Valuation of Biodiversity Benefits

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Valuation of Biodiversity Studies

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FOREWORD

Environmental problems in general and biodiversity degradation in particular are related to the failure of markets to properly value environmental services and regulate their use. Valuation methods are thus needed to assist policy makers in identifying priorities and evaluating trade-offs. The importance of valuation is widely understood in academia and is increasing in policy making and other fora. For example, the Convention on Biological Diversity (CBD), through the Conference of the Parties (COP), recognises that "economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures". Furthermore, it encourages the Parties to "take into account economic, social, cultural and ethical valuation in the development of relevant incentive measures" (CBD COP Decision IV/10). The menu of valuation techniques is vast and draws on several disciplines, with economics playing a major role.

The OECD has been actively collaborating with the CBD on *Incentive Measures* (CBD, Article 11), under which valuation is considered. The OECD Working Group on Economic Aspects of Biodiversity has been striving to "fulfil the need of OECD Member countries for pragmatic policy guidance regarding the implementation of incentives measures for the conservation and the sustainable use of biological diversity". It produced the *Handbook of Incentives Measures For Biodiversity: Design and Implementation* (OECD 1999), and identified valuation as a major area where additional work is needed.

In this context, an international workshop was organised at the OECD on the theme of "Benefit Valuation of Biodiversity Resources" (18-19 October 1999). The objectives were to identify the policy relevance of valuation techniques as applied to environmental policy making and the design of policy instruments, and to discuss the different valuation techniques, the recent advances, the synergies among them, and their applicability to OECD Member countries. Moreover, the workshop identified the need to further study the interactions between ecological and economic valuation techniques, resulting in the commissioning of an additional paper on the matter. This book edited by Dan Biller and Rosalind Bark contains the results of the workshop.

The Secretariat wishes to acknowledge the comments of the workshop participants and the contributions of the authors. The views expressed are those of the workshop contributors. The book is published under the responsibility of the Secretary-General of the OECD.

TABLE OF CONTENTS

PART 1

CHAPTER 1:
by Dan BILLER and Rosalind BARK

RATIONALE, SUMMARY AND CONCLUSIONS

Background and workshop objectives

Quantitative Valuation techniques are increasingly being applied on environmental issues in general and biodiversity resources in particular. Although fairly common in the environmental economics literature, valuation techniques have remained somewhat peripheral to environmental policymaking on major issues. As part of its work program on the valuation of biodiversity, the OECD organised a workshop to discuss the policy relevance of valuation techniques as applied to environmental policy making, the design of policy instruments, recent progress in resolving problems with techniques currently used, and the development of alternative valuation techniques.

The papers presented in this volume neither cover all the economic valuation techniques described in the literature, nor all of the theoretical and implementation issues involved. For instance, travel cost and hedonic pricing techniques are not discussed, in part because such overlook non-use values, which are particularly important in the economic evolution of biodiversity. Moreover, they are by now well understood and less controversial than, say, contingent valuation techniques. The papers in this report define the components of biodiversity value, describe valuation methods to quantify them, and discuss some contentious issues in the application of valuation techniques. Contingent valuation techniques (CV) are essential to the valuation of biodiversity because of the large non-use value component (see Figure 1.1); however, their methodologies are complex and they are costly to undertake. The three final papers in the report summarise some non-monetary valuation techniques that can be used by policy makers when non-use values, are significant.

The report

The compendium is organised in four parts. Part 1 introduces the concepts of biodiversity and biological resources and discusses how to quantify the value associated with changes in the level or quality of global environmental services at the local and national levels. Part 2 looks at the application of economic valuation to environmental policy in the United States and Europe and discusses some of the reasons why valuation techniques are not more widely used. In particular, the main issues in benefit transfer — the transfer of values found in the literature to a new policy site — are discussed. Part 3 and 4 examine some alternative methodological approaches to the valuation of biodiversity: conjoint analysis and the development of an ecological indicator (i.e. non-monetary) of ecosystem value. Part 4 also provides an initial attempt to integrate economic and ecological valuation.

Why value biodiversity?

There are three main reasons for undertaking economic valuation of biodiversity and biological resources: to facilitate cost-benefit analysis, for integration with the system of national accounts, i.e., "green" accounting, and for the proper pricing of biological resources (Pearce, Chapter 2). Cost-benefit analysis (CBA) of specific investments and policies, that properly incorporate environmental costs and benefits, are essential to enable policy makers to choose the investment or policy option that maximises total net benefits to society. Environmental accounting adjusts the standard gross domestic product (GDP) measure to take into account any depreciation in the environmental base of the economy. Finally, in the absence of markets for all of the benefits provided by biodiversity the valuation of changes in the level or quality of these services is small and may even be zero. Therefore the proper valuation of the loss due to marginal changes in biological resources or biological diversity is essential in order to level the playing field between conservation and economic development.

The role of environmental economics in biodiversity policy

Dixon and Pagiola (Chapter 3) poses two questions that are at the core of biodiversity policy making: "What role can economics play in understanding this dilemma and illuminating the trade-offs inherent in biodiversity conservation decisions, and what are the policies that are likely to help resolve this conflict?"

Valuing biodiversity

Pearce convincingly argues that demonstrating the value of biodiversity is a fundamental step in conservation, because "the pressures to reduce biodiversity are so large that the chances that we *will* introduce incentives (for conservation) without demonstrating the economic value of biodiversity are much less than if we do engage in valuation." Environmental valuation techniques can be used to place a value on changes in biodiversity, in order to level the playing field for conservation in policy making that would otherwise be dominated by the financial benefits of land use conversion. In this way valuation techniques (and other non-monetary approaches, see Chapters 6 and 7) illuminate the trade-offs inherent in biodiversity conservation decisions. Furthermore, Pearce suggests that "We learn so much from the *process* of valuation, especially if we adopted stated preference techniques. We learn what people care about, what their motives are for conservation, what their reactions would be to different management objectives." The process itself, which can be supplemented by expert opinion and information, may be a critical requirement for successful conservation.

"One of the difficult issues in placing an economic value on biodiversity is determining exactly what the object of value is" (Pearce, Chapter 2). Pearce makes a clear distinction between biological resources and biological diversity: "A biological resource is simply a given example of a gene, species, or ecosystem. Biological diversity refers to the *variability* of biological resources." He contends that many valuation studies do not make this distinction and value biological resources not biological diversity. The consequence is that many programmes to support biodiversity may not be optimal. For example, programmes to save single species, such as the spotted owl or the tiger, may in biodiversity terms be sub-optimal, if these species have a close genetic relative that is not endangered. In addition, Pearce states that to be relevant to policy making economic valuation of biodiversity should measure marginal or discrete changes in the availability of biodiversity. Land use development, increased tourism, increased pollution, etc. affect the future flow of services from an ecosystem and it is these changes that should be valued.

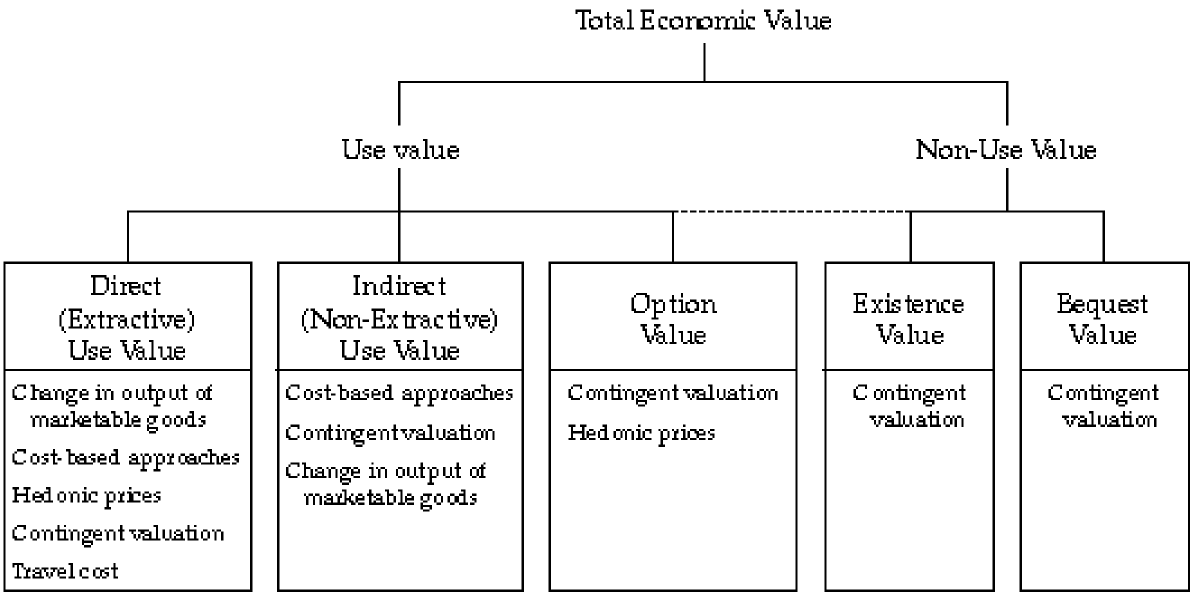
Table 1.1 shows the extinction probabilities of crane species. Conservation policies based on this data alone would probably support the protection of the Siberian and whooping crane habitats, as these are the most endangered crane species. However, Solow *et al.* (1993) show that if the policy objective is to maximise the (expected) level of diversity, then the optimal programme requires that the Siberian, white-naped and black-necked cranes should be conserved because they are more distantly related than are Siberian and whooping cranes. Focusing on the most endangered, thus does not necessarily minimise biodiversity loss. The reason for this is that the genetic distance between the endangered species and at least one of the 'safe' species is small. Minimising the probability of the number of species lost is not the same as minimising the value of potentially lost biodiversity.

Table 1.1 Biodiversity of cranes

	Species	Extinction probability
Endangered	Siberian	0.9
	Whooping	0.9
Vulnerable	Japanese	0.7
	Hooded	0.7
	White-naped	0.7
Indeterminate:	Black-necked	0.5
Safe:	All others	0.0

As indicated by Dixon and Pagiola, the economic value of a forest or a species of big cat is made up of use and non-use values. Current environmental valuation techniques can be used to elicit these values. *Stated preference* techniques, for instance, contingent valuation, contingent ranking and choice experiments, elicit value from surveys, whilst *revealed preference* techniques value traces implicit in observed behaviour and which are revealed by modelling behaviour using hedonic price, travel cost or averting behaviour models.

Figure 1.1 Total economic value: use and non-use



Source: Dixon and Pagiola (Chapter 3).

In Chapter 6, Kahn *et al.* defines ecological services: "ecological services are the functions that ecosystems perform that provide the basis for all ecological and economic activity, and include carbon sequestration, nitrogen fixation, hydrological cycles, nutrient cycles, biodiversity, production of oxygen, maintenance of global climate, soil formation and primary productivity." The valuation of ecological services and of non-use values is difficult, but nonetheless crucial because it is likely that these two components form a large part of the total value of biodiversity. Although environmental valuation techniques used to value changes in the level and quality of biodiversity are not yet capable of valuing all the benefits of biodiversity, they are important for fuller analyses of the economic values of biodiversity conservation. In response to the difficulties in valuing non-use and ecological service values, alternative methodologies have been developed to aid decision-makers in biodiversity and biological resource policy making (see Part 3).

Capturing biodiversity values

The process of valuation involves two steps: the *demonstration of value* and the *quantification* of value. Without the second step, land users and policy makers will not adequately take account of biodiversity conservation.

Biodiversity provides services at different levels. Local biodiversity benefits could include products with harvest potential and crop pollination. Hydrological regulation is an example of national level benefits and two global benefits are global carbon sequestration and genetic information. Incentives to preserve biodiversity at all these levels are constantly challenged by the financial incentives of land use development. In contrast, biodiversity is a non-market good that is difficult to value and the result often is that conservation is 'under-provided' by the market.

Valuation is only the first step. As important is how economic agents may capture biodiversity values and equitably distribute the gains. In many cases nationally important benefits of biodiversity may be ignored by local decision-makers. For example, it is unlikely that farmers when deciding whether or not to extend cultivation into forestland account for the wider watershed benefits of the forest that will be lost. There is a pressing requirement both to find ways to demonstrate biodiversity benefits at all levels and also to capture these benefits, otherwise incentives for land use change will dominate those for conservation. "In an effort to bridge the gap between local costs and global benefits, new ways are required to help increase the amounts that individuals or countries are willing to pay for protection of various national or global (or non-local) benefits" (Dixon and Pagiola).

One measure to bridge the gap between biodiversity benefits and the benefits of land use development is to demonstrate the value of various non-local (national and global) benefits and to make payments for these environmental services to land users. By creating markets in this way biodiversity benefits are properly priced and land users will have direct incentives to include these services in their land use decisions, resulting in more socially optimal land uses. However, Dixon and Pagiola points out that policies based on environmental services provided may not always result in biodiversity conservation. For example, logging forests in a sustainable way may provide the same hydrological services as an intact primary forest but biodiversity will be reduced.

'Capturing' international biodiversity values

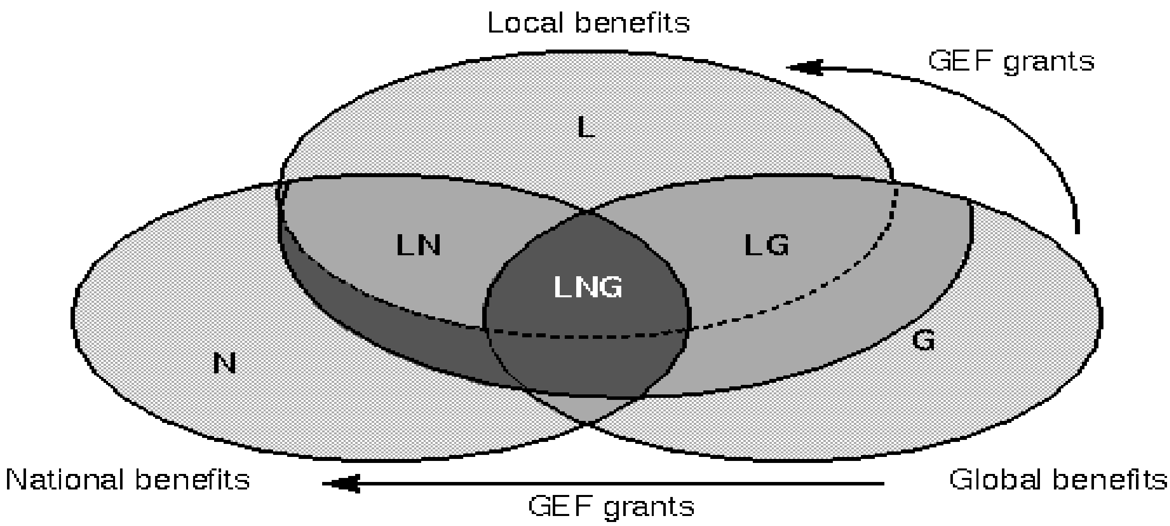
Dixon and Pagiola and Pearce examine how to quantify global use and non-use values for biodiversity and turn these into real resource flows for the local agents and national governments. Pearce identifies five categories of value for global biodiversity values:

- direct global use – tourism;
- direct global use – genetic store for agriculture and pharmaceuticals;
- indirect global use – ecosystem resilience;
- indirect global use – ecosystem services;
- non-use, including option value and quasi-option values.

Dixon and Pagiola and Pearce papers analyse the option of raising funds for biodiversity conservation through ecotourism, and identify a number of difficulties with this approach. Ecotourism receipts are very small, the tourism itself has to be environmentally sustainable, and it can be difficult to ensure that local populations have a stake in, and capture the potential income from ecotourism so as to have incentives for conservation. Another example is the direct global value of biodiversity as a genetic store for new or modified pharmaceuticals. Pearce notes that large variations in value are reported in the literature and that this can be explained by methodological differences. For example, many studies compute an average value for genetic material rather than the marginal value, earlier studies ignore that substitute drugs may be available, imperfectly most studies compute only the private not the social value of drugs, and only some studies take into account the impact of the competitive structure of the pharmaceutical industry.

Dixon and Pagiola illustrate how the Global Environment Facility or international NGOs can support biodiversity conservation at the local level by paying grants for global biodiversity benefits. Grants from such organisations in effect pay local land users for some of the global benefits that stem from biodiversity at the local level. In this way grants for global biodiversity services raise the opportunity cost of land use conversion and thus increase the incentive for conservation.

Figure 1.2 Using GEF grants to increase local and national incentives to conserve biodiversity



Real world examples of payments for environmental services to land users have been pioneered in Costa Rica. For example international payments have been made for forest conservation, sustainable logging and plantations for carbon sequestration and other benefits. The Global Environment Facility (GEF) provides grant funding to countries to undertake activities that generate global benefits. International NGOs and some large pharmaceutical companies have also financed the acquisition or management of protected areas. Further, the government has implemented a local level programme — an 'environmentally adjusted water tariff' to protect watershed areas upstream. At present the payments are very small and reflect in part the uncertainty in valuing all the benefits from biodiversity.

Practical Applications

This book includes two papers on the practical application of biodiversity valuation, including the appropriate use of benefits transfer techniques, for local and national policy making.

Valuation in policy making and the limitations of benefit transfer techniques

Navrud (Chapter 4) describes four policy situations where environmental valuation studies are used in Europe and the United States:

- as an input into cost benefit analyses of investment projects (e.g. the UK and transport investments) and other policies (e.g. CBAs of United States federal environmental regulations);

- to determine the environmental externality costs of an activity, for example the environmental damage caused by transportation (the European Commission's ExternE project), to use in investment decisions or as the basis for "green" taxation;

- to undertake "green" national accounting (e.g. the European Commission's Green Accounting Research Project) and environmental reporting; and

- to determine compensation payments for natural resource injuries, for example Natural Resource Damage Assessment used in the USA.

Navrud contends that the accuracy requirement increases and thus the applicability of benefits transfer techniques decreases, as one moves down the above list of potential policy uses of valuation studies. International organisations like the World Bank and OECD have produced guidelines on environmental valuation techniques: however, there are no similar guidelines for good practice in benefits transfer techniques.

Navrud identifies five main categories of potential obstacles for the wider use of environmental valuation:

- *cost and time constraints* – undertaking original environmental valuation studies is costly and time consuming, particularly if the study complies with the set of guidelines put forward by the NOAA panel in the United States;

- *methodological problems* – for example the validity of valuation estimates from Contingent Valuation (CV) studies, in particular for non-use values, an important component of value for biodiversity, has been questioned;

- *information and communication problems* – many decision makers lack the information and skills necessary to be confident with using the results from valuation studies;

- *administrative constraints* – the legal basis for using valuation data is often lacking and guidelines on the use of such techniques are often absent or inadequate; and

- *political constraints*.

Benefit transfer techniques

The consequence of the first obstacle, the cost and time demands for original valuation studies, is the practice of transferring benefit estimates from an original study to a new policy site. The applicability and validity of so-called "benefit transfer" is a key issue in valuation as it offers the potential to reduce the costs of incorporating environmental aspects in policy making without undertaking costly original studies. If benefit transfer methodology could be improved so that valuation estimates could be transferred across countries and across time then the only obstacle to the techniques widespread application would be the creation and maintenance of a database of good quality valuation estimates. In fact, Navrud explains that such a database has been developed — the Environmental Valuation Reference Inventory (EVRI, at http://www.evri.ec.gc.ca/EVRI/) and that it currently contains 700 valuation studies. At present the majority of the studies in the database are from North America, but the number of European and Asian studies captured in the database is steadily increasing. Navrud contends that there is a need to increase the number of valuation studies included in the database and also for undertaking new valuation studies that have been specifically designed for benefit transfer application.

Navrud discusses the two main approaches to benefit transfer, namely unit value transfer and function transfer. In *unit value transfer* the results of an environmental valuation study (e.g. average willingness to pay per household per year) are simply transferred to the policy site. The main problem with this approach is that the values experienced by the average individual in the study site may not be the same as in the policy site, as the result of differences in income, education, preferences, etc., or the sites' attributes may not be the same, or both. In addition, it is unlikely that the starting environmental quality and the same marginal changes in biodiversity or biological resources will be experienced at the original study site and the new policy site. Non-use values are most problematic to transfer between sites, and — unfortunately for biodiversity valuation — these values are thought to have a large weight in the overall value. These added complexities offer elicit strong objections to the use of standard benefit transfer techniques when valuing changes in biodiversity.

A second approach to benefit transfer is to transfer the entire *benefit function*. To implement this approach a researcher transfers a benefit function and estimates of parameters from an original study and collects data on independent variables at the new policy site. The main problem with the benefit function approach results from the exclusion of relevant variables in the bid or demand functions of the original study. One solution is to choose a policy site that is similar to the study site. Alternatively the results from several valuation studies could be combined in a *meta-analysis* to estimate one common benefit function. Meta-analysis enables researchers to test for the significance of the resources characteristics, the features of the samples, and the modelling assumptions. Again a potential problem with the results from meta-analysis is the potential for bias resulting from omitted variables and errors in the specification/measurement of the included variables.

The accuracy of benefits transfer

The validity of these transferred estimates is debated, particularly for goods with a large non-use component, or for complex goods, or both, for instance valuing ecosystems and biodiversity. Studies that compare benefit transfer with original CV studies at the same site can be used to the test the accuracy and validity of benefit transfer, both spatially and intertemporally. Navrud summarises a study by Ready *et al.* (1999) that tests the accuracy of benefits transfer (for the willingness to pay to avoid respiratory symptoms) adjusted for income (using Purchasing Power Indices) in five European countries with the results of individual country contingent valuation studies. The observed transfer error is found to be significant and is also larger than the variability in the original estimate within a country. Similar types of validity tests are needed for the international benefit transfer of use and non-use value for non-health examples, including biodiversity values.

Navrud concludes that the uncertainty revealed by such validity tests means that benefit transfer should be limited to those applications where the demand for accuracy is not high. For example, considerable caution should be exercised in using this technique to determine taxes based on valuations of environmental externalities and for determining compensation payments. He recommends undertaking more original valuation studies and comparisons between studies to test for the appropriateness of benefits transfer techniques. Such studies could also aid in the development of guidelines for undertaking benefit transfer analysis.[1] Such guidelines could encourage best practice in order to improve the validity of benefit transfers techniques.

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[1] For example, Brower (1999) proposes a seven-step protocol for good practice when benefit transfer is used in CBAs.

Benefit Transfer and Contingent Valuation

Navrud discusses some of the constraints on the wider use of environmental evaluation, such as methodological problems and their impact on benefit transfer techniques. In Chapter 5, Santos focuses on the difficulties of valuing biodiversity by using contingent valuation techniques because of the multiple benefits provided by biological resources and the consequences this has for benefits transfer.

Santos describes the basic problem as the potential for valuation bias resulting from the substitution effects between multiple impacts from biological resources policies. Biodiversity policies typically produce multiple benefits and there is evidence that the level of the various services affects people's valuation of each individual benefit provided (the substitution effect). The consequence of substitution effects is that the value for one service of biological resources is reduced if the substitute service is increased and the benefit of providing two services that are substitutes is smaller than the sum of the benefits of providing each service in isolation. The policy problem resulting from this independent valuation and summation (IVS) bias is that there is a risk that the 'wrong' policy option/mix could be chosen.

Santos considers a contingent valuation study of wildlife and landscape conservation benefits of the Pennine Dales Environmentally Sensitive Area in the UK and tests for substitution effects between different conservation policies:

- Policy 1 - stone walls and field barns;
- Policy 2 - flower-rich hay meadows; and
- Policy 3 - small broad-leaved woods.

The survey results were analysed using a model that allowed for negative or positive substitution effects thereby allowing an observation of the sign of the substitution effects, the testing of the statistical significance of these signs, and the measuring of their magnitude and hence the IVS bias. Santos found statistically significant substitution effects between two out of three policy combinations. The significance of these results is that valuing policies together reduces the value of an individual policy provided in isolation. In this example, individually valuing each policy and then aggregating lead to an overestimation (the IVS bias) of the true value of the policy mix by between 10% and 35% for the policy mixes that include two programmes and by 54% for that with all three policies.

The practical significance of the IVS bias in this case was of no consequence, because the policy recommendations, given individual policy costs, were not sensitive to the differences between model-based (corrected) and IVS biased benefit estimates. However, Santos has shown in earlier work that in some cases, policy costs offset model-based benefit estimates but not upwardly biased IVS estimates.

Santos considers some methods to reduce the IVS bias. First the best way to estimate multiple policy benefits is to value any change in a biological resource in one single step, thereby accounting for substitution effects. Second when valuing a single species or service, care must be taken to specify the levels of other services. This has significant consequences for both stated preference (e.g. CV studies) and revealed preference (e.g. travel cost) methods – all other service levels would have to be specified. Furthermore, the implementation consequences of these conclusions for benefit transfer are that the levels of other services in the original study should be similar to those in the policy context. This is highly unlikely, which may lead to important biases when transferring benefit estimates to other policy sites.

Table 1.2 Model-based WTP estimates compared to the corresponding IVS results

Policy-mix P1, P2, P3	Model-based WTP[a]	IVS[a]	IVS bias (in %)
(1, 0, 0)	56.42	56.42	0.00
(0, 1, 0)	57.78	57.78	0.00
(0, 0, 1)	59.15	59.15	0.00
(1, 1, 0)	103.87	114.20	+9.95
(1, 0, 1)	95.49	115.57	+21.03
(0, 1, 1)	86.46	116.93	+35.24
(1, 1, 1)	112.47	173.35	+54.13

Note: [a] values in £ per household per year.

One method to ensure that applied policy analyses that rely on contingent valuation estimates transferred from past studies take into account such bias is sequential cost-benefit analysis. Substitution effects indicate that the value of a particular policy component depends on which other components are included in the policy mix. Under these circumstances the only way to determine the optimal policy mix is to evaluate the (successively more inclusive) policy mixes. In this way sequential CBA allows policy makers to select an optimal policy mix. Santos describes how to undertake such sequential CBA and illustrates the methodology and policy implications with the Lake District example.

Alternative Methodologies for Environmental Valuation

Economic valuation techniques have a role to play in identifying and quantifying some aspects of biodiversity and biological resource values, however, these techniques are less able to value non-use values and the value of ecological services provided by biodiveristy. Kahn *et al.* suggests there are two main reasons why it is difficult to assign an overall value to changes in biodiversity. They argue that revealed preference models of valuation are "generally capable of measuring direct use values of environmental change but they are generally not well suited to measure either indirect use values or the value of ecological services such as biodiversity." The problem is that ecological services are not traded in markets and therefore, unlike direct use values, it is not possible to use revealed preference techniques to elicit values. Secondly people do not know about, or appreciate, the many functions and services provided by ecosystems and therefore in stated preference models are not willing to pay as much as they should to protect ecosystems. In addition to these systematic biases, especially when applied to measure indirect use values, there are problems with the implementation and the lack of internal and external validity of this technique. Therefore they conclude, "as a result of these shortcomings, *indirect use values and the value of ecological services seldom are included in cost-benefit analysis or other forms of economic or ecological risk assessment."* It is because non-use and ecosystem services are difficult to value that other mechanisms have been developed to support policy making and to set conservation priorities. Part 3 investigates two methodologies that could be used in environmental policy making - conjoint analysis and indicators of site conditions and landscape context.

Conjoint analysis

Kahn *et al.* (Chapter 6) discusses the reasons why environmental economics has experienced limited success in measuring the value of biodiversity and suggest an alternative method, conjoint analysis, which performs an indexing function rather than a monetary valuation function. However, they show how conjoint analysis could be extended to include monetary valuations.

The authors also discuss other attempts to create an operational indicator of environmental quality, for example, representative environmental variables, green GDP, satellite accounts used by the United Nations Statistical Division, and aggregate indices such as the United States Environment Protection Agency's Environmental Monitoring and Assessment Program. Their own indicator is a choice, or trade-off based indicator developed using discrete choice based conjoint analysis to present alternative states of the environment to the individual. The alternative states would be defined by different levels of physical characteristics of the environment, including characteristics of important sub-systems of the environment. These alternative states of the environment would then comprise choice sets, and both experts and stakeholders would be asked to choose their preferred set. They argue that citizens are used to making such decisions, for example between living in the suburbs or in the city, marrying or staying single. This method is based on individuals willingness to make trade-offs. In this way, although no monetary values are elicited, this approach does in Dixon and Pagiola's words "illuminate the trade-offs inherent in biodiversity conservation decisions."

The key to developing the trade-off-weighted index is to appropriately specify choice sets that provide the survey data used to estimate the preference function. Biodiversity could be included as a component of overall environmental quality. In this way changes in biodiversity can be measured as the impact of the change in these biodiversity characteristics on the environmental index. An alternative approach would be to develop a set of choice sets describing alternative states of the world, where only biodiversity characteristics change between choice sets. A difficulty with this alternative is that citizens may not have the prerequisite knowledge to evaluate such choice sets. A third approach is to include biodiversity and other variables associated with the quality of life in different choice sets. Finally a price associated with the environmental or biodiversity resources could be included in the choice set, in order to assign some value to a change in the level of biodiversity. In this approach conjoint analysis based methods can be used to derive *monetary* estimates of the value of biodiversity.

Kahn *et al.* describes the development of this index approach in measuring the importance of biodiversity resources in the Clinch River, United States. Respondents are asked to choose a choice set of policy outcomes as indicated in table 1.3. An example of such a choice set is provided below.

Question: Which option for the future of the environment in the Clinch Valley do you prefer the most, Option A, Option B, or Option C? Option C is the **status quo**, or what is currently happening and will continue to happen with no further environmental policy. Evaluation of the results provides information on the preferences of citizens for various management/policy options.

Table 1.3 Policy Options for the Clinch Valley Conjoint Example

	Option A	Option B	Option C: Status Quo
Mussels	No change	10% increase	10% decrease
Fish (all species)	No change	No change	10% decrease
Recreation (fishing quality-small mouth bass)	Catch rates increase 10%	Catch rates increase 5%	No change
Songbird diversity/population	10% Increase	10 % increase	10% decrease
Wildlife population/diversity	5% Increase	No change	10% decrease
Air Quality	10 % improvement	10% decline	No change

The authors are also developing other indices that include different environmental variables and quality of life variables. In addition, a set of surveys will be conducted where the choice sets include taxes, so that willingness to pay measures can be derived. The indices can be used to infer which state of the environment is preferred, holding non-ecological costs and benefits constant and to determine which management scheme would achieve this desired result. The indices can also be used to evaluate changes in environmental quality in terms of trade-offs with other types of costs and benefits, such as economic benefits derived from a set of management decisions or actions. The index can be used to evaluate these types of trade-offs by computing the value of the index and the net economic benefits of each management alternative. These could then be plotted and management scenarios with higher economic benefits and better ecological quality identified.

The advantages of conjoint analysis is that it avoids many of the biases associated with contingent valuation as it does not ask individuals to undertake a difficult task, i.e. to compute their willingness to pay for an environmental change. However, this method does not provide a complete decision making tool; it only eliminates inferior alternatives and then explicitly outlines the trade-offs (e.g. between environmental improvement and regional development or between environmental improvement and loss of farmers' incomes) associated with the remaining management scenarios. The final extension of the index is to develop a willingness to pay measure for changes in environmental quality.

Indicators of site conditions and landscape context

In Chapter 7 King and Wainger assess the economic value of biodiversity using indicators of site conditions and landscape context.[2] They contend that alternative methodologies to measure the value of biodiversity are needed because "most ecosystem services are 'non-marketed' and 'non-market' valuation methods aimed at monetary measures of value are expensive to apply and yield estimates that are controversial, site-dependent, and not widely accepted in legal and regulatory settings." Furthermore current techniques are not well suited to measuring ecosystem services and therefore vastly understate total value. In addition to these theoretical difficulties with using standard valuation techniques, they also provide examples of 'valuation backfires' where dollar-based ecosystem valuation techniques were used to influence environmental policy but had unintended (negative) outcomes for conservation (see Box 1.1). The remainder of their paper describes the potential of relative (non-dollar) ecosystem valuation indicators compared to absolute (dollar-based) estimates of ecosystem value. The indicator system developed in their paper, is based on the notion that it is easier, and sometimes more useful, to consider the factors that affect whether a particular site is likely to provide values above, or below average, than to have an absolute measure of the average dollar value itself.

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[2] King and Wainger also provide a good summary of valuation techniques.

Box 1.1 Backfire: benefit transfer approach

An environmental group presents testimony in Oregon based on a widely disputed study in Louisiana that generated a wetland economic value of US$ 28,000 per acre. After disputing the validity of the estimating method and of using estimates from Louisiana in Oregon, the opposing side agrees to accept the number as fact, and points out that the county already requires US$ 40,000 per acre in compensation for wetland impacts as part of its "in lieu" mitigation fee program. Later in the year a group of wetland developers who are also paying US$ 40,000 per acre as wetland impact fees sue the state to reduce the fee and, using evidence presented by the environmental group, get the fee lowered to US$ 28,000.

The authors develop relative, non-monetary indices of ecosystem value based on: the capacity and opportunity of an ecosystem to provide services[3]; the supply and demand for those services; the risk of service flow disruptions; and other factors that affect service values. Combinations of ecosystem attributes determine the level of ecosystem services, and other indicators reflect the supply and demand for services, who has access to the services, the cost of the services, etc. For each category of ecosystem service, the indicator system focuses on seven building blocks of value:

– level of service;
– scarcity of service;
– replaceability of service;
– number of people with access to service;
– cost of getting or keeping access to service;
– preferences of people for service; and
– risk of service flow disruptions.

These factors are determined in predictable ways by on site and landscape characteristics that differ significantly from one ecosystem to another. Geographical Information Systems (GIS) can be used to identify such differences. The next step is to develop a set of 'leading indicators' of ecosystem value, where the value of an ecosystem is derived from the value of the flow of beneficial services it is expected to provide over time and individuals preferences for different sets of ecosystem services.

Accepting the premise that the economic value of an ecosystem is derived from the economic value of the services that it is expected to provide over time has some clear implications for value-based indicator development. It means that effort should focus on forecasting future service flows, possible disruptions, and their values rather than describing ecosystem conditions.

The authors illustrate this indicator system using an application that involves 'scoring' two different wetland sites, i.e. assigning relative values the two sites. In the example there are two wetlands of equal size but the conservation budget is large enough to conserve only one site. The landscape context and site conditions vary for the two wetlands (see Figure 7.1) and in turn impact on the seven indicators of value above. King and Wainger develop an ecosystem value indicator — the final index of ecosystem value is based on four sets of indicators: the functional capacity, function, service, and value indices (see Figure 7.4) to compare the 'value' of the ecosystems.

[3] The authors group ecosystem services into active and passive categories. Examples of active ecosystem services are commercial agriculture, recreational fishing, drinking water purification and education opportunities and passive uses include, climate regulation, existence/option/bequest values.

The above ecosystem value indicator will need to be adjusted if, for example, one wetland provides more of some services and fewer of others compared to a second wetland, in order to decide which wetland to conserve. The authors discuss how the relative preferences that people attach to various services can be used to develop service preference weights and thereby rank preferences. The advantages of this approach is that it focuses on the benefits and services associated with marginal changes in biodiversity and elements of the demand for them, however, it does not estimate absolute values for such changes in biodiveristy. Another difficulty with this approach is that it does not attempt to establish a demand function for the service being valued in contrast with contingent valuation techniques and conjoint analysis.

Bringing together Economic and Ecological Valuation

In Chapter 8, Nunes *et. al*. attempts to bring together economic and ecological valuation. A detailed discussion on the measurement of the different forms of biodiversity (i.e. genetic, species, and ecosystem) is provided and sets the stage for an explanation of different ecological indicators. The indicators provide some notion of ranking importance by relative based on expert opinion as method of assessing biodiversity. Nunes *et. al*. also discusses economic methods of valuing biodiversity, providing a number of examples of their application.

The objective of discussing both ecological and economic methods is to explore the scope for integrating them. While examples involving an integrated approach can be found, directly transferring or linking results from economic and ecological models is not simple. Moreover, a serious integration exercise is likely to partly diminish the analytical capabilities of each of the models involved; therefore, a clear understanding of the rationale for the use of this approach is needed to assess its trade-offs and justify its usage. Nunes *et. al* provides an initial framework that can assist policymakers and researchers interested in ecological-economic integrated models to assess these trade-offs.

Conclusions: the applicability of valuation techniques in OECD countries

Pearce contends that despite the growing volume of environmental economic valuation literature, we still have little idea of biodiversity value *per se,* even if we know a lot about the local use values of biological resources such as wetlands, forests, and endangered species. There is still limited information on the value of biodiversity as a store of genetic information for agriculture and pharmaceuticals and its value in terms of ecosystem resilience. Furthermore, there is still limited understanding of global non-use values of major biological resources, though it is contentious whether this is valuing diversity. However, as Dixon and Pagliola suggest “there are ‘best practice’ examples of the applications of these approaches and… these techniques often provide valuable assistance in deciding specific, well-defined operational questions”. Future advances in the implementation of environmental valuation techniques, including guidelines for the use benefit transfer techniques, will enhance and expand the role of such exercises in demonstrating the value of biodiversity conservation.

To understand the magnitude, and hence the importance, of non-consumptive benefits of biodiversity, it is illustrative to look at an example that attempts to calculate these benefits at the local, national, and international levels. The example below shows that non-timber benefits are significant, and in particular that international biodiversity benefits account for more than half of the total non-timber benefits. It is because these non-consumptive benefits are so large that methodologies to measure such values are critical for biodiversity policy.

Box 1.2 Value of Turkey's Forests

A recent review of the non-timber benefits provided by Turkey's forests [Bann and Clemens (1999)] illustrates the value of some of the services provided by the biodiversity they contain. As can be seen in the table below, the non-timber benefits of Turkish forests – many of which depend on biodiversity – are substantial at the local, national, and global levels, and exceed the timber benefits, which are about US$ 26 per hectare per year. Equally clearly, however, decisions that focus on only one of these non-timber benefits will omit important additional benefits, and so may tend to result in policies that under protect forests.

Estimated lower-bound of non-timber benefits from Turkish forestland

		US $/hectare/year
Mainly local benefits:	Non-wood forest products	18.4
	Wildlife	2.0
	Recreation	0.1
	Informal fuelwood	2.2
Mainly national benefits:	Watershed protection	7.4
Mainly global benefits:	Carbon storage	26.0
	Genetic resources	5.0
Total		**61.5**
Additional values associated with protected areas		2.6

Source: Dixon and Pagiola, Chapter 3.
Note: All estimates are lower bounds, and dependent on the assumptions described in the paper. Estimates are average expected benefits for the entire forested area (18 million hectares) except for the additional benefits from protected areas, which apply only to the 2.5 million hectares of protected forests.

Biodiversity valuation so far has not achieved the same level of popularity in policymaking as it enjoys in academic circles. The controversies surrounding some methods may have precluded the full use of valuation as an important tool to inform policymakers. Nonetheless, its importance strengthening the case for biodiversity conservation and its sustainable use is increasingly being recognised. The Convention of Biological Diversity (CBD) Conference of the Parties (COP) through its decision IV/10 underscores the need to undertake valuation and the important role economics play in this subject (www.biodiv.org):

> "… Recognising that economic valuation of biodiversity and biological resources is an important tool for well-targeted and calibrated economic incentive measures:
>
> 1. Encourages Parties, Governments and relevant organisations:
>
> …
>
> (c) To take into account economic, social, cultural and ethical valuation in the development of relevant incentive measures; …"

As essential as the demonstration of biodiversity values for conservation is the capture of these values by stakeholders at the local level "where actual resource use decisions are made" (Dixon and Pagiola). In order to raise the economic value of biodiversity conservation more effort could be made to enable local stakeholders to capture use values from tourism and other non-consumptive uses of biological resources, and for the identification of, and capture of, the non-use values and ecological service values associated with biodiversity, including through GEF-type grant funding. Nevertheless, where there is a high level of uncertainty over the value of biodiversity but a high risk of extinction, it is important that other policy measures, including those mentioned in Part 3 and 4, such as expert opinion and safe minimum standards, support biodiversity conservation policy.

CHAPTER 2:
by David PEARCE[4]

VALUING BIOLOGICAL DIVERSITY: ISSUES AND OVERVIEW

Introduction: what are we trying to value?

One of the most difficult issues in placing an economic value on biodiversity is determining what exactly the object of value is. We make a distinction between *biological resources* and *biological diversity*. A biological resource is simply a given example of a gene, species, or ecosystem. Biological diversity refers to the *variability* of biological resources. Thus, the Convention on Biological Diversity defines biodiversity as:

> "the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part".

In simple terms, biodiversity is the 'variety of life' whereas, biological resources are the manifestation or embodiment of that variety. If we look at the literature on the economic valuation of 'biodiversity' much of it is actually about the value of biological resources and is linked only tenuously to the value of diversity. This is especially true of the studies that use stated preference techniques – questionnaire approaches which ask directly for willingness to pay for the resource (contingent valuation), or which elicit a value indirectly (conjoint analysis). Arguably, when we value an endangered species we are thinking about what we are willing to pay to avoid its extinction because, once extinct, the diversity of species has been reduced. But it is far from clear from the stated preference valuation studies themselves that what people are valuing is this loss of diversity, rather than the aesthetic, or use value, of the biological resource that is at risk.

If this is correct, then the economic valuation literature has yet to come fully to grips fully with the concept of biological diversity [Pemberton (1996)]. Nonetheless, there is a growing literature that addresses the economic value of diversity *per se* [Swanson (1997) and Heal (2000)].

An example makes the distinction between resource and diversity of the resource clearer. Associated with the idea of diversity is the concept of '*distance*', i.e. some measure of the dissimilarity of the resources in question. In the species context, efforts have been made to incorporate the concept of genetic differences between species in order to elicit the implications for conservation policy. One

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4 I am indebted to Dominic Moran of the Scottish Agricultural College, Edinburgh for comments on an earlier draft.

of the most important implications is that, without some idea of distance, it is very easy for us to conserve the 'wrong' set of species (or genes or ecosystems) if our aim is to conserve diversity. Solow *et al*. (1993) provide an example in which they assemble information on the pairwise distance between cranes and the extinction probabilities of those cranes. The situation in terms of extinction probability are shown below:

	Species	Extinction probability
Endangered	Siberian	0.9
	Whooping	0.9
Vulnerable	Japanese	0.7
	Hooded	0.7
	White-naped	0.7
Indeterminate:	Black-necked	0.5
Safe:	All others	0.0

Now suppose that budget constraints mean that we can save only three species and that, unrealistically, the marginal costs of protection are the same regardless of the species in question**.** Also assume that the probabilities of extinction are independent of each other. Which species should be saved? It is very tempting to allocate all resources to the most endangered species – the whooping and Siberian cranes. But if the policy objective is to minimise the (expected) level of diversity, then Solow *et al.* (1993) show that the optimal programme requires that the Siberian, white-naped and black-necked cranes should be conserved. Focusing on the most endangered does not in fact minimise biodiversity loss. The reason for this is that the genetic distance between the endangered species and at least one of the 'safe' species is small. Minimising the probability of the number of species lost is not the same as minimising the value of lost biodiversity.

The example appears counterintuitive, not least because conservation policy tends to operate in the mode that appears intuitively correct. Thus, the US Endangered Species Act focuses on the degree of extinction risk, and popular attempts to set priorities for ecosystem conservation through concepts such as 'hot spots' have a similar focus [Myers (1988): (1990)]. Whatever is most threatened should be conserved first. Yet the limited literature on the measurement of diversity suggests that we may not be acting optimally by focusing on priorities in this way. Additionally, of course, prioritising action according to the degree of threat of extinction tends to ignore the reason why the resource is severely threatened in the first place. If the cause of extinction is not very amenable to policy measures, allocating resources to conservation is likely to be wasteful anyway. Priorities must take account of cost-effectiveness [Pearce and Moran (1996a), (1996b), (1998); Caldecott *et al*. (1996); Salafsky and Margoluis (1999)].

In what follows, we attempt to maintain the distinction between resources and the diversity of resources.

Why do we want to value biodiversity?

The process of valuation involves two things and it is very easy to engage in the first whilst forgetting the second. The first stage is the *demonstration of value*, i.e. measuring the economic value of the resource in an effort to see how large it is. This stage can be useful in itself if, for example, the issue is one of choosing between alternative land uses using a rule that says we will choose the one with the highest (net) economic value. Since the services rendered by most biological resources have no markets, demonstration alone may show that the uncaptured market value of conservation is higher than the value of the land use that involves loss of biological resources. A relevant context for 'demonstration only' would, for example, be a land use planning regulation that automatically allocated land to its highest economic value. Even then, the example is fairy fanciful because land use planning does not work this way. Few, if any attempts are made to value conservation services in economic terms in planning applications.

Hence the second *capture* phase is needed. In this stage mechanisms are devised whereby some or all of the demonstrated economic value can be turned into real resource flows which benefit the agents who would otherwise adopt a land use that threatens biodiversity. Any number of such capture mechanisms exist, ranging from entry pricing, to donations, to debt-for-nature swaps, carbon offsets, and so on [OECD (1996)].

Many people think that we either should not place monetary values on biological resources and diversity, or that that we need not do so. Arguments against valuation are addressed in Pearce (1999) and are not discussed further here. More recently, Heal (1999) has suggested that:

> "Valuation is neither necessary nor sufficient for conservation. We conserve much that we do not value, and do not conserve much that we value," and
>
> "Incentives are critical for conservation: valuation is not necessary for establishing the correct incentives."

The second statement is technically correct. Incentives are critical and we could introduce incentives without going through the valuation stage of the demonstration-capture paradigm. The statement echoes the Baumol-Oates least-cost theorem for pollution charges, where we do not need to know the value of pollution to adopt cost-minimising incentives [Baumol and Oates (1988)]. Unfortunately, the pressures to reduce biodiversity are so large that the chances that we *will* introduce incentives without demonstrating the economic value of biodiversity are much less than if we do engage in valuation. One reason for this is that valuation does not just tell us the economic value of a resource, it also tells us something about the demand curve for biological resources, i.e. how people will respond to prices charged for the use of those resources. This principle is no different for biodiversity than it is for, say, water, another resource where, until recently, scarcity value was not recognised.

Heal's first statement also requires qualification. The factors giving rise to biodiversity loss are dominated by land use change. Thus whether biodiversity gets conserved or not depends very much on the competing values for land (or water, or coastal resources etc). Swanson (1994) has argued convincingly that it is the opportunity cost of conservation that drives the decisions relating to biodiversity. *Homo sapiens* continues to compete for land with the panda, the elephant, natural wetlands, and so on. In principle, we can of course simply say that the value of land in conservation exceeds the opportunity cost of conservation, and that is how protected area status has largely come about. No one sought to ascribe economic values to the conserved land in order to do this. But it is precisely because of this process of declaring value by *diktat* that protected areas are under threat. Put another way, simply declaring an area is protected does not guarantee it will be protected. Few 'protected areas' in the developing world are protected in any true sense of the word, and the story is also not a good one in many rich countries, as the fate of Britain's Special Sites of Scientific Interest testifies. The issue is whether by demonstrating the economic value of conservation, we can do any better than the *diktat* approach. I suggest that, while valuation also can offer no guarantee of protection, it is preferable to not valuing the resource, and that the combination of valuation and incentives (capture) is better than no-valuation and no incentives.

Additionally, valuation does not just tell us that something is valuable. We learn so much more from the *process* of valuation, especially if we adopt stated preference techniques. We learn what people care about, what their motives are for conservation, what their reactions would be to different management objectives. It is worth noting that in most stated preference techniques, the question about willingness to pay is only a tiny part of the full questionnaire. But even if we did not learn these things, the valuation itself would still be immensely important. Suppose people express a low conservation value for an area, but we go ahead and protect it nonetheless. This is a recipe for destruction because we know from the valuation exercise that people do not value the area highly. They will put the pressure on to convert it to some other use, and that is exactly what is happening in many, many so-called protected areas. So, why are we successful in protecting some areas, in a more, or less, complete fashion? One clue lies in the opportunity cost of those areas – they are often remote or have limited alternative use value. Simply put, the opportunity cost of conservation for successfully 'protected' areas is often very low, and may even be zero.

Where Heal (1999) is absolutely right is in reminding us that economic valuation is all about marginal or discrete change in the availability of environmental resources. It is categorically *not* about valuing the existence of all biological resources. The economic value of an ecosystem can, in principle, be measured by the present value of the future flow of services from the ecosystem, and noting that all ecological services are economic services as well. But one cannot go from there to estimate the value of the Earth's *total stock* of biological resources using economic values. Just such a monumental mistake is made in the much publicised paper by Costanza *et al*. (1997). For a detailed critique see Pearce (1998).

The main reasons for advocating economic valuation of biodiversity and biological resources are so that we can facilitate cost-benefit analysis, green accounting, and proper pricing of biological resources.

Cost-benefit analysis

In cost-benefit analysis (CBA), costs and benefits are measured, as far as possible, in monetary terms. CBA can be applied to *investment projects* and to *policies*. An investment project might be something like a conservation programme. Since CBA has traditionally be defined in terms of what the gains and losses are to *society*, project-oriented CBA tends to be confined to public sector projects. The idea of using CBA to evaluate policy is more recent. In principle, the techniques and considerations are exactly the same. Policies have costs and benefits. The basic rule is not to sanction anything where the costs exceed the benefits. If benefits exceed costs then the project or policy is potentially worthwhile, but may still not be the best choice. This is because there may be some alternative where the ratio of benefits to costs is even higher. To say benefits exceed costs is therefore to adopt only a 'screening rule'. Typically, we aim to choose the option that maximises the difference between benefits and costs. Values for biological resources are increasingly being incorporated in cost-benefit evaluations of projects and policies, but values for diversity tend not to be.

Green accounting

Monetised costs and benefits can be used in a more expanded measure of gross national product (GNP). The basic aim is to take GNP and observe that it is made up of output, which is truly additional, and output which is needed to cover the depreciation of the capital base of the economy. This is usually summarised by saying that GNP = NNP + depreciation on man-made assets, where NNP is 'net national product'. It is intuitively obvious that it is NNP, not GNP, which comes closest to measuring the goods and services that are potentially available for individuals to consume. Deducting depreciation on other assets, including environmental assets then follows logically. This is the essence of a modified accounting framework. But to measure depreciation on environmental assets we need monetary measures of depreciation. Once again, what is being measured is the willingness to pay of individuals to avoid that depreciation (or the compensation they require to tolerate it). So, monetary valuation is integral to a 'proper' estimate of GNP. This said, no green national product study has yet valued biological resources beyond fisheries, forest timber and similar commercially used resources. Efforts to measure 'global' values of biodiversity have attracted a lot of attention, but have little analytical validity [Pearce (1998)].

Proper pricing

The essential feature of many environmental problems is the absence of a market for the environmental goods and services in question. In the market place, prices are determined by the interaction of supply and demand. But if there is no market place we cannot know the demand for environmental services. It will appear as if it is zero and hence the price that we can expect to get for those services also appears to be zero. If we have a given piece of land which can be used for a national park or for mining developments, it will have positive economic value as a mine but zero value as a national park. This, albeit exaggerated case, explains the 'non-level playing field' between conservation and 'development'. Non-market valuation techniques can be used to obtain at least part of a demand curve, which is in fact, a willingness to pay curve. Once we know the curve we can estimate the total WTP for the national park, and this can be compared to the opportunity cost of having the park, namely the mining profits foregone. This is the cost-benefit approach above. But we can also use the demand curve to find a price for entering the national park, a price which would maximise the revenues or profits from the park. In this way we can 'capture' the WTP for the park, in the form of entry prices, using the revenues to invest in conservation of the park.

Asking the right economic question

It is important to the right question about valuing species preservation:

Species extinction vs the probability of species extinction

The first issue is whether we are valuing the species itself or the probability that the species will become extinct. The need for a valuation of an entire species is unlikely to be commonplace. Most times we are concerned with policies which change the probability of species survival.

Local vs global extinction

Where extinction is a real possibility, it is likely to be local, rather than, global. Given the importance of non-use values (see below), it is important to remind people that a given policy may make a local population extinct, or vulnerable, but that the species exists elsewhere in a given state of protection, or danger.

