CHAPTER 2

Interpreting and estimating the value of ecosystem services

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2.1 Introduction: why is valuing nature important?

Many of the critical ecosystem services generated by natural capital (such as pollination services, flood control, water filtration, and provision of habitat for biodiversity) are externalities—they are not given a price in markets. As a result, unfettered markets often lead to the compromising or collapse of ecosystems, much to the detriment of human welfare. Oftentimes society would benefit from greater protection of ecosystems and their services than results from unregulated markets.

Public policy has a crucial role to play in regulating or influencing markets so as to prevent them from producing unfortunate societal outcomes. Yet decisions about such public policies are often contentious. Agricultural interests will vie for greater ability to purchase wetlands and convert them through drainage to agricultural land. Urban developers will push to convert open space to new housing tracts.

Perhaps the most important basis for supporting a policy that would protect otherwise threatened ecosystem services is evidence that society gains more value from such protections than it gives up. Providing such evidence requires an understanding of the biophysical processes involved, that is, the various services offered by the ecosystem in question. It also requires an assessment of the benefits to well-being—or values to society—of these ecosystem services.

This chapter clarifies how such an assessment of ecosystem services can be made. It has two main components. One is to examine the philosophical basis of ecosystem service value. In considering this basis, we contrast competing approaches to value and bring out some ethical issues underlying the choice among different approaches.

The other component is to lay out various methods for measuring the values of ecosystem services, and to consider the strengths and limitations of these approaches. Quantitative assessments of ecosystem service value have become much more widespread in recent years. The expanding literature now includes estimates of the value of such ecosystem services as pollination, pest control, and water purification. These assessments are beginning to play a significant role in the formulation of land-use policies.

Setting out the values of ecosystem services to society provides a basis for making public policy decisions. However, it is not the only basis. As we discuss briefly below, it may make sense to consider as well whether a policy decision is consistent with preserving the intrinsic rights of the various organisms or ecosystems that might be affected by the decision. If intrinsic rights are involved, it is reasonable to restrict the set of serious alternatives to those that are consistent with these rights.

The chapter is organized as follows. Section 2.2 examines alternative philosophical foundations for valuing living things and ecosystems. It also considers how attention to intrinsic rights might supplement or even offer an alternative to a consideration of values. This philosophical discussion lays the groundwork for Section 2.3's examination of empirical valuation methods. In Section 2.4 we indicate some valuation problems that arise in a few specific real-world cases. Our final section draws conclusions.

2.2 Philosophical issues: values, rights, and decision-making

2.2.1 Competing philosophical approaches to value

2.2.1.1 The anthropocentric approach

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?

Among US policy analysts, the prevailing approach to value is anthropocentric. This approach claims that natural things (indeed, all things) have value to the extent that they confer satisfactions to humans. It stipulates that value is based on the ability to give *utility* (or well-being) to humans. Economists tend to support this viewpoint which, as we discuss below, is inherent in benefit–cost analysis.

At first blush, this anthropocentric approach might seem inconsistent with safeguarding the planet or protecting non-human forms of life. But the approach does not necessarily imply a ruthless exploitation of nature. On the contrary, it can be consistent with the fervent protection of non-human things, both individually and as collectivities. After all, we may feel that the protection of nature or particular non-human forms of life is important to our satisfaction or well-being, and thus we may place a high value on these forms. The anthropocentric approach does not rule out our making substantial sacrifices to protect and maintain other living things. However, it asserts that we should assign value (and therefore help other forms of life) *only insofar as we humans gain satisfaction or well-being 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).

Anthropocentric value can be categorized according to the way the satisfaction is generated. *Use value* refers to satisfaction that involves (directly or indirectly) a physical encountering with the object in question. There are *direct* use values (for example, the satisfaction from catching or eating trout) as well as *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). The anthropocentric approach does not restrict value to forms of nature that are consumed: there are both *consumptive* and *non-consumptive* use values. Examples of the former are the values that might be attached to ducks insofar as they provide food. Examples of the latter are the values ducks provide in the form of pleasure to bird-watchers.

Satisfactions also include non-use values: values that involve no 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 pure 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. As another example, many people experienced a loss of satisfaction or well-being simply from learning of the ecological damage resulting from the Bluewater Horizon oil spill in the Gulf of Mexico in 2010. This was a loss of existence value.

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. Their provision of habitat for migratory birds confers an indirect use value for people who enjoy watching them (non-consumptive) or hunting them (consumptive). Ecosystems also yield an existence value: wetlands, for example, provide such value to people who simply appreciate the fact that wetlands or their services exist.

The fundamental assertion of the anthropocentric approach is that the value of a given species or form of nature to an *individual* is entirely based on its ability to yield satisfaction to that person (directly or indirectly). Benefit–cost analysis invokes the anthropocentric approach, while introducing a further assumption—that the value to *society* of the natural thing is the sum of the values it confers to persons.

Benefit–cost analysis offers a rather convenient way of measuring the overall social values of alternative policies. Thus it provides a basis for making difficult policy decisions. It seeks to ascertain in monetary terms the gain or loss of satisfaction to different groups of humans 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 often does not differentiate between one person's valuation of a given species and another's—that is, each person's valuation receives the same weight as another's. Many times, no attempt is made to correct for differences in awareness, education, or "enlightenment" among individuals. The preferences of people who have no concern for future generations, or who have no sense of the ecological implications of their actions, are often counted equally with those of people who are more altruistic or who recognize more fully the fragility of ecosystems.

Such benefit–cost analyses are non-discriminating, perhaps to a fault. Consider the fact 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 certain 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 *et al.* (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. However, some benefit–cost analyses do in fact give special attention to the assessments offered by experts.

