National Ecosystem Assessment (NEA): Economic Analysis

Coastal Margin and Marine Habitats, Final Report

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1. Definition of study parameters

The research presented within this report is based heavily upon the natural science analysis undertaken as part of the Natural Ecosystem Assessment, specifically the Marine Habitat (Austen *et al.* 2010) and Coastal Margin Habitat (Jones *et al.* 2010) chapters. The remit and scope of this report was therefore significantly bounded by these supporting documents.

1.1 Services included in the economic analysis

The services arising from the coastal margin and marine habitats are considered together to avoid double counting. Jones *et al.* (2010) and Austen *et al.* (2010) provide extensive qualitative detail on the many goods and services which are provided by these habitats. However, the availability of quantitative natural science data varies considerably both between and within these services, and there are not sufficient data to determine the marginal values for all services. To determine which services to include within the economic analysis a prioritisation exercise was undertaken with the natural scientists, and the services were scored on a basis of significance and data availability. Four services were selected as viable for analysis within the time and resource framework of this report:

1. Climate regulation
2. Recreation and tourism
3. Disturbance prevention, including coastal flood defence (coastal margin only)
4. Extractable – food provision (marine only)

Cultural services are covered in a separate section within the economic report.

Only ecosystem services (i.e. biotic services) are considered within this analysis. Ecosystem services either tend to be drawn from stocks which are renewable, or are non-extractive. Environmental services (i.e. abiotic services) such as aggregates and oil and gas are not included within this report, as they are already detailed extensively elsewhere (Pugh 2008, Saunders *et al.* 2010). It is essential to recognise that although some services are not assessed in this report this does not imply an absence of value. Table 1 documents a broad range of services, and their static values, which are attributed to the marine and coastal

Table 1. Review of UK per annum values of goods and services provided by marine and coastal margin habitats, including values of abiotic commercial activities (shaded grey)

Marine and coastal margin services	Beaumont *et al.* 2006 (£million, 2004)	Pugh 2008 (GVA, £million)	Saunders *et al.* 2010 (£ million, 2008)
Extractable biotic resources – Food Provision	£513 (fish)	£808 (fish, including processing, 2004/05)	£520 (GVA, fisheries and aquaculture, excluding processing)
Extractable biotic resources – Raw Materials	£81.5 (fish meal, fish oil and seaweed)	X	£89.41 (Turnover, fertiliser/feed)
Waste breakdown and detoxification	Insufficient data	£364 (2005)	Insufficient data
Climate regulation	£400 – 8470	X	Insufficient data
Disturbance prevention	£300	X	Insufficient data
Aesthetic, inspirational	Insufficient data	X	Insufficient data
Education, research and development opportunities	£317	£478 (2006)	£67 (research funding 2006/7) £95 (education, turnover)
Cultural and spiritual well being	Insufficient data	X	£1000 (stated preference)
Leisure and recreation	£11770	£3326 (2005-6)	£2500 (tourism, GVA) £1960 (leisure boating, turnover) £200 (surfing, turnover, 2007) £800 (Recreational angling , expenditure) £1.8 (Whale watching, expenditure)
Option and non-use values	£500 - £1100 million	X	X
Nutrient cycling	£800,000 – 2,320,000	X	X
Biologically mediated habitat	Insufficient data	X	X
Resilience and resistance	Insufficient data	X	X
Oil and gas	X	£19,845 (2005)	£36,814 (GVA, non-sustainable)
Aggregates	X	£114 (2006)	£31 (GVA)
Cooling water	X	X	£100 (replacement)
Salt	X	X	£4
Ship and boat building	X	£1223 (2004)	X
Marine equipment and materials	X	£3268 (2005)	X
Marine renewable energy	X	£10 (2005-6)	£62 (value of avoided emissions)
Construction	X	£228(2005-6)	X
Shipping operations	X	£3399 (2005)	£7100 (GVA, maritime transport)
Ports	X	£5045 (2005)	X
Navigation and safety	X	£150 (2005-6)	X
Cables	X	£2705 (2005)	Insufficient data
Business services	X	£2086 (2004)	X
Licence and rental	X	£90 (2005-6)	X
Defence	X	£2814 (2005-6)	£300 (GVA)

X – not included in report.

margin habitats. The benefits shaded in grey are not considered to be true ecosystem services, but are abiotic commercially based activities which are based within the marine environment.

An additional study of note applied a choice experiment to undertake a top down valuation of the benefits of marine conservation zones (MCZ) (McVittae and Moran, 2010). The total aggregate value for a policy that halts UK marine biodiversity loss through the introduction of a UK MCZ network was estimated to be £1714 million per annum.

It is of interest that previously all studies have focussed on static values, where as the analysis presented here aims to address the issue of change in provision through quantification of marginal values.

1.2 Spatial definition

Both the Marine and Coastal Margin habitat types are further divided into sub-habitat types. The type and extent of services provided will depend upon the specific sub-habitat type, thus it is important to determine the areas of sub-habitats, how they have changed and how they are likely to change in the future. The coastal margin habitats comprise: sand dunes & sandy beaches; saltmarsh; vegetated shingle & shingle beaches; machair; maritime cliffs & slopes & small islands; saline lagoons. There are good data availability on the areas and distributions of the sub habitats (Table 2), with additional information provided by Jones *et al.* (2010).

There is a downward trend in most of the areas, the causes of which are described in detail by Jones *et al.* (2010). In the case of sand dunes this decline is mainly due to urban expansion, forestry planting, agricultural improvement, tourism e.g. golf and caravan parks, and sea level rise. The decrease in saltmarsh area is primarily due to land grab from agriculture and industry, in addition to sea level rise. The downward trend in the shingle and machair areas is due mainly to erosion and sea level rise. For some sub-habitats the areas have not changed significantly but the specific type has, for example, maritime cliffs (extent measured in km length) remain reasonably constant, but the quality has changed with more cliff armouring that has negative implications for provision of some services, particularly coastal defence via reduced sediment supply. Finally there has been little net change in the area of saline lagoons as construction of artificial lagoons has largely compensated for lagoon area lost elsewhere. Recently increased statutory protection has slowed the rate of loss of coastal margin habitats, but sea level rise and coastal erosion continue to pose a significant threat to many of these habitats.

Table 2. UK areas (ha) of coastal margin sub-habitat. Projections to 2060 in brackets assume no net change in extent (for additional information see Jones *et al.* 2010)

	Area (ha)	**Year**						
		1900	1945	1970	1990	2000	2010	2060
Sand dune	**England**	16996	14446	12407		11897	11778	10707
	N. Ireland	2244	1908	1638		1571	1555	1430
	Scotland	71429	60714	52143		50000	49500	45857
	Wales	11573	9837	8448		8101	8020	7534
	UK	102241	86905	74636		71569	70853	65528
Saltmarsh	**England**		37331			32462	32462	32462
	N. Ireland		288			250	250	250
	Scotland		6900			6000	6000	6000
	Wales		6670			5800	5800	5800
	UK		51189			44512	44512	44512
Shingle	**England**		10046		5081		5023	4822
	N. Ireland		50		50		50	50
	Scotland		670		670		670	670
	Wales		109		109		109	109
	UK		10875		5910		5852	5651
Machair	**England**							
	N. Ireland							
	Scotland		20171				19698	18516
	Wales							
	UK		20171				19698	18516
Saline Lagoons							5184	(5184)
Maritime cliffs and				4554#			(4554)	

slopes								

Cliff extent measured in km length.

The marine habitat is categorised into six broad habitat types: intertidal rock; intertidal sediment; subtidal rock; shallow subtidal sediment; shelf subtidal sediment; deep-sea habitats. Data availability on the area and spatial distribution of these is poor. In the case of the marine environment the spatial data is less essential, as most marine environments deliver most marine ecosystem services, albeit in differing amounts (Austen *et al.* 2010).

1.3 Temporal definition

Where possible a hind cast from 1945-2010 and future projections to 2060 have been provided. The assumptions associated with these trends and projections are individually detailed.

2. Benefits provided

2.1. Climate Regulation (valuing C sequestration)

Biomass and sediments in coastal margins and the marine environment offer the potential for sequestration of greenhouse gases. This sequestration has a value that is associated with the notional damage potentially caused by any release to the atmosphere. An avoided damage cost (i.e. benefit value) can be approximated by the so-called shadow price of carbon that is attributable to tonnes of carbon dioxide equivalent gases[1] sequestered in these areas.

The UK government is one of few authorities to adopt a formal shadow price of carbon as a way of mainstreaming climate change mitigation into policy appraisal and evaluation. The Department of Energy and Climate Change (DECC) guidance recommends that the long term prices of carbon, shown in Table 3, should be adopted.

[1] The release of greenhouse gases from land use (predominantly nitrous oxide, methane and carbon dioxide) is typically expressed in terms of a common global warming potential unit of carbon dioxide equivalent (CO2e).

With regard to hind casting the first date for a carbon price from DECC is for 2020. This was calculated in 2008-9 and the NEA carbon economists assume that given the policy context and technological landscape this can be used as a hind cast to 2004, but no further, with a linear interpolation of the 2020-2030 prices provided in the DECC report.

Table 3. DECC 2009 carbon prices (DECC 2009)

YEAR	carbon price (t/CO2equivilant)*
**2004	£44.00
**2010	£50.00
2020	£60.00
2040	£135.00
2050	£200.00
**2060	£265.00

*central estimate with a range of +/- 50%
** calculated from DECC values assuming a linear interpolation

2.1.1. Carbon sequestration in coastal margin habitats

Carbon sequestration is primarily provided by habitats where rapid soil development or sediment accumulation occurs, primarily sand dune, uncultivated machair and saltmarsh.

Long-term C sequestration rates in soil of the most significant coastal margin sub-habitats have been collated:

Dry dune = 0.58±0.26 t/C/ha/yr i.e. 0.32 – 0.84 t/C/ha/yr (Jones *et al.* 2008)

Dune wet slack = 0.73±0.22 t/C/ha/yr i.e. 0.51 – 0.95 t/C/ha/yr (Jones *et al.* 2008)

Saltmarsh = 0.64 - 2.19 tC/ha/yr (Cannell *et al.* 1999)

Sequestration rates are not available for machair. To simplify calculations, a composite sequestration rate for dune habitats is calculated, assuming rates are similar across the UK. Dune slacks comprise roughly 9 % of UK sand dune habitat (JNCC data ca. 1995) therefore a proportional average for C sequestration in dune habitats, taking the upper and lower bound estimates is 0.34 – 0.85 t C/ha/yr.

Carbon is converted to CO_2 by multiplying the ratios of molecular weights, that is 44/12 or 3.67. Thus, if the carbon sequestration rates are 0.34 - 0.85tC/ha/yr (composite dune) and 0.64 - 2.19tC/ha/yr (saltmarsh) the respective CO_2 sequestration rates will be 1.25 - 3.12 tonnes CO_2 (composite dune) and 2.35 – 8.04 tonnes CO_2 (salt marsh). Combining these tC/ha/yr figures with data on changes in UK areas of the sub-habitats (Table 2) estimates of the changes in the capacity of these habitats to sequester CO_2 can be derived (Figure 1 and Figure 2). There is a loss over time in the capacity of all habitats to sequester CO_2, attributable to a loss in area, with a greater decrease in the sand dune habitats relative to the saltmarsh habitats.

