ELSEVIER

Ocean & Coastal Management 45 (2002) 583–616

Ocean & Coastal Management

www.elsevier.com/locate/ocecoaman

Valuing ocean and coastal resources: a review of practical examples and issues for further action

L. Ledoux*, R.K. Turner

Centre for Social and Economic Research on the Global Environment (CSERGE), School of Environmental Science, University of East Anglia, Norwich NR4 7TJ, UK

Abstract

This review article examines the importance of valuing environmental resources in the context of sustainable development. The different values stemming from ocean and coastal resources, relevant methodologies and issues raised by valuation approaches are reviewed. The authors then present practical policy-relevant valuation examples, and conclude by outlining progress since 1992 and remaining challenges. It is argued that while the Rio summit has shifted somewhat the emphasis from classical cost–benefit analysis to safe minimum standards through the adoption of the precautionary principle, economic valuation still provides useful information to decision-makers and should be part of a holistic decision-making process. It should be recognised, however, that although valuation techniques have been refined and linked to reliability protocols, they remain imperfect and for some commentators controversial. Further progress is needed on assigning monetary values but also on decision-making systems that better integrate monetary, social, and natural science criteria.

1. Introduction

The earth summit in 1992 put sustainable development on the nations' agenda, and advocated a closer integration of environmental and socio-economic sciences in decision making [1,2]. This is particularly important in ocean and coastal zones, which have come under heavy environmental pressure in recent decades. One of the outcomes of the Earth Summit has been to recognise the need for an integrated approach to coastal management [3]. The management challenge is made even more problematic by the natural variability present in ocean and coastal systems and by

*Corresponding author. Tel.: +44-1603-593747; fax: +44-1603-593739.
E-mail address: l.ledoux@uea.ac.uk (L. Ledoux).

0964-5691/02/$ - see front matter
PII: S0964-5691(02)00088-1

the multiple stakeholder interests and competing resource uses and values typically found in such zones [4].

Ocean resources are a global common good. Future prospects are bleak if current fishing and seabed exploitation practices and trends continue in an unreformed way. All countries with a coastline have a direct interest in the sustainable management of the coastal resource systems. The main objective of such management can be seen as the sustainable utilisation of the multiple goods and services generated by coastal and ocean resources (processes, functions and their interrelationships), together with the 'socially equitable' distribution of welfare gains and losses inherent in such usages [5]. This social welfare accounting process is essential for designing sustainable development policies, and presents an evaluation challenge, which is the main focus of this review article. The first and second sections take a closer look at the importance of valuing resources in the context of sustainable development, summarising the different values stemming from ocean and coastal resources, and methodologies available, as well as reviewing how valuation studies can inform decision-making. After examining issues raised by valuation approaches, the authors then look at practical valuation examples and in the context of decision-making, considering environmental functions across different spatial scales, ranging from a 'local' site to a drainage basin at the 'international' scale. The article concludes by outlining progress since 1992 and remaining challenges.

2. An economic perspective to sustainable development

The Driver Pressure State Impact Response (DPSIR) framework has proved useful to scope sustainable development issues in oceans and coastal zones. First used by the OECD, it was further developed and adapted to the context of coastal zone management by Turner [6] (Fig. 1). At the root of environmental change are economic drivers, for example agricultural intensification, urbanisation, tourism development, which in turn will create pressures: land conversion and reclamation, nutrient emissions, waste disposal in coastal waters. These pressures, along with physical factors such as climate change, will lead to changes in the state of the environment: changes in nutrient concentration, loss of habitat and species diversity. These physical changes will in turn have an impact on human welfare, for example through fisheries productivity, health impacts, or amenity value changes. Environmental economic valuation measures these changes in human welfare due to physical changes, in terms of costs and benefits to society. These welfare changes will provide the stimulus for management action, which will seek to control socio-economic drivers and consequent environmental pressures, thus creating a continuous and dynamic cycle with feedback loops.

Sustainability essentially requires that the stock of capital and the related set of opportunities available in future is equivalent to that available at present. Although an economic use of the environment can be both efficient and sustainable, economic efficiency does not in itself guarantee sustainability [7,8]. Given that a significant proportion of human welfare is based on coastal and ocean resources and

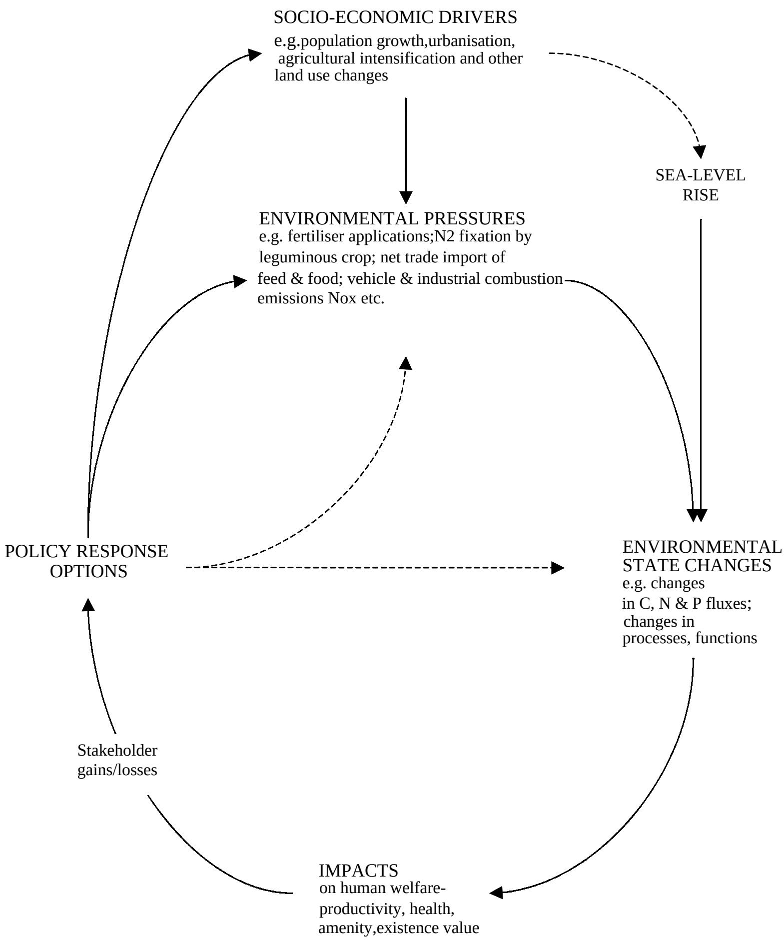

Fig. 1. D–P–S–I–R cycle and continuous feedback process (Source: [6]).

ecosystems, sustainable decision-making begs the question: what aggregate level of human welfare does society want to maintain in the future, how should it be distributed and what level of conservation is required. Decision-makers need to know people's values for ocean and coastal resources to start answering that question. In other words, how environmental state changes may influence human welfare is a key issue in developing sustainable development policies. In an attempt to aid decision-making in that context, the concept of sustainability has been roughly partitioned into two approaches: weak sustainability and strong sustainability [9–11]. Weak sustainability requires that the total stock of capital, whether man-made or natural, be maintained, and rests upon the assumption of substitutability between these two types of capital. Economic theory suggests that decreasing supplies of natural resources will tend to increase their price, encouraging more efficient use, substitution with other goods, and technological advancement. However, complete substitution will not always be possible due to physical limits to the efficiency and availability of substitution opportunities. There is also the question of whether man-made capital is able to fully compensate for all the functions provided by complex ecosystems, and the existence of 'critical' natural capital and thresholds beyond which reversal is not possible. Hence the more stringent interpretation of 'strong sustainability' which requires that the total stock of natural capital be non-declining. Natural and man-made capital, rather than being regarded as substitutes, can be interpreted as complements [12], making it important that stocks of both are maintained. Projects should either conserve the natural capital or ensure that losses incurred are capable of being replaced or fully compensated-for.

In line with the strong sustainability criterion, this paper argues that the concepts of functional diversity and functional value diversity offer sound and practical foundations for a management strategy aimed at the sustainable utilisation of coastal and ocean resources [5]. The basic notion is that ecosystem processes, composition, and functions provide outputs of goods and services, which can then be assigned monetary economic and/or other values (Fig. 2).

From an anthropocentric viewpoint all ecosystems can be classified in terms of their structural and functional aspects [13,14]. Ecosystem structure is defined as the tangible items such as plants, animals, soil, air and water of which it is composed. Thus structural benefits (of instrumental value to humans) include fish, waterfowl, peat, timber, reed and fur harvests as well as non-consumptive use benefits such as recreation and research or education. By contrast, ecosystem processes are encompassed by the dynamics of exchange of means of energy. The processes are subsequently responsible for the services—life support services, such as assimilation of pollutants, cycling of nutrients and maintenance of the balance of gases in the air.

The terminology used regards processes and functions as relationships within and between natural systems; uses refer to actual use, potential use and nonuse interactions between human and natural systems; and values refer to assessment of human preferences for a range of natural or non-natural 'objects', services and attributes. A management strategy based on the sustainable utilisation of coastal resources principle should have at its core the objective of ecosystem integrity maintenance i.e. the maintenance of system components, interactions among them and the resultant behaviour or dynamic of the system. Protecting functional diversity and functional diversity values within and between ecosystems would therefore be a vital component action in a sustainability strategy [5].

3. Sustainable development and valuation

We have argued that valuation is necessary to inform sustainable decision-making, in a market-based economy. One key to valuing a change in an ecosystem function is establishing the link between that function and some service flow valued by people. If that link can be established, then the concept of derived demand can be applied. The value of a marginal change in an ecosystem function can be derived from the change in the value of the ecosystem service flow it supports (Fig. 3).

