Valuing the UK Marine Environment – an Exploratory Study of Benthic Ecosystem Services

Appendix III – Economic Literature Review

For Defra

June 2014

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List of abbreviations and acronyms

BOD	Biochemical Oxygen Demand
CE	Choice Experiment
CICES	Common International Classification of Ecosystem Services
C	Carbon
CVM	Contingent Valuation Method
EEZ	Exclusive Economic Zone
ES	Ecosystem Service
FOCI	Features Of Conservation Importance
GVA	Gross Value Added
HEI	Higher Education Institute
MA	Millennium Ecosystem Assessment
MENE	Monitor of Engagement with the Natural Environment
MPA	Marine Protected Area
NERC	Natural Environment Research Council
NPV	Net Present Value
NUV	Non-use value
pMPA	Possible Marine Protected Area
pppd	per person per day
PSEG	Productive Seas Evidence Group
rMCZ	Recommended Marine Conservation Zone
SAC	Special Area of Conservation
SP	Stated Preference
TCM	Travel Cost Method
TEV	Total Economic Value
UKMMAS	UK Marine Monitoring and Assessment Strategy
UK NEA	UK National Ecosystem Assessment
UK NEAFO	UK National Ecosystem Assessment Follow-On
UNEP	United National Environment Programme
VT	Value Transfer
WTP	Willingness to Pay

1 Introduction

This is a review of existing valuation evidence on marine resources. It focuses, where possible, on the benthic environment. The aim of this review is to highlight the extent of the valuation of marine ecosystem services (ES) in the literature to inform a 'valuation strategy'. This 'valuation strategy' will consist of a combination of (i) value transfer to value benthic ES and (ii) recommendations where possible for primary valuation where there are gaps in the valuation literature.

Turner et al. (2013) define eight ES of interest in the context of marine ecosystems. These are broken down by provisioning, regulating and cultural ES. This classification of ES along with others from sources such as the UK NEA (Austen et al., 2010) and the Common International Classification of ES (CICES, 2011) were used to devise the list of ES that are considered for this study (see Appendix I). The list of ES assessed in this study then forms the structure of this literature review which considers provisioning, regulating, supporting and provisioning ES in the context of marine resources[1].

2 Provisioning ecosystem services

The provisioning ES considered in this study are food (fish, shellfish and seaweed) and other biological resources such as fertilizer and biofuels.

The 2010 Charting Progress 2 report by the Productive Seas Evidence Group (PSEG) for the UK Marine Monitoring and Assessment Strategy (UKMMAS) community (2010) provides an overall assessment of socio-economic activities that take place in the UK's marine environment as well the trends and impacts of these activities.

2.1 Food: fish, shellfish and seaweed

The overwhelming majority of economic aquaculture (i.e. the farming or culturing of organisms in the marine environment) is related to fish and shellfish with an increasing culture of using seaweed and marine worms for bait (UKMMAS, 2010).

The turnover from aquaculture in the UK in 2007 was £350 million per year equivalent to a gross value added (GVA) of £193 million per year (UKMMAS, 2010). Further, the processing of fish from aquaculture accounted for a GVA in 2007 of £105 million. In terms of the trend in aquaculture, it has more than doubled from 2000 to 2006 (UKMMAS, 2010).

Landings from the UK commercial fishery amounted to 611 thousand tonnes of fish and shellfish in 2007 amounting to nearly £650 million per year (UKMMAS, 2010). Out of total landings for commercial fishery in the UK, eight regions in the UK were responsible for nearly 80% of the catch resulting in a GVA of £204 million in the same year. The total value of this market is mainly broken down into (i) shellfish (40%), (ii) demersal fish (40%) and (iii) pelagic species (20%). Table 2.1 shows the breakdown of volume and value from fisheries for Scotland, England, Wales and Northern Ireland.

[1] See Appendix I in the main report for the full list of ES considered.

Table 2.1 Production and total value of cultured shellfish species in the UK in 2007 (UKMMAS, 2010).

	Scotland	England	Wales	N Ireland	Total
Total volume (Mt/yr)	5.1	3.9	10.0	8.4	27.4
Total value (£m/yr)	5.1	4.5	7.5	5.8	22.9

Mardle et al. (2002) examine the fisheries sectors in the UK, France, Spain and Denmark. In the UK, they focus specifically on the English Channel where 40 key species are considered. The overall study spanning four countries estimates that, on average, between 1993 and 1995, 37 species contributed more than €1 million per year to the value of fisheries.

More recently, Austen et al. (2010) present an assessment of fish landings for the UK as part of UK National Ecosystem Assessment (UK NEA). They estimate the gross value of these landings to be £596 million and turnover worth £327 million and £23 million from finfish and shellfish farming respectively. Crilly and Esteban (2013) look at cod fisheries in the UK and estimate a total value of landings worth £12 million for trawlers between 2006 and 2008.

Overall this ES is reasonably well-covered by the economic valuation literature with some additional market data available (e.g. for demersal fish species).

2.2 Other biological resources: fertilizer, biofuels

In the marine environment, seaweeds can be used to produce fertiliser and biofuels. The few studies that focus on the benefits of using seaweeds in these ways tend to consider multiple biological resources together rather than specific types such as fertiliser and biofuels. In effect, there are currently no UK studies on goods extracted from the marine environment other than those studies focusing on (shell-) fisheries and aquaculture.

Beaumont et al. (2006), for example, look at estimates of the turnover from seaweed harvesting as a source of information. They report that the turnover of seaweed harvesting is worth an estimated £0.41 million per year in 2008 based on inflating values from 1994.