Figure 1. Estimated change in CO_2 sequestration provided by UK sand dunes, 1900 – 2060, applying an averaged sequestration rate

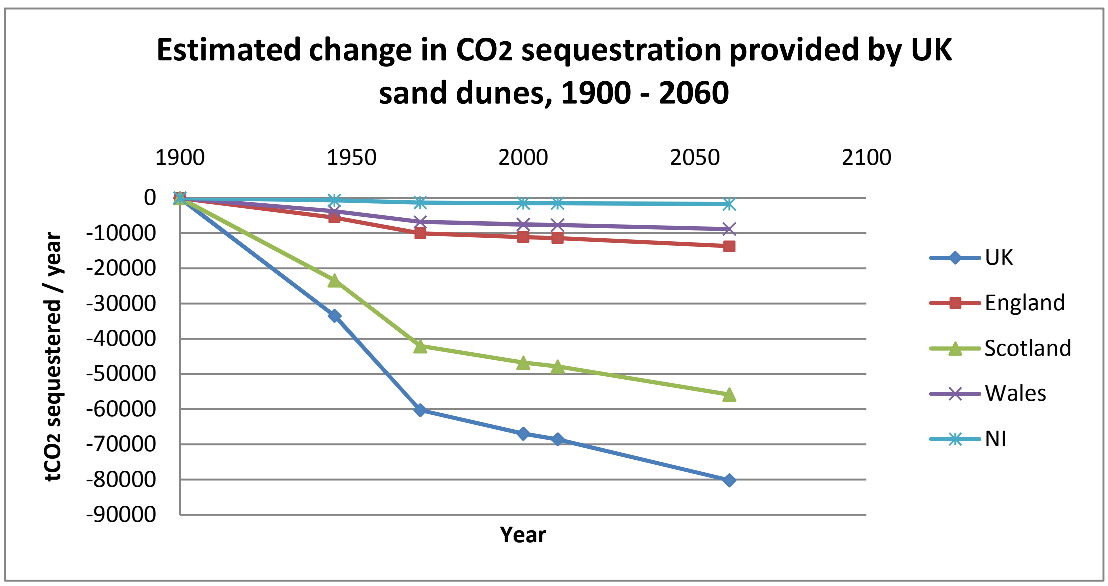
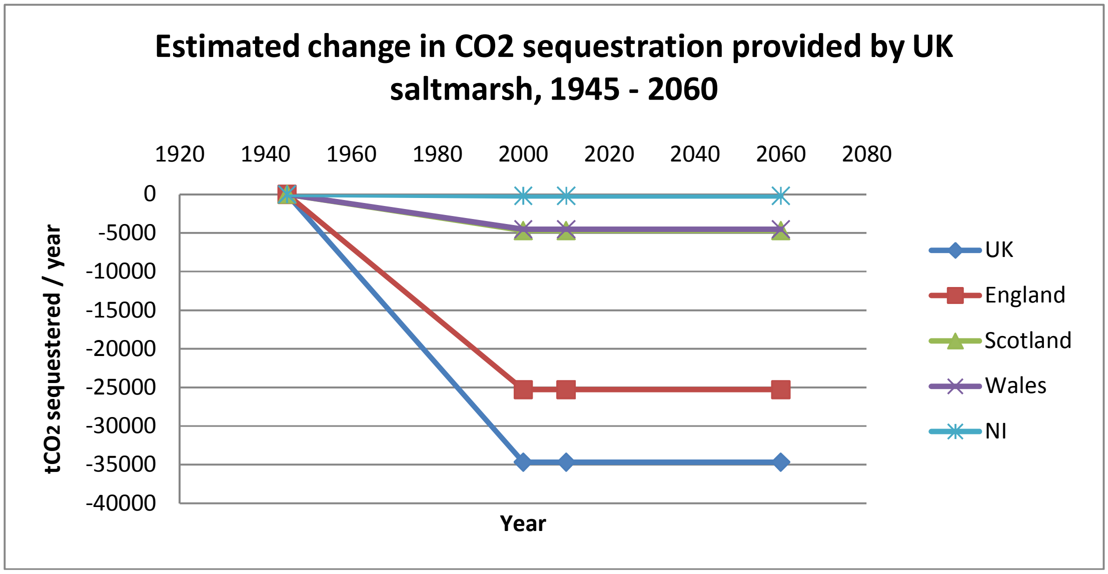


Figure 2. Estimated change in CO_2 sequestration provided by UK saltmarsh, 1945 – 2060, applying an averaged sequestration rate



Combining these sequestration rates with the 2010 DECC CO_2 price (£51.6 +/- 50%), the £/ha/yr values can be derived for the provision of C sequestration by the sub-habitats. For dune, values range from £32.25 ha/yr to £241.49/ha/yr. For saltmarsh, values range from £60.63/ha/yr to £622.30/ha/yr. Combining these £/ha/yr values with data on changes in UK areas of the sub-habitats (Table 2) lower and upper bound estimates of the changes in the value of these habitats can be derived, with regard to their C sequestration potential. Although the natural science data would enable a hind cast of C sequestration potential, the economic data are not available to support this. The results, depicted in Figures 3 and 4, show the value of this service provided by these habitats. The filled markers depict the lower sequestration rates, and the outlined markers the higher sequestration rates. The three different carbon prices (high, medium and low estimates), are also shown. The extent of carbon sequestration service provided by coastal margin habitats decreases over time, as a direct result of habitat loss, but the value of this service increases due to the C prices applied. There is considerable variation in potential values, with the significant driver of the uncertainty being attributable to the variability of the sequestration rate. This variability arises primarily due to climatic factors, soil type, and successional age.

Figure 3. Estimated value of C sequestration service provided by UK sand dunes, 2010 and 2060

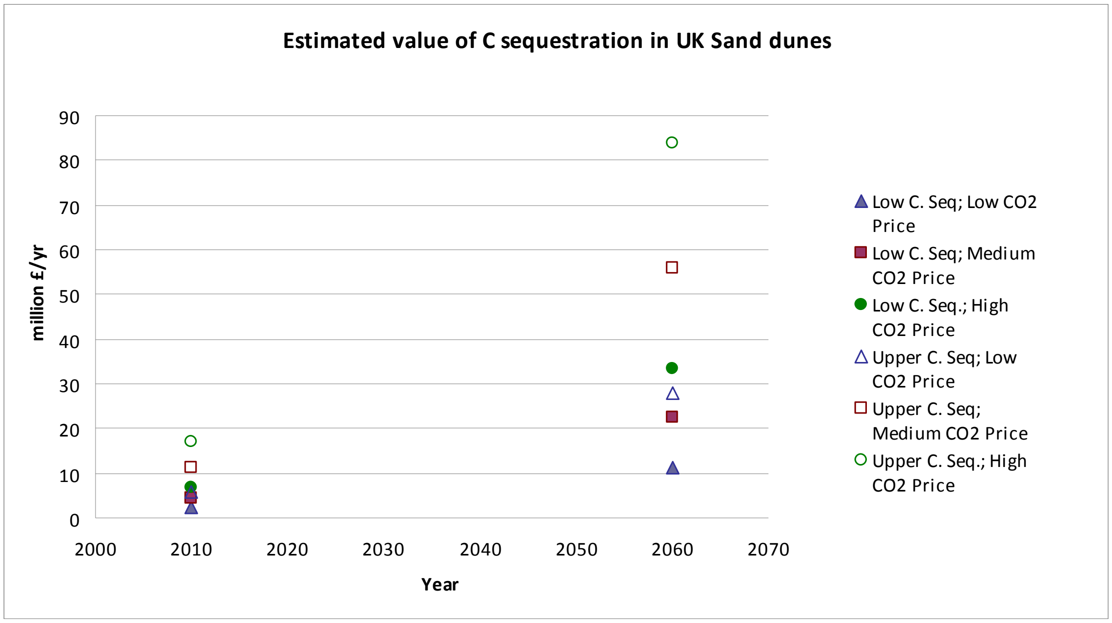
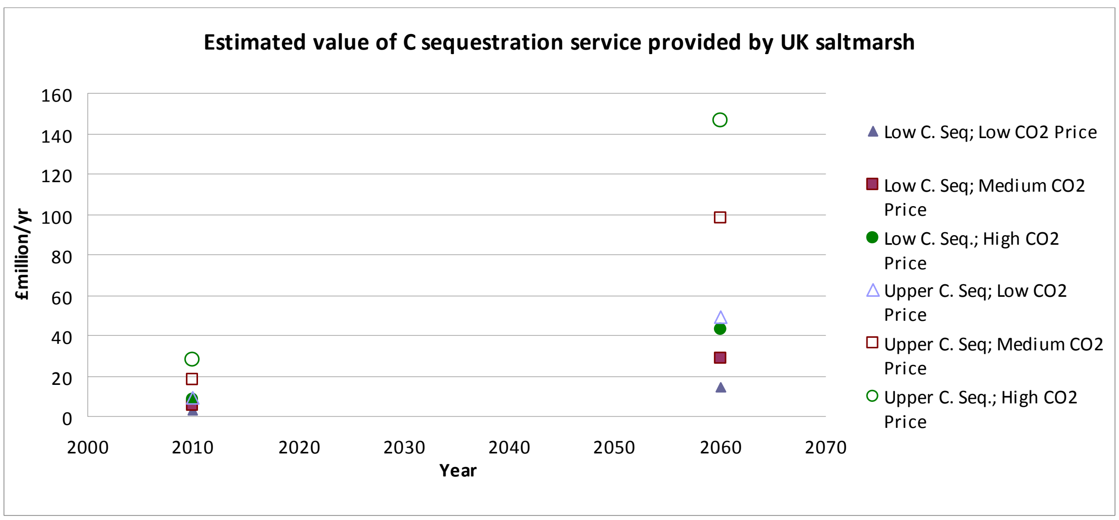


Figure 4. Estimated value of C sequestration service provided by UK saltmarshes, 2010 and 2060



The average (mid carbon price and mid sequestration rate) increase in value of sand dunes, with regard to carbon sequestration service, between 2010 and 2060, is approximately £31million. The average (mid carbon price and mid sequestration rate) increase in value of saltmarsh, with regard to carbon sequestration service, between 2010 and 2060, is approximately £51million. This is despite a loss in area and is attributable solely to the increase in carbon price.

Figures 5 and 6 provide an indication of the total value of the carbon sequestration service provided by saltmarshes and sand dunes, divided by country, and show an increase in the value of this service over time, again despite habitat loss and attributable to the increasing carbon price. The majority of the C sequestration by sand dunes is situated in Scotland, and the majority of C sequestration by saltmarsh is situated in England.

Figure 5. Estimated value of C sequestration service provided by UK sand dunes, divided by country, 2010 and 2060, using mid carbon price and mid carbon sequestration rates.

