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**Coastal and marine ecosystem services valuation
for policy and management**

By

Tiziana Luisetti, R. Kerry Turner, David Hadley, and
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COASTAL AND MARINE ECOSYSTEM SERVICES VALUATION FOR POLICY AND MANAGEMENT

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Abstract

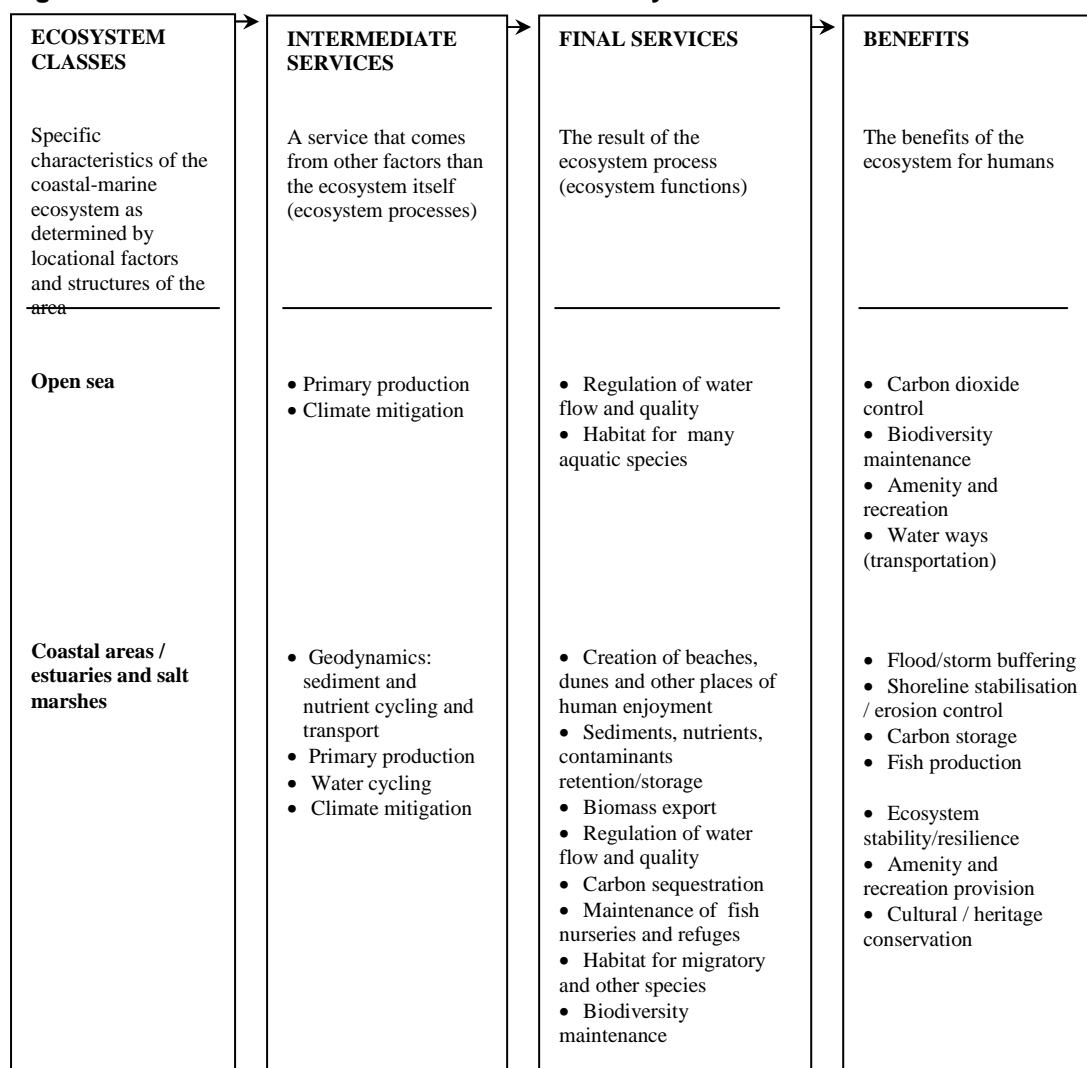
Understanding the economic value of nature and the services it provides to humanity has become increasingly important. In this paper we review the progress to date on both the necessary conceptual framework and empirical valuation studies required to bolster decision support systems targeted at integrated coastal zone management goals. We first review definitions of ecosystem services. We then highlight and discuss the importance of: spatial explicitness; marginal changes; double-counting; non-linearities; and threshold effects. Finally, using UK case studies on managed coastal realignment, we highlight the usefulness of an ecosystem services sequential decision support system to environmental valuation and policy assessment.

Keywords: Ecosystem services; Ecosystem valuation; Managed realignment; Choice experiment; Cost-benefit analysis.

1. INTRODUCTION

Depending on the precise definition used, coastal zones occupy around 20% of the earth's surface but host more than 45% of the global population and 75% of the world's largest urban agglomerations. The functioning of coastal and related marine areas is maintained through a diversity of ecosystems – coral reefs, mangroves, salt marshes and other wetlands, sea grasses and sea weeds beds, beaches and sand dunes, estuaries and lagoons, forests and grasslands. This natural capital stock provides a range of services, such as nutrient and sediment storage, water flow regulation and quality control and storm and erosion buffering (see Figure 1) (Crossland et al, 2005).

