The Value of Biodiversity

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Glossary

Accounting prices Accounting or shadow prices are measures of the social opportunity cost of resource use. The accounting price of a resource is the increase in social well-being that would be enjoyed if a unit more of the resource was made available at no cost.
Ecological functions The combinations of ecosystem component and ecosystem processes that enable ecosystems to support primary production and to maintain populations of species.
Economic valuation The methods used to identify society's willingness to pay for the nonmarket benefits obtained from ecosystems.
Ecosystem processes Biogeochemical cycling, including primary production (photosynthesis), nutrient and water cycling, and materials decomposition, and the flow, storage, and transformation of materials and energy through an ecosystem, including through food web processes such as pollination, predation, and parasitism.
Ecosystem services The benefits (i.e., goods and services) people obtain from ecosystems.
Production functions The combinations of capital stocks, material inputs, and technology to produce valued outputs.
Regulating services The benefits obtained from the regulation of ecosystem processes, including flood protection, climate regulation, human disease regulation, water purification, air quality maintenance, pollination, and pest control.
Resilience The capacity of an ecosystem to maintain functionality when subject to stress or shock.
Social discount rates The rate at which society is willing to substitute future consumption for present consumption.

Biological Resources and Biodiversity

Traditionally economists have approached the use people make of biodiversity through the individual species those people exploit: the fish that are caught in freshwater and marine capture fisheries; the trees that are harvested from either managed or natural forests; the livestock raised for meat, hides, and other animal products in pastoral systems; the food and cash crops produced in arable systems; or the medicinal plants extracted from forests, grasslands, and wetlands. To do this, they have relied on models of the growth of individual populations in which the functioning of the rest of the ecosystem is taken to be exogenous. Well known illustrations of this viewpoint include the use of the logistic function to chart the time path of the population of a single species of fish, the study of predator–prey interactions by means of variants of the Lotka–Volterra equations, the estimation of growth in biomass of a species of trees at a given site, and so on. A prominent concern has been to determine the rates at which a single resource would be harvested in different institutional settings. Thus, not only have socially optimum harvest rates been analyzed (Clark, 1976), but economists have also determined harvest rates when harvesters have free or open access to the resource (e.g., Gordon, 1954; Dasgupta, 1982).

This approach has given an understanding of the effect on harvest rates of harvesting costs, the rates at which harvesters discount future costs and benefits, the productivity of the resource *in situ*, the "worth" of the harvest to harvesters, and the property-rights regime in which the harvesting is done. It has also given an understanding of the implications of changes in either the abundance or the intrinsic growth rates of exploited species. The approach has offered much less insight, however, into the economic effects of changes in the various measures of species richness (alpha, beta, or gamma diversity) or in the diversity within and between functional groups of species.

An alternative approach has focused on the ecosystem that is exploited when a particular species is harvested. In this approach, the resource switches from the individual species to the biotic and abiotic components of the system that regulate its growth. Instead of treating the role of the ecosystem as constant, as it is in the carrying capacity of a Lotka–Volterra population growth model, the approach allows investigation of the interactions between ecosystem components induced by human exploitation. There the focus is on system feedbacks through, for example, the energy flows involved in trophic interactions and the distribution and flows of biochemical substances in soils and water bodies, and of gases and particulates in the atmosphere. Where the state variables in the traditional approach are population "quantities" (e.g., herd sizes, expressed in numbers, or biomass units), in the ecosystem approach they may also include quality indices for air, soil, or water. The motivation is to understand the way in which the exploitation of ecosystems alters their usefulness to humankind by changing the biotic and abiotic processes that underlie various ecosystem functions, and hence the services they yield.

Economic studies of global warming, eutrophication of lakes, the management of rangelands, purification of water in watersheds, and the pollution of estuaries are examples of such endeavor (Mäler, 1974; Ludwig *et al.*, 2003; Perrings and Walker, 2005; Chichilnisky and Heal, 1998). Such studies have provided valuable insights into the effects on ecosystems of the character of economic activities, as driven by technology, costs and revenues, discount rates, and the property-rights regime that governs access to the ecosystem. In this approach the focus of study is the ecosystem as a renewable natural resource system: a mix of functional groups of species that support a range of ecosystem processes, and hence a range of ecosystem services. Many ecosystems are deliberately "simplified" (through the removal of pests, pathogens, predators, or competitors) in order to increase their value for particular purposes. The best examples of this are to be found in agriculture, aquaculture, forestry, and urban systems. The biodiversity in such systems is managed to enhance production of particular services. But even the most simplified ecosystems – monocultures in agriculture and forestry and cities – are still ecosystems. The services they provide depend on the composition of species they contain, the energy and nutrient flows within the system, and the interactions between species.

This article asks how the second of these approaches informs the understanding of the value that biodiversity has to individual farmers, foresters, and fishers; how that differs from the value it has to wider society; and why the difference matters. All ecosystems generate an array of services that affect peoples' well-being in different ways. All ecosystems are also subject to property rights or access rules that affect the rights and responsibilities of users, and that determine which services they take into account and which they do not. Biodiversity change is a problem for society if it has either significant social impacts that are ignored by users or unacceptable distributional consequences. This article considers both the inefficiency that results when biodiversity use involves external costs and the inequity that results when biodiversity change bears disproportionately on the poor.

Biodiversity and Ecosystem Services

The Millennium Ecosystem Assessment (MA) defined ecosystem services to include the full array of benefits people obtain from ecosystems distinguishing four broad benefit streams: the provisioning, cultural, regulating, and supporting services (Millennium Ecosystem Assessment, 2005). The MA provisioning services describe the processes that yield foods, fibers, fuels, water, biochemicals, medicines, pharmaceuticals, and genetic resources. Many of these products are subject to well-defined property rights and are priced in the market. Others are not. The MA cultural services comprise a set of largely nonconsumptive uses of the environment including both the well-established market benefits of recreation and tourism and the spiritual, religious, esthetic, and inspirational well-being that people derive from the "natural" world, along with the value to science of the opportunity to study that world. The MA regulating services limit the effect of stresses and shocks to the system; include erosion control or soil stabilization, water purification, and waste treatment; and the regulation of air quality, climate, hydrological flows, pests, disease, and natural hazards. Finally, the MA supporting services comprise the ecosystem processes that underpin production of all these services, including soil formation, photosynthesis, primary production, nutrient, carbon, and water cycling.

The value of biodiversity derives from the value of the final goods and services it produces. To estimate this value, one needs to understand the "production functions" that link biodiversity, ecosystem functions, ecosystem services, and the goods and services that enter into final demand. Recent ecological research on the linkages between biodiversity and ecological functioning has improved the understanding of ecological processes involved in the production of a number of ecosystem services (Loreau *et al*., 2002; Naeem *et al*., 2009). It has also improved the understanding of the role played by the diversity of species in the production of these services. Species are related through functional traits that make them more or less perfect substitutes in executing particular ecological functions. Individual species near perfect functional substitutes for other species if they share a full set of traits with those other species. Conversely, they are "singular" if they possess a unique set of traits (Naeem, 1998). Species are also related through ecological interactions – trophic relationships, competition, parasitism, facilitation and so on – that make them more or less complementary in executing ecological functions (Thebault and Loreau, 2006).