Many ecologists are uneasy with the tendency of benefit–cost assessments to give considerable weight to valuations made by relatively uninformed individuals. There is a basic appeal to the idea that the preferences of some individuals—particularly those who are better informed or have more relevant expertise—should count more. But it is very difficult to arrive at an objective standard for "relevant expertise." Philosophers offer varying viewpoints as to what's appropriate here (NRC 2004).

2.2.1.2 *A biocentric approach*

The biocentric approach offers another basis for value. It asserts that value consists in the ability to provide well-being or utility to humans *and* to other species. Under the anthropocentric approach, the well-being of other species counts only indirectly: such well-being is important only to the extent that it contributes to human well-being. In contrast, the biocentric approach gives weight directly to the well-being of other species. Thus, it allows for the possibility that another species will have value even if it does not confer satisfaction directly or indirectly to humans. This independent value is sometimes referred to as intrinsic value.

Defenders of the anthropocentric approach point out that since human beings are the dominant species on the planet; they are obliged to define ethical principles in terms of human wants and needs. However, biocentrists can counter by pointing out the following implication of anthropocentric logic. Suppose that representatives of another species should arrive from outer space, a species clearly superior to human beings in intelligence, perceptiveness, and technological know-how. To the extent that defenders of anthropocentrism have invoked the "dominant species" argument, consistency would require humans to allocate some decision-making authority to this other species, no matter whether humans like their decisions or not. Human well-being would count only insofar as it served the well-being of the superior species. This may seem troubling to many of us! What if the dominant species felt it was best to exterminate humans? This *reductio ad absurdum* argument has been invoked to support a biocentric approach that gives weight directly to a range of species.

While the biocentric approach may have some appeal, it is difficult to implement. As discussed below, "willingness to pay" offers a measure of the change in well-being to humans generated by a given policy change to protect nature or environmental quality. No comparable measure is currently available for assessing changes in satisfaction to other species or communities of them. Also, it is difficult to draw a clear line between biocentric value and certain anthropocentric values. When individuals call for a biocentric approach, they may actually be expressing the anthropocentric satisfaction they would gain if that approach were followed. For example, when someone calls for the preservation of a given habitat on the grounds that the species residing there has intrinsic value, that individual may really be revealing the (anthropocentric) existence value that the species provides. It thus becomes difficult to distinguish biocentric intrinsic value from existence value, which suggests that the biocentric approach may be superfluous.

2.2.2 Intrinsic rights: a further consideration

Under the value-based approaches just discussed, social decisions are to be made based on a comparison of values. If Policy A generates greater value than Policy B, then Policy A should be given preference over Policy B. Consider in particular the anthropocentric approach to value. If a given a species or other element of nature does not convey satisfaction to human beings directly or indirectly, then according to this approach it should be given no value. It must produce no use value, either directly or indirectly. Thus, it must be something we do not enjoy eating (there is no consumptive use value) and something we do not enjoy observing (there is no non-consumptive use value). In addition, the organism must not serve any positive ecosystem function (there must be no indirect use value). Also it must be the case that we are certain that humans' tastes and ecosystem function will not change to give rise to a future use value. To complete the picture, the organism must also have 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 is not 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 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 this particular species. If destroying the habitat and putting the area involved to an alternative use (e.g., residential housing) had any value at all, then according to the anthropocentric approach this is the best option for society.

Based on examples of this sort, some philosophers argue that the fate of other species becomes too precarious when it must depend on a link to human values or satisfactions. (See, for example, Skidmore 2001.) An *intrinsic rights* approach provides an entirely different basis for decision-making. When intrinsic rights are involved, then the appropriate social decision must respect those rights. Attention to intrinsic rights can in some cases complement the weighing of values. In such cases, policy makers would first restrict the set of options to be considered to those that respect intrinsic rights. Within this restricted set, policy makers would then choose the option that yielded the highest value.

In other cases, an attention to intrinsic rights is fully dispositive. This applies, for example, when any change to a given habitat would violate the claimed intrinsic rights of the species that currently reside there. In such circumstances, a defender of intrinsic rights could argue that the value-based approach is inappropriate: any comparison of benefits (values gained) and costs (values sacrificed) is not justified. Many analysts argue that species and natural communities have intrinsic rights to exist and prosper. They claim that, consequently, society should uphold these rights irrespective of the values gained (benefits) or sacrificed (costs) in the process.

Arguments for intrinsic rights are not entirely independent of references to well-being or satisfactions of other species. For example, in *Animal Liberation*, ethicist Peter Singer argues that non-human animals have the basic right to be spared of suffering deliberately caused by humans (Singer 1975). This argument is grounded in the notion that, like humans, other animals are sentient creatures, capable of experiencing pleasure and pain, and that there is something fundamentally wrong about causing pain to any creature. However, even though the call for intrinsic rights may be based on a concern for well-being, it proposes a very different basis for decision-making: the appropriate social decision must respect intrinsic rights. A policy that satisfies a benefit–cost test should be rejected if it violates intrinsic rights.

2.2.3 Public policy's inconsistent approach to decision-making

When should a value-based approach be employed, and when should attention to intrinsic rights supply the primary basis for decision-making? And which of the two approaches do societies in fact adopt? United States environmental policy adopts both the anthropocentric value approach (via benefit–cost analysis) and an intrinsic rights approach, and often acts inconsistently. Oftentimes the mandate for a particular environmental law will embrace the intrinsic rights approach, but actual implementation yields to a value-based approach.

A key example is the US Endangered Species Act, passed in 1973 after a previous Act was brought up to date and linked to the Convention on International Trade in Endangered Species. In addition to defining the status of "endangered" and "threatened," it made eligible for protection all plants and invertebrates, and prohibited the "taking" of all endangered animals. "Taking" included destruction of essential habitat. Federal agencies were required to use their authority to conserve listed species and were prohibited from undertaking actions that would jeopardize listed species or modify their critical habitats.

