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Valuing the regulatory services provided by marine ecosystems[†]

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It is widely accepted that mankind derives benefits from ecosystem services provided by the marine environment. It is less clear how these benefits can be quantified in order to make objective and responsible environmental management decisions. Providing an economic quantification of these benefits is one approach that can help. While it is relatively simple to derive monetary values for benefits accruing from activities with an economic basis, such as food provision and tourism, different approaches must be taken to provide economic values for services with less obvious links to economic activity. Here we demonstrate how a range of approaches may be used to derive economic values for three very different ecosystem services provided by marine environments, namely the bioremediation of waste by marine biota, the fixation of carbon dioxide by photosynthetic marine organisms, and the sea defence role provided by wetlands in coastal areas of the UK. Copyright © 2010 John Wiley & Sons, Ltd.

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1. INTRODUCTION

Ecosystem functions and the benefits humans derive from them have become a central topic of research at the interface of social and natural systems (Costanza *et al*., 1997; MEA, 2005). This follows a growing recognition of the need to evaluate ecological functions and the value of the ecosystem services they provide so that they are not overlooked when management decisions are made. Without monetary valuation less apparent ecosystem services, such as nutrient cycling, can be overlooked or considered to be 'free' and therefore not considered within management strategies. This could lead to over-exploitation and degradation of the environment. Economic valuation of ecosystem services is also important for benefit–cost analysis, which is a cornerstone in the evaluation of any policy design. Benefit–cost analysis necessitates that all costs and benefits of a development or intervention are captured in monetary terms. Ecosystem service valuation studies estimate the values of not only the marketed ecosystem services, but also the non-marketed ones allowing easy comparison with all other costs and benefits. By interpreting the value of ecosystem services in monetary terms, a convincing case can be put to decision-makers to persuade them to incur the opportunity cost of their protection.

Economic valuation of ecosystem services is, however, controversial partly due to ethical reasons. Sagoff (2002) and Maguire and Justus (2008) argue that nature has a right to exist irrespective of human needs and that economic valuation is morally suspect. Heal (2000) states that what is important is not to value ecosystem services, but to demonstrate the incentives for their conservation. While valuation is not a panacea, it can do just that. The use of a common language (i.e. money and markets) provides a yardstick that measures the benefits humans derive from ecosystems. These benefits can then be compared more easily against alternative uses of the ecosystems and their assets (Turner *et al*., 2003).

Ecosystem services are a fundamental part of human survival yet many of them have no market or monetary value (Daily, 1997). This imposes a big challenge to environmental economists on how to translate the values of ecosystem services into monetary terms. To address this, economists need to work together with natural scientists to understand the components of ecosystems that provide the services. This will facilitate the selection of suitable indicators or proxies of the services that can be used in their valuation.

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The objective of this work was to develop methods to evaluate the contribution of marine ecosystems to bioremediation of wastes, gas and climate regulation service, and coastal defence, and assess how elevated biodiversity enhances these ecosystem services. Owing to the difficulties in quantifying biodiversity (Sheppard, 2006), this study focused on keystone species as proxies for biodiversity.

The rest of the paper is organised as follows: Sections 2, 3 and 4 explain the methods developed for each of the three regulatory services and presents preliminary findings from case study sites. In Section 5, the results of the valuation studies are discussed together with how the methods developed could be improved. Section 6 concludes the paper.

2. GAS AND CLIMATE REGULATION

Gas and climate regulation is an ecosystem service where living marine organisms contribute considerably to the maintenance of the chemical composition of the atmosphere and oceans (Beaumont *et al*., 2008). The marine environment strongly influences atmospheric CO_2 concentrations through the biological carbon pump (Miller *et al*., 1999; Bakker *et al*., 2001). Marine phytoplankton annually fixes ~45 Gt carbon globally of which ~16 Gt C per year is exported to the ocean interior (Falkowski *et al*., 1998). Only a small fraction of this however, enters deep-water reservoirs where it is removed for significant time periods from potential interaction with the atmosphere. The remainder that does not sink to deeper waters is re-mineralised or sequestered by secondary producers. The oceanic uptake of atmospheric CO_2 provides indirect social benefits, including the reduction of global warming impacts (Tol, 2005). Such oceanic uptake is carried out by autotrophic organisms including phytoplankton and macrophytes such as kelp and seagrass. The marine ecosystem component of the gas and climate regulation service can therefore be valued by determining the quantity of carbon fixed by living marine organisms and combining it with the marginal damage costs avoided method (e.g., Lambert, 2003).

2.1. Gas and climate regulation by the Isles of Scilly marine ecosystems

The Isles of Scilly is a geologically defined granite archipelago consisting of five inhabited islands and over 300 uninhabited islands and islets. It is located 28 miles off the southwest coast of England, and is comprised of approximately 383 km^2 of inshore waters (Figure 1). The geology of the archipelago dictates a diverse substrate range from fine mud through to sand, bare boulder and bedrock. A combination of extremely low freshwater runoff and varying strength of tidal currents around the islands results in high habitat diversity supporting a number of naturally rare species (Munro and Nunny, 1998).

The biodiversity of the Isles of Scilly has been extensively studied (e.g., Russel, 1968; Fowler, 1990, 1992; Sims, 1996). The exposed western reaches of the island system consist of bedrock and boulder substrate predominantly supporting kelp and red algae. The less exposed eastern reaches support a higher diversity of kelp, algae, sponges and seagrass beds of varying density. Within the island system there are extensive intertidal flats, bars and shallow subtidal areas of finer sand and shell gravel supporting significant seagrass beds and algae (Munro

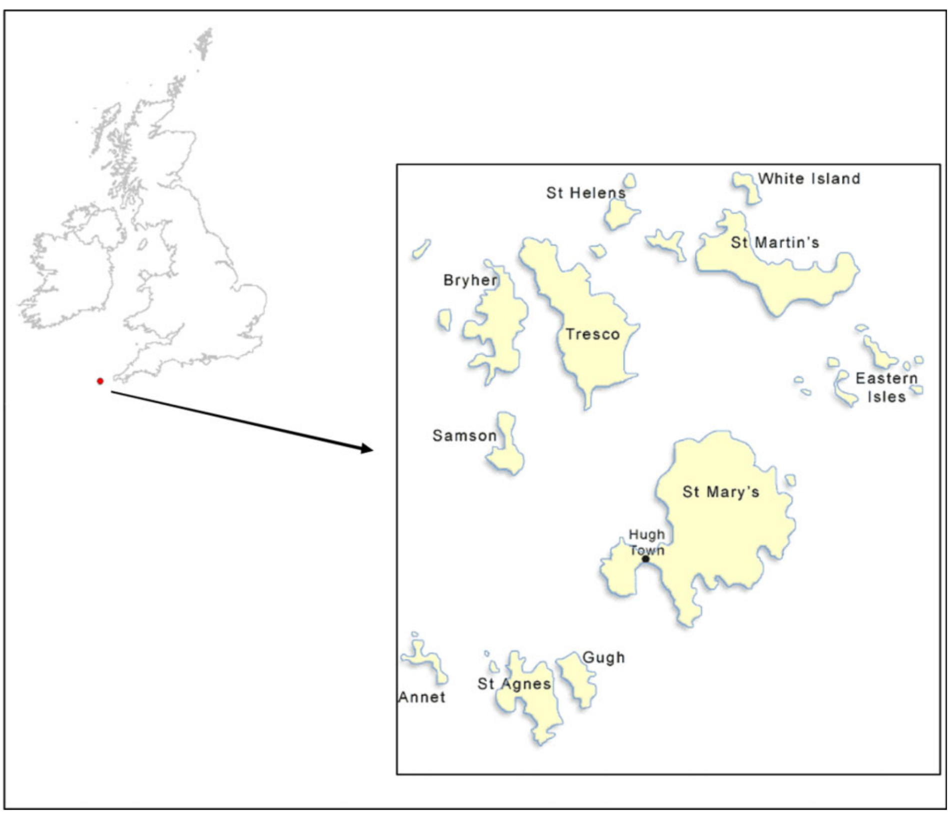

Figure 1. Map of the study area taken from www.scilly.gov.uk. This figure is available in colour online at wileyonlinelibrary.com/journal/environmetrics

and Nunny, 1998). Due to data availability, research focused on the keystone ecosystem species of kelp (*Laminaria hyperborea*) (Foslie), seagrass (*Zostera marina*) (Linnaeus), and phytoplankton productivity in the overlying water column.

