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Coastal and marine ecosystem services valuation for policy and management: Managed realignment case studies in England

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ABSTRACT

The recognition of the economic value of nature and the services it provides to humanity has become increasingly essential in the context of coastal zone management. 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 (DSS) targeted at integrated coastal zone management goals. We highlight the importance of using an ecosystem services approach and recommend some basic sequential, although overlapping, analytical steps for appropriate environmental valuation and policy assessment: spatial explicitness; marginal changes; double counting; non-linearities; and threshold effects. We illustrate the practical use of the DSS by reviewing an existing UK case study on managed coastal realignment along the Eastern coast of England, and a new UK case study on managed realignment implemented in a southern area on the same coast. Comparing the two studies, the importance of spatial explicitness and the need for a sequential decision support system when dealing with ecosystem services valuation and project policy appraisal become particularly evident.

1. Introduction

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) (Millennium Ecosystem Assessment, 2005). As the process of environmental change across local to regional and up to the global scale has intensified and increased in pace, so the risk posed to the integrity and resilience of ecosystems, not least in coastal areas, has increased in parallel. This has in turn led to calls for an ecosystem services approach to support the case for more protected areas or better management and sustainable use of ecosystems under threat of conversion or degradation (Balmford et al., 2002; Bockstael et al., 2000; Costanza et al., 1997; Daily, 1997; Daily and Ellison, 2002; Turner et al., 2003; Turner and Daily, 2008). The consequences of the loss of services and economic benefits at the margin are considerable, but they have yet to be properly recognised, quantified and evaluated (Daily, 1997; Turner et al., 2003; Mäler 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.

Coastal zone policy in the UK and Europe is being re-orientated towards a more flexible and adaptive approach, while linked water catchment management is also being reformulated under the EU Water Framework Directive (European Commission, 2000). A key component of this new thinking could be managed realignment (MR), which involves the deliberate breaching of existing sea defences with the land behind them consequentially being flooded.

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MR projects result in the creation/restoration of saltmarshes that are a soft and more sustainable flood defence helping to dissipate wave energy. Salt marshes have currently been disappearing from coasts because of what is known as the 'coastal squeeze' phenomenon: due to sea level rise the inter-tidal habitat is gradually constrained on one side by the sea and on the other by sea walls erected between the land and an inter-tidal habitat to protect the coast. MR allows the inter-tidal habitat to naturally move inland so that it can continue to protect the coast in combination with manmade or natural 'secondary defences' (using local topography), creating opportunities for biodiversity enhancement,[4] and expanding opportunities for amenity and recreation (i.e. a diversity of ecosystem services). 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 (Turner et al., 2007). In other words, 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 in which continuing uncertainty over ecosystem functioning under stress and shock condition still prevails (Turner, 2007). Argumentation will be focused on the trade-offs between greater resource efficiency, social justice, equity and compensation objectives. In general, unless MR schemes are very extensively adopted, multiplier effects on the regional or national economy are not likely to be significant. The proposed sequential approach to the appraisal process would first identify all sites that are likely to generate the minimum of social justice/ethical concerns. In these cases an efficiency-based cost-benefit analysis (CBA) could provide the decisive information in policy choice (Randall, 2002; Turner, 2007). In the cases where people, property, culture/historical assets and designated fresh water conservation sites are part of the opportunity cost calculation, CBA will not be as decisive and may have to be subsumed within a multi-criteria decision support system and deliberative process in order to tackle the ethical and other value judgments that will be in dispute (Climaco, 1997; Janssen et al., 2001; Vincke, 1992; Yu, 1985; Zeleny, 1982).

In this paper we show the actual application of a coastal decision support system (DSS) describing how the steps required for an appropriate valuation of coastal zone management policies were applied to two estuaries case studies in the UK. We focus on the East of England where major flooding can take place on the coastline because of the concomitant occurrence of sea level rise and isostatic movements. The reviewed case studies analysed the economic efficiency of implementing MR schemes in the Humber and Blackwater estuaries, which share similar risk concerns about the future of the coasts and the same legislative regime, and subsequent proposed mitigation actions. Furthermore, from a methodological point of view we test the advantages of 'improved' ecosystem services valuation methods. In the Humber case study, for example, a benefit transfer technique was used to measure the amenity and recreational benefits of recreated saltmarshes. In the Blackwater we measure the same benefit with a new stated preference study (a choice experiment) which was then incorporated in the CBA.

The paper is set out as follows: in Section 2 we present an ecosystem services approach for appropriate economic valuation; Section 3 presents the case studies; and Section 4 offers some conclusions.

[4] Depending on how successfully the saltmarsh communities can re-establish.

2. A conceptual framework for ecosystem services

As reported in Fisher et al. (2009) several definitions of ecosystem services (ES) are now found in the literature (Boyd and Banzhaf, 2007; Costanza et al., 1997; Daily and Ellison, 2002). The Millennium Ecosystem Assessment (MEA) definition that describes ecosystem services as 'the benefits that people obtain from ecosystems' is probably one of the most widely cited. Classifying 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), the MEA definition provides an excellent starting point for moving towards a more operational classification system. That definition, for example, has been adapted and re-oriented by Wallace (2007) focusing on land management, while Boyd and Banzhaf (2007) and Mäler et al. (2008) took national income accounting as their policy context. Importantly, in their paper Boyd and Banzhaf (2007) highlight that ES and benefits are not identical. Fisher and Turner (2008) have clarified this distinction between ES and benefits for economic valuation purposes: ecosystem services are the aspects of ecosystems utilised (actively or passively) to produce human well-being. Fisher et al. (2009) see ecosystem services as the link between ecosystems and things that humans benefit from, not the benefits themselves. ES include ecosystem classes as well as ecosystem processes and functions, with the latter becoming services only if there are humans that (directly or indirectly) benefit from them. In other words, ES are the ecological phenomena, and the benefit is the realisation of the impact on human welfare. More importantly for environmental valuation the Fisher et al. (2009) definition explicitly links changes in ecosystem services to changes in human welfare. The key feature of that definition is therefore the separation of ecosystem processes and functioning into intermediate services (that influence human well-being indirectly) and outcomes in terms of final services (that contribute directly to human well-being), with the latter yielding welfare benefits (Fig. 1).

Classification of ES into intermediate and final services 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). 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 this paper we first present a sequential framework (DSS) for appropriate economic valuation of ecosystem services/benefits. The

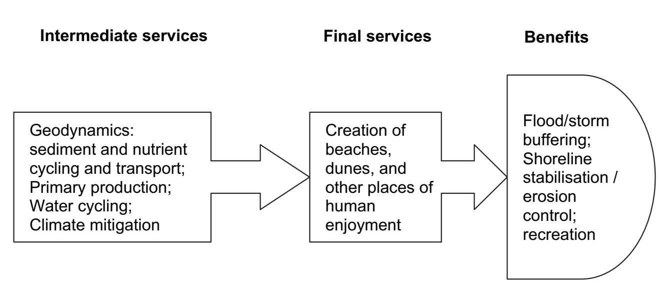
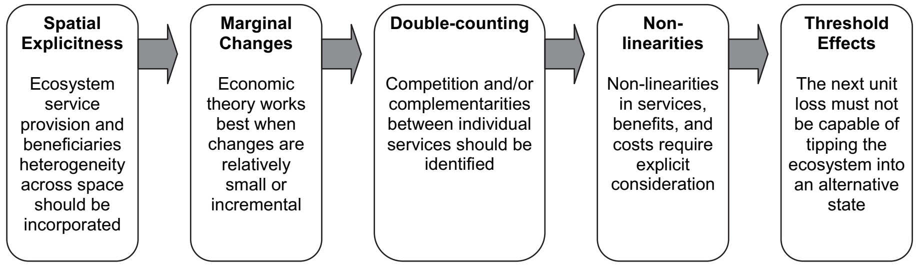


Fig. 1. Relationships among representative intermediate services, final services and benefits; an example for marine/coastal ecosystems. Adapted from Fisher and Turner (2008).



Source: Turner et al., 2010.

Fig. 2. Ecosystem Services Sequential Steps (ESSSs): A Framework for Appropriate Economic Valuation.

DSS is then applied in two UK case studies, for two different estuaries located in the East coast of England. The two case studies contain a cost-benefit model based on an ecosystem services approach, which quantifies the economic gain and loss associated with managed realignment.

