# Mapping and Valuing Ecosystem Services as an Approach for Conservation and Natural-Resource Management

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**Current approaches to conservation and natural-resource management often focus on single objectives, resulting in many unintended consequences. These outcomes often affect society through unaccounted-for ecosystem services. A major challenge in moving to a more ecosystem-based approach to management that would avoid such societal damages is the creation of practical tools that bring a scientifically sound, production function-based approach to natural-resource decision making. A new set of computer-based models is presented, the Integrated Valuation of Ecosystem Services and Trade-offs tool (InVEST) that has been designed to inform such decisions. Several of the key features of these models are discussed, including the ability to visualize relationships among multiple ecosystem services and biodiversity, the ability to focus on ecosystem services rather than biophysical processes, the ability to project service levels and values in space, sensitivity to manager-designed scenarios, and flexibility to deal with data and knowledge limitations. Sample outputs of InVEST are shown for two case applications; the Willamette Basin in Oregon and the Amazon Basin. Future challenges relating to the incorporation of social data, the projection of social distributional effects, and the design of effective policy mechanisms are discussed.**

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## The Problem with Current Decision Making

Conservation and natural-resource management have been dominated by approaches that tend to focus on a single sector and a narrow set of objectives. These approaches often fail to include a wider set of consequences of decision making. For example, maximizing profit from industrial production leads to negative impacts on air quality and human health. Maximizing agricultural production often leads to poor water quality and in some cases losses of productivity in downstream fisheries. Maximizing biodiversity conservation can come at the cost of local jobs, food, or other sources of income.

Many of these consequences are the result of management decisions that overlook a wide set of ecosystem services, by which we mean the goods and services ecosystems produce that are important for human well-being. These services include climate regulation, carbon sequestration, soil fertility, pollination, filtration of pollutants, provision of clean water, flood control, recreation, and aesthetic and spiritual values (Daily 1997). In most cases, and for most services, there is little incentive for decision makers, whether they are government officials, business managers, or local landowners, to account for the continued provision of ecosystem

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services in their decision making. For example, in tropical coastal ecosystems, mangroves have been cleared in many areas and the resultant open land used for shrimp aquaculture. Those clearing the mangroves receive high market prices for the shrimp they produce, but they do not bear the full costs associated with the loss of habitat for coastal fisheries, loss of protection from storm surges, the loss of pollution filtration, and other ecosystem services provided by intact mangroves. A more complete accounting shows that maintaining mangroves generally provides more benefits for society (Sathirathai and Barbier 2001; Barbier 2007). Some policies exist that provide incentives for a wider set of ecosystem services beyond just what is provided for by selling marketed commodities. For example, the Conservation Reserve Program in the United States pays landowners to adopt conservation-oriented land uses, and similar green payments programs exist in other countries. These programs, however, do not exist in all sectors and incentives for ecosystem services remain lacking in many areas.

A single-sector approach that ignores the multitude of connections among components of natural and social systems generally fails to provide as high a value to society from the bundle of services that the system is capable of producing as would management that accounted for the complete range of services. The connections among services and the links in ecosystem processes are, in the long run, often critical for the maintenance of ecosystem health, human well-being, and the sector of interest itself (Millennium Ecosystem Assessment 2005; Guerry 2005).

## Reforming Decision Making with Ecosystem-Based Management

Growing recognition that single-sector management leads to socially and ecologically harmful "unintended consequences" has motivated the development of ecosystem-based management (EBM), an approach capable of incorporating effects of management on multiple ecosystem services in an integrated systems approach (Christensen *et al.* 1996; National Research Council 1999; Food and Agriculture Organization [FAO] 2003; Pikitch *et al.* 2004; U.S. Commission on Ocean Policy 2004). The EBM paradigm recognizes that human and ecological well-being are tightly coupled so that sustainability only occurs when it is pursued in both arenas simultaneously (FAO 2003). More specifically, EBM:

- emphasizes the protection of ecosystem structure, functioning, and key processes;
- is place-based in focusing on a specific ecosystem and the range of activities affecting it;
- explicitly accounts for the interconnectedness within systems, recognizing the importance of interactions between many target species or key services and other nontarget species and services;
- acknowledges interconnectedness among systems, such as between air, land, and sea;
- integrates ecological, social, economic, and institutional perspectives, recognizing their strong interdependences (McLeod *et al.* 2005).

The EBM approach is meant to provide a framework to enable managers to broaden their perspectives and consider the multiple linked consequences of their decisions. What makes for a good EBM approach, in theory, is now well understood (Christensen *et al.* 1996). Further, the Millennium Ecosystem Assessment (MA) contributed substantially to our understanding of an EBM framework applied at a global scale (Millennium Ecosystem Assessment 2005). Within a year of its completion, findings from the MA were incorporated into the Convention on Biological Diversity, the RAMSAR Convention on Wetlands, and the Convention to Combat Desertification. There are also a small number of examples of successful application of EBM in specific terrestrial, aquatic, estuarine, and marine settings

(Imperial 1999). These successes are evidence that the EBM integrated approach can be practical and operational.

Despite the overall success of the MA at the global scale, we are still left with the grand challenge of bringing useful models and information to bear at local, regional, and national scales where most decisions are made. Although we have several cases where people have attempted EBM at subglobal scales (see Imperial 1999 for a review, and MA subglobal assessments), there are no systematic tools that can be applied in a general, consistent way across many sites at the spatial scales and time frames relevant to subglobal decisions. One of the most challenging aspects of creating such tools is the application of sound ecological models and understanding to develop "ecological production functions" that define how the structure and function of ecosystems ultimately leads to resulting levels of ecosystem services provided. This challenge is particularly acute with ecosystem functions and ecosystem services that act across ecosystem boundaries (such as nutrient transport from land to sea) and across scales (Engel *et al.* 2008).

A second major challenge of applying EBM to specific ecosystems centers on generating estimates of the value of ecosystem services, a task that requires linking ecological models with social and economic models to reveal the values people hold for different ecological services (National Research Council 2005). This task is easier for many provisioning services that are traded in markets with observable prices. The same task is especially hard for the many ecosystem services that generate public goods, such as climate regulation or existence value of species, for which there is no market price or other readily available signals of value. Over the past 40 years or so, economists have developed a number of methods and tools for non-market valuation that can be applied to estimate the value of ecosystem services (Freeman 2003; National Research Council 2005). Whether for market or nonmarket values, appropriately linking social and economic valuation with ecological production functions is necessary to ensure that values reflect underlying ecological conditions.

## Ecological Production Functions and Economic Valuation for Mapping and Valuing Ecosystem Services

In economics, a production function specifies the feasible output of goods and services that can be produced with a given set of inputs (labor, machinery, natural resources, etc.). The level of technology determines the economic production function. A technological improvement will mean that more goods and services can be produced from a given set of inputs. An ecological production function specifies the feasible output of ecosystem services that are provided ("produced") by an ecosystem. In an ecological production function it is ecosystem processes that determine the feasible output of services. Changes in ecosystem conditions from natural disturbances or human modification alter ecological production functions shifting the amount of various services that can be provided. In the twentieth century, human alteration of ecosystems on a large scale, such as the conversion of native ecosystems to monoculture agriculture, led to an increase in some provisioning services (e.g., food production) at the expense of many regulating, supporting, and cultural services (Vitousek *et al.* 1997; Millennium Ecosystem Assessment 2005).

