Title:

The value of carbon sequestration and storage in coastal habitats

Authors:

N.J. Beaumont [a,*], Jones, L. [b], Garbutt, A.[b], Hansom, J. D.[c], Tobermann, M.[b]

[a] Plymouth Marine Laboratory, Prospect Place, Plymouth, DEVON, PL1 3DH

[b] Centre for Ecology and Hydrology Bangor, Environment Centre Wales, Deiniol Road, Bangor, LL57 2UW, UK

[c] Department of Geographical and Earth Sciences, University of Glasgow, Glasgow, G12 8QQ, U.K.

*corresponding author, e-mail: nijb@pml.ac.uk

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Abstract:

Coastal margin habitats are globally significant in terms of their capacity to sequester and store carbon, but their continuing decline, due to environmental change and human land use decisions, is reducing their capacity to provide this ecosystem service. In this paper the UK is used as a case study area to develop methodologies to quantify and calculate a monetary value for the service of carbon sequestration and storage in coastal margin habitats. Specific changes in UK coastal habitat area between 1900 and 2060 are documented, the long term stocks of carbon stored by these habitats are calculated, and the capacity of these habitats to sequester CO_2 is outlined. Changes in value of the carbon sequestration and storage service of coastal habitats were then projected for 2000-2060 under two scenarios, the maintenance of the current state of the habitat and the continuation of current trends of habitat loss. If coastal habitats were to be maintained at their current extent, their sequestration capacity over the period 2000-2060 is valued to be in the region of £1-3 billion UK sterling. However, if current trends of habitat loss continue, the capacity of the coastal habitats both to sequester and store CO_2 will be significantly reduced, with a reduction in value of around £0.25 -1 billion (2000-2060). If loss-trends due to sea level rise or land reclamation worsen, this loss in value will be greater. This case study provides valuable site specific information, but also highlights global issues regarding the quantification and valuation of carbon sequestration and storage. Whilst our ability to value ecosystem services is improving, considerable uncertainty remains. If such ecosystem valuations are to be incorporated with confidence into national and global policy and legislative frameworks, it is necessary to further address this uncertainty. Recommendations on achieving this are outlined.

1. Introduction

Ecosystem services are commonly defined as "the outputs of ecosystems from which people derive benefits" (NEA 2011). The ecosystem service of carbon sequestration and storage, linked to the provision of an equable climate, is a rapidly growing research field (Chung et al. 2011). Whilst there is extensive literature regarding the role of terrestrial habitats as a source and sink of greenhouse gases, the role of marine and coastal habitats is comparatively unknown. Recent research has shown however that "blue carbon", that is carbon sequestrated and stored by marine and coastal habitats (Nellemann et al. 2009), could play a significant role in the global carbon budget (McLeod et al. 2011, Chung et al. 2011). At present, an estimated one third (~2 Gt C yr-1) of anthropogenic CO_2 emissions are sequestered by the oceans (Orr et al. 2001, Takahashi et al. 2002). In addition, coastal habitats such as mangroves, sand dunes and saltmarsh have the capacity to sequester carbon at a rapid rate (Alongi et al. 2012, Jones et al. 2008, Chmura et al. 2003) and, on accreting coasts, this may occur to considerable depth or lateral extent (Chmura et al. 2003). The relative carbon storage potential of coastal habitats is now considered to play a significant role in the regulation of both local and global climate (Pendleton et al. 2012, Nellemann et al. 2009, Irving et al. 2011).

Coastal habitats are at risk and in decline across the world (Martinez et al. 2004, French 1997). Drivers of this decline include urban and industrial development, aquaculture, agriculture, tourism, forestry, coastal erosion and sea level rise (Jones et al. 2011). For example, 'reclamation' of coastal land for agricultural or industrial use alone, here termed 'land claim', has accounted for an estimated 25% loss of intertidal land in estuaries worldwide (French 1997). In the UK, coastal margin habitats have been subject to considerable land use change over the last 100 years (Delbaere 1998, French 1997), with land claim through draining occurring on an industrial scale since the 1700s (Hansom et al. 2001). With conversion, degradation or loss comes a decline in their potential to sequester and store carbon. Pendleton et al. (2012) estimate that 0.15 – 1.02 Pg (billion tons) of CO_2 are released annually through the conversion of vegetated coastal ecosystems resulting in economic damage, estimated to be in the order of $US 6-42 billion annually (£ 4-27 billion).

The social and economic significance of this ecosystem service in coastal systems is however poorly represented in policy and management decisions, and rarely features in global climate change mitigation discussions or documentation. In the context of coastal management, it is critical to recognise that any change in type, functioning and area of these ecosystems has the potential to influence carbon sequestration and storage (Everard et al. 2010). In addition to policy at a global scale, this capacity is also of importance for local scale ecosystem service accounting, for example when making decisions on coastal flood defence options such as managed realignment (Andrews et al. 2006).

Global level studies have raised awareness of coastal carbon (Pendleton et al. 2012), yet there is a continuing need for methodological development regarding the calculation of carbon sequestration rates, carbon stocks, and the valuation of this ecosystem service. In addition there is an on-going requirement for site specific data to support meaningful national and local scale policy.

We address these issues using the UK as a case study, partly because few studies have been published in this area. For example, published carbon sequestration rates in saltmarsh rely predominantly on US studies of saltmarsh (Kirwan and Mudd 2012) which are geomorphologically different from European saltmarshes. Sequestration rates in saltmarsh have been estimated from extrapolation of sedimentation rates and carbon content of established saltmarsh sediments (Cannell et al. 1999; Adams et al. 2012) but do not quantify carbon stocks. In sand dune grasslands and dune wetlands, chronosequence approaches have been used to estimate carbon sequestration rates (Jones et al. 2008). However, no study has yet attempted an inventory of carbon stocks in these habitats and the implications of coastal change for carbon stocks are largely unquantified.

