses (Link 7), even if the economists and ecologists hold different views about the motivations underlying those trade-offs or the other considerations (such as distributional equity or process-related issues) that should also be considered when making policy choices.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

In the following sections, we discuss in further detail the opportunities and challenges of integrating economics and ecology that are reflected in **Figure 1**. We first discuss issues involved in positive analysis (understanding decisions and their effect on ecosystems and the services they provide). We then turn to the more difficult issues involved in normative analysis (valuing those changes and using information about values to assess alternative policy options or outcomes).

6. ANALYSIS OF INCENTIVES AND CONSEQUENCES OF DECISIONS ON ECOSYSTEMS

Economics is a behavioral science. Understanding choices and how choices change under different policies or market conditions is a central focus of economic analysis. Environmental economics analyzes incentives created by various environmental policies, such as taxes and cap-and-trade mechanisms. For the most part, this analysis has been directed at policies to achieve environmental improvement by reducing emissions of pollutants (e.g., Baumol & Oates 1988, Hanley et al. 1996). Increasingly though, environmental economists are asked for analysis of policies to create incentives to provide ecosystem services. Much of what we know about the effects of policies for pollution control is directly applicable to understanding policies to promote the provision of ecosystem services.

The standard set of environmental policy tools can be applied to the provision of ecosystem services. Price mechanisms, typically in the form of subsidies for provision of services (payments for ecosystem services), are the most common policy approaches. Examples of such programs include Costa Rica's program *Pago por Servicios Ambientales*, which pays landowners for carbon sequestration, habitat conservation, water quality, and aesthetics (Pagiola 2008). In the United States, the Conservation Reserve Program pays farmers to set land aside for conservation. Originally the Conservation Reserve Program was directed primarily toward preventing soil erosion, but it is now also being used to promote habitat conservation, water quality improvement, and other environmental benefits (see Claassen et al. 2008). Taxes on activities that degrade ecosystems and lessen the provision of services are also possible, but they are not often used in practice. The strong preference for using positive rewards rather than negative sanctions to provide ecosystem services stems largely from the necessity of working with many private landowners and the political difficulties of imposing and enforcing sanctions.

It is also possible to use quantity mechanisms, such as cap-and-trade, for the provision of ecosystem services. For example, programs such as wetland banking or tradable development rights can be used to maintain or ensure a given level of ecosystem protection (Boyd et al. 2001, National Research Council 2001). These programs set a quantity limit (cap) on the total amount of habitat destruction that can occur, but they allow trade in permits among landowners. Such policies can be set up to require no net loss, so that any habitat destruction must be offset by the restoration of habitat somewhere else (as in wetland banking).

Though most of the positive analysis of incentives for the provision of ecosystem services is like that in other areas of environmental economics, there are several issues that are novel or require greater attention when used in this context. Among issues that gain a higher profile when applied to ecosystem services are land use and management as well as the spatial and temporal dimensions of policies and responses.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org by 187.234.107.18 on 03/29/13. For personal use only.

Economists have developed econometric models that predict land use and land-use change based on the economic returns from alternative potential uses (e.g., Feng et al. 2006, Lubowski et al. 2006). These models are often based on econometric specifications of discrete choice models (Train 2003) and estimates of economic returns based on observables characteristics of the land (Lubowski 2002). Furthermore, this type of model can be used to predict land-use change in response to policies, such as payments for ecosystem services, which result in changes to the economic returns on alternative land uses (e.g., Lewis & Plantinga 2007, Nelson et al. 2008). Other studies of land use and land-use change have also incorporated the impacts of neighborhood effects on the relative utility of alternative decisions (e.g., Irwin & Bockstael 2002, 2004).

The provision of ecosystem services often depends on the spatial pattern of the decisions of many landowners. For example, species conservation outcomes are typically a function of the spatial pattern of habitat across a landscape. An important issue in such contexts is coordinating decisions across many landowners. Recent papers have investigated improving conservation solutions by making payments to landowners a function of decisions of all landowners in a neighborhood (e.g., Parkhurst et al. 2002, Parkhurst & Shogren 2007). This literature shows that it is possible to coordinate landowner decisions by making payoffs contingent on neighbors' decisions. In addition, empirical studies and simulations show that using biological information to target incentives can improve performance (e.g., Lewis et al. 2009, Nelson et al. 2008).

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

7. FROM ECOSYSTEMS TO SERVICES: ECOLOGICAL PRODUCTION FUNCTIONS AND ECOSYSTEM SERVICES

Beyond estimating how policies and market conditions affect choices that impact ecosystems, other necessary steps in the analysis of ecosystem services involve linking impacts to understanding ecosystem processes and further linking these processes to the provision of ecosystem services. In this section, we focus on the translation required from a primarily ecological focus on ecosystems to information about the provision of ecosystem services, i.e., the ecological production function.

The field of bioeconomics developed an extensive literature on the translation from biological systems to the provision of an economic good long before the term ecosystem service was in common usage. Starting from the 1950s, fisheries biologists and economists worked jointly to develop models of the economic returns from fish harvesting (for a review of bioeconomic models of fisheries and other renewable resources, see Clark 1990). Much of this work focused on a single good or service (e.g., fish harvest) and typically assumed a simple biological growth function (e.g., logistic growth). Over the past 30 years, the bioeconomic framework has been used to show how changes in ecosystems lead to changes in productivity through changes in the biological growth function (e.g., Barbier 1994, 2000, 2003; Barbier & Strand 1998; Barbier et al. 2002; Bell 1997; Ellis & Fisher 1987; Freeman 1991; Lynne et al. 1981; Swallow 1994).

A different strand of analysis starts from land use and habitat type to predict the provision of services. For example, there is a large natural science literature on the amount of carbon stored in a landscape as a function of land cover and land use (e.g., Brown & Schroeder 1999, Cairns et al. 2000, Canadell & Raupach 2008, Nascimento & Laurance 2002). Similarly, agricultural and timber production models use soil quality and the quantity of various inputs (e.g., fertilizer, water) to predict yield. Several papers have used

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

this approach to model agricultural and timber returns across space in conservation and ecosystem service assessments (e.g., Naidoo & Adamowicz 2006; Naidoo & Iwamura 2007; Polasky et al. 2005, 2008; and for a review, see Naidoo et al. 2006).

It is important to distinguish between an approach using land use and land cover to generate spatially explicit maps of particular ecosystem services based on ecological production functions, as done in the papers mentioned above, versus an approach to valuation of ecosystem services based on habitat types and benefit transfer. As noted above, Costanza et al. (1997) used estimates of the value of ecosystem services per hectare at specific locations within a habitat type, and they multiplied these estimates by the total amount of the habitat type to generate global estimates of value for ecosystem services. Other recent papers have used a similar approach to generating estimates of the value of ecosystem services (e.g., Ingraham & Foster 2008, Troy & Wilson 2006, Turner et al. 2007). Assuming a constant per hectare value of ecosystem services by habitat type ignores issues of rarity, spatial configuration, size, habitat quality and condition, and demand for ecosystem services generated by the number of nearby people or their preferences, all of which should matter in the determination of both the quantity of services produced and the value of those services (Nelson et al. 2009). For example, for ecosystem services such as storm protection along coastlines, Barbier et al. (2008) and Koch et al. (2009) examined the details of the pattern and amount of habitat, including temporal and spatial variation, to show why ecosystem service provision is unlikely to be linearly related to the amount of habitat.