As with efforts to apply the concept of EBM, most applications of an ecological production function modeling approach have been done at small scales or for a single service. There are a growing number of such studies (e.g., Ellis and Fisher 1987; Barbier and Strand 1998; Wilson and Carpenter 1999; Barbier 2000; Kaiser and Roumasset 2002; Ricketts *et al.* 2004), and useful overviews and summaries have been compiled (Barbier 2007; National Research Council 2005; Pagiola *et al.* 2004). One of the most challenging tenets of EBM in general, and production functions specifically, is to integrate modeling across multiple services. The essential next step toward informing decision making is a systematic approach that combines the rigor of the small-scale studies with the breadth of broad-scale assessments. Recent work has taken strides in this vein (e.g., Boody *et al.* 2005; Jackson *et al.* 2005; Antle and Stoorvogel 2006; Naidoo and Ricketts 2006; Nelson *et al.* 2008; Nelson *et al.* 2009).

There are some cases where understanding ecological production functions alone is sufficient for the successful application of EBM. For example, many government agencies make decisions about what activities will be allowed based on how they affect the ability of an ecosystem to provide water that passes a quality standard. Their decision is not based on how much it would cost to treat that water for consumption or the value of access to clean drinking water, but rather on the expected change in contaminant levels. In these cases, simply knowing how ecosystem services will change in biophysical terms is very informative and useful.

Many other decisions are more tied to economic costs and benefits, and many decision makers are used to analyzing policy alternatives in terms of the net benefits measured in monetary terms. A concern is that ecosystem services that are not measured in monetary terms may not be given full weight in decision making or may be ignored altogether (Daily 1997). In these cases, it may be very useful to combine ecological production functions with economic valuation methods to estimate and report the monetary values of ecosystem services. Some ecological production function approaches have been able to use appropriate market prices and nonmarket valuation methods to estimate economic value, and show how the dollar value of services change with different environmental conditions (e.g., Swallow 1994; Naidoo and Ricketts 2006; Barbier 2007). At present, with the possible exception of Naidoo and Ricketts (2006), we lack comprehensive studies that tie together economic valuation methods with ecological production functions to estimate the value of ecosystem services for a significant range of ecosystem services provided by an ecosystem (National Research Council 2005).

Doing this kind of comprehensive integrated study of ecosystem services requires detailed ecological and economic understanding and data, which is lacking for many systems. Generating new data on each system studied can be expensive and time-consuming. Benefit transfer is one approach to overcoming the lack of system-specific information relatively cheaply and quickly. Benefit transfer is a method in which research results on the value of ecosystem services generated in one setting is used (transferred) to value ecosystem services in another setting. If care is taken to closely match the settings and methods are used to ensure consistency with basic economic principles, benefit transfer can be a useful approach to estimate the value of ecosystem services (Smith *et al.* 2002). Combining ecological production functions that predict biophysical changes along with benefit transfer for economic valuation can be used to generate estimates of the value of ecosystem services for systems that lack economic valuation studies.

Use of benefit transfer in valuing ecosystem services has garnered significant international attention, especially following the study by Costanza *et al.* (1997) that estimated the value of ecosystem services for the entire planet. Costanza *et al.* (1997) used estimates of the value of services per hectare for an ecosystem type from specific locations and applied these estimates to value all hectares of that ecosystem type across all regions. Other papers have also used this approach (e.g., Ingraham and Foster 2008; Troy and Wilson 2006; Turner *et al.* 2007). This approach, however, has significant disadvantages that limit its social, economic, and ecological realism. Assuming that every hectare of a given habitat type is of equal value ignores factors of rarity, spatial configuration, size, quality of habitat, number of nearby people or their social practices, and preferences that are often crucial in determining the value of services. In most cases the approach provides an estimate of total economic value rather than estimates of value for individual services. When limited to total economic value estimates, we cannot analyze how the provision and value of each individual service will change under new conditions. If a wetland is converted to agricultural land, how does this subsequently affect the provision of clean drinking water, floods downstream, climate regulation, or soil fertility? Without service-specific information, it is impossible to design effective policies or payment programs that ensure the continued provision of ecosystem services. For these reasons, we do not believe that application of benefit transfer based on value per hectare by habitat type is a promising direction to pursue. We direct attention instead toward development of ecological production functions linked to economic and social valuation methods.

A major advance that would allow the application of EBM to diverse decisions at diverse scales is to develop general tools capable of bringing an ecological production function approach to bear in ecosystems across the globe that incorporate a broad suite of important ecosystem services. At present, there are no systematic tools that can be applied in a general, consistent way across many sites at the spatial scales and time frames relevant to subglobal decisions. In other words, with existing methods and information, we have to start over each time we want to bring an ecosystem approach to bear on subglobal decisions, a time-consuming process often out of step with the pace of decision making. In the next section, we describe the development of a new general tool aimed at filling this gap.

## Mind the Gap: A New Tool for Linking Ecological Production Functions and Economic Valuation to Map and Value Ecosystem Services

The Natural Capital Project (www.naturalcapitalproject.org) has developed a new tool designed to address the principles of EBM, bringing together credible, useful models based on ecological production functions and economic valuation methods, with the intention of bringing biophysical and economic information about ecosystem services to bear on conservation and natural-resource decisions at an appropriate scale. The tool is called the Integrated Valuation of Ecosystem Services and Tradeoffs tool (InVEST). We have built in several key features that make this a flexible, yet scientifically grounded tool. InVEST is a set of computer-based models that

- clearly reveals relationships among multiple services;
- focuses on ecosystem services rather than biophysical processes;
- is spatially explicit;
- provides output in both biophysical and economic terms;
- is scenario driven;
- has a tiered approach to deal with data availability and the state of system knowledge. Several of these features are discussed in greater detail later in the chapter.

## A Multiple Ecosystem-Service Approach

Managers are often forced to make tradeoffs among sectors and goals. A fundamental mathematical truth is that one cannot simultaneously maximize many different objectives. A fundamental socioeconomic truth is that a manager cannot simultaneously maximize returns for all sectors of society at once. As the old saying goes, we cannot have our cake and eat it too. Despite the ubiquity of situations involving trade-offs, managers frequently lack a set of tools to inform them about the trade-offs they face. Often, mental assessments of the existence and magnitude of trade-offs are wrong and lead to decisions that result in poor outcomes. Although management actions that strike a balance among goals may exist, providing acceptable outcomes in multiple dimensions, these actions may be hard to identify in a highly charged political environment where arguments are based on qualitative assumptions and there is no way to bring evidence to the table in a systematic and clearly understood way. A modeling framework that can reveal the likely relationship among services can help dispel incorrect assumptions and identify management options that minimize trade-offs. There is growing evidence that decision makers are ready for this kind of information, if only they had tools to help them move forward (Box 1).

## Box 1. Decision Makers Ready for Tools of Change

### Colombia Ministry of the Environment, Housing, and Territorial Development

Colombia's national environmental regulation is changing the face of environmental permitting and mitigation. The Ministry of the Environment, Housing and Territorial Development is invested in a 7-year project with several universities, research institutes, and nongovernment organizations to create a set of tools that will allow them to take a more comprehensive approach to permitting and offsetting. The Ministry regulates all major economic sectors (agriculture, mining, transportation, etc.). They aim to assess all major projects proposed in each sector over the next 5 years, avoid permitting projects in areas identified as priority areas for conservation of biodiversity and ecosystem services, set levels of compensation and mitigation for damages that cannot be avoided, and target offsetting in areas of high biodiversity and ecosystem-service provision. InVEST will be one of several tools used in this new permitting process.