This study provides the first comprehensive inventory of carbon stocks and sequestration for the principal UK coastal margin habitats of saltmarsh, sand dune and machair dune grassland, including change over time from 1900 to 2060. Information is collated from published sources, the grey literature and unpublished data to calculate C stocks, and estimate the impact of future change. Given the rate of conversion of coastal habitats to other land uses globally and within Europe, the implications of this decline, both for the carbon stocks held and for future carbon sequestration, are explored. It is essential to understand the extent and stability of those carbon stocks and therefore to understand the permanence of storage.

"Carbon stock" is used to define the carbon stored in the given ecosystem, often shown in static units of g/m^2 or g/m^3. This is different to an "ecosystem service stock", which in the case of carbon storage and sequestration is the ecosystem structure and processes (Luisetti et al. 2013), sometimes known as the natural capital. Neither stock is valued here. The carbon stock currently stored in coastal ecosystems is not valued as the carbon prices are designed to be applied to marginal changes in stock, not total stock values. In addition, although stand alone environmental values are useful in raising awareness (Costanza et al. 1997, Beaumont et al. 2008), they and do not aid the decision making process with regard to balancing trade-offs and selecting between different development options.

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From the ecosystem service stock flows a variety of ecosystem services, one of which is carbon sequestration, or the rate of carbon uptake, which is generally measured in dynamic units such as g m^{-2} yr^{-1}. The carbon stock has the potential to increase, via the ecosystem service of carbon sequestration (a positive flow of this ecosystem service), or decrease via habitat destruction and an accompanying release of CO_2 (which could be interpreted as a negative flow of this ecosystem service, or dis-service). It is this change in carbon stock which is valued here, and both aspects, the potential service and dis-service, are explored. This approach will provide information to policy makers and coastal managers, and an improved methodology which will be transferable to coastal habitats elsewhere.

2. Carbon sequestration and storage by UK coastal margin habitats

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Sand dune habitats and sandy beaches, saltmarsh and machair dune grassland comprise 93% of the UK coastal margin habitat, the remainder consisting of vegetated shingle and shingle beaches; saline lagoons; and maritime cliffs and slopes and small islands. In this study we focus on these first three habitats, henceforth termed sand dunes, saltmarsh and machair since almost nothing is known about the carbon stock or sequestration rates in vegetated shingle, maritime cliff grasslands or saline lagoons, and they occupy less than 10%

of the UK coastal margin area. However, we acknowledge that further work needs to be done to study carbon sequestration and storage in these habitats. Sand dune systems contain a variety of vegetation types, from mobile sand dunes to fixed dune grassland, scrub and dune slacks, and seasonal wetland habitat. Saltmarsh comprises vegetated inter-tidal habitat in a range of communities defined primarily by the frequency of tidal inundation. Machair systems are unique to Britain and Ireland and typically consist of a cordon of mobile sand dunes bordering fixed machair grassland which is occasionally cultivated and fertilised with seaweed. Machair may also contain seasonal wetlands and may grade to peaty wetlands inland.

2.1 Trends and drivers of change in coastal margin habitat area

There is a downward trend in the area of all UK coastal margin habitats (Jones et al. 2011). In sand dunes this decline is mainly due to urban expansion, forestry planting, agricultural improvement, tourism and leisure (e.g. golf and caravan parks), and sea level rise. Decreases in saltmarsh area are primarily a result of land claim from agriculture and industry, and coastal erosion. The downward trend in machair area is due to infrastructure development, coastal erosion and sea level rise. Increased statutory protection over the last few decades has slowed the rate of loss of coastal margin habitats, but coastal erosion and sea level rise continue to pose a significant threat. Changes in habitat quality, and natural successional development within existing areas, will also alter rates of carbon sequestration (Jones et al. 2008). Table 1 summarises habitat area by country, and changes in their extent over time.

Current sand dune area (Table 1) is ca. 71,000 ha (JNCC; Dargie 2000), of which 71.4% is in Scotland (Angus et al 2011). Since 1900 some 30% of the UK dune area has been lost, including 9,127ha of the Scottish resource (Delbaere 1998; Angus et al 2011). Prior to 1945 losses were mainly due to urban expansion and forestry planting, in the period 1945-1970 losses were primarily due to agricultural improvement and continued infrastructure development for tourism, while between 1970-2000 losses decreased due to statutory protection of most large sites, however smaller or unprotected sites continued to be lost to agricultural improvement or tourism infrastructure (e.g. golf) (Doody 1989; Sturgess and Atkinson 1993). In the future, sand dune extent is projected to decline by 2% over 20 years due to coastal erosion and sea level rise (JNCC 2007), with linear extrapolation in this study to a decline of 6% over the period 2000 to 2060. By region, the percentage of habitat lost will be greater in England and lower in Scotland due to differences in the rates of relative sea level rise (Shennan et al. 2009), while losses in Wales and Northern Ireland will remain more-or-less constant. Further losses are projected to occur in small sites lacking statutory protection (Packham and Willis, 1997) as a result of agriculture or tourism development.

UK saltmarsh area (Table 1) is ca. 47,000 ha (JNCC; Burd 1989) of which between 6,000ha and 7,000ha is in Scotland (Angus et al 2011). Since 1945 there has been approximately 15% loss in UK saltmarsh area (Cooper et al. 2001, Baily and Pearson 2001) much of this loss being in estuaries and inlets. For example, 2860ha or 51% of the mudflat and saltmarsh area of the Forth estuary in Scotland has been lost to agricultural and industrial reclamation over the last 400 years (Hansom et al 2001). Future losses to sea level rise were projected to be 4.5% over 20 years (French 1997), extrapolated forward to 2060.