Despite the availability of market data on other marine provisioning ES, the valuation literature is extremely limited in its coverage of other biological resources (e.g. fertiliser and biofuels). This significantly limits the scope for value transfer to be conducted with respect to this ES.

3 Regulating ecosystem services

3.1 Natural hazard protection

Erosion control

Bateman et al. (2001) is the only UK study focused on the valuation of erosion control and coastal protection management in England since 2000. The study assesses recreation values of the freshwater Cley Marshes Natural Reserve in the UK which are protected by a shingle bank. The method involves a combination of the travel cost and contingent valuation methods. The potential for value transfer of the values from this study is limited. This is because of the iconic value of this site, and the values can only be applied to shingles protecting freshwater marshes.

Other than the study by Bateman et al. (2001), no studies have been conducted to value the benefits of erosion control in the UK in the context of marine resources. It follows that value transfer is extremely limited due to a lack of studies focusing on habitats which are relevant to the benthic environment.

Sea defence

The economic valuation literature has tended to focus on sea defence in habitats which are not strictly benthic with the majority of research efforts focusing on saltmarshes. Whilst saltmarshes are not strictly part of the benthic environment, their ability to provide sea defence as an ES is impacted by the health of the benthic environment.

The valuation of sea defence by saltmarshes is well-researched in the literature. Notably, Andrews et al. (2006) value the benefits from sea defence by saltmarshes and mudflats in the Humber estuary, UK in the form of replacement costs. They estimate that saltmarsh and mudflat creation would save £0.8 million to replace hard defence with an additional saving from avoided maintenance costs.

Beaumont et al. (2010) value four coastal margin and marine ES in the UK including disturbance prevention by coastal margins. Using results from King and Lester (1995), Beaumont et al. (2010) estimate the cost of replacing UK saltmarsh with man-made sea defences to be in the order of £5.5 - £9.7 billion. There are multiple issues associated with the method used to derive this estimate. For example, replacements costs do not factor in the value of land which is being protected through the provision of sea defence as a regulating ES. Further, the replacement cost approach does not factor in the full economic value associated with the benefits of ES provision. This approach does not make the distinction between the benefits of sea defence from saltmarshes and from man-made structures. Man-made structures in place of saltmarshes may result in the loss of other ES provided by saltmarshes (e.g. carbon sequestration and nutrient cycling).

Overall, the coverage of the economic valuation literature is moderate for natural hazard protection as an ES. There are several potential studies to conduct VT from particularly for the benefits of sea defence. However, the focus on mostly coastal natural hazard protection will require adjustments to be made if possible.

3.2 Climate Regulation

The definition of climate regulation used in this study is taken from the UK NEA (Austen et al., 2010). Climate regulation entails a series of biogeochemical processes regulated by living marine organisms. There is minimal quantitative evidence available on the stock of carbon currently stored within the marine habitat and also on the value of climate regulation as an ES. Further, studies that attempt to cover this ES tend to focus on valuing the sequestration of carbon by non-benthic habitats.

Duarte et al. (2005) estimate carbon sequestration rates for seagrasses globally. They find a C sequestration rate of 0.83 tC/ha/year. McLeod et al. (2011) also estimate the C sequestration rate of seagrass beds at a global level. They find a rate of 1.38 ± 0.38 tC/ha/year. Kennedy et al. (2010) focus on 88 locations (including the UK) and find a C sequestration rate of 0.41-0.66 tC/ha/year.

Beaumont et al. (2006) provide a valuation (in terms of Total Economic Value (TEV)) of the goods and services resulting from marine biodiversity in UK waters to inform further protection of marine biodiversity in UK waters.

Beaumont et al. (2010) value four coastal margin and marine ES in the UK including climate regulation. Given the gap in data availability highlighted above, these authors use average annual primary production (carbon sequestered by phytoplankton) in the UK shelf seas as a proxy for carbon sequestration by marine habitats. Their study therefore focuses on the euphotic zone (below the pelagic zone). They find that there is significant annual variation in the C sequestration rate by phytoplankton in the UK from 1960 to 2004. Thereafter, the rate of C sequestration does not present a discernible long term trend. Variation in the temperate coastal area that they consider is driven by light and nutrient supply. They combine these data with the carbon price to calculate an estimated value of carbon sequestration in UK waters since 2004. The estimated value of carbon sequestration in the UK shelf sea was £7 billion in 2004 and is expected to be £32 billion in 2050 (assuming a 'medium' carbon price).

As stated above, the study by Beaumont et al. (2010) does not focus on C sequestration by benthic habitats and uses a proxy which does not provide an accurate estimate of marine C sequestration.

Focusing on the C sequestration of non-benthic habitats is problematic in two respects:

1. On the one hand, valuing C sequestration in shallow zones is an underestimate because it excludes the contribution to this ES of macroalgae and benthic microalgal production on intertidal sand and mudflats especially within estuaries. Macroalgae plays a particularly significant role in marine C sequestration. The study by Davis (2007) uses the spatial mapping of data on net annual photosynthetic fixation values for kelp, seagrass and phytoplankton to value C sequestration in the Isles of Scilly. They find a rate of C sequestration of 136,405 tC per year with macroalgae playing a significant role in the provision of this ES.

2. On the other hand, valuing C sequestration in shallow zones is an overestimate of deep sea C sequestration because it focuses on shallow shelf seas when C sequestration refers to carbon being stored in the deep ocean or being buried in the benthic environment. The carbon sequestered in the euphotic zone which is valued by Beaumont et al. (2010) is therefore unlikely to be permanently sequestered in the benthic zone and results in a misleading valuation of this ES.

