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# Ecosystem valuation: some principles and a partial application[†]

Sian Morse-Jones[a*], Tiziana Luisetti[a], R. Kerry Turner[a] and Brendan Fisher[a,b]

**Understanding the economic value of nature and the services it provides to humanity has become increasingly important for local, national and global policy, and decision-making. However, problems arise in that it is difficult to obtain meaningful values for goods and services that ecosystems provide which have no formal market, or are characteristically intangible. Additional problems occur when economic methods are applied inappropriately and when the importance of ecosystem maintenance for human welfare is underestimated. In this article, we provide clarification to practitioners on important considerations in ecosystem services valuation. We first review and adapt definitions of ecosystem services in order to make an operational link to valuation methods. We make a distinction between intermediate and final ecosystem services and also identify non-monetary ways to incorporate regulatory and support services into decision-making. We then discuss the spatially explicit nature of ecosystem service provision and benefits capture, and highlight the issues surrounding the valuing of marginal changes, nonlinearities in service benefits, and the significance of non-convexities (threshold effects). Finally, we argue for a sequential decision support system that can lead to a more integrated and rigorous approach to ecosystem valuation and illustrate some of its features in a coastal ecosystem management context.**


**Keywords:** ecosystem services; ecosystem valuation; economic valuation

## 1. INTRODUCTION

Despite increasing recognition of the importance of ecosystems and biodiversity for human welfare, they continue to decline at an unprecedented rate (GEF, 1998; Chapin *et al.*, 2000; Koziell, 2001; CBD, 2006). In many cases, the losses are irreversible, posing a serious threat to sustainable development and to human well-being in general (Millennium Ecosystem Assessment, 2005; Loreau *et al.*, 2006). In view of this, efforts have been made to increase the understanding of the importance of ecosystems and their conservation in the face of developmental pressures. However, despite a dramatic increase in the number of studies aiming to value ecosystem services, there is a lack of consensus over the validity and implications of ecosystem valuation (Pagiola *et al.*, 2004).

The valuation of ecosystem services is a complex process that is reliant on the availability of relevant and accurate biophysical data on ecosystem processes and functions but also the appropriate applications of economic valuation. This presents a challenge to policymakers looking to make practical use of such studies. In this paper, we provide clarification to practitioners on important considerations in ecosystem services valuation and review how the literature has dealt with these issues to date. We propose a sequential decision support system aimed at producing a more rigorous approach to ecosystem valuation by ensuring explicit consideration of these issues. We illustrate how this system is being used to derive meaningful values for ecosystem service benefits provided by coastal resources in the U.K. We begin by reviewing where valuation fits into the overall ecosystem services framework and discuss ecosystem service definitions.

## 2. ECOSYSTEM SERVICES: FRAMEWORK AND DEFINITIONS

In the past, conservation efforts have been justified on scientific and moral/ethical grounds, yet continued and increasing pressures on natural resources suggests that these arguments may no longer be enough (Turner and Daily, 2008). Due to the serious and urgent nature of the problem, considerable effort has recently been directed toward understanding the relationship between natural capital and human welfare. This has seen a move toward more systematic approaches to measuring, modeling and mapping of ecosystem services, governance analysis, and valuation (Cowling *et al.*, 2008; Naidoo *et al.*, 2008; Turner and Daily, 2008; Fisher *et al.*, 2008b).

* *Correspondence to: S. Morse-Jones, CSERGE, School of Environmental Sciences, University of East Anglia, ZICER Building, Norwich NR4 7TJ, U.K. E-mail: s.morse-jones@uea.ac.uk*

a *S. Morse-Jones, T. Luisetti, R. K. Turner*
*The Centre for Social and Economic Research on the Global Environment (CSERGE), University of East Anglia, Norwich NR4 7TJ, UK*

b *Program in Science, Technology and Environmental Policy, Woodrow Wilson School of Public and International Affairs, Princeton University, Princeton, NJ 08544, USA*

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For example, the ecosystem services approach (ESApp), after Fisher *et al.* (2008b), explicitly recognizes the critical role healthy ecosystems play in sustaining human welfare, in poverty alleviation, and in economic development. It sets out a semi-cyclical process for combining information on ecosystem services production, flow, beneficiaries, costs and benefits, and governance structures, in a spatial context, to help redress natural capital depletion (see Figure 1). It begins with a scoping exercise where the overall landscape (or seascape) is categorized, services are identified and key stakeholders are engaged. Service production, flow, beneficiaries, and cost and benefits are then measured and mapped—this involves expertise in understanding biophysical processes and functioning—to produce a map of the distribution of "winners and losers." Modeling is often required in order to aggregate service flows and benefit values over a range of spatial scales. This process is then repeated under different future scenarios. The process "ends" with an assessment of policy outcomes and re-evaluation, which may necessitate another cycle.

Valuation fits into this framework at steps 5 and 6, measuring, modeling and mapping of benefits and costs of conserving services. By estimating the economic value of ecosystem services in monetary terms we have a common, comparable unit with which to assess trade-offs. This information can then be used to demonstrate the importance of ecosystem services; to evaluate different policy interventions; to examine how costs and benefits are distributed across society; to design appropriate mechanisms for benefits capture; and so on. In short, the primary aim of ecosystem valuation is to be able to make better (more efficient or more cost effective) decisions regarding the sustainable use and management of ecosystem services. But the conceptual framework itself is not restricted to processes culminating in monetary valuation.