Habitat vs inhabitants

A third distinction concerns habitat and its constituent parts. Maintaining habitat maintains the inhabitants, but maintaining the inhabitants does not always necessarily mean maintaining the habitat. For example, relocation of valued species may be possible. More importantly, if the object of value is the habitat, many different ecological (and therefore economic) services are involved. Habitats are ‘gestalts’ of values, going beyond the value that may be attached to individual components.

To this decade old list we now need to add whether it is diversity, or the biological resource, (however defined) that we are valuing.

What are the economic values of biodiversity?

The usual framework for analysing the economic value of biological resources is that of *total economic value* (TEV). TEV comprises use and non-use values, the former related to an actual use made of the resource, the latter to a willingness to pay for the resource independently of any use made of it (also known as passive use value). Use values may be direct, as with, say, timber from a forest, or indirect, as with the role played by the forest in regulating micro-climate or water flow. A substantial literature exists on the direct use values of biological resources – a slightly dated survey can be found in Pearce and Moran (1994, currently being revised as Pearce, Krug and Moran, forthcoming). Direct use values tend to relate to ecosystem products such a timber, fish, medicinal or edible plants. As such they have private good characteristics and also tend to have markets. Markets will tend to under-provide the public good features of biological resources and hence the policy focus should be more on the public good aspects. In turn, biological resources will have local public good aspects and global features. We focus on the global features because these are most germane to international policy.

Classifications of the various economic values of diversity vary. In what follows we focus on the categories set out below:

- Direct international use values: tourism.

- Direct global use values: information value in agriculture, pharmaceuticals.

- Indirect global use values: ecosystem resilience.

- Indirect global use values: ecosystem services.

- Non-use value.

Direct international use values: tourism

Nature tourism is one of the fastest growing industries in the world. It seems fair to say that it is based on both biological resources and diversity. It is well known that wildlife viewing is more attractive the more species there are to view, with exceptions surrounding the exotic rare species such as pandas, mountain gorillas, etc. [Pearce *et al.* (1999)]. Analysis of five African national parks suggests that international tourists account for substantially the greater proportion of measured consumer surplus from visiting the parks.

Direct global use values: agriculture

Swanson (1997) reports the results of a survey of plant breeding companies. These companies relied on wild species and landraces for 6.5% of the germplasm used in their research. Expressing the 6.5% as a percentage of the 82.9% well understood and standardised material, this suggests that an 8% wild germplasm 'injection' is needed each year to maintain the system as it is (see Table 2.1). Put another way, the stock of agricultural germplasm is totally renewing itself every 12 years or so (100÷8) from wild sources. As the wild sources degrade, this renewal/maintenance function will be lost.

Table 2.1 The role of biodiversity in agriculture: sources of germplasm in a sample of plant breeding corporations

Source of germplasm	Percentage from each source
Commercial cultivar	81.5
Related minor crop	1.4
Wild species: ex situ genebanks	2.5
Wild species: in situ	1.0
Landrace: ex situ genebank	1.6
Landrace: in situ	1.4
Induced mutation	2.2
Biotechnology	4.5
Relatively wild species	6.5

Source: Swanson (1997).

Self-evidently, germplasm from these wild sources has an economic value, but it is not clear what it is. Following the example of studies on the pharmaceutical value of biodiversity, one could estimate what plant breeding companies are willing to pay to conserve the 'remote' sources of germplasm. A start might be made by estimating what proportion of their current R&D budgets goes on such sources, but this will be only a minimum estimate of gross willingness to pay.

Alternative approaches would estimate what the value of lost crop output would be if this material did not exist, and another approach would be to ask what it will cost to substitute alternative sources. More indirectly, in most developed countries there are crop insurance schemes and it could be argued that such schemes are a substitute for the reduced risk of failure that would come about if natural diversity was increased. Farmers prefer the insurance scheme to adopting more diverse output, because it is cheaper for them: diverse crops may not match demand so well, and tend to be lower productivity crops (and, of course, subsidy systems favour uniformity of output). Additionally, not all crop failure will be due to uniformity of crops, although there is growing evidence that crop uniformity is correlated with ever widening instability in crop output [Anderson and Hazell (1989)]. Insurance payments will therefore be a maximum estimate of the economic value of crop diversity. WCMC (1992) estimate that total crop insurance premia in the USA in 1990 amounted to US$ 0.82 billion. Far more work is needed on the agricultural value of biodiversity.

Direct global use values: pharmaceuticals

Far more has been written about the pharmaceutical value of diversity, mainly because of early, popular arguments to the effect that there were massive stores of untapped wealth in tropical forests. While Oldfield (1984) drew attention to the commercial uses of biodiversity, the 'high value' approach was popularised by Myers (1979, 1983). Myers focused particularly on the potential value of wild species as sources of anti-cancer drugs, noting that the value of any drug should be measured against the costs to society of the diseases that may be cured or alleviated. However, Myers offered no detailed estimates of economic value. Economic studies which suggest relatively high valuations include Principe (1989, 1991), Farnsworth *et al.* (1985), and Farnsworth and Soejarto (1985). Pearce and Puroshothaman (1995) also adapt Principe's estimates to secure high global values for plant-based drugs, although when translated into values per hectare of biodiversity-rich land, the resulting values are very modest. More recently, new work by Rausser and Small (1998a, 1998b) argues that work claiming to demonstrate only very small economic values for plant genetic material is flawed, and that when due allowance is made for the competitive structure of the pharmaceutical industry, the resulting economic values are significant. These arguments are controversial, however.

Far more modest estimates of economic value are obtained by Aylward (1993), Simpson *et al*. (1994, 1996) and Simpson and Craft (1996). Mostly, these estimates are based on simulations of what drug companies are willing to pay for plant genetic material. But Simpson and Craft (1996) attempt to extend this 'private' valuation to a wider social dimension, i.e. to secure a value based on the worth of plant genetic material in social, rather than, private terms.

To understand the explanations for the differences in values it is essential to note the following points:

- Many studies compute the *total* value of the plant genetic material, i.e. they look at the market value of a drug and multiply the price of the drug (e.g. the prescription charge) by the quantity. Dividing total values by the number of drugs gives an *average* value for the drug. But several authors have pointed out that averages are misleading and that what is required is the *marginal* value of the genetic material, i.e. the value attached to one extra unit of genetic material [Simpson *et al*., (1994); Simpson and Craft (1996)].

- Early studies also ascribe the total market value of a drug to the plant material. This is legitimate if an only if there are no substitutes for that material, but, as Aylward *et al.* (1993) point out, this is not the case. Substitutes exist in the form of alternatives to plants and in the form of synthesis of plant material. Much the same goes for those studies that attempt to allocate the health benefits of drugs to the plant genetic material alone [Principe (1991); Pearce and Puroshothaman (1995)]. Clearly, there are many inputs into the health benefits that drugs may bring about.

- Most studies confine themselves to the *private* value of the drugs derived from plant material, i.e. the value to drug companies or the value to the public through payments in the market place. The relevant magnitude, however, is the *social* value of the drugs, e.g. in terms of lives saved or illness avoided, net of the costs of developing the drug. Studies, which attempt a social valuation, include Principe (1989, 1991), Pearce and Puroshothaman (1995) and Simpson and Craft (1996). Principe's study, however, used a very high value of statistical life of C$12 million in 1998 prices. As noted above, there are additional problems with this social value approach, namely ascribing health benefits to the drug alone. The Simpson-Craft study also uses a particularly narrow definition of the 'social' benefit, but deliberately so – see below.

- The early studies rely on US prescription data that is heavily influenced by the success of a few drugs, notably from the *Dioscerea* family. This raises the issue of how representative the early analyses would be for future success rates [Aylward *et al.*, (1993)].

- Markedly different estimates of economic value arise when the competitive structure of the drugs industry is taken into account – Rausser and Small (1998a, 1998b), for example, secure very much larger estimates than Simpson *et al.* (1996) but using the same data. The Rausser-Small work remains open to question, however – see below.

- Valuation procedures vary not only according to whether the value is 'private' (WTP of drug companies) or social (value of health benefits), but also according to the focus of the valuation. In the early studies the focus is on the value of the drug. In the Ruitenbeek (1989) study the focus is on the value of the research discovery as revealed by the renewal fee for patenting the resulting compound. Aylward *et al.* (1993) note that patent fees impart a downward bias to the estimates since renewal fees are small relative to other costs of R&D. The Pearce and Puroshothaman (1995) study, as well as Reid *et al.* (1993) studies, adopt a royalty approach, i.e. by looking at what a 'prospecting' company would pay to a host country for the rights to prospect for plant genetic material.

Table 2.2 brings together the estimates. The original figures have been converted to 1998 estimates using an average inflation rate of 3%. While the numbers have been standardised, caution should be exercised in interpreting the table. For the reasons stated above, the figures from different studies are not comparable. More generally, the studies are of very varied quality. The best studies are those of Simpson *et al.* and Rausser and Small. For more detailed discussion see Pearce *et al.* (1999).

Ecologists also draw attention to a wider insurance value of diversity in terms of its value in ecosystem integrity and functioning. The diversity of plants, animals and micro-organisms appears to have a role in helping ecosystems organise themselves to cope with shocks and stresses. Put another way, diversity would appear to be linked to resilience, the capacity of ecosystems to deal with externally imposed change. Diversity also matters at various levels. Genetic variability enables a given species to survive since, as changes to the environment occur, so some individuals stand more chance of adapting to the change than do others. A totally genetically homogeneous species could easily be at risk of being unsuitable to the new conditions, hence failing to survive. But just how important individual species are is debated. It appears that ecosystem functioning depends mainly on a comparatively small number of 'keystone' species, with others 'going along for the ride', so-called 'passenger species'. The less important species are certainly affected by ecosystem change but are not instrumental in effecting that change. In the same way, there may be a limited number of processes that drive ecosystem change. Does this mean that many species and processes can be dispensed with in an ecosystem without that ecosystem failing? The answer to this question is disputed. Many ecologists would argue that all species are 'necessary' in some sense, not least because they occupy niches that, if the species disappeared, would become empty and hence potentially subject to invasion by some other species that could easily be destabilising. Passenger species may also not be static, transforming themselves into keystone species at a later stage [Barbier *et al*., (1994), ch.2].

Indirect global use values: ecosystem resilience

Clearly, there is extensive uncertainty about what species 'matter' in the sense of ensuring that an ecosystem stands more, rather than less, chance of survival. Precautionary approaches would suggest that, in the face of such uncertainty, caution be exercised in eliminating species from within ecosystems simply because they appear to be redundant, in the sense of being 'passengers'.

Table 2.2 Estimates of the Medicinal Value of Plants, 1998 prices

Study	Value	Comment
Farnsworth and Soejarto (1985)	US$ 298 million per plant based drug, USA	Value of prescriptions for plant based drugs divided by 40 drugs based on plants. NB an average value.
Farnsworth and Soejarto (1985)	US$ 2.4 million per year per single untested plant species, USA	40 successful plants out of 5 000 tested entails 1 success per 125 tested plants. Total value of plant based drugs (US$ 298 million) divided by 125 gives value of untested species. NB an average value.
Principe (1991)	US$ 0.5 million per year per untested plant species, OECD wide	Based on Farnsworth and Soejarto but with modified probability of success in deriving a drug from a plant test. OECD total value of US$ 600 million (1980US$) x 1 in 2000 probability of success = US$ 300,000 per untested drug = US$ 510,000 per untested drug 1998 prices. Average value.
McAllister (1991)	US$ 9 500 per untested tree species, Canada, per annum	3 in 100 Canadian trees estimated to have marketable medicinal properties. Value of untested species = annual global value of a drug = US$ 250,000 x 0.03 = US$ 7500 in 1990 prices. Average value (low value due to low assumed value of successful drug)
Principe (1991)	US$ 28.4 million per untested species, OECD, per annum	US$ 37.5 billion annual value per successful species divided by 1 in 2000 probability of success = US$ 18.8 billion per untested species, or US$ 28.4 billion in 1998 prices. Value based on value of statistical life saved of US$ 8 million (1984 prices).
Ruitenbeek (1989)	US$ 190 per untested species per annum	Assumed 10 research discoveries in Camerounian rainforest each with patent value of US$ 7 500 pa. Divided by 500 species = US$ 150 or US$ 190 in 1998 prices. Note use of patent values as measure of value.
Pearce and Puroshothaman (1995)	US$ 743 to US$ 1.33 million per untested species, OECD, per annum.	Uses Principe and Farnsworth data. Lower value is private value and upper is social value based on VOSL of US$ 7 million.
Reid *et al*. (1993)	US$ 4 to US$ 4 600 per untested species per annum, hypothetical deal (annuitised at 5% over 20 years)	Royalty of 3% assumed, 1 in 10 000 success rate.
Artuso (1994)	Present value of US$ 866 per sample extract in terms of private WTP; US$ 9900 per extract in social terms	Detailed analysis of cash flows associated with sampling 25,000 extracts. Average value
Mendelsohn and Balick (1995)	Net revenue to drug companies = US$ 2.8 to 4.1 billion from rights of access to all tropical forests. Around US$ 1 her hectare.	Average value based on likely discoveries and their market value.
Simpson *et al.* (1994, 1996)	'Private' WTP of US$ 0.02 to US$ 2.29 per hectare of 'hotspot' land. Max WTP of marginal species = US$ 9 410	See Pearce *et al.* (1999)
Simpson and Craft (1996)	'Social' WTP of US$ 29 to US$ 2 888 per hectare of hotspot' land .Max WTP of marginal species = US$ 33 000	See Pearce *et al.* (1999)
Rausser and Small (1998a)	'Private' WTP of US$ 0 to US$ 9177 per hectare of 'hotspot' land.	See Pearce *et al.* (1999)

What then is the economic value of diversity in this broader sense? If diversity is critical to ecosystem functioning, albeit in an uncertain fashion, then reduced diversity reduces the probability that the ecosystem will survive. Its functions will then be lost. Hence the value of diversity can be approximated by the value of the functions of the ecosystem. The kinds of values estimated for the Thai mangrove system would then be relevant: loss of local, indirect, and global values. If this is correct, analysing the value of the functions of ecosystems amounts to conferring a value on diversity. But, uncertainty makes it difficult to know if measuring the value of these functions amounts to a complete coverage of the value of biodiversity. Given the context of uncertainty, conservation of diversity may have a value greater than the values that can be assigned to the 'outputs' that the ecosystem produces. To some extent these additional values are captured in the idea of *option value*, the value of conserving diversity because we may wish to make use of its beneficial effects in the future, and *quasi-option value*, the value of any information that might accrue by maintaining diversity rather than sacrificing it now.

The pervasiveness of the uncertainty about the value of diversity suggests two things:

– First, that, while placing economic values on the functions that we can value is essential, that activity may not capture the 'total' or 'true' economic value of diversity. Unless we have a better feel for option-type values, our valuation is incomplete; and

– Second, we should be cautious about sacrificing diversity.

The 'precaution' in the second implication needs further explanation. A traditional cost-benefit analysis has the capacity to incorporate a good deal of our concerns about uncertainty. For example, it is possible to introduce probabilities, if they are known, and work with expected values rather than certain values. If probabilities are known, we can also build risk aversion into the picture, i.e. we can weight quite heavily the losses that might occur if diversity is lost. But the problem with this expected value approach (or *expected utility* approach where risk aversion is incorporated) is that the probabilities tend not to be known. We have *pure uncertainty*. Cost-benefit analysis is less adaptable in this case and some authors have argued for use of the *safe minimum standards* approach (SMS) in such contexts. Under SMS there is a presumption in favour of conservation unless the opportunity costs are 'high' [Bishop (1978)]. Arguably, as people are better and better informed about the value of diversity, i.e. as we learn more and disseminate that learning, so the SMS approach will approximate a cost-benefit approach, because people's preferences will be better informed. In the meantime, SMS is intuitively attractive because it forces us to ask whether we are really sure that the benefits of sacrificing diversity are 'high'. The problem remains, of course, that we have to be clear on what we mean by 'unacceptably high' costs.

Indirect global use values: ecosystem services

The kinds of services that ecosystems provide include the protection of watersheds, climate regulation, waste assimilation, and nutrient cycling. Many of the benefits of these *ecosystem functions* accrue locally, and do not therefore appear to be of global significance. But since all living organisms depend on these functions, there is a real sense in which the local benefits contribute to an overall global benefit. Barbier (1994) classifies these ecosystem functions as regulation, production, carrier, and information functions. Regulatory functions include the regulation of climate, water flow, waste flows, and nutrients. Production functions refer to the useful outputs of ecosystems, such as water and fuel. Carrier functions relate to the support roles that ecosystems have for recreation, industry, fishing, agriculture, etc. Finally, information functions relate to aesthetic, cultural and scientific benefits.

The importance of indirect use value can be illustrated in the context of mangroves. Bann (1998) lists the potential benefits of a mangrove resource as shown in Table 2.3 (with slight modifications). The table reveals the wide range of functions that can be served by any one ecosystem. In each case, the ecological functions have economic value.

Table 2.3 Economic value of a mangrove resource

Use values			Non-use values
Direct value	Indirect value	Option value	
Timber, fuelwood, charcoal	Shoreline, riverbank stabilisation	Future direct and indirect values	Cultural, aesthetic
Fisheries	Groundwater recharge/discharge		Spiritual, religious
Forest products: food, medicine, wildlife etc	Flood and flow control		Global existence value
Agricultural resources	Waste storage and recycling		
Water supply	Biodiversity maintenance		
Water transport	Provision of migration habitat		
Genetic resources	Nursery/breeding grounds for fish		
Tourism and recreation	Nutrient retention		
Human habitat	Coral reef maintenance and protection		
Information	Prevention of saline water intrusion		

Source: Bann (1998) (with modifications).

Table 2.4 provides monetary estimates for one mangrove system in Surat Thani in South Thailand [Sathirathai (1998)]. The notable feature of these estimates is the dominant role played by indirect use values. Local use values relate to the use of the mangrove to supply fish, timber, and fuelwood. The offshore fishery benefit relates to the benefit that the mangrove has for the productivity of the offshore fishery. The coastal protection benefit is estimated by what it would cost to replace the beneficial protection function produced by the mangrove, i.e. what it would cost in coastal defences. This 'replacement cost' approach assumes that if the mangrove was not there then the protective function would have to be replaced, and this somewhat begs the question of the whether coastal defence would be worthwhile. Nonetheless, the use of replacement costs is quite widespread in cost-benefit studies. Finally, the carbon value is estimated in terms of the sequestration of carbon by the mangrove. Overall, some two-thirds of the economic value of the mangrove comes from coastal protection, and a further 9 per cent from carbon sequestration. Local values amount to just one quarter of the total. In terms of the local/global distinction, the estimates still omit many of the linkages discussed above. Nonetheless, the estimates are illustrative of the importance of indirect use values.

Table 2.4 Economic value of mangrove functions: Surat Thani, Thailand

Type of economic value	Net return per rai, US$	As % of total returns
Local use value	169	23
Indirect use value:		
off-shore fishery	13	2
coastal protection	498	67
carbon sequestration*	68	9
Total economic value	748	100

Source: Sathirathai, 1998.
Note: * Sathirathai uses a value of US$ 5.67 per tonne of carbon which is too low – see Annex B. We have multiplied this value by a factor of 5 here.

Global non-use values

Finally, we turn to global non-use values. There are very few studies which seek to elicit the cross-border willingness to pay to conserve resources. There are many more that focus on the ‘local’ willingness to pay to conserve species and habitats. Kramer and Mercer (1997) sought the WTP of US residents to conserve 5 per cent of the world’s tropical forests, and Swanson *et al.* (1998) estimate UK residents WTP for the conservation of the black rhinoceros in Africa. The results were that US residents were WTP US$ 21-31 per household for rainforest conservation via a rainforest ‘tax’ or fund. Aggregated over all US households this would be some US$ 1.9 to 2.8 billion. In the rhinoceros case, UK residents were WTP US$ 20-24 per person (say US$ 50 per household) for rhinoceros conservation in Namibia, the lower value relating to an option which included trophy hunting (aimed at raising revenues).

Pearce (1996) looks at the available WTP studies, the ‘implicit’ WTP of Western conservationists for developing country biodiversity via debt-for-nature swaps, and the funding of the Global Environment facility. He concludes that ‘world’ WTP for conservation of biodiversity might not exceed a few billions of dollars each year.

Conclusions

This brief overview raises several questions about the economic value of biodiversity.

First, it is not clear that the many willingness to pay studies of biological resources such as wetlands, forests, endangered species, etc. are also studies of biological diversity. Studies tend to focus on individual ecosystem services, a given ecosystem, or particular species. While people may be valuing these resources because they ‘represent’ diversity, we cannot be sure.

Second, we currently have only the most limited idea of the economic value of diversity as a store of genetic information for agriculture. We have a far better idea of the value of diversity for pharmaceutical use, although the literature remains controversial. At the moment, it would appear that pharmaceutical values for natural diversity are low, in the sense that they would do little to overcome the opportunity costs of conservation.

Third, we have extensive studies on the local use values of biological resources. To the extent that we can add individual ecosystem services and regard the sum as a 'total' value, we have good studies of such ecosystems (these are not surveyed here).

Fourth, we have a strong literature that links diversity to ecosystem resilience in the face of shocks and stress. One avenue of research exploration would be to assign a value to the instability caused by ecosystem uniformity, as in modern agriculture. This should be no different in concept to the value of maintaining diverse financial portfolios.

Fifth, we have a limited idea of the global non-use values of major biological resources. Whether they are values of diversity as such, is open to question but they probably have a strong element of diversity value in them. However, the sum of these values may not exceed a few billions of dollars a year.

The reality is that, despite the massive growth of economic valuation literature in recent years, we still have little idea of the value of diversity *per se*, even if we know a lot about the local use values of biological resources.

References

Anderson, J. and Hazell, P. Eds. (1989). *Variability in Grain Yields: implications for agricultural research and policy in developing country*. Johns Hopkins University Press for the International Food Policy Research Institute, Baltimore.

Artuso, A.. (1994). *Economic Analysis of Biodiversity as a Source of Pharmaceuticals*, Pan American Health Organisation/ Inter American Institute for Cupertino in Agriculture, Conference on Biodiversity, Biotechnology and Sustainable Development, San Jose, Costa Rica, *mimeo*.

Aylward, B. (1993). *The Economic value of Pharmaceutical Prospecting and Its Role in Biodiversity Conservation*, London Environmental Economics Centre, International Institute for Environment and Development, London, LEEC Paper DP 93-05.

Aylward, B., Echeverria, J., Fendt, L and Barbier, E. (1993), *The Economic Value of Species Information and its Role in Biodiversity Conservation*, Report to the Swedish international Development Authority (SIDA), London Environmental Economics Centre, International Institute for Environment and Development, London.

Aylward, B. (1995). "The role of plant screening and plant supply in biodiversity conservation, drug development and health care", in Swanson, T (ed), *Intellectual Property Rights and Biodiversity Conservation: an Interdisciplinary Analysis of the Values of Medicinal Plants*, Cambridge University Press, Cambridge. 93-126.

Bann, C. (1998). *The Economic Valuation of Mangroves: a Manual for Researchers*, Economy and Environment Program for Southeast Asia, Singapore.

Barbier, E., Burgess, J. and Folke, C. (1994). *Paradise Lost? The Ecological Economics of Biodiversity*, Earthscan, London.

Baumol, W and Oates, W. (1988). *The Theory of Environmental Policy*, Prentice Hall, New York.

Bishop, R. (1978). "Economics of Endangered Species". *American Journal of Agricultural Economics*, **60**, 10-18.

Caldecott, J., Jenkins, M., Johnson, T and Groombridge, B. (1996). "Priorities for conserving global species richness and endemism", *Biodiversity and Conservation*, **5**, 699-727.

Costanza, R *et al*. (1997). "The value of the world's ecosystem services and natural capital", *Nature*, **387**, May 15, 253.

Heal, G. (1999). "Valuing ecosystem services", Columbia Business School, *mimeo.*

Heal, G. (2000). Biodiversity as a Commodity", in *Encyclopaedia of Biodiversity*, Academic Press, New York, forthcoming.

Kramer.R and Mercer, D (1997). "Valuing a global environmental good: US residents' willingness to pay to protect tropical rain forests", *Land Economics*, **73**, 2, 196-210.

McAllister, D., (1991). Estimating the pharmaceutical value of forests, Canadian and Tropical, *Canadian Biodiversity*, **1**(3), 16-25.

Mendelsohn, R and Balick, M. (1995). The value of undiscovered pharmaceuticals in tropical forests, *Economic Botany*, **49**, 2, 223-228.

Myers, N. (1988). “Threatened biotas: ‘hot spots’ in tropical forests”, *The Environmentalist*, **8**, 187-208.

Myers, N. (1990). “The biodiversity challenge: Expanded ‘hot spots’ analysis”, *The Environmentalist*, **10**, 243-256.

Myers, N. (1979). *The Sinking Ark*, Pergamon Press, Oxford.

Myers, N. (1983). *A Wealth of Wild Species*, Westview Press, Boulder, Co.

OECD (1996). *Saving Biological Diversity: Economic Incentives*, OECD, Paris.

Oldfield, M. (1984). *The Value of Conserving Genetic Resources,* National Park Service, US Department of the Interior, Washington DC.

Pearce, D.W.(1996). “Global environmental value and the tropical forests: demonstration and capture”, in Adamowicz, V., Boxall, P., Luckett, M, Philips, W and White W (eds), *Forestry, Economics and the Environment*, CAB International, Reading, 11-48.

Pearce, D.W. (1998). “Auditing the earth”, *Environment*, **40**, 2, 23-28.

Pearce, D.W. (1999). “Valuing the environment”. In D.W.Pearce, *Economics and Environment: Essays in Ecological Economics and Sustainable Development*, Edward Elgar: Cheltenham. 13-33.

Pearce, D.W and Moran, D. (1994). *The Economic Value of Biodiversity*, Earthscan, London.

Pearce, D.W and Puroshothaman, S. (1995). “The economic value of plant-based pharmaceuticals”, in Swanson, T (ed), *Intellectual Property Rights and Biodiversity Conservation: an Interdisciplinary Analysis of the Values of Medicinal Plants*, Cambridge University Press, Cambridge, 127-138.

Pearce, D.W., Moran, D and A.Wendelaar. (1996*a*) “Global biodiversity priorities: a cost effectiveness index for investments”, *Global Environmental Change*, **6**, 2, 103-119.

Pearce, D.W and Moran, D. (1996b). “Investing in biodiversity: an economic perspective on global priority setting”, *Biodiversity and Conservation*.

Pearce, D.W and Moran, D. (1998). The economics of biological diversity conservation, in P.Fiedler and P.Kareiva (eds), *Conservation Biology for the Coming Decade*, Chapman and Hall, London, 384-409.

Pearce, D.W, Krug, W and Moran, D (1999). *The Global Value of Biological Diversity*, a Report to UNEP, Nairobi.

Pearce, D.W, Krug, W and Moran, D. (forthcoming). *The Economics of Biological Diversity*, Earthscan, London.

Pemberton, M. (1996). "The Measurement of Biodiversity", CSERGE, University College London, *mimeo.*

Principe, P. (1991). *Monetising the Pharmacological Benefits of Plants*, US Environmental Protection Agency, Washington DC.

Rausser, G and Small, A. (1998*a)*. "Valuing Research Leads: Bioprospecting and the Conservation of Genetic Resources", Department of Agriculture and natural Resources, University of California at Berkeley, May, *mimeo.*

Rausser, G and Small, A. (1998*b*). "Bioprospecting with Patent Races", Department of Agriculture and natural Resources, University of California at Berkeley, October, *mimeo.*

Reid, W., Laird, S., Gamez, R., Sittenfield, A., Janzen, D, Gollin, M and Juma, C. (1993). "A new lease of life", in Reid, W et al (eds), *Biodiversity Prospecting: Using Genetic Resources for Sustainable Development,* World Resources Institute, Washington DC, 1-52.

Ruitenbeek, J. (1992). "The rainforest supply price: a tool for evaluating rainforest conservation expenditures", *Ecological Economics*, **6**, 57-78.

Salafsky, N and Margoluis, R. (1999). "Threat reduction assessment: a practical and cost-effective approach to evaluating conservation and development projects", *Conservation Biology*, **13**, 4, 830-841.

Sathirathai, S. (1998). *Economic Valuation of Mangroves and the Roles of Local Communities in the Conservation of Natural Resources: Case Study of Surat Thani, South of Thailand*, EEPSEA Research Report Series, Economy and Environment Program for Southeast Asia, Singapore.

Simpson, D. (1998*a*). *Biodiversity Prospecting: Shopping the Wilds is Not the Key to Conservation*, Resources for the Future Web Page: www.rff/org/resources_articles/files/biodprospect.http.

Simpson, D. (1998*b*). "Some Comments on 'Bioprospecting with Patent Races" and "Valuing Research Leads: Bioprospecting and the Conservation of Genetic Resources" by Gordon Rausser and Arthur Small, Resources for the Future, Washington DC, *mimeo*.

Simpson, D., Sedjo, R and Reid, J. (1994). "Valuing Biodiversity: an Application to Genetic Prospecting", Resources for the Future, Discussion Paper 94-20, Washington DC.

(Early version of Simpson et al. 1996).

Simpson, D and Sedjo, R. (1996). "Valuation of Biodiversity for Use in New Product Research in a Model of Sequential Search", Resources for the Future, Discussion Paper 96-27, Washington DC.

Simpson, D., Sedjo, R and Reid, J. (1996). "Valuing biodiversity for use in pharmaceutical research", *Journal of Political Economy*, **104**, 1, 163-185.

Simpson, D and Craft, A.. (1996). "The Social Value of Biodiversity in New Pharmaceutical Product Research", Resources for the Future, Discussion Paper 96-33, Washington DC.

Simpson, D and Sedjo, R. (1996). "Investments in Biodiversity Prospecting and Incentives for Conservation", Resources for the Future, Discussion Paper 96-14, Washington DC.

Solow, A., Polasky, S and J.Broadus (1993). "On the measurement of biological diversity", *Journal of Environmental Economics and Management*, **24**, 60-68.

Swanson, T. (1994). *The International Regulation of Extinction*, Macmillan, London.

Swanson, T. (1997). *Global Action for Biodiversity*, Earthscan, London.

Swanson, T., Mourato, S., Swierzbinski, J and Kontoleon, A. (1998). "Conflicts in conservation: aggregating total economic values". Paper presented to World Congress on Environmental and Resource Economists, June, Venice, Italy. (Available from T Swanson at CSERGE, University College London).

CHAPTER 3:
by John A. DIXON and Stefano PAGIOLA

LOCAL COSTS, GLOBAL BENEFITS: VALUING BIODIVERSITY IN DEVELOPING COUNTRIES

Introduction

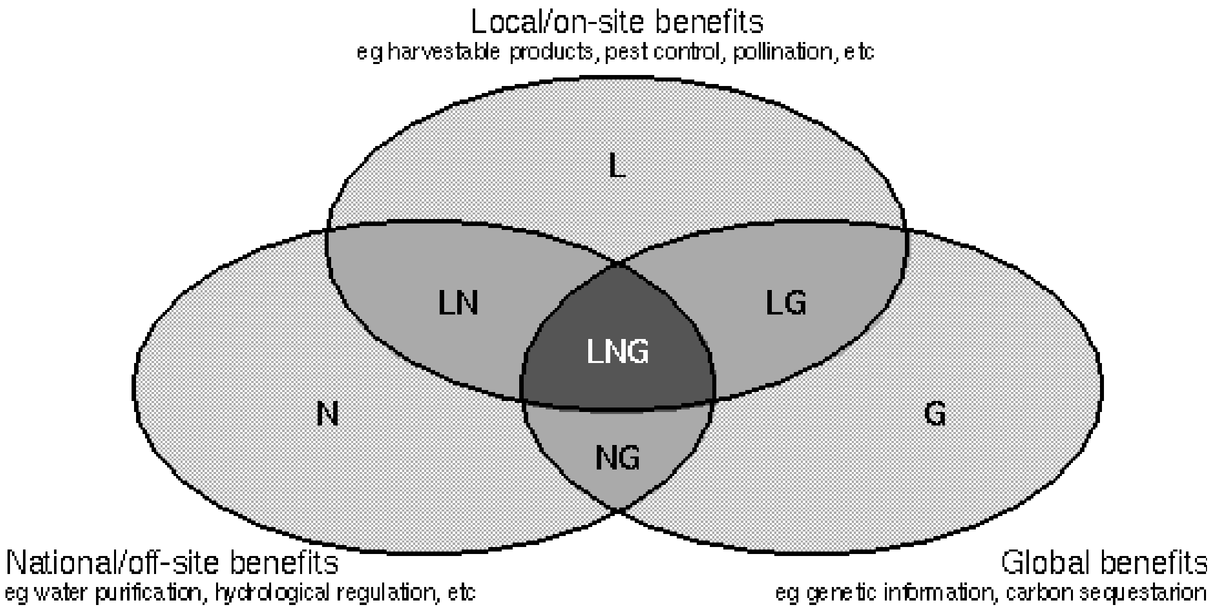
The question of the conservation of global biodiversity presents an interesting paradox: although biodiversity provides us with many benefits – and, indeed, may be indispensable for our very existence – it is being lost at unprecedented rates.

Biodiversity, and the ecosystems that contain it, provide benefits at multiple levels [Daily (1997)]. Locally, it provides benefits to farmers, villagers, and other land users such as harvestable products, and services such as crop pollination. Nationally, it provides benefits such as hydrological regulation and water purification to populations living downstream. Globally, it provides benefits such a carbon sequestration and genetic information. Why, then, is biodiversity so threatened? Why are we not doing more to protect it?

To the extent that biodiversity produces benefits at the local level, individual land users and countries have an incentive to conserve it. Likewise, national governments have an incentive to provide the resources needed to protect biodiversity to the extent that it provides benefits at the national level. Neither local land users nor national governments, however, have any incentive to protect the global benefits provided by biodiversity. Moreover, even at the national level, the benefits provided by biodiversity are often very poorly understood, if at all. As a result, national governments all too often view biodiversity conservation in terms of the development options that must be given up to ensure conservation. At the local level, land users receive but a small fraction of the total benefits of biodiversity. Conversely, the forgone benefits of biodiversity protection – in terms of increased agricultural or livestock production, or the cutting and sale of forest products – loom large to the local population.

Hence the paradox: biodiversity conservation is usually “under-provided” by the market – that is, market forces will lead to more conversion of habitat, and biodiversity loss, than would be either optimal or economically justified, precisely because of the divergence between local costs and global benefits. This situation is illustrated in Figure 3.1, which shows the sets of land-use that would maximise local, national, or global benefits. In an ideal world, these three sets would coincide. In practice, the overlap between these sets is only partial: activities that maximise local benefits overlap only partially with those which maximise national or global benefits (the area LNG in Figure 3.1).

Figure 3.1 Benefits of Biodiversity Use at the Local, National, and Global Level

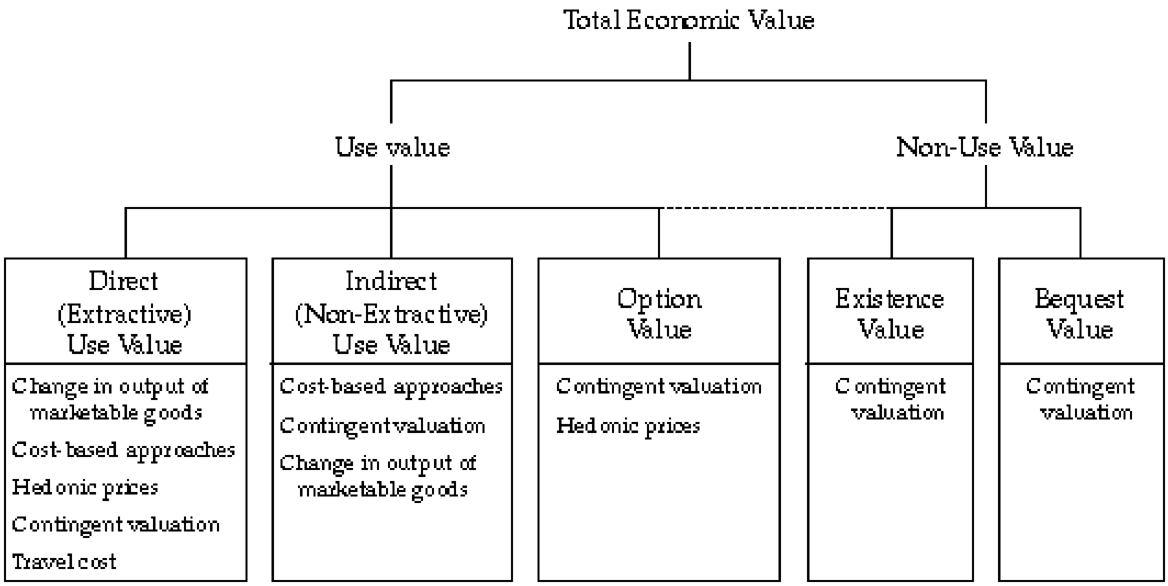


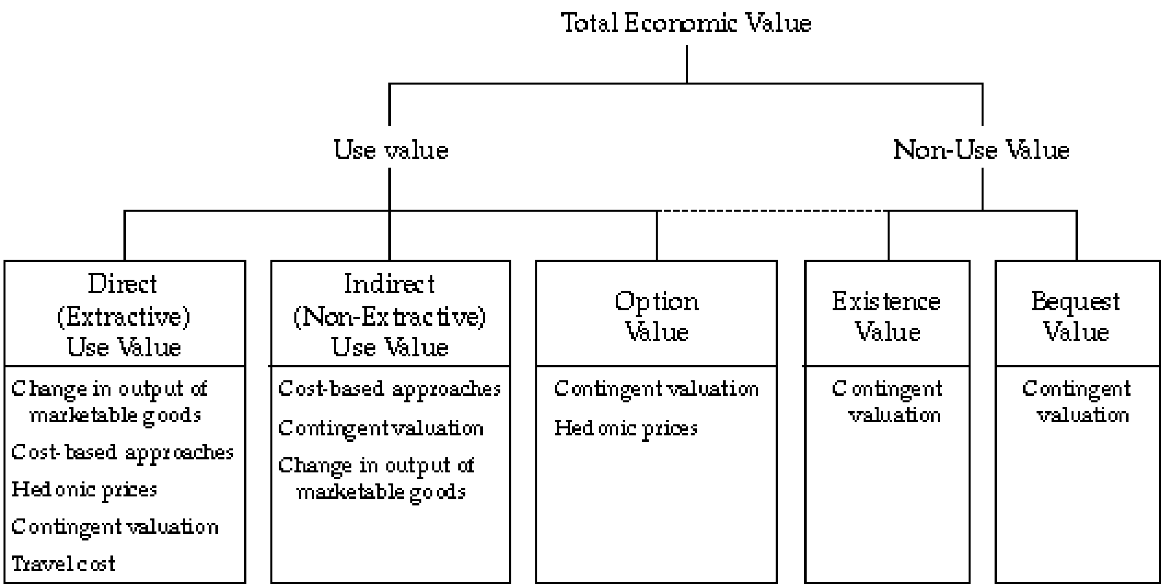
What role can economics play in understanding this dilemma and illuminating the trade-offs inherent in biodiversity conservation decisions, and what are the policies that are likely to help resolve this conflict? This paper discusses these issues and illustrates some of the approaches being tried in a number of countries. At the international level, the role of the Global Environment Facility (GEF) as one means of bridging this gap is also described. (Additional discussion on valuing biodiversity can be found in Pearce, Chapter 2, in this volume).

A Question of Values

Biodiversity is notoriously difficult to define, let alone measure. In light of these difficulties, it should come as no surprise that estimating its value is extremely difficult. A helpful first step is to decompose the many benefits that biodiversity provides. This can be done, for example, by using the Total Economic Value (TEV) framework [Pearce and Warford (1993)]. As can be seen in Figure 3.2, the TEV concept recognises that the value of biodiversity depends on the total benefits provided by each of a series of types of benefits, which range from the very tangible 'direct use' values (e.g. biodiversity as a producer of medicines, genetic stock for agricultural production, or actively experiencing an ecosystem or particular plant or animal species), through a series of increasingly less tangible 'indirect use' values (e.g. watershed protection, or storm protection from coral reefs), to completely intangible 'non-use' values (eg the pleasure people derive from the knowledge that biodiversity in general or certain specific components such as charismatic species exist). There is a dotted line between the Non-use Values and Option Value, since this component of value straddles the two broad groupings.

Figure 3.2 Total Economic Value and Valuation Techniques





Decomposing the value of biodiversity into components helps to avoid the common trap of calling biodiversity 'invaluable.' At one level, biodiversity could certainly be said to be infinitely valuable, since life would be impossible without it. But most decisions do not concern choices between complete preservation or complete destruction of biodiversity, but rather small changes in the level and quality of biodiversity. Thus, estimates such as those by Costanza *et al.* (1997) that the world's ecosystems are worth some US$ 33 trillion are of very little use even if they are correct – which in this particular case they are unlikely to be [Toman (1998)]. Most current economic work is aimed at giving more precise quantitative answers to more narrowly defined – and more policy-relevant – questions. Although such answers will of necessity not include all 'values', they can often be a useful starting point for fuller analyses of the economic values of biodiversity conservation.

Economic valuation techniques exist to measure all of the many component values of biodiversity, some with greater degrees of confidence than others [Dixon *et al*., (1994)]. The distinction can also be made between biodiversity *per se* (biological or genetic material) and biological resources. Biological resources are often easier to identify and value and include many of the goods and services produced by healthy ecosystems. In this paper the terms are often used interchangeably. Figure 3.2 shows some of the techniques which might be used for each type of value. Assuming the necessary data are available, many *use values* are relatively straightforward to estimate. In the case of products that are harvested directly, the biggest constraint is generally in obtaining accurate measurements of the quantities harvested. In addition, it is often difficult to determine whether current harvest rates are sustainable or whether they are degrading the biodiversity upon which the harvests depend. Considerable work has also been undertaken on valuing recreational use, and we now have several analytical techniques at our disposal, such as travel costs and contingent valuation (CV) studies, that can place fairly precise economic values on such use. Even in these cases, however, the '*option value'* of preserving potential future uses of biodiversity is difficult to assess. An interesting example of the application of these approaches to value non-timber forest benefits in Turkey is given in Box 3.1.

Box 3.1 Value of Turkey's Forests

A recent review of the non-timber benefits provided by Turkey's forests [Bann and Clemens (1999)] illustrates both the TEV technique and – although the study was not intended to value biodiversity *per se* – the value of some of the services provided by the biodiversity they contain. As can be seen in the table below, the non-timber benefits of Turkish forests – many of which depend on biodiversity – are substantial at the local, national, and global levels, and exceed the timber benefits, which are of about US$ 26 per hectare per year. Equally clearly, however, decisions which focus on only one of these levels will omit important additional benefits, and so will tend to under-protect forests.

Estimated lower-bound of non-timber benefits from Turkish forestland

		US$/ha/year
Mainly local benefits:	Non-wood forest products	18.4
	Wildlife	2.0
	Recreation	0.1
	Informal fuelwood	2.2
Mainly national benefits:	Watershed protection	7.4
Mainly global benefits:	Carbon storage	26.0
	Genetic resources	5.0
Total		**61.5**
Additional values associated with protected areas		2.6

Source: Adapted from Bann and Clemens, 1999.

Note: All estimates are lower-bounds, and dependent on the assumptions described in the paper. Estimates are average expected benefits for the entire forested area (18 million hectares) except for the additional benefits from protected areas, which apply only to the 2.5 million hectares of protected forests.

Other values associated with biodiversity can be frustratingly hard to assess. For example, the genetic information contained in intact ecosystems is often identified as an important benefit of biodiversity. However, both the extent of this information and its potential future usefulness are largely unknown, making it hard if not impossible to place an economic value on it. The potential benefits from as yet undiscovered genetic information is included in the economists' notion of '*quasi-option value*', which arises from the desire to preserve a resource in the expectation that as information and knowledge develop, we can learn more about the values of the resource and whether or not we want to preserve it. How much should we be willing to pay for this, however? The record of actual payments from potential users of this information, such as pharmaceutical companies, is hardly encouraging. Even in Costa Rica, which is a megabiodiversity country with well established protected areas and an internationally known biological research center, National Institute of Biology (INBio), actual payments for the option of the use of this biological information is only several million dollars in total – hardly the sorts of sums that are likely to meet the opportunity costs of protecting significant areas of tropical ecosystems![5]

The last major categories of non-use value are *existence value* and *bequest values*. Existence values arise from the pleasure people derive from the existence of biodiversity, even if they do not plan to use it. This value can be important, especially for charismatic species such as elephants, pandas, whales, or tigers, and for special, unique ecosystems such as certain pristine mountain, arctic, forest, or coastal areas. Bequest values are those values that come from leaving something for our children and future generations. Because existence value reflects preferences, which are not reflected in any market behaviour, survey-based methods such as CV are the sole means of measuring it.[6]

As a result of these problems, there are obviously differing levels of confidence in economic estimates for different resources and for different countries. Table 3.1 lists selected important uses and values associated with biodiversity and the level of confidence in the economic value estimates for each. In general the level of confidence is highest for direct-use, consumptive uses (often of biological resources rather than biodiversity *per se*). Here one usually has a definite P (price) and Q (quantity). The lowest level of confidence is for existence or option values for genetic material, not surprising since the level of certainty here is also the lowest.

Table 3.1 Levels of Confidence in Estimates of the Economic Value of Biodiversity

Value	Confidence
Direct use values	High
Tourism/ recreation	Medium
Ecosystem services	Low-Medium
Existence/ option values (individual)	Medium
Existence/ option values (genetic)	Very low-Medium

It is interesting that there are many similarities between biodiversity valuation and another new valuation area – the valuation of cultural heritage [Pagiola (1996)]. In both cases, much of the measurable value resides in various direct or indirect uses by individuals and societies, not in intrinsic

5 There are some interesting, and instructive, parallels between prospecting for oil and 'prospecting' for biodiversity. In the case of oil, companies pay very large sums for prospecting rights. These costs often total 80% of total production costs. In contrast, actual payments for biodiversity 'prospecting' have been very small. Part of this difference arises from the high degree of certainty on the market value of oil, once found. In the case of biodiversity, in contrast, there is uncertainty over both discovering new genetic material, and over its value if found. In addition, the time horizon from new discovery to commercial sales may be very long. All of these factors result in a much lower willingness-to-pay for biodiversity prospecting rights as compared to oil and gas prospecting rights.

6 A vast literature has developed on contingent valuation techniques. Although the technique has long been controversial, a 'blue-ribbon' panel composed of several Nobel prize winners found that, properly used, it can lead to reliable estimates of existence value (which the panel called 'passive use' value). The report of this panel [NOAA (1993)] is generally regarded as authoritative on appropriate use of the technique. For an overview of recent work in this field, see Carson (1997).

structural values. There are a number of parallels: both biodiversity and cultural heritage are often said to be 'invaluable', both 'use' and 'non-use' values are common, there are a number of difficult valuation areas, and there is widespread uncertainty and examples of market failures.

Values for Whom?

Simply measuring the benefits provided by biodiversity – no matter how accurately we may be able to do so – will not be sufficient to ensure its preservation. What matters is the incentives faced by individual decision makers – such as the farmers who decide what crops to plant and what inputs to use, whether to increase their cultivated area by clearing some forests, and which and how much of an ecosystem's products they will harvest. These, and many other decisions that affect biodiversity, will be made by individual decision makers in light of their own objectives and constraints and not according to any theory of the social good.

As indicated in Figure 3.1, many nationally important benefits of biodiversity may not be felt at the local level, and so are likely to be ignored by local decision makers. Consider the case of farmers deciding whether to clear a particular area of natural habitat – thus destroying the biodiversity it contains – for agricultural use. In making this decision, they would certainly consider the net benefits they expect to derive from increased crop production. They may also consider the loss of some goods and services from the uncleared area, such as fuelwood or pasture for livestock, since converting the area would mean having to find alternative sources of fuel and fodder. They will almost never consider the loss of benefits such as watershed protection, however, since they will not bear the costs of downstream flooding and sedimentation – these costs will be borne by people living far downstream. In some cases, activities designed to maximise local benefits will happen to coincide with those that maximise national, or global benefits, or both. In many cases, however, they will not.

It is important to understand that this divergence is not due to ignorance *per se* (although it may well be that local land users have no knowledge of the downstream consequences of their actions). Rather, it is a perfectly rational response to the incentives they face.

It is also important to understand how government policies affect biodiversity. In doing so, it must be borne in mind that only a few of the decisions that affect biodiversity directly are made by governments. Rather, government policies are important because they can have a significant—albeit often inadvertent—influence on the actions of individual decision makers both directly (through legislation, regulation, and zoning ordinances) and indirectly (through taxation, subsidies, and price policies).

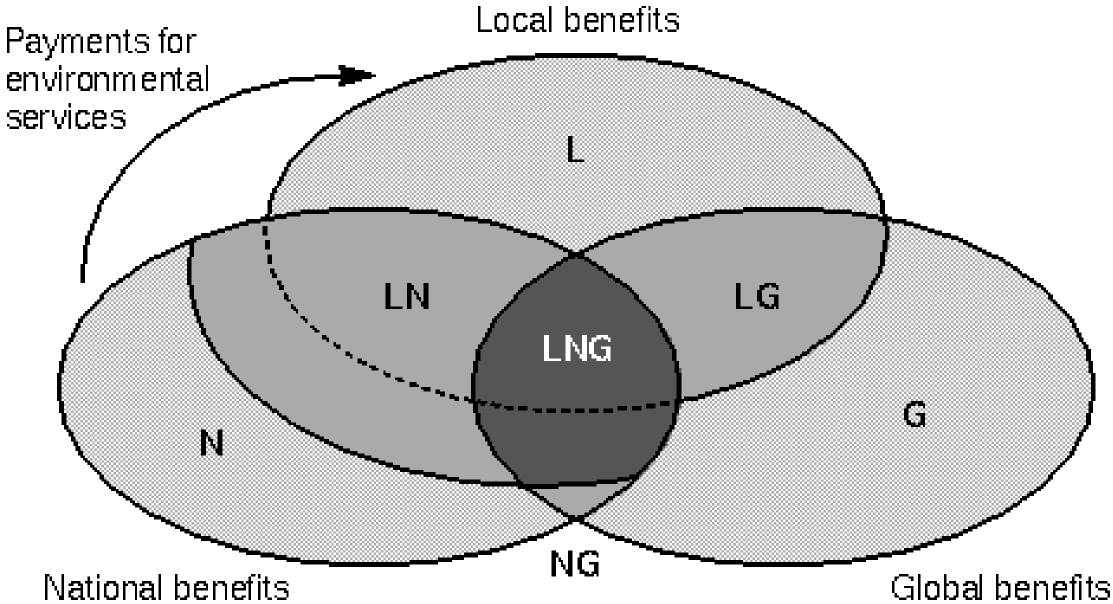
In an effort to help bridge the gap between local costs and global benefits, new ways are required to help increase the amounts that individuals or countries are willing to pay for protection of various national or global (or non-local) benefits. This is equivalent to expanding the overlap between the local/on-site benefit space in Figure 3.1 (and hence increases the overlap found in those areas identified as LN, LG, and LNG). Three ways of doing this are considered: Payment for national environmental services; payments for global environmental services, and the use of tourism/ecotourism to generate increased local economic benefits. In each case the objective is the same: to increase the perception at the local level of the amount of biodiversity conservation that is justified because it generates "local" economic benefits.

Paying for Environmental Services

Concern over the loss of valuable ecosystem services and the increase in problems such as reservoir siltation and downstream flooding, has led many governments to attempt to encourage land uses that preserve these services. Some governments adopted legislation and regulations intended to prevent land users from undertaking degrading activities or to compel them to adopt conservation practices, while others opted to subsidise the adoption of particular practices. The results of these efforts have often fallen far short of expectations. Land use rules have proven exceedingly difficult to enforce because of the vast spatial dispersion of agricultural activities and the often weak enforcement powers available to developing country governments. Subsidies have often succeeded in stimulating the adoption of conservation measures, but farmers frequently abandon their use – and sometimes actively destroy conservation structures – once subsidies cease [Pagiola (1999) and Lutz, Pagiola, and Reiche (1994)]. At other times, efforts to encourage conservation have achieved only token co-operation by farmers [Enters (1997)].

