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ANALYSIS

Valuing nature: lessons learned and future research directions☆

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Abstract

This paper critically reviews the literature on environmental valuation of ecosystem services across the range of global biomes. The main objective of this review is to assess the policy relevance of the information encompassed by the wide range of valuation studies that have been undertaken so far. Published and other studies now cover most ecosystems, with aquatic and marine contexts attracting the least attention. There is also a predominance of single function valuation studies. Studies valuing multiple functions and uses, and studies which seek to capture the 'before and after' states as environmental changes take place, are rare. By and large it is the latter types of analyses that are most important as aids to more rational decision taking in ecosystem conservation versus development situations involving different stakeholders (local, national and global). Aggregate (global scale) estimates of ecosystems value are problematic, given the fact that only 'marginal' values are consistent with conventional decision-aiding tools such as economic cost–benefit analysis. In general, valuation data provide prima facie support for the hypothesis that net ecosystem service value diminishes with biodiversity and ecosystem loss [Balmford et al. (2002), Science 297, p. 950]. Future research effort should include complementary research on multiple ecosystem services that seeks to capture the temporal disturbance profile and its causal factors. The explicit recognition of multiple, interdependent ecosystem services and values, poses both conceptual and empirical research challenges. It would serve to transform the practice of research in this sub-field via the a priori assumption of multiple (and inter-dependent) use, instead of independent single use. This line of reasoning can then be extended to the institutional arrangements that determine which values are captured. New institutional processes and arrangements are probably required in order to best realise benefit streams from multiple ecosystem use and non-use provision, across a range of different stakeholders.

Keywords: Environmental values; Ecosystem services; Cost–benefit analysis

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1. Introduction: the purpose of valuation

In the last 30 years or so, valuation of environmental services and change has become one of the most significant and fastest evolving areas of research in environmental and ecological economics. From the outset, one important motivation for valuation studies has been to generate a better and more comprehensive informational base for the policy formulation and decision taking process. Such studies can inform societal decision mechanisms trying to cope with the allocation of scarce resources among competing demands. In particular, they support preference-based approaches (consumer and/or citizen preferences) and are compatible with a common monetary metric deployed across competing uses. The fundamental aim is not to put a "$ price tag" on the environment, or its component parts, but to express the effect of a marginal change in ecosystem services provision in terms of a rate of trade off against other things people value (Randall, 2002; Hanley and Shogren, 2002). While we believe that there is a strong case in favour of environmental economic valuation as a decision aid, we also recognise that there are limits to its use (Turner, 2000).

In the next section of the paper the concept of natural value is assessed from a range of perspectives. This is followed by a discussion designed to highlight factors, which have an important bearing on the design of valuation studies and interpretation of the results of such studies. Finally, case study evidence is presented drawn from developed and developing countries to provide support for nature conservation strategies. On a global scale such strategies need to be a combination of protected area zoning, sustainable utilisation practices and recreation/restoration programmes. At the margin, particularly in developing country contexts, there will also continue to be cases where economic development needs outweigh nature conservation requirements; or cases where the latter are feasibly met only through international compensation schemes (e.g. debt for nature swaps, other financial transfers, education and training provision etc.).

2. What is value

2.1. The value of nature

The debate over what value resides in nature, or what is the value of nature, has highlighted the fact that the core concept is complex and multidimensional. In the literature a useful general value typology, summarised in Table 1, has found support from a range of disciplines concerned with environmental science and management (Hargrove, 1992; Turner, 2000). The valuation data presented in the case studies section relate to the first and, more problematically, the second category in Table 1, and are anchored to individual human preferences and valuation. Economists have generally settled for a taxonomy of environmental value, the components of which add up to total economic value (TEV). The key distinction made is between use values and a remainder called non-use value. The latter component reflects value in addition to that, which arises from usage. Thus individuals may have little or no use for a given environmental asset or attribute but would nevertheless feel a 'loss' if such things were to disappear. However, the boundaries of the non-use value category are not clear cut and some human motivations which may underlie the position that nature should be conserved 'in its own right', and labelled existence value, are arguably outside the scope of conventional economic thought (see category two in Table 1). In practice, what is at issue here is whether it is meaningful to say that individuals can assign a quantified value to nature or its component parts, reflecting what they consider to be intrinsic value.

Economic valuation can be combined with an ecosystem function (and related goods and services outputs) approach. It is important to note that what is, therefore, being valued is not biodiversity per se, but rather interdependent elements of ecological services. The aggregation of the main function-based values provided by a given ecosystem has been labelled TEV. But the aggregate TEV of a given ecosystem's functions, or combinations of such systems at the landscape level, may not be equivalent to the total system value. The continued functioning of a healthy ecosystem is more than

Table 1
A general value typology

●	Anthropocentric value
1	Anthropocentric instrumental value
	This is equivalent to: 'Total economic value' = use + non-use value. The non-use category is bounded by the existence value concept, which has itself been the subject of much debate. Existence value may, therefore, encompass some or all of the following motivations
i	Intragenerational altruism: resource conservation to ensure availability for others; vicarious use value linked to self-interested altruism and the "warm glow" effect of purchased moral satisfaction
ii	Intergenerational altruism (bequest motivation and value): resource conservation to ensure availability for future generations
iii	stewardship motivation: human responsibilities for resource conservation on behalf of all nature; this motivation may be based on the belief that non-human resources have rights and/or interests and as far as possible should be left undisturbed
	If existence value is defined to include stewardship then it will overlap into the next value category outlined below
2	Anthropocentric intrinsic value
	This value category is linked to stewardship in a subjectivist sense of the term value. It is culturally dependent. The value attribution is to entities which have a 'sake' or 'good of their own', and instrumentally use other parts of nature for their own intrinsic ends. It remains an anthropocentrically related concept because it is still a human valuer that is ascribing intrinsic value to non-human nature
●	Non-anthropocentric value
3	Non-anthropocentric instrumental value
	In this value category entities are assumed to have sakes or goods of their own independent of human interests. It also encompasses the good of collective entities, e.g. ecosystems, in a way that is not irreducible to that of its members
	But this category may not demand moral considerability as far as humans are concerned
4	Non-anthropocentric intrinsic values
	This value category is viewed in an objective value sense, i.e. 'inherent worth' in nature, the value that an object possesses independently of the valuation of valuers. It is a meta-ethical claim, and usually involves the search for constitute rules or trump cards with which to constrain anthropocentric instrumental values and policy