Economic valuation cannot value everything—that is, not all benefits provided by ecosystem services are fully translatable into economic terms. For example, it cannot be applied to measure values that are non-anthropocentric, or anthropocentric values that cannot be

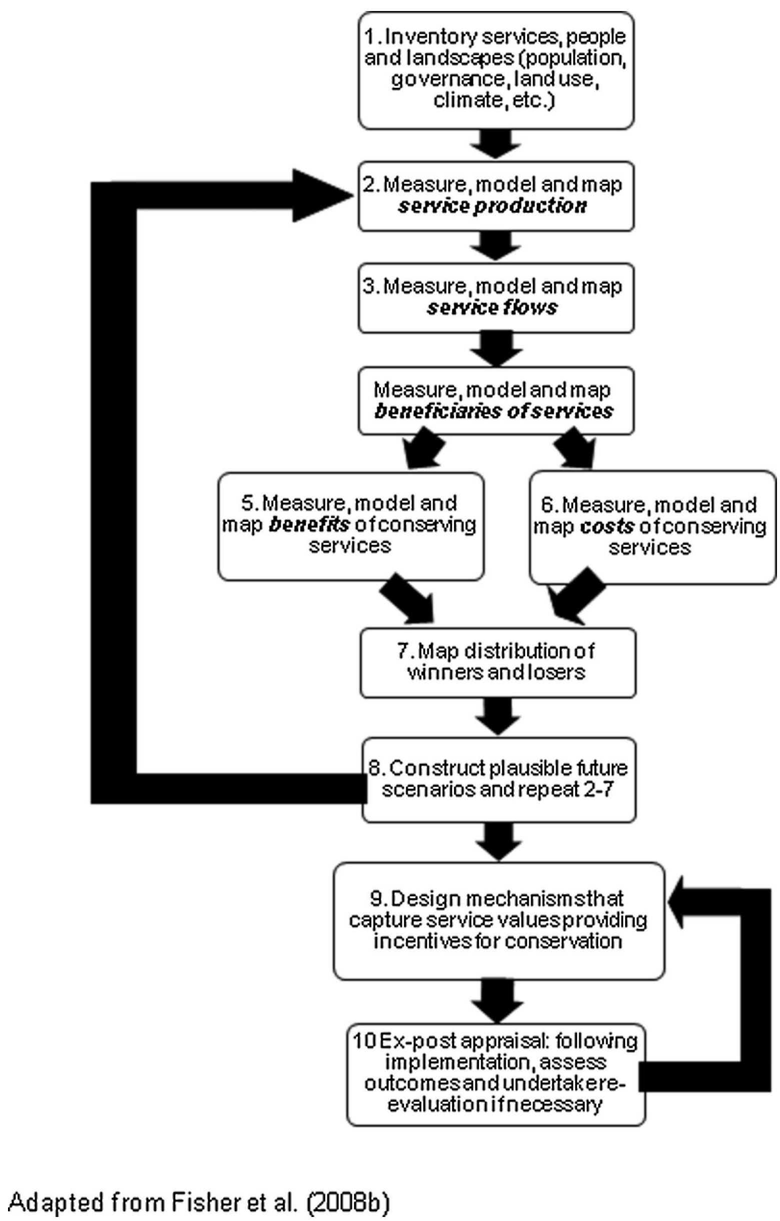


**Figure 1.** The ecosystem services approach

meaningfully expressed in monetary terms due to irreversibility concerns. To elaborate, ecosystems have been argued to provide socio-cultural, intrinsic values, or ecological values (Millennium Ecosystem Assessment, 2003) which may not be captured by economic valuation techniques, but may be important to decisions concerning the conservation (or conversion) of ecosystem services.

Socio-cultural values include, for example, a person's sense of identity with a particular place or landscape, the value attached to sacred or spiritual forests, or historical values; these may be identified through a wide range of qualitative exercises such as focus group discussions or citizen juries.

Ecological values relate to the importance of the various attributes or functions of an ecosystem, irrespective of any direct contribution to human well-being. For example, the value of one species to the survival of another species (Farber *et al.*, 2002), or the indirect use value associated with support and regulatory environmental services that underpin the eventual provision of service benefits, such as, for example, clean water. The latter acknowledges that an ecosystem must have a minimum level of process and structure in order to deliver services. This ''glue'' value (and what others variously term, infrastructure value, insurance value, and option and quasi-option value (Barbier, 1994; Gren *et al.*, 1994; Costanza *et al.*, 1997)), is over and above the value of the services themselves and is not a monetary value. In view of this, aggregation of the main service-based economic values provided by a given ecosystem will provide an estimate of the total economic value but this may not be the equivalent of the total system value. Clearly, the preservation of these regulatory and support services (as defined by the Millennium Ecosystem Assessment (MEA) see below), is critical to the sustainable supply of ecosystem services that do directly contribute to human welfare. However, it is frequently the case that due to ecosystem complexity we do not know the minimum level of stock, structure or process required to deliver these services. Indeed, forecasting future changes in the state of ecosystems is very difficult (Balmford and Bond, 2005). A huge range of ecosystems is already under pressure and showing signs of stress. Even broadly extrapolating these trends suggests that both the amount and variability of nature will continue to decline over much of the planet (Balmford and Bond, 2005). While establishing early warning systems to signal the proximity of the threshold points would be desirable, it is not yet clear that we are able to anticipate such ''tipping points'' with sufficient accuracy (see Lenton *et al.*, 2008). Consequently, ensuring the adequate provision of ecosystem services will require taking a precautionary approach when evaluating trade-offs and recognizing the ''insurance'' value of a sufficient amount of intact ecosystem structure and process.

In order to place an economic value on ecosystem service benefits (or costs) we need to understand precisely what is being valued (Barbier, 1994; Costanza *et al.*, 1997; Daily, 1997; Farber *et al.*, 2002; Pagiola *et al.*, 2004; Turner *et al.*, 2008). A multitude of definitions and classification schemes for ecosystem services exist (Costanza *et al.*, 1997; Daily, 1997; Turner *et al.*, 2000; Millennium Ecosystem Assessment, 2005). One of the most widely cited is the MEA definition, which describes ecosystem services as ''the benefits that people obtain from ecosystems'' (Millennium Ecosystem Assessment, 2005). It classifies services into *supporting*, *regulating*, *provisioning*, and *cultural*. This framework provides an excellent platform for moving toward a more operational classification system which explicitly links changes in ecosystem services to changes in human welfare. By adapting and re-orienting this definition, it can be better suited to the purpose of this paper, with little loss of functionality. Wallace (2007), for example, has focused on land management, while Boyd and Banzhaf (2007) and Maler *et al.* (2008) take national income accounting as their policy context and make a critical distinction between services and benefits.

