Coastal Zone Ecosystem Services: From science to values and decision making; a case study

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Abstract

This research is concerned with the following environmental research questions: socio-ecological system complexity, especially when valuing ecosystem services; ecosystems stock and services flow sustainability and valuation; the incorporation of scale issues when valuing ecosystem services; and the integration of knowledge from diverse disciplines for governance and decision making. In this case study, we focused on ecosystem services that can be jointly supplied but independently valued in economic terms: healthy climate (via carbon sequestration and storage), food (via fisheries production in nursery grounds), and nature recreation (nature watching and enjoyment). We also explored the issue of ecosystem stock and services flow, and we provide recommendations on how to value stock and flows of ecosystem services via accounting and economic values respectively. We considered broadly comparable estuarine systems located on the English North Sea coast: the Blackwater estuary and the Humber estuary. In the past, these two estuaries have undergone major land-claim. Managed realignment is a policy through which previously claimed intertidal habitats are recreated allowing the enhancement of the ecosystem services provided by saltmarshes. In this context, we investigated ecosystem service values, through biophysical estimates and welfare value estimates. Using an optimistic (extended conservation of coastal ecosystems) and a pessimistic (loss of coastal ecosystems because of, for example, European policy reversal) scenario, we find that context dependency, and hence value transfer possibilities, vary among ecosystem services and benefits. As a result, careful consideration in the use and application of value transfer, both in biophysical estimates and welfare value estimates, is advocated to supply reliable information for policy making.

Keywords: System complexity, Ecosystem services, Stock, Flows, Context dependency, Value transfer

1. Introduction

Although our understanding of the functioning of coastal and marine ecosystems, their reactions to pressures, and their contribution to human well-being is still limited, it is clear that very valuable assets are at risk and require more sustainable and adaptive management. The flexibility focus of adaptive management allows us to adjust to the new knowledge and new drivers and pressures, including climate change. Adaptive management provides a practical framework for implementation of the Ecosystem Approach and ecosystem services concept.

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Ecosystem services valuation is currently faced with several challenges, including the need to better understand: socio-ecological system complexity, especially when valuing ecosystem services; ecosystems stock and services flow sustainability and valuation; the incorporation of the issues of scale and aggregation when valuing ecosystem services; and the integration of knowledge from diverse disciplines for governance and decision making (Valuing Nature Network, 2011).

Here we begin to look for insights into these issues by considering saltmarshes in two estuarine systems located on the English North Sea coast: the Blackwater and the Humber estuaries. We also acknowledge that evidence drawn from a small number of case study sites does not in itself provide robust evidence base. Although these estuaries are close geographically and geomorphically, and agricultural impacts from the catchments are probably broadly comparable (Shepherd et al., 2007), there are marked differences in other characteristics such as the legacy of varied industrial activities within the catchments (Neal and Robson, 2000; Neal and Davies, 2003).

The valuation of ecosystem services can become complex when, within a given ecosystem, interdependent ecosystem services are potentially available and fundamentally interlinked (Fisher et al., 2009). Our case study focuses on ecosystem services that can be jointly supplied by saltmarshes but can be independently valued in economic terms. The biophysical links between the configuration of ecosystem structure and processes that are present at a given time (stock) and the ecosystem services they provide over time (flows) are investigated through an examination of the carbon cycle. One possible way to tackle the problem of aggregation in valuation analysis is through value transfer, a method by which the value estimated for an ecosystem service or environmental benefit in a specific area (the study site) and time is applied to another area (the policy site) and time (Navrud and Ready, 2007). Valid and reliable benefit transfer requires a set of quite stringent conditions (Eftec, 2009); hence we highlight the need for the integration of knowledge about the value of ecosystem services from diverse disciplines from natural sciences (e.g. biogeochemistry, ecology, marine biology) to social sciences and environmental economics for improved decision making.

A set of global and regional socio-economic drivers such as land conversion and reclamation, agricultural runoff, pollution etc. leads to multiple environmental pressures on the coastal areas. Another key pressure on estuarine habitats, and their associated ecosystem services, is sea level rise. Because of sea level rise intertidal habitats are often ‘squeezed’ between the sea and sea defence or coastal protection structures (Doody, 2004).

The Fifth Assessment Report of the United Nations Intergovernmental Panel on Climate Change (IPCC 5) (Church et al., 2013) predicts between 19 and 83 cm of sea level rise by the end of the century, depending on greenhouse gas emission scenario, which concurs with an earlier estimate of 32 cm sea level rise by 2050 (Rignot et al., 2011). These are likely to be conservative estimates as the models are not yet able to accurately predict rapid dynamic changes in the Antarctic ice sheet (Church et al., 2013). If these more extreme predictions prove correct, such a rate of sea-level rise would be unprecedented on the timescale of human settlements. Response strategies and adaptation measures may well require a radical shift away from the ‘hold the line’ hard defences approach. A different flood defence and coastal protection strategy, able to selectively protect ‘high value’ coastal areas/assets while at the same time being able to maintain or expand intertidal habitats may need to be implemented. One option is coastal

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managed realignment (MR) (Elliott et al., 2014; Mazik et al., 2010), which has been implemented with different degrees of success in several estuaries in our study region (see the Online Managed Realignment Guide on-line map - http://www.abpmer.net/omreg/).

The case study includes a scenario analysis to investigate the future evolution of saltmarshes. Coastal and marine ecosystem services and related benefits may either be enhanced or put at risk depending on whether conservation policies such as the EU Directives on Habitats (92/43/EEC) and Birds (2009/147/EC) and the Marine Strategy Framework Directive (2008/56/EC) remain in place and/or are strengthened; or, on the other hand, a European policy dilution trend may become a reality. At the time of writing, extreme weather and flooding took place in Great Britain with devastating consequences (e.g. Somerset). We consider a baseline scenario in which some conservation policies such as managed realignment (MR) have already been put in place, securing a minimum of ecosystem services (business as usual scenario) and two extreme but still plausible future scenarios. These are a scenario in which implementation of conservation policies (e.g. an extended MR) is reinforced allowing maximisation of ecosystem services provision (optimistic scenario), and a scenario in which the implementation of conservation policies is weakened leading to continued degradation of saltmarshes at current estimated rates and to the loss of ecosystem services (pessimistic scenario). To test our ability to integrate biophysical sciences and economics, we explore the complexity of saltmarsh ecosystems through the stock and flows of carbon, and assess the economic value of the estuarine services through the goods/benefits they supply to society ('healthy' climate; food; and nature recreation) (Luisetti et al., 2011a). At the same time we clarify the meaning of 'marginality' (in economics terms) for the stock and the flow of ecosystem services when environmental changes such as those suggested by our optimistic and pessimistic scenarios are involved (Turner et al., 2003). We also explore the extent to which value transfer techniques could be applied.

2. Methodological challenges

In this paper we focus on some of the challenges faced when trying to construct a more robust evidence base for the incorporation of the ecosystem services concept into practical coastal management. We recognise that the provision of 'evidence', the process of policymaking and the delivery of policy are not separate activities. Coastal/marine management is part of a wider system that is composed of hierarchies and networks (including networked publics) and the interrelationships between them (Potts et al., 2014). Our focus is on the building and testing of the evidence base but with due regard for user needs.