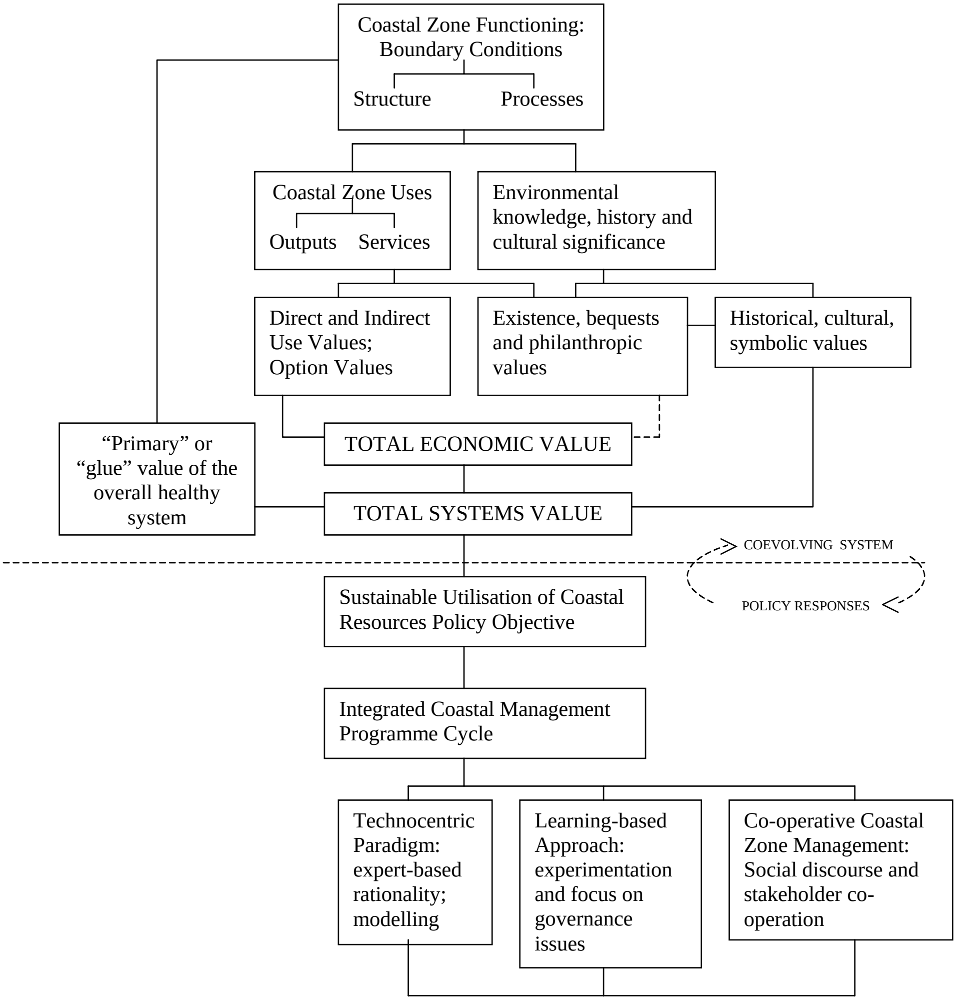

Fig. 2. Functional and other dimensions of coastal zone values (Source: [5]).

The main problem when including the range of environmental services in economic choices is that many of these services are not valued on markets. There is a gap between market valuation and the economic value of environmental resources. To fill these gaps, the non-market values must first be identified and then where possible monetised. The mainstream economic approach to valuation takes a more or less instrumental (usage-based) approach (as opposed to an intrinsic value which resides in the object itself), and seeks to combine various components of value into an aggregate measure of resource value labelled total economic value (TEV). This total economic value (TEV) can be usefully broken down into a number of categories as shown in Fig. 3. The initial distinction is between use (direct and indirect) value

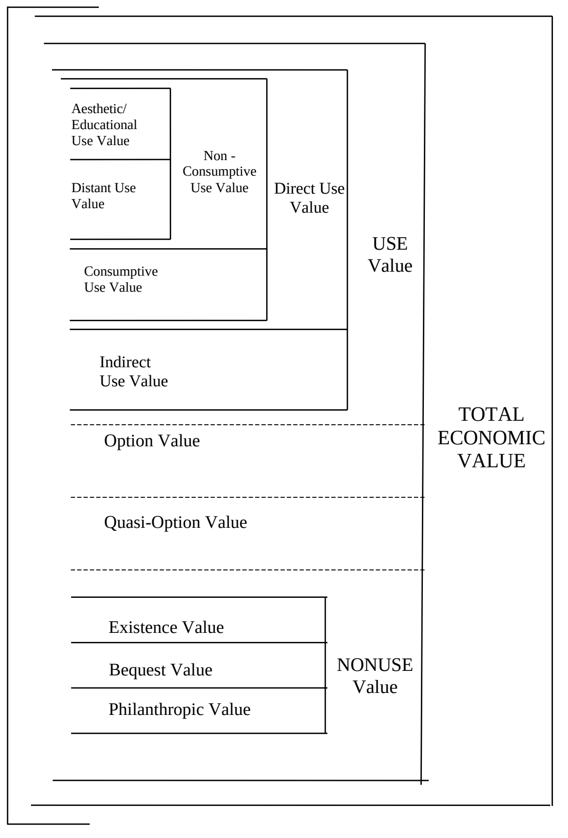

Fig. 3. Components of economic value (Source: [5]).

and non-use value, but there is really a continuum in place and the boundaries between use and non-use categories are not clearcut [15].

A range of valuation methodologies can be applied, that capture different components of the TEV, as described in Table 1, using a wetland ecosystem as an example (see [7,5] for a more complete description). Monetary valuation is still, however, controversial. There is a growing body of evidence to suggest that some of the conventional economic axioms are systematically violated by humans in controlled experiments and in their everyday life. As citizens, individuals are influenced by held values, attitudes, and beliefs about public-type goods and their provisions. In this context, property rights (actual and/or perceived), social choices and moral concerns can all be involved in a nature conservation versus development conflict. The polar opposite view to the conventional economic approach would hold that market boundaries should not be extended to cover as many environmental assets as is possible. Instead, society should give greater consideration to the nature of deliberative institutions for resolving environmental problems and of the social and economic framework that will sustain them [16]. The counterbalancing

Table 1
Valuation methodologies relating to ecosystem functions: e.g. wetlands

Valuation method	Description	Direct use values	Indirect use values[a]	Non-use values
Market analysis	Where market prices of outputs (and inputs) are available. Marginal productivity net of human effort/cost. Could approximate with market price of close substitute. Requires shadow pricing.	√	√	
(Productivity losses)	Change in net return from marketed goods: a form of (does-response) market analysis.	√	√	
(Production functions)	Wetland treated as one input into the production of other goods: based on ecological linkages and market analysis.		√	
(Public pricing)	Public investment, for instance via land purchase or monetary incentives, as a surrogate for market transactions.	√	√	√[b]
Hedonic price method (HPM)	Derive an implicit price for an environmental good from analysis of goods for which markets exist and which incorporate particular environmental characteristics.	√	√	
Travel cost method (TCM)	Cost incurred in reaching a recreation site as a proxy for the value of recreation. Expenses differ between sites (or for the same site over time) with different environmental attributes.	√	√	
Contingent valuation (CVM)	Construction of a hypothetical market by direct surveying of a sample of individuals and aggregation to encompass the relevant population. Problems of potential biases.	√	√	√
Damage costs avoided	The costs that would be incurred if the wetland function were not present; e.g., flood prevention.		√	
Defensive expenditures	Costs incurred in mitigating the effects of reduced environmental quality. Represents a minimum value for the environmental function.		√	
(Relocation costs)	Expenditures involved in relocation of affected agents or facilities: a particular form of defensive expenditure.		√	
Replacement/ substitute costs	Potential expenditures incurred in replacing the function that is lost; for	√	√	√[c]

Table 1 (*continued*)

Valuation method	Description	Direct use values	Indirect use values[a]	Non-use values
	instance by the use of substitute facilities or 'shadow projects'.			
Restoration costs	Costs of returning the degraded wetland to its original state. A total value approach; important ecological, temporal and cultural dimensions.	√	√	√[c]

Source: [5].

[a] Indirect use values associated with functions performed by a wetland will generally be associated with benefits derived off-site. Thus, methodologies such hedonic pricing and travel cost analysis, which necessarily involve direct contact with a feature of the environment, can be used to assess the value of indirect benefits downstream from the wetland.

[b] Investment by public bodies in conserving wetlands (most often for maintaining biodiversity) can be interpreted as the total value attributed to the wetland by society. This could therefore encapsulate potential non-use values, although such a valuation technique is an extremely rough approximation of the theoretically correct economic measure of social value, which is the sum of individual WTP.

[c] Perfect restoration of the wetland or creation of a perfectly substitutable 'shadow project' wetland, which maintains key features of the original, might have the potential to provide the same non-use benefits as the original. However, cultural and historical aspects as well as a desire for 'authenticity' may limit the extent to which non-use values can be 'transferred' in this manner to newer versions of the original. This is in addition to spatial and temporal complexities involved in the physical location of the new wetland or the time frame for restoration.

argument would be that some environmental goods/services which have a mixed public/private good set of characteristics (e.g. coastal resource) could be privatised or securitised (shares issued), so that self-interest and profit motive can be made to work in favour of environmental conservation [17]. Fig. 4 summarises three highly simplified and probably overlapping worldviews about the valuation and assessment of environmental quality. Section 4 develops further the effect of these standpoints on decision-making.

4. Issues in valuation

4.1. Quantitative and qualitative approaches to valuation

A combination of quantitative and qualitative research methods can be advocated in order to generate a blend of different types of policy relevant information. This applies to both the biophysical assessment of management options, and the evaluation of the welfare gains and losses people perceive to be associated with the environmental changes and the management options that may be entailed.

Social research dependent on quantitative research methods and techniques is premised on the assumption that opinions, feelings, perceptions, beliefs, attitudes or behaviour can be expressed in meaningful numerical ways within a given context. It

Simplified Typology

[Mainstream Position]

- Individualism
- Exogenous preferences
- fixed preferences
- marginal/discrete environmental changes
- revealed preferences via markets; household production functions
- positive rates of discount
- efficiency criterion
- economic welfare

[Extended CBAPosition]

- environmental systems-functions
- individualism vs collectivism-consumer and citizen motivations
- endogenous preferences-psychosocial and cultural theory variables
- expressed and revealed preferences
- focus group testing
- benefits transfer
- non-use values
- validity/reliability testing protocols
- equity criterion-policy trade offs econ welfare ≠ social welfare
- standards/regulations-cost effectiveness

[Environmental Management without prices]

- preferences not appropriate basis for valuation
- rights-based approaches environmental ‘trump’ cards
- citizen motivations as a distinct and separate category
- keep ‘markets’ out of the environment
- expert opinion
- contingent valuation as opinion polls
- deliberative and participatory processes to assign values via social discourse

Fig. 4. Approaches to valuation (Source: [5]).

is most often criticised for its overly reductionist character in the face of real world complexity and diversity, i.e. social, cultural, economic, political and environmental. Its technical nature may also act as a shroud, obscuring its ‘proper’ interpretation by the public.

Qualitative research methods, on the other hand, are, in principle, more comprehensive in their coverage of the variety of contexts found in society. But such research usually produces a vast amount of ethnographic data not amenable to scrutiny via traditional statistical or related analysis as in quantitative approaches. Consequently, interpretation of the results is perhaps an even more difficult task and the risk of manipulation and value judgement masking no less apparent [18].

More research is needed since the individual- and group-based approaches place the whole process of eliciting environmental values, monetised or not, in different social settings and therefore provides us with different kinds of information. Typically, qualitative research will provide in-depth information on fewer cases, whereas quantitative procedures will allow for more breadth of information across a larger number of cases. A combination of both approaches offers future promise for environmental valuation.

In summary, the main generic approaches which can form the methodological basis for strategic socio-economic options appraisal are:

- Cost effectiveness analysis—assesses the impact of different options in physical terms, and compare these to the costs of the different options to determine which option, or mix of options, achieves the target at least cost.
- Cost–benefit analysis—values all costs and benefits in monetary terms to establish a stream of costs and benefits associated with a particular policy over time and compare these.

- Multi-criteria analysis—where policy options are compared by reference to an explicit set of evaluation criteria, which can include both quantitative (monetary or not) and qualitative data, together with a weighting scheme.