Recent work integrating ecological production functions into an economic framework has made several important advances. First, models have become spatially explicit. In a series of papers, Sanchirico & Wilen (1999, 2001, 2005) developed a bioeconomic model with distinct habitat patches and dispersal of a harvested species between patches. Other papers have developed spatially explicit models of ecosystem services as a function of land use and land cover (e.g., Lewis & Plantinga 2007, Nelson et al. 2008, 2009, Polasky et al. 2005, 2008). Second, models have expanded beyond a single service focus to consider provision of the bundle of ecosystem services jointly produced by the ecosystem (e.g., Antle & Stoorvogel 2006; Boody et al. 2005; Coiner et al. 2001; Naidoo & Ricketts 2006; Nelson et al. 2008, 2009). Such work allows an analysis of the production possibility frontier for services and the trade-offs in provision among services (e.g., Jackson et al. 2005; Nelson et al. 2008, 2009; Polasky et al. 2008).

8. TOWARD NORMATIVE ANALYSIS: VALUING CHANGES IN ECOSYSTEM SERVICES

As noted above, a critical problem motivating recent interest in integrating ecology and economics is the recognition of the need to incorporate more fully the value of ecosystem services into policy decisions. By not valuing changes in these services in formal policy evaluations (e.g., benefit-cost analyses), ecosystem services are effectively assigned a value of zero (Daily et al. 2000). All the interdisciplinary efforts discussed in Section 3 emphasize the importance of valuing ecosystem services to better inform decision making. In this section, we discuss recent efforts to combine economics and ecology in the valuation of ecosystem services. However, as also noted, valuation that is closely tied to policy evaluation raises issues about the sources and nature of value as well as the appropriate basis for evaluating alternatives, which can be significant roadblocks to collaboration between economists and researchers from other disciplines, including ecology. This section also discusses some of these issues and the challenges they present.

The valuation of the change in ecosystem services resulting from a particular policy choice includes three basic components: (*a*) identification of the services provided by the ecosystems that are affected by the policy choice, (*b*) prediction of the changes in the flow of these services measured in biophysical terms, and (*c*) estimation of the value of these predicted changes measured in terms of human welfare (National Research Council 2005). The first two components rely primarily on positive analysis of the sort described in the preceding sections, including use of economic models that predict how policy choices affect ecosystems through changes in incentives and decisions as well as the use of ecological models that predict how changes in ecosystems affect the provision of ecosystem services (see Links 1, 2, and 4 in **Figure 1**). However, the third component, estimating value, requires the adoption and application of a specific concept of value.

The literature on environmental values identifies a number of different concepts along various dimensions of value: (*a*) intrinsic and instrumental, (*b*) anthropocentric and biocentric (or ecocentric), and (*c*) utilitarian and deontological (Rolston 1991). Something has intrinsic value if it is valued as an end in itself, whereas it has instrumental value if it is valued as a means to achieve a desired objective. For anthropocentric values, only humans have intrinsic value, whereas for biocentric (ecocentric) values, the natural world beyond humans has intrinsic value. Utilitarian values are defined in terms of how they contribute to the desired objective, which for anthropocentric values is assumed to be the maximization of human utility or welfare. In contrast, under deontological approaches, values are based on rights and obligations rather than consequences or contributions to an objective.

In welfare economics, values are defined in instrumental anthropocentric utilitarian terms and reflect trade-offs individuals are willing to make. In the context of valuing ecosystem services, values reflect the willingness of the affected individuals to trade the increase or decrease in ecosystem services for a decrease or increase in other goods and services. When values are expressed in monetary terms, they are measured in terms of compensating or equivalent variation, typically measured as willingness to pay or willingness to accept (Just et al. 2004). Compensating (equivalent) variation is defined as the amount of money that would exactly compensate individuals for a given change, i.e., allow them to maintain the same level of utility as before (after) the change.

The concept of ecosystem services fits easily within the anthropocentric framework underlying this concept of value, because it is based on the notion that ecosystems contribute to human welfare and hence have instrumental value. In addition, it is conceptually straightforward to combine ecology and economics to value ecosystem services using economic valuation techniques. Ecology provides the understanding of ecological systems necessary to construct ecological production functions that are used to predict changes in the flow of these services, and economics defines services and provides methods for estimating the value of changes in these services.

There is a vast literature within economics on the valuation of market and nonmarket goods (see Freeman 2003), which includes applications to valuing ecosystem services. In fact, most of the methods of valuing the environment developed by economists are directly applicable to valuing ecosystem services. As noted above, ecological inputs can be valued in the same way that other inputs (such as labor and capital) are valued. For example, because wetlands can serve as a breeding ground and nursery for marine populations such as shrimp and crabs, the quantity and/or quality of wetlands can be viewed as an input

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org by 187.234.107.18 on 03/29/13. For personal use only.

affecting the productivity of the fishery. Several studies have estimated the value of the contribution of coastal wetlands to increased productivity of commercial or recreational fisheries (e.g., Barbier 1994, 2000, 2003; Barbier & Strand 1998; Barbier et al. 2002; Batie & Wilson 1978; Bell 1997; Ellis & Fisher 1987; Freeman 1991; Lynne et al. 1981; Swallow 1994). Other ecosystem services provided by wetlands that have been valued in a similar way include increased productivity of waterfowl (Hammack & Brown 1974), storm protection (Barbier et al. 2008, Costanza et al. 2008, Sathirathai & Barbier 2001), groundwater recharge (Acharya 2000, Acharya & Barbier 2000, 2002), and water quality improvements (Breaux et al. 1995, Kahn 1987, Kahn & Kemp 1985, McConnell & Strand 1989, Wu et al. 2000). A number of other papers have applied hedonic property price models or contingent valuation to estimate the value of wetlands (e.g., Bin & Polasky 2005, Boyer & Polasky 2004, Doss & Taff 1996, Earnhart 2001, Hammitt et al. 2001, Hoehn et al. 2002, Mahan et al. 2000, Morrison et al. 1999, Woodward & Wui 2001).

Use of an integrated ecological-economic model for valuation allows for a more realistic prediction of changes in the flow of ecosystem services that result from a given policy alternative. It also allows for an examination of the complementarity or trade-offs associated with the provision of various ecosystem services. Integrated models have been applied in a variety of contexts, including eutrophication (Batabyal et al. 2003, Carpenter et al. 1999, Hart 2003, Iwasa et al. 2007), habitat protection and restoration (Ando et al. 1998; Balmford et al. 2000; Montgomery et al. 1994, 1999; Naidoo et al. 2006; Nalle et al. 2004; Polasky et al. 2005, 2008; Wu et al. 2003), biodiversity (Brock & Xepapadeas 2003, Polasky & Solow 1995, Simpson et al. 1996), and the joint provision of multiple ecosystem services (e.g., Antle & Stoorvogel 2006; Coiner et al. 2001; Naidoo & Ricketts 2006; Nelson et al. 2008, 2009).