Source, adapted from Hargrove (1992).

the sum of its individual functions (components). The difference lies in that the operating system yields or possesses primary, 'glue' or infrastructure value, i.e. value related to the fact that some combinations of ecosystem structure and composition is necessary to ensure the 'healthy' functioning of the system, or system status (Gren et al., 1994). Society may also regard nature or some of its attributes as socio-culturally, historically or symbolically valuable; and for some people such value cannot be meaningfully expressed in monetary terms. Fig. 1 summarises these issues using the example of a wetland ecosystem and serves to highlight the complexities of valuing multiple uses.

2.2. Economic value

An economic perspective on nature portrays it as an asset providing a flow of goods and services, physical as well as aesthetic, intrinsic, and moral. This provision is the means of life support as well as of quality of life enhancement. When other means of provision of these goods and services are acceptable, or compensate for their loss, these means can be used to value losses in nature's services. As market and monetised economies are so pervasive, the use of money as the yardstick of measurement of benefits provided by nature establishes a transparent relationship with other uses of its assets or attributes.

There are likely to be instances when the substitutability of other means is not feasible or 'socially' acceptable. Feasible valuation may be precluded because of scientific complexity and uncertainty and consequent ignorance about the welfare consequences of severe ecosystem degradation or collapse. Human cognitive limitations can also constrain monetary valuation (particularly non-use estimation) in contexts where individual survey respondents are unable to properly reference frame the problem despite state of the art survey designs. For example, when the gas regulation functions of the atmosphere are so severely degraded that all human life is at risk, it is unlikely

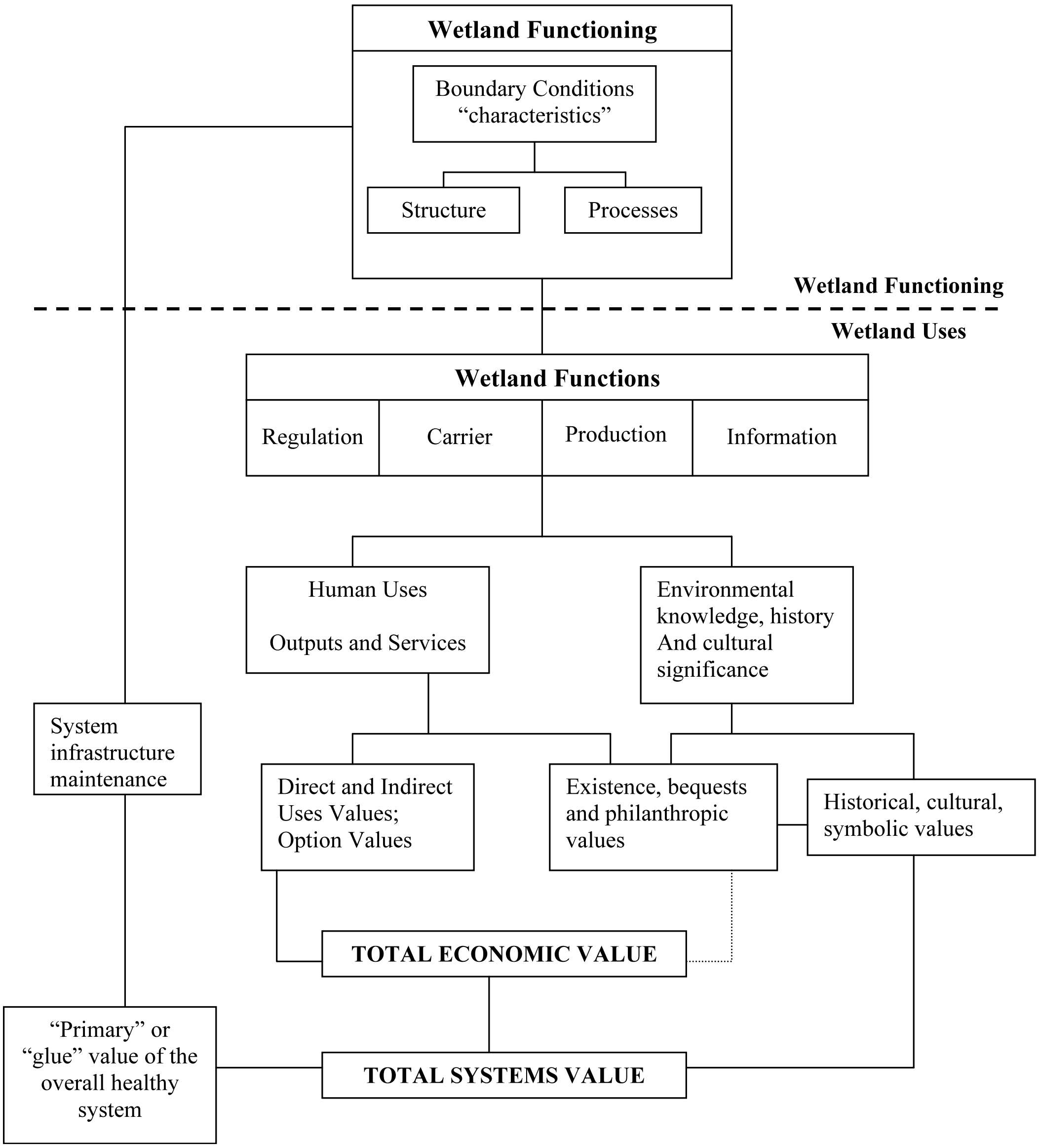

Fig. 1. From ecosystem structure and processes to ecosystem functions and values. Source: Turner et al. (2001).

to be feasible to think of substitutes for those functions, much less place monetary values on them. Or when cultural functions, such as spiritual values of certain species, are under consideration, in some societies their values may be on a completely different "moral" plane, within which there is no acceptable substitute, or monetary measure of acceptable compensation, because of prevailing social norms. These are cases where monetary valuation of nature's services may be judged to be inappropriate.