In recent years, recognition of the problem and of the failure of previous approaches to deal with it has led to efforts to develop systems in which land users are compensated for the environmental services they generate. In this way, land users would have a direct incentive to include these services in their land use decisions, resulting in more socially optimal land uses. The logic behind this approach is shown in Figure 3.3: by paying local land users for the environmental services they generate, the overlap between land uses that maximise local and national benefits is increased (as seen by the increased area found in LN and LNG in Figure 3.3).

Figure 3.3 Using Payments for Environmental Services to Increase Local Incentives to Conserve Biodiversity



Reid (forthcoming) discusses the potential for capturing part of the value of ecosystem services to finance biodiversity conservation efforts. He argues that there is significant scope for doing so, especially in the case of services associated with water quantity and quality carbon sequestration, while the scope for capturing part of the benefits of services such as pollination, pest control, waste treatment, and flood and storm protection, is more limited.

This approach is not a panacea, however. In some cases, the desired environmental services can be generated by land uses, which bring limited or no benefits in terms of biodiversity conservation. For example, forests logged with sustainable logging practices might provide the same hydrological services as intact primary forest, but have much lower biodiversity [Bruijnzeel (1990) and Hamilton and King (1983)].

In Central America, the payment for land use changes to increase carbon sequestration and provide other benefits has been pioneered by Costa Rica [Castro *et al*., (1997) and Chomitz *et al*., (forthcoming)]. Under the 1997 Forestry Law, land users can receive payments for specified land uses – including new plantations, sustainable logging, and conservation of natural forests – which are intended to generate a variety of environmental services, including regulation of hydrological flows, biodiversity conservation, and carbon sequestration. Since 1995, over 200,000 hectares have been incorporated into the programme, at a cost of about US$ 47 million. Other Central American countries are adopting similar approaches [Pagiola (2000)]. Costa Rica is now extending this approach to the local level. In another recent development, in April of 2000, Costa Rica approved a law that established a trial “environmentally adjusted water tariff” the proceeds of which will be used to help maintain watershed areas near Heredia. This payment for local environmental services is notable since it is a transfer from downstream beneficiaries (the user of water and payers of the water tariff) to upstream resource managers.

Paying for Global Environmental Services

Just as local land users have no incentive to conserve biodiversity except insofar as it generates local benefits, national governments have no incentive to conserve biodiversity except insofar as it generates national benefits. The Global Environment Facility (GEF) was created to help fill this gap. The GEF provides grant financing to countries to undertake activities that generate global benefits, such as biodiversity conservation, but which are not in the country’s direct interest. The logic here is similar to that of national governments attempting to encourage activities that generate national benefits: by providing grant financing, the GEF increases the overlap between activities that generate global benefits and those that generate local and national benefits (Figure 3.4). Since 1992, the GEF has provided over US$ 400 million in financing for biodiversity conservation through the World Bank alone.[7]

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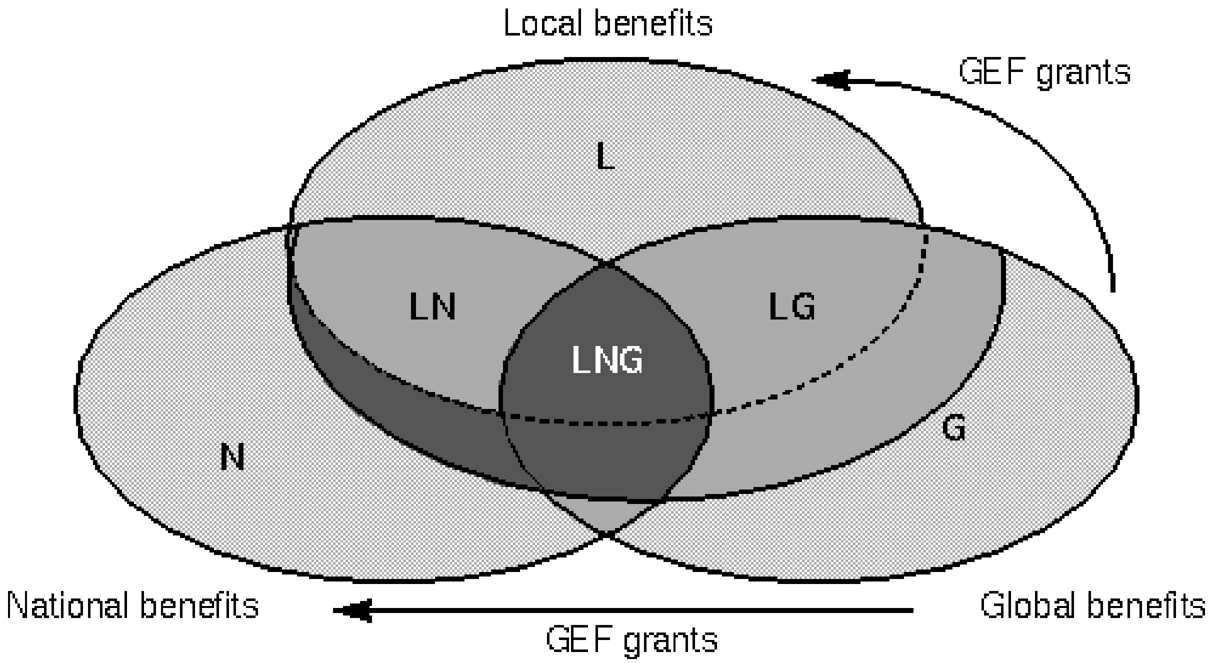
[7] The World Bank is one of three implementing agencies for the GEF. Additional GEF financing for biodiversity conservation has been channeled through the other two implementing agencies, the United Nations Development Programme (UNDP) and the United Nations Environment Programme (UNEP). During the same time period, the World Bank and its soft-loan window, IDA, provided an additional US$750 million in financing for biodiversity conservation [World Bank (2000)].

Figure 3.4 Using GEF grants to Increase Local and National Incentives to Conserve Biodiversity



The GEF is not the only organisation paying for global environmental services. Many conservation NGOs have financed the acquisition or management of protected areas. Some commercial organisations have also played a role. Merck and other pharmaceutical companies, for example have established contracts with Costa Rica's INBio that gives them exclusive rights to first use of selected genetic material in exchange for support for INBio's program of collection and analysis of genetic material. Unfortunately the amounts generated by this programme have been modest – usually several million dollars per contract spread over a number of years – and would not be sufficient, in and of themselves, to provide compensation to local populations who are asked to forgo conversion options in order to maintain these ecosystems intact.

In a 1998 GEF project, the World Bank provided grant funds to Costa Rica to INBio to help support its work on biodiversity resource development. The economic analysis of the project estimated the additional national benefits to Costa Rica from the proposed activities and supported only those costs that were not offset by increased national benefits [World Bank (1998)]. The analysis found that the limited increase in national benefits, largely from additional tourism, with smaller amounts from new bioprospecting contracts, would yield less that US$ 0.13 million per year. In contrast the proposed project would cost more than US$ 1.0 million per year. Hence the increase in economic benefits that Costa Rica would receive from the project activities were small (less than US$ 0.9 million) over the seven year life of the project, and almost all of the project costs were thus eligible for GEF financing. In essence, this is an international payment to Costa Rica for providing global benefits above what it would be justified in doing itself from a national benefit-cost perspective.

Ecotourism as a Way to Generate Local Benefits

As long as the benefits local decision makers derive from biodiversity are limited primarily to extractive use, their incentive to conserve it will remain limited. Ecotourism[8] has been seen as a way of solving this problem by generating income from aspects of biodiversity and biological resources which had hitherto been of little local interest, and so helping to bridge the gap between local costs and global benefits [Dixon (1999) and Brandon (1996)]. With tourism in general and ecotourism in particular growing rapidly worldwide[9], considerable hopes have been placed on this solution. The reality, of course, has often fallen short of these high expectations. Rather than seeing ecotourism as a panacea, it is more useful to ask to what degree, and under what conditions, ecotourism can help meet the twin challenges of biodiversity conservation and local income generation.

Type of tourists **-** The potential for ecotourism to generate income from biodiversity obviously depends on the type and number of tourist concerned. As noted, the ‘ideal’ ecotourist is often thought to be low-impact and high-spending. Even when such an ideal ecotourist exists, however, there are some important trade-offs to be considered. For example, visitors to Rwanda’s Parc des Volcans paid several hundred dollars per person in entrance fees, plus substantial additional amounts for supplies, to visit the area inhabited by the mountain gorillas made famous by naturalist Dian Fossey. Because the need for low impact resulted in a very low permissible number of visitors, however, the total revenue generated was very low (and even at these low numbers, Fossey believed the impact was unacceptably high). In general, though, the number of tourists willing to pay large amounts is likely to be small and – as ecotourism destinations multiply around the world – spread thin. Generating sufficiently high levels of revenue may thus require attracting a larger number of tourists, which is likely to have a higher impact on biodiversity, both from the larger numbers of visitors and from the infrastructure they require. This does not automatically mean that mass-based tourism is the only viable option. While small-scale tourism may generate low total revenues, it may also have lower costs, since mass-based tourism is likely to require substantially greater investment in support infrastructure (such as transport and lodging). Whether *net* revenues are higher with a small number of higher-paying tourists or a large number of lower-paying tourists will vary from case to case.

Type of ecosystem **-** The sensitivity of different ecosystems to visitation can vary substantially. Sites such as the Parc des Volcans in Uganda are very sensitive to outside intrusions, and will quickly deteriorate if mismanaged. Other sites are much more resilient. Thus, while the mountain gorillas of the Parc des Volcans are easily disturbed even by low levels of visitation on foot, wildlife in Kenya’s Masai Mara is often all but oblivious to gaggles of minibuses. This is not to say that sites

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8 Definitions of ecotourism have varied substantially. While some use the term very broadly to cover any form of tourism in which nature-based activities are important, others make a sharp distinction between ecotourism and ‘nature-based tourism’. The Ecotourism Society (TES) defines ecotourism as “travel to natural areas that conserves the environment and sustains the well-being of local people”. Brandon (1996) emphasizes that ecotourism must be small scale “with limited ecological and social impacts.”

9 No specific data are collected on ecotourism, and so estimates of its importance, both in terms of numbers and economic impact, vary substantially. The Ecotourism Society estimates the number of nature tourists in 1994 at 211-317 million, and the number of wildlife-related tourists at 106-211 million. Note that the two definitions provided focus solely on the *purpose* of the trip, and say nothing about some of the other aspects contained in the Society’s own definition of ecotourism. Note also that these purposes are themselves defined extremely broadly, and that it is not clear whether the two categories are meant to be mutually exclusive, or if, for example, wildlife-related tourists are a subset of nature tourists. These kinds of problems bedevil all available statistics on ecotourism.

such as the Masai Mara cannot be damaged — perhaps irreversibly so — if mismanaged, but that the requirements for "low impact" visitation differ.

Type of site **-** Depending on the nature of the site, different types of visits will be most appropriate. Some will hold visitors' interest over several days, while others will only attract visitors for a few hours. Sites with particularly unusual or popular attractions may bring in tourists from far away, while others may be visited only as an adjunct to a different trip.

Depending on the specific situation both the ideal and the feasible trade-offs will differ greatly; it would be wrong, therefore, to expect that a unique 'ecotourism' solution exists which would work in every case. The extent to which the objectives of biodiversity conservation and local income generation can be met will likewise vary substantially.

Involving local populations **-** Even if ecotourism does generate high net revenues, this will not necessarily contribute to biodiversity conservation unless a large proportion of these revenues is retained locally. In most countries, however, the funds collected by entrance fees, tourist taxes, and other mechanisms are often channelled to the central government with only a small proportion, if any, remaining on-site. Likewise, the benefits of economic activities supporting ecotourism (lodging, food, transport) are often captured by economic agents from outside the immediate area. Involving local stakeholders is both practical and equitable. Since a primary challenge is to enlarge the circle of benefits that the local population is willing to help provide (see Figure 3.1), the active planning of biodiversity conservation with local populations is essential – both to obtain their support for the effort, but also to make sure that they share in the generation (and capture of) economic benefits.

Box 3.2 Ecotourism in Mantadia National Park, Madagascar

Madagascar is one of the world's 'mega-diversity' countries, with both a very high level of biodiversity and high rates of endemism. This rich biodiversity is an important factor in attracting tourists from all over the world, but has been under considerable stress from conversion of natural habitats to alternative uses. The government is attempting to protect biodiversity by creating a system of protected areas. The government, however, lacks the budgetary resources necessary to cover the expenses of park maintenance and to compensate local communities for the losses they bear as a result of the creation of protected areas. A set of studies carried out in the early 1990s examined the benefits tourists obtained from visiting national parks in Madagascar, as well as the cost to local communities of giving up their traditional uses in areas brought into the protected area system [Kramer *et al*., (1993)].

Two different methodologies were used to estimate the benefits tourists would obtain from the creation of a new national park at Mantadia. The travel cost method, which uses information on the costs borne by tourists to visit a location to derive their demand curve for the location, and hence the enjoyment they receive from visiting it. The contingent valuation method, in which visitors are asked directly for their willingness to pay for such visits, was also used. Both methods have their strengths and weaknesses, but both gave similar estimates of the benefits, namely about US$ 24-65 per trip. Assuming that the number of visitors to this new park is about the same as in neighboring parks, the total benefit generated would be about US$ 0.8-2.2 million.

The costs to local communities of losing their traditional access to the protected area were also estimated in two different ways – using contingent valuation, and by estimating the opportunity cost of lost income from the park area. These methods also gave very similar estimates of costs of about US$ 90-110 per household, for a total cost of about US$ 0.6-0.7 million. It would thus seem that even at the lower end of likely tourism benefits, income might be sufficient to compensate local communities. Some of this compensation will occur indirectly, through employment and other income opportunities generated by tourism, but it will also be necessary to capture at least a portion of the tourism benefits directly and to redistribute them to local communities. How much of this benefit the park is ultimately able to capture will depend partly on the approach adopted, and on the desires of the tourists.

The case of the Mantadia National Park, Madagascar, is one example (see Box 3.2). In another example, villagers living around the Royal Chitwan National Park in Nepal, the habitat for the endangered Indian rhino and other threatened species, are allowed into the park on certain times each year to collect grass for roof thatch. In this way the villagers benefit from protection of the ecosystem, and the rhino and tall grasses, while restricting their encroachment into the National Park. Such symbiotic approaches are increasingly finding favor with protected area managers around the world.

Conclusions and Recommendations

Recent years have seen a considerable expansion in the use of economic valuation of biodiversity and biological resources, both in the academic literature and in operational work such as in World Bank projects. Considerably experience has been accumulated in the use of a variety of valuation techniques to estimate the benefits of various components of the total economic value of biodiversity. These techniques tend to be most robust when used to estimate direct-use values, but non-use values can also be estimated (although with less certainty in many cases). In many cases there are 'best practice' examples of the applications of these approaches and they are increasingly being used for decision making by governments. Although many conceptual, methodological, and empirical problems remain, these techniques often provide valuable assistance in deciding specific, well-defined operational questions.

Valuation by itself is not sufficient to guarantee that biodiversity will be conserved. First, it may well be that in certain specific instances biodiversity conservation is in fact more costly than alternative courses of action, even when all the many benefits of biodiversity are accounted for.

Perhaps most important, however, valuation alone will not save biodiversity if it remains an academic exercise. What matters is the incentives that individual decision makers have to conserve or not to conserve individual bits of biodiversity – the actual costs and benefits they will receive from conservation compared to alternative courses of action. Often, these decision makers stand to gain only a small part of biodiversity's total benefits, while they stand to gain the full benefits from alternative courses of action such as clearing virgin forest for agricultural use. And so they quite understandably will tend to under-protect biodiversity. *Mechanisms must thus be found by which at least part of these broader benefits of biodiversity are channelled to the local level where actual resource use decisions are made.* Systems of payments for environmental services and GEF grants are two such mechanisms, which aim to preserve the national and global benefits provided by biodiversity, respectively. Developing ecotourism is another mechanism.

The process of implementing these incentive measures is complicated and time is often an enemy of biodiversity conservation. Three broad lessons flow from the previous statements:

1. *Avoid extinction and other irreversible actions* **-** Once a species is extinct, or a unique habitat is destroyed, we will never know what we have lost (or its potential economic or scientific value). When irreversible actions are possible the cautionary principles of the Safe Minimum Standard (SMS) and the use of the Opportunity Cost approach are helpful [see Ciriacy-Wantrup (1952) or Bishop (1978), for a fuller discussion of the cautionary principle].

2. *Capture important direct use values, especially from tourism and other non-consumptive uses of biological resources* **-** Since a major challenge is to increase the generation of economic benefits from biodiversity conservation, sometimes the easiest (and potentially most lucrative) approach is to capture a larger share of the economic rents associated with various direct uses (both consumptive and non-consumptive) of biological resources. This both helps to generate economic returns, and also demonstrates to decision makers that there are concrete (and capturable) economic values associated with these resources.

3. *Identify and attempt to capture part of the non-use values associated with biodiversity conservation.* Although harder to do than capturing rents from users of the resource, there is a substantial willingness-to-pay for certain types of biodiversity and biological resources. These altruistic payments (whether they are for bequest, or existence, or option values) can be important sources of income that can help offset some of the local costs of biodiversity conservation. The major problem with this aspect is that non-use values may be small, and may heavily favour certain "charismatic" species, rather than those in greatest danger, or of greatest potential scientific values.

Final thoughts on what can be done

We started this paper by saying that there was a paradox between the amount of biodiversity that would be provided by the market due to the divergence between local costs of conservation and the national and global benefits provided from the same biodiversity and associated habitats. Even if the latter benefits greatly exceeded the local costs, unless there is a change in the accounting framework (or a mechanism to capture part of the national and global benefits and transfer them to offset local costs) we will end up with less biodiversity conserved worldwide than is optimal – from either a scientific or economic perspective.

We see that economic valuation has an important role to play in identifying and quantifying some of these values. However, in most cases, those parts that we can value (various forms of direct use values, both consumptive and non-consumptive), are often not the most important components or what is potentially the most valuable (genetic material, genetic diversity, the "unknown" discovery). In this situation, how does one set priorities for biodiversity conservation, especially when there are limited quantities of financial and human resources available and more demands for conservation than can be financed. Two options can be offered:

First, there is an important role for various sorts of expert opinion to select priority areas (especially those that can "justify" their conservation through application of the economic valuation techniques presented in this paper). Expert opinion may take the form of a Delphi approach (a process of multiple questioning of experts to arrive at a consensus decision on areas to protect) or some other expert-based process.

Second, the high levels of uncertainty over biodiversity values, and the irreversible nature of extinction, argue persuasively for serious attention to the need to maintain representative ecosystems. Whether this is an application of the Safe Minimum Standard approach of Ciriacy-Wantrup (1952) or just an injunction to "avoid extinction" this need illustrates why one cannot exclusively rely on market solutions to meet all biodiversity conservation needs.

Squaring the circle of this challenge will not be easy. What may be required is some form of a Global Option Value or Global Existence Value, and the search for some effective mechanism to make this a realistic payment mechanism to help bridge the gap when there is a divergence between local costs and global benefits.

References

Bann, C., and M. Clemens (1999). "Turkey: Forest Sector Review – Global Overlays Program Final Report." Washington: World Bank (processed).

Brandon, K. (1996). "Ecotourism and Conservation: A Review of Key Issues." Environment Department Paper No.33. Washington: World Bank.

Bishop, R. (1978). "Endangered Species and Uncertainty: The Economics of a Safe Minimum Standard" *American Journal of Agricultural Economics* 60: 10-18.

Bruijnzeel, L.A. (1990). *Hydrology of Moist Tropical Forests and Effects of Conservation: A State of Knowledge Review*. UNESCO International Hydrological Programme. Paris: UNESCO.

Carson, R.T. (1997). "Contingent Valuation: Theoretical Advances and Empirical Tests since the NOAA Panel." *American Journal of Agricultural Economics*, **79**, pp.1501-1507.

Castro, R., F. Tattenbach, with N. Olson and L. Gamez (1997). "The Costa Rican Experience with Market Instruments to Mitigate Climate Change and Conserve Biodiversity." Paper presented at the Global Conference on Knowledge for Development in the Information Age, Toronto, Canada, 24 June 1997 (processed).

Chomitz, K.M., E. Brenes, and L. Constantino (forthcoming). "Financing Environmental Services: The Costa Rican Experience and its Implications." *Science of the Total Environment*.

Ciariacy-Wantrup,S.V. (1952). *Resource Conservation Economics and Policies*. Berkeley, University of California Press.

Costanza, R., R. D'Arge, R. de Groot, S. Farber, M. Grasso, B. Hannon, K. Limburg, S. Naeem, R.V. O'Neill, J. Paruelo, R.G. Raskin, P. Sutton, and M. van den Belt (1997). "The Value of the World's Ecosystem Services and Natural Capital." *Nature*, **387**, pp.253-260.

Daily, G.C. (ed.) (1997). *Nature's Services: Societal Dependance on Natural Ecosystems*. Washington: Island Press.

Dixon, J.A. (1999). "Ecotourism: Incentives for Conservation Success." Paper presented at the Workshop on Sustainable Tourism in Natural Areas in North America, Playa del Carmen, Quintana Roo, Mexico, 27-28 May 1999 (processed).

Dixon, J.A., L.F. Scura, R.A. Carpenter, and P.B. Sherman (1994). *Economic Analysis of Environmental Impacts*. London: Earthscan.

Enters, T. (1997). "The Token Line: Adoption and Non-Adoption of Soil Conservation Practices in the Highlands of Northern Thailand." In S. Sombatpanit, M.A. Zðbisch, D.W. Sanders, and M.G. Cook, eds. *Soil Conservation Extension: From Concepts to Adoption*. Enfield: Soil Publishers.

Hamilton, L.S., and P.N. King (1983). *Tropical Forest Watersheds: Hydrologic and Soils Response to Major Uses and Conversions*. Boulder: Westview Press.

Kramer, R.A., M. Munasinghe, N. Sharma, E. Mercer, and P. Shyamsundar (1993). "Valuation of Biophysical Resources in Madagascar." In M. Munasinghe, ed, *Environmental Economics and Sustainable Development*. Environment Paper No.3. Washington: World Bank.

Lindberg, K. (1998). "Economic Aspects of Ecotourism." In K. Lindberg, M. Epler-Wood, and D. Engeldrum (eds.), *Ecotourism: A Guide for Planners and Managers*. N. Bennington: The Ecotourism Society.

Lutz, E., S. Pagiola, and C. Reiche (1994). "The Costs and Benefits of Soil Conservation: The Farmer's Viewpoint." *World Bank Research Observer*, **9**, pp.273-295.

NOAA. (1993). "Report of the NOAA Panel on Contingent Valuation." *Federal Register,* **58**, pp.4602-4614.

Pagiola, S. (1996). "Economic Analysis of Investments in Cultural Heritage: Insights from Environmental Economics." Washington: World Bank (processed).

Pagiola, S. (1999). "Economic Analysis of Incentives for Soil Conservation." In D.W. Sanders, P.C. Huszar, S. Sombatpanit, and T. Enters (eds), *Incentives in Soil Conservation: From Theory to Practice*. New Delhi: IBH Publishing.

Pagiola, S. (2000). "Payments for Environmental Services." http://www-esd.worldbank.org/eei/.

Pearce, D.W., and J.W. Warford (1993). *World Without End: Economics, Environment, and Sustainable Development*. Oxford: Oxford University Press.

Reid, W.V. (forthcoming). "Capturing the Value of Ecosystem Services to Protect Biodiversity." In *Managing Human Dominated Ecosystems*. Washington: Island Press.

Toman, M. (1998). "Forum on Valuation of Ecosystem Services." *Ecological Economics*, 25, pp.57-60.

World Bank (1998). *Costa Rica: Biodiversity Resources Development Project*. Project Document, Jan.21.Environment Department. Washington: World Bank.

World Bank (2000). Supporting the Web of Life: The World Bank and Biodiversity - A Portfolio Update (1988 -1999). Washington: World Bank.

PART 2

CHAPTER 4:
by Ståle NAVRUD

COMPARING VALUATION EXERCISES IN EUROPE AND THE UNITED STATES - CHALLENGES FOR BENEFIT TRANSFER AND SOME POLICY IMPLICATIONS

Introduction

Environmental valuation studies have four main types of use [Navrud and Pruckner (1997)]:

i. Cost-benefit analysis (CBA) of investment projects and policies.

ii. Environmental externality costing in order to map the marginal environmental and health damages of e.g. air, water and soil pollution from energy production, waste treatment and other production and consumption activities. These marginal external cost can be used in investment decisions and policy making (e.g. as the basis for "green taxes").

iii. Environmental accounting at the national level (green national accounts), local level (community green accounts) and firm level (environmental reporting).

iv. Natural Resource Damage Assessment (NRDA); i.e. compensation payments for natural resource damage from e.g. pollution accidents.

Environmental valuation techniques have mostly been used in CBAs, but are now increasingly used also in NDRAs in the United States; in environmental externality costing of electricity production from different energy sources in both the US and Europe [Rowe *et al.* (1995), Desvousges *et al.* (1998) and European Commission – DG XII (1995, 1999)]; and in green national accounting exercises, e.g. the Green Accounting Research Project (GARP) of the European Commission [Tamborra (1999) and GARP II (1999)]. The accuracy requirements increase, and thus the applicability of benefit transfer techniques decrease, as we move down the above list of potential policy uses of valuation studies [Navrud and Pruckner (1997)].

CBA has a long tradition in the US as a project evaluation tool, and has also been used extensively as an input in decision making following President Reagan's Executive Order (EO) 12292 issued in 1981, necessitating a formal analysis of the costs and benefits of federal environmental regulations that impose significant costs or economic impacts (i.e. Regulatory Impact Analysis). In many European countries, CBA has a long tradition in the evaluation of transportation investment projects, however, environmental valuation techniques were seldom used. With the exception of the UK Environment Act, there is no legal basis for environmental CBA in any European country, however, some countries have administrative CBA guidelines for project and policy evaluation, and in a few cases these include a section on environmental valuation techniques.

Paragraph 130r of the Maastricht Treaty, which focuses on European Union´s environmental goals, environmental protection measures and international co-operation in general, states that the EU will consider the burden and advantage of environmental action or non-action. Furthermore, the “Fifth Activity Programme for Environmental Protection Towards Sustainability” (1993 – 2000) asserts: *In accordance with the Treaty, an analysis of the potential costs and benefits of action and non-action will be undertaken in developing specific formal proposals within the Commission. In developing such proposals every care will be taken as far as possible to avoid the imposition of disproportionate costs and to ensure that the benefits will outweigh the costs over time* (European Community 1993, p. 142). The 1994 Communication from the Commission to the Council of the European Parliament, entitled: “Directions for the EU on Environmental Indicators and Green National accounting - The Integration of Environmental and Economic Information Systems” (COM (94)670, final 21.12.94) states a specific action for *improving the methodology and enlarging the scope for monetary valuation of environmental damage*. More recently, the EC´s Green Paper, entitled: “For a European Union Energy Policy”, states that the *internalisation of external costs is central to energy and environmental policy*. During the last few years the European Commission has performed CBAs of two new regulations; the large combustion plant directive and the air quality standards. Both analyses rely heavily on the work done within the EC Directorate General (DG) XII´s ExternE project (European Commission - DG XII 1995, 1999). The Environment Directorate (DG XI) of the EC has also started training courses in CBA for their administrative staff to promote better priority setting. It can be seen that in both the United States and in Europe, there is an increased interest in using environmental valuation both for CBA, environmental externality costing and environmental accounting.

International organisations like the OECD, the World Bank, regional development banks and UNEP (United Nations Environment Program) have produced guidelines on environmental valuation techniques; e.g. OECD (1989, 1994, 1995); Asian Development Bank (1996), and UNEP (1995, Chapter 12). In many cases these institutions have used valuation techniques as an integral part of CBA of investment projects, e.g. the World Bank’s evaluation of water and sanitation projects [Whittington (1998)]. The UN´s statistical division, UNSTAT, has also actively supported the development of resource accounting systems (e.g. the Handbook on Integrated Environmental Economic Accounts).

Although there have been numerous environmental valuation studies of biodiversity and ecosystem functions in the US and in Europe [Navrud (1992, 1999)] for an overview of European valuation studies), the policy use of valuation studies seem to have concentrated on air and water pollution impacts and policies [Navrud and Pruckner (1997)]. Increased policy use of environmental valuation estimates raises the need for more original studies of environmental goods and the health effects from air, water and soil pollution, and also improved techniques for transferring valuation estimates from one geographical area and context to another area and context (i.e. benefit transfer techniques).

This paper will first review the use of selected original biodiversity valuation studies by decision makers in Norway. Based on the Norwegian experience, obstacles to the more widespread use of valuation techniques in project evaluation and policy analysis are discussed in Section 2. Increased demand for biodiversity values and restricted time and money to do new, original valuation studies, have paved the way for benefit transfer. Section 3 provides an overview of benefit transfer techniques, and results from tests on the accuracy of benefit transfer exercises compared to doing new, original studies. Section 4 concludes with the main challenges for benefit transfer and some policy implications.

Policy use of biodiversity valuation estimates in Norway

In Norway about 30 empirical environmental valuation studies of biodiversity have been carried out since the early 1980s. Many of these studies have been initiated and funded by the Ministry of Environment and its affiliated institutions: the State Pollution Control Authority (SFT) and the Directorate of Nature Management (DN). Both Travel Cost (TC) and Contingent Valuation (CV) methods have been used to estimate the use value of freshwater fish (recreational fishing) and wildlife (hunting and viewing). There have also been CV studies of the non-use value (and some use value) of endangered species, such as the three big carnivores in Norway (brown bear, wolverine, and wolf); insects, fungi and lichens growing in old growth coniferous forests; and the genetic diversity of local trout and salmon stocks threatened by acidification.

Benefit estimates from CV studies on the impacts from improved water and air quality have been utilised by SFT in a CBA of measures to reduce air and water pollution on the local, regional and national level since the mid-80s. During the past 5 years SFT has funded new CV studies to create an updated list of values to use in CBA evaluations. In the early 1990s the Directorate of Public Roads (DPR) also funded new CV studies on valuations for noise, local air pollution, and soiling from road traffic. The DPR has included these valuations in their new manual and software package for undertaking CBAs of road projects. A CBA is required for all new road projects in Norway. Both the SFT and DPR analyses plays an important administrative role. The SFT analyses have also had a large influence on policy decisions, while the DPR analyses play a relatively minor role in the political decision making on road projects.

The administrative and political use of benefit estimates of biodiversity is much less regular. In most cases the economic estimates of biodiversity and a CBA are used to support decisions, but they are seldom decisive. However, there are also examples where the valuation estimates have played an important role in the decision making process.

Three case studies are discussed [Navrud (1993)]:

a) CBA of preservation of the brown bear, wolf, and wolverine;

b) CBA of liming acidified lakes and rivers to preserve the genetic diversity of local salmon and trout stocks; and

c) CBA of the preservation of biodiversity in old growth coniferous forests

In case A the benefit estimate was used to support the policy decision, in case B the benefit estimate played a much more decisive role, while in case C the benefit estimates had little or no effect on the decision taken.

Obstacles to the wider use of environmental valuation

Five main categories of potential obstacles for wider use of environmental valuation can be identified.

(1) *Cost and time constraints*

The cost of and the time required to undertake a new CV study have increased, in particular if the study complies with the set of guidelines put forward by the NOAA panel in the US [Arrow *et al.* (1993)]. For these reasons there is an increased interest in benefit transfer, i.e. transferring estimates from previous valuation studies to value impacts of another project or policy. However, little is known about the validity of these new estimates. One should be very careful when transferring estimates, particularly for complex goods, goods with a large non-use component, or both, for instance for ecosystems and biodiversity. More work on the validity of benefit transfers is needed.

(2) *Methodological problems*

Methodological problems due to the uncertainty both in the measurement of biodiversity and how it is affected by a new project or policy, and the measurement of the economic value of biodiversity, may lead to invalid estimates. This is the most important obstacle to the wider use of benefits transfer. The validity of economic estimates from CV studies, especially for non-use values, has been questioned. The main question asked by policymakers is: "Can we trust the economic estimates of biodiversity and ecosystems - Will people actually pay the amounts they state they are willing to pay in CV surveys?" Navrud and Veisten (1997) combined hypothetical donations from a CV study with actual donations to preserve biodiversity in old growth coniferous forests in Oslomarka, Norway. The same respondents were asked to actually donate what they stated in the CV study to create an upper and lower bound, respectively, for the welfare measure, WTP to preserve biodiversity. Everybody had to pay, if, aggregated WTP for preservation exceeded the social costs of preservation. The bound proved to be sufficiently small (i.e. mean WTP of about 3-13 euro/household/year) to be informative in a CBA of preserving 13 old growth forest areas. It would be useful to have more studies comparing actual and hypothetical WTP for biodiversity preservation.

(3) *Information and communication problems*

Lack of information about valuation techniques among decision makers at different levels makes it difficult to defend the results of valuation studies against, often unjustified, critique from affected interest groups. Ethical concerns combined with scepticism towards neoclassical welfare economics, which is the foundation of CBA and environmental valuation methods, are the most important reasons for such criticism. Increased and improved dissemination activities, and training of bureaucrats and decision-makers by valuation researchers, has proved effective in improving acceptance of environmental valuation techniques in Norway.

(4) *Administrative constraints*

The lack of legal basis for doing CBAs, and the absence of environmental valuation techniques in administrative guidelines for CBAs, discourages environmental benefit estimation. In Norway, a theoretical CBA manual, and practical CBA guidelines for evaluation of public projects and policies, have been prepared [NOU (1997:27) and NOU (1998:16)]. These manuals have special sections on valuing environmental impacts. The previous CBA manual, issued in 1979, contained no information on environmental valuation techniques. The new CBA guidelines recommend using environmental valuation techniques for local environmental problems, e.g. soiling and noise from road traffic and recreational areas. However, they raise caution on including non-use values in CBAs because of the uncertainty of CV studies for valuing non-use values (see also (2) above).

(5) Political constraints

Politicians often tend to give higher priority to avoiding controversy than to cost-effectiveness. This is partly due to corrections for income and regional distributional effects. Thus, descriptions of distributional effects should be included in CBAs.

Of these obstacles (2), and, in particular the validity of CV estimates of non-use values, is the most problematic.

Benefit transfer approaches and their accuracy

There are two main approaches to benefit transfer:

i) Unit value transfer
   a) Simple unit transfer
   b) Unit transfer with income adjustments
ii) Function transfer
   a) Benefit function transfer
   b) Meta-analysis

Unit value transfer

Simple unit transfer is the easiest approach to transferring benefit estimates from one site to another. This approach assumes that the well being experienced by an average individual at the study site is the same as that which will be experienced by the average individual at the policy site. Thus, we can directly transfer the mean benefit estimate (e.g. mean WTP/household/year) from the study site to the policy site.

For the past few decades such a procedure has often been used in the United States to estimate the recreational benefits associated with multipurpose reservoir developments and forest management. The selection of these unit values could be based on estimates from only one or a few valuation studies considered to be close to the policy site, or based on mean values from literature reviews of existing values. Walsh *et al*. (1992, Table 1) presents a summary of unit values of days spent in various recreational activities, obtained from 287 CV and TC studies. More recently the US Oil Spill Act recommends transfer of unit values for assessing the damages resulting from small “Type A” spills or accidents using the National Resource Damage Assessment Model for Coastal and Marine Environment. This model transfers benefit estimates from various sources to produce damage assessments that are however based on limited physical information from the spill site.

The obvious problem with this transfer of unit values for recreational activities is that individuals at the policy site may not value recreational activities the same as the average individual at the study sites. There are two principal reasons for this difference. First, people at the policy site might be different from individuals at the study sites in terms of income, education, religion, ethnic group, or other socio-economic characteristics that affect their demand for recreation. Second, even if individuals’ preferences for recreation at the policy and study sites were the same, the recreational opportunities might not be.

Unit values for non-use values of e.g. ecosystems from CV studies might be even more difficult to transfer than recreational (use) values for at least two reasons. First, the unit of transfer is more difficult to define. While the obvious choice of unit for use values are consumer surplus (CS) per activity day, there is greater variability in reporting non-use values from CV surveys, both in terms of WTP for whom, and for what time period. WTP is reported both per household and per individual, and as a one-time payment, an annual payment for a limited time period, an annual payment for an indefinite time, or even monthly payments. Second, the WTP is reported for one or more specified discrete changes in environmental quality, and not on a marginal basis. Therefore, the magnitude of the change, should be similar, in order to get valid transfers of estimates of mean, annual WTP per household. Also the initial levels of environmental quality should be similar if one expects non-linearity in the benefit estimate or underlying physical impacts.

For health impacts the question of which units to transfer seems somewhat simpler. With regards to mortality the unit would be the Value of a Statistical Life (VSL) or the newer, and disputed measure, Years of Life Lost (YOLL). It is more complex to measure morbidity, therefore several units of value are used. For light symptoms like coughing, headaches and itching eyes, symptom days (defined as a specified symptom experienced one day by one individual) are often used. Values for more serious illnesses are reported in terms of value per case. However, the description of these different symptoms and illnesses varies in terms of severity. A better measure would be to construct values for episodes of illness defined in terms of symptoms, duration and severity (in terms of restrictions in activity levels, whether one would have to go to the hospital, etc.).

On the issue of units to transfer, one should also keep in mind that often the valuation step is part of a larger damage function approach. The study objective is often to estimate values for the endpoints of dose-response and exposure-response functions for environmental and health impacts, respectively, due to changes in, for example, emissions of air pollutants. Thus, a linkage has to be developed between the units the endpoints are expressed in, and the unit of the economic estimates. This has been done successfully for e.g. changes in visibility range [Smith and Osborne (1996)], but is more difficult as the complexity of changes in environmental resources increase.

The simple unit transfer approach is not fit for transfer between countries with different income levels and standards of living. Therefore, unit transfer with income adjustments has been applied, for example, by using Purchase Power Parity indexes. However, this adjustment does not take correct for differences in preferences, environmental conditions, and cultural and institutional conditions between countries. Very few studies have tested the impacts of these factors on valuation results. Ready *et al*. (1999) conducted the same CV study in five European countries (the Netherlands, Norway, Portugal, Spain, and the United Kingdom). They found that the transfer error in valuing respiratory symptoms was ±38% in terms of predicting mean willingness-to-pay (WTP) to avoid the symptom in one country from the data of the other countries. The WTP estimates were adjusted with PPP indexes (for the cities the studies were conducted in, since national PPP indexes were not representative for these specific cities). Thus, the remaining differences in the valuations are due to other factors than income.

This same study is also a test of the accuracy of benefit transfer. The observed transfer error should be compared with the variability in the original estimate within a country of ±16 % (estimated using Monte Carlo simulations). These results relate to the valuation of respiratory symptoms that can be linked to air or water pollution, and it is not certain that they are transferable to other environmental goods. Thus, we should perform similar types of validity tests of international benefit transfer for both use and non-use value of biodiversity. This would provide very useful information about the possibilities and uncertainty involved in using the international stock of biodiversity valuation studies when valuing biodiversity in CBA of projects or new policies.

Function transfer

Benefit Function Transfer

Instead of transferring the benefit estimates, the analyst could transfer the entire benefit function. This approach is conceptually more appealing than transferring unit values, because more information is effectively transferred. The benefit relationship to be transferred from the study site(s) to the policy site could again be estimated using either revealed preference (RP) approaches, like TC and HP methods, or stated preferences (SP) approaches, like the CV method and Choice Experiments (CE). For a CV study, the benefit function could be:

$$WTP_i = b_0 + b_1 G_{ij} + b_2 C_i + e \qquad (1)$$

Where WTP_i = the willingness-to-pay of household i, Gij = the characteristics of the environmental good and site j, and Ci = characteristics of household i, and b_0 , b_1 and b_2 are parameters and e is the random error.

To implement this approach the analyst would have to find a study in the existing literature with estimates of the parameters, b_0, b_1, and b_2. Then the analyst would have to collect data on the two independent variables at the policy site. The values of these independent variables from the policy site and the estimates of b_0, b_1, and b_2 from the study site would be replaced in the CV model (1), and this equation could then be used to calculate households' WTP at the policy site.

The main problem with the benefit function approach is the exclusion of relevant variables in the bid or demand functions estimated in a single study. When the estimation is based on observations from a single study of one or a small number of recreational sites or a particular change in environmental quality, a lack of variation in some of the independent variables usually prohibits inclusion of these variables. For domestic benefit transfers researchers tackle this problem by choosing the study site to be as similar as possible to the policy site. The exclusion of methodological variables makes the benefit function approach susceptible to methodological flaws in the original study. In practice, researchers tackle this problem by choosing scientifically sound original studies.

Meta-analysis

Instead of transferring the benefit function from one valuation study, results from several valuation studies could be combined in a meta-analysis to estimate one common benefit function. Meta-analysis has been used to synthesise research findings and improve the quality of literature reviews of valuation studies to arrive at adjusted unit values. In a meta-analysis original studies are analysed as a group, where the results from each study are treated as a single observation into new analysis of the combined data set.[10] This allows researchers to evaluate the influence of the resources' characteristics, the features of the samples used in each analysis (including characteristics of the population affected by the change in environmental quality), and the modelling assumptions. The resulting regression equations explaining variations in unit values can then be used together with data collected on the independent variables in the model that describes the policy site, to construct an adjusted unit value. The regression from a meta-analysis would look like equation (1), but with one

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10 To gain more information from the studies, one can use several estimates from different models within each study (but note that then the estimates are not independent), or the complete data sets (if the studies are quite similar with respect to the type of variables for which data has been or can be collected).

added independent variable C_s = characteristics of the study *s* (and the dependent variable would be WTP_s = mean willingness-to-pay from study *s*.

Smith and Karou's (1990) meta-analysis of TC recreation demand models using both summary TC and CV studies for the US Forest Service's resource planning program, were the first attempts to apply meta-analysis to environmental valuation. Later there have been applications to HP models valuing air quality [Smith and Huang (1993)], CV studies of both use and non-use values of water quality improvements [Magnussen (1993)], and CV studies of groundwater protection [Boyle *et al.* (1994)]. There have also been CV studies of visibility changes at national parks [Smith and Osborne (1996)], of morbidity using Quality of Life Years (QUALY) indexes [Johnson *et al.* (1996)], of endangered species [Loomis and White (1996)], and of environmental functions of wetlands [Brouwer *et al.* (1997)]. Other examples are TC studies of freshwater fishing [Sturtevant *et al.* (1995)] and HP studies of aircraft noise [Schipper *et al.* (1998)]. Only the last two studies are international meta-analyses, including both European and North American studies. All the others, except Magnussen (1993), analyse US studies only.

Many of these meta-analyses of relatively homogenous environmental goods and health effects are not particularly useful for benefit transfer even within the US, where most of these analyses have taken place, because they focus mostly on methodological differences.[11] Methodological variables, like "payment vehicle", "elicitation format", and "response rates" (as a general indicator of quality of mail surveys) in CV studies, and model assumptions, specifications and estimators in TC and HP studies, are not particularly useful in predicting values for a specified change in environmental quality at the policy site. This focus on methodological variables is partly due to the fact that some of these analyses were not constructed for benefit transfer [Smith and Kauro (1990), Smith and Huang (1993) and Smith and Osborne (1996)]. In addition, insufficient, or inadequate information is often reported in the published studies with regards to characteristics of the study site, the change in environmental quality valued, income, and other socio-economic characteristics of the sampled population. Information on the last class of variables would be necessary for international benefit transfer, assuming cross-country heterogeneity in preferences for environmental goods and health effects. [12]

In most of the meta-analyses secondary information was collected on at least some of these site and population characteristics variables or some proxy for them. These variables make it possible to value impacts outside the domain of a single valuation study, which is the main advantage of meta-analysis over the benefit function transfer approach. However, often the use of secondary data, or proxy variables introduces added uncertainty, for example, using income data for a regional

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[11] Carson *et al.* (1996) is an example of a meta analysis of different environmental goods and health effects, which was performed with the sole purpose of comparing results from valuation studies using both stated preference (CV) and revealed preference methods (TC, HP, defensive expenditures and actual market data).

[12] There is some evidence from the literature that attempts to explain cross-country patterns of consumption by allowing tastes to vary across countries; rejecting the controversial hypothesis of Stigler and Becker (1977) that tastes are the same across countries. Pollack and Wales (1987) and Selvanathan and Selvanathan (1993) test this hypothesis by pooling international data and testing the acceptability of restricting the equations of different countries to share common parameters. Even if the hypothesis of common tastes is rejected in both cases, the results of these tests do not determine whether tastes differ between countries, or whether patterns differ as a consequence of differences in household production technologies, or the endowment of non-market environmental amenities used in household production processes. It is also unclear whether these results can be generalised to cross-country preferences for *non-market* environmental goods.

population instead of income data for fishermen at the study site. On the other hand, this secondary data is more readily available at the policy site.

Most meta-analyses caution against the results for adjusting unit values because of the potential biases from omitted variables and specification/measurement of included variables. To increase the applicability of meta-analysis for benefit transfer, one could select studies that are as similar as possible with regards to methodology, and thus single out the effects of site and population characteristics on the value estimates. However, there are usually so few valuation studies of a specific environmental good or health impact, that one cannot undertake a statistically sound analysis.

Accuracy of benefit transfer

There are very detailed guidelines, although disputed, on how to carry out high-quality original valuation studies, i.e. Arrow *et al*. (1993) for CV surveys, however, no such protocol exists for benefit transfer. Smith (1992) has called for the development of a standard protocol or guidelines for conducting benefit transfer studies. Recent studies comparing benefit transfers with new CV studies of the same site, test the validity of benefit transfer, and could provide a valuable input in the development of such guidelines.

Loomis (1992) argues that cross-state benefit transfer in the U.S. (even for identically defined activities) are likely to be inaccurate. His work rejects the hypotheses that the demand equations and average benefits per trips are equal for ocean sport salmon fishing in Oregon versus Washington, and for freshwater steelhead fishing in Oregon versus Idaho.

There are examples of such studies in Norway. Bergland *et al*. (1995) conducted the same CV study of increased use and non-use values for water quality improvements at two Norwegian lakes (let us call them A and B for simplicity). They constructed benefit functions for A and B, and then transferred the benefit function of lake A to value the water quality improvement in lake B, and visa versa. The mean values were also transferred and compared with the original CV estimate, since the two lakes are rather similar with regard to size and type of pollution problem. When selecting the independent variables for the demand function two different approaches were used: i) selecting variables which give the largest explanatory power, and ii) selecting variables for which it is possible to obtain data for at the policy site without having to do a costly survey. The last approach would ease future transfers, but could give less reliable estimates. Several tests for transferability were conducted, but all indicated a lack of transferability, in statistical terms (i.e. transferred and original values are significantly different at the 5 % level). However, the mean values differed by “only” 20-30 %, and for many uses (e.g. cost-benefit analysis) this level of accuracy could be acceptable. In one lake the transferred values were higher and in the other they were lower than the estimate from the original study. Thus, from this study one cannot conclude what procedure would produce the highest values.

While Bergland *et al.* (1995) test benefit transfers spatially by conducting two CV studies at the same time, Downing and Ozuno (1996) test benefit transfer (use values only) both spatially and intertemporally, through CV and TC models of recreational angling at eight bays along the Texas coast. Using a 5 % significance level, they found that 91-100 % of the estimates were not transferable across bays (but 50-63% of within-bay estimates were transferable across time). Like Bergland *et al.* (1995) they conclude that geographical benefit transfer is generally not statistically reliable. Brouwer and Spaninks (1999) reached the same conclusion in their CV studies of use and non-use values of amenities (meadow birds and flowery ditch sides) of two Dutch peat meadow sites. The original CV study gave significantly higher estimates than transferred CV estimates from the other peat meadow area, but estimates of the mean WTP/household/year were only 20 per cent different with benefit function transfer. In their international benefit transfer test Ready *et al*. (1999) found that the transfer error in valuing respiratory symptoms (that could be caused by air and water pollution) of ±38%. Whether errors of this size are acceptable depend on the use of the value estimates [Navrud and Pruckner (1997)].

Challenges to benefit transfer and policy implications

Results from validity tests show that the uncertainty in benefit transfers both spatially and intertemporally could be quite large. Thus, benefit transfer should be applied to uses of environmental valuation where the demand for accuracy is not too high. This means using benefit transfer in cost-benefit analyses of projects and policies, but more caution should be exercised in using transferred values in environmental costing and accounting exercises, and in particular Natural Resource Damage Assessments (NRDA) and calculations of compensation payments in general (see Section 1).

Benefit transfer is less than ideal, but so are most valuation efforts in the sense that better estimates could be obtained if more time and money were available. Analysts must constantly judge how to provide policy advice in a timely manner, subject to the resource constraints they face. Benefit transfer methods may be particularly useful in policy contexts where rough or crude economic benefits may be sufficient to make a judgement regarding the advisability of a policy or project. Therefore analysts should compare the benefits of increased accuracy of the benefit estimates (when going from a benefit transfer exercise to a new, original valuation study) with the costs of making the wrong decision based on the benefit transfer estimate.

There are four main difficulties or challenges in benefit transfer:

1. Availability and quality of existing studies;

2. Valuation of new policies or projects are difficult in respect of:

   expected change resulting from a policy is outside the range of previous experience

   discrete versus marginal change

   increase versus decrease in environmental quality;

3. Differences in the study site(s) and policy sites that are not accounted for in the specification of the valuation model or in the procedure used to adjust the unit value; and

4. The determination of the "extent of the market". To calculate aggregated benefits the mean benefit estimate has to multiplied with the total number of affected households (i.e. households that find their well-being affected by the change in the quality of the environmental good). There is a need for guidelines on how to determine the size of the affected population.

The policy response to these main challenges in benefit transfer could be i) the development of improved benefit transfer techniques and a protocol for benefit transfer (including guidelines on how to determine the "size of the market", and ii) the establishment of a database of environmental valuation studies. There have been recent advances in both these areas. Based on a review of value transfer studies and validity tests of transfer, Brouwer (1999) propose a seven-step protocol for good practice when benefit transfer is used in CBAs. The web-based database EVRI (Environmental Valuation Reference Inventory, www.evri.ec.gc.ca/EVRI/) now contains about 700 valuation studies. The majority of these studies are from North America, but the number of European and Asian studies captured in this database is steadily increasing (see also Navrud 1999 for a favourable evaluation of the suitability of EVRI for capturing European valuation studies). Thus, there is a need to increase the number of existing valuation studies captured in this database, but there is also a need for new, original valuation studies, which have been designed with benefit transfer in mind.

References

Arrow, K.J., R. Solow, E. Leamer, P. Portney, R. Radner and H. Schuman (1993). “Report of the NOAA Panel on Contingent Valuation”. Federal Register, 58, 4601-4614 (January 15, 1993).

Asian Development Bank (1996): Economic Evaluation of Environmental Impacts. A Workbook. Parts I and II. Environment Division, Manilla, Philippines. March 1996.

Bergland, O., K. Magnussen and S. Navrud (1995). “Benefit transfer: Testing for Accuracy and Reliability”. Discussion Paper D-03/95, Department of Economics, Agricultural University of Norway. Paper presented at the sixth annual conference of The European Association of Environmental and Resource Economists (EAERE), Umeaa, Sweden, June 17-20, 1995. In Florax, R.J.G.M., P. Nijkamp and K. Willis (eds.) 1999: Comparative Environmental Economic Assessment: Meta Analysis and Benefit Transfer. Kluwer Academic Publishers.

Boyle, K.J., G.L. Poe and J.C. Bergstrom (1994). “What do we know about groundwater values? preliminary Implications from a meta analysis of contingent valuation studies”. American Journal of Agricultural Economics 76 (December 1994); 1055-1061.

Brouwer, R. and F.A. Spaninks (1999). “The validity of environmental benefit transfer: further empirical testing”. Environmental and Resource Economics, 14 (1); 95-117.

Brouwer, R., I.H. Langford, I.J. Bateman, T.C. Crowards and R.K. Turner (1997). “A Meta-analysis of Wetland Contingent Valuation Studies”. CSERGE Working Paper GEC 97-20. Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia and Unviersity College London; 76 pp.

