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Quantifying and valuing carbon flows and stores in coastal and shelf ecosystems in the UK

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ABSTRACT

Evidence shows that habitats with potential to mitigate against greenhouse gases emissions, by taking up and storing CO_2, are being lost due to the effects of on-going human activities and climate change. The carbon storage by terrestrial habitats (e.g. tropical forests) and the role of coastal habitats ('Blue Carbon') as carbon storage sinks is well recognised.

Offshore shelf sediments are also a manageable carbon store, covering ~9% of global marine area, but not currently protected by international agreements to enable their conservation. Through a scenario analysis, we explore the economic value of the damage of human activities and climate change can inflict on UK marine habitats, including shelf sea sediments.

In a scenario of increased human and climate pressures over a 25-year period, we estimate damage costs up to US$12.5 billion from carbon release linked to disturbance of coastal and shelf sea sediment carbon stores.

It may be possible to manage socio-economic pressure to maintain sedimentary carbon storage, but the trade-offs with other global social welfare benefits such as food security will have to be taken into account. To develop effective incentive mechanisms to preserve these valuable coastal and marine ecosystems within a sustainability governance framework, robust evidence is required.

1. Introduction

There is now evidence that vegetated habitats with potential to mitigate against greenhouse gases (GHG) emissions, by taking up and storing CO_2, are being lost due to the effects of both on-going human activities and climate change. The carbon storage by terrestrial habitats (e.g. tropical forests) is included in international agreements and economic incentive schemes to enable their conservation and realise their local and global community benefits (e.g. Reducing Emissions from Deforestation and Forest Degradation – REDD). The role of coastal habitats (i.e., mangroves, saltmarshes, and seagrasses) as carbon storage sinks, *i.e.* 'Blue Carbon', is now recognised too.

With this analysis we aim to contribute to fill in two important gaps in the literature of ecosystem services valuation. The first gap is that, in general, coastal and marine ecosystem services have not received the same attention as terrestrial ecosystem services, with some gaps still present in terms of their monetary valuation for the benefits they provide to humans (Turner et al., 2014). It may be argued that this has been because of the uncertainty related to the complexities of the functions and processes of the marine ecosystems, aggravated by lack of relevant data. Even if the benefits provided by the coastal shelf are among those that have been most frequently valued in the literature, Turner et al. (2014) find that, globally, only few value estimates are available for the regulating service of carbon sequestration and storage in coastal and marine ecosystems. Moreover, given the nature of public good of most of the services provided by marine ecosystems, some services, and especially the regulating services, are usually not well recognized by individuals, which leads to their limited developed governance and management (Costanza et al., 2017; Watson et al., 2016).

Offshore shelf sediments cover ~9% of global marine area and are also a substantial and potentially manageable carbon store (Diesing,

