Pricing Biodiversity and Ecosystem Services: The Never-Ending Story

BY MARINO GATTO AND GIULIO A. DE LEO

In 1844, the French engineer Jules Juvénal Dupuit introduced cost–benefit analysis to evaluate investment projects. This methodology relies on the concept of consumer surplus (which is the difference between willingness-to-pay and actual payment; see Nijkamp 1977), which was also defined by Dupuit (1844). The application of cost–benefit analysis to ecological issues fell out of favor three decades ago, and it was gradually replaced by multicriteria analysis in the decision-making process for projects that have an impact on the environment. Although multicriteria analysis is currently used for environmental impact assessments in many nations, in the last 5 years the concept of cost–benefit analysis has again become fashionable, along with the various pricing techniques associated with it, such as contingent valuation methods, hedonic prices, and costs of replacement of ecological services. For example, during the First World Congress of Environmental and Resource Economists held in Venice, Italy, in June 1998, 12 of the 88 sessions were focused on theoretical and empirical problems related to contingent valuation methods. Overall, almost 100 of 500 contributions were related to issues of pricing environmental goods and services. By contrast, only a small number of papers used multicriteria analysis. Economists have generated a wealth of virtuosic variations on the theme of assessing the societal value of biodiversity, but most of these techniques are invariably based on price—that is, on a single scale of values, that of goods currently traded on world markets.

Perhaps the most famous recent study on the issue of pricing biodiversity and ecological services is that by Costanza et al. (1997), who argued that if the importance of nature's free benefits could be adequately quantified in economic terms, then policy decisions would better reflect the value of ecosystem services and natural capital. Drawing on earlier studies aimed at estimating the value of a wide variety of ecosystem goods and services, Costanza et al. (1997) estimated the current economic value of the entire biosphere at $16–54 trillion per year, with an average value of approximately $33 trillion per year. By contrast, the gross national product of the United States totals approximately $18 trillion per year (Costanza et al. 1997). The paper, as its authors intended, stimulated much discussion, media attention, and debate. A special issue of *Ecological Economics* (April 1998) was devoted to commentaries on the paper, which, with few exceptions, were laudatory. Some economists (Pearce 1998) have questioned the actual numbers, but many scientists have praised the attempt to value biodiversity and ecosystem functions.

Although Costanza et al. acknowledged that their estimates were crude and imperfect, they also pointed the way to improved assessments. In particular, they noted the need to develop comprehensive ecological economic models that could adequately incorporate the complex interdependencies between ecosystems and economic systems, as well as the complex individual dynamics of both types of systems. Despite the authors' caveats and the fact that many economists have been circumspect in applying their own tools to decisions regarding natural systems, the monetary approach is perceived by scientists, policymakers, and the general public as extremely appealing; a number of biologists are also of the opinion that attaching economic values to ecological services is of paramount importance for preserving the biosphere and for effective decision-making in all cases where the environment is concerned (Daily 1997, Pimentel et al. 1997).

In this article, we espouse a contrary view, stressing that, for most of the values that humans attach to biodiversity and ecosystem services, the pricing approach is inadequate—if not misleading and obsolete—because it implies erroneously that complex decisions with important environmental impacts can be based on a single scale of values. We contend that the use of cost–benefit analysis as the exclusive tool for decision-making about environmental policy represents a setback relative to the existing legislation of the United States, Canada, the European Union, and Australia on environmental impact assessment, which explicitly incorporates multiple criteria (technical, economic, environmental, and social) in the process of evaluating different alternatives. We show that there are sound methodologies, mainly developed in business and administration schools by regional economists and by urban planners, that can assist decision-makers in evaluating projects and drafting policies while accounting for the nonmarket values of environmental services.

Marino Gatto (e-mail: gatto@elet.polimi.it) is a professor of Applied Ecology in the Dipartimento di Elettronica e Informazione, Politecnico di Milano, Milano, Italy. Giulio A. De Leo (e-mail: deleo@dsa.unipr.it) is an associate professor of Applied Ecology and Environmental Impact Assessment in the Dipartimento di Scienze Ambientali, Università degli Studi di Parma, Parma, Italy.

Type of value	Function (examples)	Type of function (examples)	Possible methods for economic evaluation
Utilitarian value	Commercial production	Hunting; fishing; plant harvesting	Monetary market value
	Protection	Shelter for man-made development; shelter for other habitats	Cost of flooding and erosion resulting from lack of protection; cost of replacement
	Pollution control; disposal of waste products; climate regulation	Absorption of pollution/sewage	Contingent valuation methods; unknown/indirect
User value	Quality of life	Providing natural beauty and a healthy and enjoyable environment	Hedonic house prices; contingent valuation methods
	Recreation	Bird watching; sport fishing; walking; jogging	Travel cost methods
	Education	School visits; projects	User willingness to pay
Nonuser value	Existence/bequest	Ecological role; existence for own sake	Incalculable; indirect
		Option for future use; bequest for future generations	Nonuser willingness to pay

Table 1. A schematic approach to economic evaluation of environmental services (adapted from Tunstall and Coker 1992).