The flow of many ecosystem service benefits can be enjoyed (subject to appropriate sustainable management) over long periods of time but long time horizons can prove problematic for the cost-benefit analysis method and its time discounting procedure. There is in the economic literature an ongoing debate around the correct discount rate to be applied in the social appraisal of long-term environmental projects which has not reached a satisfactory conclusion (Broome, 1995; Dasgupta, 2008; Frederick et al., 2002). Traditionally the discount rate conventionally used for assessing public sector projects was a positive and constant rate of discount. Only recently, declining discounting approaches have been recognised as an alternative[5] method when long-term environmental projects are being appraised. 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 predetermined without any moral reasoning in support of the chosen rate of pure time preference, or the ethical intra-generational decision related to the elasticity of marginal utility, or the forecasted future path of consumption growth. Although recognising the existence of hyperbolic discounting in individuals' behaviour, Dasgupta (2008) argues that in the case of public evaluation, the decision maker should be consistent over time with his/her social choice. If consumption is growing, the social rate of discount used in the CBA should be positive; otherwise the social discount rate should be declining, or even negative. That is because the declining discount effect that comes with a declining rate of economic growth does not lead to time inconsistency over project valuation. In the case studies presented, three discount approaches are applied in the CBA to investigate sensitivity to different discount rates.

As reported by Turner et al. (2010) to be most useful for policy, ES 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. We discuss each of these issues summarised in Fig. 2 in the analysis of the case studies referring to each of them as an ecosystem services sequential step (ESSS) and adding a numerical order for convenience as the ESSSs may overlap: spatial explicitness (ESSS1); marginality (ESSS2); double counting (ESSS3); non-linearities (ESSS4); threshold effects (ESSS5). For a deeper analysis of the ESSS see Turner et al. (2010).

3. Managed coastal realignment in two UK case studies: methods and results

To show how a combined ecosystem services approach and a sequential decision support system can operate in practice, we illustrate and then compare a fairly recent UK case study on the Humber estuary (North-East of England) (Turner et al., 2007) with a new UK case study on the Blackwater estuary in the southern part of the same English coastline in which managed realignment (MR) schemes could be implemented. These two case studies have been chosen because of their physical similarities and risk profiles and because they are subject to the same legislative regime. Both estuaries are located on the east coast of England where major issues of flooding because of sea surge can occur and which may be exacerbated due to climate change effects such as sea level rise and vertical land movements associated with the melting of ice sheets (glacial isostatic adjustment). Actions will be different depending on the area at risk. In estuaries along the English east coast some managed realignment (MR) experimental schemes have already been implemented. MR is controversial because some previously reclaimed coastal land (usually agricultural land) is sacrificed in order to reduce the threats of coastal erosion and flooding along the coast (Royal Commission on Environmental Pollution, 2010). Sea level rise, increased cliff erosion and more extensive and frequent flooding, is potentially putting at risk private owned houses and business assets – estimated to encompass 4 million people and collateral worth £200 billion across England and Wales (O'Riordan et al., 2008).

Adaptive coastal governance is currently evolving and inevitably it is fragmented with little integration among public, private and voluntary organisations interests concerned with coastal adaptation (Nicholson-Cole and O'Riordan, 2009). The EU has promoted an integrated coastal zone management (ICZM) strategy, which is slowly becoming embedded in national strategies. The governance of coastal areas for England and Wales takes place at three levels: political (central government departments), executive (statutory but non-departmental agencies) and through civic organisations and coastal communities (other organisations concerned with the coastal management). The first level includes: the Department of Food and Rural Affairs (DEFRA) that provides the overall policy as well as the supervision of the non-departmental agencies; the Department for Communities and Local Government (DCLG) that provides planning policy and local government finance; and HM Treasury, which provides general funding to DEFRA, the non-departmental agencies and local authorities. Local Government County and District Councils operate both at the political and at the executive level as they are concerned with local planning, economic

[5] Official guidance on UK public sector project/policy assessment now advocates such an approach (HMT, 2003).

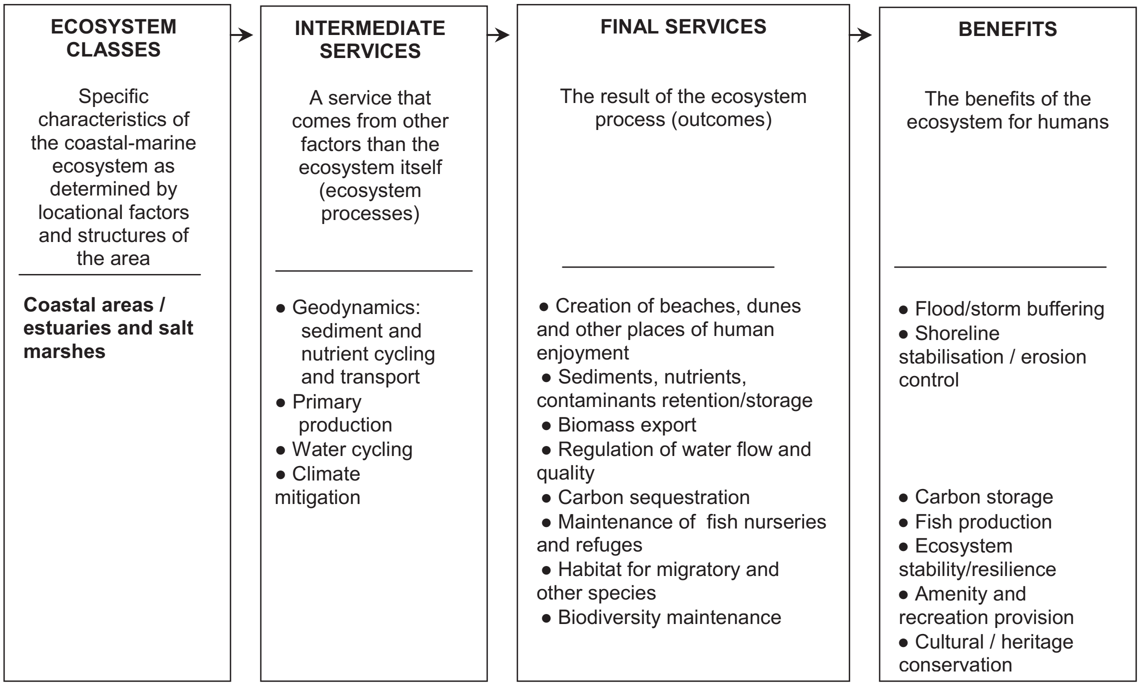


Fig. 3. Classification of Coastal and Marine Ecosystem Services.

development, and social well-being providing funding for flood and coastal erosion protection. At the second level we find the two relevant governmental agencies, Environment Agency (EA) and Natural England (NE), and the Local Maritime Authority. At the third level there are local residents and communities, and non-governmental bodies such as the National Trust and the Royal Society for the Protection of Birds, which own coastal land managed as natural reserves (O'Riordan et al., 2008). In general, there is distrust among local communities about government intentions and plans for coastal adaptation measures. This context has been conditioned by uncertainty surrounding the goals of coastal adaptation because of the scientific uncertainties linked to future climate change, and uncertainty about compensation measures for those that will be adversely affected by the changes in the coastline.

For these reasons two main instruments have been recently used in England and Wales: Shoreline Management Plans (SMPs) and the Pathfinder. SMPs are the instrument, implemented in the mid 1990s, to reach a large-scale assessment of the risks associated with coastal processes and the technical feasibility of policies that work with natural processes. The second generation of SMPs, which are delivered by 'Regional Coastal Groups' in partnership with the EA, is due to completion by December 2010 and will cover the entire coast of England and Wales (6000 km) (Postnote, 2009; Environment Agency, 2010). Another new 18 months (2009–2011) governmental scheme called Pathfinder will support (£11 million budget) fifteen pathfinder authorities (County and District Councils) to explore new ways of adapting to coastal change (Defra, 2010). The North Norfolk District Council received the largest amount (£3 million) to implement the scheme in its local area, and it is the first to have put forward, in 2010, a 'compensation' plan using some of the funding of the Pathfinder scheme. The council is making individual offers to the owners of cliff top homes of between 40 and 50% of the theoretical value of their homes if they were inland and not at any risk of coastal erosion. However, it is not yet clear precisely which kind of support mechanism will be taken up more generally by pathfinder projects (North Norfolk District Council, 2010).

As shown in Fig. 3, estuaries and saltmarshes yield benefits in terms of ecosystem services: flood protection; carbon storage benefits (climate change mitigation effect); fisheries' productivity (fish industry and recreation); recreation and amenity benefits (e.g. walking, bird watching etc); existence value benefits (e.g. biodiversity maintenance); and sea walls maintenance cost savings.[6]

In the next section similarities and differences in the methodology adopted for the analysis of the two case studies is explained. The Humber estuary comes first chronologically and most of its methodology is then used for the Blackwater estuary case study. However, some improvements in the valuation of the ecosystem services identified for the Blackwater were introduced in the new analysis. The section concludes by reporting on and comparing the results of the two case studies in which an ecosystem services approach (Fisher and Turner, 2008) is used.

