Valuing Biodiversity

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Contents

Abstract

Any assessment of the value of biodiversity should begin with an account of why we need to value it and the reasons market values would not be expected to suffice for the purpose. Sections 1-3 discuss these matters in the wider context of valuing natural resources (biodiversity is but a special case). A transitional section (Section 4) shows how values can be translated into prescriptions for economic policy. Sections 5-7 discuss the special problems that arise in valuing biodiversity and the techniques that are available for coping with those problems.

Keywords: investment projects, accounting prices, social discount rates, social cost-benefit analysis, bifurcation points, irreversibilities, substitution possibilities.

1. The Subject

Resource economists view the natural environment through the lens of population ecology. Since the latter's focus is the dynamics of interacting populations, it has proven useful in resource economics to regard the functioning of the rest of the ecosystem as exogenously given. 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 in resource economics 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 (e.g., Spence, 1974; Clark, 1976; Dasgupta and Heal, 1979; Hartwick, 1993), economists have also determined harvest rates when harvesters have free access to the resource (e.g., Gordon, 1954; Dasgupta and Heal, 1979; Dasgupta, 1982). We now have 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.

Environmental economists, in contrast, base their studies on ecosystem ecology. There the focus is on such objects as energy at different trophic levels and its rate of flow among them, the distribution and flows of bio-chemical substances in soils and bodies of water, and of gases and particulates in the atmosphere. The motivation is to study the biotic and abiotic processes which underlie various ecosystem functions. 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 endeavour (e.g., Mäler, 1974; Costanza, 1991; Nordhaus, 1994; Perrings and Walker, 1995; 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 the ecosystem.

Formally, it is useful to interpret differences between resource and environmental economics in terms of the state variables that are taken to characterize complex systems (state variables are frequently called stock variables). In resource economics state variables are "quantities" (biomass units for forests, cowdung and crop residue; herd size, expressed in numbers, for animal populations; acre-feet for aquifers; and so on). In environmental economics state variables are frequently "quality" indices, such as those for air, soil, or water (although quality - for example, salinity - is typically inferred from quantity indices), the intention being to draw attention to ecosystem services. State variables are summary statistics, reflecting as they do different forms of aggregation. Therein lies their virtue: they enable the analyst to study complex systems by means of a few strategically chosen variables.[1]

The viewpoint just offered, that of distinguishing resource and environmental economics in terms of the state variables that summarize complex systems, has enabled economists to integrate problems of resource management with those of environmental pollution (Dasgupta, 1982). Insights from one field of study have been used for gaining an understanding of the others. The viewpoint also reminds us that environmental and resource economics is the study of renewable natural resource systems, when subject to human predation. Since it would be convenient to refer to environmental and resource economics by an overarching name, I do so in this article by the term ecological economics.[2]

The economics of biodiversity, still in its infancy (Perrings et al., 1994, 1995, are pioneering collections of studies on the subject), is all of the above. But it includes an additional complication, in that it recognizes that aggregate statistics can mislead; that, for example, to devise schemes to manage a resource without understanding its function within the ecosystem of which it is a part can be a road to disaster, because the ecosystem's biodiversity can itself be a source of its productivity, including, for example, the system's stability (Tilman and Downing, 1994; Tilman, 1997). In this article I first provide an account of valuation problems in ecological economics; they are, naturally, valuation problems as well in the economics of biodiversity. I then sketch some of the special problems that arise in the latter and of techniques available for resolving them.

2. Market Failure

If you were to browse among such leading Western journals in ecological economics as the Journal of Environmental Economics and Management, you would discover that a central concern in the field has been to devise ways by which it would be possible to ascertain the "value" of natural resources and the services they provide. A question that would inevitably occur to you is, why? Why should there be a special need to determine the worth of such resources? Why not rely on their market prices? More generally, we may ask why it would not do to rely on markets to guide decisions bearing on the natural-resource base, be they global or local, in the way we do for so many other goods and services. To put it another way, we may

[1] This two-fold classification, based on the character of state variables, is implicit in contemporary ecology (Ehrlich, Ehrlich and Holdren, 1977; Roughgarden, May and Levin, 1989).

[2] I am able to usurp the term from the literature, for the reason that it appears to have no fixed meaning: "ecological economics" seems to mean different things to different people.

ask why markets are not an adequate set of institutions for protecting the "environment".