The majority of machair, 67%, is found in Scotland and the remainder in Ireland (Jones et al. 2011). The total area of machair in the UK (i.e. that occurring in Scotland) is around 20,000 ha (Dargie 2000) (Table 1). Past extent has been estimated based on losses of 1.2 ha per linear km over the period 1945-2010 due to coastal erosion (Hansom 2010), calculated assuming the average width of a machair system is 0.5km. However, losses due to grassland improvement (particularly in the Orkney Islands and Tiree) and infrastructure are as yet unquantified. Future losses are projected to be 2% loss in area over 20 years due to sea level rise (JNCC 2007) with linear extrapolation to 6% by 2060.

[Insert Table 1: UK areas (ha) of the three main UK coastal margin habitats, by country.]

2.2 Carbon sequestered and stored by UK coastal margin habitats

Long-term C sequestration rates in soil of the most significant coastal margin sub-habitats were derived from the literature as follows: In dune habitats a chronosequence study covering 160 years (Jones et al. 2008) calculated mean sequestration rates (+/- s.d.) of 58.2±26.2 g C m^{-2} yr^{-1} in dry dune grasslands and 73.0±26.2 g C m^{-2} yr^{-1} in wet dune slack habitats. Based on the relative area of dry dune and dune slack habitats (91% and 9% of UK dune area respectively) a proportional average sequestration rate for dune habitats of 59.5±25.8 g C m^{-2} yr^{-1} was calculated. There have been no estimates of long-term C sequestration in machair systems. However, since they are ecologically similar to sand dune grasslands, we estimated a sequestration rate of 34.9±15.7 g C m^{-2} yr^{-1} based on the ratio of soil carbon measurements in the two habitats. We make the assumption that they have been developing over similar time periods, based on common trends of change in dune systems across north west Europe (Provoost et al. 2011). We acknowledge that this may be an under-estimate of sequestration rates since machair systems undergo frequent disturbance through cultivation (Hansom and Angus 2001, 2006) and so the observed stocks may have built up over shorter timescales. Sequestration rates in UK saltmarsh range from 64 – 219 g C m^{-2} yr^{-1} (Cannell et al. 1999; Chmura et al. 2003; Adams et al. 2012), with typical figures around 120 – 150 g C m^{-2} yr^{-1}. Using the wider ranges above and converting to CO_2 equivalents, we obtain the following: 1.25 - 3.12 tonnes CO_2 ha^{-1} yr^{-1} (composite dune), 0.70 - 1.87 tonnes CO_2 ha^{-1} yr^{-1} (machair) and 2.35 – 8.04 tonnes CO_2 ha^{-1} yr^{-1} (salt marsh). By combining these figures with data on changes in UK areas of the sub-habitats (Table 1), estimates of the changes in the capacity of these habitats to sequester CO_2 can be derived (Figure 1)

[Insert Figure 1: Estimated change in annual CO2 sequestration provided by UK coastal habitats, 1900 – 2060, due to changes in area, applying a constant sequestration rate.]

These values relate to gross changes due to loss of coastal margin habitat. Calculating net values requires detailed information on what land classes the coastal margins are being converted into, and the carbon storage and sequestration potential of these new land classes. The implications of this are discussed in detail below.

2.3 Carbon stocks in UK coastal margin habitats

In addition to sequestration rates, stocks of carbon stored can be calculated for coastal margin habitats for the period 2000 – 2010. In terms of stock, coastal margin vegetation and soils are estimated to hold at least 9.4 MtC (Table 3).

[Insert Table 2: Underpinning data used to calculate carbon stocks in soils and vegetation for sand dunes, machair and saltmarsh. For data sources please refer to section 2.3]

Carbon stock in sand dunes was calculated separately for carbon stored in above-ground and below-ground (root) plant biomass, and for soil (to 15 cm), excluding roots. Calculations were made separately for three classes of dune habitat: mobile and semi-fixed dunes, fixed dune grassland and dune slacks. Strandline communities and dune scrub or woodland were not included. The area of each community type from JNCC inventory data of the sand dune survey of Great Britain (Dargie 1995, Dargie 2000, Radley 1994) was summed to these habitat classes for England, Scotland and Wales. Data on the proportions of these habitats in Northern Ireland were not available therefore the sand dune resource was allocated to habitat classes in proportion to the average proportions of these classes in Great Britain. Above-ground biomass carbon data for these habitat classes were based on biomass samples and bulk %C content from a survey of eleven dune systems around Great Britain (Jones et al. 2002, 2004). Below-ground (root) carbon for each habitat class was scaled from above-ground data based on biomass ratios and %C content of dune habitats from Jones et al. (2005). Soil C stocks to 15 cm, which included the full organic profile in all but a few dune slack soils, were calculated from the survey of eleven sites, where soils were acidified to remove carbonates prior to C content being measured on a CSN analyser (Jones et al. 2004). Data for machair grassland used the same above and below-ground plant carbon content and biomass as those of fixed dune grassland, scaled by machair area (Dargie 2000). Soil carbon stocks to 15 cm in machair were based on %loss on ignition (LOI) of eight soils (Table 2) sampled from six sites in the Outer Hebrides: Seilebost, Horgabost (Harris); Berneray; Clachan Sands, Balranald (North Uist); Bornish (South Uist) in 2010, of which three samples were taken in areas which had been recently cultivated. The carbon proportion of the machair soil organic matter was 0.3, calculated for a subset of nine machair soils sampled from the Outer Hebrides in a previous study (Jones et al. 2004), which were acidified to remove carbonates before combustion on a CSN analyser (most non-carbonate soils have an organic carbon ratio of ~0.5-0.6). Machair soils from the 2004 were not used as they lacked bulk density information.

