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VALUING WILDLANDS

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One of the central problems of land and water management is "the way in which scarce resources are allocated among alternative uses and users. The question is, of course, fundamental to economic thinking, and it is for this reason that we have seen the introduction of essentially economic models and modes of thought in ecology" (Rapport and Turner 1977). Many questions that are at the heart of environmental management may be answered not only through the use and advancement of landscape ecology and EcoRAs (the primary topics of this book), but also through resource valuation. The value of wildlands is derived from human use of resources, as well as ecological functions such as provision of habitat, that support non-use or existence values of organisms, populations, communities, and ecosystems. Ecological valuation entails both the description of valued attributes of the environment, as well as quantitative methods for comparing these attributes and alternative scenarios. The valuation of wildlands can support several types of decisions, such as which lands to conserve, which lands to develop, which waters to impound, how much flow to leave in rivers, which lands or waters to remediate, and which lands or waters to set aside for research. Moreover, various United States federal agencies are increasingly required to evaluate benefits of conservation and environmental research programs, both of which rely on valuation methods. For example, the US Department of Agriculture evaluates benefits of its Conservation Reserve Program (USDA 2004), and the US Department

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of Defense is increasingly interested in valuing its lands that are exclusion zones or buffer areas for military training or testing (R. Pinkham, Booz Allen Hamilton, personal communication, September 2006).

Wildlands

The use of the term "wildland" implies that value is somehow derived from wildness. Wildlands are lands and waters where natural processes dominate and human impact is minimized. The term "wilderness" can be a synonym for wildlands, but is more narrowly defined by law, though the US Wilderness Act of 1964 took the rather broad definition "area where the earth and its community of life are untrammeled by man, where man himself is a visitor who does not remain" (Public Law 88–577). For the purpose of this chapter, we assume a gradient of "wildness" or lack of human impact, and only completely exclude from discussion areas of extensive urbanization, industrial development, intense resource extraction (e.g., oil and gas development, agriculture, timber extraction), and stream impoundment. Thus, most forests, grasslands, rangelands, streams, and natural lakes would fit our definition of wildlands, as would small natural areas such as riparian zones that are surrounded by urban, suburban, or industrial development. Although some readers would dispute that powerline rights-of-way are wildlands, for example, those that are managed for dense scrub vegetation provide substantial pollination services (Russell et al. 2005). Similarly, many military installations have large wildland communities that serve as reservoirs for protected species, despite the proximate disturbances from training (Tazik and Martin 2002).

For the purposes of this chapter, we include aquatic ecosystems within the definition of wildlands. In the United States, some rivers are designated *Wild and Scenic Rivers*: "certain selected rivers of the Nation which, with their immediate environments, possess outstandingly remarkable scenic, recreational, geologic, fish and wildlife, historic, cultural or other similar values, shall be preserved in free-flowing condition, and ... they and their immediate environments shall be protected for the benefit and enjoyment of present and future generations" (16 U.S.C. §§ 1271–1287). "Free-flowing" is defined as "existing or flowing in a natural condition without impoundment, diversion, straightening, rip-rapping, or other modification of the waterway." Dams upstream of Wild and Scenic portions of rivers are typically required to
Q1 maintain natural flow regimes (Jager and Bevelhimer, 2007). The "wild and scenic rivers" designation recognizes the public's interest in maintaining a subset of rivers in a relatively pristine state.

Similarly, lakes without shoreline development have enhanced value as wildlands. In 1965, the US Congress established the Land and Water Conservation Fund (16 U.S.C. §§ 460l–460l-11) to purchase and protect undeveloped shoreline along critical lakes and streams. These lands are often placed in the custody of the USDA Forest Service.

Types of Value

The value of wildlands is not derived primarily from human extractive use, even where hunting, fishing, and timber harvesting are common. Although game fish and wildlife are sometimes classified as market entities (e.g., US EPA 2006), most people who engage in these activities are not recouping their travel or other costs from sales. These activities are valued because of the cultural experience and environment as well as the resource product. Human use values of wildlands include recreational and aesthetic value. They also include other ecological service values, many of which are not well-quantified or well-monetized: supportive functions such as nutrient cycling and pollination, regulating services such as climate modulation and soil retention, provisioning services such as water supply, and cultural services such as historical or spiritual symbolism (see the *Millennium Ecosystem Assessment*, WRI 2005, for more detail).

Non-use values are existence values or bequest values that are unrelated to use of or visits to wildlands. For example, we value rare species just because they exist. Likewise, we value the fact that we could visit the African Plains even if we never travel there. Option value is an additional type of value related to preserving the opportunity of possible future use of the resource (e.g., for genes or medicines), but it may also be viewed as belonging to the non-use category of values. This taxonomy of ecological valuation is described in more detail at http://www.ecosystemvaluation.org/(viewed June 2009).

Preservation value (a combination of option value for recreational use, existence value, and bequest value) contributes most of the value of wildlands, but willingness to pay for preservation declines as the number of protected resources becomes large. For example, in a study of the protection of rivers in the Rocky Mountains of Colorado, USA, Sanders et al. (1990) found that preservation value was higher than recreational use value, but declined as the number of protected rivers increased. Consequently, total value reached a peak at an intermediate number of protected rivers (Fig. 9.1).

In its *Ecological Benefits Assessment Strategic Plan*, the US EPA defines "indirect-use" values as those that indirectly benefit society though the "support [of] offsite ecological resources or [maintenance of] the biological . . . or biochemical processes

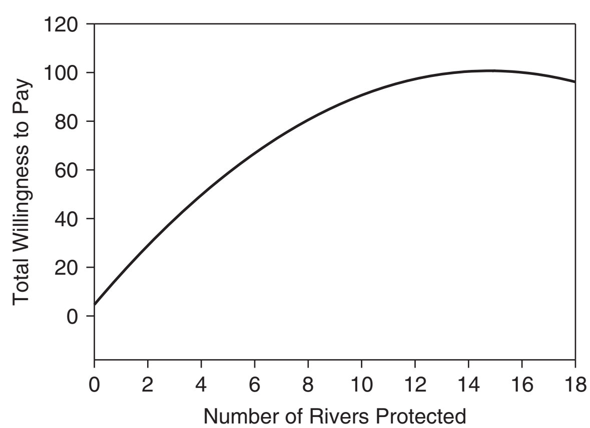

Figure 9.1. Total benefits of river preservation, including preservation values for protection of wild and scenic rivers, Colorado, 1983. [Redrawn from Sanders et al. (1990).]

required for life support." In this definition, EPA includes many "benefits" that are relevant to wildlands, including maintenance of biodiversity, protection of habitat, pollination, seed dispersal, flood protection, water supply (quantity), water purification, pest and pathogen control, and energy and nutrient flow (US EPA 2006). Many of these benefits are not well-quantified.

Many environmentalists are reluctant to value natural environments from an economic or even an ecological service perspective. For example, McCauley (2006) argues that conservation must be a moral or ethical enterprise and that "Nature has an intrinsic value that makes it priceless." While this cultural belief is valid, it does not help environmental managers choose which lands to conserve or which to restore first.