The limitations of cost–benefit analysis and contingent valuation methods

Historically, the first important implementation of cost–benefit analysis at the political level came in 1936, with passage of the US Flood Control Act. This legislation stated that a public project can be given a green light if the benefits, to whomsoever they accrue, are in excess of estimated costs. This concept implies that all benefits and costs are to be considered, not just actual cash flows from and to government coffers. However, public agencies (e.g., the US Army Corps of Engineers) quickly ran into a problem: They were not able to give a monetary value to many environmental effects, even those that were predictable in quantitative terms. For instance, engineers could calculate the reduction of downstream water flow resulting from construction of a dam, and biologists could predict the river species most likely to become extinct as a consequence of this flow reduction. However, public agencies were not able to calculate the cost of each lost species. Therefore, many ingenious techniques for the monetary valuation of environmental goods and services have been devised since the 1940s (Table 1). These techniques fall into four basic categories (Pearce 1993):

- **Conventional market approaches.** These approaches, such as the replacement cost technique, use market prices for the environmental service that is affected. For example, degradation of vegetation in developing countries leads to a decrease in available fuelwood. Consequently, animal dung has to be used as a fuel instead of a fertilizer, and farmers must therefore replace dung with chemical fertilizers. By computing the cost of these chemical fertilizers, a monetary value for the degradation of vegetation can then be calculated.
- **Household production functions.** These approaches, such as the travel cost method, use expenditures on commodities that are substitutes or complements for the environmental service that is affected. The travel cost method was first proposed in 1947 by the economist Harold Hotelling, who, in a letter to the director of the US National Park Service (see Prewitt 1949), suggested that the actual traveling costs incurred by visitors could be used to develop a measure of the recreation value of the sites visited.
- **Hedonic pricing.** This form of pricing occurs when a price is imputed for an environmental good by examining the effect that its presence has on a relevant market-priced good. For instance, the cost of air and noise pollution is reflected in the price of plots of land that are characterized by different levels of pollution, because people are willing to pay more to build their houses in places with good air quality and little noise. This notion can be traced back to David Ricardo, who, together with Adam Smith, is one of the founding fathers of modern economics.
- **Experimental methods.** These methods include contingent valuation methods, which were devised by the resource economist Siegfried V. Ciriacy-Wantrup (1947). Contingent valuation methods require that individuals express their preferences for some environmental resources by answering questions about hypothetical choices. In particular, respondents to a contingent valuation methods questionnaire will be asked how much they would be willing to pay to ensure a welfare gain from a change in the provision of a nonmarket environmental commodity, or how much they would be willing to accept in compensation to endure a welfare loss from a reduced provision of the commodity.

Among these pricing techniques, the contingent valuation methods approach is the only one (Simpson 1998) that is capable of providing an estimate of existence values, in which biologists have a special interest. Existence value was first defined by Krutilla (1967) as the value that individuals may attach to the mere knowledge that rare and diverse species, unique natural environments, or other "goods" exist, even if these individuals do not contemplate ever making active use of or benefiting in a more direct way from them. The name "contingent valuation" comes from the fact that the procedure is contingent on a constructed or simulated market, in which people are asked to manifest, through questionnaires and interviews, their demand function for a certain environmental good (i.e., the price they would pay for one extra unit of the good versus the availability of the good).

The rationale underlying contingent valuation methods can be understood in the context of the classical approach of welfare economics (e.g., Herfindahl and Kneese 1974). In this approach, each individual has a certain utility, which is a measure of his or her welfare and is a function of the various goods and services that are available to him or her. To simplify, suppose that the welfare of each individual depends on only two quantities: the consumption (Y) of market-priced goods, measured in dollars, and the availability (X) of an unpriced environmental good, such as the water quality of a river, measured as dissolved oxygen concentration. Figure 1 shows the "indifference curves" (i.e., the geometric loci of all X and Y corresponding to the same utility for a certain consumer). The indifference curves are downsloping because the same individual welfare can be achieved by a smaller consumption of marketed goods and a larger provision of environmental goods: Instead of buying a ticket to a movie theater, one can enjoy the same welfare by strolling along a river with good water quality. In general, indifference curves are also convex, as well as downsloping, because lower and lower levels of environmental quality can be compensated for only by larger and larger increases in the consumption of marketed goods.

Assume that the initial river quality is X_0 and that the government wants to increase this quality by ΔX. If this increase could be achieved at no expense, the individual utility would jump from U_0 to U^+. However, consumers know that this increase cannot be achieved at no expense and therefore are prepared to give up some money in exchange for a higher-quality river. The maximum amount of money an individual is willing to pay is the amount that returns him or her to the initial utility, U_0 (Figure 1). Conversely, suppose that the authorities want to start a development project that will decrease the river quality by the same ΔX. If the income of each individual is not raised, the project implementation will result in a decrease of the utility, from U_0 to U^-. The minimum amount of money an individual is willing to accept for compensation is the amount that returns him or her to the initial utility, U_0 (Figure 1). Because of the convexity of indifference curves, "willingness to accept" is larger than "willingness to pay" (see Hanemann 1991 for a thorough discussion).

