Chapter 3

VALUING ECOSYSTEM SERVICES: PHILOSOPHICAL BASES AND EMPIRICAL METHODS

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Societies often must choose between alternative uses of the natural environment. Should a given wetland be preserved, or should the land be drained and converted to agricultural use? Should a particular timberland be maintained in its current state, or should it be opened to forestry or other development? Should a certain park be maintained, or converted to a parking lot? These are difficult questions. The way they are answered has critical importance for the viability of species in the habitats involved as well as the performance of the complex ecosystems of which they are a part.

To make rational choices among alternative uses of a given natural environment, it is important to know both what ecosystem services are provided by that environment and what those services are worth. The first item lies in the realm of fact; the second, the realm of value. Societies cannot escape the value issue: whenever societies choose among alternative uses of nature, they indicate (at least implicitly) which alternative is deemed to be worth more. In many instances, environmentally concerned individuals sense that the wrong decision has been made—that society has imputed insufficient value to nature in its current state and has thereby permitted conversion to take place for the sake of an inferior alternative. Indeed, one may sense that nature routinely is undervalued. No matter how strong suspicions are along these lines, one cannot make a convincing case that nature is undervalued without having a philosophical and empirical framework for assessing nature's values. The philosophical element seeks to identify the ethical or philosophical basis of value, that is, articulate what constitutes the source of value. The empirical element aims to find techniques for the measurement of value, as defined according to a given philosophical notion.

This chapter considers both components in offering a framework for valuing ecosystem services. While most of the other chapters in this volume examine valuation issues as they apply to particular ecosystem services (soil conservation, pest control, pollination, etc.), this chapter is more philosophical and broader in its focus. Our attention to philosophical underpinnings helps clarify the ethical issues underlying different approaches to value. And our general approach to empirical valuation methods helps convey the range of empirical approaches available to researchers, as well as the strengths and limitations of these approaches.

The Philosophical Basis of Value

From what do nature's values derive? When we claim that a given living thing or species or habitat is worth such and such, what is the basis of that claim?

Competing Approaches

A broad class of approaches to value is represented by anthropocentric viewpoints: elements of nature are valuable insofar as they serve human beings in one way or another. Within the anthropocentric group is utilitarianism, which maintains that natural things (indeed, all things) have value to the extent that they confer satisfactions to humans. Economists endorse the utilitarian viewpoint; as we will discuss later, this approach is inherent in benefit-cost analysis.

At first blush, it might seem that a utilitarian basis for value cannot be consistent with safeguarding the planet or protecting "lower" forms of life. But utilitarianism does not necessarily imply a ruthless exploitation of nature. On the contrary, it can be consistent with fervently protecting nonhuman things, both individually and as collectivities. After all, we may feel that the protection of certain forms of life is important to our satisfaction or well-being, and thus be led to place a high value on these forms. Utilitarianism doesn't rule out making substantial sacrifices to protect and maintain other living things. But it asserts that we can assign value (and therefore help other forms of life) only insofar as we humans take satisfaction from doing so. The notion of satisfaction here should be interpreted broadly, to encompass not only mundane enjoyments (as with consuming plants or animals for food) but also more lofty pursuits (such as marveling at the beauty of an eagle).

The utilitarian approach allows value to arise in a number of ways. It embraces both direct use values (for example, the satisfaction from eating fish) and indirect use values (for example, the value that can be attached to plankton because it provides nutrients for other living things that in turn feed humans). This approach does not restrict value to forms of nature that are consumed: there are both consumptive and nonconsumptive use values. An example of the former are the values that might be attached to ducks insofar as they provide food. An example of the latter are the values we attribute to ducks that provide pleasure to bird watchers. This approach also includes non-use values: values that do not involve any actual direct or indirect physical involvement with the natural thing in question. The most important value of this type may be existence value (or passive use value)—the satisfaction one enjoys from the mere contemplation of the existence of some entity. For example, a New Jersey resident who has never seen the Grand Canyon and who never intends to visit it can derive satisfaction simply from knowing it exists.[1] The array of services provided by ecosystems spans all of these categories of values. The pest-control and flood-control services they offer have direct use value to nearby agricultural producers.[2] Their provision of habitats for migratory birds implies an indirect use value to people who enjoy watching or hunting these animals; depending on whether such birds are hunted or just observed, the indirect use value may be consumptive or nonconsumptive. Ecosystems also yield an existence value: wetlands, for example, provide such value to people who simply appreciate the fact that wetlands exist.

One can distinguish weak and strong forms of utilitarianism. The weak form asserts that the value of a given species or form of nature to an individual is entirely based on its ability to yield satisfaction to the person (directly or indirectly). The stronger form makes an assertion about the value of a species (or other natural thing) to society. It claims that the value to society of the natural thing is the sum of the values it confers to persons.

This stronger form of utilitarianism is inherent in benefit-cost analysis. An attraction of strong utilitarianism is that it provides a rather convenient way of ascertaining social values of alternative policies and thus offers a way to make difficult decisions. Benefit-cost analysis seeks to ascertain in monetary terms the gain or loss of satisfaction to different groups of human beings under each of various policy alternatives. Under each alternative, it adds up the gains and subtracts the losses, and then compares the net gains across policy options. Importantly, benefit-cost analysis doesn't cast judgment on the differences between one person's valuation of a given species and another's. Each person's valuation receives the same weight. It makes no attempt to correct for differences in awareness, education, or "enlightenment" among individuals. The preferences of people who have no concern for fubenefit-cost analysis falls short. To the extent that ecologists can show that the general public was unaware of significant ecological issues in forming their own valuations, this seems relevant to decision making. Moreover, even if individual valuations were based on very good information, there is an ethical dimension to the decision—associated with how the benefits and costs are distributed across affected parties or generations—that is not addressed by the simple adding up of individual benefits and costs. At the same time, we would affirm that benefit-cost information—in particular, the aggregate net benefits from various alternatives—remains useful in weighing the various policy alternatives.