### BC Hydro

British Columbia's major power provider has set an ambitious environmental goal for the next 20 years. They plan to achieve "no net environmental impact" by identifying how their infrastructure and activities affect air, land, and water quality, and then designing programs that avoid and reduce all impacts as much as possible. They will then use mitigation to offset any remaining unavoidable changes to the environment. To achieve this goal, they need to be able to project how much environmental impact different activities incur and how effective alternative programs might be. BC Hydro is currently considering ways to conduct such assessments, including the use of InVEST.

### Kamehameha Schools

With a mission to provide economic, cultural, educational, environmental, and community values from its land holdings, Hawaii's largest private landowner has a major challenge in assessing their land assets. How does each parcel measure up across these five diverse objectives? As many of their long-term land leases are coming up for renewal, they have the opportunity to change the face of much of Hawaii's landscape. What set of management practices would optimize returns across all objectives in the future? Kamehameha Schools is exploring different ways to reveal the level of each set of values provided by their lands. They currently apply a rigorous scoring rubric similar to that used to assess art, and they are exploring the use of InVEST.

There are several reasons why current management decisions lead to trade-offs among sectors, or among ecosystem services. One reason is that not all services are positively correlated. Using data from the Willamette Basin in Oregon, Nelson *et al*. (2008) found that targeting policies to provide carbon sequestration, by limiting enrollment to landowners who would grow forests on their land, was effective at increasing carbon storage, but not effective for species conservation. Alternatively, targeting policies to meet species-conservation objectives, by limiting enrollment to landowners who would restore rare habitat types (e.g., oak savannah and prairie), was effective at increasing species conservation but not effective for carbon sequestration. More generally, the Millennium Ecosystem Assessment (2005) found pervasive trade-offs between provisioning services (e.g., food and timber production) and other types of services (regulating, supporting and cultural services). A modeling framework that allows assessments of biodiversity and multiple ecosystem services can identify policies or geographies that would lead to a win-win outcome, where all objectives can be increased relative to the status quo, and those situations where outcomes necessarily would lead to trade-offs.

We address the need to reveal synergies and trade-offs by providing models for a suite of ecosystem services in InVEST. The services we currently model are climate regulation through carbon sequestration, water supply for hydropower or irrigation, erosion control for infrastructure (reservoir) maintenance, water quality control for regulatory compliance (phosphorous loading, not contaminants related to drinking water such as fecal coliform bacteria), storm peak flow mitigation, recreation and tourism, provision of native pollination for commercial agricultural crops, agriculture production, timber production, nontimber forest product production, provision of cultural values and nonuse values. We also provide models for terrestrial biodiversity as an attribute of natural systems that underpins the delivery of ecosystem services. All of these models have been described in detail elsewhere (Nelson *et al.* 2008; Nelson *et al.* 2009; Kareiva *et al.* in press).

Additional ecosystem services should be considered in any natural-resource decision-making process besides those just listed. We have focused on this subset because of its importance, relevance to major decisions being made currently, and proximity of many of these services to markets. As the modeling effort progresses, the aim is to include other important services that likely provide large value to society, such as forage production and the regulation of pests and diseases. InVEST would also be greatly improved by the ability to model freshwater biodiversity and marine ecosystem services. The potential for inclusion of marine ecosystem services is embodied in the ecosystem-based management literature, but practical tools are still lacking. We are early in our development of marine models, but some initial steps have been taken by mangers and scientists in the Puget Sound region of Washington State (Anne Guerry, personal communication, 2008).

## Biophysical Processes vs. Ecosystem Services: An Important Distinction

Biophysical processes are essential for the provision of ecosystem services, but processes are not synonymous with services. Until there is some person somewhere who is benefiting from a given process it is only a process and not a service. This is a critical distinction to make as we look at what science and tools are currently available for decision makers. Extensive research has been applied to the modeling and measurement of biophysical processes, and it is tempting to simply apply those to ecosystem service–related decisions. However, biophysical processes only tell us about the supply side of the ecosystem service–provision equation. It is equally critical to include the demand side. In other words, where are the people who use services, and how much do they use? To truly model ecosystem services, we need to adjust the supply of ecosystem services (biophysical processes) based on the location, type, and intensity of use of each service.

Water-related services provide good examples for how to think about this distinction. Consider the provision of clean water for drinking. Many useful biophysical models exist that can help us predict the concentration of contaminants in waterways (e.g., SWAT, Gassman *et al.* 2007; AGNPS, Yuan *et al.* 2006). However, the presence of clean water that could be consumed is not a service unless there is someone there to drink it. This does not mean that a natural system providing clean water in a remote area with no people does not provide any services. Clean water in remote areas can maintain biodiversity existence values and may also be a value as clean water for drinking in the future. But if no one currently makes use of the water for drinking, then there is no provision of that ecosystem service in that particular place at that particular time. Ecosystem services have to be connected to beneficiaries (demand) to gain an accurate picture of what the provision of the service provided by the landscape (supply) is worth.

Consider another example with the provision of pollination to agricultural crops. Many agricultural crops require insect pollination (e.g., almonds, strawberries), but many other crops do not (e.g., rice, corn). A patch of native habitat in an agricultural landscape may house healthy native bee populations, but if there are no agricultural fields within foraging distance with a crop in need of pollination, that native habitat patch does not provide pollination benefits. In this case, a model of native pollinator metapopulation dynamics could give us a very clear sense of how much pollination service could be supplied by patches of native habitat in an agricultural landscape. Until that information is paired with information on the identity of crops grown, their distribution in the area, and the crop-specific yield benefits of native pollination, we cannot estimate the amount or value of pollination service being provided.

InVEST deals with this challenge by first modeling the biophysical processes that constitute the supply side of the ecosystem-service equation. These models draw heavily from existing knowledge. For example, our water-related service models start out with the same fundamental hydrologic processes included in models such as SWAT (Gassman *et al.* 2007). Our sediment retention for infrastructure maintenance model draws heavily from the universal soil-loss equation (USLE) (Brooks *et al.* 1982). Our biodiversity model is based on species–area relationships (Connor and McCoy 1979; Pereira and Daily 2006). To determine the value of the ecosystem services provided we combine supply-side models about provision with the likely level of ecosystem-service demand. For example, in our model of water supply for irrigation, we consider how much water is demanded by crops grown in the region that is not met by rainfall. The value of increased water for irrigation equals the increase in the value of crops that can be grown with an increase in available water input. This increase in value from an increase in available irrigation water can arise because more land can be irrigated or because existing cropland can receive more water, which could increase yields of crops or allow more water-intensive but higher value crops to be grown. To the extent that vegetation and management practices in an ecosystem lead to an increase in water supply, such practices provide a valuable ecosystem service in places where there is demand for water (for irrigation or other purposes). Consider another example. The role that vegetation and management practices play in keeping sediment out of waterways can provide services to society, including avoided infrastructure maintenance costs (as reservoirs silt in and need to be dredged) and avoided flood risk (as rivers or reservoirs silt in and lose their capacity to control or buffer floods). To estimate the value of avoided siltation in reservoirs, we ask how much of the erosion control predicted by the USLE is upstream of a reservoir, and consider the characteristics of that reservoir to estimate the level and value of service provided by avoiding dredging and other maintenance costs. If there are no reservoirs, then this specific service is not being provided.