In practice, indifference curves are very difficult to estimate. Furthermore, they differ for each consumer because each human has a different perception of welfare. Therefore, contingent valuation methods are intended to elicit the willingness to pay or willingness to accept through questionnaires or interviews of a sample of consumers. The results of the sampling are then extended, through appropriate statistical methods, to all consumers, and the sum of all individual willingnesses to pay (or willingnesses to accept) is the monetary benefit (or cost) to be used in cost–benefit analysis. Once all the effects of a proposed project—be they environmental, social, or aesthetic—are converted to monetary benefits or costs, cost–benefit analysis requires simply that actual cash flows, such as costs of construction, and the benefits and costs evaluated by the above pricing techniques be algebraically summed. If the net benefit is positive, then the project is considered to be acceptable in all respects. If there are alternative projects, then the optimal project is the one with the maximum net benefit.

The limits of cost–benefit analysis were discussed in the 1960s, after more than two decades of experimentation (for review, see Nijkamp 1977). In particular, many authors pointed out that cost–benefit analysis encouraged policymakers to focus on things that can be measured and quantified, especially in cash terms, and to disregard problems that are too large to be assessed easily (Adams 1992).

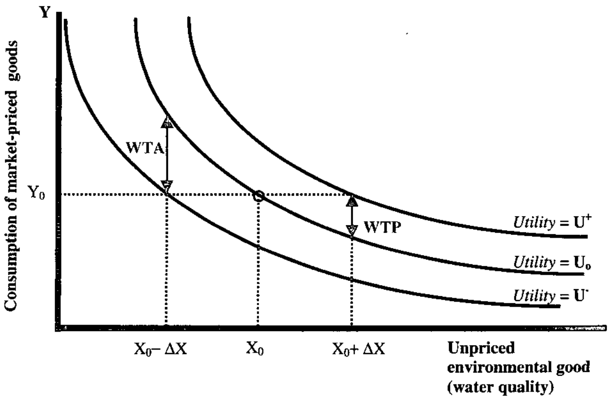

Figure 1. Indifference curves of a hypothetical consumer whose welfare depends on only two goods: a nearby river, with water of quality X (measured, for example, by its dissolved oxygen concentration) and the consumption, Y (measured in dollars), of market-priced goods. Each indifference curve is characterized by the same value of the utility function, which is the function that maps X and Y onto individual welfare. The open circle marks an initial condition that is characterized by water quality X_0, consumption Y_0, and utility U_0. The figure shows the willingness to pay for an increase ΔX of water quality that would increase utility to U^+ and the willingness to accept a decrease ΔX of water quality that would decrease utility to U^-.

Therefore, the associated price might not reflect the "true" value of social equity, environmental services, natural capital, or human health (Sagoff 1998). In particular, economists themselves recognize that the increasingly popular contingent valuation methods are undermined by several conceptual problems, such as free-riding, overbidding (MacMillan et al. 1998), and preference reversal (Chilton et al. 1998).

When it comes to monetary valuation of the goods and services provided by natural ecosystems and landscapes specifically, a number of additional problems undermine the effectiveness of pricing techniques and cost–benefit analysis. These problems include the very definition of "existence" value, the dependence of pricing techniques on the composition of the reference group, and the significance of the simulated market used in contingent valuation.

The definition of "existence" value. A classic example of contingent valuation methods is to ask for the amount of money individuals are willing to pay to ensure the continued existence of a species such as the blue whale. However, the existence value of whales does not take into account potential indirect services and benefits provided by these mammals. It is just the value of the existence of whales for humans, that is, the satisfaction that the existence of blue whales provides to people who want them to continue to exist (Holland and Cox 1992). Therefore, there is a real risk that species with very low or no aesthetic appeal or whose biological role has not been properly advertised will be given a low value, even if they play a fundamental ecological function. Without adequate information, most people do not understand the extent, importance, and gravity of most environmental problems. As a consequence, people may react emotionally and either underestimate or overestimate risks and effects.

Therefore, it is not surprising that five of the seven guidelines issued by the National Oceanic and Atmospheric Administration (NOAA; Federal Register 4601 1993) about how to conduct contingent valuation discuss how to properly inform and question respondents to produce reliable estimates (e.g., in-person interviews are preferred to telephone surveys to elicit values). Of course, acquisition of reliable and complete information is always possible in theory, but in practice strict adherence to NOAA guidelines makes contingent valuation methods expensive and time consuming (Portney 1994).

Difficulties with the reference group for pricing. Pricing techniques such as contingent valuation methods provide information about individual willingnesses to pay or willingnesses to accept, which must be summed up in the final balance of cost–benefit analysis. Therefore, the outcome of cost–benefit analysis depends strongly on the group of people that is taken as a reference for valuation—particularly on their income. Van der Straaten (1998) noted that the Exxon *Valdez* oil spill in 1989 provides a good example of this dependence. The population of the United States was used as a reference group to calculate the damage to the existence value of the affected species and ecosystems using contingent valuation methods. Exxon was ultimately ordered to pay $5 billion to compensate the people of Alaska for their losses (Van der Straaten 1998). This huge figure was a consequence of the high income of the US population. If the same accident had occurred in Siberia, where salaries are lower, the outcome would certainly have been different.