3.1. Methods

The methods applied in the analysis of each estuary case study are examined following the ESSSs (sequential DSS). The anchor point for the whole analysis is the use of a cost-benefit analysis (CBA) for managed realignment schemes appraisal, which was originally applied to the Humber case study (Turner et al., 2007). The same CBA approach was then used for the analysis of the Blackwater MR scheme. The general approach is the following (Equation (1)–(3)): comparison of the present value (PV) of the 'status quo' (SQ) existing protection system and the present value of MR schemes.

The PV of the 'status quo' scenario can be appraised on the basis of Equation (1):

[6] The implementation of managed realignment schemes at appropriate sites can recreate saltmarshes, which can act either as a more natural coastal defence for the coast or provide protection in front of existing engineered defences, implying savings on the sea walls maintenance costs.

1. Location of the area below the high spring tide level. 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

Fig. 4. Geographical Information System (GIS) based realignment site location criteria (adapted from Turner et al., 2007).

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

where: PV_t^{sq} is the present value of total costs of current defences (£ million) as there are no future benefits in the status quo scenario, r is the discount rate, l^{sq} is the length of defences, and $C_{m,t}^{sq}$ are the maintenance costs (£/km/yr).

The present value of the costs and benefits of the managed realignment scheme is given in Equation (2):

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

where: PV_t^{mr} is the present value of the managed realignment scheme (£ million), r is the discount rate, a_h^{mr} is the area of inter-tidal habitat created, $B_{e,t}$ are the ecosystem value benefits (£/ha), l^{mr} is the length of managed realignment (km), $C_{k,t}^{mr}$ are the capital costs of realignment, $C_{m,t}^{mr}$ are the maintenance costs, a_t^{mr} is the agricultural land lost, and $L_{agr,t}^{agr}$ is the forgone agricultural land value.

The overall CBA result is found via Equation (3):

$$NPV_t^{mr} = \left(PV_t^{mr} - PV_t^{sq}\right) \tag{3}$$

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).

In both cases, the capital costs of implementing the scheme usually include the costs of realigning defences whenever a secondary line of defence was necessary further inland; and opportunity costs associated with any agricultural land that would be converted in inter-tidal habitat. Other costs are related to the maintenance costs of old and new sea walls.

The ecosystem value benefits gained recreating inter-tidal habitats are identified via an ecosystem services approach as reported in Section 3.1.1 and in further detail in Section 3.1.2.

3.1.1. ESSS1 – spatial explicitness

Knowledge of the underlying biophysical structure and processes of any given ecosystem service is crucial. Spatially explicit analysis must then encompass appropriate socio-economic, political and cultural parameters in order to properly identify ecosystem services supply and demand side beneficiaries. Importantly, ecosystem services are context dependent in terms of their provision and their associated benefits and costs, which means that service values may change across the landscape due to geographical variations (scale) in biophysical supply or demand of a service.

A generic ecosystem services approach was used for the Humber estuary. The ecosystem services/benefits identified were: habitat values, which were treated as a composite value related to intermediate services such as biodiversity, nutrient cycling and water quality given that those service contributions are captured by the estimation of their human welfare impact via amenity and recreational values; and carbon storage value. For the Blackwater case study the ecosystem services approach (Fisher and Turner, 2008) as presented in Section 2 was used. That allowed us to identify similar benefits to the Humber (amenity and recreation as a composite environmental benefit, and carbon storage) plus another benefit, fisheries production values. The implementation of MR schemes in both estuaries can provide savings on sea walls maintenance. In some cases when new saltmarshes are recreated in front of a secondary defence inland wall additional costs must be borne, but if the land elevation provides enough flood defence a new sea wall is not required.

To identify the more appropriate sites for managed realignment for both estuaries Geographical Information System (GIS) techniques were used in combination with four possible future scenarios. The GIS methodology was originally developed for the Humber estuary case study and then replicated for the Blackwater estuary (Fig. 4). The GIS-based realignment site location criteria were used to minimise the opportunity costs of realignment involving significant social justice/ethical concerns: urban centres were assumed to be protected and therefore excluded from the analysis; and only the sacrifice of agricultural land for the purpose of saltmarsh recreation was taken into account. However, as food security issues continue to rise up the policy agenda the sacrifice of any usable agricultural land will result in escalating opportunity costs.

3.1.1.1. Humber estuary.

The Humber estuary is one of the largest in the UK with a maximum tidal length of 147 km from Cromwell Weir on the Trent to the Humber's mouth, and a maximum width of 15 km. The Humber is of international importance for wildlife with a large area of inter-tidal habitat, mostly mudflats and sandflats and some saltmarsh (Turner et al., 2007).

The suitable areas of realignment identified via GIS layers were combined with four possible future policy scenarios: business-as-usual (BAU), which takes into account existing realignment schemes; policy targets (PT), in which economic growth is combined with environmental protection; deep green (DG) is a scenario in which environmental protection takes priority over economic growth; and extended deep green (EDG) places even a greater emphasis on habitat creation than the DG scenario. 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 which other scenarios were evaluated. In the HTL scenario existing defences are maintained to a satisfactory standard, but loss of inter-tidal habitat will continue. 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

Table 1
Details of areas suitable for realignment in the Humber and Blackwater estuary (Adapted from Turner et al. (2007)).

	Scenarios				
	HTL	BAU	PT	DG	EDG
Humber estuary					
Length of defences before realignment (km)	405.3	405.3	405.3	405.3	405.3
Length of defences after realignment (km)	405.3	396.8	361.6	318.2	284.5
Length of realigned defences (km)	0.0	7.0	30.8	69.0	102.7
Amount of inter-tidal habitat created by realignment (ha)	64.6	61.9	42.2	38.2	34.0
Estimated amount of lost agricultural areas (ha)	0.0	80.0	1320.9	2332.4	7493.6
Estimated tonnes of carbon stored each year (t)[a]	0.0	38.4	634.1	1119.4	3597.1
Blackwater estuary					
Length of defences before realignment (km)	124.1	NA	124.1	124.1	124.1
Length of defences after realignment (km)	124.1	NA	121.2	106.5	85.66
Length of realigned defences (km)	0	NA	2.9	18.4	40.17
Amount of inter-tidal habitat created by realignment (ha)	0	NA	81.6	816.5	2404.1
Estimated amount of lost agricultural areas (ha)	0	NA	9.9	365.9	886.8
Estimated tonnes of carbon stored each year (t)[a]	0	NA	21.70	217.19	639.49

[a] Estimates of the carbon storage capacity of newly created inter-tidal habitat are derived from (Andrews et al., 2000) for the Humber estuary and from (Adams, 2008) for the Blackwater estuary.

current sea defence policy includes an allowance for increased risk in line with the United Kingdom Climate Impacts Programme (2002) (UKCIP02) information. However, due to uncertainty over the loss of inter-tidal habitat because of coastal squeeze over the next 50 years, it was assumed that no further coastal squeeze takes place.[7] Details of the areas that were identified as suitable for realignment in the Humber (length of defences before and after realignment; length of realigned defences; amount of inter-tidal habitat created by realignment; estimated amount of lost agricultural areas; and estimated tonnes of carbon stored each year) are illustrated in Table 1.

3.1.1.2. Blackwater estuary. The Blackwater estuary covers 5500 ha with open water, mudflats and saltmarshes.[8] This case study builds upon the Humber case study methodology. The same scenarios were considered with the exception of the BAU scenario, because in this study existing experimental realignment areas in the estuary created between 1991–2002 – Tollesbury (21 ha), Orplands (40 ha), Northy Island (0.6 ha) and Abbotts Hall (84 ha) – were considered as current saltmarshes (in 2006). Utilising GIS methods, we investigated only the potential for newly created saltmarsh areas. The same GIS methodology and criteria applied in the Humber (see Fig. 4) were focused on the Blackwater, see Table 1.

3.1.2. ESSS2 and ESSS3 – marginality and double counting (cost and benefit values of managed realignment schemes)

All CBA studies utilise marginal costs/benefits changes and we deal with some of the relevant issues in Section 3.3. In this section we focus on the double counting dilemma. The potential problem of double counting may occur where competing ecosystem services are valued separately and the values aggregated (e.g. separate valuation of recreational and industrial fish nurseries of the same fish species); or, where an intermediate service (e.g. nutrient cycling, or water quality) is first valued separately, but also subsequently through its contribution to a final service benefit (e.g. recreation gains from improved water quality). As in the Humber, case study care was taken to ensure that the aggregation of benefits values was appropriate.