The value of individual species depends both on their substitutability/complementarity in the functioning of ecological systems and on the way in that functioning has been affected by land use. The simplification of agroecosystems to privilege particular crops or livestock strains necessarily affects the array of services that system delivers, partly because the number of functions performed reduces with the number of species (Hector and Bagchi, 2007) and partly because each species in a system generally performs multiple functions. Ecosystems are systems of "joint production." Individual systems generate multiple services. It follows that part of the cost of simplification is the ecosystem services foregone as a result. So, for example, modern agriculture has focused on increased yields per hectare, but has compromised a range of other ecosystem services including water supply, water quality, habitat provision, pollination, and soil erosion control (Millennium Ecosystem Assessment, 2005). It has also affected the capacity of the modified system to function over a range of environmental conditions – that is, its stability or resilience.

There is a vast and confusing literature on the relationship between diversity (or complexity) and stability or resilience (see, e.g., May, 1972; Pimm, 1984; Schulze and Mooney, 1993; Tilman *et al.*, 1996; Kinzig *et al.*, 2001), with many studies appearing to contradict each other. This is in part due to definitional ambiguities, in part due to analysis at different levels of organization, and in part due to differing results in theoretical versus experimental versus observational approaches. Most studies suggest that diversity (in the form of a greater number of species) enhances stability for community-level properties, though these results do not necessarily extend to the population level, where diversity can enhance, erode, or have little impact on stability (Tilman *et al.*, 1996; Mccann, 2000; Muradian, 2001). Most of these studies, though, address stability and resilience in a particular sense – how diversity affects the ability of a system to return to a previous (possibly equilibrium) state (Gunderson, 2000). They say less about resilience in the alternative sense (sensu Holling) – the contributions that diversity might play in diminishing the probability that a system might access entirely new system configurations, or ''flipping'' into a new basin of attraction. (A system could be highly resilient in the second sense but not in the first – highly dynamic within a given basin of attraction, never returning to any given state for very long, but never leaving that basin of attraction, either.)

The relationship between diversity and resilience in this second sense is less clear. The vulnerability of low-diversity systems, such as agricultural monocultures, and the persistence of high-diversity systems, such as tropical forests, have often been cited as evidence of a relationship between diversity and resilience. Goodman (1975) takes these examples to task. In addition, there are equally compelling examples of low diversity systems that appear to be highly resilient – rice paddies have persisted in the same places for hundreds of years (Geertz, 1963; Lansing, 1991), and desertified and biologically depauperate regions are highly resistant to restoration efforts. (This latter example illustrates that resilience is not always a good thing.) These contradictions indicate the hazards of associating system-level properties with system-level outcomes. Degradation of soils, for instance, can cause both low diversity and high resilience, without there being a direct causal link between the two. Similar observations have been made about the role of productivity in high-diversity systems – are high-diversity systems productive, or are highly productive systems diverse?

Ecological theory suggests that increasing species number or complexity within a community should increase the number of equilibrium points or stable states (May, 1977; Deangelis and Goldstein, 1978). All else being equal, that would seem to imply more thresholds and lower resilience. Of course, all else would not be equal – it is entirely possible that increased diversity and complexity increase both the number of alternative states that can be accessed and the tenacity with which the system remains in any particular state. But both the theoretical and the empirical evidence for a strong relationship between diversity and resilience (sensu Holling), when correctly sorted, are still ambiguous.

Diversity can, of course, play a role in enhancing the functioning of ecosystems. More functional guilds will mean provisioning of a greater number of ecosystem services. Those services may or may not be resilient. What will enhance resilience (continued delivery) of a particular service is the diversity of species within the functional guild responsible for that service (e.g., within the guild of nitrogen fixers). If each species within a guild has a differential response to the environment, this would allow sustained function under environmental variability. This is known as functional redundancy.

Why should one necessarily expect species within a functional guild to have differential responses to the environment? Ecological theory suggests, in fact, that this is exactly what one should expect. In spite of perceptions of ''nature red in tooth and claw,'' evolution by natural selection, in fact, serves to diversify the niches of similar species and reduce competition among them (hence the diversification of beak sizes and shapes among the finches on the Galapagos Islands, each suited to different seed-gathering strategies). In most ecological communities it was also observed that just a few species make up the bulk of the biomass, with many more occurring in low abundance. The hypothesis is that both the breadth of services provided and their resilience are facilitated by these patterns. In particular, one might expect the dominant species to be functionally dissimilar (providing different services, but reducing direct competition through functional dissimilarity), with the lower-abundance species being functional redundants of the dominant species (but with different responses to environmental variability or stressors). Walker *et al.*, 1999 demonstrated just these outcomes in a grassland ecosystem subjected to grazing pressure . In the absence of grazing, dominant species were functionally dissimilar, whereas minor species were functionally similar to a dominant species. When dominant species declined or were eliminated by grazing, their functionally similar minor counterparts increased in abundance, thus maintaining certain functions within the disturbed grassland.

Although much remains to be done to understand the joint production of ecosystem services in particular ecosystems, the interactions between services, and the impact of changes in the relative abundance of species within and between functional groups, this article can now identify at least some of the consequences that biodiversity change for ancillary ecosystem services. The canonical bioeconomic models developed by Clark to understand the exploitation of marine mammals and fisheries (Clark and Munro, 1979) clarified the conditions required for the optimal extraction of particular populations, establishing the capital theoretical basis for exploiting biological stocks. The extension of such work to cover the exploitation of multiple species has made it possible to understand the effect of species interactions on co-produced ecosystem services (Tilman *et al.*, 2005; Eichner and Pethig, 2005; Brock and Xepapadeas, 2002; Perrings and Walker, 1997, 2005). This, in turn, has made it possible to track the physical effect of biodiversity change that lies outside the market, and to begin to estimate its social cost. This article first considers the generic issues involved in the diagnosis and cure of market failures, and then identifies their implications for the economics of biodiversity change.

Market Failure

A central concern among economists concerned with the environment has been to devise ways by which it would be possible to ascertain the ''value'' of natural resources and the services they provide. This is because for many natural resources and the services they support, markets simply do not exist. The interactions between biodiversity, ecological functioning, ecosystem services, and the production of the things that people care about most directly are not signaled by market prices. In some cases, markets do not exist because the costs of negotiation and monitoring are too high. One class of examples is provided by economic activities affected by ecological interactions involving long geographical distances (e.g., the effects of upland deforestation on downstream activities hundreds of miles away); another, by large temporal distances (e.g., the effect of carbon emission on climate in the distant future, in a world where forward markets are nonexistent because future generations are not present today to negotiate with the present generation). Then there are cases (e.g., the atmosphere and the open seas) where the nature of the resource makes private property rights impractical and so keeps markets from existing; whereas in others, ill-specified or unprotected property rights prevent their existence, or make markets function wrongly even when they do exist. In short, market failures mean that people are not confronted by the real cost of their behavior (but *see* Market Failure). This gives rise to the phenomenon of externalities (i.e., exchanges among people that take place without their consent). (The early literature on ecological economics identified market failure as the underlying cause of environmental problems. (Pigou, 1920; Lindahl, 1958; Arrow, 1971; Meade, 1973; Mäler, 1974; Baumol and Oates, 1975; Dasgupta and Heal, 1979)

Problems arising from an absence of forward markets for ''transactions'' between the present generation and those to appear in the distant future are no doubt ameliorated by the fact that humans care about their children's well-being and know that they, in turn, will care for theirs, in an intergenerational sequence. This means, by recursion, that even if people do not care directly about the well-being of their distant descendants, they do care about them indirectly. However, there is a distinct possibility that one's implicit concern for the distant future via such recursion is inadequate, due, say, to institutional failure in other spheres of economic activity. This is why economists have argued that market rates of interest do not reflect socially desirable discount rates (Lind, 1982; Arrow *et al.*, 1996; Portney and Weyant, 1999). In short, market failure involves misallocation of resources not only in the present but also across time.