This example shows that contingent valuation methods simply provide information about the preferences of a particular group of people but do not necessarily reflect the ecological importance of ecosystem goods and services. Moreover, the outcome of cost–benefit analysis depends on which individual willingnesses to pay or willingnesses to accept are included in the cost–benefit analysis. If the quality of the Mississippi River is at issue, should the analysis be restricted to US citizens living close to the river, or should the willingness to pay of Californians and New Yorkers be included too? According to Krutilla's (1967) definition of existence value, for many environmental goods and ecological services that may ultimately affect ecosystem integrity at the global level, the preferences of the entire human population should potentially be considered in the analysis. Because practical reasons obviously preclude doing so, contingent valuation methods will inevitably only provide information about the preferences of specific groups of people. For many of the ecological services that may be considered the heritage of humanity, contingent valuation methods analyses performed locally in a particular economic situation should be extrapolated only with great caution to other areas. The process of placing a monetary value on biodiversity and ecosystem functioning through nonuser willingness to pay is performed in the same way as for user willingness to pay, but the identification of people who do not use an environmental good directly and still have a legitimate interest in its preservation is problematic.

Significance of the simulated market. Contingent valuation methods are contingent on a market that is constructed or simulated, not real. It is difficult to believe in the efficiency of what Adam Smith called the "invisible hand" of the market for a process that is the artificial production of economic advisors and does not possess the dynamic feedback that characterizes real competitive markets. Is it even possible to simulate a market where units of biodiversity are bought and sold? As Friend (1997) stated, "these contingency evaluation methods (CVM) tend to create an illusion of choice based on psychology (willingness) and ideology (the need to pay) which is supposed, somewhat mysteriously, to reflect an equilibrium between the consumer demand for and producer supply of environmental goods and services."

Many additional criticisms of pricing ecological services are more familiar to biologists (e.g., Ehrenfeld 1988, Rees 1998). For many ecological services, there is simply no possibility of technological substitution. Moreover, the precise contribution of many species is not known, and it may not be known until the species is close to extinction (see O'Neill and Kahn 2000). In addition, specific ecosystem services, as evaluated by Costanza et al. (1997), should not be separated from one another and valued individually (Norgaard et al. 1998) because the importance of any piece of biodiversity cannot be determined without considering the value of biodiversity in the aggregate. And finally, the use of marginal value theory may be invalidated by the erratic and catastrophic behavior of many ecological systems (Holling 1992), resulting in potentially detrimental effects on the health of humans, the productivity of renewable resources, and the vitality and stability of societies themselves (Rees 1998).

Despite the efforts of many economists, we believe that some goods and services, especially those related to ecosystems, cannot reasonably be given a monetary value, although they are of great value to humans. Economists coined the term "intangibles" to define these goods. Cost–benefit analysis cannot easily deal with intangibles. As Nijkamp (1977) wrote, more than 20 years ago, "the only reasonable way to take account of intangibles in the traditional cost–benefit analysis seems to be the use of a balance with a debit and a credit side in which all intangible project effects (both positive and negative) are represented in their own (qualitative or quantitative) dimensions" as secondary information (p. 147). In other words, the result of cost–benefit analysis is primarily a single number, the net monetary benefit that comprises all the effects that can be sensibly converted into monetary returns and costs.

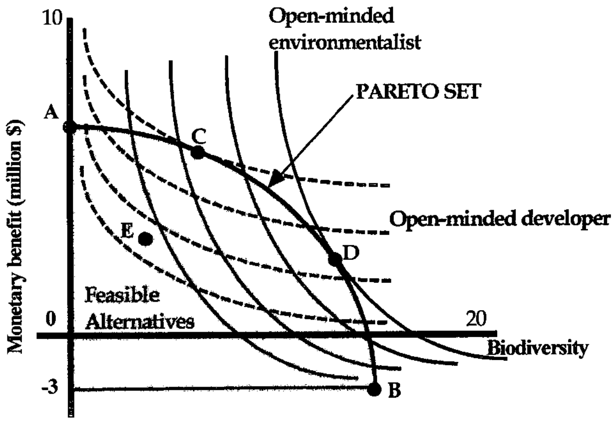

Figure 2. Possible outcomes (in terms of biodiversity and monetary net benefit) of hypothetical mining projects located in a natural area. Mining the whole area by the most cost-effective technology corresponds to maximum monetary benefit and no biodiversity (point A). Not developing the project corresponds to no profit and maximum biodiversity (point B). The gray region corresponds to the set of projects that are feasible, given available technology. There is a tradeoff between profit and biodiversity along the feasibility boundary, which coincides with the Pareto set. Indifference curves (see Figure 1) are shown for a developer who has some consideration for environmental problems (dashed lines) and for an environmentalist who cares also about monetary benefits (solid lines). Indifference curves closer to the origin correspond to lower utility. The developer would favor project C because it maximizes his or her welfare; the environmentalist would favor project D for the same reason. Alternatives on the Pareto set that fall between C and D represent the real dilemma for the deciding authority. By contrast, cost–benefit analysis would give a green light to project E because it corresponds to positive net monetary benefit.