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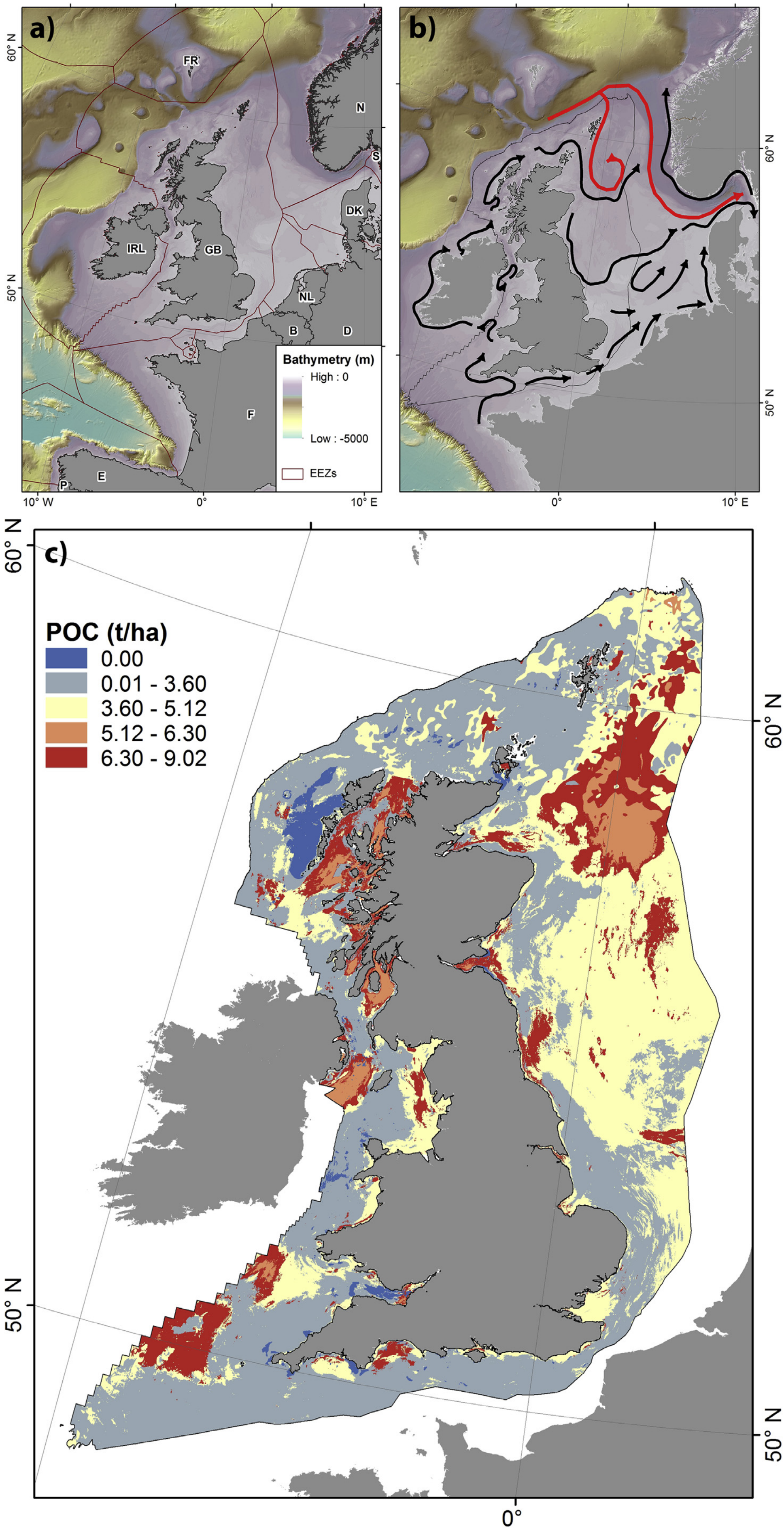

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Fig. 1. a) Exclusive Economic Zones (Claus et al., 2018) of the UK and neighbouring countries. Bathymetry from Marine Information Service (2016); b) Generalised water circulation patterns (after Hill *et al.*, 2008, modified); c) Particulate organic carbon (POC) areal stocks of the upper 10 cm of the sediment column in t/ha; based on mean POC concentrations and dry bulk densities for different substrate types (Diesing et al., 2017: Table 4).

et al., 2017). However, they are not yet recognised as having the potential to contribute to climate change mitigation, and are not currently protected by international agreements to enable their conservation and realise their local and global community benefits (da Silva Copertino, 2011). Using the United Kingdom's coastal habitats and territorial waters – comprising 2% of the global ocean shelf area (Harris et al., 2014) – as a preliminary regional study (Fig. 1), we demonstrate the scale and location of shelf carbon storage and explore the economic value of the damage of human activities and climate change inflicted on the UK marine habitats, including shelf sea sediments, through a scenario analysis over a 25-year period. This allows, for the first time, a demonstration of the importance of the ecosystem service of offshore shelf sea carbon storage and the economic value of the damage caused if carbon stored is released. The study aims to demonstrate the need for an appropriate governance framework that takes into account the value of marine habitats, including shelf sea sediments.

Management of coastal and marine habitats is challenging because they are subject to multiple pressures threatening their conservation. Managing the differing scales of these pressures together is challenging, as shown by the continuing ongoing loss of saltmarsh globally (Rupp-Armstrong and Nicholls, 2007) despite local restoration efforts. Marine ecosystem management focuses on maintaining ecosystem structure and process and is done at national level. However, the economic value of the carbon storage service provided by these ecosystems is a global social welfare benefit with international agreement implications.