2.2. Estimating productivity

2.2.1. *Phytoplankton*

Sea-viewing Wide Field-of-view Sensor (SeaWiFS) ocean-colour data from 1998–2005 was used to obtain phytoplankton production estimates from surface chlorophyll concentrations through the application of a productivity model (Smyth *et al*., 2005). A Case I water algorithm was applied to the data, which was obtained at 18 km resolution for the coordinates 49.55 N′, 06.19 W′ (Isles of Scilly). Resultant mean daily production values for each month were converted to monthly totals, annual totals and finally mean net annual primary production values per unit area. For months where no data were available, values were interpolated by averaging values from the previous and subsequent months.

2.2.2. *Kelp and seagrass*

An extensive literature search was conducted to establish the most representative values for photosynthetic carbon fixation by the two major benthic primary producers in the Isles of Scilly: *Laminaria hyperborea* (kelp) and *Zostera marina* (seagrass). *L. hyperborea* is the most abundant macroalgae in the Isles of Scilly (Munro and Nunny, 1998). Productivity values for *L. hyperborea* were determined from the literature (Table 1). Abdullah and Fredriksen's (2004) productivity values from a study of *L. hyperborea* on the southwest coast of Norway were selected due to their accuracy, species and geographical proximity relative to other similar studies (e.g., Drew *et al*., 1976; Borum *et al*., 2002). Drew *et al*. (1976) obtained productivity values for *L. hyperborea* in British waters but did not provide supplementary data on the fate of carbon. Since their value fell within the high-end range of that produced by Abdullah and Fredriksen (2004) this supported the notion that the latter values were suitable.

Productivity data for *Z. marina* of the closest geographical and environmental proximity was that of Wium-Andersen and Borum (1984) from a random biomass study on the Danish coast (Table 2). The mean water depth over *Z. marina* beds was 0.8 m and the substrate consisted of coarse sand comparable to that of the Isles of Scilly. These values were transferable to the Isles of Scilly as production was found to be independent of salinity and temperature, and dependent on irradiance.

Table 1. Summary of the values from the literature on the amount of carbon sequestration by species of kelp

Author	Specification	Value	Units	Study area
Nybakken and Bertness (2003)	Net productivity	800	$gC\,m^{-2}\,y^{-1}$	California
		2000	$gC\,m^{-2}\,y^{-1}$	Indian Ocean
Mann *et al*. (1980)	Fixation rates	5.0–10	$gC\,m^{-2}\,d^{-1}$	
Johnson *et al*. (1998)	Maximum gross photosynthesis	7	$\mu molC\,m^{-2}\,s^{-1}$	Model
	Respiration rate	0.002	$\mu molCg$ (dry wt)$^{-1}\,s^{-1}$	
	L. sacc total biomass	0.12–2.6	$gC\,m^{-2}$	NE Greenland
	Cummulated respiration	15	$\mu molC\,mg$ (dry wt)$^{-1}$	
	Net annual gain	8	$\mu molC\,mg$ (dry wt)$^{-1}$	
	Annual primary production	0.1–1.6	$gC\,m^{-2}\,y^{-1}$	
Dunton and Schell (1986)	*L. soliclungula*			Alaska
	Net carbon production	0.7	$gC\,y^{-1}\,plant^{-1}$	
	Annual primary production	6.0–10	$gC\,m^{-2}\,y^{-1}$	
	L. sacc fixation rate	0.2–0.8	$\mu gC\,mg^{-1}\,h^{-1}$	
Willenbrink *et al*. (1975)	*L. hyperborea*			
	Photosynthetic rate	12.0–39	$\mu molCO_2\,dm^{-1}\,h^{-1}$	
Johnston *et al*. (1977)	*L. sacc* net production	120	$gC\,m^{-2}\,y^{-1}$	Scottish sea loch
	Distal decay loss	40–50	%	
Abdullah and Fredriksen (2004)	*L. hyperborea*			
	Biomass	10	kg (wet wt) m^{-1}	West Norway
	Production	600–1000	$gC\,m^{-2}\,y^{-1}$	
	Night exudation DOC	0.28–4.09	$\mu molC\,g^{-1}\,h^{-1}$	
	Day exudation DOC	0.18–4.7	$\mu molC\,g^{-1}\,h^{-1}$	
	Annual exudation	1.29	$kgC\,m^{-2}\,y^{-1}$	
	Mean density of kelp	16.27	$kg\,m^{-2}$	
	Fixation rates	3	$kgC\,m^{-2}\,y^{-1}$	
Drew *et al*. (1976)	*L. hyperborea*			
	Productivity	2.6	$gC\,m^{-2}\,d^{-1}$	British waters
	Fixation rates	1.0–10	$\mu gC\,cm^{-1}\,h^{-1}$	20 m–0 m depth

Table 2. Summary of the values from the literature on the amount of carbon sequestration for *Z. marina*

Author	Specification	Value	Units	Study area
McRoy and McMillan (1977)	Estimated production	500–1000	$gC\,m^{-2}\,y^{-1}$	Temperate
Zieman and Wetzel (1990)	Production	8	$gC\,m^{-2}\,d^{-1}$	
Suzuki *et al*. (2003)	Primary production	0.4–1.5	$gC\,m^{-2}\,d^{-1}$	Tropical West Pacific
	Total ecosystem	13	$tC\,km^{-1}\,y^{-1}$	
	Net carbon fixation	14	$tC\,km^{-1}\,y^{-1}$	
Thom (1996)	Net primary productivity	300–1000	$gC\,m^{-2}\,y^{-1}$	
Short and Neckles (1999)	Net photosynthesis	14	$\mu gC\,cm^{-2}\,d^{-1}$	
Gazeau *et al*. (2005)	Average sink	1.7–2.7	$mmolC\,m^{-2}\,d^{-1}$	NW Mediterranean
	Fluxes	2.9–8.7	$mmolC\,m^{-2}\,d^{-1}$	
Penhale (1977)	Mean fixation	0.923	$mgC\,g^{-1}\,h^{-1}$	North Carolina, USA
	Mean excretion	0.007	$mgC\,g^{-1}\,h^{-1}$	
	Epiphyte-colonised fixation	1.192	$mgC\,g^{-1}\,h^{-1}$	
	Clean plant fixation	1.517	$mgC\,g^{-1}\,h^{-1}$	
	Primary productivity	330	$gC\,m^{-2}\,y^{-1}$	
	Excretion	5	$gC\,m^{-2}\,y^{-1}$	
Wium-Andersen and Borum (1984)	Leaf production	554	$gC\,m^{-2}\,y^{-1}$	Danish Coast
	Root rhizome production	260	$gC\,m^{-2}\,y^{-1}$	
	Total eelgrass production	814	$gC\,m^{-2}\,y^{-1}$	
	Total community production	884	$gC\,m^{-2}\,y^{-1}$	
Sand-Jensen (1975)	Leaf production	328	$gC\,m^{-2}$ April–Oct	Denmark
	Rhizome production	87	$gC\,m^{-2}$ April–Oct	
	Total production	415	$gC\,m^{-2}$ April–Oct	

An aerial photograph, a subtidal biotope map (Munro and Nunny, 1998) and a visual field study enabled identification of the spatial extent of habitats. The habitat categories of seagrass bed and kelp were mapped using the software package ArcMap 9.2 on a British National Grid projected coordinate system spatially referenced base map of the archipelago provided by EDINA Digimap.

2.3. Calculation of net present value

The social cost of carbon is the damage done by emitting an additional unit of carbon dioxide. It is a measure of the seriousness of climate change, and a yardstick against which to judge actual and proposed climate policy. The social cost of carbon is usually estimated as the net present value of climate change impacts over the next 100 years (or longer) of one additional tonne of carbon emitted to the atmosphere today. It is the marginal global damage costs of carbon emissions. Three values of the social cost of carbon were selected based on their derivation and representation of the range of values currently available (Table 3). All values are quoted for the period 2001–2010 and were converted to pounds sterling (£) using purchasing power parity tables (Antweiler, 2006).