Commensurability of different objectives and multicriteria analysis

Cost–benefit analysis includes intangibles in the decision-making process only as ancillary information, with the main focus being on those effects that can be converted to monetary value. This approach is not a balanced solution to the problem of making political decisions that are acceptable to a wide number of social groups with a range of legitimate interests. We also recognize that this argument against cost–benefit analysis is not new: For example, Herfindahl and Kneese (1974) wrote that "a final approach [to the problem of multiple planning objectives] is that of viewing the various possible objectives of public policy as being substantially incommensurable on any simple scale and therefore necessitating the generation of various kinds of information, not summable into a single number, as a basis for political decision" (p. 223). It is unfortunate that more than 20 years later, the opinion of these distinguished economists is still overlooked.

However, even if the attempt to put a price on everything is abandoned, it is not necessary to give up the attempt to reconcile economic issues with social and environmental ones. Social scientists long ago developed multicriteria techniques to reach a decision in the face of multiple different and structurally incommensurable goals. The most important concept in multicriteria analysis was actually conceived by an Italian economist, Vilfredo Pareto, at the end of the nineteenth century. It is best explained by a simple example. Suppose that a natural area hosting several rare species is a target for the development of a mining activity. Alternative mining projects can have different effects in terms of profits from mining (measured in dollars) and in terms of sustained biodiversity (measured in suitable units, for instance, through the Shannon index). Profit from mining can be corrected using welfare economics to include those environmental and social effects that can be priced (e.g., the benefit of providing jobs to otherwise unemployed people, the cost of treating lung disease of miners, and the cost of the loss of the tourists who used to visit the natural area).

Figure 2 summarizes the consequences of all of the alternative mining projects: Each point in the gray region represents the biodiversity and the net monetary benefit corresponding to a certain feasible alternative. There is an obvious tradeoff between monetary benefit and biodiversity that is reflected in the downsloping boundary of the gray region: Even if the most environmentally friendly technology is employed, mining a larger area will yield a greater profit but result in lower biodiversity. Consequently, all of the projects interior to the gray region would be discarded by a rational decision-maker because they are nonoptimal. In fact, there are projects lying on the boundary (the so-called Pareto set) that are better in terms of both monetary benefit and biodiversity. Therefore, the Pareto set contains all of the rational alternatives, including that of using the most cost-effective mining technology across the whole area, effectively destroying it (Figure 2, decision A) and that of preserving the whole area as it is (Figure 2, decision B).

However, compromise decisions will typically lie in the central part of the Pareto set. This point can be shown by using the concept of indifference curves (Figure 1). An open-minded developer—for whom biodiversity plays a role in his or her utility function but monetary benefit is more important—will be characterized by the dashed indifference curves in Figure 2 and will favor decision C, which maximizes his or her welfare. An open-minded environmentalist—whose welfare is more sensitive to changes in biodiversity than in monetary benefit—will have the solid indifference curves and will prefer decision D. Thus, in practice, a rational decision-maker has to make a choice among the alternatives lying on the Pareto set between points C and D.

The methods of multicriteria analysis are intended to assist the decision-maker in choosing among the alternatives belonging to the Pareto set (a task that is particularly difficult when there are several incommensurable objectives, not just two). Nevertheless, the initial step of determining the Pareto alternatives is of enormous importance, for three reasons. First, the Pareto set makes perfect sense even if there is no way of pricing a certain environmental good because each objective can be expressed in its own proper units without reduction to a common scale. Second, the determination of all the feasible alternatives and, subsequently, of the Pareto alternatives requires the joint effort of a multidisciplinary team that includes, for example, economists, engineers, and biologists and that must predict the effects of alternative decisions on all of the different environmental and social components to which humans are sensitive and which, therefore, deserve consideration. Third, the determination of the Pareto set allows the objective elimination of inadequate alternatives because the Pareto set is independent of the subjective perception of welfare (as described, for example, by indifference curves). Therefore, the Pareto set in essence describes the tradeoff between the various incommensurable objectives when every effort is made to achieve the best results in all respects; the attention of the authority that must make the final decision is thus directed toward genuine potential solutions because nonoptimal decisions have already been discarded.

It should be noted that a cost–benefit analysis does not elicit tradeoffs between incommensurable goods because it also gives a green light to projects that are not on the Pareto boundary, provided that the benefits that can be converted into a monetary scale exceed the costs. For instance, in the example of Figure 2, cost–benefit analysis, being unable to provide a price for biodiversity units, would consider alternative E to be acceptable because it corresponds to a positive monetary benefit. The information that decision E would reduce biodiversity to well below the original level is auxiliary to the analysis. This same example, however, clarifies that cost–benefit analysis has an important role to play within multicriteria techniques; in fact, for each possible level of biodiversity, a cost–benefit analysis will indicate which project provides the maximum monetary benefit. In this way, it is possible to construct the Pareto set, which conveys the tradeoff between dollars and biodiversity. Cost–benefit analysis, however, is not useful for eliciting the tradeoffs between two incommensurable goods, neither of which is monetary. For instance, there might be a conflict between the goals of preserving wildlife within a populated area and minimizing the risk that wild animals are vectors of dangerous diseases. A multicriteria analysis can describe this tradeoff, whereas a cost–benefit analysis cannot.