There is an assumption here that certain species under threat have an intrinsic right to exist. However, when it comes to actual implementation of the Endangered Species Act, the Congress only allocated funds sufficient for protecting a small fraction of species that may qualify for the designation of threatened or endangered. Based on threat criteria and the availability of appropriated funds, the Interior Department decided that some species are more worthy of protection than others. In effect, it adopted an anthropogenic, value-based approach, with the priorities reflecting the range of values that people place on different species. Charismatic megafauna like the wolf or the peregrine falcon get more protection than the white-footed mouse. The anthropocentric, value-based approach involved in implementation contradicts the intrinsic rights basis of the mandate declared by Congress.

One might be tempted to fault Congress for failing to allocate enough funds to protect all species. However, the allocation reflects the preferences of the broader public: to protect every species would require far more funds than the public generally wishes to devote to this purpose. People want species protection, but they also want funds for other things (e.g., education, defense, and their own consumption).

The case of the Endangered Species Act is not unusual. The US Clean Air offers another example. The mandate is broad: setting air quality standards tight enough to assure an "adequate margin of safety" to all individuals. There is no reference to a comparison of benefits and costs. Yet in the actual establishment of the standards, the EPA paid close attention to costs, and the ultimate standards imposed were not tight enough to prevent serious health problems or premature mortality to the most pollution-sensitive individuals.

In many instances there is a fundamental inconsistency between the stated objectives of environmental laws and the way the laws are implemented. Lawmakers and the public may experience rewards from establishing broad mandates that declare intrinsic rights. At the same time, they are free to implement the laws much more restrictively; so that people need not sacrifice as many other things as they would had they enforced intrinsic rights fully.

We do not suggest that society must choose between the universal application of an intrinsic rights approach and the across-the-board adoption of a value-based approach. In certain circumstances, it may be best to invoke intrinsic rights. In the Clean Water Act, Congress essentially found that the population of the United States had an intrinsic right to clean water. In other circumstances the appeals to intrinsic rates are fairly rare; for example, there are few claims that farmers enjoying the pollination services provided to agriculture by bees inhabiting nearby natural habitats have an intrinsic right to such services. Are these pollination services somewhat less "fundamental" than the various services offered by clean water? Does this explain the relative infrequency of claims that woodland-based pollination services are an intrinsic right?

Even if one adopts a value-based approach rather than an intrinsic rights approach in making policy decisions, this does not necessarily preclude invoking additional evaluation criteria in the decision process. Benefit–cost analysis considers the aggregate values gained (benefits) and values sacrificed (costs) of a given policy option. It usually does not focus on how the benefits and costs are distributed across members of society (rich vs. poor, current generations vs. future generations, etc.; see Chapter 16 for further discussion). The distribution of policy impacts is important and deserves attention as well. Other evaluative criteria (minimization of risk and political feasibility) can also be important. Thus, the results of a benefit–cost study may not be sufficient to settle the question of which policy is best.

2.3 Measuring ecosystem values

Although attention to values need not be the sole criterion for decision-making, we believe it is sufficiently important to justify a focus on how to measure various values. Here we describe central methods for measuring anthropocentric value. Considerable progress has been made over the years in developing such methods. However, the science is far from perfect. Controversies persist.

Ecosystem services are especially difficult to measure for the same reason that ecosystems 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 for providing support for reasonable public policies to protect important habitats. This makes it all the more important to determine the values of these services.

2.3.1 Willingness to pay

As indicated, under the anthropocentric approach 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 the measure of satisfaction.

It is important to be clear as to what is meant by "willingness to pay." It 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 represents a kind of psychological equivalence. Suppose a project would improve water quality an individual enjoys. That individual's willingness to pay is the income sacrifice that just brings the individual back to his or her original utility level after the improvement in water quality. More formally, the willingness to pay W for a given improvement to environmental quality Q is the value W that leads to the following equality:

$$U(Q + \Delta Q, Y - W) = U(Q, Y), \qquad (2.1)$$

where U stands for the individual's utility and Y is the individual's income. Willingness to pay expresses the maximum payment an individual would make that just compensates for (or undoes) the utility gain from the environmental improvement. It is the size of the payment that, if made, would keep the person's utility from changing. It is not necessarily what people say they are willing to pay. In some cases, markets indicate individuals' true willingness to pay as defined above: for example, the market price of a tomato might indicate what consumers are willing to pay at the margin for this product. But in other circumstances researchers need to rely on other, more indirect methods to fathom it.

2.3.2 Methods for measuring the values of ecosystem services

Above, we have distinguished various types of ecosystem service values. The ecosystem services themselves can be placed in various categories as well. As in the Millennium Ecosystem Assessment (2003), we will distinguish provisioning, regulating, and other services offered by ecosystems. Below, we consider the types of values associated with each of these major categories of service. Different types of

Table 2.1 Ecosystem services and valuation methods

Services	*Types of values offered*	*Valuation method*
Provisioning services		
Sustenance of plants and animals	Direct use values	
	—Consumptive	Direct valuations based on market prices
	—Non-consumptive	Indirect valuations (revealed expenditure methods, contingent valuation method)
	Indirect use values	(No valuations necessary if plants/animals with direct values are counted)
Regulating services		
Water filtration, flood control, pest control, pollination, climate stabilization	Direct and indirect use values	Estimation of service's contribution to profit (holding all else constant)
Other services		
Generation of spiritual, esthetic, and cultural satisfaction	Existence value	Indirect valuations (contingent valuation method)
	Direct, non-consumptive use value	Indirect valuations (revealed expenditure methods, contingent valuation method)
Recreational services	Non-consumptive direct use value (e.g., from bird-watching)	Indirect valuations (revealed expenditure methods, contingent valuation method)
Generation of option value*	Option value	Empirical assessments of individual risk-aversion

* Option value represents a component of the overall value offered by a potential future ecosystem service, supplementing other values attributed to this potential service. See discussion text.

valuation techniques are called for, depending on the category of service involved. Table 2.1 shows the relationships between service types and valuation methods.