There will be circumstances where one approach is preferable to the others. Where there are clear and commonly agreed objectives or targets to be reached in a specific policy context, then the most appropriate approach is likely to be to look at cost-effective options. On the other hand, when targets cannot be pre-defined but must be determined within the assessment exercise, and all or most of the impacts can be expressed in money terms then cost–benefit analysis will be favoured. In contrast, if impacts cannot be monetised, but are instead expressed via a variety of measurement units, if they are to be included in a less aggregated way, and with more explicit input from stakeholders, then multi-criteria analysis may be most appropriate. In practice a variety of approaches are used which provide decision-makers with a range of different types of information about the nature and level of costs and benefits allowing them to make better decisions.

Usually all of the above approaches will involve some form of stakeholder analysis—in that they can involve stakeholders at a number of different points within the appraisal process. Stakeholders could for example be involved in the setting of management objectives, or in the determination of values. Deciding how stakeholders should be involved is a key issue and may influence the choice of the valuation approach.

4.2. Irreversibility, thresholds and the precautionary approach

An important aspect of the economics–science interface is the existence of thresholds and the potential for irreversible change. Where an incremental change in a parameter has a disproportionate effect, this might be associated with relatively high economic costs. And if the change is irreversible, account needs to be taken of the uncertain future losses that might be associated with this change, and the possible imposition of a Safe Minimum Standards (SMS) decision rule [19–21]. This rule recommends that when an impact on the environment threatens to breach an irreversible threshold, that the conservation option be adopted unless the costs of forgoing the development are regarded as 'unacceptable'. It is based on a principle of minimising the maximum possible loss, rather than cost–benefit and risk analysis which is based on maximising expected gains. Given our absolute uncertainty regarding the benefits in the future that might have been derived from the threatened resource, our maximum possible loss must be associated with loss of that resource. We can calculate the net benefits that we expect to derive from the development project which threatens this resource, and so long as forgoing these benefits is regarded as an acceptable sacrifice. The conservation option is always preferred in contexts involving potentially irreversible damages.

Clearly the critical factor in SMS is what is regarded as an 'unacceptable' sacrifice of present benefits for the sake of possible future losses. The degree of sacrifice entailed involves a full cost–benefit assessment of the development option, including the quantifiable costs of damage to the environment. It is then largely a political decision within the constraints of society's other goals, as to whether avoiding potentially massive, but wholly uncertain, future costs can be justified. In this sense, it provides a mechanism for incorporating the Precautionary Principle into decision-making, where society may choose to conserve even in the absence of proof that damage will occur, in order to limit potential costs in the future [22].

The concept of SMS has usually been applied to endangered species. In this context it may well be applicable to a number of ecosystems given their role in supporting a variety of threatened species. However, it could equally well apply to irreversible impacts threatening ecosystems as a whole. One complication is the need to identify what is a truly irreversible change in the ecosystem, since any change that can be reversed in the future will not necessarily entail the maximum possible costs. It will also be necessary to determine whether or not thresholds in current ecosystem functioning exist, and whether these may be threatened by proposed developments. Where it is decided that thresholds of ecosystem functioning are threatened with irreversible change, SMS as a decision framework that gives more weight to concerns of future generations and promotes a more sustainable approach to current development might represent an appropriate supplement to purely monetary analysis.

4.3. Discounting and temporal scale

Temporal scale, in combination with the rate of discount applied, will influence the value assigned to ecosystem functions. It is frequently necessary within cost–benefit analysis to choose between alternative projects which may have different intertemporal patterns of benefits and costs extending over varying durations. Costs and benefits which occur at different times need to be compared within a common matrix, and this is the rationale behind discounting effects which occur in the future. It is common practice in economic appraisal to convert the stream of future costs and benefits into 'present' values to allow them to be directly compared, the difference between total benefits and costs being referred to as 'net present value' (NPV). A project is only accepted if NPV is positive.

Discounting future values stems from the observation that costs and benefits in the future are not valued as highly as equivalent costs and benefits occurring now. The choice of discount rate can have a significant influence on which projects pass the cost–benefit criterion. Options that involve high initial costs and a stream of benefits far into the future, such as the creation or restoration of wetlands, are less likely to be accepted when employing a higher rate of discount. Options for which the benefits are more immediate and the costs are not incurred until far into the future, will become more viable with a higher discount rate. For projects that produce hazardous wastes that must be stored for lengthy periods, such as nuclear power generation, the potentially disastrous costs can become insignificant when discounted to present value. A higher rate of discount is also more likely to encourage more rapid depletion of non-renewable natural resources and over-exploitation of renewable natural resources, thereby reducing the inheritance of natural capital for future generations. However, lower rates of discount will tend to encourage investments in non-environmentally beneficial projects that might not otherwise have been viable and could conceivably result in more rapid depletion of resources. The link between the size of discount rate and the degree to which options will impinge upon the environment is therefore ambiguous, and it is not clear that the traditional call for lower discount rates in order to incorporate environmental concerns is generally valid.

It is the social rate of discount that should be used when assessing developments that will influence intergenerational welfare. Maintaining future welfare could be regarded as a public good, if it is seen as an obligation of society as a whole, in which private individuals will tend to under-invest. As a result, the social discount rate—measured as either the social rate of time preference (SRTP) or the social opportunity cost of capital (SOC)—can be expected to be lower than the equivalent individual rate of discount. The rates currently recommended for project evaluation by the UK Treasury, for example, are 8% for commercial investments, 6% for public sector projects and 3% for the forestry sector. Pearce [23] in a recent study of the factors determining the social rate of discount in the UK, measured as the consumption rate of interest, argue that a rate of nearer 2% would be more appropriate.

The discount rate does not take into account effects that developments might have which are irreversible, for instance the extinction of species or exhaustion of minerals. An approach to rectify this has been proposed in which future benefits forgone are treated as additional costs. These net benefits of preservation are likely to increase over time as demand for environmental services rise, with limited or declining supply, while net benefits from development projects are likely to decline relatively as alternative technologies improve. These temporal trends in benefits can be incorporated into the decision rule by applying adjustments to the social discount rate: in effect, decreasing the discount rate applied to preservation benefits while increasing the rate applied to development benefits.

4.4. Aggregation and double counting

If each output provided by an ecosystem is identified separately, and then attributed to underlying functions, there is the likelihood that benefits will be double counted. Benefits might therefore have to be explicitly allocated between functions. For instance, Barbier [24] notes that if the nutrient retention function is integral to the maintenance of biodiversity, then if both functions are valued separately and aggregated this would double count the nutrient retention which is already 'captured' in the biodiversity value. Some functions might also be incompatible, such as water extraction and groundwater recharge, so that combining these values would overestimate the feasible benefits to be derived from the ecosystem.

Double counting will be particularly important with partial analysis and total valuation of an ecosystem, although some approximations to total valuation do not encounter this problem. A study by Costanza [25] has engaged environmental scientists and policy makers, but the global, biome scale, economic value calculations contained in this study risk criticism from both scientists and economists. On the basis of the data and methods cited in the article and supporting inventory, the conclusion that the value of the biosphere services is, on average US $33 trillion per year, is not to be taken literally. Apart from raising policymaker, scientist and citizen awareness about the environment's economic value and the possible significance of the loss of that value over time, the global value calculations do not serve to advance meaningful policy debate in efficiency and equity terms, in practical conservation versus development contexts. Such calculations with their 'single number' outcomes shroud a number of fundamental 'scaling' problems to do with valuation contexts i.e. the temporal, spatial and cultural specificity of economic value estimates. Such values can also only meaningfully be assigned to relatively small ('marginal') changes in ecosystem capabilities (Table 2). The practical problem is that determining precisely what is and what is not a discrete and marginal change in complex ecological systems is not straight forward (Turner [26]).

4.5. *Benefit transfer*

Environmental value transfer is commonly defined as the transposition of monetary environmental values estimated at one site (study site) through market-based or non-market-based economic valuation techniques to another site (policy site). The most important reason for using previous research results in new policy

Table 2
Estimated impact of four development scenarios for Buccoo Reef Marine Park area on the economic, social and ecological criteria[a]

Criteria	Scenario			
	A	B	C	D
Economic				
(1) Economic revenues to Tobago (US$ million)	9	11	17	19
(2) Visitor enjoyment of BRMP (US$ million)	1.2	2.5	0.9	1.7
Social				
(3) Local employment (no. jobs)	2500	2600	6400	6500
(4) Informal sector benefits (score)	5	4	3	2
(5) Local access (score)	6	5	6	7
Ecological				
(6) Water quality (μg N/l)	1.5	1.4	2.2	1.9
(7) Sea grass health (g dry weight/m^2)	18	19	12	15
(8) Coral reef viability (% live stony coral)	19	20	17	18
(9) Mangrove health (ha)	65	73	41	65

Source: [45].

[a] Scenarios: (A) limited tourism development without complementary environmental management; (B) limited tourism development with complementary environmental management; (C) expansive tourism development without complementary environmental management; (D) expansive tourism development with complementary environmental management.

contexts is cost-effectiveness. Applying previous research findings to similar decision situations is a very attractive alternative to expensive and time consuming original research to quickly inform decision-making. The criteria for selecting studies for environmental value transfer suggested in the literature focus on the environmental goods involved, the sites in which the goods are found, the stakeholders and the study quality [27]. However, very little published evidence exists of studies that test the validity of environmental value transfer. Moreover, in the few studies that have been carried out, the transfer errors are substantial [28].

Bateman [29] review approaches to benefit transfer in theory and practice, of which a summary of findings follow. The simplest approach to transferring benefits is to apply the unit value estimate of the site at which the original valuation study was conducted to the target site where benefit estimates are required. In practice, the assumption of identical unit values across sites may not hold for a variety of reasons, including differences in the socio-economic characteristics of the relevant population, differences in the physical characteristics of the study and policy site, difference in the proposed change in provision between the sites, differences in the market conditions applying to the sites. The extent to which these and other potential differences hold can be regarded as the criteria for acceptability of unadjusted unit value transfer. Clearly, ideal conditions for this simple approach will rarely hold, and unit values should be adapted to the new site. Three adjustment strategies can be considered:

- Expert judgement—while there may be certain cases for which this is acceptable, more objective adjustment techniques are obviously preferable;
- Re-analyses of existing study samples to identify subsamples of data suitable for transferral—the extent to which this can be done depends crucially upon initial sample size, and this problem becomes exacerbated where sub-division is required across a number of variables; and
- Meta-analysis, which is the statistical analysis of the summary findings of prior empirical studies for the purpose of integrating findings [30]. Meta-analysis assumes relatively standard designs and standard measurements. The further the raw data deviates from such specifications, the more it is difficult to rely on results from a cross-analysis. Another problem is that studies published in the available literature may over represent studies which produce "positive" or significant results if studies yielding "negative" or non-significant findings tend not to be published. Nevertheless, meta-analysis offers a transparent structure allowing the derivation of useful generalisations, and permitting extraction of information from large masses of data in a way that would be difficult with narrative or qualitative analysis only.

The last approach to benefit transfer is to transfer the entire benefit function from the study to policy site, instead of using unit values. Since benefit estimates are often a complex function of the site and user characteristics, function transfer can directly account for these by using the relationship between characteristics and the benefit estimate. Bateman et al. review practical examples of benefit function transfer, and conclude that in some cases the transferability of the benefit function transfer is rejected. Brouwer [28] shows that the errors in transferring monetary value estimates for seemingly similar environmental goods over sites can be as large as 56% in the case of average unit value transfer and 475% in the case of benefit function transfer. Nevertheless, carefully designed benefit functions can yield useful results, and Bateman [29] suggest that the application of GIS can substantially benefit function transfer through improved and systematised data access.