Building on this, Fisher and Turner (2008) proposed that ''ecosystem services are the aspects of ecosystems utilized (actively or passively) to produce human well-being.'' Their definition of services differs from that of Boyd and Banzhaf (2007) in that they consider processes or functions to be services as long as there are human beneficiaries. Also, an intermediate service is one which influences human well-being indirectly, whereas a final service contributes directly. Classification of this type is context dependent, for example, clean water provision is a final service and benefit (i.e., direct change in human welfare) to a person requiring drinking water, but it is an intermediate service to a recreational angler, who requires a final service in terms of the provision of the fish population in order to get recreation enjoyment (final welfare benefit). For the purpose of economic valuation, we are interested in the final benefits only.

## 3. IMPORTANT CONSIDERATIONS IN ECONOMIC VALUATION

A number of problems have to be resolved if an *appropriate* economic valuation of ecosystem services is to be carried out. These are: spatial explicitness, marginality, the double-counting trap, and the challenges of dealing with nonlinearities in benefits and threshold effects. If resulting welfare estimates (the monetary equivalents in terms of, for example, willingness to pay (WTP) or to be compensated for a loss) are to be meaningful and appropriate it is considered imperative that these issues receive due consideration when undertaking ecosystem valuation. In this section, we review why these issues are important and consider how the valuation literature has dealt with these issues to date.

### 3.1. Spatial explicitness

First and foremost, it is critically important to clarify the degree of certainty/uncertainty surrounding the underlying biophysical structure and processes through spatially explicit models of any given ecosystem service. This contextual analysis must then encompass appropriate socio-economic, political, and cultural parameters in order to properly identify ecosystem services supply and demand, and gainers, and losers.

The requirement for spatially explicit ecosystem valuation is based on the recognition that ecosystem services are context dependent in terms of their provision and their associated benefits and costs. That is, service values may change across the landscape due to geographical variations in biophysical supply or demand, for example, how scarce or abundant clean water is, or how large the adjacent population is or how wealthy they are.



  

The importance of this point for decision-making is neatly illustrated by Naidoo and Ricketts (2006) in a cost–benefit analysis (CBA) of three potential conservation corridors in Mbaracayu Biosphere Reserve, Eastern Paraguay. The three corridors were potentially equivalent; however, one corridor was found to generate net benefits approximately three times greater than the other corridors. The disparity was due to variability in spatial factors, such as, land tenure, slope, and soil type which affected ecosystem service provision across the landscape. Explicitly incorporating the spatial context into the CBA was critical in obtaining unbiased estimates of both the costs and benefits of ecosystem provision, and, crucially, in enabling conservation planners to identify the most economically efficient location for the conservation corridor.

In a different example, Luisetti *et al.* (2008) illustrate the importance of spatial context in aggregating benefits of new wetland creation on the east coast of England. The authors used a site-specific choice experiment survey to elicit ecosystem service values from the regional population closest to the proposed scheme.[1] Importantly, the distance attribute was found to be significant and negative, indicating that utility of the salt marsh declines as distance from the site increases—the so called "distance decay effect." This meant that assuming a constant unit value across populations for a specified change in ecosystem service provision would have led to biased estimates. Correspondingly, aggregate WTP was estimated using a spatially sensitive valuation function with distance bands at 8-, 15-, 23-, and 32-mile intervals from the proposed site. WTP was calculated by multiplying the mean household WTP for each band by the total population within the band. By incorporating the "distance decay" effect, Luisetti *et al.* (2008) were able to sensitize aggregate benefits to the socio-economic context.

The key point illustrated by these examples is that if we fail to take into account spatial variability in ecosystem supply and demand, we risk over- or under-estimating ecosystem values. Notably, a central feature in both was the use of a GIS, which is emerging as a valuable tool in ecosystem valuation (see for example: Lovett *et al.*, 1997; Troy and Wilson, 2006). Providing ecosystem values in a spatial format, through the use of a geographical information system (GIS), can be extremely useful to planners and policymakers, for example, by highlighting where opportunities for payments for environmental services may occur. It is anticipated that the incorporation of spatial factors in ecosystem valuation is likely to become easier and more commonplace as access to GIS software and expertise increases.

Temporal aspects are also very important. Ecosystems are subject to both "natural" and human-induced change over a range of temporal scales. The rate and extent of the environmental change process is often uncertain. But it is also the case that human preferences, values, and social norms change over time. In economic analysis, the discounting procedure is used to transform a flow of costs and benefits which are incurred over a long period of time into a so-called net present value (NPV). We return to discuss the discounting procedure and the controversy that surrounds its application in the case study section.

### 3.2. "Marginal" analysis

Economic valuation works best when so-called "marginal" environmental changes are being assessed.[2] The focus is typically on relatively small, incremental changes rather than large state changing impacts (Pearce and Turner, 1990; Pearce, 1998; Turner *et al.*, 1998; Bockstael *et al.*, 2000; Turner *et al.*, 2003).