The links between ecosystem processes, ecosystem services, and the value of ecosystem services for the ecosystem services currently

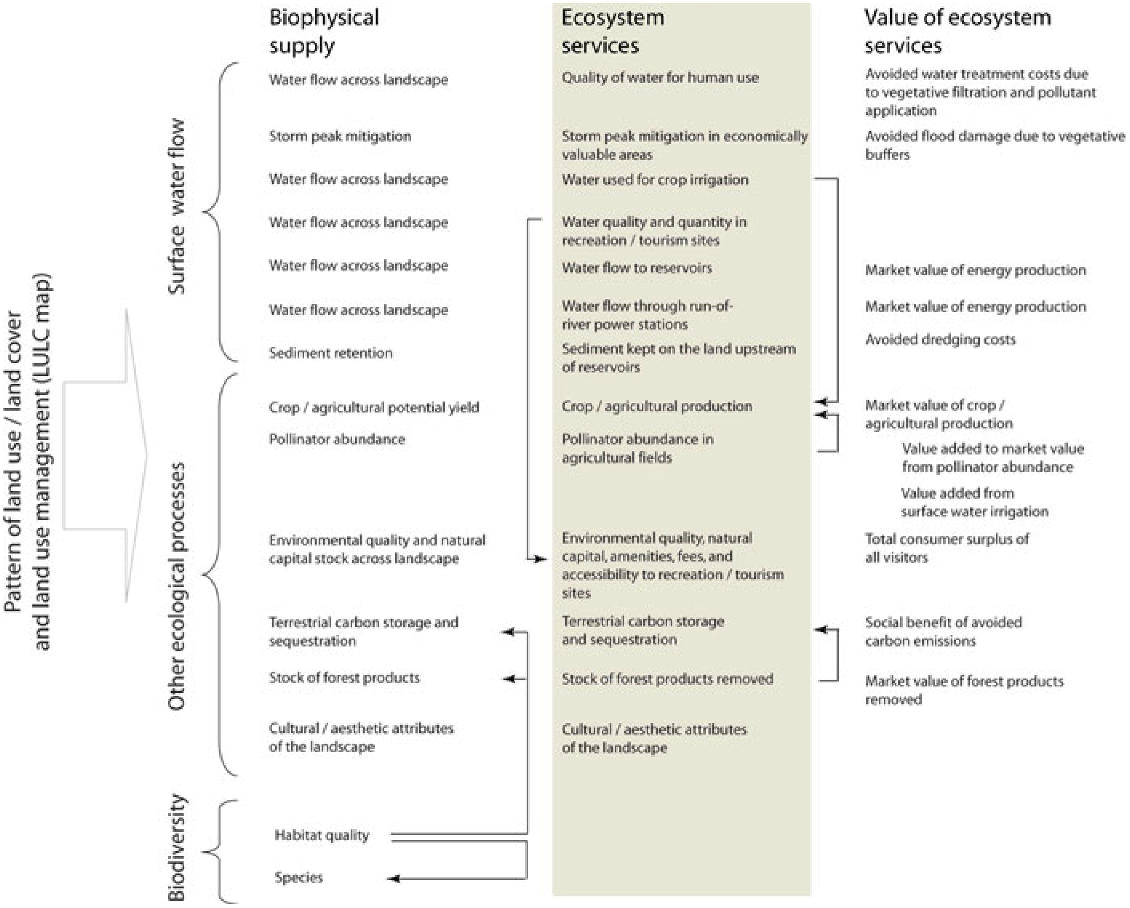


**Figure 1.** Links between the biophysical supply, ecosystem services (which combine supply and demand for services), and the value of ecosystem services for ecosystem services currently modeled in InVEST. Land-use and management decisions impact on ecosystems and ecological processes (*far left*). Ecological processes give rise to the biophysical supply of ecosystem services (*first column*). Ecosystem-service supply combined with demand generates realized ecosystem services (*second column*). Applying economic or social valuation methods yield estimates of the value of ecosystem services (*third column*). Important links among ecological processes or ecosystem services are indicated with *arrows*.

incorporated into InVEST is illustrated in Figure 1. The specifics of how supply and demand are combined in specific models for these and other services can be found in greater detail elsewhere (Nelson *et al.* 2008; Nelson *et al.* 2009; Kareiva *et al.* in press).

## Spatially Explicit

The value of ecosystem services is determined both by the location of ecological processes that create the provision of services (supply) and the location of people who derive benefits from the services (demand). Thus, any ecosystem-service modeling effort should be spatially explicit.

We consider two key elements of space in the application of InVEST: the role of spatial pattern and heterogeneity in the landscape in controlling the provision of services, and the scale across which different services act. In regard to the first element, decision makers often want to know where to invest or how to target programs to get the greatest return from their investment. For instance, where should protected areas be located to gain the largest biodiversity and climate-regulation benefits? Should a new agricultural subsidy program to control water quality be targeted at riparian areas in headwaters or further downstream? Will a tree planting program in a poor district help with flood control? All of these questions have a spatial element, but many existing biophysical process models are aspatial and do not allow analysis of landscape locations that are best for investment. All of the models in InVEST focus on identifying how much each parcel (or pixel) on the landscape controls, or contributes to, each service.

In addressing the contribution of a parcel to the provision of services, we must consider the scale across which services are provided. Some services, such as pollination and some water-related services are provided at a very local scale, while other services such as climate regulation are provided at a global scale. Trees fixing carbon in the Amazon forest are providing a benefit to you as you read this chapter, no matter where you are in the world. Each model in InVEST looks across the appropriate scale for the service of interest. For example, the pollination model uses the foraging distance of native pollinators as the scope of assessment, while the carbon-sequestration model assumes that tree growth on any parcel provides a benefit since the global atmosphere is well-mixed.

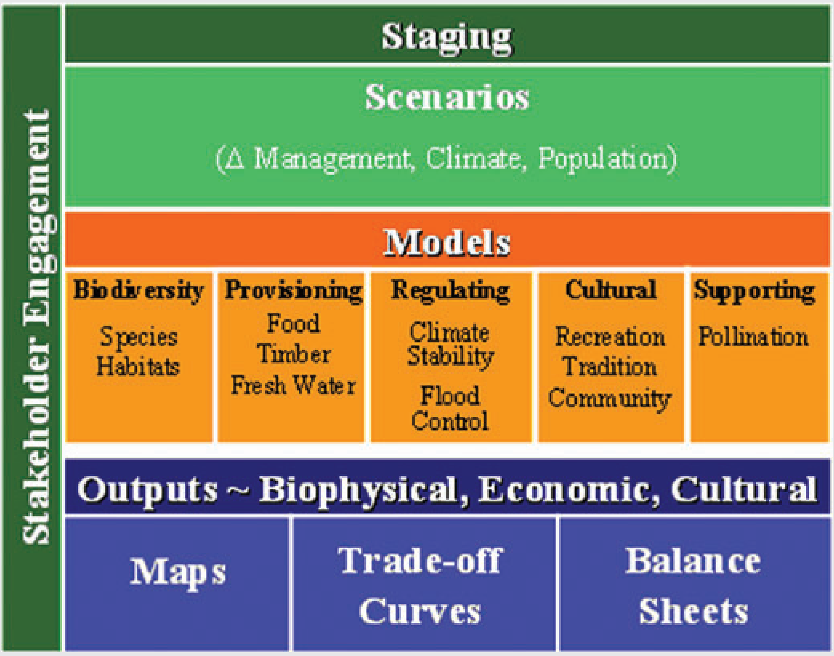
Considerations of scale raise two important issues, one related to modeling and one related to policy. It may be more difficult to apply models for local services since input data on land use and cover patterns need to be at a high enough resolution to capture important features of the service. One would not learn much from the pollination model if native pollinators at the site of interest foraged 1.5 km, but land use and cover data were only available at 50 km$^2$ resolution. When modeling multiple services, the scale of data resolution needed corresponds to the most finely detailed ecosystem service model.