The appropriate context for economic valuation is conditioned, among other things, by the scale of the environmental changes. Monetary valuation is most meaningful when considering small, or marginal, changes in the conditions of natural assets. For example, determining the biogeophysical value of a forest at a local scale is more worthwhile than attempts to determine the global value of all forests. The loss or degradation of forests on a local or regional scale is imaginable, and the consequent loss of services may not result in such dramatic alterations in ecosystem processes as to place human survival at risk (although cultural and spiritual losses may effectively destroy a culture). In contrast, the loss of all forests on a global scale would result in such profound human survival consequences as to be "beyond the margin of analysis".

When monetary valuation is feasible, nature's service flows can be considered as having economic value as represented by Fig. 2. This figure illustrates the marginal valuation of subsequent units of service flow. The units of service flow may be physical assets, such as trees and fish; or biogeochemical processes, such as nutrient throughput and water release; or cultural and social processes, such as recreational use and level of aesthetic enjoyment. The marginal units of service flow are then valued based upon the willingness to pay for their provision, or willingness to accept compensation for their loss. This willingness to pay and accept is based on the availability and costs of substitute provision, or measures of psychic loss.

As shown in Fig. 2, economists expect that marginal values decline as the service flow increases (say from B to A), and vice versa, over some non-critical range. However, below some critical threshold defined by for example the safe minimum standard for endangered species (Ready and Bishop, 1991). There may be no meaningful marginal WTP-based value available. The aggregate of the marginal values over some non-critical range is the TEV.

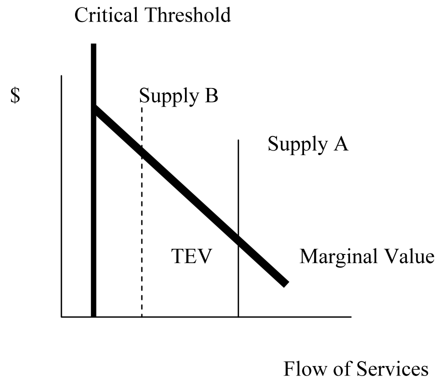

Fig. 2. Valuing nature's services.

A further crucial concept in valuation is net value, viz. while nature's services are provided by natural ecosystem structures and processes outside of economic systems, they are not free to access. For example, the necessary time and resource costs of harvesting timber must be subtracted from timber sales revenue, in order to estimate the value of this service flow.

The requirement to deduct costs of procurement in determining the values of nature's services should not be confused with the opportunity costs of preserving the flows of services. For example, assuring a sustained flow of non-timber forest products may result in the sacrifice of other uses of the forest, say for timber production or cattle grazing. This is a different cost to that of procuring the non-timber services. In determining whether to preserve the non-timber services, the TEV of these services, net of their procurement costs, should be compared with the values of alternative uses of the forest.

While the value of resources we procure from nature, such as the timber and fish we harvest, may have straightforward, readily estimable values (because of the existence of markets) nature's services may have to be more indirectly valued. Initially, the biogeophysical processes and functioning which yield the services flows must be characterised and quantified, in order to identify all relevant benefits. For example, the water retention/release function of a forested ecosystem may generate value through moderating stream flows, which, in turn, may have value for irrigation, flood protection, and recreation. Their implied values must then be "constructed", since markets may not value these services directly. Economists have developed a variety of tools with which to make these valuations once the biogeophysical services are characterised and quantified (van den Bergh, 1999).

The ability to value nature's services is constrained by the complexity of nature itself. The "production function" of nature is so complex, and little understood in many instances, that reliable estimates of all services could not be made. An aspect of this complexity is that joint products are inherent in most of nature's processes; for example, trees perform valuable hydrologic, nutrient cycle, and climate functions. Accounting for value must recognise all these joint product values.

Where economic valuation perhaps becomes most difficult is in assessing biodiversity, per se. If biodiversity is considered as the "glue" that holds all of nature's structure and process together, it is indispensable. Clearly certain species can be valued independently, as resources or aesthetically. But even this becomes complicated when an entire web is dependent on a species. As diversity and biocomplexity is degraded the valuable productivity of natural systems is threatened and its flexibility to reconfigure itself (resilience) in ways that we consider valuable may be diminished. This suggests an "insurance value" of biodiversity, which is both highly significant and yet formidably difficult to value.

3. Ecosystem valuation difficulties

3.1. Marginality

Previous estimates of the main ecosystem service values at the global level have 'engaged' science and policy at this scale, but there is a remaining requirement to better inform the local decision making level because of the everyday pressure imposed on ecosystems (Costanza et al., 1997; Turner et al., 1998). Annual data relating to the physical loss of ecosystems and the costs of conservation strategies are good examples of such policy relevant information. When it comes to valuation data it is 'marginal' values that are required, rather than aggregated global values, which do not fit into formal cost–benefit appraisal systems and methods. At the margin, it is important to know what is the value of lost ecosystem services as, for example, parts of the stock of tropical forests in certain locations are degraded or destroyed. A range of case study evidence is reviewed in order to indicate the order of magnitude of such losses across different biomes.

We have argued that the values of, and in, nature are complex and multi-dimensional. However, the case studies we present are predominately based on a functional (use values), instrumental and anthropocentric view of nature's value. This is not meant to downplay the significance of anthropocentric intrinsic (non-use) values; indeed, empirical estimations using the survey-based contingent valuation method are included, but merely to indicate that capturable market or near-market values can be assigned to nature and, in combination, form a powerful case in favour of nature conservation and/or restoration strategies across a spectrum of biomes.