The answer is that for many natural resources markets simply do not exist. In some cases they do not exist because the costs of negotiation and monitoring are too high. One class of examples is provided by economic activities which are affected by ecological interactions involving long geographical distances (e.g., the effects of uplands 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 non-existent because future generations are not present today to negotiate with us). Then there are cases (e.g., the atmosphere, aquifers, and the open seas) where the nature of the physical situation (viz., the migratory nature of the resource) makes private property rights impractical and so keeps markets from existing; while in others, ill-specified or unprotected property rights prevent their existence, or make markets function wrongly even when they do exist. In short, environmental problems are often caused by market failure (but see Section 3). Indeed, the phenomenon of externalities (that is, exchanges among people which take place without their consent) looms large in what has traditionally been called environmental economics.[3]

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 we care about our 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 we do not care directly about the well-being of our distant descendants, we do care about them indirectly. However, there is a distinct possibility that our 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 (see, e.g., Lind, 1982; Arrow et al., 1996; Portney and Weyant, 1999). In short, market failure involves not only misallocation of resources in the present, but also misallocation 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 are called accounting prices (sometimes, shadow prices). The accounting price of a resource is the increase in social well-being which would be enjoyed if a unit more of the resource were made available costlessly. So a resource's accounting price is the difference

[3] 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).

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, being 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 insufficent one.

One way to improve matters would be to impose regulations on resource users; for example, restrictions on effluent discharges, quotas on fish harvests, and bans on logging. Another would be to introduce a system of taxes, often called Pigovian taxes (in honour of Pigou, 1920, who first discussed the difference between private and social costs in the context of environmental pollution). Pollution charges, charges on the amount of fish harvested, and stumpage fees are examples. The idea underlying Pigouvian 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 we recognize not only that ecological processes are stochastic, but also that resource users and government agencies do not have the same information about local ecology, say, for example, the cost of waste disposal (Meade, 1973; Weitzman, 1974; Baumol and Oates, 1975; Dasgupta, 1982). We cannot enter into details here, but three points are worth noting. First, the two schemes are distributionally not equivalent: under a quota, resource rents are captured by harvesters and polluters, whereas under a tax system they are collected by the tax authority. Secondly, the imposition of Pigouvian 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. Thirdly, 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 van der Ploeg, 1994; Goulder, 1995; Bovenberg and Goulder, 1996; but see Bohm, 1996), a rhetorical phrase that has been much used in recent years to persuade governments to impose "green" taxes. Matters of public finance have been a recurrent theme in ecological economics (see, especially, Baumol and Oates, 1975; Cropper and Oates, 1992; Carraro and Siniscalco, 1996).[4]

3. Institutional Failures and Poverty: Global vs. Local Environmental Problems

Thus far, market failure. Recently, however, certain patterns of environmental deterioration have been traced to government failure. For example, Binswanger (1991) has argued that, in Brazil, the exemption from taxation of virtually all agricultural income (allied to the fact that logging is regarded as proof of land occupancy) has provided strong incentives to the rich to acquire forest land and to then deforest it. He has argued that the subsidy the government has thereby provided to the private sector has been so large, that a reduction in deforestation (via a removal of subsidies) is in Brazil's interests, not merely in the interest of the rest of the world. This has implications for international negotiations. The current consensus appears to be that, as a country, Brazil has much to lose from reducing the rate of deforestation she is engaged in. If this were true, there would be a case for the rest of the world to subsidize her, as compensation for losses she would sustain if she were to restrain herself. But, as Binswanger's account suggests, it is not clear if the consensus is correct.[5] Elsewhere (and by extension), one would imagine that the massive agricultural subsidies in the European Union considerably influence agricultural practices in ways that inflict substantial damage on the environment.

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. We 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, we should expect women to favour planting for fuelwood and men for fruit trees, because it is the women and children who collect fuelwood, while 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, 1993a).

That political instability (at the extreme, civil war) is a direct cause of resource

[4] A hybrid policy instrument, which involves the government issuing a fixed number of transferable licenses, combines some of the features of quotas and Pigouvian taxes. For example, the scheme resembles quotas, in that, resource rents are not captured by government; and it resembles Pigouvian taxes, in that, at the margin license holders pay the accounting price of the resource for its use. See Tietenberg (1980, 1990) for a good discussion of transferable licenses, both in theory and in practice.

[5] 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 have also been a potent force.

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 (1994) has offered statistical evidence of a positive link between political instability and forest depletion.