ECOLOGICAL VALUATION

We believe that the future of wildland valuation will be driven by the increased incorporation of ecological relationships. Ecologists can quantify many economic concepts that are at the heart of valuation, such as rarity, complementarity (i.e., value in context), and substitutability. Values of wildlands depend on spatial relationships, temporal systems dynamics, and thresholds. Ecological models can be used to transfer estimated value from one previously valued (e.g., by surveys) entity to a related, unvalued one (such as a predator, forage, habitat, etc.). It is unlikely that new economic methods of valuation of direct-use benefits, such as recreation, will advance the science of valuation as much as ecology. Therefore, we believe that a discussion of the future of wildland valuation is a discussion of the future of *ecological* valuation, involving valuation of populations and their habitats, communities, and ecosystem function. (See also Chapters 16, 17, and 18, all of which address economic ecology.)

The science of ecological valuation is moving in two directions at once—toward increased simplification and toward increased complexity. Simple approaches include several nonmonetary valuation methods: (2006) semiquantitative lists of valued attributes, such as aspects of habitat value; (2005) environmental benefit indicators; (2005) environmental benefit indices (aggregations of indicators); and (1999) areal equivalencies for ecological services. Simple approaches are often chosen when funding is not available to monetize, direct measurement of value is important, all relevant ecological benefits cannot be monetized, monetization is not in the interest of the land owner or manager (e.g., if a high value might prompt a sale of land that is not desired by all stakeholders), or valuation is being used primarily as a communication tool (e.g., if users want maps of value). More complex approaches use dynamic models that include feedbacks between ecology and economics. These are typically used when adequate funding is available to support a large valuation effort, value can be monetized, and mechanistic relationships are understood.

EXAMPLE APPROACHES TO ECOLOGICAL VALUATION

We now review some of the methods that are available for quantitative and semiquantitative valuation of wildlands. These include simple models of value (e.g., habitat valuation metrics, and indicators/indices of environmental benefits) and more complex models of value (integrated models, mechanistic models of ecology). We also discuss the use of ecological values that are derived using these methods in optimizations to address objectives that combine ecological and nonecological values.

Simple Models of Value

Ecological value can be decomposed into measurable characteristics. One of the important questions is, What makes wildlands wild? Remoteness is a characteristic of wildlands that is valued by many hunters, fishers, hikers, and wide-ranging vertebrate species. Remoteness is often correlated with valued ecological services and attributes of habitat. For example, bird densities are reduced near automobile traffic (Reijnen et al. 1995). One could represent remoteness by using a simple measure such as average road density in the area (the value would be sensitive to the area chosen) or distance to closest road. By the latter measure, R. T. Forman asserts that the most remote location in the eastern United States would be somewhere in the Florida Everglades (Cromie 2001), coinciding with prime Florida panther (*Puma concolor coryi*) and American alligator (*Alligator mississippiensis*) habitat. However, this quality of remoteness raises the dilemma presented in Banzhaf and Boyd (2005): If an ecosystem benefit is enjoyed by many, rather than a few, is a higher level of ecological service being provided? Is ecological value higher?

Wildness also implies a lack of disturbance from other stressors, not just roads and their vehicles. Therefore, measures of extent or intensity of disturbance might be viewed as other broad indicators of wildness, or more precisely, a lack of wildness. However, the term *disturbance* has many meanings, sometimes representing exposure to physical (e.g., noise, erosion) and biological (e.g., invasive plant species) agents and sometimes biological effect. Disturbance is not easily measured as a broad value metric, but descriptions of specific disturbances have been used in valuation studies. For example, in a habitat valuation study, Efroymson et al. (2008a, 2008b) included examples of disturbances or management practices as part of the site descriptions that were used in the analysis of habitat complexity, land cover, and ecological corridors: presence of invasive biota, presence of weir, presence of concrete liner, absence of riparian zone, erosion, substantial nutrient influx, presence of chemical contamination, pine beetle damage, plantation land cover, presence of burial ground, mowing, presence of roads, presence of buildings, and presence of scrap metal.

Moreover, in some instances, disturbed lands may be more ecologically valuable than wilder lands, depending on the ecosystem service under consideration. For example, some species benefit from disturbance at explosives-contaminated military ranges. These include early successional plant species, kangaroo rats (*Dipodomys merriami*), Sonoran pronghorns (*Antilocapra americana sonoriensis*), and frogs that use impact craters. Other species [e.g., black-capped vireo (*Vireo atricapillus*) and Karner blue butterfly (*Lycaecides melissa samuelis*)] use early successional habitats that persist only in the presence of wildfire (Efroymson et al. 2009 and references within).

Habitat Valuation Metrics. Attributes of lands and waters that make them good habitat for multiple species or rare species have been used to estimate habitat value. As early as the 1970s, land areas were prioritized for conservation using one or more of five typical value metrics: quantity of habitat, biodiversity supported, naturalness, rarity, and threat of human interference (Margules and Usher 1981). Although economic factors have always been considered in conservation decisions, habitat benefits are typically described, but not monetized.

Habitat Quantity. Area is a measure of relative habitat value for sites within a single ecosystem. A larger, contiguous habitat patch or stream reach is generally more valuable to a species than a smaller one of the same habitat quality. Rates of species loss are dependent on land or water body area (Margules and Usher 1981). However, area is not a reasonable habitat value metric for comparisons across ecosystem types.

Rarity of Species and Communities. Another determinant of habitat value is rarity, or the lack of substitute habitats. A rare vegetation community is arguably more valuable than a common association, especially if organisms are closely adapted to that vegetation association. The presence of rare species increases the existence value of a community (Rossi and Kuitunen, 1996). Moreover, rare plant or bird species are often indicative of rare vegetation associations (SAMAB 1996). An important dimension of rarity is the region, land area, or stream reach within which a species or biotic community is rare.

Biodiversity Supported. Species diversity or taxa richness are direct measures of use of a site by organisms. Biodiversity is also related to the functional value of ecosystems (Hooper et al. 2005). Some ecologists view biodiversity as insurance against major functional changes in an ecosystem because higher diversity ensures redundancy in ecosystem function among individual species (Doherty et al. 2000). Habitat structural complexity has been found to increase biodiversity by many researchers (Crowder and Cooper 1982, Downes et al. 1998, Benton et al. 2003, Johnson et al. 2003), but not by all (e.g., Doherty et al. 2000). Quantitative methods for assessing habitat structural complexity are much less common in terrestrial systems (Newsome and Catling 1979) and lacustrine systems than in streams (Barbour et al. 1999). Kapustka et al. (2004) modified a model developed by Short (1984) to estimate potential for biodiversity and ecological recovery of habitat. They predicted wildlife species richness for locations surrounding a contaminated copper mine site, based on vertical and horizontal diversity of vegetation cover types.