2.1. Challenge 1: socio-ecological system complexity when valuing ecosystem services

Complexity in valuing ecosystem services can arise when within a given ecosystem interdependent ecosystem services are potentially available and fundamentally interlinked (Fisher et al., 2009). Coastal and marine system complexity (Fig. 1), following the framework set out by Fisher et al., can be conceptually separated into four components: basic pressures and structure, intermediate ecosystem services, final ecosystem services, and goods/benefits (Potts et al., 2014).

We investigate complexity in valuing ecosystem services by using a case study focusing on marine and coastal ecosystem services that can be jointly supplied but independently valued in economic terms. The economic efficiency of coastal MR as a policy response to the loss of

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intertidal ecosystem services has been analysed in previous studies on the Humber (Turner et al., 2007) and Blackwater (Luisetti et al., 2011a) estuaries. In our case study we first revisit the results of Luisetti et al. (2011a) for the Blackwater estuary study but re-analyse them under different future environmental scenarios. Based on these new biophysical and economic estimates, and using value transfer techniques (Navrud and Ready, 2007), we assess the ecosystem services and benefits provided by the MR sites in the Humber estuary.

The benefits identified and estimated in this study are: 'healthy climate' contributed to via carbon sequestration and storage, which is linked to climate regulation by removing CO2 from the atmosphere); food provision (fish, via fisheries production in nursery grounds), and nature recreation (nature watching and enjoyment via socially valued seascapes). We recognise that in some estuaries the supply of raw material and energy are also important but less so in our case study estuaries. Other benefits provided by the intertidal areas of these estuaries such as, for example, flood defence and erosion control and bioremediation of waste may also be important (see the Adaptation Sub-Committee Progress Report (2013) for a review of the services provided by coastal habitats specifically in the UK).

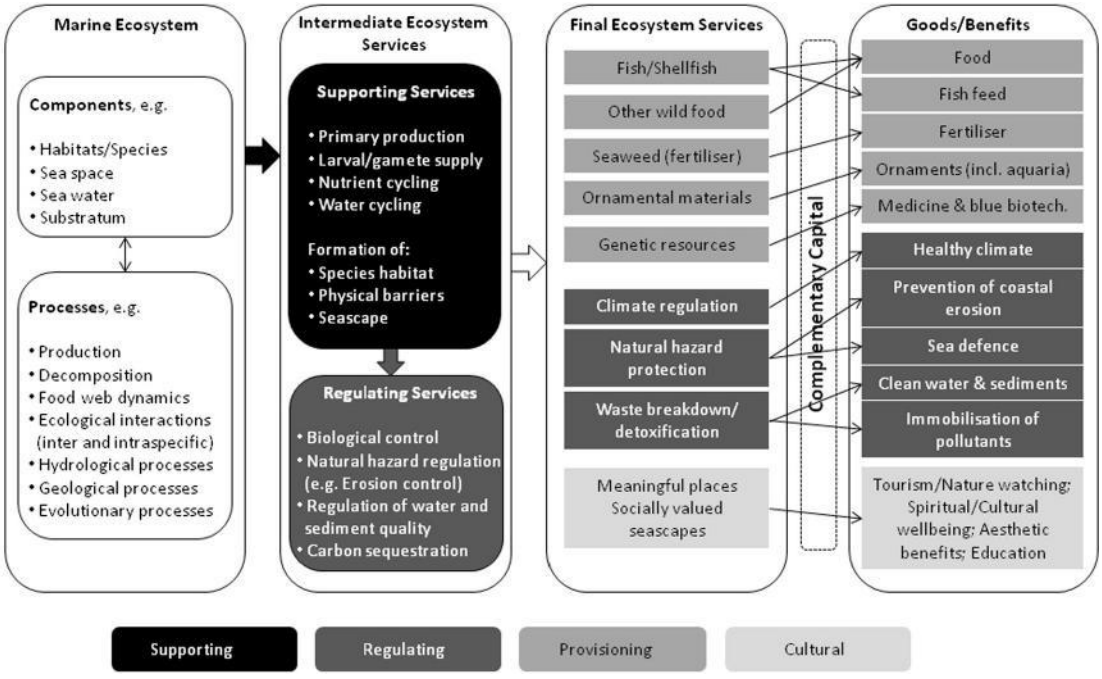

Figure 1: Classification of Coastal and Marine Ecosystem Services (Source: Potts et al., 2014).

2.2. Challenge 2: ecosystem stock and services flow sustainability and valuation

Coastal ecosystem stocks of natural capital (the ecosystem structure and processes and links to the abiotic environment) possess high biological productivity (McLusky and Elliott, 2004) and provide a diverse set of habitats and species, with a consequent flow of ecosystem services (such as carbon sequestration and storage) of significant benefit (value) to society (Fisher et al., 2009). We focus on the carbon sequestration and storage processes in order to help clarify the issues of stocks, flows and values through an interdisciplinary approach. We begin by examining organic carbon cycling, burial, and storage in the temperate estuarine mudflat–saltmarsh system (the

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stock). Coastal zone sedimentary carbon cycling has been reviewed by Andrews et al. (2011) and the role of sediments in sustaining ecosystem services has been reviewed by Apitz (2012) and the reader is referred to these papers for a fuller overview. Organic carbon input and cycling within the estuarine water column is itself complex (e.g. see Boyes and Elliott, 2006) and this complexity is maintained as organic carbon is deposited in estuarine sediments. Organic matter typically arrives in sediments either as coatings on minerogenic particles or as discrete particles (Andrews et al., 2011). The situation differs between mudflats, where organic carbon is almost wholly detrital or due to the microphytobenthos (e.g. diatoms), and saltmarshes, where much organic carbon is directly coupled to plant growth and particularly manifests as root material in sediments. Once deposited in sediment, organic matter is typically subject to a series of aerobic and anaerobic microbial decay mechanisms (e.g. Bianchi, 2007). The most labile organic compounds (e.g. phytoplankton cells) decay at a faster rate than the more refractory ones such as higher plant roots. This early stage of 'burial' in sediments is not simple and is particularly complicated by bioturbation (Gray and Elliott, 2009). However, overall these decay mechanisms can serve as an intermediate ecosystem service; for example the microbiallydriven denitrification reaction uses the dissolved nutrient compound nitrate in seawater to help oxidise organic matter (e.g. see Bianchi, 2007), a reaction that consumes the nitrate, acting as a natural 'waste treatment' process, that helps improve water quality (Seitzinger et al., 2006; Andrews et al., 2006; Turner et al., 2008). However, this reaction also produces a small but significant quantity of the potent greenhouse gas nitrous oxide (Andrews et al., 2006; Middelburg and Levin, 2009; Jickells and Weston, 2011; Adams et al., 2012), which through its role in global warming is a dis-benefit rather than 'a service'.

Even at the level of a single example reaction there are clear environmental positives and negatives that need balancing. Most of the above processes, which we can broadly class as 'cycling', operate on timescales of hours to decades, but at some point, on timescales of decades, the rate of cycling becomes sufficiently slow that we can consider it negligible. Then the residual (or net) organic carbon burial becomes semipermanent and classed as storage (Andrews et al., 2011). This net organic carbon storage (final ecosystem service) can be directly measured (e.g. Andrews et al., 2008; Sousa et al., 2010). Moreover it can be viewed as an ecosystem service benefit in terms of an atmospheric CO2 equivalent that has been removed from the atmosphere, thus reducing greenhouse gas impacts. Storage (a service flow over time) in estuarine mudflat–saltmarsh systems (the stock) operates on timescales of decades (Andrews et al., 2008) and centuries (Allen and Rae, 1987) to millennia (Rees et al., 2000; Andrews et al., 2000).