Taken together, these examples reflect the environmental consequences of <u>institutional failure</u>. 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 and Mäler, 1991, 1995; Dasgupta, 1993a, 1999; Ehrlich, Ehrlich, and Daily, 1995). This reverse causality occurs because some natural resources (e.g., ponds and rivers) are essential for survival in normal times, while 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 centres), social norms which 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, while 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 which earlier were subject to communitarian regulations become "open access", with all the attended 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 upon one another in a synergistic manner over time (Dasgupta, 1993a, 1995, 2000). Recent experiences in sub-Saharan Africa and Pakistan are 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 destitutes 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 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 rains, which sweep across regions, nations and continents; and about those matters (such as, for example, 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 I have 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, while others identify them with wealth and unprecedented expenditure patterns in the North (I am using the geographical terms in their current geo-political sense). 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 (Dasgupta and Mäler, 1995; Reardon and Vosti, 1995; Vincent and Ali, 1997). Thus, growth in industrial wastes and resource use have been allied to increased economic activity; and in the former Socialist block 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 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.

On the other hand, 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 air-borne 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. I would guess that this lacuna is 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 need constantly to be kept in mind in discussions of the value of biodiversity in different locations.

4. Valuing Resources and Evaluating Projects

Since institutional failures abound in our 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.

Analysing the consequences of a project would involve estimating the need for labour, 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 Section 7 for a formalisation). 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.[6]

In order 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; see Lind, 1986; Arrow et al., 1996; Portney and Weyant, 1998), the net social benefits yielded by a project in each period are added. Projects which yield a positive present discounted value of net social benefits are recommended for acceptance, those yielding 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 fifty years, and is now, to all intents and purposes, complete (Dasgupta, Marglin and Sen, 1972; Little and Mirrlees, 1974; Dasgupta and Mäler, 2000).[7]

A prior exercize (that is, prior to conducting social cost-benefit analysis) is to estimate

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[6] But there are signs of change. See, for example, Perrings et al. (1994, 1995), Vincent and Ali (1997), Chopra and Kadekodi (1999), and various issues of the new journal, Environment and Development Economics.

[7] Daily et al. (1999) provides a non-technical account of the role of social cost-benefit analysis in environmental management.

accounting prices. A great deal of work in ecological economics has been directed at discovering methods for estimating accounting prices of natural resources. I turn to this.

It is as well to remember that the kinds of resources we are thinking of here 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 aesthetic (e.g., a shore-line), or it may be intrinsic (see below); 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, 1986; 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, qua 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 used also 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, for example, World Bank, 1992). Other techniques have been devised for valuing "amenities", such as places of scenic beauty.[8] And so on.

Methods have also been devised for giving expression to the "precautionary principle", which assumes a particularly subtle form when applied to such resources as genetic material in tropical forests. The subtlety arises from a combination of two things: 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. 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

[8] 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 which offer the amenity (this is called the "hedonic price" of land). See Mäler and Wyzga (1976), Mitchell and Carson (1989) and Freeman (1992).

accounting price of a resource is, at the very least, the sum of its direct use-value and its option value.

These techniques enable us 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 resource. An additional consideration is applicable to living resources: their intrinsic worth <u>as</u> 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 <u>intrinsic worth</u> of living things is inherent not only in traditional religious systems of ethics, but also in modern ethical theories. So the question is not so much whether living creatures have intrinsic worth, but rather, of ways of assessing this worth. As it is almost impossible to get a quantitative handle on intrinsic worth, the correct thing to do is 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.

We may conclude that the social worth of natural resources can be decomposed into three parts: their <u>use value</u>, their <u>option value</u>, and their <u>intrinsic value</u>. The components appear in different proportions, depending on the resource. For example, oil and natural gas would not be thought to have intrinsic value, nor perhaps an option value, but they have use value. On the other hand, primates would be thought to be intrinsically valuable. And so on.

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 <u>changes</u> 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 (for example, 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.

Contrast such an estimate of the value of an incremental change in the natural-resource base with the one which says that, world-wide, the flow of environmental services is currently worth, <u>in total</u>, 33 trillion US dollars annually (Costanza <u>et al</u>., 1997). The former is meaningful because it presumes that humanity will survive the incremental change and be there to experience and assess the change. The reason the latter should cause us to balk is that if environmental services were to cease, life would not exist. But then who would be there to receive 33 trillion dollars of annual benefits if humanity wished to exchange its very existence for them? This is a case where the value of an entire something has no meaning and, therefore, is of no use, even though the value of incremental changes to that same something not only has meaning, it also has use.