In terms of mitigating climate change, a key terrestrial and marine habitat management issue is maintaining carbon uptake and minimising CO_2 release from the remineralisation of organic carbon, due to habitat loss or disturbance, with consequent societal global damage costs (Dasgupta and Ehrlich, 2013). For coastal habitats such as saltmarshes, areal loss is the main disturbance, resulting in loss of carbon storage capacity and remineralisation of sedimentary carbon stores. For shelf sediment management, bottom trawling is likely to be the most widespread pressure. Fishing gear towed along the seabed disturbs ecosystem function and resuspends sediment. This is a global scale issue (Hiddink et al., 2017). Trawling impacts up to 75% of continental shelf sediments globally, with almost 20 million km^2 of sediments subject to trawling once or more per annum (Kaiser et al., 2002). Bottom trawling affects sedimentary carbon storage through remineralisation of the resuspended sedimentary organic carbon, altering the depth and rate of organic carbon burial and by changing the seabed communities involved in bioturbation and bio-irrigation (Duplisea et al., 2001). While the long-term effects of trawling on carbon storage are poorly known, the amounts of sediment resuspended globally are large (~22 Gt/y) (Oberle et al., 2016), and a recent study showed trawling affecting sediments to a depth of 10 cm with a 52% reduction in organic carbon storage, slower carbon turnover and reduced meiofauna abundance and biodiversity (Pusceddu et al., 2014).

2. Material and methods

2.1. Case study location and scenarios

Using a scenario analysis, we investigate the economic value of the global damage costs to society due to remineralisation following disturbance or loss of the UK's coastal and shelf sea carbon stores. The case study area includes the coastal regions of England, Scotland, Wales and Northern Ireland, and associated UK territorial waters shelf seas (i.e. the Celtic, Irish and North Seas) to the 200 m depth contour. UK overseas territories or territorial deep-sea regions were not considered.

Clearly a range of scenarios could be considered with reference to the marine environment. Two basic scenarios of environmental change are considered over 25 years (2016–2040): Business as usual (BAU) and Continuous growth and climate change (CG&CC). Based on Griscom et al. (2017), we consider also a Restoration scenario. Griscom et al. investigate the mitigation potential linked to the conservation and restoration of what they define 'natural climate solution' (NCS), a set of cost-effective and low-cost actions aimed to increase carbon storage and/or avoid greenhouse gas emissions in forests, wetlands, grasslands, and agricultural lands. Similar to what we highlight here, Griscom et al. assume that the ecosystems they consider can contribute to climate mitigation both through enhanced carbon sinks, via restoration, and reduced emissions, via conservation. Griscom et al. analysis is based on well-defined international governance agreements such as the Paris Climate Agreement to keep the global average temperature to below 2C above pre-industrial levels, and the Intergovernmental Panel on Climate Change (IPCC) scenarios as these are consistent with Paris Climate Agreement temperature target. In our analysis, we are challenged by the lack of governance structures and agreements for the management of carbon in shelf ecosystems. However, Gallo et al (2017) report that Ocean issues are now receiving more attention in climate negotiations such as, for example, in the nationally determined contributions (NDCs). Also, conservation and restoration actions are under development for both seagrasses (van Katwijk et al., 2016) and shelf sea sediments (Cooper et al., 2013). To shed some initial light on this context, we report here a scenario including the restoration of saltmarshes. We limit our restoration analysis to saltmarshes because of the uncertainty surrounding seagrass restoration, which is considered challenging, and large-scale planting is required to increase chances of success (van Katwijk et al., 2016). According to Cooper et al. (2013) it is the case that seabed restoration costs have been judged as not cost-effective on an industrial scale when carried out at a specific site only. They also argue that the physical and biological impacts on the seabed can persist long after aggregate dredging has taken place (Cooper et al., 2013). Further, it is worth noting that Griscom et al. (2017) highlight the fact that wetland restoration is in general more expensive than conservation, while van Katwijk et al. (2016) confirm the need to remove the environmental change threats prior to replanting of seagrasses for restoration.