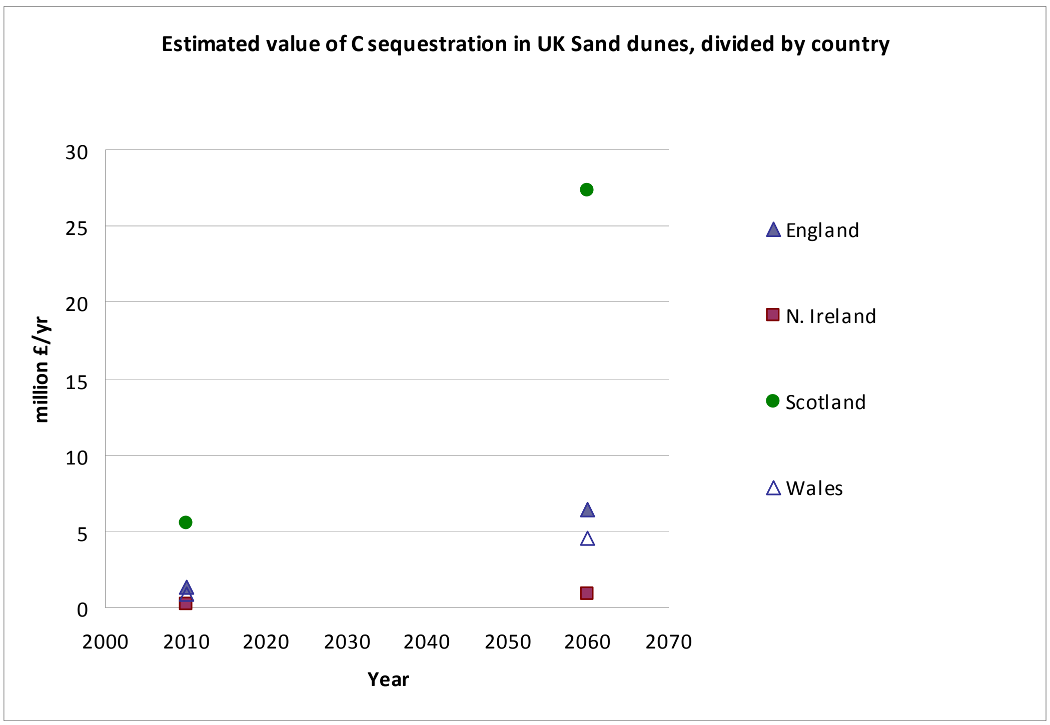
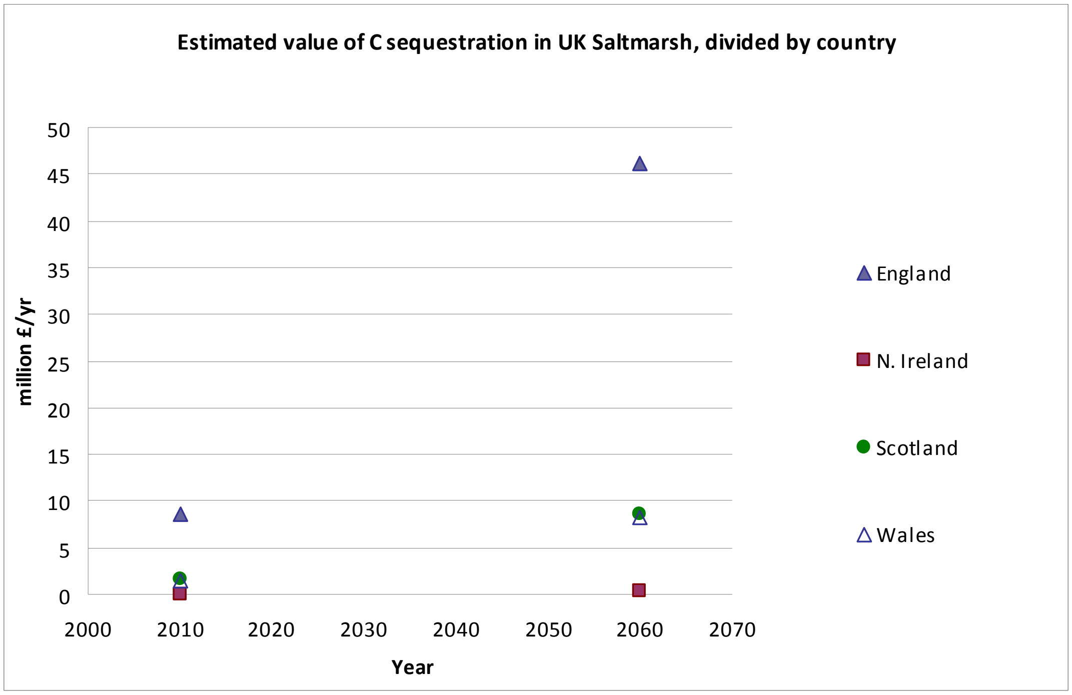


Figure 6. Estimated value of C sequestration service provided by UK saltmarsh, divided by country, 2010 and 2060, using mid carbon price and mid carbon sequestration rates.



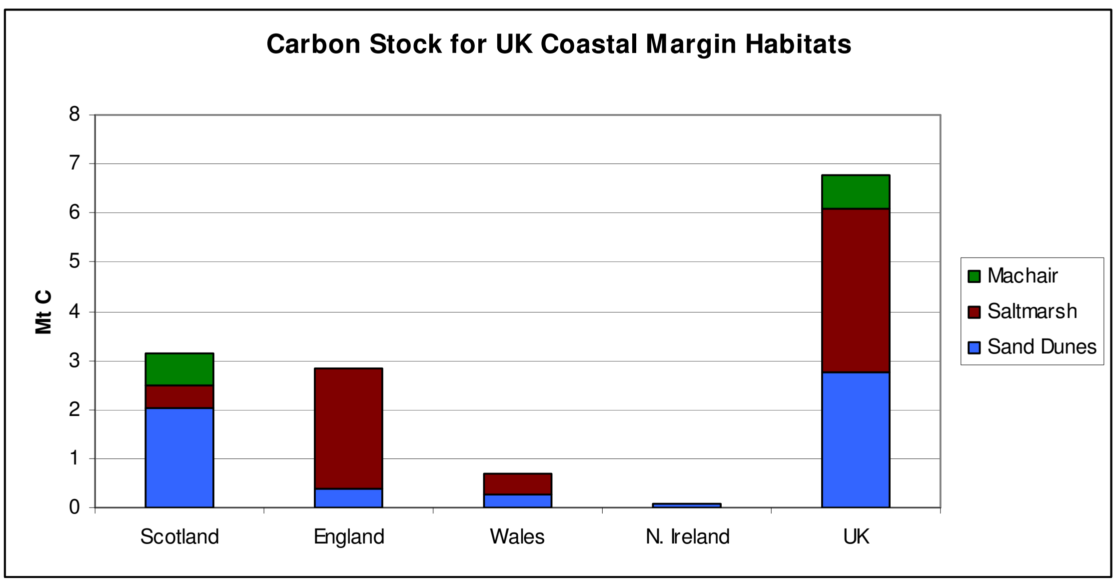
These calculations assume that sequestration rates remain the same over time and UK location. Sequestration rates in dunes vary with successional stage, but the figures used here represent average sequestration rates over a 60-year period. They are for a mid-latitude west-coast site, Newborough Warren, rainfall 850 mm/a, and probably represent an acceptable UK average between slower C accumulation in the dry south and east, and faster accumulation in the wetter north and west UK, so the assumption of transferability of sequestration value is not unreasonable. In the future additional information may become available to show how the carbon sequestration rates vary with factors such as temperature, CO_2 concentrations and UK location, but currently these types of data do not exist.

Details of the basis of the future projections are discussed by Jones *et al.* (2010), but are extrapolated from a range of sources including the sand dune Habitat Action Plan, and academic publications (French, 1997). In both sub-habitats there is a loss of habitat extent over time, with an accompanying decline in C sequestration, but the monetary value of this service increases albeit solely due to the carbon price. However, these are not net values. When a coastal margin habitat is lost it will be replaced with an alternative habitat, which in most cases will have a capacity for C sequestration. In the case of sand dunes, conversion is primarily to urban expansion, forestry planting, agricultural improvement and tourism e.g.

golf. In the case of saltmarsh it is converted to land for agriculture and industry. Ideally, the carbon sequestration rates of the areas of the new habitats would be calculated to determine the overall net change of C sequestered. However, although some of the sequestration rates are available, the areas are not, thus at the present time this is cannot be undertaken due to poor data availability. However, the bulk of sand dune loss occurs to land uses with lower C sequestration rates, and saltmarsh has higher C sequestration rates than all the alternative land uses. Therefore, while the net loss in sequestration cannot be calculated exactly, overall there will be a decrease in the provision of this service.

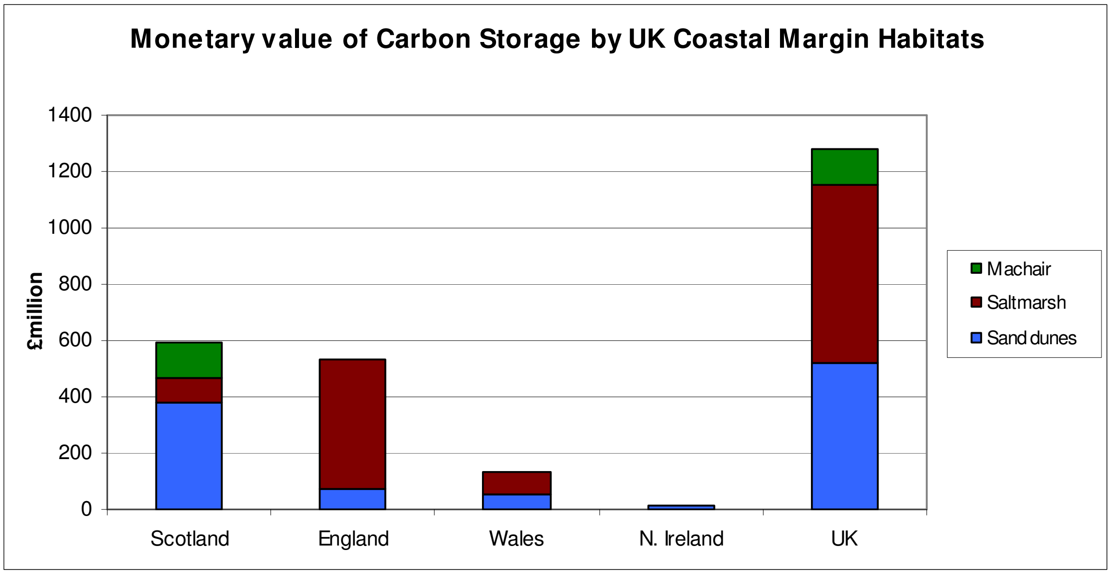
In addition to sequestration rates data on carbon stored are also available for some of the coastal margin habitats. In terms of stock, coastal margin vegetation and soils (to 15cm) are estimated to hold at least 6.8 MtC, shown by habitat and by region in Figure 7.

Figure 7. Carbon stock for the coastal margin habitats. Data shown are total C stocks in above- and below-ground vegetation and in soils to 15 cm depth (soils only for Machair grasslands due to lack of data). Data re-worked from Jones *et al.* (2004; 2008); and unpublished CEH data



This carbon stock value can be converted to CO_2e by multiplying the ratios of molecular weights, that is 44/12 or 3.67. The result can then be combined with the monetary value data, enabling the derivation of £/habitat values for the provision of C storage, as depicted in Figure 8.

Figure 8. Monetary value of Carbon storage by UK coastal margin habitats, using a 2010 DECC C price of £51.6 per tonne of CO_2e



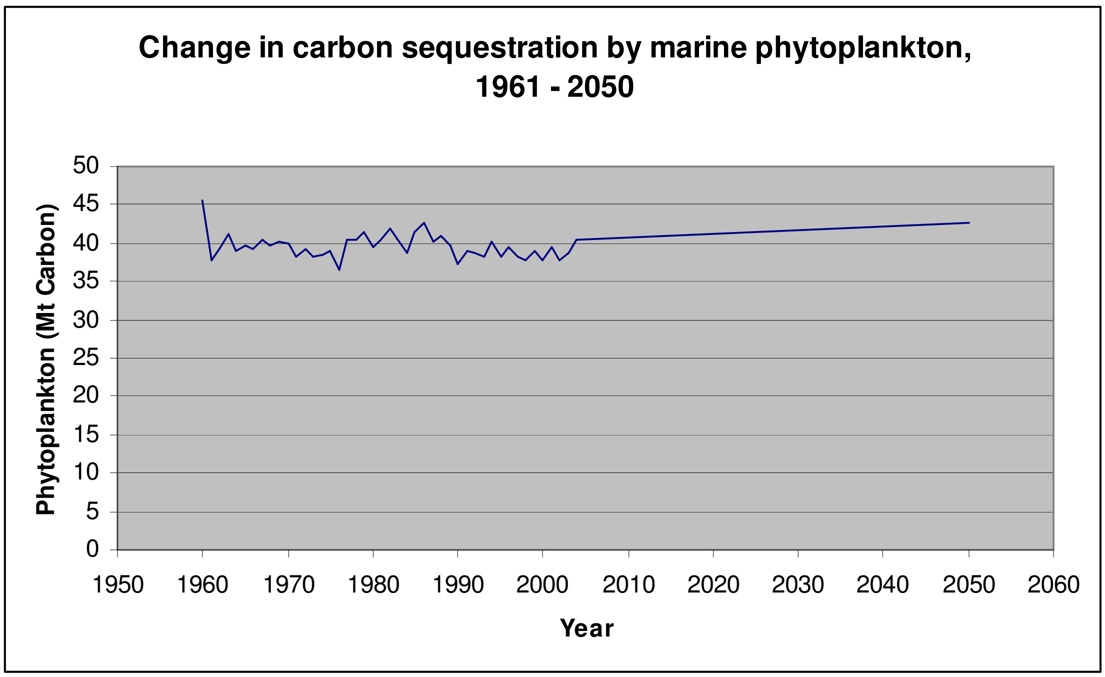
These three habitats may also emit greenhouse gases to an unknown extent, methane (CH_4) emissions from saltmarsh are thought to be negligible due to sulphate inhibition of methanogenesis, but nitrous oxide (N_2O) emissions may be important (Andrews *et al.,* 2006). The net effect on climate regulation is likely to be beneficial; however the contribution to climate regulation is probably small at the UK scale due to the low total area of these habitats.