Habitat valuation schemes based on biodiversity can be refined to account for the fact that species are not valued equally by society. One measure of naturalness and an important determinant of habitat value is the presence, abundance, or land area covered by nonnative and especially invasive species (Burger et al. 2004). The diversity of nonnative species has been used as an indicator of reduced habitat value for native species (Efroymson et al. 2008a). The susceptibility to invasion by exotic species is strongly influenced by species composition, as well as disturbance by stressors such as roads, noise, chemical contaminants, and so on. Invasive exotic plant species are typically assumed to have lower habitat value than their less-invasive counterparts, because some invasive species have the potential to increase their abundances so rapidly that they can dominate the landscape.

Threats to Habitat. Some valuation schemes assume that threatened systems are more valuable for conservation (Margules and Usher 1981). For example, US EPA Region 7 has developed tools for identifying critical terrestrial ecosystems (Missouri Resource Assessment Partnership 2004). In addition to species richness, low number or intensity of stressors, high percentage of public ownership, and connectivity, value in these ecological assessments is based on absence of threats. Threats include land demand, agriculture, and toxic releases.

Case Study. Habitat value metrics representing some of these environmental attributes were recently applied to environmental remediation decisions for chemical contaminants. We conducted a study that was intended to identify metrics of habitat value that might supplement formal EcoRA of contaminants to help decision-makers prioritize wildland and non-wildland sites for remediation (Efroymson et al. 2008a, 2008b). Methods were developed to summarize dimensions of habitat value for several aquatic and terrestrial contaminated sites at the East Tennessee Technology Park (ETTP) on the US Department of Energy (DOE) Oak Ridge Reservation in Oak Ridge, TN, USA. Many locations on Department of Defense (DOD) and DOE reservations where security buffers have been in place for decades have high habitat value (Mann et al. 1996). In this study, an industrialized area with low ecological habitat value and chemical concentrations associated with high ecological risk (but low human health risk) might have a lower priority for remediation than a more natural area with lower ecological risk, but high habitat value. Similarly, the baseline habitat value would provide evidence concerning the potential harm that might be caused by remedial technologies (Whicker et al. 2004, Efroymson et al. 2004).

For this habitat valuation study at ETTP, we developed three broad categories of valuation metrics: onsite use by groups of organisms, value added to onsite use value from spatial context, and rarity (Efroymson et al. 2008a). Use value was measured by taxa richness, a direct measure of number of species that inhabit an area; complexity of habitat structure, an indirect measure of potential number of species that may use the area; and land use designation, a measure of the length of time that the area would be available for use (Table 9.1). Value derived from spatial context was measured by similarity or complementarities of neighboring habitat patches and presence of habitat corridors. Value derived from rarity was measured by the presence of rare species or communities.

Metrics that were more specific to groups of organisms in contaminated streams, ponds, and terrestrial ecosystems, as well those that applied to the east Tennessee region, were selected as examples of the general metrics. Examples of use of value metrics were taxa richness of fish, number of sensitive benthic invertebrate species, riparian wetland coverage relative to Southern Appalachian regional average, and taxa richness of edge-associated breeding birds (Efroymson et al. 2008a). Examples of metrics of rarity were the presence of a rare vegetation community as well as the

Table 9.1. Metrics for Valuing Habitat at Six Contaminated Sites[a]

Type of Value	Metric	Explanation
	Value from Site Alone	
Use	Taxa richness	Direct measure of number of species that inhabit area.
	Number of sensitive species	Subset of diversity and number of species that use area. Absence provides indication of level of degradation of area.
	Complexity of habitat structure	Indirect measure of potential number of species that may use area.
	Presence of special wildlife habitat services	Presence of bird rookeries, bat maternity roosts, male display areas, vernal pools, or other wildlife breeding areas that indicate greater use and importance compared to similar areas without features.
	Habitat suitability relationship for broad taxa	Relationships provide information on whether particular vegetation associations or other environmental quality variables are highly suitable or not suitable for particular broad taxa.
	Number of invasive or nonnative species	Nonnative species decrease use by native species. Invasive species also decrease use by native species, and footprint increases with time, if unchecked (therefore, area-weighted use value for native species decreases with time).
	Land cover designation	If the majority of land area is paved or covered with buildings, habitat value is low because of lack of vegetation, minimal habitat structure, and fragmentation.
	Land use designation	If land used is designated as industrial area, habitat use value may not continue for as long as it would if area were conserved.
	Offsite Value Added	
Rarity	Presence of rare species	Current value of habitat is high if rare species use it. State and federal listed and candidate species considered rare for this study.
	Presence of rare community with respect to ORR, the region, Ridge and Valley ecoregion, or Southern Appalachians	Rare community implies little redundancy or substitutability for habitat services, along with potentially high demand for site.

Table 9.1. (*Continued*)

Type of Value	Metric	Explanation
Use from spatial context	Presence of similar, adjacent habitat patch	Use value of habitat patch increases with area, because some species need minimal patch areas for home ranges, territories, or viable populations. In addition, size of habitat patch correlated with diversity.
	Presence of ecological corridor	Presence of migration and other movement corridors indicates that community of site in question adds use value to surrounding habitat and that surrounding communities add use value to habitat on site.
	Adjacency to complementary land or water	Arrangement of communities can add value to organisms that enjoy services of each (e.g., terrestrial zones around wetlands and riparian habitats).
	Adjacency to conservation land use area	Habitat value of site adjacent to reserve would probably persist longer than habitat value of other sites.

[a]The major components of value are use, rarity, and use value added from spatial context.

presence of listed species, such as fish and bats (Efroymson et al. 2008a). Examples of metrics for value derived from spatial context were adjacency to a conservation area or part of an ecological corridor linking forests from the Cumberland Plateau to the Smoky Mountains (Efroymson et al. 2008a). For each of these metrics, cutoff values for high, medium, and low habitat value were recommended in the study, based on distributions of organisms and landscape features, as well as habitat use information.

Habitat Equivalency Analysis. Habitat Equivalency Analysis (HEA) is a non-monetary valuation method used to determine locations and land or water areas that provide equivalent ecological services. The method is typically used in Natural Resource Damage Assessment applications or other ecological restoration analyses (NOAA 2000). The HEA might be applied to assign ecological value to alternative wildlands being considered for preservation to compensate for injured ecosystems. An HEA could also be used to evaluate restoration efforts that recreate wildlands from injured resources.

In HEA, ecosystem functions are assumed to be proportional to monetary value; that is, people derive utility from ecological entities correlated with their ecological function(s) (Roach and Wade 2006, Dunford et al. 2004). Thus, resource equivalencies are usually expressed in units of service-acre-years. The relationship between ecological function and economic utility is most likely to apply to relatively small (marginal) changes in habitat services in which changes in scarcity of injured habitat are insignificant (Dunford et al. 2004).

Although ecological restoration decisions commonly rely on HEA, the analysis becomes difficult when the services provided by prospective compensatory resources are not of the same type as those that have been lost. The value of apples may be compared with the value of oranges by gauging human preferences, but the ecological service relationships that HEA draws from are less helpful for comparisons of unrelated ecological entities. For example, the DOE transferred Black Oak Ridge forest land to the state of Tennessee to offset the losses of aquatic resources from chemical contamination in Watts Barr Reservoir from the DOE Oak Ridge Reservation. This exchange of forest for fish and benthic invertebrates could not have been justified by HEA or by comparing ecological relationships because the forest and fish did not belong to the same ecosystem.