Figure 1 Classification of Coastal and Marine Ecosystem Services.



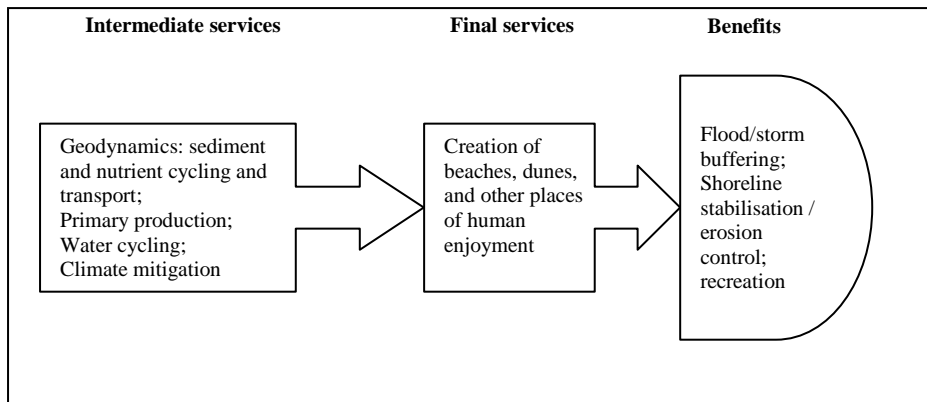
Coastal zone ecosystems are impacted by dynamic environmental change that occurs both ways across the land-ocean boundary. The natural and anthropogenic drivers of change (including climate change) cause impacts ranging from erosion, siltation, eutrophication and over-fishing to expansion of the built environment and inundation due to sea level rise. All coastal zone natural capital assets have suffered significant loss over the last three decades (e.g. 50% of marshes lost or degraded, 35% of mangroves and 30% of reefs) (MEA, 2005).

The consequences for services and economic benefits value of this loss at the margin is considerable, but has yet to be properly recognised and more precisely quantified and evaluated (Daily, 1997; Turner et al., 2003; Maler et al. 2008; Barbier et al. 2008). In this paper we review the progress to date on both the necessary conceptual framework and empirical valuation studies required to bolster decision support systems ambitiously targeted at integrated coastal zone management goals.

2. A CONCEPTUAL FRAMEWORK FOR ECOSYSTEM SERVICES

Many definitions and classification schemes for ecosystem services exist (Daily, 1997; Costanza *et al.*, 1997; Boyd and Banzhaf, 2007). One of the most widely cited is the Millennium Ecosystem Assessment definition, which describes ecosystem services as ‘the benefits that people obtain from ecosystems’. It classifies ecosystem services into: *supporting* services (e.g. nutrient cycling, soil formation, primary production), *regulating* services (e.g. climate regulation, flood regulation, water purification), *provisioning* services (e.g. food, fresh water), and *cultural* services (e.g. aesthetic, spiritual, recreational and other non-material benefits). 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. 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. For economic valuation purposes the definition proposed by Fisher *et al.* (2009) clarifies the distinction between ecosystem services and benefits: *ecosystem services are the aspects of ecosystems utilised (actively or passively) to produce human well-being*. Fisher *et al.* see ecosystem services as being the link between ecosystems and things that humans benefit from, not the benefits themselves. Ecosystem services include ecosystem organisation or structure (the ecosystem classes) as well as ecosystem processes and functions (the way in which the ecosystem operates). The processes and functions become services only if there are humans that (directly or indirectly) benefit from them. In other words, ecosystem services are the ecological phenomena, and the benefit is the realisation of the direct impact on human welfare. The key feature of this definition is the separation of ecosystem processes and functions into intermediate and final services, with the latter yielding welfare benefits (Figure 2).

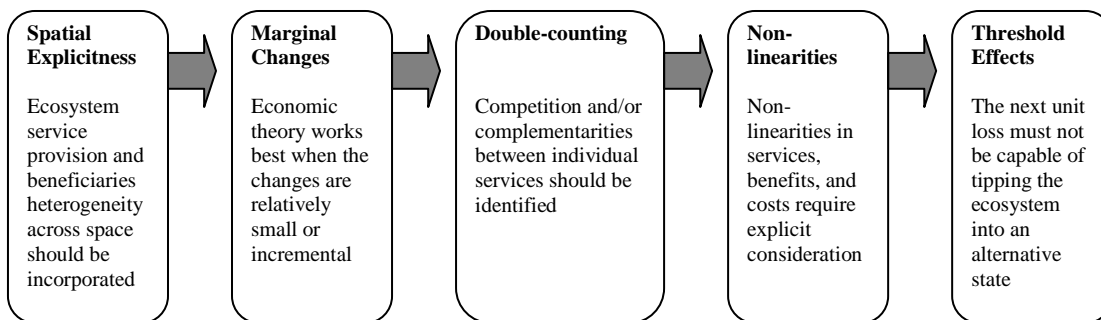
Figure 2 Example of relationships among representative intermediate services, final services and benefits. Adapted from Fisher and Turner (2008).



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. This approach seeks to provide a transparent method for identifying the aspects of ecosystem services which are of direct relevance to economic valuation, and critically, to avoid the problem of double-counting.