In each of these cases, the market prices of goods and services fail to reflect their social worth; typically, they are less than their social worth. In economics, the social worth of goods and services is called accounting prices (sometimes, shadow prices). The accounting price of a resource is the increase in social well-being that would be enjoyed if a unit more of the resource were made available at no cost. So a resource's accounting price is the difference between its market price and the tax (or subsidy) that ought to be imposed on it. Needless to say, accounting prices reflect social objectives, ecological and technological constraints, and the extent to which resources are available.

It should be noted that externalities do not create market distortions; they are a form of market distortion. The presence of externalities leads to a wedge between market prices and accounting prices. Generally speaking, laissez-faire economies are not much good at producing publicly observable signals of environmental scarcities. To illustrate, if there were free access to a resource base, the market price of the resource, *in situ*, would be zero. However, if the resource were in limited supply, its accounting price would be positive. So, there is a directional bias in environmental externalities: market failure typically results in an excessive use of the natural resource base, not an insufficient one.

One way to improve matters would be to impose regulations on resource users, for example, restrictions on effluent discharges, quota on fish harvests, and bans on logging. Another would be to introduce a system of taxes, often called Pigovian taxes (in honor of Pigou, who first discussed the difference between private and social costs in the context of environmental pollution (Pigou, 1920)). Pollution charges, charges on the amount of fish harvested, and stumpage fees are examples. The idea underlying Pigovian taxes is to bring market prices (inclusive of taxes) in line with accounting prices. Each of the two schemes, quotas and taxes, has its advantages and disadvantages, some of the differences between the two becoming salient once it is recognized not only that ecological processes are stochastic, but also resource users and government agencies do not have the same information about local ecology, say, for example, the cost of waste disposal (Weitzman, 1974; Dasgupta, 1982). This article cannot discuss this in details, but three points are worth noting. First, the two schemes are distributionally not equivalent: under a quota system resource rents are captured by harvesters and polluters, whereas under a tax system they are collected by the tax authority. Second, the imposition of Pigovian taxes provides greater incentives to resource users to explore resource-saving technological improvements. This is because if the users are taxed, they pay more for the resource than they would have if they had been issued quotas instead. Third, environmental taxes, when properly designed, remove market distortions. In addition, there is a presumption that tax revenues, thus collected, would enable the government to reduce distortionary taxes (e.g., taxes on earned income). There is, thus, a presumption that Pigovian taxes yield a ''double dividend'' (Bovenberg and Goulder, 1996; Goulder, 1995) (but see Bohm, 1996), a rhetorical phrase that has been much used to persuade governments to impose ''green'' taxes. Matters of public finance have been a recurrent theme in ecological economics since the 1970s (see, especially, Carraro and Siniscalco, 1996; Baumol and Oates, 1975; Cropper and Oates, 1992). A hybrid policy instrument, which involves the government issuing a fixed number of transferable licenses, combines some of the features of quotas and Pigovian taxes. For example, the scheme resembles quotas, in that, resource rents are not captured by government, and it resembles Pigovian taxes, in that at the margin license, holders pay the accounting price of the resource for its use. (See Tietenberg (1990) for a good discussion of transferable licenses, both in theory and in practice.)

Institutional Failures and Poverty: Global Versus Local Environmental Problems

Aside from market failure, another long-recognized source of difficulty is what may be called government failure. For example, Binswanger argued that, in Brazil, the exemption from taxation of virtually all agricultural income (allied to the fact that logging was regarded as proof of land occupancy) provided strong incentives to the rich to acquire forest land and to then deforest it (Binswanger, 1991). He argued that the subsidy the government thereby provided to the private sector was so large, that a reduction in deforestation (via a removal of subsidies) was in Brazil's interests, not merely in the interests of the rest of the world. That is, he concluded that Brazil had much to gain from reducing then current rates of deforestation. (In a wider discussion of the conversion of forests into ranches in the Amazon basin Schneider (1995) has demonstrated that the construction of roads through the forests has also been a potent force.) Since that time the government of Brazil has reached a similar conclusion. It has changed the incentives it offers to landowners and has entered into negotiations with other countries that are potential beneficiaries of reduced deforestation in Brazil. At the heart of this decision is the recognition that there are interactions between ecosystem services generated by tropical forests. Land clearance for agriculture has consequences for biodiversity, water supplies, microclimatic regulation, and macroclimatic regulation (via carbon sequestration). Indeed, Brazil is now one of the main players in the development of the Reducing Emissions from Deforestation and Forest Degradation in Developing Countries (REDD) scheme. The scheme aims to create a financial value for the carbon stored in forests, offering incentives for developing countries to reduce emissions from forested lands (Malhi *et al.*, 2008; Phelps *et al.*, 2010; Arriagada and Perrings, 2011).

This said, it is important to note that the causes of environmental problems are not limited to market and government failure; problems also arise because such micro-institutions as the household can function badly. In poor communities, for example, men typically have the bulk of the political voice. One should then expect public investment in, say, resource regeneration to be guided by male preferences, not female needs. On matters of afforestation in the drylands, for instance, one should expect women to favor planting for fuelwood and men for fruit trees, because it is the women and children who collect fuelwood, whereas men control cash income (and fruit can be sold in the market). This explains why, even as the sources of fuelwood continue to recede, fruit trees are often planted (Dasgupta, 1993b).

That political instability (at the extreme, civil war) is a direct cause of resource degradation is obvious. What is not obvious is that it is a hidden cause as well. Political instability creates uncertainty in property rights. In its presence, people are reluctant to make the investments that are necessary for environmental protection and improvement: the expected returns on such forms of investment are low. In a study comprising 120 countries, Deacon has offered statistical evidence of a positive link between political instability and forest depletion (Deacon, 1994).

Taken together, these examples reflect the environmental consequences of institutional failure. They have a wide reach, and in recent years, they have often been discussed within the context of the thesis that environmental degradation, such as eroding soil, receding forests, and vanishing water supplies, is a cause of accentuated poverty among the rural poor in poor countries. There is truth in this. But there is also accumulated evidence that poverty itself can be a cause of environmental degradation (Dasgupta, 1993a, 2001). This reverse causality occurs because some natural resources (e.g., ponds and rivers) are essential for survival in normal times, whereas others (e.g., forest products) are also a source of supplementary income in times of acute economic stress. Under changing circumstances (e.g., economic development in urban centers), social norms that previously had maintained long-term economic relationships among members of a community tend to break down. Some (e.g., the able bodied and mobile) gain, whereas others (e.g., women, the old, and the very young) lose and become poorer. In extreme cases the breakdown of social norms also means that local resources that earlier were subject to communitarian regulations become "open access," with all the attendant consequences.