Table 3. Description on how the values of the social cost of carbon used were derived by the different literature sources

Source	Value	Derivation
Fankhauser (1994)	\$23 tC^{-1}	Calculations were based on a stochastic greenhouse model in which all key parameters were random. Employing expected values instead of best guess values gave high damage values due to uncertainties in future emission estimates. The model neglected catastrophic events, which might skew the data, and partly accounted for damage aversion and adaptation, without employing equity weights
Tol (2005)	\$5 tC^{-1}	Based on a review of estimates produced by 28 studies of the marginal costs of CO_2 from both peer-reviewed and grey literature. The mean was much higher at \$104 tC^{-1}, reflecting large uncertainties and the greater probability of negative impacts over positive ones. Exclusion of grey literature results in lower estimates: the best guess is still \$5 tC^{-1} but the mean estimate drops to \$50 tC^{-1}. The benefit of this estimate is that it theoretically gives a representative figure of current thinking. Not all studies include equity weighting and discount rates varied between studies giving a large standard deviation
Stern (2006)	\$16 tC^{-1}	Value reflects prices generated by traders based on supply and demand information for emissions allowances under the European Union Emission Trading Scheme, Phase I. Tradable values were taken to be the most representative of the current market (August 2006) and most applicable value in terms of influencing modem policy making

Table 4. Summary of productivity estimates for kelp (*L. hyperborea*), seagrass (*Z. marina*) and phytoplankton for the Isles of Scilly showing the total annual production

Primary producers	Area, km^{-2}	Productivity, $gC\,m^{-2}\,y^{-1}$	Annual production, $tC\,y^{-1}$
Kelp	23.5	800	18 776.6
Seagrass	3.1	814	2519.8
Phyto plankton	383.1	300.7	115 198.8
Total			136 495.1

Damages associated with the emission of greenhouse gases result over long time periods and so economic discounting is necessary to calculate the net present value (NPV) of carbon fixation. The NPV can be defined as the present gains, plus all the future benefits minus both initial and future costs (Brainard *et al*., 2005). Different discount rates have been used in the literature to produce estimates of global warming damages (e.g. Brainard *et al*., 2005). Discount rates of 1, 3.5 and 5% were chosen based on a review of discount rates used in previous studies (Brainard *et al*., 2005) and the UK Government's preferred discount rate for projects with social benefits of 3.5% (Brainard *et al*., 2005). Net present value (NPV) was calculated using the above discount rates over a 100-year period since damages associated with a tonne of carbon dioxide emissions would occur over a period in excess of 100 years (Clarkson and Deyes, 2002).

The productivity values for each biotope or habitat were combined with the area coverage data to produce annual photosynthetic carbon fixation estimates for the archipelago. This data was multiplied by the selected values for carbon and discounted over 100 years to produce a range of net present values for the marine ecosystem service of gas and climate regulation in the Isles of Scilly.

2.4. Results of gas and climate regulation in the Isles of Scilly

Mapped area coverage for both kelp and seagrass indicated that kelp was much more prevalent than seagrass at approximately 6 and 0.8%, respectively. However, photosynthetic production in the Isles of Scilly (around 136 495 $tC\,y^{-1}$) is mostly due to phytoplankton productivity (Table 4).

The net present value estimates for the marine ecosystem service of gas and climate regulation in the Isles of Scilly based on annual photosynthetic carbon fixation and averages of the three values of social cost of carbon ranged from £25.4 million to £80.6 million (Table 5). The large range of values resulted from the variability in the values of the social cost of carbon from the literature and the different discount rates applied. Variation in values due to discount rates indicates the importance of using an appropriate rate. As expected the lower discount rate (1%) showed the highest mean value (£80.6 million) while the highest discount rate (5%), the lowest mean value (£25.4 million). The results indicated that lowering the UK Government's recommended discount rate from 3.5% to 1% led to an increase of ~£45 million in the mean value while increasing the discount rate to 5% only reduced the mean value by £10 million. The use of different discount rates acted as a sensitivity analysis of the values estimated.

3. BIOREMEDIATION OF WASTES

The marine environment is where a significant amount of human waste (organic and inorganic) finally settles. Marine living organisms store, bury and transform many wastes through assimilation and chemical de - and re-composition, either directly or indirectly. For example, the bioturbation (reworking and mixing of sediments) process as carried out by mega- and macro-faunal organisms on the deep-sea floor will serve to bury and recycle wastes through assimilation and chemical re-composition (and simple burial with no assimilation at all). These detoxification and purification processes are of critical importance to the health of the marine environment. This bioremediation service provided by marine biota can be valued indirectly by estimating the cost of an alternative approach i.e. the cost of using some other resource to achieve the same end as the one in question. For example, where raw sewage is discharged directly into the marine environment, it is marine biota which effectively performs remediation of such waste. Therefore, the value of this service can be quantified by estimating the cost required to treat raw sewage industrially before discharge to the same standard as that achieved by the ecosystem.

Table 5. Calculated net present value in £ of gas and climate regulation at the Isles of Scilly showing the range of values estimated using the different discount rates. The net present values were calculated over a 100-year period

Source	Value, £	$r = 1\%$	$r = 3.5\%$	$r = 5\%$
Fankhauser (1994)	14.3	122 680 714	53 829 092	38 632 383
Tol (2005)	3.1	26 239 563	10 513 235	8 262 887
Stern (2006)	10.8	93 171 958	40 881 421	29 340 021
Average	9.4	80 697 412	35 407 916	25 411 764

r is the discount rate.

3.1. Bioremediation of wastes by the Isles of Scilly marine ecosystems

The Isles of Scilly Council is responsible for waste disposal on the islands. An interview was conducted with the Chief Engineer to quantify the amount of water consumed on the islands and the amount of raw sewage discharged directly into the sea. The amount of BOD discharged per day through a sea outfall on the Island was estimated between July 2006 and June 2007. Using 2006 data on costs for sewage treatment published through the June Returns by OFWAT (www.ofwat.gov.uk), the cost of treating the volume of sewage discharge in the Isles of Scilly was calculated. This was based on how much money would be spent to treat the raw waste to tertiary level. The sewage treatment works loads (kg BOD/day) and treatment costs used were those from South West Water Regional Band 1 that reported the amount of BOD discharged for a population of <25 000 people. The cost of treating 1 kg BOD of sewage based on the reported costs from South West Water was £2.1.

3.2. Results of the bioremediation of wastes in the Isles of Scilly

The annual discharge of raw sewage in the Isles of Scilly was estimated at 59,761 m^3 containing 108 521 kg BOD per year. This waste would have had an annual treatment cost of £227 894 if it were treated to a tertiary level by a sewage treatment plant. Since the marine biota around the Isles of Scilly performs a service equivalent to the cost of treating the waste to tertiary level, the bioremediation of waste service can be valued at £227 894 per year for the islands.

4. COASTAL SEA DEFENCE BY WETLANDS

Coastal wetlands, defined as the coupled system of mudflats and salt marsh, provide natural sea defence by acting as buffer zones to wave action and storm surges (Brampton, 1992; Pethick, 1992). The presence of biogenic structures as frontline sea defences can dampen and prevent the impact of tidal surges, storms and floods through binding of sediments and wave attenuation (Moeller and Spencer, 2002; Widdows and Brinsley, 2002). The attenuation of wave energy may vary with the hydrodynamic factors such as incident wave height and water depth (Möller, 2006). For instance, results from field measurements on UK salt marsh by Möller *et al.* (2001) showed that at low inundation depths, wave attenuation was 87% while in high inundation depths this was reduced to 72%. Besides hydrodynamic factors, wave dissipation over a vegetated surface is affected by the roughness of the bed. Salt marsh vegetation can be considered as a form of macro-roughness, where reduction in flow is greatest in the densest canopy (Neumeier, 2007). Wetlands not only attenuate wave energy, but also play a role in reducing erosion of the coastline. Mudflats dissipate tidal and wave energy to a level low enough to permit net sediment deposition, allowing colonisation by salt marsh or reed-bed vegetation on the upper intertidal and hence reducing erosion (Nottage and Robertson, 2005).

Two valuation methods were used to illustrate the benefits of the sea defence role of wetlands: a contingent valuation survey asking coastal households their willingness to pay to avoid a reduction in wetlands acreage, and a preventative costs method that estimates the money saved in sea defence construction from a fronting wetland. The two valuation approaches were applied to the Humber and Exe Estuaries, North Norfolk and Essex coastlines in order to demonstrate the value of wetlands as a sea defence for UK coastal populations and to assess changes in the value based on different wave exposure levels and marsh profiles. Both Essex and North Norfolk have extensive open-coast wetlands. Essex has approximately 4636 ha of salt marsh (Doody, 1992) while North Norfolk is fringed by a 35 km stretch of back-barrier salt marsh (Pye, 1992). The Humber Estuary also has significant coverage by salt marsh of ~648 ha (Doody, 1992), whereas the southerly location of the Exe Estuary has salt marsh covering 66 ha (JNCC, 1997).