In the economic literature, a number of issues can be identified as critical to the *appropriate* economic valuation of ecosystem services. These are: spatial explicitness, marginality, the double-counting trap, non-linearities in benefits, and threshold effects (see Figure 3).

Figure 3 Ecosystem Services Sequential Steps: A Framework for Appropriate Economic Valuation.



Source: Morse-Jones et al. (2008).

Spatial explicitness

It is critically important to first and foremost clarify the level of understanding (or ignorance) of 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 side beneficiaries. The requirement for spatially explicit ecosystem valuation is based on recognition that ecosystem services are context dependent in terms of

their provision and their associated benefits and costs. The importance of this point can be illustrated in the example of coastal wetland services provision (Andrews et al., 2006; Shephard et al., 2007; and Turner et al., 2007). One of the services provided by wetlands is carbon storage but the net effect of this service is conditioned by the simultaneous release of methane. It turns out that the spatial location of the wetland and in particular the salinity condition plays a significant role in the carbon storage to methane emission ratio and the consequent global warming effect.

An essential component of the valuation approach that has rapidly emerged is the use of GIS techniques. Explicitly incorporating the spatial context is critical in obtaining unbiased estimates of both the costs and benefits of ecosystem provision, and, crucially, in enabling planners to identify the most economically efficient trade-offs. 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.

Marginality

Economics requires that for the valuation of ecosystem services to be meaningful such analysis should be conducted “at the margin”. This means focusing on relatively small, incremental changes rather than large state changing impacts. Given the scientific uncertainties which shroud ecosystem functioning, it is often difficult to discern whether a given change is ‘marginal’ or not and when thresholds are being approached or crossed.

Knowledge of the drivers and pressures on the ecosystems under study, as well as an understanding of how the system is changing or might change from its current state is crucial. This has been called the system’s transition path (Turner et al., 2003; Fisher et al., 2009). It is important to know if the transition path is “stepped” as in the loss of a full coral reef system or shallow lake, or it is “relatively smooth” such as in species invasion into an area. By identifying the transition path, we can force the analysis to consider losses or gains in service provision or economic value between two distinct states of the systems.

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. 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 wellbeing for a *10 hectare* change in the stock of 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, or the actual service flow. In some cases these analyses have been placed in a context of ‘change’ by drawing comparisons with alternative land use options.

Double counting

Another widely recognised issue concerns the potential problem of double-counting. 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. The value of a marine ecosystem for industrial fishing, for example, should not be added to the value of the same marine area for recreational fishing, since the former will likely preclude the later. Farber et al (2006) similarly note the problem of including aesthetic services and nutrient regulation in a case study of Plum Island coastal ecosystem. In essence, double-counting is a feature of the complexity of ecosystem services and the difficulty in understanding their multiple interactions.

Unfortunately, there are numerous cases where researchers have incorrectly summed values in order to obtain aggregate estimates of ecosystem value (evidence from Fisher et al., 2009). It is thus essential that the analyst has a clear understanding of the various overlaps and feedbacks between services when undertaking aggregation. 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). Alternatively, the classification scheme recommended by Fisher and Turner (2008) as shown in Figure 2 helps to avoid the problem by drawing a clear distinction between intermediate services, final services, and benefits, the latter being the focus of economic valuation (see also Maler et al., 2008).

Non-linearities

The existence of non-linearities in ecosystem services provision adds further complexity to their valuation and subsequent management. Because many ecosystems typically respond non-linearly to disturbances, their supply may seem to be relatively unaffected by increasing perturbation, until they suddenly reach a point 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. Further, in situations where non-linearities occur, one cannot 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). If a cost-benefit appraisal assumes linearity, but service provision is in fact non-linear, policy option outcomes may be unnecessarily polarised. Correspondingly, for ecosystem valuation to better inform policy decisions, non-linearities need to be clearly understood and reflected in both ecological and economic analysis.

Barbier et al. (2008) have stressed that for some ecosystems (such as: coastal mangroves, salt marshes and other marine ecosystems) the services provided change in a non-linear way as habitat variables such as size of area alter. 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. 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.

Threshold effects

A threshold effect refers to the point at which an ecosystem may change abruptly into an alternative steady state. 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)¹. In practise, this requires knowledge of the ecosystem's resilience to stress and shock and its possible tipping threshold. 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. For this reason, threshold effects pose especially complex policy and analysis challenges. Identifying this hazardous zone, in fact, will require expert input from ecologists, risk analysts and others, and may ultimately require ethical/political choices to be made and deliberatively agreed.

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. 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 change, appropriate for marginal analysis, it is possible that the cumulative effect of doubling fish catch per hour could result in flipping the recreational fishery into an alternative state. 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 (see Figure 3), to be most useful for policy, 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.

3. CASE STUDIES: MANAGED COASTAL REALIGNMENT IN THE UK

In this section we present two recent UK case studies examining the re-orientation of the UK coastal zone policy towards a more flexible and adaptive approach. A key component of this new thinking is managed realignment (MR), which involves the deliberate breaching of existing sea defences with the land behind them consequentially being flooded. These projects result in the creation/restoration of salt marshes, which are a soft and more sustainable flood defence which help to dissipate wave energy. Salt marshes have been disappearing from coasts because of what is known as the 'coastal squeeze' phenomenon. Due to sea level rise, the intertidal habitat is gradually constrained on one side by the sea and on the other by sea walls erected between the land and an intertidal habitat to protect the coast. Managed realignment allows the intertidal habitat to naturally move inland so that it can continue to protect the coast in combination with manmade or natural 'secondary defences' (using the land elevation of the area).