These links between rural poverty and the state of the local natural resource base in poor countries offer a possible pathway along which poverty, resource degradation, and even high fertility feed on one another in a synergistic manner over time (Dasgupta, 1993a, 1995, 2000). Experience in sub-Saharan Africa and Pakistan is not inconsistent with this (Cleaver and Schreiber, 1994; Filmer and Pritchett, 1996). Indeed, an erosion of the local natural-resource base can make certain categories of people destitute even while the economy's gross national product (GNP) increases. The thought that entire populations can always be relied upon to make the shift from resource-based, subsistence existence to a high-income, industrial one is belied both by evidence and by theory.

These two causes of resource degradation, namely, institutional failure and poverty, pull in different directions and are together not unrelated to an intellectual tension between the concerns people share about global warming and acid rain, which sweep across regions, nations and continents, and those matters (such as, for e.g., the decline in firewood or water sources) which are specific to the needs and concerns of the poor in as small a group as a village community. Environmental problems present themselves differently to different people. In part, it is a reflection of the tension this article has just noted and is a source of misunderstanding of people's attitudes. Some people, for example, identify environmental problems with poverty and unprecedented population growth in the South, whereas others identify them with wealth and unprecedented expenditure patterns in the North. Even though debates between the two groups often become shrill, each vision is in part correct. There is no single environmental problem and, so, no single valuation problem; rather, there is a large collection of them. Thus, growth in industrial waste and resource use has been allied to increased economic activity; in the former Socialist bloc neither preventive nor curative measures have kept pace with the production of waste. Moreover, the scale of the human enterprise, both by virtue of unprecedented increases in the size of the world's population and by the extent of economic activity, has so stretched the capabilities of ecosystems that humankind can today rightly be characterized as Earth's dominant species (Vitousek *et al.*, 1997). These observations loom large not only in ecological economics, but also in the more general writings of environmentalists and in the professional writings of ecologists in the West.

However, economic growth itself has brought with it improvements in the quality of a number of natural resources. The large-scale availability of potable water, and the increased protection of human populations against both water- and airborne diseases in industrial countries, have in great measure come in the wake of growth in national income these countries have enjoyed over the past 200 years or so. Moreover, the physical environment inside the home has improved beyond measure with economic growth. For example, cooking in South Asia continues to be a central route to respiratory illnesses among women. Such positive links between economic growth and environmental quality often go unnoted by environmentalists in the North. This lacuna may be yet another reflection of the fact that it is all too easy to overlook the enormous heterogeneity of Earth's natural resource base, ranging as it does from the atmosphere, oceans, and landscapes to water holes, grazing fields, and sources of fuelwood. Both this heterogeneity and the diversity of the human condition across the globe constantly need to be kept in mind in discussions of the value of biodiversity in different locations.

Valuing Resources and Evaluating Projects

Since institutional failures abound in human dealings with Earth, the commercial profitability of economic activities, say, of investment projects (projects for short), is frequently not an adequate measure of their social worth. So recourse should be taken to social cost–benefit analysis, the purpose of which is to estimate the impact of projects on human well-being, now and in the future. Notice that, if undertaken, a project would be a perturbation to the economy. So, for example, a project consisting of the construction of a dam would be a perturbation to an economy without the dam. The economic forecast *sans* the project can be thought of as the *status quo*.

Analyzing the consequences of a project would involve estimating the need for labor, intermediate products, raw materials, and output, as well as predicting the ecological effects of the project. These consequences need to be specified for each future period (*see* Discontinuous Value Functions for a formalization). Since there is never sufficient knowledge to make precise estimates of the consequences, project evaluators should quantify estimates of the uncertainties, preferably in terms of probabilities. This means that, in general, project designers ought to model the integrated ecological and economic system. Unhappily, in practice this is infrequently done.

To arrive at a good estimate of a project's social benefits and costs, one should in principle value each and every commodity involved in it. The procedure devised by economists is to select some readily measurable bundle of goods ordinarily consumed and to define the "value" of any other commodity as the amount of the bundle society would be willing to give up for it. This is a workable way for estimating the commodity's accounting price. The net social benefit of a project in any given period of its life is obtained by multiplying the project's inputs and outputs in that period by their corresponding accounting prices and adding them (outputs of "goods" are taken to be positive; output of "bads" and inputs are taken to be negative). Using a suitable discount rate (often called the social discount rate), the net social benefits yielded by a project in each period are added. Projects that yield a positive present discounted value of net social benefits are recommended for acceptance; those that yield a negative present discounted value of net social benefits are rejected. The theory of social cost-benefit analysis has been developed by economists over the past 50 years, and is now, to all intents and purposes, complete (Dasgupta *et al.*, 1972; Little and Mirrlees, 1974; Dasgupta and Mäler, 2000). Daily *et al.* (1999) provides a nontechnical account of the role of social cost–benefit analysis in environmental management.

A prior exercise (i.e., prior to conducting social cost–benefit analysis) is to estimate accounting prices. A great deal of work in ecological economics has been directed at discovering methods for estimating accounting prices of natural resources. It is as well to remember that the kinds of resources this article mentions are on occasion of direct use in consumption (as with fisheries), on occasion indirectly, as inputs in production (as with plankton, which serves as food for fish), and sometimes in both (as with drinking and irrigation water). The value may be utilitarian (e.g., as a source of food, or as a keystone species), it may be esthetic (e.g., a shoreline), or it may be intrinsic; indeed, it may be all these things.

Economists have devised various methods for estimating accounting prices. As would be expected, the prices of some natural resources are easier to estimate than those of others. There are now standard techniques for determining the accounting prices of irrigation water, fisheries, timber, and agricultural soil (Anderson, 1987; Repetto *et al.*, 1989; Solorzano *et al.*, 1991; Vincent and Ali, 1997). They involve estimating the resource's use-value. For example, the value of a piece of agricultural land would be the present discounted value of the flow of net profits it is expected to generate from cultivation, minus the environmental damage caused by the pesticides and herbicides to be used. Such an approach can be also used for estimating losses associated with water logging and overgrazing. Reductions in air- or water-borne pollution can be valued in terms of improvements in health (e.g., reductions in the number of days people would be expected to be ill; see, e.g., (World Bank, 1992)). Other techniques have been devised for valuing "amenities," such as places of scenic beauty. One popular method involves asking people hypothetical questions concerning their willingness to pay for preserving the amenity (this is called the "contingent-valuation method," or CVM, for short); another involves estimating from sample surveys the distribution of costs visitors from different locations have incurred to view the site (this is called the "travel-cost method"). A third involves inferring how much people are willing to pay for enjoying the amenity (e.g., clean air) from the commercial value of land at sites that offer the amenity (this is called the "hedonic price" of land). (See Mitchell and Carson, (1989) and Freeman (2003).) Good descriptions of these methods and their application to valuing ecosystem services can be found in Freeman (2003).

Two things that complicate the valuation of biodiversity change are uncertainty in the future use-values of these resources and irreversibility when they are lost. The twin presence of uncertainty and irreversibility implies that preservation of the stock has a value in addition to its current use-value, namely, the value of extending society's set of future options. (For a review of the challenges posed by these two issues, see Perrings and Brock (2009).) Future options have an additional worth because with the passage of time, more information is expected to be forthcoming about the resource's use-value. This additional worth is often called an option value (Arrow and Fisher, 1974; Henry, 1974; Fisher and Hanemann, 1986). The accounting price of a resource is, at the very least, the sum of its direct use-value and its option value.