Clearly the net storage term increases as long as there is sediment available for deposition and the carbon supply to that sediment is maintained; it then follows that the final ecosystem service becomes more valuable, which implies we should protect, and even enable deposition of muddy estuarine accumulation (Turner et al., 2007; Luisetti et al., 2011a). In the UK east coast setting appropriate for this case study, coastal erosion and reworking of offshore sediments provide adequate sediment to allow sediment accretion at managed realignment sites (Shepherd et al., 2007); clearly this may not be the case in other geographic settings. Such deposition cannot occur when man-made defences interfere with sedimentation processes, but MR schemes can augment this mechanism. This is important given that there is potential for a significant decline in the C sequestration service and accompanying benefits due, for example, to sea level change, climatic change and change in sediment supply. Carbon storage and the related climate change mitigation effect can be difficult to assess because the (positive) capture of greenhouse gases

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has to be balanced against the (negative) release of greenhouse gases into the atmosphere during carbon storage.

However, the release of greenhouse gases, particularly methane is lower in saltmarsh areas than in fresh water marsh areas (Andrews et al., 2006). Net C sequestration has an economic value based on the environmental damage avoided by storing rather than emitting each ton of CO2, which may help to meet reduction targets set by the UK Climate Change Act 2008, as detailed in the Low Carbon Transition Plan (DECC, 2009). From an evaluation point of view, we consider the economic value of the flow of benefits provided by intertidal ecosystems and also the current ecosystem stock and its potential supply of benefits. The ecosystem stock accounting price value is estimated, in accounting terms, at a precise point in time (Banzhaf and Boyd, 2012); we review the debate following Costanza et al. (1997) over the valuation of such entities. The paper in the journal Nature by Costanza et al. (1997) estimated the value of global ecosystem services at $33 trillion and led to a protracted debate and controversy over the 'true' value of the natural environment. The value of nature is a multidimensional concept, which includes monetary value but also more qualitative measures. The complete 'commodification' of nature is an ever-present danger to be avoided according to critics of monetary valuation. The position adopted here is that many (but not all) ecosystem services can be meaningfully expressed in monetary terms and that this type of calculus has 'political' purchase that can be used to further conservation efforts in the real world. We do not agree with the argument put forward by some that the ecosystem services approach results in 'cherry picking' and therefore is an unsound basis for biodiversity conservation (Seppelt et al., 2011; Apitz, 2013). Conservation and management at the landscape scale can encompass one or more habitats or ecosystems and therefore a range of ecosystem services. The framework set out by Fisher et al. (2009) and adopted by the UK National Ecosystem Assessment (UKNEA, 2011) distinguishes between final services and the supporting ecological processes and structure. In any given policy trade off context involving development pressures and ecosystem conservation it is often the case that valuing a restricted bundle of final ecosystem services is sufficient to demonstrate an economic case in favour of conservation (Balmford et al., 2002). If this is not the case, it is necessary to bear in mind that the total economic value of an ecosystem is always less than the total system value (Turner et al., 2003) and therefore the economic data presented to policymakers is only one component of the evidence base which can be supplemented by other scientific or ethical/moral evidence.

The Costanza et al. (1997) global ecosystem services estimation has been attacked on a number of grounds including that the aggregate value was not necessarily the sum of the parts, and that US$33 trillion was more than global income and therefore peoples' ability to pay (Heal, 2000). Further work (Howarth and Farber, 2002) sought to defend the Costanza et al., 1997 approach by arguing that the estimates of ecosystem services value were analogous to National Income Accounting entities such as GDP with a constant set of value weights. The underlying rationale here is that the aggregate measure is a quantity parameter (the stock concept), and, while it is related to value, it does not directly value the planet's ecosystem services in total. In this sense it is an accounting price measure of the quantity of ecosystem services holding prices constant, where the measures are not based on economic theory but on accounting rules. In this stock accounting context the criticism related to peoples' budget constraint and ability to pay is not relevant, because the measure is based on virtual (not real)

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prices and virtual incomes (i.e. incomes adjusted to enable individuals to hypothetically pay for the services).

For the income and expenditure accounts to balance, the total expenditure must be less than actual and virtual income. The current extent of European coastal blue carbon (the carbon storage service provided by salt marshes and sea grasses) has, for example, an accounting stock price (value) of about US$180 million, valued against a Social Cost of Carbon estimate (Luisetti et al., 2013). Such total (stock) values can be estimated and compared for two different points in time as a heuristic to help to appreciate the change in natural capital. This viewpoint is, however, controversial and is not supported by many mainstream economists. For them the only relevant measure is the marginal economic value. However, we argue that estimates of aggregate stock accounting price value can play a valuable role in the 'politics' of the environment and can heighten awareness of the overall importance of ecosystem services relative to and in combination with other contributors to human well-being.

For economic valuation it is important to be able to quantify and evaluate gains or losses in stock assets and consequent service flows (analogous to net GDP)[1]. Now instead of holding prices constant, we seek to determine marginal economic value as it relates to an incremental increase in a set of ecosystem services over time and space. When the final ecosystem services value relates only to non-market services, it can be combined with GDP (in the same way as relevant pollution and other externalities are internalised) to yield a more green GDP measure. The present value of a discounted flow of ecosystem services values can contribute to stock of wealth accounts such as the Inclusive Wealth account (UNU-IHDP and UNEP, 2012). An important consideration is that the flow economic values and the stock accounting price values cannot be aggregated. They are complementary in the sense that they provide different dimensions of the magnitude and significance of ecosystem services.

2.3. Challenge 3: the incorporation of the issue of scale when valuing ecosystem services

The issue of scale is investigated because of ecosystem services context dependency, both in terms of biophysical estimates and welfare value estimates. We examine the extent to which context dependency is a binding constraint limiting aggregation. As fieldwork can be very expensive, we investigate the possibility of transferring biophysical estimates and welfare value estimates for the ecosystem services provided by the Blackwater estuary to the Humber estuary. As both estuaries are on the same coastline we assume that they are both geomorphologically and socio-economically similar even if not identical. Here we assume similarity and explore the validity of this assumption.

Transferability of biophysical estimates appears to be broadly valid for regional carbon storage, as Chmura et al. (2003) suggests that saltmarshes in a similar climate will have similar sequestration rates. Moreover, for the case study described here there is empirical evidence that the carbon storage terms for the Southern North Sea tidal fringe are remarkably similar (see Andrews et al., 2011). We are less certain about the variability of greenhouse gas emissions although Sousa et al. (2010, 2012) suggest that while carbon, nitrogen and methane cycling in

[1] GDP reflects the financial (market) value of all final goods and services produced within a country within a certain period. Net Domestic Product (net GDP) is GDP net of the depreciation on capital goods, and thereby reflects how much capital has been consumed over the year.

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intertidal areas are variable in space and time, they do not vary systematically between plant species or even between saltmarshes or mudflats. On this basis we make the reasonable assumption that Adams (2012) net C sequestration estimate in mature managed realignment marsh for the Blackwater is a suitable estimate for the other estuaries located on this same coastline. Clearly there will be some variability in the carbon storage terms, but when the much bigger uncertainty around greenhouse gas emissions is factored in, these become trivial. The transferability of this function on a global scale is much more challenging. As the issue of the sustainability, the store is also complex: at the century scale natural saltmarsh systems are transitory and should migrate geomorphically as climate and sea level change (Chmura, 2011); although in many situations such natural migration is no longer possible due to coastal development.