The drawing of generic lessons, and even less straightforwardly the aggregation of site (spatial), temporal and cultural specific data should be approached cautiously. The case study data reviewed here offer strong evidence against further ecosystem degradation, biodiversity loss and/or increased disturbance activities, but it is not in itself a globally conclusive or uniform outcome. A number of important caveats must be borne in mind in the context of environmental valuation studies of single or multiple ecosystem functions goods and services, and especially if TEV estimates are under scrutiny. Because of the uncertainties surrounding threshold effects and the true extent of intact or relatively undisturbed global biomes, judging what is and what is not a 'marginal' change, for example, is a far from straightforward problem (Turner et al., 1998).

3.2. Double counting

Going beyond the 'marginality' problem, it is also important to identify sources of 'double counting' in any TEV study. In other words, many ecosystem services are not complementary, the provision of one (say recreation in a wetland) is precluded by others (for example, using the same wetland for effluent treatment and storage). The full range of complementary and competitive services must be distinguished before any aggregated valuation is completed.

3.3. Typological issues

Our review of the literature has also highlighted other significant typological issues related to the strategy and design of valuation studies. It is important to differentiate between valuations of the in situ ecosystem stock and estimates of the value of the flow of goods and services from a given stock. The flow values may also be potential or based on actual income from harvesting (Batagoda et al., 2000). The published studies reflect the real world historical environmental change process, driven by economic development and globalisation, in the sense that some are ex ante analyses, while others are ex post. The former are estimates of the social value potentially lost if pristine (or relatively un-disturbed) ecosystems were not to be conserved or sustainably utilised, the latter represent estimations of what the potential gains value would be if previously degraded or destroyed ecosystems were restored or recreated.

The ex ante/ex post distinction serves to emphasise the importance of the starting point (i.e. the position in the temporal profile of any ecosystem disturbance that the study adopts), and its policy reference. Some ecosystems go through a number of transitional stages as their exploitation and levels of disturbance increase over time. Other ecosystems are subjected to a more or less discrete conversion from one use to a completely different one in a relatively short period of time (e.g. wetland to farmland). The distinction also demarcates fairly well the division between developed country and developing country contexts and studies, with the latter providing the bulk of ex ante valuations and the former more ex post valuations.

Generalising, it seems reasonable to argue that harvestable resources in terrestrial systems have been subjected to disturbance pressure which has been caused by economic activities seeking to increase the intensity of exploitation of given ecosystem functions. The result has been a relatively smooth disturbance profile over time. Other terrestrial resources and systems have also been affected by land use change that exhibits a much more discrete disturbance profile. Aquatic resources and systems, however, are dominated by intensity of functional use changes and pressures principally because of the 'fugitive' nature of most of the resources. Only coral reefs and benthic-based resources are truly in situ and, therefore, prone to more discrete disturbance profiles. Transitional ecosystems such as wetlands (temperate and tropical) are closer to terrestrial cases than they are to aquatic ones.

3.4. Spatial and temporal data transfer

In order to estimate benefits given limited funds and in a relatively short time period, it may be possible to transfer data from other studies as a rough guide to appropriate values this technique of 'benefits transfer' is, however, fraught with difficulties. Problems include: a requirement for good quality studies of similar situations, the potential for characteristics to change between different time periods and an inability to deal with the valuation of novel impacts the quality of studies carried out using transferred values can be no better than the quality of the data in its original context and there is little published evidence that tests the validity of environmental value transfer. In the few studies that have been carried out, transfer errors have been found to be substantial (Brouwer, 1998). It may be possible to make value transfer more robust if, as well as socio-economic variables, essential physical variables, e.g. ecosystem characteristics and processes, are considered at the different sites. As more information about factors that influence environmental values becomes available, e.g. through meta-analysis (Brouwer et al., 1999), the transfer of values across populations and sites becomes more practicable using either only existing data or supplementing this with new original data. Following this brief review of methodological difficulties we now turn to an important policy issue.

3.5. Distribution of benefits and costs

The scale at which benefits and costs are 'captured' is also a very important issue in terms of practicality and equity. Local users often have a preference for direct, short term, gains derived from the consumption and/or sale of products harvested from forests, wetlands, etc. Other stakeholders in the same country or in the international community might express conservation-based preferences tied to the more indirect environmental services provided by forests, etc. Differences across stakeholders in rates of private time preference and tenure security, among other factors, will be important in this context. Typically, though not exclusively, developing countries conserving ecosystems and biodiversity, incur high local costs for the sake of often large global benefits (Kremen et al., 2000). In Kenya, for example, it has been estimated that the nation incurs a penalty of some \$203 million per annum in forgone benefits because of its protected area policy (Norton-Griffiths and Southey, 1995).

In contrast to developing countries, developed countries tend to incur relatively low local costs yielding more modest global benefits. However, they enjoy the social, global benefits of conservation policies enacted domestically and in the developing world. They should, therefore, compensate those developing countries that incur net losses from conservation policies, through, for example, international resource transfer mechanisms. Currently, such mechanisms are deficient and are not able to meet the challenges posed (Keohane and Levy, 1996; Richards, 2000). They are also not panaceas, and a sustainable development strategy needs to encompass complementary local actions.

At least 25% of developing country forests, for example, are owned or administered by local communities. Currently the contribution that local forests can make to local livelihoods is limited by market, policy and information 'failures' of various sorts. Correction of at least some of these failures might open market niches where large numbers of low-income producers could develop competitive advantage (Scherr et al., 2002). This 'conservation through commercialisation' thesis should not, however, be accepted without question. In the case of tropical forests and their provision of non-timber forests products (NTFPs) market demand for only selected products can still result in forest stock degradation. These same pressures may also disadvantage the poorest local households as the commercialisation process intensifies (Arnold and Periz, 2001). Nevertheless, there is some evidence to suggest that the local financial benefits derived from intact rain forests do contribute substantially to household consumption and earnings, 38 and 23%, respectively, on an annual basis for communities in Bolivia and Honduras (Godoy et al., 2002). A study in Sri Lanka (Batagoda et al., 2000) has also shown that local forest communities, which harvest NTFPs, gain in terms of the overall income distribution and that there is a consequent reduction in rural income inequalities.