These techniques enable one to estimate the use-value of a given resource. As it happens, the resource's accounting price may well exceed this. Why? The reason is that there may be additional values "embodied" in a species, an assemblage of species, or a landscape. This is sometimes captured in the idea that living resources have an "intrinsic" worth as living resources. It would be absurd to suppose that the value of a blue whale is embodied entirely in its flesh and oil or that the value of game in Kenyan safari parks is simply the present-discounted value of the flow of tourists' willingness-to-pay to view them. The idea of intrinsic worth of living things is inherent not only in traditional religious systems of ethics but also in modern ethical theories. Economists tend to think of this as the nonuse value of resources. The question is not so much whether living creatures have nonuse value (intrinsic worth), but rather of ways of assessing this worth. Since it is very difficult to get a quantitative handle on intrinsic worth, the correct thing to do may often be to take note of it, keep an eye on it, and call attention to it in public debate if the stock is threatened with destruction.

It can be concluded that the social worth of natural resources can be decomposed into three parts: their use value, their option value, and their nonuse value. The components appear in different proportions, depending on the resource. For example, oil and natural gas would not be thought to have very low nonuse value relative to their use value. In contrast, endangered primates would be thought to have very limited use value relative to their nonuse value. And so on.

For both use and nonuse values, wherever species are identified in the functions that describe ecosystem processes and functions, one can also identify their marginal impact on the output of things people care about. The marginal value of an incremental change in the abundance of any species derives from the value of the services it yields. Derivation of that value requires specification of the functions that connect species to directly valued goods or services (Barbier, 2007) or that connect ecosystems and the services they produce (Barbier, 2008). Whether one uses market prices or revealed or stated preference methods to obtain an estimate of willingness to pay for the directly valued goods or services is more or less irrelevant. What is important is that some form of "production function" approach is needed to estimate the value of a marginal change in the biodiversity that supports the directly valued good or service. A good example of the combination of methods is Allen and Loomis, (2006). They combine willingness-to-pay estimates obtained using stated preference methods for the conservation of directly valued higher-trophic-level species with ecological data on trophic relationships to derive estimates of implicit willingness to pay for the conservation of species lower down the food chain.

It is as well to emphasize that the purpose of estimating environmental accounting prices is not to value the entire environment; rather, it is to evaluate the benefits and costs associated with changes made to the environment due to human activities. Prices, whether actual or accounting, have significance only when there are potential exchanges from which choices have to be made (e.g., when one has to choose among alternative investment projects). Thus, the statement that a particular act of investment can be expected to degrade the environment by, say, 1 million dollars annually has meaning, because it says, among other things, that if the investment were not to be undertaken, humanity would enjoy an additional 1 million dollars of annual benefits in the form of environmental services. The statement also has operational significance: the estimate could (and should) be used for calculating the rate of return attributable to the investment in question.

Biodiversity: Necessity or Luxury?

The starting point is that the value of biodiversity derives from its role in the production of things that people care about. This extends from clean water, through the production of foods, fuels, fibers, and pharmaceuticals, all the way to the spiritual, cultural, or scientific value of intact ecosystems or totemic species. It follows that the value people implicitly place on biodiversity change depends on the value of the ecosystem services they want. For those who are most directly threatened by pests or diseases, the most important consideration may be pest or pathogen elimination. For those engaged in the production of foods, fuels, or fibers, the most important consideration may be the elimination of crop predators (e.g., insect pests), competitors (e.g., weed species), and diseases (e.g., blights and rusts). For those engaged in conservation or ecotourism the most important consideration may be the preservation of native species and the elimination of exotic species. For those engaged in managing the impacts of trade or travel, the most important consideration might be preventing movement of potentially invasive plants, insects, or diseases. Since the services that people care about are likely to differ depending on geographical location, culture, income, and so on, the value they attach to changes in biodiversity is also likely to differ. And since many services are increased by focusing on a relatively small number of species, people may be expected to put a positive value on reducing species richness as often as they put a positive value on enhancing it.

This helps putting into perspective the common view that the rich value biodiversity more than the poor: that biodiversity conservation is a "luxury" good. The natural resource base is far from being a luxury. As the source of a multitude of ecosystem services necessary to the lives and livelihoods of the majority of people on the planet, the natural-resource base is a necessity. The way that the poorest of people depend on the resource base may differ from the way that the rich do so, but both do depend on it. (The way that the poor depend on the resource base is explored in greater detail in Dasgupta (2001, 1993a) and Dasgupta and Mäler (1997).

This perspective, of viewing natural resources as luxuries, first found expression in World Bank (1992), where it was suggested that there is an empirical relationship between gross domestic product (GDP) per head and concentrations of industrial pollutants. Based on the historical experience of OECD countries, it was argued in the document that when GDP per head is low, concentrations of atmospheric pollutants (e.g., sulfur dioxide (SO_2)) increase as GDP per head increases, but when GDP per head is high, concentrations decrease as GDP per head increases further. In short, it was found that the functional relationship between GDP per head and concentrations of industrial pollutants has an inverted-U shape (**Figure 1**). Among economists this relationship has been christened the "environmental Kuznets curve." (It is a misnomer. The original Kuznets curve, which was an inverted U, related income inequality to real national income per head on the basis of historical cross-country evidence.)

Panayotou had reported the inverted-U shape in cross-country data on GDP per head and deforestation and emissions of SO_2, nitrogen oxides (NO_x), and particulate matters (Panayotou, 1992). The logic underlying the environmental

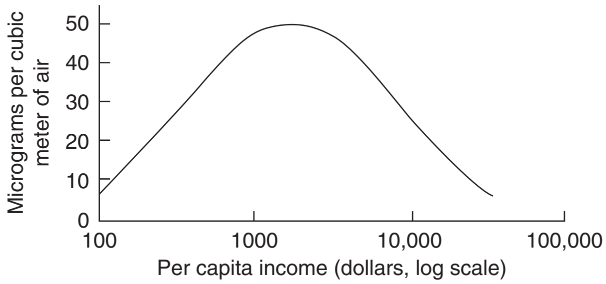

Figure 1 Urban concentrations of sulfur dioxide. Reprinted from figure 4 in *World Development Report* (1992), with permission from World Bank.

Kuznets curve is that resource degradation is reversible: degrade all you want now, you can always recover the stock later, because Earth can be relied upon to rejuvenate it. It is known, however, that this presumption does not apply to biodiversity loss. The presence of ecological thresholds implies that damage to ecosystems can be irreversible. As an overarching metaphor for ''trade-offs'' between manufactured wealth and resource degradation, the environmental Kuznets curve has to be rejected. (For more extensive discussions of the environmental Kuznets curve, see Arrow *et al.* (1995), and the responses it elicited in symposia built round the article in *Ecological Economics*, 1995, 15(1); *Ecological Applications*, 1996, 6(1); and *Environment and Development Economics*, 1996, 1(1); and see the special issue of *Environment and Development*, 1997, 2(4).)