A second main insight is that the leading alternatives to utilitarianism (and benefit-cost analysis) usually do not deal with "values" at all! The exercise of imputing values to different elements of nature is part and parcel of utilitarianism, but is not an essential ingredient of intrinsic rights or Kantian approaches to decision making. If one adopts an intrinsic rights or Kantian approach, the choice as to how to choose among policy alternatives usually reflects issues of whether fundamental rights are violated or whether the action in question is able to be universalized. Values may not be a central part of the consideration. This is important, because it suggests that making arguments for social policy by referring to the "value of ecosystem services" is to conform, to a degree, to the utilitarian approach. Ecologically concerned individuals should recognize that this is the case, and realize what issues can and cannot be addressed by a focus on values.

Measuring Ecosystem Values

It is difficult enough to agree on a philosophical basis for value. Further difficulties arise in attempting to measure nature's values (after assuming a given basis for value). This section presents some important measurement methods. Considerable progress has been made over the years in developing such methods. But the science is far from perfect. Controversies persist.

Ecosystem services are especially difficult to measure for the same reason that ecosystems themselves are threatened. Many of the services provided by ecosystems are positive externalities. The flood-control benefits, water-filtration services, and species-sustaining services offered by ecosystems are usually external to the parties involved in the market decision as to whether and at what price a given habitat will be sold. As a result, the habitats that support complex ecosystems tend to be sold too cheaply in the absence of public intervention, since important social benefits are not captured in the price. Public attention to the values of these (largely external) benefits is important to provide support for reasonable public policies to protect important habitats. This makes it all the more important to determine the values of these services. At the same time, it explains why gauging these values is so difficult: in many cases the values of these services are not directly expressed in market prices.

The prevailing approach to ascertaining value is benefit-cost analysis. As indicated, benefit-cost analysis implicitly adopts the utilitarian basis for value. The value of a given living thing is the amount of human satisfaction that thing provides. How could such satisfaction be measured? Nearly every empirical approach assumes that the value of a given natural amenity is revealed by the amount that people would be willing to pay or sacrifice in order to enjoy it. Willingness to pay is thus regarded as the measure of satisfaction.

It is important to note that willingness to pay is not always an actual, expressed willingness; it is not restricted to what we observe from people's actual payments in market transactions. Rather, it expresses how much people would be willing to pay for a given good or service, whether or not they actually have opportunities to do so. Market behavior often gives evidence of willingness to pay, but in many instances researchers must rely on other, more indirect methods to fathom it.

Ecosystem Services and Valuation Methods

The myriad services offered by ecosystems can be divided into three main categories: (1) the provision of production inputs, (2) the sustenance of plant and animal life, and (3) the provision of non-use values, which include existence and option values. Different types of valuation techniques are called for, depending on the category of service involved. Table 3.1 shows the relationships between service types and valuation methods.

Valuing Production Inputs

Table 3.1 lists four examples of production inputs from ecosystems: pest control, flood control, soil fertilization, and water filtration. These services are inputs to the sustained production of agricultural products in the sense that it would be difficult to maintain agricultural production without pest control, flood control, fertile soil, or (at least in some cases) relatively pure water.

One can place a value on these production inputs by recognizing what costs or expenditures agricultural producers manage to avoid by virtue of the availability of these inputs. For example, where ecosystems provide effective pest control, farmers can avoid undertaking expenditure on alterna-

Table 3.1. Ecosystem services and valuation methods

Service	Valuation Method
Provision of Production Inputs	
Pest Control	Avoided Cost
Flood Control	Avoided Cost
Soil Fertilization	Avoided Cost
Water Filtration	Avoided Cost
Sustenance of Plant and Animal Life	
Plants/Animals with Direct Use Values	
• consumptive uses	Direct valuations based on market prices
• nonconsumptive uses	Indirect valuations (travel cost method, contingent valuation method)
Plants/Animals with Indirect Use Values	(No valuations necessary if plants/animals with direct use values are counted)
Provision of Existence Values	Indirect valuations (contingent valuation method)
Provision of Option Values	Empirical assessments of individual risk-aversion

tive pest-control methods such as the use of synthetic pesticides (Naylor and Ehrlich, this volume). To the extent that data are available on expenditures on synthetic pesticides, they provide an indication of the value of the pest-control services provided by ecosystems.[9]

Similarly, the flood-control services offered by ecosystems eliminate farmers' needs to undertake alternative flood control expenditures. The avoided costs of flood-control again indicate the value of the services provided by ecosystems; here the cost may be avoided by taxpayers (who otherwise pay for flood-control projects), rather than farmers, but the principle still applies. The same logic applies to soil fertilization and water filtration services.

Of course, farmers' circumstances vary, and the avoided costs associated with these ecosystem services will therefore vary for different farming enterprises. Given this heterogeneity, it becomes difficult to pinpoint the ecosystem values. Nevertheless, attention to avoided costs offers a very useful gauge of the values of the production inputs supplied by ecosystems.

Valuing Plant and Animal Life

As suggested by table 3.1, a second main type of service provided by ecosystems is the sustenance of plant and animal life. In choosing a method for valuing this type of service, it helps to distinguish living things with direct use values from those with indirect use values. Examples of the former are plants or animals that are consumed as food or that directly offer recreational values (sightseeing, nature watching, etc.). Examples of the latter are plants and animals (such as organisms that are lower on the food chain) that help sustain other plants and animals that we enjoy directly. To give specific examples: ecosystems generate direct use values by supporting the various types of birds that we enjoy either nonconsumptively as bird watchers or consumptively as bird hunters. They generate indirect use values by supporting the life of various plants or insects that in turn enable birds to thrive.