On the other hand, context dependency was considered to be a major issue for fisheries productivity in saltmarshes because ecological conditions and functions may differ between estuaries. Hence we concluded that while transfer may be possible, in this study the existing biophysical estimates could not be reliably transferred without a more in depth analysis and possibly a targeted data collection (e.g. fish community analysis, age of population, survival rate, etc.) (Burdon et al., 2011).

Value transfer of willingness to pay (WTP) values is accepted practice in environmental economics studies, for example, studies of value function transfer of wetlands are reported (Brander et al., 2011). In our case study we transfer the WTP estimates for the value of the Black-water re-created saltmarshes (the amenity and recreational value) into an optimistic scenario of re-created saltmarsh in the Humber. We also consider a pessimistic scenario of lost saltmarshes because of European policy reversal for both the Blackwater and the Humber. However it is necessary to consider various constraints on value transfer such as, for example, the fact that gains and losses are not valued equivalently by people in some contexts and marginal value functions can be nonlinear. Hence we make a number of assumptions in order to adjust the estimates of Luisetti et al. (2011a). We consider three main issues:

a) The primary valuation study in the Blackwater addressed saltmarsh creation, so the WTP values refer to gains in saltmarsh area. Gains are valued by assessing WTP, but environmental losses are typically valued by assessing public Willingness-to-Accept (WTA) compensation for a loss. The difference between WTP and WTA estimates is a technical and empirical issue and is the subject of continued debate in the valuation literature. In empirical studies, WTA for losses has been found to be larger than WTP for gains, which can be attributed primarily to substitution effects and loss aversion (Pearce, 2002). Loss aversion indicates that people weigh losses more heavily than gains (Kahneman and Tversky, 1984). Theoretically, the divergence between WTP and WTA is minor when there are sufficient substitution options for the good lost (Hanemann, 1999). Economic values for losses (WTA) can therefore be approximated with WTP estimates for gains when substitutes are plentiful, where a WTP value will give a conservative estimate. This is therefore the assumption made in our analysis.

b) The original Blackwater study (Luisetti et al., 2011a) estimated a value function for gains between 10 and 70 hectares. Therefore, the preferences of people for habitat changes beyond this range are less certain. Application of this WTP function to larger

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changes in habitat may be problematic and the area changes in our VNN scenarios are greater than 70 hectares.[2]

c) The original WTP function is additive, i.e. total WTP is a summation of the monetary values of area, distance, access and species recovered. This means that WTP values for distance, access and species are not scaled to the extent of the saltmarsh area. So even for very small increases in saltmarsh area this function will produce relatively high WTP estimates, whilst the added hectares may not actually be able to support species or be of recreational interest. In addition, the effect of area changes is included in the WTP function after applying a log transformation. This generally implies that the additional WTP for an additional hectare decreases as the total area increases, reflecting theoretically expected satiation effects. However, it also suggests that for very small changes (e.g., <5 ha) the WTP per ha for the attribute AREA is unrealistically high, whilst for additional saltmarshes of much more than 70 ha (the upper boundary of the variable), the WTP value is not much higher than for marshes of 70 ha. For example, the WTP of a typical household living 11 miles from a 70 ha new saltmarsh is £12/yr, while for an area of 1084 ha (>15 times larger) the WTP is £15/yr (only 1.25 times larger).

The gain/loss issue is easily addressed because a test showing the respondents two maps with different scales, which included multiple substitutes in the area, was conducted in the Luisetti et al. (2011a) valuation study. This revealed that respondents were aware of the many substitutes close to the Blackwater. Therefore, we used WTA for losses as a proxy for WTP estimates for gains. To address the second and third issues described above, we assume that the function is applicable to areas <10 hectares. This relates to the first 4 years of the pessimistic scenario when the new saltmarsh loss will be <10 ha. For the large gains in area, as well as for losses of >70 ha, we provide two estimates. The first estimate (option A) is based on the assumption that the WTP functions can be used beyond the range of the AREA variable without adjustments. This option provides a lower bound estimate of the saltmarsh values in the optimistic scenario. The second option (B) uses the additional value per household for an increase from 69 to 70 ha new saltmarsh (e.g. £0.0156/yr/ha). This value is then applied to any additional ha of saltmarsh beyond >70 ha, assuming a linear function from that point onwards. Under this assumption, the WTP of a typical household living at 11 miles from a 1084 ha new saltmarsh is now £28/yr.[3]

[2] Note that in the aggregation procedure we also have an issue with the range of the at¬tribute distance used in the original study. Therefore, the aggregated WTP values do not ac¬count for potential positive WTP/WTA of people beyond 32 km from the newly created saltmarshes.

[3] Option C would be to split the new/lost areas into sections of 70 ha, apply the equa¬tions to each section, and aggregate those values. This would lead to much higher esti¬mates than option B and is therefore not included in this paper

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3. Case study analysis

3.1. Case study areas: the Humber and the Blackwater estuaries (English North Sea coast)

The Humber estuary is a relatively large and complex estuary with two major river systems (the Rivers Trent and Ouse) meeting within the estuary and draining more than 20% of England. The catchment includes areas of intensive agriculture and upland areas used for low intensity sheep grazing. This region has a history of mining and industrial activity dating back to Roman times (Neal and Robson, 2000; Neal and Davies, 2003). Although most of the estuarine industrial activities are now closed or operating on a reduced scale with less environmental impact, a significant estuarine contamination legacy remains (GarcíaAlonso et al., 2011). The Humber continues to be an important commercial and population centre (Cave et al., 2003). In the past, the estuary has been subject to major land claim of intertidal areas with a loss of >90% of its intertidal area, which has greatly reduced the capacity for the storage and processing of carbon, nutrients and contaminants (Jickells et al., 2000; Andrews et al., 2006; McLusky and Elliott, 2004). According to the Humber Estuary Shoreline Management Plan, given the estimated loss of saltmarshes and mudflats due to future sea level rise, the recreation of intertidal areas is highlighted as a priority (Edwards and Winn, 2006). MR schemes are currently in progress in the estuary and an extension of these is planned at a number of sites (Mazik et al., 2010).

The Blackwater has also been subject to land claim of intertidal areas, the re-creation of which is also considered a priority because of the risks related to sea level rise. The Blackwater is located in the Essex coastal region, which has relatively small lowland rivers with small catchments and low flows. The region is largely semi-rural with a catchment dominated by intensive agriculture with small amounts of industrial legacy contaminants. Nitrate concentrations from agriculture are however high (Shepherd et al., 2007).