The relation between income and biodiversity loss has been approached in three ways in this literature. In one approach, deforestation has been used as a proxy for biodiversity loss (using the species area relationship to explain the link between changes in forest area and biodiversity loss). The evidence for any well-defined relation between income and biodiversity loss using this metric is extremely weak (Dietz and Adger, 2003; Mills and Waite, 2009). A second approach uses the National Biodiversity Risk Assessment Index (NABRAI) developed by Reyers *et al.* (1998). This too has failed to find evidence for a statistically significant relation between biodiversity loss and income (Mozumder *et al.*, 2006). A third approach has focused on the direct measures of threat contained in the IUCN's Red List, and finds a statistically significant relation between the natural log of *per capita* income and the number of threatened species that is linear in the case of plants; ''U'' shaped in the case of amphibians, reptiles, fishes and invertebrates, and inverted ''U'' shaped in the case of birds (Naidoo and Adamowicz, 2001). More recently, Perrings and Halkos have reconsidered this latter approach using the most recent IUCN data. Controlling for climate, population density and protected areas, they find a strongly quadratic relation between *per capita* Gross National Income and the number of threatened species. They conclude that the interest people have in the abundance of other species is indeed sensitive to income. In poor countries where agriculture is expanding most rapidly at the extensive margin (by bringing new land into cultivation), people have a direct interest in habitat conversion, and in the elimination of weeds, pests and predators. They may also have an interest in conserving their biological heritage, and many poor countries have committed a significant share of their total land area as protected areas, but their willingness to pay for conservation is strictly limited by their income – their ability to pay (Perrings and Halkos, 2010).

Substitution Possibilities

This section and the following one are based on Dasgupta *et al.* (2000a, b) and Dasgupta and Maler (2004). In fact the belief that constraints arising from resource depletion can be overcome as countries become wealthier in terms of their manufactured- and human-capital assets is frequently based on a subtler thought than the one that underlies the environmental Kuznets curve. The belief is based on possibilities of substitution.

Resource constraints facing an economy can be eased by four types of substitution. First, there can be substitution of one thing for another in consumption (nylon and rayon cloth substituting for cotton and wool, pulses substituting for meat, and so forth). Second, manufactured capital can substitute for labor and natural resources in production (the wheel and double-glazing are two extreme examples). Third, novel production techniques can substitute for old ones. For example, the discovery of effective ways to replace the piston by the steam turbine (i.e., converting from reciprocating to rotary motion) was introduced into power plants and ships a little more than a hundred years ago. The innovation was an enormous energy saver in engines. Fourth, and for this article most importantly, natural resources themselves can substitute for one another. This involves the thought that, as each resource (e.g., each species) is depleted, there are close substitutes lying in wait, either at the same site or elsewhere. If this were true, then even as constraints increasingly bite on any one resource base, humanity would be able move to other resource bases, either at the same site or elsewhere. The enormous additions to the sources of industrial energy that have been realized (successively human and animal power, wind, timber, coal, oil and natural gas, and most recently, nuclear) are a prime historical illustration of this possibility. (But these shifts have not been without unanticipated collective costs. Global warming, associated with the burning of fossil fuels (an ''externality''), did not feature in economic computations in earlier decades. (See Dasgupta (1993b) for a less coarse partition of substitution possibilities than the four-way classification.)

Humans have been ''substituting'' one thing for another since time immemorial. Even the conversion of forests into agricultural land in England in the middle ages was a form of substitution: large ecosystems were transformed to produce more food. However, the pace and scale of substitution in recent centuries have been unprecedented. Landes has argued that substitution made the Industrial Revolution in England in the eighteenth century (Landes, 1998). The extraordinary economic progress experienced in Western Europe and North America since then (during the past two centuries GDP per head in Western Europe has increased some twelve-fold), and in East Asia more recently, has also been a consequence of substitution. Spatial dispersion of ecosystems has enabled this to happen. The ecological transformation of rural England in the middle ages presumably reduced the nation's biodiversity, but it increased income without any direct effect on global productivity.

But that was then and this article discusses the here and now. A question currently much debated is whether it is possible for the scale of human activity to be increased substantially beyond what it is today, without placing undue stress on the major ecosystems that remain. In any event, the cost of substituting manufactured capital for natural resources can be high. Low-cost substitutes could turn out to be not so low cost if accounting prices were used in the costing, not market prices. Even when accounting prices are not used, degrading natural capital and substituting it with manufactured capital can be uneconomic. Studies of the well-known Catskill Watershed ecosystem in New York State demonstrated the overwhelming economic advantages of conservation of the watershed over construction of water-purification plants. Chichilnisky and Heal, for example, showed that independent of the other services the Catskill Watershed provides, and ignoring the annual running costs of 300 million US dollars for a filtration plant, the capital costs alone showed a more than six-fold advantage for investing in the natural-capital base (Chichilnisky and Heal, 1998).

Degradation of a natural-resource base (e.g., destruction of native populations of flora and fauna) not only affects the volume and quality of ecosystem services the base provides, but it also challenges the system's resilience, which is its capacity to absorb disturbances, or perturbations, without undergoing fundamental changes in its functional characteristics. The way to interpret an ecosystem's loss of resilience is to view it as having moved to a new stability domain, which is another way of saying that the system, having crossed a "threshold," has been captured by a different attractor (Levin *et al.*, 1998; Levin, 1999; Brock *et al.*, 1999). Sudden changes in the character of shallow lakes (e.g., from clear to eutrophied water), owing to increases in the input of nutrients (Scheffer and Carpenter, 2003) and the transformation of grasslands into shrublands, consequent upon nonadaptive cattle-management practices (Perrings and Walker, 2004) provide two examples. Human societies have on occasions been unable to avoid suffering from unexpected flips in their local ecosystems because of this.

In particular ecosystems, biodiversity has been shown to be key to ecosystem resilience. However, even today it is a popular belief that the utilitarian value of biodiversity is located mainly in the potential uses of genetic material (e.g., for pharmaceutical purposes), or in other words, that its social worth is almost wholly an option value. Preservation of biodiversity is seen as a way to hold a diverse portfolio of assets with uncertain payoff. But as other contributions to this Encyclopedia make clear, biodiversity, appropriately conceived, is essential for the maintenance of a wide variety of services on which humans depend for survival. This has the important corollary that to invoke the idea of substitutability among natural resources in order to play down the use value of biodiversity, as has often been done (Simon, 1981), is a wrong intellectual move. The point is this: if an ecosystem's biodiversity is necessary for it to be able to continue providing humans with its services, the importance of that same biodiversity cannot be downplayed by the mere hope that for every species, there are substitute species lying in wait within that same ecosystem. In short, there is an inconsistency in this line of reasoning. Recall Ehrlich and Ehrlich's famous analogy relating species in an ecosystem to rivets in an airplane (Ehrlich and Ehrlich, 1981). One by one, perhaps, species may disappear and may not be missed. Eventually, however, the cumulative effect of loss of biodiversity will lead to the crash of ecosystem functioning, just as the cumulative loss of redundant rivets will lead to the crash of an airplane.

Discontinuous Value Functions

How do discontinuities in the social worth of ecosystems affect valuation exercises and social cost-benefit analysis? To answer this, it helps to formalize.

Consider an ecosystem describable by N state variables, indexed by i and j ($i, j = 1, 2, \ldots, N$). For concreteness, one may think of each state variable as reflecting the population size of a particular species. (As noted in the section Biological Resources and Biodiversity, problems of environmental pollution can be formulated in a similar manner.) Denote time by t (≥ 0) and let S_{it} be the population size of i at t. Time is taken to be a continuous variable. Imagine, therefore, that the dynamics of the ecosystem can be described by a system of (nonlinear) differential equations. For expositional ease, one can assume for the moment that the system is deterministic.