A CBA assessing the efficiency of MR schemes was undertaken comparing costs and benefits of the scheme. In the Blackwater case study (in line with the Humber analysis) it was assumed that there was no reduction in the level of protection (vis-à-vis hard defences) where new saltmarshes were put in place. Market analysis was used to estimate the cost savings on hard defences. In the Humber case study, sea defences cost savings were estimated on the basis of maintenance cost values at the time of the analysis (Black and Veatch/Halcrow, 2005). For the Blackwater case study the maintenance costs of non-realigned defences were based on data in the Environment Agency Blackwater management strategy (Halcrow, forthcoming) for the Blackwater and Colne estuaries. The cost of realigned defences was assumed to be 50% of the non-realigned defences.

Compared to the Humber study, in the Blackwater new geochemical and biological data from study site(s) fieldwork campaigns (Adams, 2008; Fonseca, 2009) were used to estimate the value of carbon storage (climate change impacts mitigation) and the value of an extra benefit (fish production). Table 2 shows the details of the fisheries production.

Table 2
Value of bass (£ per hectare) contributing to inshore fishery after five years.

Survival parameter estimates:	Upper	Mean	Lower
Value per hectare at average wholesale price (£7/kg)	47.45	11.55	1.93
Value per hectare at lowest wholesale price (£4.50/kg)	30.50	7.43	1.24
Total weight (kg) of juvenile bass per hectare surviving to 36 cm after 5 (or 4) years	6.78	1.65	0.28

An estimate of the fish production value in the newly created saltmarsh nurseries (saltmarsh creeks) is obtained multiplying the value per hectare of the fish species considered times the hectares of new saltmarsh created. Linearity is assumed in the PT through the EDG scenario (the bigger the extent of the saltmarsh, the higher the fish productivity). The fish production function is calculated in Fonseca (2009), and it is based on quantitative estimates of the abundance of juvenile bass up to two years old. The estimated annual abundance[9] range of juvenile bass per hectare of saltmarsh was applied to a range of average survival rates, to an approximate length of 36 cm, the minimum size for legal commercial capture of wild sea bass in the UK. These estimates[10] were then applied to a range of local market values for wild-caught bass[11] to estimate their economic contribution to the local inshore fishery in units of kg of bass per hectare of saltmarsh. The fish considered were only

[7] If future sea level rise exceeds current expectations, or new estimates become available, both traditional and MR strategies will need to be reappraised.

[8] http://www.maldon.gov.uk/Leisure/ParksAndOpenSpaces/Parks/Blackwater+Estuary.htm.

[9] The abundance is based on Fonseca's catch per 0.1 ha and extrapolated for the size of the site.

[10] These estimates are only a guideline and are based on a number of ecological and economic assumptions. The assumptions are that the following are constant: water volume, fish distribution, habitat quality, catchability, the total yield of bass from a number of different cohorts in any one year equals one cohort over the whole of its life span, the ecological carrying capacity and resilience of the estuary, the biological and physical processes determining supply of 0 to 3-group bass to the saltmarshes over time, the natural and fishing mortality rate, the migration of 3- and 4 group bass away from the local market, and the price at local market.

[11] The price of wild-caught bass is very volatile and can be influenced by the farmed bass market (Fishing News, January–September 2007; Campbell I., Essex Sea Fisheries Officer, personal communication).

Table 3
Net carbon burial per year, amended to take account of greenhouse gases production (CH_4 and N_2O converted in CO_2eq using Greenhouse Warming Potential – GWPs). 1.5 and 6 mm assumed sedimentation rate. All figures are in tonnes, per hectare, per year.

Relic and spartina marsh	C burial	CH_4 (actual flux)	CH_4 (CO_2eq)	N_2O (actual flux)	N_2O (CO_2eq)	Net C burial (C sequestration)
1.5 mm	*1.027*	0.0012	*−0.025*	0.00237	*−0.735*	*0.266*
6 mm	*4.108*	0.0012	*−0.025*	0.00237	*−0.735*	*3.347*

those species sold in the market[12] and were therefore valued using market analysis. Table 2 shows the value of bass per hectare contributing to inshore fishery after five years and for every year thereafter, so for the first five years in the CBA tables the value of fisheries is set to zero. Following a conservative approach, the value of bass chosen for the CBA is the mean value per hectare at lowest wholesale price.[13] It is also the case that other fish species (non commercial) benefit but this value was not estimated.

Climate change impacts mitigation can be difficult to value because the (positive) capture of greenhouses gases has to be balanced against the (negative) release of greenhouse gases in the atmosphere in the process of carbon storage. However, it has been shown that this releasing phenomenon is less likely to occur in saltmarsh areas than in fresh water marsh areas. Table 3 shows the details of the carbon storage for the Blackwater estuary. The net carbon burial (positive numbers give net carbon burial/sequestration, negative numbers give net release of greenhouse gases) per year, amended to take account of greenhouse gases production (CH_4 and N_2O converted in CO_2eq using Greenhouse Warming Potential GWPs),[14] assuming 1.5 mm and 6 mm sedimentation rates respectively in areas with relic and spartina marsh. In both cases there is a positive carbon sequestration: 0.266/t/ha/yr; and 3.347/t/ha/yr. Carbon storage was valued, as in the Humber study, with the damage cost avoided method i.e. the monetary value of the avoided carbon releases to the atmosphere because of storage (Pearce, 2003; Stern, 2007; Tol, 2005).

For the purpose of sensitivity analysis in both estuary case studies different values of carbon storage were used. In the Humber estuary three distinct values were used: £45 in 2005 (Tol, 2005); £222/tC (Tol, 2005); £4/tC (Pearce, 2003). In the Blackwater case study four different prices were considered: the highest price that could be used is the figure suggested formerly by Pearce et al. (1996) and confirmed by Tol (2005) of about £30 in 2007; another much lower figure that could be applied is £7, which is in the range of the estimates recommended by the Second Assessment Report (Arrow et al., 1996) as well as in the range suggested by Pearce et al. (1996) and confirmed by Li et al. (2004); another more recent estimate for the social cost of carbon (SCC) is the figure proposed by the Stern Review (2007): \$85/tCO_2, which is equal to \$350/tC in 2005 prices (Stern, 2007; Nordhaus, 2007), which is around £230/tC in 2007. However, as reported by Tol (2008) in a meta-analysis on estimates of the social cost of carbon, the Stern (2007) estimate is almost an outlier. Another figure is given by the mean price of traded carbon that in October 2007 was equal to €21.50[15] which is about £15.

In the Humber estuary case study an indicative value of recreation and amenity benefits was obtained using benefit transfer values on the basis of the results of a meta-analysis on wetland values (Woodward and Wui, 2001). Environmental benefits transfer is a technique by which the results of previous monetary environmental values originally estimated at one site are 'transferred' to another site in a new policy or new decision making context. The advantages of using this technique include financial and time resource savings. Potential disadvantages are the fact that the valuation studies used for the meta-analysis on which a benefit transfer value is based, might not have the exact same characteristics of the good or the area under investigation.

In the Blackwater estuary case study, the amenity and recreation benefit was estimated (willingness to pay) *in situ* with a site-specific stated preferences study: a choice experiment (CE). Stated preference techniques are survey based studies in which respondents are asked to state their willingness to pay (or willingness to accept) a policy against a status quo policy or other policies. With this technique there is the advantage of having a valuation specific to the area of interest. However, the application of this methodology is not without criticism because respondents might not be familiar with the object of the investigation and the valuation results can be biased by unexpected behavioural effects. In a choice experiment respondents are asked to make a choice between alternative goods or policies that are specified with several attributes and different levels of those attributes (choice set). Conventionally, in one questionnaire an individual is asked to state his/her preference over more than one choice set. Based on the ecosystem services approach, the expected water quality improvement and biodiversity enhancement (including fisheries not sold in the market) are assumed to be captured by the value estimated (willingness to pay) for the recreational value of the newly created saltmarshes. The choice experiment design, survey and results are illustrated in detail in Section 3.1.3.

3.1.3. Blackwater saltmarsh amenity and recreation choice experiment

3.1.3.1. Design and survey. In the Blackwater estuary, management strategies have been proposed to provide sustainable flood defence, and experiments have been running since 1990s on the managed realignment of the coast (at Tollesbury, Orplands, Northy Island and Abbott's Hall) to mitigate flood risk. Fig. 5 shows the study and sampling area, and the managed realignment sites already implemented in the estuary.

Following a series of preliminary meetings and focus group investigations the following attributes and their units of measurement were selected (Table 4):

- The area of new saltmarshes to be created[16] (variable label '*AREA*'); measured both as acres[17] and as the corresponding number of football pitches[18];

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

[13] The survival rates were calculated after: Lorenzen (1996).