Direct, Consumptive Use Values. When direct use values are involved, two main valuation methods may apply. In the case of direct consumptive use values, it may be possible to employ direct valuation methods based on market prices. When natural ecosystems provide a habitat for animals that are harvested and sold commercially, the commercial market value provides a gauge of the value of the habitat services. For example, part of the value of marine ecosystems is conveyed by the value of the commercial fish that they help sustain. Of course, this only represents a portion of the value of the ecosystem—namely, the value of the ecosystem's potential to sustain fish with a market value.

There is an important difference between the marginal and the total value associated with market prices or the willingness to pay of consumers in markets. Economists regard the prices that people are willing to pay as indicators of the marginal value—the value they place on the last unit purchased. Consider what a homeowner would be willing to pay for residential water in a given month. He might be willing to pay a huge sum for the privilege of consuming the first ten cubic feet, because doing without them would deprive him of even the most fundamental (and valuable) uses of water for that month: drinking water, the occasional shower, etc. The next ten cubic feet would probably not be worth quite as much. They would allow him additional opportunities to fill a glass from the faucet, and an extra shower or two, but these would not be as critical to him (or to the people with whom he associates!) as the first ten cubic feet. Thus the marginal value of water—

the amount one is willing to pay for each successive increment—falls steadily.

Figure 3.1 displays a typical willingness-to-pay schedule. The first cubic foot is shown to be worth a great deal more than the fiftieth, which in turn is worth much more than the hundredth. In reality, of course, households don't have to purchase each unit of water at its marginal value. If they did, they would be charged larger amounts for the first increments than for later ones. Instead, households generally pay a given price per unit of water, regardless of how much they consume.[10]

In figure 3.1, the horizontal line at $0.02 represents the price charged for the water. (We use this number arbitrarily.) The standard economic assumption is that users will continue to purchase water until the marginal value of the water (or marginal willingness to pay) is equal to the marginal sacrifice (or price). Under these circumstances, the price is an expression of the marginal willingness to pay, or marginal value. (In the example of figure 3.1, the user would demand four hundred cubic feet of water per month at this price.)

The total value of the water consumed is much more than the price, however. The total value is the area under the marginal willingness-to-pay schedule (the sum of areas I and II in the diagram). Note that to ascertain total value (as opposed to marginal value), researchers need to have infor-

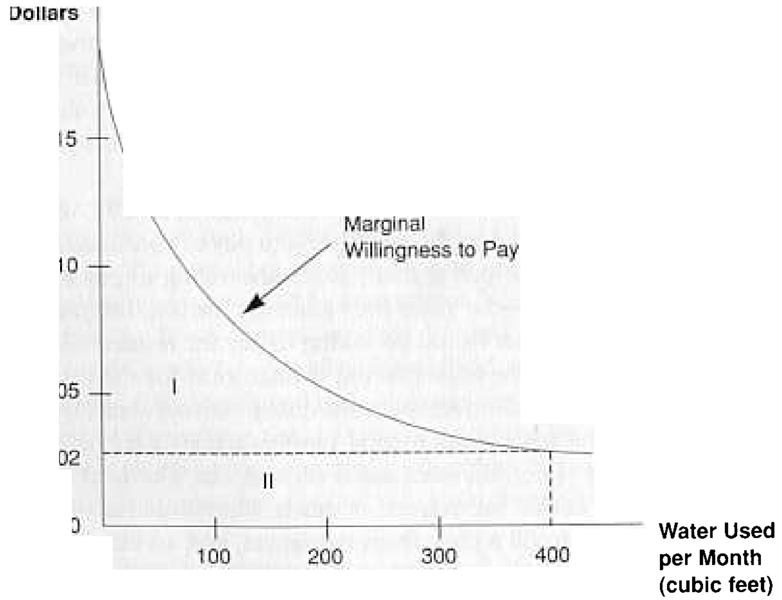

Figure 3.1. Relationship between water use and marginal willingness to pay.

mation on the entire marginal willingness-to-pay schedule (or demand curve), not just the price paid.[11] A main challenge of empirical valuation techniques is to trace out marginal willingness-to-pay schedules.

In the context of commercial products of ecosystems, this means that market prices represent only the marginal value of these products. The value of the total sales of these products corresponds to area II in figure 3.1. Note that this is less than the total value to consumers, which is the sum of areas I and II. Thus market sales understate the overall value of the commercially viable forms of life supported by ecosystems.

Direct, Nonconsumptive Use Values. Within the category of direct use values from living things maintained by ecosystems, we have another case to consider. This is the case where the life forms are used nonconsumptively. For such uses, the relevant markets do not usually arise, and thus it is not possible to gauge values directly by observing market prices.[12] For example, there usually are no markets for the bird-watching opportunities that ecosystems provide by offering suitable habitats. In these cases, it is necessary to apply more inferential methods to ascertain the relevant values.

The travel-cost method is a widely used inferential approach. The method has been applied to ascertain some of the values provided by parks, lakes, and rivers—or, equivalently, the costs that result from the loss of these elements of nature. The nonconsumptive uses are not directly bought or sold in markets; prices are not usually charged for their use. And in those instances when use prices are charged (through entry fees, etc.), the prices are unlikely to be good indicators of (marginal) value. That is because the users of these resources actually "pay" more than the entry fees to use them. For example, the cost of the family visit to Yosemite National Park is much greater than the $15-per-day use fee. The travel-cost method recognizes that by adding to the entry fee (if any) the transportation cost and time cost expended to visit a particular site, one can ascertain the overall travel cost. This method regards the overall travel cost as a measure of the marginal willingness to pay by a visitor to the park; this is considered to be the same as the marginal value of the park to the visitor. The underlying assumption is that people will continue to visit the park until the value of the last unit (that is, the marginal value) is just equal to the travel cost.[13]

It is also possible to use survey methods, such as the contingent valuation method, to determine how much value people place on the nonconsumptive uses.[14] Many economists distrust results from survey approaches, claiming that individuals' asserted preferences in the hypothetical circumstances posed by surveys bear no systematic relationship to their true preferences. Defenders of survey methods counter that, in many cases, surveys are the only method available. This "only game in town" argument may have force when existence values are involved, as discussed below.