4. Case studies

4.1. Typology

Our literature review uncovered a large number of single ecosystem functions valuation studies but relatively few that encompass multiple services, or that seek to capture the stages in a smooth disturbance profile, or that consider "before and after states" in the case of a discrete land use change. The selected studies below more or less meet these caveats and also serve to make the case that conservation is invariably an economically viable strategy. These findings are we believe highly policy relevant.

We now present the case study evidence organised via a simple four quadrant typology, based on developed versus developing country contexts and smooth versus discrete disturbance profiles (see Fig. 3). The monetary values have been standardised (to year 2000 US \$, using the US consumer price index, and translated at purchasing power parity rates where necessary). Discount rates applied in the studies vary but we refer to upper and lower-bound discount rates of 2/3 and 8/10%, respectively.

4.2. Temperate forests and rangelands—type 1 examples

We examine the value of Nordic boreal forests under different management regimes—as an example of temperate forests in general—and the

		COUNTRY CONTEXT	
		DEVELOPED	**DEVELOPING**
ECOSYSTEM DISTURBANCE/ DEGRADATION PROFILE	**RELATIVELY SMOOTH**	**TYPE 1** • ***Examples:*** temperate rangeland temperate forests regional seas fisheries • *Ex post* studies dominate	**TYPE 2** • ***Example:*** tropical forests • *Ex ante* studies for the most part
	STEPPED/ DISCRETE	**TYPE 3** • ***Examples:*** wetlands and shallow lakes hydrological regime changes in river systems (dams etc.) • Some ex *ante* studies but more *ex post* studies	**TYPE 4** • ***Examples:*** mangroves coral reefs semi-arid rangelands • *Ex ante* studies dominate

Fig. 3. Valuation contexts.

values associated with the loss of rangeland due to urbanisation.

Taking the boreal forests first, they are multiple use resources that produce timber, berries, mushrooms, game, recreational opportunities for hiking, camping and hunting, and provide storage for carbon dioxide, erosion control, retardation of run-off, and recharge of aquifers (see Secretariat of the Convention on Biological Diversity, 2001). Intensive management for timber production alone may, therefore, reduce or eliminate the value of other forest uses. Correspondingly, management alternatives that cater for a broad set of forest uses and services may generate a higher aggregate level of benefits than timber production, or the conversion of forests to agricultural use.

Private timber income from Nordic forests in the early 1990s represented the bulk of benefits from forest use (see Table 2). However, it is also clear that social benefits from non-timber forest outputs were substantial (some 60–80% of timber income), contradicting conventional wisdom that non-timber products are not significant in temperate and borial forests (Secretariat of the Convention on Biological Diversity, 2001). Nevertheless, the private value of forest land is determined by the value of timber production. In the early 1990s this was around \$1750 ha^{-1} in Finland and of comparable value elsewhere in Scandinavia because of the common influence of the world markets for timber and pulpwood (Finnish Forest Research Institute, 1996; Rantala, 1998). By contrast, the price of agricultural land was twice as high—\$3500 ha^{-1} on average (Peltola, 1997; Rantala, 1998)—reflecting higher rates of income from farming. This simple price comparison is distorted by the existence of agricultural subsidies, some 70% of agricultural income, for example, in Finland and Norway in the early 1990s. Furthermore, the comparison ignores the potential loss of social benefits if forest land was converted to agricultural use. Consequently, the conversion of forests has been subject to regulatory control (e.g. by means of a permit system in the case of Finland).

Ex ante studies of virgin forest preservation are much more uncommon than ex post studies of alternative uses such as those considered above. Furthermore, those that have been undertaken do not tend to consider individual benefit flows. A case in point is a contingent valuation study conducted by Hoen and Winther (1993) to estimate the benefits associated with the preservation of virgin forests in Norway. They elicited households' annual WTP for a 10-year-period for three perpetual preservation alternatives. Extending their results to the whole of the Norwegian population, we estimate that Norwegians were

Table 2
Monetary values associated with Nordic forests

Benefit stream	Value ($ ha^{-1} per year)		Capitalised values (£ ha^{-1})
	Lower-bound	Upper-bound	
Net benefits from timber production	45	85	1700[a]
Berries, mushrooms. Lichen and peat	10	15	
Recreation	15	20	
CO_2 sequestration	10	15	
Losses			
Total benefits of forest use	80	135	2500[b]
Benefits of preserving virgin forests			4060–5800[c], 4505–6450[d], 5265–7544[e]

Adapted from Nordic Council of Ministers (1995), Hoffren (1997), Holgen et al. (2000) and Hoen and Winther (1993).
[a] Market value of forest land is assumed to capitalise annual timber revenue.
[b] This estimate of total benefits of forest use is based on the assumption that the market price of forest land capitalises upper-bound annual timber revenue. Upper-bound estimates for annual benefits from non-timber forest products and services are about 50% of upper-bound annual timber revenues. Their capitalised value should thus be around 50% of market value of forest land. Upper-bound figures are used to generate a conservative estimate; lower-bound estimates for non-timber benefits are almost as high as annual timber revenue and would translate to higher capitalised total benefit figure.
[c] These lower benefit estimates of preservation are calculated using 10% discount rate. Lower figures represent WTP for larger preservation programme and higher WTP figures represent WTP for smaller preservation programme.
[d] These benefit estimates of preservation are calculated using 7% discount rate, the baseline assumption used by Hoen and Winther (1993). Lower figures represent WTP for larger preservation programme and higher WTP figure represent WTP for smaller preservation programme.
[e] These higher benefit estimates of preservation are calculated using 3% discount rate. Lower figures represent WTP for larger preservation programme and higher WTP figures represent WTP for smaller preservation programme.

willing to pay at least \$4500 ha^{-1} for preservation (and a maximum of about \$6500 ha^{-1}). Even at the lower estimate, the results suggest a substantial premium on the retention of existing virgin forest stocks.