The existing literature illustrates the potential for applying standard economic valuation methods to the valuation of ecosystem services. However, attempts to apply economic valuation methods to a broad set of interrelated services, both intermediate and final, face a number of practical difficulties that arise in either predicting changes in service flows or estimating the associated values. Among the more important practical difficulties in assessing the value of ecosystem services are the following (National Research Council 2005, US Environmental Protection Agency 2009):

- Limited understanding of the structure and functions of the relevant ecosystem(s), including important dynamics, nonlinearities and threshold effects, irreversibilities, and interconnections
- Lack of multiproduct, ecological production functions to quantitatively map ecosystem structure and function to a flow of services that can then be valued
- Limited public understanding of the services, and ultimately the contributions to welfare, provided by ecosystems
- Reliance on valuation methods that do not adequately capture the role of important underlying ecological relationships

Despite these challenges, at least at a conceptual level, economics and ecology can be integrated to measure the value of ecosystem services using the economic concept of value, as illustrated by the studies referenced above. However, in interdisciplinary contexts, philosophical differences often arise over the sources of value and the appropriate way to measure it. Although well-accepted within neoclassical economics, the welfare economic

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

approach to defining and estimating values discussed above is not universally endorsed (Goulder & Kennedy 1997, Norton & Noonar 2007, Turner 1999, Winkler 2006). Key areas of divergence in views about defining values relate to the questions addressed in the following sections.

8.1. Should Values Be Defined in Anthropocentric Terms?

This question is essentially about who/what has intrinsic value. Some people believe that the natural world has value apart from how it contributes to human welfare. Under this biocentric or ecocentric view, all species may have intrinsic value or the condition of the ecosystem as a whole (health, integrity, resilience) may have intrinsic value (Callicott 1989, Light & Rolston 2003, Norton 1987). Although anthropocentric and nonanthropocentric systems of value can lead to similar policy prescriptions (see further discussion below), they are fundamentally different approaches. Even the concept of existence value, which economists sometimes describe as capturing the intrinsic value of nature, is still fundamentally defined from an anthropocentric perspective because it is based on human preferences (Edwards 1992). Examples of alternative definitions of value that are not explicitly based on human preferences include values defined in terms of energy content or contributions to a prespecified conservation goal such as biodiversity preservation or sustainability (Costanza 2000, 2004; Grossman & Comer 2004; Winkler 2006). If the underlying motivation for these goals is their importance for humans, the associated value concepts are anthropocentric. If alternatively, the motivation is based on intrinsic values of nature, they should be interpreted as biocentric or ecocentric concepts of value.

8.2. Should Values Be Defined in Utilitarian or Deontological Terms?

Even with agreement on an anthropocentric approach to value, the notion of defining values in terms of consequences or contributions to human welfare, rather than rights or moral obligations, is not universally accepted (Anderson 1993; Spash 1997, 2006). For example, some might argue that a given ecosystem should be protected because all people have a right to the services it provides (e.g., access to clean water or clean air). Although in principle the utilitarian approach allows for the possibility of lexicographical preferences, under which some considerations "trump" all others, in general it implies substitutability between ecosystem services and other goods and services that also contribute to utility. For example, the economic (utilitarian) concept of the existence value of a species is not defined in terms of its intrinsic right to exist but rather in terms of the amounts of other goods and services that an individual is willing to give up to ensure its existence. Deontological rights-based approaches do not embody principles of substitutability and acceptable trade-offs, except when there is a conflict between rights.

8.3. Who Should Determine Value When the Public Does Not Have Full Information?

Consumer sovereignty assumes that individuals are the best judges of what is in their own best interest and that their preferences are valid regardless of how they are viewed by others. Under consumer sovereignty, public preferences rather than those of a set of experts or government officials determine the relevant values (National Research Council 1996).

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

However, the public may not understand or “appreciate” the contributions of ecosystems. For example, the public might not understand the importance of microorganisms that contribute to soil formation and hence, if asked, might assign a value of zero to changes in their abundance. This type of answer can cause some to suggest that expert judgment, rather than public values, should be the basis for policy decisions. The economic approach to this information problem would not be to substitute expert judgment for public values; rather, it would be either to provide the public with the scientific information necessary to understand the contributions provided by the microorganisms, or to recast the valuation question in terms of final services that the public does understand and value (e.g., seek to estimate the value of the change in water quality, crop yield from improved soil fertility, or other final services resulting from changes in microorganism abundance).

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

8.4. Are Preferences over Ecosystem Services Well-Defined or Constructed?

A foundational assumption of economics is that individuals are rational and have well-defined preferences over bundles of goods and services, which they reveal consistently in a number of ways (e.g., through behavior or surveys). An alternative view within parts of psychology and decision science is that preferences are constructed through a cognitive process, particularly for complex and unfamiliar goods (Lichtenstein & Slovic 2006). This view implies that different contexts or settings can yield varying, and sometimes inconsistent, rankings of alternatives. These alternative views imply not only fundamentally different theoretical constructs for value but also a different set of methods that are deemed appropriate for value elicitation (Gregory et al. 1993).

8.5. Are Values Associated Only with Changes in Services?

In economics, the concept of a change is fundamental to defining values. Economists measure the value of a change from a baseline (prepolicy) level of some price or quantity variable to an alternative (postpolicy) level (Just et al. 2004). However, within some realms, values are conceived as a set of principles, concepts, or beliefs that guide decisions and evaluations (see, for example, Dietz et al. 2005, Hitlin & Piliavin 2004). In addition, even when defined in terms of contributions to human welfare, some valuation studies have sought to estimate the value of an entire ecosystem or even the set of ecosystems that make up a given landscape at a local, regional, or global scale (Costanza et al. 1997). Valuing an entire ecosystem is consistent with an economic approach if the value of the ecosystem is based on a comparison of human welfare with and without the ecosystem. However, typically, eliminating the ecosystem(s) is neither a policy-relevant change (Daily et al. 2000), nor a sensible comparison (Toman 1998). Even if it were, the standard economic valuation methods, which were derived primarily in the context of marginal analysis, would not likely adequately capture the associated value.

8.6. What Is the Relevant Baseline?

Although economic values are defined in terms of a change from some current or alternative level, they do not embody any judgment about whether that level is “good” or “bad,” or about the process by which the level was reached (in particular, whether it is higher or lower than some other reference level). However, when an ecosystem is already degraded, some might advocate a baseline defined in terms of an historical level that was deemed "acceptable" (e.g., the system 50 or 100 years ago), or perhaps even a "pristine" level (the system prior to human contact). This can reflect an implicit goal of restoration to that baseline. Defining the baseline may also be tied to views on entitlements or rights such as a view that the public has the right to clean water or a pristine habitat.

The debates surrounding the above issues reflect different views about the sources and nature of value. Focusing on these differences can lead to lengthy debates that generally cannot be resolved through scientific inquiry. In fact, much of the discussion and debate that occupied the C-VPESS for five years in its effort to produce a consensus report surrounded these issues. In contrast, the NRC committee, whose charge explicitly focused on the economic approach to valuation, avoided these debates and as a result reached consensus much more quickly.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org by 187.234.107.18 on 03/29/13. For personal use only.