Indirect Use Values. Ecosystems support many "lower" forms of life that provide only indirect use value. It is sometimes argued that the value of ecosystem services should include the values of the services provided by these life forms. But in fact there is no need to include the values of these services in an accounting of the overall value of an ecosystem. The values of these services are already captured in the values attached to the life forms that humans enjoy directly. Consider the value of certain plants whose fruits are eaten by birds and other "higher" life forms; assume humans obtain no direct use value from these plants. If we abide by the utilitarian approach to value, then there is no value to these plants over and above the value that we attach to the higher life forms to which they contribute.[15] To add their indirect use values to the direct use values would be double-counting.[16]

Non-Use Values

Some of the values from ecosystems do not involve direct or indirect uses of the good or service in question. These are non-use values. There are two main types of non-use value.

Existence Value. This is the value that derives from the sheer contemplation of the existence of ecosystems—apart from any direct or indirect uses of goods and services they provide.

Survey approaches such as contingent valuation assessments may be the only way of ascertaining existence value, since actual market and nonmarket behavior gives little hint of its magnitude. As mentioned, survey approaches are controversial. Yet they may be the only way of measuring existence values because people's actions do not leave a "behavioral trail" from which their valuations can be inferred. In this limited space we cannot offer an appraisal of survey approaches.[17] But we can point out what seems to be the key underlying question: whether the information obtained from surveys, however imperfect, is better than no information at all. In the next section we revisit issues of uncertainty and imperfect information.

The existence value could include a pure biodiversity component. This is the appreciation for the variation or richness we observe in the ecosystem; it is based on the contemplation of the ecosystem as an ensemble of life forms, as contrasted with an appreciation for each of its members individually. Although we mention this value in connection with existence value, the pure biodiversity value may also have a use-value component: we take pleasure in the ecosystem's heterogeneity when we visit the habitat in question and observe the diversity of life forms that reside there.

Option Value. As developed in the economics literature,[18] the term "option value" refers to a premium that people are willing to pay to preserve an environmental amenity, over and above the mean value (or expected value) of the use values anticipated from the amenity.[19] This premium reflects individual risk-aversion: in the absence of risk-aversion, people's willingness to pay would equal the mean use value (its expected value), and option value would be zero. It is much easier to define option value than to measure it. Its measurement requires a gauging of individuals' risk-aversion, and this may depend on the specific context: persons are not equally averse to different types of risk. For an empirical assessment of option value, see Fisher and Hanemann 1986.

Marginal vs. Total Value

In much of the preceding discussion, we have concentrated on measurement of the total value of ecosystems. But in many real-world circumstances, the policy debate concerns the change in value or marginal loss of value that results from alteration or conversion of a part of the region that occupies an ecosystem. In benefit-cost analyses, when a portion of the ecosystem is threatened with conversion, it may be more important to know the change or loss of ecosystem value associated with such conversion than to know the total value of the entire original ecosystem. Does a "minor" encroachment on the land area of an ecosystem generate small losses in ecosystem value, or do small encroachments precipitate large damages?

To examine this issue, we can begin with a very large area of a (relatively) undisturbed ecosystem.[20] The value we place on a given amount of area lost to other uses depends on the area of this system.[21] Let A represent the land area of our ecosystem, and suppose that the initial area is A_0. This ecosystem, valued for its natural beauty and its biological diversity, is being decreased marginally in area by being converted to farmland. Suppose first (counter to fact) that this decrease takes place without changing the ecosystem's character through species loss. Since a larger area is worth more than a small one, the marginal value of each withdrawn unit rises gradually as the area (A) decreases. But in the limit, an area of size zero is worthless, and tiny areas are less attractive because they have a rather zoo-like character. Thus at small values of A, the marginal value begins to fall again. This relationship is shown in the path marked "1" in figure 3.2. The relationship between area and value expresses the pure ecosystem-scale effect.

In fact we know that the biological diversity of the ecosystem—one of the features contributing to its value to nature lovers—is not area-independent. The relationship, established mainly in studies on islands and (to a more limited extent) on tropical forests, is a nonlinear one. The precise form varies, but in a variety of studies the number of species lost is slight until a quarter to a half of the area is lost, and rises precipitously after about three-

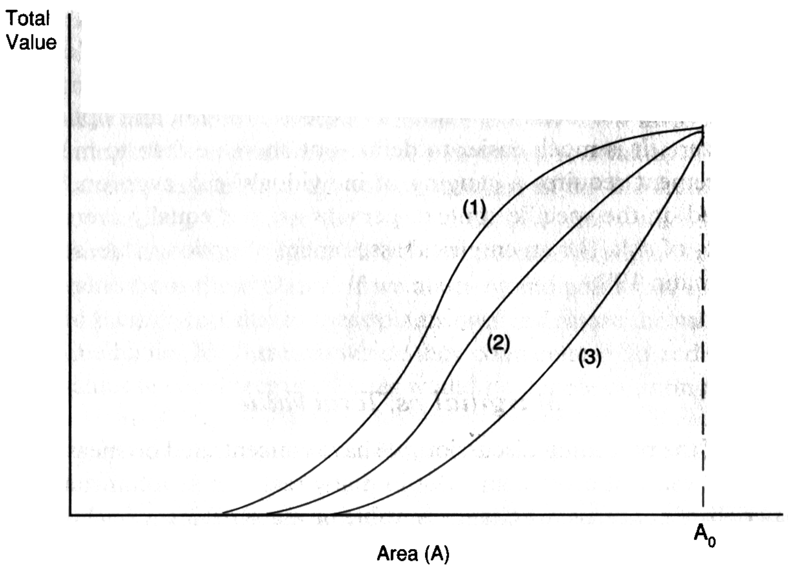

Figure 3.2. Habitat area and ecosystem value: ecosystem-scale, diversity, and species-composition effects.