Let the net reproduction rate of i at t be F_{it}. Since the ecosystem is coupled, F_{it} is a function of the stocks at t. This can be written as $F_{it}(S_{1t}, S_{2t}, \ldots, S_{Nt})$, for $i = 1, 2, \ldots, N$. It is assumed that ecologists have estimated these functions. Assume next that the ecosystem dynamics are autonomous. This means that F_{it} is not an explicit function of t. So drop the subscript t from F_{it} and write the function as $F_i(S_{1t}, S_{2t}, \ldots, S_{Nt})$. In all the familiar applications of this framework with which we are familiar, F_i is taken to be a differentiable function. Assume this.

The analysis begins at $t = 0$ (the "present"). Denote by X_{it} the rate at which species i is harvested at time t. Now imagine that economists have studied the human–ecosystem interactions in question. They have enquired into the structure of property rights, demand conditions, government policies, and so forth. On the basis of this they have concluded that harvests are based on an implicit policy, in that they are time autonomous and are functions solely of stocks. So one may write $X_{it} = X_i(S_{1t}, S_{2t}, \ldots, S_{Nt})$. Assume that X_i is piece-wise continuous and possesses right- and left-partial derivatives everywhere. This is a technical assumption and a good one. For example, optimal policy functions for those ecosystem management problems that have been studied have been found to possess this property (Skiba, 1978; Brock *et al.*, 1999). Moreover, actual harvest rates have frequently been known to be approximately constant over time. So both sets of example satisfy the assumption. (Actual harvest rates frequently display time trends, say, because population and income grow. Time trends in X_{it} would render the system of eqn [1] nonautonomous. This article restricts the discussion to autonomous systems because it is understood that the mathematics of autonomous system is better than that of nonautonomus ones. But experience with simple nonautonomous systems suggests that the arguments offered next cover them as well.) No doubt some of the X_{it}s would be zero. For example, it could be that only one species in the ecosystem is ever harvested (because, say, it is the only one that has economic worth). One should think of X_{it} as a forecast. It should be stressed that $X_i(S_{1t}, S_{2t}, ..., S_{Nt})$ is not necessarily a socially optimal harvest-policy function. It can be an actual policy function within an imperfect institution (e.g., the ecosystem could be one to which there is free access).

The rate of increase of S_{it} is the difference between F_i and X_{it}. Therefore, given the economists' forecast for X_{it}, mathematicians would be able to forecast S_{it} by solving the "coupled" system of differential equations:

$$dS_{it}/dt = F_i(S_{1t}, S_{2t}, ..., S_{Nt}) - X_i(S_{1t}, S_{2t}, ..., S_{Nt}), \quad \text{for all } i \qquad [1]$$

For simplicity of exposition, assume that the social worth of the ecosystem is autonomous in time. Then express that worth by a scalar V. Since V would be a function of the stocks, and may be written as $V(S_{1t}, S_{2t}, ..., S_{Nt})$. V is the value at t of the entire ecosystem. It is the maximum amount society should be "willing to pay" at t for the ecosystem's survival if the stocks of the N resources were S_{1t}, S_{2t}, ..., and S_{Nt}, respectively. Any alternative use of the site (e.g., conversion into an urban center) would have to be worth at least V if the alternative were to be acceptable. The form of V would depend on the availability of substitutes for those species that are harvested. Again, to keep the mathematical notation from getting out of hand, assume that there are no substitutes available at low cost from outside the ecosystem (e.g., because the community doing the harvesting is not near other sources of livelihood). From eqn [1] it is possible to use the forecast on harvest rates to determine forecasts on stocks. This in turn makes it possible to forecast the time path of V.

At one level the valuation problem is now "solved": $V(S_{1t}, S_{2t}, ..., S_{Nt})$ would be the value of the ecosystem at t. It would be the social worth of the ecosystem at t. The problem is that V is typically a nonlinear function, which means that it is hugely difficult to estimate. The task of valuing ecosystems would be made much easier if recourse were taken to estimating accounting prices. The advantage would be this: since accounting prices reflect the social worth of marginal units of the various populations, one could use such prices to construct a linear index of the ecosystem's value. This article turns to this discussion.

Assume for the moment that V is differentiable everywhere. Let P_{it} be the accounting price of i at t. From the discussion in the section Institutional Failures and Poverty: Global versus Local Environmental Problems and from the assumption that no substitute resources are near in hand for the human community in question, it is known that (if substitutes were available, P_{it} would be the minimum of $\partial V/\partial S_{it}$ and the accounting price of the substitute. It is better to avoid such complications here.)

$$P_{it} \alpha \partial V/\partial S_{it}, \quad \text{for all } i \text{ and all } t \qquad [2]$$

At time t, the value of species i would be $P_{it}S_{it}$. It follows that the value of the ecosystem itself would be $C_i P_{it} S_{it}$. Notice that this is a linear function of stocks, the weights being accounting prices.

In the section Institutional Failures and Poverty: Global versus Local Environmental Problems it is noted that a "project" can be thought of as a perturbation of the forecast X_{it}. So a project can be denoted as $(\otimes X_{1t} \otimes X_{1t}, \otimes X_{2t}, ..., \otimes X_{Nt})$, for $t \geq 0$. ($\otimes$ denotes an operator signifying "small difference".) (Note that for all i and all t, $\Delta X_{it} = \Sigma_j[\partial X_i/\partial S_{jt}][dS_{jt}/dt]\Delta t$.) Some of the $\otimes X_{it}$s would be zero. Nevertheless, the project would be expected to perturb future stocks of all the resources, since this is what a strongly coupled ecosystem would be expected to display.

Let r be the social rate of discount and let C_{it} be the unit cost of harvesting i at t. (C_{it} could depend on stock sizes at t. For example, the unit cost of fishing depends not only on the technology available for fishing and the price of fishing equipment, but also on the stock in the fishery: the larger the stock, the smaller the unit cost.) It follows that the present discounted value of the flow of net social benefits from the project is:

$$\int_0^\infty e^{-rt}[\copyright_i(P_{it} - C_{it}) \otimes X_{it}]dt \qquad [3]$$

If expression [3] is positive, the project should be accepted; if it is negative, the project should be rejected.

It can be argued that projects, as it has been defined here, are merely "small" perturbations, whereas redirecting economic activity so as to avoid damaging an ecosystem irrevocably could involve drastic change. But it should be noted that one way to conceptualize a "large" perturbation is to regard it as the sum of a large number of small perturbations. A large perturbation (i.e., a large project) could then be evaluated by repeated use of expression [3].

However, if this route is not adopted, social cost–benefit analysis of large projects requires the project evaluator to estimate the large changes in V consequent upon the adoption of large changes in economic policy. Accounting prices, reflecting as they do the social worth of marginal units of the various resources (expression [2]), would then not suffice: the evaluator would need to integrate over the marginal units so as to estimate "consumer surpluses," to use a term familiar in economics.

So far, so good. But there is a problem with the account: it is unreasonable to assume that V is differentiable, even continuous, everywhere. Ecosystems are nonlinear systems (eqn [1]). So even if it were reasonable to suppose that V is differentiable everywhere else, it would be wrong to suppose that it is even continuous at loci of points separating different basins of attraction (i.e., at separatrices). (An important early contribution, showed via an example that if harvest functions are optimal, V is continuous even at points where harvests are discontinuous. This means that accounting prices are not uniquely specified at such points (Skiba, 1978). However, V can be shown to possess right- and left-partial derivatives there. So, accounting prices can be used for evaluation purposes, even though they are not uniquely given at every point on the space of resource stocks.)