A weakness of HEA is that it assumes that ecosystem function (and therefore ecological value) is proportional to land or water area. Kremen and Ostfeld (2005) recommend that mitigation banks to compensate for damage to wetlands, as well as other applications of HEA, allow factors such as shape of land area, location, connectivity, and species composition to contribute to the relative ecological value of a parcel of land. Landscape Equivalency Analysis is a modification of HEA that incorporates the habitat connectivity value of a particular habitat patch and the tradeoffs between connectivity and area (Bruggeman et al. 2005). In this method, the habitat value of a wildland patch derives from its marginal contribution to metapopulation (group of interacting, spatially separated populations) persistence or the marginal decline in habitat service flows that result from removal of the patch. We believe that habitat connectivity represents an important future direction for habitat valuation (see below).

Environmental Benefits Indicators. Environmental benefits indicators (EBI) are being used as nonmonetary measures of ecological value. They take advantage of the increased availability of spatial data and growing literature of ecological indicators. Boyd and others (Boyd 2004, Boyd and Wainger 2002) have pioneered some of these ideas, arguing for the affordability and ease of use of indicators intended to represent some of the same dimensions of ecological value as the habitat valuation metrics described earlier, as well as relative human demand (Table 9.2).

Researchers have used similar types of indicators to represent benefits of ecological services, such as providing habitat, regulating water, and assimilating wastes on military installations (Richard Pinkham, Booz Allen Hamilton, personal communication, September 2006). Pilot tests of these indicators and environmental benefit indices (demand index, scarcity index, risk index) have been conducted to assess the ecosystem service value of providing habitat at Vandenberg Air Force Base and Fort Lewis Army Base (R. Pinkham, personal communication, September 2006). A combined habitat index shows hotspots for habitat value.

Q2 Dale and Polasky (2007) discussed the potential use of environmental benefits indicators in measuring ecosystem services from agriculture. Examples of ecological services pertinent to wildlands include pollination, soil retention, nutrient cycling, and maintenance of biodiversity. They argue that useful EBIs must be linked to and predictive of the production of ecosystem services.

Table 9.2. Example Attributes of Value and Related Indicators

Value Attribute	Example Indicator
Demand	Proximity to population
Scarcity, substitutability	Local prevalence
	Abundance of population, ecosystem, land-cover type providing identical service
Complementary inputs	Landscape characteristic or infrastructure allowing access to recreation
Low probability or magnitude of future risks	Measure of stressor such as invasive species, low elevation (vulnerability to flood), etc.

Source: Modified from information in Boyd and Wainger (2002).

Multimetric Environmental-Benefits Indices. Natural systems are inherently multidimensional. Valuation joins the ranks of scientific efforts to project the many dimensions that define ecological systems into one dimension. Measures or indicators of environmental benefits are sometimes aggregated into multimetric indices. Many indices add the component EBI values, often weighting the factors differently. The reductionism of indices is most reasonable if the relationship between environmental variables is well understood [e.g., the relationship between vegetation structure and wildlife habitat and species richness in the habitat model of Kapustka et al. (2004)]. One of the fundamental underpinnings of economic valuation is that different components of value are independent and additive and that the total value of a system or scenario does not either include doubly counted component values or exclude component values. An example of double counting would be adding the contributory value of a prey item (i.e., the value it has as a result of contributing biomass to a valued predator) to the value of the predator.

Multimetric indices are commonly used among aquatic toxicologists and aquatic ecologists to estimate and compare status and trends of ecosystems (Bruins and Heberling 2005). One common multimetric index used in rivers is the index of biotic integrity (Karr 1981), which measures the deviation of a stream invertebrate community from that in a group of pristine reference streams. A challenge for using the index is finding reference streams of approximately the same size and in the same geographic region. An example of a multimetric index that comes closer to measuring ecological value is the index of "ecosystem ecological significance," which is calculated by the US EPA Region 5 Critical Ecosystem Assessment Model (CrEAM). CrEAM is a geographic information system (GIS)-based tool that incorporates ecological diversity, ecological sustainability, rare species, and land cover into one multimetric index of ecosystem value (White and Maurice 2004). More specific habitat quality indices are also available, such as the 64 benthic habitat quality indices summarized in Diaz et al. (2004).

The US Department of Agriculture has developed an EBI to rank offers to enroll lands in the Conservation Reserve Program. Although they are not strictly wildlands, these lands are taken out of agricultural production temporarily or permanently, and participants must show ecological benefits, such as reduced erosion or restoration of vegetation cover for wildlife habitat (USDA 2004). The USDA EBI is the sum of several weighted factors and subfactors. Up to 100 points (of 395 possible points for environmental benefits, exclusive of costs) may be assigned to the "wildlife habitat cover benefits" factor, the only factor that represents ecological benefits.

Within the "wildlife" habitat cover benefits factor, the "cover" subfactor measures management options and seeding mixes that provide habitat for wildlife species of national, regional, state, or local significance (USDA 2004). The "wildlife enhancements" subfactor measures the provision of water to wildlife as well as the degree of conversion of land from a monoculture of vegetation to native species. The "wildlife priority zones" subfactor add points if the land may contribute to the restoration of habitat of threatened or endangered species or other important or declining species (USDA 2004). However, the tracts of land are not formally examined in their spatial context (e.g., whether they are part of an existing wildlife corridor). Additional environmental benefits in the index relate to water quality, prevention of wind erosion, air quality, and carbon sequestration (USDA 2004).

Banzhaf and Boyd (2005) described how an ecological services index might be developed to summarize beneficial environmental services through time. The index would be based on a comprehensive list of ecological services weighted by proxies for willingness to pay (e.g., human population measure), location-specific quality factors (e.g., proximity of wetlands to polluted runoff), substitution factors (availability of close substitutes), and complementarity factors (i.e., availability of adjacent assets that increase the value of the ecological service) (Banzhaf and Boyd 2005).

Although environmental benefits indices are easily used, their assumptions are not easily understood. Indices can have several disadvantages for valuing ecological stocks and services, such as habitat services. First, if managers or stakeholders have not fully expressed their relative preference for different ecosystem services, then a multimetric index is not useful for estimating ecological value (Efroymson et al. 2008a). Moreover, different weightings of the various indicators might be appropriate for different potential users of environmental benefits indices; a single index is not very useful. Furthermore, indicators developed at one spatial scale may be not be useful to a decision that targets a different spatial scale (Efroymson et al. 2008a, 2008b). Some of Suter's (1993) criticisms of ecosystem health indices also apply to the aggregation of variables into a multimetric index of environmental benefit. Several of his arguments against the use of indices include:

- *Ambiguity.* If the value of an index is low, one cannot tell how many components were low.
- *Arbitrariness of Combining Functions.* An index may be very sensitive to the methods used to calculate it.
- *Arbitrariness of Variance.* The variance of an index does not have a clear relationship to a biological response.
- *Unreality.* Indices do not measure actual biophysical properties.
- *Disconnection from Testing.* Indices cannot be tested in the laboratory or verified in the field.