Another philosophical point concerning the issue of commensurability is the question of implicit pricing. Economists often argue that to make a decision is to put an implicit price on such intangibles as human life or aesthetics and, therefore, to reduce their value to a common scale (as pointed out also by Costanza et al. 1997). This process of implicit pricing is summarized in Figure 3, which reconsiders the example of Figure 2. Suppose that the decision-maker, after pondering the pros and cons of the different projects on the Pareto boundary, reaches a final decision, which is represented by the decision point in Figure 3. At this stage of the analysis, it is possible to do the exercise of calculating the implicit price of one unit of biodiversity or the implicit ecological value of $1. For instance, in the example in Figure 3, the price of each unit of biodiversity, which is calculated by drawing the tangent line to the Pareto boundary through the decision point and determining the absolute value of its slope, is $0.5 million (10/20 = 0.5). In fact, if each unit of biodiversity had been given a price of $0.5 million, then the choice of the project by the decision-maker could be interpreted as the decision that maximizes an artificially defined total net benefit, where

total net benefit = monetary benefit + 0.5 × biodiversity

This argument is easily understood by considering that all of the alternative projects providing equal total benefit lie on straight lines parallel to the tangent. Lines closer to the origin of the coordinate axes define alternatives with lower total benefits, whereas lines farther away define projects with larger total benefits. Among the feasible alternatives, the choice of the decision-maker corresponds to maximum total net benefit because it lies on the line that is farthest from the origin. This maximum total net benefit is $10 million (or 20 biodiversity units).

This argument for implicit pricing has one basic flaw, however: It is a purely academic statistical abstraction or descriptive convention that can only be done a posteriori (i.e., after a decision is made). As pointed out by Sagoff (1998), the retrospective interpretation of many governmental regulations would give human life any value ranging between $20 (motorcycle helmet requirements) and $20 billion (benzene emission control at rubber-tire manufacturing plants), depending on the contingent situation and the perception of the risk! Given this enormous range of variability (nine orders of magnitude), it is doubtful that retrospective studies of past decisions can be useful for future decisions. Also, as was pointed out in the 1960s, the application of implicit values used in the past would tend to perpetuate erroneous decisions. Indeed, pricing techniques must be used a priori if they are to be useful for arriving at a decision. However, if they are used a priori, the argument of an implicit price for intangibles cannot be invoked as a scientific foundation for pricing techniques.

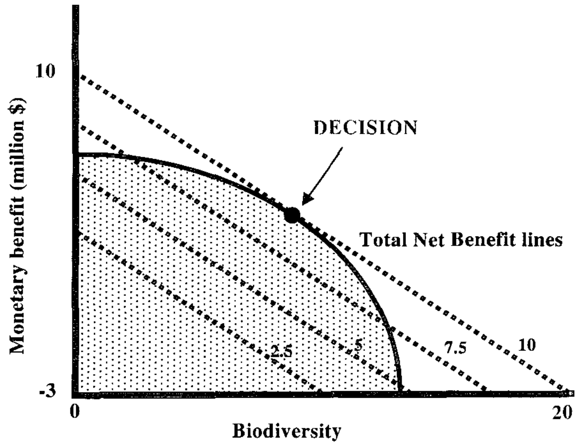

Figure 3. The argument of implicit pricing. Suppose that the authority for the example of Figure 2 has reached a final decision belonging to the Pareto set. The decision is equivalent to maximizing a measure of total net benefit, which is calculated as monetary benefit + 0.5 × biodiversity. In fact, projects lying on each dashed line correspond to equal total benefit (given by the numbers that label the lines). The project corresponding to the tangency point with the Pareto boundary is thus the one providing the maximum total benefit. Therefore, a posteriori we can say that each unit of biodiversity is implicitly given the price of $0.5 million.

Environmental impact assessment and multiattribute decision-making

Because of the flaws of cost–benefit analysis, many countries have taken a different approach to decision-making through the use of environmental impact assessment legislation (e.g., the United States in 1970, with the signing of the National Environmental Policy Act, NEPA; France in 1976, with the act 76/629; the European Union in 1985, with the directive 85/337). Environmental impact assessment procedures, if properly carried out, represent a wiser approach than setting an a priori value of biodiversity and ecosystem services because these procedures explicitly recognize that each situation, and every regulatory decision, responds to different ethical, economic, political, historical, and other conditions (Dietz and Stern 1998) and that the final decision must be reached by giving appropriate consideration to several different objectives. As Canter (1996) noted, all projects, plans, and policies that are expected to have a significant environmental impact would ideally be subject to environmental impact assessment.