For BAU, we assume saltmarsh reduction is due to tidal squeeze from sea level rise, but without significant erosion of the sediment POC (particulate organic carbon) stock (*i.e.* the existing carbon stored remains buried, but the lost saltmarsh area can no longer take up CO_2). Seagrasses are similarly lost due to a range of impacts and again we assume that existing stored carbon is not remineralised to CO_2. We assume bottom trawling resuspends shelf sediment dependent on trawling frequency without causing loss of shelf sea habitat but remineralises the resuspended organic carbon.

The second scenario, CG&CC, predicts a larger areal loss of seagrass habitats, major sediment carbon loss from saltmarshes (*i.e.* the saltmarshes are lost and the carbon previously stored is remineralised), and higher trawling frequency for shelf sediments resulting in greater total organic carbon remineralisation.

We assume that all organic carbon resuspended by bottom trawling is remineralised and released to the atmosphere for both scenarios and consider this assumption further below. This represent a critical issue in any estimates of the carbon vulnerability as also discussed by Lovelocket al. (2017). Given the considerable uncertainties in estimating the effect of trawling on carbon storage, this approach allows us to quantify an upper bound on the potential value lost due to trawling-driven carbon release. The amounts of organic carbon oxidised and the valuation of the carbon scale linearly, so if we alternatively assume that

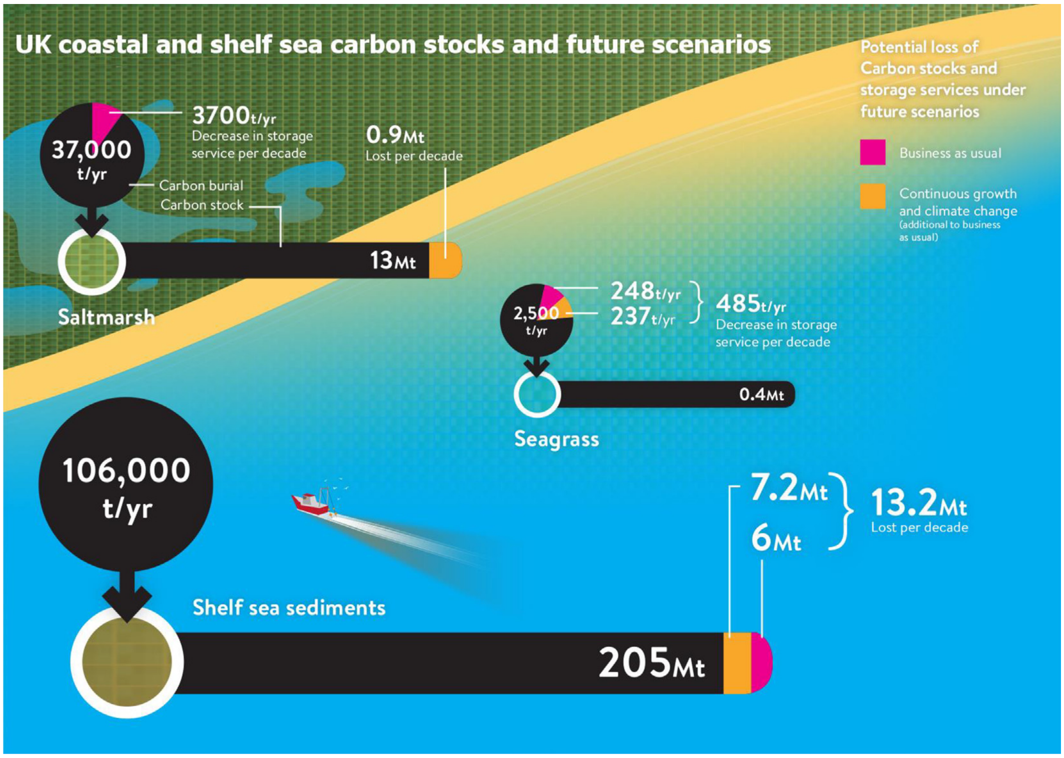

Fig. 2. United Kingdom's coastal and shelf sea carbon stocks and potential loss of carbon stocks and storage services under future scenarios with colours representing different scenarios and magnitudes and fluxes as indicated. For the economic values calculated for the loss of carbon sequestration and storage fluxes and previously stored carbon for each ecosystem considered (saltmarshes, seagrasses, and shelf sea sediments) in the two scenarios presented as discussed in the text, see the economic analysis results in Table 1. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

half the resuspended carbon is oxidised then the implied societal costs also halve.