[Insert Table 3: Carbon stocks (t), by habitat, vegetation and soil pools and by country]

Saltmarsh above and below-ground biomass and loss on ignition data were obtained from nine sites on the west coast of England and Wales, representing a range of livestock grazing intensities, including grazed and ungrazed systems, and from nine sites in the south east of England to adequately characterise regional differences (Table 2). The salt marshes of west coast UK generally have a shallow organic-rich clay layer (<1m) underlain by sandy substrate and are frequently grazed by livestock (May and Hansom 2003). The marshes of the south and east UK coasts are characterised by a deep (>10m) organic-rich clay substrate and are most commonly ungrazed. The west coast sites were: Four Mile Bridge and Newborough (Anglesey); Y Foryd and Morfa Harlech (Gwynedd); the Dyfi and Dyfi RSPB reserve (Powys); the Dee estuary (Flintshire/Wirral); Llanfairfechan (Conwy); Banks Marsh on the Ribble estuary (Merseyside). On the west coast sites two samples of soil and vegetation were taken from grazed and ungrazed, mid or upper marsh at each site. Above-ground biomass was harvested from 25x25cm plots. Below ground biomass C was estimated for all sites based on the ratio of below: above ground plant C from a separate study on the Ribble estuary (Ford 2012b). Soils were sampled to 15 cm to incorporate the majority of the organic-rich sediment, following methods in Ford et al. (2012a). The carbon content of the underlying sandier substrate was calculated using soils sampled at 30 cm depth from the nine sites, and this figure was extrapolated for the remaining soil profile to a depth of 50 cm, taken to be the average sediment depth for these systems. Bulk density for west coast soils was calculated from 36 samples on the Ribble estuary, to 15 cm depth (Ford et al. 2012b). On the East coast of England nine salt marshes were sampled along the coast of the county of Essex, following methods in Burden et al. (2013). Four 30cm soil cores were taken from each marsh, homogenised and %loss on ignition and bulk density measured on each. Calculations for underlying substrate to 100 cm used a LOI of 5% for data from intertidal mudflats (Centre for Ecology and Hydrology, unpublished data). Above and below-ground vegetation biomass carbon for the east coast sites was based on west coast data, but are similar to published values for the Blackwater estuary (Burden et al. 2013).

Above and below ground biomass was dried at 105 deg C over 24 hours to obtain oven dry weight. Loss on ignition was measured for saltmarsh soils at 375 degree C for 8 hours, which minimises loss of carbonates (Ball 1964). A ratio of 0.52 was used to convert %LOI to soil C for all saltmarsh soils. For west coast and particularly for east coast sites, we acknowledge that the %carbon ratio may be an overestimate due to loss of carbonates during combustion, but overall this is likely to be an under-estimate of saltmarsh soil carbon stocks since organic sediments may be considerably deeper than sampled in this study. For upscaling of soil C stock data, values for the sandier west coast-type sediments were applied to sites in Wales, North West England and Scotland, while the deeper, more organic rich south and east coast-type sediments were used to upscale C stocks for sites in the South West, Southern, Eastern and North east UK coasts.

3. Valuing the ecosystem service of carbon sequestration and storage by UK coastal margin habitats

Understanding the underlying processes which support C sequestration and storage enables the quantification of this ecosystem service, and in turn more effective sustainable environmental management. However, valuation, both monetary and non-monetary, can facilitate transparency in the discussion of trade-offs between different ecosystem services (and associated beneficiaries) when different development options are considered.

To avoid double counting valuation of ecosystem services needs to focus on the benefits provided, rather on the services themselves (NEA 2011, Fisher et al. 2009). In this case the final ecosystem service is that of carbon sequestration and storage. The benefits of this service include a stable and equable climate, which can be assessed using non-monetary techniques (NEA 2011) or valued monetarily using a proxy such as a carbon price. In this case a carbon price approach is taken. A wide range of studies present a variety of carbon prices, many of which are based on the estimation of damage costs avoided, or the value of the welfare loss to society of emitting an extra tonne of CO_2, in the form of human health impacts, environmental disasters etc. (Tol 2005; Defra 2007, Stern 2006). In 2009 the UK Department of Energy and Climate Change (DECC 2009) changed their CO_2 value appraisal guidance to use the costs of mitigation (DECC 2011), or the costs of reducing emissions. The DECC (2011) values are slightly higher, but fall within a similar range to previous studies. As the case study of this research is UK based, and as the research aims to be policy relevant, the recommended DECC values are applied (DECC 2011). DECC (2011) provide a low, medium and high range of carbon prices which are all applied to investigate the sensitivity of the analysis to the price.

Both traded and non-traded values of carbon are provided by the DECC (2011) guidance, but as environmentally sequestered carbon is not traded in the EU Emission Trading System, it is recommended to use the non-traded price. These non-traded values represent the maximum marginal abatement cost that will need to be incurred to ensure we can meet our emissions reductions targets in the non-traded sector, based on current emissions projections. In the case of environmental carbon sequestration the validity of this approach depends upon the implication that this is the additional cost that would have to be incurred elsewhere in the UK in order to meet our reductions targets if this carbon sequestration were not to occur in the environment. This is currently not the case, but with an increasing understanding and recognition of the role of habitat type in CO_2 budgets it seems increasingly possible that coastal habitats could, and potentially should, be mainstreamed and incorporated into overall country-wide carbon budgets.

With regard to hindcasting the first date for a carbon price from the DECC non-traded carbon values is for 2008. Given the policy context and technological landscape, the 2008 value is considered to hold for the years 2000-2008, but no further back due to the different and rapidly changing policy landscape prior to this point in time (NEA 2011).