5. Valuing coasts and oceans: practical examples

In general terms, ecosystem degradation and loss are often due to a failure to appreciate the full value of beneficial functions provided by such systems. In oceans and coastal zones, these include: sustaining biodiversity, storage of sediment, flood defence and storm buffering, maintenance of water quality and supporting commercial coastal and marine food chains. There has been a multiplicity of recent valuation studies on coastal and ocean resources (see for example [25,5,31] for an overview). Appendix A summarises some of these studies, going back 30 years, and classifies them in terms of ecosystem function and use, spatial scale, and valuation method. Values were converted to international dollars using the Purchase Power Parity (PPP) conversion factor developed by the World Bank (see for example [32]), and normalised to $2000 for ease of comparison. Some studies in developing countries were not included because of lack of PPP data. It is immediately apparent from the sample presented in Appendix A that there is a heavy bias towards studies focusing on recreational benefits, and located in the USA. Although not all valuation results can be compared directly, the range of studies examined allows to derive a range of estimates for some categories. The recreational value of beaches, for example, vary from 1.75 to 56 $2000 per person and per visit, and seem to have a lower value in the UK than in the US. Studies on valuing benefits from improved water quality show reasonably consistent values ranging from 66.8 to 263.5 $2000 per household and per year. Commercial fisheries can range from a value from 3.4 to 260 million $2000 per year, the most productive systems appearing to be coral reefs. Coastal and ocean resources also sustain biodiversity, and the review sample show values from 35.4 to 181.2 $2000 per person per year for a variety of habitats and species. Finally, we were only able to include two studies providing total economic value of ecosystems due to lack of data. Both the Indonesian coral reef and the Louisiana coastal wetland reveal high values per acre, of between 1.16 million to 3.90 million per km^2, to be compared to use values (direct and indirect) from 744,800 to 880,000 $2000/$km^2$ for mangroves, for example, suggesting significant non-use values. Figures vary according to ecosystems and functions evaluated and scale, but the results are reasonably consistent and of the same order of magnitude per category, generally confirming that oceans and coasts generate extremely high private and social values.

Within a sustainable development context, the next question is how these values can usefully inform decision-making. The four examples presented below were chosen to illustrate valuation approaches in a range of contexts and at a variety of geographical and temporal scales, and also to demonstrate how economic valuation can be used to aid decision-making through different approaches. The case study in the Mediterranean investigates non-use values for a threatened species in the context of a conservation scheme, and shows that even people with economic motives contemplate ethical issues in their construction of environmental values. The Vietnam mangrove case study presents a straightforward cost–benefit analysis, and illustrates how even when including solely market values, it can be economically desirable to rehabilitate coastal ecosystems. The study of nutrient pollution reduction in the Baltic looks at basin-wide value estimates and shows how valuation can be used in cost-effectiveness analysis when pollution reduction targets have been fixed, and/or in cost–benefit analysis when the targets themselves are to be determined. Finally, the example of a marine park in the Carribean illustrates how monetary economic values can be used in a wider multi-criteria analysis context incorporating stake-holder views.

5.1. Use and non-use values for conserving endangered species: the case of the Mediterranean Monk Seal and coastal wetlands

The Mediterranean Monk Seal (Monachus-monachus) is the most endangered seal in the world. The seals are killed by inshore fishermen because they damage fishing gear while trying to fish directly from their nets. In spite of numerous restrictions, regulations and conservation measures initiated by the Greek state, the problem should be seen as a real economic competition between fishermen and seals. Based on a belief that economics should play an important role in species conservation and following a two-year monitoring of the habitat and the population of the monk seal, the Mediterranean Monk Seal Project was proposed as a key contribution to a sustained conservation solution, involving a compensation scheme for the fishermen.

The CV method was used to estimate the willingness to pay (WTP) of respondents to financially support a public fund for the protection of the Mediterranean Monk Seal [33] (Langford in [5]). The fund would be used to compensate fishermen for damages caused by the seals in the hope that they would no longer have an incentive to kill the seals. The survey was designed and carried out on the island of Lesvos, the surrounding waters of which hosts one of the three biggest Mediterranean monk seal subpopulations in the Aegean. Because of the lack of previous estimations which would have supplied a reliable range of WTP, a dichotomous choice elicitation format [34] could not be used; the survey was therefore designed as an open-ended questionnaire trying to elicit with face-to-face interviews the maximum WTP of the respondents.

In order to understand individual's motives for their WTP choices, and not to bound the study too firmly in the concept of anthropocentric instrumental value, a series of questions were asked, scored on a five-point Likert-type scale regarding the individuals attitudes and beliefs about the possible extinction of the monk seal and environmental issues in general. These questions were again limited by the nature of the questionnaire survey in both number, length and complexity, but represented an attempt to gain some insight into individuals interpretative repertoires, in a somewhat general and arguably superficial sense, via the survey procedure. In addition, respondents were asked about their previous exposure to and knowledge of monk seals.

The statistical analysis of WTP responses demonstrated that there are a wide range of opinions which individuals hold towards preservation of the monk seal, from protection and moral considerations as largely esoteric evaluations of the monks seal issue to more practical and pragmatic views on competition between human and monk seals for limited resources and tourism potential. Responses to willingness to pay questions are associated to some degree with these attitudinal variables, as well as socio-demographic factors, and even interview setting and interviewer attitudes to the respondents. This suggests that the motivations for stating particular WTP amounts are complex and context dependent. However, the low proportions of variation explained also suggest that many other unmeasured variables may be influencing WTP strategies for individuals. A recent study by some of the authors on perception of risks from polluted coastal bathing waters in the UK suggests that psychosocial factors such as self efficacy, behavioural expectations, importance values, critical incidents and worldviews influence not only perception of risk but willingness to pay for risk reduction [35,36].

Another factor is that in this study, only outcomes have been measured, in terms of attitudinal statements and stated WTP. A more in-depth analysis is required, using qualitative data obtained from depth interviews and focus groups, both in individual cognitive terms and with reference to social and cultural contexts, to explain the process of attitude formation and justification, and the expression of these through WTP. This work was not possible with the monk seal study, but has been carried out with reference to conservation of an internationally important wetland site at Kalloni Bay on the island of Lesvos [37]. Results from this study suggest that different stakeholders (for example, farmers, fishermen, hoteliers) build definite strategies towards conservation and development based on their own interests but also their perception of others, including competing stakeholders and institutions such as the Greek Government and the European Commission, as well as their attitudes towards the environment. In this study, fishermen were interviewed, and their attitudes showed quite a wide diversity, reflecting different worldviews and moral positions towards the monk seals. This may be in part because only a few fishermen work full-time in the industry, with most supplementing their income with agricultural and other activities to greater or lesser degrees. Hence, they are not completely dependent on fishing for their livelihood. However, this study has shown that even people with potential economic motives for self interest regarding the monk seals do consider wider issues and contemplate the moral, ethical and temporal implications of their choices in relation to the natural environment. This suggests that further detailed research on nonuse values would be of great value to sustainable development policy analysis.

5.2. Cost-benefit analysis of mangrove planting in coastal Vietnam

In this case study, authors examine costs and benefits of mangrove rehabilitation in areas where they have been lost in coastal Vietnam [38–40]. The study focuses on

Table 3
Costs and benefits of direct and indirect use values of mangrove restoration compared

Discount rate	Direct benefits (PV million VND per ha)	Indirect benefits (PV million VND per ha)	Costs (PV million VND per ha)	Overall B/C ratio
3	18.26	1.40	3.45	5.69
6	12.08	1.04	2.51	5.22
10	7.72	0.75	1.82	4.65

Source: [39]. *Note*: US$1 = VND 11,000; B/C ratio = NPV total benefits/NPV costs.

direct and indirect use benefits as these are the most relevant aspects of value for local decision-making, and for the differential impacts of global change. Authors argue that option and existence values may also exist for mangroves, but often accrue at a more global scale, and would not be normally associated with local management decisions. The direct benefits of rehabilitation include: the value of the timber, as it becomes available through the first rotation of the mangrove stands; and other locally used products including shellfish and crabs, and honey from bee-keeping. The major indirect benefit is the role of mangroves in protecting the extensive sea-dike systems present along much of the low-lying deltaic coast of northern Vietnam. In the study, this indirect benefit (valued at replacement cost through work days saved) is estimated through a model using parameters including the width and age of the stand, and the local hydrological features. The deterministic model is calibrated for the area, and gives plausible results for regular maintenance costs. The model does not consider the impact of global change, such as change in the incidence of severe storms, or of mean sea level rise.

The results of the cost benefit analysis are presented in Table 3. The calculations compare only establishment and extraction costs, with the direct benefits from extracted marketable products, and the indirect benefits of avoided maintenance of the sea dike system. Biodiversity values or values of the links to offshore fisheries are not included in the analysis.

The results show a benefit to cost ratio of 4–5, for a range of discount rate, which indicates that mangrove rehabilitation can be justified on economic grounds for that range of discount rates. Sensitivity analysis shows that mangrove rehabilitation would be desirable from an economic perspective based solely on the direct use benefits of local communities.

5.3. Nutrient pollution in the Baltic drainage basin—costs and benefits

A concerted attempt by a consortium of European researchers was made to estimate the costs and economic benefits of environmental improvements in the Baltic drainage basin [41,42]. A cost-effectiveness analysis was first carried out, to determine how to reach reductions in the nutrient load to the Baltic sea specified by international conventions. Measures involved the agricultural sector, sewage treatment plants, wetland restoration and traffic and other nitrogen oxides emission sources. Marginal costs of these measures for nitrogen and phosphorus reductions were calculated for all countries in the drainage basin that had coastal zones coincident with the Baltic Sea. The relationship between possible nutrient reduction targets and associated minimum costs for their achievement was thus derived. Although the results relied on some simplistic assumptions and suffered from missing information such as the retention and leaching of nitrogen and phosphorus, some general lessons could be learnt from the study: nitrogen costs were much higher than phosphorus costs for the same percentage reductions; there are rapid increases in costs at reduction targets exceeding 40–45% reductions; the cost of simultaneous reductions in both nitrogen and phosphorus loads is less than the cost of separate reductions. The cost-effective allocation of measures for a 50% reduction reveals that for nitrogen reductions, sewage treatment plants in the entire baltic sea drainage basin account for about 33% of the reduction, wetland restoration contribute 33%, and the agricultural sector contributes mainly by reduction in nitrogen fertilisers, the cultivation of other crops, and changed practices for manure treatment. For Phosphorus, sewage treatment accounts for 80% of reductions, and wetland restoration for 15%. A uniform 50% reduction also implies that the highest burdens are carried out by Poland, Latvia, Lithuania, Estonia and Russia, raising compensation issues.