Managed realignment schemes yield benefits in terms of ecosystem services. They generate carbon storage benefits (via salt marsh creation) which can be valued in terms of the damage cost avoided per tonne of CO₂. The sites also serve to improve fisheries'

¹ The safe minimum standard represents the minimum level of a well-functioning ecosystem which is capable of producing a sustainable supply of service.

productivity via nursery areas and this gain can be valued via market prices for commercial species. They also generate recreation and amenity benefits related to walking, bird watching and other recreational activities, as well as biodiversity maintenance and existence value benefits. 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 (see Humber estuary study below). Or, more properly, site-specific contingent valuation/choice experiment studies can be conducted to estimate willingness-to-pay values (see Blackwater estuary study below). Finally, the maintenance costs of the existing engineered defences will be saved as realignment schemes are implemented. On the costs side, secondary defences may be required further inland and there are opportunity costs associated with any agriculture land that is sacrificed as the old defences are breached.

Cost-benefit analysis of managed realignment schemes can take the following approach. The 'status quo' existing protection system can be appraised on the basis of equation (1):

$$(1) \quad PV_t^{sq} = -\sum_{t=0}^T \frac{1}{(1+r)^t} [(l^{sq} C_{m,t}^{sq})],$$

where: PV_t^{sq} = present value of total costs of current defences (£ million)

r = discount rate

l^{sq} = length of defences

$C_{m,t}^{sq}$ = maintenance costs (£/km/yr); and

The costs and benefits of managed realignment is given in equation (2):

$$(2) \quad PV_t^{mr} = \sum_{t=0}^T \frac{1}{(1+r)^t} [l^{mr} (C_{k,t}^{mr} + C_{m,t}^{mr}) - (a_t^{mr} L_{agr,t}^{agr}) - (a_h^{mr} B_{e,t})]$$

where: PV_t^{mr} = present value of managed realignment schemes (£ million)

r = discount rate

l^{mr} = length of managed realignment (km)

$C_{k,t}^{mr}$ = capital cost of realignment

$C_{m,t}^{mr}$ = maintenance costs

a_t^{mr} = agricultural land lost

$L_{agr,t}^{agr}$ = forgone agricultural land value

a_h^{mr} = area of intertidal habitat created

$B_{e,t}$ = ecosystem value benefits (£/ha)

Finally, the overall CBA result is found via equation (3):

$$(3) \quad NPV_t^{mr} = (PV_t^{sq} - PV_t^{mr})$$

where: NPV_t^{mr} = net present value of managed realignment compared to hold-the-line for a given stretch of coastline at time t (£ million).

Previous research has highlighted that managed realignment policy needs to be appraised across a more extensive spatial and temporal scale than has been the case in the traditional scheme-by-scheme coastal management system. Whole estuaries or multiple coastal cells need to be treated as a single 'project' encompassing a number of realignment sites. Any comprehensive deployment of this coastal strategy, however, will be conditioned by a complex set of factors in a highly 'contested' political economy context (Turner, 2007). Argumentation will be focused on the trade offs between greater resource efficiency, social justice, equity and compensation objectives. A sequential approach to the appraisal process should first identify all sites in which the opportunity costs of realignment minimise significant social justice/ethical concerns. In these cases an efficiency-based CBA could provide the decisive information in policy choice (Randall, 2002; Turner, 2007). In the cases where people, property, culture/historical assets and designated freshwater conservation sites are part of the opportunity cost calculation, CBA will not be as decisive and must be subsumed within a multi-criteria decision support system and process.

3.1 The Humber study

In the UK, the coastal managed realignment study on the Humber estuary (North-east England) Turner et al. (2007) utilised GIS techniques to identify possible realignment sites in which the opportunity costs of realignment minimise significant social justice/ethical concerns. Urban centres were assumed to be protected and therefore excluded from the analysis. The GIS identified the suitable areas of realignment for four realignment scenarios: business-as-usual (BAU); policy targets (PT); deep green (DG); 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. The layers of the GIS reflected five geographical and socio-economic criteria:

1. Firstly the area below the high spring tide level was located. This area represents the maximum area of intertidal habitat that could be created before considering other factors.
2. The land use in the area previously identified was investigated. Undeveloped areas were considered the most suitable for realignment. Amongst the agricultural land areas, only those of grade 3, 4 and 5 were judged suitable for conversion. Sites of Special Scientific Interest (SSSI), Special Areas of Conservation (SAC) and other similar protected areas together with historically significant buildings, as well as archaeological and military sites were excluded from the realignment areas.
3. The transport network (roads, railway lines and canals) was assumed to be protected and therefore precluded from realignment.
4. Areas historically reclaimed from the estuary were considered suitable for realignment. However, the historical land use of an area is less important than its

current land use. Thus, this criterion was not considered to be compulsory, merely beneficial.

5. This last criterion is applied once the preceding four criteria have provided the basis for identifying areas physically suitable for realignment. It takes account of the size, shape, land elevation and proximity to existing intertidal habitats, and it aids the identification of the position to which defences may be realigned, and which areas may offer the greatest benefits in relation to costs.