To estimate the size of the UK's coastal and shelf sea carbon store, we calculated organic carbon storage in saltmarshes, seagrass meadows and shelf sea sediments resulting in an estimate of ~220 megatonnes (Mt) (Fig. 2). Shelf sea sediments are the dominant component (~93%) of coastal and shelf sea carbon stores; saltmarshes and seagrass store more carbon per unit area, but their areas are small relative to shelf sediments (Fig. 2). This emphasises that shelf sediments are an important carbon stores both locally and indeed globally (Bauer et al., 2013) and management action to retain and enhance this store is clearly of importance.

Following Griscom et al. (2017), the Restoration scenario aims to analyse the potential to counterbalance the CO_2 emissions following land-use conversion of saltmarshes with enhanced carbon storage in newly re-created saltmarshes, contributing to the achievement of keeping the global warming temperature to below 2C. Despite its limitations (Mossman et al., 2012), managed realignment (MR), a soft defence technique by which sea walls are breached to let the sea inundate the land behind the seawall (Sullivan et al., 2017), is an effective option to restore and re-create saltmarshes. In fact, even if MR usually requires the sacrifice of low-lying agricultural land, and there are differences in functioning between restored and natural saltmarshes, Lawrence et al. (2018) find that compared to agricultural land restored saltmarshes still provide important ecosystem services.

2.2. Biogeochemistry – UK coastal and shelf sea carbon budget data

All habitats when disturbed can release previously stored carbon. Here we considered CO_2 from disturbed sediment carbon stores in saltmarshes, seagrasses and shelf sediments. We used appropriate styles and values for carbon release depending on habitat. For saltmarsh, we use new estimates, while for seagrasses we rely on literature values (Dickie et al., 2014). For shelf sediments, we used a combination of North Sea organic carbon data, estimates of trawling resuspension and 1D modelling to simulate remineralization. We concentrated on the uppermost 10 cm of shelf sediment because sediment in this depth zone is most vulnerable to trawling disturbance (Pusceddu et al., 2014) and as such can be influenced by management decisions.

Coastal and shelf sea habitats can also produce other greenhouse gases during the burial of organic carbon, particularly N_2O, that can potentially offset the organic carbon storage benefits. Our values are corrected for N_2O flux for saltmarshes (Adams et al., 2012). N_2O fluxes from shelf sediments to the water column are low compared to saltmarsh and may be negative (Laursen and Seitzinger, 2002); our calculations for offshore sediment carbon burial are therefore not corrected for such releases.

We have not considered some aspects of shelf sea carbon cycling. The exchanges of carbon (principally inorganic) between the shelf and the open ocean, primarily across the northern boundary of the North Sea, are not realistically manageable (notwithstanding changes caused by ocean acidification and global warming) and are not considered. The total organic carbon stock in UK Holocene (last 10,000 years) intertidal sediments was not included because this carbon stored at depths below 10 cm is largely invulnerable to human activity impacts due to its burial depth.

There are intertidal mud and sand flats that probably have carbon storage values per unit area an order of magnitude higher than shelf sediments, although we do not have reliable area and carbon estimates for these habitats; therefore, we did not consider them as their area is < 1% of the shelf sediment area and excluding these does not impact our final carbon storage estimates.

2.2.1. UK saltmarsh

UK saltmarsh cover an area of approximately 42,712 ha. Modern UK estuarine areas (hectares of saltmarsh, mudflat and sandflat and totals) were taken from Davidson and Buck (1997). For each modern area, a carbon density was calculated based on organic carbon content, bulk density and assumed annual sedimentation rate appropriate for each geographic region. Using carbon density, a carbon burial per unit area per annum was calculated.

To estimate carbon storage in the intertidal Holocene sediment prism the Humber sediment volumes and facies types published in Rees et al. (2000) were used as they were the best quality data available. Volumes were converted to mass assuming bulk densities in Adams (2008) and Parkes (2003).