On the benefit valuation side, a total of 14 empirical valuation studies in three countries—Poland, Sweden, and Lithuania—were carried out to look at benefit estimation issues. These included the total economic value of reducing the effects of eutrophication, as well as sub-components of this total value such as: beach recreation benefits; existence and option values of preserving species and their habitats; and the benefits from preserving and restoring wetlands. The willingness to pay (WTP) data obtained allowed aggregate estimates for the three countries, but also more controversially for the two groups of economies around the Baltic Sea, in order to give total basin wide benefit estimates (Table 4). While the studies outlined provide a large amount of information about the value of the Baltic's resources, there are still gaps in our knowledge of total basin wide benefit estimates. The data on benefits corresponds to that which people can perceive, and so the benefits are 'total' only in a special sense. It is extremely difficult to communicate a detailed description of all effects of reduced eutrophication, which means that the perceived benefits from reduced eutrophication may differ from the total ones. A further problem is the incomplete scientific knowledge of eutrophication and its effects. Nevertheless, the estimates that are available indicate the significant value of the limited number of resource types considered (Table 5).

The full results of all the studies are presented in [43,44]. For illustration, we describe the study carried out in Sweden, which was a contingent valuation study focusing on Baltic Sea use and non-use values in Sweden. This study was designed as a mail questionnaire survey. The questionnaire was sent to about 600 randomly selected adult Swedes. The response rate turned out to be about 60%, which is quite similar to other contingent valuation mail questionnaire surveys that have been undertaken in Sweden. The respondents were asked to assume that an action plan

Table 4
Basin wide benefit estimates of environmental improvement in the Baltic

Country	GDP per capita at PPP[a] (US$)	Annual WTP per person (SEK)		National WTP, year 1 (MSEK)		National WTP, present value[b] (MSEK)		National WTP, present value per year (MSEK)	
Transition economies									
Estonia	3823	*355*	(375)	*401*	(423)	*4248*	(4476)	*212*	(224)
Latvia	3058	*284*	(257)	*549*	(497)	*5816*	(5260)	*291*	(263)
Lithuania	3632	*337*	(182)	*883*	(477)	*9355*	(5050)	*468*	(253)
Poland	4588	*426*	(300)	*11,136*	(7842)	*117,974*	(83,077)	*5899*	(4154)
Russia	4970	*461*	(246)	*3340*	(1782)	*35,384*	(18,883)	*1769*	(944)
Market economies									
Denmark	19,306	*3790*	(3515)	*13,080*	(12,131)	*138,570*	(128,514)	*6929*	(6426)
Finland	15,483	*3040*	(2229)	*11,414*	(8369)	*120,920*	(88,661)	*6046*	(4433)
Germany	18,541	*3640*	(3334)	*8848*	(8104)	*93,736*	(85,852)	*4687*	(4293)
Sweden	16,821	*3300*	(3000)	*21,882*	(19,893)	*231,818*	(210,750)	*11,591*	(10,537)
Total				*71,533*	(59,518)	*757,821*	(630,523)	*37,892*	(31,527)

Source: [42]. *Note*: Figures in brackets are for benefit figures derived using Polish mean WTP of SEK 300 and Swedish mean WTP estimate of SEK 3000.
[a] PPP—purchasing power parity.
[b] Time horizon: 20 years (specified in the CVM studies). Discount rate: 7% (this rate was also used in the estimation of nutrient reduction costs).

Table 5
Composition of value elements for selected ecosystems

Coral reefs		Mangroves $ per ha per year	
Coastal protection	2750	Coastal protection	1839
Waste treatment	58	Nutrient cycling	6696
Food	259	Food	797
Production/biol		Production/biol	
Control		Control	
Recreation	3008	Recreation	658
Total	6075	Total	9990

Source: [25].

against eutrophication had been suggested, and that this action plan would imply that the eutrophication in 20 years would decrease to a level that the Baltic Sea can sustain. The types of action that this plan would involve were briefly described. It was also explained that the way to finance the actions would be to introduce an extra environmental tax in all countries around the Baltic Sea.

In order to calculate basin wide benefit estimates, the values for the different activities carried out had to be added up, taking care not to double-count, and using the relevant correct populations. Since there are benefit estimates available for the same valuation scenario in only two of the 14 countries that are included in the Baltic

Drainage Basin, any aggregation to the whole basin has to rely on strong assumptions. The aggregate benefit estimates should thus not be taken too literally. However, they may give useful information regarding the order of magnitude of basin wide benefit estimates.

The costs of pollution abatement and related economic benefit estimates were then brought together in a cost–benefit analysis framework. The results showed that there is considerable merit in the adoption of a basin-wide approach to pollution abatement policy in the Baltic and therefore in the implementation of an integrated coastal zone management strategy. Despite the pioneering nature (i.e. in the 'transition' economies) of some of the economic benefits research, there seems to be little doubt that a cost-effective pollution abatement strategy roughly equivalent to the 50% nutrients reduction target adopted by the Helsinki Commission would generate positive net economic benefits (benefits minus costs). Results also indicated that a policy of uniform pollution reduction targets is neither environmentally nor economically optimal. Rather, what is required is a differentiated approach with abatement measures being concentrated on nutrient loads entering the Baltic proper from surrounding southern sub-drainage basins. The northern sub-drainage basins possess quite effective nutrient traps and contribute a much smaller proportionate impact on the Baltic's environmental quality state. The market economy countries such as Sweden, within whose national jurisdiction some of the southern sub-basins lie are also the biggest net economic gainers from the abatement strategy.

5.4. *Trade-off analysis in the management of a marine park in the Caribbean*

A recent study in Tobago, West Indies applied a 'trade-off analysis' approach to determine the potential trade-offs between users of a marine protected area [45]. The study incorporated multiple objectives for protected area management, namely economic growth, social well-being and environmental health within a decision-making framework. The authors used complementary approaches including stakeholder analysis to derive criteria for decision-making and weights attached to them, to be used in a multi-criteria analysis (Table 3). The results of the analysis were discussed in a consensus-building workshop.

Social, economic and ecological criteria were defined in consultation with stakeholders. Economic criteria included tourism benefits, which were estimated through a contingent valuation study. The contingent valuation survey estimated the total WTP of visitors to south west Tobago (both users and non-users of BRMP), based on a randomised survey of 1000 visitors and residents. The survey revealed how much the sample was WTP to prevent further deterioration in the quality of Buccoo Reef. The equivalent surplus generated from visitor and resident use of BRMP was estimated for the existing level of environmental quality, and at different levels of environmental quality. Both open-ended and dichotomous choice questions were used in order to estimate two final values for mean WTP. The mean WTP by all respondents ranged from US $3.70 to $9.30. An annual equivalent surplus generated under each scenario was estimated, and then extrapolated over a 10-year period and discounted at a rate of 10%, to ensure comparable values with other criteria. To determine the effects of different scenarios on WTP, respondents were asked if they would still be WTP under different scenarios of changes in the level of tourism development and environmental management. The results suggest that the Net Present Value of the Buccoo Reef Marine Park (BRMP) may range between US $2.5 and $3.7 million, under different future scenarios.

This example shows how valuation studies can be used in a wider decision-making framework, which includes both qualitative and quantitative information. Here, the multi-criteria analysis is used in a process-oriented rather than outcome-oriented use. It facilitates the deliberation of stakeholders through supporting a process by which diverse stakeholders can examine information on criteria and impacts, and explore the outcomes and impacts of decisions made, through applying different weights to economic, social and ecological criteria. Scenario B was ranked higher across the range of weighting from all stakeholders. There were differences, however, between different stakeholder weightings in the subsequent ordering of scenarios. Although the authors did not tackle the problem of aggregating different stakeholders' weightings, they argue that the inclusion of stakeholder views and values within a rigorous framework such as MCA can, potentially, provide rich information for regulators seeking to manage resources in partnership with other stakeholders.

6. Conclusion

Moving towards sustainable use of coastal resources means that we must take stock of the resources we have, determine the full range of costs and benefits that management options provide and develop flexible policies accordingly [46]. Valuing ocean and coastal resources can provide significant insights into decision-making with sustainable development goals. Even when environmental sustainability standards are advocated, it is still necessary to quantify the opportunity costs of such standards, and in general to compare the costs of current and prospective protection measures. The critical importance of making value-laden assumptions transparent in sustainable development policies needs to be highlighted.

Valuation studies were undertaken before the earth Summit, but Agenda 21 has pushed the need for the integration of environmental and economic dimensions in decision-making to the foreground, and encouraged research on economic approaches of sustainable development. Valuation studies have been used by local authorities, national governments and international organisations more extensively since, as summarised in Appendix A, providing a useful indication of the range of estimates for different categories of values.

The emphasis of the earth summit on various issues of environmental and economic integration has influenced economic valuation in a variety of ways. The perspective of risk and uncertainty has lead to better tools to deal with these aspects in decision-making [47]. Sensitivity analysis and scenarios are used more and more frequently in cost–benefit analysis and in other approaches. Agenda 21 also emphasised the need to include stakeholders at a variety of stages in decision-making. This has lead to a whole range of alternatives to cost–benefit analysis, but also to the refining of valuation methods such as contingent valuation to make them more inclusionary in nature, with the use of focus groups and other devices to engage with relevant stakeholders.

One of the most important aspects of the Summit was the introduction and widespread dissemination of the notion of precautionary principle. The impact in decision-making was to shift the emphasis somewhat from classical cost–benefit analysis to approaches such as SMS. But the latter still has a cost dimension and therefore some quantification of the damage costs avoided in monetary terms will be useful in the political economy context.

Human welfare depends on ocean and coastal resources in many ways. Further research is needed to understand the basic functions which ecosystem provide and translate these into the various socio-economic and cultural (monetary and non-monetary) values to society. Although techniques have gained in reliability through numerous empirical applications, and more refined statistical tests and controls have been introduced, economic valuation approaches are still controversial. Issues such as scale and aggregation, risk and uncertainty, and benefit transfer continue to pose challenges to environmental economists. More benefit transfer studies are needed to avoid the need for too many costly original valuation exercises. A combination of quantitative and qualitative methods can in general be advocated in order to generate a blend of different types of policy relevant information. It is also useful to emphasise that evaluation is more than just the assignment of monetary values and includes multi-criteria methods and techniques in order to identify practicable trade-offs. The latter, however, are themselves not free of technical limitations and should not seek to shroud subjective judgements behind a veil of technical analysis.

Finally, it is worth emphasising that the section of Agenda 21 on integration of environment and development in decision-making also underlines the need for a broad range of analytical methods to provide a variety of points of view. Sustainable development requires mechanisms to allow holistic decision making. Economic valuation is an important but not the only component of this process. Capacity building for sustainable development should include more practical applications of economic valuation methods, but also dissemination of decision-making systems integrating economic, social and natural science components.