The value from Beaumont et al. (2010) can however be considered to provide an order of magnitude for the value of C sequestration in the marine environment.

Mangi et al. (2010) assume an average social cost of carbon £9.4 per tC in their study. They estimate the NPV over 100 years of C sequestration by kelp forests, seagrass beds and phytoplankton to be £25.4 million (5% discount rate) and £80.6 million (1% discount rate).

More recently Mangi et al. (2011) use estimates from Wium-Anderson & Borum (1984) for the Isles of Scilly in the UK. Mangi et al. (2011) provide NPV for seagrasses, kelp forests and phytoplankton together. However, in Schaafsma and Turner (2013), a carbon price of USD 5 – USD 23 USD/tC used to find the value of the benefits from C sequestration for 3.1 km^2 of seagrass (8.14 tC/ha/yr) which amount to USD 12,600 – USD 57,960 per year. Mangi et al. (2011) also estimate C sequestration by kelp forests. An area of 23.5 km^2 of kelp is assumed to have an annual production using 800 gC/m^2 (8 tC/ha) resulting in an annual production of 18777 tC/yr. The carbon sequestration rate is also overestimated in this instance.

Overall, a review of the literature on the value of C sequestration in the marine environment shows that there are no studies that focus on deep sea C sequestration. Notably, data on the permanence of storage, the sequestration by macroalgae and benthic microalgae and the total stock of C stored in the marine ecosystem are missing.

Further, there a no studies which currently focus on cold water coral reefs, rocky bottoms and the open ocean with further research needed for the valuation of climate regulation by seagrasses and kelp forests (UK NEAFO, Turner et al, 2013).

3.3 Waste breakdown and detox

Waste breakdown is a regulating ES concerning water and purification as well as treatment of waste arising naturally as a result of human action (CICES, 2011).

Mangi et al. (2011) use the avoided cost approach to value coastal zone water purification in the Isles of Scilly. They find that treating 1 kg BOD (Biochemical Oxygen Demand) of sewage is £2.39 amounting to £259,365 per year for sewage treatment. It is worth noting that this avoided cost is dependent upon the available sewage treatment technology which is deemed equivalent to the natural treatment of waste in marine ecosystems. Further, the waste impurity being treated, which is highly dependent on location and on waste-generating activities, will affect replacement costs.

Overall, this ES is not particularly well-covered by the economic valuation literature. Further, of the very few available studies, most tend to value waste breakdown and detoxification based on the avoided costs of treating a specific form of waste which is spatially dependent.

4 Supporting ecosystem services

4.1 Nutrient cycling

Although nutrient cycling is a central marine ES, the valuation literature's coverage of this supporting ES has been extremely limited. To date there are two studies which could potentially be used. But any value transfer exercise will require very strong assumptions and will present multiple limitations.

The first study by Beaumont et al. (2006) uses a TEV approach to value. This is discussed in more detail in Section 6. It is worth noting here that the applicability of the results from their study is limited due to the method used to value the benefits of nutrient cycling. In effect, the study values nutrient cycling as an ES in the form of replacement costs. The result is a value of £800 - £2,320 billion in 2004 prices which is not particularly representative of the value of nutrient cycling as this ES cannot effectively be replaced.

The second study by Shepherd et al. (2012) provides useful information on the effects and economics of managed realignment on the cycling and storage of nutrients. Clearly, the focus of this study on the example of managed realignment in the Blackwater estuary in the UK renders the applicability of this study to the benthic context extremely limited. The study can, however, arguably be indicative of how nutrient cycling can impact the benthic environment. A study by Nixon (1982) on the remineralisation and nutrient cycling of coastal marine ecosystems mentions a growing awareness of the relationship between benthic and pelagic communities in coastal waters. The study notes a strong linear correlation between the organic matter produced in the pelagic zone and the amount of organic matter consumed on the bottom in almost all of the coastal environments for which annual data are available. The benthos is thought to consume a large amount of the organic matter produced in the pelagic zone (25% - 50%). This illustrates the fact that nutrient cycling in the pelagic zone, which leads to nutrient consumption in the benthic zone, is an important factor in the health of benthic communities.

4.2 Biologically mediated habitat

The ES termed 'biologically mediated habitat' refers to organisms that provide structured space or living habitat for other organisms through their normal growth in the form of essential feeding, breeding (spawning grounds) and nursery space for other plants and animals which can be particularly important for the continued recruitment of commercial and/or subsistence fish and shellfish species (Beaumont et al., 2007; Austen et al., 2010). As such this ES supports high biodiversity. The biologically mediated habitat also supports the regulation of the biotic environment which includes habitat maintenance.

Despite the importance of this ES in determining the health of the benthic environment, there are no studies to date to value the benefits of the biologically mediated habitat as a supporting ES.

5 Cultural ecosystem services

5.1 Background

Cultural Ecosystem Services identified in the Millennium Ecosystem Assessment (UNEP, 2006) as relevant to marine and coastal environments were tourism and recreation, aesthetic and spiritual services, traditional knowledge and educational and research services (Gee and Burkhard 2010). Other definitions of services considered as important by Beaumont et al. (2007) and the UK NEA Follow On follow very similar definitions of: Cultural heritage and identity, cognitive benefits, leisure and recreation and non-use benefits; or Sense of place, Stewardship Obligation, Recreational, Aesthetics, Education/scientific, Cultural/historical and Spiritual respectively. However, there are distinct problems within the ecosystem services framework in relation to the identification of services which are often "intangible" in their nature (Gee and Burkhard 2010)[2].