For all other Holocene sediment volumes/masses, it was assumed that Humber/Fenland represented half the sedimentary mass of eastern England and it was further assumed that eastern England represents about one third the mass of UK saltmarsh sediments as a whole. Saltmarshes in the UK are predicted to suffer a further 8% loss of the current habitat area by 2060 (Jones et al., 2011) because of coastal erosion, compounded by sea-level rise and reduced sediment supply, which corresponds to a yearly loss of saltmarsh of 0.16%. According to French (1997) 4.5% of saltmarsh area have been lost over the last twenty years (i.e. 1990–2010) because of sea level rise, which is equivalent to an annual loss of 0.225%. Other authors estimated a net annual loss of saltmarsh habitat of 2% (Nottage and Robertson, 2005).

Under the BAU scenario, we assume a constant annual reduction of UK saltmarsh, equivalent to 1% of the original area. This is mainly due to tidal squeeze from sea-level rise, but without significant erosion of the sediment carbon stock. This means a 1% reduction of the current 42,712 hectares of UK saltmarsh that store 36,732 tonnes of sediment carbon per year. This amounts to about 367 tonnes of organic matter that is not stored per year. In the year of disturbance, we have to add the loss of 1% of the standing stock of plants, currently storing 132 tonnes of C according to Beaumont et al. (2014). This therefore means 1.3 tonnes of C not removed from the atmosphere, to give a total of around 369 tonnes per year not removed from the atmosphere. This is an upper figure as it assumes all lost C is converted to CO_2. In the case of saltmarshes, much of the sediment carbon is root material (Andrews et al., 2011) which may not readily oxidise, although we might speculate that some will.

For the re-emission of carbon (as CO_2) into the atmosphere due to loss or erosion of saltmarsh under the CG&CC scenario, we also envisage a constant annual reduction of UK saltmarsh extent, equivalent to 1% of the original area. However, there is also an additional impact, albeit in a rather extreme way, from invasive initiatives like port building or substantial erosion caused by increased storminess. We envisage that this results in the top 1 m of sediment carbon removed from the 1% reduced area and oxidised to CO_2. From this, we calculate that 300 tonnes C/ha (~1000 tonnes CO_2/ha) would be lost, a figure similar to that of Pendleton et al. (2012) who estimated 250 tonnes C/ha.

In the Restoration scenario, we assume that over the 25-year time horizon considered (2016–2040), 1% of the current 42,712 hectares of UK saltmarshes is re-created annually via MR to compensate the recorded annual loss of saltmarshes. Similar to the BAU scenario, this means an extra 367 tonnes of organic matter stored per year. However, we consider a full C storage service at these new areas functioning properly starting 10 years after their re-creation. In fact, it has been observed that the functioning of the newly re-created saltmarshes reaches a capacity similar to that of reference marshes after decades (Mossman et al., 2012). We assume the same C burial rate used for the BAU and CC&CG scenarios, although it has been argued that re-created saltmarshes might even increase carbon storage capacity due to the community diversity (Lawrence et al., 2018).

2.2.2. UK seagrasses

Data on seagrasses is patchy both at the global and the UK level, although there is evidence of their past decline and subsequent recovery both globally and nationally. Seagrass areas (*Zostera marina* and *Zostera noltii*) are poorly mapped in the UK, but are probably of the order of 9000 ha (2300 ha in Northern Ireland (Department of Agriculture Environment and Rural Affairs, 2003), 1600 ha in Scotland (Burrows et al., 2014) and 4500 ha in England and Wales (Dickie et al., 2014)). Mapped *Z. marina* beds total 4887 ha, i.e. approximately 50% of the total UK sea grass bed area, thus an average storage and burial rate (N. Atlantic seagrass data from Fourqurean et al. (2012) was used for budget calculations). Seagrasses may continue to decline in the future but those located within marine protected areas (MPAs) where physical impact may decrease could show a future increase in quality and extent. Here we use the range 25–49% of seagrasses decline estimated by Hiscock et al. (2005) between 1980 and the beginning of the 21st century (about 25 years) and assume a future decline of seagrasses equal to 1% per year (BAU) and 2% per year (CG&CC).