Acknowledgements

The support of the Economic and Social Research Council (ESRC) is gratefully acknowledged. This work is part of the interdisciplinary research programme of the ESRC Research Centre for Social and Economic Research on the Global Environment (CSERGE). This review paper was written for the Global Conference on Oceans and Coasts at Rio + 10: Towards the 2002 World Summit on Sustainable Development; Assessing Progress, Addressing Continuing and New Challenges, UNESCO, Paris, 3–7 December 2001. The authors thank the participants for their feedback. This article was written in the context of a European project on integrated catchment-coastal zone management, EUROCAT, Project No

Appendix A. Summary overview of valuation studies on ocean and coastal resources

Function/use	System	Spatial scale	Country	Author(s)	Title	Bibliographical details	Publication year	Issue addressed in study/General Function-Use Identification	Technique	Year data	Mean/total	Measurement unit	Equivalent int. $2000
Recreation	Beaches	Regional	USA	Bell, F.W. and V.R. Leeworthy.	"Recreational Demand by Tourists for Saltwater Beach Days,"	*Journal of Environmental Economics and Management*, *18*, *189–205*.	1990	Recreation beach days for out of state tourists. Function-use: Recreation.	TC	1984	33.91.	$/person/day.	56.2
	Beaches and lake	Regional	USA	Bockstael, N.E., W.M. Hanemann, and C.L. Kling.	"Estimating the Value of Water Quality Improvements in a Recreational Demand Framework,"	Water Resources Research, 23 (5), 951–960.	1987	Valuation of water quality improvements Function-use: recreation.	TC, DCM	1975	DCM-0.50/12.04 TCM: $450/visit	In dollars, per visit, per choice occasion and per season.	1.75–42; 1570
	Beaches	National	United Kingdom	Green, C.H. et al.	"The benefits of coastal protection: Result form testing the CVM for beach recreation,"	Annual Conference of River and Coastal Engineers, Loughborough University, Loughborough.	1990	Recreational value of beaches.	CV		WTP: 4.9.	Pounds per annum per capita.	4.03
	Lakes, river and beaches	Regional	Ireland	O'Neill, C.E. and J. Davis.	"Alternative Definitions of Demand for Recreational Angling in Northern Ireland,"	*Journal of Agricultural Economics*, *42 (2)*, *174–179*.	1991	TC-study of aggregate demand for recreational angling.	TC	1988	Estimated user benefits. 9.1–22.21	Millions of UK Pounds.	14.54–35.43 million
	Beaches	National	United Kingdom	Penning-Rowsell et al.	"Recreational aspects of coast protection benefits,"	Paper presented to the Conference of River and Coastal Engineers, Loughborough University, Loughborough.	1989	Beach replenishment.	CV		7.8.	£/adult visit.	6.53
	Sea, Beaches	Local	USA	Silberman, J., D.A. Gerlowski, and N.A. Williams.	"Estimating Existence Value for Users and Nonusers of New Jersey Beaches,"	*Land Economics*, *68 (2)*, *225–236*.	1992	Use and non-use value for beach nourishment.	CV	1985	9.34–15.21	$ per visit.	14.95–24.34
	Sea, Beaches	Regional	USA	Silberman, J. and M. Klock.	"The Recreation Benefits of Beach Renourishment,"	*Ocean and Shoreline Management*, *11*, *73–90*.	1988	Recreational benefits associated with beach renourishment.	CV	1988	Mean WTP per person per beach day for use of the renourished beach is reported: 3.9.	$/person/trip.	5.68

Sea, Beaches	Regional	USA	Smith, V.K., X. Zhang, and R.B. Palmquist.	"Marine Debris, Beach Quality, and Non-Market Values,"	*Environmental and Resource Economics, 10, 223–247.*	1997	Economic importance of controlling marine debris to enhance the quality of beaches.	CV	1991	21.38–72.18	$ per yr.	27.03–91.26
SNF: beaches and river.	Regional	USA	Loomis, J.B.	"A Bioeconomic Approach to Estimating the Economic Effects of Watershed Disturbance on Recreational and Commercial Fisheries,"	*Journal of Soil and Water Conservation, 44(1), 83–87.*	1989	Change in catchable fish populations due to watershed disturbances from road building and timber harvesting.	TC	SNF: 1981; GNF: 1980–1984 and 1987.	Economic value of the lost fish or recreational and commercial anglers is $2 million (30-year period).	In dollars and per trip.	3.14 million
Pond	Local	USA	Anderson, G.D. and S.F. Edwards.	"Protecting Rhode Island's Coastal Salt Ponds: An Economic Assessment of Downzoning,"	*Coastal Zone Management Journal, 14 (1l2), 67–91.*	1986	Economic benefits of coastal amenities (swimmable water).Function Use: Recreation (fishing, swimming, wildlife)	HP	1983	11.16–101,71 (frontage); 4275 to over 20000 (view)	Per foot (frontage) and per house (view)	13.3–121; 5090–23800
Coastal wetlands	Regional	USA	Bergstrom, J.C., J.R. Stoll, J.P. Titre, and V.L. Wright.	"Economic Value of Wetlands-Based Recreation,"	*Ecological Economics, 2, 129–147.*	1990	Wetlands loss and recreational value. Function-use: recreation (fishing and hunting)	CV	1986–1987.	360/$27, 360,000.	$/user	566
Coast	Regional	USA	Downing, M. and T. Ozuna Jr.	"Testing the Reliability of the Benefit Function Transfer Approach,"	*Journal of Environmental Economics and Management, 30, 316–322.*	1996	Test of benefit transfer using CV. Function-Use: Recreation.	CV	1987, 1988, 1989	2,195–3,404.	$/yr.	3050–4730
Coast	Local	Uruguay	McConnell, K.E. and Ducci, J.H.	"Valuing Environmental Quality in Developing Countries: Two Case Studies	Paper presented at AERE session on contingent valuation surveys in developing countries, American Economics Association meetings.	1989	WTP for water quality improvement at local beach	CV		14	$/household	20.38
Bay	Regional	USA	Lindsey, G.	"Market Models, Protest Bids, and Outliers in Contingent Valuation,"	*Journal of Water Resources Planning and Management, 20 (1), 121–129.*	1994	Estimation of total benefits of nutrient reduction. Recreation value.	CV	1989	Unadjusted mean annual bid (bench mark): $42.00.	In dollars per year and per person.	58.3
Bay	Regional	USA	Strand, I.E., N.E. Bockstael and C.L. Kling.	"Chesapeake Bay Water Quality and Public Beach Use in Maryland,"	Paper presented at the Conference on Economics of Chesapeake Bay Management II,	1986	Water quality improvement of the Chesapeake Bay for water related recreation.	CV		(a.: 72;(users) (b. 34. (non users)	$/Person/yr.	115.23; 54.41

Appendix *Continued*

Function/use	System	Spatial scale	Country	Author(s)	Title	Bibliographical details	Publication year	Issue addressed in study/General Function-Use Identification	Technique	Year data	Mean/total	Measurement unit	Equivalent int. $2000
						Annapolis, Maryland, May 28–29, 1986.							
	Bay	Regional	USA	Wang, H.	"Treatment of "Don't-Know" Responses in Contingent Valuation Surveys: A Random Valuation Model,"	*Journal of Environmental Economics and Management, 32, 219–232.*	1997	Improving the environmental quality of Galveston Bay, Texas.	CV	1994	2.647–11.860	$/months	3.1–13.8
	Sea (coast)	Regional	USA	Agnello, R.J.	"The Economic Value of Fishing Success: An Application of Socio-economic Survey Data"	*Fishery Bulletin, 87 (1), 223–232.*	1988	Valuing fishing success to marine recreational sport anglers targeting three Atlantic coast species.	TC	1987	4.66.	$/additional fish per trip.	7.06
	Sea	Regional	USA	Bockstael, N.E., K.E. McConnell, and I.E. Strand.	"Benefits from Improvements in Chesapeake Bay Water Quality (Volume II),"	Washington, D.C.: Office of Policy and Resource Management, US Environmental Protection Agency, USA.	1988	Benefits from cleaning up Chesapeake Bay	CV	1984	Annual WTP of users: 121.	$/household/yr.	201
	Sea	Regional	USA	Bockstael, N.E., K.E. McConnell, and I.E. Strand.	"A Random Utility Model for Sportfishing: Some Preliminary Results for Florida,"	*Marine Resource Economics 6, 245–260.*	1989	Recreational shore and private boat fishing.	TC	1988	(a. WTA: 3.64; (b. WTP: 0.32.	$/person/trip.	5.3;0.47
	Sea, coast	Regional	Tobago	Brown, K., Adger, N., Tompkins, E., Bacon, P., Shim, D., Young, K.	"Trade-Off Analysis for Marine Protected Area Management,"	*Ecological Economics, 37, 417–434.*	2001	WTP to prevent further deterioration in the quality of a coral reef	CV		3.70–9.30	$/household	3.97–9.98
	Sea, Coast	Local	United Kingdom	Turner, R.K. and J. Brooke.	"A benefits assessment for the Aldeburgh Sea defence scheme,"	In J.-Ph. Barde and D.W. Pearce (eds.) Valuing the Environment, Ch. 6, Earthscan, London.	1988	Coastal recreation and amenity.	CV		Mean WTP: (a. locals: 15; (b. non-locals: 18.8.	Pounds per household per annum.	40.38–50.61
	River, sea	Regional	Philippines	Choe, K., D. Whittington, and D.T. Lauria.	"The Economic Benefits of Surface Water Quality Improvements	*Land Economics, 72 (4), 519–537.*	1996	Value of improving water quality of nearby rivers and	CV	1992	37	Pesos/dollar per month per	6.1

				in Developing Countries: A Case Study of Davao, Phillippines,"			sea. Function-use: recreation				househ.	
Sea	Regional	USA	Falk, J.M. and V.C. Crouse.	"Beach Management Survey: An Examination of Attitudes and Concerns of Coastal Property Owners, Resort Merchants and Realtors in Sussex County, Delaware,"	Newark: DEL-SG-05-88, Sea Grant College Program, University of Delaware, USA.	1988	Concerns of coastal property owners (PO), resort merchants (RM) and Sussex County (SCR) realtors regarding erosion issues and beach management practices.	CV	1987	PO: $200-$250	In dollars, per year and per person.	303–379
Sea	Regional	USA	Falk, J.M., A.R. Graefe, and M.E. Suddleson.	"Recreational Benefits of Delaware's Public Beaches: Attitudes and Perceptions of Beach Users and Residents of the Mid-Atlantic Region,"	Newark: DEL-SG-05-94, report prepared for Delaware Department of Natural Resources and Environmental Control Division of Soil and Water Conservation and the US Army Corps of Engineers, Philadelphia District, University of Delaware, USA.	1994	Beach users' and non-beach users' WTP for using Delaware beaches, willingness to contribute to a voluntary annual beach protection fund Function-Use: Recreation.	CV	1993	On-site: overall average: daily fee for natural beach/enhanced beach/ voluntary beach protection fund (in $): 3.01/ 3.70/63.69.	In dollars, per person and per visit and per yr.	3.59–4.41; 75.9
Sea	Regional	USA	Lindsay, B.E., J.M. Halstead, H.C. Tupper, and J.J. Vaske.	"Factors Influencing the Willingness to Pay for Coastal Beach Protection,"	*Coastal Management*, *20*, *291–302*.	1992	To examine coastal beach visitors' WTP for a beach erosion control program.	CV	1988	Mean of FBP (maximum WTP value): $30.85.	In dollars, per year and per person.	44.9
Sea	Regional	USA	McConnell, K.E.	"The Damages to Recreational Activities from PCBs in the New Bedford Harbor,"	Rockville, MD: Ocean Assessment Division, National Oceanic and Atmospheric Administration, University of Maryland, USA.	1986	To estimate the present value of damages to beach use.	TC	1986	$11.4 million (=without capacity contraints) and $8.3 (=with capacity contraints).	In dollars, per year and per household.	13.28–18.24
Sea	Local	Norway	Navrud, S.	"Social profitability of liming River Audna,'	Report from the Directorate for Nature Management.	1991	Recreational value of angling.	TC, CV		(a. TC: 27–56; (b. CV: 40-65.	NOK per angler per day.	3.65–7.58; 5.42–8.80
Sea	Regional	USA	Parsons, G.R. and	"The Opportunity Cost	*Land Economics*, *67*	1991	Value of lost coast-	HP	1985	3.3	$ per	5.71