Within the ecosystems services and functions framework there are two key issues which limit the identification of cultural ecosystem services associated with the marine environment generally and benthic ecosystem functions more specifically. Firstly, there is a lack of public knowledge and understanding of the functions and services provided by, what is to most, an 'alien' environment. Secondly, there are difficulties associated with identification of cultural services generally, which is exacerbated by the relative paucity of research into the marine environment. Liquete et al. (2013) identify a catalogue of indicators identified from all previous literature for ecosystem services generally from marine environments. They identified that benthic habitats are essentially missing from the literature and also that cultural services were broadly under-represented with identified studies focusing on coastal environments (i.e. beaches, dunes and coastal wetlands).

Of the literature identified as relevant to the current research, there is a significant weight upon tourism and recreation. Whilst the other relevant cultural services can be identified as having value, quantifying that value is not easy given the lack of research focused on these services in these environments.

5.2 Introduction

Benthic systems are relatively unknown to members of the public (Hynes et al. 2013). Whilst the results of a healthy benthic system such as increased species diversity and ecosystem functions such

[2] See Appendix I in the main report for the list of marine ES that this study focuses on.

as nutrient cycling and pollution assimilation can directly impact upon utility, the identification of value of a healthy benthos has previously been determined in relation to more direct values. Of the available literature, only one research project (Hynes et al., 2013) was identified which directly valued cultural services changes from benthic health. However, a range of studies focused upon cultural services which in part depend on the 'health of the seas' in terms of benthic function are present in the literature.

The main focus of the literature is upon the value of recreational visits to the coast and beaches. These values involve both use and non-use values and the perceived quality of the coastal environment is an important determinant of value. However, not all of these values are direct values from the benthic environment so for the purposes of benefits transfer it will be necessary to identify what if any relationship there is between the value of a visit and the underlying benthic health.

The Millennium Ecosystem Assessment (MA) described cultural services as "the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation and aesthetic experiences" (MA 2005a p.29). The MA also identified difficulties in relation to context specificity of cultural benefits in developing a consistent and coherent framework for the assessment of such services. There was little quantitative data beyond measures of volumes of global tourism, leading the MA to conclude that whilst difficult to measure the loss of cultural services are significant for many people (MA 2005a). Every national assessment of the cultural component of ecosystem services faces similar problems with data, partly because the 'subjective' elements of human-nature relationships—supposedly captured in the concept of 'non-material benefits'—have not, to date, been of central concern either to the natural sciences or to economics. Fisher et al. (2008) argue that "couching ecosystem service research within economic theory gives us one way to move to a more structured engagement between biophysical science, social science research, and policy". One important challenge is how to develop a conceptual and/or methodological approach which allows the humanities and more interpretive social science disciplines to make their distinctive contributions to the assessment in such a way as to strengthen the integration of scientific, economic, cultural and socio-political evidence for policy.

5.3 Recreational services

In the NEA the total value of recreational services from the coast and marine environment was estimated to be £17 billion annually. However, this is an estimate of total value for all recreational activities and it is not related in the NEA in any way to the underlying benthic ecosystem services or functions.

Previous meta-analyses

Based upon 79 studies (since the 1990's) Ghermandi and Nunes (2013) present maps of relative values globally for coastal recreation (Figure 5.1).

