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Ecosystem valuation: Some principles and a partial application

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**Ecosystem valuation: Some principles and a
partial application**

by

Sian Morse-Jones, R. Kerry Turner, Brendan Fisher
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CSERGE Working Paper EDM 10-01

ECOSYSTEM VALUATION: SOME PRINCIPLES AND A PARTIAL APPLICATION

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Abstract: 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, non-linearities 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 an Afro-montane and a coastal ecosystem context.

Keywords: Ecosystem services; Ecosystem valuation; Economic valuation.

Acknowledgements

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1 INTRODUCTION

Despite increasing recognition of the importance of ecosystems and biodiversity for human welfare, they continue to decline at an unprecedented rate (CBD 2006;Chapin et al. 2000;GEF 1998;Koziell 2001;MA 2005). In many cases the losses are irreversible, posing a serious threat to sustainable development and to human wellbeing in general (Loreau et al. 2006;MA 2005). 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 on 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 the Eastern Arc Mountains, Tanzania (www.valuingthearc.org); and its partial application in coastal zone management in the UK. We begin by reviewing where valuation fits into the overall ecosystem services framework and discuss ecosystem service definitions.

2 ECOSYSTEM SERVICES: FRAMEWORK AND DEFINITIONS

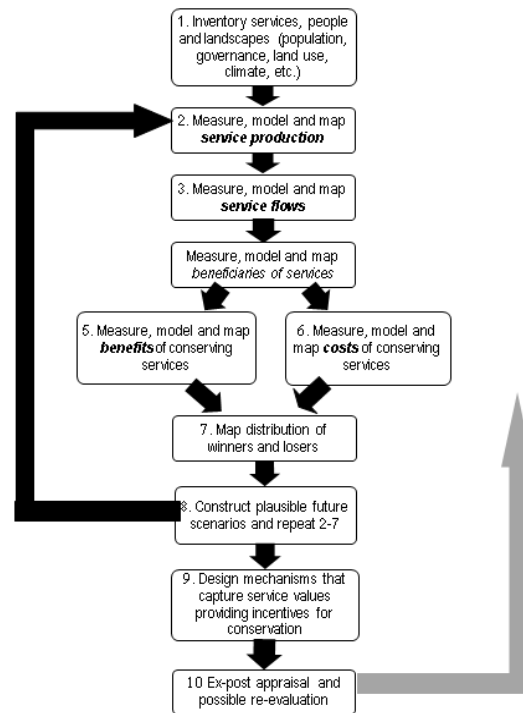
In the past, conservation efforts have been justified on scientific and moral/ethical grounds, yet continued and increasing pressures on natural resources suggests that these arguments may no longer be enough (Turner and Daily, 2008). Due to the serious and urgent nature of the problem, considerable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

For example, the Ecosystem Services Approach (ESApp), after Fisher et al (2008a), explicitly recognises the critical role healthy ecosystems play in sustaining human welfare, in poverty alleviation and 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, cost and benefits are then measured, modelled and mapped - this involves expertise in understanding biophysical processes and functioning - to produce a map of the distribution of 'winners and losers'. This process is then repeated under different future scenarios. The process 'ends' with policy appraisal and possible re-evaluation which may necessitate another cycle.

Valuation fits into this framework at steps 5 and 6, measuring, modelling 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.

Figure 1: The Ecosystem Services Approach



Adapted from Fisher et al 2008a

However, 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 meaningfully expressed in monetary terms due to irreversibility concerns. To elaborate, ecosystems have been argued to provide socio-cultural, intrinsic values or ecological values (MA, 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 nature area, the value attached to sacred or spiritual forests, or historical values; these may be identified through 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 wellbeing. For example, the value of one species to the survival of another species (Farber et al 2002), or the indirect use value associated with regulatory environmental services that support economic activities. 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, option and quasi-option value), 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 MA, 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 are 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 recognising 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 (Turner et al, 2008). A multitude of definitions and classification schemes for ecosystem services exist (MA, 2005; Daily et al, 1997; Turner et al 2000; Costanza et al 1997). One of the most widely cited is the Millennium Ecosystem Assessment definition, which describes ecosystem services as ‘the benefits that people obtain from ecosystems’ (MA, 2005). It classifies services into *supporting*, *regulating*, *provisioning* and *cultural*. This framework provides an excellent platform for moving towards 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 at hand, with little loss of functionality. For example, Wallace (2007), 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 wellbeing indirectly, whereas a final service contributes directly. Classification is context dependent, for example, clean water provision is a final service to a person requiring drinking water, but it is an intermediate service to a recreational angler. Importantly, a final service is often but not always the same as a benefit. For example, recreation is a benefit to the recreational angler, but the final ecosystem service is the provision of the fish population. For the purpose of economic valuation we are interested in the final benefits only.