Appendix *Continued*

Function/use	System	Spatial scale	Country	Author(s)	Title	Bibliographical details	Publication year	Issue addressed in study/General Function-Use Identification	Technique	Year data	Mean/total	Measurement unit	Equivalent int. $2000
				Y. Wu.	of Coastal Land-Use Controls: An Empirical Analysis,"	*(3), 308–316.*		al access amenities due to land-use controls.				house.	
	Sea	Regional	USA	Roberts, K.J., M.E. Thompson, and P.W. Pawlyk.	"Contingent Valuation of Recreational Diving at Petroleum Rigs, Gulf of Mexico,"	*Transactions of the American Fisheries Society, 114, 214–219.*	1985	Economic value of sport diving around the many petroleum structures located off Louisiana's coast.	CV	1982	$163.	In dollars, per year and per person.	291
	Sea	Local, regional	USA	Shaikh, S. and D.M. Larson.	"Methods for Combining Travel Cost and Contingent Valuation Data,"	Report prepared for the World Congress of Environmental and Resource Economists, Venice, Italy.	1998	Two-constraint recreation demand model, which uses combined TC and CV data.	CV	Winter 1991–1992.	Max money WTP: $13.79/$19.56; max time WTP: $9.92/$11.76 50% increase	$.	16.93–24.01; 12.18–14.36
Storm protection	Coastal Wetland	Regional	USA	Farber, S.	"The Value of Coastal Wetlands for Protection of Property Against Hurricane Wind Damage"	*Journal of Environmental Economics and Management, 14(2), 143–51.*	1987	Coastal wetland value for protection against hurricane damage (indirect use)	MV	July 1984 to June 1985.	6.82–22.94	$/acre	11.3–38
Water Quality	Lakes and Bays	Regional	USA	Ribaudo, M.O. and J.E. Epp.	"The Importance of Sample Discrimination in Using the Travel Cost Method to Estimate the Benefits of Improved Water Quality,"	*Land Economics, 60 (4), 397–403.*	1984	Benefits from improvement in environmental quality	TC	1982	Mean level of benefits for current users: $123.00; for former users: $97.00.	In dollars, per year and per number of trips.	161–204
	Coastal pond	Regional	USA	Kaoru, Y.	"Differential Use and Non Use Values for Coastal Pond Water Quality Improvements,"	*Environmental and Resource Economics, 3, 487–494*	1993	Components of the total value of coastal pond water quality improvements. Function-use:water quality	CV	1987	Total WTP for the water quality improvements: $131.03	Dollars, years.	199
	Coast	Regional	USA	Hayes, K.M., Tyrrell, T.J., Anderson, G.	"Estimating the Benefits of Water Quality Improvements in the Upper Narragansett Bay"	*Marine Resource Economics, 7(1), 75–85.*	1992	WTP for water quality improvement	CV		80–187	$/household	105–246

	Sea	Regional	USA	Carson, R.T., W.M. Hanemann, R.J. Kopp, J.A. Krosnick, R.C. Mitchell, S. Presser, P.A. Ruud, and V.K. Smith, with M. Conaway and K. Martin.	"Temporal Reliability of Estimates from Contingent Valuation,"	Discussion paper No. 95-37, Resources for the Future, Washington D.C., USA.	1995		Temporal reliability of CV estimates Function-use: water quality.	CV	1991 and 1993.	Original 1991 study: WTP \$52.80	In dollars, per year and per household.	66.8
	Sea	Regional	United Kingdom	Georgiou, S., I.H. Langford, I.J. Bateman, and R.K. Turner.	"Determinants of Individuals' Willingness to Pay for Perceived Reductions in Environmental Health Risk: A Case Study of Bathing Water Quality,"	*Environment and Planning A*, *30*, *577–594*.	1998		This study calculated the WTP for perceived reduction in environmental health risks.	CV	1995	WTP questions (95% confidence intervals): 12.64 (9.00:16.28)	£.	8.57
	Sea	Local	Norway	Magnussen, K. and S. Navrud.	"Valuing reduced pollution of the North Sea,"	Report B-015-92. Norwegian Agricultural Economics Research Institute.	1992		WTP for water quality improvement.	CV		Average WTP: 1500–2000.	NOK per household.	197.61–263.48
	Sea	Regional	USA	Niklitschek, M. and J. Leon.	"Combining Intended Demand and Yes/No Responses in the Estimation of Contingent Valuation Models,"	*Journal of Environmental Economics and Management*, *31*, *387–402*.	1996		Use and non-use values for improvements in water quality.	CV (including TC).	1992	8.46.	\$ per household per month.	10.38
	Sea	National	Sweden	Sandstrom, M.	"Recreational Benefits from Improved Water Quality: A Random Utility Model of Swedish Seaside Recreation,"	Working Paper No. 121, Stockholm School of Economics, The Economic Research Institute, Sweden.	1996		Benefits from reduced eutrophication of the seas around Sweden. Water quality	TC	1990–1994	240–540	In Swedish crownes per household and per trip.	28.18–63.41
Fishing	Bay	Local	USA	Anderson, E.	"Economic Benefits of Habitat Restoration: Seagrass and the Virginia Hard-Shell Blue Crab Fishery,"	*North American Journal of Fisheries Management*, *9*, *140–149*.	1989		Aquatic vegetation (seagrass) used as habitat by Virginia hard-shell blue crabs	SM		2,438,000	\$.	3,390,000
	Gulf	Regional	Mexico	Grant, W.E. and W.L. Griffin.	"A Bioeconomic Model of the Gulf of Mexico Shrimp Fishery,"	*Transactions of the American Fisheries Society*, *108(1)*, *1–13*.	1979		Bioeconomic simulation model of the Gulf shrimp fishery, assessing the impact of alternative management strategies on the harvest of shrimp.	SM		4.2	\$Millions/yr.	11.1

Appendix *Continued*

Function/use	System	Spatial scale	Country	Author(s)	Title	Bibliographical details	Publication year	Issue addressed in study/General Function-Use Identification	Technique	Year data	Mean/total	Measurement unit	Equivalent int. $2000
	Gulf	Regional	Mexico, USA	Griffin, W.L. and B.R. Beattie.	"Economic Impact of Mexico's 200-Mile Offshore Fishing Zone on the United States Gulf of Mexico Shrimp Fishery,"	*Land Economics, 54(1), 27–38.*	1978	Shrimp fishery.	MV	1973	1.74.	$/lb. of shrimp.	6.75
	Coral reef	Regional	Philippines	Hodgson, G., Dixon, J.A.	" Logging vs Fisheries and Tourism in Palawan"	East-West Environment and Policy Institute Occasional Paper No 7, East-West Centre, Honolulu.	1988	Loss of production value of coral reefs	MV		40 million	$ over 10 yr	60.63 million
	Coral reefs	Regional	Philippines	McAllister, D.E.	"Environmental, Economic and Social Costs of Coral Reef Destruction in the Philippines"	Galaxea, 7, 161–178	1988	Loss of production value of coral reefs	MV		160 million (domestically consumed fish) 11 million (exports)	$/yr	242/54 million 16.67 million
	Coastal wetland	Regional	USA	Lynne,G.D., Conroy, P., Prochaska, F.J.	"Economic Valuation of Marsh Areas for Marine Production Processes"	*Journal of Environmental Economics and Management, 8, 175–186.*	1981	Indirect use of coastal wetland in the production of blue crab	MV		3 (total present value)	$/acre	6.27
Habitats and species	Coastal (lagoon)	Local	France	Boisson, J.M. and M.A. Rudolff.	"Second-Thoughts on Long Term and Supra Long Term Valuation of Natural Assets in a CVM Application to the Filling of a Coastal Lagoon,"	*Montpellier: Faculté de Sciences Economiques, LAMETA (CNRS), Université de Montpellier I, France.*	1998	Context of the filling of a natural lagoon. Test difference between valuation of two types of natural assets, with use and bequest values (30 years) and without (100 years). Function-use: habitat	CV		FF52,72	FF.	57
	Sea	Regional	USA	Loomis, J.B. and D.M. Larson.	"Total Economic Values of Increasing Gray Whale Populations: Results from a Contingent Valuation Survey of Visitors and Households,"	*Marine Resource Economics, 9, 275–286.*	1994	Value of several different levels of increased gray whale populations to both users (whale-watchers) and nonusers (CA households).	CV		29.73.	$/Visitor/yr.	35.4

	Sea	National	USA	Samples, K.C., J.A. Dixon and M.M. Gowen.	"Information Disclosure and Endangered Species Valuation,"	*Land Economics*, *62(3)*, *306–312*.	1986	Humpback whale preservation.	CV		36.33–57.06	$/person/yr.	58.14–91.32
	Sea	Regional	USA	Samples, K.C. and J.R. Hollyer.	"Contingent Valuation of Wildlife Resources in the Presence of Substitutes and Complements,"	Ch. 11 in Economic Valuation of Natural Resources: Issues, Theory, and Applications. Rebecca L. Johnson and Gary V. Johnson (eds.) Boulder: Westview Press, 177–191.	1990	Endangered marine mammal preservation (humpback whale and Hawaiian monk seal).	CV		131	$/person.	181.92
	Sea	International	Netherlands	Spaninks, F.A., O.J. Kuik, and J.G.M. Hoogeveen.	"Willingness to Pay of Dutch Household for Natural Wadden Sea. An Application of the Contingent Valuation,"	Report No. E-96/6, Institute for Environmental Studies, Free University Amsterdam, The Netherlands.	1996	WTP of households in the Netherlands for measures needed to restore the Dutch Wadden Sea Area from its present state to its "natural state".	CV	1995	67.37	Dfl/yr.	82.74
	Sea	Regional	USA	Stevens, T.H., N.E. DeCoteau, and C.E. Willis.	"Sensitivity of Contingent Valuation to Alternative Payment Schedules,"	*Land Economics*, *73 (1)*, *140–148*.	1997	Benefits of restoration of Atlantic salmon Nature conservation.	CV	1994	1. 8.5–20 (Annual payments) 2. 25 (Lump Sum payments)	$.	9.88–23.24; 29.05
Multiple Functions	Mangrove	Local	Malaysia	Bennet, E.L., and Reynolds, C.J.	"The Value of a Mangrove Area in Sarawak"	*Biodiversity and Conservation*, *2*, *359–75*.	1993	Valuing damage to fisheries and tourism from destruction of mangrove area Function-use: fishing and recreation	MV		24.8 million	$/yr.	30.44 million
	Mangrove	Regional	Fiji	Lal, P.N.	"Conservation or Conservation of Mangroves in Fiji"	*Occasional Paper11. Environment and Policy Institute*, *East-West Centre*, *Honolulu*	1990	Direct and indirect use values of wetlands	MV		3,277 forestry 3,051 fishing	$/ha.	4,550 4,237
	Mangrove	Regional	Indonesia	Ruitenbeek, H.J.	"Modelling Economy-Ecology Linkages in Mangroves: Economic Evidence for Promoting conservation in Bintuni Bay, Sarawak"	*Ecological Economics*, *10(3)*, *233–247*.	1994	Direct and Indirect Use Value	MV		6,250 (Total present value)	$/ha.	7,448
	Mangrove	Regional		Christensen, B.	"Management and Utilization of Mangroves	FAO Environment Paper3, Food and	1982	Direct and indirect use values of man-	MV		318 (charcoal production)	$/ha/yr.	602; 4 36; 57; 189.4;