Complex Models of Value

Complex models of ecological value tend to be used in situations where decision-makers want ecological and economic factors to be integrated, ecological and economic data availability is high, relationships between ecology and economics are understood in a mechanistic way, and adequate funding is available. Although state-of-the-art ecological models produce highly uncertain results, the data to support these models are becoming more readily available, and it is not clear that they are any less predictive than complex economic models.

We identify three classes of complex models. These include (1) integrated models of ecology and economics, (2) models supporting habitat-based replacement costs, and (3) multivariate analysis and optimization.

Integrated Models. "Full ecological-economic models may be the gold standard for establishing the full range of ecosystem service possibilities and management options" (Farber et al. 2006). Integrated ecological–economic systems fit the characteristics of complex systems described in Costanza et al. (1996): strong and usually nonlinear interactions, feedback loops that make cause indistinguishable from effect, lags in time from cause to effect, distance between cause and effect, thresholds, and hierarchical behavior (failure of small-scale results to easily predict large-scale behavior). Costanza et al. (1996) argue that "reductionist thinking fails in its quest to understand complex systems." Thus, previously described simple indicators do not capture all of the dynamics of ecological–economic systems that must be understood in order to inform particular decisions about wildlands. Such dynamics can be simulated, however. Understanding the dynamic behavior of ecological–economic systems and the interdependencies of human and ecological processes has been attempted at the regional scale using ecological–economic models. These have been used to evaluate tradeoffs among policies related to land-use change, development, and ecological value (Costanza et al. 1996). For example, Costanza et al. (2002) developed and demonstrated an integrated ecological economics model for the Patuxent River watershed in Maryland, USA. The goal of these models was "to test alternative scenarios of land-use patterns and management" (Costanza et al. 2002). Simulations incorporated topography, hydrology, nutrient dynamics, and vegetation dynamics with changes in land use.

Habitat-Based Replacement Cost. The Habitat-Based Replacement Cost Method (HRC), a method derived from HEA, generates the habitat restoration (and its cost) needed to offset the losses of a specific number of organisms (Allen et al. 2005, Strange et al. 2004). This method for transferring value from organisms to habitat has been used in the context of replacement of fish lost by impingement and entrainment by power plants. The challenge in HRC is to estimate fish survival, growth, density, movement, and other determinants of productivity in various habitat areas. If HRC is estimated through the use of population models, this method is appropriately included as a complex valuation model.

In the context of HRC, we consider the cost of river habitat required to raise sturgeon—the largest freshwater fish in North America. Maintaining the river as

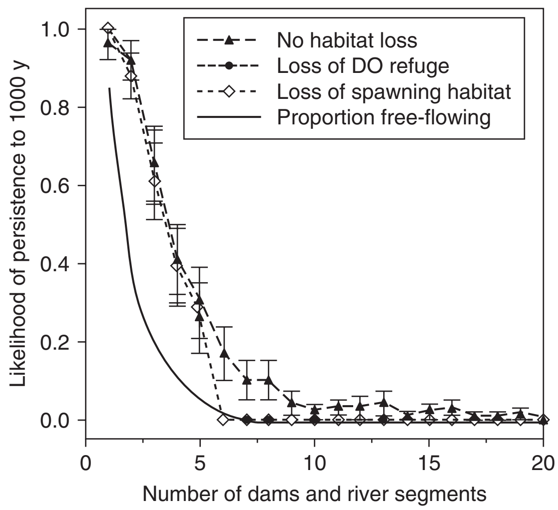

Figure 9.2. Simulated effect of increased fragmentation on the average likelihood of persistence, P_{1000}, for isolated white sturgeon populations. Results are shown for simulations with no loss of habitat and for two habitat-loss scenarios. Circles indicate the average of P_{1000} over populations, and error bars show the standard error in P_{1000} among replicate simulations, averaged over populations.

habitat for sturgeon places constraints on other uses of the river. For example, short river segments appear to be less suitable as white sturgeon (*Acipenser transmontanus*) habitat because they do not provide free-flowing areas used for spawning and for refuge from low oxygen levels in reservoirs (Jager et al. 2002). A population viability analysis model predicted an increase in the likelihood of persistence for white sturgeon populations as a function of the length of river habitat available (Fig. 9.2). Thus, preserving a spawning population may preclude the option of placing dams close together, which reduces the amount of hydropower that can be generated from the same parcel of water. The actual value of this energy depends on the specific characteristics of the site and the local value of alternative fuel sources.

The value of wild rivers may be estimated in part from the difference between the value of wild and hatchery fish. The cost of hatchery operation underestimates total replacement value of fish, because owners assume only the minimum costs by keeping fish until it is no longer cost-effective to do so, and they rely on a continued supply of wild broodstock to persist in the river. The number of adult fish that can practically be kept in a hatchery is low [e.g., 5–15 sturgeon broodstock in Logan et al. (1995)] because it is expensive to house and maintain large enough tanks to accommodate older and considerably larger fish. In addition, the cost of feed increases with sturgeon age due to decreased feed conversion efficiency. Survivorship of various life stages of fish, which can be factored into population models, also addresses differences in value between wild and hatchery fish. For example, in the wild, female white sturgeon produce 5600 eggs/kg compared with 3200 eggs from domestic broodstock, and egg survival increases from 18% to 41% (Logan et al. 1995). However, post-hatch survival of age-zero juveniles is lower in the river than in the hatchery (Jager 2005). Thus, the cost of operating hatcheries to replace reproduction is subsidized by the continued persistence of a wild spawning population and preservation of adequate spawning habitat in rivers.

Multivariate Analysis and Optimization. Ecological valuation brings us a step closer to making optimal decisions that combine ecological and nonecological objectives. This is because multiobjective optimization is facilitated by using a single currency to quantify different objectives. Valuation has been previously used in
Q3 an optimization context. For example, Field et al. (2004) used decision theory to maximize the ecological value of an endangered koala species. Various mathematical algorithms have been developed to optimize natural reserve design and reserve site selection (Church et al. 2000), two important applied problems associated with wildlands valuation. These focused on one type of ecological objective, thereby maximizing the number of species represented. Root et al. (2003) refined this objective by weighting species by proxies of extinction risk from organizations such as the World Conservation Union and the US Fish and Wildlife Service. Ferraro (2004) criticized the use of a single characteristic (e.g., genetic diversity measures, habitat suitability indices, number of species) to represent environmental amenities that are desired at least cost. He provided an alternative optimization approach to allocate funds for conservation cost-effectively by combining multiple biophysical and economic dimensions that contribute to value, using a distance function that can be estimated using nonparametric methods. Church et al. (2000) argue that the "quality" of species representation is just as important to include as number of species in optimizations for reserve site selection—that is, habitat value, adequate population size, presence of critical resources, and presence or absence of nonnative competitors. Moreover, in an examination of the optimal use of conservation funds by Wu and Boggess (1999), the marginal benefits of additional expenditures on wildlands preservation depended on cumulative benefits and correlations among benefits. Thresholds in ecological parameters translate into important thresholds in value that influence on the optimal spatial allocation of conservation funds (Wu and Boggess 1999, Johst et al. 2002, Wu and Skelton-Groth 2002). Wu and Skelton-Groth studied the optimal allocation of riparian conservation funds for salmonid restoration in Pacific Northwest in this context.