9. POLICY EVALUATION FOR COMBINED ECOLOGICAL-ECONOMIC SYSTEMS

A key motivation for conducting valuation of ecosystem services is to improve public policy decisions. This requires evaluation of policy options using some criteria. Different criteria can be used, and these often reflect various views about why people do or should value protection of ecosystem services.

The standard economic efficiency approach to policy evaluation (Link 7 in **Figure 1**), as embodied in benefit-cost analysis, uses net benefits based on estimates of market and nonmarket values to evaluate alternatives. An economically efficient policy is one that maximizes the aggregate net benefits of individuals in society. Efficiency does not require that all people are actually better off compared with the baseline, only that the gains exceed the losses so that the "winners" from the policy change could, in principle, compensate the "losers" in a way that would make everyone better off. Economists generally recommend using benefit-cost analysis as an input into societal decision making rather than as a decision rule, because other considerations not reflected in measures of aggregate net benefits, such as equity considerations, are also important in social choice (Arrow et al. 1996).

There is now a large literature that has applied standard benefit-cost principles to evaluate policy options using integrated ecological-economic models: e.g., Carpenter et al. (1999) on lake eutrophication; Barbier et al. (2008) on coastal protection; Archer & Shogren (2001) on pesticide use; Settle & Shogren (2006) on the introduction of non-native species; Wu et al. (2003) on restoration of salmon habitat; Costello & Polasky (2008), Eichner & Tschirhart (2007), and Sanchirico & Wilen (1999, 2001, 2005) on harvesting marine species. The explicit incorporation of ecological relationships within a benefit-cost analysis will generally highlight the importance of considering uncertainty, irreversibility, risk aversion, and threshold effects, all of which are concerns often raised by ecologists. These considerations can be incorporated through the use of expected utility theory based on appropriately specified utility functions that incorporate risk aversion and probability distributions that incorporate threshold effects and nonlinear dynamics. In addition, the concept of (quasi) option value incorporates irreversibility and uncertainty into an economic efficiency paradigm (Arrow & Fisher 1974, Dixit & Pindyck 1994). Option-value arguments, which place value on avoiding irreversible decisions until uncertainty is resolved, are similar to noneconomists' calls for the use of the precautionary principle, which places a high burden of proof on proponents of actions that might cause irreversible harm (for an economic evaluation of the precautionary principle, see Gollier et al. 2000, Gollier & Treich 2003). Similar arguments can be made for "safe minimum standards" (Ciriacy-Wantrup 1952) to avoid decisions that might result in crossing a threshold with serious negative consequences. Arguments for other risk-averse approaches to decision making can be justified if the uncertainty about the ecological-economic systems is so profound that probability functions over potential outcomes cannot be defined in any meaningful way (termed Knightian uncertainty) (for a recent discussion, see Margolis & Naevdal 2008).

Concerns about ecosystem degradation and resource depletion have also generated calls for using sustainability as a broad normative goal (see, e.g., Arrow et al. 2004, Costanza 1991). Sustainability is generally defined as meeting the "needs of the present without compromising the ability of future generations to meet their own needs" (World Commission on Environment and Development 1987). Although the specific interpretation of this goal can vary (e.g., Pezzey & Toman 2002), sustainability is often viewed as an alternative to standard economic efficiency as a normative goal (e.g., Arrow et al. 2004, Ayres 2008, Costanza 1991). However, Heal (1998) derived sustainability as an efficient outcome when society places a positive value on the very long run and people intrinsically value environmental assets. Including environmental stocks in the utility function can also yield a sustainable solution to an otherwise standard model of the efficient use of an exhaustible resource (Krautkraemer 1985). As these works show, a potentially fruitful approach is to characterize cases in which efficient paths are also sustainable.

Because of the complexity of joint ecological-economic systems, some natural scientists prefer to think about system resilience rather than any deliberate attempt to maximize expected net benefits (e.g., Gunderson & Holling 2002). One definition of resilience is the ability of a system to withstand shocks without fundamentally changing the character of the system (for a more complete set of definitions of resilience, see **http://www.resalliance.org**). Under the resilience approach, actions are desirable if they increase an ecological system's ability to tolerate disturbances or promote flexible human response through learning or adaptation. The resilience approach is a general philosophy of what types of system properties are important, but it does not have well-developed decision-making rules (analogous, for example, to maximizing expected utility).

Despite the differing principles or philosophies that can underlie various decision rules, the various rules can lead to policy prescriptions that are similar, at least qualitatively. For example, Eichner & Pethig (2006) showed that protection of nature can be justified through a variety of channels, using both anthropocentric and non-anthropocentric social welfare functions. Likewise, Arrow et al. (2004) argued that some policies designed to promote efficiency, such as those designed to correct the underpricing of natural resources, can contribute to both efficiency and sustainability goals. Perrings (1995) showed that improvement in ecological resilience increases the likelihood that economic development will be sustainable.

Thus, different perspectives on appropriate normative criteria may have little practical difference in terms of decision making. For example, a person with anthropocentric nonuse values such as existence value and a person with biocentric values may favor similar management alternatives, despite the differences in their underlying motivations. In each case, the person may place much weight on the conservation of biodiversity and ecosystems. Nonetheless, the process of getting to a common policy prescription can be

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

challenging when individuals apply different evaluation criteria and when the debate focuses not on the commonality of the final prescription, but rather on the ideological differences underlying the various ways of reaching that end.

10. INCREASING THE SCOPE FOR COLLABORATION IN NORMATIVE ANALYSIS

The previous sections overview some of the challenges that can arise in valuing ecosystem services for use in policy evaluations. The process of trying to overcome these challenges can be long, frustrating, and discouraging for the parties involved. However, we believe that we could take a significant step toward this end by recasting or reframing the discussion. We suggest the following in the spirit of providing a practical rather than a purist approach.

As noted above, much of the debate over normative analysis of ecosystem services ultimately stems from differing views about why ecosystems are important and should be protected. Views that do not consider the benefits of ecosystems to human well-being (see Links 3 and 5 in **Figure 1**) cannot be reconciled with an economic efficiency approach to policy evaluation; hence, they preclude integration of economics and ecology in normative analyses. However, as the review above suggests, there is considerable potential for integrating economics and ecology in valuing ecosystem services and incorporating those values into policy evaluations that consider trade-offs and economic efficiency (Links 6 and 7 in **Figure 1**). As a step toward this goal, we suggest that, rather than focusing on why individuals value nature and protection of ecological systems, economists and ecologists seeking to collaborate on normative analyses should focus on reaching agreement on the basis of the following premises:

1. Evaluation of trade-offs is an important consideration in evaluating policy options.
2. Trade-offs should be assessed on the basis of what affected individuals are willing to give up to secure, or demand in exchange for foregoing, an ecological improvement, regardless of why they are willing to make these exchanges.
3. Individuals can reveal the exchanges or trade-offs they are willing to make through actual and/or stated choices.