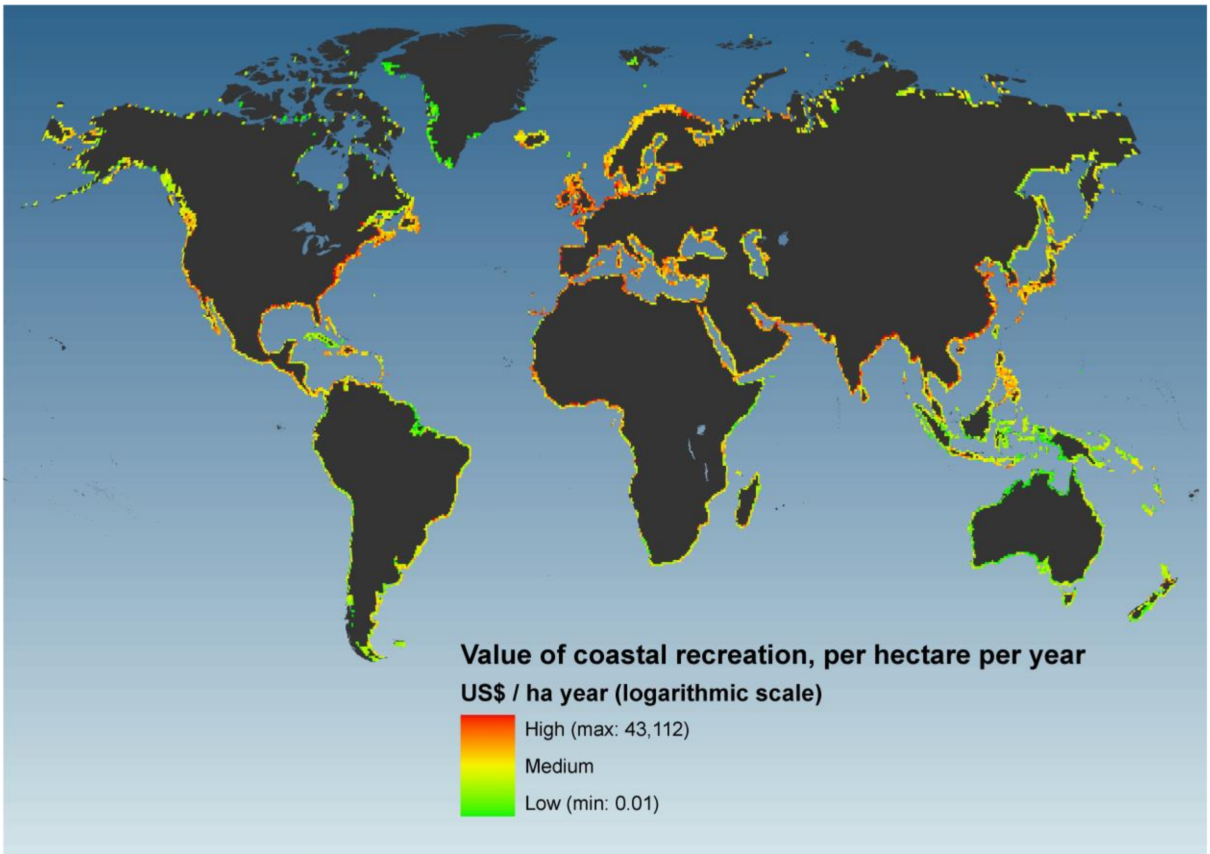

Figure 5.1 Global map of recreational values of coastal ecosystems (Ghermandi and Nunes, 2013).

Values (indexed to 2003 and for purchasing power parity) are distributed over a wide span around an average of 4,698 $/ha/year (±11,283 $/ha/year) and a median value of 453 $/ha/year. Most (60%) fall between 100 and 10,000 $/ha/year (Ghermandi and Nunes, 2013).

In a meta-analysis of the recreational value of coral reefs, Brander et al. (2007) found an average value of 3726 $/ha/year (2000 prices), with values ranging between 0.25 and 57,470 $/ha/year.

Lui and Stern (2009) develop a meta-analysis (based on 39 papers) of coastal and near shore marine habitats and find a mean annual willingness to pay per household for coastal ecosystem services – of £521. However, the median value is found to only be £60, suggesting that there is a skew of a few very highly valued services / studies at the tail impacting on the mean value. Given this and the relatively low number of surveys upon which the meta-analysis is based, they suggest that the errors involved are too high for the results to be used for transfer.

Multi-site surveys

Most of the previous research into recreational values of coastal and marine environments has focused upon the implications of specific policy changes such as the bathing water directive. As such they have tended to focus upon water quality impacts or general values for a visit to the coast. And although some of the value captured in the research may be related either directly or indirectly to benthic ecosystem goods and services the nature of this link is not clear which could inhibit the potential for value transfer. Tables 5.1, 5.2 and 5.3 summarise the results of such studies.

Table 5.1 Review of previous research involved in the identification of values of previous changes to the EU Bathing Water Directive.

Survey	Details	Annual or per visit WTP[3]
Brouwer and Bronda (2004)	CVM - value of for a reduction in illness risk level by 5% for swimmers in Holland.	£27.60
Day et al (2001)	CVM for improvements to the EU mandatory levels for beaches failing the (then) current standards, in South West Scotland.	£6.80 to £15.50
Georgiou et al (1998)	CVM for achievement of the (then) current mandatory standards in East Anglia.	£18.30
Georgiou et al (2000)	CVM to achieve the (then) revised EU standards in East Anglia.	£39.70
Hanley et al (2001)	CVM to achieve the then current EU mandatory standards in South West Scotland.	£10.20 (£0.60 per visit)
Hanley et al (2003)	TCM / SP for an improvement in bathing water standards in South West Scotland.	£7.30 (per individual)
Machado and Mourato (1999)	CVM to achieve the then current EU Mandatory standards in Portugal	£46.20 (£9.20 per trip)
Machado and Mourato (1999)	CVM to achieve the then current EU Guideline standards in Portugal	£117.60 (£15.90 per trip)
Mourato et al (2003)	CE for a 1% reduction in the risk of stomach illness in the UK.	£1.40

*CVM – Contingent valuation method, CE – Choice Experiment, TCM – Travel cost method. SP – stated preference in this case the additional trips which would be made under changed water quality.

Table 5.2 Previous research relating to bathing water quality generally (i.e. not related to the Bathing Water Directive).

Survey	Details	Annual or per visit WTP[3]
Barton (1998)	Improvements to swimmable in Costa Rica	£114.40 – 137.30
Beharry-Borg et al (2010)	CE – Low chance of ear infection related to snorkelers in Trinidad and Tobago	£2.50 – 3.10
Bocksteal et al. (1987)	TCM - Improvement in water quality for Boston, USA	£11.80 for a 10% improvement (£37.40for 30%)
Bocksteal et al (1989)	CVM – improvement from unacceptable to acceptable water standards in Chesapeake Bay, USA	£151.60
Choe et al (1996)	CVM for a shift to swimmable standards in the Philippines	£11.40 – 22.90
Eggert and Olsson (2010)	CE – Bathing water quality, biodiversity and cod stock attributes, Swedish West Coast.	£56 Water, £117 Fish, £59 High Biodiversity

[3] Prices from different studies do not use the same base year in adjusting for inflation.

Survey	Details	Annual or per visit WTP[3]
Feenberg and Mills (1980)	TCM - 10% improvement in water quality in Boston, USA	£4.40
Le Goffe (1995)	CVM - change in water quality to swimmable (and sufficient to safely take shellfish)	£41.70
Mantymaa (1999)	CVM – change in water quality related to sewage levels in Barbados	£13.40-180.50
McConnel and Ducci (1989)	CVM - From swimmable to drinkable in Finland	£38.10
Nikilitshek and Leon (1996)	CVM and TCM – To swimmable in Chile	£101.00 (CVM) £133.40 (TCM)
Sandstrom (1997)	TCM 50% Improvement in terms of nutrients in Sweden	£31.60 – 43.80 per trip values.
Zylicz et al (1995)	CVM – To swimmable (in terms of eutrophication) in Poland	£17.20 – 67.70

*CVM – Contingent valuation method, CE – Choice Experiment, TCM – Travel cost method.