2.1.2. Carbon sequestration in marine habitats

The marine habitat plays a significant role in the regulation of our climate, not least through marine organisms acting as a reserve or sink for CO_2 in living tissue and by facilitating burial of carbon in sea bed sediments (Austen *et al.* 2010). However, as detailed in Austen *et al.* (2010) there is minimal data readily available to quantify the extent of this role, or indeed even the total stock of carbon stored within the marine habitat.

In the absence of other data, average annual primary production (carbon sequestered by phytoplankton) in the UK shelf seas is used as a proxy for this service. Natural scientists at the Plymouth Marine Laboratory provided a hind cast of primary productivity in UK shelf seas over the last 50 years (Momme Butenschön unpublished, Austen *et al.* 2010), using coupled hydrographic-ecosystem modelling, producing estimates of annual biomass of carbon in the pelagic components of bacteria, phytoplankton and zooplankton (Figure 9). A projection has also been provided (Rob Holmes, unpublished), based upon the Special Report Emissions Scenario (SRES) AIB (=BAU), and adapted from QUEST fish data, although there is significant uncertainty associated with this (Figure 9).

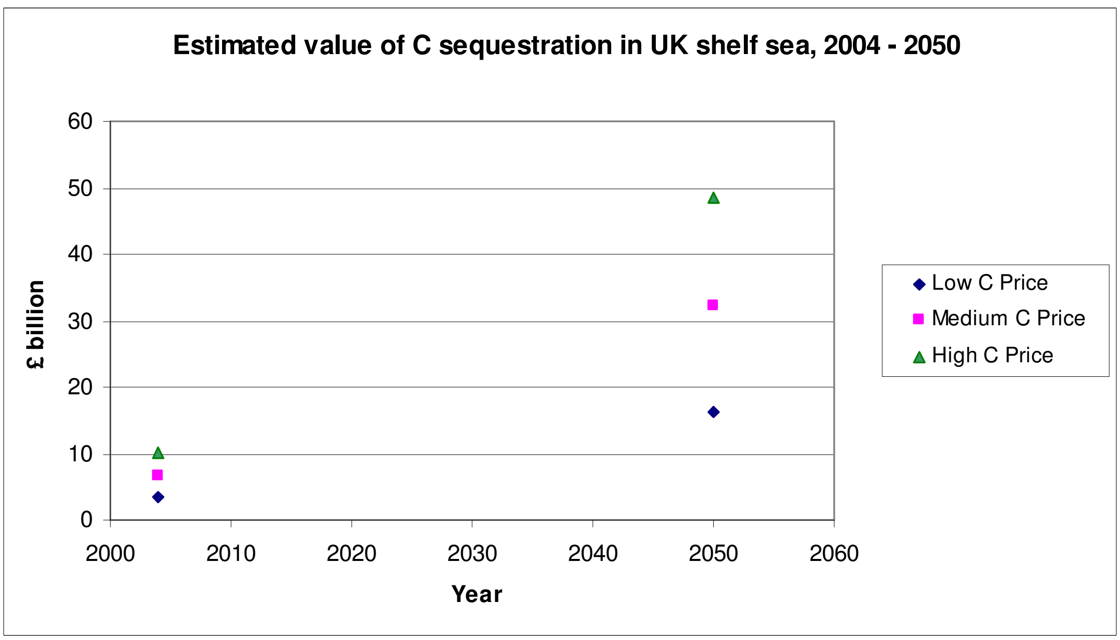
Figure 9. Estimated carbon sequestration by marine phytoplankton in UK shelf seas, 1961 – 2050 (Momme Butenschön, unpublished Rob Holmes, unpublished)



There is considerable annual variation in the hind cast data, but no clear long term trends. As this is a temperate coastal area there are many different causes for the variability observed in primary productivity, and thus the value of the C sequestration service. The main causes are light and nutrient supply. In winter the area is light limited, in summer it is nutrient limited. Temperature can have an effect but it is second order. The projected increase in primary productivity, following 2004, is in part due to an increase in temperature which in turn increases production, but the relationship is complex and non-linear so a simple analytical relationship between production and temperature cannot be identified. In addition, as the projected increase in C sequestration lies within the bounds of previous variability it cannot truly be considered as a future trend.

These data are combined with the carbon price data to provide an estimate of the change in value of C sequestration in UK waters, since 2004 (see Figure 10). The increase in value of this service is attributable to both an increase in primary productivity and the increase in carbon price. Economic data are not currently available to undertake hind cast analysis beyond 2004.

Figure 10. Estimated value of carbon sequestration by marine phytoplankton in UK shelf seas, 2004 - 2050



There are two significant problems using primary productivity as a proxy for carbon sequestration. Firstly, this will result in an underestimate of total primary production as this figure does not include primary production from macro algae and benthic micro algal production on intertidal sand and mudflats, especially within estuaries. Davis (2007) undertook a study on the value of C sequestration in the Isles of Scilly, spatially mapping the net annual C photosynthetic fixation values for kelp, sea grass, and phytoplankton. The result indicated that 136405 tC are fixed annually, with macro algae playing a significant role.

The second and more significant problem is that for marine carbon to be considered permanently sequestered it must either sink to the deep ocean, via the "biological carbon pump", or be buried in the benthic environment. The UK waters assessed in this analysis are primarily shallow shelf seas and the currents in these waters mean that it is unlikely that the carbon fixed by primary productivity in UK waters will be transported to the deep oceans. It is also unlikely that the carbon will be buried in the benthic environment as the carbon is more likely to be labile, and therefore more accessible and likely to be "processed" and kept within the marine ecosystem. The amount of carbon permanently sequestered cannot currently be quantified, but increasing primary productivity may not necessarily lead to increased carbon storage, and therefore these figures may be an over estimate.

2.2 Recreation and tourism

Sen *et al.* (2010) provides a full analysis of the value of marine and coastal recreation and tourism.

2.2.1 Coastal margin recreation and tourism overview, taken directly from Jones et al. (2010) and Austen et al. (2010)

The most obvious cultural benefit that society receives from the coastal margin and marine environment is the opportunity for leisure and recreational activities. Coastal margins and marine habitats are highly valued by the public, as living space for coastal communities, as a symbol of identity, a place for rest and relaxation, with a sense of freedom, where people can enjoy scenery and wildlife and specific activities including sunbathing, walking, bird watching, boating, swimming, and specialist outdoor sports. There are over 250 million visits per year to the UK coast, of which around one-third are to natural habitats such as beaches, sand dunes, shingle and cliffs.

Cultural ecosystem services provided by the coast are very important to the UK with seaside tourism valued at £17 billion (Jones *et al.* 2010). Tourism patterns have changed recently with overnight leisure visits being replaced by day trips. However, UK overnight trips to the seaside (worth £4.8 billion in 2009) still exceed overnight stays in the rest of the UK countryside and villages combined. The UK Leisure Day Visits Survey (2002 - 2003) reports 267 million visits to the seaside during 2002, approximately five per cent of all UK leisure day visits. This is an increase from previous surveys: in 1994, seaside visits accounted for only 3.5 per cent of all UK day leisure visits (although this figure varies across England, Scotland and Wales.) Expenditure at the seaside as a proportion of all expenditure on leisure day visits has remained more or less constant, at around four percent, between 1994 and 2002-03, although the actual amount has increased over this period from £2.2 billion to £3.1 billion.

These economic benefits are particularly significant in the more remote areas of the UK. In Wales in 2005, seaside tourism accounted for 42 % of domestic tourism spend, supporting nearly 100,000 direct and indirect jobs, together contributing £5 billion income (Valuing our Environment Partnership, 2006), while the value of tourism to the Western Isles of Scotland in 2006 is £49.9 million per year (Taylor et al 2006).

2.3 Disturbance prevention, including coastal flood defence (coastal margin habitat only)

2.3.1 Quantification of coastal defence

Coastal defence in the UK is provided by both natural and manmade structures. Currently approximately 18% of the coast is protected by defence works and artificial beaches, specifically 46% of England, 28% of Wales, 20% of Northern Ireland and 7% of Scotland (Masselink and Russell 2008). All the coastal margin sub-habitats play a role in coastal defence, with saltmarshes and sand dunes providing the major contribution to disturbance prevention (Paramor and Hughes 2004, Everard *et al*., 2010). Saltmarshes attenuate and dissipate wave and tidal energy and thereby substantially reduce the cost of flood defence measures (Morris *et al*. 2004, Brampton 1992, Möller 1996), while sand dunes provide direct protection, often replacing the need for artificial sea defence structures providing the dune system is wide enough, or the primary dune ridge is large enough.

2.3.2 The value of UK coastal defence

One approach to valuing coastal defence is to estimate the coastal defence expenditure avoided, for example, to calculate the cost of replacing a habitat with a sea wall. King and Lester (1995) estimated that in sea defence terms, assuming an 80m width saltmarsh, UK saltmarshes could result in cost savings relative to building man-made structures of £0.47 million to £0.94 million per hectare in terms of capital costs, and £9400 per hectare in terms of annual maintenance costs (adjusted to 2010 prices). By coupling this cost savings data with saltmarsh area (Table 2), an estimate of the cost of replacing UK saltmarsh with man-made sea defences can be made (Table 4). Flood risk is assumed to be constant.

Table 4. Estimate of cost of replacing UK saltmarsh with man-made sea defences (combining cost data from King and Lester (1995) with area data from Jones *et al.* (2010)

Country	**Year**	**ha saltmarsh**	**Capital costs (2010 £million)**	**Maintenance costs (2010 £million)**
England	1945	37331	17546 - 35091	351
	2000	32462	15257 - 30514	305
	2010	32462	15257 - 30514	305
	2060	32462	15257 - 30514	305
N. Ireland	1945	288	135 - 271	2.71
	2000	250	118 - 235	2.35
	2010	250	118 - 235	2.35
	2060	250	118 - 235	2.35
Scotland	1945	6900	3243 - 6486	65
	2000	6000	2820 - 5640	56
	2010	6000	2820 - 5640	56
	2060	6000	2820 - 5640	56
Wales	1945	6670	3135 - 6270	63
	2000	5800	2726 – 5452	55
	2010	5800	2726 – 5452	55

	2060	5800	2726 – 5452	55
UK	1945	51189	24059 - 48118	481
	2000	44512	20921 - 41841	418
	2010	44512	20921 - 41841	418
	2060	44512	20921 - 41841	418

It is noteworthy that despite the errors associated with this method, discussed in the following text, this is the only available estimate for the entire UK including a hind cast and forecast. King and Lester (1995) include a number of caveats to these data, and in addition it should be noted that these figures are based on an Essex saltmarsh, so there will be some error associated with extrapolating these across the UK. Furthermore, sea defence measures and their associated costs may have changed since this study.