quarters of the area is lost. As A is reduced the effect on marginal value is to exaggerate the loss of ecosystem value. The impact of the loss in numbers of species as A is reduced may be termed the diversity effect. This effect is taken into account in the path marked "2" in figure 3.2. As indicated by the differences between paths 1 and 2, this intensifies the marginal loss of value from a given reduction in A.

There is a third effect that needs to be considered. The species in ecosystem A are not considered to be of equal value to humans. People seem to care more about eagles and panthers than about mosses and bacteria. We also know that species are related to one another in a complex, co-evolved web of dependencies: prey and predator, plant and pollinator. Trophic relationships are also vitally important. Often, higher-order species on the food chain have the most exacting environmental requirements and are thus valuable indicators of the health of the entire ecosystem; they or others may also be critical "keystone" species because they are located at the center of a network of interdependencies. Thus, as a practical matter, species values become proxies for ecosystem values: the Endangered Species Act in the United States is an embodiment of this principle in policy. And of course we regularly justify large expenditures to save some species (e.g., the African rhinoceros) but not others (there is no Save the Furbish Lousewort Society).

On what basis do we assign value to species? The following are some axes along which different people make selections.

Taxonomic Proximity. We like animals that are like us. Primates attract human attention not only because there may be utility in the relationship ("animal models" for human disease) but because we respond to their quasi-human qualities.

Rarity. All other things being equal, we have more interest in rare things than in common ones. This is not simply a matter of vulnerability, although it is true that rare organisms are more vulnerable to extinction than abundant ones. Rarity itself can be the attraction; in some sense animals and plants in nature are "collectibles," if only in the sense of finding and listing them, and collections of the rare are more desired than collections of the commonplace. Indeed, "collecting" in the form of listing is a motive with powerful economic consequences. Many bird watchers will undertake extreme expenditures to visit ecosystems harboring rare species for the purpose of expanding their "life-lists."

Genetic Uniqueness. If a species represents a unique evolutionary line—is, for example, the only extant member of its genus or family—then it may be entitled to higher value than it would otherwise. Scientists especially would favor the use of this criterion.

Importance to Ecosystem Function. Certain species (such as "dominant" and "keystone" species) create conditions that permit the maintenance of the entire ecosystem. The dominant trees in a forest, or birds that dig nest holes in trees that are used by other species, or insects vital to the pollination of a dominant plant, would be examples.

How can these preferences be related to the marginal value calculation? Biological diversity is reduced as A shrinks, but species do not fall out randomly; certain kinds tend to drop out relatively early, others only when A becomes quite small. For conservation biologists and others, this means that wise policies cannot be made unless some value is attached to the different kinds.[22]

Obviously the number of possible criteria is large enough to prohibit development of a precise relationship among area, species loss, and value. But larger organisms with broad ranges that are especially area-sensitive would be likely to be rarer, on average closer taxonomically to humans, favored for "charm," and important to ecosystem function. Thus it is reasonable to assume a species-composition effect: that as A is reduced, the species lost early in the reduction are more valuable than those lost later. When this effect is taken into account, the marginal loss from a reduction in species area is even greater than indicated by path 2. Path 3 incorporates this effect (and the others).[23]

Obviously some of these relationships are uncertain, and the exercise could be applied to real natural areas only after substantial research. But it points up the importance of thinking about value in marginal rather than aggregate terms and suggests a discipline that could be applied in the framing of general conservation policy.

Uncertainty and Policy Making

It is evident that precise information on the values of ecosystem services will often be lacking. That is a fact of life, yet we still need to make choices. How can they be made as rationally as possible? What is the right framework for decision making in a realm of uncertainty?

An important first principle emerges from the still developing discipline of risk assessment, best known in the context of efforts to make quantitative estimates of the risks (to health, safety, and the environment) of pollutants. Plainly the same analytical format is applicable to other kinds of global change, including loss of ecosystem services. The frequent criticisms of the use of risk assessment in toxic substances regulation have usually singled out uncertainties in the extension of animal data to humans and in the extrapolation of results from high dosages to low. But a more serious problem has been a tendency to deliver assessments in the form of point estimates rather than probability distributions.

In the even more uncertain domain of ecological risk assessment, the density of probabilities around any estimate may be as important to the formation of policy as the point estimate itself. This is the case for several reasons.

First, many people, including those involved in the policy process, are risk averse and likely to concentrate attention on the unfavorable or high-cost side of the distribution of possible outcomes. If society as a whole approaches risk from a conservative or risk-averse position, the information contained in a distribution may be vitally important; that is especially true if the variance among estimates is high, that is, if the distribution is broad.

Second, the distribution of risk estimates is sometimes skewed, often toward the downside. The asymmetry is often not revealed until independent estimates are pooled. Furthermore, experience often modifies our view of a certain class of risk. For example, the adverse environmental consequences of introducing exotic organisms for control purposes have nearly always been unforeseen—so much so that it is probably now prudent to assume the worst.