It is possible to argue that in the context of the next 50–100 years deleterious system change is likely to result in both the Humber and the Blackwater estuary. This will be driven by a combination of rapid sea-level rise, coupled with climatic change resulting in increased winter rainfall, increased incidence of extreme events and increased storminess of the North Sea. Rapid sea-level rise is expected to cause erosion of mudflats and sediments due to substantial changes in the symmetry of the tidal prism compounded by a lack of sediment supply. Under natural conditions the estuary would widen with sea level rise (Allen and Rae, 1987), but the presence of sea walls in most of the modern estuary will result, under a ‘business as usual’ scenario, in rapidly deepening water with no intertidal space for sedimentation (Andrews et al., 2000). Under these conditions, even without erosion, the present intertidal area will be drowned and its ecosystem service provision potentially reduced. If, however, the present rate of global sea-level change (3.1 ± 0.4 mm yr^{-1}) (Nerem et al., 2010) increases, it is arguable that coastal protection will not be able to operate under ‘business as usual’ conditions and estuaries will need to ‘make space for water’ (Defra, 2004, 2005), but also for sediment to help rebuild mudflat and saltmarsh and to restart ecosystem service provision. It is also of note that isostatic adjustment of the English east coast will increase the relative sea level rise (Ducrotoy and Elliott, 2008). The case study area is presented in Fig. 2.

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3.2. Scenario analysis

Two scenarios and a baseline are considered, over the period 2010– 2110, and are related to previous scenario studies (Langmead et al., 2007; Turner et al., 2007; Luisetti et al., 2011a). These are:

- Baseline/minimum ecosystem services (MinES): economic growth is combined with environmental protection, with realignment undertaken to reduce flood defence expenditure and compensate for past and future intertidal habitat loss in compliance with the Habitats Directive.

- Optimistic/maximum ecosystem services (MaxES): primary objective is habitat creation.

- Pessimistic/loss of ecosystem services (LESS): the existing defences are maintained to a satisfactory standard, wherever possible, but intertidal habitat will be lost due to continued development and coastal squeeze.

Calculations for the stock and the flow estimates of ecosystem services provided by the estuaries are based on the extent of saltmarshes estimated for each scenario. As reported in Table 1, the extent of the areas of saltmarsh for both estuaries in the baseline and optimistic scenario comes from previous studies (Luisetti et al., 2011a; Turner et al., 2007; Andrews et al., 2000). Evidence from the literature suggests that the colonization of managed realignment sites may take place relatively quickly with large numbers of plants establishing, especially in areas with rapid accretion such as the Humber estuary (Boyes & Allen, 2007), however with respect to community development this may take decades to develop (Mazik et al., 2007; Mossman et al., 2012). Dagley (1995) observed that on the Blackwater the colonization rate and order of establishment at the Brancaster and Northey Island MR sites was similar, the second with 23 plant species established in 3 years. Mossman et al. (2012) found evidence of convergence between the vegetation communities in the MR site and those of reference marshes after 5 years. This was also shown by Wolters et al. (2008) whose state that it took approximately 5 years for species diversity in the Tollesbury restoration site to become similar to a local reference marsh. In the United states, Morgan and Short (2002) for marshes in the Great Bay Estuary suggested that species richness could be within 95% confidence limits of reference marshes within 7 years, reaching their mean value after 12 years. Based on the evidence provided by the literature, especially for the Blackwater and Humber estuary we assume that full provision of ecosystem services will be effective after 5 year from realignment in the Blackwater and 15 years in the Humber estuary and that this will be maintained until 2110.

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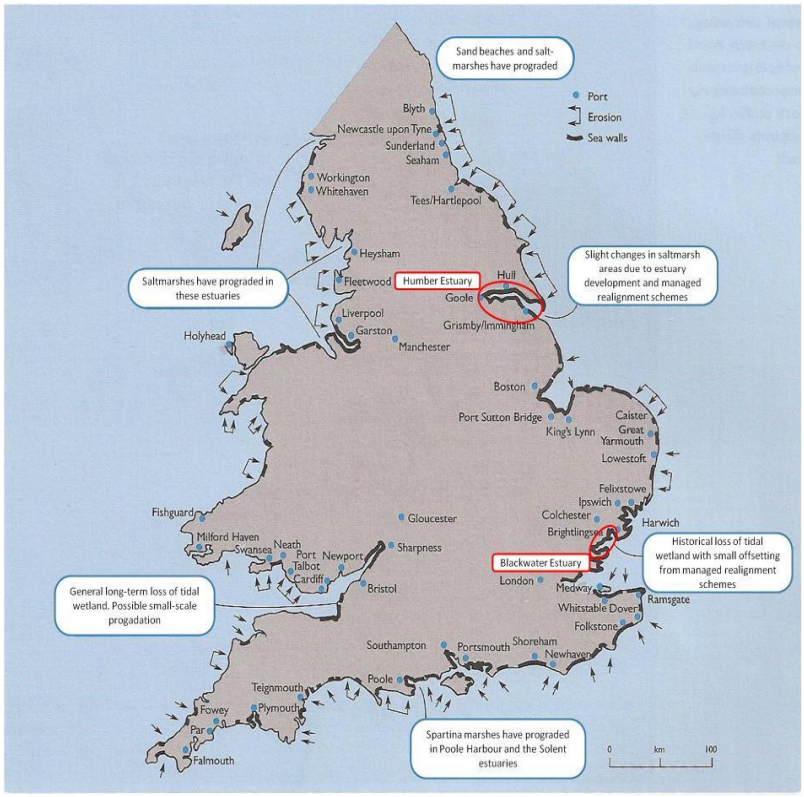

Figure 2: Coastline changes in England and Wales where arrows show zones of erosion and the heavy lines show areas of accretion. The study sites discussed in this paper are highlighted in red. Modified from Figure 9.2 in O'Riordan et al. (2000).

Table 1: Area extent (ha) per scenario in the Blackwater and the Humber estuaries.

Scenario/Estuaries	Blackwater[1]	Humber[2]
Baseline/MinES	1,084	11,100
Optimistic/MaxES	+2,404	+176
Pessimistic/LESS	−244	−2,500

Notes: 1. Luisetti et al. (2011a); 2. Turner et al. (2007) and Andrews et al. (2000). The figures for the pessimistic scenarios have been calculated in this study.

Table 1 also shows the potential future loss of saltmarsh under the pessimistic scenario. In this scenario, implementation of the EU Habitats (92/43/EEC) and Birds (2009/147/EC) Directives allowing for re-creating of saltmarshes will become limited, and intertidal habitats are squeezed between the defences and the rising sea level. This scenario follows the predictions for saltmarsh loss because of sea level rise reported in Jones (2011): 4.5% of saltmarshes lost over the last twenty years (i.e. 1990–2010), although losses are projected to reach 8% by 2060. A decrease of 4.5% for 20 years is equal to a saltmarsh loss of 0.225% per annum. The uncertainty around future climate change is high. The future may present us with smooth as well as abrupt changes in the loss of our coastlines. Although we acknowledge that other evidence may suggest a different percentage loss of saltmarshes for the UK coastline, for the illustrative

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purposes of this study we assume that the loss of saltmarshes will continue at the fixed pace of 0.225% per annum calculated on the baseline in 2010 till 2110 (i.e. 2.44 ha/year loss in the Blackwater, and 25 ha/year loss in the Humber estuaries). For the Blackwater this is equal to a loss of 244 ha after 100 years and for the Humber to 2,500 ha after 100 years.

A more likely scenario would be a mix of realignment and continued loss of saltmarshes. What we show here is an example of what we could gain or lose in plausible but extreme circumstances.