King and Lester (1995) also give a cost saving per metre of wall (assuming an 80m width saltmarsh) ranging from £2600 – £4600. Scaling these values by linear length rather than area gives a value range of £3.7 – £6.55 billion, which in 2010 prices would be equivalent to £5.8 – £10.26 billion. Scaling by habitat length gives a more realistic estimate of value than scaling by unit area, as area is not directly related to the length of coastline protected by the habitat, although it is important to note that the ability of the habitat to provide this function is partially dependent on its width. However, most of the physical changes to coastal margin habitat are in width due to land-grab from landward side (with the possible exception of saltmarsh), and there are no historical data on changes in linear length. The difference in scaling by area and by linear length may account for some of the discrepancy between the two estimates of saltmarsh value.

In Table 5, the replacement cost is calculated for England as the difference in cost between constructing a sea wall, and the cost of maintaining the equivalent natural habitats (Environment Agency, 2007), multiplied up by linear length of the habitat. Comparable data on linear length of habitat and of sea-defence costs are not available for Wales, Scotland or Northern Ireland.

Table 5. Estimate of cost to replace coastal margin sub-habitat with sea wall, for England only (Environment Agency, 2007)

Sub-habitat type	Total habitat length (km) (Pat Doody, unpublished data).	Average cost to replace habitat with man-made seawall (£ per metre)	Total replacement cost for habitat (£ billion)
Shingle shore	536	1468	0.786
Saltmarsh	1424	1522#	2.167
Sand dune	346	1487	0.515
Total			3.498

cost of maintaining saltmarsh not given, assumed £0/m.

There are a number of sources of inaccuracy associated with this method. Firstly, scaling by either linear length or by habitat leads to an over-estimate of value as this assumes that the habitat provides a coastal defence function at all locations where it occurs. This may not always be the case, for example some dune systems abut steeply rising land and therefore do not provide a direct sea-defence service to the land behind, although they may provide an indirect service in regulating sediment supply to lower-lying land downdrift. Pye *et al.* (2007) aim to overcome this issue, and derive a more realistic estimate, by understanding the context of coastal margin habitat locations. Pye *et al.* (2007) detail dune systems in England and Wales which protect high value land and lack any artificial defence structures. A revised estimate was calculated using the linear length of dune systems with a protective function but lacking artificial defence structures (Pye *et al.* 2007), combined with the Environment Agency costs (Table 6). This gives a sea defence value for dunes in England of £173.7 million and £54.2 million for Wales. This is an under-estimate, as the calculation methodology excludes dunes where the sea defence function is supplemented by artificial structures of some kind. In common with saltmarsh, when dunes and shingle are retained in combination with artificial structures, they reduce the necessary size and therefore the cost of the man-made sea defences. Information was not available for Northern Ireland or Scotland, or regarding any temporal changes.

Table 6. Replacement cost of sand dunes with coastal defence function, for England & Wales, assuming an average cost to replace habitat with man-made seawall of £1487 per metre (Environment Agency, 2007)

	Sand dune length (km) (calculated from data in: Pye *et al.* 2007).	Cost of sea wall replacement (£million)
Wales	36.45	£54
England	116.79	£174
Subtotal	153.24	£228

A second source of inaccuracy associated with the replacement cost method is that it does not consider the value of the land which is being protected. The importance of a defence is dependent upon the value of the land use behind it, for example, high density residential or industrial developments, high grade agricultural land or habitats of international conservation importance have high value to society (Pye *et al*. 2007). Thirdly, as mentioned above, this approach also does not take into account the risk of flooding, not least as the topography of the land behind the defence is not considered.

Fourthly the values presented above should be treated with caution because the replacement cost approach fails to capture the full economic value of the ecosystem service being valued. This approach does not consider that the replacement of a natural structure with a man made alternative may result in the loss of other services which the habitat was providing, for example carbon storage and nutrient cycling. Finally, it is entirely unclear whether society would actually be willing to pay the necessary amounts to replace the natural structure, and rather than investing in coastal defences they may adjust their activity in other ways. For example, an Environment Agency (2004) study showed that areas of coastal Essex were no longer economical to defend from flooding, as in some cases the cost of maintaining the many existing defences exceeds the benefits (Defra 2004). A final problem is the variability of data available, with some data being only available for some habitats, and some data being available only for some countries. There is little UK wide data for all habitats, and even less data regarding temporal changes.

An alternative to the replacement cost approach could be to use the damage cost avoided method, as promoted by Penning-Roswell *et al.* (2010) in their handbook of assessment techniques for measuring the benefits of flood and coastal risk management. They present a series of methods for assessing the vulnerability of an area to flooding (from both rivers and the coast) and coastal erosion; calculating potential damage costs to land, property and recreational uses, and emergency costs (e.g. police, fire and ambulance; local authorities; and environment agency); estimating the probability of flooding based on topographical data; and calculating the damage costs not avoided by the defence scheme (i.e. defence schemes are built that will not protect against all floods, some flooding may still therefore occur). While following this approach may provide a more accurate assessment of the costs and benefits to a flood protection scheme, it still does not consider the willingness of individuals and society to replace a natural structure with a man-made alternative or whether the two options are indeed perfect substitutes. In addition data are not available on a UK wide scale.

In the absence of specific studies of individual's preferences for or against the use of coastal margin habitats in flood defence and coastal protection, Eftec (2010) recommend the use of value transfer. This approach transposes values estimated at one site (an original study site) to another (the site of interest). They illustrate their approach using three case study sites in the Humber Estuary. The potential effects on habitats and ecosystem services by potential flood and coastal erosion risk management scheme options (maintain the line, do nothing, managed retreat) are identified, values generated by other studies (e.g. for carbon storage, habitat change, change in recreational use) are then transferred and adjusted according to site and over time (100years), and a sensitivity analysis is then undertaken. While this approach potentially considers some aspects of individual preferences (although the extent to which they are incorporated will depend on the methods applied in the original studies undertaken), it faces short-comings. For example, the extent to which it is realistic to transfer values from one site to another has been questioned and good studies may not be available from which to transfer values (benefit transfers can only be as good as the original studies used).

The approach used by Eftec (2010) only explores impacts on ecosystem services from the management approaches and the losses and gains that may result over a 100 year timeframe. It does not consider the risk of flooding and the potential damages caused as a result. When undertaking assessments regarding the use of natural structures in coastal defence (e.g. through Shoreline Management Plans), all of these factors need to be taken into account and valuation tools need to be developed to meet this need.

2.3.3 Value of coastal defence: future projections 2010 - 2060

The net area of saltmarsh habitat is projected to remain the same (Jones *et al.* 2010) but sand dune, shingle and machair habitats are predicted to decline, thus impacting the provision of coastal defence in these areas. Sea level rise will increase the pressure on coastal defences, and projections compared to the 1980 – 1999 baseline, suggest a 12cm (1.2mm per year) to 76 cm (7.6mm per year) increase by 2095, with a greater rise in southern regions than north (MCCIP 2010). This increase of pressure on coastal defences will in turn increase the value of this service. Increased vulnerability of the coast to flooding and erosion as a result of climate change has resulted in a doubling of investment in the English and Welsh coastal defence sector in the past 10 years, with a recent spend of £600million in 2007, £650 million in 2008/2009 and £700million in 2009/10 (Defra 2008). It is estimated that investment will continue to increase to £1040million a year, plus inflation, by 2035 to maintain current standards of flood protection in England (Environment Agency, 2009), with an increase in the use of managed realignment and other forms of soft coastal defence measures. In the case of Scotland the total cost of sea defences, including construction and maintenance, was estimated at £76 million from 2009 to December 2015 (£12.67 million per annum), with a running cost of £13.5 million per annum thereafter (The Highlands Council, 2008).

2.4 Extractable – food and raw materials (marine only)

The extensive provisioning services provided by UK seas benefit people from both within the UK and abroad, and include fish and shellfish from wild capture and aquaculture, for consumption; fishmeal and oil as inputs to aquaculture and food supplements; algae and seaweed as inputs to pharmaceuticals and biofuels; and bait used during sea angling. From a socio-economic perspective the most significant activity, and the focus of this section, is the provision of fish and shellfish from wild capture and aquaculture.

Consideration of fisheries is divided into two sections: firstly, the assessment of flows resulting from fishing activity; secondly, the analysis of fish stocks, which then enables the assessment of the sustainability of the given flows. This analysis is based on data presented in Austen *et al.* (2010), and as a result the remit and scope is significantly bounded to focus primarily on landings of pelagic and demersal finfish and shellfish into the UK by domestic and foreign vessels. Aquaculture is not included, but is discussed in depth by Austen *et al.* (2010).

2.4.1 Quantification of UK fisheries (flows)

Austen *et al.* (2010) provides a detailed overview of the recent trends in the fish landings into the UK by domestic and foreign registered vessels, taken as a proxy for the volume of fish provided by UK waters. These landings data are currently the best available long term estimate of the total fisheries service provided by marine habitats in UK waters. However, there are inaccuracies associated with this estimate as not all fish caught in UK waters will be landed in the UK, and equally some of the fish landed in the UK will have been caught outside of UK waters. For example, in 2006 it was estimated that more than 75% of the volume of fish caught in UK seas was captured by non-UK vessels, notably by French, Danish, Norwegian and Dutch fishing fleets (www.seaaroundus.org), only some of which will be landed in the UK.

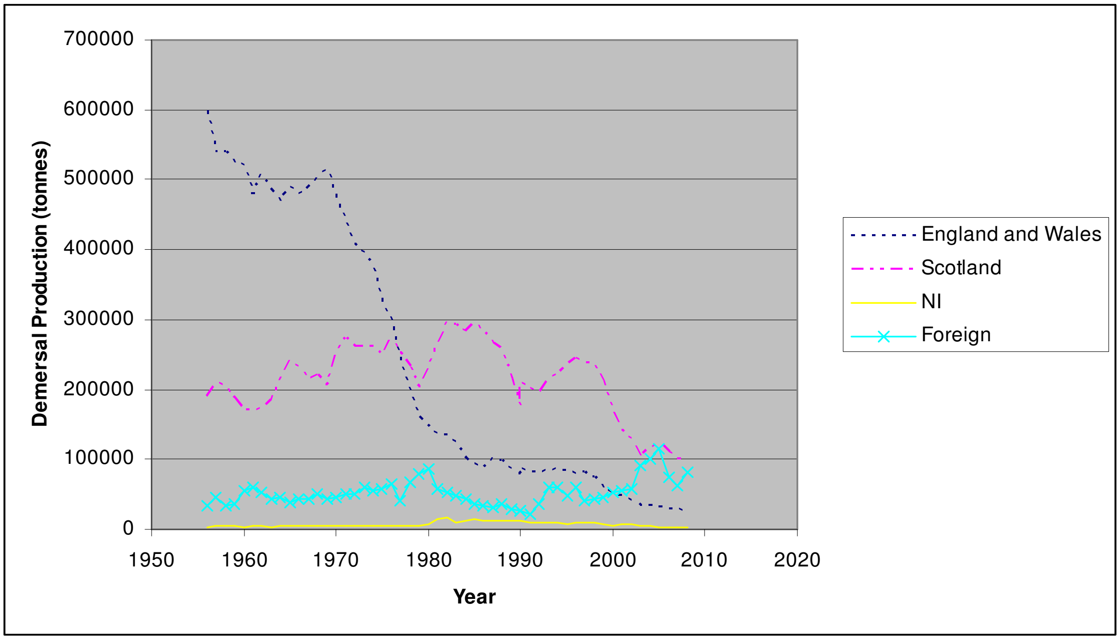
Landings from marine ecosystems can be divided into three separate categories: 1) Demersal fish species which live on or near the sea bed including cod, haddock, plaice, whiting, pollack, and soles; 2) Pelagic fish species, such as herring and mackerel which are typically found in mid and upper waters; and 3) Shellfish including molluscs (e.g. scallops, oysters, mussels, cockles), crustacea (e.g. prawns, crabs, lobsters) and cephalopods (e.g.

octopus, squid, cuttlefish). Total landings into the UK increased from 1938 to 1948 (from 1.1 million tonnes to 1.2 million tonnes per year) after which they declined steadily to 0.5 million tonnes in 2000 (Marine Fisheries Agency 2008). Thereafter, landings have remained stable (Figure 11). The decline in landings has not been consistent across all landing categories. Total landings of demersals have declined over time, however, foreign vessel landings and UK vessel landings into Northern Ireland have increased. There are several potential reasons for the decline in demersal landings. These include: declining fish stock sizes due to high levels of fishing mortality; reduced catch quotas; restrictions on number of days allowed at sea; a shift to shellfish harvesting; decommissioning schemes that have resulted in reductions in the overall size of the fishing fleet, imposed fishing effort reductions in the North Sea.