In terms of policy, the scale and location of the provision of ecosystem services by natural systems and the scale and location of beneficiaries of the services in society are often disconnected. As we have mentioned, trees fixing carbon in the Amazon rainforest are providing a global benefit enjoyed by people far removed from the Amazon. For other services, such as pollination or provision of clean water, benefits are fairly local. But even where provision and benefits of services are local, supply and demand may be disconnected in space. Because InVEST is a spatially explicit model, it can highlight the locations on the landscape important for the supply of services as well as the location of people who benefit from services that show patterns of overlap or disconnect between supply and demand. A prime example of a disconnect is where upstream landowners divert water or increase nutrient loading that harms downstream water users. Such spatial disconnects between provision and benefits have important implications for policy. With spatial disconnects explicit policies will be needed to give incentives to decision makers who control the provision of services so that they can recognize the benefits that their provision provides to others, a point that we discuss further in the final section of the chapter. Such policies can be explored through the development of scenarios.

## Scenario-Driven Modeling: Making a Decision-Relevant Tool

To be effective in a decision-making or policy arena, analyses should be relevant to the needs and questions of managers and decision makers. To apply InVEST in such situations, we envision embedding the modeling within a stakeholder engagement process that allows managers to identify the choices of interest to them (Box 2). InVEST is designed to respond to many different kinds of scenarios derived through many different types of stakeholder engagement processes.

## Box 2. InVEST in a Stakeholder Process



InVEST is meant to be used as part of a stakeholder engagement process with decision makers who give input at every stage of the modeling process. First, scenarios, or spatially explicit maps of how the future could look, are created based on specific choices being considered. These landscapes are then fed into a set of biophysical models that project the level of ecosystem services that will be provided by that landscape. InVEST models several ecosystem services and biodiversity.

If economic valuation is of interest, there is a second set of models that can be used to derive the value of each of the ecosystem services. Biodiversity, as an attribute of the system, is never directly valued, rather we reveal its value through the multiple services it supports. Finally, the outputs of either the biophysical or economic models can be viewed as maps of the landscape, trade-off curves, or balance sheets. After seeing the implications of their scenarios, stakeholders may choose to create new scenarios and use the process iteratively.

InVEST can take input from stakeholders to set what land-use and land-management scenarios to analyze. InVEST is quite flexible in that it can consider a wide range of land-use and resource-management alternatives. Each ecosystem-service model uses land-use and land-cover (LULC) patterns as inputs to predict the biodiversity and the production of ecosystem-services outputs. The inclusion of both land use and land cover means that we can consider choices that affect the type of land cover (urban, wetland, closed-canopy, deciduous forest, etc.) and choices that keep land cover the same but alter management practices on any particular part of the landscape (change in release pattern from an existing dam, change in crop type planted in existing agricultural areas, change in fertilizer type or amount used, change in rotation time in existing plantation forests, etc.). Most natural-resource management decisions will affect land-use or land-cover patterns, either directly or indirectly, so sensitivity of the models to LULC patterns translates into broad sensitivity to management choices.

To represent the choices that managers or other stakeholders want to consider in a way that InVEST can recognize and assess them, we need to translate those choices into likely future LULC patterns. Just as there are many different types of choices that managers consider, there are many different methods for turning choices into LULC patterns. When decision makers hold full control over the area of interest, their own planning processes usually include the development of scenarios. For example, we conducted a very simple scenario-development workshop with a large private landowner in Hawaii. Kamehameha Schools, the owner of 10% of Hawaii's land, is considering future management options for a relatively large parcel of land (26,000 acres) and have a few alternative options in mind. Since they are the sole owners of the entire parcel, they were able to draw simple maps of these options in a short workshop that we then turned into scenarios that were assessed by InVEST for likely ecosystem-service provision.

When the area of interest is a more complex landscape with multiple ownership and multiple drivers of change, more complex scenario generation options are available. One can add predictions of how landowners will react given various institutions and incentives that they face. For example, we can use models of landowner decision making to predict how landowners would react to changes in crop prices or government policies to generate scenarios of LULC that can be input into InVEST models. We used this kind of scenario-generation approach to analyze the effect of alternative policies with some of the early InVEST models (Nelson *et al.* 2008). In more demonstrative applications, users may want to explore what is possible on a particular landscape given the fundamental ecosystem-service relationships in place. In these cases, landscape optimization modeling can be used to create scenarios of how the landscape could look under optimal conditions. We have also used this approach to investigate optimal land-use patterns using early InVEST models (Polasky *et al.* 2008).

We have emphasized management decisions as drivers of landscape change, but there are obviously other factors at work. Climate change and human population growth will undeniably change LULC patterns and climate conditions in the future. Scenarios can be built to include these drivers of change in addition to management practices. Many efforts have now downscaled global climate models and used regional predictions to evolve vegetation patterns, giving us maps of likely future land cover and climate. Similarly, several projections of human population growth have been made, and some groups are in the process of turning these numeric projections into spatially explicit human population–density estimates or urban/rural-area extents (Salvatore *et al.* 2005). When models or maps of these drivers are available, they can be combined with any approaches that project management impacts, giving scenarios that represent all three major drivers of future change.

## Tiered Modeling: Flexibility for a Data-Limited World

There is always a trade-off in modeling between making a model more complex and detailed and keeping it simple. Simple models require fewer data, are often less prone to parameter estimation errors and subsequent error propagation, and can be easier to explain and understand. Complex models require more information, but they often inspire greater confidence because they more faithfully depict the details and underlying intricacies of processes. Different applications and different users will have specific needs for either complicated or simple models of ecosystem services and valuation. For this reason, we have developed a tiered system of models in InVEST (Fig. 2).

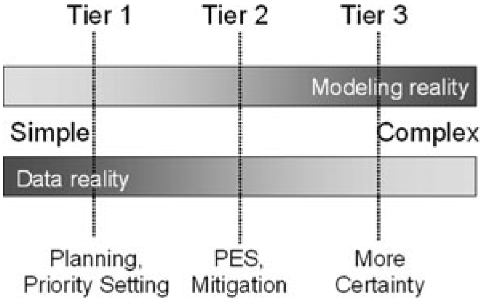


**Figure 2.** A tiered approach to modeling ecosystem services. Given the difficulty of matching models with the desired level of complexity with often sparse data, we created InVEST with different types of models. Tier 1 models are simple, require few data, and are best used for planning and priority setting. Tier 2 models are more complex, require more data, and are necessary for setting payment levels or informing mitigation programs. Tier 3 models add even greater confidence and will often be site-specific models created by other research groups.

Tier 1 models are the simplest models. We developed these models to require few data, be easy to understand and explain to others, and yet retain sufficient credibility to guide some management decisions. Their distinguishing feature is a reliance on readily available data that are generally accessible everywhere in the world. Tier 1 models can draw information from the published literature, global data sets, site-specific data sources, local traditional knowledge, or expert opinion. Because of their simplicity, these models will be most appropriately applied in general scoping and planning activities where the purpose is to understand the general lay of the land.