3 IMPORTANT CONSIDERATIONS IN ECONOMIC VALUATION

In the economics literature 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 non-linearities in benefits, and threshold effects. If resulting welfare estimates 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 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 scheme¹. Importantly, the distance attribute was found to be significant and negatively signed, indicating that utility 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 willingness to pay (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. were able to sensitise 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 GIS, can be extremely useful to planners and policy makers, 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.

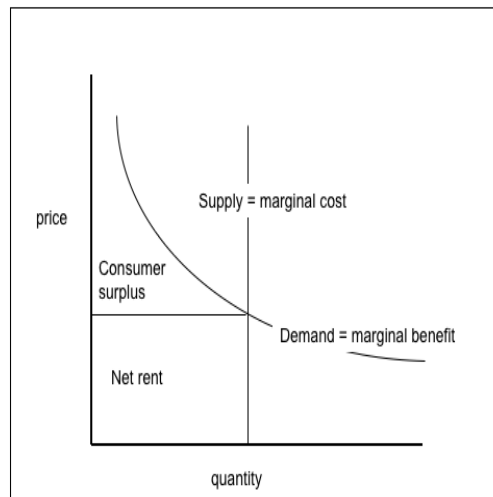
¹ The good was described in terms of five attributes: salt marsh area, number of bird species observable, distance from home, accessibility and price.

3.2 'Marginal' Analysis

Economic valuation works best when so-called 'marginal' environmental changes are being assessed². The focus is typically on relatively small, incremental changes rather than large state changing impacts (Pearce, 1998; Turner et al 1998; 2003; Bockstael et al. 2000; Pearce and Turner 1990).

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 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).

Figure 2: Supply and Demand Curves



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).

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, Polasky and Adams (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 wellbeing for a *10 hectare* change in a mangrove forest of *4000 hectares* in size, obtaining an accounting price of \$200/hectare. 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., 2008b). 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. (2008b) 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, 2008a).

3.3 Double-counting

Another widely recognised issue concerns the potential problem of double-counting (Barbier, 1994; Turner et al 1998; 2003; de Groot et al, 2002; MA 2003; Boyd and Banzhaf, 2006). 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., 2008b). It is thus essential 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 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 (Woodward and Wui, 2001), they used an estimate of £621/ha/yr 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 saltmarsh 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 Non-linearities

The existence of non-linearities in ecosystem functioning and service provision adds further complexities to the process of valuation and subsequent management. Many ecosystems typically respond non-linearly 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 non-linearities 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 non-linear way. They claim that recognising such non-linearities 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 non-linear 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 mill). 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 mill) when up to 2km² of mangrove is converted to shrimp farm and

the remainder is preserved. Consequently, very different policy outcomes may be supported depending on whether non-linearities are recognised or not.

Critically, the authors note that an 'up to 20%' conversion rule seems to be an emerging policy principle. But such generalisations 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; 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, 1998; Fisher et al., 2008b).