2.2.3. UK shelf sea sediments

The total area of the UK shelf is approximately 51,797,389 ha. Organic carbon concentrations in the top 10 cm of shelf sea sediments were spatially predicted using a validated Random Forest regression model[1] based on 849 measurements of organic carbon in surficial seabed sediments at varying depths between 0 and 10 cm and a number of important and uncorrelated predictor variables (mud fraction of the surficial seabed, annual average bottom water temperature, geographic position (eastings), Euclidean distance to coast, gravel fraction of the surficial seabed and peak wave orbital velocity at the seabed) for parts of the north-west European continental shelf (Diesing et al., 2017).

However, this model did not cover the whole UK shelf area. We therefore utilised mean values of organic carbon concentrations and dry bulk densities for various substrate types provided by Diesing et al. (2017: Table 4) and a map of substrate types on the UK shelf. The latter was compiled by combining data from Stephens and Diesing (2015) and the British Geological Survey's digital marine sediment map called DigSBS250 (https://www.bgs.ac.uk/products/offshore/DigSBS250.html). Areal stocks of substrate types (Fig. 1) were calculated by multiplying dry bulk density, organic carbon concentration (as a fraction) and the sediment depth (0.1 m).

We concentrated on the uppermost 10 cm of shelf sediment because sediment in this depth zone is most vulnerable to trawling disturbance and as such can be influenced by management decisions. A mean annual burial rate for organic carbon (2 kg/ha) was derived from a North Sea average value (de Haas et al., 1997) and applied for the whole UK shelf. An estimated value of sediment resuspension due to bottom trawling of 18.84 kgC/ha/year per trawl pass was used for the whole of the shelf sediment area of 51,797,389 ha. This was calculated by combining release data from trawling plume studies in a temperate shelf sea region with the ratio of sediment types (muds and sand) for the North Sea (van der Molen et al., 2013), and assuming this sediment distribution is representative of the whole UK shelf. For mud-rich areas an average release of 27.5 kgC/ha was derived from measured concentrations associated with a trawl pass (Dounas et al., 2007; Durrieu de Madron et al., 2005) and applied to an area of 8,765,500 ha. For the sandier areas, direct measurements of plume release are unavailable. To account for expected lower sediment carbon concentrations (Koster and Meyer-Reil, 2001) in these sandier sediments, and shallower gear penetration (Ivanović et al., 2011), an estimated scaled release of 1 kgC/ha was estimated for an area of 4,255,100 ha.

The assumption that all resuspended carbon is remineralised to CO_2 was made due to a lack of other suitable data and represents an upper limit but is similar to assumptions made elsewhere (Lovelock et al., 2017; Pendleton et al., 2012). The dynamics of resuspension and settling following a trawl pass are complex, as are the kinetics of bacterial remineralisation. 100% remineralisation was thus assumed, to set an upper bound. Our analysis also assumes that all areas of the shelf are trawled equally, which is not likely. There is some evidence that repeat trawl passes have little effect on carbon loss, or even lead to small gains in carbon concentration (Hale et al., 2017). However, the most frequently trawled sediments tend to be the most carbon rich (as they are associated with greater numbers of fish). These two considerations act in opposing directions: carbon rich sediments being trawled in preference increasing the likely carbon loss, while diminished effects of secondary trawls would lead to a decrease, potentially somewhat mitigating the uncertainty due to this assumption. Overall the analysis is coarse, but indicates likely upper magnitude estimates for carbon release. It should also be noted that the distribution of sediment carbon stores within the region are not uniform (Fig. 1d) and neither is trawling pressure (Eigaard et al., 2016).