The breadth of goals embraced by environmental impact assessment is much wider than that of cost–benefit analysis. Environmental impact assessment provides a conceptual framework and formal procedures for comparing different alternatives to a proposed project (including the possibilities of not developing a site, employing different management rules, or using mitigation measures); for fostering interdisciplinary team formation (see Barrett and Odum 2000) to investigate all possible environmental, social, and economic consequences of a proposed activity; for enhancing administrative review procedures and coordination among the agencies involved in the process; for producing the necessary documentation to enhance transparency in the decision-making process and the possibility of reviewing all the objective and subjective steps that resulted in a given conclusion; for encouraging broad public participation and the input of different interest groups; and for including monitoring and feedback procedures. Classical multiattribute analysis can be used to rank different alternatives belonging to the Pareto set (e.g., Keeney and Raiffa 1976). Ranking usually requires the use of value functions to transform environmental and other indicators (e.g., biological oxygen demand or animal density) to levels of satisfaction on a normalized scale, and the weighting of factors to combine value functions and to rank the alternatives (Barrett 1985). These weights explicitly reflect the relative importance of the different environmental, social, and economic compartments and indicators.

A wide range of software packages for decision support can assist experts in organizing the collected information; in documenting the various phases of EIA; in guiding the assignment of importance weights; in scaling, rating, and ranking alternatives; and in conducting sensitivity analysis for the overall decision-making process (Barrett 1985). This last step, of testing the robustness and consistency of multiattribute analysis results, is especially important because it shows how sensitive the final ranking is to small or large changes in the set of weights and value functions, which often reflect different and subjective perspectives. It is important to stress that, although the majority of environmental impact assessments have been conducted on specific projects, such as road construction or the location of chemical plants, there is no conceptual barrier to extending the procedure to evaluation of plans, programs, policies, and regulations. In fact, according to NEPA, the procedure is mandatory for any federal action with an important impact on the environment. The extension of environmental impact assessment to a level higher than a single project is termed "strategic environmental assessment" and has received considerable attention (Lee and Walsh 1992).

Conclusions

An impressive literature is available on environmental impact assessment and multiattribute analysis that documents the experience gained through 30 years of study and application. Nevertheless, these studies seem to be confined to the area of urban planning and are almost completely ignored by present-day economists as well as by many ecologists. Somewhere between the assignment of a zero value to biodiversity (the old-fashioned but still used practice, in which environmental impacts are viewed as externalities to be discarded from the balance sheet) and the assignment of an infinite value (as advocated by some radical environmentalists), lie more sensible methods to assign value to biodiversity than the price tag techniques suggested by the new wave of environmental economists. Rather than collapsing every measure of social and environmental value onto a monetary axis, environmental impact assessment and multiattribute analysis allow for explicit consideration of intangible nonmonetary values along with classical economic assessment, which, of course, remains important. It is, in fact, possible to assess ecosystem values and the ecological impact of human activity without using prices. Concepts such as Odum's eMergy (1996) and Rees' ecological footprint (Wackernagel and Rees 1996), although perceived by some as naive, may aid both ecologists and economists in addressing this important need.

To summarize our viewpoint, economists should recognize that cost–benefit analysis is only part of the decision-making process and that it lies at the same level as other considerations. Ecologists should accept that monetary valuation of biodiversity and ecosystem services is possible (and even helpful) for part of its value, typically its use value. We contend that the realistic substitute for markets, when they fail, is a transparent decision-making process, not old-style cost–benefit analysis. The idea that, if one could get the price right, the best and most effective decisions at both the individual and public levels would automatically follow is, for many scientists, a sort of Panglossian obsession. In reality, there is no simple solution to complex problems. We fear that putting an a priori monetary value on biodiversity and ecosystem services will prevent humans from valuing the environment other than as a commodity to be exploited, thus reinvigorating the old economic paradigm that assumes a perfect substitution between natural and human-made capital (Ehrenfeld 1988). As Rees (1998) wrote, "for all its theoretical attractiveness, ascribing money values to nature's services is only a partial solution to the present dilemma and, if relied on exclusively, may actually be counterproductive" (p. 52).

Acknowledgments

The work has been in part supported by Consiglio Nazionale delle Ricerche. We thank Rebecca Chasan, Gary Barrett, Helena Muller-Landau, and two referees for the very useful comments on a first draft of our manuscript.

References cited

Adams J. 1992. Horse and rabbit stew. Pages 65–73 in Coker A, Richards C, eds. Valuing the Environment: Economic Approaches to Environmental Evaluation. New York: John Wiley & Sons.

Barrett GW. 1985. A problem-solving approach to resource management. BioScience 35: 423–427.

Barrett GW, Odum EP. 2000. The twenty-first century: The world at carrying capacity. BioScience 50: 363–368.

Canter LW. 1996. Environmental Impact Assessment. 2nd ed. London: McGraw-Hill.

Chilton S, Burton T, Jones M, Loomes G. 1998. A qualitative examination of preference reversals. Paper presented at the First World Congress of Environmental and Resource Economists; June 25–27 1998; Venice, Italy. <www.feem.it/worldcongress/abs1/chilton. html>.

Ciriacy-Wantrup SV. 1947. Capital returns from soil conservation practices. Journal of Farm Economics 29: 1181–1196.

Costanza R, et al. 1997. The value of the world's ecosystem services and natural capital. Nature 387: 253–260.

Daily GC, ed. 1997. Nature's Services: Societal Dependence on Natural Ecosystems. Washington (DC): Island Press.

Dietz T, Stern PC. 1998. Science, values, and biodiversity. BioScience 48: 441–444.