[14] Since CO_2, CH_4 and N_2O all absorb different amounts and frequencies of radiation, and have different atmospheric residence times, the amounts of each gas cannot be compared directly, either as mass to mass or volume to volume. To make such a comparison the masses of each gas are first converted using the Greenhouse Warming Potential (GWP). Over a 100 year time frame the GWP of CH_4 is 21 meaning that it contributes 21 times more to anthropogenic global warming than an equal amount of CO_2. Over the same time frame the GWP of N_2O is 310 (Forster et al., 2007). The conversion gives carbon dioxide equivalent (CO_2eq) values of the greenhouse gases allowing a meaningful comparison of carbon burial and greenhouse gas fluxes in inter-tidal areas.

[15] That value is reported by http://www.pointcarbon.com. It should be noticed that this price is higher than usual due to the dollar price that in the same period was anomalously going down.

[16] The extent of saltmarsh for the *AREA* attribute based on the future scenarios assumed was determined using GIS techniques as described for the Humber estuary.

[17] For the econometric analysis the acres were then converted in hectares.

[18] In British English a football pitch is the expression to define a football field. In other words, the football pitch defines the extent of the play ground of a football stadium.

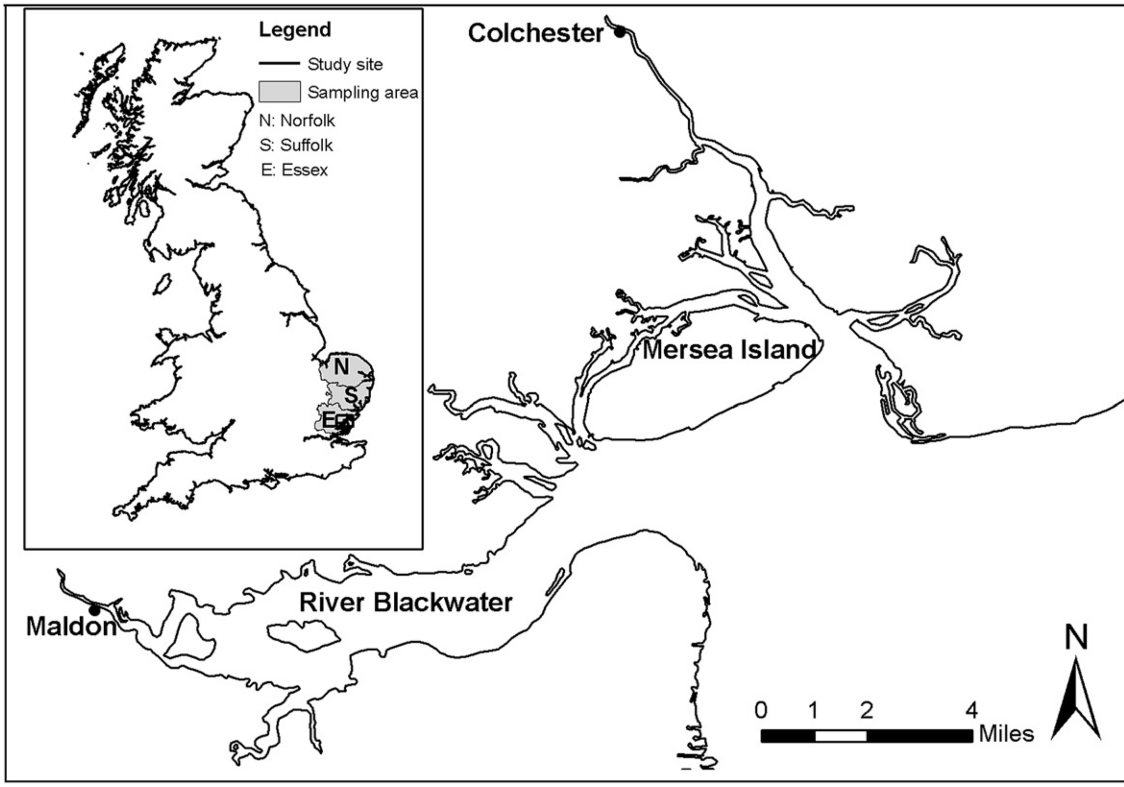


Fig. 5. Study and sampling area map.

- Bird species observable (‘*BIRDS*’): measured as the number of protected species returning in the estuary and reflecting the quality level of the environment in the area[19];
- Distance from respondent's home to the nearest site (‘*DISTANCE*’); measured in miles. As shown in Table 4, the correspondent attribute levels are divided into those seen by the ‘*Far*’ sample (living in Norfolk and Suffolk) and those seen by the ‘*Near*’ sample (living in Essex);
- Whether the created saltmarsh 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.

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 saltmarshes were created. The questionnaire included some introductory sections to inform respondents about saltmarshes and to test their familiarity with such environments. The hypothetical policy of MR for the Blackwater was introduced explaining that the policy would not change the level of flood protection and listing the ecosystem service benefits of saltmarshes explicitly. The number of bird species in a saltmarsh was used as a good indicator of general environmental quality. A specific section introduced the respondents to the choice sets of the CE in which each attribute was fully explained. To verify the level of understanding of the respondent (data quality check) a final question openly asking what the respondent was thinking while making each choice was asked soon after the last choice set. Respondents' socio-demographic characteristics were also recorded. Interviews were conducted by a team of trained interviewers at various locations within both the ‘*Near*’ (within Essex) and the ‘*Far*’ (within Norfolk and Suffolk) distance zones.[20]

3.1.3.2. Results and discussion. 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.

To value the composite environmental benefit for the CBA, the willingness to pay (WTP) for saltmarsh creation in the Blackwater was calculated on the econometric model estimated for the ‘*Near*’ sample.[21] To take account of the fact that each individual made eight choices, a random effects logit model[22] was estimated. The specification for the model estimated is the following:

$$V_{\mathrm{ij}} = \beta_1 CONST_{\mathrm{ij}} + \beta_2 LnAREA_{\mathrm{ij}} + \beta_3 BIRD3_{\mathrm{ij}} + \beta_4 BIRD4_{\mathrm{ij}} + \beta_5 \mathrm{BIRD5}_{\mathrm{ij}} - \beta_6 \mathrm{DIST}_{\mathrm{ij}} + \beta_7 ACC_{\mathrm{ij}} - \beta_8 TAX_{\mathrm{ij}}$$

[19] Birds represent a final link in the food chain of salt marshes, and they also represent saltmarsh biodiversity in general (non-use values), so higher numbers of protected bird species within the estuary were taken as a proxy for greater environmental quality in the area.

[20] 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.

[21] The results of the econometric model for the ‘*Far*’, and a *Pooled* sample, are presented elsewhere (Luisetti et al., in press).

[22] The logit model is used to model the probability of occurrence of binary responses (yes/no; or, in the case of a choice experiment, preference – choice – over alternative options). In a binary choice experiment the probability that individual *i* prefers alternative 1 over alternative 0 (the status quo) is modelled by: $P(Y_i = 1) = \frac{Exp(X_i'\beta)}{1+Exp(X_i'\beta)}$ where Y_i is a bivariate random variable (the choice of individual *i*); X_i' is a vector of explanatory variables (attributes etc.) and β is the corresponding parameter vector. The model is then estimated by maximum likelihood (Greene, 2007; Kanninen, 2000).

Table 4
Attribute levels used in the choice experiment design.

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

[a] fp stands for football pitches.

where V_{ij} is the deterministic part of utility influenced by the choice set attributes presented to individual i (i = 1,…,N) in choice question j (j = 1,…,Ji − in the general case the choice questions range from one to a person specific J_i; in this study J_i is equal eight for all respondents) and the βs are the estimated coefficients of the variables defined in Table 4.

The results of the model estimation are shown in Table 5.

The *CONSTANT* is positive but not significant meaning that, everything else held constant, the respondents receive more utility from the new saltmarsh projects going ahead than not.

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. Although a random effects logit model with the *AREA* coefficient provided a positive and highly significant (99%) coefficient, from an economic point of view this result did not seem sensible. For that reason, the functional form of the *AREA* attribute is assumed to be non-linear. We therefore 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*.

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 and of the expected positive sign indicating that providing habitat for a higher number of endangered bird species results in higher utility levels. An examination of parameter values suggests that the satiation point is attained at the *BIRD4* level suggesting a positive but declining marginal WTP for bird habitat.

Table 5
Random effects logit model for the '*Near*' sample.

Variables	Coefficient	Standard error	*P*-value
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	
Choice questions		8	

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

The *DISTANCE* variable has the expected negative parameter value and is significant. This result shows that, as expected, the utility of new wetland sites diminishes the further away they are from the individual's home (an effect known as 'distance decay'). A deeper analysis of the distance variable highlights the non-linearity of this attribute (ESSS4). 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 perceived to be higher. The following attribute levels have, as predicted by the distance decay, a lower utility than the first and the second attribute level. A behaviour similar to the one observed by the hedonic literature.