3.1 Valuing carbon sequestration by UK coastal margins

Combining the previously calculated sequestration rates with the mid DECC (2011) CO_2 price the £/ha/yr values can be derived for the provision of CO_2 sequestration by the sub-habitats in 2009:

Sand dune	= 18.36 – 45.9 £/ha/yr
Machair	= 10.26 – 27.54 £/ha/yr
Saltmarsh	= 34.56 – 118.26 £/ha/yr

Combining these £/ha/yr values with data on changes in UK areas of the sub-habitats (Table 1) it is possible to value these coastal habitats with regard to their CO_2 sequestration potential (Figure 2). The extent of carbon sequestration service provided by coastal margin habitats decreases over time (Figure 1), as a direct result of habitat loss, but the value of this service increases due to the fact that the applied carbon prices will increase over time. This reinforces the increasing importance, and value, of this service into the future, and as a result the increasing importance of maintaining the habitats which provide this service.

[Insert Figure 2: Estimated value of annual C02 sequestration service provided by UK coastal habitats, sand dune, saltmarsh and machair, 2000 – 2060. Applying average C sequestration rate and mid DECC (2011) non-traded carbon value.]

There is uncertainty in both the carbon sequestration rates, and also in the DECC (2011) carbon value. Figure 3 provides a combined UK annual value for the carbon sequestration service of the UK coastal habitats of sand dune, saltmarsh and machair, but also incorporates the uncertainty both the DECC price and the carbon sequestration rate. As a result 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.

[Insert Figure 3. Uncertainty in the estimated value of annual C02 sequestration service provided by UK coastal habitats (combined sand dune, saltmarsh and machair) 2000 – 2060. Applying a range of C sequestration rates and low, mid and high DECC (2011) non-traded carbon values.]

3.2 Long term values of carbon sequestration by UK coastal margin habitats

Annual values of this ecosystem service do not adequately reflect the true long term value of this habitat with regard to carbon sequestration. It is particularly important to consider this long term value if these habitats are being compared to other long term development options. Thus in addition to calculating values for the annual flow of the carbon sequestration service, the Net Present Value (NPV) of this service is also calculated. To sum the amount of carbon sequestered each year in order to produce a 60 year total of carbon sequestered is complex; it cannot be assumed the carbon sequestered in any given year will be permanent (Table 1), as if habitat is lost previously sequestered carbon will be released. To calculate total carbon sequestration, and long term storage over a 60 year period, the areas of habitat which are lost, and the likely associated release of carbon, must also be considered. This loss of carbon could be termed the potential “dis-service” associated with these habitats if they are destroyed.

It is the change in carbon stock which is valued here, and both aspects, the potential service and dis-service, are explored. Firstly the positive value of the service of carbon sequestration is calculated. Using a baseline year of 2000, and applying an average sequestration rate, it is possible to determine the yearly amount of carbon sequestered and stored using equation 1 (below), where *R* is the rate of CO_2 sequestration (tonnes/ha/yr) and *At* is the area (ha) of habitat at year *t*.

Secondly the potential dis-service, of CO_2 release, associated with these habitats is calculated. Since coastal habitat is anticipated to be lost in the future (Table 1) it is possible to estimate the associated CO_2 released through the application of a linear loss of habitat between 2000 and 2060. The amount of carbon released is calculated in two stages. The long term carbon stored within each habitat type at the baseline year, 2000, is known (table 3), thus if we lose 1 ha of habitat it is possible to calculate the CO_2 released using equation 2, where Q is the CO_2 stored (tonnes/ha) and *ALt* is the area (ha) of habitat lost in year *t*. In addition any CO_2 sequestered since 2000 will also be released (equation 2). It is notable that the amount released per given area is dependent upon the time since sequestration began, in this case the year 2000, but older and more established habitats will have accumulated more carbon per ha.

The net carbon stored by the habitat is then calculated using equation 3. However, this is not the true net value as to calculate this it would be necessary to know what the coastal habitat was converted into when it was lost. However, as it is most likely that the coastal habitats will be converted into man made alternatives, such as urban developments and for industrial and tourism uses, it is likely these alternatives will provide minimal carbon sequestration capacity so this is not expected to be a major source of error.

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Eq. 1: *Total* CO_2 *Sequestered (S)* = $\sum_{t=0}^{N} Rate\ (R) * Area\ (At)$

Eq. 2. *Total* CO_2 *Released (L)* =

$$\sum_{t=0}^{N} Stock\ (Q) * Area\ lost\ (ALt)$$

+

$$\sum_{t=0}^{N} Rate\ (R) * Area\ lost\ (ALt) * \ Number\ of\ years\ of\ sequestration\ (t-1)$$

Eq. 3: *Net* CO_2 *sequestered = S - L*

The mid DECC (2011) carbon price is applied to calculate the difference in value between the predicted future scenarios of habitat loss, and no further habitat loss. In the case of highly uncertain, long term flows of environmental benefits the appropriate discount rate is disputed (Pearce and Ulph 1995, Price 2010). Previous authors have argued that the discount rate should take its lowest possible limiting value (Weitzman 1998), whilst others go further and propose that in such cases a zero discount rate should be applied (Broome 1992). Others have argued against this (Fisher and Krutilla 1975), and the current recommended social discount rate from the HM Treasury Green Book (2011) is of 3.5% diminishing with time. Given the variability in the literature two discount rates are applied: 0% to explore what happens to the values under this scenario, and 3.5% to be more closely in line with UK policy.