In many applications the predictions of Tier 1 models may be too crude or too prone to errors of "averaging" or "aggregating" to meet the needs of decision makers. This will likely be the case when it is critical to get quantitative estimates of ecosystem services for decisions related to making actual payments for ecosystem services or for setting the levels of mitigation required. For these cases, we provide more detailed Tier 2 models. These models require more data, have more parameters, and are more time-consuming and difficult to apply. However, Tier 2 models are generally "better" in the sense of addressing more ecological complexity, allowing for greater spatial and temporal heterogeneity, and allowing more refined estimates of everything from carbon production to species richness to pollination services. Instead of representing a world of "average trees" and the "average pollinator" or habitat types sans species lists, Tier 2 models disaggregate groups to include age structure of trees, a variety of pollinator guilds or species, and monthly precipitation patterns.

InVEST provides a general framework within which one can mix and match Tier 1 and Tier 2 models, depending on differing data availability or need for precision among services. By allowing modular mixing of tiers, InVEST allows users to customize its application to specific problems. While we have developed only Tier 1 and Tier 2 models, it is important to realize that it is possible to also use InVEST with what we call Tier 3 models—research level, state-of-the art models (e.g., the CENTURY carbon model, Parton *et al.* 1994).

## InVESTing in Real Decisions

InVEST has been applied in decision-making processes at several sites including the Willamette Basin (Oregon), Oahu (Hawaii), the state of California, Puget Sound (Washington State), the Eastern Arc Mountains (Tanzania), the Upper Yangtze Basin (China), the Amazon Basin and Northern Andes (South America) and at a national level in Ecuador and Colombia. Here, we highlight some of the outputs the InVEST models can provide in a decision-making context by presenting the findings from the Amazon Basin and the Willamette Basin.

### Projecting Short-Term Ecosystem-Service Loss in the Amazon Basin

The United Kingdom's Natural Environment Research Council (NERC), Department for International Development (DFID) and the Economic and Social Research Council (ESRC) launched the Ecosystem Services and Poverty Alleviation Program in 2007. The goal of the program is to promote research and capacity building to achieve sustainable ecosystem management and well-being in developing countries (Porro *et al.* 2008). One of their focal regions is the Amazon Basin, a nearly 10 million km$^2$ area with some of the globe's highest diversity and endemism, and largest remaining tracts of intact rainforest. This is a region of relatively low rural population density, but high poverty and extremely rapid rates of forest loss and degradation (Porro *et al.* 2008). The agencies funding the work are interested in what is at stake to be lost in the Basin, who is likely to lose, and how losses can be avoided.

Through a collaboration with The Nature Conservancy, InVEST was used as one of several tools to predict likely future changes in biodiversity and ecosystem services in the region. We considered the year 2000 as the base case, and the future scenario created predicts how the basin will look in 2020 after likely levels of deforestation (projected by the Instituto de Pesquisa Ambiental da Amazonia) and road development (projected by the Initiative for Integration of Regional Infrastructure in the South) (Porro *et al.* 2008). Tier 1 models for habitat quality and several forest products (timber, fruits and nuts, fiber, medicinal plants, and bushmeat) were applied to the whole basin at a 1-km$^2$ resolution. Forest products are essential staples for some indigenous groups in the region. One study estimated that people consume 148.2 tons of bushmeat per year in the Amazon Basin (Fa *et al.* 2002).

Areas along roadways and near population centers are projected to lose the most overall in terms of forest product provision (e.g., Fig. 3; Porro *et al.* 2008). These areas are mostly outside of protected areas and indigenous territories. However, in montane forests, people living in indigenous territories will experience the greatest losses in both marketed and subsistence products (Fig. 4; Porro *et al.* 2008). The same pattern was projected for losses of marketed products in savanna and varzea ecosystems. Further work is underway to apply higher-resolution poverty data to ask whether the poor in the Basin are at especially high risk of losing access to ecosystem services in the next 20 years (Kareiva *et al.* in press). For now, the message to policymakers in Brazil and elsewhere who influence decisions affecting the Amazon Basin

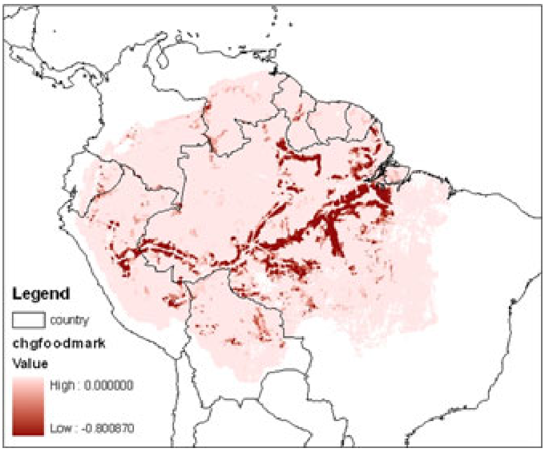


**Figure 3.** Projected change in the provision of food (fruits and nuts) for market sale between 2000 and 2020. The *pink area* shows the extent of analysis. The measure of food provision is a relative index where the parcel with the highest initial abundance of fruit and nut species for market sale in the region received score of 1.0 in the year 2000. This map shows the difference between the index score for each parcel in 2020 and 2000.

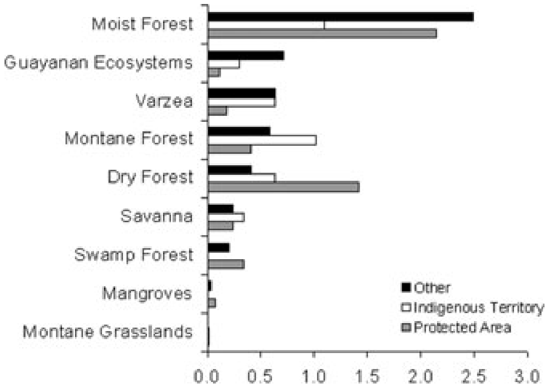


**Figure 4.** Loss in marketed forest products by habitat type in the Amazon Basin between 2000 and 2020. The highest initial provision of forest products (in 2000) had a score of 5.0.

is the importance of stemming the loss and degradation of habitat and ecosystem services in the Basin while levels are still relatively high. The Amazon is a unique system given its relatively large remaining intact habitat and relatively low rural population density. The Amazon has lost ~84 million hectares of native forest in the last few decades. Other regions that failed to stop such drastic losses now struggle in an extremely costly and often futile effort to rebuild ecosystem services in severely degraded areas with overwhelming numbers of rural poor (Porro *et al.* 2008).

## Willamette Basin Example

The Willamette Basin in Oregon (USA) has been the site of intense debates over land and water management in recent decades. In the late 1990s, there were conflicts between environmental groups and the timber industry about the impacts of logging on the spotted owl, a species dependent on old-growth forest and listed under the U.S. Endangered Species Act. In an effort to resolve the debate President Clinton met with various stakeholders at the Pacific Northwest Forest Conference in 1993. Clinton directed federal agencies to develop a Pacific Northwest Forest Plan to protect the spotted owl while addressing other economic and social concerns. Other conflicts in the region revolve around efforts to save imperiled populations of salmon hurt by the cumulative impact of timber harvests, hydroelectric power dams, and land use that increased siltation and water temperature. Because of these various conflicts and efforts to find solutions, much effort has gone into collecting data on species, resources, and land use, making the Willamette Basin a perfect laboratory on which to develop and test models of ecosystem services.