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

Finally, the rho coefficient is statistically different from zero confirming that the use of a random effects logit model for our specification was appropriate.

3.1.3.3. Willingness to pay estimates. Aggregated 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 for a change in a hypothetical policy and it is possible to derive as many WTP estimates as the policies considered depending on the attributes and attribute levels selected. For each policy, instead, MWTPs are the implicit prices for each attribute of the choice experiment (CE). So, in order to calculate the WTP, we need to define the policy options and to estimate the MWTPs (Table 6). Multiplying attribute's MWTPs with the selected attribute levels, and summing them up together, we get the value of the WTP (Kanninen, 2000; Haab and McConnell, 2002).

In the specification of the model for the econometric analysis of the choice experiment, we assumed that the attribute *AREA* was not linear. After investigation, the logarithmic functional form for this attribute seemed to be most appropriate.[23] That means that the WTP for the attribute *AREA* does not tell us the respondents' WTP for 1 ha increase in the creation of new saltmarshes. Instead, the result in Table 6 describes the WTP for a marginal increment of the natural logarithm of the area.

The use of dummy variables for the *BIRDS* coefficient (*BIRD3*, *BIRD4* and *BIRD5*) allows us to quantify the MWTP of the respondents when they were offered one, two, or three more endangered species respectively, over the basic conservation level of two species. This attribute was used in the CE as an index of general environmental quality of the area and hence representing the

[23] Non-linearity was investigated with two random effects logit models: a model with the variable *AREA* and *AREA square* and a model with the natural logarithm of the variable *AREA*. The first model was run to verify if a non-linear model was actually superior. The second model was run because intuitively considered the more sensible functional form for the attribute *AREA*. In fact, the quadratic form of the attribute assumes, after a specific point, a declining preference for that attribute. The logarithmic form, instead, assumes that once reached a specific point the preference for the attribute *AREA* would stay the same. The first model shows an *AREA* square coefficient negative, but not significant indicating that a linear model would fit better. However, the model with the logarithm shows a positive and highly significant (99% level) coefficient, providing evidence of the appropriateness of this model.

Table 6
Implicit prices for the choice experiment attributes (Marginal willingness to pay - Marginal WTP) in the *Near* sample.

Attribute	Unit	Attribute coefficients	Marginal WTP (£)[1]
LnAREA	Hectares	0.2519	1.11
BIRD3	Number of bird species	0.4172	1.84
BIRD4	Number of bird species	0.6937	3.07
BIRD5	Number of bird species	0.8078	3.57
ACCESS	yes/no	0.9746	4.31

Notes: 1. Tax coefficients: *Near* sample = −0.2261.

non-use values. The interaction effects between the dummies for *BIRDS* and the attribute *ACCESS* were not significant revealing that respondents did not assign any use values to the attribute *BIRDS*. So the variables *BIRD3*, *BIRD4* and *BIRD5* represent ascending levels of environmental quality and their MWTPs represent their respective implicit prices. It is worth noting that respondents assigned similar economic values to the higher levels of environmental quality and to the *ACCESS* attribute.

The attribute *ACCESS* was dummy coded. The MWTP for that attribute represents the value that respondents give to the access to new saltmarshes, and the potential to visit and enjoy them representing the use value of the saltmarshes.

An aggregated WTP for two different policies creating new saltmarshes in the Blackwater estuary was then calculated. That WTP represents the estimated value for the amenity and recreation benefits (as a composite environmental benefit, biodiversity is included in that benefit as a final service so to avoid double counting) that will be inserted in the CBA. Policy 1 in Equation (4), encompasses use and non-use values of the saltmarshes, and the aggregated WTP is defined for a policy involving a given extent of new saltmarsh area created (corresponding to the realignment scenarios) and for a high (*BIRD5* variable) level of environmental quality, involving different distances, and with the possibility of access.

$$\mathrm{WTP} = \frac{\beta_2}{\beta_8} LnAREA + \frac{\beta_5}{\beta_8} BIRD5 - \frac{\beta_6}{\beta_8} DIST + \frac{\beta_7}{\beta_8} ACC \tag{4}$$

where β_8 is the coefficient for the cost attribute, β_2, β_5, β_6, andβ_7 are the coefficients for the *LnAREA*, *BIRD5*, *DISTANCE*, and *ACCESS* attribute respectively.

Policy 2 is similar to Policy 1 but follows a conservative approach (utilising only use values) as shown in Equation (5) where the variable *BIRD5* is dropped from the equation:

$$\mathrm{WTP} = \frac{\beta_2}{\beta_8} LnAREA - \frac{\beta_6}{\beta_8} DIST + \frac{\beta_7}{\beta_8} ACC \tag{5}$$

Given the non-significance of the constant coefficient, and based on Morrison et al. (2002), the authors 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.

The variable *DISTANCE* provides a key determinant of the aggregated WTP that allows for non-linearities (ESSS4). 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. 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 saltmarsh area created for each scenario, the aggregated WTP for the PT, DG and EDG scenarios was calculated as reported in Table 7.

In the PT scenario the area of saltmarsh created is 81.6 ha. 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 ha) 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. Similarly, while the EDG scenario further increases saltmarsh area (2404.1 ha) 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). 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 enhancements. 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.

3.1.4. Costs and benefits – summary

A summary of the costs and of the ecosystem benefits and related values for both estuary case studies is reported in Table 8. The costs identified in both case studies are those common to any MR scheme: capital costs of realignment; lost agricultural land costs based on sale prices (Defra, 2004, 2005). Replacement costs of defences were only considered for the Humber case study as for the

Table 7
Aggregated willingness to pay (WTP) for the Blackwater saltmarshes under the Policy Targets (PT), Deep Green (DG) and Extended Deep Green (EDG) scenarios (£ thousand).

Distance to Abbott's Hall (miles)	Households population	PT: 81.6 Hectares		DG: 816.5 Hectares		EDG: 2404.1 Hectares	
		WTP	WTP	WTP	WTP	WTP	WTP
		use and non-use values	use values only	use and non-use values	use values only	use and non-use values	use values only
		Benefits (£/yr)	Benefits (£/yr)	Benefits (£/yr)	Benefits (£/yr)	Benefits (£/yr)	Benefits (£/yr)
8	63706	771	547	930	706	1005	781
15	349836	4107	2875	4981	3749	5392	4160
23	97974	1109	764	1354	1009	1469	1124
32	33185	360	243	443	326	482	365
		6347	4429	7708	5790	8348	6430

Table 8
Values used to estimate the costs and the benefits of realignment for the Blackwater estuary (2007 base year).

Costs and benefits	Humber	Blackwater
	Value at time of reference (2005)[a]	Value at time of reference (2007)[a]
Capital costs of realignment[b]	£878,159/km	£929,252/km
Replacement costs[c]	£668,441/km	n/a
Maintenance cost of non-realigned defences[d]	£3560/km/yr	£866/km/yr
Maintenance cost of realigned defences[e]	£1780/km/yr	£433/km/yr
Opportunity costs		
Grade 1 and 2 agricultural land[f]	£4790/ha	NA
Grade 3 agricultural land[f]	£5458/ha	£5138/ha
Benefits		
Amenity and recreation[g]	£621/ha/yr	see Table 7
Carbon sequestration benefits	£222; £45; £4/tC	£230; £30; £15 and £7/tC
Fish nursery benefits	n/a	£7.43/ha

n/a denotes aspects of one cost or benefit that were not available for one case study or the other.
[a] All values are converted to 2005 prices for the Humber estuary and to 2007 prices for the Blackwater estuary using the GDP deflators published by HM treasury (http://www.hm-treasury.gov.uk/).
[b] Costs based on contemporary realignment schemes (Halcrow, 2000).
[c] Only the costs of replacing unsatisfactory defences (Defra, 2001)not affected by realignment are included.
[d] Maintenance costs are taken from Black and Veatch/Halcrow (2005) for the Humber case study. These are assumed to increase in the future due to the effects of climate change. Following current government guidance (Penning-Rowsell et al., 2005) maintenance costs are increased by a factor of 1.5 for the period between 20 and 50 years into the future and by a factor of 2 for years further into the future. For the Blackwater, values are based on EA Blackwater management strategy (Halcrow, 2007) for the Blackwater and Colne estuaries.
[e] Assuming maintenance cost being 50% than for non-realigned defences.
[f] Based on sale prices (Humber – (Defra, 2004)); Blackwater, (Defra, 2005)and adjusted downwards for the effects of the single farm payment following (Penning-Rowsell et al., 2005).
[g] For the Humber, benefit transfer values were used. For the Blackwater the WTP values reported in Table 7 were used.