It is assumed that, subject to appropriate sustainable management, the flow of many ecosystem service benefits provided by saltmarshes (e.g. 'healthy' climate (carbon sequestration and storage); food (fish), and nature recreation) can be enjoyed over time into the future. As a consequence, future flows of services value have to be discounted to give their present value. The concept of sustainability aims to guarantee to future generations at least the same level of environmental resources enjoyed by the present generation. However, the future is by definition uncertain. Whether and how future technology will be able to help future generation is uncertain. For this reason it has been observed (Frederick et al., 2002) that individuals prefer rather to have something today than in the future. This is known as individual time preference. If the belief of the policy maker is that future generations will be better-off, then a relatively high discount rate will be applied to estimate the efficiency of a project/policy. On the contrary if the belief is that future generations will be worse-off, then a relatively low discount rate will be applied. Some authors even advocate a zero discount rate (Broome, 1995). Current British policy appraisal (Treasury Green Book) suggests the use of a declining discount rate — the further away in the future the benefits manifest themselves, the lower the discount rate. For this reason, in this application, we use the Green Book suggested declining discount rates schedule. In our case study, the present value of the services flow for a time horizon of 100 years in each scenario is calculated applying the declining discount rate scheme as recommended by the UK Government (HMT, 2011).

3.3. 'Healthy' climate valuation

The latest estimates of net carbon storage per year (Adams et al., 2012) for the Blackwater estuary, amended to take account of greenhouse gases production and global warming potential of CH4 and N2O, have been used in this study (Table 2). Following atmospheric science convention, emissions are reported as positive and sequestration as negative values. Table 2 illustrates that while the N2O issue is important, the overall saltmarsh development represents a net sink for greenhouse gases, i.e. there is a net climate change mitigation benefit. A first estimate of net carbon burial (−1.15) is based on mature MR marsh sampling in the Blackwater. The other estimate (– 0.94) shows the C sequestration potential in natural saltmarshes (e.g. Atriplex portuacoides, Limonium vulgare, Armeria maritima, Triglochin maritimum). We use the first estimate for the MR scenario analysis (MaxES), and the second for the calculation of the current stock of the C sequestration service and for the pessimistic scenario (LESS). Table 2 is based on the estimate of a 5.4 mm sedimentation rate, which is an acceptable estimate for long term surface elevation change, given a regional sea-level rise of ~6 mm per year in the Blackwater estuary (NRA, 1994) and assuming that the sampling area is currently in equilibrium with sea-level rise and remains so.

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Table 2: Estimated net carbon burial per year in the Blackwater estuary.

	Gross C burial tC/ha/yr	CH_4 tC/ha/yr	N_2O tC/ha/yr	Net C burial[1] (C sequestration) tC/ha/yr
Mature managed realignment marsh	−1.19	0.012	0.028	−1.15
Natural Saltmarsh	−1.27	0.008	0.315	−0.94

Note: 1. Figures are amended to take account of greenhouse gases production (i.e. net figures). 5.4 mm assumed sedimentation rate.

Table 3 summarises the estimated flow changes per year in carbon storage capacity (based on the estimates in Table 2) for the Blackwater and Humber estuaries under two different scenarios: optimistic/MaxES (increase) and pessimistic/LESS (decrease). In the pessimistic scenario, it is assumed that losses of intertidal habitat occur. In this scenario, we assume that intertidal mud and sand with saltmarsh is lost and replaced by subtidal sediment. We assume this subtidal sediment produces GHGs, although the flux is not yet constrained. To try and account for this, as shown in Table 3, we assume that the actual C storage capacity lost will be equal to 90% of the net annual C storage capacity shown in Table 2.

It was assumed that carbon sequestration is linearly related to saltmarsh area since, as discussed above, we assume that the carbon storage rate per gram of sediment is rather uniform in these habitats. Table 4 shows the value of the C sequestration potential for the intertidal habitats that could be gained with the optimistic scenario or lost with the pessimistic one in the Blackwater and Humber estuary. For the purpose of sensitivity analysis, the changes of carbon storage service flows in the two scenarios are valued by two different methods: the social cost of carbon (SCC), which measures the monetary value of the avoided carbon releases to the atmosphere because of storage (damage cost avoided method), and the Department of Energy and Climate Change (DECC) prices for non-traded carbon (DECC, 2011), for which figures are calculated using the marginal abatement cost (clean-up cost) method. As in Luisetti et al. (2011a) three different SCC prices were considered: a lower value of £7/tC (Arrow et al., 1996; Pearce et al., 1996; Li et al., 2004); a middle value of £30/tC (Pearce et al., 1996; Tol, 2005); and an upper value of £230/tC (Stern, 2007; Nordhaus, 2007). DECC (2011) provides non-traded C prices till 2100. For the illustrative purposes of this study, we assume that the trend showed in the relevant year values between 2096 and 2100 will continue till 2110. Also, since DECC prices are defined for units of tCO2, the net carbon (C) sequestration values of Table 2 were converted[44] to net carbon dioxide (CO2) values in Table 3 and used for the calculation of the present value results presented in Table 4.

[4] The mass of C is first divided by 12.01 to bring everything to a base value, and the re¬sult is then multiplied by the mass of CO2 (44.01).

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Table 3: Estimated change over the time period 2010–2110 in C sequestration potential for the Blackwater and Humber estuaries under two different scenarios: optimistic/MaxES and pessimistic/ LESS.

	Intertidal Area (ha)	Net C sequestration (tC/ha/year)	Total C storage in tC/yr	Total C storage in tCO_2/yr
Blackwater				
Optimistic/MaxES Scenario	+2,404	−1.15	+2,765	+10,132
Pessimistic/LESS Scenario	−2.44/yr	−0.846	−2.06	−7.55
(loss of 90% of C storage capacity)		(90% of −0.94)		
Humber				
Optimistic/MaxES Scenario	+176	−1.15	+202	+743
Pessimistic/LESS Scenario	−25/yr	−0.846	−21.15	−77.5
(loss of 90% of C storage capacity)		(90% of −0.94)		

Following Howarth and Farber (2002), we use the 'market' price to calculate the value of the stock of saltmarshes in terms of C sequestration and storage in the Blackwater and Humber estuaries, measuring the aggregate service provision at a specific point in time: i.e. 2010. To calculate the stock of carbon storage service we have used the mean price of traded carbon (a proxy for the current market price of CO2). The average EU Allowance spot (i.e. market) price in 2010 was € 15/tCO2 (Rickels et al., 2010; Chevallier, 2010). At those prices, the accounting value of the saltmarsh stock in terms of C sequestration (natural saltmarshes) in the Blackwater and Humber estuaries in 2010 is reported in Table 5.

3.4. Amenity and recreation

In the original Blackwater estuary case study (Luisetti et al., 2011a), a specific stated preferences (survey based) technique to estimate the amenity and recreation benefit potential of re-created intertidal habitats after MR in the estuary was undertaken. Based on the ecosystem services framework, the benefits of expected water quality improvements and the non-use value of biodiversity maintenance (including fisheries output not sold in the market) were assumed to be captured by the value estimated for the amenity and recreational value of the newly created intertidal habitats. The valuation procedure is service and location specific. In another context water quality may be a final service and should be independently valued. The survey asked the respondents to answer a choice experiment (CE) exercise. In a CE, respondents are asked to choose between alternative goods or policies that are described according to several characteristics (attributes) with varying levels. In Luisetti et al. (2011a), the attributes considered were: the extent of the salt marsh area re-created (AREA), the number of protected bird species observable (BIRD), the distance from respondent's home to the new recreational site (DIST), whether there is public access to the new area (ACCESS), and the hypothetical annual cost expressed by an increment of the local council tax paid by each respondent to implement MR schemes and re-create saltmarshes (TAX). The attribute ACCESS reveals how much people value the possibility of 'using' the newly created intertidal habitat. Use values are related to the actual or possible use of the good in question. Non-use values are the values that individuals assign to some good because its existence is thought to be important even though there is no actual or possible use (Turner et al., 2003). Here, we use the variable BIRD as a proxy for non-use (e.g. biodiversity) value.