5. Biodiversity: Necessity or Luxury?

All said, though, biodiversity has been neglected in economics. Scratch an economist, and you are all too likely to find someone who regards the natural-resource base as an "amenity". Thus, it is even today a commonplace that, to quote a recent editorial in London's Independent (4 December 1999), "... (economic) growth is good for the environment because countries need to put poverty behind them in order to care", or that, to quote the Economist (4 December, 1999, p. 17), "... trade improves the environment, because it raises incomes, and the richer people are, the more willing they are to devote resources to cleaning up their living space."

I quote these views not so as to question that poverty should be a phenomenon of the past, nor to suggest that arbitrary restrictions on trade do not cause much harm, but to show that natural resources are widely viewed as luxuries. This is, of course, a wrong view: the natural-resource base is not a luxury. Producing, as it does a multitude of ecosystem services, the natural-resource base is a necessity.[9] Indeed, the ecological economics to be found in the North does not exactly resonate among those who worry about degradation of the natural-resource base in the South and the additional hardship this brings to the many among the poorest of people whose lives depend directly on that base.[10] Needless to say, it does not resonate among ecologists, qua ecologists, either.

This perspective, of viewing natural resources as luxuries, 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., sulphur dioxide (SO_2)) increase as GDP per head increases, but when GDP per head is high, concentrations decrease as GDP per

[9] As stressed elsewhere in the Enclopedia, these services include maintaining a genetic library, preserving and regenerating soil, recycling nutrients, controlling floods, filtering pollutants, assimilating waste, pollinating crops, operating the hydrological cycle, and maintaining the gaseous composition of the atmosphere.

[10] I have gone into these concerns in greater detail in Dasgupta (1982, 1993a, 1996, 1997).

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".[11]

Panayotou (1992) has 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. Sweden, for example, was found to lie on the downward part of the curve. Indeed, time series on timber stocks and sulphur and nitrogen emissions in Sweden, covering the decade of the 1980s, are consistent with this: timber stocks have increased, and the emission rates of sulphur and nitrogen oxides have declined.

It will be noticed that, forests excepted (but see Section 6), the above findings concern mobile pollutants. Their mobility hides the fact that earlier emissions have had to find somewhere to lodge: since matter is conserved, inputs and outputs of material must balance (Ayres and Kneese, 1969; d'Arge, Ayres and Kneese, 1970; Mäler, 1974). The point is that my emissions would affect not only me, it would affect you as well if you lived down-wind or down-stream. But even if we were to leave that aside, the logic underlying the environmental 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. The science of biodiversity has shown this presumption to be false. The presence of ecological thresholds implies that damage to ecosystems can be irreversible. As an overarching metaphor for "tradeoffs" between manufactured wealth and resource degradation, the environmental Kuznets curve has to be rejected.[12]

6. Substitution Possibilities[13]

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,

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[11] 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.

[12] 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).

[13] This and the following section are based on Dasgupta and Mäler (2000), Dasgupta, Levin and Lubchenco (2000) and Dasgupta, Levin, Lubchenco and Mäler (2000).

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). Secondly, manufactured capital can substitute for labour and natural resources in production (the wheel and double-glazing are two extreme examples). Thirdly, 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 over a hundred years ago. The innovation was an enormous energy saver in engines. Fourthly, and for us here 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.[14]

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 (1998) has argued that substitution made the Industrial Revolution in England in the eighteenth century. 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 we are in 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

[14] 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 above, four-way classification.

substituting it with manufactured capital can be uneconomic. Chichilnisky and Heal (1998) compared the costs of restoring the ecological functioning of the Catskill Watershed ecosystem in New York State, to the costs of replacing the natural water purification and filtration services the ecosystem has provided in the past by building a water-purification plant costing 8 billion US dollars. They showed the overwhelming economic advantages of preservation over construction: 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 6-fold advantage for investing in the natural-capital base.

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; 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, Mäler and Perrings, 2000). Sudden changes in the character of shallow lakes (e.g., from clear to eutrophied water), owing to increases in the input of nutrients (Scheffer 1997, Carpenter et al., 1998) and the transformation of grasslands into shrublands, consequent upon non-adaptive cattle-management practices (Perrings and Walker, 1995), provide two examples. Human societies have on occasions been unable to avoid suffering from unexpected flips in their local ecosystems because of this. Fishermen on Lake Victoria and the nomads in the now-shrublands of southern Africa are examples from recent years.