Figure 2 shows the marginal supply (cost) and marginal benefit (demand) curves for an essential ecosystem service. The vertical axis measures economic value or price, and the horizontal axis measures the flow or quantity of ecosystem service. The marginal supply curve is vertical because the supply of ecosystem services does not tend to increase or decrease in relation to economic systems (Costanza *et al.*,

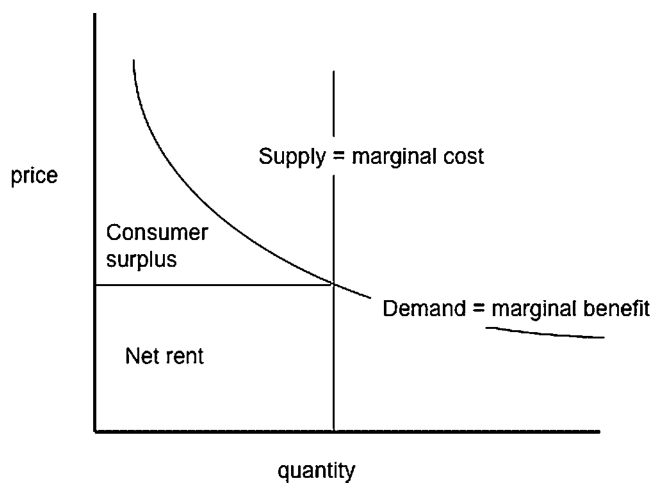


**Figure 2.** Supply and demand curves showing the definitions of cost, net rent, and consumer surplus for an essential ecosystem service

[1]The good was described in terms of five attributes: salt marsh area; number of protected bird species observable; distance from the respondent's home; accessibility; and price.
[2]Specifically, the focus on small-scale changes was to avoid large income effects (which may change the marginal utility of income) and violations of the ceteris paribus assumption (changes in other prices) (Spash and Vatn, 2006).

  

1997). The downward sloping demand curve shows how much individuals are willing to pay for an incremental amount of ecosystem service. The shape indicates that the more scarce the ecosystem service, the more an additional unit is valued (or, conversely, the more of a service we have, the less we value an additional unit). Given ecosystem services are only substitutable up to a point, as the quantity available approaches zero (or some necessary minimum provision of ecosystem structure and processes) the demand curve approaches infinity. The area beneath the demand curve indicates the *total* economic value of the ecosystem service, however, as the curve is not bounded on the horizontal axis, this area cannot be properly defined. More specifically, policy decisions related to natural resources, as with decisions relating to marketed goods, typically involve trade-offs which occur at the margin. This means examining the value of the ‘‘next unit’’ at some point along the marginal benefit curve. However, given the scientific uncertainties which shroud ecosystem functioning it remains the case that it is often difficult to discern whether a given change is ‘‘marginal’’ or not and when thresholds are being approached or crossed (Turner *et al.*, 1998).

While it is appropriate to consider, as far as is feasible, economic value in terms of marginal changes, a review of the existing empirical literature suggests that in fact very few studies do so. Moreover, even fewer have undertaken marginal analysis of ecosystem transitions (Bulte and van Kooten, 2000; Balmford *et al.*, 2002; Turner *et al.*, 2003), instead the majority focus on ‘‘point estimates’’. Mahan *et al.* (2000), for example, produce marginal value estimates of the value of wetland amenities to properties in Portland, Oregon. The results indicate a property’s value increases by \$24.39 per *one acre* increase in the size of the nearest wetland. Maler *et al.* (2008) explicitly undertake marginal analysis in estimating the accounting price for the habitat service provided by a mangrove ecosystem to a shrimp population. Their model evaluates changes to fisherman well-being for a 10 ha change in a mangrove forest of 4000 ha in size, obtaining an accounting price of \$200 $ha^{-1}$. In most cases, the ecosystem valuation literature has focused on valuing the stock, for example, Peters *et al.* (1989) estimate the value of non-timber forest product (NTFP) services based on a stock inventory. Or, the actual service flow is valued, for example, Godoy *et al.* (2000) value actual NTFP service flows from a Central American rainforest, and Croitoru (2007) estimates annual flow of NTFP benefits for the Mediterranean region. While Adger *et al.* (1995) estimate the total economic value of Mexican forest services. In some cases, these analyses have been placed in a context of ‘‘change’’ by drawing comparisons with alternative land use options. For example, Peters *et al*. (1989) and Bann (2002) compare commercial timber extraction and NTFP harvesting for forests in Amazon and Cambodia, respectively. Yaron (2001) examines the total economic value of three service flows, sustainable forest use, small-scale agriculture and plantation agriculture, from forested lowland in the Mount Cameroon region, from the perspective of global, national, and local stakeholders. While point estimates are of interest, for policy it is most useful to have an understanding of the impact on welfare of a transition in state from ‘‘before’’ and ‘‘after’’ some policy since this helps with decisions regarding trade-offs.

At the practical policy level, since ecosystem services often cross scales, the decision on whether the ‘‘next unit’’ is meaningful in terms of marginal analysis is conditioned by the scale of the policy decision (Fisher *et al*., 2008). For example, at the local scale, the loss of an entire forest on which livelihoods are dependent, may be so catastrophic as to render the change meaningless (since lives would be so dramatically altered). A more relevant change may be to consider the loss of *part* of the forest. Conversely, at the global scale, the loss of an entire forest may be ‘‘marginal’’, at this scale it is not likely to be perceived as a catastrophic loss for life support or a severe impediment to the functioning of international economic systems. In practise, applying the concept to multi-scale systems may be confusing. As a guide, Fisher *et al*. (2008a) recommend considering the ‘‘next unit’’ in terms of the geographic extent a policy decision could encompass, for example, extending a forest within its national borders. Scenario-building is also proving to be a useful tool for defining marginal changes, and offers the opportunity for stakeholder participation which can result in the identification of more realistic and relevant changes (see Fisher *et al*., 2008b).

While a range of valuation methods and techniques are available the economics literature contains agreed guidance on which techniques work ‘‘best’’ in any given context (Bockstael *et al.*, 2000; Bateman *et al.*, 2002; Pagiola *et al.*, 2004; Soderqvist and Soutukorva, 2006).