Acceptance of these premises would not require adoption of a particular individual value system or adherence to a set of philosophical or ethical principles relating to people's values. In addition, it would not require agreement about why individuals want to protect ecosystems and would be willing to give up other goods and services toward that end. For these reasons, we believe it would be easier to get agreement among an interdisciplinary group of researchers on these premises than to resolve all of the issues and debates that can hamper collaborative efforts, as discussed in the previous section. We believe that refocusing the discussion around the relevance and assessment of trade-offs could allow interdisciplinary groups of people with differing views to move forward in integrating ecology and economics in normative analyses.

11. CONCLUSION

In this paper, we draw on our collective experience with interdisciplinary committees working on ecosystem services to reflect on both the opportunities and challenges asso-

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

ciated with integrating economics and ecology in this context. In terms of positive analysis, there is rapid growth in research that combines the two disciplines to provide insight and better understanding of the bidirectional linkage between economic and ecological systems. This research is a crucial part of addressing the growing large-scale environmental challenges facing the world today. The integration of ecology and economics is equally important, but potentially much more difficult, in normative analyses, because of different views regarding why people value nature and the criteria that should be used in making policy decisions relating to ecosystem protection. Some of these differences are irreconcilable, even though the policy prescriptions implied by the different perspectives can be very similar. Nonetheless, we believe there is still considerable room for collaboration among economists and ecologists in normative analysis, even when they do not fully agree on these two issues. Successful collaboration does not require full agreement on individuals' underlying motivations or appropriate social choice rules. Rather, it requires agreement that the trade-offs that affected individuals are willing to make to preserve or enhance ecosystem services are relevant for public policy decisions and that those trade-offs are reflected in the choices individuals make. Researchers who can agree on these basic premises and then focus collaborative efforts on the study of trade-offs should be able to collaborate successfully on normative analyses, even if they hold disparate views on other issues. Our hope is that such an approach will expand the scope for collaboration by highlighting common ground rather than philosophical and other differences and thus help in moving forward interdisciplinary research on ecosystem services.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

DISCLOSURE STATEMENT

The authors are not aware of any affiliations, memberships, funding, or financial holdings that might be perceived as affecting the objectivity of this review.

ACKNOWLEDGMENT

We thank V. Kerry Smith for very useful comments on an earlier draft of this article, which greatly improved its contents.

LITERATURE CITED

Acharya G. 2000. Approaches to valuing the hidden hydrological services of wetland ecosystems. *Ecol. Econ.* 35:63–74

Acharya G, Barbier EB. 2000. Valuing groundwater recharge through agricultural production in the Hadejia'Jama'are wetlands in Northern Nigeria. *Agric. Econ.* 22:247–59

Acharya G, Barbier EB. 2002. Using domestic water analysis to value groundwater recharge in the Hadejia'Jama'are Floodplain in northern Nigeria. *Am. J. Agric. Econ.* 84:415–26

Alchian AA. 1950. Uncertainty, evolution, and economic theory. *J. Polit. Econ.* 58:211–21

Anderson JC. 1993. Species equality and the foundations of moral theory. *Environ. Values* 2:347–65

Ando A, Camm J, Polasky S, Solow A. 1998. Species distributions, land values, and efficient conservation. *Science* 279:2126–28

Antle JM, Stoorvogel JJ. 2006. Incorporating systems dynamics and spatial heterogeneity in integrated assessment of agricultural production systems. *Environ. Dev. Econ.* 11:39–58

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Archer DW, Shogren JF. 2001. Risk-indexed herbicide taxes to reduce ground and surface water pollution: an integrated ecological economics evaluation. *Ecol. Econ.* 38:227–50

Arrow KJ, Cropper ML, Eads GC, Hahn RW, Lave LB, et al. 1996. Is there a role for benefit-cost analysis in environmental, health, and safety regulation? *Science* 272:221–22

Arrow K, Dasgupta P, Goulder L, Daily G, Ehrlich P, et al. 2004. Are we consuming too much? *J. Econ. Perspect.* 18:147–72

Arrow KJ, Fisher AC. 1974. Environmental preservation, uncertainty, and irreversibility. *Q. J. Econ.* 88:312–19

Ayres RU. 2008. Sustainability economics: Where do we stand? *Ecol. Econ.* 67:281–310

Balmford A, Gaston KJ, Rodrigues ASL, James A. 2000. Integrating conservation costs into international priority setting. *Conserv. Biol.* 11:597–605

Barbier EB. 1994. Valuing environmental functions: tropical wetlands. *Land Econ.* 70:155–73

Barbier EB. 2000. Valuing the environment as input: applications to mangrove-fishery linkages. *Ecol. Econ.* 35:47–61

Barbier EB. 2003. Habitat-fishery linkages and mangrove loss in Thailand. *Contemp. Econ. Policy* 21:59–77

Barbier EB, Koch EW, Silliman BR, Hacker SD, Wolanski E, et al. 2008. Coastal ecosystem-based management with non-linear ecological functions and values. *Science* 319:321–23

Barbier EB, Strand I. 1998. Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. *Environ. Resour. Econ.* 12:151–66

Barbier EB, Strand I, Sathirathai S. 2002. Do open access conditions affect the valuation of an externality? Estimating the welfare effects of mangrove-fishery linkages in Thailand. *Environ. Resour. Econ.* 21:343–67

Batabyal AA, Kahn JR, O'Neill RV. 2003. On the scarcity value of ecosystem services. *J. Environ. Econ. Manag.* 46:334–52

Batie SS, Wilson JR. 1978. Economic values attributable to Virginia's coastal wetlands as inputs in oyster production. *South. J. Agric. Econ.* 10:111–8

Baumol WJ, Oates WE. 1988. *The Theory of Environmental Policy.* New York: Cambridge Univ. Press. 2nd ed.

Bell FW. 1997. The economic value of saltwater marsh supporting marine recreational fishing in the Southeastern United States. *Ecol. Econ.* 21:243–54

Bin O, Polasky S. 2005. Evidence on the amenity value of wetlands in a rural setting. *J. Agric. Appl. Econ.* 37:589–602

Bockstael N, Freeman AM III, Kopp RJ, Portney PR, Smith VK. 2000. On measuring economic values for nature. *Environ. Sci. Technol.* 34:1384–89

Boody G, Vondracek B, Andow DA, Krinke M, Westra J, et al. 2005. Multifunctional agriculture in the United States. *Bioscience* 55:27–38

Boyd J, King D, Wainger L. 2001. Compensation for lost ecosystem services: the need for benefit-based transfer ratios and restoration criteria. *Stanford Environ. Law J.* 20:393–412

Boyer T, Polasky S. 2004. Valuing urban wetlands: a review of non-market valuation studies. *Wetlands* 24:744–55

Breaux AM, Farber S, Day J. 1995. Using natural coastal wetlands systems for wastewater treatment and economic benefit analysis. *J. Environ. Manag.* 44:285–91

Brock WA, Xepapadeas A. 2003. Valuing biodiversity from an economic perspective: a unified economic, ecological and genetic approach. *Am. Econ. Rev.* 93:1597–614