To conclude, values associated with the use of boreal forests indicate that preservation may be the highest-valued use for marginal units of virgin forests. The results suggest that management of other temperate forests for multiple use makes economic sense. The values associated with timber production and carbon sequestration are not location-specific because they are effectively determined in the world market (they do, of course, vary across forest types and stands). However, the value of non-timber products and recreational services is sensitive to location. Therefore, the Nordic figures must be considered conservative estimates because of relatively low population density and abundance of forests. For example, generalisation of the results of Bateman et al. (1996) indicates that in Oxfordshire, England the value of a forest established for recreational use would be \$2290 ha^{-1} per annum (see also Garrod and Willis, 1997; Scarpa et al., 2000a,b).

The results also raise the question of what institutional arrangements can best realise a broad set of forest use benefits. The answer is likely to be private property rights complemented by public access and other rights that foster the production of non-timber goods and services (see, e.g. Vail and Hultkrantz, 2000). Finally, it is note-worthy that agricultural subsidies have, over time, resulted in land use decisions with both economically and environmentally adverse consequences. In the future a reorientation of subsidy regimes could foster the concept of “multi-functional agriculture” (OECD, 2001). Estimates of multifunctional values per hectare are not yet available, but according to Yrjölä and Kola (2001)a 30% reduction in agricultural subsidies in Finland, rather than a redirection, would decrease multifunctional benefits to such an extent as to result in a social welfare loss.

Turning to the case of rangelands, we focus on a recent, comprehensive ex post study reported by Kreuter et al. (2001). On the basis of satellite data, they track land use changes over an area of approximately 140 000 ha in three major watersheds around San Antonio, TX, a rapidly growing metropolitan area. By reference to six categories of land cover, they find that rangeland cover was initially dominant but fell by some 65% during the period 1976–1991. The distribution of land use changes in the study area is shown in Table 3, applying categories and related ecosystem service coefficients provided by Costanza et al. (1997). Thus, the Costanza et al. coefficient for temperate/boreal forest (woodland) is treated as a proxy for areas subject to "pervasive woody plant invasion" and that for cropland is treated as a proxy for (post-harvest) bare soil.

The increase in land cover attributed to "woodland" and the associated increase in service value per hectare largely offset losses in services due to conversion of rangeland to "urban sprawl". Thus, the loss of ecosystem services for the 15-year-period is \$6.49 ha^{-1} per year (equivalent to \$216 ha^{-1} capitalised at 3%; \$65 ha^{-1} at 10%). However, if the conversion of rangeland to woodland is treated as having no net effect on ecosystem services then the average loss is increased to \$27.00 ha^{-1} per year.

While the valuations are based on service coefficients from earlier studies, even in the "better case" scenario, the study highlights the appreciable losses in ecosystem service value that can accompany urbanisation. Furthermore, the estimates are likely to be conservative given increasing urbanisation (i.e. as the stock of virgin, or at least non-urban, land diminishes). However, the study does not attempt to value the economic benefits of increased residential and commercial capacity resulting from land use change in the study area so that it is not possible to conclude on the changes in TEV that accompanied this conversion of rangeland. Nevertheless, the study thus also demonstrates the complexities of assessing land use changes in developed countries.

4.3. *Tropical forests—type 2 example*

Our survey of the available literature identified several studies that estimated (TEV) but only a few cases attempted to compare these values for alternative land uses. Most cases relied on benefits transfer to provide estimates of the TEV for the set of services valued. Comparison of the available data revealed that the quantifiable benefits from direct and indirect uses of forests are highly variable (Adger et al., 1995; Torras, 2000; Andersen, 1997; Chomitz and Kumari, 1998). For example, values for NTFPs ranged from \$9 to 1407 ha^{-1} per year.

Table 4 summarises the results from three study sites, which came closest to meeting our selection

Table 3
Valuation of changes in ecosystem services on loss of rangeland

Land cover category	Change in cover 1976–1991 (ha)[a]	Ecosystem service coefficient 1994 (\$ ha^{-1} per year)	Change in ecosystem services 1976–1991 (\$ per year)
Rangeland	−52 601	232	−14 187 931
Woodland	35 769	302	12 558 877
Bare soil	6694	92	715 996
Residential	5156	0	0
Commercial	9246	0	0
Transportation	−1891	0	0
Total	2373	–	−913 058
Average loss of value in ecosystem services (\$ ha^{-1})			6.49

Adapted from Kreuter et al. (2001).

[a] The land use changes given by Kreuter et al. (2001) in Table 2 do not sum to zero. Hence, there is a small residual in the average annual change in cover.

Table 4
Case studies showing comparison in private, social and global benefits under alternative management options

Management options	Cameroon[1]			Sri Lanka[2]		Malaysia[3]		
	Sustainable forest	Conversion to oil palm	Conversion to small scale agriculture	Sustainable timber potential	Cultivation of tea (land clearance)	Unsustainable timber logging	Sustainable logging—option 1	Sustainable logging—option 2
Total private benefits	309	(1695)	1472	1212 (20 years), 1403 (50 years)[b]	4596 (20 years)[b]	2361	1922	1401
Total social benefits	206	10	34	158 (20 years), 196 (50 years)[b]	Not calculated	610	1230	1547
Total global benefits	2055	601	608	Not calculated	Not calculated	8270	10 048	10 076
TEV	2570[a]	(1084)[a]	2114[a]	Not calculated	Not calculated	11 242[c]	13 200[c]	13 024[c]
Total private benefit—3%				1829 (20 years)[d], 3165 (50 years)[d]	6978 (20 years)			
TEV at 2% discount rate						30 333	39 104	39 898

All values in ha^{-1}. Adapted from [1]Yaron (2001), [2]Batagoda et al. (2000) and [3]Kumari (1994).

[a] NPV at 10% discount rate for 32 year cycle; private benefits for oil palm based on removal of market distortions (taxes, subsidies); social benefits include: NTFPs, flood prevention and sediment control; global benefits include: carbon storage and undiscovered plant drugs.

[b] NPV at 8% discount rate for 20 and 50 year cycles; social benefits include: NTFPs both flora and fauna; externalities of tea cultivation such as soil erosion and sedimentation are not included.