[Insert Table 4: Estimation of long term CO_2 *sequestration and release by UK coastal habitats, 2000 – 2060, and associated monetary values. Monetary values expressed as NPV UK£]*

If we maintain our current coastal habitats in the time period 2000- 2060 they will have provided an ecosystem service valued at over £UK 3 billion (not discounted) in terms of CO_2 sequestered. If however current projections of habitat loss are followed £UK 1 billion of this service will not be realised as the CO_2 will instead either remain in the atmosphere, or will be released into the atmosphere following the conversion of coastal habitats. The habitat loss projections made in Table 1 may turn out to be conservative but are based on expert judgement projections of coastal erosion, and do not account for other habitat losses due to land conversion which are difficult to predict. As a result, the difference in value could be greater, with a greater significant reduction in the value of this service.

The application of the different discount rates causes a significant difference in the value of the ecosystem service. It should be recalled that this variability is coupled with the uncertainty observed in Figure 3 when sensitivity to the carbon price and carbon sequestration rate was also calculated. This case study demonstrates the potential variability and uncertainty in the results depending on the approach taken, which has the potential to make a significant difference to the outcome when a variety of development options are being considered.

4. Discussion

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Coastal habitats are declining in both area and habitat quality in the face of increasing pressures related to climate change (such as the effect of coastal erosion and sea level rise) as well as increasing pressure resulting from human-driven change (conversion of habitat due to development). This study has used the UK as a case study to quantify the extent of habitat decline that has already occurred, and provided plausible future projections of continued habitat loss. The potential implications of this loss in terms of carbon sequestration and storage are explored and a monetary value attributed to the value of carbon emitted or stored.

This study explores how coastal carbon might be valued. In order to successfully value habitats it is critical to understand both the "stock" of service (in this case the carbon stored at the present time) and also the "flow" of service (what the ecosystem will continue to provide into the future). In this study both the carbon sequestered (the flow) and the carbon stored (the stock) are considered, enabling the development of an approach which quantifies and values the dynamic long term sequestration and release of CO_2 from coastal habitats under two realistic, but different, future scenarios.

In addition the results provide UK specific data which can be actively applied in a policy context and to support evidence-based ecosystem management. The results shown here are presented at a national scale, but could be readily applied at a smaller scale, enabling local decision making for policy decisions regarding different coastal development options. Given the extent of CO_2 stored and sequestered, the accompanying value of this service, and the vulnerability and decline of these habitats, it seems increasingly relevant to mainstream the role of these habitats into overall country-wide carbon budgets.

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With an increasing tendency to value the suite of ecosystem services provided by natural habitats and to utilise these values in the decision making process, comes a requirement to ensure both that these values are as accurate as possible and that uncertainty is transparent. As shown in Figure 3 there is potential for considerable uncertainty in valuation estimates, arising from uncertainty in the natural science, the CO_2 price and the choice of discount rate. The estimates presented here reflect the best information that is available at present. However, there remain sources of uncertainty in four key areas which are discussed below along with recommendations for future research aimed at improving and refining the estimates:

i. Stocks and sequestration rates

Soil carbon stocks accumulate over time and this can be both rapid and measurable in the case of coastal habitats. Our assessment of saltmarsh C stocks down to 0.5 m (west coast) and 1.0 m depth (east coast) is an improvement on previous methodologies but is probably still an under-estimate primarily because the depth of organic sediment on accreting east coast sites has not been consistently measured. Information from the limited chronosequences and dateable sites on the east coast suggests that accretion is keeping pace with sea level rise of between 2-6mm a year (van der Wal and Pye 2004), and deep soil cores made on the east coast suggest that carbon rich sediments may extend to much greater depth than 1 m. Therefore, a substantial soil C pool may not be accounted for here. On the sandy north and west UK coasts of the UK, extrapolation to 50 cm depth captures the majority of the sediment C stock. Nevertheless, there remains a need to more widely assess carbon stocks and organic sediment depths on these coasts also. There is considerable potential to improve the estimates of carbon sequestration rates in saltmarsh through the sampling of sites of known age, and this could provide validation of the sequestration rates derived from first principles by Cannell et al. (1999). On the other hand, for saltmarsh soils, the majority of these estimates use %loss on ignition values rather than measures of C content, which may overestimate actual carbon content in marine and calcareous sediments due to combustion of carbonates. The low furnace temperature of 375°C used minimises this but future data collection should address this issue. In this paper, we publish carbon densities and calculate for the first time carbon stocks in machair. However, calculation of total stocks is made difficult by varying opinions on the true areal extent of this habitat, which partly depends on the definition of machair (Angus, 2006). Latest estimates suggest this lies somewhere between 17,500 (Angus et al 2011) and 20,000 ha (Dargie, 2000).

The calculations presented here assume that sequestration rates remain the same over time and UK location. Sequestration rates vary with successional stage, being slowest at the point of initial vegetation colonisation, then increasing rapidly for a while before slowing as the organic profile develops (Jones et al. 2008), a pattern also shown over much longer timescales (Syers et al. 1970). For dunes and, by extrapolation, machair grassland, the figures used here represent average sequestration rates over a 60-year period. They are based on sequestration at 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 may not be 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, water table depth and UK location, but currently these data do not exist.

ii. Net values

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In all three habitats there is a loss of habitat extent over time, with an accompanying decline in C sequestration and release of previously stored CO_2. The monetary values associated with this decline have been estimated but fall short of true net values. This occurs because coastal margin habitat is replaced with an alternative habitat, which may also have some capacity for C sequestration. Conversions to urban or infrastructure expansion or agricultural land claim will lower sequestration potentials yet conversion to forestry or leisure activities may not. 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 that might be converted are not and so at the present time this aspect cannot be accounted for. Historically, the bulk of coastal margin loss has occurred to land uses with lower C sequestration rates. Therefore, whilst an exact net loss in sequestration cannot be calculated, overall there will still be a decrease in the provision of this service.