4.1. Wetlands valuation using the contingent valuation method (CVM)

The contingent valuation method (CVM) is widely used in the environmental economic literature to estimate benefits from ecosystems since it is capable of measuring direct use, indirect use and non-use values of environmental goods and services (Spurgeon, 1992). CVM elicits people's preferences for public goods by asking how much they would be willing to pay or accept for a specified change in the availability of the goods, using a questionnaire. It addresses the absence of traditional markets for public goods by describing a hypothetical market in which respondents are able to purchase the goods in question. The elicited willingness to pay (WTP) amounts are therefore contingent upon the hypothetical market presented to the respondent (Mitchell and Carson, 1989).

CVM has been used extensively in the valuation of biodiversity including recreational use of biodiversity (e.g., Mercer *et al*. 1995; Slee, 2007), and the perceived value of conservation policies (e.g., Garrod and Willis, 1997; Gurluk, 2006). Although the use of CVM is widespread, Hanley and Spash (1995) identified a number of potential sources of bias such as: (1) strategic bias caused by individuals being able to secure benefits greater than the costs they have to pay by answering dishonestly; (2) vehicle bias which arises from the hypothetical instrument of payment used (for example, some individuals may regard 1% increase in taxation as more 'costly' than one dollar increase in price. However, the use of a neutral payment vehicle can resolve this source of bias); (3) information bias based on the amount of information provided to respondents; and (4) hypothetical bias, where respondents answer a question in such a way as to please the interviewer, or state that they are willing to pay more than they truly would, knowing that they will not have to spend real money. In response to criticisms, the National Oceanic and Atmospheric Administration (NOAA) Contingent Valuation Panel recommended guidelines to help ensure the reliability of contingent valuation surveys including the use of in-person interviews, a binary discrete choice question, a careful description of the good and its substitutes, and several different tests that should be included in the report on the survey results (Arrow *et al*., 1993).

The contingent valuation method was used to estimate the value of the defence role provided by coastal wetlands in the UK. Local coastal residents in each of the four case study sites were asked to state the maximum amount they were willing to pay each year to conserve the wetlands of their area using an open-ended question format. The open-ended format was preferred over closed-ended since it is straightforward both in terms of question design and data analysis. Further, since the sample would comprise of coastal households with a wide range of incomes and abilities to pay, it was considered that the discrete WTP values that respondents would need to select from in a closed format would be too restrictive. The payment vehicle used was an annual trust fund payment to the Environment Agency, which is responsible for coastal defence in the UK. The questionnaire was pre-tested with 21 households around the Exe Estuary during March 2008 and was administered as a postal survey to 221 households during May and June 2008. Where possible, the households that were used in the study were located within 500 m of shoreline and so were most likely to benefit from the existence of coastal defence systems.

Obtaining accurate estimates using CVM requires a detailed description of the resource being valued. This is evident from the name of the method, which produces values, contingent upon the description of the good and method of payment. A pre-notification letter was therefore sent to all households detailing the process of the CVM survey. Further, the questionnaire presented details of three hypothetical management plans related to wetlands in the UK. These management plans included: (1) improvement of wetlands with a limited maintenance of hard defences. This plan would ensure that in appropriate areas, sea walls would be breached to allow new wetlands to develop thereby resulting in roughly 25% hard defence and 75% wetlands; (2) construction of new hard defences while holding the existing line of wetlands. This plan would include squeezing out the present wetlands between the possible rising sea level and hard defences thereby ending with roughly 75% hard defence and 25% wetlands and (3) a mixture of both natural and hard defences with the improvement of wetlands in certain areas and the construction of hard defences in appropriate areas resulting in roughly 50% hard defence and 50% wetlands. Respondents were then asked if they would be willing to pay to support the stated management plans. Those who answered yes to this question were asked to state the maximum amount they were willing to pay for each plan.

A total of 87 households responded out of 221 contacted giving a response rate of 39%. No responses were rejected as being unrealistic bids in relation to respondent's income but 24 responses were rejected as protest bids. The mean and median willingness to pay (WTP) was calculated for each case study site from the remaining 63 responses. The maximum revenue that would be generated by each hypothetical management plan was calculated by summing individuals WTP amounts for each plan.

4.2. Valuation using savings made on sea defence construction

The money that could be saved in sea defence construction due to a fronting wetland was estimated for each site. The height of an offshore wave was calculated and transformed to a nearshore height. Unbroken nearshore waves at the start of the mudflat were modelled across five different intertidal attenuation scenarios ranging from full wetland to no wetland based on recent UK field obtained attenuation data (Moller and Spencer, 2002). In addition, different storm surges were applied in order to determine the height of a seawall that would be required to protect coastal settlements.

Nearshore wave data needed to accurately design sea defence walls are rare and therefore engineers regularly rely upon offshore data from a network of buoys around the UK to construct defence walls. Data for wave direction and wave period were obtained from CEFAS (Centre for Environment, Fisheries and Aquaculture Science) data archive for four buoys located as close to the chosen wetland sites as possible. The data were sorted into weekly peak values and plotted on a Gumbel and Weibull II probability distribution chart. The slope for each best-fit model was recorded and used to calculate the wave height and tidal regime.

Bathymetric and navigational charts obtained from Edina Marine Digimap service (http://edina.ac.uk/digimap) were used to define bottom contours from the buoy to the start of mudflats at 0-m chart datum. The shoaling coefficient, refraction coefficient and the refraction diagram orthogonal method (CERC, 1984) were used to calculate the nearshore height of the wave. The depth at which the wave would break was calculated to determine whether the wave was broken prior to travelling over the wetland at each site. If the wave was not broken at the 0-m chart datum, then the wave was attenuated by the wetland, whereas if the wave had broken prior to the mudflat then the wetland did not attenuate the wave.

Depending on the sediment supply and wave energy, the transition between mudflat and salt marsh may take the form of a ramp or cliff profile (Allen, 1989). A gradual sloped margin, typical of an accreting salt marsh, will dissipate wave energy differently from a salt marsh with a cliff transition, normally associated with erosion. The profile of the salt marsh for each case study site was identified and used to compare the savings made on sea defence construction. Since the height of a wave is determined by the depth of water it travels over, water depth was estimated for various widths of vegetated marsh using the navigational charts and shoreline profiles.

The depth of water at the base of a hypothetical seawall was determined by the influence of wave action and different storm surges. It was assumed that the upper limit of the wetland would be at the high water spring tide level due to the ecology of marsh zonation. From the water depth at the shoreline, the crest elevation (height of the wall) of a seawall structure was calculated using the relationship between still water depth and relative crest elevation presented by Goda (1985). Seawall heights were calculated for a vertical revetment (a simple seawall construction) with a 1/30 slope.

Using the costs of seawall construction reported by King and Lester (1995), new price bands at 2007–2008 market values were determined for various wall heights using the retail price index. Savings in hypothetical seawall construction were determined by comparing seawall heights required under different scenarios from having a full wetland, through no vegetation hence mudflat only, to no wetland at all. The sites at North Norfolk and Exe Estuary were not subject to this valuation approach since the calculated wave height and tidal regime showed that the offshore waves break prior to the start of the wetlands (0-m chart datum). Therefore, the valuation results presented are those from Essex and Humber Estuary.

4.3. Results of valuation of sea defence role of wetlands

Coastal residents in Humber, Essex, North Norfolk and Exe estuary were willing to pay £213 per year (median £100) to prevent a reduction in the present coverage of wetlands. Local residents in the Exe Estuary placed the highest value on the sea defence role of wetlands (mean £362 per year) while Essex households provided the lowest value of wetlands (mean £39 per year) (Table 6). The management plan that received the largest revenue from the respondents WTP bids was having 50% wetlands together with 50% hard defences which received total revenue of £10 547 per year (Table 6).

The wetlands of Humber (Tetney Marshes) and Essex (St. Peter's Flats) attenuated waves at all water depths while those of North Norfolk and Exe Estuary showed that waves broke prior to the start of wetlands. From a shoreline profile, the wetlands at St. Peter's Flats in Essex showed a cliff profile while those at Tetney Marshes in the Humber had a ramped profile. The present widths of salt marsh at St Peter's Flats and Tetney Marshes therefore would reduce the height required for a seawall through offshore wave attenuation. Having wetlands of over 60 m vegetation in both sites meant that there was no need for a seawall. However, all storm surge scenarios reduced the function of the present wetlands thereby implying that a seawall would be required. The height and therefore the cost of constructing a seawall that would be required for effective protection from flood and storms in the Humber Estuary was consistently higher across all wetland width scenarios compared to that required for the Essex coastline (Figure 2). The wetlands at St. Peter's Flats in Essex were associated with lower savings in seawall construction than the Tetney Marshes in the Humber (Figure 3). At Tetney Marshes with present vegetated marsh width at approximately 1648 m, the potential mean saving on seawall construction cost ranged from £12 237–30 057 per m^2 across the storm surge scenarios, while the savings at St. Peter's Flats in Essex ranged from £889–6115 per m^2 (Figure 3).