Third, ecological risk factors are frequently multiple and interconnected. The leverage provided by such relationships is difficult to predict, but it is clear that apparently independent events may summate to produce levels of effect much greater than the sum of disaggregated risk estimates. Thus the distribution of outcomes for the system as a whole is much broader than one would expect based on risk assessments that concentrate on the individual risk factors separately. For example, a major environmental issue concerns the state of Everglades National Park in Florida, where a unique wetland ecosystem is threatened because historic flows of fresh water into the area have been slowed by human activity. It is interesting to note that in the complex history of development that has led to the present state of affairs in the Everglades, no single development—the Tamiami Canal, the engineering projects on the Okechobee drainage, or the intensification of agriculture—would by itself have been predicted to interrupt sheet-flow into the central Everglades and thereby disrupt the entire ecosystem.

Finally, the time dimension is often ignored in traditional risk assessment, yet the dynamic character of ecological risk often raises the time issue in a way that should amplify our policy concerns. In the first place, ecological change often shows a strong hysteresis: restoration processes work slowly, and intense perturbations may exact costs over a very long period. Second, and perhaps more important, human preferences—in this case, our interest in natural ecosystems—have changed with industrialization and affluence. Such changes pose a challenge for traditional benefit-cost analysis, since our assumptions about future preferences may err in understating value for future generations. Problems of intergenerational equity, difficult as they are to resolve, are at the heart of ecosystem valuation.

Examples of Real-World Valuation Challenges

The challenge in real-world decision making about land use is to evaluate the costs of altered use against the benefits. The latter are relatively easy to measure, but the costs of conversion—that is, the value of the loss associated with the "native" ecosystem—are much more difficult. The following examples illustrate some of the problems.

Wetlands

Consider the following situation applying to Jason Shifflet, a hypothetical farmer in the lower Mississippi valley. Shifflet has on his property a fifty-acre forested wetland. He wishes to drain this wetland, harvest the trees, and convert it to productive cropland. The parcel is connected (barely) to a larger swamp on state land; the entire wetland has been used heavily by local duck hunters and bird watchers.

In order to accomplish the conversion, Shifflet must follow provisions under two different federal laws. Under the terms of the "swampbuster" section of the 1990 farm bill, he would become ineligible for Department of Agriculture farm program benefits. Shifflet is not bothered by this, since his operation has been subsidy free, but he is concerned with meeting the requirements of Section 404 of the Clean Water Act. In order to ditch and drain the property, he is required under that law to obtain a permit from the U.S. Army Corps of Engineers. The law requires that steps be taken to minimize or avoid impacts on wetlands and to provide compensation for unavoidable impacts by other activities to restore or create wetlands.

Shifflet applies, emphasizing the care with which he proposes to accomplish the drainage. He will leave the portion of the property closest to the state land untouched and create a new wetland of nearly equal area on another piece of land he owns. His application will be examined not only by the Corps of Engineers but by the Environmental Protection Agency as well. They will look carefully at other values of the wetland parcel that Shifflet proposes to convert. They may apply value estimation methods that would encompass both consumptive (duck-hunting) and nonconsumptive (bird-watching) uses, by using travel-cost and other measures. Existence values may or may not be considered; contingent valuation techniques have been applied to some situations.[24]

In the end, Shifflet's application is denied; when added to the state parcel, his wetland generated substantial recreational values and represented—in the view of EPA reviewers consulted by the Corps of Engineers—an unacceptable loss. The Corps was inclined to agree, since Shifflet's drainage plan would have altered the remaining wetland area in unpredictable ways.

Shifflet has since become an active member of the Lower Mississippi Property Rights Forum, an organization dedicated to lobbying in favor of the application of "takings" provisions to lands devalued as a result of regulatory decisions.

The Galapagos Islands

A second example, international in character, is provided by the Galapagos Islands. This archipelago, located six hundred miles west of the Ecuador coast, consists of thirteen large islands and a number of smaller ones. All are of recent volcanic origin (100,000 to a million years old), and they contain a unique assemblage of plants and animals. They were visited by Charles

Darwin during the voyage of the *Beagle*, and now are an important site for contemporary studies of evolutionary biology—many of them carried out under the auspices of the Darwin Research Station located on the largest island.

Managed as a national park by Ecuador since the 1950s, the islands have also become a favorite destination for tourists, who explore the islands from boats and debark on the islands to follow carefully marked trails in the company of trained naturalist-guides. With the growth in popularity of "ecotourism," the Galapagos now attract about fifty thousand visitors each year.

There is a resident population on several of the larger islands, with a few service industries and a subsistence economy that depends on agriculture and fishing. These have been augmented by other direct uses that compete with the "natural" state of the islands. A significant fishery for sea cucumbers, a delicacy prized in Asian and French cuisine, has developed. Not only does it threaten the rich intertidal fauna; it has posed significant risks to the terrestrial ecosystem, through the introduction of "exotic" species and destructive camping on some islands. Other extractive industries are either established or in prospect.

Arrayed against these direct, consumptive use values are two other values. The first is the direct, nonconsumptive use value from ecotourism, which brings significant revenue. A sample calculation of this value would estimate that the average visitor spends the equivalent of $3,000 (a week on a boat is a typical excursion). If the visitor is from the United States, additional revenue will accrue to the Ecuadorian economy through accommodations on the mainland, the flight to the islands, and (if a national carrier is used) the flight to Quito or Guayaquil. A total per-visit value of $5,000 would be a reasonable figure for the "overseas" visitor: if half were Ecuadorian nationals and half from elsewhere, the value of the industry would be $200 million annually.

Local residents, however, would make quite a different calculation. The T-shirt shops and restaurants at Puerto Ayora collect some money, and the support of the Darwin Station by tourists flows into the local economy. Some boat operators are islanders, and some services for all vessels are locally provided. But the vast majority of the revenue flows to tour operators, many of them non-Ecuadorian, and to other off-island entities.