iii. Other gases sequestered and emitted

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The coastal margin habitats may also emit greenhouse gases to an unknown extent. Methane (CH_4) emissions from saltmarsh have previously been thought to be negligible due to sulphate inhibition of methanogenesis, but recent evidence suggests they can be locally high, particularly in grazed systems (Ford et al. 2012b). Nitrous oxide (N_2O) emissions may be important (Andrews et al., 2006; Dausse et al. 2012) but are relatively un-studied. Ford et al. (2012b) suggest for saltmarsh that the carbon budget represents the bulk of the greenhouse gas forcing, therefore the contribution of other gases to climate regulation can probably be ignored for this habitat. Both nitrous oxide and methane emissions are unstudied in dune grassland and machair systems but are likely to be negligible in these predominantly dry habitats.

iv. A satisfactory carbon price

None of the carbon prices which are currently available are truly applicable to carbon sequestration and storage in natural environments. The price used in this case is the DECC non-traded Marginal Abatement Costs, but as environmental carbon is currently not included in the mainstream carbon budgets this is not ideal. Other prices also suffer limitations, such as a focus on CO_2 release rather than on CO_2 sequestration and storage. Carbon prices are increasingly applied to natural environments often with little consideration of what this price means. Given the rapid growth in the application of these prices, it is strongly advised that greater consideration and resource is invested in developing robust values which are specific to the natural environment. Despite the uncertainty associated with these estimates Tol (2005) concludes that they do provide a useful benchmark to compare against costs of emission reduction policies.

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5. Conclusions

This study has investigated how the value of the carbon sequestration service in coastal margin habitats changes over time providing, where possible a hindcast and a forecast. It has demonstrated that coastal habitats can have significant value in terms of CO_2 stored and sequestered. If the current extent of UK coastal habitat is maintained their sequestration capacity over the period 2000-2060 is valued to be in the region of £3 billion (not discounted). However, if current habitat loss trends continue, the capacity of the coastal habitats both to sequester and store CO_2 will be significantly reduced, with a reduction in value estimated to be in the region of £1 billion over the period 2000-2060, (not discounted). If habitat loss is greater due to greater sea level rise, more rapid coastal erosion or unforeseen land conversion, this value has the potential to increase significantly.

Finally, it is noteworthy that carbon sequestration and storage is only one of many ecosystem services provided by coastal habitats. Other services from coastal habitats include coastal defence, recreation, nutrient and contaminant storage and cycling, and fish nursery grounds. Thus whilst the values of carbon sequestration and storage presented here make a convincing case to maintain these habitats, the argument to conserve these margin coastal habitats becomes all the more compelling when the full range of ecosystem services provided by coastal habitats is considered.

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Tables and Figures

Table 1. UK areas (ha) of the three main UK coastal margin habitats by country. See Section 2.1 for details.

	Area (ha)	**Year**					
		1900	1945	1970	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**		39476		34327	33572	29795
	N. Ireland		288		250	244	216
	Scotland		6900		6000	5865	5190
	Wales		8173		7107	6950	6168
	UK		54836		47683	46631	41369
Machair	**England**						
	N. Ireland						
	Scotland		20171			19698	18516
	Wales						
	UK		20171			19698	18516

Table 2: Underpinning data used to calculate carbon stocks in soils and vegetation for sand dunes, machair and saltmarsh. For data sources and methods please refer to section 2.3. Mean ± standard deviation. '–' not applicable.

	Vegetation				Soils			
	N	Above-ground biomass (g m^{-2})	Above ground plant %C	Ratio Below ground: Above ground C	N	Bulk density (g cm^{-3})	Soil %C	Soil C density (g cm^{-3})
Mobile dunes	62	1375 ± 855	43 ± 1.8	1.77	62	1.449 ± 0.155	0.33 ± 0.22	0.0047 ± 0.0003
Fixed dunes	66	1221 ± 783	41.9 ± 1.7	2.10	66	1.206 ± 0.244	1.84 ± 1.71	0.0222 ± 0.0042
Dune slacks	28	1257 ± 750	43.2 ± 1.5	3.02	28	0.97 ± 0.33	4.38 ± 3.32	0.0425 ± 0.011
Machair[1]	-	1221 ± 783	41.9 ± 1.7	2.10	8	1.107 ± 0.259	1.16 ± 0.51	0.0128 ± 0.0013
Saltmarsh, west coast (and soils 0-15 cm)	18	470 ± 390	60.0	3.43	18	0.766 ± 0.103	4.27 ± 1.98	0.0327 ± 0.002
west coast soils 15-50 cm	18	-	-	-	18	0.766 ± 0.103	1.27 ± 1.34	0.0097 ± 0.0014
Saltmarsh, east coast (and soils 0-30 cm)	36	470 ± 390	60.0	3.43	36	0.448 ± 0.03	5.45 ± 1.35	0.0244 ± 0.0004
east coast soils 30-100 cm[2]	-	-	-	-	-	0.448 ± 0.03	2.6	0.0116

[1]Machair biomass data use that for fixed dune grassland

[2]East coast saltmarsh soils (30-100 cm) use estimates of organic matter content from intertidal mudflats.

Table 3. Carbon stocks (t) by habitat, vegetation and soil pools and by country. AG = Above Ground.