Table 6. Amount of money in £ that coastal residents in the four sites were willing to pay to preserve wetlands for their sea defence role. The revenue that would be generated from respondents' willingness to pay bids is presented for each hypothetical management plan

WTP statistics	Exe Estuary	North Norfolk	Essex	Humber Estuary	Total
Mean WTP	362	244	39	231	213
Median WTP	110	100	5	75	100
Sample size	10	10	10	10	63
Management plan					
A (25% hard defence, 75% wetland)	1620	3457	205	1810	7092
B (75% hard defence, 25% wetland)	1870	1600	153	4210	7833
C (50% hard defence, 50% wetland)	3020	3102	535	3890	10 547

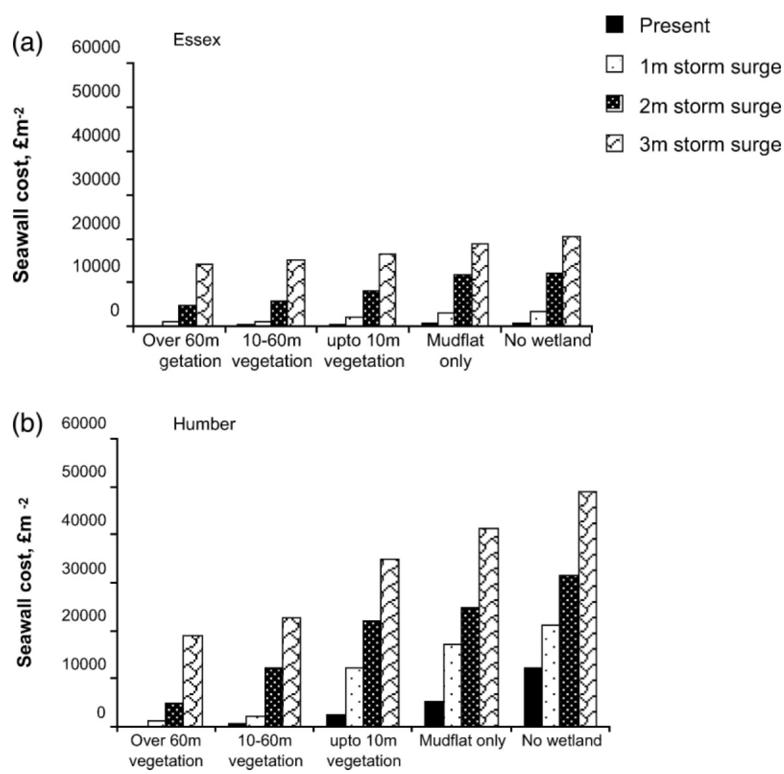

Figure 2. Seawall construction costs for (a) Essex and (b) Humber for five wetland widths scenarios. The costs for each scenario were estimated based on the present height of waves and possible wave heights for three storm events

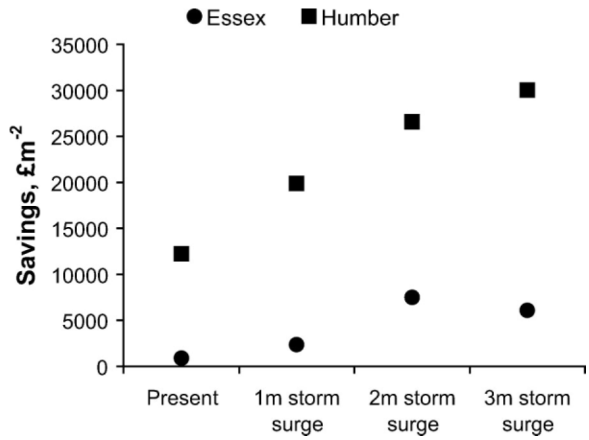

Figure 3. Amount (£) of savings made on seawall construction costs for Essex and Humber Estuary comparing the savings made based on present wave height and possible storm surge events. Savings were calculated as the difference between the amount of money required to build a seawall where there is no wetland and the money required where a wetland of over 60 m vegetation is present

5. DISCUSSION

There is a considerable increase in the desire by ecologists and economists for biodiversity and ecosystem functioning to be expressed as part of mainstream economic philosophy of value. Here, three methods were developed based on value transfer, stated preferences, replacement costs and damage cost avoided based on social cost of carbon. The aim was to estimate the value of bioremediation of wastes and marine ecosystem gas and climate regulation service provided by marine ecosystems in the Isles of Scilly, and sea defence provided by wetlands around the UK. The quantification of the services using the equivalent costs of treating sewage to a tertiary level; marginal damage costs avoided through the mapping of habitats of keystone species and phytoplankton productivity; and stated preferences and savings on sea defence construction are relatively simple ways to evaluate complex ecosystem services that are an integration of many different ecological processes.

5.1. Valuing gas and climate regulation

The purpose of valuing ecosystem services is, ultimately, to provide information that will lead to improved management of marine ecosystems. A range of values were estimated that represent the costs that would be incurred if the marine ecosystems in the Isles of Scilly were lost or damaged. Carbon is cycled between photo-autotrophic primary producers and secondary producers via the microbial loop which itself is populated by a wide range of organisms (Azam *et al*., 1983). Some of the carbon is also exported to deeper waters primarily as particulate organic carbon (POC) and many recent models suggest a global surface POC export of 10 Pg C y^{-1} (Boyd and Trull, 2007). Only a small fraction of POC enters deep water reservoirs where it is removed for significant time periods from potential interaction with the atmosphere. The POC that does not sink to deeper waters is re-mineralised or sequestered by secondary producers thereby contributing to other ecosystem services such as fisheries production. Consequently, the method developed is limited since it focused primarily on the carbon sequestered by phytoplankton and macrophytes. However, it indicates the value of maintaining these ecosystems as dynamic carbon stores and thus their conservation is essential in sustaining their contribution to ecosystem services.

Phytoplankton primary productivity estimates were highly accurate owing to the net westerly flow of water around the Isles of Scilly and lack of significant freshwater runoff providing uniform salinity and low turbidity around the islands. As such the influence of coloured dissolved organic matter (CDOM) associated with riverine inputs, suspended particulate matter (SPM), and associated with turbidity were reduced. Therefore, there were fewer complications in the optical signal and the Case I water algorithm assumption that all ocean colour variations were due to phytoplankton and their by-products was appropriate (Smyth *et al*., 2005). The model used, as described in Smyth *et al*. (2005) was found to be among the most accurate models to date in a recent study (Carr *et al*., 2006). Therefore, the confidence levels in the phytoplankton primary productivity estimates are high.

Nonetheless, the method requires further development. A notable limitation of this method was the estimation of benthic vascular plant and macro-algae productivity. The assumption that productivity of kelp and seagrass in the Isles of Scilly is comparable to that of Norwegian kelp and Danish seagrass probably reduced the accuracy of estimates. Productivity of these plants is influenced by physical conditions such as temperature, water clarity, nutrient availability and competition with other flora and grazing fauna (Hasegaw *et al*., 2007), which may vary between the Isles of Scilly and the study sites where the productivity rates were determined. Exudate rates of seaweeds have been found to vary in the natural environment by up to 35% of the net assimilation value (Hatcher *et al*., 1977), which may have resulted in inaccuracies when the Norwegian kelp productivity data were transferred to the Isles of Scilly. The Wium-Andersen and Borum (1984) study from which seagrass productivity was derived was located near the Kattegat and the Baltic Sea so it is possible that the nutrient status of the study areas differed. The presence of competitive species may have varied between the Isles of Scilly and the Danish coast leading to potential inaccuracies when estimating seagrass productivity. Ecosystem service valuations based on functioning of keystone species alone are likely to be underestimates.