Appendix *Continued*

Function/use	System	Spatial scale	Country	Author(s)	Title	Bibliographical details	Publication year	Issue addressed in study/General Function-Use Identification	Technique	Year data	Mean/total	Measure-ment unit	Equivalent int. $2000
					in Asia and the Pacific."	Agriculture Organization of the United Nations, Rome.		groves Fishing and agriculture			230 (Nipa cultivation) 30 (interestuary esfishery) 100 (out-estuary fishery) 60 (Oyster culture) 206 (shrimp farming) 220 (rice cultivation)		113.7; 390.2; 416.3
	Coral reef	Regional	Indonesia	Cesar, H.	"Economic Analysis of Indonesian Coral Reefs"	Environment Department Working Papers, World Bank, Washington DC.	1996	Value of coral reef for tourism and fisherie S Function-use: fishing and recreation	MV		3,000–$503,000 (tourism) 2, 108,900 (fishery)	$/km^2 of reef	3,390–56,850 2,382,891
	Harbour, Bay	Local	France	Goffe, Ph., Le.	"The Benefits of Improvements in Coastal Water Quality: A Contingent Approach,"	*Journal of Environmental Management*, *45*, *305–317*.	1995	Non-market value which local people give to water quality in the harbour. Water quality, habitat	CV	1993	Mean WTP "salubrity" 214–215. Mean WTP "ecosystem" 158–162	FF per household per year.	256; 191
Total Economic Value	Coastal wetland	Regional	USA	Farber, S.	"Welfare Loss of Wetlands Disintegration: a Louisiana Study"	*Contemporary Economic Policy*, *14*, *92–106*.	1996	Total present value of coastal wetlands	MV, CV, etc.		8,437–15,763	$/acre $/km^2	9,530–17,800 2,084,800 3,895,000
	Coral reef	Regional	Indonesia	Riopelle, J.M.	"The Economic Valuation of Coral Reefs: a Case Studyof West Lombok, Indonesia"	Thesis, Dalhousie University, Halifax, Nova Scotia.	1995	Total value of coral reefs	Mixed		$1,000,000 (NPV)	$/km^2	1,160,000

Notes: Valuation technique. SM, simulation models; MV, market valuation; TC, travel cost method; DCM, direct choice model; CV, contingent valuation method; HP, hedonic price approach.

EVK1-CT-2000-00044. Finally, we would like to thank the authors of the case studies examined for allowing us to use them as illustrations.

References

[1] UNCED. The Earth Summit'92, United Nations Conference on Environment and Development, Rio de Janeiro 1992, London: The Regency Press, 1992.
[2] Grubb M, Koch M, Thomson K, Munson A, Sullivan F. The "Earth Summit" agreements: a guide and assessment. London: Earthscan, 1993.
[3] Cincin-Sain B. Earth Summit implementation: progress since Rio. Marine Policy 1996;20(2):123–43.
[4] Bower BT, Turner RK. Characterising and analysing benefits from integrated coastal management (ICM). Ocean & Coastal Management 1998;38:41–66.
[5] Turner RK, Bateman IJ, Adger N, editors. Economics of coastal and water resources: valuing environmental functions. Dordrecht: Kluwer Academic Publishers, 2001.
[6] Turner RK, Lorenzoni I, Beaumont N, Bateman IJ, Langford IH, McDonald AL. Coastal management for sustainable development: analysing environmental and socioeconomic changes on the UK coast. The Geographical Journal 1998;3:164.
[7] Pearce DW, Turner RK. Economics of natural resources and the environment. New York: Springer, 1990.
[8] Common M, Perrings C. Towards an ecological economics of sustainability. Ecological Economics 1992;6:7–34.
[9] Daly HE, Cobb JE. For the common good: redirecting the economy toward community, the environment, and a sustainable future. Boston: Beacon Press, 1989.
[10] Pearce DW, Markandya A, Barbier EB. Blueprint for a green economy. London: Earthscan, 1989.
[11] Turner RK, editor. Sustainable environmental economics and management: principles and practice. London: Belhaven Press, 1993.
[12] Daly H. On Wilfred Beckerman's critique of sustainable development. Environmental Values 1995;4(1):49–55.
[13] Westman RE. Ecology, impact assessment and environmental planning. Chichester: Wiley, 1985.
[14] Turner RK. Wetland conservation: economics and ethics. In: Collard D et al., editors. Economic, growth and sustainable development. London: Macmillan, 1988.
[15] Turner RK. The place of economic values in environmental valuation. In: Bateman IJ, Willis KG, editors. Valuing environmental preferences. Oxford: Oxford University Press, 1999. p. 17–41.
[16] O'Neil J. Managing without prices: the monetary valuation of biodiversity. Ambio 1997;26:546–50.
[17] Chichilnisky G, Heal G. Economic returns from the biosphere. Nature 1998;391:629–30.
[18] Brouwer R, Turner RK, Voisey H. Investigating public perception of overcrowding in a National Park: managing a multi-purpose open access reserve. Journal of Sustainable Tourism 2002;9:6.
[19] Ciriacy-Wantrup SV. Resource conservation: economics and policies. Berkeley: University of California Press, 1952.
[20] Bishop R. Endangered species and uncertainty: the economics of the safe minimum standard. American Journal of Agricultural Economics 1978;60:10–8.
[21] Crowards TM. Addressing uncertainty in project evaluation: the costs and benefits of safe minimum standards. Global Environmental Change Working Paper GEC 96-04, Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia and University College London, 1996.
[22] Crowards TM. Combining economics, ecology and philosophy: safe minimum standards of environmental protection. In: O'Connor M, Spash C, editors. Valuation and environment: principles and practices. Aldershot: Edward Elgar, 1997.
[23] Pearce D, Ulph D. A social discount rate for the United Kingdom. CSERGE Working Paper GEC 95-01. University College London and University of East Anglia, 1995.
[24] Barbier EB. Sustainable use of wetlands. Valuing tropical wetland benefits: economic methodologies and applications. The Geographical Journal 1993;159(1):22–32.

[25] Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, van den Belt M. The value of the world's ecosystem services and natural capital. Nature 1997;387:253–60.
[26] Turner RK, Adger N, Brouwer R. Ecosystem services value, research needs and policy relevance. Ecological Economics 1998;25:61–5.
[27] Desvousges WH, Johnson FR, Dunford RW, Boyle KJ, Hudson SP, Wilson KN. Measuring non-use damages using contingent valuation: an experimental evaluation of accuracy. Research Triangle Monograph 92-1. North Carolina: Research Triangle Institute, 1992.
[28] Brouwer R, Lanford IH, Bateman IJ, Turner RK. A meta-analysis of wetland contingent valuation studies. Regional Environmental Change 1999;1:47–57.
[29] Bateman IJ, Jones AJ, Nishikausa N, Brouwer R. Benefits transfer in theory and practice. CSERGE Working Paper No. 2000-25. University of East Anglia, 2000.
[30] Wolfe FM. Meta-analysis. Beverly Hills: Sage Publications, 1986.
[31] Turner RK, Bateman IJ, editors. Water resources and coastal management. Cheltenham: Edward Elgar, 2001.
[32] World Bank. World Bank indicators, 2002.
[33] Langford IH, Skourtos MS, Kontogianni A, Day RJ, Georgiou S, Bateman IJ. Use and nonuse values for conserving endangered species: the case of the Mediterranean Monk Seal. In: Turner RK, Bateman IJ, Adger N, editors. Economics of coastal and water resources: valuing environmental functions. Dordrecht: Kluwer Academic Publishers, 2001.
[34] Langford IH. Using a generalized linear mixed model to analyse dichotomous choice contingent valuation data. Land Economics 1994;70(4):507–14.
[35] Langford IH, Marris C, O'Riordan T. Public reactions to risk: social structures, images of science and the role of trust. In: Bennett PG, Calman KC, editors. Risk communication and public health: policy, science and participation. Oxford: Oxford University Press, 1999.
[36] Langford IH, Georgiou S, Bateman I, Day RJ, Turner RK. Public perceptions of health risks from polluted coastal bathing waters: a mixed methodological analysis using Cultural Theory. Risk Analysis 2000;20(5):691–704.
[37] Kontogianni A, Skourtos MS, Langford IH, Bateman IJ, Georgiou S. Integrating stakeholder analysis in non-market valuation of environmental assets. Ecological Economics 2001;37(1):123–38.
[38] Turner RK, Adger WN, Lorenzoni I. Towards integrated modelling and analysis in coastal zones: principles and practices. LOICZ Reports and Studies No 11, IGBP/LOICZ, Texel, the Netherlands, 1998.
[39] Tri NH, Adger WN, Kelly PM. Natural resource management in mitigating climate impacts: mangrove restoration in Vietnam. Global Environmental Change 1998;8:49–61.
[40] Adger WN, Kelly PM, Tri NH. Valuing the products and services of mangrove restoration. Commonwealth Forestry Review 1997;76:198–202.
[41] Gren IM, Turner RK, Wulff F, editors. Managing a sea: the ecological economics of the Baltic Sea. London: Earthscan, 2000.
[42] Georgiou S, Turner RK, Bateman IJ. Valuation and management of nutrient pollution in the baltic drainage basin. In: Turner RK, Bateman IJ, Adger N, editors. Economics of coastal and water resources: valuing environmental functions. Dordrecht: Kluwer Academic Publishers, 2001.
[43] Georgiou S, Bateman IJ, Söderqvist T, Markowska A, Zylicz T. Benefits valuation. In: Turner RK, Gren I-M, Wulff F, editors. The Baltic drainage basin report. EV5V-CT-92-0183. Brussels: European Commission, 1995.
[44] Söderqvist T. The benefits of a less eutrophicated Baltic Sea. In: Gren I-M, Turner RK, Wulff F, editors. Managing a sea: the ecological economics of the Baltic. London: Earthscan, 2000.
[45] Brown K, Adger NW, Tompkins E, Bacon P, Shim D, Young K. Trade-off analysis for marine protected area management. Ecological Economics 2001;37:417–34.
[46] Crooks S, Turner S. Integrated coastal management: sustaining estuarine natural resources. Advances in Ecological Research 1999;29:241–89.
[47] Russell CS, Vaughan WJ, Clark CD, Rodriguez DJ, Darling AH. Investing in water quality: measuring benefits, costs and risks. Washington, DC: Inter-American Development Bank, 2001.