Ideally, wildlands are protected from human influences, but in many cases these lands (or waters) are also used for resource extraction, and the goal of optimization becomes minimizing impacts of resource use on the value of wildlands. Optimization of ecological value has been applied to other applied environmental problems such
Q4 as timber harvest and reservoir operations. Hof and Bevers (1998) offered numerous examples of spatial resource management decisions aided by spatial optimization, including harvest schedules, containment of pests by optimally treating areas of forest, and harvesting to minimize water quality impacts. In one study, they maximized the long-term diversity of species in a forest, measured by the joint viabilities of multiple
Q5 species (Hof and Bevers 1998). In general, studies have attempted to optimize land use with regard to either ecological objectives (species preservation; Haight 1995)

or human-use objectives (timber production, Nalle et al. 2004). However, ecological optimizations that consider both ecological and economic objectives together are rare.

Not all applications of ecological valuation truly maximize ecological objectives. For example, a recent review characterized the state of the art in reservoir operation toward ecological sustainability (Jager and Smith 2008). The majority of studies, and all that were implemented in practice, used legally mandated restrictions (e.g., minimum flows) as constraints on efforts to maximize other values, such as the amount of hydropower or revenue generated. Consequently, the value of water was not optimized, because the analyses assumed that a fixed amount of instream flow would be best—neglecting the considerable value, as measured by willingness to pay, of higher instream flows (Loomis 1998).

Four approaches to measuring ecological value as a function of flow were considered in reservoir optimizations: (1) the effect of flow regime on water quality in the upstream reservoir, downstream tailwater, or downstream estuary; (2) the effect of flow regime on fish habitat; (3) the deviation of flow regime from a natural flow regime; and (4) the effect of flow regime on simulated fish population viability. At least two model-based approaches have been used to optimize flow regimes, one emphasizing fish population responses to flow and the other emphasizing water allocation aspects of the problem. In an example of the fish modeling approach, Jager and Rose (2003) identified flow regimes to maximize salmon recruitment. In an example of a water-allocation approach, Sale et al. (1982) included more-realistic restrictions on water availability, while treating adequate fish habitat as a constraint.

Some argue that wildlands have the highest value if they not only provide good habitat and associated existence value but also facilitate human access (e.g., with trails or navigable waters) and therefore provide some human use value. However, evidence that willingness to pay for preservation far exceeds other components of ecological value (e.g., Loomis 1998) suggests that access is not an important part of value. In addition, roads are strongly correlated with human disturbance and consequent loss of ecological value as wildlands for ecosystems ranging from lakes to forests.

FUTURE DIRECTIONS

We believe that three main directions in wildlands valuation share great promise for advancing the science: (1) developing theories and methods for representing temporal variation in ecological value, (2) developing theories and methods for understanding how spatial context influences ecological value, and (3) developing theories and methods for representing ecological relationships in ecological value.

Incorporating the Future in Wildlands Valuation

The future plays a different role in ecological valuation from its role in valuation of nonecological services and commodities. Time is traditionally considered in valuation through discounting—that is, representing the fact that goods and services that are anticipated in the future have lower value than the same goods and services today

(Ludwig et al. 2005). Ecological thresholds can be reached beyond which related goods and services will cease to be available. For example, harvest of a fish population today can result in its economic collapse in future. This outcome likely reduces discounted use value for future users and nullifies existence value. It has been shown that making environmental management decisions based on conventional statistics (low Type I error rate) leads to suboptimal results, because the risk of reaching an ecological threshold is not taken into account (Field et al. 2004). Q6

Quantifying the risk of future extirpation should be a priority for valuation of populations that are rare. Rarity influences value in two major ways. With respect to use value, scarcity leads to increased marginal value of an individual and decreased total value of the population. Rarity also inflates the existence value of ecological entities, because long-term persistence is threatened. Both future use value and existence value are lost when extirpation/extinction thresholds are reached.

A simple approach to assign value based on extirpation risk is to quantify rarity. Value is sometimes assigned to rarity based on semiquantitative indicators (e.g., Efroymson et al. 2008a). A more quantitative and complex approach is to use population models to estimate future risk of extirpation via population viability analysis (PVA). PVA models have only occasionally been used as tools in ecosystem valuation (see HRC discussion above). One use of PVA models is to identify extirpation/extinction thresholds such as the minimum viable population size or the minimum area of suitable habitat (MASH) for a particular species. These thresholds may be important for estimating existence value of a population or the value of a service that is uniquely provided by that population. PVA models can estimate MASH by linking habitat quality and quantity to population processes such as survival and reproduction. The effects of temporal variation on extirpation/extinction risk are simulated by representing (1) environmental stochasticity (year-to-year variation in weather or other environmental variables that influence individual survival or reproduction), (2) demographic stochasticity (chance of extirpation due to small population size), and (3) catastrophes. The use of PVA models has been identified as a priority for advancing the science of ecosystem valuation (US EPA 2006).

Whereas populations face a risk of extirpation, other ecological entities face different risks of irrevocable loss. For example, functioning ecosystems can be destroyed or altered by unnatural and permanent disturbances (e.g., processes of residential or industrial development), particularly when no sources of reintroduction or restorative processes are operating.

Even when extirpation or functional thresholds are remote and the risk of irrevocable loss is zero, changes in ecological value over time can be important. For example, in rivers below dams, both the economic value of hydropower and the ecological value of flow to fishes vary seasonally. If one were trying to design optimal flow regime to permit sustainable coexistence of salmon populations and hydropower generation, it would be important to consider two things. The first is that salmon require higher flows during spawning migration and outmigration than during other times of the year. The second is that hydropower is more valuable during certain times of the day and week (Jager and Smith 2008). Changes in rarity of species and their habitats are also important components of ecological value.

Additional temporal factors that may influence the value of wildlands are worthy of consideration. Trends in ecological recovery and succession and the values of associated ecological attributes will be altered with climatic change. Species niches may change dramatically in the future, with some increasing in suitable area and others disappearing entirely (Hoffman et al. 2005, Best et al. 2007). The challenge will be to describe not only the dynamic changes in ecological systems, but also changes in human preferences.

The vagaries of human preference have a dynamic influence on value, but one that we often neglect. Combined models that forecast changes in human preferences in response to ecological futures can be used to estimate future changes in the value of ecological entities.