2.3.2.1 *Valuing the provisioning services of ecosystems*

As suggested by Table 2.1, a general type of service provided by ecosystems is the sustenance of plants and animals. In choosing a method for valuing this type of service, it helps to distinguish plants and animals 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 either enjoy non-consumptively 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, one can 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 those fish that have a market value.

There is an important difference between the *marginal* and *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 2.1 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 do not 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, utilities charge households a given price per unit of water, regardless of how much they consume.

In Figure 2.1, 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). In these circumstances, the price is an expression of the marginal willingness to pay, or marginal value. (In the example of Figure 2.1, the user would demand 400 cubic feet of water per month at this price.)

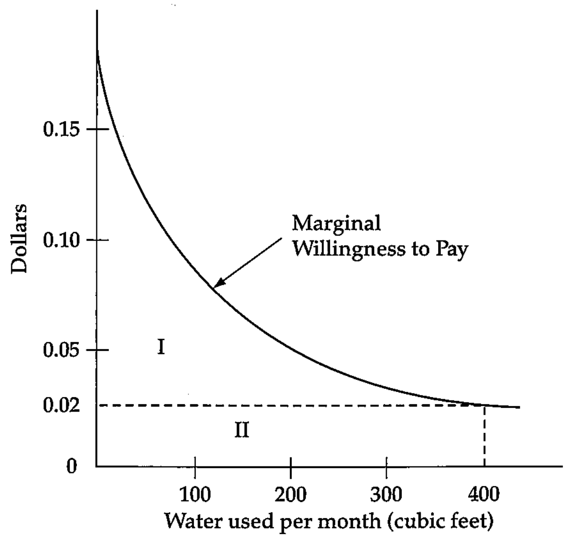

Figure 2.1 Relationship between water use and marginal willingness to pay. From Daily (1997) reproduced by permission of Island Press, Washington, DC.

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 information on the entire marginal willingness-to-pay schedule (or demand curve), not just the price paid. 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 2.1. 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, non-consumptive use values. Within the category of direct use values from living things maintained by ecosystems, we have another case to consider: the case where the life forms are used *non-consumptively*. For such uses, the relevant markets do not usually arise, and thus it is not possible to gauge values directly by observing market prices. 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 non-consumptive use values (like birdwatching), it is difficult to establish a market because people cannot easily be excluded from enjoying the good or service. In these cases, it is necessary to apply more inferential methods to ascertain the relevant values.

Revealed expenditure methods represent a broad category of inferential approaches (NRC 2004). Revealed expenditure methods have been applied to ascertain some of the values provided by parks, lakes, and rivers—or, equivalently, the costs that results from the loss of these elements of nature. Here we describe one of the first and simplest revealed expenditure methods: the travel cost method (Freeman 1993). In recent years several more general and sophisticated approaches have tended to supplant the travel cost method, but the basic logic of the newer approaches is the same as that of the travel cost method.

Non-consumptive uses are not directly bought or sold in markets; prices are not usually charged for their use. 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 a family visit to Yosemite National Park is much greater than the daily 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.

It is also possible to employ survey methods, such as the contingent valuation method, to determine how much value people place on the non-consumptive uses. In contingent valuation assessments of value, interviewees are asked what they would be willing to pay in order to prevent some real or hypothetical amenity. For an exposition of this approach, see Mitchell and Carson (1989). 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 contain many living organisms that support other, often "higher" forms of life that provide direct or indirect value to humans. It could be assumed that the value of ecosystem services should include the values of the services provided by these life forms. In fact, there is no need to include the values of these services in an accounting of the overall value of an ecosystem! These values are already captured in the values attached to the life forms that humans enjoy. 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 anthropocentric 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. To add their indirect use values to the direct use values would be double counting. The accounting here is perfectly analogous to the economic valuation of net economic output, which disregards the value of intermediate inputs, that is, inputs used up in the process of producing final goods such as consumer goods and capital goods.

2.3.2.2 *Valuing the regulating services of ecosystems*

Table 2.1 lists four examples of production inputs from ecosystems: water filtration, flood control, pollination, and climate stabilization. These services are inputs to the sustained production of agricultural products in the sense that it would be difficult to maintain agricultural production without relatively pure water, flood control, pest control, or a stable climate.

An appropriate measure of the value of productive inputs is the additional economic income or profit that they provide, holding everything else constant. Thus, for example, the value of pest control services provided by ecosystems is their contribution to profits. To assess these values, agronomists and other researchers develop models in which the profitability of various agricultural products is assessed in the presence and absence of ecosystem-provided pest control, a key production input. The difference is the value of the pest control services. Similarly, one can gauge the value of flood control services by comparing profitability in the presence and absence of such services. A favorable climate can be considered a productive input. Numerous studies have aimed to assess the damage from climate change to agriculture by comparing yields and agricultural profits under current climate with those that would apply after predicted future climate change (e.g. Mendelsohn *et al.* 1994; Schlenker *et al.* 2005). The damage from a changed climate is equivalent to the monetized benefit or value from avoiding this change.

Pollination services are another example of an important production input. In the Central Valley of northern California, various specialty crops, including melons, nuts, and tree fruits, depend upon the pollination services supplied by wild bees whose population is maintained by breeding sites in nearby "natural" areas such as undeveloped forestland. The value of these pollination services is the additional profit generated by the populations of wild bees.