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 UKCIP02 information. As reported by Haweks et al. (2003), the recommended precautionary allowances for future mean sea-level rise are appropriate and consistent, and remain the standard approach for the moment (6mm/year for the Anglian region). However, due to uncertainty over the loss of intertidal habitat because of coastal squeeze over the next 50 years, it is 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 CBA results show positive net present values (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.

3.2 The Blackwater study

A more recent study in the Blackwater estuary (Luisetti et al., 2008a) has tried to overcome some limitations in the Humber study. As a first step, the same GIS investigation and the same criteria as for the Humber were applied to this new study to locate suitable areas of realignment in the Blackwater estuary (located in Essex in the UK). Furthermore, the same scenarios were applied with the exception of the BAU scenario². Building upon that study, the ecosystem services approach of Fisher and Turner (2008) was used to identify the benefits of creating new salt-marshes in the Blackwater estuary. The resulting benefits are: flood/storm buffering and related cost savings on hard defences; carbon storage; fish production; amenity and recreation provision (considered as a composite environmental benefit including biodiversity).

Compared with the Humber study, there is one more benefit considered here: fish production³. The fish considered were only those species sold in the market and were therefore valued using market analysis. An *in situ* analysis to quantify fish nurseries and carbon storage were carried out by a fisheries and a biogeochemical expert. These data were then used to value the two benefits of fish production and carbon storage. Carbon storage was quantified, as in the Humber study, with the damage cost avoided method (Tol, 2005; Pearce, 2003; Stern, 2007). Market analysis was used to estimate the cost savings on hard defences. But in the Blackwater study, the composite environmental benefit was estimated *in situ* with a stated preference technique: a choice experiment.

Choice experiment: design and survey

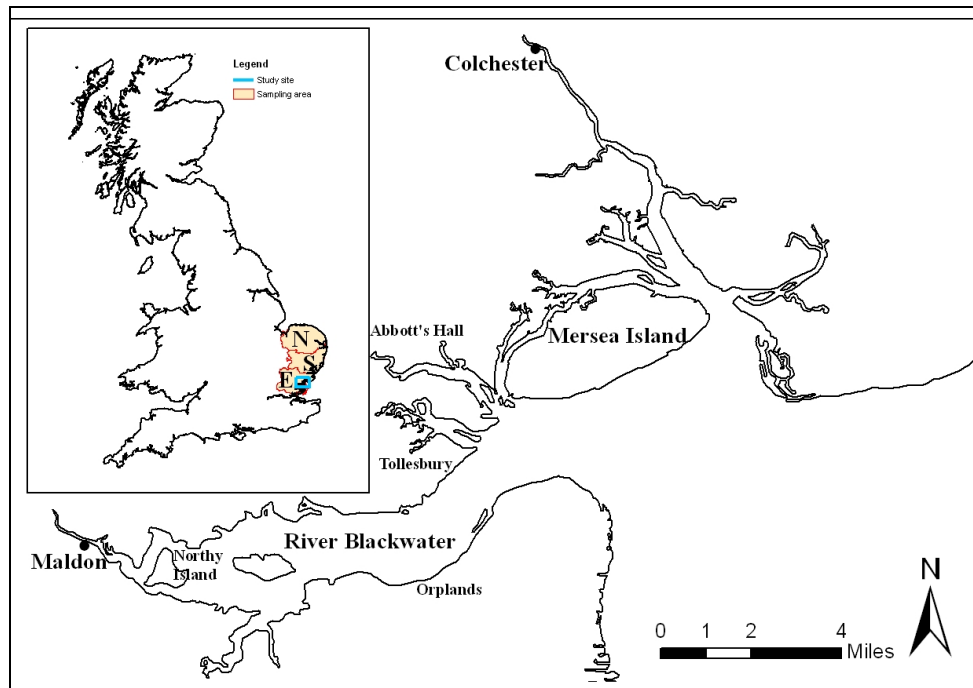
The choice experiment survey was designed, and conducted in Essex as well as the neighbouring counties of Norfolk and Suffolk. In the Blackwater estuary, management

² In this study the existing realignment areas in the estuary were considered as existing salt marshes. With the GIS map, the authors investigated the possibility of new salt marsh areas.

³ The only nursery grounds producing enough marketed fish were those of sea bass, which was therefore the only species valued.

strategies have been proposed to provide sustainable flood defence, and experiments are underway on the managed realignment of the coast (at Tollesbury, Orplands, Northy Island and Abbott's Hall) to mitigate flood risk. Figure 4 shows the study and sampling area, and the managed realignment sites already implemented in the estuary.

Figure 4 Study area map.



N = Norfolk; S = Suffolk; E = Essex

Following a series of preliminary meetings and focus group investigations with policy makers, stakeholders and members of the public, a set of attributes was identified to define the salient features of the good and policy under investigation. These attributes and their units of measurement were as follows:

- The area of new salt-marshes to be created (variable label '*AREA*'): measured both as acres (the most well understood standard unit in the UK) and as the corresponding number of football pitches (as a further approach to enhance comprehension);
- Bird species observable ('*BIRDS*'): measured as the number of protected species - a key policy focus;
- Distance from respondent's home to the nearest site ('*DISTANCE*'): measured in miles;
- Whether the created salt-marsh would be open-access or not ('*ACCESS*'): a simple binary variable;
- Increase in the respondent's annual local (council) tax to pay for the option ('*TAX*'): measured in £ per household per annum.