Carson, R.C., N.E. Flores, K.M. Martin and J.L. Wright (1996). “Contingent Valuation and Revealed Preference Methodologies: Comparing the Estimates for Quasi-Public Goods”. Land Economics, 72, 80-99.

Desvousges, W.H, F. R. Johnson and H.S. Banzhaf (1998). Environmental Policy Analysis with Limited Information.Principles and Applications of the Transfer Method. New Horizons in Environmental Economics. Edward Elgar, Cheltenham, UK and Northampton, MA, USA.

Downing, M. and T. Ozuno (1996). “Testing the reliability of the benefit function transfer approach”. Journal of Environmental and Resource Economics and Management, 30, 3; 316-322.

European Community (1993). “Towards sustainability: A European Community programme of policy and action in relation to the environment and sustainable development”. Office for Official Publications of the European Communities, Brussels.

European Commission – DG XII (1995). ExternE. Externalities of Energy. Vol. 2. Methodology. European Commission Directorate General-XII Science Research and Development, Report EUR 16521 EN, Brussels.

European Commission - DG XII (1999). ExternE - Externalities of Energy. Vol. 7: Methodology 1998 Update. European Commission (EC) - Directorate General (DG) XII. Report EUR 19083, Brussels.

GARP II (1999). Final report of the Green Accounting Research Project II to the European Commission – DG XII, RTD Programme “Environment and Climate”, Contract ENV4-CT96-0285, November 1999.

Johnson, F.R., E.E. Fries and H.S. Banzhaf (1996). Valuing Morbidity: An Integration of the Willingness-to-Pay and Health Status Index Literatures. December 1996. Triangle Economic Research and Duke University.

Loomis, J.B. (1992). “The evolution of a more rigorous approach to benefit transfer: Benefit function transfer”. Water Resources Research, 28 (3), 701-706.

Loomis, J.B. and D.S. White (1996). “Economic benefits of rare and endangered species: summary and meta-analysis”. Ecological Economics 18; 197-206.

Magnussen, K. (1993). “Mini meta analysis of Norwegian water quality improvements valuation studies. Norwegian Institute for Water Research, Oslo. Unpublished paper, 29 p.

Navrud, S. (1993). “Economic value of biological diversity in Norway”. Scandinavian Forest Economics, no. 34 (1993); 74-97.

Navrud, S. and K. Veisten (1997). “Using contingent valuation and actual donations to bound the true willingness-to-pay”. Presented at the seventh annual conference of the European Association of Environmental and Resource Economists, Lisbon, June 27-29 1996. Revised version May 1997.

Navrud, S. and G.J. Pruckner (1997). “Environmental Valuation - To Use or Not to Use?” A Comparative Study of the United States and Europe. Environmental and Resource Economics, 10; 1-26.

Navrud, S. (1992). Pricing the European Environment. Scandinavian University Press / Oxford University Press, Oslo, Oxford, New York.

Navrud, S. (1999). Pilot Project to assess Environmental Valuation Reference Inventory (EVRI) and to Expands Its Coverage to the EU. Report to the European Commission, DG XI – Environment. Report July 1, 1999.

NOU (1997). “27: Cost.-Benefit Analysis- Principles for evaluation of public projects and policies” (In Norwegian). Norwegian Ministry of Finance, Oslo.

NOU (1998). “16 Cost-Benefit Analysis – Guidelines for economic evaluation of public projects and policies” (in Norwegian). Norwegian Ministry of Finance, Oslo.

OECD (1989). Environmental Policy Benefits: Monetary Valuation. Organisation for Economic Co-operation and Development (OECD), Paris, ISBN 92-64-13182-5.

OECD (1994). Project and Policy Appraisal: Integrating Economics and Environment. Organisation for Economic Co-operation and Development (OECD), Paris, ISBN 92-64-14107-3.

OECD (1995). The Economic Appraisal of Environmental Projects and Policies. A Practical Guide. Organisation for Economic Co-Operation and Development (OECD), Paris, ISBN 92-64-14583-4.

Pollack, R. and T. Wales (1987). “Pooling International Consumption Data”, Review of Economics and Statistics, 69, 90-99.

Ready, R., S. Navrud, B.Day, R. Doubourg, F. Machado, S. Mourato, F. Spanninks and M. X. V. Rodriquez ( ). “Benefit Transfer in Europe: Are Values Consistent Across Countries?” Paper presented at the EVE Workshop on Benefit Transfer, Lillehammer, Norway, October 14-16, 1999. EU Concerted Action “Environmental Valuation in Europe” (EVE).

Rowe, R.D., C.M. Lang, L.G. Chestnut, D.A. Latimer, D.A. Rae, S.M. Bernow and D.E. White (1995). The New York Electricity Externality Study. Volumes I and II. Hagler Bailley Consulting, Inc., Oceana Publications Inc.

Selvanathan, S. and E. Selvanathan (1993). “A Cross Country Analysis of Consumption Patterns”, Applied Economics 25, 1245-1259.

Schipper, Y., P. Nijkamp and P. Rietveld (1998). “Why do aircraft noise value estimates differ?” A meta-analysis. Journal of Air Transport Management 4; 117-124.

Smith, V.K. (1992). “On separating defensible benefit transfers from smoke and mirrors.” Water Resources Research 28, (3) 685-694.

Smith, V.K. and J.C. Huang (1993). “Hedonic models and air pollution. Twenty-five years and counting”. Environmental and Resource Economics 3, 381-394.

Smith, V.K. and Y. Karou (1990). “Signals or noise? Explaining the variation in recreation benefit estimates”. American Journal of Agricultural Economics, 72 (2). 419-433.

Smith, V.K. and L. Osborne (1996). “Do Contingent Valuation Estimates Pass a “scope” Test? A Meta Analysis”. Journal of Environmental Economics and Management, 31 (3), 287-301.

Stigler, G. and G. Becker (1977). De Gustibus Non Est Disputandum, American Economic Review, 67 (2), 76-90.

Sturtevant, L.A., Johnson, F.R. and W.H. Desvousges (1995). “A meta-analysis of recreational fishing”. Triangle Economic Research, Durham, North Carolina, USA.

Tamborra, M. (1999). “Towards a Green Accounting System for the European Union: the Contribution of GARP II”. FEEM newsletter no. 2 1999 (November 1999); 16-18. Fondazione Eni Enrico Mattei (FEEM), Milano, Italy.

UNEP (1995). “Global Biodiversity Assessment”. (Chapter 12: Economic values of Biodiversity). United Nations Environment Program (UNEP), Cambridge University Press. ISBN 0-521-56481-6.

Walsh, R.G., D.M. Johnson and J.R. McKean (1990). Nonmarket values from two decades of research on recreation demand. Advances in applied micro-economics, vol. 5, 167-193, JAI Press Inc.

Walsh, R.G., D.M. Johnson and J.R. McKean (1992). "Benefit transfer of outdoor recreation demand studies", 1968-1988. Water Resources Research, 28, (3) 707-714.

Whittington, D. (1998). "Administering Contingent Valuation Surveys in Developing Countries", World Development, 26, (1), 21-30.

CHAPTER 5:
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EVALUATING MULTIDIMENSIONAL BIODIVERSITY POLICY: WHAT CAN WE LEARN FROM CONTINGENT VALUATION STUDIES OF BIOLOGICAL RESOURCES IN THE CONTEXT OF RURAL AMENITIES?

Multidimensional biodiversity policy

Biologists have come to see the living world as a hierarchy of levels of organisation, from molecules to the biosphere, including the cell, the individual, the population, the ecosystem and the landscape levels. Each level has a particular *structure*: some building blocks or *components* are organised to form a whole building. (Cells, for example, are organised in tissues to make up individuals). In turn, a component at one level appears as a whole building when seen at lower levels. (A cell is itself made up of molecules).

The term biodiversity refers to the diversity of components at all these levels, although only three are mentioned in usual definitions, which focus on gene, species and ecosystem diversity alone.

The properties of life at any level are different from what could be predicted based on properties of components. With life, the whole is different from the sum of the parts. At each level, *structure* matters. New *functions* and *dynamics* emerge as one passes from a lower to an upper level of organisation. For example, vegetation succession is a dynamic process that only appears at the ecosystem level.

Living resources are crucial for people’s wellbeing. They supply a huge variety of goods and services: basic foodstuffs, raw materials for industry, medicines, soil cover, flood control, nutrient recycling, aesthetics and recreation. To the extent that these services are scarce, people are prepared to trade off other valuable things for them — that is: people have values for these services. Hence, living resources, the assets providing these service flows, also have value.

In this paper we refer to value in this strict sense alone. This is not to say that we ignore here ‘existence values’ people may hold for particular living creatures, ecosystems or landscapes, in the sense people would feel worse off if knowing that these things have disappeared for good. If people are prepared to give up other valuable things to avoid extinction, then a value, even in this strict sense, exists. However, for the purposes of this paper, we ignore values that living creatures, ecosystems or landscapes may have in themselves, that is: intrinsic values.

The valuable services of living resources are often identified with components themselves. For example: (1) birds for a birdwatcher; (2) particular genes for a plant breeder who creates new, disease-resistant, varieties; or (3) different landscape elements that enhance the aesthetic experience of visitors and local dwellers. Sometimes, these services do not depend on use but only on knowing about the continued existence of a living component, as it is the case with people having existence values for whales. Very often, valuable services are provided not directly by components, but by functions or dynamic processes at particular levels of organisation: (1) soil protection by vegetation; (2) nutrient recycling by bacteria in soils; or (3) the colonisation of burnt areas by seeds coming from remaining vegetation patches.

Although some valuable functions and dynamic processes are enhanced by higher levels of biodiversity, this is not always the case; even when it is, the relationship may not be simple and self-evident [Odum (1971)]. So, some valuable services of biological resources are not linked to biodiversity levels. However, in this paper, we indistinctly use the term biodiversity polices for all policies aimed at preserving the services of biological resources.

Many public policies, particularly biodiversity polices, have an impact on living resources. They can change their components, structure, function or dynamics at different levels. Thus, they will likely change the services provided by biological resources, and, therefore, the wellbeing of those benefiting from those services.

How should these welfare effects of policy affect policy decisions? The economist's answer assumes that the relevant value concept is people's values for the affected services, that is: the amounts of other valued things (income) they are prepared to give up for these services.

Accordingly, these values should directly enter as an input into any policy decision implying these trade-offs of economic income for the services of biological resources. Examples of these decisions are: (1) deciding whether to go ahead with a biodiversity scheme (by weighing conservation benefits against the corresponding social cost); (2) selecting the best biodiversity policy mix (on social-welfare grounds); and (3) prioritising lines of action for a biodiversity strategy when, as always, policy funds are scarce.

Determining how the services of biological resources are affected by policy is a task for biologists, landscape ecologists and other scientists. The applied economist's task is to measure how people's wellbeing is affected by these changes in service levels, by measuring people's willingness-to-pay (WTP) for either avoiding degradation or getting improvements in service levels.

As there are no markets for many services of living resources, WTP cannot be directly inferred from market demands for these services. Hence, it has to be measured with non-market valuation techniques. These are based on either:

1. value-eliciting surveys (that is: *stated-preference techniques*, including contingent valuation, contingent ranking and choice experiments); or
2. value-traces implicit in observed behaviour and revealed by modelling this behaviour (that is: *revealed-preference techniques*, using hedonic-price, travel-cost or averting-behaviour models).

Most policy impacts on biological resources are inherently multidimensional in that multiple services of these resources are affected. There are two possible causes for this. First, very often, more than one component (that is: more than one individual, species or ecosystem) supplying valuable services is directly affected. Second, even if only one component is directly affected, inter-component linkages usually lead to diverse components, functions or processes, and thus services, being affected. Alternatively, the affected component may deliver several services. One example is the removal of tree cover on slopes to create pasture. This may directly affect tree cover alone. However, services as diverse as aesthetics, recreation, soil conservation, flood control and habitat, will all change as a result.

This paper explores the issues raised by this multidimensional nature of biodiversity policies when evaluating these policies. Most of the empirical basis for the discussion comes from contingent-valuation studies of biological resources in the context of rural amenities (for a review see Santos, 1998). However, the general arguments and methods apply to most policymaking contexts, independently of the particular resources, types of services and valuation techniques used. The discussion is conducted with an eye on the policy relevance of the reported methodological advances. The basic questions are (1) what can we learn from the reported research about policy recommendations based on standard valuation studies of biodiversity policies; and (2) how can we improve the methods used in these studies.

The basic problem addressed here, substitution effects between multiple policy impacts on biological resources, is first analysed in Section 2. Section 3 reviews empirical evidence on the magnitude of substitution effects and the implied independent-valuation-and-summation (IVS) bias. Section 4 derives implications for valuation studies of multidimensional biodiversity policies; Section 5, implications for benefit aggregation in policy evaluation; and Section 6, implications for the selection of optimal policy mixes, namely for prioritising action lines when designing biodiversity strategies.

Substitution effects in the valuation of multiple-service changes I: theory

Biodiversity policies typically affect multiple services of living resources at the same time. Are people's values for each service dependent on the levels of the other services that are also changing? Theoretical reasons and empirical evidence discussed below lead us to believe that this is very often the case. And, if this is so, it has important practical consequences for:

(1) valid implementation of non-market valuation techniques to measure WTP for the services of biological resources affected by policy;
(2) valid benefit aggregation across multiple services, as it is usually required for policy evaluation;
(3) valid cost-benefit procedures used to select the best policy mix for biodiversity conservation.

If these issues are not adequately addressed, the results of policy analysis will be biased and, depending on the magnitude of the bias, there will be a non-trivial probability of recommending the wrong policy decision.

So, let us return to the initial question: are people's values for a service of a biological resource dependent on the levels of other such services? Economic theory provides a strong rationale for an affirmative answer: substitution effects.

Let us exemplify with a simple multi-site problem. Suppose that we want to value the benefits of conserving the same ecosystem, say deciduous woodland, in two areas located close to each other and visited by the same population. Very likely, each individual visitor perceives the two woods as close substitutes. Thus, suppose that a conservation scheme for wood A alone is considered, with woodland disappearing from area B. Suppose an individual is asked how much is he prepared to pay for wood A and that his answer is US$ 10. Probably, because the areas are similar, he would also pay US$ 10 for wood B with no conservation of wood A. If the two woods are perfect substitutes, a programme jointly conserving woods A and B will also be valued at US$ 10. If conserving a single wood costs US$ 8, then conserving either wood A or wood B in isolation is worthwhile, but jointly conserving woods A and B is not. In cases where substitution is less severe, it may simply happen that the sum of benefits for each wood in isolation is larger than the joint benefit for the two woods together. If the two woods are perfect substitutes, the fact of one of them being conserved implies that there is no benefit at all from conserving the other. However, in general, if there is less-than-perfect substitution, we only know that the benefit of conserving one wood will depend on whether the other wood is to be conserved.

The same example could be extended to many other settings: we could ask whether people's WTP for conserving the lynx depends on whether the tiger will also be conserved? or whether WTP for conserving all threatened felines altogether is smaller than the sum of WTP for each species considered in isolation (i.e. if all others were to remain threatened)?

Note that substitution effects between two services of biological resources has two sources [Santos (1998)]. First, both services may satisfy the same basic need; thus, they are *substitutes in utility* (as in the example of the two woods). Second, even if they are not substitutes in utility (as may be the case with felines), at least they compete for the same budget. The WTP for one service is reduced after having paid for another service because income is now lower.

There are two conclusions from the examples of substitution effects:

1 the value for one service of biological resources is reduced if the level of a substitute service is increased (the opposite occurs for services that are complements for each other);
2. the benefit of a policy providing two services that are substitutes for each other is smaller than the sum of the benefits of providing each service in isolation.

The first conclusion implies that, to value a change in one service level, the other services should be held constant at levels that are known by the evaluator—they are a relevant part of the valuation context. The second is related to a frequent bias when aggregating benefits across services: the independent valuation and summation (IVS) bias.

The second conclusion also means that, for economics as well as for biology, the whole is different from the sum of the parts. Therefore, assessing the welfare effects of multidimensional changes in biological resources is a task in which the complexity of value relationships compounds the complexity of the living world.

Substitution effects in the valuation of multiple-service changes II: empirical evidence from contingent-valuation studies

What about the empirical evidence on the magnitude of substitution effects and the related aggregation (IVS) bias?

Evidence discussed here comes from a contingent valuation (CV) study of the wildlife and landscape-conservation benefits of the Pennine Dales Environmentally Sensitive Area (ESA) scheme in the UK. This example was selected from a set of CV studies of substitution effects using similar methods and achieving similar conclusions, which have been (or are still being) conducted in the UK and Portugal [see Santos (1997) and Santos (1998)].

A discrete choice approach to questioning was adopted in this CV application. In a field survey of 422 visitors to the Pennine Dales ESA, respondents were asked to choose between (1) the continuance of a specified ESA scheme at a given tax-rise cost, and (2) giving up the scheme with no tax increase.

The valued ESA schemes represented different policy mixes across respondents. Hypothetical policy mixes were built by combining three basic programmes:

- P1 - the conservation of existing stone walls and field barns;

- P2 - the conservation of flower-rich hay meadows;

- P3 - the conservation of remaining small broad-leaved woods.

The first programme was perceived by most respondents as a purely aesthetic or cultural-landscape attribute. The second, as both an aesthetic attribute and an habitat for important species of wild flowers and ground-nesting birds and the third programme was mostly perceived as an important ecosystem and scarce wildlife habitat, and, secondarily, as an aesthetic landscape attribute.

The three programmes were expected to be complements *in utility*, at least in aesthetic terms. In fact, as some of the most characteristic visual attributes of a cherished countryside area these individual attributes were expected to magnify the visual impact of each other. Hence, the aesthetic impact of the whole experience would be larger than the sum of the partial impacts of the attributes in isolation. Yet, meadows and woods, perceived as habitats for wildlife, could satisfy similar needs of visitors and could, thus, behave as substitutes in utility.

Survey results were analysed using a model allowing for negative or positive substitution effects of any magnitude; this enabled researchers:

1. to observe the sign of substitution effects (a negative sign means programmes are substitutes, that is they reduce the value of each other; a positive sign means programmes are complements, that is they increase the value of each other);
2. to test for statistical significance of these signs; and
3. to measure their magnitude, and hence that of the associated IVS bias.

The estimated model is presented in Table 5.1. From this, we observe that the three estimated parameters for substitution effects (that is: all programme-interactions P1*P2; P1*P3, and P2*P3) are negative, implying that all of the three programmes are substitutes in valuation. Thus, in spite of some expected complementarity in utility, only substitution in valuation was found. This can easily be explained by the existence of a common budget for which WTP for all programmes compete, which, as shown above, is one of the two sources for substitution in valuation.

Let us now look at the statistical significance and compared sizes of the several estimated substitution effects. Note that only the parameter estimate for P2*P3 is significantly negative with a 1% probability of error; while that for P1*P3 is also so with a 5% probability of error; and that for P1*P2 is not significantly different from zero even if we accept a 10% probability of error for the test. Thus, as expected:

(1) the strongest substitutes are meadows and woods, both habitat attributes; and
(2) the substitution effects between aesthetic (walls and barns) and habitat (meadows and woods) attributes, that is P1*P3* and P1*P2, are weaker, with the latter not statistically significant.

Table 5.1 Model of WTP for different alternative policy mixes for the Pennine Dales ESA scheme

Dependent variable: WTP for different conservation policy mixes in the Pennine Dales (£/year)	**Prediction success (%)**				
# of observations: 2290			Actual		Total
Correct predictions: 78.0%			would	wouldn't	
–2log-likeli. ratio: 978.84	Predicted	would pay	50.6	15.9	66.6
Degrees of freedom: 13		wouldn't	6.0	27.4	33.4
Level of signif.: P<0.0001	Total		56.7	43.3	100.0

Variables	**Parameter estimates**	**t-ratios**	**sign. level**	**Labels**
P1	32.62	3.259	0.01	Programme 1: stone walls and field barns (0–1)
P2	21.80	2.274	0.05	Programme 2: flower-rich meadows (0–1)
P3	44.58	4.659	0.01	Programme 3: broad-leaved woodland (0–1)
P1*P2	–10.33	–1.158		Interaction between programmes 1 and 2 (0–1)
P1*P3	–20.08	–2.269	0.05	Interaction between programmes 1 and 3 (0–1)
P2*P3	–30.47	–3.443	0.01	Interaction between programmes 2 and 3 (0–1)
P1*INCOME	0.000755	2.516	0.01	Interaction between programme 1 and income (£)
P2*INCOME	0.000896	2.930	0.01	Interaction between programme 2 and income (£)
P3*INCOME	0.000394	1.312	0.10	Interaction between programme 3 and income (£)
P1*FIRSTP1	8.96	1.192		Programme 1 when first in preferences (0–1)
P2*FIRSTP2	38.35	4.640	0.01	Programme 2 when first in preferences (0–1)
P3*FIRSTP3	20.84	2.328	0.01	Programme 3 when first in preferences (0–1)
κ	47.17	20.300	0.01	Dispersion parameter of the logistic random term

As regards the absolute size of substitution effects, we note that P3 reduces the value of P2 by £30.47 (see Table 5.1), that is by 53% of the value of P2 when valued in isolation (this is £57.78, see Table 5.2). For programmes that are less perfect substitutes for each other than these, the reduction in value is lower. For example, P1 reduces the value of P2 by only 18% of the value of P2 when valued in isolation. Table 5.3 shows the reduction in value of each programme when one or both of the other programmes are also provided.

What does this data show about the magnitude of the IVS bias implied by substitution effects of these sizes?

To provide an answer, we used the model in Table 5.1 to estimate the WTP for all possible policy mixes (considering the average individual in the sample in terms of income and relative preferences for the three programmes). The resulting point estimates are presented in the second column of Table 5.2 as E (WTP| $\mathbf{X}$, $\boldsymbol{\beta}$). These are average WTP amounts conditional on explanatory variables, $\mathbf{X}$ and parameters, $\boldsymbol{\beta}$. The first column in this table represents the relevant policy mix in the (P1, P2, P3) vector form, with the Pi's being dummy variables with 'ones' indicating that the corresponding programme is in the policy mix and 'zeros' that it is not.

Table 5.2 Model-based point estimates of conditional means of WTP for different alternative policy mixes for the Pennine Dales ESA scheme, as compared to the corresponding IVS results

Policy-mix P1, P2, P3	E (WTP|X,β)[a]	IVS[a]	IVS bias (in %)
(1, 0, 0)	56.42	56.42	0.00
(0, 1, 0)	57.78	57.78	0.00
(0, 0, 1)	59.15	59.15	0.00
(1, 1, 0)	103.87	114.20	+9.95
(1, 0, 1)	95.49	115.57	+21.03
(0, 1, 1)	86.46	116.93	+35.24
(1, 1, 1)	112.47	173.35	+54.13

Note: [a] values in £ per household per year.

Next, the IVS results for each policy mix were calculated by summing the appropriate values in column 2. IVS results are presented in column 3. In column 4, we present IVS biases as a percentage of the 'true' value. Model-based estimates E (WTP| **X**, **β**) are considered 'true' (that is: valid) WTP values, in that they are estimated from a model accounting for substitution effects, and are, therefore, *not* prone to IVS bias. The results in the fourth column show that independently valuing each programme and then aggregating leads to overestimation of the true value of a policy mix by between 10 and 35 per cent for the policy mixes that include two programmes, and by 54 per cent for all three programmes.

Most of the estimated IVS biases are statistically very significant. But what is the practical significance, in terms of the probability of this bias leading to the wrong recommendations for policymakers?

To assess this, we compared (1) the benefits implied by valid estimates from the model, and (2) the benefits implied by IVS estimates, with policy costs estimated elsewhere [Santos (1997)]. The conclusion was that policy recommendations were *not* sensitive to the differences between model-based (valid) and IVS (biased) benefit estimates. Whatever the policy mix, the right decision was always to proceed with the policy, as costs were sufficiently low to be offset by valid (the lowest) benefit estimates. So, the fact that IVS benefit estimates overestimated true benefits is, in this case, immaterial for policymaking purposes.

However, in a similar case study in Portugal [Santos (1997)], this was not the case. For some policy mixes, costs offset the valid (model-based) benefit estimates but not IVS (biased) estimates. So, the right decision was not to proceed with those policy mixes, but based on IVS (biased) estimates we would recommend proceeding with them.

All these concerns are strongly reinforced because very general theoretical models predict that the IVS bias increases with the number of components (programmes) in the policy mix [Hoehn and Randall (1989)]. For many real-world biodiversity programmes, the number of services of biological resources affected is indeed very large, accentuating the IVS bias. Adequately accounting for this source of bias is, therefore, strongly advised.

Implications for non-market valuation studies of services of biological resources

Policy impacts on biological resources usually affect multiple services of these resources. Some other services can change as a result of non-policy-related causes, for example: the extinction of tigers could independently occur while implementing a programme to preserve the lynx. As shown in previous sections, people's values for one service depend on the levels of other services. This has important consequences for the implementation of valuation techniques for multiple-service biodiversity policies.

(1) First, the best way to estimate multidimensional policy benefits is valuing exactly the same multiple-service change in one single step. This 'automatically' takes into account substitution effects.

(2) Second, when the interest is to value a change in one single service (e.g. habitat for species A), we should need to specify not only the change in the service we want to value, but also the levels of other services (e.g. habitat for species B and C) and whether these are to change simultaneously with the service to be valued. That is: other services are also part of the relevant context for the valuation task, even when the interest is valuing a single service.

(2a) The implication of this second consequence for CV is immediate: levels of those other services should be included in the scenarios presented to CV survey respondents [see Mitchell and Carson (1989) and Arrow *et al*. (1993)]. Other stated-preference techniques, such as choice experiments, are rather flexible in this respect, as they allow researchers to ask respondents to value a variety of service changes at the same time.

(2b) For revealed preference techniques, the implication is to include other services' levels in the behavioural models used to infer values people hold for a specific resource.

(2c) When benefit estimates are to be transferred from past studies, the implication is to make sure that levels of other services in the original study (from which we want to transfer benefit estimates) are similar to those levels in the policy context (for which we want to transfer those estimates). This coincidence is highly unlikely, which may lead to important biases in transferring benefit estimates for policy evaluation [Boyle and Bergstrom (1992)].

The best solution (1) is rarely possible in practice, as it would imply carrying out an original study for each policy evaluation problem. The other implications are straightforward in principle but raise an enormous number of difficult practical implementation problems. For example, how best to describe multiple-service changes in a short CV interview? How to secure data about simultaneous changes in travel cost models?

Implications for benefit aggregation across multiple services of biological resources

For most applied policy analyses, analysts rely on WTP estimates transferred from past studies to build benefit estimates for the policy under evaluation. In general, past studies are searched for WTP estimates supposedly applying to each service that is changed by the policy under evaluation (that is: WTP for recreational fishing, for landscape amenities, for existence values for a wildlife species). Eventually all these transferred WTP estimates are aggregated across services to provide a benefit estimate for the overall multiple-service policy [Desvousges *et al*. (1998)]. As shown in previous sections, this aggregation procedure is prone to IVS bias, which (we have reasons to suppose) is potentially very large for multidimensional biodiversity policies. So, these usual procedures are likely to increase the probability of recommending the wrong policy decision.

What is the practical alternative? In most policymaking contexts, it is impossible to commission an original study for each policy, because of budget and time constraints. However, given the potential magnitude of the bias (hence, the likelihood and expected cost of wrong decisions being taken), further research on substitution effects is recommended, to gain an understanding for the magnitude of these effects under different circumstances, and with different types of biological resources. Only then can we expect to derive the appropriate adjustment factors to correct for IVS biases.

Sequential cost-benefit analysis for the selection of an optimal policy mix for biodiversity

Very often, policymakers are not concerned with evaluating a proposed policy for a biodiversity issue, but in determining an optimal policy mix from a set of possible alternatives. This is particularly clear when several lines of action need to be prioritised for inclusion in a general biodiversity strategy, under conditions of scarce funds for policy implementation. What is the proper technique to use for these purposes when substitution effects are expected to be significant, as it is the case with multidimensional biodiversity policies?

Substitution implies that the value of a particular policy component depends, in general, on which other components are to be included in the policy mix. Under these circumstances, the only way to determine an optimal policy mix is to evaluate the (successively more inclusive) policy mixes that will result from sequentially adding up more components to the mix. This is sequential cost-benefit analysis and requires sequential values of policy components. The basic approach was proposed in Santos (1996). It is analysed in detail and applied to policy evaluation in Santos (1998).

Briefly, the method proceeds as follows. First, identify the most complete biodiversity policy mix that is possible (that is: unconstrained by social cost or financial budget). Second, divide this mix into a number of ‘programmes’. Each programme comprises a particular subset of valued biodiversity components or functions. Components included in different programmes should be independent in production, so that policymakers may choose to implement any possible mix of the programmes once this is selected as the best. The problem for policy makers is how to select the single best possible mix. There are two policy settings defining what ‘best’ means: in the first, ‘best’ means welfare maximising, and identifying it implies using a social cost-benefit frame; in the second, policy funds are budget constrained, and thus ‘best’ means the one making the best use of available funds.

For the solution for the social cost-benefit problem, we should sequentially evaluate the welfare effects of adding particular programmes to a previously constituted mix. If the evaluation is considered positive (that: is if the social benefit/cost ratio is larger than one), then the programme is added to the previous mix. Another candidate programme is considered next, and so on until there are no more remaining programmes with social benefits offsetting the corresponding social costs.

If we consider the financial budget constrained approach, a simpler non-sequential approach is possible. This implies estimating the benefit of every single programme-mix and selecting the mix generating the highest benefit while not exceeding the available financial budget.

To illustrate the sequential procedure, consider the Pennine Dales ESA example we have followed throughout the paper. First, note that we need sequential benefits of each programme. These are easily calculated from Table 5.2 by taking the appropriate differences between the model-based estimates for the different mixes in column 2. The results are presented in Table 5.3. In parentheses we present the same benefit estimates, but expressed in relative terms: as a percentage of the benefits of the same programme when estimated in isolation. These provide a clearer idea of the rising magnitude of substitution effects as we go on further in the sequence, i.e.: as more and more programmes are already included in the mix.

Table 5.3 Point estimates of sequential values of programmes for the Pennine Dales ESA scheme when valued in different valuation sequences [a]

Programme	Initial policy mix						
	(0, 0, 0)	(1, 0, 0)	(0, 1, 0)	(0, 0, 1)	(1, 1, 0)	(1, 0, 1)	(0, 1, 1)
P1	56.42 (100.0)	–	46.09 (81.7)	36.33 (64.4)	–	–	26.00 (46.1)
P2	57.78 (100.0)	47.45 (82.1)	–	27.31 (47.3)	–	16.98 (29.4)	–
P3	59.15 (100.0)	39.07 (66.1)	28.68 (48.5)	–	8.60 (14.5)	–	–

Note: [a] values in £ per household per year; values in parenthesis are expressed as percentage of the value of the corresponding programme when valued in isolation.

The next stage is to estimate the sequential costs of programmes, which was done elsewhere [Santos (1997)]. Then, sequential benefits for each programme (dependent on policy mix) are divided by costs to provide sequential benefit/cost ratios. These are presented in Table 5.4, using upper-bound social cost estimates. IVS results (that is: using non-sequential, IVS, values for benefits), which do not account for substitution effects (and are, thus, biased), are presented in parentheses. The objective is to illustrate the sensitivity of results and policy recommendations to the IVS bias when selecting an optimal policy mix.

From Table 5.4, we conclude that all programmes pass the cost-benefit test whatever the evaluating sequence we adopt. Therefore, from a strict social-welfare perspective, the optimal mix should include all programmes. Of course, this is not the case if available budget for policy is not sufficient for this mix. (The criterion to use in this case is described above).

Table 5.4 Sequential benefit/cost ratios for conservation programmes in the Pennine Dales ESA, when evaluated in different valuation sequences (that is: with different initial policy mixes)

Programme	Initial policy mix						
	(0, 0, 0)	(1, 0, 0)	(0, 1, 0)	(0, 0, 1)	(1, 1, 0)	(1, 0, 1)	(0, 1, 1)
P1	3.89 (3.89)[a]	–	4.35 (5.32)	2.51 (3.90)	–	–	2.45 (5.31)
P2	3.35 (3.35)	3.55 (4.32)	–	1.58 (3.34)	–	1.27 (4.32)	–
P3	113.08 (113.08)	74.69 (113.08)	54.83 (113.08)	–	16.44 (113.08)	–	–

Note: [a] ratios in parenthesis were estimated using IVS benefit estimates.

For an example of an evaluating sequence, consider the sequence P3→P1→P2. The appropriate ratio for P3 is 113.08, which obviously passes the benefit-cost test. The ratio for P1 when

P3 is already in the mix is 2.51, still well over 1.00, and hence P1 also passes the cost-benefit test. Eventually, the ratio for P2 when P1 and P3 are already in the mix is 1.27. This is still over 1.00, but the difference may not be statistically significant (in fact, this is actually the case). Hence, the recommendation to include P2 is subject to a non-trivial risk of error.

However, using the IVS benefit-cost ratio (that is: 4.32), analysts would be very confident in recommending the inclusion of P2 in this sequence. Here the IVS result does not change the average policy recommendation (it would change if we used lower limits of 95% confidence intervals rather than averages for benefit estimation), but leads us to be overconfident about this recommendation.

In other studied cases, IVS actually leads to recommendations of the wrong policy mix [Santos (1997)]. This wrong mix will be broader than the optimal one, if substitution between programmes prevails; it will be stricter if complementarity is the prevailing relationship between programmes. The practical conclusion is, therefore, to use, when possible, the modelling approach proposed in this paper for benefit estimation, which accounts for substitution effects, and hence is not prone to IVS bias. In cases where a model such as this cannot be estimated (e.g. where benefits are to be transferred from previous studies), it would be wise to adjust biased estimates to correct for the IVS effect. This leads us to strongly recommend the promotion of further research on substitution effects under the most diverse circumstances, using the proposed modelling approach, so that the proper adjustment factors can be developed for a broad range of valuation settings.

Other practical application problems exist with the proposed sequential approach to optimal policy mix, such as the path-dependency of the optimum. These problems are discussed and solutions proposed in Santos (1998).

References

Arrow, K.; R. Solow; E. Leamer; P. Portney; R. Radner, and H. Schuman (1993). *Report to the NOAA Panel on Contingent Valuation.* Federal Register 58 (10).

Boyle, K. and J. Bergstrom (1992). “Benefit Transfer Studies: Myths, Pragmatism, and Idealism”. *Water Resources Research 28* (3): 657–663.

Desvousges, William H.; F. Reed Johnson, and H. Spencer Banzhaf (1998). *Environmental Policy Analysis with Limited Information. Principles and Applications of the Transfer Method.* Cheltenham: Edward Elgar.

Hoehn, J. and A. Randall (1989). “Too Many Proposals Pass the Benefit Cost Test”. *American Economic Review 79* (3): 544–551.

Mitchell, R. and R. Carson (1989). *Using Surveys to Value Public Goods: the Contingent Valuation Method*. Washington DC: Resources for the Future.

Odum, Eugene P. (1971). *Fundamentals of Ecology*. (3rd Edition). W. B. Saunders Company.

Santos, José M. L. (1996). *Valuing Alternative Bundles of Landscape Attributes: Cost-Benefit Analysis for the Selection of Optimal Landscapes.* Paper presented at the 7th Annual Conference of the European Association of Environmental and Resource Economists. Lisbon, June 1996. Published in: K. G. Willis, K. J. Button e P. Nijkamp (1999), Eds. Environmental Valuation, Cheltenham: Edward Elgar Publishers.

Santos, José M. L. (1997). *Valuation and Cost–Benefit Analysis of Multi-Attribute Environmental Changes. Upland Agricultural Landscapes in England and Portugal.* Unpublished PhD thesis. Newcastle, UK: University of Newcastle upon Tyne.

Santos, José M. L. (1998). *The Economic Valuation of Landscape Change. Theory and Policies for Land Use and Conservation*. Cheltenham: Edward Elgar.

PART 3

CHAPTER 6:
by James R. KAHN[13], Robert O'NEILL[14] and Steven STEWART[15]

STATED PREFERENCE APPROACHES TO THE MEASUREMENT OF THE VALUE OF BIODIVERSITY[16]

Introduction

In many ways, biodiversity is similar to love or faith. Everybody believes that these are concepts with well-defined and precise meanings, but in reality, everybody has different perspectives, often contradictory. Furthermore, the value of each of these is highly recognised, but difficult to measure. However, there is no practical need to develop monetary measures of the value of love or faith. This is because for ethical, political and practical reasons, we don't engage in cost-benefit analysis of public policy to promote love or faith. However, there is an important need to develop public policy to protect biodiversity. This policy-making requirement generates a need for valuation and/or an assessment of the societal consequences of protecting biodiversity.

This paper will discuss the issue of the valuation of biodiversity resources, focusing on alternative stated preference approaches to develop innovative ways to measure the value of biodiversity. In particular, conjoint analysis will be explored as a potential method for solving this difficult valuation problem. The paper does not attempt to present findings, which suggest an immediately implementable method for valuing biodiversity, but presents suggestions and highlights conceptual and empirical issues in an effort to suggest promising methods for valuing biodiversity. The organisation of the paper proceeds as follows.

The second section of the paper will focus on the problems of valuing biodiversity resources and incorporating these values into cost-benefit analysis. The third section of the paper explores a conjoint analysis based index of biodiversity as a means of assessing the societal consequences of changes in biodiversity resources, while the fourth section of the paper discusses preliminary plans to apply these methods to the problem of valuing the biodiversity of fresh water mussels and fish in the Clinch River, a Nature Conservancy Biodiversity Reserve in Virginia and Tennessee, USA. The fifth section of the paper examines the ability of conjoint analysis to develop more conventional monetised

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16 The authors would like to thank Dan Biller and the participants of the OECD Workshop on Benefit Valuation of Biodiversity Resources for helpful comments.

measures of value. An appendix to the paper discusses the theory and implementation method of conjoint analysis in more detail.

Problems with current approaches

Current approaches to valuing environmental change (including measuring ecological risks which affect biodiversity) tend to focus on predicting an outcome (or set of outcomes) associated with an environmental change, and using an economic valuation method to estimate the value of that change. For example, Rubin *et al.* (1991) estimate the value of preserving spotted owls to measure the benefits of preserving old growth forests.

There are two major problems associated with this approach. The first problem is that the process tends to focus on only a small subset of the values associated with the change in the environmental resource, rather than looking at all the values associated with biodiversity and the corresponding risks associated with system-wide change. The second problem is that the valuation techniques as currently developed inadequately measure the losses in social welfare associated with the environmental change. Taken together, these two problems give rise to the fundamental valuation problem. The *fundamental valuation problem* suggests that currently employed valuation techniques are inadequate to support policy associated with managing environmental resources such as biodiversity. Before discussing the fundamental valuation problem it is necessary to briefly discuss the value of ecological resources, place the values of biodiversity within this discussion, and assess the economic valuation techniques that are used to measure the value of environmental resources and environmental change. The discussion will demonstrate that as currently employed, economic valuation techniques can not provide a significant contribution to assessing the societal consequences of either changes in biodiversity or risks of changes to biodiversity, and that new and innovative techniques must be developed.

Categorising the Benefits of Environmental Resources

There are many different types of values associated with environmental resources, and they can generally be categorised into three groups. First, there are direct-use values, which occur when environmental resources are directly used in production and consumption activities. The use of rainforest plants to develop new medicines is an example of a direct use of an environmental resource. Second, there are indirect-use values, where the environmental resources are not directly used in a consumption or production activity, but still provide value.

Indirect use values include altruistic values, bequest values, and existence values. Altruistic values exist when a person values environmental quality, because of concern for the welfare of other people and the belief that increasing environmental quality increases the welfare of others. Bequest values arise from a person’s desire to pass on environmental quality to his or her descendants. Finally, existence values arise when a person’s welfare increases simply through the knowledge of the existence of environmental resources. For example, a person may never plan to go “whale watching,” but still derives utility from the knowledge that whales are present in the oceans.

The third general category of environmental values is the value of ecological services. Ecological services are the functions that ecosystems perform that provide the basis for all ecological and economic activity, and include carbon sequestration, nitrogen fixation, hydrological cycles, nutrient cycles, biodiversity, production of oxygen, maintenance of global climate, soil formation and primary productivity. Although in some circumstances these ecological services fit neatly into either direct use values and indirect use values, in the context of valuation and ecological risk assessment, it is important to break them out into a separate category. The reason for this is that even though they are necessary for virtually every human productive and consumptive activity, people do not have to directly seek these ecological services, so there is not as direct a connection as with other direct uses, such as clean water for swimming or fishing.

Methods for Measuring the Benefits of Environmental Resources

Two different types of approaches are generally used by economists to measure the benefits of environmental resources or the damages associated with degradation. The first category of approaches consists of revealed preference approaches that are based on observing actual behaviour and inferring the value of environmental quality from this behaviour. The second category of approaches consists of stated preference approaches that are based on survey techniques, which ask people to state their values for potential changes in environmental quality.

Revealed Preference Approaches

Revealed preference approaches look at people's decisions to use environmental resources, and from this infers the value. For example, if a person is observed driving to a more distant forest to hike in an area of greater biodiversity, then the travel cost incurred can provide a measure of the person's willingness to pay to obtain access to greater biodiversity. There are four major types of revealed preference approaches including demand and supply analysis, the travel cost method, the hedonic wage method, and the hedonic housing price method.

Demand and supply analysis is based on estimating demand and supply equations for a market good, in a fashion, which relates environmental change to either the demand function, the supply function, or both. The environmental change will therefore generate a new market equilibrium and the change in net social benefits can then be estimated, by computing the change in the area between the demand and supply functions (the change in consumers' surplus plus the change in producers' surplus).

Obviously, it is only possible to use this technique for a set of market goods for which either the demand curve or supply curve is affected by changes in the level of environmental quality. However, for many non-market goods, it is still possible to estimate value based on observing actual behaviour. An example of this is the travel cost model, which can be used to measure the value of environmental resources that are important to outdoor recreational activities. In the travel cost model, surveys of recreationists are conducted and the data are used to estimate a functional relationship with the number of trips as the dependent variable and with travel costs and environmental quality (among other variables) as the independent variables.[17] This functional relationship actually constitutes a demand curve for access to the environmental resource and can be used in an analogous fashion to estimate changes in social benefits. Although many areas of biodiversity (such as the Galapagos Islands, South American Rainforests, African savannahs and tropical coral reefs) are associated with ecotourism, the travel cost method has limited ability to measure the value of biodiversity.

An alternative and more general method is the hedonic pricing method, which can be applied to either housing prices or wages. The applicability of this technique can best be illustrated by an example. Assume that there is a community of identical homes, with people with identical incomes, who work in their homes. Further assume that the community is a featureless plane with no attractive

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[17] See Kahn (1998), Freeman (1993), or Cropper and Oates (1992) for a more detailed discussion of the travel cost model.

features (such as parks, beaches or rivers) and no unattractive features (such as power plants or waste sites). Under these conditions, everyone is indifferent across locations in the community and all the houses will have the same price. Now imagine that a waste disposal site is built along the western edge of the community. People will no longer be indifferent about their residential location, because it will be better to live in the more eastern parts of the community, further away from the potential health hazards and the diminished aesthetics associated with the waste disposal facility. As people try to leave the western part of the community and move to the east, the price of houses will fall in the west and increase in the east. This gap in prices will continue to grow until people are once again indifferent among locations in the community. The gap can be interpreted as the willingness to pay to avoid the disamenities associated with the waste disposal facility.

Of course, the real world does not conform to the assumptions of this example, but other influences on the price of houses can be controlled for in the statistical analysis. For example, the price of houses can be regressed on the characteristics of the houses and the characteristics of the neighbourhood, where the characteristics of the neighbourhood can include environmental quality variables. The price function can then be differentiated with respect to the level of a particular characteristic to yield the marginal willingness to pay for a change in the level of the characteristic. A similar type of analysis can be conducted by looking at inter-city variation in wages. However, since the biodiversity at a particular location generates benefits for people at all locations, the hedonic methods can not provide a general measure of the value of biodiversity. However, the aesthetic and recreational benefits associated with biodiversity are potentially measurable with these techniques.

In other words, revealed preference techniques are generally capable of estimating the direct use values of environmental change, but they are generally not well suited to measure either indirect use values or the value of ecological services such as biodiversity. In particular, it is very difficult for revealed preference approaches to measure the value of indirect uses, since indirect uses are by definition not associated with an activity that can be measured and then statistically analysed to infer a value. This is especially true for hedonic housing price and wage models, where value is inferred from locational choice. Since the value of biodiversity is associated with global public good benefits and is not specific to a person's location, these values can not be revealed by locational choice models.

Another important problem associated with revealed choice models is that it is very difficult to use them to measure the benefits of ecological services. For example, to the extent that increased biodiversity increases the quality of recreational fishing or the aesthetic characteristics of a neighbourhood, it would be possible to measure these values associated with greater biodiversity. However, all other types of value arising from biodiversity resources could not be measured with this method. For example, it would be exceedingly difficult to use a revealed preference method to measure the total value associated with increasing the abundance of freshwater mussels in a particular river or preserving the biodiversity of mussel species.

Stated Preference Approaches

Stated preference approaches are survey methods based on asking people hypothetical questions about how much they are willing to pay to obtain a more desirable level of environmental quality or to avoid a less desirable level of environmental quality. An example of a contingent valuation question follows:

- The Abrams Creek watershed in the south-western corner of the Great Smoky Mountains National Park is threatened by potential development of land bordering on the Park which contains the upstream portion of the watershed. The watershed is an important habitat of many plants and animals, including black bear, red wolf, and river otter. Hikers, recreational trout anglers, and canoeists also use the area.

- A group of non-profit organisations is collecting money to purchase the land, which totals 4 600 acres. The land would then be donated to public agencies to administer as preserved land. Three hundred acres would actually be incorporated into the National Park, with the remainder administered by the Tennessee Wildlife Resources Agency for public outdoor recreation.

- Would you be willing to donate US$ X to aid in the purchase of the land?

In theory, contingent valuation can measure both direct use values and indirect use values. However, the method is quite controversial because of its hypothetical nature, and a series of potential biases have been identified. The hypothetical nature of the technique leads to two problems. First, the survey respondents have no experience in making decisions like this in the real world. In the real world, people seldom, if ever, confront the problem of deciding how much to pay to improve environmental quality, especially those environmental resources, which provide indirect use values, or ecological services, or both. This lack of experience can lead to either over or under statements of true willingness to pay. Another problem created by the hypothetical nature of the decision framework is that people do not actually make real economic commitments. This has been hypothesised to lead to a systematic overstatement of true willingness to pay, although the empirical evidence on the existence of this bias is mixed.[18]

In addition to these problems generated with the hypothetical nature of contingent valuation, there are two other important biases, which have been identified. These are part/whole biases and embedding biases. Part/whole biases are used to describe the empirical observation that the willingness to pay for a smaller environmental resource is often measured to be essentially equal to measures of the willingness to pay for a more encompassing environmental resource. Economic theory suggests that the willingness to pay for the more encompassing good should be at least as large as the willingness to pay for the good which is encompassed. For example, the willingness to pay for air quality improvements in a system of parks should be larger than the willingness to pay for air quality improvements in one park in the system. However, empirical studies have shown there to be little difference between the two in a variety of applications. Embedding biases occur when the survey respondent includes in his statement of willingness to pay the value of goods other than those, which are the subject of the survey. For example, if a survey is about the willingness to pay for improved visibility associated with air quality improvements, the respondent may include his or her willingness to pay to avoid air pollution impacts to ecosystems or human health.

In summary, contingent valuation is quite controversial and many researchers believe it to be subject to systematic biases, especially when applied to the valuation of indirect use values. Contingent valuation models have had a mixed performance when subjected to internal and external validity tests.[19] This leads to what can be referred to as the fundamental valuation problem. Revealed preference models tend to pass external and internal validity tests, but by their very nature can not be used to measure indirect use values. Contingent valuation models can be used to measure indirect values, but there are problems with implementation and the techniques are controversial and contentious because of the potential biases associated with contingent valuation and the lack of internal and external validity. In addition, neither revealed preference models nor contingent valuation models have been successfully applied to measure the value of the full range of ecological services. As a result of these shortcomings, *indirect-use values and the value of ecological services seldom are*

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18 See Neill *et al*. (1994), Kealy *et al*. (1990), Duffield and Patterson, Siep and Strand (1992), Cummings and Harrison (1992).

19 See Bjornstad and Kahn (1996), Portney (1994), Diamond and Hauseman (1994), and Hanneman (1994).

included in cost-benefit analysis or other forms of economic or ecological risk assessment. This is why the field of environmental economics has experienced only limited success in measuring the value of biodiversity, and this problem reinforces the need for the contribution to be made by the OECD "Workshop on Benefit Valuation of Biodiversity Resources*.*"

How Do We Address the Fundamental Valuation Problem?

The obvious solution to addressing the fundamental valuation problem is to develop better valuation techniques, but this is not an easy task. Since many researchers continue to explore the development of contingent valuation, we focus on an alternative method, which we believe has much greater potential. We suggest an alternative method, which performs an indexing function rather than a monetary valuation function. However, the index-based method will still measure social preferences and allow economic analysis of alternative levels of biodiversity. Rather than looking for dollar measures of changes in social welfare, it provides an indicator of how biodiversity impacts social welfare. In addition, we also discuss another set of conjoint analysis based methods which can be used to derive monetary estimates of the value of biodiversity.

Indices as a Measure of the Societal Importance of Environmental Resources

In the environmental economics literature, most research has focused on developing dollar measures of value, rather than some other indicator of social welfare. However, an "indicator" approach may be more consistent with ecological risk assessment than a valuation approach, and may provide a better assessment of the impacts of changes in biodiversity on social welfare.

Figure 6.1 illustrates the relationships among four different aspects of the quality of life and social welfare and helps focus the discussion on the importance of indicators. These aspects of the quality of life include the health of the economy, the health of the population, and the health of the environment. Although there are many other important influences on social welfare, this proposal diagrams only these four factors, in an effort to focus on the direct and indirect impacts of the environment on social welfare. Direct impacts are depicted by the thick solid arrow and represent environmental resources (or services flowing from environmental resources) that appear as individual arguments in an individual's utility function.[20] In contrast, environmental quality also indirectly impacts social welfare through its effects on the economy, health of the population, and social justice.

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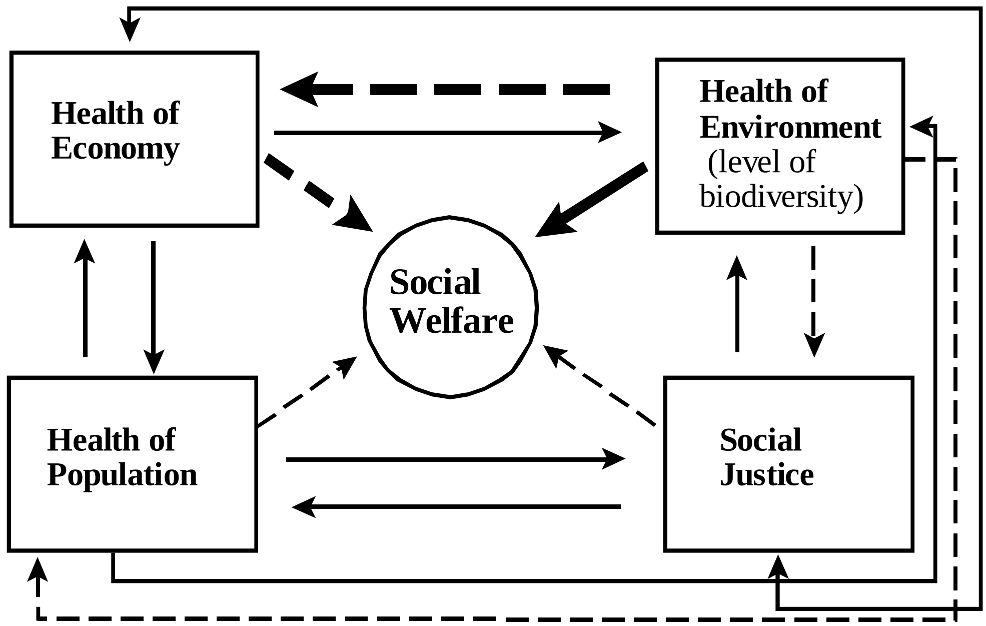
[20] If one were looking at this problem in a household production function approach, then these environmental resources and services would be inputs to the production of a final service flow.