These pollination services have declined over time as larger and larger proportions of the region have been developed for agricultural purposes. One can only assume that at some point in the developmental history of this unusually productive agriculture, wild insect populations alone were sufficient to guarantee some base level of pollination services and thus guarantee yields adequate to keep the farmers in business. Since that is clearly no longer the case, farmers now have to substitute a costly alternative—pollination services from the bees supplied by commercial beekeepers. This is now an economically significant activity in these regions. In this example, the avoided cost is the difference in cost between the case where farmers enjoyed free pollination services from wild bees and the case where they must pay for the services of bees husbanded by commercial beekeepers.

It is sometimes suggested that one can place a value on production inputs by examining what costs or expenditures agricultural producers manage to avoid by having these inputs and thus not having to substitute other inputs for them. For example, where ecosystems provide effective pest control, farmers can avoid having to pay for alternative pest control methods such as the use of synthetic pesticides. In fact, avoided cost is not a theoretically valid indicator of value. To see this, consider the following (extreme) situation. Suppose it were infinitely costly for farmers to find an alternative to wetlands in providing flood control. If avoided cost indicated value, then the value of the wetlands' flood control services would be considered infinite. In fact, although the loss of flood control services would cause a significant loss of profit to farmers, the loss would not be infinite.

Although avoided cost is not a measure of value, it is still important information. It indicates the *net* advantage of having access to the productive input provided by ecosystems, as opposed to having to achieve the same input through an alternative. It provides a rationale for preserving the ecosystem service. For example, when the New York City Water District struggled with how to preserve water quality in the Catskills, it determined that it was far less costly to do so by restoring the ecosystems surrounding the city's upstate reservoirs rather than by constructing a new water treatment plant. The very high avoided cost motivated the decision to pursue ecosystem-generated water quality control (Daily and Ellison 2002).

2.3.2.3 *Valuing ecosystem services offering non-use values*

Other important services include the generation of spiritual, esthetic, and cultural satisfaction, the provision of recreational services, and the generation of option value. Recreational services provide a non-consumptive direct use value. For example, a National Park offers opportunities for hiking, swimming, and bird-watching. Park visitors engaging in these activities physically encounter the ecosystems involved (implying a use value), but (hopefully) do not use up the hiking trails, lakes, or birds in the process of enjoying them. The non-consumptive use values from these activities can be estimated using the methods described for such values under Section 2.1 above.

Ecosystems also provide services with values other than use values—values that do not derive from a physical encounter with the item of nature in question. The values provided here 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. While much of our enjoyment of biodiversity involves use value—that is, it derives from a physical encountering with various plants and animals—we also derive satisfaction from simply recognizing that these forms of nature exist. Thus existence value is an important element of the value people attach to nature or the functioning of ecosystems. It may reflect the spiritual, esthetic, or cultural satisfaction we obtain when we contemplate the diversity, beauty, complexity, or power of nature.

Survey approaches such as contingent valuation assessments may be the only way of ascertaining existence value, since actual market and non-market behavior gives little hint of its magnitude. As mentioned, survey approaches are controversial. Yet, when it comes to existence values, surveys may be the only way of ascertaining 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. However, we can point out what seems to be the key underlying question: Is the information obtained from surveys, however imperfect, better than no information at all?

Option value. As developed in the economics literature (e.g., Bishop 1982), 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. Suppose, for example, a habitat is threatened with destruction. And suppose that, if the habitat is preserved, there is a 50% chance you would visit it, and a 50% chance you would not. If you were to 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). 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 0.

We follow general practice in subsuming option value under the general category of non-use values. However, the case can be made that option value is so closely connected with (potential) use that it should be placed in the use-value category.

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 Cameron (1992).

2.3.3 Marginal vs. total value

In discussions of ecosystems, one often might have in mind their *total* value. However, 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. The value we place on a given amount of area lost to other uses depends on the area of this system. 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 through conversion 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 0 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 labeled "1" in Figure 2.2. 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

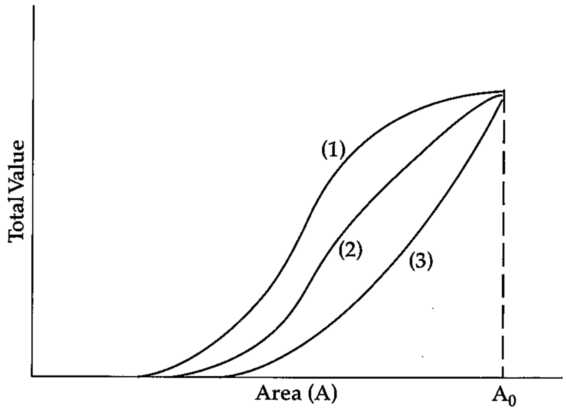

Figure 2.2 Relationship between loss of habitat area and loss of ecosystem value viewed through three different dimensions: ecosystem scale effect (1), diversity effect (2), and species effects (3). A_0 denotes the initial area of the ecosystem and the curves show how value declines as that initial area is reduced. In all cases there is some threshold area that is too small to support any value.

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 non-linear 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-quarters of the area is lost. The effect on marginal value is to exaggerate the loss of ecosystem value as A is reduced. 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 labeled "2" in Figure 2.2. As indicated by the differences between paths 1 and 2, this intensifies the marginal loss of value from a given reduction in A.

A third effect 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. Of course we regularly justify large expenditures to save some species (e.g., the Black 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, "collection" in the form of listing is a motive with powerful economic consequences. Many birdwatchers 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 (often called "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. 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.

Obviously the number of possible criteria is large enough to prohibit development of a precise relationship among area, species loss, and value. However, 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). 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 (Tilman 1997).

Clearly some of these relationships are uncertain, and the exercise could be applied to real natural areas only after substantial research. However, 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.