From the stated choices between alternative policy-scenarios, the WTP of respondents for the attributes can be estimated. The resulting WTP/household/year from the study by Luisetti et al. (2011a) is presented in Eq. (1):

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$$WTP = 1.087 * Ln(\Delta AREA) + 3.522 * 1\ (BIRD\ 5) - 0.052 * DIST + 4.217 * 1\ (ACCESS) \quad (1)$$

The variable AREA is logtransformed (reflecting diminishing values per additional ha) and BIRD5 is a dummy variable representing the highest level of interest by respondents in non-use values.

We use this WTP function to consider two different policy scenarios: optimistic/MaxES and pessimistic/LESS for saltmarshes in the Blackwater estuary. We present two WTP estimates for these scenarios: one based on use and non-use values of the saltmarshes, and a conservative estimate based predominantly on use values and in which the WTP related to the variable BIRD is excluded.

The variable DIST provides a key determinant of the aggregated WTP. So called 'distance–decay effects' reflect that the respondent WTP diminishes with increasing distance between the good (related to the final ecosystem service) of the investigation and the respondent residence. Significant distance decay values were revealed in the econometric analysis of the CE in Luisetti et al. (2011a). We account for this distance-decay effect in the aggregation of WTP estimates per households over the relevant population.

We based the aggregation on the number of households in urban centres in a range of 32 miles from the MR site Abbott's Hall, an internationally recognised and locally well-known successful MR example in the Blackwater estuary[5]. The number of households per urban centre was based on the 2001 Census Statistics (Office for National Statistics, 2004)[6]. This provides a representative, yet lower-bound estimated of aggregate WTP for the area, since not all households within 32 miles are included. Distance from each urban centre to the MR site was calculated using Google Maps. We assume that people can only reach the new saltmarshes by car.

Aggregate WTP is calculated by multiplying the number of households in a centre by the mean household WTP of that centre for the policy under consideration, taking the travel distance between the centre and the MR site into account, and summed over all relevant centres. The estimated present values (PV) of the recreation benefit flows over a time horizon of 100 years are reported in Table 6. For the positive scenario, the PV are estimated based on the assumption that benefits only start to accrue after 5 years in the Blackwater estuary MR site. Values of this scenario range from GBP 78 million (use values only, option A) to GBP 328 million (use and non-use values, option B). The (negative) benefit flows of the pessimistic/LESS scenario increase each year as more saltmarsh is lost and vary between GBP 43 million (option B, use values) to GBP 86 million (option B, use and non-use values).

[5] Luisetti et al. (2011b) show that the distance-decay effect depends on the range of DIST attribute show to respondents in the CE survey. Respondents of a subsample for which the option included a MR site nearby (2–32) showed significantly different distance-decay values than respondents in a 'distant' subsample.

[6] Adapted from data from the Office for National Statistics licensed under the Open Government Licence v.1.0 (Table KS20 Household composition).

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Table 4: Present value of carbon sequestration flows for the Blackwater and Humber estuaries over 100 years (2010–2110) discounted following the declining discount rate scheme of the UK Green Book (HMT, 2011) under two different scenarios: optimistic/MaxES; pessimistic/LESS (£*1000).* DECC (2013) provides non-traded C prices till 2100. Here, we assume that the trend showed in the relevant year values between 2096–2100 will continue till 2110.

	SCC/tC			DECC* Price/tCO_2
	£ 7	£ 30	£ 230	All relevant year values (£)
Blackwater				
MaxES Scenario	527	2,257	36,670	45,500
LESS Scenario	0.5	2	15	36
Humber				
MaxES Scenario	28	123	941	3,000
LESS Scenario	3	15	117	325

In option A, the WTP function is used beyond the range of the AREA variable without adjustments, which provides a lower bound estimate of the WTP per area increase. In Option B, we apply the additional value per household for an increase from 69 to 70 ha new saltmarsh (e.g. £0.0156/yr/ha for any additional ha of saltmarsh beyond >70 ha). Option B results in considerably higher benefit flows than option A for the optimistic/MaxES scenario, whilst for the pessimistic/LESS scenario the differences are fairly small, partly because the total saltmarsh loss in 2110 is 244 ha.

Here the economic value of the amenity and recreation service estimated for the Blackwater estuary is transferred to the Humber estuary making use of a benefit transfer value technique. In previous studies of the Humber estuary (Turner et al., 2007), the value of recreation and amenity was based on a meta-analysis of wetlands areas around the world. We argue that using a UK based study is likely to provide more accurate estimates because the socio-economic conditions and ecosystem services are likely to be more similar. For this benefit transfer exercise, we use the coefficients in Eq. (1), and for each variable the data of the Humber estuary. Although the area around the Humber is more populated than the area surrounding the Blackwater estuary (i.e. Essex county), socio-economic characteristics appear to be similar (Office for National Statistics, 2011). Again, we choose some urban centres in the Humber estuary surrounding the MR site of interest, assume that people can only reach the new saltmarshes by car, and calculate distances between the centre and the MR site.

Table 7 presents the estimated present values for the optimistic/ MaxES and pessimistic/LESS scenarios for the MR site in the Humber estuary. For the Humber study, the recreation and amenity benefits in the optimistic scenario are assumed to start after 15 years after the establishment of the MR site (176 ha). In the pessimistic/LESS scenario, the total area lost in 2110 is 2,500 ha, immediately leading to losses in recreation and amenity values.

The estimated benefits of the optimistic/MaxES scenario range from GBP 33 million (option A, use values) to GBP 48 million (option B, use and non-use values). The difference between the estimated benefits of the optimistic/MaxES scenario and the losses under the pessimistic/ LESS

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scenario are larger using the calculation method of option B than under option A. This is because the diminishing marginal value assumed for any additional ha under option A, which implies that the much larger area lost in the pessimistic/LESS scenario (compared with the optimistic scenario), does not result in a proportional decrease in monetary value.

Table 5: Accounting C stock value in the Blackwater and Humber estuaries in 2010.

Estuaries	Area extent in the baseline (ha)	Net C tCO_2/year in natural saltmarshes	Total stock net C sequestration potential/year	Accounting value of the stock in 2010 (€*1,000) at € 15/tCO_2	Accounting value of the stock in 2010 (£*1,000) Conversion rate: 1€ = £ 0.858347
Blackwater	1,084	−0.94	3,740	56	48
Humber	11,100	−0.94	38,295	581	499

Table 6: Present value of the aggregated willingness to pay (WTP) flows for recreation in the Blackwater saltmarshes and over a time horizon of 100 years (2010–2110) discounted following the declining discount rate scheme of the UK Green Book (HMT, 2011) under the two different scenarios (£ *1000).