Figure 6.1 Selected Determinants of Social Welfare



It is important to note that we have already developed operational indicators associated with non-environmental direct impacts (the inner thick dashed arrow and the inner thin dashed arrows). For example, GDP, green GDP, unemployment rates, and inflation rates are all operational indicators of the health of the economy. Similarly, infant mortality, birth weights and longevity are used as operational indicators of the health of the population, while variables such as the inequality of income distribution (Gini Coefficient), literacy rates and incarcerated proportion of the population are used as indicators of social justice.

One can use these indicators to evaluate policy, even in the absence of a social welfare function. For instance, ceteris paribus, if longevity or literacy rates increase, social welfare will increase. The difficult policy question arises in terms of trade-offs, for example, if reducing infant mortality rates requires a sacrifice in terms of potential GDP. Methods for examining this trade-off will be further discussed below, but for now, the crucial point is that we have not developed a corresponding environmental indicator for the health of the environment. One can develop indices for environmental quality in general, or for categories of environmental resources, which contribute to environmental quality. For example, one could develop indices directly related to biodiversity or based on the health of the individual ecosystems, which provide the biodiversity.

It should be noted that any choice of an operational indicator of either the environment in general or biodiversity in particular, will have both advantages and disadvantages, and will not be an "ideal indicator." For example, the unemployment rate is an operational indicator of the health of the economy, but it has its flaws. It does not include those who are so discouraged that they are not actively seeking work, and aggregate unemployment rates understate the true unemployment rates for some sectors of society, such as young people in the inner cities. However, if one understands the idiosyncracies of an operational indicator, it can be used in the policy formulation process.

A Discussion of Past Efforts to Develop an Operational Indicator of Environmental Quality

At least four methods have been suggested or employed to develop operational indicators of environmental quality. These include use of "representative" environmental variables, the development of satellite accounts for the National Income and Product Accounts, green GDP/NDP, and indices of sets of environmental variables.

Representative environmental variables

Measures of individual types of pollution have been used in many studies, with the underlying assumption being that the trends associated with these individual pollutants are somehow representative of environmental quality. For example, sulphur dioxide pollution has been used as an indicator of overall environmental quality in the estimation of "environmental Kuznet's curves," which purportedly show a u-shaped relationship between environmental quality and income (an inverse u-shaped relationship between income and concentrations of pollution). The use of a particular pollutant to proxy for environmental quality in general is conceptually similar to using output in the steel industry as a proxy for GDP, or the incidence of lung cancer as a general indicator of the health of the population. This measure has been criticised by Arrow *et al*. (1993) and O'Neill *et al*. (1996) among others. In particular, the measure is completely unrelated to both land use changes (such as deforestation and desertification) and water quality changes (with the exception of acidification of water bodies). In addition, sulphur dioxide is a fund or flow pollutant, which does not accumulate over time, unlike carbon dioxide, chlorofluorocarbons, or heavy metals. This method could be applied to biodiversity by using an indicator (canary in the coal mine) species as the representative environmental variables. For example, the spotted owl is suggested as an indicator species for the Pacific old growth forests (US and Canada), and the abundance and biodiversity of mussels in general has been suggested as an environmental indicator for rivers in the southeast United States. In these two cases, it may be possible to define appropriate indicator species. However, what would be the appropriate indicator species for an ecosystem as complex and diverse as the Amazonian rainforests?

An alternative to direct measurement of biodiversity may be defining representative environmental variables which represent threats to biodiversity. Two variables, which might be useful in this regard, include the percentage of particular ecosystems, which remain intact, and the presence of invasive species in particular ecosystems.

Green GDP

Many economists, including Daly (1991), Peskin (1976), Prince and Gordon (1994), and Repetto (1989) have argued that disastrous consequences can occur when macroeconomic policy is based on promoting the growth of GDP. They argue that not only does this ignore other aspects of the quality of life, but that GDP has a serious flaw as a measure of economic progress.

This flaw has to do with the fact that measures of Net Domestic Product (NDP) subtract the depreciation of human-made capital, but do not subtract the depreciation of natural capital. Thus, when a machine is worn out in the production of current income, the loss in income producing ability is subtracted from the measure of current income. The depreciation of human-made capital is subtracted from GDP to give NDP, a more accurate measure of the current economic wellbeing of a nation. However, when a forest is clear cut, soil is degraded, or stocks of minerals are depleted in order to produce current income, a similar debit is not made.

Although one can argue that this is just a definitional issue, and that all definitions are arbitrary in nature, there are serious implications when national economic policy is based on this flawed measure of NDP. If increasing current NDP is a primary policy goal, then natural capital and its ability to produce future income (or other services) may be expended even if this is detrimental to producing future social welfare. Although this is a serious problem in developed countries such as the United States, it is perhaps even more important in developing countries, where pressing needs to increase current income have caused catastrophic deforestation, pollution, soil erosion and desertification.

As Repetto indicates, this difference in the treatment of natural capital and human-made capital

> ...reinforces the false dichotomy between the economy and the "environment" that leads policy makers to ignore or destroy the latter in the name of economic development. It confuses the depletion of valuable assets with the generation of income. Thus, it promotes and seems to validate the idea that rapid rates of growth can be achieved and sustained by exploiting the resource base. The result can be illusory growth and permanent losses in wealth.

Therefore, Repetto and others argue that the depreciation of natural capital should be factored into GDP in a fashion analogous to the depreciation of human-made capital. Repetto recomputes Indonesia's national income and product accounts making corrections for deforestation, soil erosion and oil reserves. Repetto found that the measured 7.1% annual growth rate of GDP is actually only 4% when these corrections are made.

Although this type of analysis can aid in the formulation of macro environmental policy, it does not give complete information about the relationship between the health of the environment and social welfare. This is because this measure only takes into account one pathway for the environment to affect social welfare, and most importantly, it ignores the direct effect of the health of the environment on social welfare. Although the value of these direct effects or non-pecuniary environmental services could be incorporated into national income and product accounts, this would be difficult, if not impossible to do. It is much more difficult to monetise these other aspects of environmental quality than it is to monetise environmental effects such as oil depletion and soil erosion. Thus, the monetary conversion problem associated with willingness-to-pay measures is not eliminated with national income and product accounts, it has simply been transferred to the national income and product accounts.

An additional problem with these macroeconomic approaches is that there is a tendency to focus on environmental resources that are part of the economic production process and to focus less on environmental resources that contribute more basic life support services or amenity benefits. More importantly, GDP and NDP are measures of the health of the economy, not the health of the environment. Separate measures of the health of the environment must be developed to better understand the relationship between environmental quality and social welfare. This is particularly true for the measurement of the value of biodiversity, since the path by which biodiversity contributes to GDP is indirect. The contributions and potential contributions of biodiversity to GDP would represent only a small subset of the impacts of biodiversity on social welfare.

Even though the "greening" of the National Income and Product accounts can not create an operational indicator of the health of the environment and would add little to the understanding of the value of biodiversity, this effort should be pursued to give a more accurate indication of the health of the economy. Prince and Gordon (1996) provide a detailed discussion of this type of modification of the U.S. National Income and Product Accounts.

Satellite Accounts

In addition to the types of modifications of GDP discussed above, the United Nations Statistical Division recommends the development of a system of environmental satellite accounts, to monitor environmental change:

> "Satellite Accounts try to integrate environmental data sets with existing national accounts information, while maintaining SNA [System of National Accounts] concepts and principles as far as possible. Environmental costs, benefits and natural resource assets, as well as expenditures for environmental protection, are presented in flow accounts and balance sheets in a consistent manner. That way, the accounting identities of the SNA are maintained. One of the values of the SEEA [System of Integrated Economic and Environmental Accounts] framework compared to more partial approaches is that it permits balancing, so that rough monetary estimates can be made for residual categories [Hamilton and Lutz (1996)]."

However, satellite accounts (by intent) represent a disaggregation of measures of environmental change, rather than an aggregation. They could serve as inputs for developing operational indicators of environmental change, but as an independent set of indicators they would suffer from the same problem as the indicators that United States Environment Protection Agency's (EPA) Environmental Monitoring and Assessment Program (EMAP) collects. This problem revolves around the large number of measures, which make the examination of trade-offs and overall trends more difficult. This is particularly true for biodiversity, as the number of individual species totals in the tens of millions. However, these satellite accounts could serve as a basis of an aggregate measure, in which the individual variables categorised in the satellite accounts are aggregated into a more general index.

Aggregate Indices

The EPA's EMAP represents an effort to develop indicators of environmental quality. EMAP attempts to develop overall indicators for individual ecosystems (such as forests, wetlands or estuaries). In the case of estuaries, EMAP develops a series of over twenty indicators, but creates an aggregate index by summing the indicators based on water clarity, the benthic index, and the presence of trash [Schimmel *et al.* (1994)]. This is indicative of a general procedure employed by natural scientists to create aggregate indicators by summing all individual environmental indicators and dividing by the number of indicators to create an unweighted index. This unweighted index is virtually meaningless, because it implicitly and arbitrarily uses equal weights for each individual indicator. For instance, why should a 10% increase in the benthic index and a ten percent increase in the presence of trash receive the same weight in the index? Additionally, there is a potential problem of the level of the index being a function of the choice of the unit for measurement of each of the individual variables. For example, if one variable is measured in parts per million and another is measured in parts per billion, they will have very different impacts on the index. One way to get around this measurement problem is to normalise each variable by dividing by its maximum level, so that all the variables are then numbers between zero and one.

While normalisation solves the unit of measurement problem, it does not solve the problem associated with an arbitrary choice of equal weights for each variable. One way of developing more meaningful weights is to base them on expert opinion. This could be done through a Delphi process or through averaging the weights that each expert has assigned. While this is certainly an improvement over the arbitrary choice of equal weights, the following section outlines a more formal procedure for defining weights that is consistent with the way decisions are made with respect to private goods, by examining individuals' willingness to make trade-offs.

The Trade-Off Weighted Index of Environmental Quality

An operational indicator can be developed, *that is consistent with the economic paradigm*, by looking at how people prefer one state of the world to another. While it is a difficult task to ask people to place a value on a state of the world, stating a preference for one state of the world over another is consistent with the way people make many decisions such as locational choice, whether to get married, or have children, or both, or whether to vote Democrat or Republican. It should be noted that people do not contemplate the marriage decision by asking themselves what their willingness to pay for being married to one potential spouse versus being married to another potential spouse versus being single, but which choice gives them the highest quality of life. These optimising choices may be constrained by ethical or moral rules that people have imposed upon themselves.

A choice, or trade-off based indicator can be developed by using discrete choice based conjoint analysis to present alternative states of the environment to the individual.[21] The alternative states would be defined by different levels of physical characteristics of the environment, including the characteristics of important sub-systems of the environment. Ecosystem characteristics would be varied according to actual and potential risks, as well as potential environmental improvements. These alternative states of the environment would then comprise choice sets, and both experts and stakeholders would be asked to choose between alternative choice sets.

As discussed above, the level of the physical environmental characteristics in the choice sets would be varied both within the choice sets presented to individuals, and across individuals. This variation in the level of the characteristics of the alternative states of the environment would allow the estimation of a preference function. In this preference function, the probability of preference is estimated as a function of the levels of the physical characteristics. The derivatives (with respect to each physical characteristics) of the preference function can then be used as weights to aggregate the physical characteristics into a single index or set of indices. In other words, if the estimated preference function was of the form,

$$\mathbf{PROB} = \theta(C_1, C_2, C_3, ..., C_n) \qquad \textbf{(1)}$$

where the C_i's refer to the levels of the environmental characteristics that define the alternative states of the world, the index could be computed as,

$$\mathbf{I} = \sum_{i=1}^{n} \frac{\partial\theta}{\partial \mathbf{C}_i}\mathbf{C}_i \qquad \textbf{(2)}$$

At first glance, this method for deriving these weights may seem to be a re-cast version of contingent valuation, as both this method and contingent valuation ask the respondent hypothetical questions about willingness to make trade-offs concerning environmental quality. However, the two are based on fundamentally different mental models. Contingent valuation asks people to state a willingness to pay for a non-market good, but people are not accustomed to purchasing non-market goods. This forced employment of an unfamiliar mental model may be what gives rise to the biases

[21] See Louviere (1988, 1996) for a general discussion of conjoint analysis and Kahn and Maynard (1996) for a discussion of conjoint analysis in environmental applications. Conjoint analysis could be applied to evaluate other areas of social policy, but the focus of this paper is environmental policy.

which many people argue are associated with contingent valuation. However, conjoint analysis asks people to chose among alternative states of the world. Even though these alternative states may involve non-market goods, people are accustomed to making this type of choice. For example, the choices of whether to get married or stay single, have children or not have children, vote Republican or Democrat, live downtown versus the suburbs, or go into academics versus consulting, are all alternative states of existence which are associated with different bundles of non-market goods. (They may also be associated with changes in income and bundles of market goods.)

Although conjoint analysis remains largely untested with respect to environmental goods, its performance with market goods indicates a high degree of internal and external validity. In particular, in comparisons of hypothetical and actual responses, conjoint analysis has been a good predictor of actual responses.[22] In comparison, in many experimental studies, contingent valuation has been a poor predictor of actual responses [see Cummings (1996)].

Even though the method proposed for deriving indexes is not based on willingness to pay, one of its attractive features is that it is still based on the willingness of individuals to make trade-offs. In contrast, the EMAP of EPA has developed indices of physical characteristics that are unweighted indices. In addition to the inherent desirability (at least from the point of view of an economist) of basing the indicators on willingness to make trade-offs, the trade-off based foundation of the indicators would make the measures of the health of the environment analogous to the primary measure of the health of the economy (GDP), as GDP is a set of physical quantities which are weighted by people’s willingness to make trade-offs, which in the case of GDP are measured by prices. An indicator of the health of the environment which is based on “trade-off weighted” physical quantities would be completely analogous to GDP, except in this case, the trade-offs are measured through a survey process, since market prices do not exist for the physical characteristics of the environment.[23]

One of the major criticisms by non-economists of willingness to pay measures and other methods which are based on individual choice is that they do not take into account expert knowledge of the consequences of environmental change. It is possible to incorporate expert knowledge into the trade-off based indicator of environmental quality or sustainability by implementing a parallel choice process and separate index among a sample of experts.

An important policy consideration is the determination of how much importance to place on the expert index in comparison to the ordinary citizen index. This expert index could be kept separately and then policies could be evaluated with respect to both indices. Alternatively, the indices could be merged into one index. One way to do this would be to include a statement in the survey to which citizens respond. This statement would indicate that experts are being asked to state preferences for alternative states of the environment in the same fashion. The citizens could then be asked (as part of the survey questionnaire) how much weight in the decision-making process expert opinion should be given relative to citizen opinion.

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[22] See Louviere and Hensher (1982); Levin, Louviere, Schepanski and Norman (1983); Louviere and Woodworth (1983); Louviere (1988); Horowitz and Louviere (1990); Elrod, Louviere, and Davey (1992); Louviere, Fox, and Moore (1993); and Adamowicz, Louviere, Williams (1994).

[23] The same procedures could be used to develop a single indicator variable for the other areas of social concern. For example, this process could be used to aggregate longevity, infant mortality and other health indicators into a single indicator variable of the health of the population.

Use of the Trade-off Weighted Index in Valuing Biodiversity

As indicated above, the key to developing the trade-off weighted index is to appropriately specify the choice sets that provide the survey data used to estimate the preference function. This can be done is several ways.

The most general method of developing a biodiversity index would be to include biodiversity as a component of overall environmental quality. For example, characteristics of the choice sets could include levels of different types of pollution, quantity and quality measures of characteristics related to major ecosystem types (estuaries, wetlands, forests, etc.) and quantity and quality measures of characteristics related to biodiversity. Then the social consequences of a reduction in biodiversity or a risk of a reduction in biodiversity can be measured as the impact of the change in these biodiversity characteristics on the environmental index.

A difficulty associated with both this method and the measurement of the value of biodiversity in general is that ecosystem health and biodiversity are jointly produced, and both contribute to social welfare. A biodiverse system, *ceteris paribus*, is a healthier, more stable system that provides a greater flow of ecological services. At the same time, a healthier, more productive and more stable ecosystem provides greater protection of biodiversity. Since both greater environmental quality and greater biodiversity both contribute to social welfare, and since both are functions of each other, it may not be possible to completely attribute changes in social welfare to one factor or the other.

An alternative approach would be to develop a set of choice sets describing alternative states of the world, where only biodiversity characteristics change between choice sets. This would allow the measurement of an index of biodiversity. However, it is not clear that such a choice experiment could be successfully implemented within a survey population of ordinary citizens, as the ordinary citizens might not be able to successfully evaluate alternative sets of biodiversity variables. Such a choice experiment and development of a separate index of biodiversity might be better implemented with a survey population consisting solely of experts.

A third approach is to define choice sets which include not only biodiversity and environmental variables, but other variables associated with the quality of life, such as regional economic variables (changes in per capita income, quality of education, transportation increases, changes in crime rates, and so on). Then, the index would express trade-offs among all these factors, which affect the quality of life.

A final approach would be to include a price associated with the environmental or biodiversity resources in the choice set. Under these circumstances, a Hicksian measure of value can be associated with a change in the level of biodiversity (or other environmental variable). This willingness to pay based choice method will be discussed in Section 5.

Application to Biodiversity in the Clinch River, USA

We are in the process of implementing the index approach to measuring the importance of biodiversity resources in the Clinch River (Virginia and Tennessee, USA) under funding from a co-operative agreement with the US Environmental Protection Agency program that is developing methods for integrating ecological economics and ecological risk assessment. Two other projects are being implemented by other universities on the Middle Platte River (University of Nebraska) and Big Darby Creek (Miami (Ohio) University). The Middle Platte River project is using a game theoretical approach to competing uses of water resources, while the Big Darby Creek project is using hedonic pricing methods and contingent valuation.

Since the objective of this research program is to integrate economic analysis and ecological risk assessment, our planned empirical research will not focus on a pure biodiversity index, because such an index will not allow a broad examination of people's willingness to make trade-offs of biodiversity for other aspects of the quality of life. Instead, we will focus on the other two index methods (all environmental variables in the choice sets and a set of environmental and quality of life variables in the choice set). In addition, a set of surveys will be conducted where the choice sets include a tax price, so that willingness to pay measures can be derived.

The Clinch River is remarkable in that it contains a great diversity of freshwater mussel and fish species. However, the abundance and biodiversity of both the mussels and fish are threatened by the changes in environmental quality resulting from coal mining, deforestation of the riparian corridor, cattle ranching, crop cultivation, chemical spills and run off from highways and construction areas.

One complication in the valuation process is that the mussels affect social welfare in a very indirect path. They provide few direct ecological services (even their role in filtration is of limited importance) but they represent a unique reservoir of freshwater mussel species.[24] In addition, the mussels serve as an indicator species. Since mussels are immobile (as adults), they are located in the sediments, and they filter water to attain sustenance, they are very vulnerable to changes in environmental quality.

Since the connection to individual utility is indirect, our surveys must first describe the significance of the mussel populations and biodiversity, its role in the uniqueness and character of the system, and its role as a biodiversity reserve in North America and the world. Of course, any time an informational process is part of the survey technique, there is a danger that the surveyors are creating value rather than measuring value.

The first set of choice sets involve only environmental characteristics. Choice sets will involve characteristics associated with water quality, mussel abundance, mussel biodiversity, quality of fresh water fishing (predominantly smallmouth bass), landscape of the neighbouring mountain ridges and landscape of the riparian corridor. A variation of this method will also include different environmental management plans as characteristics (restrictions on coal mining, restrictions on agriculture, etc.). Then in addition to having indices to guide policy making, one can have ordinal preference rankings of bundles of environmental characteristics and management policies.

An abbreviated and illustrative presentation of such a choice set is presented below. It should be noted that the characteristics are chosen solely for the purposes of this example and have not yet been translated into "user-friendly" wording and units of measurement, which would be utilised in the actual surveys.

Question: Which option for the future of the environment in the Clinch Valley do you prefer the most, Option A, Option B, or Option C? Option C is the **status quo**, or what is currently happening and will continue to happen with no further environmental policy.

[24] The Clinch/Powell River system contains seven fish species and 28 mussel species which are federally listed as threatened or endangered.

	Option A	Option B	Option C: Status Quo
Mussels	No change	10% increase	10% decrease
Fish (all species)	No change	No change	10% decrease
Recreation (fishing quality-smallmouth bass)	Catch rates increase 10%	Catch rates increase 5%	No change
Songbird diversity/population	10% Increase	10 % increase	10% decrease
Wildlife population/diversity	5% Increase	No change	10% decrease
Air Quality	10% improvement	10% decline	No change

The respondent would be asked to choose between Options A, B, and C. After making this choice, the respondent would be presented with new states with new levels of the characteristics. This would be repeated four to eight more times depending on how long each choice set takes to be evaluated (we don't want the survey to be too long). Then the data will be used to estimate a preference function. The dependent variable will have a value equal to one, if Option A is chosen and a value of zero, if Option B is chosen. The probability of choosing Option A will be estimated as a function of the levels of the characteristics, using a discrete choice statistical method, such as, multinomial logit analysis.

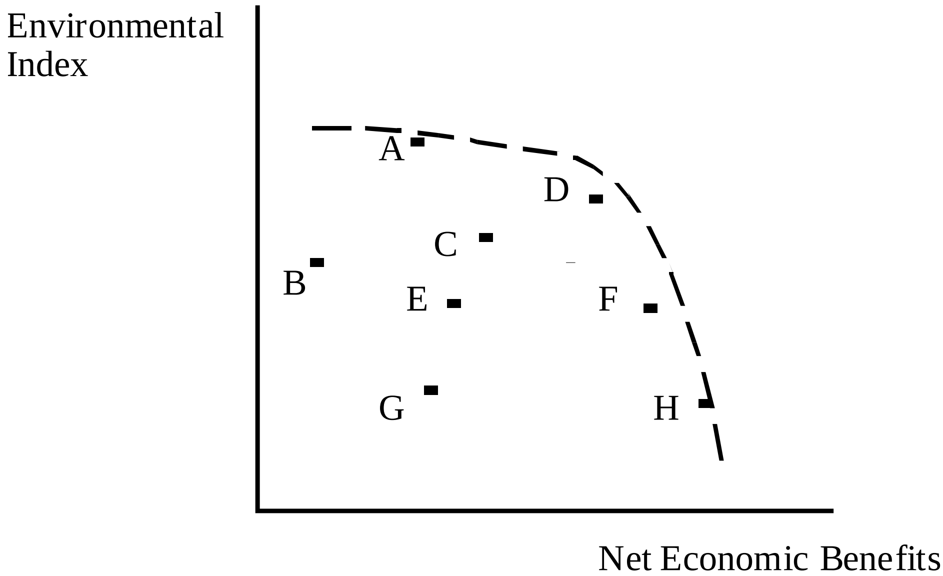
As mentioned in the earlier discussion, the derivatives of this probability function can be used as weights to aggregate the characteristics into a single index. The value of the index would be equal to sum over the characteristics of the multiplicative product of the levels of the characteristics and the partial derivatives of the probability function with respect to the characteristics. Therefore, as long as the levels of the characteristics can be determined for a management scenario, a value of the index can be computed.

How can this index be used in the evaluation of management decisions and actions? First, it can allow us to compute which state of the environment is preferred, holding non-ecological costs and benefits constant. If a particular management scheme causes some characteristics to improve, and others to decline, we can compute if the index goes up or down and if the management alternative gives us a better environment or diminished environment.

However, in most circumstances we will want to evaluate changes in environmental quality in terms of trade-offs of other types of costs and benefits, such as the economic benefits derived from a set of management decisions or actions. This index can be quite useful in performing this type of analysis.

The way the index can be used in evaluating these types of trade-offs is to compute the value of the index and the net economic benefits of each management alternative. Benefit transfer studies and expert opinion in the manner previously discussed would compute net economic benefits. Then, the value of the index and the net economic benefits associated with each project could be plotted, as in Figure 6.2. In Figure 6.2, each lettered dot corresponds to a particular project or development scenario. Dots can also represent a combination of projects that are not mutually exclusive. In Figure 6.2, the vertical axis shows the environmental quality associated with the management scenario, while the horizontal axis shows the economic benefits.

Figure 6.2 Trade-Off Analysis



Plotting the alternatives in this fashion gives a variety of information. First, it shows that points to the interior of the outer envelope (such as point G) are inferior. Point G is inferior because there are many management scenarios which have higher economic benefits and better ecological quality (such as Points E, C, D and F). The points along the outer envelope (points A, D, F, and H) are characterised by what economists term technological efficiency, as there are no other alternatives that have both higher environmental quality and greater economic benefits. The question remains, how do we choose among points A, D, F, and H? The answer to this question is that the method does not tell us which of these is better, but it does explicitly diagram the trade-offs between economic benefits and environmental quality, allowing a more educated debate among policy makers, stakeholders and other relevant groups. Thus, this method does not provide a decision making tool, but it eliminates inferior alternatives and then explicitly outlines the trade-offs associated with remaining alternatives. Although it does not decide which alternative is best, it informs the decision making process and the discussion of which alternative is best.

The second set of choice sets involve characteristics of the environment as well as characteristics associated with the quality of life of the region. These quality of life characteristics would include regional economic growth, unemployment, quality of education and similar variables. The index then will allow the measurement of trade-offs among environmental characteristics and these other quality of life variables. It is interesting to note that the inclusion of these variables precludes the need for the type of analysis depicted in Figure 6.2. It is important to incorporate these types of variables in the analysis, as this Appalachian section of Virginia and Tennessee is economically disadvantaged compared to other parts of these states and the United States in general.

Another interesting aspect of the index-based method is that it incorporates the willingness to make trade-offs without giving a disproportionate weight in the decision making process to higher income people. If the analysis is based on monetary measures of willingness to pay, people with more money and higher willingness to pay (for one outcome or another) will have a greater impact on the cost-benefit analysis based on the willingness to pay analysis. In many respects, cost-benefit analysis is based on the principle of “one dollar, one vote.” However, the index based analysis is a more equitable approach in that if all segments of the population are proportionately represented in the survey analysis, the results will be determined by the principle of "one person, one vote," which is the principle upon which most modern constitutional democracies are based.

Conjoint Analysis and Willingness to Pay Measures

In addition to developing indices or ordinal rankings of alternative states of the world, or both, conjoint analysis can be used to develop willingness to pay measures for changes in environmental quality. Discrete choice conjoint analysis (picking between two alternative sets of characteristics) is consistent with a Random Utility modelling of consumer preferences and can yield "utility theoretic" measures of value such as compensating or equivalent surplus.[25] In this case, the choice sets would include a set of environmental/biodiversity characteristics, as well as a "price." In general, the price will be a tax necessary to finance a certain management plan.

For example, one of the greatest stressors to the mussels is agriculture, both in terms of run-off and in terms of cattle entering the shallow river and trampling mussels. An effective management policy is to take land in proximity of the river out of crop production, and to fence the river so that cattle can not enter the river or contribute to the erosion of its banks. This policy scenario could be expressed in the information in the conjoint analysis survey, and then a tax associated with compensating the farmers for their remediation efforts could be explicitly incorporated as one of the characteristics in the choice sets. As more fully explained in the appendix, value estimates can be derived directly random utility function estimated as part of the conjoint analysis method. A question structured to produce this type of information is below. Again, this question is not formulated in "user friendly" terms, but is solely designed to convey information about the potential question to the reader of this paper.

Evaluating Changes in Agriculture to Protect the Environment sample question

As mentioned earlier, one problem that affects the quality of the river is that livestock get into the river, crushing mussels, eroding river banks, and muddying the water. Also cultivation (growing crops) near the river allows fertilisers, pesticides, soil and other substances to contaminate the river.

These problems could be lessened by the development of an "agriculture free zone" in the immediate proximity of the river. This zone, where no activities would be allowed to take place, could be of different widths. In our study, we ask you to compare no zone with a zone that is 10 yards wide with one which is 25 yards wide.

Farmers that keep cattle would need to construct fences that keep the livestock out of the exclusion zones. This would keep the cattle from trampling the mussels, reduce erosion and sedimentation of the river. Trees would shade the river water, reducing its summertime temperature and increasing the dissolved oxygen level, which would benefit aquatic life. As the pastures revert to more naturally occurring types of vegetation, this could also increase songbird and wildlife populations. The construction of fences and substitute watering facilities for the cattle, and the loss of the use of the land are costly for farmers. Farmers who grow crops would not be able to plant in the zones, which may be among their most fertile (and flattest) land holdings.

[25] See the appendix for a more detailed discussion of conjoint analysis.

However, the farmers need not bear the full cost of the policy. A pilot project has been underway where The Nature Conservancy has been paying farmers to construct fences and take the land out of production. This could be expanded and could be funded through a small increase in taxes for everyone in the Clinch valley. The survey below asks you to compare alternative policies. One primary difference between the policies that are evaluated is the extent of who bears the costs, the farmer or the taxpayer. Farmers could be fully or partially compensated for their losses. In fact, the payments could potentially cause farmer income to increase. Another set of differences involves the levels of the environmental characteristics.

The percentage change given in each characteristic would occur gradually over the next five years, except for changes in taxes and farmer income, which would occur immediately and remain at this new level.

Question: Which option for the future of agriculture and the environment in the Clinch Valley do you prefer the most, Option A, Option B, or Option C? Option C is the status quo, or what is currently happening and will continue to happen with no further environmental or agricultural policies.

POLICY	**Option A** 10 yard wide agricultural free zone	**Option B** 25 yard wide agricultural free zone	**Option C** Status Quo
Mussels	no further decline	10% increase	5% decrease/year
Fish	no further decline	20%increase	5% decrease/year
Recreation (fishing quality-smallmouth bass)	catch rates increase 10%	Catch rates increase 10%	No change
Songbird diversity/population	5% Increase	5 % increase	10% decline
Wildlife population/diversity	5% increase	10% Increase	10% decline
Aggregate Income	1% decrease	5% decrease	No change
Tax policies	+US $ 5/household/year	+US $ 15/household/year	No change

Please check the option that you think is the best:

Option A___, Option B____, Option C____

This question could be used to measure both, the individual’s willingness to pay to protect the environment, and the individual’s willingness to pay to protect the income of family farmers. In addition, one can measure the value of environmental protection in terms of the public’s willingness to accept reductions in farmers’ incomes.

One can do a similar analysis of other policies, including regional economic characteristics (such as regional income, income of a specific sector (eg coal mining) or individual tax variables. Such policies could include additional agricultural restrictions, environmental restrictions on coal mining or general development policies (with differing degrees of environmental and economic impacts).

Conclusions

The estimation of indirect use values and the value of ecological services has proven to be a daunting task. The measurement of the value of biodiversity is no exception to this problem. Although contingent valuation may not be the most appropriate method for deriving these value estimates, a related stated preference method, conjoint analysis, may prove very useful in this regard. Conjoint analysis has the potential to avoid many of the biases associated with contingent valuation since it does not directly ask a person to compute their willingness to pay for an environmental change.

Conjoint analysis can be employed to tackle this problem in several ways. First, it can be used to derive indices, which can serve a vital role in the assessment of alternative management policies. In addition, monetary measures of the value of biodiversity can be developed. One way of doing this is to put regional economic variables into the choice set, while an alternative would be to use individual changes in the tax burden. The latter method would allow the derivation of conventional (Hicksian) willingness to pay measures.

Appendix 1: Theory and Application of Conjoint Analysis

Introduction

Conjoint analysis belongs to the group of valuation techniques known as stated preference models. Contingent valuation (CVM) is also a stated preference approach where individuals are simply asked how much they would be willing to pay to obtain an increment in environmental quality, or how much they would be willing to accept a decrement in quality. Conjoint analysis (CJ) asks individuals to make choices about which state of the world they would prefer to be in given that different states have differing levels of attributes in each of them.

The use of CVM is written into administrative law in the United States as a tool to measure economic loss in environmental damage cases, the Exxon Valdez oil spill being the most famous. Although a complete discussion of CVM is beyond the scope of this paper [see Carson and Mitchell (1989) and Diamond (1994)], it is important to point out that CVM has been subject to a number of criticisms, including hypothetical bias, yea-saying, insensitivity to scope, framing, protest, non-response and failure to consider substitutes.

As pointed out earlier, CJ asks questions that may be more familiar to individuals. Individuals are asked to rank, rate, or choose from bundles of goods according to the level of attributes each bundle has. For example, individuals routinely make choices among goods that have multiple attributes such as the choice among five automobiles having different colours, engines, interiors, etc. A typical CJ question might ask the subject to either rate the different autos on a scale of 1-10, rank them according from most desirable to least desirable, or ask them to choose the best of the five. In contrast to CVM, which would ask the respondent whether he would pay US$ x dollars to purchase a given car, CJ makes the individual consider explicitly the tradeoffs involved in one object over another. This balancing of pros and cons is arguably more representative of the choices that individuals regularly face in making transactions involving either market or non-market goods.

Conjoint analysis is receiving increasing attention in the economics literature as well as in policy circles. Its use has been legitimised by NOAA's proposed Habitat Equivalency ruling, which arose in part due to the criticisms that CVM was subjected to during the Exxon Valdez damage assessment case (60 FR 39816).[26] In particular, NOAA recommended conjoint analysis as a tool to measure in-kind compensation substitution for damaged natural assets.

Within the survey, the consideration of substitutes is part of the selection process. Goods are differentiated based on attributes of the goods. Thus the role of substitute levels is explicitly recognised. In traditional CVM, the role of substitutes is relegated to a reminder sentence or two in the description of the good.

Theory

CJ analysis allows the investigator to infer the implicit weights (part-worths) of the attributes of the package to the package's total value. In particular, it is extremely useful for multi-dimensional

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26 Habitat equivalency argues that the appropriate measure of natural resource damages is the replacement of critical natural capital. That is, damages should be equal to the cost of substitute services and the services foregone while recovery/replacement takes place. In the case of an oil spill that damages wetlands, the offending party would have to provide like ecological services (perhaps purchase unspoiled private wetlands in an area and place a conservation easement on them).

changes - varying the level of the alternatives' attributes allows measurement of the individual's willingness to substitute one attribute for another. Economic values may be estimated if one of the attributes is measured in economic terms (dollars, taxes, jobs, etc.).

The traditional CJ ratings models have tried to explain the ratings difference between programs having differing attributes using a linear relation ship between the attributes such that the rating for program is given by

$$r^i = k + \beta_1 q_1^i + \beta_2 q_2^i + ... + \beta_k q_k^i + \beta_p p^i \qquad (3)$$

Differentiating (3) totally gives

$$dr^i = 0 = \beta_1 dq_1^i + \beta_2 dq_2^i + ... + \beta_k dq_k^i + \beta_p dp^i \qquad (4)$$

From (4), one can find the marginal rate of substitution between any of the quality attributes or price in the bundle. For example, $dp^i / dq_1^i = -b_1 / b_p$ gives the implicit price of attribute q_1. The majority of the conjoint studies before 1990 used this implicit price format.

In non-market valuation and natural resource damage assessment, the policy maker needs to assess welfare changes from changes in environmental quality. An individual's rating of a single commodity does not provide the information necessary to estimate changes in welfare [Roe *et al.* (1996)]. CJ models are increasingly being formulated in a random utility framework, which does allow the measurement of changes in welfare.

Random utility models (RUM), which are widely used in dichotomous choice contingent valuation and travel cost models, rely on choice behaviour. RUM models estimate the probability that an individual will select a choice based on the attributes of each possible choice. If the utility of alternative, *i* is greater than the utility of alternative *j*, the individual will choose *i*. Utility is comprised of both deterministic and random components.

The RUM framework is directly estimable from conjoint rankings and binary choice models. Recently, alternative CJ models that allow the estimation of welfare impacts from ratings data, which contain cardinal information, have been formulated as well.

Following Roe *et al*. (1996) and Stevens *et al*. (1997),

$$U^i(p^i, q^i, m, z) \qquad (5)$$

Where the utility of program i for the individual is a function of the price of i (explicit or implicit), the attributes of i, m is income, and z represents individual characteristics. Utility is related to an individual's rating of a program by a transformation function ϕ.

$$r^i(p^i, q^i, m, z) = \phi[v^i(p^i, q^i, m, z] \qquad (6)$$

A move from q^0 (attribute bundle zero) to q^1 is given by,

$$r^i(p^i, q^i, m - C^i, z) - r^0(p^0, q^0, m, z) = 0 \qquad (7)$$

where C^i is the income adjustment necessary to leave the individual as well of with bundle i as she was with bundle

0 (compensating variation). Rearranging (3) and solving for C^i yields,

$$C^i = m - g[r^0(p^0, q^0, m, z), p^i, q^i, z] \qquad (8)$$

Where g is the inverse of r^i with respect to income. If we assume that marginal utility of income is linear, then the difference in ratings is given by,[27]

$$r^i(p^i, q^i, m, z) = r(q^i, z) + a(m - p^i) \qquad (9)$$

By taking differences the income variable drops out, thus,

$$\Delta\, r^i(p^i, q^i, q^0, z) = r(q^i, z) - r(q^0, z) - a(p^i - p^o) \qquad (10)$$

Roe *et al*. (1996) show that compensating variation can be obtained from the above by adding or subtracting dollars from $(p^i - p^0)$ until the change in ratings (Δr) equals zero. Then compensating variation for a change from

q^0 to q^i is given by

$$C^i = [\{r(q^0, z) - r(q^i, z)\}/a] - (p^i - p^0) \qquad (11)$$

Binary response (choose one) conjoint can be estimated from (7) using the standard random utility model:

$$\Pr(i) = \Pr\{v^i(p^i, q^i, m, z) + \varepsilon^i > v^0(p^0, q^0, m, z) + \varepsilon^0\} \qquad (12)$$

The probability that the program having attributes *i* is chosen is the probability that the indirect utility of program *i* plus a random error is greater than the indirect utility of program 0 and its error term.

Methodology: Survey Design and Statistical Analysis

Conjoint models individuals' preferences by considering the tradeoffs that they are willing to make. The use of focus groups comprised of individuals drawn from the population of interest allows the researcher to determine what attributes are important to the survey population. Further, the focus group allows the researcher to hone in on changes in the levels of an attribute that are salient to the survey respondent.

Conjoint surveys by their nature are complex. Each possible choice comprises bundles of attributes, with each attribute having different levels. These choices are more readily presented by providing an information packet that may include black and white illustrations and descriptions of attributes and levels. Because of this complexity, most studies to date have used a mail-in format.

Four types of conjoint surveys can be constructed. A ranking format asks the subject to rank alternative scenarios (1, 2…7, etc.) each with different attributes and levels from most to least

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27 This is standard in the extant literature, however marginal utility of income need not be linear.

desirable. Closely related to ranking is the binary choice format, in which the subject is presented with two or more scenarios and is asked to choose the scenario that is most preferred. The binary choice format is mathematically similar to dichotomous choice contingent valuation [Adamowicz *et al.* (1994)], although the survey process is very different. Both rankings and binary choice responses are easily modelled in a random utility framework.

The rating format asks the subject to rate the different scenarios on a bounded integer scale (1,2,…6) from very desirable to undesirable. The ratings format is desirable because it captures the intensity of the individual's preferences, which gives a measure of cardinality and it allows ties between scenarios. Ratings differ from rankings in that ranking occurs relative to the other scenarios and ratings are independent of the alternative scenarios [Matthews *et al.* (1998)]. Roe *et al.* (1996) illustrate that ratings allow a cardinal interpretation of rankings on a bounded integer scale. Rankings force individuals to make distinctions across scenarios i.e., ties are not allowed. Ratings allow an individual to be truly indifferent between one or more scenarios.

A fourth format is the graded-pair comparison. Subjects are asked to consider two scenarios at a time and are asked to rate the intensity of their preference for one scenario over the other.

Which format is best has yet to be proven — there is considerable evidence supporting the validity of all of the formats and some studies have shown that similar results are obtained from all of the models.[28] Which format to use depends on the researcher's output requirements and the nature of the good being examined. Both rankings and binary choice provide ordinal information. An ordinal utility scale captures the respondent's preferences if she is asked to rank the bundles. If the individual is asked to consider the intensity of her preferences for the different bundles of goods and rate them, cardinal utility may be measured. Ratings and graded-pair formats provide cardinal information.

There is considerable debate in the profession about whether cardinal utility is even measurable. The marketing research literature and the more recent literature of conjoint analysis in non-market valuation have largely been able to show that preferences recovered from ratings (cardinal utility) and rankings (ordinal utility) are similar.

The majority of work in CJ thus far has attempted to show the relationships between ratings, rankings, and choice type responses.

The work of Roe *et al.* (1996) presented above takes advantage of the fact that the recovery of rankings and binary choice information is possible if one assumes that preferences are transitive. Given this the ratings format may the most flexible of the formats to use.

Making choices across bundles of goods requires effort. Individuals may first limit their choice sets by separating bundles by segregating them into "good" and "bad" bundles. The bad bundles receive no more attention. If only one bundle is in the good category, the individual's task is

[28] Ben-Akiva *et al.* (1991) and Feather (1973) address the cognitive burden that goes with trying to differentiate between bundles of goods. Ben-Akiva found that the reliability of rankings data decreases as the scenario decreases in rank. Feather was able to illustrate that the ease of selecting ratings may reduce subjects' willingness to make detailed distinctions about the desired attributes. In a survey of parental vales for children, Alwin and Krosnick (1985), Krosnick and Alwin (1988), find that ratings and rankings lead to similar aggregate measures of preferences, and there is a high correlation for the two measures for most subjects. Kalish and Nelson (1991) compare conjoint results using ratings and rankings and find little to choose between the two. The performance of the models was undifferentiated especially when trying to predict choice behavior. Elrod, Louviere, and Davey (1992) found that ratings and ratings worked equally well at predicting choice.

complete. If there are more than one in the "good" category, additional effort is needed. Arguably, many consumer decision processes end here. From the good bundle, the consumer may choose one and stop their choice process. This choice or "choose one" may mimic actual choice behaviour the best, but provides the least amount of information about preferences.

Choice experiments are derived from conjoint and are similar (and can be informationally identical) to CVM. The choice format relies upon the presentation of a choice between one or more alternatives involving changes in one or more of the alternatives' attributes. In contrast to CVM, it relies less on the information contained in the description of the scenario and more on the description of the attributes of each alternative [Boxall *et al.* (1996)].

Conjoint analysis models are estimated using standard limited dependent variable statistical methods. For example, conjoint studies involving discrete choices, categorical, censored, and ordered choices are typically analysed various versions of logit, probit, and tobit models. Ratings data may be analysed using double-hurdle tobit if the cardinality in the data is important [Boyle *et al.* (1998) or probit and ordered probit if only ordinality is assumed [Boyle *et al.* (1998)]. Rank data are analysed with ordered probit/logit and conditional logit, choose-one is analysed with binary probit/logit.

References

Adamowicz, W., J. J. Louviere, and M. Williams (1994). "Combining Revealed and Stated Methods for Valuing Environmental Amenities." *Journal of Environmental Economics and Management* 26:271-292.

Alwin, Duane F. and Jon A. Krosnick, 1985. The Measurement of Values in Surveys: A Comparison of Ratings and Rankings." Public Opinion Quarterly 49:535-552

Arrow, Kenneth *et al*. (1995). "Economic Growth, Carrying Capacity and the Environment", *Science* 268:520-21

Bartik. Timothy J. 1987. Measuring the Benefits of Amenity Improvements in Hedonic Price models, Land Economics, 64:81-88.

Ben-Akiva, Moshe, Takayuki Morikawa, and Fumiaki Shiroishi. 1991. "Analysis of the Reliability of Preference Ranking Data." Journal of Business Research 24(149-164).

Boadway, Robin W. and Neil Bruce (1984). *Welfare Economics*, Oxford, Blackwell.

Boxall, Peter C. Wiktor L. Adamowicz, Joffre Swait, Michael Williams, Jordan Louviere (1996). "A Comparison of Stated Preference Methods for Environmental Valuation." *Ecological Economics* 18:243-253.

Boyle, Holmes *et al* (1998). "Public Preferences for Timber Harvesting Using Conjoint Analysis: A Comparison of Response Formats".

Cummings, Ronald (1996). "Relating stated and revealed preferences: challenges and opportunities", in Bjornstad, David and James R. Kahn (eds.), *The Contingent Valuation of Environmental Resources: Methodological Issues and Research Needs*, London, Edward Elgar Publishing.

Daly, Herman E. (1991). "Towards an environmental macroeconomics", *Land Econ.* 67: 255-59.

Diamond, P.A. and J.A. Hauseman (1994). Contingent valuation: is some number better than no number, *Journal of Economic Perspectives*, 8:45-64.

Elrod, T., J. J. Louviere and K. S. Davey (1992). "An Empirical Comparison of Ratings-Based and Choice-Based Conjoint Models." *Journal of Marketing Research* 29 : 368-77.

Feather, N.T. 1973. "The Measurement of Values." *Journal of Psychology* (5):221-231

Freeman, A.M. III, *The Measurement of Environmental and resource Values: Theory and Methods*, Resources for the Future, Washington, D.C.

Hamilton, K. and E. Lutz, 1996. *Green national Accounts: Policy Uses and Empirical Experience*, Environment Department Paper, the World Bank, paper no 039, Washington, D.C.

Hanemann, W.M., 1994, "Valuing the Environment through Contingent Valuation", *Journal of Economic Perspectives* 8, pp.19-44.

Horowitz, Joel. L. and Jordan. J. Louviere (1993). "The External Validity of Discrete Choice Models Based on Laboratory Choice Experiments." *Marketing Science* 12 : 270-9.

Johnson, F. Reed, Douglas MacNair, and Erin E. Fries (19). "Ill-Considered Values: Contingent Valuation in Light of Evidence form Multiple-Response Stated-Preference Surveys. Manuscript.

Kalish, Shlomo and Paul Nelson. 1991. "A Comparison of Ranking, Rating and Reservation Price Measurement in Conjoint Analysis." Marketing Letters.

Kahn, James R. (1998). *The Economic Approach to Environmental and Natural Resources*, 2nd edition, Harcourt Brace/Dryden Press, Fort Worth.

Krosnick, Jon A. and Duane F. Alwin. 1988. "A Test of the Form-Resistant Correlation Hypothesis." *Public Opinion Quarterly* 52:526-538.

Levin, I. P., J. J. Louviere, A. A. Schepanski, K. L. Norman (1983). "External Validity Tests of Laboratory Studies of Information Integration." *Organizational Behavior and Human Performance* 31 : 173-193.

Louviere, Jordan J. (1988). "Conjoint analysis modeling of stated preferences :a review of theory, methods, recent developments and external validity", *Journal of Transport Economics and Policy* pp 93-119.

Louviere, Jordan (1996). Relating stated preference methods and models to choices in real markets: calibration of CV responses, in Bjornstad, David and James R. Kahn (eds.), *The Contingent Valuation of Environmental Resources: Methodological Issues and Research Needs*, London, Edward Elgar Publishing.

Louviere, J. J., M. F. Fox, and W. L. Moore (1993). "Cross-Task Validity Comparisons of Stated Preference Choice Models" *Marketing Letters* 4 : 205-213.

Louviere, J. J. and D. A. Hensher (1982). "On the Design and Analysis of Simulated Choice or Allocation Experiments in Travel Choice Modeling." *Transportation Research Record* 890 : 11-17.

Louviere, J. J. and G. G. Woodworth (1983). "Design and Analysis of Simulated Consumer Choice or Allocation Experiments: an Approach Based on Aggregate Data." *Journal of Marketing Research* 20: 350-67.

Louviere, J. J. and D. A. Hensher (1982): "On the Design and Analysis of Simulated Choice or Allocation Experiments in Travel Choice Modeling." *Transportation Research Record* 890 11-17.

Louviere, J. J. and G. G. Woodworth (1983). "Design and Analysis of Simulated Consumer Choice or Allocation Experiments: an Approach Based on Aggregate Data." *Journal of Marketing Research* 20: 350-67.

Matthews, Kristy, F. Reed Johnson, Richard W. Dunford, and William Desvousges (1998). "The Potential Role of Conjoint Analysis in Natural Resource Damage Assessments. Triangle Economic Research working paper.

Moffat, I. and M.D. Wilson (1994). "Index of Sustainable Economic Welfare for Scotland", 1980-91, *International Journal of Sustainable Development and World Ecology*, 1:264-291.

Neill, Helen R., Ronald Cummings, Phillip Ganderton, Glenn Harrison, and Thomas McGuckin (1994). "Hypothetical surveys and real economic commitments", *Land Economics*, 70:145-54.

Peskin, Henry M. (1976). A national accounting framework for environmental assets, *Journal of Environmental Economics and Management*, 2: 255-262.

Portney, P.R. (1994). "The contingent valuation debate: why should economists care?", *Journal of Economic Perspectives*, 8:1-18.

Prince, Raymond and Pratice L. Gordon (1994). Greening the national accounts, CBO Papers, Congressional Budget Office, Washington, DC.

O'Neill R.V. , *et al* (1996). "Economic Growth and Sustainability: A New Challenge", *Ecological Applications,* (vol 26, pp 23-24).

Repetto, Robert *et al*. (1989). *Wasting Assets: Natural Resources in the National Income and Product Accounts*, World Resources Institute, Washington D.C.

Roe, Brian, Kevin J. Boyle and Mario F. Teisl (1996). "Using Conjoint Analysis to Derive Estimates of Compensating Variation." *Journal of Environmental Economics and Management* 31(2):145-159.

Rubin, Jonathon, Gloria Helfand and John Loomis (1991). A Benefit-Cost Analysis of the Northern Spotted Owl, Journal of Forestry Bjornstad, David and James R. Kahn (eds.) (1996). *The Contingent Valuation of Environmental Resources: Methodological Issues and Research Needs,* London, Edward Elgar Publishing.

Schimmel, Steven C. *et al*. (1994). Statistical Summary: EMAP - Estuaries, Virginia Province-1991, USEPA, EPA/620/R-94/005, March.

Stevens, Thomas H., Christopher Barret, and Cleve Willis. (1997). "Conjoint Analysis of Groundwater Protection Programs." Agriculture and Resource Economics Review October 229-236.

CHAPTER 7:
by Dennis M. KING and Lisa A. WAINGER[29]

ASSESSING THE ECONOMIC VALUE OF BIODIVERSITY USING INDICATORS OF SITE CONDITIONS AND LANDSCAPE CONTEXT

Introduction

Biodiversity and Ecosystem Services

Of the millions of bacteria, insect, plant, and animal species, a few individual species contribute in direct and measurable ways to economic welfare (e.g., timber, crops, and edible fish); a few more contribute to the quality of life in less measurable but noticeable ways (e.g., dolphins, songbirds, and wildflowers).[30] However, most of the millions of species that exist in nature contribute in obscure and roundabout ways to human welfare (e.g., pollinators and decomposers on land; benthic organisms, plankton, coral, and forage fish at sea). Their lives and functions are so intertwined with each other and with surrounding ecological landscapes that their individual contributions to human welfare, as a practical matter, cannot be isolated.

The term "biodiversity" refers generally to the number and variety of life forms that inhabit an area and was popularised in a 1988 book *Biodiversity*, by the well-known Harvard ecologist E.O. Wilson. It is a vague term that is useful for some purposes, but not useful as a focus for valuation exercises or as a goal of environmental policy. What aspects of "biodiversity" are important? How much can we afford? How much do we need and where? Answering these questions requires research that goes beyond measuring biodiversity or changes in biodiversity. We may never have a clear understanding of our dependence on biodiversity, but there are advantages to knowing which aspects of biodiversity contribute to specific measures of human welfare.

In the introduction of the book, *Biodiversity*, Wilson states that the book "offers an overall view of biological diversity and carries the urgent warning that we are rapidly altering and destroying the environments that have fostered the diversity of life forms for more than a billion years." This

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29 Sponsored by the OECD. Prepared under Cooperative Agreement: 65-7482-8-335 between the US Environmental Protection Agency, Office of Policy Analysis, US Department of Agriculture, Natural Resource and Conservation Service and University of Maryland Center for Environmental Science, Chesapeake Biological Laboratory.