Biodiversity would appear to be a 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 people frequently do (e.g., Simon 1981, 1994), is a wrong intellectual move. The point is this: if an ecosystem's biodiversity is necessary for it to be able to continue providing us 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 the famous analogy in Ehrlich and Ehrlich (1981) relating species in an ecosystem to rivets in an airplane. One by one, perhaps, species may disappear and 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.

7. Discontinuous Value Functions

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

Consider an ecosystem describable by N state variables, indexed by i and j (i, j = 1, 2, ..., N). For concreteness, we may think of each state variable as reflecting the population size of a particular species. (As noted in Section 1, 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. We imagine, therefore, that the dynamics of the ecosystem can be described by a system of (non-linear) differential equations. For expositional ease, we 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 I write as $F_{it}(S_{1t}, S_{2t}, ..., S_{Nt})$, for i = 1, 2, ..., N. I assume 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 I drop the subscript t from F_{it} and write the function as $F_i(S_{1t}, S_{2t}, ..., S_{Nt})$. In all the applications of this framework with which I am familiar, F_i is taken to be a differentiable function. Let us 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. We 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 we may write $X_{it} = X_i(S_{1t}, S_{2t}, ..., S_{Nt})$. 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.[15] 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). We 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 functions 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:

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

For simplicity of exposition, let us assume that the social worth of the ecosystem is autonomous in time. We may then express that worth by a scalar V. Since V would be a function of the stocks, we may write it 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 centre) would have to be worth at least V if the 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, I 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). Using equations (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 non-linear 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, we could use such prices to construct a linear index of the ecosystem's value. I turn to this.

Assume for the moment that V is differentiable everywhere. Let P_{it} be the accounting

[15] Actual harvest rates frequently display time trends, say, because population and income grow. Time trends in X_{it} would render the system of equations (1) below non-autonomous. In the text I am restricting the discussion to autonomous systems because I understand the mathematics of autonomous better than that of non-autonomus ones. But experience with simple non-autonomous systems suggests that the arguments I offer below in the text covers them as well.

price of i at t. From the discussion in Section 4 and from our assumption that no substitute resources are near in hand for the human community in question, we know that

$$P_{it} \equiv \partial V / \partial S_{it}, \text{ for all i and all t.}^{16} \qquad (2)$$

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 $\Sigma_i P_{it} S_{it}$. Notice that this is a linear function of stocks, the weights being accounting prices.

In Section 4 we noted that a "project" can be thought of as a perturbation of the forecast X_{it}. So a project can be denoted as $(\Delta X_{1t}, \Delta X_{2t}, ..., \Delta X_{Nt})$, for $t \geq 0$. (Δ denotes an operator signifying "small difference".)[17] Some of the ΔX_{it}s would be zero. Nevertheless, the project would be expected to perturb future stocks of <u>all</u> 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.[18] It follows that the present discounted value of the flow of net social benefits from the project is:

$$_0\int^{\infty} e^{-rt}[\Sigma_i (P_{it} - C_{it}) \Delta X_{it}] dt. \qquad (3)$$

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 I have defined them 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 conceptualise 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 <u>un</u>reasonable to assume that V is differentiable, even continuous, everywhere. Ecosystems are non-linear systems (equation

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[16] If substitutes were available, P_{it} would be the minimum of $\partial V / \partial S_{it}$ and the accounting price of the substitute. I want to avoid such complications here.

[17] Note that, for all i and all t, $\Delta X_{it} = \Sigma_j [\partial X_i / \partial S_{jt}][dS_{jt} / dt] \Delta t$.

[18] 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, it also depends on the stock in the fishery: the larger the stock, the smaller the unit cost.

(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 separatrixes).[19]

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 us study the implications of this for biodiversity valuation and social cost-benefit analysis.

Experience with non-linear models of ecosystems tells us that, under the assumptions we have made, there could be at most a countable number of separatrixes. This is fortunate, because it means that points on the stock space that are "troublesome" are non-generic. So let us 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 were in excess of 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, excepting by fluke the stock at t=0 would be different from S*. So then let us assume it is different. If the project is sufficiently small, the account of social cost-benefit analysis given above 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 that acceptance of a project or its rejection could mean that the ecosystem is eventually in one basin of attraction rather than in another. How do we extend 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 we were to try using expression (3). The problem lies in that an accounting price cannot be defined at S*. This means that a project which

[19] In an important early contribution, Skiba (1978) 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. 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.

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*. Estimating 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 which 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. We 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 <u>expected value</u> 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.

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