The assumption that vegetation density was uniform will also have affected productivity estimates, particularly with respect to the patchy nature of *Z. Marina* beds. In the case of *L. hyperborea*, mapping deeper kelp forests was not possible using the aerial photograph as they could not be seen and it was therefore assumed that kelp cover corresponded to hard substrates to a water depth of 30 m, determined through reference to SeaZone electronic chart data (www.seazone.com). This seemed a reasonable assumption since Drew *et al*. (1976) found that the lower limit of *L. hyperborea* production in Cornwall (~37 m) was related to light penetration. However, their study indicated that plant density and productivity both decreased with depth due to the attenuation of light and grazing pressure. This in turn would influence the accuracy of the productivity data used here as constant productivity was assumed throughout the archipelago.

The largest sensitivity of this method relates to the economic data. Conservationists have long been concerned that economic theory rationalises discounting the future. From the perspective of a utility maximizing individual or profit maximizing firm, discounting the future makes sense (Norgaard and Howarth, 1991). However, discounting seems highly descriptive of how individuals and firms behave. Whether or not discounting should take place, the broader academic community concerned with sustainable development accept this conceptual argument for one of the same reasons economists do: because it has considerable economic power. However, since a positive discount rate means that effectively no weight is given to resource use or welfare beyond the current generation, discounting appears to be inconsistent with sustainability. Thus, many people question whether discounting is appropriate for public decision making. If the method described in this paper were to be adopted as part of decision support systems for management, standardisation of discount rates and the social cost of carbon would be required.

It is important to consider the impact that climate change and global warming may have on the functional capacity of ecosystems, with respect to sea level rise and ocean temperature rise (Stern, 2006). In the Isles of Scilly, *Z. marina* beds are present in the shallow regions between the islands (Munro and Nunny, 1998) and will be sensitive to increased sea level and water depth as there are minimal opportunities for migration to shallower water. Since *L. hyperborea* is productive to much greater depths sea level rise is unlikely to be as detrimental to its productivity. In the North Sea *Laminaria* species were found to be incapable of surviving water temperatures of 23°C or higher (Luning, 1984). The June sea temperature for the Isles of Scilly was recorded as 13°C in 2005 (MarLIN, 2007), thus in the future the productivity of *L. hyperborea* may be reduced due to increased water temperatures.

5.2. Valuing bioremediation service provided by marine biota

The annual value of waste remediation by marine biota was estimated to be £227,894 using the replacement cost method. It should be noted that this estimate excludes the initial construction costs of sewage treatment works, which may be considerable given the logistics and transport of materials to build a plant on the Island. The value estimated here could therefore be an underestimate. However, it might be an overestimate considering that much of this remediation will be a consequence of dilution and dispersal. Further, the amount of waste remediated will vary according to the type of discharge system used, proximity to the discharge system, and how much of the waste actually reaches the benthos. It is also important to emphasise that there is no official monitoring of sewage discharge rates in place and therefore data to estimate this value is limited.

Pingree and Mardell (1986) showed that the coastal tidal jet current system around the Isles of Scilly archipelago consists of two tidal jets originating at opposing latitudes, which circumvent the islands in a clockwise direction. Sewage discharged at the sea outfall (Morning Point) is therefore likely to be transported on the dominant tidal jet current and circulated around the boundary of the islands rather than being transported offshore to open sea. The findings of Pingree and Mardell (1986) justify this study's assumption that Isles of Scilly marine microbial community is responsible for the bioremediation of the sewage discharged.

This study valued the bioremediation of wastes through the storage of organic BOD by the local flora, fauna and microbial communities. It could be argued that since organic waste would still be present in the microbial and plant matter within the ecosystem then bioremediation has not fully occurred as the waste is still available but in a different form. Micro-organisms do not only facilitate the remediation of organics. Resistant microbial communities have been found to bioremediate heavy metal contamination associated with sewage sludge pollution (Watanabe, 2001), and macro fauna exudates can modify the distribution of metals, their bioavailability and toxicity through complexation (Abdullah and Fredriksen, 2004). The issues of bioremediation of associated pharmaceuticals (Jones *et al*., 2003) and pathogens (Edwards *et al*., 1998) due to the biological processes present in the archipelago ecosystem have not been addressed in this study. Hypothetically, if this marine ecosystem service were to be removed completely there would be a heavy impact on local tourism due to aesthetic degradation of the environment. Since these aspects were excluded the produced value is an underestimate of the true value of bioremediation of untreated sewage waste in the Isles of Scilly. The possible contribution of new nutrients provided by the untreated sewage to the gas and climate regulation service and the release of CO_2 through the respiration of microbial communities which will counteract the potentially enhanced productivity are also excluded from this study.

Consideration of the levels of bioremediation highlights the fact that each habitat has a number of beneficial functions other than those included in this study, such as nursery grounds for fish stocks and disturbance prevention through sediment accretion and stabilisation and dissipation of wave energy (Bos *et al*., 2007). Therefore, the values produced here will only represent a fraction of the total value of the biodiversity of the Isles of Scilly.

5.3. Valuation of coastal wetlands

Coastal residents around the Exe, Humber, North Norfolk and Essex coastlines were willing to pay considerable amounts of money to preserve wetlands for their sea defence role. The management plan of having equal proportions of hard defences and wetlands protecting the coastline received the highest bid revenues. This suggests that respondents have confidence in the ability of wetlands to protect their settlements too and reveals that the UK coastal population would like to see mixture of both hard and soft defences along the coastline. The results showed that a mixture of soft and hard defence schemes could produce effective protection from waves and storms and at lower costs.

This is good news for the current UK government policy that considers a long-term holistic approach in sea defence that works with nature. The current UK flood management policy (Making Space for Water 2004) lays out the UK policy for managing flood and erosion risk (DEFRA, 2005). The policy has four strategies: (1) to set up an adaptable programme enabling modifications in light of climate change; (2) a whole catchment and whole shoreline approach; (3) extensive stakeholder consultation and (4) ensuring that the Environment Agency has a strategic overview of coastal risks. The results obtained in this study not only provide an insight into the value of wetlands, but also highlight the UK coastal populations' preference in sea defence provision. The UK coastal population would support a policy that includes both wetlands and hard defences in protecting their coastal settlements from flood and storm damage.

In UK, sea walls are commonly constructed to withstand a 1 in 50 year (1:50) significant wave height, with current new sea walls around residential settlements being required to withstand a 1 in 200 year event (EUROTOP, 2007). The results of this study showed that wetlands of varying widths could attenuate a 1:50 wave, resulting in lower seawall requirements and subsequent savings in construction costs. The results also showed that the greatest potential savings were obtained with current wetland widths, where it was found that no sea walls were needed. For instance, if there had been no wetland fronting the Humber Estuary then a sea wall of 6 m would have been needed whereas a sea wall of around 2 m would be needed in Essex to offer the same protection as that offered by the present wetland. These results demonstrate that fronting wetlands have high economic value as a sea defence. Further, mudflats, a habitat not widely recognised as providing economic benefit to coastal populations, were associated with savings on sea defence construction since they led to a reduction in the height of the sea wall required.

Parts of the preventative cost method developed here need further improvement. For instance, the use of the pre-analysed data from Möller and Spencer (2002) opened this method to potential inaccuracies. The field observations used in formulating the wetland attenuation scenarios included the stepped transition within the mudflat of a cliff profile wetland in the Humber. These data were transferred and used to calculate the wave attenuation for a similar cliff profile wetland in Essex which resulted in an increase in wave height in Essex as the wave was modelled over this transition. A cliff transition is due to the geology of the formation and is specific to a site and therefore further work will need to repeat the calculations with data from Essex.

Both the CVM and preventative cost method applied to value the buffer role of wetlands have demonstrated that the function of wetlands is not necessarily to be the sole protector of coastal settlements, but have greater value when used together with hard defences. This coupled defence system is under threat in many areas of the UK from pressures such as sea level rise and habitat degradation. Making Space for Water 2004 was the first UK flood-risk strategy to aim for a sustainable long-term approach. The potential financial benefits and public support for wetlands as a sea defence demonstrated in this paper should help push forward new policies under this strategy devised to create a sustainable relationship between constructed defences and wetlands. This valuation exercise has supplied the means for increasing or at least maintaining wetlands in open coasts and estuarine areas of the UK, by determining the financial benefits of conserving this multi-function habitat.

6. CONCLUSION

The services of bioremediation of wastes, and marine gas and climate regulation through the photosynthetic fixation of carbon dioxide are of particular socio-economic importance in light of increased pollution in the sea and global warming. This study applied the damage costs avoided economic method with remote sensing, benthic habitat mapping, and ecosystem productivity data from literature to produce an economic value of the marine service of gas and climate regulation in the Isles of Scilly. Given the exclusion of some species and habitats from the study this is most likely an underestimate of the true value of the marine service.