2.4 Some case studies

2.4.1 Wetlands in the United States

Wetlands provide important ecosystem services, including flood control, water purification, and provision of habitat for numerous species. The values of these and many other services are very difficult to quantify. Perhaps even more important to the measurement challenge is the complexity of the network that links wetlands to groundwater and thence to streams and lakes and other "navigable waters."

In theory, society could decide on the desired level of protection of wetlands and their ecosystem services by calculating the values of the numerous ecosystem services, determining the extent of wetlands that maximizes the net benefits from these services minus the opportunity cost to society, and then implementing laws that protect just this amount of wetland. In fact, wetland protection has largely ignored valuation. In the United States, the law does not invoke an explicit comparison of benefits or costs as a basis for the protection offered; this may partly reflect the measurement difficulties just mentioned. Indeed, US law does not even acknowledge cost as a consideration in determining the extent to which wetlands or their services are to be protected. Rather, these broad ecosystem services are treated like a public "right," something to be safeguarded irrespective of the cost of protection.

That right is protected by two public agencies under several laws. The Clean Water Act, administered through the Environmental Protection Agency, has several sections devoted to wetlands protection, and these refer to the general set of the "navigable waters" referred to above. The Food Quality Protection Act has a provision colloquially known as the "swampbuster" clause that prohibits the drainage or alteration of wetlands for farming purposes. That provision is administered by the US Army Corps of Engineers. These two agencies are required to issue permits when a landowner undertakes measures that would contribute fill or pollutants to wetlands that lay within the drainage system of the owner's property. In an important wetlands case called Borden Ranch, the Supreme Court ruled that a California farmer could not be issued a permit for a technique of plowing called "deep ripping," on the grounds that it violated provisions of the Clean Water Act. In short, the Court found that the connection of the groundwater under the farmer's plow to the navigable waters could not be disturbed or interrupted.

In a much later case, the Supreme Court again split about a proposal to fill some wetlands near Lake St. Claire in Michigan. This particular wetland was some distance from the lake, which clearly fit any ordinary definition of a "navigable water." Once again, the issue rested on the question on whether a wetland that is distant from clearly navigable surface waters is nevertheless entitled to protection. The responsibility, again shared by the Environmental Protection Agency and the Army Corps of Engineers, was complicated once again by the ambiguity of the Clean Water Act's language. A four-four split on the Court was eventually decided by Justice Kennedy, who wrote in his opinion that the Clean Water Act intended to apply its provisions to the nation's waters generally, not restricted to surface waters. However, he also argued that the case should be ultimately decided scientifically by the federal agencies responsible for applying the protection.

This case exemplifies the difficulty of interpreting Congressional intention. It also highlights the difficulties of measuring ecosystem values in a network of rivers, lakes, wetlands, and groundwater that is diffuse, interconnected, and complex. In the face of such difficulties, policy-makers might prefer simply to establish a broad right to protection, rather than aim to compare values gained and values sacrificed under alternative levels of protection.

2.4.2 Vegetation and coastal protection

About one-third of the world's population lives in coastal areas or small islands, and they are at risk from buffeting and damage from storms and extreme weather events, like the hurricanes that regularly visit Caribbean and Gulf of Mexico coastlines and the recent tsunamis that swept across Indonesia and coastlines in the south Pacific and Indian Oceans. In coastal areas, societies often must make difficult choices between economic development activities and risk-reducing conservation measures. Decision-makers practicing ecosystem-based management are required to address both of these competing needs in a way that balances the welfare of residents. In order to do this, they must be able to measure the values associated with the ecosystem services provided by conservation measures (see Box 2.1).

This problem has been analyzed by Barbier and a team of fifteen authors (Barbier *et al.* 2008; Box 17.1 in this book) from various international institutions. Their analysis examined the trade-offs between conservation of coastal mangroves in Thailand and the conversion of mangrove lands to shrimp aquaculture. The authors of the study started with the reasonable assumption that the buffering capacity of the mangroves would depend on their area. On measuring the ocean wave attenuation at various distances inland from the shore, they showed that the storm buffering service provided by the mangroves was in fact nonlinear owing to the shape of the declining wave attenuation.

Using the estimated nonlinear relationship, Barbier *et al.* (2008) found that the policy that provides the greatest overall benefit to the local population is one that prohibits shrimp aquaculture in much, but not all, of the area in question, and thus reserves some area for shrimp farming ponds. Some economic benefits to aquaculture investors were retained. (In contrast, the assumption of a linear relationship between area conserved and buffering capacity would have suggested that prohibition of all shrimp farming was optimal.) This significant effort at ecosystem service valuation provided the basis for a solution favoring both stakeholders—both conservation groups and investors in aquaculture.

2.4.3 The Galapagos Islands

A third example, international in character, is provided by the Galapagos Islands. This archipelago, located 600 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 1 000 000 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.

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

Box 2.1 Designing coastal protection based on the valuation of natural coastal ecosystems

R. K. Turner

Depending on the precise definition used, coastal zones occupy around 20% of the earth's surface but host more than 45% of the global population and 75% of the world's megacities (> 10 million inhabitants). The zone's underpinning coastal ecosystems—coral reefs, mangroves, salt marshes and other wetlands, sea grasses and seaweed beds, beaches and sand dunes, estuaries and lagoons, forests and grasslands—necessary to sustain human occupation, are highly diverse, productive, and biocomplex. These ecosystems provide a range of services, such as, nutrient and sediment storage, water flow regulation and quality control, and storm and erosion buffering, summarized in Table 2.A.1 (Crossland *et al.* 2005). Coastal zones are impacted by dynamic environmental changes that occur both ways across the land–ocean boundary. The natural and anthropogenic drivers of change (including climate change) cause impacts ranging from erosion, siltation, eutrophication, and over-fishing to expansion of the built environment, and inundation due to sea level rise. All coastal zone natural capital assets have suffered significant losses over the past three decades (e.g., 50% of marshes lost or degraded, 35% of mangroves, and 30% of reefs) (MA 2005). The consequences for services and economic value of this loss at the margin are considerable but have yet to be properly recognized and more precisely quantified and evaluated (Daily 1997; Turner *et al.* 2003; Barbier *et al.* 2008).