Table 5.3 Values of beach for recreation

Survey	Details	Value[3]
Adamson-Badilla et al. (1997)	Value per user day for beaches in Costa-Rica	£14.20 – 63.90
Bin et al. (2004)	Per user day for beaches in North Carolina, USA	£8.80 to 64.00
Costanza et al. (2006)	Meta-analysis and hedonic valuation - per acre analysis for services from beaches.	£20,671 disturbance control, £11,241 recreation, £18 cultural heritage. Hedonic £24,000 - 55,000
Lew and Larson (2005)	Per beach user day in San Diego County, USA[4].	£19 to 29
Peters and Hawkins (2009)	Meta-analysis of WTP for entrance fees for beaches in marine parks (US).	Up to £2.31 million annually per mile of beach (2010).

In terms of the direct impact of benthic systems there are certain recreational activities likely to be impacted. As identified by Beaumont et al. (2008) and Rees et al (2010), these are diving, recreational fishing and nature watching (both birds and marine mammals). Beaumont (2008) identify the study of Pugh and Skinner (2002) which identifies marine leisure and recreation to be worth an estimated £11.77 billion. However, this is considered to be an overestimate as it included cruising and leisure craft services.

In 2004, sea fish anglers resident in England and Wales spent an estimate £538 million per year and in the 1990's income from whale watching was estimated to be roughly £8 million pounds annually

[4] Also showed a loss of only $1-2 per user day as the result of a particular beach closure assuming other beaches remained open. This highlights differences between average values and marginal changes in service.

and seal watching £36 million. Dickie (2006) showed that certain key sites in the UK for watching seabirds attracted between 250,000 and 400,000 visitors in 2005, taking into account watching in other areas, bird watching is therefore likely to have a significant economic value albeit one which has not been fully quantified (indicative values for certain sites were given). There is a direct link between the value, the populations of sea birds and the benthic ecosystem functions upon which they rely.

Ruiz-Frau et al. 2011 identify values for various recreational users: scuba-divers cost of diving trip £71 per person per day (pppd); sea kayaking £27 pppd; wildlife watching boat trips £44 pppd; and sea bird watching £28 pppd. They found that total value of these "non-extractive recreational activities" in 2008 was estimated as between £21.8 and £33 million.

Kenter et al. (2013) identify the value of MPA's for sea anglers and divers. Headline monetary valuation figures from the study are presented in Table 5.4.

Table 5.4 Headline monetary valuation figures (£ million) (Kenter et al., 2013)

England (rMCZs)		*Scotland (pMPAs)*[1]		*Wales (marine SACs)*	
Sites nominated by Defra for designation in 2013		*Sites not nominated for designation in 2013*		*Total*	
Annual current recreational use value					
Divers	46-76	58-97	104-173	33-56	11-19
Anglers	498-906	1,271-2,311	1,769-3,217	34-61	57-103
*Maximum added recreational use value resulting from restrictions associated with protection**					
Divers† ‡	8-14	12-20	20-34	5-8	1-2
Anglers‡	51-93	136-247	187-340	6-10	10-18
Non-use value that would result from protection					
Divers – base level	26-43	76-127	102-167	20-33	10-16
Divers – max. value restrictions† †† *	30-51	89-148	119-199	22-37	11-19
Anglers – base level	159-289	470-854	628-1,143	105-191	56-97
Anglers – max. value restrictions† †† *	186-339	552-1,004	738-1,343	120-218	62-113

† No potting and gillnetting; ‡ no anchoring or mooring; †† no dredging and trawling.
* The actual value associated with restrictions cannot be calculated because it is uncertain which restrictions will come into place at each site; hence the actual value will lie somewhere between zero and the stated maxima.

The economic value of recreational sea angling in England was also assessed as part of a study for Defra in 2012. The study looked at the number of anglers in England, their total catch, and the economic and social values of sea angling as a recreational activity. Online surveys and face-to-face interviews were conducted at five case study locations – Weymouth, Deal, Liverpool, Northumberland and Lowestoft – with an overall sample size of 11,000 sea anglers. Results are show in Table 5.5 below.

Table 5.5 Headline results on economic value of recreational sea angling (Armstrong et al., 2013)

	Total Spending	Spending excluding imports & taxes	GVA
Direct	£1.23 billion	£831 million	£360 million
Total (direct, indirect and induced effects)	£2.1 billion	NA	~ £980 million

Single-site surveys

Rees et al (2010) identify marine recreation (fishing, diving, charter boats) expenditure based on valuation for Lyme Bay. As such, they identify market values of recreational activities assuming that the consumer surplus associated with the biodiversity hot spot will be captured in the amount paid to partake in the recreational activities. They then relate this value to the closure of the area to dredged shellfish activities. However, the value for a single bay on the South Coast of England of only these marketed recreational activities is found to be circa £18 million per year. It is suggested in the research that much of the value is related to the biodiversity levels and as such is related to benthic health. For example, diving is focused on sites with high biodiversity and fishing relies upon catch. However, the approach does not necessarily identify the specific value of the premium associated with the high levels of biodiversity, as this will only make up an unquantified part of the overall value of the experience.