Brown S, Schroeder PE. 1999. Spatial patterns of aboveground production and mortality of woody biomass for eastern US forests. *Ecol. Appl.* 9:968–80

Bulte EH, Lipper L, Stringer R, Zilberman D. 2008. Payments for ecosystem services and poverty reduction: concepts, issues, and empirical perspectives. *Environ. Dev. Econ.* 13(3):245–54

Cairns MA, Haggerty PK, Alvarez R, De Jong BHJ, Olmsted I. 2000. Tropical Mexico's recent land-use change: a region's contribution to the global carbon cycle. *Ecol. Appl.* 10:1426–41

Callicott JB. 1989. *In Defense of the Land Ethic: Essays in Environmental Philosophy*. Albany, NY: SUNY Press

Canadell JG, Raupach MR. 2008. Managing forests for climate change mitigation. *Science* 320:1456–57

Carpenter S, Defries R, Dietz T, Mooney HA, Polasky S, et al. 2006. Research needs revealed by the Millennium Ecosystem Assessment. *Science* 314:257–58

Carpenter SR, Ludwig D, Brock WA. 1999. Management of eutrophication for lakes subject to potentially irreversible change. *Ecol. Appl.* 9:751–71

Ciriacy-Wantrup SV. 1952. *Resources Conservation: Economics and Policy*. Berkeley, CA: Univ. Calif. Press

Claassen R, Cattaneo A, Johansson R. 2008. Cost-effective design of agri-environmental payment programs: U.S. experience in theory and practice. *Ecol. Econ.* 65:737–52

Clark C. 1990. *Mathematical Bioeconomics*. New York: Wiley. 2nd ed.

Coiner C, Wu J, Polasky S. 2001. Economic and environmental implications of alternative landscape designs in the Walnut Creek Watershed of Iowa. *Ecol. Econ.* 38:119–39

Costanza R, ed. 1991. *Ecological Economics: The Science and Management of Sustainability*. New York: Columbia Univ. Press

Costanza R. 2000. Social goals and the valuation of ecosystem services. *Ecosystems* 3:4–10

Costanza R. 2004. Value theory and energy. In *Encyclopedia of Energy*, Vol. 6, ed. C Cleveland, pp. 337–46. Amsterdam: Elsevier Sci.

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M. 1997. The value of the world's ecosystem services and natural capital. *Nature* 387:253–60

Costanza R, Pérez-Maqueo O, Martinez ML, Sutton P, Anderson SJ, Mulder KV. 2008. The value of coastal wetlands for hurricane protection. *AMBIO J. Hum. Environ.* 37:241--48

Costello C, Polasky S. 2008. Optimal harvesting of stochastic spatial resources. *J. Environ. Econ. Manag.* 56:1–18

Daily GC, ed. 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Washington, DC: Island Press

Daily G, Ellison K. 1999. *The New Economy of Nature: The Quest to Make Conservation Profitable*. Cheltenham, UK: Edward Elgar

Daily G, Polasky S, Goldstein J, Kareiva PM, Mooney HA, et al. 2009. Ecosystem services in decision-making: time to deliver. In *Frontiers in Ecology and the Environment*. In press

Daily GC, Söderqvist T, Aniyar S, Arrow K, Dasgupta P, et al. 2000. The value of nature and the nature of value. *Science* 289:395–96

Dietz T, Fitzgerald A, Shwom R. 2005. Environmental values. *Annu. Rev. Environ. Resour.* 30:335–72

Dixit AK, Pindyck RS. 1994. *Investment under Uncertainty*. Princeton, NJ: Princeton Univ. Press

Doss CR, Taff SJ. 1996. The influence of wetland type and wetland proximity on residential property values. *J. Agric. Resour. Econ.* 21:120–29

Earnhart D. 2001. Combining revealed and stated preferences methods to value environmental amenities at residential locations. *Land Econ.* 77:12–29

Edwards S. 1992. Rethinking existence values. *Land Econ.* 68:120–22

Eichner T, Pethig R. 2006. Efficient non-anthropocentric nature protection. *Soc. Choice Welfare* 26:47–74

Eichner T, Tschirhart J. 2007. Efficient ecosystem services and naturalness in an ecological/economic model. *Environ. Resour. Econ.* 37:733–55

Ellis GM, Fisher AC. 1987. Valuing the environment as input. *J. Environ. Manag.* 25:149–56

Feng H, Kurkalova LA, Kling CA, Gassman PW. 2006. Environmental conservation in agriculture: land retirement vs. changing practices on working land. *J. Environ. Econ. Manag.* 52:600–14

Freeman AM. 1991. Valuing environmental resources under alternative management regimes. *Ecol. Econ.* 3:247–56

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Freeman AM. 2003. *The Measurement of Environmental and Resource Values*. Washington, DC: Resour. Future

Gollier C, Treich N. 2003. Decision-making under scientific uncertainty: the economics of the Precautionary Principle. *J. Risk Uncertain.* 27:77–103

Gollier C, Treich N, Jullien B. 2000. Scientific progress and irreversibility: an economic interpretation of the precautionary principle. *J. Public Econ.* 75:229–53

Gordon HS. 1954. The economic theory of a common-property resource: the fishery. *J. Polit. Econ.* 62:124–42

Goulder LH, Kennedy D. 1997. Valuing ecosystem services: philosophical bases and empirical methods. In *Nature's Services: Societal Dependence on Natural Ecosystems*, ed. GC Daily, pp. 23–47. Washington, DC: Island Press

Gregory R, Lichtenstein S, Slovic P. 1993. Valuing environmental resources: a constructive approach. *J. Risk Uncertain.* 7:177–97

Grossman DH, Comer PJ. 2004. Setting priorities for biodiversity conservation in Puerto Rico. *NatureServ Techn. Rep.*, Arlington, VA

Gunderson LH, Holling CS. 2002. *Panarchy: Understanding Transformations in Human and Natural Systems*. Washington, DC: Island Press

Hammack J, Brown GM Jr. 1974. *Waterfowl and Wetlands: Toward Bioeconomic Analysis*. Baltimore, MD: Resour. Future/Johns Hopkins Univ. Press

Hammitt JK, Liu JT, Liu JL. 2001. Contingent valuation of Taiwanese wetlands. *Environ. Dev. Econ.* 6:259–68

Hanley N, Shogren JF, White B. 1996. *Environmental Economics in Theory and Practice*. New York: Oxford Univ. Press

Hart R. 2003. Dynamic pollution control—time lags and optimal restoration of marine ecosystems. *Ecol. Econ.* 47:79–93

Heal G. 1998. Interpreting sustainability. In *Sustainability: Dynamics and Uncertainty*, ed. G Chichilnisky, G Heal, A Vercelli, pp. 3–22. Dorcrecht: Kluwer Acad.