30 Estimates of the number of species are less precise than estimates of the number of species that have become extinct. Wilson (1988) estimates that there are 1.3 million species of plants and animals, excluding insects, and that if insects are included the number of species ranges from 5 million to 30 million.

provides the correct context for developing valuation methods related to biodiversity. The focus should not be on biodiversity *per se*, but on protecting and restoring environments on which biodiversity depends. The biodiversity in the Florida Everglades, for example, is relatively low. However, the capacity of the Everglades to filter nutrients, sediments, and contaminants from surface water as it flows to the coast makes it enormously important to the biodiversity of coastal ecosystems and nearby coral reefs. Increasing the biodiversity of the Everglades at the expense of its capacity to trap contaminants and regulate the flow of surface water to adjacent coastal ecosystems may lead to an overall loss of biodiversity and other sources of ecosystem values. Strategies for managing the Everglades and most other ecosystems depend on what aspects of biodiversity and what other environmental services we want to protect or restore.

Focus of this Paper

In this paper, we focus on indicators of economic value that are related to ecosystem services. We describe a method of developing indicators in which the economic value of an ecosystem is derived from the value of the flow of beneficial services it is expected to provide over time. These services, in turn, depend on site characteristics (e.g., soil, hydrology, vegetative cover, species richness and diversity) and landscape context (proximity to other natural and man-made features and to people).

We use an ecosystem "production function" to reflect how combinations of on-site and off-site inputs (site and landscape indicators) determine the level of ecosystem *services* which we define generally as the beneficial outcomes of ecosystem *functions* (see Box 7.1). Other indicators of landscape features reflect supply and demand for services, who has access to services, the cost of access to services, and natural and man-made threats to site and landscape conditions that put service flows at risk. Box 7.1 defines some key terms including some that we use as building blocks of ecosystem value as we develop an indicator system that can be used to assess and compare ecosystems.

Box 7.1 Definition of terms related to ecosystem valuation

Functions, services, values, risk and several other terms are used in different ways in the ecosystem assessment literature and in the economics literature. The following definitions are offered here to minimise confusion over what will be used in the following sections as building blocks of ecosystem value indices

Functions – the biophysical processes that take place within an ecosystem (e.g., fish and waterfowl habitat, carbon cycling, nutrient trapping). The level of ecosystem function depends on site and landscape characteristics and can be assessed independently of any human context.

Services – the beneficial outcomes that result from ecosystem functions (e.g., better fishing and hunting, cleaner water, better views, reduced human health and ecological risks). These require some interaction with, or at least some appreciation by, humans. However, they can be measured in physical terms (e.g., increased catch rates, greater carrying capacity, more user days, reduced risk, property damage avoided). The capacity of an ecosystem to provide services can be estimated without any ethical or subjective judgements about how much the services are worth. The types of potential services depend to some degree on the level of functions, but predominantly on other factors (e.g., access, proximity to people or problems caused by people).

Values – Defined in strict economic terms, the full range of ecosystem benefits includes each person's *willingness-to-pay* in dollars for each ecosystem service summed across all people and all services. In most cases, tracing and estimating the absolute (dollar) value of an ecosystem is impossible. However, overall willingness to pay for a ecosystem services depends on the number of people with access, their income and tastes, the cost of access, the availability of substitutes, and other factors related to local, regional, and national supply and demand.

Features – on-site characteristics of an ecosystem (e.g., soil, vegetative cover, hydrology,) that establish its capacity to perform or support various environmental functions (e.g., support fish, birds, wildlife)

Landscape context – proximity of the ecosystem to other natural and man-made features in the surrounding landscape. Landscape context influences: a) the ecosystem's opportunity to function at capacity, b) the services that will flow from various functions, c) the value of those services, and d) the risk that the services will not persist.

Preferences – the identification of one ecosystem service as being more important or valuable than another. For purposes of comparing ecosystem values, individual and community indicators of ranked preferences can be as useful as absolute (dollar-based) estimates of value and are much easier to determine.

Risk – the volatility of potential outcomes. In the case of ecosystem values, the important risk factors are those that affect the possibility of service flow disruptions and the reversibility of service flow disruptions. These are associated with controllable and uncontrollable on-site risk factors (e.g., invasive plants, overuse, and restoration failure) and landscape risk factors (e.g., changes in adjacent land uses, water diversions).

Building Blocks of Ecosystem Values

Four of the terms listed in Box 7.1, *features, functions, services*, and *values* refer to attributes of ecosystems that are related to one another and are often used to represent one another. Improvements in fish habitat or in the abundance of fish, for example, are often used to represent improvements in the value of a fishery. An increase in the vegetative cover of a wetland (a feature) is sometimes assumed to imply an increase in the wetlands nutrient trapping capacity (a function) which is presumed to improve downstream fish habitat (a function) which leads to better fishing (a service) and an increase in willingness to pay for fishing (a measure of value). However, there are three reasons why it is important to maintain clear distinctions between these terms when attempting to assess and compare ecosystem values.

First, the information needed to evaluate each of them and the criteria used to assess ecosystems with respect to each of them are significantly different. The features of an ecosystem that might give it a high capacity to provide a particular function (e.g., suitability as waterfowl habitat) do not guarantee that it will actually provide a high level of *function* (e.g., attract or support the greatest number of waterfowl). Ecosystems that provide the highest level of *function* may not provide the highest level of *service* (e.g., birding, hunting, educational, scientific opportunities). Those that provide the highest levels of service may not provide the greatest *value* (e.g., aggregate "willingness-to-pay" for birding, hunting, educational opportunities). And, lastly, the ecosystem that generates the greatest value may not result in a distribution of benefits considered *equitable* (e.g., opportunities for rich compared to. poor or urban compared to. rural).

Second, there are significant differences in the attributes of ecosystems that allow them to provide different types of ecosystem functions and services. Box 7.2 provides a list of the most frequently cited ecosystem services. Some of these are provided best by ecosystems that are located away from people and are surrounded by undisturbed natural landscapes (e.g., endangered species habitats). Others require that the ecosystem be relatively close to people (e.g., educational and recreational opportunities, flood damage prevention, aesthetics). Similarly, some services, such as those associated with sediment, nutrient, or contaminant trapping, are provided only if the ecosystem is located near disturbed landscapes where runoff is a problem. Others, such as breeding habitat for migratory waterfowl, are provided more effectively at sites in undisturbed landscapes. Unless the factors affecting the various functions, services, and values of an ecosystem are considered separately, it is impossible to compare ecosystems with similar features in terms of specific or overall measures of value.

The third and most important reason for distinguishing between these terms is that most widely used analytical methods for assessing and comparing ecosystems do not focus on the biophysical or socio-economic linkages that are necessary for an ecosystem to contribute to human welfare. Standard ecosystem assessment methods were developed primarily by scientists and evolved as extensions of morphological studies — studies of the form and structure of biological systems.[31] They focus on ecosystem features and usually employ indicators that refer to an ecosystem's biophysical capacity to provide functions. Many of them refer to functional values or value indices when describing functional capacity, but they rarely address outcomes related to services or values as the terms are defined in Box 7.1.

Ecosystem valuation methods, on the other hand, have been developed primarily by economists and are limited in scope at the opposite extreme. They attempt to assign values to ecosystem services, usually in absolute (dollar) terms, without much regard for the specific ecosystem features or functions that generated them.[32] The fact that many ecosystem services generate off-site benefits that are difficult to trace and measure and result primarily in non-marketed benefits make them extremely difficult and expensive to measure and trace back to specific ecosystems. The methods that must be used are usually too expensive to be applied to the full range of ecosystem services and, to date, have been applied to only a small sub-set of them. The following section provides a critical overview of dollar-based ecosystem valuation methods and outlines the need for an alternative.

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[31] The most comprehensive ecosystem assessment methods are those developed for wetlands. Most of these result in numerical indicators of specific wetland functions. Some are used to "score" wetland trades and to assess the success of wetland restoration. A recent review of these methods is provided in Bartoldus (1999).

[32] Several recent textbooks outline non-market methods of assigning value to ecosystem services and discuss the practical problems of applying the methods usefully. In particular, see Smith (1996) and Bateman and Willis (1999).

Box 7.2 Categories of Ecosystem Services

Active

1. ***Commercial uses***
   - 1.1 Agriculture
   - 1.2 Trapping
   - 1.3 Mining (including genetic)
   - 1.4 Forestry
   - 1.5 Fisheries

2. ***Recreational uses***
   - 2.1 Fishing
   - 2.2 Swimming
   - 2.3 Hiking
   - 2.4 Nature viewing
   - 2.5 Hunting
   - 2.6 Birding
   - 2.7 Boating

3. ***Municipal uses***
   - 3.1 Groundwater: recharge/discharge
   - 3.2 Drinking water purification
   - 3.3 Pollution prevention

4. ***Other uses***
   - 4.1 Aesthetics - visibility, odor, noise
   - 4.2 Education/learning opportunities
   - 4.3 Research/scientific opportunities
   - 4.4 Cultural/spiritual enrichment

Passive

5. ***Property damage avoided***
   - 5.1 Flooding
   - 5.2 Storm, waves, surge
   - 5.3 Siltation/sedimentation
   - 5.4 Over-nutrification
   - 5.5 Noxious weed infestations

6. ***Human health risks/costs avoided***
   - 6.1 Nutrient cycling
   - 6.2 Carbon cycling
   - 6.3 Chemical cycling
   - 6.4 Oxygen cycling

7. ***Ecosystem health risks avoided***
   - 7.1 Biodiversity support
   - 7.2 Endangered species protection
   - 7.3 Protection of ecological infrastructure

8. ***Climate regulation***
   - 8.1 Global climate effects/attenuation
   - 8.2 Microclimate effects/attenuation

9. ***General non-use*** (can be attached to places, species, features, etc.)
   - 9.1 Existence values
   - 9.2 Option values
   - 9.3 Bequest values

Monetary and Non-Monetary Measures of Ecosystem Value

Monetary measures of ecosystem value

Economic values are not the only useful measure of value for ecosystems or anything else. However, in conventional economics, it is generally accepted that a measure of value should be based on what people want, and that people, not the government, scientists, or preachers, should be the judge of what they want. Based on this individualistic notion of value, the amount of one thing a person is willing to give up to get more of something else is considered a fair measure of the relative value of the two things in the eyes of that person. Dollars are an enormously useful and universally accepted basis for expressing and comparing economic values because the number of dollars that people are willing to pay for something reflects how much of all other for-sale goods and services they are willing to give up to get it. In the case of ecosystems it is important that measuring the economic value of something based on this notion does not require that it be bought and sold in markets. It only requires that someone estimate how much purchasing power (dollars) people would be willing to give up to get it (or would need to be paid to give it up) if they were forced to make a choice.

The three general approaches to estimating the economic value of ecosystem services are outlined in Box 7.3. People can *reveal* the dollar value they place on some services by their purchasing decisions; people can *express* the dollar value they place on some services through "willingness to pay" surveys; and people's "willingness to pay" for some services can be *imputed* based on the costs they would incur if the services were not provided.

Box 7.3 Three accepted approaches for estimating economic value

Revealed willingness to pay - (e.g., market prices). – When people purchase something (e.g., a home near a wetland) or spend time and money to get somewhere (e.g., a fishing spot or bird watching dependent on a nearby wetland) they reveal that they are willing to pay at least what they actually spend; they may be willing to pay more.

Expressed willingness to pay, (e.g., survey results). – Many ecosystem services are not traded in markets (e.g., a scenic view or a day of bird watching) so people may never "reveal" what they are willing to pay for them. Simply asking them what they would be willing to pay can sometimes yields useful results. Surveys of "willingness to pay" are expensive and controversial and usually yield results that are reliable only when questions are asked about specific wetland services provided in specific contexts.

Derived willingness to pay (e.g., circumstantial evidence). – This method involves tracing and measuring the functions provided by an ecosystem (e.g., retaining floodwater, reducing wave energy, maintaining water quality) and estimating what people would be "willing to pay" to avoid the adverse effects of losing them. The dollar value of flood and siltation damage avoided because of a wetland is an example of derived willingness to pay for ecosystem services.

Convincing arguments can be made that it does not make sense to try to assign economic values to ecosystems using these three generally acceptable methods because: a) most important ecosystem services are not traded in markets so people cannot *reveal* the dollar value they place on them; b) people do not know about or appreciate the many functions and services that ecosystems provide and therefore will not *express* that they are willing to pay as much as they should to protect ecosystems; and c) ecosystems generate so many diverse functions, services, and products that the cost of tracing and measuring all of them to *impute* their economic value is prohibitive.

For example, consider what would be involved in developing a comprehensive estimate of the economic value of just one type of ecosystem, a wetland, that could provide nearly all of the services depicted in Box 7.2. Measuring the flow of some services would require tracing biophysical linkages across vast distances in space and time and assigning values to them using a host of different valuation studies. The dollar value of next year's catch of flounder off the central Atlantic coast of the U.S., for example, depends in critical ways on conditions in spawning and feeding areas supported by wetlands hundreds of miles to the north in the North Atlantic. Moreover, the effects of changing fish spawning and nursery conditions in the North Atlantic may take several years to affect fishery values further south. Countless other examples of indirect and induced ecosystem benefits could be used to illustrate the difficulty of estimating the overall value of ecosystems in terms of dollars. In fact, after about twenty years of applied research the literature on ecosystem valuation still consists primarily of conceptual and theoretical papers and partial estimates of ecosystem values associates with a few specific services. Box 7.4 provides brief descriptions of nine ecosystem valuation methods. Detailed description of the pros and cons of each of these methods and illustrations of how they have been used are available in many recent texts and are available on line.[33]

[33] The authors are in the process of constructing a website, which describes ecosystem valuation methods and contains links to other related sites. A prototype of that website can be accessed at: *http://cbl.umces.edu/~dkingweb*

Box 7.4 Dollar-based ecosystem valuation methods

Market price method: Estimates the economic value of commercially traded products and services from wetlands (e.g. peat, hay, hunting rights) on the basis of their market prices. Does not deduct market value of other resources used to bring wetland products to market.

Net factor income method: Estimates the value of wetland resources in commercial production by estimating the profits of wetland-dependent commercial activities after payments are made for other factors of production.

Travel cost (TC) method: Used to estimate the value of recreational benefits generated by wetlands. Assumes that the value of a site is reflected in how much people are willing to pay to get there.

Hedonic pricing - Property value (PV) analysis: Hedonic techniques assume that the price paid for a commodity is directly related to the supply of the commodity's attributes. Most common is the PV approach, which uses variations in property values to reveal implicit values and demand for environmental amenities.

Contingent valuation (CV) method: The only available technique for estimating most non-use (CV) values. Questions are posed to individuals directly about their willingness to pay (WTP) or willingness to accept (WTA) payment. Controversial and expensive; often impractical.

Valuation transfer: the activity day method: simple transfer: an activity day valued at one site is used to value the same activity at the study site. Values are usually site/location/user specific, but transfers can be useful for gross estimates of recreational values.

Replacement cost (RC) method: Estimates the value of a non-market service based on the cost of substitution. This involves three steps: estimate level of service provided, identify least cost alternative, and establish public demand for this alternative.

Damage cost (DC) method: Estimates the value of a service based on the cost of damage that may result from its loss. Steps include assess service level, estimate potential damage, translate to dollar terms, and identify possible substitute.

Opportunity cost (OC) method: Proxy value for uncertain wetland functions/services calculated on the basis of the cost of foregone development values and appropriate least-cost substitutes.

The difficulties associated with dollar-based ecosystem valuation do not end even when researchers have generated dollar estimates and made them available to policy makers. Box 7.5 is provided as a warning to those who decide to use dollar-based ecosystem valuation to influence environmental policy without understanding how the numbers were generated or how they can be abused. It contains brief case summaries of "valuation backfires" based on experiences of the authors. In each case dollar estimates of wetland value were offered in an attempt to influence wetland policy, but did not have the intended outcome. Some of the situations described took place in unofficial hearings and conference settings where actually they had very little impact on wetland policy. They are presented here as they would unfold if they took place in more formal legal or administrative proceedings where they would have a more direct impact on the fate of wetlands.

Box 7.5 Risks of using dollar-based wetland values

Case # 1 The "Willingness to Pay" survey

At a coastal zone hearing, a wetland advocate bases his testimony on survey results published in the journal *Wetlands* (July, 1995) showing that people are willing to pay US$ 100 per household to protect wetlands within
25 miles of their homes. An opposing expert points out that the survey did not specify the type, size, or condition of the wetland or how many other wetlands were in the survey area. She presents evidence that the survey results in a nearly infinite dollar value being placed on tiny degraded wetlands in an urban settings – where there are many households – and a very low value being assigned to large pristine wetlands in rural areas. Admitting that the cost of doing the survey correctly would be prohibitive the wetland advocate withdraws his testimony and the wetland in question is permitted for development. Bad willingness to pay surveys do not hold up; good ones are expensive and still may not hold up.

Case # 2 The derived fishery value approach

Studies show that coastal wetlands in Massachusetts support over 75% of commercially valuable fish species. However, Massachusetts fisheries have been so mismanaged and over-fished over the past twenty years that their economic value is near zero. A contracted study to estimate the "derived value" of wetlands to the state's fisheries yields estimates that are less than one dollar per wetland acre. Using this method, the mismanagement of fisheries results in very little justification for protecting the wetlands on which fishery recovery may depend.

Case # 3 The hedonic housing value approach

At a public hearing to consider a wetland development, a wetland advocate cites a wetland valuation study showing that the average price of a home adjacent to a wetland in the Chesapeake Bay area is US$ 10,000 higher than an identical home that is not adjacent to a wetland. Later in the day experts representing the prospective wetland developer accept this as a valid basis for comparing economic value. They then provide results from a similar study showing that the average price of a home adjacent to a wetland area that has been filled and bulkheaded with a dock on an adjacent water body is US$ 50,000 to US$ 80,000 higher than a house not adjacent to a wetland. It is dangerous to validate a statistical method unless you know how it can be used against you.

Case # 4 Benefit transfer approach

An environmental group presents testimony in Oregon based on a widely disputed study in Louisiana that generated a wetland economic value of US$ 28,000 per acre. After disputing the validity of the estimating method and of using estimates from Louisiana in Oregon, the opposing side agrees to accept the number as fact, and points out that the county already requires US$ 40,000 per acre in compensation for wetland impacts as part of its "in lieu" mitigation fee program. Later in the year a group of wetland developers who are also paying US$ 40,000 per acre as wetland impact fees sue the state to reduce the fee and, using evidence presented by the environmental group, get the fee lowered to US$ 28,000.

Case # 5 The replacement cost approach

At the request of state wetland managers, local engineers estimate that the cost of trying to restore a 1,000 acre bottom land non-tidal wetland area that is being threatened with development to pre-colonial conditions is over US$ 300,000 per acre. This figure is used at a public hearing as an indicator of "wetland value." Under questioning, the wetland manager agrees that "no one in his right mind" would spend US$ 300 million to try to restore this 1,000-acre site to pre-colonial conditions. When asked if it was fair to offer the US$ 300,000 per acre figure as an estimate of the economic value of this wetland area the wetland manager admits he is not sure it is. When asked if the US$ 3,000 in fees paid to the engineering firm to estimate restoration cost was a good use of tax dollars he admits that he is sure it wasn't.

Non-monetary indicators of ecosystem value

Difference between Absolute and Relative Measures

To illustrate the potential of relative (non-dollar) ecosystem valuation indicators and how they differ from absolute (dollar-based) estimates of value consider the wetlands and landscape setting depicted in Figure 7.1. Assume that the two wetlands at Site A and Site B are the same size, the same shape, and have identical biophysical characteristics. Based on site conditions alone they can be expected to have the same *capacity* to provide all wetland function. However, the different landscape contexts of the two sites affect the functions they will actually provide, the services associated with them, the value of the services, and the risk of service flow disruptions.

If the dollar-based valuation methods outlined in the previous section were applied to estimate the typical or average economic value of this type of wetland, the same dollar amount (e.g., US$ 1 per acre or US$ 1 million per acre) would be assigned to both sites. However, as a practical matter, the purpose of the valuation exercise may be to compare the economic value of Site A or Site B. This would be the case, for example, if limited funds are available to restore Site A or Site B, or if Site A is being traded for Site B, or if investments are being considered to control harmful invasive species at Site A or Site B. In such situations having an "average" dollar value to assign to both sites would not be very useful. It would be more useful to know how far above or below the average value the two sites are expected to be. In fact, having a relative measure of economic value for each site (e.g., percent above or below average) would be useful for comparing the two sites even if the average value itself had never been estimated in absolute (dollars) terms. The indicator system proposed in this paper is based on the notion that it is easier and sometimes more useful to consider the factors that affect whether a particular site is likely to provide values above or below average than to have an absolute measure of the average itself.

Illustration of Indicators

To illustrate the logic of the indicator system that will be developed in the following section consider the relative "value" of the two wetland sites depicted in Figure 7.1 with respect to three specific functions: wildlife habitat, fishery support, and water quality improvements.[34] The biophysical features of the two sites are identical so they have exactly the same on-site capacity to provide all three functions. However, because of differences in landscape contexts, Site A can be expected to generate significantly more services and values than those provided at Site B. This is the case because:

- Site A has more opportunity than Site B to provide wildlife support because it is accessible to wildlife from the upland wildlife refuge area whereas the road blocks the wildlife corridor to Site B.
- Site A has more opportunity to support fish habitat than Site B because it is adjacent to fish habitat whereas Site B is not.
- Site A has more opportunity to improve water quality than Site B because of its proximity to the coast and because its longest dimension is parallel to the coast therefore providing greater "buffering" potential.

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[34] This illustration is adapted from a paper that describes a method of conducting "habitat equivalency analysis" and contains a more detailed accounting of site differences that affect wetland values. See King (1997a).

- Site A is downslope of agricultural land uses that generate harmful levels of nutrients that without a wetland at Site A would reach the water body.
- Site B, on the other hand, creates a narrow "buffer" away from the coast and has no significant upslope source of nutrients to filter.
- Even with a source of nutrients, the payoff from filtering nutrients at Site B would be less than at Site A because Site B is adjacent to a polluted and fast-moving section of the water body where harmful effects would be negligible.

To make Site A seem even more "valuable", Figure 7.1 also depicts Site A where it provides aesthetic and educational opportunities to a nearby residential population whereas Site B is surrounded by industrial sites and private forest lands which limits its amenity values. To be even more sure that the relative value of Site A is greater than Site B, consider differences in the exposure of the two sites to natural and man-made risks that could disrupt the flow of beneficial services from the two sites. For example, assume that a new 10-year land use plan for the region designates the area around Site A as "environmentally sensitive, no industrial use" whereas the area around Site B as "industrial use, fast-track permitting". Not only are the current environmental values provided at Site A higher than those provided by Site B, they are less likely to be decline in the future as a result of land use changes. The same type of differences in site risks might be associated with the exposure and vulnerability of the two sites to threats from water diversion, sea level rise, invasive species, or other factors.

Based purely on the geographic information presented in Figure 7.1, it is possible to pose a technically and legally defensible argument that a wetland at Site A is more valuable than an identical wetland at Site B. This demonstrates the logic of the indicator system that we outline in the following section, which relies on geographic information systems (GIS) to compare ecosystem value.[35] Without judging the merits of protecting or improving wetland capacity at both Site A and Site B, the illustration shows that indicators can be used to reflect important valuation tradeoffs without resorting to conventional (dollar-based) valuation.

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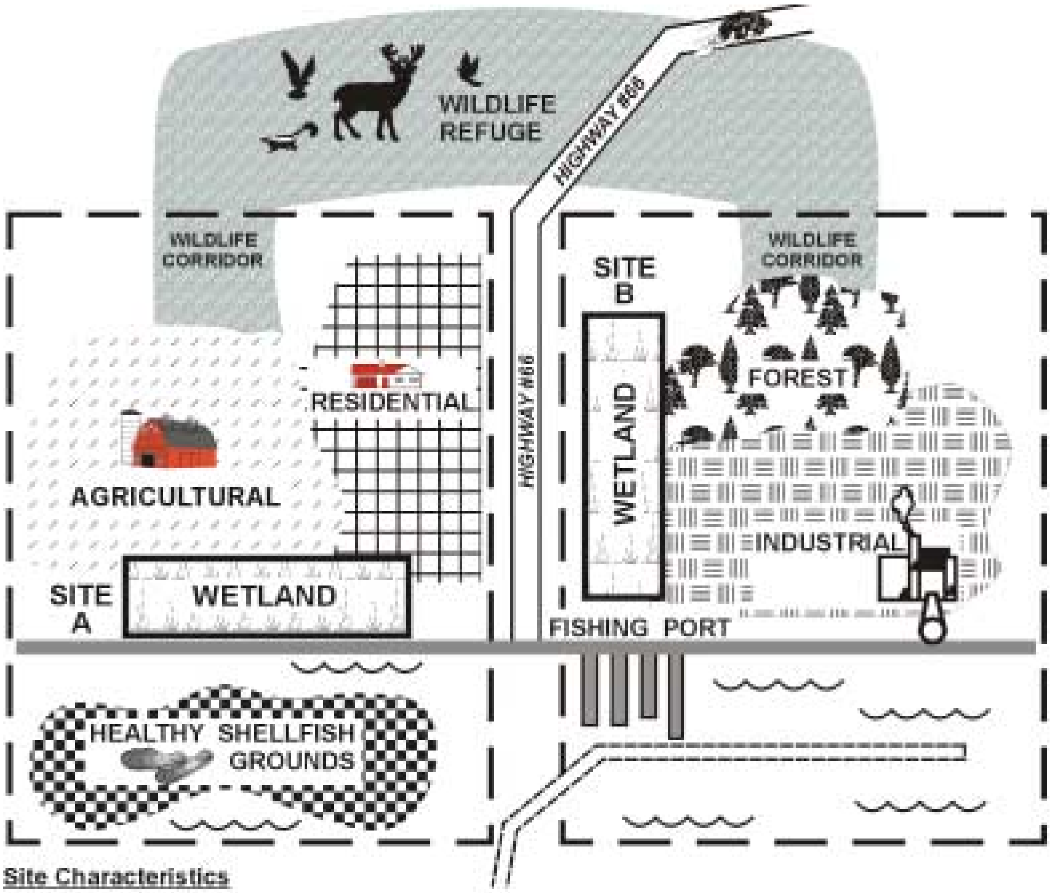
[35] A useful description of how applications of Geographic Information Systems (GIS) can be used in the social sciences is provided in *People and Pixels* published by the U.S. National Research Council (1998).

Figure 7.1 Effects of Wetland Location on Function, Service and Value



<u>Site Characteristics</u>

Wetland Site A and Wetland Site B are identical in size, shape and bio-physical characteristics and are located in the same sub-watershed on either side of Highway 66.

<u>Landscape Context</u>

SITE A

- near the coast, downstream is a beach area
- adjacent to large healthy shellfish grounds that are accessible to the community
- upslope is agricultural land (nutrient run off)
- wildlife corridor open from the North
- near residential areas (aesthetics, scenic)
- good access, adjacent public lands
- access to many urban poor people

SITE B

- slightly off coast, downstream is industrial site
- adjacent to fishing port and small shellfish beds that are contaminated and remote
- upslope is forest (no nutrient runoff)
- wildlife corridor is blocked by Highway 66
- nearby industrial sites (no proximity to people)
- poor access, surrounded by private lands
- access to few suburban rich people

Developing ecosystem value indices

The Basis of an Indicator System

The conceptual basis of the proposed indicator system is a widely used analytical tool called a production function.[36] This is a relationship that shows how the quantity and composition of the outputs of a productive process are related to the quantity and composition of inputs that are used in the process. Production functions are at the core of most economic studies related to industrial, agricultural, and manufacturing operations. They are also the source of many indicator systems that are used to determine the risks and potential payoffs from corporate investment portfolios. The availability of inputs and the likely effects of controllable and uncontrollable risks on the availability of inputs are critical determinants of how reliable and how valuable a productive process is. Rules of "comparative advantage", "marginal value product", and "derived asset value" are all based on the concept of the production function.[37]

Mathematical production functions are usually based on underlying engineering relationships, but many times they include uncontrollable relationships related to natural systems. In fisheries and agriculture, for example, many of the input categories that are used in production functions are natural and uncontrollable and are frequently represented by indicators (e.g., soil productivity, fish abundance, weather).[38] Production functions that treat ecosystem services as an output and on-site and off-site indicators of resource conditions as inputs are not much different from other forms of production functions. In the indicator system that is introduced here, on-site inputs are assumed to affect the capacity of the site to provide functions and certain offsite or landscape inputs determine the "rate of capacity utilization." Other landscape indicators reflect the likelihood that the functions provided at the site will generate services, that the services will have value, and that the flow of services is sustainable.

Types of Indicators

Accepting the premise that the economic value of an ecosystem is derived from the economic value of the services it is expected to provide over time has some clear implications for value-based indicator development. It means that the effort should focus on forecasting, not on

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[36] The concept of the production function as a general relationship between inputs and outputs is described in all standard microeconomics texts (e.g., Samuelson and Nordhaus1995, elementary; Mankiw 1997, elementary; and Varian 1992, intermediate). Clark (1976) describes the use of mathematical production functions in natural resource industries where the results of natural processes provide a basis for developing indices of critical inputs.

[37] One production process is said to have a "comparative advantage" over another if it can provide the necessary conditions for production at a lower cost. The "marginal value product" of an input (MVP) is the incremental increase in the output that results from an incremental increase in the use of the input multiplied by the price of the output; inputs are usually purchased until the MVP declines to the input price level. "Derived asset value" refers to the fact that the economic value of an asset can be estimated as the sum of the net economic value of the stream of services it is expected to provide over time discounted to their present value.

[38] The production function used in fisheries, for example, is referred to as the classic catch equation ($C=qEP$) and includes three indices representing; fish abundance (P), the catchability of fish and the power of fishing gear (q), and fishing effort (E). A description of bioeconomic production functions and the characteristics of capital and resource indicators in them is provided in Clark (1976).

description, and that the usefulness of descriptive indicators depends on what clues they provide about future service flows and their values. It also means that indicators of potential adverse changes in site or landscape conditions are important because they reflect the likelihood of service flow disruptions. This focus on forecasting service flows rather than describing ecosystem conditions distinguishes the proposed indicator system from most other ecosystem indicator systems.[39]

Box 7.6 lists the basic questions that are addressed by the indicator system. The following topics summarise the types of information used within the indicator system to answer these questions.

On-site features – biophysical characteristics that determine if the wetland has the capacity to provide particular functions.

- Characteristics of vegetative cover.
- Characteristics of soil and topography.
- Characteristics of hydrology.

Landscape context – off-site features that determine:

- How much of a wetland's functional capacity will be utilised
- Whether functions that are performed will generate services
- How valuable the services that are generated will be
- The likelihood of service flow disruptions

Components of landscape context include:

- *Topographical characteristics* – adjacent and nearby hydrological/ geological features (e.g., upslope/downslope gradients, proximity to water bodies, floodplains)
- *Habitat characteristics* – connectedness to fish, wildlife, fur-bearer habitats (e.g., flyways, wildlife corridors, other wetland areas)
- *Man-made characteristics* – proximity to residential, commercial, industrial land uses, including proximity to roads, parking lots, right of ways, etc.
- *Demographic characteristics* – size, age, mobility, ethnicity, and geographic distribution of human populations that benefits from specific wetland services.
- *Socioeconomic characteristics* – income, assets, and other characteristics of the population that benefits from specific wetland services.
- *Scarcity of services* **–** the overall abundance of wetland services in the region and the availability of similar services in nearby regions. All other things equal, fewer perfect and near-perfect substitutes mean higher "willingness to pay" per unit service.
- *Scale of services* **–** the size of the population that has access to the service. All other things equal, the greater the number of people with access to wetland services the greater the economic value of the services.

[39] A description of environmental indicators and how they can be used is provided in Hammond, et al (1995). A review of numerical indicator systems used to assess and compare ecosystems and their potential applications in forecasting ecosystem service flows is provided in King (1997b).

- *Cost of service access* **–** time and money required to take advantage of the wetland service. All other things equal, the lower the cost of access to a wetland service, the greater the value of the services to those who have access.
- *Revealed Preferences* **–** participation rates, purchasing patterns, subscriptions, donations, and other decisions that reveal preferences for wetland services.
- *Stated Preferences* – relative values assigned to wetland services by individuals, community leaders, elected officials, or citizen "valuation" juries.
- *Imputed Preferences* – individual and community preferences assigned to wetland services and imputed to wetland functions and features as a result of choice modelling, conjoint analysis, and other forms of multi-attribute analysis.

Box 7.6 Essential Questions About Relative Ecosystem Values

Functions

What environmental functions does this ecosystem have the capacity to provide?
Does the ecosystem's landscape context allow it to provide these functions?
If so, are there factors that will cause it to function at less than full capacity?
Are there factors that may cause it to function beyond its sustainable capacity?

Services

What services, products, and amenities will these ecosystem functions generate?
Over what geographic area will people benefit from these services and products?

Values

How scarce are these services, products, and amenities in this area?
How many people benefit from them; what is their income, ethnicity, etc.?
How much does it cost in money or time for people to enjoy these services?
Are there near-perfect *natural* substitutes that exist or could be developed?
Are there near-perfect *man-made* substitutes that exist or could be developed?
How could the affected population adapt to having fewer of these services?
How much would the affected population benefit from having more of these services?

Risk

How might future development make the services provided here more/less important?
How vulnerable are services generated by this site to temporary/permanent disruptions?
How restorable are these services in this region compared to other regions?
How might future development make the services provided here more/less vulnerable?
Will demographic/land use change increase/decrease preferences for these services?
Will demographic/land use changes increase/decrease availability of these services?

The system we propose uses the following four sets of indicators that combine to generate a single overall index of ecosystem value.

1) Capacity utilisation sub-index

Indicators of landscape conditions that determine how much of the functional capacity of the site is likely to be used.

2) Service capacity sub-index

Indicators of landscape conditions that limit or enhance the level of services expected per unit of function.

3) Service value sub-index

Indicators of local, regional, and national supply and demand conditions, individual and community preferences, and the substitutability and replaceability of the service which reflect the expected value per unit service.

4) Service risk sub-index

Indicators of the likelihood of future disruptions in service flows that affect the value of expected ecosystem services. These are related to the exposure and vulnerability of the site or other critical landscape features to such threats as floods, droughts, fire, disease, infestations, water diversion, pollution, and industrial development.

Flow of Indicator Development

The flow of indicator development using site capacity indicators and the four sub-indices listed above is displayed in Figure 7.4. The initial assessment of the site's functional capacity can be expressed using any of the biophysical ecosystem assessment methods and ecosystem conditions with respect to particular functions.[40] A recent paper Bartoldus (1999) describes approximately 30 ecosystem assessment methods that generate "functional capacity indicators" or "habitat suitability indicators".[41] The sub-indices listed above are then used to adjust site indices of functional capacity to reflect how landscape factors influence expected levels of service, value, and risk. If necessary the "adjusted service value" estimates associated with each function can be weighted using service preference weights and aggregated to form a single overall value index.

The details of how individual sub-indices are developed are provided in the last section of this paper. The following section provides an illustration showing how the indicator system can be used to assess wetland trades.

Illustrations of Indicator Applications

Wetland trading: For illustration, assume that each of the four sub-indices listed above is designed to range around a mean or average value of one. At a site in a landscape context that is

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40 The review of ecosystem assessment methods provided in Bartoldus (1999) provides evidence that most of them focus on site conditions and ignore landscape linkages. They establish the capacity of an ecosystem site to provided functions, not the opportunity the ecosystem site has to provide functions, and usually do not address landscape factors that link biophysical functions with services that matter to people.

41 Most "functional capacity indicators" and habitat suitability indicators" range from zero to one with zero representing no functional capacity and one representing a high level of functional capacity. Several recent methods rely on high quality specimens of an ecosystem as a reference to establish benchmark indicator values, which are then used to rank other sites. See the description of the "hydrogeomorphic"(HGM) wetland assessment method in Brinson, et al (1993).

average in every way, for example, an increase in functional capacity of 10% would be expected to result in a similar 10% increase in expected levels of function, service, and value. (.1 X 1 X 1 X 1= .1)

Now consider Site A and Site B in Figure 7.1 and assume that Site A is in a location that is 20% above average in every way and that Site B is in a location that is 20% below average in every way. Since we assumed that the size, shape, and biophysical characteristics of Site A and Site B are the same they would rank the same in terms of functional capacity, and an investment to achieve a 10% increase in functional capacity at both sites would yield the same on-site results and cost about the same. However, because Site A is 20% above average in every way, a 10% increase in functional capacity at that site would result in a 12% increase in function, a 14% increase in service, and a 17.3% increase in value (.1 X 1.2 X 1.2 X 1.2 = .173) (see Figures 7.2 and 7.3). Similarly, since Site B is 20% below average in every way the same 10% increase in on-site functional capacity would result in only an 8% increase in function, a 6.4% increase in service, and a 5.1% increase in value (.1 X .8 X .8 X .8 =5.1) (see Figures 7.2 and 7.3). This implies that investments aimed at improving functional capacity at Site A would result in 340% more economic value than similar investment at Site B.[42] It also implies that allowing wetland mitigation trading that involved gaining an acre of wetland at Site B and losing an acre of wetland at Site A would result in an economic loss of 340% even though the sites themselves are identical.

Value-based analyses such as this could support significant changes in the way ecosystems are compared. In the case of wetland mitigation trading, for example, gains and losses are frequently "scored" using area measures or biophysical measures that reflect functional capacity. In such instances, wetland acreage at Site A and Site B in Figure 7.1 would be considered an even trade. However, if trading rules were designed to achieve "no net loss" of wetland value, the trading rules, based on the indicator system outlined above, would require a "compensation ratio" of at least 3 or 4 acres of wetlands at Site B to compensate for each 1 acre of wetland lost at Site A.[43]

Specification and Measurement of Indicators

Functional Capacity Index

The Functional Capacity Index reflects the capacity of the site to provide a particular function independent of its landscape context. It is based on biophysical characteristics of the site including soil, topography, vegetative cover, and hydrology. The approach described here assumes that an accepted ecosystem assessment method has been used to "score" sites in terms of their functional capacity. The recently developed "hydrogeomorphic" or HGM method of assessing

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[42] The 340% difference her results because the indicators are designed to be multiplicative. For example, if one site provides 20% more services than another and each unit of service provided is worth 20% more than at the other site the difference in the value of services at between the sites is 44% (1.2 X 1.2). In the illustration Site A is 20% above average with respect to four multiplicative indicators and Site B is 20% less than average in each indicator category. As a result Site A achieves a cumulative value of 1.73 (1 X1.2X1.2X1.2) and Site B attains a value of, 51 (1X0.8X0.8X0.8); using the differences in these indicator scores Site A is 340% more "valuable" than Site B ((1.73 – 0.51) / 0.51).

[43] In wetland mitigation the phrase "compensation ratio" is used to refer to the number of acres of created, restored, or enhanced wetland areas required to offset the loss of one acre of natural wetland. An economic interpretation of these ratios and a formula for estimating them using a net present value formulation is presented in King et al (1993).

wetlands, for example, results in Functional Capacity Indicators (FCIs) for around ten wetland functions from sediment and nutrient trapping to waterfowl habitat.[44]

Figure 7.2 Illustration of Indicator Development for Site A and Site B

Prototype indicators

Each site is ranked for each of three wetland functions using four sub-indices that range from 0 to 2 around an "average" value of 1. For a site that is average in every way, an x% change in functional capacity is expected to result in an x% change in functions, services and values.

Site A and Site B have the same functional capacity. However, the landscape context of Site A is above average in every way and the landscape context of Site B is below average in every way. In this illustration, Site A is assumed to be 20% above average (Indices of 1.2 in all categories) and Site B is assumed to be 20% below average (Indices of 0.8 in all categories).

Relative value of Site A and Site B

	Wildlife habitat		**Fishery support**		**Nutrient trapping**	
	Site A	Site B	Site A	Site B	Site A	Site B
Functional Capacity Index	Identical size, shape, bio-physical characteristics ***Score: 1.0***	Identical size, shape, bio-physical characteristics ***Score: 1.0***	Identical size, shape, bio-physical characteristics ***Score: 1.0***	Identical size, shape, bio-physical characteristics ***Score: 1.0***	Identical size, shape, bio-physical characteristics ***Score: 1.0***	Identical size, shape, bio-physical characteristics ***Score: 1.0***
Capacity Utilisation Sub-index	Wildlife Corridor <u>open</u> from North Score: 1.2	Wildlife Corridor <u>blocked</u> from North by Highway 66 Score: 0.8	Traps agricultural sediment, near coast, adjacent to fishing grounds Score: 1.2	Little sediment to trap, off the coast, adjacent to boat channel Score: 0.8	Upslope is farm land generating nutrient flow, natural water flow, non-point discharge Score: 1.2	Upslope is industrial sites and forests (little nutrients) channelized water flow to point discharge Score: 0.8
Service Capacity Sub-index	Near residential areas, accessible, public land Score: 1.2	Surrounded by industrial sites, inaccessible, private land Score: 0.8	Adjacent to large healthy shellfish area, public access, nearby parking Score: 1.2	Few shellfish nearby, little access if there were, near point source discharge Score 0.8	Adjacent to large healthy shellfish area, public access, nearby parking Score: 1.2	Few shellfish nearby, little access if there were, near point source discharge Score 0.8
Value Of Service Sub-index	Accessible to residential population Score 1.2	Access limited to few rich Score: 0.8	Aesthetic recreational opportunities for many poor Score: 1.2	Access limited to few rich Score: 0.8	General water quality improvements, and adjacent to healthy shellfish area, accessible to poor Score: 1.2	General water quality improvements only Score: 0.8
Risk Of Service Sub-index	Ag land in ag preservation district (corridor remains open) Score 1.2	Forest zoned for development (habitat likely to decrease) Score 0.8	Local area is built out – no new sources of sediment Score 1.2	Any future development will generate more sediments (developed replaces forest) Score 0.8	New development upstream would be on sewer Score 1.2	Future development on septic Score 0.8

44 The "hydrogeomorphic" or HGM wetland assessment method results in wetland functional capacity indicators that take very little account of landscape context in determining expected levels of wetland function, and no account of how landscape context affects wetland services and values. The authors are involved in research to extend the HGM method using many of the concepts and applications described in this paper. We are estimating prototype indicators to assess gains and losses associated with actual wetland mitigation trades in the state of Florida.

Figure 7.3 Illustration of indicator development and application:

Improving functional capacity at Site A compared to Site B

	Wildlife habitat		**Fishery support**		**Nutrient trapping**	
	Site A	Site B	Site A	Site B	Site A	Site B
Functional Capacity Index	1.0	1.0	1.0	1.0	1.0	1.0
Capacity Utilisation Sub-index	X 1.2	X 0.8	X 1.2	X 0.8	X 1.2	X 0.8
Level of Function	1.2	0.8	1.2	0.8	1.2	0.8
Service Capacity Sub-index	X 1.2	X 0.8	X 1.2	X 0.8	X 1.2	X 0.8
Level of Service	1.44	0.64	1.44	0.64	1.44	0.64
Value of Service Sub-index	X 1.2	X 0.8	X 1.2	X 0.8	X 1.2	X 0.8
Value	1.73	0.51	1.73	0.51	1.73	0.51
Risk of Service Sub-index	X 1.2	X 0.8	X 1.2	X 0.8	X 1.2	X 0.8
Adjusted Value	**2.07**	**0.41**	**2.07**	**0.41**	**2.07**	**0.41**

Note: Value Index per unit Functional Capacity and, within limits, per unit change in Functional Capacity.

Site A 2.074 (1 x 1.2 x 1.2 x 1.2 x 1.2)

Site B 0.410 (1 x 0.8 x 0.8 x 0.8 x 0.8).

Conclusion: Site A and Site B are identical in size, shape and biophysical characteristics. However, based on differences in landscape context, the "value" of Site A is 4 times higher than the "value" of Site B [((2.074 – 0.410)/0.410)=4.06]. Within limits, increasing the functional capacity at Site A will result in 4 times the economic benefits as increasing the capacity at Site B. If costs per unit of capacity restoration are the same at both sites, the benefit-cost ratio of restoration investments at Site A is 4 times higher than Site B. Note that investments to improve the landscape context of Site B could yield greater payoff than investing directly at either site.

Figure 7.4 Flow of Ecosystem Value Indicator Development

Links between indicators	Question addressed by indictors	Basis of indicators
(1) Functional Capacity Index	(1) Does this site have biophysical characteristics that will allow it to provide this function?	(1)Soil, hydrology, vegetative cover, biological diversity
(2) Capacity Utilisation Sub-index	(2) Does the landscape context of this site enhance or limit its ability to provide this function?	(2) Proximity to other natural and man-made features and to people.
(3) Function index	(3) Based on site condition and landscape context, what level of function will be provided?	(3) Derived from above.
(4) Service Capacity Sub-index	(4) Does the landscape context of this site enhance or limit the amount of services that will result from these functions?	(4) Proximity to other natural and man0made features and to people (series # 2)
(5) Service Index	(5) Based on expected level of function and landscape context, what levels of service will be provided?	(5) Derived from above.
(6) Service Value Sub-index	(6) Based on supply and demand for the service, what is the relative value (WTP) for an additional unit of service?	(6) Availability of substitutes, number of people with access, cost of access, income, wealth, preferences
(7) Value Index	(7) Based on level of service and per unit service value, what is the relative value (WTP) for the service provided by this site?	(7) Derived from above.
(8) Service Risk Sub-index	(8) What is the risk of service flow disruptions as a result of natural or man-made changes affecting site conditions, or landscape context, or both?	(8) Exposure and vulnerability of the site to floods, droughts, fire, invasive weeds, development and other factors.
(9) Adjusted Value Index	(9) Based on the relative value of services and the risk of service flow disruptions, what is the risk-adjusted value of services?	(9) Derived from above.

In this indicator system off-site features are measured by considering landscape characteristics that act to enhance or detract from the on-site features that are the basis of an ecosystem function. A variety of descriptive statistics, particularly spatial statistics (e.g., fragmentation indices), can be used to describe landscape context effects at various scales from data that is generally readily available in the US. These statistics can be derived within a Geographic Information System (GIS) alone or in combination with spatial statistical programs (e.g., FRAGSTATS). They can be less easily developed from other data sources such as paper maps and accessible regional data sets (e.g., Census Bureau data, state planning office data).

Functional Capacity Utilisation Sub-index

The level of function provided at a site depends partly on the functional capacity of the site, but also on the rate of capacity utilisation. A site with high capacity to support waterfowl, for example, may not attract waterfowl if it is adjacent to a highway or a polluted river. The expected rate of capacity utilisation at a site depends in predictable ways on its proximity to other natural and man-made features of the landscape and to people and problems caused by people. Measures of distances to various landscape features that increase or decrease the expected rate of capacity utilisation form the basis of this sub-index.

Service Capacity Sub-index: Connections between Landscape Setting and Services

The level of output (services) that should be expected to flow from an ecosystem depends on more than the level of biophysical function it provides. Conditions of access, adjacent land use, and downstream resources have some obvious effects on the Levels of Service associates with a given level of function. The Level of Service associated with a system's flood storage capacity, for example, depends on the presence of downstream property that would be damaged by flooding. Similarly, the opportunity for an ecosystem to provide certain services, such as fishing, depends on adjacent land uses, the connection between the ecosystem and open water habitat, and accessibility of the water and the fish to humans.

Each type of service associated with each ecosystem function can be examined in terms of landscape features that limit or enhance that specific service. Service Capacity Sub-indices can be developed as multipliers that adjust Levels of Function scores to arrive at Level of Service scores (Figure 7.4).

To illustrate, we selected three specific ecosystem functions: wildlife habitat, water chemistry, and hydrological regulation, and used examples of four potential services (hunting, fishing, flood damage avoided, and aesthetics). The indices focus on aspects of landscape context (e.g., land arrangement, upslope sources, accessibility) that enhance or limit the ability of the site to provide a given service.

A variety of descriptive statistics, particularly spatial statistics, can be used to describe landscape contexts that affect service flows at various scales. These statistics can be derived from a GIS system (where available), or other data sources such as paper maps and generally accessible regional data sets. In particular, variables describing the ability of humans to get to a wetland or adjacent site supported by the wetland will be useful in determining which active use services are being provided. In many cases, the presence of such features as boat ramps, restrooms, and parking areas will reflect Service Capacity. Otherwise, roads or housing density can serve to represent access (Service Capacity) levels. Some useful indices for assessing Service Capacity are described in Table 7.1.

Services that accrue at the regional or state level may be assessed by determining what role an ecosystem site plays in regional planning goals. Zoning maps and/or GIS analysis can be used to assess whether a wetland is part of a coastal protection zone, a drinking water protection area, or a wildlife corridor and therefore fulfilling regional habitat protection services. A variety of land arrangement variables, especially habitat fragmentation measures, may be used to supplement functional indices and relate them to broader scale services. For example, fragmentation measures have been related to such functions as the attractiveness of areas to migrant bird species [Flather and Sauer (1996)]. These established relationships, measured in terms of landscape fragmentation (e.g., perimeter to area ratio of habitat patches or contiguity of patches) offer a measurable variable to use in analysis if migrant species or habitat corridors are not considered in functional assessments.

Physical and biological distinctions that allow an ecosystem site to provide services can be reflected by landscape variables that measure upstream and downstream land uses and land configurations. For instance, using GIS flow network features, site differences can easily be quantified by calculating the upland area that would contribute to surface runoff or shallow underground flow (throughflow) passing into any particular site. While this would not be a highly accurate assessment of how much runoff would be received by a wetland, it would provide a way to differentiate the abilities of wetlands to provide certain types of services. Other landscape analysis can determine whether a wetland is upstream of a water body used by swimmers or fishers, which will partially determine the wetland's ability to contribute to a swimming or fishing service. The likely constituents of runoff can be predicted by considering the types of land covers and land uses in areas generating the runoff. The proximity of toxic discharges will be a factor in whether a wetland provides the service of water decontamination.

Value of Service Sub-index

Arriving at an index of the level of service provided by a site provides a useful basis for beginning to consider its relative economic value. However, the services provided at or by different sites may not be equally valuable. The Value of Service Sub-index is used to weight service flows from different sites in terms of their relative value. These sub-indices are based on conventional economic concepts and are intended to reflect aggregate "willingness to pay" for an incremental unit of ecosystem service based on the economic definition given earlier. Even without specific dollar estimates, a great deal can be determined about the values people place on special services. General measures of the supply and demand of various services exist that can be used to develop indices of relative service values. Other things equal, for example, services that are provided where they are relatively scarce and for which there are few substitutes are more valuable than services provided where they are abundant and for which there are many substitutes. All other things equal, services provided where they benefit more people are worth more than services that benefit few people. All other things equal, reducing risk to species or habitat where it is the least reversible is more important than reducing risk where it is the most reversible.

Our Value of Service Sub-index relies on five measurable factors that can be shown to affect the aggregate willingness to pay for a particular ecosystem service provided at a particular location. These include: 1) the number of people with access to the service, 2) their incomes and wealth holdings, 3) the cost in time or money of getting or keeping access to the service, 4) the availability of perfect or near-perfect substitutes for the service, and 5) people's expressed or revealed preferences for this service compared with other competing services.

Once the geographic range of wetland services is determined, relevant indices of these factors are developed from many different data sources. Zip code-scale demographic, socio-economic, and land use statistics are generally available and are analyzed routinely by regional economists and market analysts to compare regions and sites for various kinds of investments. The purpose of those analyses, like the purpose here, is to compare regional and local supply and demand conditions and population preferences that affect the value (marketability) of certain services. The Nominal Service Value resulting from services provided during any particular time period, then, would be considered the product of the Level of Service multiplied by the Value of Service Sub-index (Figure 7.4). Table 7.2 provides a list of other factors that may be examined as potentially useful indices of the value of wetland services.