Table 9
Net present values (NPVs) in the Blackwater case study using conservative values (only use values) different discount rates (£ million)[a].

	25yrs	50 yrs	100 yrs
Discount rate: **constant**			
Policy targets (PT)			
NPV = PV(PT) − PV(HTL)[b]	**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**			
Policy targets (PT)			
NPV = PV(PT) − PV(HTL)	**70.29**	**156.21**	**315**
Deep green (DG)			
NPV = PV(DG) − PV(HTL)	**76.71**	**189.31**	**397.39**
Extended deep green (EDG)			
NPV = PV(EDG) − PV(HTL)	**64.71**	**190.18**	**422.05**
Discount rate: **gamma**			
Policy targets (PT)			
NPV = PV(PT) − PV(HTL)	**82.9**	**169.41**	**353.5**
Deep green (DG)			
NPV = PV(DG) − PV(HTL)	**93.24**	**206.6**	**447.84**
Extended deep green (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, 2.5% for years 76–125) (HMT, 2003); gamma discounting method (4% for years 1–5, 3% for years 6–25, 2% for years 26–75, 1% for years 76–300) (Weitzman, 2001).
[a] Carbon valued at £7.
[b] The present value (PV) of the hold the line (HTL) scenario is always negative because it is assumed that, in the present scenario, benefits are zero.

Blackwater there was no data available. The benefits values shown in the table are those reported in Sections 3.1.2 and 3.1.3.

3.2. *Results – a comparison*

For both case studies the PV of the proposed managed realignment projects have been calculated for a time horizon of 25, 50 and 100 years and with different positive discount rates: a constant rate (3.5%); a declining rate (3.5% for years 1–30, 3% for years 31–75, 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, 1% for years 76–300). The results of the analysis showed positive net present values (NPVs) for both estuaries.

As Table 10 shows, in the Humber NPVs are positive for periods of time longer than 25 years – the longer the time horizon, the more economically efficient is the policy. The results of the CBA for the Blackwater estuary are even stronger than those of the Humber. NPVs are positive under any MR scenario examined and for any period of time considered. These results are achieved even when a set of quite conservative assumptions (use values only) are adopted for possible realignment policies in the Blackwater. Evidently, the use of site-specific (as opposed to benefit transfer data) value estimates derived via choice experiment for the amenity and recreation benefit (composite environmental benefit), which dominates over carbon storage[24] and fisheries benefits, has served to reinforce the positive NPV findings of the Humber study. The benefit transfer values used for the Humber gave a general value of wetlands per hectare. Our choice experiment was designed to be site-specific and to calculate more properly the WTP values which differ depending on the distance between the respondents' home and the new inter-tidal habitat created. The aggregated WTP value is then calculated in the policy equations (Equations (4) and (5)) for any size of wetlands as we did for each scenario. A sensitivity analysis applying different discounting approaches was conducted allowing 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 9): the higher the discount rate the lower the value of future costs and benefits. As expected, the gamma discounting suggesting lower discount rates for longer time horizons than the other discounting approaches gives more weight to future costs and benefits than the other discount methods resulting in a higher PV of benefits. Because of its inter-generational fairness bias, we believe that a more pragmatic approach to discounting for public project appraisal seems to be the application of a declining discount rate. On the related matter of inter-temporal consistency, we assume that the decision makers are in the end only human beings in a policy world of continually changing circumstances and inevitable inconsistencies.

It is also worth noticing that the opportunity costs involved at the selected realignment sites were by design relatively low i.e. absence of people and property etc. However, in the future the 'value' of agricultural land may rise substantially as food production requirements escalate. This is likely to increase opportunity costs significantly.

[24] Little sensitivity is found also when the figure for the social cost of carbon used by the Stern Review (Stern, 2007) – £230, which is one of the highest figures found in literature – is applied in the analysis.

Table 10
Comparing net present values (NPVs) for the Humber and the Blackwater estuary studies using conservative values (only use values) and a declining discount rate (HMT); (£ million).

	25yrs		50 yrs		100 yrs	
Scenario	Humber	Blackwater*	Humber	Blackwater*	Humber	Blackwater*
Policy targets (PT)						
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**
Deep green (DG)						
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**
Extended deep green (EDG)						
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**

3.2.1. ESSS4 — non-linearities

A comparison between the results of the Humber case study published in Turner et al. (2007) and the results of the Blackwater case study analysed in this paper is shown in Table 10.

Closer inspection of Tables 9 and 10 reveals a possible anomaly in the NPVs. We would expect to see the benefits growing from the PT to the EDG scenario. However, for example, in the Blackwater it is the DG scenario that has the highest positive value over the 25 years time horizon. Over the other time horizons the difference between the NPV of the two scenarios (DG and EDG) is quite small. 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, although most of 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, among other factors, on the extent of the MR scheme and the specific topography of the region we are looking at.

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. In situations where non-linearities occur, one also cannot make the assumption that marginal benefit values are equally distributed. 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.

3.3. ESSS2 and ESSS5 — marginality and threshold effects

Economics traditionally 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. Since a threshold effect (ESSS5) 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).[25] On the basis of current scientific knowledge no threshold effects were thought to be relevant in this analysis.

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 the attribute coefficients of the choice experiment used in the Blackwater case study, with the discovery of a distance decay effect when respondents were asked to value the creation of new marshes. These results show that, although MR schemes were revealed to be efficient for both the Humber and the Blackwater estuary, this was only the case at carefully selected sites. While it is possible to argue that a number of similar sites exist elsewhere in the coastal zone, it is not appropriate to claim that as a generic rule managed realignment is an economically efficient option. Sites or zones in which significant numbers of people, property, and other assets are potentially at risk present complex social decision making contexts. In such circumstances CBA will provide useful but not necessarily decisive information on trade-offs.

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References

Adams, C., 2008. Carbon burial and greenhouse gas fluxes of new intertidal and saltmarsh sediments. PhD thesis, University of East Anglia, Norwich (UK).

[25] The safe minimum standard represents the minimum level of a well-functioning ecosystem which is capable of producing a sustainable supply of service (Bishop, 1978; Ciriacy-Wantrup, 1952; Crowards, 1998).

Andrews, J.E., Samways, G., Dennis, P.F., Maher, B.A., 2000. Origin, abundance and storage of organic carbon and sulphur in the Holocene Humber Estuary: emphasising human impact on storage changes. In: Shennan, I., Andrews, J. (Eds.), Holocene Land–Ocean Interaction and Environmental Change Around the North Sea. Geological Society Special Publication, vol. 166. Geological Society Publishing House, Bath, UK, pp. 145–170.

Arrow, K., Cline, W.R., Maeler, K.G., Munasinghe, M., Squitieri, R., Stiglitz, J.E., 1996. Intertemporal equity, discounting, and economic efficiency. In: Climate Change 1995: Economic and Social Dimensions – Contribution of Working Group III to the Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, pp. 125–144.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, J., Turner, R.K., 2002. Economic reasons for conserving wild nature. Science 297, 950–953.

Barbier, E.B., Koch, E.W., Silliman, B.R., Hacker, S.D., Wolanski, E., Primavera, J., Granek, E.F., Polasky, S., Aswani, S., Cramer, L.A., Stoms, D.M., Kennedy, C.J., Bael, D., Kappel, C.V., Perillo, G.M.E., Reed, D.J., 2008. Coastal ecosystem-based management with nonlinear ecological functions and values. Science 319 (5861), 321–323.

Bateman, I.J., Day, B.H., Georgiou, S., Lake, I., 2006. The aggregation of environmental benefit values: welfare measures, distance decay and total WTP. Ecological Economics 60 (2), 450–460.

Bishop, R.C., 1978. Endangered species and uncertainty: the economics of a safe minimum standard. American journal of agricultural economics 60, 10–18.

Black and Veatch/Halcrow, 2005. Humber Estuary Flood Defence Strategy: strategy development study. Technical Report, Report to the Environment Agency, Leeds.

Bockstael, N., Freeman, A.M., Kopp, R., Portney, P., Smith, V.K., 2000. On valuing nature. Environmental Science and Technology 34, 1384–1389.

Boyd, J., Banzhaf, S., 2007. What are ecosystem services? Ecological Economics 63 (2–3), 616–626.

Broome, J., 1995. Discounting the future. Philosophy and Public Affairs 20, 128–156.

Ciriacy-Wantrup, S.V., 1952. Resource Conservation: Economics and Policies. University of California Press, Berkeley.

Climaco, J., 1997. In: Multicriteria Analysis. Springer-Verlag, Berlin.