Heal G. 2000. *Nature and the Marketplace: Capturing the Value of Ecosystem Services*. Washington, DC: Island Press

Hitlin S, Piliavin JA. 2004. Values: reviving a dormant concept. *Annu. Rev. Sociol.* 30:359–93

Hoehn JP, Lupi F, Kaplowitz MD. 2002. The economic equivalency of drained and restored wetlands in Michigan. *Am. J. Agric. Econ.* 84:1355–61

Ingraham MW, Foster SG. 2008. The value of ecosystem services provided by the U.S. National Wildlife Refuge System in the contiguous U.S. *Ecol. Econ.* 67:608–18

Irwin EG, Bockstael NE. 2002. Interacting agents, spatial externalities, and the endogenous evolution of residential land use pattern. *J. Econ. Geogr.* 2:31–54

Irwin EG, Bockstael NE. 2004. Land use externalities, growth management policies, and urban sprawl. *Reg. Sci. Urban Econ.* 34:705–25

Iwasa Y, Uchida T, Yokomizo H. 2007. Nonlinear behavior of the socio-economic dynamics for lake eutrophication control. *Ecol. Econ.* 63:219–29

Jack BK, Kousky C, Sims KRE. 2008. Designing payments for ecosystem services: lessons from previous experience with incentive-based mechanisms. *Proc. Natl. Acad. Sci. USA* 105:9465–70

Jackson RB, Jobbagy EG, Avissar R, Roy SB, Barrett DJ, et al. 2005. Trading water for carbon with biological sequestration. *Science* 310:1944–47

Just RE, Hueth DL, Schmitz A. 2004. *The Welfare Economics of Public Policy: A Practical Approach to Project and Policy Evaluation*. Cheltenham, UK/Northampton, MA: Edward Elgar

Kahn JR. 1987. Measuring the economic damages associated with terrestrial pollution of marine ecosystems. *Mar. Resour. Econ.* 4:193–209

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Kahn JR, Kemp WM. 1985. Economic losses associated with the degradation of an ecosystem: the case of submerged aquatic vegetation in the Chesapeake Bay. *J. Environ. Econ. Manag.* 12:246–63

Koch EW, Barbier EB, Silliman BR, Reed DJ, Perillo GME, et al. 2009. Non-linearity in ecosystem services: temporal and spatial variability in coastal protection. *Front. Ecol. Environ.* 7:29–37

Krautkraemer J. 1985. Optimal growth, resource amenities, and the preservation of natural environments. *Rev. Econ. Stud.* 52:153–70

Lewis DJ, Nelson E, Plantinga AJ, Polasky S. 2009. *The efficiency of voluntary incentive policies for preventing biodiversity loss*. Work. Pap., Dep. Agric. Appl. Econ., Univ. Wisc.-Madison

Lewis DJ, Plantinga AJ. 2007. Policies for habitat fragmentation: combining econometrics with GIS-based landscape simulations. *Land Econ.* 83:109–27

Lichtenstein S, Slovic P. 2006. *The Construction of Preferences*. New York: Cambridge Univ. Press

Light A, Rolston III H, eds. 2003. *Environmental Ethics: An Anthology.* Chichester, UK: Blackwell

Lubowski RN. 2002. *Determinants of land-use transitions in the United States: econometric analysis of changes among the major land-use categories*. Ph.D. thesis. Harvard Univ., Cambridge, MA

Lubowski RN, Plantinga AJ, Stavins RN. 2006. Land-use change and carbon sinks: econometric estimation of the carbon sequestration supply function. *J. Environ. Econ. Manag.* 51:135–52

Lynne GD, Conroy P, Prochaska F. 1981. Economic valuation of marsh areas for marine protection. *J. Environ. Econ. Manag.* 8:175–81

Mahan BL, Polasky S, Adams R. 2000. Valuing urban wetlands: a property price approach. *Land Econ.* 76:100–13

Margolis M, Naevdal E. 2008. Safe minimum standards in dynamic resource problems: conditions for living on the edge of risk. *Environ. Resour. Econ.* 40:401–23

McConnell KE, Strand IE. 1989. Benefits from commercial fisheries when demand and supply depend on water quality. *J. Environ. Econ. Manag.* 17:284–92

Millennium Ecosystem Assessment. 2005. *Ecosystems and Human Well-Being. Synthesis*. Washington, DC: Island Press

Montgomery CA, Brown GM, Adams DM. 1994. The marginal cost of species preservation: the northern spotted owl. *J. Environ. Econ. Manag.* 26:111–28

Montgomery CA, Pollak RA, Freemark K, White D. 1999. Pricing biodiversity. *J. Environ. Econ. Manag.* 38:1–19

Morrison M, Bennett J, Blamey R. 1999. Valuing improved wetland quality using choice modeling. *Water Resour. Res.* 35:2805–14

Naidoo R, Adamowicz WL. 2006. Modeling opportunity costs of conservation in transitional landscapes. *Conserv. Biol.* 20:490–500

Naidoo R, Balmford A, Ferraro PJ, Polasky S, Ricketts TH, et al. 2006. Integrating economic cost into conservation planning. *Trends Ecol. Evol.* 21:681–87

Naidoo R, Iwamura T. 2007. Global-scale mapping of economic benefits from agricultural lands: implications for conservation priorities. *Biol. Conserv.* 140:40–49

Naidoo R, Ricketts TH. 2006. Mapping the economic costs and benefits of conservation. *PLoS Biol.* 4:2153–64

Nalle DJ, Montgomery CA, Arthur JL, Polasky S, Schumaker NH. 2004. Modeling joint production of wildlife and timber in forests. *J. Environ. Econ. Manag.* 48:997–1017

Nascimento HEM, Laurance WF. 2002. Total aboveground biomass in central Amazonian rainforests: a landscape-scale study. *For. Ecol. Manag.* 168:311–21

National Research Council. 1996. *Understanding Risk: Informing Decisions in a Democratic Society.* Washington, DC: Natl. Acad. Press

National Research Council. 2001. *Compensating for Wetland Losses under the Clean Water Act*. Washington, DC: Natl. Acad. Press

National Research Council. 2005. *Valuing Ecosystem Services: Toward Better Environmental Decision-Making*. Washington, DC: Natl. Acad. Press

Nelson E, Polasky S, Lewis DJ, Plantinga AJ, Lonsdorf E, et al. 2008. Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. *Proc. Natl. Acad. Sci. USA* 105:9471–76

Nelson EN, Mendoza GM, Regetz J, Polasky S, Tallis H, et al. 2009. Modeling multiple ecosystem services, biodiversity conservation, commodity production and tradeoffs at landscape scales. *Front. Ecol. Environ.* In press

Nordhaus WD, Boyer JG. 2000. *Warming the World: Economic Models of Global Warming*. Cambridge, MA: MIT Press

Norton BG. 1987. *Why Preserve Natural Variety?* Princeton, NJ: Princeton Univ. Press

Norton BG, Noonar D. 2007. Ecology and valuation: big changes needed. *Ecol. Econ.* 63:664–75

Pagiola S. 2008. Payments for environmental services in Costa Rica. *Ecol. Econ.* 65:712–24

Parkhurst GM, Shogren JF. 2007. Spatial incentives to coordinate contiguous habitat. *Ecol. Econ.* 64:344–55

Parkhurst GM, Shogren JF, Bastian C, Kivi P, Donner J, et al. 2002. Agglomeration bonus: an incentive mechanism to reunite fragmented habitat for biodiversity conservation. *Ecol. Econ.* 41:305–28