Table 7.1 Service Capacity Sub-index Development

Service (On Or Off Site)	Wetland Contribution	Necessary Conditions	Measures Of Necessary Conditions	Potential Components Of Service Capacity Index
Recreational Fishing Opportunities	Provide feeding, breeding and nursery habitat	Game fish present in adjacent or connected water body and fish/larvae have access; People have access	Presence of game fish or larvae; Infrastructure to support fishing in connected waterway	•Obstructions to fish movement; • Fishable classification downstream; • % of time wetland hydrologically connected to adjacent waterway; •Fish population surveys; •Recreational infrastructure (fishing pier, fishing bank area, parking lot size, boat ramp, restroom capacity)
Birding, Hunting and Gathering Opportunities	Provide habitat for fungi, plants, birds and animals that use wetlands	Support of appropriate (esp. diverse or rare) habitat; Access by enthusiasts	Presence of rare or desirable species; Access by birders, hunters, gatherers	•Biodiversity indices; presence/absence data; •Property ownership; •Trail miles; •Hunting restrictions;
Water Quality Maintenance	Trap sediments; Cycle nutrients; Filter contaminants	Surface water usage; Water quality (fishable, swimmable, drinkable)	Sources of erosion and contaminants upslope/ upstream; Runoff, shallow throughflow received, low gradients; Access; Significant contribution to water quality given current conditions	Sources: •Presence of industrial or agricultural activity; •Area in unsewered residential; •Volume of waste water discharged upstream. Usage/Access: •Beaches, •Parking, •Restrooms, •Municipal water intakes
Flood Damage Avoidance	Hydrologic regulation	Vulnerable property downstream; Ability of wetland to hold water	Structures and crops in 100-year flood plain downstream; Depressional area volume; upslope area drained by wetland	• # of structures in 100-year flood plain downstream; ownership type, land use •Depressional area volume; •Upslope area drained by wetland

Table 7.2 Value of Service Sub-index Development

Service (on or off site)	Geographic extent of service	Population benefiting	Supply conditions	Demand conditions	Potential components value of service index
Recreational fishing opportunities	Local	5-mile radius	Alternative sites within 5 miles; Quantity, quality, and capacity of alternative sites within 60 miles	Level and frequency of participation; Expressed / revealed preferences; Leisure time	• # Fishing permits in zip code; •User days; •Contributions and memberships; • # Fishing related businesses; • #Alternative sites; •Preference survey results; •Average income, property value
	Regional	60-mile radius	Alternative sites within 60 miles	Level and frequency of participation; Expressed and revealed preferences; Leisure time	# Fishing permits in state; #Fishing related businesses; •Contributions and memberships; •Average income
Birding, Hunting and Gathering Opportunities	Local	5-mile radius	Alternative sites within 5 miles; Alternative sites within 60 miles	Level and frequency of participation; Expressed, revealed preferences; Leisure time	• # Hunting permits; • #Related businesses; • #Alternative sites; •Value of gathered goods; •Access fees; •Contributions and memberships; •Average income
	Regional	60-mile radius	Alternative sites within 60 miles	Level and frequency of participation; Expressed, revealed preferences; Leisure time	• Number of hunting permits; • #Related businesses; • #Alternative sites; •Value of gathered goods; •Access fees; •Contributions and memberships; •Average income
Water quality maintenance	Regional	Regional	Existing water quality; Safe alternatives	Expressed, revealed, imputed preferences;	•Stream order; Stream designation (swimmable / fishable); •Water volume; •Salinity; •Flow rates, •Residence time; •Types of use
Flood damage avoided	Local	Owners of downstream property within flood plain	Alternative natural or humanmade stormwater control	Potential property and income losses; Proportion of industrial, residential and business property; Insurance costs	•Volume of runoff controlled by stormwater devices and natural depressions / vegetation; •Value of property at risk; •Proportion of industrial, residential, business property

The level of some services and the value of some services may be adequately reflected by functional capacity indices. This is most likely the case when assessing certain passive use services where location and access are not very important. Total ecosystem area or total core area of wetland in a region is an obvious and potentially useful indicator for examining habitat scarcity for species that may be of special interest (e.g., eagles). However, from the point of view of a particular species, it is not merely quantity of habitat that matters, but how it is arranged on the landscape. It may be useful to further describe the expected level of function using the maximum or average nearest-neighbor distance between wetlands to reflect the scarcity of the habitat at some relevant scale. Consider stepping stones used to cross a river, if a person's stride is a maximum of 5 feet and the maximum nearest neighbor distance between stones in a river crossing is 4 feet or less, a person can cross the river. If removing just one stone increases the maximum distance between neighboring stones to 7 feet, the person can no longer cross the river. That one stone was a necessary condition for that person to be able to cross the river. In terms of habitat configuration, this nearest-neighbor distance measure can be used to identify a site (patch) that has particular value as a critical "stepping stone" [Keitt *et al.* (1998)].

Service Risk Sub-index

The economic value of a wetland depends on the *expected* flow of services it provides over time. Risks of wetland service disruptions can result from the vulnerability of the wetland site itself, but also from threats to important landscape features that affect its "economic productivity." Risk can arise from natural processes alone (biological, chemical, physical), from human actions in the wetland (illegal draining, vandalism), or from human activities (construction, road travel) outside the wetland (see Table 7.3). We envision accounting for risk by introducing another multiplier (the Service Risk Sub-index) into the framework for assessing wetland values (Figure 7.4).

The market value of assets, such as homes, factories, and farms, reflects whether or not they are located on earthquake faults or flood plains or where there is a high level of political instability or crime. The reason is that the risks of service flow disruptions are higher in those areas. Similarly, the expected services from a wetland with a low FCI score in a stable location could be much more highly valued than those expected from a pristine wetland in a rapidly deteriorating landscape. Urban sprawl, toxic runoff, noxious weed invasions, and water diversions do not threaten all wetlands in all locations in the same way. Knowing where these threats are the greatest and which sites are at risk provides critical information for assessing and comparing wetland values. Indices that reflect these risks provide an important basis for distinguishing between wetlands on the basis of their economic value.

Many sources can provide reliable information about what threats exist and which wetlands are at risk. Zoning plans, sewer extensions, and road construction, in combination with population projections, can suggest a great deal about the potential effects of population growth on wetland services and values. Intensive agriculture or feedlot operations and unsewered medium and low-density residential land use have also been shown to be strong predictors of pollutants in groundwater and surface water [Harper *et al*. (1992) and Hall *et al*. (1994)] and so plans for these types of land uses adjacent to or upstream of a wetland would also suggest increased risk. Factors that would tend to mitigate or exacerbate risk from development include: limits on allowable population densities, limits on land parcel size, and stormwater zoning regulations. These factors can be incorporated directly into Service Risk Sub-indices once they are measured from a GIS or other sources.

Table 7.3 Service Risk Sub-index Development

Service	Major threats to function*		Major threats to services		Potential components of service risk sub-index
	On-site	Off-site	On-site	Off-site	
Recreational fishing opportunities	Biological, physical and chemical threats to wetland	Biological, physical and chemical threats to landscape features	n/a	Change in access, property ownership, regulation and zoning; Change in land use	•Projected population growth rates (by locale / zipcode / watershed); •Expected development patterns in area (e.g. % impervious surface at buildout); •Disturbance level in adjacent area (mowing, boat traffic, agriculture, unsewered residential, channelization, invasive species); •Planned changes to nutrient loads, hydrologic regimes (e.g., waste water discharges, reservoirs, water diversions, groundwater drawdowns); •Invasive species spread rates.
Birding, hunting and gathering opportunities	Biological, physical and chemical threats to wetland	Delivery of excess sediments, nutrients or contaminants (beyond wetland filtering capacity)	Change in access / property ownership; Conversion to developed use (agricultural/ residential); excavation; draining	Change in access, property ownership, regulation and zoning; Change in land use	
Water quality maintenance	Change in water table depth; Alien invasive plant / animal species; Erosion; Sea level rise; Change in soil or plant characteristics	Activities that generate delivery of excess sediments, nutrients or contaminants (beyond wetland filtering capacity)	Conversion to developed use: excavation, draining; Logging; Fire	Projected land or water use that precludes service; Water diversion; Logging; Fire; Dredging	•Existing land use risk factors: agriculture, feed lots, septic fields; •Projected land use risk factors: water withdrawals; new feed lots, septic fields, logging , etc.; •Fire frequency; •Changes to hydrologic regime: flood control structures, water diversions, groundwater drawdowns.
Flood damage avoided	Change in water table depth; Alien invasive plant / animal species; Decrease in floodplain storage or roughness	Excess sediment delivery; Increased runoff upslope; Change in flood frequency; Change in floodplain slope; Change in channel	Conversion to developed use: excavation; draining	Change in land use	•Homes in flood plain modified, moved, destroyed; •Changes to hydrologic regime: flood control structures, water diversions, groundwater drawdowns.

* Since services depend on function, the service risk sub-index includes risk factors related to threats to functions as well as threats to service flows.

The explicit distribution of development in many regions can be examined through “scenario analysis” of growth under current zoning and regulatory conditions, or by using simpler geo-referenced indices of development. A regional planning tool called “build out” analysis is a type of scenario analysis that considers multiple sources of risk. It combines forecasts of population growth, landscape and land use data, and zoning and regulatory restrictions to evaluate effects of development on economic and environmental conditions. In Maryland, a statistical scenario analysis has been used specifically to identify which parcels are most likely to be developed under various regulatory and zoning conditions [Bockstael (1996) and Geoghegan *et al*. (1997)].

Scenario analysis can involve complex models or simple correlations. A variety of variables contribute to the level and location of development, but zoning and sewer extensions in particular have been shown to be strong predictors of future development. Many jurisdictions generate predictions of population growth by locale or zip code to facilitate this kind of analysis. Such projection tools can be used to show how changes in regulatory conditions will affect the size and location of wetlands at risk. The results of scenario analyses can also be used to show how the unavoidable degradation of certain wetlands will increase the expected value of the services provided by other wetlands that can be protected. Table 7.3 provides some risk factors that might be included in indices of wetland site risks for some wetland services.

The most commonly used tools for risk assessment are process-based or statistical models that explicitly link risk factors with demonstrated risk. However, where data or budgets are insufficient, risks can be assessed using qualitative methods such as cumulative scoring or ranking systems. In such a system, sources of previously identified disturbances or risks (e.g., upland unsewered medium density residential, planned highway, etc.) are summed to produce a total score for a site. Sites are then lumped into risk quartiles (very high, high, medium, low) based on the score distribution of a range of sites. Another option is a simple binary risk adjustment factor (0.5 if risk factor present, 1 if risk factor not present) which might be suitable to adjust the expected level of service to account for whether a wetland site is likely to be developed during any given year. The Expected Service Value during any given period of time, therefore, would be the product of multiplying the Service Risk Sub-index (reflecting such factors as the likelihood of the site being developed) by the Nominal Service Value (Figure 7.4).

Natural processes (or those controlled indirectly by human activities), such as apparent sea level rise, will be important in many regions. Known risk factors can be assessed through trend or scenario analysis (e.g., dispersion of invasive plants), but the potential for new natural risk factors may be difficult to characterize. In areas where human risk factors markedly outweigh natural risk factors, it may be sufficient to characterize anthropogenic risk factors only. The converse will also be true.

It is useful to note that the site characteristics that have adverse effects on the Service Risk Sub-index may tend to be the same ones that cause the Level of Service and Expected Service Value at a site to be high. The value of bird watching, for example, goes up with proximity to residential development or access, but so does site risk. Tradeoffs associated with these kinds of conflicting goals are better understood by developing site-based wetland service, value, and risk indices. It may even be possible to develop decision rules or response protocols that are based on simple indices that include service and risk and reflect how close a wetland may be to service capacity overload.

Using Service Preference Weights to Adjust Values

Using Service Preference Weights

In general, if one wetland provides more of all services than another does, it can be said to be more "valuable" regardless of the weights assigned to individual services. However, if one wetland provides more of some services and less of other services the relative value of the two wetlands could depend on the relative preferences that people attach to various services. The development of service preference weights can be based on direct survey data, secondary source information about the wetland service area, or some combination of both. The approach outlined in the following section, for example, employs a two-tier method that gives attention to the relative value of a given service provided by wetlands at different locations, and to the relative preferences that people have for different wetland services. Service Value Indices that reflect the relative value of a unit of service provided at a site are developed using secondary information about local and regional supply and demand conditions, numbers of users, participation rates, access costs, established watershed goals, and planning, zoning and permitting decisions. Public preference surveys based on paired service comparisons are then used to develop Service Preference Weights that rank individual services. The extent and frequency of the surveys that would be required to assign credible weights to various services, and how results might be affected by the geographic scale of the wetland service area and the geographic scale of the survey, still remain to be determined.

Measuring Service Preference Weights

There are several promising new areas of "non-market valuation" research, which could contribute significantly to the development of wetland benefit indicators. These differ from previous methods in that they are not designed to assign dollar values to natural resources or to environmental services. Instead, they use stated preferences for alternative sets of resource attributes or resource use options to rank and assign *relative* values to environmental assets. Applications of these techniques are beginning to appear in the literature under several different names including ranked preference analysis, contingent choice analysis, conjoint analysis, and choice modeling.

Interest in "ranked preference" studies is growing because they are relatively inexpensive and provide results that are easier to interpret and defend than those provided by "willingness to pay" surveys that focus on dollars. There is a growing consensus that the "cognitive difficulty " of attaching dollar values to non-marketed environmental attributes or services make "willingness to pay " surveys unusable in the case of wetlands. On the other hand, people seem to find it relatively easy to determine and express their preferences for one set of environmental attributes or services over another, and find it not too much more difficult to express by how much they prefer one set over another. The statistical properties and overall credibility of "ranked preference" studies seem to be exceptionally good compared to dollar-based stated preference surveys. Although new and still developmental, it promises to be an extremely useful focus for non-market valuation research.

One of the attributes that can be included in attribute sets that respondents are asked to rank can be a hypothetical access fee or tax assessment expressed in dollars. Surveys can be designed to include a range of dollar costs associated with similar sets of environmental attributes. Differences in the rankings that result from differences in the "dollar cost" attribute provide a statistical basis for estimating "willingness to pay". Results so far suggest that when respondents encounter dollar costs as one of many attributes associated with a set of environmental services they find it more natural than when they are asked directly what they would be "willing to pay" for a specific non-market service. Ironically, techniques developed to overcome the problems associated with attaching dollar values to environmental goods, in time, may provide the best opportunity for getting the job done.

The Expected Service Value Index that would be developed by assessing Service Capacity, Value of Service and Risk of Service Sub-indices would reflect the relative value of a particular service provided at a particular wetland site at a period in time. It does not provide a basis for assessing the relative value of that wetland service with respect to other wetland services provided at that site or other sites. The final step in assessing wetland value, therefore, involves assigning weights to wetland services based on people's preferences. In our indicator system these are called Service Preference Weights. For each service category, the Adjusted Value of a wetland is calculated by weighting the Expected Service Value Indices, calculated for each service, to account for how people rank the various services the wetland provides. In the proposed framework, Service Preference Weights are used to show the relative preferences that people have for individual services in a particular geographic region, not at each wetland site.

A variety of methods exist for estimating and ranking preferences [Nijkamp *et al*. (1990)]. In Box 7.7, we illustrate the use of pairwise preference comparisons. In the illustration, a five-step process is used to evaluate how respondents rank their absolute preference for one type of service over another. An alternative method could involve asking respondents to express the intensity of their preferences for one service over another by ranking pairs of services on a 1-5 scale (equal importance to absolute importance). Weights from individuals can be aggregated for purposes of statistical analysis, or sets of weights representing different points of view can be compared to examine the distributional or equity affects. The differences in Service Preference Weights assigned by sample respondents selected from populations at different geographic scales would be particularly instructive for examining equity issues.

Box 7.7 Using paired comparisons to assign preference weight to wetland services

Paired comparisons is one survey technique used to derive ranked preferences from a group. To simplify the cognitive demands of such a ranking, participants are asked to compare pairs of options and to say which one is absolutely preferred or to rank on numbered scale the degree of preference. The steps to developing preferences are:

Step 1: List wetland services

This can be a subset of the services listed in Box 7.2 that are important in a particular watershed or are the focus of important tradeoffs.

Step 2: Develop paired comparison matrix

The purpose here is to identify all possible pairs of wetland services in a matrix as illustrated for a limited set of wetland services in Table 7.4A.

Step 3 Select a representative sample of respondents

The selection and stratification of samples would depend on the nature of the comparisons. It might be useful to sample from the public in general or from selected stakeholder groups. Since the geographic ranges of wetland services differ widely it might be useful to test for preference differences in sampled populations selected at different geographic scales.

Step 4: Develop paired preference rankings

Have respondents select their preferred wetland service from each pair presented in Table 7.4A. Service Preferences are then aggregated to develop Service Preference Rankings as illustrated in Table 7.4C.

If there are ties or close rankings it may be useful to provide respondents with additional information and conduct a second iteration of paired preference ranking for selected pairs of services.

Step 5: Use statistical methods to develop rank orderings

Various statistical methods can be applied to the results of paired preference comparison surveys to arrive at relative service weights or rank orderings of services [USDA (1997) and David (1988)].

Overall Ecosystem Value Index

The final step in developing an overall ecosystem value is to sum the Adjusted Service Value Index calculated for each time interval over the time period of interest (Figure 7.4). This step allows the aggregate effect of changes in risk and value through time to be evaluated. At present, we are assuming that service preference weights remain constant while value and risk sub-indices can reflect spatial and temporal changes. It may be useful to retest service preference weights whenever significant changes occur in supply and demand conditions.

Finding information

In recent years a wide range of landscape, land use, and demographic information has become available in regional databases, often using GIS applications. The site and landscape factors that influence Level of Service, Value, and Risk are different for different services. However, it should be possible to develop indicators for each of them by tapping into the same general pool of information. Table 7.4 provides a preliminary regional checklist that might be used to help gather initial information for a particular region.

Table 7.4 Illustration of Paired Comparison Approach to Service Preference Ranking A. Pairs of Selected Wetland Service Presented to Respondents

	Waterfowl hunting	**Waterfowl viewing**	**Local freshwater fishing**	**Downstream water quality**	**Reduced flood damage**	**Biodiversity protection**
Waterfowl hunting						
Waterfowl viewing						
Local freshwater fishing						
Downstream water quality						
Reduced flood damage						
Biodiversity protection						

B. List of Paired Service Comparisons and Preferred Choice Based on Local, Regional, National Surveys

Comparison	**Preferred Choice**		
	Local	**Regional**	**National**
Waterfowl hunting/waterfowl viewing	Hunting	Hunting	Viewing
Waterfowl hunting/local freshwater fishing	Fishing	Fishing	Hunting
Waterfowl hunting/downstream water quality	Hunting	Water Quality	Water Quality
Waterfowl hunting/reduced flood damage	Hunting	Reduced Damage	Reduced Damage
Waterfowl hunting/biodiversity protection	Hunting	Hunting	Biodiversity
Waterfowl viewing/local freshwater fishing	Fishing	Fishing	Viewing
Waterfowl viewing/downstream water quality	Viewing	Water Quality	Water Quality
Waterfowl viewing/reduced flood damage	Reduced damage	Reduced damage	Reduced damage
Waterfowl viewing/biodiversity protection	Viewing	Viewing	Biodiversity
Local freshwater fishing/downstream water quality	Fishing	Water Quality	Water Quality
Local freshwater fishing/reduced flood damage	Fishing	Fishing	Reduced Damage
Local freshwater fishing/biodiversity protection	Fishing	Fishing	Biodiversity
Downstream water quality/reduced flood damage	Reduced damage	Reduced damage	Water quality
Downstream water quality/biodiversity protection	Water quality	Water quality	Water quality
Reduced flood damage/biodiversity protection	Reduced damage	Reduced damage	Biodiversity

C. Illustrative Service Preference Ranking Based on Paired Comparison Surveys

Wetland service	**Local**		**Regional**		**National**	
Waterfowl hunting	IIII	4	III	2	I	1
Waterfowl viewing	II	2	II	1	II	2
Local freshwater fishing	~~IIII~~ IIII	5	~~IIII~~ I	5		
Downstream water quality	I	1	III	3	~~IIII~~ II	5
Reduced flood damage	III	3	~~IIII~~	4	IIII	3
Biodiversity protection					~~IIII~~	4

Table 7.5 Example Checklist of Regional Information to Develop Wetland Benefit Indicators

A. Maps	Coarse	Fine	Coarse	Fine	Isolated	Format
Flood zones						
Soil types						
Hydrologic features						
Topography						
Geology						
Vegetation						
Wetlands						
Land use						
Infrastructure						
Natural resource						
Fish/wildlife inventories						
Natural hazards						
Pollution inventory (RCRA/Superfund)						
Endangered species						
Critical habitat						
Natural areas						
Historical/archaeological sites						
Motor traffic						
Population						
Income						
B. <u>Plans and forecasts</u>						
Coastal zone						
Shoreline and shore land						
Wild and scenic rivers						
Floodplains and greenways						
Environmental corridors						
Water quality						
Critical area						
Local land use						
Watershed restoration						
Growth projections						
C. Parcel/plot information						
Parcel size/ownership						
Use of parcel						
Property value						
Taxes						
Zoning						
Easements/restrictions						
Utilities available						
D. Regional survey results						
Regional surveys of outdoor recreation: participation rates, willingness to pay, etc						
Regional surveys of preferences for environmental amenities.						

PART 4

CHAPTER 8:

by **Paulo A.L.D. NUNES, Jeroen C.J.M. VAN DEN BERGH, Peter NIJKAMP**[45]

INTEGRATION OF ECONOMIC AND ECOLOGICAL INDICATORS OF BIODIVERSITY[46]

Introduction: context and scope of the study

In the spirit of the growing environmental awareness in the past decades, we witness nowadays an increasing interest in biodiversity, both locally and world-wide. Biodiversity requires our attention for two reasons. First, it provides a wide range of benefits to mankind and human activities. Second, many human activities have contributed to unprecedented rates of biodiversity loss, which threaten the stability and continuity of ecosystems as well as their provision of goods and services to mankind. Consequently, in recent years much attention has been directed towards the analysis and valuation of (the loss of) biodiversity. The valuation of biodiversity can be approached from an ecological, economic, or combined perspective. This contribution presents an overview of economic and ecological indicators of biodiversity, identifies the underlying valuation approaches, discusses basic concepts and theories, and reviews the respective applications.

The organisation of this study is as follows. Section 2 starts with a concise review of the most important frameworks that focus on the conceptualisation of biodiversity and its role in ecological processes. It continues with the identification and discussion of alternative ecological valuation approaches of biodiversity, including biotic-richness and ecosystem health approaches. Section 3 addresses the economic concepts and methods relevant to the valuation of biodiversity. It starts with a discussion of the reasons why economists are interested in pursuing monetary value assessment of biodiversity and why some scientists disagree with this approach. A survey of empirical valuation studies is presented. The distinction between 'resource valuation' and 'biodiversity valuation' is discussed. Finally, respective valuation results and measurement methods are discussed. Section 4 identifies a combined economic-ecological valuation perspective. According to this perspective, biodiversity is analysed, modelled and valued from an integrated point of view, combining natural and social-economic aspects, and thus can enable the construction of mutually consistent indicators of biodiversity. Finally, Section 5 presents conclusions and outlines some recommendations for use of valuation methods and findings in environmental policy.

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[46] An extended version is available at the *Tinbergen Institute* discussion paper series (code # 00-100/3).

Ecological foundations for biodiversity analysis and valuation

Conceptualisation of biodiversity and its role in ecological processes

The analysis of biodiversity is rooted in the domain of both natural and social sciences and its modelling implies a review of knowledge about the relationship between biodiversity, the dynamics of ecosystems, and the level of human economic activities. One reason why biodiversity modelling has been so difficult relates to the complex and partly unobservable nature of the biodiversity-ecosystem relationships involved, such as biotic-abiotic interactions[47], food webs nutrient flows, and species interrelations. Independently of the complex nature of the biodiversity-ecosystem relationships, an important aspect is the recognition that the variability of the biological resources influences the functioning and structure of ecosystems. In the literature one can find three main approaches to modelling. These are reviewed in the present section.

Holling (1987 and 1992) proposed a model to describe and explain the dynamics of a terrestrial ecosystem in terms of a structure that is characterised by a sequential interaction between four basic functions or phases - the "4-box model". The functions are: (1) exploitation; (2) conservation; (3) release; and (4) reorganisation. Within this model, ecosystems develop from the exploitation phase during which systems capture easily accessible resources, to the conservation phase during which systems build and store increasingly complex structures, and then evolve to the release phase during which systems free some of the mature structures. The released structure is then available for reorganisation and uptake in the exploitation phase. The exploitation function refers to the ecosystem's processes that are responsible for "colonising disturbed sites". The conservation function refers to the ecosystem's processes that are responsible for "resource accumulation that builds and stores energy and material". The release or "creative destruction" function refers to an abrupt change in the ecosystem caused by external disturbance, releasing energy and material that have been accumulated during the conservation phase. Examples of the release phase are fire, storms, and pests (Costanza *et al.* 1995). Finally, the reorganisation function refers to the ecosystem's processes that are responsible for mobilising released energy and materials and making them available for the next exploitation phase.

Based on a categorisation of the ecosystem's functions initially made by Odum (1971), de Groot (1994) describes the relationship between biodiversity and ecosystem. In general terms, de Groot characterises the ecosystem structure in terms of four categories of biodiversity functions: (1) life support functions, (2) carrier functions, (3) production functions, and (4) information functions. Biodiversity is seen to have a life support function, i.e., a regulation of essential ecological processes. The life support functions refer to the group of the biodiversity service flows that contribute to the maintenance of a healthy environment, by providing clean air, water and soil, by providing flood control, and by providing carbon storage and waste absorption. Most of the life support functions are often not easy to define or identify(e.g. provision of carbon storage). The carrier functions refer to the provision of space for human activities such as habitation, agriculture, and recreational activities. The production functions refer to the provision of natural resources, ranging from raw materials for industrial use to water and energy resources. The information functions refer to the maintenance of mental health, providing opportunities for reflection, spiritual enrichment and aesthetic experience.

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47 By definition, the abiotic environment counts for the *no* living components of the ecosystem. Therefore, abiotic interactions count for interactions between ecosystem hydrology, geomorphology, and physical characteristics of the landscape (e.g., variables on soil and vegetation structures, temperature, sediments, water, soil pH and salinity). In contrast, biotic interactions embody the interactions between the living components of the ecosystem, including the dynamics of transformation of matter or energy between the fauna and flora communities.

More recently, Norberg (1999) proposed an alternative approach to classify ecosystem functions and services. Norberg selected groups of ecosystem services to which common ecological concepts apply: (1) are the goods and services internal to the ecosystem or shared with other systems?, or, (2) are the goods and services of biotic or abiotic origin?, and (3) at which level of ecological hierarchy are goods and services maintained? Bearing in mind such selection criteria, ecosystem functions and services are classified in three categories: (1) maintenance of populations; (2) regulation of material and energy flows; and (3) organisation of biological units through selective processes. These categories represent three major fields in ecology that have well-established theoretical foundations and refer to population/community ecology research, ecosystem research, and biological organisation, respectively (Levin *et al.* 1997, Levin 1998).

The first category corresponds to the group of ecosystem services that are “(…) associated with certain species or a group of similar species (…)” (Norberg, page 185). Examples of such services include valuable foods and goods such as fish, timber, pharmaceuticals chemicals, and flowers. The second category consists of processes that regulate exogenous chemical or physical cycles, i.e. processes that drive material and energy flows in ecosystems. The biota takes a significant part in most global cycles of chemical compounds such as water, CO_2, and nitrogen. Finally, the third category of ecosystem services is related to the organisation of biotic entities. Organisation is virtually present at all scales: organisation of genes through natural selection, spatial distribution of a population through dispersal and competitive exclusion, or the development of food webs and ecosystems through invasion and extinction processes.

Ecological indicators of biodiversity: the biotic-richness approach

From a biotic approach, the definition of biodiversity ‘value’ indicates the range magnitude of biological products and services flows provided by nature. Traditionally, the measurement of biological diversity has been undertaken with the use of genetic, species, and ecosystem richness or variety indices.

Measurement of genetic diversity

The analysis, conceptualisation and measurement of genetic differences can be done in terms of (1) allelic frequencies, (2) phenotypic traits and (3) DNA sequences. The same gene can exist in different frequencies or variants. These variants are called alleles. Thus, allelic diversity measures the different gene composition variants of individuals. In general, the more alleles, the more diverse their frequencies and the greater the genetic diversity. Average expected heterozygosity, the probability that two alleles sampled at random are different, is commonly used as an overall measure. A number of different indices can be applied to the measurements to assess average expected heterozygosity (Antonovic 1990). A commonly used index to heterozygosity refers to the proportion of the total expected heterozygosity due to allele-frequency differences among populations (G_{ST}). Values on G_{ST} range from zero (no variability among populations) to near one (fixed allell-frequency differences). From Table 8.1, it is clear that the mean value of G_{ST} is largest for some molluscs, amphibians, reptiles and mammals; i.e. populations of most species in these groups have more subdivided populations, and larger amounts of population sub-division harbour more genetic diversity than small-subdivided populations. On the contrary, birds and insects tend to show less variability among populations, because of the high gene flow between them (Avise and Aquadro 1982).

Phenotic diversity is a measure based on an individual's phenotype traits[48], checking whether individuals share the same characteristics. This valuation method is focused on the measurement of the variance of certain traits and, in general, involves readily measurable morphological and physiological characteristics of the individual. However, individual genetic information is often difficult to assess, and comparisons are difficult when the individuals or populations are measured in terms of different qualitative traits. To cope with this, scientists may now use DNA sequence variation to measure genetic variety. The DNA sequence information is obtained by the use of the polymerase chain reaction[49]. For this reason, only a small amount of material, ultimately one single cell, is required to obtain the DNA sequence data. Closely related species may share up to 95% of their DNA sequences, thus implying little diversity in the overall genetic information.

Table 8.1 G_{ST} in major taxanomic groups

	G_{ST} ± Standard Error	Number of species
Vertebrates		
Mammals	0.242 ± 0.030	57
Birds	0.076 ± 0.020	16
Reptiles	0.258 ± 0.050	22
Amphibians	0.315 ± 0.040	33
Fishes	0.135 ± 0.021	79
Invertebrates		
Insects	0.097 ± 0.015	46
Crustaceans	0.169 ± 0.061	19
Molluscs	0.263 ± 0.036	44
Others	0.060 ± 0.021	5

Source: Ward *et al.* (1992).

Measurement of species diversity

The measurement of species diversity, in its ideal form, consists of a complete catalogue of the distribution and abundance of all species in the area under consideration. However, this measurement is usually not possible unless the area is small. Therefore, in practice the measurement of species diversity is often based on samples. The main measures of species diversity are the α, β and γ species diversity, as originally proposed by Whittaker (1960 and 1972). α-diversity refers to the number of species using only their presence (and not abundance) in a given area. It therefore measures the species richness of a given sample plot. The ranking of natural areas in terms of species diversity is, according to this measure, done in terms of the overall number of species surveyed in that same

48 Whereas heterozygosity refers either to the genetic variability within species and among populations, phenotype refers to the genetic variability among individuals.

49 Technique used in molecular biology to analyse DNA that is characterised by the use of several cycles of denaturing of the DNA molecule through heat (for additional information see Hoelzel and Green 1992).

area, where areas containing a higher number of species, and thus described by larger α, would have a higher ranking position than the an area with a fractionally smaller number of species (Huston 1994).

Another measure of species richness is γ-diversity. It is often used to assess the overall diversity within a large region and its comprehension has direct implications when dealing with biodiversity at the landscape level (Noss 1983, Franklin 1993). National species lists, usually the only information available, can be treated as lower bounds on gamma diversity. Colombia and Kenya, for example, are the homes to over 1,000 species of birds, while the UK and the forests of eastern North America are homes to only about 200. A coral reef of northern Australia may be a home to 500 species, while the rocky shoreline of Japan may be a home to only 100 species (UNEP 1995).

Finally, β-diversity measures the turnover of species between local areas, i.e. the rate of change in species composition among discrete sites or habitat units (Cody 1986 and 1993). As such it cannot be expressed in the number of species: it is represented in terms of an index and it is interpreted as a species turnover rate. β-diversity is generally used to estimate average changes in species in response to site or habitat heterogeneity.

Species richness measurement is useful, but it may depict a biased diversity assessment. First, scientists face great uncertainties about the total number of species. Therefore, species richness can only be measured for some species (the species studied), but not for all the species in the area. As a matter of fact, only in a very few places on the planet are there rough estimates for the total number of species. Second, the size of the area is often arbitrary. Species diversity is associated with habitat scale in a complex way. Thus, one needs to be cautious when comparing the species diversity of areas that differ greatly in size. In addition, species diversity is a result of complex genealogical relationships that are not measured here. Alternative species diversity measures supplement species richness with measures of the degree of genealogical difference. Such diversity measures include the weighting of close-to-root species, higher-taxon richness, spanning tree length and taxomonic dispersion (Williams *et al.* 1991). However, for the time being, practical difficulties regarding the implementation of such measures force reliance on the simpler indicators of species richness.

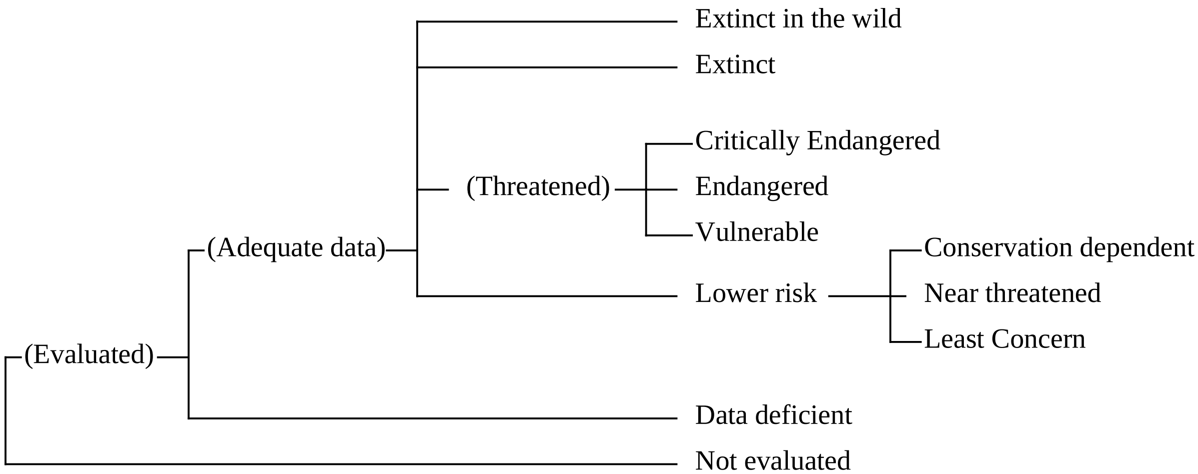
Measurement of ecosystem diversity

The measurement of biodiversity at the organisational level of ecosystem diversity encompasses multi-complex relationships, both at the intra- and supra-species level, that play a crucial role in defining the overall distribution and abundance of species. For this reason, there are a number of factors that make the assessment of ecosystem diversity less clearly defined. Actually, at the ecosystem level many different units of diversity are involved, ranging from the patterns of habitats to the age structure of populations, including the patterns of communities on the landscape and patch dynamics. At these levels it is not clear where to draw boundaries delineating units of biodiversity. When a wetland, for example, is disturbed or degraded, we need to look at the impacts of the disturbance on the larger level of the landscape. Emphasis on a system-wide approach also serves to remind us that the ecological value of an ecosystem may not be equivalent to the aggregate value of that same system’s components. In other words, the system is more than just the aggregation of its individual parts; it possesses a primary value (Gren *et al.* 1994; Turner, Perrings and Folke, 1997). Furthermore, the conservation of biodiversity at the ecosystem level does not only underlie the preservation of species, but also contributes to the safeguard of the ecosystem functions and services. Thus, the full range of biodiversity values depends on the processes that support the functioning of larger-scale ecological systems. Given such unambiguous boundaries, there are different measurement approaches. These include bio-geographical realms or provinces, based on the distribution of species, and eco-regions or eco-zones, based on physical attributes such as soil and climate (UNEP 1995).

Operationalisation of the biotic-richness approach: examples

Ecologists are often asked to contribute with their expertise and help policy makers define conservation priorities. A considerable part of the ecologist's help is, in some way, related to Usher's ecological approach to environmental protection (Usher 1989). According to Usher's conservation assessment, decisions are characterised by three steps. First, attributes are identified and are used to reflect the conservation interest of the species or site. Second, criteria are developed for the expression of the attributes in a form that allows evaluation. Finally, values are attached to particular levels of criteria.

Figure 8.1 Structure of species categories



Source: http://www.iucn.org/themes/ssc/redlists/categor.htm

One important instrument for assessing species variety is the Red Data Book (e.g. IUCN 1993). Red Data Book identifies threats or causes of decline of different species around the world (Fitter and Fitter 1987, Mace and Lande 1991, Mace *et al.* 1992, IUCN 1993, Mace and Stuart 1994). In short, it comprises an evaluation technique characterised by the selection of a species list (the attribute) to assess the species richness (the criterion). Red Data Books classify species in one of eight different categories: extinct, extinct in wild, critically endangered, endangered, vulnerable, lower risk, data deficient and not evaluated – see Figure 8.1. The goal is to provide an easily and widely understood method to classify species in categories related to their threat of extinction under current circumstances[50] in order to (1) provide information on which to base conservation programs; (2) assist the drafting of legislation, and (3) to convey information comprehensible to a non-specialist. For this reason, Red Data Books are frequently used by numerous governmental and non-governmental organisations for policy guidance and the establishment of conservation priorities.[51]

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[50] For example, "critically endangered" category refers to the species that is facing an extremely high risk of extinction in the wild in the immediate future. For further details see Nunes *et al.* (2000).

[51] Most obviously these were required for selecting and designating the European Union 8,819 nature reserves under the Natura 2000 network, 6,977 within the Habitat Directive (92/43/CEE), and 1,842 within the Birds Directive (79/409/CEE).

Several problems however exist, rendering category assessment quite difficult to apply in policy making. While the classification of a species in a category relies on an objective evaluation, the actual definitions of these categories rely on a subjective view. In practice, the large number of available criteria (e.g. α, β and γ criterion) used for evaluation in some way already reflect the difficulties that exist in determining value. Furthermore, the information provided by the list of species under the category labelled as "threat" is not necessarily sufficient to determine priorities for conservation action Indeed, if Red Data Books are elected as the ecological method to establish biodiversity priorities, it is likely that choices would be made based on the likelihood of species extinction rather than on the (differences) level of among the gene flow between the survival species and the species with higher probability of extinction. Finally, given the scientific understanding of population and ecosystems, it is possible to develop alternative and more complete indicators involving the use of numerous other criteria concerning conservation action.

An early example of a multi-criteria rating is the method proposed by Randwell (1969). The method was used to evaluate coastal habitats and combines the use of eight criteria into a single score, the Comparative Biological Value Index (CBVI). Each of these criteria are rated according to the scale shown in Table 8.2 and the final score is obtained by summing up the scores for all the nine criteria:

$$CBVI = Ph + O + D + G + S + P + E + C$$

The maximum potential value is 28 and the minimum value is 7. The higher the CBVI value the greater the requirement for site protection. Since Randwell, the use of indices constitutes a popular practice in ecological valuation and management (see Spellerberg 1992 for an extensive review of CBVI assessment of landscape and urban habitats). However, this valuation approach relies on input criteria that require some subjective valuation. As such they may not be helpful to the decision-making process as originally intended.

Recently, computer-based systems have been used to develop a general, integrated framework where one can simulate natural/management changes and assess the respective conservation evaluations. One specific application of this valuation approach is to study species population dynamics and estimate a minimum viable population, defined as the smallest population which has an acceptable probability of persisting over a given time period (Soulé 1987).[52] This in turn allows for the calculation of the minimum dynamic area, i.e. the geographical area of suitable habitat required to support the minimum viable population. Recent published studies on the leadbeaster's possum (Lindenmayer *et al.* 1993), the eastern barred bandicoot (Lacy and Clark 1990), the Scottish wild boar, and the giant panda (Zhou and Pan 1997) confirm the use of the population viability analysis in policy decision circles and help conservation decisions. Another example of the application of a computer system to ecological evaluation is the System for Evaluating Rivers for Conservation (SERCON) recently undertaken by the Scottish Natural Heritage, the nature conservation agency in Scotland (Boon *et al.* 1997). The overall objective was to predict the impact of different development scenarios on river ecological conservation value as well as to provide a simple way of communicating such results to planners, developers and policy-makers.

Ecological valuation based on computer modelling presents important advantages. First, it encourages greater rigour in data evaluation since it permits the introduction of elements of subjectivity within a transparent and consistent framework. Second, it allows for a direct comparison of alternative conservation policies, independently of the number of criteria involved and respective attributes. For example, it permits the comparison of a conservation policy involving a criterion with

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[52] Several general models have been developed for this task, e.g. VORTEX (Lacy 1993) and METAPOP (Akcakaya 1994), and these can be used to consider the complex interactions between demographic, environmental and genetic influences on a population.

ten attributes that could score a maximum of 50, with another strategy involving a criterion with two attributes that could only score a maximum of 10. Finally, it permits an evaluation even when some of the attribute data are missing, which is common in practice.

Table 8.2 Rating of the criteria used by Randwell for evaluating coastal habitats

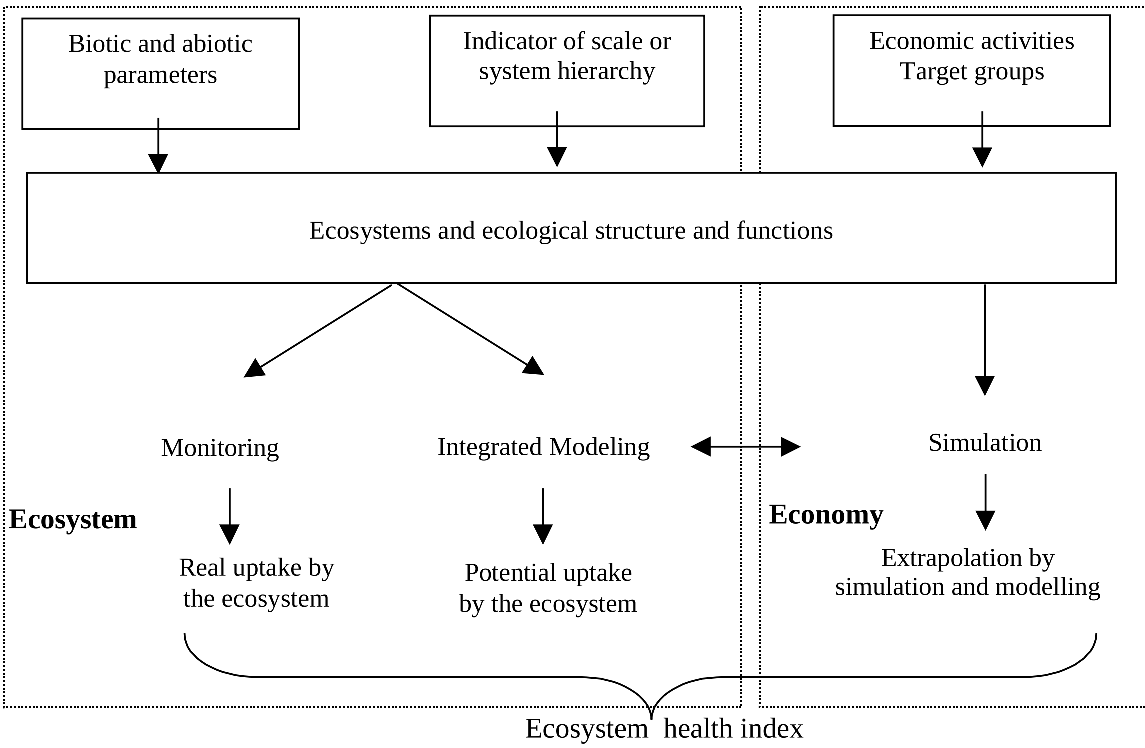
Criteria	Type		Rating
Physicochemical features (Ph)	High speciality		3
	Some special features		2
	Type example		1
Optimum populations (O)	Best populations of one or more local species		4
	Large populations of local species		3
	Large populations of common species and small populations of local species		2
Diversity (D)	Representative populations		1
	Outstanding populations		3
	High diversity		2
	Species ranges small		1
Geographic units (G)	Many species at limit		3
	Some species at limit		2
	Dew species or no species at limit		1
Size (S)	Mud-flats (ha)	Cliffs (km)	
	> 4000	> 80	5
	1600 – 3999	40-79	4
	800 – 1599	24-39	3
	400 – 799	8-23	2
	< 400	<8	1
Purity (P)	Little disturbance		3
	Moderate disturbance		2
	Much ground disturbed or polluted		1
Education and research use (E)	Much used		3
	Some use		2
	Potential use		1
Combinations of localities (C)	Adjacent to another habitat of likely national value		4
	Adjacent to another habitat of likely regional value		3
	Adjacent to another coast habitat site not spoilt by development		1
	Surrounded by developed coastline		0

Source: Randwell (1969).

Ecological indicators of biodiversity: the ecosystem health approach

Ecological valuation methods are not only aimed at assessing diversity and rarity of species, but also the complex interactions between the biotic and abiotic environments, based on the assumption that the variety of abiotic conditions is equally important as variety of species. For instance, abiotic diversity (e.g. physical characteristics of the landscape such as soil pH and salinity) is expected to be linked to the prevalence of endemic species and thus to biotic diversity and rarity in a natural way (Bertollo 1998). Therefore, from an ecosystem perspective, the definition of biodiversity value is intrinsically related to ecosystem performance and integrity. Furthermore, the term "value" indicates how well an ecosystem is functioning when compared to its own potential and how important this is for the functioning of other ecosystems and, ultimately, for the functioning of the global ecosystem (Sijtsma *et al.* 1998).[53]

Figure 8.2 Definition of ecosystem health index



Ecosystem health

Most of the biotic and abiotic parameters or indicators, such as indexes describing the soil, flora, and fauna, have emerged from the ecological literature (Odum 1971). On the contrary, ecosystem health is an overall indicator of the ecosystem functioning (or ecosystem integrity), taking into account both ecological and human processes. In this context, an ecological system is said to be healthy if it is stable and sustainable, i.e. if it is active and maintains its organisation and vigour over time and is resilient to stress. Ecosystem health can be defined as "(…) a measure of the overall performance of a complex system that is built up from the behaviour of its parts (…)" (Costanza 1992, pp. 241). Before we can measure the health of an ecosystem it is, however, necessary to identify biotic and abiotic parameters or indicators, target human economic activities, and choose the scale or hierarchy of analysis – see Figure 8.2.

[53] From an ecological perspective, the definition of 'value' indicates how well an ecosystem is functioning compared to its own potential (Sijtsma *et al.* 1998).

The choice of the scale relates to important decisions concerning boundary and temporal perspective of analysis (Norton and Ulanowicz 1992). It is common that boundaries are drawn in accordance with the ecosystem's land features or its geography (e.g. wetlands ecosystem). Finally, the measurement of ecosystem health also requires the identification of the human actions that influence the ecological structure and processes. The underlining idea is that human economic activity or target groups, e.g. consumers, industry, influence their environment. As a result, information on the impact of economic activity on the ecological structure and processes, in general, and on economic indicators, in particular, needs to be taken into account when assessing ecosystem health. It is also clear that both ecological and human dimensions are dynamic. Consequently, biotic, abiotic, and economic indicators must be sufficiently flexible to accommodate changes. Even if we have clearly defined the spatial and temporal boundaries of analysis, important questions regarding the structure of the ecosystem, originally described as a whole, could remain. Therefore, it would be necessary to represent boundaries on a smaller scale. Each part is characterised by its own set of indicators and is assessed individually (Costanza *et al.* 1992). This set of indicators can differ substantially from part to part, i.e., the set of from indicators needed to assess the integrity of a forest ecosystem can differ from the set of from indicators needed to assess the integrity of a desert grasslands ecosystem (for case studies see Rapport *et al.* 1998).

After having decided upon the biotic and abiotic indicators, scale and hierarchy of analysis and the target economic groups, the scientist is in a good position to proceed with the measurement of the overall ecosystem performance, i.e. ecosystem health. One possible approach is to use the available data provided by monitoring activities directly. This data provides information on the real uptake by the ecosystem and allows the overall ecosystem health to be measured. Alternatively, one can integrate the relevant concepts and develop an analytical framework. The combination of the available data and such a model formulation allows for a further step in the assessment of ecosystem health. At this stage, the scientist is not only to use the available data to measure ecosystem health, but also to explore the dynamics of integrated modelling so as to estimate the potential uptake by the ecosystem. Finally, the scientist is able to simulate different conservation scenarios by manipulating the observed characteristics, or 'controlling variables' - e.g., introduction of a new set of water nutrient loads. For each conservation scenario the scientist is able to compute the associated ecosystem health index and thus provide crucial information in order to rank the alternative management policy scenarios.

Examples of ecosystem health indicators

The construction of ecosystem health indexes allows policy makers to predict ecosystem response to various specific management alternatives and natural changes. In practice, this formulation leads to the following form for an overall system health index (HI):

$$HI = V*O*R$$

where *V* designates system 'Vigour' and represents a cardinal measure of the ecological system activity, metabolism, or primary productivity; *O* designates system 'Organisation' and represents a 0-1 index of the relative degree of the ecological system's organisation, including its diversity and the degree of connectivity, or interaction, between system's biotic and abiotic components; and *R* designates system 'Resilience' and represents a 0-1 index of the relative degree of the ecological system's resilience – see the components in Table 8.3. The overall ecological system health is given by its activity weight through indices for the relative system's organisation and resilience. To operationalise the vigour, organisation and resilience components, the health index will require the application of different measurement solutions to current data, involving the use of expertise from both economics and ecology.

Table 8.3 Indices of vigour, organisation and resilience

Components of health	Related concepts	Related measures	Field of origin	Measurement via
Vigour	Function Productivity Throughput	Gross primary production Gross national product Ecosystem metabolism	Ecology Economics Biology	Monitoring
Organisation	Structure Biodiversity	Diversity index Mutual information predictability	Ecology Ecology	Network analysis
Resilience		Scope for growth	Ecology	Simulation modelling

Source: Costanza (1922).

Ulanowicz's ascendancy index

One important example of ecosystem-health index, Ulanowicz's ascendancy index, allows for an integrated, quantitative, hierarchical measurement of ecosystem health (Ulanowicz 1992). In simple terms, the ascendancy index reflects any degradation of the system. However, such an indicator requires data on all transfers occurring in the ecosystem under consideration. The collection of this data is usually an expensive task; therefore, fully quantified networks of ecosystems remain scarce (Costanza 1992). In addition, it is important to remember the distinction between a scientific-orientated methodology and a policy-orientated approach. In practice one can find several valuation methods that combine a set of ecosystem integrity indicators into one common denominator that has a socio-political appeal and thus a meaning to policy makers. The following sub-sections review the operationalisation of some of the ecological valuation methods, most of them with an application to the Dutch regional and national policy agenda (see Ruijgrok 1999).[54]

The ecosystem classification method

Brink and Hosper (1989) developed a General Method for the Description and Evaluation of Ecosystems, known by the Dutch acronym AMOEBE.[55] The method was originally used to assess the quality of aquatic ecosystems by comparing the presence of selected species with their presence in a benchmark situation of 1930. The selected species, which Brink and Hosper called 'target variables', were chosen on the basis of: (1) their representativeness (i.e. do they represent a healthy aquatic ecosystem); (2) their flexibility (i.e. can they be influenced by human interventions), and, finally (3) their measurability and data availability (i.e. are they easy to measure and/or are there data-bases available). The AMOEBE does not indicate whether one ecosystem is more valuable than another, since it describes the value of the ecosystem in terms of the selected 'target variables', and the respective deviation measurement from its own potential, i.e. when measured at an optimal management situation. In other words, it is able to evaluate an intertemporal value path of an ecosystem, but comparisons across different ecosystems are not feasible. For such a reason, this valuation method is not often selected for policy formulation guidance.

[54] The characteristics of the Netherlands, a country with the highest population density in the European Union and home of important human economic activity, suggest the use of this ecological valuation method more than any other country.

[55] Algemene Methode voor OEcosysteembeschrijving en Beooordeling.