Thus it is not surprising that a sometimes violent controversy has arisen over the protection of the islands. When the government closed the sea-cucumber fishery in 1994 because the catch limit was being vastly exceeded, fishermen and some other local residents seized the Darwin Station and took scientists hostage. In a political controversy over a bill that would have given the fifteen thousand islanders more local autonomy (and relaxed many of the ecological protections) there was another takeover. The tense contest between extraction and conservation in the Galapagos is, at least with respect to this particular indirect use value, the result of distributional effects. The economic potential of ecotourism is almost certainly greater than that of the resource-extraction uses. Yet the residents retain most of the rents from the second and little from the first.

A second use value stems from the (uncertain) future benefits that would emanate from the scientific research under way on the Galapagos. The large number of endemic species found there, and the recency of their evolutionary divergence from mainland relatives, make the islands a living laboratory for studies of species formation. Important recent work (see Grant 1986) depends on the integrity of the ecosystems of certain islands. Calculating its value, of course, would be extremely difficult.

Finally, there are two important non-use values. First, as in the case of the wetland example, people who have never been to the Galapagos and never expect to, may experience a loss of existence value that they would willingly pay to avoid. The unique quality of the islands and the considerable publicity they have received as a mecca for naturalists gives this consideration a weight it might lack in less special areas. In addition, in the presence of uncertainty, people might be willing to pay a premium (over and above the expected future use value) to ensure the preservation of the unique flora and fauna of the islands. This is the option value.

Conclusions

To assess the value of ecosystem services we must choose among alternative philosophical bases of value as well as alternative measurement techniques. Philosophers will continue to debate the relative merits of alternative philosophical approaches, and we cannot hope to settle this debate here. We have given special emphasis to the utilitarian basis for value, in part because it underlies nearly all empirical assessments of value, including all benefit-cost analyses. Selecting the utilitarian approach does not eliminate from consideration nonconsumptive enjoyments of nature, nor does it disregard satisfactions that do not entail direct or indirect use, such as existence value.

The problems of measurement are at least as daunting as the problems of selecting or justifying a philosophical basis. Empirical assessments tend to disfavor the "natural" state in comparison with economically desired alternative uses for the same land, simply because the benefits from the alternative uses are usually more easily measured than are the benefits of ecosystem services. Many of the most important beneficial services of ecosystems are public (that is, jointly enjoyed) goods whose values are not expressed in market prices. The values of these services are therefore especially difficult

to measure. Evaluators often concentrate on the most easily measured impacts and ignore the difficult ones. As a consequence, the unacknowledged pathways may fade from view, yielding an overall value estimate of ecosystem value that is far too low. This indicates the critical importance of developing and improving measurement techniques oriented toward those ecosystems services whose values are not expressed directly in markets.

We have noted the importance of distinguishing carefully between aggregate value and marginal value. In many instances public projects or private developments encroach on portions of ecosystems, rather than the entire system, and in these cases the relevant question is the change in ecosystem value (or marginal loss of value), not the overall ecosystem value. Whether a particular ecosystem is or is not "worth saving" may depend critically on how much of the total area devoted to that ecosystem is still extant. Earlier, we illustrated the dependence of marginal value on total area, taking account of area-diversity relations and the differential value of species. That same kind of analysis could be modified to extend beyond the continuous-patch model we used to apply to the global distribution of certain plant and animal assemblages. Such considerations could help develop a more comprehensive strategy for allocating scarce conservation resources among competing needs.

Even granting our fondest hopes for success in this venture, however, for some time the values we can associate with natural ecosystems will be full of uncertainty. That uncertainty has to be incorporated into the estimates that serve decision makers; point estimates without probability distributions often lead to wrong conclusions, especially when—as often happens—the unstated distributions are skewed toward the more costly outcomes.

Although our discussion acknowledges a key role for benefit-cost analysis in the valuation of ecosystem services, we would emphasize that such analysis does not yield a sufficient criterion for deciding policy. Fundamental issues of fairness or distribution are ignored in benefit-cost assessments. At best, benefit-cost analyses yield useful information on aggregate net benefits under alternative policy scenarios. This information needs to be accompanied by a recognition of the distribution of the gains and losses, both across the current generation and between current and future generations. When the distributions of benefits and cost differ, the ethical issue of "who decides" becomes central to policy making. How much weight should we give to the well-being of future generations, as compared to that of current inhabitants of the planet? And how can we gauge the preferences of future generations in attempts to ascertain the gains or losses they might experience under different policies? Among the members of the current generation, do we give preference to particular members of society? Are sophisticated ecologists worth more votes than city dwellers who evidence neither knowledge of nor interest in "nature"? And what do we do about the enormous variation in risk aversion among our citizens? Do we owe extra deference to those who truly believe that we are threatening our very futures?

These questions reach to the very center of our views about the design of society and the appropriate relationship between state and citizen. The fact that they have no easy answers need not make us pessimistic about the prospects for sensible public policy. We can go a long way toward improving policy making simply by calling attention to the underlying philosophical questions, by developing empirical methods that generate better information about the gains and losses at stake under alternative public policies, and by developing channels for communicating this information to the general public.

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Notes

1. As another example, many people experienced a loss of satisfaction or well-being upon learning of the ecological damage resulting from the 1989 *Exxon Valdez* oil spill. The spill caused a lot of existence value.

2. These are direct, nonconsumptive use values in that the enjoyment of the wetland's flood-control or pest-control services does not use up the potential of the wetland to continue to provide these services.