Attributes and their levels are presented in Table 1 which also details the division of the '*DISTANCE*' attribute into those seen by the '*Far*' sample (living in Norfolk and Suffolk) and those seen by the '*Near*' sample (living in Essex).

Table 1 Attribute levels used in the choice experiment design.

Attribute	Variable label	Levels
Area of new salt-marshes	<i>AREA</i>	25acres = 10fp*
		74acres = 30fp
		123acres = 50fp
		173acres = 70fp
Number of protected bird species observable	<i>BIRDS</i>	2, 3, 4, 5 species
Distance from respondent's home (in miles)	<i>DISTANCE</i>	<i>Near</i> sample: 2, 12, 22, 32 miles <i>Far</i> sample: 42, 52, 62, 72 miles
Access to the salt-marshes	<i>ACCESS</i>	Yes; No
Increase in respondent's council tax per year	<i>TAX</i>	£2, £6, £10, £14

*fp stands for football pitches

Choice sets were determined based on a fractional factorial design. Each respondent answered eight choice questions, each consisting of two options: a status quo option, representing the current situation; and an alternative, in which new salt marshes were created.

Interviews were conducted by a team of trained interviewers at various locations within both the '*Near*' (within Essex) and '*Far*' (within Norfolk and Suffolk) distance zones⁴. After cleaning for yea-saying and protest responses a total sample of some 508 completed questionnaires was obtained of which 162 originated within the '*Far*' zone and 346 from the '*Near*' area. A *t*-test and a Kruskal-Wallis test on socio-economic variables showed that the sub-samples are highly similar in their characteristics. The data also fit quite well with the statistics of the national population.

Choice experiment: results

To value the composite environmental benefit for the CBA, the willingness to pay (WTP) for salt marsh creation in the Blackwater was calculated on the econometric model estimated for the '*Near*' sample⁵. To take account of the fact that each individual made eight choices, a random effects binomial logit model was estimated to investigate the significance and the adequacy of the model fit as well as the presence of heterogeneity in the model. The specification for the model estimated is the following:

$$V_{it} = \beta_1 Const_{it} + \beta_2 LnArea_{it} + \beta_3 Bird3_{it} + \beta_4 Bird4_{it} + \beta_5 Bird5_{it} - \beta_6 Dist_{it} + \beta_7 Acc_{it} - \beta_8 Tax_{it}$$

where V_{it} is the vector of the response variable (choosing Option A or not) for individual i in group t , the β s are the estimated coefficients of the variables presented in Table 1. In a random effects logit model there are two unobserved components of utility: one is the unobserved influences on respondents' choice (a pure random term); and the other represents the individual-specific random element, which takes into account the fact that the same respondent answered to more than one question.

⁴ Non-probability sampling techniques were adopted, a convenient and frequently used approach for hypothesis testing purposes. The survey was conducted over six weeks during summer 2006.

⁵ The results of the econometric model for the '*Far*', and a *Pooled* sample, are presented elsewhere (Luisetti et al., 2008b).

The results of the model estimation are shown in Table 2.

Table 2: Random effects logit model for the 'Near' sample.

<i>Variables</i>	<i>Coefficient</i>	<i>Standard error</i>	<i>P-value</i>
Constant	0.2754	0.2581	0.2860
LnArea	0.2519	0.0609	0.0000
Bird3	0.4172	0.1137	0.0002
Bird4	0.6936	0.1224	0.0000
Bird5	0.8078	0.1070	0.0000
Distance	-0.0118	0.0035	0.0010
Access	0.9745	0.0615	0.0000
Tax	-0.2261	0.0080	0.0000
Rho	0.2368	0.0212	0.0000
LL		-1399.659	
P-value		0.00010	
Correct predictions		73.48%	
Individuals		346	
Periods		8	

The interpretation of the output is the following:

The *CONSTANT* is positive but not significant meaning that, everything else held constant, the respondents receive more utility from the new salt marsh projects going ahead than not. That also means that other elements that the respondents might have considered in making their choice had a little weight in comparison to the attributes presented in the choice experiment.

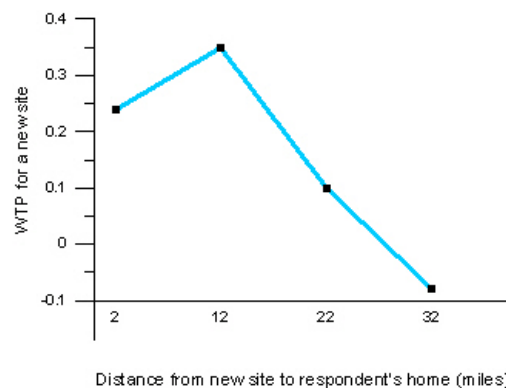
The attribute *LnAREA* is highly significant and of the expected sign. It is reasonable to think that locals prefer a bigger area to a smaller one. However, we assume a log-linear relationship for the attribute *AREA* rather than a linear relationship because the natural logarithm of the *AREA* seems intuitively the correct functional form for the attribute *AREA*, and therefore is the selected form for the analysis.