### 3.3. Double counting

Another widely recognized issue concerns the potential problem of double counting (Barbier, 1994; Turner *et al.*, 1998; de Groot *et al.*, 2002; Millennium Ecosystem Assessment, 2003; Turner *et al.*, 2003; Boyd and Banzhaf, 2007). This may occur where competing ecosystem services are valued separately and the values aggregated; or, where an intermediate service is first valued separately but also subsequently through its contribution to a final service benefit. For example, the value of a forest ecosystem for clearance timber logging should not be added to the value of the same forest patch for recreational benefits since the former will likely preclude the later. Nor should the value of a pollination service, which is already embodied in the market price of a crop, be counted separately unless the value of its input to the crop is deducted.

In essence, double counting is a feature of the complexity of ecosystem functioning and uncertainty surrounding our understanding of the systems and their interlinkages. Unfortunately, there are cases where researchers have incorrectly summed values in order to obtain aggregate estimates of ecosystem value (evidence from Fisher *et al.*, 2008a). It is thus essential that that the analyst has a clear understanding of the various overlaps and feedbacks between services when undertaking aggregation (de Groot *et al.*, 2002; Turner *et al.*, 2003). Only a relatively small number of studies in the valuation literature have directly sought to address this. For example, in a CBA of a UK coastal managed realignment (MR) policy, Turner *et al.* (2007) avoid the double-counting trap by treating the environmental benefits provided by the creation of intertidal habitats as a composite value. Based on a recent meta-analysis (see Woodward and Wui, 2001), they used an estimate of £621 $ha^{-1}$ $year^{-1}$ which incorporated the contribution of nutrient storage function (for nitrogen and particle reactive phosphorus) on the basis that this provided an intermediate service, via improved water quality, to the final benefit of enhanced amenity and recreational quality. Other benefits provided by the salt marsh which could be valued and legitimately aggregated include storm buffering costs savings on hard defences, carbon storage, and fish productivity gains (Luisetti *et al*., 2008). In another example, Hein *et al*. (2006) suggest only including regulation services in valuations if ‘‘(i) they have an impact outside the ecosystem to be valued; and/or (ii) if they provide a direct benefit to people living in the area (i.e., not through sustaining or improving another service)'' (p. 214). A classification scheme recommended by Fisher and Turner (2008) helps to avoid the problem by drawing a clear distinction between intermediate services, final services, and benefits, with only the latter being subject to economic valuation.

### 3.4. Nonlinearities

The existence of nonlinearities in ecosystem functioning and service provision adds further complexities to the process of valuation and subsequent management. Many ecosystems typically respond nonlinearly to disturbances, their functioning may seem to be relatively unaffected by increasing perturbation, until suddenly a point is reached at which a dramatic system changing response occurs, for example, in the ecology of phosphorus-limited shallow lakes which can flip suddenly from one state to another (Arrow *et al.*, 2000; Turner *et al.*, 2003). We take a closer look at so-called threshold effects in the next section. At the level of individual service provision one cannot always make the assumption that marginal benefit values are equally distributed. For example, the storm protection benefit of a unit increase in mangrove habitat area may not be assumed to be constant for mangroves of all sizes due to nonlinearities in wave attenuation (Barbier *et al*., 2008).

Indeed, Barbier *et al*. (2008) have stressed that for some ecosystems, for example, coastal mangroves, salt marshes, and other marine ecosystems, as habitat variables such as size of area alter, the services provided tend to change in a nonlinear way. They claim that recognizing such nonlinearities opens up the choice set available to policymakers. In the case of mangroves and the storm buffering service they provide, it is argued that the nonlinear supply of the buffering service (i.e., reducing as successive landward zones of the mangrove forest are crossed) means that some mangrove conversion (e.g., to provide space for shrimp ponds) can be economically justified in cost–benefit terms. This is illustrated using a case study of a mangrove in Thailand where choices exist between conversion of the mangrove to shrimp aquaculture and preservation to ensure the supply of key ecosystem services of coastal protection, fish habitat, and wood products. Data for mangroves indicated there are quadratic and exponential decreases in wave height with increasing habitat distance inland from shorelines. If linearity is assumed, the aggregate value is highest when the entire mangrove is preserved ($18.98 million). However, if a nonlinear mangrove wave attenuation function is assumed, results indicate that small losses in mangrove area will not cause major losses in economic benefits, so that the aggregate value is now highest ($17.5 million) when up to 2 $km^2$ of mangrove is converted to shrimp farm and the remainder is preserved. Consequently, very different policy outcomes may be supported depending on whether nonlinearities are recognized or not.

Critically, the authors note that an ''up to 20%'' conversion rule seems to be an emerging policy principle. But such generalizations are dangerous because ecosystem services must be assessed in a spatially explicit manner and with due regard for uncertainties surrounding possible threshold effects. In the mangrove example, it matters crucially where the shrimp ponds are located and what the current degradation status of the mangrove forest is. If the shrimp ponds are located on the seaward edge of the mangroves they will be prone to storm damage and lost productivity. If the mangrove has already experienced significant degradation it may be at or close to a threshold tipping point. Finally, mangroves (and other ecosystems) supply a range of interconnected services the value of which needs to be included in any economic benefit and loss account.

### 3.5. Threshold effects

The threshold effect refers to the point at which an ecosystem may change abruptly into an alternative steady state (Scheffer *et al*., 1993; Scheffer *et al*., 2001). For marginal analysis to hold true, the ''next unit'' to be valued should not be capable of tipping the system over a functional threshold or safe minimum standard (SMS) (Turner *et al*., 2003; Fisher *et al*., 2008a).