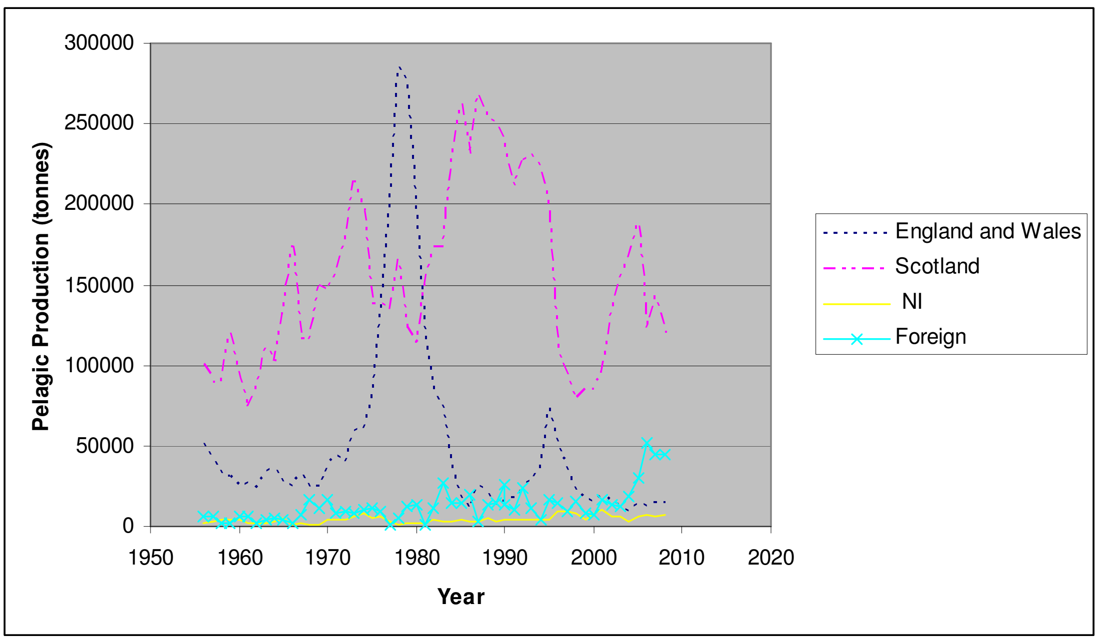
Pelagic species exhibit a more erratic pattern with no overall trend, although there is a slight increase in UK vessel landings into Scotland and Northern Ireland, and landings by foreign vessels. Finally, landings for shellfish increased from 32, 055 tonnes in 1938 to 127, 744 tonnes in 2000, and have remained relatively stable since. Demersal landings remain the largest in terms of weight. A more detailed analysis of landings data is presented in Austen *et al.* (2010).

Figure 11. Landings of: a. demersals (1956 – 2008); b. pelagics (1956-2008); c. shellfish (1966 – 2008), into England and Wales, Scotland, and Northern Ireland by UK vessels, and landings into the UK by foreign vessels (Marine and Fisheries Agency, 1956-2009)

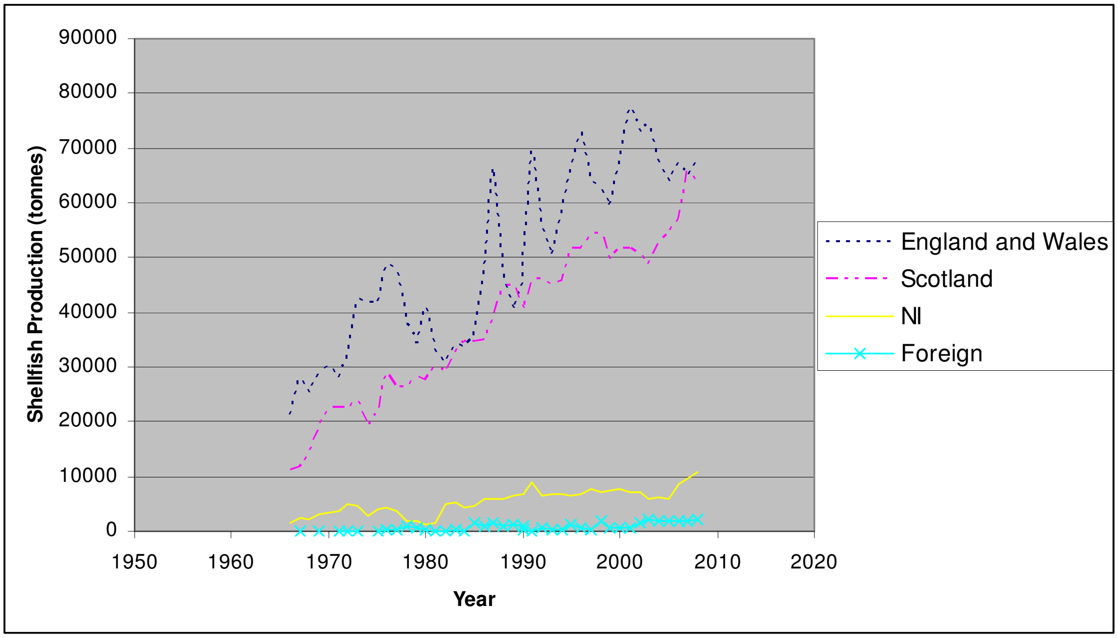
a. demersals



b. Pelagics



c. Shellfish

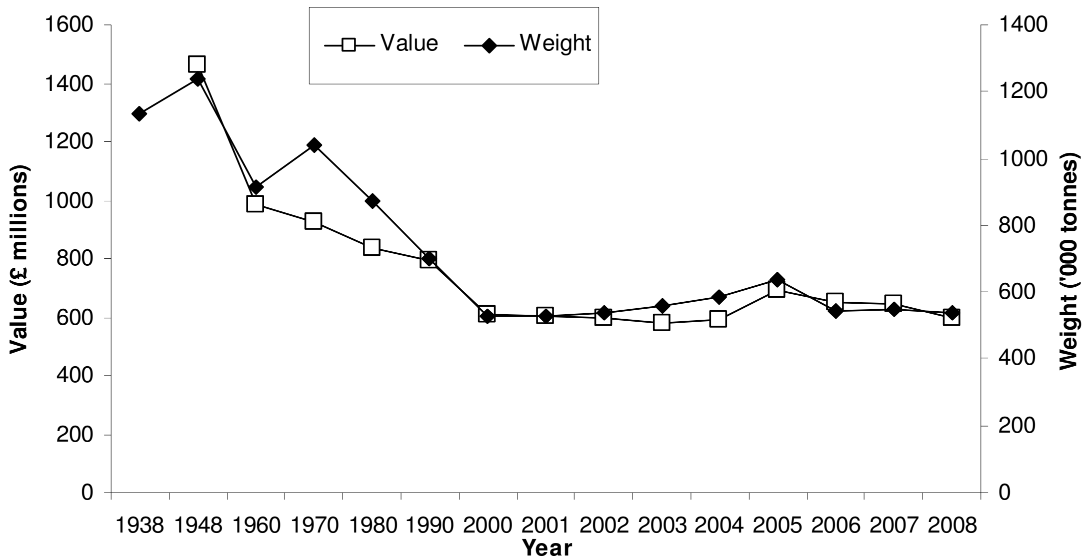


2.4.2 The value of UK fisheries (flows)

The output from a fishery is a flow of fish which is then sold at market price, thus flow can be valued using this market price. The total value[2] of the landings by UK and foreign registered vessels into UK ports is also detailed in Austen *et al.* (2010), and depicted here in Figure 12. The value of landings follows a similar decline to that of volume caught. Shellfish landings have overtaken pelagic and demersal landings in terms of value, but remain the smallest in terms of volume.

Figure 12. Landings into the UK by UK and foreign vessels: 1938 to 2008 adjusted to 2008 prices (Marine and Fisheries Agency United Kingdom Sea Fisheries Statistics 2008).

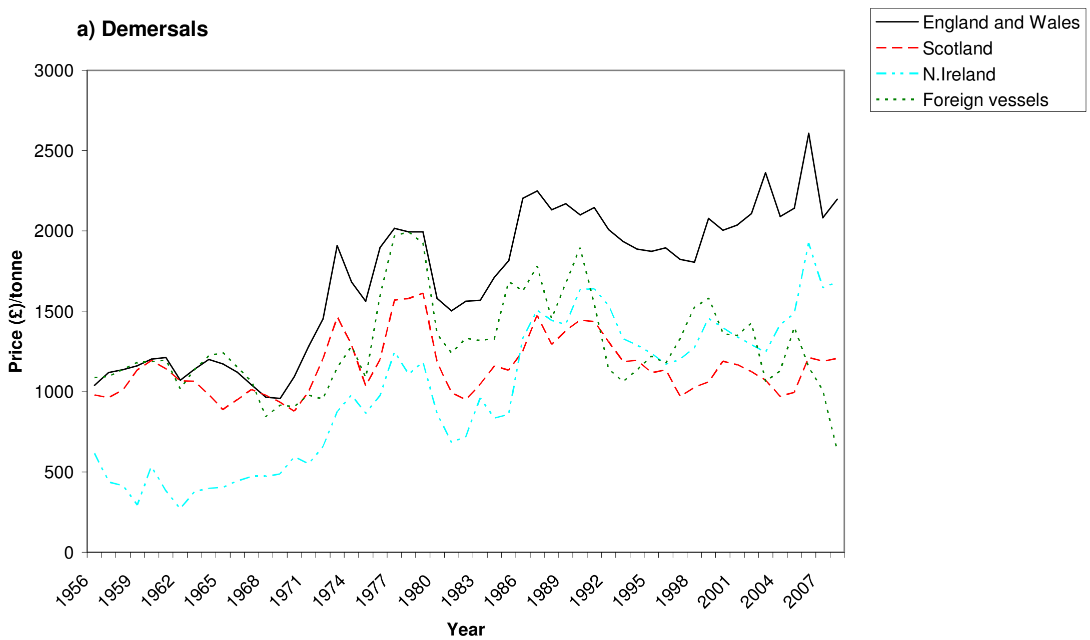
[2] To aid comparison, all values reported have been adjusted to 2008 prices using Retail Price Index.