This study has demonstrated that wetlands have an economic value through financial savings in sea defence construction costs. Ramped wetlands showed the greatest protection and value whereas cliff-profile wetlands potentially offered similar protection from a wide wetland. Further, having mudflats in the transition zone resulted in a decrease in seawall construction costs. The stakeholder perceptions and values estimated in this study should inform future decisions on shoreline policy in the UK. It is unclear whether engineers account for the role of wetlands when designing seawall structures, or when designing sea walls to prevent storm surge events. This project has demonstrated that maintaining coastal wetlands not only ensures an increase in coastal biodiversity but can also have considerable economic benefits.

The success of these valuation methods has potentially large implications for future policy-making. Monetary value is a currency that is universally understood by policy makers, economists, scientists and politicians alike. Provision of such values would help policy makers distribute funds for conservation, and assist in marine planning which is a likely requirement in future policy directives such as the Marine Bill and the European Maritime Strategy.

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REFERENCES

Abdullah MI, Fredriksen S. 2004. Production, respiration and exudation of dissolved organic matter by the kelp Laminaria hyperborea along the west coast of Norway. *Journal of the Marine Biological Association of the United Kingdom* **84**: 887–894.

Allen JRL. 1989. Evolution of salt marsh cliffs in muddy and sand systems: a qualitative comparison in British West Coast estuaries. *Earth Surface Processes and Landforms* **14**: 85–92.
Antweiler W. 2006. *Pacific Exchange Rate Service - Purchasing Power Parity*. Sauder School of Business, University of British Columbia. Available at http://fx.sauder.ubc.ca/
Arrow K, Solow R, Portney PR, Leamer EE, Radner R, Schuman EH. 1993. Report of the NOAA panel on contingent valuation. *Federal Register* **58**: 4602–4614.
Azam F, Fenchel T, Field JG, Gray JS, Meyer-Reil LA, Thingstad F. 1983. The ecological role of water-column microbes in the sea. *Marine Ecology Progress Series* **10**: 257–263.
Bakker DCE, Watson AJ, Law CS. 2001. Southern Ocean iron enrichment promotes inorganic carbon drawdown. *Deep Sea Research Part II – Topical Studies in Oceanography* **48**: 2483–2507.
Beaumont NJ, Austen MC, Mangi SC, Townsend M. 2008. Economic valuation for the conservation of marine biodiversity. *Marine Pollution Bulletin* **56**: 386–396.
Borum J, Pedersen MF, Krause-Jensen D, Christensen PB, Nielsen K. 2002. Biomass, photosynthesis and growth of Laminaria saccharina in a high-arctic fjord, NE Greenland. *Marine Biology* **141**: 11–19.
Bos AR, Bouma TJ, Kort GLJ, van Katwijk MM. 2007. Ecosystem engineering by annual intertidal seagrass beds: sediment accretion and modification. *Estuarine, Coastal and Shelf Science* **74**: 344–348.
Boyd PW, Trull TW. 2007. Understanding the export of biogenic particles in oceanic waters: is there consensus? *Progress in Oceanography* **72**: 276–312.
Brainard J, Bateman IJ, Lovett AA. 2005. The social value of carbon sequestered in Great Britain's woodlands. *CSERGE Working Paper EDM 05-03*.
Brampton AH. 1992. Engineering significance of British salt marshes. In *Salt Marshes Morphodynamics, Conservation and Engineering Significance*, Allen JLR, Pye K (eds). Cambridge University Press: Cambridge; 115–122.
Carr ME, Friedrichs MAM, Schmeltz M, Aita MN, Antoine D, Arrigo KR, Asanuma I, Aumont O, Barber R, Behrenfeld M, Bidigare R, Buitenhuis ET, Campbell J, Ciotti A, Dierssen H, Dowell M, Dunne J, Esaias W, Gentili B, Gregg W, Groom S, Hoepffner N, Ishizaka J, Kameda T, Le Quere C, Lohrenz S, Marra J, Melin F, Moore K, Morel A, Reddy TE, Ryan J, Scardi M, Smyth T, Turpie K, Tilstone G, Waters K, Yamanaha Y. 2006. A comparison of global estimates of marine primary production from ocean colour. *Deep Sea Research* **53**: 741–770.
Coastal Engineering Research Center (CERC). 1984. Department of the Army Waterways Experiment Station, Corps of Engineers, 1984. *Shore Protection Manual, 2* (4th edn). U.S. Government Printing Office: Washington DC.
Clarkson R, Deyes K. 2002. Estimating the social cost of carbon emissions. *Government Economic Service Working Paper 140*.
Costanza R, d'Arge R, DeGroot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, Raskin RG, Sutton P, Van Den Belt M. 1997. The value of the world's ecosystem services and natural capital. *Nature* **387**: 253–260.
Daily GC. 1997. *Nature's Services: Societal Dependence on Natural Ecosystems*. Island Press: Washington DC.
Department of Environment Food and Rural Affairs (DEFRA). 2005. Making space for water - first response. Available at: http://www.defra.gov.uk/environ/fcd/policy/strategy/1stres.pdf.
Doody JP. 1992. Conservation of British salt marshes. In *Salt Marshes Morphodynamics, Conservation and Engineering Significance*, Allen JLR, Pye K, (eds). Cambridge University Press: Cambridge; 80–114.
Drew EA, Jupp BP, Robertson WAA. 1976. Photosynthesis and growth of Laminaria hyperborea in British water. In *Underwater Research*, Drew EA, Lythgoe JN, Woods JD (eds). Academic Press Inc: US; 369–380.
Dunton KH, Schell DM. 1986. Seasonal carbon budget and growth of Laminaria solidungula in the Alaskan high Arctic. *Marine Ecology Progress Series* **31**: 57–66.
Edwards DD, McFeters GA, Venkatesan MI. 1998. Distrbution of Clostridium perfingens and fecal sterols in a benthic coastal marine environment influenced by the sewage outfall from McMurdo Station, Antarctica. *Applied and Environmental Microbiology* **64**: 2596–2600.
Eurotop. 2007. *Wave Overtopping of Sea Defences and Related Structures: Assessment Manual*. Environment Agency, UK; Expertise Netwerk Waterkeren, Netherlands; Kuratorium fur Forschung im Kusteningenieurwesen, Germany; 201 pp.
Falkowski PG, Barber RT, Smetacek V. 1998. Biogeochemical controls and feedbacks on ocean primary productivity. *Science* **281**: 200–206.
Fankhauser S. 1994. Evaluating the social costs of greenhouse gas emissions. *CSERGE Working Paper GEC- 94-01*.
Fowler SL. 1990. Marine nature conservation importance of the Isles of Scilly. *Porcupine Newsletter* **4**: 182–183.
Fowler SL. 1992. Marine monitoring in the Isles of Scilly. *English Nature Research Reports Number 9*, 31 pp.
Garrod GD, Willis KG. 1997. The non-use benefits of enhancing forest biodiversity: a contingent ranking study. *Ecological Economics* **21**: 45–61.
Gazeau F, Duarte CM, Gattuso JP, Barron C, Navarro N, Ruiz S, Prairie YT, Calleja M, Delille B, Frankignoulle M, Borges AV. 2005. Whole ecosystem metabolism and CO_2 fluxes in a Mediterranean Bay dominated by seagrass beds (Palma Bay, NW Mediterranean). *Biogeosciences* **2**: 43–60.
Goda Y. 1985. *Random Seas and Design of Maritime Structures*. University of Tokyo Press: Tokyo.
Gurluk S. 2006. The estimation of ecosystem services' value in the region of Misi Rural Development Project: results from a contingent valuation survey. *Forest Policy and Economics* **9**: 209–218.
Hanley N, Spash CL. 1995. *Cost-Benefit Analysis and the Environment*. Edward Elgar Publishing: Aldershot.
Hasegaw N, Hori M, Mukai H. 2007. Seasonal shifts in seagrass bed primary producers in a cold-temperature estuary: dynamics of eelgrass Zostera marina and associated epiphytic algae. *Aquatic Botany* **86**: 337–345.
Hatcher BG, Chapman ARO, Mann KH. 1977. An annual carbon budget for the kelp Laminaria longicruris. *Marine Biology* **44**: 85–96.
Heal G. 2000. Valuing ecosystem services. *Ecosystems* **3**: 24–30.
Joint Nature Conservation Committee (JNCC). 1997. *An Inventory of UK Estuaries*, Volume 6 Southern England. JNCC: Peterborough; 144.1–144.2.
Johnson MP, Hawkins SJ, Hartnoll RG, Norton TR. 1998. The establishment of fucoid zonation on algal dominated rocky shores: hypotheses derived from a simulation model. *Functional Ecology* **12**: 259–269.
Johnston CS, Jones RG, Hunt RD. 1977. A seasonal carbon budget for a Laminarian population in a Scottish sea loch. *Helgoland Marine Research* **30**: 527–545.
Jones OAH, Voulvoulis N, Lester JN. 2003. Potential impact of pharmaceuticals on environmental health. *Bulletin of the World Health Organisation* **81**: 768–769.
King SE, Lester JN. 1995. The value of salt marsh as a sea defence. *Marine Pollution Bulletin* **30**: 180–189.
Lambert A. 2003. Economic valuation of wetlands: an important component of wetland management strategies at the river basin scale. *Discussion Paper*: Ramsar Convention on Wetlands.
Luning K. 1984. Temperature tolerance and biogeography of seaweeds: the marine algal flora of Helgoland (North Sea) as an example. *Helgoland Marine Research* **38**: 305–317.
Maguire LA, Justus J. 2008. Why intrinsic value is a poor basis for conservation decisions. *BioScience* **58**: 910–911.
Mann KH, Chapman ARO, Gagne JA. 1980. Productivity of seaweeds: the potential and the reality. In *Primary Productivity in the Sea*, Falkowski PG (ed.). Plenum press: New York and London; 363–380.
MarLIN. 2007. Data archive for seabird species and habitats. The Marine Life Information Network for Britain and Ireland, part of Marine Biological Association of the United Kingdom, Citadel Hill, Plymouth, Devon.