Pearce D. 1998. Auditing the earth. *Environment* 40:23–28

Perrings C. 1995. Ecological resilience in the sustainability of economic development. *Econ. Appl.* 48:121–42

Pezzey JCV, Toman MA. 2002. Progress and problems in the economics of sustainability. In *The International Yearbook of Environmental and Resource Economics 2002/2003*, ed. T Tietenberg, H Folmer, pp. 165–232. Cheltenham, UK: Edward Elgar

Polasky S, Nelson E, Camm J, Csuti B, Fackler P, et al. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biol. Conserv.* 141:1505–24

Polasky S, Nelson E, Lonsdorf E, Fackler P, Starfield A. 2005. Conserving species in a working landscape: land use with biological and economic objectives. *Ecol. Appl.* 15:1387–401

Polasky S, Solow A. 1995. On the value of a collection of species. *J. Environ. Econ. Manag.* 29:298–303

Rapport DJ, Turner JE. 1977. Economic models in ecology. *Science* 195:367–73

Rolston H. 1991. Environmental ethics: values in and duties to the natural world. In *Ecology, Economics and Ethics: The Broken Circle*, ed. FH Bormann, SR Kellert. New Haven, CT: Yale Univ. Press

Sanchirico J, Wilen J. 1999. Bioeconomics of spatial exploitation in a patchy environment. *J. Environ. Econ. Manag.* 37:129–50

Sanchirico J, Wilen J. 2001. A bioeconomic model of marine reserve creation. *J. Environ. Econ. Manag.* 42:257–76

Sanchirico J, Wilen J. 2005. Optimal spatial management of renewable resources: matching policy scope to ecosystem scale. *J. Environ. Econ. Manag.* 50:23–46

Sathirathai S, Barbier EB. 2001. Valuing mangrove conservation in Southern Thailand. *Contemp. Econ. Policy* 19:109–22

Schaefer MB. 1957. Some considerations of population dynamics and economics in relation to the management of marine fishes. *J. Fish. Res. Board Can.* 14:669–81

Scott A. 1955. The fishery: the objectives of sole ownership. *J. Polit. Econ.* 63:116–24

Settle C, Shogren JF. 2006. Does integrating economic and biological systems matter for public policy? The case of Yellowstone Lake. *B.E. J. Econ. Anal. Policy* 6:1–46

Simpson D, Sedjo R, Reid J. 1996. Valuing biodiversity for use in pharmaceutical research. *J. Polit. Econ.* 104:163–85

Spash CL. 1997. Ethics and environmental attitudes with implications for economic valuation. *J. Environ. Manag.* 50:403–16

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Spash CL. 2006. Non-economic motivation for contingent values: rights and attitudinal beliefs in willingness to pay for environmental improvements. *Land Econ.* 82:602–22

Stern N. 2007. *Stern review on the economics of climate change. Executive summary.* HM Treasury, London. http://www.hm-treasury.gov.uk/sternreview_index.htm

Swallow SK. 1994. Renewable and nonrenewable resource theory applied to coastal agriculture, forest, wetland and fishery linkages. *Mar. Resour. Econ.* 9:291–310

Tilman D, Polasky S, Lehman C. 2005. Diversity, productivity and temporal stability in the economies of humans and nature. *J. Environ. Econ. Manag.* 49:405–26

Tol RSJ. 2005. The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. *Energy Policy* 33:2064–74

Toman M. 1998. Why not to calculate the value of the world's ecosystem services and natural capital. *Ecol. Econ.* 25:57–60

Train KE. 2003. *Discrete Choice Methods with Simulation.* New York: Cambridge Univ. Press

Troy A, Wilson MA. 2006. Mapping ecosystem services: practical challenges and opportunities in linking GIS and value transfer. *Ecol. Econ.* 60:435–49

Turner RK. 1999. The place of economic values in environmental valuation. In *Valuing Environmental Preferences*, ed. I Bateman, KG Willis, pp. 17–42. London: Oxford Univ. Press

Turner WR, Brandon K, Brooks TM, Costanza R, Fonseca GAB, et al. 2007. Global conservation of biodiversity and ecosystem services. *Bioscience* 57:868–73

US Environmental Protection Agency. 2009. *Valuing the protection of ecological systems and services.* EPA Sci. Advis. Board Rep., Washington, DC

Vermeij GJ. 2004. *Nature: An Economic History.* Princeton, NJ: Princeton Univ. Press

Winkler R. 2006. Valuation of ecosystem goods and services: part 1. An integrated dynamic approach. *Ecol. Econ.* 59:82–93

Woodward RT, Wui YS. 2001. The economic value of wetland services: a meta-analysis. *Ecol. Econ.* 37:257–70

World Commission Environment Development. 1987. *Our Common Future.* New York: Oxford Univ. Press

Wu J, Adams RM, Boggess WG. 2000. Cumulative effects and optimal targeting of conservation efforts: steelhead trout habitat enhancement in Oregon. *Am. J. Agric. Econ.* 82:400–13

Wu J, Skelton-Groth K, Boggess WG, Adams RM. 2003. Pacific salmon restoration: trade-offs between economic efficiency and political acceptance. *Contemp. Econ. Policy* 21:78–89

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

80 YEARS

ANNUAL REVIEWS

Celebrating 80 Years of Publishing Excellence

NEW FROM ANNUAL REVIEWS:

EconScholar App

Available for iOS and Android

ECON SCHOLAR

Economics scholars can now access highly cited, mobile-optimized review articles from a variety of mobile devices. The ***EconScholar*** app, from Annual Reviews, allows immediate access to full-text review articles for users with personal or institutional subscriptions to the *Annual Review of Economics*, the *Annual Review of Financial Economics,* and the *Annual Review of Resource Economics*.

Also, non-subscribers and new users can access selected complimentary articles and all abstracts, and discover firsthand the breadth and quality of these review articles.

The app allows users to:

- read and cache full-text articles on a mobile device
- view high-resolution images and video
- bookmark articles (saving full-text indefinitely)
- search journal content
- read and share content through social media tools

Subscribers can either enter their personal login information or connect via institutional access to view full-text content.

To download the free *EconScholar* app, please visit the Apple *AppStore* or *GooglePlay* store.

For more information visit: **http://www.annualreviews.org/r/econscholarpdf**

ANNUAL REVIEWS | ***Guiding Scientists to Essential Research Since 1932***

Tel: 800.523.8635 (US/CAN) | Tel: 650.843.6647 | Fax: 650.424.0910 | Email: service@annualreviews.org

Annual Review of Resource Economics

Volume 1, 2009

Contents

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org by 187.234.107.18 on 03/29/13. For personal use only.

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.

Resource Dynamics

Ecology and Space

Technology and Innovation

Errata

An online log of corrections to *Annual Review of Resource Economics* articles may be found at http://resource.AnnualReviews.org

Annu. Rev. Resour. Econ. 2009.1:409-434. Downloaded from www.annualreviews.org
by 187.234.107.18 on 03/29/13. For personal use only.