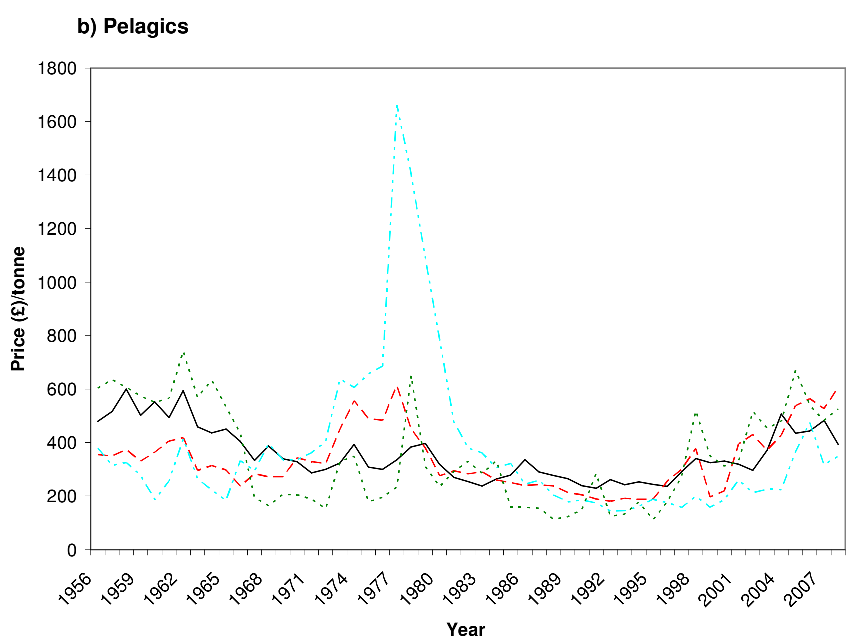


Landings value at first sale has decreased in line with the volume of landings, as would be expected, but what is of greater interest is the impact of declining fish stocks on their marginal value (i.e. how has the price of a unit of fish changed over time?). In a closed simplified system, due to the interaction of supply and demand, it is predicted that as the availability of a good decreases (e.g. landings of fish) the price of a unit of that good (e.g. £/tonne of fish) will increase, and vice versa. Fisheries are clearly not a closed system, but this hypothesis provides a useful starting point for discussion of these marginal values.

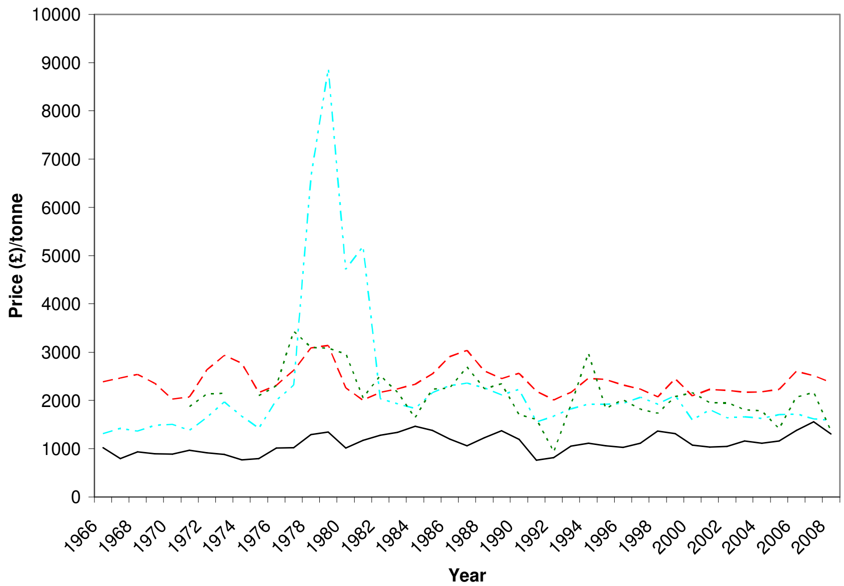
Price per tonne is used as a proxy for marginal value with the results depicted in Figure 13. In the case of demersal fish species landed in England, Wales and Northern Ireland, there is an upward trend in price per tonne following substantial reductions in catch and fleet size (figure 13a). There is, however, little change in the price per tonne for demersals landed into Scotland and for those landed by foreign vessels. In the case of the pelagic fish, price per tonne remains fairly constant, with some variability, which mostly follows fluctuations in the landings data. There is an overall slight increase in price per tonne, following a decrease in total UK landings.

Figure 13. £/tonne of demersal, pelagic and shellfish species landed into England and Wales, Scotland, Northern Ireland and the UK as a whole by UK and foreign vessels (Prices adjusted to 2008 prices using the RPI). Source: MFA (1956-2008).

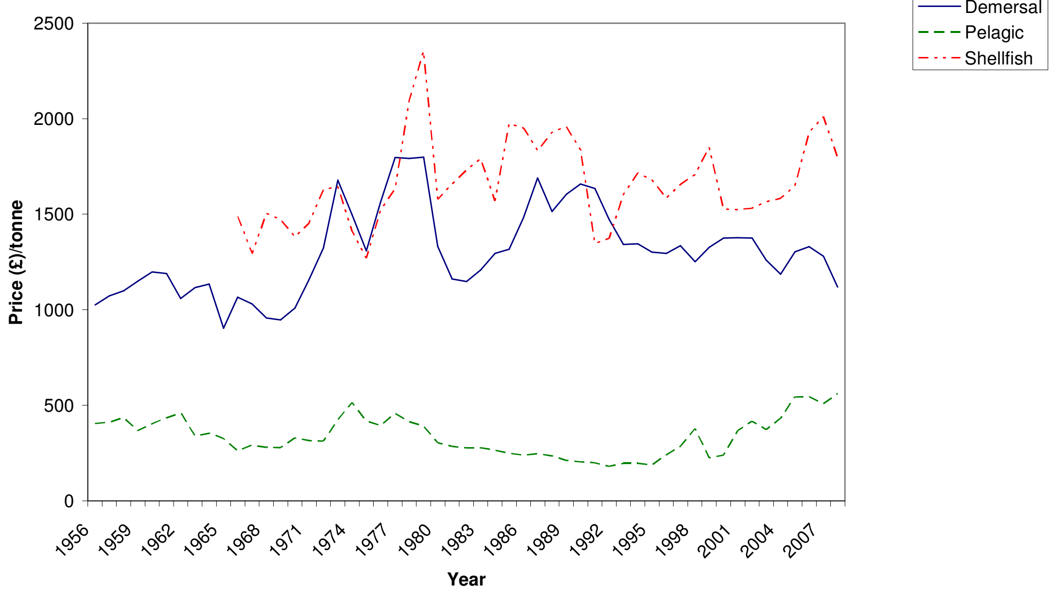




c) Shellfish



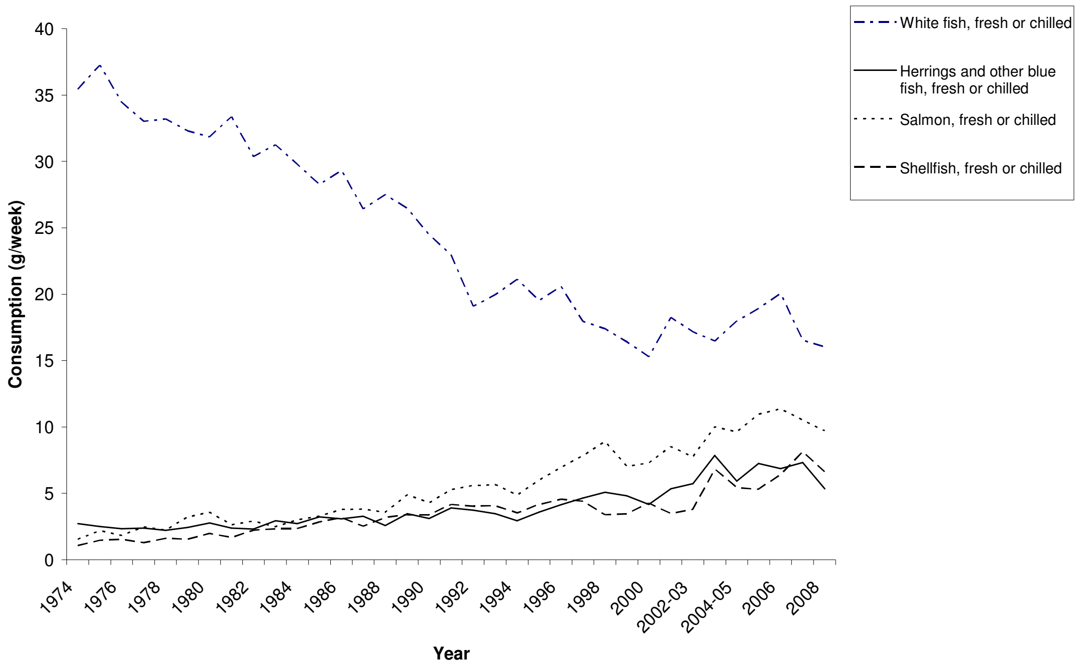
d) UK and foreign vessel landings combined



It is difficult to draw conclusions about how and why the marginal values have changed as they are influenced by the distorting effects of policy instruments such as subsidies and quotas on the market for fish. The Sea Around Us project (www.seaaroundus.org) estimated that subsidies provided to the UK fishing fleet contributed 26.6% of the value of landings in 2000. Quotas and days at sea restrictions prevent excessive catches, as do stock declines, but quotas also encourage fishers to catch the amount of quota available irrespective of whether those quotas are set in terms of maximum sustainable yields.

Secondly, the system is not closed and there is ready substitution with imports and alternative fish species which influences the marginal value. The fish landings data clearly do not equate to UK fish consumption. Fish imports into the UK have grown in recent years, with imports exceeding exports by 364 000 tonnes in 2008, a 30% increase from 2007 (MFA, 2009), and making the UK a net importer of fish. The main species imported are cod, haddock, tuna, shrimps and prawns, with imports of key species such as cod and haddock well in excess of exports. However, in the pelagic fishing sector, exports of herring and mackerel still exceed the imports. Consumers easily switch their consumption between species according to availability or to other non-fish products. Figure 14 shows the UK consumption of fish since 1974 and demonstrates how the consumption of white fish has decreased dramatically while the consumption of salmon and oily fish has increased. This may in part reflect the supply of these fish species, and the resultant change in price, but may also indicate a response to recent government advice on benefits from the consumption of oily fish.

Figure 14. Consumption of fresh or chilled white fish, blue fish, salmon and shellfish within the UK. The source of these fish is not identified and may not be from UK waters. Source: Expenditure and Food Survey 2008.



An additional complication associated with these data are that the value of fisheries presented here is a combination of both the ecosystem service value and human and manufactured capital needed to extract the fish. Ideally the total value of the fisheries would be divided into two components: ecosystem service values and the cost of the human and manufactured capital. Attributing the entire value of landings to the value of the fisheries ecosystem service is a simplification and an over-estimation. To disaggregate the ecosystem service value from the non-ecosystem service values it is necessary to quantify all the human and manufactured costs used in fishing activities across different vessel and gear types. These include fixed costs such as one off payments for fishing boats and gear; variable continuous costs such as fuel, wages, ice, boat and gear repair and maintenance, the purchase of quota and insurance; and compliance costs such as licences. Subsidies provided to the fishing industry would also have to be taken into consideration (e.g. exemption from fuel duty on marine diesel, support for vessel refits), as would capital investment and debt levels.

There are some data available on the human and manufactured capital costs, but they are by no means complete. For example, there are data on number of fishermen, 39,380 in 1948, falling to 10, 242 in 2008 (Austen *et al.* 2010), but there are little data on salaries or income. Equally, there are some data on number of vessels at sea and their capacity (gross tonnage) and engine power (Marine Fisheries Agency, 2009), but the temporal scope of these data is poor, and there is a paucity of data relating to the capital and maintenance costs of the vessels. Given the scarcity and irregularity of these data it was decided that it was not possible to separate the ecosystem service value from the natural and man made capital costs within the scope of this report.