McRoy CP, McMillan C. 1977. Production ecology and physiology of sea grasses. In *Seagrass Ecosystems*, McRoy CP, Helfferich C (eds). Dekker: New York; 53–88.
Mercer E, Kramer R, Sharma N. 1995. Rain forest tourism: estimating the benefits of tourism development in a new national park in Madagascar. *Journal of Forest Economics* **1**: 239–269.
Millennium Ecosystem Assessment (MEA). 2005. *Ecosystem and Human Well-Being: Wetlands and Water Synthesis*. World Resources Institute: Washington, DC.
Miller LA, Chierica M, Johannessen T, Noji TT, Rey F, Skjelvan I. 1999. Seasonal dissolved inorganic carbon variations in the Greenland Sea and implication for atmospheric CO_2 exchange. *Deep Sea Research Part II – Topical Studies in Oceanography* **46**: 1473–1496.
Mitchell RC, Carson RT. 1989. *Using Surveys to Value Public Goods: The Contingent Valuation Method*. Resources for the Future: Washington DC.
Möller I, Spencer T. 2002. Wave dissipation over macro-tidal salt marshes: effects of marsh edge typology and vegetation change. *Journal of Coastal Research* **I36**: 506–521.
Möller I, Spencer T, French JR, Leggett DJ, Dixon M. 2001. The sea-defence value of salt marshes: field evidence from north Norfolk. *Journal of the Chartered Institution, Water and Environment* **15**: 109–116.
Möller I. 2006. Quantifying salt marsh vegetation and its effect on wave height dissipation: results from a UK East coast salt marsh. *Estuarine Coastal Shelf Science* **69**: 337–351.
Munro C, Nunny R. 1998. Isles of Scilly subtidal habitat and biotope mapping survey. *English Nature Research Reports No. 276*.
Neumeier U. 2007. Velocity and turbulence variations at the edge of salt marshes. *Continental Shelf Research* **27**: 1046–1059.
Norgaard RB, Howarth RB. 1991. Sustainability and discounting the future. In *Ecological Economics: The Science and Management of Sustainability*, Constanza R (ed.). Columbia University Press: New York; 88–101.
Nottage AS, Robertson PA. 2005. *The Salt Marsh Creation Handbook: A Project Managers Guide to the Creation of Salt Marsh and Intertidal Mudflat*. The RSPB, Sandy and CIWEM: London.
Nybakken JW, Bertness MD. 2003. Shallow water subtidal benthic associations. In *Marine Biology: An Ecological Approach* (6th edn), Pearson Benjamin Cummings Publications: San Francisco, California, USA; 579.
Penhale PA. 1977. Excretion of dissolved organic carbon by eelgrass (Zostera marina) and its epiphytes. *Limnology and Oceanography* **22**: 400–414.
Pethick JS. 1992. Salt marsh geomorphology. In *Salt Marshes Morphodynamics, Conservation and Engineering Significance*, Allen JLR, Pye K (eds). Cambridge University Press: Cambridge; 41–62.
Pingree RD, Mardell GT. 1986. Coastal tidal jets and tidal fringe development around the Isles of Scilly. *Estuarine, Coastal and Shelf Science* **23**: 581–594.
Pye K. 1992. *North Norfolk Marshes*. In *Salt Marshes Morphodynamics, Conservation and Engineering Significance*, Allen JLR, Pye K (eds). Cambridge University Press: Cambridge; 148–173.
Russel G. 1968. List of marine algae from the Isles of Scilly. *British Phycological Bulletin* **3**: 579–584.
Sagoff M. 2002. On the value of natural ecosystems: the catskills parable. *Politics and the Life Sciences* **21**: 19–25.
Sand-Jensen K. 1975. Biomass, net production and growth dynamics in an eelgrass (Zostera marina L.) population in Vellerup Vig, Denmark. *Ophelia* **14**: 185–201.
Sheppard C. 2006. The muddle of biodiversity. *Marine Pollution Bulletin* **52**: 123–124.
Short FT, Neckles HA. 1999. Effects of global climate change on seagrasses. *Aquatic Botany* **63**: 169–196.
Sims PA (ed.). 1996. *An Atlas of British Diatoms*. Biopress Limited: Bristol.
Slee B. 2007. Social indicators of multifunctional rural land use: the case of forestry in the UK. *Agriculture Ecosystems and Environment* **120**: 31–40.
Smyth T, Tilstone GH, Groom SB. 2005. Integration of radiative transfer into satellite models of ocean primary production. *Journal of Geophysical Research* **110**: C10014.
Spurgeon JPG. 1992. The economic valuation of coral reefs. *Marine Pollution Bulletin* **24**: 529–536.
Stern N. 2006. The economics of climate change. Part II: The impacts of climate change on growth and development. Report produced for Her Majesty's Treasury, available online http://www.hm-treasury.gov.uk/independent_reviews/stern_review_economics_climate_change/stern_review_report.cfm Accessed October 2008.
Suzuki Y, Fujii M, Casareto BE, Furuta A, Ishikawa Y. 2003. CO_2 sequestration and fate of organic matter within seagrass (Zostera marina) ecosystem. *Journal of Chemical Engineering of Japan* **36**: 417–427.
Thom R. 1996. CO_2 enrichment effects on eelgrass (Z. marina) and bull kelp (Nereocystis Luekena). *Water, Air and Soil Pollution* **88**: 383–391.
Tol RSJ. 2005. The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. *Energy Policy* **33**: 2064–2074.
Turner KT, Paavola J, Cooper P, Farber S, Jessamy V, Georgiou S. 2003. Valuing nature: lessons learned and future research directions. *Ecological Economics* **46**: 493–510.
Watanabe K. 2001. Microorganisms relevant to bioremediation. *Current opinion in Biotechnology* **12**: 237–241.
Widdows J, Brinsley M. 2002. Impact of biotic and abiotic processes on sediment dynamics and the consequences to the structure and functioning of the intertidal zone. *Journal of Sea Research* **48**: 143–156.
Willenbrink J, Rangoni-Kubbeler M, Tersky B. 1975. Frond development and carbon dioxide fixation in Laminaria hyperborean. *Planta* **125**: 161–170.
Wium-Andersen S, Borum J. 1984. Biomass variation and autotrophic production of an epiphyte-macrophyte community in a coastal Danish area: I. Eelgrass (Zostera Marina L.) biomass and net production. *Ophelia* **23**: 33–46.
Zieman JC, Wetzel RG. 1990. Productivity in seagrasses: methods and rates. In *Handbook of Seagrass Biology: An Ecosystem Perspective*, Phillips RC, McRoy CP (eds). Garland STPM Press: New York; 87–116.