3. A further, and related, issue is that preferences change. They may change for a given person over his or her lifetime, or from generation to generation. To impute values for future generations (such as the value that future generations might place on ecosystem functions), benefit-cost analysis must impute preferences to these generations. Clearly, this can only involve guesswork. Usually, benefit-cost analyses assume that future generations' preferences are similar to those of the current generation. Costanza, Norton, and Bishop (1995) indicate that preferences seem to evolve toward an increasing concern for sustainability. They consider the notion that this natural evolution of preferences ought to be accounted for in social decisions—that more evolved, developed preferences deserve greater weight in analyses of policy options.

4. Costanza, Norton, and Bishop (1995) and the chapter by Costanza and Folke in this volume consider this issue in some detail.

5. This organism must produce no use value, either directly or indirectly. Thus it must be something we don't enjoy eating (there is no consumptive use value) and something we don't enjoy observing (there is no nonconsumptive use value). In addition, the organism must not serve any positive ecosystem function (there must be no indirect use value). And it must be the case that we're certain that human's tastes and ecosystem function won't change to give rise to a future use value. To complete the picture, the organism must also have a zero existence value—humans must not enjoy contemplating this thing. Is there any real-world organism that fits this picture? Perhaps some lowly species of cockroach comes close. Whether it exactly fits the picture isn't important. The key point is that such a creature would be given virtually no value in a benefit-cost analysis. This means that if we are considering a development project that threatens its existence, this threat does not cause us (as utilitarians) to refrain from undertaking the project. As long as there are some benefits from the project and no other, "significant" form of life is put at risk, we would not prevent the loss of the particular species.

6. The animal rights position is sometimes extended to embrace other "rights" such as freedom, hence the occasionally observed bumper sticker, "Pet Breeders are Pimps

7. We thank Partha Dasgupta for pointing this idea out to us.

8. The Kantian emphasis on removing one's own identity from the consideration is inherent in John Rawls's notion of the original position. This notion gives rise to a Rawlsian conception of justice that is close in many respects to the Kantian conception. See Rawls 1973.

9. If the pest-control services provided by the ecosystem in question are perfect substitutes for the pest-control services offered by the alternative (e.g., synthetic substitutes), then the avoided expenditure is a fairly good measure of the pest-control benefit provided by the ecosystem. However, if the services are imperfect substitutes for one another, the avoided costs can significantly understate the value of pest-control services generated by ecosystems. For details on this issue see Freeman 1993.

10. There are exceptions. In some cases, there is one unit rate or price for up to a certain quantity of water, then another unit for consumption in excess of that quantity. This is a case in which two prices are charged, but it does not constitute a charge based on willingness to pay for each unit. That would require a multitude of prices

11. It may be noted that the total value or benefit from the water consumed (areas I and II) exceeds the sacrifice associated with paying for the water (area II). Thus there is a consumer surplus given by area I.

12. Markets tend to arise for goods or services that are excludable: the failure to pay for the good or service implies an inability to enjoy or consume the good. For nonconsumptive use values (like bird watching) it is difficult to establish a market because people cannot easily be excluded from enjoying the good or service.

13. For an illustration of the use of the travel-cost method, see Goulder and Kennedy 1995. For a detailed exposition, see Freeman 1993.

14. In contingent valuation assessments of value, interviewees are asked what they would be willing to pay in order to provide some real or hypothetical amenity.

15. The accounting here is perfectly analogous to the economic valuation of net economic output, which disregards the value of intermediate inputs, that is, inputs that are used up in the process of producing final goods such as consumer goods and capital goods.

16. Our attention to the possibilities for double-counting should not be misinterpreted. We do not mean to suggest that there is a general tendency to overvalue ecosystems services. To the contrary, these services are often undervalued because important direct use values and production services are ignored. But we do wish to indicate that if these types of services are valued correctly, there is no need to add further values attributed to indirect contributions by various life forms.

17. A collection of thoughtful examinations of the contingent valuation method is provided in the fall 1994 issue of the *Journal of Economic Perspectives*.

18. For a detailed discussion, see Bishop 1982. A closely related concept is that of the quasi-option value, which relates to the value of flexibility in situations involving the irreversibilities; on this see, for example, Dasgupta (1982, ch. 10). We follow general practice in subsuming option value is so closely connected with (potential) use that it should be placed in the use-value category.

19. For example, suppose a habitat is threatened with destruction. Suppose that, if the habitat is preserved, there is a 50 percent chance you would visit it, you would derive a use value of 10; if you didn't, you would enjoy no use value. In this case the expected value of the use value is 5. But you might be willing to pay, say, 7 to ensure the preservation of the habitat. If so, your option value is 2 (7–5).

20. The degree of historical disturbance, of course, is difficult to estimate. It is usually underestimated by human observers, whose decisions often are based on what they believe the ecosystem was like in their grandfather's time.

21. The same principle applies to other resources: as indicated earlier, the marginal value of water to households dwindles as the amount of water consumed increases. Working in the other direction, the marginal value rises the lower the amount of water available for consumption.

22. If, for example, the ones we view as most valuable did well in relatively small areas, we might argue for a patchwork of little parks; whereas, if the opposite were true, we would insist on large refuges.

23. Indeed, our analysis applies specifically to the simple case in which A is reduced by shrinkage from the outside edges. In many situations, the reduction occurs by fragmentation—a patch here, a patch there, leading to a checkerboard of "natural" and "modified" areas. The new habitats provided by "edge effects" can raise local biodiversity (at least transiently). In the longer run the area/diversity rule will apply over the entire region, but the value of species lost may differ. In recent studies of plant diversity in grassland patches, the first species lost are the most effective, narrow-niche competitors: fragmentation gives an advantage to those species adept at dispersal and at rapid colonization. (See, for example, the results discussed in the chapter by David Tilman in the volume.)

24. A technical discussion of these models is in Bergstrom and Stoll 1993.

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