Costanza, R., d'Arge, R., de Groot, R.S., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neil, R.V., Paruelo, J., Raskin, R.G., Sutton, P., van den Belt, M., 1997. The value of the world's ecosystem services and natural capital. Nature 387, 253–260.

Crowards, T.M., 1998. Safe minimum standards: costs and opportunities. Ecological Economics 25, 303–314.

Daily, G.C., 1997. Nature's Services. Island Press, Covelo California.

Daily, G.C., Ellison, K., 2002. The New Economy of Nature: The Quest to Make Conservation Profitable. Island Press, Washington.

Dasgupta, P., 2008. Discounting climate change. Journal of Risk and Uncertainty 37, 141–169.

Defra, 2001. FCDPAG3 flood and coastal defence project appraisal guidance: economic appraisal. http://www.defra.gov.uk/environ/fcd/pubs/pagn/fcdpag3/default.htm.

Defra, 2004. Agricultural land sales and prices in England quarterly. On-line edition (16/06/04).

Defra, 2005. Agricultural land sales and prices in England. On-line edition (18/12/06).

Defra, 2010. http://www.defra.gov.uk/environment/flooding/manage/pathfinder/index.htm

Environment Agency, 2010. http://environment-agency.gov.uk/research/planning/104939.aspx

European Commission, 2000. Directive 2000/60/EC (Water Framework Directive). Official Journal, 22 December 2000.

Fisher, B., Turner, R.K., 2008. Ecosystem services: classification for valuation. Biological conservation 141, 1167–1169.

Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. Ecological Economics 68 (3), 643–653.

Fonseca, L., 2009. Fish utilisation of managed realignment areas and saltmarshes in the Blackwater Estuary, Essex, S. E. England. PhD thesis, Queen Mary University of London.

Forster, P., Ramaswamy, V., Artaxo, P., Berntsen, T., Betts, R., Fahey, D.W., Haywood, J., Lean, J., Lowe, D.C., Myhre, G., Nganga, J., Prinn, R., Raga, G., Schulz, M., Van Dorland, R., 2007. Changes in atmospheric constituents and in radiative forcing. In: Solomon, S., Qin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Frederick, S., Loewenstein, G., O'Donoghue, T., 2002. Time discounting and time preference: a critical review. Journal of Economic Literature XL, 351–401.

Greene, W., 2007. Econometric Analysis. Prentice Hall, Upper Saddle River, N.J.

HMT, 2003. Green Book: Appraisal and Evaluation in Central Government HMSO, London.

Haab, Timothy C., McConnell, Kenneth E, 2002. Valuing Environmental and Natural Resources: The Econometrics of Non-Market Valuation. Edward Elgar, Cheltenham.

Halcrow UK., 2000. Humber estuary tidal defences, urgent works 1—Little Humber to Thorngumbald Clough. Engineer's Report Prepared for the Environment Agency.

Halcrow, Blackwater and Colne estuaries flood defence strategy. Technical Report, Report to the Environment Agency, forthcoming.

Janssen, R., van Herwijnen M., Beinat E., 2001. DEFINITE. Institute for Environmental Studies Vrije Universiteit Amsterdam, The Netherlands.

Kanninen, B., 2000. Theory and Design of Stated Preference Methods. Stated Preference: What Do We Know? Where Do We Go? Doubletree Hotel, Park Terrace, Washington, DC.

Li, H., Berrens, R.P., Bohara, A.K., Jenkins-Smith, H.C., Silva, C.L., Weimer, D.L., 2004. Would developing country commitments affect US households support for a modified Kyoto protocol? Ecological Economics 48, 329–343.

Lorenzen, K., 1996. The relationship between body weight and natural mortality in juvenile and adult fish: a comparison of natural ecosystems and aquaculture. Journal of Fish Biology 49 (4), 627–642.

Luisetti, T., Bateman, I., Turner R. K. Testing the fundamental assumption of choice experiments: are values absolute or relative? in Land Economics, in press.

Mäler, K.G., Aniyar, S., Jansson, Å, 2008. Accounting for ecosystem services as a way to understand the requirements for sustainable development. PNAS 105 (28), 9501–9506.

Millennium Ecosystem Assessment, 2005. Chapter 19 – Coastal Systems. Island Press, Washington, DC. World Resources Institute.

Morrison, M., Bennet, J., Blamey, R., Louviere, J., 2002. Choice modeling and tests of benefit transfer. American Journal of Agricultural Economics 84 (1), 161–170.

Nicholson-Cole, S.A., O'Riordan, T., 2009. Adaptive governance for a changing coastline: science, policy and publics in search of a sustainable future. In: Adger, W.N., Lorenzoni, I., O'Brien, K. (Eds.), Adapting to Climate Change: Thresholds, Values, Governance. Cambridge University Press.

Nordhaus, W., 2007. A review of the Stern review on the economics of climate change. Journal of Economic Literature XLV, 686–702.

North Norfolk District Council, 2010. http://www.northnorfolk.org/.

O'Riordan, T., Nicholson-Cole, S.A., Milligan, J., 2008. Designing sustainable coastal futures. Twenty-First Century Society Vol. 3, 145–157.

Pearce, D., 2003. The social cost of carbon and its policy implications. Oxford Review Economic Policy 19 (3), 1–32.

Pearce, D., Cline, W.R., Achanta, A.N., Fankhauser, S., Pachauri, R.K., Tol, R.S.J., Vellinga, P., 1996. The social cost of climate change: greenhouse damage and the benefits of control. In: Bruce, J.P., Lee, H., Haites, E.F. (Eds.), Climate Change 1995: economic and Social Dimensions – Contirbution of Working Group III to the Second Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, pp. 179–224.

Penning-Rowsell, E.C., Jonhnson, C., Tunstall, S., Tapsell, S., Morris, J., Chatterton, J., Green, C., 2005. The Benefits of Flood and Coastal Risk Management: a Handbook of Assessment Techniques. Middlesex University Press, London.

Postnote, 2009. Coastal Management. Parliamentary Office of Science and Technology, n. 342.

Randall, A., 2002. Benefit – cost considerations should be decisive when there is nothing more important at Stake. In: Bromley, D., Paavola, J. (Eds.), Economies, Ethics and Environmental Policy. Blackwell, Oxford.

Royal Commission on Environmental Pollution (RECP), 2010. 28th Report: Adapting Institutions to Climate Change.

Stern, N., 2007. The Economics of Climate Change: the Stern Review. Cambridge University Press, Cambridge, UK.

Tol, R.S.J., 2005. The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. Energy Policy 33, 2064–2074.

Tol, R.S.J., 2008. The social cost of carbon: trends, Outliers and Catastrophes. Economics-ejournal 2. www.economics-ejournal.org.

Turner, R.K., 2007. Limits to CBA in UK and European environmental policy: retrospects and future prospects. Environment and Resource Economics 37, 253–269.

Turner, R.K., Daily, G.C., 2008. The ecosystem services framework and natural capital conservation. Environmental & Resource Economics 39 (1), 25–35.

Turner, R.K., Paavola, J., Cooper, P., Farber, S., Jessamy, V., Georgiu, S., 2003. Valuing nature: lessons learned and future research directions. Ecological Economics 46, 493–510.

Turner, R.K., Burgess, D., Hadley, D., Coombes, E.G., Jackson, N., 2007. A cost-benefit appraisal of coastal managed realignment policy. Global Environmental Change 17, 397–407.

Turner, R.K., Morse-Jones, S., Fisher, B., 2010. Ecosystem valuation: a sequential decision support system and quality assessment issues. Annals of the New York Academy of Sciences 1185, 79–101. New York Academy of Sciences.

United Kingdom Climate Impacts Programme, 2002. Climate Change Scenarios for the United Kingdom: The UKCIP02 Scientific Report. Tyndall Centre for Climate Change Research, School of Environmental Sciences, University of East Anglia, Norwich, April 2002.

Vincke, P., 1992. Multicriteria Decision-Aid. John Wiley & Sons Ltd, Chichester.

Wallace, K.J., 2007. Classification of ecosystem services: problems and solutions. Biological Conservation 139, 235–246.

Weitzman, M.L., 1998. Why the far distant future should be discounted as its lowest possible rate. Journal of Environmental Economic Management 36, 201–208.

Weitzman, M.L., 2001. Gamma discounting. American Economic Review 91, 260–271.

Woodward, R.T., Wui, Y., 2001. The economic value of wetland services: a meta-analysis. Ecological Economics 37, 257–270.

Yu, P.L., 1985. Multi Criteria Decision Making: concepts, Techniques and Extensions. Pelenum Press, New York.

Zeleny, M., 1982. Multiple Criteria Decision Making. McGraw Hill, New York.