Incorporating Spatial Context in Wildlands Valuation

Some aspects of wildland value, like those associated with habitat connectivity and species rarity, come not from qualities intrinsic to individual patches of habitat, but from characteristics of their surrounding landscapes. These contributions arise from the physical placement of the wildland patch and its spatial relationship and juxtaposition with the other patches in the surrounding matrix. Changes to the landscape matrix and to other wildland patches in the constellation can have cascading effects on the value of other wildland patches, even those far from the change. The fact that the ecological value of a wildland site, such as species existence value or value for hunting, derives not only from the site itself, but also from its contextual location, is ignored by EBIs (e.g., USDA 2004).

We anticipate that ecological value will be refined in the future through more complete consideration of the complementarity of ecological services in adjacent lands and waters. Many examples demonstrate how the ecological services of adjacent communities add value to plant and animal habitat (Table 9.1). Lakes and rivers provide critical sources of drinking water for terrestrial organisms. Wetlands increase the habitat value of adjacent land parcels and water bodies by removing toxicants, reducing sediment loads, transforming nutrients, and providing specific habitat needs (e.g., breeding habitat for amphibians) (King et al. 2000, Rosensteel and Awl 1995). Different life stages may require different habitats in close juxtaposition. For example, floodplains provide slow, shallow river habitats that serve as nursery areas and refuge from predators for fishes (Welcomme 1979). Similarly, wooded riparian zones provide maternity roost sites for bats that forage above adjacent ponds. Another illustration of adjacent and complementary ecological services relates to pollination. Kremen et al. (2004) developed a relationship between (a) the proportion of upland natural habitat within several kilometers of an agricultural site and (b) the magnitude and reliability of crop pollination services performed by native bees.

Although the importance of landscape juxtaposition is increasingly recognized in measures of habitat suitability, it is rarely included in ecosystem valuation. Geographic information systems (GIS) are useful to measure distances between areas with particular land-cover or land-use classifications.

Corridors and Connectivity. Movement corridors improve the habitat quality or suitability of adjacent land areas and water bodies. Connectivity increases habitat value of metapopulations because populations in local patches are more likely to be rescued from chance extirpation by immigration from other, connected patches. The presence of habitat corridors has been shown to be correlated with increased native plant species richness in connected patches (Damschen et al. 2006). However, connectivity can also encourage the encroachment of weedy and invasive species, competitors, predators, parasites, and diseases.

The next challenge will be to quantify connectivity and its influence on habitat quality and, ultimately, its contribution to perceived value of a wildland to humans. Many approaches have been used to detect and quantify connectivity among patches within a landscape. Researchers at the Savannah River Site in South Carolina, USA, have taken a direct experimental approach to quantifying connectivity effects by (a) cutting voids in a pine forest to create negative "patches" connected by negative "corridors" and (b) studying the resulting impacts on seeds, plants, rodents, butterflies, and birds (Tewksbury et al. 2002, Haddad et al. 2003, Damschen et al. 2006). Morphometric image analysis, involving sequential dilation and erosion of patches and matrix, has been used to determine the degree of direct and indirect landscape connectivity (Vogt et al. 2007). Even electrical circuit theory has been used to simulate metapopulation connectivity via estimates of impedance and current flow through the habitat patches and surrounding matrix (McRae 2006, McRae et al. 2005). Individual-based models using virtual "walkers" as software agents have also been used to simulate movement preferences of a target species to quantify connectivity and to locate potential optimum movement pathways through a landscape (Gustafson and Gardner 1996, Gardner and Gustafson 2004, Hargrove et al. 2005).

GIS-based analysis of Least-Cost Path (LCP), originally developed to help plan roadway construction routes, was among the first analytical techniques to be borrowed for connectivity analysis. Once parameterized for the cost of movement or friction through each habitat type, LCP results in the pathway of lowest cost between two specified patches of habitat. In one application, the Southeastern Ecological Framework, funded by the US EPA, used GIS-based LCP methods to create a network of forest patches and "linkages" across the southeastern United States (Hoctor et al. 2000).

Graph theory represents individual habitat patches as nodes connected by line segment "edges" to form a connected network (Keitt et al. 1997, Urban and Keitt 2001). Edges may represent simple Euclidean distance, or they may reflect more complex costs of movement. The importance of any connecting edge can be calculated by the number of connections emanating from its two nodes. The minimum spanning tree is the shortest set of edges connecting all nodes. This tree, which shows how to connect all habitat patches with minimum cost, solves problems similar to the famous traveling salesman problem. Graph-theoretic approaches quantify connectivity, but do not explicitly map movement corridors geographically on the landscape.

One should distinguish structural habitat corridors (narrow portions of patches of high-quality habitat) from functional habitat corridors (paths between different patches of high-quality habitat that pass through an intervening matrix of lower-quality habitat). Both structural and functional connectivity affect the habitat value of a particular patch to wildlife, no matter where that patch falls in the continuum of habitat quality for a particular species. In the future, these methods for quantifying connectivity could be integrated into measures of habitat value. Habitat value influences the human use and existence values of relevant species.

Percolation Thresholds. Percolation theory (Stauffer 1985) predicts abrupt thresholds of connectivity as the number and quality of individual connections increases. Nonlinear percolation thresholds, which have been observed empirically in many fields, should have similar, dramatic effects on connectivity-based habitat value (Fig. 9.3). As the number and strength of connections increases, a critical percolation threshold is reached, and connections span the landscape. Spanning connections suddenly and abruptly allow even patches that are separated by significant geographic distances to be open to migrating individuals. Wildland valuations based on habitat connectivity should show a similar nonlinear jump in value near this percolation threshold.

There may, however, be an optimal level of connectivity for patches within a particular landscape. The best degree of connectivity should be one that allows for communication among all patches throughout the metapopulation, but no more. Connectivity in excess of this sufficient ideal may make metapopulations too vulnerable to epidemic processes like species invasions, parasitism, disease, and wildfire (Simberloff and Cox 1987, Minor and Urban 2008). If such disturbances can sweep across multiple, connected patches, metapopulations are less likely to find refugia. The connectivity-based value of wildlands could also decrease beyond this optimum connectivity (the dotted line in Fig. 9.3). However, connectivity is both species- and landscape-specific.

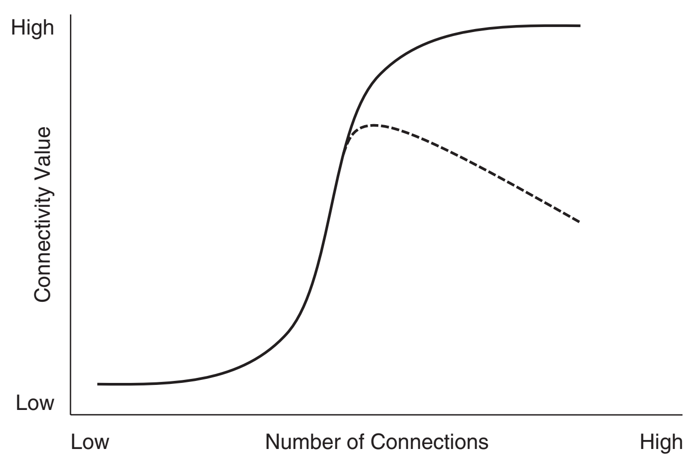

Figure 9.3. Habitat value in relation to landscape connectivity. The dotted line represents situations where connectivity may promote species invasion, disease, or other negative consequences.