Nelson *et al.* (2009) applied InVEST to model multiple ecosystem services and biodiversity conservation in the Willamette Basin under three alternative land-use scenarios (plan trend, development, and conservation) that track land-use trajectories from 1990 to 2050. The scenarios were developed by the Pacific Northwest Ecosystem Research Consortium, composed of representatives from government agencies, nongovernment organizations, and universities (see details at http://www.fsl.orst.edu/pnwerc/wrb/access.html). The scenarios were used as input into InVEST models that then tracked changes in water quality, storm peak mitigation, erosion control, carbon sequestration, biodiversity conservation, and market returns for agriculture, timber, and housing development. Nelson *et al.* (2009) found that all ecosystem services (water quality, storm peak mitigation, erosion control, and carbon sequestration) along with biodiversity conservation were highest in the conservation scenario, while market returns were higher in the plan trend and development scenarios. According to these results, programs designed to enhance the provision of ecosystem services would likely enhance biodiversity conservation. However, letting landowners pursue their own self-interest would likely harm both ecosystem services provision and biodiversity conservation. If, however, payments for ecosystem services could be arranged so that landowners received some benefit from the provision of ecosystem services, then the apparent conflict between landowner interests and ecosystem services disappears. Paying for just one ecosystem service, carbon sequestration, using a price of $43 per ton of carbon (Tol 2005), made the conservation scenario the most valuable outcome rather than the least valuable outcome in terms of market value of goods and services produced (see Fig. 5).

# The Next Frontiers

InVEST provides a means to map and value multiple ecosystem services that can be used to inform conservation and natural-resource management, but it is not a panacea. Many of the components of InVEST are relatively new and untested. Predicting how ecosystem

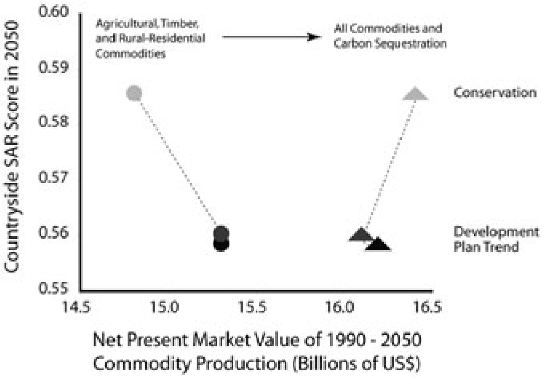


**Figure 5.** Trade-off between biodiversity and commodity production in the Willamette Basin in 2050 under different scenarios. The Conservation scenario (*gray circle*) has the highest biodiversity score, but lowest commodity value, given current markets. This scenario would have the highest commodity and biodiversity values if the carbon market was formalized (*gray triangle*). SAR is species–area relationship.

services will change as ecosystems are altered and what the value of those changes will be in various socioeconomic systems present a number of novel challenges. Increasing our working understanding of these processes will be an on-going task. The InVEST tools in their current form are sufficient for providing information about priority areas on landscapes for the provision of various services, but in most cases have not been tested sufficiently to provide quantitative estimates to underpin payments for ecosystem services or other regulatory mechanisms. Developing methods to assess the validity and reliability of model predictions at multiple scales are needed. The models will assuredly undergo modification and updating as experience and applications increase.

A large unmet challenge in ecosystem-based management is to understand the distribution of ecosystem service among different groups in society, and how this distribution will likely change as a consequence of management decisions. While it is important to know the total amount of ecosystem services provided and their overall value to society, it is also important to know who benefits from the provision of services and their social and economic status. Without information about the distribution of benefits from ecosystem services, management decisions can lead to serious unintended consequences for equity and well-being. This concern is especially strong for management actions that negatively affect underprivileged parts of society. For example, in developing countries, establishing a new national park or conservation area has in some cases resulted in removing and separating people and their work from ecosystems they rely on (Kareiva and Marvier 2007), leading to a decline in their livelihood or well-being.

Another example of distributional issues at play occurs under "wetland banking" programs allowed under the U.S. Clean Water Act. A policy objective in the United States is to have "no net loss" of wetlands. Under wetland banking, a developer who destroys wetlands in one place can offset this loss by buying credits from a wetland bank that has gained credits by restoring wetlands elsewhere. One of the unintentional consequences of this program has been the reallocation of wetlands from urban to rural areas, with a corresponding shift in wetland service provision away from the poor in urban areas to areas with significantly lower population densities (Ruhl and Salzman 2006). This type of regulatory effect is often overlooked in the decision-making process, but it is an important outcome of the functioning of the program and needs to be addressed (BenDor *et al.* 2008).

Whether a particular social group wins, loses, or remains unaffected by a decision is determined by several factors. Of utmost importance is access. Underprivileged members of society will not benefit or lose from changes in ecosystem services if they cannot access them. Access has two critical components in this context, physical overlap in space and legal rights. InVEST can show clearly where services will be provided on a landscape and how their provision is likely to change in space. This ability can give insights into the spatial-overlap part of access. Rapid advances are being made in the mapping of social indicators of poverty (CIESIN 2006; World Resources Institute 2007), and we can draw from these approaches to ask where ecosystem services and the poor overlap on the landscape. However, we currently do not have a standardized way to bring information about institutions and their level of enforcement into a mapping and valuation context. Developing ways to represent and predict the interactions among ecosystem services, people, and institutions will be critical to the assessment of the distributional effects of management decisions.

A related challenge in ecosystem services is to alter the financial and institutional infrastructure to give incentives to maintain and enhance the provision of ecosystem services. Often there is a spatial or temporal mismatch between those who control the provision of ecosystem services and those who benefit from the services. Without the ability to connect the demand for services from beneficiaries with the supply from those who control the supply, there will be insufficient incentive for suppliers to protect ecosystems to maintain the supply of ecosystem services. InVEST can help to highlight the spatial patterns of both supply and demand and illustrate and the spatial mismatch between them. Addressing the mismatch and providing proper incentives for provision of ecosystem services will require changes in policies of local and national governments and in international agreements. Some progress on these fronts can be seen, as with the recent emergence of carbon markets, international policy discussions on reduced emissions from deforestation in developing countries (REDD), and expansion of programs of payments for ecosystem services (PES). Successfully linking the science of mapping and valuing ecosystem services with proper institutions and policies will likely remain a major challenge for decades to come.

## Conflicts of Interest

The authors declare no conflicts of interest.

## References

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

Barbier, E.B. 2007. Valuing ecosystem services as productive inputs. *Econ. Policy* **22:** 177–229.

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

Barbier, E.B. & I. Strand. 1998. Valuing mangrove-fishery linkages: a case study of Campeche, Mexico. *Environ. Resour. Econ.* **12:** 151–166.

BenDor, T., N. Brozovic & V.G. Pallathucheril. 2008. The social impacts of wetland mitigation policies in the United States. *J. Plann. Lit.* **22:** 10.1177/0885412207314011.

Boody, G., B. Vondracek, D.A. Andow, *et al*. 2005. Multifunctional agriculture in the United States. *Bioscience* **55:** 27–38.

Brooks, K.N., H.M. Gregersen, E.R. Berglund & M. Tayaa. 1982. Economic evaluation of watershed projects—An overview methodology and application. *Water Res. Bull.* **18:** 245.

Christensen, N.L., A.M. Bartuska, J.H. Brown, *et al*. 1996. The Report of the Ecological Society of America Committee on the Scientific Basis for Ecosystem Management. *Ecol. Appl*. **6**(3): 665–691.

CIESIN (Center for International Earth Science Information Network). 2006. *Where the poor are: an atlas of poverty*. Columbia University, Palisades, NY. Available at www.ciesin.columbia.edu/povmap/. [Accessed February 13, 2009]

Connor, E.F. & E.D. McCoy. 1979. The statistics and biology of the species-area relationship. *Am. Nat.* **113:** 791–833.

Costanza, R., R. d'Arge, R. de Groot, *et al*. 1997. The value of the world's ecosystem services and natural capital. *Nature* **387:** 253–2260.