Blackwater		Option A	Option B
MaxES Scenario	*WTP use and non-use values*	+102,972	+327,780
	WTP mainly use values	+78,728	+303,556
LESS Scenario	*WTP use and non-use values*	−84,395	−85,682
	WTP mainly use values	−56,594	−42,501

Note: In option A, the WTP function is used beyond the range of the AREA variable without adjustments. Option (B) applies the additional value per household for an increase from 69 to 70 ha new saltmarsh (e.g. £0.0156/yr/ha for any additional ha of saltmarsh beyond >70 ha.

Table 7: Present value of the aggregated willingness to pay (WTP) flows for recreation in the Humber saltmarshes under the different scenarios and over a time horizon of 100 years (2010–2110) discounted following the declining discount rate scheme of the UK Green Book (HMT, 2011) (£ *1000).

Humber		Option A	Option B
MaxES Scenario	*WTP use and non-use values*	45,901	48,268
	WTP mainly use values	33,187	35,554
LESS Scenario	*WTP use and non-use values*	−76,874	−126,611
	WTP mainly use values	−57,281	−106,281

Note: In option A, the WTP function is used beyond the range of the AREA variable without adjustments. Option (B) applies the additional value per household for an increase from 69 to 70 ha new saltmarsh (e.g. £0.0156/yr/ha for any additional ha of saltmarsh beyond >70 ha.

4. Conclusions

One aim of this study was to test the possibility of providing ecosystem services benefit estimates within the same country, or at least the same region, for subsequent use in cost-benefit analysis (CBA). For this purpose, the study builds on the previous CBA results of Turner et al. (2007) and

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Luisetti et al. (2011a) that evaluated managed realignment (MR) schemes for the estuaries located on the East coast of England (the Humber estuary and the Blackwater estuary respectively). The schemes were appraised over short (25 years) and mid/long (50/100 years) time horizons and against three scenarios of future sea level rise and more or less extensive MR. The advantages of a "low regret" policy such as MR were apparent under any of the three scenarios for the Blackwater estuary when a time horizon of 50/100 years was considered. Tradeoffs such as lost agricultural land because of realignment were incorporated in the analysis. In Luisetti et al. (2011a) an anomaly was found when the most ecological scenario and a short time period such as 25 years were taken into account — the short term realignment costs (breaching existing walls and building secondary lines of defense) where much higher than the short run benefits. However, it is also the case that benefits (co-benefits) other than flood defense may be delivered if MR schemes are implemented. Other findings of this study relate to the spatial scale of biophysical and welfare value estimates and their transferability across areas and different policy contexts. Among the ecosystem services and benefits considered in this analysis (see Table 8), the C sequestration and storage service, underpinning the 'healthy' climate societal benefit, was found to be particularly suited to cross-disciplinary analysis.

Table 8: Case study expert judgment on reliability of function transfer results.

Ecosystem goods/benefits	BIOPHISICAL ESTIMATE OF FINAL ECOSYSTEM SERVICE	ECONOMIC VALUE	TRANSFERABILITY	METHODS
'Healthy' climate	+++	++	++	- Assumptions about biophysical data transferability supported by real data findings; - Use of SCC and DECC prices.
Food	−−−	+++	+/−	Further investigation needed as currently biophysical data are lacking and different methodologies may be applied.
Recreation	NA	++	+/−	- Assumptions to overcome methodological limitations encountered in the original case study source of the main data (WTP); - Assumptions about socio-economic characteristic similarities for transferability of WTP values supported by real data.

NA: not available; +++ max transferability/max availability of data; −−− not transferable/no data available; +/− transferability is be possible under specific conditions.

Within the case study, for this ecosystem service, biophysical estimates are transferable without any adjustment and the accounting and economic value for carbon sequestration can be estimated using carbon prices and global social cost of carbon estimates respectively. The same argument may not be applicable at a bigger scale (e.g. European level) where the differences in the climate zones may result in more significant differences between biophysical estimates as well as socio-economic characteristics. The transferability of the amenity and recreational benefit in both scenarios was possible but subject to several assumptions. On a more extensive scale, those assumptions may increase in number and complexity mitigating against an extensive use of the value transfer practice. Food (fish, via fisheries production in nursery grounds) transferability may also be possible. However transferability of fisheries production function is not straightforward. Fish production estimates within managed realignment sites in the Humber estuary at the time of the analysis were less-advanced than for the Blackwater estuary (Fonseca, 2009; Luisetti et al., 2011a), therefore it was not possible to carry out a similar valuation for food as in Luisetti et al. (2011a), nor to investigate transferability options for the Humber due to lack of data (Burdon et al., 2011). Future research should test the appropriateness of using the same production function estimated for the Blackwater and applying the data collected for the Humber to this production function. It would also be valuable to estimate a new specific production function for the Humber using Humber data; and transfer the estimates of the Blackwater estuary to the Humber. Therefore, we can conclude that, although subject to the well-known methodological limitations affecting the benefit transfer technique, depending on the similarity between the biophysical conditions of an ecosystem service in the case study and the policy site

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and the data availability of that service, transferability of ecosystem benefits may be possible at the same country level. However, careful examination of the similarities of biophysical conditions and socio-economic characteristics between sites, as well as the technical transferability possibilities of production and socioeconomic functions, are essential before undertaking any benefit transfer. Outside the boundaries of the same country, biophysical conditions and socio-economic characteristics may differ significantly from the case study site limiting the meaningful application of value transfer in the policy site when ecosystem services, which are context dependent by definition, are concerned.

There are many more ecosystem services and benefits which arise from estuarine habitats in addition to the three detailed here. Two significant ecosystem services are flood defence and erosion control and contaminant and nutrient cycling, future research should seek to include these and other services as sustainable management will require a holistic inclusive approach to ecosystem services. The recently completed TIDE project has taken further the estimation of these ecosystem services (see http://www.tide-toolbox.eu/reports/). It can be argued that informed decision making requires integration between disciplines helping to tackle societal and environmental change issues from a pluralistic perspective and leading to policy relevant tools and means to adapt the governance structure for better future coastal management. To achieve this aim a holistic and inclusive approach to ecosystem management is necessary and the ecosystem approach can play a key role in this. Issues key to the successful valuation of ecosystem services and benefits, and addressed in this study, are:

- A recognition of socio-ecological system complexity when valuing ecosystem services, dealt with in this study in biophysical and economic terms through the application of an ecosystem approach and a related ecosystem services valuation framework for the assessment and valuation of the goods/benefits provided by the marine ecosystem and ecosystem services;
- The need to distinguish between ecosystem stock and service flow sustainability and valuation, investigated in this study through the carbon cycle in saltmarshes, with recommendations on how to value stocks and flows of ecosystem services via accounting and economic values respectively; and
- An acknowledgement of the difficulties posed by the issues of scale and transferability when valuing ecosystem services, and their incorporation in policy appraisal methods such as CBA, investigated using a case study highlighting ecosystem services context dependency and potential transfer of ecosystem services values, both in terms of biophysical estimates and welfare value estimates. Another sister VNN project “Uncertainty and Scale” reviewed a selection of ecosystem services in order to identify common themes related to scale and uncertainty.

Results from this study suggest that because of socio-ecological system complexity, careful consideration is required in the use and application of benefit transfer to ecosystem service values, both in terms of biophysical data and welfare value estimates to supply reliable information for policy making.

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Conflict of interest

The authors of this manuscript do not have any actual or potential conflict of interest to disclose to Science of the Total Environment.

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