The SMS represents the minimum level of a well-functioning ecosystem which is capable of producing a sustainable supply of service. Conceptually, looking at Figure 2, this means that marginal analysis should be conducted far away from the point at which the demand curve increases sharply to infinity. In practise, this requires knowledge of the location of the SMS zone. Of course, due to the considerable uncertainty surrounding ecosystem functioning this introduces complexity since it is often far from clear when a threshold may be reached (Turner *et al*., 2003). Lenton *et al*. (2008) illustrate how to identify, at the sub-continental scale, *systems* which are likely to cross ''tipping points'' (threshold effects) and are relevant to policy and accessed by humans (tipping elements). For example, the overturning of the Atlantic Thermohaline Circulation (ATC) due to freshwater input to North Atlantic may have a significant impact on climate change, with regional cooling, sea level changes, and a shift in the intertropical convergence zone; this represents a key ''tipping element'' for the Earth's climate system. The authors suggest that tipping points may then be located using predictive models and historical data (Lenton *et al*., 2008); for instance, a ''degenerate fingerprinting'' method has been used to detect thresholds in the ATC by using time series output from a model of intermediate complexity (Held and Kleinen, 2004). This promising line of inquiry is still in an embryonic state and will take time to mature into an operational strategy.

While identifying this hazardous zone will require expert input from ecologists, risk analysts and others, it ultimately requires ethical/political choices to be made. These choices can be informed by economic opportunity cost analysis related to the range of conservation options that may be proposed. The challenge in incorporating threshold effects in ecosystem services valuation lies in our relatively limited knowledge of ecosystem complexity and interrelationships. Moreover, individual valuation studies frequently do not have the resources to undertake complex biophysical modeling. Consequently, the importance of threshold effects is often acknowledged in the valuation literature but rarely explicitly incorporated (Dasgupta and Maler, 2003). Soderqvist *et al*. (2005) apply the travel cost method to value the benefit of a bigger fish catch to recreational fishers in the Stockholm Archipelago. The results indicate that doubling the average spring catch per hour of Perch from 0.8 to 1.6 kg amounts to a WTP of 56 SEK per angler. While on the surface this appears to be a small (i.e., marginal) change, it is possible that the cumulative effect of doubling fish catch per hour could result in flipping the recreational fishery into an alternative state.



  

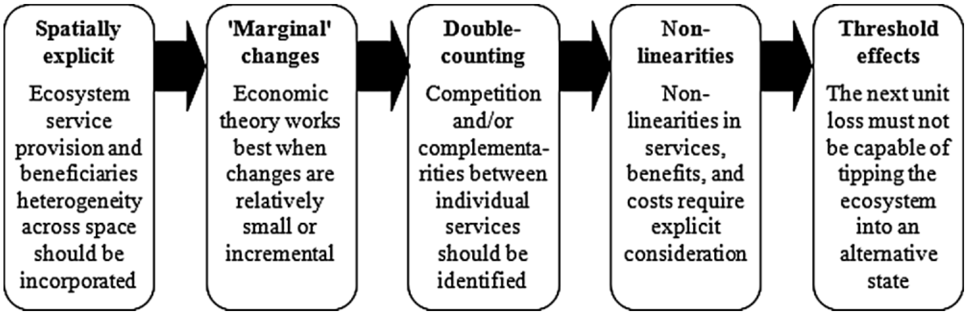


**Figure 3.** Ecosystem services: a framework for appropriate economic valuation

Indeed, the authors note the need for further work in order to assess the potential effects (and costs) of measures that would improve fishing conditions in the archipelago. In a different example, Hein (2006) explicitly incorporates threshold effects in modeling the optimum eutrophication control for a shallow lake ecosystem. Information on the supply of ecosystem services, the costs of eutrophication control measures and the response of the lake to reduced nutrient loading (including the threshold effect) was combined in one ecological–economic model, to calculate the net benefit of eutrophication controls for the four biggest lakes in De Wieden wetland, the Netherlands. Uncertainty regarding the point at which a switch to a clear water system occurs (the threshold) was incorporated via a sensitivity analysis. Threshold values were found to have a significant impact on the analysis.

In summary, to be most useful for policy, ecosystem services must be assessed within their appropriate spatial context and economic valuation should provide marginal estimates of value (avoiding double counting) that can feed into decisions at the appropriate scale, and which recognize possible nonlinearities and are well within the bounds of SMS. Progress toward this goal has so far been steady but many uncertainties remain.

## 4. SEQUENTIAL DECISION SUPPORT SYSTEM FOR ECOSYSTEM VALUATION

In order to operationalize these issues in ecosystem valuation, we propose a sequential decision support system (see Figure 3) which begins with: (i) understanding and incorporating the heterogeneous nature of ecosystem service provision and beneficiaries across space; (ii) invoking the concept of marginal analysis *albeit* with inevitable data deficiencies; (iii) the identification of and subsequent avoidance of the double-counting trap; (iv) explicit consideration of nonlinearities in services, benefits and costs; and finally (v) a coping strategy for dealing with threshold effects in marginal analysis.

The support system is structured as a linear process rather than cyclical because we are moving from a starting point of ‘‘ecosystem services’’ to an output of ‘‘values.’’ By starting with ‘‘spatial explicitness,’’ we help to avoid confusion later by clearly identifying at the outset what ecosystem services are present, how their stocks and flows vary across the landscape and the potential supply and demand side beneficiaries. In this sense, it forms a ‘‘scoping exercise’’ and provides a definitional position from which to decide on the appropriate type of valuation approach. If economic valuation is decided upon, the next step is to define the policy change of interest in terms of a ‘‘marginal’’ change in ecosystem provision, taking into account context and scale, and ideally, in terms of a ‘‘before’’ and ‘‘after’’ transition. Having identified the relevant marginal change, impacts on service provision can then be classified into intermediate services, final services, and benefits; the latter being the focus of economic valuation. This classification helps to avoid double counting. Moreover, the refinement of the policy change of interest into impacts on ecosystem services production and potential human welfare impacts brings us to a point where consideration can be given to possible nonlinearities and threshold effects. These are key features that need to be identified and analyzed, and which may require considerable expertise. The sequential order is extremely important since ‘‘jumping in’’ at any point increases the risk of under or over estimates and/or the need for additional post-valuation re-appraisal. For example, adopting this sequence in the Barbier *et al.* (2008) case study, offers the advantage of highlighting at the outset the importance of considering spatial variations in service production and benefits flow thereby producing a more realistic set of policy outcomes. We argue that these sequential steps are the necessary and sufficient elements in any ecosystem services assessment and decision support system, and may help to guide future ecosystem valuation studies toward more rigorous valuation estimates.