Daily, G.C. 1997. *Nature's Services*. Island Press. Washington, DC.

Ecosystem Principles Advisory Panel. 1999. *Ecosystem-based fishery management: A report to Congress by the Ecosystem Principles Advisory Panel.* U.S. National Marine Fisheries Service. Washington, DC.

Ellis, G.M. & A.C. Fisher. 1987. Valuing the environment as an input. *J. Environ. Manage.* **25:** 149–156.

Engel, S., S. Pagiola & S. Wunder. 2008. Designing payments for environmental services in theory and practice: an overview of the issues. *Ecol. Econ.* **65:** 663–674.

Fa, J.E., C.A. Peres & J. Meewig. 2002. Bushmeat exploitation in tropical forests: an intercontinental comparison. *Conserv. Biol.* **16:** 232–237.

Food and Agriculture Organization. 2003. *FAO Fisheries Department Technical Report No. 4 (Supply 2)*. Food and Agriculture Organization. Rome, Italy.

Freeman, A.M., III. 2003. *The Measurement of Environmental and Resource Values: Theory and Methods*. Resources for the Future. Washington, DC.

Gassman, P.W., M.R. Reyes, C.H. Green & J.G. Arnold. 2007. The soil and water assessment tool: historical development, applications and future research directions. *Trans. Am. Soc. Agric. Biol. Eng.* **50:** 1211–1250.

Guerry, A. 2005. Icarus and Daedalus: conceptual and tactical lessons for marine ecosystem-based management. *Front. Ecol. Environ.* **3:** 202.

Imperial, M.T. 1999. Institutional analysis and ecosystem-based management: the institutional analysis and development framework. *Environ. Manage.* **24:** 449.

Ingraham, M.W. & S.G. Foster. 2008. The value of ecosystem services provided by the U.S. National Wildlife Refuge System in the contiguous U.S. *Ecol. Econ.* **67:** 608–618.

Jackson, R.B., E.G. Jobbagy, R. Avissar, *et al*. 2005. Trading water for carbon with biological sequestration. *Science* **310:** 1944–1947.

Kaiser, B. & J. Roumasset. 2002. Valuing indirect ecosystem services: the case of tropical watersheds. *Environ. Dev. Econ.* **7:** 701–714.

Kareiva, P., G. Daily, T. Ricketts, *et al*. 2009. *The Theory and Practice of Ecosystem Service Valuation in Conservation*. Oxford University Press. Oxford. In press.

Kareiva, P. & M. Marvier. 2007. Conversation for the people—Pitting nature and biodiversity against people makes little sense. Many conservationists now argue that human health and well-being should be central to conservation efforts. *Sci. Am.* **297:** 50–57.

McLeod, K.L., J. Lubchenco, S.R. Palumbi & A.A. Rosenberg. 2005. *Scientific consensus statement on marine ecosystem-based management*. Communication Partnership for Science and the Sea. Available at www.compassonline.org/pdf_files/EBM_Consensus_Statement_v12.pdf [Accessed February 13, 2009]

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

Naidoo, R. & T.H. Ricketts. 2006. Mapping the economic costs and benefits of conservation. *PLoS Biol.* **4:** 2153–2164.

National Research Council. 1999. *Sustaining marine fisheries*. National Academy Press. Washington, DC.

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

Nelson, E., G. Mendoza, J. Regetz, *et al*. 2009. Modeling multiple ecosystems services, biodiversity conservation, commodity production and tradeoffs at landscape scales. *Front. Ecol. Environ.* **7**(1): 4–11.

Nelson, E., S. Polasky, D.J. Lewis, *et al*. 2008. Efficiency of incentives to jointly increase carbon sequestration and species conservation on a landscape. *Proc. Nat. Acad. Sci.* **105:** 9471–9476.

Pagiola, S., K. von Ritter & J. Bishop. 2004. *How Much is an Ecosystem Worth? Assessing the Economic Value of Conservation*. The World Bank. Washington, DC.

Parton, W.J., D.S. Schimel, D.S. Ojima & C.V. Cole. 1994. *Quantitative Modeling of Soil Forming Processes. In Special Publication*. Soil Science Society of America. Madison, WI.

Pereira, H.M. & G.C. Daily. 2006. Modeling biodiversity dynamics in countryside landscapes. *Ecology* **87:** 1877–1885.

Pikitch, E.K., C. Santora, E.A. Babcock, *et al*. 2004. Ecosystem-based fisheries management. *Science* **305:** 346–347.

Polasky, S., E. Nelson, J. Camm, *et al*. 2008. Where to put things? Spatial land management to sustain biodiversity and economic returns. *Biol. Conserv.* **141:** 1505–1524.

Porro, R., J. Borner, A. Jarvis, *et al*. 2008. *Challenges to Managing Ecosystems Sustainably for Poverty Alleviation: Securing Well-Being in the Andes/Amazon*. Final Report, May 2008. Situation Analysis prepared for the ESPA Program. Amazon Initiative Consortium, Belém, Brazil. Available at www.ecosystemsandpoverty.org/wp-content/uploads/2008/05/espa-aa-final-report-small-version_.pdf [Accessed February 13, 2009].

Ricketts, H.T., G.C. Daily, P.R. Ehrlich & C.D. Michener. 2004. Economic value of tropical forest to coffee production. *Proc. Nat. Acad. Sci.* **101:** 12579–12582.

Ruhl, J.B. & J. Salzman. 2006. The effects of wetland mitigation banking on people. *Nat. Wetl. News* **28:** 1, 9–14.

Salvatore, M., F. Pozzi, E. Ataman, *et al*. 2005. *Mapping global urban and rural population distributions* (Environment and Natural Resources Report No. 24). Food and Agriculture Organization. Rome.

Sathirathai, S. & E.B. Barbier. 2001. Valuing mangrove conservation in southern Thailand. *Contemp. Econ. Policy* **19:** 109.

Smith, V.K., G. Van Houtven & S.K. Pattanayak. 2002. A benefit transfer via preference calibration: 'prudential algebra' for policy. *Land Econ.* **78:** 132–152.

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

Tol, R.S.J. 2005. The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. *Energy Policy* **33:** 2064–2074.

Troy, A. & M.A. Wilson. 2006. Mapping ecosystem services: Practical challenges and opportunities in linking GIS and value transfer. *Ecol. Econ.* **60:** 435–449.

Turner, W.R., K. Brandon, T.M. Brooks, *et al*. 2007. Global conservation of biodiversity and ecosystem services. *Bioscience* **57:** 868–873.

U.S. Commission on Ocean Policy. 2004. *Preliminary Report of the U.S. Commission on Ocean Policy, Governor's Draft*. U.S. Commission on Ocean Policy. Washington, DC.

Vitousek, P.M., J.D. Aber, R.W. Howarth, *et al*. 1997. Human alteration of the global nitrogen cycle: sources and consequences. *Ecol. Appl*. **7:** 737–750.

Wilson, M.A. & S.R. Carpenter. 1999. Economic valuation of freshwater ecosystem services in the United States: 1971–1997. *Ecol. Apps.* **9:** 772–783.

World Resources Institute. 2007. *Nature's Benefits in Kenya: An Atlas of Ecosystem Services and Human Well-being*. World Resources Institute. Washington, DC.

Yuan, Y., R.L. Bingner & J. Boydstun. 2006. Development of TMDL watershed implementation plan using annualized AGNPS. *Land Use Wat. Resour. Res.* **6:** 2.1–2.8.