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 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 (ITCZ); 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 Atlantic Thermohaline Circulation 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 modelling. 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 0.8kg 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. Indeed, the authors note the need for further work in order to assess the potential

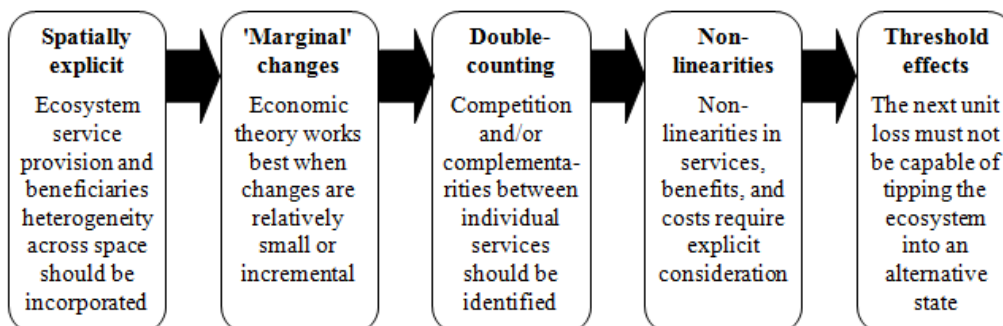
effects (and costs) of measures that would improve fishing conditions in the archipelago. In a different example, Hein (2006) explicitly incorporates threshold effects in modelling the optimum eutrophication control for a shallow lake ecosystem. Information on the supply of ecosystem services, the costs 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, 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 recognise possible non-linearities and are well within the bounds of SMS. Progress towards this goal has so far been steady but many uncertainties remain.

4 A SEQUENTIAL DECISION SUPPORT SYSTEM FOR ECOSYSTEM VALUATION

In order to operationalise 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 inequitable data deficiencies; (iii) the identification of and subsequent avoidance of the double-counting trap; (iv) explicit consideration of non-linearities in services, benefits and costs; and finally, (v) a coping strategy for dealing with threshold effects in marginal analysis.

Figure 3: Ecosystem Services: A Framework for Appropriate Economic Valuation



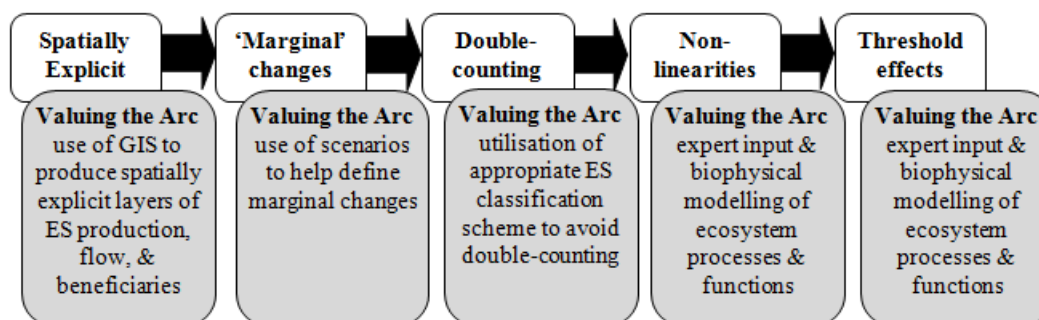
The support system is structured as a linear process rather than cyclical because we are moving from a starting point of 'ecosystem services' to an output of 'values'. By starting with 'spatial explicitness' we help to avoid confusion later by clearly identifying at the outset what ecosystem services 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 ES production and potential human welfare impacts brings us to a point where consideration can be given to possible non-linearities and

threshold effects. These are key features that need to be identified and analysed, 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 towards more rigorous valuation estimates.

5 CASE STUDIES: VALUING ECOSYSTEM SERVICES IN THE EASTERN ARC MOUNTAINS, TANZANIA; AND COASTAL REALIGNMENT SCHEME APPRAISAL, EASTERN ENGLAND.

This framework is being used to value a range of ecosystem services provided by the Eastern Arc Mountains of Tanzania as part of a five year research and policy programme (Valuing the Arc). The Eastern Arc Mountains are recognised as a globally important area for biodiversity, forming part of the Afromontane biodiversity hotspot (Mittermeier et al. 2004), and a globally important ecoregion in its own right (Burgess et al. 2004, Burgess et al. 2006). The mountains provide a range of services and related human benefits at local, regional and global scales - including timber and fuel wood; water for irrigation, domestic use and hydroelectricity; carbon storage and nature-based tourism (Doggart and Burgess, 2005). At the same time this is an area of rapid land cover change, having lost 11% of its primary forests and 41% of its woodlands since 1970 (FBD 2006). This conversion is driven by clearance for farmland as well as increasing demand for timber and fuel wood. These pressures, subsistence and commercial, are rational in the short term, especially in a country where 44% of the population is food insecure (UN 2005) and over 90% of energy comes from burning biomass (Sheya and Mushi 2000). The focus of the research programme is to better understand the links between the services provided by the mountains and human welfare. The work is guided by the Ecosystem Services APPROach (ESAPP) described earlier. A key part of this approach is the assignment of meaningful and appropriate economic values to the various benefits (and costs) associated with ecosystem services provision in the Eastern Arc Mountains. To do this, we are implementing the sequential decision support system as outlined in Figure 4.