## 5. CASE STUDY: COASTAL REALIGNMENT SCHEME APPRAISAL, EASTERN ENGLAND

A partial deployment of the framework outlined in Figure 3 has been undertaken within an analysis of salt marsh ecosystem creation linked to MR schemes in the coastal areas of south east England (Luisetti *et al*., 2008).

Two recent UK case studies on estuaries (Humber and Blackwater, England) examining the re-orientation of the UK coastal zone policy toward a more flexible and adaptive approach implementing MR schemes, followed an ESApp and put into practice the sequential steps scheme in Figure 3. Estuaries and salt marshes yield a range of benefits in terms of ecosystem services. Re-created salt marsh areas generate carbon storage benefits, which can be valued in terms of the damage cost avoided per tonne of $CO_2$; they improve fisheries’ productivity via nursery areas, and this gain can be valued via market prices for commercial species. They also generate recreation and amenity benefits as walking, bird watching and other recreational activities, and existence value benefits based on the services of biodiversity maintenance. An indication of the composite value of some of these amenity and related benefits can be got by transferring benefits data from the published literature if the spatial and other contextual variables are similar (this was done in the Humber estuary case study; Turner *et al*., 2007). Or, more properly, site-specific stated preferences studies can be conducted to estimate WTP values (this was the approach adopted in the Blackwater estuary case study; Luisetti *et al*., 2008).

The implementation of MR schemes recreating salt marshes, which can act as a more natural coastal defence for the coast (if the land elevation is appropriate) or provide protection in front of existing engineered defences, implies monetary savings on the sea walls maintenance costs. The capital costs of implementing the scheme would be: capital costs for realigning defences whenever a secondary line of defence may be required further inland; and opportunity costs associated with any agriculture land that is sacrificed as the old defences are breached.

In the coastal MR study on the Humber estuary (North-east England, UK) Turner *et al*. (2007) utilized GIS techniques to identify possible realignment sites in which the opportunity costs of realignment involving significant social justice/ethical concerns were minimized. Urban centers were assumed to be protected and therefore excluded from the analysis. However, the sacrifice of low quality agricultural land for the purpose of salt marsh re-creation was taken into account. The suitable areas of realignment identified via GIS layers were based on five geographical and socio-economic criteria (e.g., land use, transport links and other infrastructures, etc.), and four possible future policy scenarios: business-as-usual (BAU); policy targets (PT); deep green (DG); and extended deep green (EDP). The extent of realignment areas increases from the BAU through the EDG scenario. A status quo scenario, called hold the line (HTL), acted as a baseline against changes under the other scenarios evaluated. A complete do nothing strategy was not considered given the statutory duties imposed on coastal protection and sea defence agencies. In the analysis future sea level rise was taken into account in so far as current sea defence policy includes an allowance for increased risk in line with current climate science predictions. However, due to uncertainty over the loss of intertidal habitat because of coastal squeeze over the next 50 years, it was assumed that no further coastal squeeze takes place. If future sea level rise exceeds current expectations, or new estimates become available, both traditional and MR strategies will need to be reappraised.

The efficiency of MR was assessed by comparing benefits and costs of the scheme (see Table 1). The benefits of the intertidal ecosystem in the Humber estuary were identified using a basic ESApp: sea defences cost savings; carbon storage; and amenity and recreation. The results of the analysis showed positive NPVs for periods of time longer than 25 years—the longer the time horizon, the more economically efficient is the policy. Results are sensitive to the discounting procedure applied shifting the time at which the NPV becomes positive closer to the present, as one moves from the constant rate, to the declining and then to the gamma discount rate. We explain these terms in the following section.

A more recent study in the Blackwater estuary (Luisetti *et al*., 2008), located in Essex, UK, has tried to overcome some limitations in the Humber study. The same GIS methodology for the investigation of the suitable areas for realignment with the same criteria as for the Humber was applied. The same scenarios were considered with the exception of the BAU scenario. Building upon that study, a CBA assessing the efficiency of MR schemes was undertaken based on the ESApp of Fisher and Turner (2008) to identify the benefits of creating new salt marshes in the Blackwater estuary: flood/storm buffering and related cost savings on hard defences; carbon storage; fish production; and amenity and recreation provision (considered as a composite environmental benefit including biodiversity to avoid double counting of ecosystem services). Compared to the Humber study (see Table 1), a further benefit was considered namely fish production and the amenity and recreation benefit was estimated *in situ* with a stated preference technique: a survey-based choice experiment.