Dupuit JJ. 1844. De la mesure de l'utilité de travaux publics. Annales des Ponts et Chaussées. Série 8(2).

Ehrenfeld D. 1988. Why put a value on biodiversity. Pages 212–216 in Wilson EO, Peter FMP, eds. Biodiversity. Washington (DC): National Academy Press.

Friend AM. 1997. Unearthing the true exchange rate between the physical-valued and the monetary-valued environment: Concepts and methods for ecological pricing. Paper presented at the Second Biennial Meeting of the Canadian Society for Ecological Economics; October 6–7 1997; Hamilton, Ontario.

Hanemann WM. 1991. Willingness to pay and willingness to accept: How much can they differ? American Economic Review 81: 635–647.

Herfindahl OC, Kneese AV. 1974. Economic Theory of Natural Resources. Columbus (OH): Charles E. Merrill.

Holland A, Cox JR. 1992. The valuing of environmental goods: A modest proposal. Pages 65–73 in Coker A, Richards C, eds. Valuing the Environment: Economic Approaches to Environmental Evaluation. New York: John Wiley & Sons.

Holling CS. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. Ecological Monographs 62: 447–502.

Keeney RL, Raiffa H. 1976. Decisions with Multiple Objectives: Preferences and Value Tradeoffs. New York: John Wiley & Sons.

Krutilla J. 1967. Conservation reconsidered. American Economic Review 56: 777–786.

Lee N, Walsh F. 1992. Strategic environmental assessment: An overview. Project Appraisal 7: 126–136.

MacMillan D, Smart TS, Thorburn A. 1998. A comparison of real and hypothetical donations when incentives to free-ride and over-bid are equivalent across surveys. Paper presented at the First World Congress of Environmental and Resource Economists; June 25–27 1998; Venice, Italy. <www.feem.it/worldcongress/abs1/mcmill.html>.

Nijkamp P. 1977. Theory and Application of Environmental Economics. Amsterdam: North-Holland.

Norgaard RB, Bode C, Values Reading Group. 1998. Next, the value of God, and other reactions. Ecological Economics 25:37–39.

Odum HT. 1996. Environmental Accounting: Emergy and Environmental Decision Making. New York: John Wiley & Sons.

O'Neill RV, Kahn JR. 2000. *Homo economus* as a keystone species. BioScience 50: 333–337.

Pearce DW. 1993. Economic Values and the Natural World. Cambridge (MA): MIT Press.

______. 1998. Auditing the Earth: The value of the world's ecosystem services and natural capital. Environment 40: 23–28.

Pimentel D, Wilson C, McCullum C, Huang R, Dwen P, Flack J, Tran Q, Saltman T, Cliff B. 1997. Economic and environmental benefits of biodiversity. BioScience 47: 747–757.

Portney PR. 1994. The contingent valuation debate: Why economists should care. Journal of Economic Perspectives 8: 3–17.

Prewitt RA. 1949. The Economics of Public Recreation—An Economic Survey of the Monetary Evaluation of Recreation in the National Parks. Washington (DC): National Park Service.

Rees W. 1998. How should a parasite value its host? Ecological Economics 25: 49–52.

Sagoff M.1998. Can we put a price on nature's services? Report from the Institute of Philosophy and Public Policy 17 (3). <www.puaf.umd.edu/ippp/nature.htm> (24 Jan 2000).

Simpson RD. 1998. Economic analysis and ecosystems: Some concepts and issues. Ecological Applications 8: 342–349.

Tunstall RK, Coker A. 1992. Survey-based valuation methods. Pages 104–126 in Coker A, Richards C, eds. Valuing the Environment: Economic Approaches to Environmental Evaluation. New York: John Wiley & Sons.

Van der Straaten J. 1998. Is economic value the same as ecological value? Paper presented at the Seventh International Congress of Ecology (INTECOL); 19–25 Jul 1998; Florence, Italy.

Wackernagel M, Rees W. 1996 Our Ecological Footprint: Reducing Human Impact on the Earth. Philadelphia: New Society Publishers.

The natural areas of our country are shrinking fast. And some types, once extensive, have disappeared altogether.

The same is true of the wildlife species, both plant and animal, which depend on them.

By preserving just a fraction of America's undeveloped land, we can still save a small but reasonably complete representation of our natural heritage.

This is no dream. The Nature Conservancy is doing it.

The Conservancy scientifically searches out, acquires and manages the most critical natural lands to keep them and their flora and fauna undisturbed.

We pay the bill with private funds, contributed by our 300,000 members, by foundations and corporations.

Since 1951, The Nature Conservancy has brought under protection more than 2.5 million acres of the best remaining forests, wetlands, beaches and prairies.

We own over 900 preserves, most of them open to the public.

But the cost is high, and the job only half done.

We need to protect another two million acres by the year 2000 or see numerous habitats lost and countless species face total extinction.

With your help, that need can be met. To become a member and to learn about our projects in your state, write: The Nature Conservancy, 1800 North Kent Street, Box CD0004, Arlington, Virginia 22209. Or call (703) 841-5349.

The Nature Conservancy isn't out to save the world. Just the most important parts.