The ecosystem services (storm buffering) valuation example illustrated below is drawn from the European coastal zone context and in particular the east coast of England, which is one of UK's most "at-risk" areas from climate change and other impacts. The European coastal zone is around 600 000 km^2 (within 10 km of shorelines) and home to 80 million people and 280 major cities. The annual value of coastal tourism alone is 75 billion EUR (*The Changing Faces of Europe's Coastal Areas* (2006): http://reports.eea.europa.eu/eea_report_2006_6/en). In response to the multitude of pressures bearing down on coastal areas, coastal protection and sea defense policy in the UK and Europe is being reappraised and reoriented toward a more flexible and adaptive strategy anchored to an ecosystem services approach and decision support system (Turner and Daily 2008; Turner *et al.* 2008).

The Millennium Ecosystem Assessment (MA 2005) states that "ecosystem services are the benefits people obtain from ecosystems" and subdivides them into supporting, regulating, provisioning, and cultural services. Building on this platform, it can be argued that when the focus is on national accounting (Boyd and Banzhaf 2007), or landscape management (Wallace 2007), or in our example valuation of service benefits (Fisher *et al.* 2008), further elaboration is required for actual choice-making involving human welfare. A key step is the separation of ecosystem processes and functions into intermediate and final services, with the latter yielding welfare benefits (see Table 2.A.1).

The generation of services and the enjoyment of benefits is spatially conditioned and therefore a key step in any evaluation process must be the setting of the ecosystems in their appropriate contexts. The valuation process must also be restricted to marginal gains/losses and should avoid double counting and should note possible threshold effects and any nonlinearity between change in ecosystem services and habitat variables such as size of area. Failure to account for these limitations will lead to under-/overestimated economic values and unnecessarily polarized cost–benefit decision choices (Turner 2007; Barbier *et al.* 2008).

Traditional sea defense and coastal protection strategies in Europe have sought to provide rigid engineered "hold-the-line" protection for people, property, and other assets against the vagaries of dynamic coastal environments. Given the growing awareness of the consequences of climate change a policy switch is taking place toward a "coping strategy" based on a mixed approach with engineered protection focused on high commercial value areas, and the rest of the coastline left to adapt to change more naturally. Measures such as "managed realignment," which involves the deliberate breaching of engineered defenses to allow coastal migration and the creation of extended intertidal marshes and mudbanks, at the expense of agricultural land, are now being tested. Testing sites have been carefully chosen to minimize impacts on existing people, property, and environmental assets that enjoy engineered protection from the sea. The question is do they represent cost-effective strategies for society?

Managed realignment schemes yield benefits in terms of ecosystem services. They generate carbon storage benefits (via saltmarsh creation) that can be valued in terms of the damage costs avoided per tonne of CO_2. The sites also serve to improve fisheries' productivity via nursery areas and this gain can be valued via market prices for commercial species. There are also general recreation and amenity benefits related to walking, bird-watching, and other recreational activities, as well as biodiversity

continues

Box 2.1 *continued*

Table 2.A.1 Coastal zone services* and benefits

Intermediate services	Final services	Benefits	Econ valuation methods
e.g:	e.g:	e.g:	e.g:
• Geodynamics: sediment and nutrient cycling and transport	• Creation of beaches, dunes, estuaries	• Flood/storm buffering	• Market prices/damage cost avoided
• Primary production	• Sediment, nutrients, contaminants retention/storage	• Shoreline stabilization/erosion control	• Production function market prices
• Water cycling	• Biomass export	• Fish production	• Survey-based contingent valuation/ choice experimentation
• Climate mitigation	• Maintenance of fish nurseries and refuges	• Biodiversity maintenance	• Damage cost avoided
	• Regulation of water flow and quality	• Carbon storage	• Travel cost, hedonic pricing, survey based CV or CE
	• Carbon sequestration	• Amenity and recreation activities	• Survey based CV or CE
	• Recreation and amenity	• Cultural/heritage conservation	
	• Cultural heritage		

*. European coastal areas including estuaries and saltmarshes.

maintenance and existence value benefits. An indication of the composite value of some of these amenity and related benefits can be estimated by transferring benefits data from the published literature if the spatial and other contextual variables are similar, or, more properly, by conducting site-specific contingent valuation/choice experiment studies to estimate willingness-to-pay values. Finally, the maintenance costs of the existing engineered defenses will be saved as realignment schemes are implemented. On the costs side, secondary defenses may be required further inland and there are opportunity costs associated with any agricultural land that is sacrificed as the old defenses are breached.

Cost–benefit analysis of managed realignment schemes took the following approach.

The "status quo" existing protection system is appraised as follows:

$$PV_t^{sq} = -\sum_{t=0}^{T} \frac{1}{(1+r)^t} [(l^{sq} C^{sq}_{m,t})], \qquad (2.A.1)$$

where PV_t^{sq} is the present value of total costs of current defenses (£million), r is the discount rate, l^{sq} is the length of defenses and $C^{sq}_{m,t}$ is the maintenance costs (£km^{-1} yr^{-1}).