A landscape feature that serves as a movement corridor for one species can be a barrier to the movement of another. Thus, future research in ecosystem valuation should include methods for optimizing connectivity for multiple species within the same landscape.

Incorporating Spatial Scale into Wildlands Valuation. In the future, ecological valuation also will have to deal more explicitly with notions of spatial scale. Hein et al. (2006) (and references within) have noted that "to date, relatively little elaboration of the scales of ecosystem services has taken place." Thus, research should clarify these spatial scales. Moreover, the relative importance of global value versus national value versus regional value versus local value will have to be negotiated on a case-by-case basis and, more generally, where national or other policy is involved. For example, if wildlands support carbon sequestration (a global value), species or community existence value (variable with scale), and hunting value (primarily regional value), how should these scale-dependent values be weighted? The answer will influence the relative emphasis of ecological valuation research efforts at different scales. One reason that existence value is often higher than other components of ecological value is that estimates are scaled by the number of individuals. Individuals surveyed from distant areas may express preferences for preservation of a given ecosystem or species, but individuals from these same areas may not be counted in the estimated use value for hunting or fishing.

Earlier we described the importance of incorporating influences of the spatial arrangement of the landscape in wildland value. We note that effects of both connectivity and juxtaposition on wildland value are scale-dependent. All maps are finite; consequently, edge effects could cause connectivity effects on habitat value to be underestimated. Likewise, boundaries can cause estimates of how juxtaposition will influence habitat value to be inaccurate. Therefore, it may be important to consider connections with outlying areas in estimates of value of ecosystem components in a smaller area.

Incorporating Ecological Linkages in Wildlands Valuation

A common complaint regarding ecological valuation is that ecological entities are not fully valued, especially in scenarios where monetization is required. Relationships among species and their food, consumers, habitat, limiting nutrients, and functions are only rarely reflected in relative human preferences. Values of populations or services may be extended from one site to another through "benefits transfer." However, until now, benefits transfer methods have rarely taken advantage of ecological relationships to transfer values among related ecological entities, such as habitats and populations, or predators and prey. Transfers of ecological value have previously been extended from predator to prey (Allen and Loomis 2006), ecosystem to ecosystem (compensatory natural resources restoration, NOAA 2000), organism to habitat (Allen et al. 2005), commodity to enabling ecological service (e.g., crop to pollination; Losey and Vaughan 2006), and ecological service on one site to service on another (pollination; Kremen et al. 2004). Relative ecological valuation may also be used to transfer value from function to structure, population to individual, or population to habitat. We believe that extending monetary values to heretofore unvalued ecological entities through ecological modeling is an important new direction for wildlands valuation.

Integrating the results of ecological models with estimates of monetary value also requires economic research. In addition to developing models of ecological properties that influence value, it is necessary to estimate use and non-use value for different ecological entities. For example, the willingness to pay for a wildlife or plant population of different sizes—that is, those further from versus closer to an extirpation/extinction threshold—may be integrated with PVA results. Likewise, one might estimate willingness to pay for ecosystems that are perceived as more and less wild and ecosystems described as having more or less capacity to recover from disturbance. Efforts are needed to generalize from contingent valuation surveys using meta-analysis and to understand the functional form followed by human values. Development of such general economic models is needed.

Landscapes by Design

In the future, we would like to see spatial optimization used to design efficient, sustainable arrangements of uses and services on the landscape. We envision maximization of ecological value as the objective integrated over a long time horizon. The time horizon is critical, because optimal decisions based on short-term returns inevitably result in poor resource management decisions, as evidenced by numerous overharvested marine fish stocks. Field et al. (2004) demonstrated that management decisions involving rare species based on traditional statistical hypothesis tests resulted in much higher costs than those derived by minimizing long-term management costs. This is because the economic cost of Type II errors (risk of extinction due to a poor decision) is high, and hypothesis tests do not provide a cost-efficient way of deciding whether management intervention is needed.

Another issue is whether to optimize landscapes holistically, permitting mixed arrangements of wildlands with more intensively managed lands. Kareiva et al. (2007) write of the “domestication” of nature, and they suggest that we need to have a willingness to shape such domestication. They assert that we should shun the notion that “wilder is better.” Others counter that humans are not capable of understanding ecosystem–human systems well enough for such a utopian vision and that our best bet is to set aside wildlands. From a theoretical standpoint, solutions obtained to problems that permit mixed use will be better than those obtained by separate optimizations of the two types.

We stand to learn a great deal by developing and applying tools that can identify optimal arrangements of alternative land uses that maximize the value of wildlands, possibly along with those of human land uses (e.g., agriculture, rangeland, and urban). Spatial optimization, which allocates human uses and ecosystem services on the landscape, is a tool used in landscape architecture and design (Nassauer et al. 2002, Santelman et al. 2004). Designs may be optimized, tested and evaluated in simulations before they are physically wrought on the landscape (Fernandez et al. 2005). Competing land uses must be evaluated in an even-handed way and must consider all requirements, costs, and benefits (Musacchio and Wu 2004). However, current social and political systems may not allow us to enact, control, enable, and enforce such optimal landscape design solutions (Musacchio et al. 2005). History suggests that governments with the centralized decision-making authority required to implement such regional plans ultimately further political goals rather than scientific strategies for achieving long-term sustainability.

The need to evaluate alternative design schemes will increase as the human population grows and our ecological footprints spread. Landscape construction is a constrained, zero-sum game, because the total available area is fixed. The objective will be to maximize the value of wildlands, and the best designs will harmonize conflicting or competing land uses for optimal value and sustainability. The promise and challenge of wildland valuation will be to provide the tools and functions needed to design better landscapes for our environment and our society.

CONCLUSION

Valuing wildlands is essential to environmental decision-making and landscape design. Without wildland valuation methods, wildlands will be assumed to have no value. Economic valuation methods need to incorporate ecological models to provide reasonable estimates of total value. Limburg et al. (2002) note that "from a purely ecological perspective, valuation begins with identifying the key structures, functions, and interactions of systems, and probing these (via models or experiments) to understand which are important in maintaining their condition, dynamics, and production of ecosystem services." Population dynamics and spatial ecology are disciplines that will come to the forefront of ecosystem valuation. The valuation of wildlands will increasingly incorporate the spatial context of the land and temporal aspects of organisms and their functions, and methods will be selected that are appropriate to the decision context. Research involving extirpation/extinction thresholds and their equivalents at higher levels of ecological organization will achieve prominence in ecosystem valuation. Applications of wildlands valuation will be as diverse as the selection of land areas to conserve, the selection of remediation alternatives, the valuation of benefits of environmental research and development, and the design of multipurpose landscapes.

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Queries in Chapter 9

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