Habitat	Pool	Scotland	England	Wales	N. Ireland	UK
Mobile and semi-fixed dune	AG Veg C (t)	30.3	24.4	20.4	2.7	77.8
	BG (Root) C (t)	53.5	43.1	36.1	4.8	137.6
	Soil C (t)[1]	Negligible	Negligible	Negligible	Negligible	Negligible
Subtotal		83.8	67.5	56.5	7.6	215.4
Dune grassland	AG Veg C (t)	193.7	30.2	15.6	4.5	244.0
	BG (Root) C (t)	406.7	63.5	32.7	9.5	512.3
	Soil C (t)[1]	1145.2	178.7	92.0	26.9	1442.9
Subtotal		1745.6	272.4	140.3	40.9	2199.2
Dune slack	AG Veg C (t)	14.1	4.7	5.6	0.7	25.2
	BG (Root) C (t)	42.7	14.3	17.0	2.2	76.2
	Soil C (t)[1]	137.3	46.0	54.7	7.1	245.1
Subtotal		194.1	65.1	77.3	10.1	346.6
Machair	AG Veg C (t)	103.8	n/a	n/a	n/a	103.8
	BG (Root) C (t)	217.9	n/a	n/a	n/a	217.9
	Soil C (t)[1]	361.1	n/a	n/a	n/a	361.1
Subtotal		682.9				682.9
Saltmarsh	AG Veg C (t)	16.9	94.7	19.6	0.7	132.0
	BG (Root) C (t)	58.1	324.9	67.3	2.4	452.6
	Soil C (t)[2]	494.8	4324.7	573.1	20.6	5413.2
Subtotal		569.7	4744.3	660.0	23.7	5997.8
Total all habitats		3276.1	5149.3	934.1	82.3	9441.8

[1] Soil C to 15 cm

[2] Soil C to 50 cm on north and west UK coasts, and to 100 cm on south and east UK coasts.

Table 4: Estimation of long term CO_2 sequestration and release by UK coastal habitats, 2000 – 2060, and associated monetary values, expressed as Net Present Value, NPV UK£

	Habitat Type			
	Sand dune	Saltmarsh	Machair	Total
CO_2 sequestered, tonnes	9,073,004	13,955,301	1,504,711	24,533,016
CO_2 released, tonnes	1,433,463	4,026,735	202,996	5,663,194
Net CO_2 carbon stored*, tonnes	7,639,542	9,928,566	1,301,714	18,869,822
Value of CO_2 sequestered No habitat loss scenario Not discounted	£1135 million	£1794 million	£185 million	£3114 million
Value of CO_2 sequestered – value of CO_2 released Habitat loss scenario Not discounted	£878 million	£1109 million	£150 million	£2137 million
Loss in value as a result of habitat loss scenario Not discounted	£257 million	£685 million	£35 million	£977 million
Value of CO_2 sequestered No habitat loss scenario Discounted	£342 million	£541 million	£56 million	£939 million
Value of CO_2 sequestered – value of CO_2 released Habitat loss scenario Discounted	£278 million	£362 million	£48 million	£688 million
Loss in value as a result of habitat loss scenario Discounted	£64 million	£179 million	£8 million	£251 million

*net stored by given habitat, not incorporating calculations of sequestration by replacement habitat.

Figure 1: Estimated change in annual C02 sequestration provided by UK coastal habitats, 1900-2060, due to changes in area, applying a constant sequestration rate

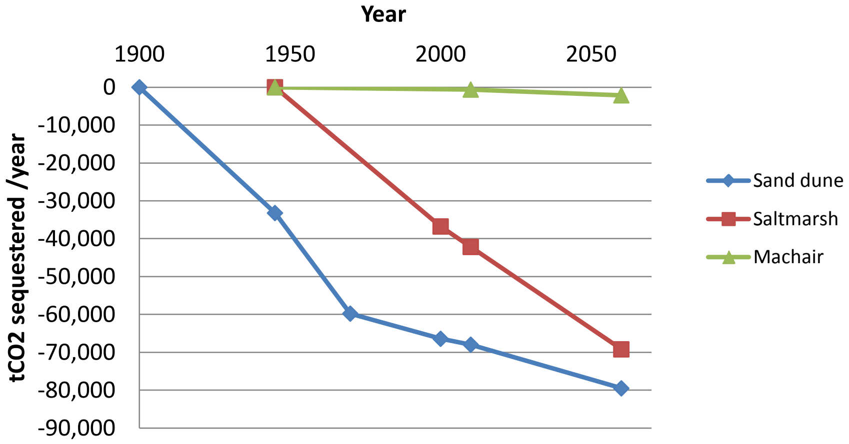

Figure 2: Estimated value of annual C02 sequestration service provided by UK coastal habitats, Sandune, Saltmarsh and Machair, 2000 – 2060. Applying average C sequestration rate and mid DECC (2011) non-traded carbon value.

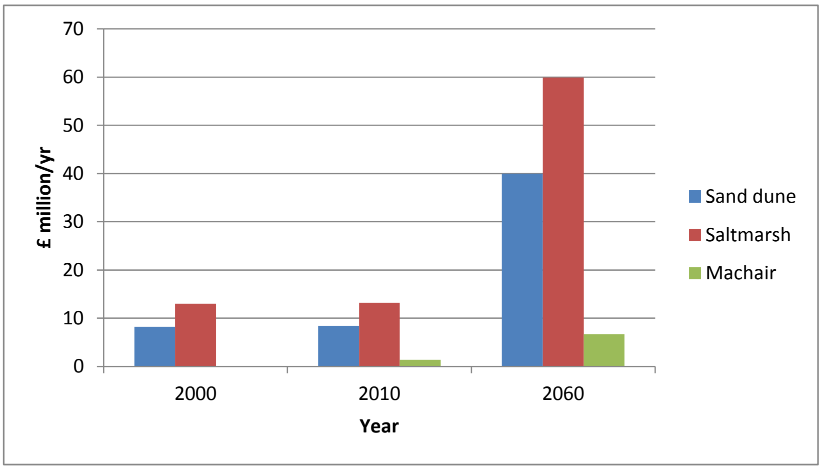

Figure 3: Uncertainty in the estimated value of annual C02 sequestration service provided by UK coastal habitats (combined Sandune, Saltmarsh and Machair) 2000 – 2060. Applying a range of C sequestration rates and low, mid and high DECC (2011) non-traded carbon values.