The costs and benefits of managed realignment is given as follows:

$$PV_t^{mr} = \sum_{t=0}^{T} \frac{1}{(1+r)^t} \begin{bmatrix} l^{mr}(C^{mr}_{k,t} + C^{mr}_{m,t}) \\ -(a_t^{mr} L^{agr}_{agr,t}) - (a_h^{mr} B_{e,t}) \end{bmatrix} \qquad (2.A.2)$$

where PV_t^{mr} is the present value of managed realignment schemes (£million), l^{mr} is the length of managed realignment (km), $C^{mr}_{k,t}$ is the capital cost of realignment, $C^{mr}_{m,t}$ is the maintenance costs, a_t^{mr} is the agricultural land lost, $L^{agr}_{agr,t}$ is the forgone agricultural land value, a_h^{mr} is the area of intertidal habitat created, and $B_{e,t}$ is the ecosystem value benefits (£ha^{-1}).

Finally, the overall CBA result is found as:

$$NPV_t^{mr} = (PV_t^{sq} - PV_t^{mr}) \qquad (2.A.3)$$

where NPV_t^{mr} is the net present value of managed realignment compared to hold-the-line for a given stretch of coastline at time t (£million).

The analysis was carried out with data from a number of different estuaries, and indicates that appropriately sited schemes do represent gains in economic efficiency if declining discount rates are applied over a 100-year time horizon (see Turner *et al.* 2007 and Luisetti *et al.* 2008 for further details).

"ecotourism," the Galapagos now attract over 150 000 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 larger islands, whereas recent reports suggest that the less occupied islands are doing better than they did 25 years ago. A significant fishery for sea cucumbers, a delicacy prized in Asian and French cuisine, developed in the 1990s and still exists, although the catch is declining. Illegal long-line shark fishing continues to create a problem. Not only do these activities threaten the intertidal fauna, they pose significant risks to the terrestrial ecosystem through the introduction of "exotic" species and destructive camping on some islands. Fortunately, the Park's protection system is much more effective now than in the past, and efforts to eradicate goats, cats, and other invasive species are continuing, most effectively on the four smaller, less-inhabited islands.

Arrayed against these direct, consumptive use values are two other values. The first is the direct, non-consumptive use value from ecotourism, which brings significant revenue. A sample calculation of this value would be that the average visitor (a week on a boat is a typical excursion) spends well more than \$10 000. If the visitor is from the USA, 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 \$15 000 would be a reasonable figure for the "overseas" visitor: if half were Ecuadorian nationals and half from elsewhere, the value of the industry would approach one billion annually.

Local residents, however, would make quite a different calculation. The 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. However, 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 islanders more local autonomy (and relaxed many of the ecological protections) there was another takeover. The tense historical 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 underway on the Galapagos. The large number of endemic species found there, and the recentness 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.

2.5 Conclusions

Society must often make difficult choices about how and how much to protect natural capital and the ecosystem services such capital generates. Perhaps the most important basis for supporting a policy that would protect otherwise threatened ecosystem services is evidence that society gains more value from such protections than it gives up. This requires an assessment of the values that human beings place on such services—values that often are not expressed in markets.

In this chapter we have aimed to clarify the philosophical underpinnings of these values. The most prevalent and perhaps most workable philosophical basis is anthropocentric—value consists in the ability to provide satisfaction or well-being to humans. Although anthropocentric, this approach is consistent with society's making great sacrifices in order to protect valued species and ecosystem services.

We have also indicated the various types of value generated by ecosystem services, and laid out principal empirical methods for measuring these values. None of the empirical approaches is perfect; the uncertainties in measurement can be vast. However, even with the imperfections the methods generally are good enough to provide a basis for public policies to protect ecosystem services. The InVEST models described throughout this book represent the frontier in valuing ecosystem services. These models exemplify the substantial progress of the past decade in researchers' abilities to depict the gains and losses associated with the protection of a wide range of ecosystems and their services.

Many of the benefits from ecosystem services are not captured by unregulated markets. An individual's private gain from protecting ecosystem services falls short of the value of such protection to society. Hence private markets tend to fail to provide sufficient protection, and there is an important role for public policy to protect these services.

Two types of public policy could stem from the information offered by the InVEST models and other empirical studies. One is the introduction of quantitative restrictions that restrict the way natural capital gets used or converted and thereby protect the services such capital generates. Limits on wetland conversion, for example, help protect the various ecosystem services (flood control, water purification, and habitat provision) that wetlands offer. Another approach is the introduction of prices for ecosystem services—prices that the market would not generate on its own. For example, a tax on wetland conversion would serve to reduce the rate of such conversion. According to economic theory, the tax rate should be set equal to the marginal value of the ecosystem services provided by the natural capital or ecosystem in question. This tax rate would lead to a lowered frequency of conversion that maximizes the net gain to society from intact wetlands—the benefit to agriculture minus the lost ecosystem service value. In many cases, the tax would prevent any conversion from taking place. These are instances in which the marginal value of the existing ecosystem services exceeds the marginal value generated by any conversion or alternative use of the natural capital.

Although our discussion acknowledges a key role for benefit–cost analysis in the valuation of ecosystem services, we would emphasize that such analysis is usually not sufficient for deciding policy. Benefit–cost analyses yield useful information on aggregate net benefits under alternative policy scenarios, but usually ignore issues of fairness or distribution. These analyses need to be accompanied by an assessment of the distribution of the gains and losses, both across the current generation and between current and future generations. If a proposed policy clearly would lead to serious inequities, it is reasonable to reject the policy, even if it passes a benefit–cost test.

The topics of valuation and policy choice raise a number of imponderables. In arriving at the social value of a given option, should every person's willingness to pay count equally, or should some members be given more weight than others? Are the preferences of sophisticated ecologists worth more than those of city-dwellers who evidence neither knowledge of nor interest in "nature"? How much weight should we give to the preferences or well-being of future generations, as compared to that of current inhabitants of the planet? How can we gauge the preferences of future generations in attempts to ascertain the gains or losses they might experience under different policies?

The fact that these questions 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 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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