[c] NPV at 8% discount rate for 100 year cycle; social benefits include: NTFPs, domestic water, fish, recreation, hydrological; global benefits include: carbon storage and endangered species; unsustainable logging – 50% damage using traxcavator + canal; sustainable 1–20% damage using traxcavator + tramline; sustainable 2–20% damage using winch + tramline.

[d] The inclusion of carbon sequestration and potential recreation benefits via benefits transfer data from other tropical forest sites would make the sustainable management option much more competitive with the land clearance option adjusted for its negative externality costs. It was also found to be the case that local income inequalities were reduced by NTFPs harvesting activities.

criteria. Each study considers values from alternative land uses in a given area or region, across a degradation profile, taking account of multiple service benefits, at both national and global levels, given different intensity and type of use. The benefits provided by tropical forests seem to be location specific, scale dependent, and influenced by the heterogeneity of ecosystem attributes (soil, species types and densities, terrain, accessibility to markets). These factors in turn affect the intensity of use of an area, the potentially available resources and the demand/supply of goods and services. Local cultural values also determine the type of goods and services exploited as well as the extraction methods used and intensity of use.

In the Cameroon study, local producers would gain financially if the forest was converted to either subsidised oil palm plantation, or to agriculture. In the Sri Lanka study, tea cultivation private benefits represent a significant short to medium term investment for local growers. The study also highlighted the importance of harvesting NTFPs for consumption and sale, which played a significant role in household income generation and in supporting income equality across local communities. Finally, in Malaysia unsustainable logging practices yield immediate and large financial gains. Nevertheless, in all three studies forest conservation almost always made economic sense if the full social global benefits were accounted for. Therefore, implementing a conservation strategy may only be feasible where those in the global community who would gain from such a strategy provide incentives to those who would lose locally. The Malaysian example also serves to emphasise that with sustainable management practices (see options 1 and 2 in Table 4) significant timber harvesting revenues can still be earned over the long run in addition to non-timber benefits.

4.4. Temperate wetlands—type 3 example

Temperate wetlands have been under persistent land conversion and pollution pressures for more than 100 years. They represent examples of ecosystems, which are characterised by stepped/discrete disturbance profiles. In the second half of the 20th century the rate of destruction and degradation was increased because of the perverse effect of agricultural subsidy regimes. In Europe, for example, under the Common Agricultural Policy (CAP) large areas of wetlands were drained and converted into arable cropping enterprises, because of the subsidies available for conversion works and the guaranteed crop price support system. Financial returns of up to 20% were not uncommon (Bowers, 1983; Turner et al., 1983). Economic analysis of typical conversion schemes, even with optimistic assumptions about yield improvements and other factors, however, proved that such investments were inefficient (Turner and Brooke, 1988). Table 5 contains the results of a study, which estimated the value of three wetlands in Canada threatened with conversion. The economic case in favour of conservation is clear once the subsidy element has been removed from the agricultural land use value and the retained wetland value is represented by its recreational value.

This result would have been reinforced if the full conserved wetland environmental services had also been factored in. Table 6 sets out just such a comparison of agricultural use values and other ecosystem service flows such as nitrogen fixation, water supply, habitat provision, etc. in Sweden (Gren, 1995; van Vuuren and Roy, 1993).

In Europe, the 1990s saw an easing of the environmental change pressure (as the subsidy regimes have been reformed) but areas of remaining pristine wetlands in Eastern Europe are now coming under development threat as the European Union expands its membership and globalisation trends intensify.

4.5. Mangroves and coral reefs—type 4 examples

The literature review revealed that several studies have estimated the economic value of mangrove ecosystems. However, only a few attempts have been made to evaluate changes in TEV. Two case studies, which evaluate direct benefits under different scenarios, are presented as follows.

Gammage (1997) reports a study of a mangrove ecosystem in El Salvador, which is under develop-

Table 5
Wetland conversion costs and benefits

Ecosystem profile	Annual value ($ ha^{-1} per year)	Value over 30 year cycle ($ ha^{-1})	
		@ 6% DR	@ 3% DR
Converted wetland			
Agricultural value (financial with subsidies)		1789	3100
Agricultural value (economic)		−757 to −4443	−1201 to −7047
Conserved wetland			
Recreational value[a]	372–237[b]	6216–3965	9860–6287

Source, van Vuuren and Roy (1993).
[a] Hunting, fishing and trapping value, based on travel cost valuation method.
[b] For a 20 and 300 ha wetland, respectively.

ment pressure, on the basis of three different management scenarios.

1) Under the current management strategy and even allowing for some natural regeneration, it is predicted that the mangroves will disappear in 26 years due to deforestation and land clearance.
2) In the Partial Mangrove Conversion Scenario, it is assumed that 240 ha will be converted to shrimp ponds and the remaining mangrove depleted for community timber and fuelwood.
3) In the Sustainable Management Scenario, it is assumed that only mature trees are felled.

As shown in Table 7 the latter management strategy would be economically superior to the others as far as local beneficiaries are concerned, over the long run. This result is achieved without consideration of other services that intact mangrove areas can provide, e.g. coastal erosion prevention and carbon sequestration benefit. At the local level, however, there may well be other constraints on the implementation of conservation measures such as property rights and cultural practices. Also, the actual distribution of local benefits is often skewed away from the very poorest in society.

The wider benefits of mangrove conservation are recognised by Sathirathai (1998) in a study of a site in Thailand, as summarised in Table 8.

As in the case of tropical forests (type 2), we see that the degradation of the ecosystem is motivated by private benefits with social costs being ignored (e.g. original mangrove coverage in the study area was over 1100 ha but 640 ha had been cleared for aquaculture). However, a notable distinction is that in the case of the mangroves the predominant

Table 6
Agricultural use values

Service flow	Annual value ($ ha^{-1} per year)		Value over 30 year cycle ($ ha^{-1})			
	Lower-bound	Upper-bound	@ 6% DR		@ 3% DR	
			Lower-bound	Upper-bound	Lower-bound	Upper-bound
Nitrogen fixation[a]	32	305	461	4424	641	6121
Other ecosystem services	225	225	3251	3251	4498	4498
Total ecosystem services	257	530	3712	7675	5139	10 619
Agricultural use	446					

Source, Gren (1995).
[a] Based on WTP for nitrogen reduction of $0.32–0.61 kg^{-1} N per year and assuming natural wetlands fix 100–500 kg N ha^{-1} per year.