Specific investigation of the benthic environments

Only one research project (Hynes et al., 2013) was found which investigated the value to the public of the health of benthic systems in relation to recreational activity. The work covered 4 countries / regions: Ireland, Northern Ireland, Scotland and England. To date only results for Ireland and Northern Ireland are available, although the remainder of the data is in the process of analysis. The research utilised a choice experiment to identify public preference for benthic health, health risk from bathing and beach litter management. The benthic health attribute was related to the public in relation to the likely impacts upon overall species diversity including to fish, birds and marine mammals. This simplified representation of the impact of improvement in the quality of benthic habitats was found to be necessary when presenting the topic to focus groups. The attribute had three levels, no change, small improvement in benthic health which would result in higher species diversity and numbers but no increased chance of seeing species and a large improvement in benthic health which would result in higher species diversity and numbers to the extent that there would be a good chance of seeing species. Table 5.6 summarises the results from this study in terms of WTP for changes of differing magnitudes for the various attributes considered in the choice experiment.

Table 5.6 Summary of results in Hynes et al. (2013)

	Northern Ireland All Users	Northern Ireland Active Users	Ireland
Benthic Health – small increase	£4.67 (±£1.03)	£5.40 (±£3.37)	€4.41
Benthic Health – large increase	£5.97 (±£1.03)	£9.34 (±£3.82)	€5.11
Health Risk 5%	£5.36 (±£1.42)	£9.49 (±£5.51)	€3.91
Health Risk – very little	£7.22 (±£1.31)	£13.56 (±£5.49)	€8.58
Debris – Prevention	£7.37 (±£1.01)	£10.77 (±£3.88)	€6.31
Debris – Collection & Prevention	£8.72 (±£1.19)	£12.54 (±£5.00)	€7.04

The study by Hynes et al. (2013) and Tinch et al. (2013) found that there was positive willingness to pay associated with improvements to benthic health of a comparable scale to those for health improvements and beach litter / debris management. The relative value did vary somewhat for different user groups (although the Irish sample only looked at active recreationalists – that is those taking part in activities likely to result in submersion of the face). Surveys were carried out on site during visits to the beach, and values relate to additional willingness to pay to travel to an alternative beach with the given characteristics.

5.4 Bequest, existence and option values

Again, little research relating to the bequest and existence value of benthic systems was found in the literature. However, certain studies which may be indicative of the scale of the value have been found. Hagman (1985) and Loomis and White (1996) estimated the value of protecting various marine mammal species at between £19 and £45 per household per year. Beaumont (2008) draws on these values to give an estimate of between £469 million and £1,136 million per year as an (underestimate) of the total non-use value of UK.

More recently, Jobstvogt et al. (2013) assess the value of biodiversity using a discrete choice experiment focusing on Scottish households' WTP for additional MPAs in the Scottish deep-sea[5]. The study examines two specific dimensions of biodiversity as an ES: (i) the existence value of deep-sea species measured by the number of protected species[6] and (ii) the option value of deep-sea organisms as a source of future medicinal products. As such, the study is one of the first to specifically focus on deep sea habitats.

The Jobstvogt et al. study determines the value of deep sea environments off the UK coast to be £34.83 per household per year for a high level of species protection. In addition, the choice experiment includes an attribute for the potential medical research potential of the genetic diversity of the habitats with a WTP value per household of £35.43 per year. Adopting the methodology of Beaumont et al (2008) of applying these values to the total number of households

[5] The study focuses specifically on the deep-sea area of the UK's North and Northwest Exclusive Economic Zone.
[6] Expressed as a change of species numbers between 0% and 60% based on the scientific literature with a maximum of 1600 species compared to the hypothetical baseline of 1000 species.

gives a value of approximately £850 to £900 million per year for both non-use and option values for UK marine habitats generally, which is a safe underestimate.

This is a growing area of research with studies offering the potential for further value transfer to be conducted in the future. For example, one such study by Aanesen et al. (forthcoming) looks at WTP for the protection of cold-water corals (CWC) off the Norwegian coast by the Norwegian population. For the same respondents, an individual-response and a group-response survey on the willingness to pay (WTP) for protection of CWC among the Norwegian population is conducted. Both surveys employ strategies to reduce unfamiliarity with the good before eliciting values. Results from the study indicate that giving people time to think and discuss the good to be valued in small groups (4-6 people) does not necessarily change median WTP, even for such an unfamiliar resource. Economic values from these surveys can inform coastal and fisheries authorities when developing management plans for deep-sea areas.

5.5 Spiritual and cultural wellbeing

In a recent analysis of data collected in the Monitor of Engagement with the Natural Environment (MENE) White et al. (2013) found that the coast was the natural environment which contributed most to a self-reported feeling of restoration during the visit. That is, a visit to the coast was found to contribute more to feelings of being "calm and relaxed" and "refreshed and revitalised". These results are indicative that the coast and marine environments are important environments in terms of the spiritual, aesthetic and cultural well-being they provide to the public. However, these impacts are not easily quantifiable and arguably not specifically related to the benthic environment.

In terms of religious elements of the coastal environment, Chapter 16 of the UK NEA (Church et al., 2010: 637) identifies that the contribution of coastal habitats to spiritual and religious experiences at holy islands of the UK (Iona, Lindisarne and Bardsley) are not unique in terms of other popular sites of pilgrimage. Suggesting a link to these cultural services and benthic ecosystems would appear even more tenuous given this argument that the location of these sites is more a result of historical fate rather than continuing services from the marine environment. However, sediment supply dynamics to predominantly dune systems have some religious or cultural significance. Machair environments and the Moray Firth are certainly examples but again the examples tend to be extremely site-specific and, as such, the capacity for consideration of benefits transfer, even if values were available, is extremely limited or non-existent.