But if the X_is are not optimal, V can be discontinuous at points on a separatrix. This causes problems, because accounting prices cannot even be defined at such points. Let the

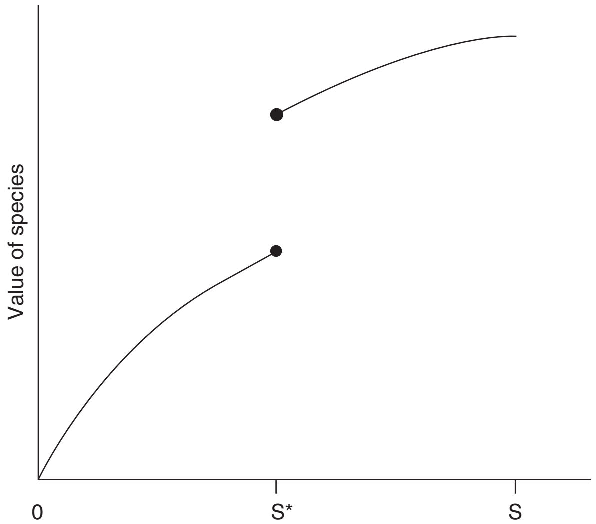

Figure 2 Nonlinear model depicting value of species in relation to function of the stock, where the ecosystem comprises a single species.

implications of this for biodiversity valuation and social cost-benefit analysis be studied.

Experience with nonlinear models of ecosystems tells that, under the assumptions made here, there could be at most a countable number of separatrices. This is fortunate, because it means that points on the stock space that are ''troublesome'' are nongeneric. So assume this. In Figure 2 the matter is illustrated in the context of an ecosystem comprising a single species. The figure depicts the case where the separatrix is a single point, S^*, reflecting a threshold. For example, it could be that, under the harvesting policy $X(S)$, the species would become extinct if its population were below S^*, but would be harvested in a sustainable manner if the population more than S^*. So stocks to the right and left of S^* represent different basins of attraction. Reasonably enough, Figure 2 depicts a case where the value of the species, $V(S)$, is an increasing function of the stock. It is assumed to be continuous (indeed, differentiable) everywhere except at S^*, where it jumps (an irrevocably dying population being a lot less valuable than a sustainable one). Of course, the location of S^* depends on $X(S)$: change the harvesting policy slightly, and S^* will shift slightly. The influence of $X(S)$ on S^* is something that has to be estimated if ecologists and economists are to offer policy advice.

Now, except by fluke the stock at $t=0$ would be different from S^*. So then assume it is different. If the project is sufficiently small, the account of social cost–benefit analysis discussed earlier remains valid: the system would not cross into a different basin of attraction. But a good theory should be extendable to fluke cases. Moreover, actual projects are frequently not ''small,'' so acceptance of a project or its rejection could mean that the ecosystem is eventually in one basin of attraction rather than in another. How can this theory be extended the theory to handle the possibility that the ecosystem crosses into a different basin of attraction? In particular, is the repeated use of expression [3] a feasible means of evaluating projects?

It is as well to be clear where the problem lies if one were to try using expression [3]. The problem lies in that an accounting price cannot be defined at S^*. This means that a project that involves the stock passing through S^* cannot be evaluated by means of a linear index of social profitability. The height of the jump would have to be estimated and put to use in social cost–benefit analysis. Ecologists and economists would have to combine their expertise to locate S^* and identify the functional form of $V(S)$, both on the right and on the left of S^*. Estimate the height of the jump involves measuring ''consumer surpluses'', a point noted earlier. In short, at least one small project in the series of small projects that add up to the large project in question would not be assessable by means of expression [3]. This makes for difficulties.

Having noted this, there is a way to avoid the problem. The authors have been studying deterministic systems. Introducing uncertainty about the location of S^* can help matters by smoothing the value function. To see how, imagine that $V(S)$ represents the expected value of the resource's social worth at S. If the location of S^* were a smooth probability distribution, $V(S)$ would be a continuous, even a differentiable function. In this case an accounting price of the resource would be definable at all S (S^* being a smooth random variable). A linear index of the social profitability of projects could then be constructed. The methods of social cost-benefit analysis outlined earlier would remain valid.

It is not often that introducing realism simplifies analysis. Valuing biodiversity would seem to be an exception.

Conclusions

The US National Academy of Science described the fundamental challenge in valuing ecosystem services as ''providing an explicit description and adequate assessment of the links between the structure and functions of natural systems, the benefits (i.e., goods and services) derived by humanity, and their subsequent values'' (Heal *et al.*, 2005, p. 2). Despite the strides already taken in meeting this challenge, however, there are few economic studies of the value avoiding thresholds. Nor are there many studies of the value of the capacity of systems to keep functioning when subject to shocks or stresses. This is partly because there is still disagreement among ecologists of the role of biodiversity in maintaining either the stability or the resilience of ecosystems. At small scales an increase in species richness and the diversity of overlapping functional groups of species enhances the level of functional diversity, and hence both ecological stability (Tilman *et al.*, 2005) and resilience. At larger scales, the evidence is less clear. However, it is likely that maintaining a portfolio of options in ecosystems does strengthen their capacity to respond to shocks and stresses in constructive and creative ways. There is certainly evidence for this is coupled social–ecological system. In agricultural systems, for example, greater crop diversity has been shown to reduce the variance of both agricultural yields and farm incomes (Di Falco, 2007). Whereas the homogenization of agroecosystems has sometimes increased yields in the short run, it has often been at the cost of increasing variance in yields due to the increased vulnerability to pests and pathogens it brings. This in turn increases reliance on pesticides that are potentially damaging to both humans and other species. In this case, the value of biodiversity – at least the value of the portfolio of cultivated species in the system – is the value to society of keeping crop production risks within acceptable bounds.

The wider economic literature has tended to identify biodiversity with conservation, and sometimes even more narrowly with the preservation of endangered species. It has also tended identify biodiversity change with biodiversity loss. But the extirpation or extinction of endangered species, important though it is, is only one dimension of biodiversity change. Biodiversity change affects the functioning of ecosystems in ways that potentially affect a large array of interdependent ecosystem services. Actions designed to enhance one service frequently compromise other services. In very many cases the effect of biodiversity change is to alter the capacity of the system to function over a range of environmental conditions. So the effect may not be immediate. In fact it may not be felt for many years. The valuation of biodiversity requires that one understands how a change in the mix of species within some functional group affects human wellbeing at various spatial and temporal scales. It also requires that one understands which effects of biodiversity change are captured in market prices and which are not.

It is not always the case that biodiversity loss is bad. Many very important ecosystem services depend on the simplification of ecosystems, and that frequently means the extirpation of particular populations. Protecting people against the effects of disease means controlling pathogens, even to the point of driving them to the extinction in the case of smallpox and rinderpest. The economic problem of biodiversity change is not that ecosystems have been simplified; it is that many of the less direct consequences of simplification are not taken into account by those responsible. The indirect effects of simplification on other ecosystem services, on other people, or on other places are frequently ignored. Such externalities of biodiversity change frequently impose costs on those who either have no voice (future generations) or are least able to cope with the effects (the poorest among the present generation). That is the economic problem of biodiversity change.

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