The *BIRDS* attribute, representing the quality of the environment, was initially modelled using a single linear variable. However, such a form yields a significantly lower degree of explanation than the more flexible, non-linear specification permitted by the use of the dummy variables *BIRD3*, *BIRD4* and *BIRD5*. All three dummy variables were significant at the 99% level and of the expected positive sign indicating that providing habitat for a higher number of endangered bird species results in higher utility levels. Examining parameter values suggests that the satiation point is attained at the *BIRD4* level suggesting a positive but declining marginal *WTP* for bird habitat.

The attribute *ACCESS* turns out to be positive, as expected, and significant at 99%. That means that respondents are interested in seeing and enjoying the natural environment of salt marshes.

The *DISTANCE* variable has the expected negative parameter value and is significant at the 99% level. This result shows that, as expected, the utility of new wetland sites diminishes the further away they are from the individual (an effect known as 'distance decay'). A deeper analysis of the distance variable highlights the non-linearity of this attribute. The use of dummy variables for each level of the attribute distance for the 'Near' sample shows that respondents have a lower level of utility for the first attribute level (2 miles) than for the second (12 miles). The former was probably perceived as a 'too' close home location to the coast, where the risk of flooding is higher. The following attribute levels have, as predicted by the distance decay, a lower utility than the first and the second attribute level. The distance decay and the non-linearity effect are shown in Figure 5.

Figure 5 Distance decay and non-linearity effect for the 'Near' sample.



The cost variable, *TAX*, is highly significant at the 99% level and of the expected negative sign. It represents the cost the respondents were asked to bear for the creation of new salt marsh areas in the Blackwater estuary. The results show that the probability of choosing Option A diminishes as its cost increases.

Willingness to pay estimates

Willingness to pay (WTP) estimates are more appropriate for use in the cost-benefit analysis than the marginal WTP (MWTP) for single attributes, because the WTP is basically the value of a hypothetical policy and it is possible to derive numerous WTP estimates from the choice experiment designed depending on the levels of the attributes selected. In order to calculate the WTP, we need to estimate MWTPs, we then multiply attribute's MWTPs and selected attribute levels, and sum them together to get the value of the WTP (Kanninen, 2000).

An aggregated WTP for two different policies that create new salt marshes in the Blackwater estuary, which represent the composite environmental benefit, was then calculated and subsequently inserted in the CBA. Two equations were elaborated to express the two hypothetical decision policies. In the first equation (4), use and non-use values of the salt marshes are considered, and the aggregated WTP is defined for a policy involving a defined extent of new salt marsh area created (corresponding to the realignment scenarios) and for a high (*BIRD5* variable) level of environmental quality⁶, involving different distances, and with the possibility of access.

⁶ The interaction between the attribute *BIRDS* (representing non-use values) and the attribute *ACCESS* (representing use values) was tested. The interaction resulted not significant meaning that that the respondents made their choices distinguishing between the attribute *BIRDS* and attribute *ACCESS*, without assigning any use value to the attribute *BIRDS*.

$$(4) \quad WTP = \frac{\beta_2}{\beta_8} LnAREA + \frac{\beta_5}{\beta_8} BIRD5 - \frac{\beta_6}{\beta_8} DIST + \frac{\beta_7}{\beta_8} ACC$$

where β_8 is the coefficient for the cost attribute, β_2 , β_5 , β_6 , and β_7 are the coefficients for the *LnAREA*, *BIRD5*, *DISTANCE*, and *ACCESS* attribute respectively.

The other equation (5) is defined as the first, but following a conservative approach the variable *BIRD5* is dropped from the equation:

$$(5) \quad WTP = \frac{\beta_2}{\beta_8} LnAREA - \frac{\beta_6}{\beta_8} DIST + \frac{\beta_7}{\beta_8} ACC$$

It was decided not to include a constant in these equations because its inclusion requires the assumption that unobserved aspects of improved wetland quality (those aspects not measured by the attributes) are the same across wetlands if surplus estimates are to be used for benefit transfer (Morrison *et al.*, 2002). The variable *DISTANCE* provides a key determinant of the aggregated WTP. Considering that distance decay values were revealed in the econometric analysis, in this study, an approach similar to the one used by Bateman *et al.* (2006) for the aggregation of the WTP is applied. Although some positive WTP was also found in Suffolk and Norfolk, the districts wholly within of the Essex County were used as a lower bound on aggregated WTP. The GIS was used to calculate the distances from each town representing an Essex district to Abbott's Hall (Essex), a well known managed realignment site located in the estuary. Districts and relative distances were categorised in four groups: 8, 15, 23 and 32 miles. WTP at each distance band is calculated by multiplying the population of that area by the mean household WTP for that area for the improvement under consideration. The total WTP for Essex is obtained by simply summing across areas. Inserting in the policy equations the extent of the new salt marsh area created for each scenario, the aggregated WTP for the PT, DG and EDG scenarios was calculated as reported in Table 3:

Table 3 Aggregated WTP for the Blackwater salt marshes under the PT, DG and EDG scenarios (£ thousand).