5.6 Education and research

Education, training and research in relation to marine science is significant. In 2002, Pugh and Skinner (2002) valued education and training at £24.8 million with marine research funding being estimated to be worth £292 million value added to the economy. There can be little doubt that funding has increased over the intervening years. Pugh (2008) estimate gross value added of research and development to be £426 million. They found higher education institutions (HEI) were responsible for £31 million, under a tenth of the value from industry at £330 million (although this captures very significant values from the ship building and resource extraction sectors of the economy). NERC alone is currently investing £50 million each year into marine research. However, up to date total expenditure could not be found nor could up to date multipliers with which to calculate value added.

The UKMMAS's (2010) PSEG estimate the turnover from education and training undertaken in the marine environment to be £132 million with a GVA of £95 million. This is not synonymous with education and research as it is thought to underpin multiple other principal activities such as leisure, research and development. The figures do however, provide an order of magnitude for what the value of the ES could be in the context of the marine environment. Although, again, identifying values associated with specific habitat types is not possible from these general marine values currently available in the literature.

5.7 The value of MPA habitat types

A recent case study (Kenter et al., 2013) for the UK NEAFO identified the value of Marine Protected Areas using a sample of 1075 sea anglers and divers identified using a choice experiment on the value of various sea bed habitats. While the results of this study may be relevant to the value of the benthic health of the identified habitat types, consideration should be given to (i) whether participants were aware of the full range of possible habitat types (18 habitat types were considered in total), and (ii) whether the design of the choice experiment was sufficiently efficient.

The results from Kenter et al. (2013) relate to separate models for anglers and divers. The main findings from the study are as follows:

- Travel distance as a measure of cost showed a negative sign and was highly significant in both angler and diver models;
- The chance of finding specimen fish was one of the most important aspects of recreation to anglers (£23.58 in travel costs per mile). Consideration was also given to FOCI (Features Of Conservation Importance) habitats;
- Restriction of dredging and trawling or anchoring and mooring does not influence recreational preferences, while restriction of potting and gillnetting is favoured (WTP £4.76),
- Amongst divers, knowing that species would be protected, even whilst the chance of encountering was very low was highly valued (WTP £0.44 per species) as was the presence of large fish (WTP £7.64); and
- Preferences for habitats were more evenly spread with more habitats being favoured by divers than anglers.

6 Bundled ecosystem services

Some studies in the valuation literature assess multiple ES together. Such valuation studies can look at the value of a group or 'bundle' of services in contrast to examining the values of individual ecosystem services. This idea has been present in environmental-economic valuation literature for some time. For example, Luck et al. (2009) discuss the idea of extending the concept of organisms or communities contributing to a single ES to their contribution to bundles of services (i.e. multiple services that are provided by a collection of organisms.

Most, but not all, such studies focus on the valuation of biodiversity as an ES. There are multiple way of classifying biodiversity as an ES. It can be thought of as a supporting service that underpins the provision of all ES flows from the benthic environment. However, in this study, biodiversity is viewed as a cultural ecosystem service.

Beaumont et al. (2006) is an example of a study that values of biodiversity in the UK by breaking it down into goods and services which are affected by declines in marine biodiversity. The table below (Table 6.1) outlines the main results which overlap with the benthic ES considered in this study.

Table 6.1 ES contributing to the TEV of UK marine biodiversity (Beaumont et al., 2006)

Good/Service	Value (£ per year in 2004)	Valuation method	Underestimate	Overestimate
Food provision	£513 million	Market	x	
Raw materials[7]	£81.5 million	Market	x	
Disturbance prevention and alleviation	£0.3 billion[8]	Avoidance	x	
Gas and climate regulation[9]	£0.4 - 8.47 billion	Avoidance	x	
Leisure and recreation	£11.8 billion[10]	Market		x

The table above does not mention the value of nutrient cycling in Beaumont et al. (2006) which is derived using the replacement cost method (£800 - £2,320 billion in 2004 prices) because in reality, the replacement of this ES is not possible. In addition, if all nutrient cycling stopped, the marine system would break down thereby arguably making this value meaningless. Estimates presented in Beaumont et al. (2006) provide a rough indication of the value of certain benthic ES. However, these values should be considered as mere components of total goods or services that can be valued with currently available data. Further, some values are derived using value transfer based on US studies extrapolated to the UK in the absence of UK or European data. It is also worth noting that the various methods used in calculating the value of biodiversity in the marine environment pose problems when it comes to aggregating values.

McVittie and Moran (2008) estimate the use and non-use value of marine biodiversity in the UK. The study derives a primary estimate of the benefits derived from the implementation of the nature conservation measures in the draft Marine Bill, specifically, Marine Conservation Zones (MCZs). A stated preference approach is used to value environmental benefits arising from designations of MCZs described as the value of 'halting biodiversity loss'. The average UK household WTP from the contingent valuation survey was found to be £26.91 and aggregated at the UK level to £698 million. Individual country models are used to produce a more conservative estimate of £487 million per year representing the TEV for the MCZ provisions. McVittie and Moran (2008) suggest that a high proportion of this value is non-use value (NUV).

[7] Defined in Beaumont et al. (2006) as the extraction of marine organisms for all purposes except human consumption.
[8] In addition to £17 - £32 billion capital costs.
[9] Calculated on a per area basis and multiplied up to the UK area.
[10] Based on 2002 values.

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