The use of site-specific (as opposed to benefit transfer data) value estimates derived via choice experiment has served to reinforce the positive NPV findings from the Humber study. As shown in Table 2, in the Blackwater study the present values (PVs) are positive for any MR

**Table 1.** Ecosystem services value estimates (£) used in the CBA assessing MR schemes in the Blackwater and the Humber estuary, respectively

| Blackwater estuary* | | Humber estuary† | |
|---|---|---|---|
| | £2007 | | £2005 |
| Carbon storage | £7–30/tC‡ | Carbon storage | £45/tC |
| Storm buffering (maintenance costs savings) | £433 $km^{-1}$ $year^{-1}$ | Storm buffering (maintenance costs savings) | £1780 $km^{-1}$ $year^{-1}$ |
| Recreation and amenity§ | £4,428,961, £5,790,713 and £6,430,145/year¶ | Composite environmental benefit|| (recreation and amenity) | £621 $ha^{-1}$ $year^{-1}$ |
| Fish productivity | £7.43/ha | | |

*Luisetti *et al*. (2008).
†Turner *et al*. (2007).
‡For the CBA discounting procedure a figure of £7/tC and £7.43 for fish productivity was adopted.
§Onsite survey.
¶Benefit estimates based on aggregate WTP values, using spatially sensitive valuation function.
||Benefit estimates based on benefits transfer.



  

**Table 2.** NPVs in the Blackwater case study using different discount rates (£ million)

| | 25 years | 50 years | 100 years |
|---|---|---|---|
| Discount rate: constant | | | |
| Policy targets (PT) | | | |
| NPV = PV(PT) – PV(HTL)* | 70.29 | 101.18 | 119.8 |
| Deep green (DG) | | | |
| NPV = PV(DG) – PV(HTL) | 76.71 | 117.21 | 141.59 |
| Extended deep green (EDG) | | | |
| NPV = PV(EDG) – PV(HTL) | 64.71 | 109.84 | 137.01 |
| Discount rate: declining | | | |
| PT | | | |
| NPV = PV(PT) – PV(HTL) | 70.29 | 156.21 | 315 |
| DG | | | |
| NPV = PV(DG) – PV(HTL) | 76.71 | 189.31 | 397.39 |
| EDG | | | |
| NPV = PV(EDG) – PV(HTL) | 64.71 | 190.18 | 422.05 |
| Discount rate: gamma | | | |
| PT | | | |
| NPV = PV(PT) – PV(HTL) | 82.9 | 169.41 | 353.5 |
| DG | | | |
| NPV = PV(DG) – PV(HTL) | 93.24 | 206.6 | 447.84 |
| EDG | | | |
| NPV = PV(EDG) – PV(HTL) | 83.14 | 209.46 | 478.26 |

Discount approaches and rates applied: constant (3.5%); declining (3.5% for years 1–30, 3% for years 31–75, and 2.5% for years 76–125) (HMT, 2003); and gamma discounting method (4% for years 1–5, 3% for years 6–25, 2% for years 26–75, and 1% for years 76–300) (Weitzman, 2001).
*The PV of the HTL scenario is always negative because it is assumed that, in the present scenario, benefits are zero.

scenario examined and for any period of time considered. This is mainly due to the higher benefits of realignment in comparison to its costs. In particular, due to the amenity and recreation benefit (composite environmental benefit), which dominate over carbon storage and fisheries benefits.

A sensitivity analysis applying different discounting approaches was conducted. That allows us to study the influence that each discount approach has on the costs and benefits involved in the appraisal of MR projects under the four scenarios considered (see Table 2). Constant rate discounting has traditionally been applied to assess public sector projects, and hence that was the discount rate conventionally used for assessing environmental projects. In simple terms, the higher the discount rate the lower the value of future costs and benefits. So if the discount rate is varied (i.e., lowered) after discrete periods of time (5 years, 25 years, etc.), the effect is to give more weight to long-term future benefits and costs. Declining discounting approaches have only recently been recognized as an alternative method to apply when long-term environmental projects are being appraised. Official guidance on UK public sector project/policy assessment now advocates such an approach (HMT, 2003). The reasoning behind the concept of declining discounting is its ability to deal more appropriately with the inter-generational dilemma. Recent research on discounting, such as Weitzman (1998), has suggested that a declining discount rate might be more appropriate in some social appraisal contexts. However, it can be argued that the weakness of Weitzman's (2001) gamma discounting approach is that the social discount rate is pre-determined without any moral reasoning in support of the chosen rate of pure time preference, or the ethical intra-generational decision (Dasgupta, 2008).

The ongoing debate around the correct discount rate to be applied in the social appraisal of long-term environmental projects has not reached a satisfactory conclusion. In the meantime, from our point of view, we believe that a more pragmatic approach to discounting for public project appraisal seems to be the application of a declining discount rate because of its inter-generational fairness bias. Following Turner *et al*. (2007), the PV of the proposed MR project has been calculated for a time horizon of 25, 50, and 100 years and with different discount rates: a constant rate (3.5%); a declining rate (3.5% for years 1–30, 3% for years 31–75, and 2.5% for years 76–125) following current HM treasury guidance for project appraisal (HMT, 2003); and the declining gamma (Weitzman, 2001) discounting method (4% for years 1–5, 3% for years 6–25, 2% for years 26–75, and 1% for years 76–300). The results of the CBA show that MR in the Blackwater estuary under any of the realignment scenarios presented yields an efficiency gain with any discount rate applied. As expected the gamma discount approach gives more weight to future costs and benefits than the other discount methods resulting in a higher PV of benefits.

## 6. CONCLUSIONS

In this paper, we have discussed the necessary conditions for making progress in ecosystem valuation and subsequent improved management, in line with advances in scientific knowledge and appropriate economic analysis. To achieve this, we have argued that it is necessary to consider a sequential analytical process which encompasses: (i) the spatial context of ecosystem service provision and beneficiaries; (ii) appropriate application of the concept of marginal economic analysis; (iii) avoidance of the double-counting trap; (iv) as far as is feasible a comprehensive understanding of the underlying biophysical relationships so that nonlinearities may be identified; and (v) full consideration of possible threshold effects via a precautionary approach based on an SMSs methodology. The latter will require a combination of economic opportunity cost analysis and political and ethical assessments. By stressing the importance of tackling these issues in a logical order this decision support system offers a structured approach to dealing with the inherent challenges associated with ecosystem valuation to deliver more robust value estimates and consequently more efficient and effective management.

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