Nature measurement method

In 1995, the Dutch Centre for Agriculture and Environment in Utrecht developed the nature measurement method to assess the natural values of agricultural areas. The natural values were measured in terms of species abundance and their deviation from their own diversity potential. (Buys 1995). A similar formula was developed by the Foundation for Spatial Economics of the University of Groningen to assess the costs and benefits of the National Ecological Network (Sijtsma and Strijker 1995). The costs were valued in monetary terms and the benefits mostly in ecological terms. The National Ecological Network used the identification of 'nature target types' (i.e. pre-defined types of nature such as the European CORINE network) that are assumed to provide the habitat mosaic for 'target species'. For this purpose, digital thematic maps are generated as geographic information systems land habitats types. Target species are, in turn, classified and selected on the basis of national and international rarity. Zurlini, Amadio and Rossi have recently applied this valuation method to create a map of Italian nature (see Zurlini *et al.* 2000).

Ecological effect measurement method

In 1996, the Centre for Environmental Studies in Leiden developed an ecological effect measurement method to value the effects of housing development projects on nature and landscape. This method involves the description of the reference situation and human intervention measures, followed by the determination of the effects on nature and, finally, proceeds to the aggregation of effects (Cuperus and Canters 1995). The main ecosystem biotic and abiotic characteristics were used for valuation, including spatial diversity (e.g. variables on soil and vegetation structures), abiotic functioning (e.g. variables on temperature, sediments, water and soil), fauna, and flora communities, as well as their relationship with their surroundings (e.g. variables on hydrologic, geomorphologic and biomass relations). Since this valuation method takes into account both biotic and abiotic diversity assessed by means of the deviation from a reference situation, it can easily be transferred to the policy arena and used, for example, to determine compensation measures in the case of damage to existing natural areas.

Ecological capital index

More recently, the Dutch Environmental Planning Bureau in Bilthoven developed the ecological capital index with the objective to assess the state of both natural and cultural ecosystems in relation to human activities. This ecological capital index is calculated by multiplying the ecosystem's quantity by its quality. The abiotic environment is here regarded as a conditional variable for the selected biodiversity reference situation, or status quo. (RIZA 1999). The international application of the ecological capital index respects the recommendations for a core set of indicators as proposed by the Convention on Biological Diversity (UNEP 1997) and thus are compatible with the international classification (IUCN 1991) of ecosystems on the basis of the degree of human influence.[56]

Ecosystem health indexes allow the scientist to assess the overall ecosystem performance. A number of examples have been identified. . These indexes play a crucial role for policy guidance since they let us predict ecosystem response as a result of alternative management scenarios, and make it possible to compare scenario rankings.

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[56] The International Union for Nature Conservation, under the auspices of the United Nations, makes a distinction between natural, adapted, cultivated, built and deteriorated ecosystem.

Economic foundations for biodiversity analysis and valuation

Why economists pursue economic valuation

The economic valuation of natural resources in general, and biodiversity, in particular, is among the most challenging issues confronting environmental economists. Economists value biodiversity because valuation allows for a direct comparison with economic values of alternative options and facilitates, for example, cost-benefit analysis – a crucial tool for policy formulation. In addition, the monetary valuation of biodiversity allows economists to perform environmental accounting, natural resource damage assessment, and to carry out benefit assessment. Valuation is essential in the research of individual consumer behaviour. It indicates the opinion of individual consumers about certain biodiversity management objectives and identifies individual consumer motivations with respect to biodiversity conservation.

Economic valuation methods are controversial because placing monetary value on biodiversity is often questioned. Arguments against it are rooted in the human preference orientation that 'guides' consumer behaviour with respect to biodiversity. Albeit oversimplified, we can distinguish two broad ranges of value orientation - see Table 8.4. According to the 'anthropocentric' orientation, the value of biodiversity is related to its role in human welfare, as humans conceive it "(…) whether they are selfish, altruistic, loyal, spiteful, or masochistic (…)" (Becker 1993). A second valuation perspective is rooted in 'biocentric' or 'ecocentric' value orientation, which claims that nature has an intrinsic value[57] and therefore deserves protection. In reality, however, value orientations are overlapping and several versions of 'anthropocentrism' and 'ecocentrism' can exist within one individual. Stewardship and Q-altruism are examples of such 'mixed' attitudes (Sagoff 1980, Norton 1982, Van der Veer and Pierce 1986). Stewardship is a form of altruism that is fully divorced from any explicit notion of consumption. It corresponds to a sense of responsibility for the conservation and maintenance of the resource. Q-altruism is rooted in the firm belief that living organisms are incapable of protecting themselves against human actions. Therefore, the conservation of living organisms merits human sympathy or compassion.[58]

Biodiversity as a source of economic values

The concept of total economic value of an environmental resource has its foundations in welfare economics. It focuses on changes in the economic welfare of humans. Therefore, the terms 'economic value' and 'welfare change' can be used interchangeably.[59] The total economic value (TEV) of an environmental resource consists of its use value (UV) and non-use value (NUV). Use values arise from the actual use. It can be further divided into direct use values (DUV), indirect use values (IUV) and option values (OV) (Pearce and Moran 1994). The non-use values are usually divided into a bequest value (BV) and existence value (XV).

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57 By definition an intrinsic value is the value of the resource *per se* without a subject attaching a value to it (Ehrenfeld 1988).

58 Recently, Nunes (2000a) developed the use of multiple motivation items scales to measure consumers' level of "anthropocentricism" and integrate such a construct in a contingent valuation application.

59 It should be clear that economists do not pursue total value assessment of an environmental system, but of system changes (preferably marginal or small).

Table 8.4 Value orientations and environmental attitudes

Value orientation	Valuation perspective	Ethical approach	Biodiversity attitude
Ecocentrism	rights conferred to all living organisms	Nature has intrinsic value, regardless of human recognition	Biodiversity first
Antrophocentrism	rights and interests conferred to individual humans	Value of nature is value conferred by humans	Humans first

Source: Adapted from Lockwood (1999).

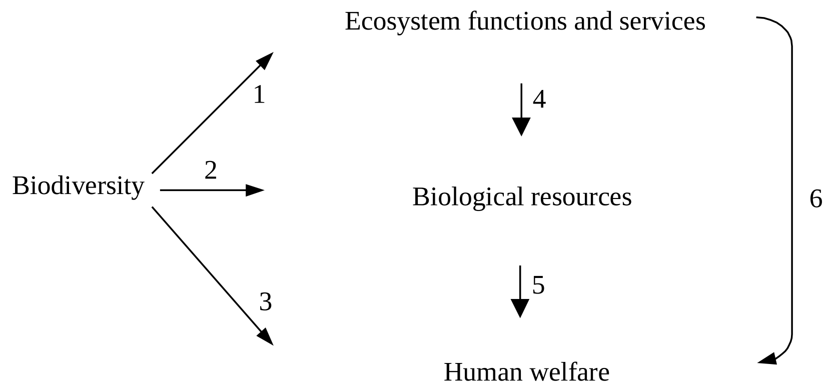
Bequest value refers to the benefit to any individual from the knowledge that others might benefit from an environmental resource in the future. Existence value refers to the benefit derived simply from the knowledge that the resource is protected. This leads to the following equation.

$$TEV = UV + NUV = (DUV + IUV + OV) + (XV + BV)$$

Economic values of changes in biodiversity require special attention. We classify such values into four categories. These are illustrated in Figure 8.3. A first category, denoted by link 1→6, depicts biodiversity benefits in terms of ecosystem functions and services We can interpret this biodiversity value category as a direct use value component. A second category, denoted by link 1→4→5, defines the value of biodiversity in terms of supply of ecosystem space or natural habitat protection. We can interpret this biodiversity value category as an indirect use value component. A third category, denoted by link 2→5, defines the benefits to society in terms of an overall diversity provision of biological resources, notably specific animals and species for use in agriculture and medicine. We can interpret this biodiversity value category as a direct use value component. A fourth category, denoted by link 3, refers to the direct impact of biodiversity on human welfare. The economic value of biodiversity is then measured in terms of philanthropic considerations, independently of biodiversity use or consumption. One can interpret this biodiversity value category as a non-use component. The non-use values have a pure public good[60] character and no market price is available. Consequently, policy makers have mostly based their decisions on an undervaluation of biodiversity benefits, which has resulted in misuse and misallocation of scarce resources. The monetary assessment of biodiversity use and non-use benefits requires special valuation tools. These are discussed in the following section.

60 There are goods that are either will be not supplied by the market or, if supplied, will be supplied in insufficient quantity. These are called public goods. They have two important properties. First, it does not cost anything for an additional individual to enjoy the benefits of the public goods. Second, it is, in general, difficult or impossible to exclude individuals from the enjoyment of the public good.

Figure 8.3 Economic values of biodiversity



Alternative monetary valuation methods

In the absence of market prices, certain techniques are needed to value consumers' preferences for biodiversity benefits. On the basis of the biodiversity component to be valued, one can distinguish alternative valuation methods: travel cost, hedonic price, averting expenditures cost, production function, and contingent valuation [61] – see Table 8.5. Travel cost (Bockstael *et al.* 1991) is especially designed to assess recreational benefits by looking at the generalised travel costs (site entrance fees, accommodation and journey costs) when visiting a natural park or reserve. Alternatively, when using the hedonic price method (Palmquist 1991) to estimate the economic value of a biodiversity benefit, say, good soil conditions, researchers explore the land market values and soil quality characteristics (e.g. depth of topsoil). For example, King and Sinden (1988) used the hedonic price method in order to assess the value of soil conservation in the farm land market of Manilla Shire, in New South Wales, Australia. The hedonic land market price regression results show that the implicit marginal price of land is $2.28/ha. Researchers can also estimate the economic value of biodiversity on the basis of consumer expenditure made to avert or mitigate the adverse effects (Cropper and Freeman 1991) of the loss of biodiversity benefits. Huszar (1989) estimated that the total annual costs of erosion caused by wind in New Mexico from additional household's expenditures in house cleaning and maintenance activities were $454 million. Alternatively, the economic value of biodiversity can be assessed through a production function. Barbier (1994) conducted a study focused on the value assessment of the economic importance of Hadejia-Jama'are wetlands, Nigeria. The valuation is based on the estimation of some key direct use values that the floodplain provides to local population through crop production, fuelwood and fishing.

[61] Many other classifications of valuation exist. See Freeman (1979).

Table 8.5 Identification of monetary valuation methods according to the different biodiversity value components

TOTAL ECONOMIC VALUE (TEV)	USE VALUES (UV)	Direct use value (DUV)	Recreation benefits, e.g. sight-seeing, fishing, swimming Methods: Travel cost, contingent valuation
		Indirect use value (IUV)	Ecosystem functional benefits, e.g. regulating local chemical composition of the water Methods: production function, averting behaviour, hedonic price
		Option Value (OV)	Insurance for having the asset on *stand-by*, e.g. future visits, future genetic manipulation Methods: contingent valuation
	NON-USE VALUES (NUV)	Bequest Value (BV)	Legacy benefits e.g. habitat conservation for future generations Methods: contingent valuation
		Existence value (XV)	Existence benefits, e.g. knowledge of existence of marine wildlife diversity Methods: contingent valuation

Finally, the contingent valuation method collects data to retrieve individuals' preferences on biodiversity by means of questionnaires, often supported by multidisciplinary research teams (Carson *et al.* 1992, NOAA 1993). Contingent valuation presents important features. First, it is able to express the Hicksian welfare measure[62] directly in monetary terms. Second, the CV method is the only valuation technique that is capable of shedding light on the monetary valuation of the non-use values, i.e., the benefit value component that is not directly associated with its direct use or consumption. Ignoring such values would lead to a systematic bias in the estimation (essentially an underestimation) of the total benefits of biodiversity. Moreover, CV brings with it the advantage that environmental changes may be valued, even if they have not yet occurred (*ex ante* valuation), offering a greater potential scope and flexibility in specifying different states of nature that may lie outside the current institutional arrangements or levels of provision. For the same reasons CV can be used to assess almost all value components of biodiversity and thus is frequently used in environmental policy.[63] Contingent valuation, and other methods are reviewed and discussed in more detail in the following section.

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62 The Hicksian welfare measure is a particular measurement of the loss of individual enjoyment whenever a certain (public) good is not supplied by the market or, if supplied, will be supplied in insufficient quantity. The interesting characteristic of such a measurement is that it is expressed in monetary units.

63 An exception is made for the ecosystem functional benefits since these are hard to describe in a questionnaire format and they are often related to issues not particularly familiar to most respondents.

Empirical valuation studies

This section reviews some biodiversity valuation studies, presenting a valuation range for each biodiversity value component.[64] The discussion is organised according to Figure 8.3. First, we focus on studies that carry out a value assessment of biodiversity in terms of the benefits associated with the protection of species diversity (see link 2→5). Second, we focus on valuation studies that pursue the assessment of biodiversity benefits in terms of protection of natural habitat or ecosystem space (see link 1→4→5). Third, we discuss valuation studies that focus on biodiversity's benefits on society in terms of ecosystem functions and services, (see link 1→6). Finally, whenever the CV method is applied, one is able to assess the non-use value component of biodiversity, and thus assess the monetary value of link 3.[65]

Single, multiple species valuation surveys

Boyle and Bishop (1987) conducted one of the first CV studies to estimate the existence values for wildlife species. The estimated values suggest that substantial existence values are associated with the Bighorn Sheep in Wyoming (see results in Table 8.6). More recently, Van Kooten (1993) studied the economic value of waterfowl in Canada. According to the results, the shadow values of marginal land converted to waterfowl habitat were estimated to be \$50 to \$60 per acre. Loomis and Larson (1994) valued an "emblematic" endangered species, namely the Gray Whale. The willingness to pay (WTP) was estimated between sixteen and eighteen dollars per household per year. Boman and Bosdedt (1995) carried out the economic valuation of the conservation of the Wolf in Sweden. The estimation results show that the mean WTP is about \$126 per year.

Table 8.6 also shows multiple species valuation studies. Johnansson (1989) conducted a CV study that focuses on the preservation of 300 endangered species in Sweden., The final estimate, \$ 194 per year, is higher than the economic valuation of the conservation of the Wolf in Sweden, about \$126 per year, though not so high as one would expect. This is partly because the single species valuation study refers to popular, socially attractive, charming and beautiful species. Therefore, when preserving the Wolf *in situ*, as one of Sweden's fauna emblems, the final value estimate may also include other characteristics not directly related to the species to be valued. In other words, the valuation results may also reflect individual social esteem motives or warm glow (Nunes 2000a). Furthermore, the estimate valuation for single species can be affected by the availability of related species, i.e. single species values can be affected by the respondent's perception of substitutes and complementary species (Samples and Hollyer 1989).

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[64] Nunes and van den Bergh (2000) list a more extensive bibliography of work related to valuation of biodiversity.

[65] Since non-use values have no behavioural market trace, economists cannot retrieve information about these values by relying on market-based valuation approaches.

Table 8.6 Valuation studies

	Valuation study	Method
Single species		
Minimum range: $5	Big Horn, endangered species in Wyoming, US Boyle and Bishop (1987)	Contingent valuation
Maximum range: $126	Wolf, endangered species in Sweden Boman and Bosdedt (1995)	Contingent valuation
Multiple species		
Minimum range: $18	Preservation of threatened and endangered species populations in the US, Hageman (1985)	Contingent valuation
Maximum range: $194	Preservation of 300 endangered species in Sweden, Johnansson (1989)	Contingent valuation
Habitat: Terrestrial (non-use)		
Minimum range: $27	Protection of the Nadgee Nature Reserve, Australia, Bennett (1984)	Contingent valuation
Maximum range: $101	Desert Protection in California, US Richer (1995)	Contingent valuation
Habitat: Coastal (non-use)		
Minimum range: $10	Protection of New Jersey beaches, US Silberman *et al.* (1992)	Contingent valuation
Maximum range: $51	Protection of a wilderness coastal area, Portugal Nunes (2000)	Contingent valuation
Habitat: Wetland (non-use)		
Minimum range: $8	Protection of the Norfolk Broads, UK Batemann *et al.* (1992)	Contingent valuation
Maximum range: $96	Enhancing wetland habitat in California, US Hoehn and Loomis (1993)	Contingent valuation
Habitat: Ecosystem space (recreation)		
Minimum range: $23/trip	Forest recreation activities in Flanders, Belgium Moons (1999)	Travel cost
Maximum range: $23 million/year	Tourism in Ecuador WTO (1997)	Tourism revenue
Ecosystems functions		
Minimum range: $1.2 million	Life-support value of a wetland ecosystem in the a Swedish island, Baltic Sea, Turner *et al.* (1995)	Production function
Maximum range: $4.4 billion	Water ecosystem benefits in ten regions in US Ribaudo (1989)	Averting behaviour

Diversity at the ecosystem level

The majority of CV studies that focus on biodiversity at the ecosystem level link it directly to the non-use or recreational valuation of habitat protection programs. The main reason for this link is primarily the difficulty associated with defining such an abstract concept as ecosystem diversity in a survey. Indeed, some CV studies indicate that the concept of biodiversity is ill understood among the general population (Hanley *et al.* 1995). A number of valuation studies have also attempted to value biodiversity conservation policies through other methods. Generally, we find studies that focus on ecosystem functions and the value assessment of life-support, soil and wind erosion, or water quality benefits. Some of these studies are listed in Table 8.6.

During the 1980s many contingent valuation studies dealt with the measurement of the non-use benefits derived from the conservation of national parks and nature reserves - see Bennett (1984) and Richer (1995). The valuation applications continued through the 1990s – see Silberman *et al.* (1992), Batemann *et al.* (1992), and Hoehn and Loomis (1993) – but now also tackling the valuation of non-use benefits of coastal and wetland habitats. Silberman *et al.* estimated the existence value for users and non-users of New Jersey beaches. The results show that the mean WTP for a user is about $15.1 per year, while the mean WTP for a non-user is about $9.26 per year. Batemann *et al.* (1992) undertook a contingent valuation to assess the monetary value of conserving the Norfolk Broads, a wetland site in the UK with three National Nature Reserves. A mail survey across Britain showed that non-visitor respondents were willing to pay, on average, 4 pounds (circa $8) for an annual and once-for-all-payment. More recently, Nunes (2000) used the CV method for the first time in Portugal to assess the national WTP for the protection of a coastal natural area. The mean WTP results ranged from $40 to $51 also for an annual and once-for-all-payment.

In the recreation domain, the World Tourism Organisation (WTO 1997) estimated that Ecuador earned $255 million from eco-tourism in 1995. A major part came from a single park, the Galapagos Islands. Studies of less popular areas indicate lower values. The recreational value of a Regional Forest Park in Belgium was estimated to be around $23 per trip (Moons 1999). Norton and Southey (1995) calculated the economic value of biodiversity protection in Kenya by assessing the associated opportunity costs in terms of forgone agricultural production, which is estimated to be $203 million. This value can be compared with $42 million in net financial revenues from wildlife tourism and forestry. More recently, Chase *et al.* (1998) studied the eco-tourism demand in Costa Rica. The value estimates result from the survey of foreign visitors to three national parks: Volcan Irazu, Volcan Poas, and Manuel Antonio. The highest WTP registered was about $25 per visitor per year for the Manuel Antonio national park.

When it comes to the monetary valuation of ecosystem functions, CV may not be the first method of choice. This is because ecosystem life support is not an issue familiar to the general public. In addition, the complexity of the relationships involved makes an accurate and comprehensive survey description more difficult. Researchers frequently end up using other valuation methods such as averting behaviour, production function, or hedonic pricing. In 1991, Andreasson-Gren (1991) estimated the costs of nitrogen abatement via wetlands restoration with the market costs associated with the use of standard abatement technologies. The estimated nitrogen purification capacity of wetlands was based on the results of a Swedish island in the Baltic Sea, Gotland. According to the study results, the total value of a marginal increase in nitrogen abatement on Gotland was about SEK 968 per kilogram. Turner *et al*. (1995) addressed the valuation of a wetland ecosystem in the Swedish island in the Baltic Sea exploring the use of the production function method. Their value estimations confirmed that a considerable amount of additional energy would be necessary to the production of substitute market goods in order to replace the loss of the wetland life-support functions – see results in Table 8.6. Ribaudo (1989) is responsible for one of the most comprehensive studies valuing water ecosystems. The author valued the economic benefits from the reduction in the discharge of pollutants in waterway systems for nine impact categories: recreational fishing, navigation, water storage, irrigation ditches, water treatment, industrial water use, steam cooling, and flooding. Benefits were defined in terms of changes in defensive expenditures[66], changes in production costs, or changes in consumer surplus, depending on the damage category and the availability of data. The total water quality benefits were estimated to be $4.4 billion.

Integrated ecological-economic modelling and valuation of biodiversity

Economics – Ecology interface

The analysis and the modelling of biodiversity are rooted in both natural and social sciences and thus imply the study of human economic activities, their relationship to biodiversity and the structure and functions of ecosystems. The combination or integration of the two approaches implies a somewhat qualitative, formal, sequentially integrated framework. Interdisciplinary work involves economists or ecologists transferring elements or even theories and models from one discipline to another and transforming them for their specific purpose (Perrings *et al.* 1995). The underlying objective of this approach is the development of a common way of thinking about modelling and valuation of biodiversity. For instance, if economic and ecological models fit in a general system frame, then they may be blended in a single model structure where compartments or modules may represent the original models and certain outputs of one module serve as input for another. Nevertheless, it is not often easy to link models directly. Alternatively, if both the economic and ecological systems are represented in the form of programming or optimisation models then several options are available: look for a new, aggregate objective; adopt a multi-objective or conflict analysis framework; or, when possible, derive multiple sets of optimality conditions and solve these simultaneously. Finally, when economic and ecological systems are represented by different model types, it is harder to suggest how they can be linked to one another. Where economic models have an optimisation format and ecosystem models have a descriptive format, then a direct technical integration seems feasible, otherwise heuristic approaches are needed. This may require operations such as reduction, simplifying or summarising. For example, one can come up with a simple dynamic model summarising and simplifying some of the causal relationships of the spatial hydrological model and the statistical vegetation model, and linking the outcomes to a simplified economic interaction and values model.

Before discussing specific methods and models it is useful to briefly analyse the pros and cons of integration frameworks and respective conceptual perspectives. These are discussed in the following section.

Integrated modelling and assessment

A general method to develop integrated models is a systems approach (also ‘systems dynamics’). This covers a wide range of model types: linear versus non-linear, continuous versus discrete, deterministic versus stochastic, and optimising versus descriptive. Such system approaches allow one to deal with concepts like ecological dynamic processes, feedback mechanisms, and controlling strategies (see Bennett and Chorley 1978; Costanza *et al.* 1993). One can integrate two subsystems, or have a hierarchy or nesting of systems. The systems approach is suitable for integrating existing models and incorporating temporal as well as spatial processes. Costanza *et al.* distinguish economic, ecological and integrated approaches on the basis of the following criteria: (1) generality,

[66] These are expenditures incurred with the purchase of market goods with the goal to mitigate the externality of flooding, e.g. river silp up activities.

characterised by simple theoretical or conceptual models that aggregate, caricature and exaggerate; (2) precision, characterised by statistical, short-term, partial, static or linear models with one element examined in much detail; and (3) realism, characterised by causal, non-linear, dynamic-evolutionary, and complex models. These three criteria are usually conflicting so that a trade-off between them is inevitable. A very general and almost non-theoretical ('no assumptions') framework is the Driver-Pressure-State-Impact-Response (DPSIR) framework, a variation on the framework proposed for environmental data classification by Turner *et al.* (1999) and Rotmans and de Vries (1997) for integrated analysis and modelling. The components have the following interpretation:

- 'Driver' = economic and social activities and processes.
- 'Pressure' = pressures on the human (health) and environmental system (resources and ecosystems).
- 'State' = the physical, chemical and biological changes in the biosphere, human population, resources and artefacts (buildings, infrastructure, machines).
- 'Impact' = the social, economic and ecological impacts of natural or human-induced changes in the biosphere.
- 'Responses' = human interventions on the level of drivers (prevention, changing behaviour), pressures (mitigation), states (relocation) or impacts (restoration, health care).

According to Rotmans and de Vries (1997) integration can include various types. Vertical integration means that the causal chain in the DPSIR framework is completely described in a model. Horizontal integration (of subsystems) in this context is defined as the coupling of various global biogeochemical cycles and earth system compartments (atmosphere, terrestrial biosphere, hydrosphere, lithosphere and cryosphere). An alternative and relevant distinction is between analytical and heuristic integration. Analytical integration means combining all aspects studied in a single model (and therefore model type). Heuristic integration can proceed by using the output of one model as input to another, and vice versa, as well as extending this by an (finite) iterative interaction. In this case different model types can be combined, such as optimisation models and descriptive models. If one desires to attain a great deal of analytical power, the analytical integration seems attractive, whereas striving for realism would imply the use of a heuristically linked set of models from different disciplines. Full or total economic-environmental integration requires a combination, leading to the complex linking of various drivers, pressures, states, impacts and responses, thus allowing for various synergies and feedback. The literature shows various examples of such integrated economic-environmental frameworks. Surveys are offered by Barbier (1990), van den Bergh and Nijkamp (1991), van den Bergh (1996), Costanza *et al.* (1997), Ayres *et al.* (1999) and Turner *et al.* (1999).

Integrating modelling and monetary valuation

Progress on improving methods for providing such economic information (particularly predictive information) will require a strong and dynamic interdisciplinary dialogue. At this multidisciplinary level, integrating modelling and monetary valuation can present important advantages for policy guidance, presenting important interactions.

First, values estimated in a valuation study can be used as a parameter value in a model study. Benefits or value transfer (e.g. meta-analysis exercise) can be used to translate value estimates to other contexts, conditions, locations or temporal settings that do not allow for direct valuation in ‘primary studies’ (due to technical or financial constraints).

Second, models can be used to generate values under particular scenarios. In particular, dynamic models can be used to generate a flow of benefits over time and to compute the net present value, which can serve as a value relating to a particular scenario of ecosystem change or management.

Third, models can be used to generate detailed scenarios that enter valuation experiments. An input scenario can describe a general environmental change, regional development or ecosystem management. This can be fed into a model calculation, which in turn provides an output scenario with more detailed spatial or temporal information. The latter can then serve, for example, as a hypothetical scenario for valuation, which is presented to respondents in a certain format (graphs, tables, story, diagrams, pictures) so as to inform them about potential consequences of the general policy or exogenous change. Computer software can be used in such a process. Finally, the output of model and valuation studies can be compared. For instance, when studying a scenario for wetland transformation one can model the consequences in multiple dimensions (physical, ecological and costs/benefits), and aggregate these via a multi-criteria evaluation procedure, with weights being set by a decision-maker or a representative panel of stakeholders. Alternatively, one can ask respondents to provide value estimates, such as a willingness to pay for not experiencing the change. If such information is available for multiple management scenarios, then rankings based on either approach can be compared. So far no empirical study has accomplished this.

Conclusions and recommendations

How can we use the ideas presented in Sections 2, 3 and 4 to formulate an integrated, effective framework for the economic valuation of biodiversity? And what do we learn from the empirical valuation studies focused on biodiversity? A crucial, and initial step, is to carefully describe the object of analysis and valuation. Therefore, researchers face two important decisions when valuing biodiversity: (1) at which level should biodiversity be examined; and, (2) what biodiversity value types should be measured?

Clearly, it is necessary to attain information about the nature, type, and persistence of stress or shocks experienced by ecosystems, their functions and stability, and respective impacts on human welfare. A comprehensive assessment of ecosystem biodiversity characteristics, structure and functioning requires the analyst to undertake various important actions. First, the causes of biodiversity loss should be determined, in order to improve understanding of socio-economic impacts on biodiversity processes and attributes. Second, the range and degree of biodiversity functioning should be assessed, especially in terms of ecosystem-functional relationships. Third, the sustainability of biodiversity uses should be assessed and the consequences of biodiversity loss should be determined. Finally, alternative biodiversity management strategies should be assessed and spatial and temporal systems analysis of alternative biodiversity conservation scenarios should be carried out.

Ecologists contribute to identification of the range of policy decisions with respect to the biodiversity strategies, by exploring the use of ecological valuation methods such as red data species lists and biological value indexes. More recently, computer models have often been used by ecologists to aid conservation evaluation decisions, namely in the domain of species population. Several general models have been developed for this task and applied, for example, to calculate the dimension of a geographic area of suitable habitat required to support the minimum viable population (e.g., the giant panda). Finally, computer models have also been used for habitat evaluation with the objective of simulating the impact of different development scenarios and predicting the respective ecological conservation values. This approach to ecological evaluation has attractive features. First, it permits the introduction of evaluation criteria within a transparent and repeatable framework. Second, it allows a direct comparison of management or conservation strategies. Finally, it permits an evaluation even when some of the data concerning the functions performed by biodiversity are missing, which is quite common in practice.

The concept of economic value has its foundations in welfare economics. Therefore, valuation in an economic sense is always the result of an interaction between the subject and an object. Moreover, economists do not pursue total value assessment of an environmental system, but rather system changes. The goal is then to assess the human welfare significance of biodiversity change under consideration, through the determination of the changes in provision of biodiversity related goods and services and consequent impacts on the well-being of humans who enjoy both use or non-use benefits from such a provision. Different instruments are available to assess the economic value of biodiversity. The choice is not always evident. Survey valuation studies have often been used because the use of revealed preference methods leaves out important biodiversity value types, notably non-use values. Alternatively, researchers can combine valuation techniques. Special attention, however, should then be given to value aggregation across the resulting values so as to avoid double counting.

There is a clear need to obtain information about the cause, type, and persistence of stress on biodiversity and the estimation of the respective impacts on human welfare. The combination or integration of ecology and economics to assess and value biodiversity leads to an integrated framework. Interdisciplinary work is thus required, involving both economists and ecologists transferring elements or even theories and models from one discipline to another and transforming them for their specific, mutually consistent purpose. In other words, the underlying objective is the development of a common way of thinking about modelling and valuing biodiversity.

References

Akcakaya, H.R. (1994). "RAMAS/METAPOP: viability analysis for stage-structured metapopulations, *Applied Biomathematics,* Setauket, New York, US.

Andreasson-Gren, I.M. (1991). "Costs for Nitrogen Source Reduction in a Eutrophicated Bay in Sweden" in Folke C. and T. Kraberger (eds), *Linking the Natural Environment and the Economy: Essays from the Eco-group,* Kluwer Academic Publishers.

Antonovic (1990). "Genetically Based Measures of Uniqueness", in Orians, G.H., Brown, G.M., Kunin, W.E. and J.E. Swierzinbinski (eds) *The Preservation and Valuation of Genetic Resources,* University of Washington Press, pp. 94-118.

Ayres, R.U., K.J. Button and P. Nijkamp (1999). *Global Aspects of the Environment,* Vol. 1 and 2. Edward Elgar, Cheltenham, UK.

Avise, J.C. and C.F. Aquadro (1982). "A comparative summary of genetic differences in vertebrates", *Evolutionary Biology,* 15, pp. 151-185.

Barbier, E.B. (1990). "Alternative approaches to economic-environmental interactions", *Ecological Economics,* 2, pp. 7-26.

Barbier, E.B. (1994). "Valuing Environmental Functions: Tropical Wetlands", *Land Economics,* 70(2), pp. 155-173.

Batemann, I.J., Willis, K.G., Garrod, G.D., Doktor, P., Langford I. and R.K. Turner (1992). *Recreation and Environmental Preservation Value of the Norfolk Broads: A Contingent Valuation Study,* Report to the National Rivers Authority, Environmental Appraisal Group, University of East Anglia, valuation results cited in Turner et al. (1995).

Becker, G.S. (1993). "Nobel Lecture: the Economic Way of Looking at Behaviour", *Journal of Political Economy,* 101(3), pp. 385-409.

Bennett, J. W. (1984). "Using Direct Questioning to Value the Existence Benefits of Preserved Natural Areas", *Australian Journal of Agricultural Economics,* 28, pp. 136-152.

Bennett, R.J. and R.J. Chorley, 1978. *Environmental Systems*. Methuen & Co., London.

Bertollo, P. (1998). "Assessing Ecosystem Health in Governed Landscapes: a Framework for Developing Core Indicators", *Ecosystem Health,* 4(1), pp. 33-51.

Bockstael, N. E., McConnell, K. E., and I. Strand (1991). "Recreation", in Braden, J. B. and C. D. Kolstad (eds). *"Measuring the Demand for Environment Quality"*, Chapter VIII, Elsevier Science Publishers, North-Holland.

Boman, M. and G. Bostedt (1995). "Valuing the Wolf in Sweden", *Report No. 110, Swedish University of Agricultural Sciences,* Department of Forest Economics, Umeä.

Boon, P.J., Holmes, N.T, Maitland, P.S., Rowell, T.A. and J. Davies (1997). "A system for evaluating rivers conservation (SERCON): development, structure and function", in P.J. Boon and D.L. Howell (eds). *Freshwater Quality: Defining the Indefinable,* HMSO, Edinburgh, UK.

Boyle, K.J. and R.C. Bishop (1987). "Valuing wildlife in benefit-cost analysis: a case study involving endangered species", *Water Resources Research*, 23, pp. 943-950.

Braden, J. B., Kolstad, C.D. and D. Miltz (eds) (1991) "Measuring the Demand for Environment Quality", Elsevier Science Publishers, North-Holland.

Brink, B.J and S.H. Hosper (1989). "Naar Toetsbare Ecoloische Doelstllingen voor het Waterbeheer: de AMOEBE benadering", *H_2O*, 22(20), pp. 607-617.

Buys, S. (1995). "Urbanisation Alternatives in the Fourth Report on Physical Planning", *Netherlands Journal of Housing and the Built Environment"*, 7(2), pp. 135-154.

Carson, R. T., R. C. Mitchell, W. M. Hanemann, R. J. Kopp, S. Presser and P. A. Ruud (1992). "A Contingent Valuation Study of Lost Passive Use Values Resulting from the Exxon Valdez Oil Spill", *Report prepared for the Attorney General of the State of Alaska*, Washington.

Chase, L.C., Lee, D.R., Schulze, W.D. and D.J. Anderson (1998). "Ecotourism Demand and Differential Pricing of National Park Access in Costa Rica", *Land Economics*, 74(4), pp. 466-482.

Cody, M.L. (1986). "Diversity , Rarity and Conservation in Mediterranean-Climate Regions", in M.E. Soule (ed) *Conservation Biology: the Science of Scarcity and Diversity*, pp. 123-152, Sinauer Associates, Sunderland, Mass.

Cody, M.L. (1993). "Bird Diversity Components within and between Habitats in Australia, in R.E. Ricklefs and D. Schluter (eds), *Species Diversity in Ecological Communities: Historical and Geographic Perspectives*, pp. 147-158, University of Chicago Press, Chicago, US.

Costanza, R. (1992). "Toward an Operational Definition of Ecosystem Health", in Costanza, R., Norton, B. and B. Haskell (eds), *Ecosystem Health: New Goals for Environmental Management*, Island Press, Washington, US.

Costanza, R., Norton, B. and B. Haskell (eds) (1992). "Ecosystem Health: New Goals for Environmental Management", Island Press, Washington, US.

Costanza, R., L. Wainger, C. Folke and K.-G. Mäler (1993). "Modelling complex ecological economic systems", *Bioscience*, Vol. 43: 545-555.

Costanza, R., Kemp, M. and W. Boynton (1995). "Scale and Biodiversity in Coastal and Estuarine Ecosystems" in Perrings, C., C.S. Holing, K-G. Miler, B-O. Janssen and C. Folke (eds) *Biodiversity Loss: Economic and Ecological Issues*, Cambridge University Press.

Cropper, M. L. and A. M. Freeman III (1991). "Environmental Health Effects", in Braden, J. B. and C. D. Kolstad (eds) *"Measuring the Demand for Environment Quality"*, Chapter III, Elsevier Science Publishers, North-Holland.

Cuperus, R. and K.J. Canters (1995). "Op near natuurcompositie voor autosnelwegen", Rijksuniversiteir Leiden, Leiden.

de Groot, R.S. (1994). "Environmental functions and the economic value of natural ecosystems", in *Investing in Natural Capital: the Ecological Economics Approach to Sustainability*, A. Jansson *et al.* (eds), Island Press.

Ehrenfeld, D. (1988). "Why put a value in biodiversity" in *Biodiversity*, E.O. Wilson (ed), National Academy Press.

Fitter, R., and M. Fitter (ed.) (1987). *The Road to Extinction*, Gland, Switzerland: IUCN.

Folke C. and T. Kraberger (eds) (1991). *Linking the Natural Environment and the Economy: Essays from the Eco-group*, Kluwer Academic Publishers.

Folke, C., C.S. Holling and C. Perrings (1996). "Biological diversity, ecosystems and the human scale", *Ecological Applications*, 6, pp. 1018-1024.

Franklin, J.F. (1993). "Preserving Biodiversity: species, ecosystems or landscapes", *Ecological Applications*, 3, pp. 202-205.

Freeman, A. M. III (1979). "Approaches to Measuring Public Goods Demands", *American Journal of Agricultural Economics*, 61, pp. 915-920.

Gren, I-M., Folke, C., Turner, R.K. and Bateman, I. (1994). Primary and Secondary Values of Wetland Ecosystems. *Environmental and Resource Economics*. Vol.4: pp. 55-74.

Hageman, R. (1985). "Valuing Marine Mammal Populations: Benefits Valuations in Multi-species Ecosystems", in *Administrative Report*, Southwest Fisheries, values cited in Eagle J. and D. Betters (1998). "The Endangered Species Act and Economic Values: a Comparison of Fines and Contingent Valuation Studies", *Ecological Economics*, 26, pp. 165-171.

Hanley, N., C. L. Spash and L. Walter (1995). "Problems in Valuing the Benefits of Biodiversity Protection", *Environment and Resources Economics*, 5(3), pp. 249-272.

Hoehn, J.P. and J.B. Loomis (1993). "Substitution Effects in the Valuation of Multiple Environmental Programs", *Journal of Environmental Economics and Management*, 25(1), pp. 56-75.

Hoelzel, A.R. and A. Green (1992). "Analysis of population-level variation by sequencing PCR-amplified DNA".

Hoelzel, A.R. (ed.) Molecular Genetic Analysis of Populations, Oxford University Press, Oxford, UK.

Holling, C.S. (1987). Simplifying the complex, the paradigms of ecological function and structure", *European Journal of Operational Research*, 30, pp. 139-146.

Holling, C.S. (1992). "Cross-scale morphology geometry and dynamics of the ecosystems", *Ecological Monographs*, 62, pp. 447-502.

Howels, O. and Edward-Jones (1997). "A feasibility study of reintroducing wild boar to Scotland: would existing woodlands support minimum viable populations?", *Biological Conservation*, 81, pp. 77-89.

Huston, M.A. (1994). *Biological Diversity – the coexistence of species on changing landscapes*, Cambridge University Press, Cambridge, UK.

Huszar, P.C. (1989). "Economics of Reducing Off-Site Costs of Wind Erosion", *Land Economics*, 65(4), pp. 333-340.

ICN-Instituto de Conservação da Natureza (1993). "Livro vermelho dos vertebrados em Portugal", Secretatia de Estado do Ambiente e Consumidor, ICN, Lisboa, Portugal.

IUCN (1991). "United Nation Environmental Program and World Wide Fund for Nature, Caring for the Earth", International Union of Nature Conservation, Gland.

IUCN (1993). *Draft IUCN Red List Categories,* Gland, Switzerland: IUCN.

IUCN (2000). *IUCN Internet Page*, http://www.iucn.org/themes/ssc/redlists/categor.htm

Johnansson, P-O. (1989). "Valuing Public Goods in a Risky World: an Experiment", in Folmer, H. and E. Ierland (eds) *Evaluation Methods and Policy Making in Environmental Economics*, Amsterdam, North Holland.

King, D.A. and J.A. Sinden (1988). "Influence of Soil Conservation on Farm Land Values", *Ecological Economics,* 64(3), pp. 242-255.

Lacy, R.C. (1993). "VORTEX: a computer simulation model for population viability analysis", *Wildlife Research*, 20, pp. 45-65.

Lacy, R.C and T.W. Clark (1990). "Population viability assessment of eastern barred bandicoot, in T.W. Clark and J.H. Seebeck (eds) *Management and Conservation of Small Populations,* Chicago Zoological Society, Brookfield, IL.

Levin, S.A. (1998). "Ecosystems and the Biosphere as Complex Adaptive Systems", *Ecosystems*, 1(5), pp. 431-436.

Levin, S.A., Greenfell, B., Hastings, A. and A. Perelson (1997). "Mathematical and Computational Challenges in Population Biology and Ecosystem Science", *Science*, 275, pp. 334-343.

Lindenmayer, D.B., Lacy, R.C., Thomas, V.C. and T.W. Clark (1993). "Predictions of the impacts of changes in population size and environmental variability on Leadbeater's possum using population viability analysis", *Wildlife Research,* 20, pp. 67-86.

Lockwood, M. (1999). "Humans Valuing Nature: Synthesising Insights from Philosophy, Psychology and Economics", *Environment Values,* 8, pp. 381-401.

Loomis, J.B. and D.M. Larson (1994). "Total economic value of increasing Gray Whale populations: results from a contingent valuation survey of visitors and households", *Marine Resource Economics*, 9, pp. 275-286.

Mace, G. M. et al. (1992). "The development of new criteria for listing species on the IUCN Red List" *Species,* 19, pp. 16-22.

Mace, G. M., and R. Lande. (1991). "Assessing extinction threats: toward a reevaluation of IUCN threatened species categories", *Conservation Biology,* 5(2), pp. 148-57.

Mace, G. M. & S. N. Stuart. (1994). "Draft IUCN Red List Categories, Version 2.2", *Species,* 21-22, pp. 13-24.

Moons, E. (1999). "Estimation of the recreation values of a forest", *M.Sc. dissertation,* Faculty of Economics and Applied Economics, Catholic University of Leuven.

NOAA - National Oceanic and Atmospheric Administration (1993). "Report of the NOAA Panel on Contingent Valuation", *Federal Register,* 58(10), pp. 4601-4614, US.

Norberg, J. (1999). "Linking Nature´s Services to Ecosystems: Some General Ecological Concepts", *Ecological Economics*, 29, pp. 183-202.

Norton, B. G. (1982). "Environmental Ethics and Nonhuman Rights", *Environmental Ethics*, 4, 17-36.

Norton, G. M. and C. Southey (1995). "The opportunity costs of biodiversity protection in Kenya", *Ecological Economics,* 11, pp. 135-151.

Norton and Ulanowicz (1992). "Scale and Biodiversity Policy: A Hierarchical Approach", *Ambio,* 21(3), pp. 244-249.

Noss, R.F. (1983). "A Regional Landscape Approach to Maintain Diversity", *BioScience,* 33, pp. 700-706.

Nunes, P.A. (2000). "Contingent Valuation of the benefits of Natural Areas and its warm glow component", *Ph.D. dissertation,* Faculty of Economics and Applied Economics, Catholic University of Leuven, Leuven, Belgium.

Nunes, P.A. (2001). "Using Factor Analysis to Identify Consumer Preferences for the Protection of a Natural Area: evidence from a Valuation Survey in Portugal", forthcoming, *European Journal of Operational Research.*

Nunes, P.A. and van den Bergh, C. J. M. (2000). "Economic analysis and valuation of biodiversity: a critical review", out for review, *Journal of Ecological Economics*.

Nunes, P.A. and van den Bergh, C. J. M. and P/ Nijkamp (2000). "Biodiversity: ecological-economic analysis and valuation", Tinbergen Institute Discussion Paper Series, Rotterdam, Netherlands.

Odum, E.P. (1971). "The Fundamentals of Ecology", Saunders College Publishing, Philadelphia, US.

Palmquist, R. B. (1991). "Hedonic Methods", in Braden, J. B. and C. D. Kolstad (eds) "*Measuring the Demand for Environment Quality*", Chapter IV, Elsevier Science Publishers, North-Holland.

Pearce, D. (1999). "Valuing biological diversity: issues and overview", draft paper presented at *OECD Workshop on Benefit Valuation for Biodiversity Resources*, Paris, France.

Pearce, D. and D. Moran (1994). "The Economic Value of Biodiversity", Earthscan Publications Limited, London, UK.

Perrings, C. (1998). "Resilience in the dynamics of economic-environment systems", *Environmental and Resource Economics: Special Issue*, 11, pp. 503-520.

Perrings, C., K-G. Maler, C. Folke, C.S. Holling and B-O. Jansson (eds) (1995). "Biodiversity Loss: Economic and Ecological Issues", Cambridge University Press.

Randwell, D.S. (1969). pers. comm. cited in Ratcliffe, D.A. (1986). "Selection of important areas for wildlife conservation in Great Britain: the Nature Conservancy Council's Approach, in M.B. Usher (ed.), *Wildlife Conservation Evaluation,* Chapman & Hall, London, UK.

Rapport, D., Costanza,R., Epstein, P.R., Gaudet, C. and R. Levins (1998). "Ecosystem Health", Blackwell Science, UK.

Ribaudo, M.O. (1989). "Targeting the Conservation Reserve Program to Maximize Water Quality Benefits", *Land Economics*, 65(4), pp. 320-332.

Richer, J. (1995). " Willingness to Pay for Desert Protection", *Contemporary Economic Policy*, XIII, pp. 93-104.

RIZA (1999). "Uitganspunten voor een Graadmeteer Natuur-waarde voor Nederland: het Ecologisch Kapitaal Index concept, definitiestudie in opdracht van het Natuurplanbureau", unpublished.

Ruijgrok, E. (1999). "Valuation of Nature in Coastal Zones", *Ph.D. dissertation*, Free University of Amsterdam.

Sagoff, M. (1980). "On the Preservation of Species", *Columbia Journal of Environmental Law*, 7, pp. 33-67.

Samples, K. and J. Hollyer (1989). "Contingent Valuation of Wildlife Resources in the Presence of Substitutes and Complement", in Johnson, R. and G. Johnson (eds) "Economic Valuation of Natural Resources: Issues, Theory and Application", Boulder, CO.

Sijtsma, F.J.and D. Strijker (1995). "Effect analyse ecologische hoodstructuur; Deel II: Natuurwaarde, Stiching Ruimtelijke Economie Groningen, Groningen, The Netherlands.

Sijtsma, F.J., Strijker, D. and M.L. Hoefsloot (1998). "Duurzame Ontwikkeling in het Waddengebied", University of Groningen, Groningen, The Netherlands.

Silberman, J., Gerlowski, D.A. and N.A. Williams (1992). "Estimating Existence Value for Users and Nonusers of New Jersey Beaches" , *Land Economics*, 68(2), pp. 225-236.

Soulé, M.E. (ed) (1987). *Viable Population for Conservation*, Cambridge University Press, Cambridge, UK.

Spellerberg, I.F. (1992). *Evaluation and Assessment for Conservation,* Chapman & Hall, London, UK.

Turner, R.K., Folke, C., Gren I.M. and I.J. Batemann (1995). "Wetland Valuation: Three Case Studies", in Perrings C., Mäler, K-G., Folke, C., Holling, C.S. and B-O. Jansson (eds), *Biodiversity Loss: Economic and Ecological Issues*, Cambridge University Press.

Turner, R.K., Perrings, C. and Folke, C. (1997). *Ecological Economics: Paradigm or Perspective?* in J. Van den Bergh and J. Van der Straaten (eds.) Economy and Ecosystems in Change, Edward Elgar Publishers, Cheltenham.

Turner, R.K., K. Button and P. Nijkamp (eds) (1999). "Ecosystems and Nature: Economics, Science and Policy", Elgar.

Ulanowicz, R.E. (1992). "Ecosystem Health and Tropic Flows Networks" in Costanza, R., Norton, B. and B. Haskell (eds), *Ecosystem Health: New Goals for Environmental Management,* Island Press, Washington, US.

UNEP (1995). "Global Biodiversity Assessment" report by Watson, R.T., Heywood, V.H., Baste, I., Dias, B., Gamez, R. and W. Reid, United Nations Environment Program, US.

UNEP (1997). "Recommendations for a Core Set of Indicators of Biological Diversity", Convention on Biological Diversity, Montreal, United Nations Environment Program, US.

Usher, M.B. (1989). "Nature Conservation in UK", *Journal of Applied Ecology,* 26, pp. 813-824.

van den Bergh and P. Nijkamp (1991). "Aggregate economic-ecological models for sustainable development", *Environment and Planning,* 23: pp. 187-206.

van den Bergh and J. van der Straaten (eds) (1997). Economy and Ecosystems in Change, Edward Elgar Publishers, Cheltenham.

van der Veer D. and C. Pierce (1986). "People, Penguins and Plastic Trees: Basic Issues in Environmental Ethics" Wadsworth, Belmont, US.

van Kooten, G.C. (1993). "Bioeconomic Evaluation of Government Agricultural Programs on Wetland Conservation", *Land Economics*, 69(1), pp. 27-38.

Ward, R., Skibinski, D. and M. Woodmark (1992). "Protein heterozygosity, protein structure, and toxonomic differentiation", *Evolutionary Biology,* 26, pp.73-159.

Whittaker, R.H. (1960). "Vegetation of the Siskiyou Mountains, Oregon and California", *Ecological Monographs,* 30, pp. 279-338.

Whittaker, R.H. (1972). "Evolution and Measurement of Species Diversity", *Taxon,* 21, pp. 1-67.

Williams, P.H., Vane-Wright, R.I. and RI Humphries (1991). "Measuring Biodiversity for Choosing Conservation Areas" in LaSalle (ed) *Hymenoptera and Biodiversity*, CAB International.

WTO (1997). *Statistical Year Book of Tourism,* World Tourism Organisation, Madrid.

Zhou, Z. and W. Pan (1997). "Analysis of the viability if a giant panda population", *Journal of Applied Ecology,* 34, pp. 363-374.

Zurlini, G., Amadio, G. and O. Rossi (2000). "A Landscape Approach to Biodiversity and Biological Planning: the Map of Italian Nature, *Ecosystem Health,* 5(4), pp. 294-311.

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Dennis King earned his Ph.D. in natural resource economics in 1977 from the University of Rhode Island (USA). His research and consulting practice focuses on natural resources industries and markets and the economics of environmental conservation and restoration. He is the author of over one hundred reports, papers, and book chapters and has been project manager on over fifty interdisciplinary research projects addressing international, national, and regional economic, development, and environmental issues. He developed and taught pioneering undergraduate and graduate level courses in natural resource economics, marine resource economics, and ecological economics, has provided expert testimony on environmental-economic conflicts before U.S. congressional committees, and has been an expert witness in over 40 cases involving private litigation over natural resource related economic losses. Dr. King serves on several national and state environmental and economic advisory boards, has been a member of National Research Council and National Academy of Science committees, and is a frequent consultant to congressional committees, regulatory bodies, and industry/government councils.

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Dr. O'Neill received his doctorate in Ecology in 1967 and spent his research career at Oak Ridge National Laboratory where he is a Corporate Fellow in the Environmental Sciences Division. He has been a pioneer in ecosystem modeling, error/risk analysis, hierarchy theory, and landscape ecology. He has 250 publications, including 5 books. He has received a number of awards including the prestigious McArthur Award from the Ecological Society of America.

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Peter Nijkamp (1946) graduated from the Erasmus University in Rotterdam, in the area of econometrics. He holds a Ph.D. (cum laude) in non-linear mathematical programming for industrial planning from the same University. Since 1975 he is professor in regional and urban economics and in economic geography at the Free University, Amsterdam. His main research interests cover plan evaluation, multicriteria analysis, regional and urban planning, transport systems analysis, mathematical modelling, technological innovation, and resource management. In the past years he has focused his research in particular on quantitative methods for policy analysis, as well as on behavioural analysis of economic subjects. He has a broad expertise in the area of public policy, services planning, infrastructure management and environmental protection. In all these fields he has published various books and numerous articles. He has been an advisor to several Dutch Ministries, regional and local policy councils, employers' organizations, private institutions, the Commission of the European Union (EU), the Organisation for Economic Cooperation and Development (OECD), the European Conference of Ministers in Transport (ECMT), the Asian Development Bank (ADB), the European Roundtable of Industrialists, ICOMOS, the World Bank, and many other institutions.

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OECD PUBLICATIONS, 2, rue André-Pascal, 75775 PARIS CEDEX 16
PRINTED IN FRANCE
(97 2001 15 1 P) ISBN 92-64-19665-X – No. 52213 2001