Distance to Abbott's Hall (miles)	Households population	PT: 81.6 Hectares		DG: 816.5 Hectares		EDG: 2404.1 Hectares	
		WTP use and non-use values Benefits (£/yr)	WTP use values only Benefits (£/yr)	WTP use and non-use values Benefits (£/yr)	WTP use values only Benefits (£/yr)	WTP use and non-use values Benefits (£/yr)	WTP use values only Benefits (£/yr)
8	63706	771	547	930	706	1,005	781
15	349836	4,107	2,875	4,981	3,749	5,392	4,160
23	97974	1,109	764	1,354	1,009	1,469	1,124
32	33185	360	243	443	326	482	365
		6,347	4,429	7,708	5,790	8,348	6,430

In the PT scenario the area of salt marsh created is 81.6 hectares. This generates an aggregate WTP of over £6 million of which more than £4 million is use value. The DG generates 10 times the area of saltmarsh (816.5 hectares) in respect to the PT scenario, however total and use values only increase by just over 20% and 30% respectively to £7.7 and £5.8 million in turn. Similarly, while the EDG scenario further increases saltmarsh area (2404.1 hectares) to nearly 30 times the PT area, total and use-only values are only just over 30% and 45% larger (at £8.3 and £6.4 million) than under the latter initial improvement. Such results conform strongly to prior economic expectations regarding diminishing marginal utility generated by provision of additional areas of environmental quality. Essentially the initial units of improvement are considered to be of significantly higher value than are the subsequent additions.

The results highlight the error inherent in calculating constant unit values of provision of environmental enhancements⁷. What cannot be observed from what is effectively a single site model is the further variation in values which would arise through locating these enhancements in alternative areas. We would expect such values to rise as we locate enhancements nearer to larger populations and fall as we locate within areas of greater substitute availability.

CBA: results

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 4, these results are achieved even when a set of quite conservative assumptions are adopted for possible realignment policies in the Blackwater. Unlike in the Humber study, in this study the NPVs are positive in any scenario and for any period of time considered, mainly because of the composite environmental benefit (CEB) dominance over carbon sequestration⁸ and fisheries benefit. It is worth noticing that the WTP estimates for the Blackwater are a lot lower than those found in some of the published literature for wetland areas (see Brouwer et al., 1999).

Table 4 Comparing NPVs for the Humber and the Blackwater estuary studies using conservative values (only use values) and a declining discount rate (HMT); (£ million).

Scenario	25yrs		50 yrs		100 yrs	
	Humber	Blackwater*	Humber	Blackwater*	Humber	Blackwater*
<i>Policy Targets (PT)</i>						
NPV PT	-73.23	68.41	-82.22	152.25	-92.27	307.19
NPV HTL	-70.40	-1.88	-86.01	-3.96	-100.93	-7.81
NPV(PT) - NPV(HTL)	-2.83	70.29	3.79	156.21	8.66	315
<i>Deep Green (DG)</i>						
NPV DG	-97.32	74.83	-101.42	185.35	-107.92	389.58
NPV HTL	-70.40	-1.88	-86.01	-3.96	-100.93	-7.81
NPV(DG) - NPV(HTL)	-26.92	76.71	-15.41	189.31	-6.99	397.39
<i>Extended Deep Green (EDG)</i>						
NPV EDG	-94.30	62.83	-74.48	186.22	-63.83	414.24
NPV HTL	-70.40	-1.88	-86.01	-3.96	-100.93	-7.81
NPV(EDG) - NPV(HTL)	-23.90	64.71	11.53	190.18	37.10	422.05

⁷ Just for the sake of comparison, these WTP values are much lower than the mean WTPs per person per year reported in the meta-analysis on wetland valuations of Brouwer *et al.* (1999). The values obtained by Woodward and Wui (2001) in their meta-analysis of wetlands, and used by Turner *et al.* (2007), are not comparable with the one obtained in this study because the calculated values are per hectare per year.

⁸ Little sensitivity is found also when the social cost of carbon used by the Stern Review (2007) - £230 – is applied in the analysis.

Closer inspection of Table 4 reveals a possible anomaly for the Blackwater estuary NPVs. We would expect to see the benefits growing from the PT to the EDG scenario, but it is actually the DG scenario that has the highest positive value over the 25 years time horizon. A possible explanation relates to the length of defences to be realigned. The EDG scenario has the longest length of defences to be realigned, incurring higher costs of realignment. However, the areas of realignment were chosen where the elevation of the land would not then require a secondary line of defence. When it is assumed that none of the areas to be realigned in any scenarios requires a secondary line of defences, the anomaly disappears. The results for the two case studies highlight that the values are sensitive to the scale of the realignment scheme and local topography. It means that we cannot deliver a general rule saying that the DG is better than the EDG scenario, because the different values depend on the extent of the MR scheme and the specific topography of the region we are looking at. In the Blackwater estuary study we believe that an EDG scenario without a second defence line is a realistic situation. Nevertheless, other areas may not have such a convenient topography.

4. Conclusions

The case studies highlight the usefulness of the conceptual framework and the sequential approach to valuation and policy assessment outlined earlier in this paper. Spatial explicitness and the use of GIS techniques were fundamentally important to scheme appraisal. Marginality and double counting avoidance rules were observed, which was especially important for the choice experiment and for the overall benefits aggregation. Non-linearities were evident in attributes used in the choice experiment and in the discovery of a distance decay effect when respondents were asked to value the creation of new marshes.

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