In addition to the first sale price of fish there are an extensive range of secondary services which are supported by fisheries, both up and down the supply chain (Austen *et al.* 2010.) These vary from boat builders and repairers, and gear merchants to fish processors and food industries, including 480 fish processing sites that employ around 15,000 people (Seafish 2009). In 2005 31,633 people were employed in the catching, processing and aquaculture sector in the UK, representing 3.5% of the total employment in all 24 maritime industries in the UK (Pugh, 2008).

In 2007, shellfish production in England was worth £4.5m producing primarily mussels with small quantities of Pacific oyster, native oyster and very small quantities of clam and cockle. In Wales, shellfish farming is concentrated almost entirely on mussels and was valued at £7.5 million in 2007. Mussels also dominate shellfish farming in Northern Ireland, although there is some production of oysters and clams; in 2007 it was valued at £5.8 million. The Scottish shellfish industry was worth approximately £5 million in 2007, producing mussels, Pacific oysters, queen scallops, scallops and native oyster (CEFAS, 2008; FRS, 2008).

Although the UK commercial fishing industry is declining and makes a relatively low contribution to overall Gross Domestic Product (GDP) it remains an important socio-economic activity particularly in remote coastal regions in Scotland, Wales and south-west England, where it provides employment, through fishing, in aquaculture farms and fish processing and associated industries (e.g. boat building and maintenance, gear supply, markets and transport for fish).

2.4.3 Assessment of sustainability of UK fisheries (stocks)

The landings data do not necessarily reflect the size of the fish stocks in UK waters. Volume of fish landed is linked to stock size, but there a number of variables which prevent it from being a true indication. For one, the recorded volume is dependent upon accurate recording at point of first sale and it is possible that some fish may not be recorded, it also hides the fact that not all fish caught are landed (i.e. some are discarded) and that black fishing is known to occur but the quantities of black fish landed are not known (Pew Trusts, 2010). Landings are also heavily influenced by policy instruments, including catch quotas and subsidies.

Austen *et al.* (2010) provide minimal data on the size of the fish stocks or the degree to which they are being sustainably harvested, both of which are ideally required for the economic analysis of extractable resources. Measuring these however is very difficult because of natural variation in fish behaviour, variability in the data collection due to observer bias, time of day, tidal influences and other sampling factors (Mangi and Roberts, 2007).

With regard to sustainable extraction level, 18 fin fish are routinely monitored and used to create a sustainability index for marine fin fish stocks around the UK. This index includes demersal round fish (cod, haddock, saith, hake), flatfish (sole, plaice), pelagic (mackerel, herring) and widely displaced (blue whiting). ICES (the International Council for the Exploration of the Seas) uses estimates of fishing mortality and spawning stock biomass to assess the sustainability of these fish stocks[3]. Focusing on just 18 species is clearly not representative of the UK fisheries provisioning service, and cannot be extrapolated to be so, but it does provide useful data for discussion. It also highlights the lack of UK wide species stock data.

Armstrong and Holmes (2010) report that for 2008 50% of assessed UK stocks were at full reproductive capacity and were being harvested sustainably. This is an increase from the 1990s when only 5-15% of monitored stocks were considered to be harvested sustainably, and 2000 when this figure stood at 20-40%. While this is a positive trend, a number of

[3] ICES www.ices.dk

scientifically assessed UK stocks continue to be fished at levels considered to be unsustainable, the majority are fished at rates well above the values expected to provide the highest long term yield (Saunders, 2010), and a number of other commercially important species remain unassessed due to inadequacies in the available data. Armstrong and Holmes (2010) therefore state that while the proportion of stocks being harvested sustainably has increased, fishing mortality in most stocks remains high and above levels that will support the maximum sustainable long-term yields or economic returns under current environmental conditions.

As fish stocks have declined there has been an increase in the levels of manmade input to substitute for the decreasing natural capital (i.e. fish) to maintain landings. Indeed, Thurston *et al.* (2010) report that despite changes in the size of the fishing fleet, technological advancements, and improvements in fishing efficiency, landings per unit of fishing power (LPUP) have reduced by 94% over the past 118 years. The authors suggest that this decrease in LPUP reflects a decrease in fish stocks and indicates that fish catch globally has only remained stable in recent years because of an increase in fishing effort. This increase in expenditure on manufactured capital per tonne of fish landed could potentially result in an increased market price for fish, thus reducing demand for this species by consumers, and helping to ensure protection of the good from over exploitation. However, as discussed previously the market for fish is complex, and as detailed in Austen *et al.* (2010) in the past demand has not declined and fisheries resources have been over-exploited. This non-sustainable extraction has occurred for several reasons including: i. as commonly observed with natural resources, the original market price for the fisheries resource was an underestimate of the true value; ii. the price has been distorted by policies including subsidies; iii. although there has been an increase in man made input, improved technology has led to an overall increase in efficiency of fish catch; iv. there has been substitution to alternative fish species with larger stocks. In recent years, however, this trend has started to be reversed, in part due to an improvement in the management of the fisheries resource.

2.4.4 Value of finfish and shellfish landings: future projections

Given the complexity of the social and natural drivers affecting fisheries it is very difficult to make any future projections beyond the next few years, and even these are prone to significant error. To make realistic projections two criteria need to be met: models need to capture the complexity of the social, economic and environmental systems in which fisheries are located and to have sufficient knowledge of how these systems might change into the future. Both of these criteria are unlikely to be met (Garcia and Granger, 2005). Nonetheless, qualitative assessments of general trends have been made. It is widely agreed that the demand for fish will increase globally, although fish consumption rates within the EU are expected to remain stable. Wild capture fish landings are expected to show limited or no growth (and may even decline as many stocks are over-exploited), with the increased demand for fish protein being met through aquaculture (Delgado *et al.* 2003; Pinnegar *et al.* 2006).

An additional variable is climate change which will certainly influence future fisheries, and is included alongside fishing pressure as one of the two main drivers affecting fish stocks in the north east Atlantic. Climatic factors have been shown to alter fish community structure through changes in distribution, migration, recruitment and growth (Pinnegar, 2010), and as a result some fish distributions are moving both deeper and northwards (MCCIP 2010).

A recent report by the UK Government's Cabinet Office Strategy Unit (2004) suggests a number of potential changes within the UK fishing industry according to three alternative futures: i. Market World: expansion of free trade, removal of tariffs and increased application of technology; ii, Green World: rise in environmental values, demand for sustainable local produce; iii. Fortress Europe: failure of international institutions, high tariffs, low investment in technology and aquaculture. The industry is expected to expand in the next 10-15 years by about 20%, assuming stocks are sustainably managed. If stocks are managed badly, the industry can expect to contract by 30%. In the worst case scenario involving stock collapses, 50% of employment in the catch sector could be lost with knock-on effect in local communities. In the most likely scenario, prices for most major UK fish stocks, however, are expected to remain stable or fall as barriers to trade are reduced.

3. Discussion

This study is unique in investigating how the value of UK marine and coastal margin ecosystem services has changed over time, providing, where possible, a hind cast and a forecast. Four of the most significant ecosystem (i.e. biotic) services are assessed.

The sub-habitat type plays a significant role in the type and extent of services provided in the coastal margins, indeed provision can even vary within sub-habitat type. It has been possible to provide service values on a £/ha and £/country basis. Service provision in the marine environment is more generic, and the data available have little spatial definition, and tend to be provided on a UK, or at best country level. Marine services data often have uncertain boundaries, and stocks which move within and out with any designated boundaries. In the case of the marine environment the spatial data are less essential, as most marine environments deliver most marine ecosystem services, albeit in differing amounts (Austen *et al.* 2010). As a result service values are provided only on a £/country basis. Table 7 provides a summary of the data provided.

Significant knowledge gaps have hindered this analysis considerably. Areas suggested for future research include:

1. Carbon sequestration and storage by coastal margin habitats.
   Data required: Machair C sequestration rates; the areas of the replacement habitats and their associated sequestration rates to enable the calculation of net change in carbon sequestration and storage; permanence of storage
2. Carbon sequestration and storage by marine habitats
   Data required: Permanence of storage; sequestration by macro algae and benthic micro algae; total stock of C stored in marine
3. Disturbance prevention
   Data required: UK wide coastal defence data, including spatial maps detailing value of land being protected, if the coastal margin habitat is providing a protective coastal defence function; risk of flooding; public preference and willingness to pay for continued protection.

Table 7. Summary of data provided, 2010 prices unless specified otherwise

Service	Method	Units	Time Series	Values
C sequestration – coastal margin	Avoided damage cost	tCO_2/yr	1900 - 2060	Sand dune: Decrease of 80,168 tCO_2/yr
			1945 - 2060	Saltmarsh: Decrease of 34, 774 tCO_2/yr
		£/ha/yr	2010	Sand dune: £32.25 - £241.49/ha/yr
			2010	Saltmarsh: £60.63 – 622.30/ha/yr
		£/UK/yr	2010 - 2060	Sand dune: 2010: £7.98 million/UK/yr 2060: £39.13 million/UK/yr **Increase of £31.15 million/UK/yr**
			2010 - 2060	Saltmarsh: 2010: £11.93 million/UK/yr 2060: £63.22 million/UK/yr **Increase of £51.29 million/UK/yr**
		Stock value £/UK	2010	Sand dune, Saltmarsh and Machair: £1282 million
C sequestration - marine	Avoided damage cost	tCO_2/yr	1961 - 2050	Variable, no clear trend
		£/UK/yr	2004 - 2050	2004: £6.74 billion/UK/yr 2050: £32.35 billion/UK/yr **Increase of £25.61 billion/UK/yr**
Disturbance prevention	Cost savings	£/ha £/ha/yr	2010	Saltmarsh: Capital costs: £0.47 – 0.94 million/ha Maintenance costs: £9400/ha/yr
		£/UK £/UK/yr	1945 - 2060	Saltmarsh: 1945: £481million/UK/yr 2060: £418 million/UK/yr **Decrease of £63million/UK/yr**
Recreation and tourism		£/UK/yr	2002	£17 billion
Fisheries (2008 prices)	Market prices	UK tonne/yr	1948 - 2000	1948: 1.2 million tonnes/yr 2000: 0.5 million tonnes/yr
		UK £/yr		1938: £1465 million/UK/yr 2008: £596 million/UK/yr **Decrease of £869 million/UK/yr**
		UK £/tonne	1956 - 2008	Demersal 1956: £1026/tonne Demersal 2008: £1119/tonne Pelagic 1956: £404/tonne Pelagic 2008: £561/tonne Shellfish 1966: £1488/tonne Shellfish 2008: £1796/tonne

4. Fisheries

   Data required: data on fish landed specifically from UK waters, VMS data is now available, but there is no historic information of this kind; expenditure data on human and manufactured capital needed to extract fish; data on UK fish stocks, beyond the 18 currently assessed species.
5. In addition to improving the valuation of the four services detailed here, expending research effort on the many other goods and services would also be very valuable, instead of continued focus on those which are well understood and often commercially based, e.g. recreation. Significant, but as yet unvalued, goods and services include bioremediation of waste and resilience and resistance.
6. Improving the quantification of the linkages between ecosystem function and provision of services would be useful to determine the long term sustainability of service provision. This includes investigating how the provision of services will change in response to environmental changes, such as temperature and pH. Potential tipping points and thresholds also need further investigation. For example, changes in marine biodiversity will influence the biogeochemical cycling of C and nutrients within the marine system, resulting in changes in the capacity of the marine environment to act as a carbon sink (Legendre and Rivkin 2005), but this relationship has yet to be quantified. Carbon sequestration rates in coastal margin habitats are likely to vary with temperature, but again this change cannot be quantified at the current time. Research effort is required in this area to enable the sustainable management of ecosystem services in the future.

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