Table 7
Net present value of different management scenarios, 1994–2050[a]

Goods/services	Management options ($ ha^{-1})[b]		
	Current management strategy	Partial mangrove conversion	Sustainable management option
Clearance logging		58	
Fuelwood and timber[c]	18	11	25
Artisanal shrimp and fish	755	736	800
Industrial shrimp[d]	902	761	1516
Rustic salt and shrimp	3		3
Shrimp ponds		91	
Total	1678	1657	2344

Source, adapted from Gammage (1997).
[a] Assuming a discount rate of 7.08%.
[b] Given a total mangrove area of 487 ha.
[c] Costs and benefits were calculated assuming that all timber needs would be met and that fuelwood consumption would be determined by the remainder.
[d] All fisheries benefits are net of primary producer costs; all capital goods are amortised over their lifetimes and discounted at the cost of borrowing for these firms.

element of social cost (loss of storm protection) is borne locally, suggesting that local public policy intervention is required. Nevertheless, it remains the case that to the extent the benefits of conservation are enjoyed on a wider scale (e.g. carbon sequestration, inter-national ecotourism and non-use values), some international compensatory transfer maybe necessary to support a conservation strategy.

Coral reefs represent another ecosystem under severe pressure and a comparative analysis of costs and benefits is shown in Table 9 (based on studies by under different resource extraction and management scenarios; Cesar, 1996; White et al., 2000). The results highlight, as in the case of mangroves, the pressures on local ecosystems as local users seek to maximise short term private financial benefits from resource exploitation. Over

Table 8
Benefits and costs of mangrove conversion and conservation options in Thailand

Goods/service (US $ ha^{-1})	Conserved mangrove	Degraded mangrove/shrimp farming
Direct extractive use (timber and NTFPs, including charcoal production)	1188	154
Off-shore fisheries (open access and managed conditions)	91	–
Storm protection (based on historical costs avoided for not having to replace engineered structures)	3285	–
Carbon sequestration @ US $6.23 tonnes^{-1} and 15.125 tonnes c ha^{-1} per year)	94	94
Shrimp revenues (ponds last for 5 years)	–	3513
Pollution damages to rice farmers	–	(401)
Total annual private benefits	1279	3667
NPV per ha at 6% for 30 years	52 875–67 920	14 831–18 540[a]
NPV per ha at 3% for 30 years	73 161–94 045	23 995–30 627[a]
NPV excluding storm protection	4952–27 735	

Source, adapted from Sathirathai (1998).
[a] NPV includes net revenues and other costs, and gradual rehabilitation to a fully functioning mangrove at a cost of $16 138 ha^{-1}.

Table 9
Coral reef exploitation and conservation options (NPV at 10% discount rate over 25 years)

Options	Private benefits ($ km^{-2})	Social benefits/costs[a]
Poison fishing	37	47–523
Blast fishing	16	108–836
Coral mining	133	193–991[b]
Intensive fishing	42	120
Hook and line fishing (sustainable)[a]	4	698

Source: adapted from Cesar (1996) and White et al. (2000).
[a] Aggregation of fisheries, coastal protection and tourism functions values forgone.
[b] Includes forest logging damage cost (of 74), when timber is used as a fuel for processing coral into lime.

time large social costs are incurred as the reefs' capacity to provide tourism recreation, storm protection and other services diminish.

5. Findings and future research directions

Our survey of the valuation literature has shown that there are very few studies which encompass a range of interdependent ecological functions, uses and values at a given site; or which track site changes in values across different states of ecological disturbance. But it is just this type of study that is of great relevance to decision makers faced with the complex trade-off between local, national and global conservation net benefits and development (requiring land use change) net benefits. There is an urgent need for more research studies of this type to complement and extend the current environmental valuation knowledge stock. This data can then be combined with socio-political and socio-cultural knowledge to better inform sustainable development projects, policies and programmes.

It is also the case that the research that has been undertaken is increasingly pointing to the fact that ecosystem conservation strategies cannot be fully justified on economic grounds without taking into account a reasonably full complement of functional uses/non-uses and values. But the conservation costs and benefits are distributed across markedly different types of stakeholder recipients, from local users deriving short-term benefits in terms of consumption and/or income from locally marketed products, to citizens of other countries deriving welfare from the long-term indirect-benefits from ecosystem services such as carbon sequestration, direct uses such as tourism and non-use motivations.

Current global development trajectories confirm that income disparities are increasing and most countries will not meet the United Nation's goals for human development and poverty eradication by 2015. Our findings show one reason why this is the case. Continuing conversion of natural ecosystems is usually undertaken for private benefits and even when locally captured these benefits often fail to filter down to subsistence users-increasing poverty and inequality. This cost is in addition to the social costs incurred if ecosystem conservation strategies are not pursued. Conversely, conservation of natural ecosystems in a way that balances environmental and developmental goals—for example, by sharing benefits and by maintaining regulated local access to important subsistence uses—offers one way to address rural poverty and inequality. This is also necessary for the effectiveness of conservation programs, as their viability ultimately depends on compliance at the local level.

A variety of institutional arrangements, often tailored to local circumstances, will be required in order to facilitate an efficient and more equitable 'capture' of ecosystem benefits. A strategy based solely on maximum commercialisation of ecosystems, or international compensatory transfers, is unlikely to be optimal. Local property rights arrangements will also be significant factors to take into account. At the macro-scale, any future ecosystem conservation strategy will need to combine protected area/zoning provisions, sustainable utilisation practices and supporting appropriate tenure systems, and international resource transfers to compensate for 'local' foregone opportunities. International compensation is a particular requirement in our type 4 ecosystem conversion versus conservation contexts, mangroves and coral reefs, as well as in the case of type 2 tropical forests.

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