Figure 4: Implementing a Sequential Decision Support System in Valuing the Arc



A GIS is a central feature in our study. We aim to produce a series of spatially explicit layers which recognize heterogeneity in ecosystem service production, flow, and beneficiaries across space, and then combine this information to understand the value of these services through benefits and costs layers. In line with the sequence described in the decision

support system, the first step is to use spatial analysis to identify ES production, flow and beneficiaries. We then use participatory scenario building to help define realistic, marginal changes based on transitions in ecosystem. This starts with broad storylines, leading to possible impacts on TZ landscape, and then translating these into policy packages and thinking about how this impacts on service flows. The process necessitates moving from qualitative descriptions to quantitative changes (see Fisher et al, 2009). For example, continued extraction of NTFP and timber and agricultural encroachment is expected to lead to 'declines' in woodland area. Stakeholder input and expert opinion then translate this into a figure, for example, a decline of 3%. Having identified relevant 'marginal' changes, utilising the classification scheme developed by Fisher and Turner (2008), conflicts and complementarities between individual services can then be identified. By adopting this classification, we explicitly recognize that in valuation only the benefits generated by final services can be aggregated, thereby avoiding double-counting. In our case study the key ecosystem services benefits are identified as timber and non-timber forest products, drinking water provision, HEP, irrigation water, carbon sequestration, ecotourism, and existence values. In the final stages of the sequence we use expert input and biophysical modelling of ecosystem processes and functions to recognise where non-linearities and threshold effects are expected to occur. Through this process we expect to deliver meaningful and appropriate value estimates for the key ecosystem services provided by the Eastern Arc. In line with the ESApp approach, this spatially explicit information can then be combined to produce a map of the winners and losers in ecosystem services provision which can be used to design appropriate benefits capture mechanisms.

A partial deployment of the framework has also been undertaken within an analysis of saltmarsh ecosystem creation linked to managed realignment schemes in the coastal areas of south east England (Luisetti et al 2008). A GIS was used to identify candidate sites and scenario analysis was used to configure a maximum and minimum spatial coverage. A range of final benefits were identified and valued, see Table 1. The benefit values were then compared with the scheme costs in a full cost benefit model with sensitivity testing (Luisetti et al 2008). Results indicate that, given the right spatial location, management realignment schemes can provide economic efficiency gains. Comparing these benefit estimates with another site, Humber estuary in north-east England, further illustrates the importance of accounting for spatial variation in value estimates for certain services. For example, in the case of storm buffering cost savings, which typically vary between sites according to wave climate, coastal topography and consequent defence works (Turner et al 2007).

Table 1: Benefit value estimates for ecosystem services provided by coastal realignment schemes in south-east and north-east England.

	Blackwater estuary^a		Humber estuary^b
	(£2007)		(£2005)
Carbon storage	£7, £15 & £30/tC	Carbon storage	£45/tC
Storm buffering (maintenance costs savings)	£433/km/yr	Storm buffering (maintenance costs savings)	£1780/km/yr
Recreation & amenity ^c	£4,428,961, £5,790,713 & £6,430,145 /yr ^d	Composite environmental benefit ^e (recreation & amenity)	£621/ha/yr
Fish productivity	£7.43 & £11.55/kg		

Notes: ^a Luisetti et al. (2008); ^b Turner et al (2007); ^c Onsite survey; ^d benefit estimates based on aggregate WTP values, using spatially sensitive valuation function; ^e benefit estimates based on benefits transfer.

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 non-linearities may be identified; and, (v) full consideration of possible threshold effects via a precautionary approach based on a safe minimum standards 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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