Given the complex seasonal cycling in shelf seas, both in well-mixed and seasonally stratifying regimes, it is not a given that remineralised carbon is automatically available for exchange with the atmosphere. This was tested in trawling scenarios using a simple 1-dimensional model of water-column processes which is detailed in the Supplementary material. Water column temperature was varied to represent future warming and POC was injected into the bottom layer of the model to represent release from sediments due to trawling. Both single box (well mixed water column) and 2-box (seasonally stratifying, with separation in spring and mixing of the 2 boxes in autumn) were explored, taking a Monte Carlo approach to exploring parameter space and providing a probability density function of the resultant fate of the DIC (dissolved inorganic carbon) released from POC disturbance. Overall the dominant control on net release of carbon to the atmosphere was found to be the intensity of trawling (a function of the depth to which carbon was disturbed, the POC content of the sediment, and the fraction redeposited without mineralisation). The effect of warming was to enhance trawl-related DIC release to the atmosphere by 21% and 23% on average, for seasonally-stratifying and well-mixed waters respectively. This is due to a combination of more rapid remineralisation of resuspended material and reduced solubility of CO_2 under warmer conditions. The timing of trawling also affected the amount of CO_2 released to the atmosphere, with mean release broadly following temperature i.e. summer trawling tended to lead to greater remineralisation of resuspended POC due to higher temperatures. In all cases the modelled release of CO_2 to the atmosphere was not significantly different from 100% of the DIC released from resuspended trawled material. Hence, we conclude that effectively all organic carbon oxidised will be released to the atmosphere as CO_2. The model does not account for 3-dimensional processes such as transport off-shelf, which may be locally important in shelf sediments adjacent to downwelling regions but is not thought to affect results in most parts of the shelf.

2.3. Economic analysis

We present the cost to society of the carbon released by human activities in both BAU and CG&CC scenarios using a range of valuations of carbon given the uncertainty surrounding the carbon value used for appraisals. The same reasoning is applicable when considering the benefits of restoration. The only difference is a negative sign for any change that encompasses a loss of habitat or ecosystem service, which determine a cost to society, and a positive sign for any change including an increase in ecosystem areas and ecosystem service provision, which implies a benefit to society. The range includes two estimates based on social cost of carbon calculations, which take into account the damage costs of current GHG emissions imposed on future generations (Tol, 2005; Nordhaus, 2017); and one based on the abatement cost, which considers the cost of mitigating carbon emissions and is the method adopted by the UK government (BEIS, 2017). Within the time horizon considered, 25 years, BEIS (2017) provides increasing annual carbon economic values per tonne of C (between 2016 and 2040 the range of central estimates of carbon values in the non-traded sector starts with US\$282.69 and ends with US\$669.53 – mean conversion rate in 2016: 1US\$ = 0.74£); US\$50 per tonne of C is the suggested upper value by Tol (2005) and we assume that this stays constant over the whole time horizon; Nordhaus (2017) provides increasing carbon values, per tonne of C, for the years 2015 (US\$114.5), 2020 (US\$136.891), 2025 (US \$161.48), 2030 (US\$189.37), 2050. Between these years we assume that the values remain constant.

The present value (PV) of a change in the amount of the carbon sequestration and storage benefit lost at time t (for t between zero and T) ($PVCB_t$) is presented in Eq. (1). The discount factor $\frac{1}{(1+r)^t}$ yields the value of the flows of lost carbon sequestration and storage service at time t, ($\Delta s_t + V_t$), where Δs_t represents a change in the carbon sequestration and storage service at time t, and V_t is the economic value at time t used to measure the carbon sequestration and storage benefit lost in monetary units. The discount rate (r) chosen is the declining discount rate used in UK for policy appraisal (HM Treasury, 2018); for the first 30 years of an environmental project a discount rate of 3.5% is advised.

$$PVCB_t = \sum_{t=0}^{T} \frac{1}{(1+r)^t}(\Delta s_t \times V_t) \tag{1}$$

The magnitude of Δs_t (see Eq. (2)) is determined by the change in the area of the coastal ecosystem that is disturbed(Δa_t^P) multiplied by the consequent change (e.g. loss) of carbon burial measured in tonnes of C per unit area at time t, (ΔCb_t), and the consequent change in carbon fluxes released in the atmosphere from the disturbed stock also measured in tonnes of C per unit area at time t, (ΔCf_t).

$$\Delta s_t = \Delta a_t^P \times (\Delta Cb_t + \Delta Cf_t) \tag{2}$$

ΔCb_t and ΔCf_t can be determined by biogeochemical sampling and/or modelling. In our study, $\Delta s_t = \Delta a_t^P \times (\Delta Cb_t + \Delta Cf_t)$ relates to the case of saltmarshes and seagrasses. For shelf sea sediments we simply write $\Delta s_t = \Delta a_t^P \times \Delta Cf_t$ because in the case of shelf sea sediments we assume that the carbon burial process does not change, whereas that is lost when saltmarshes or seagrasses are lost.

From an economic point of view, it is important to distinguish between the two fluxes ($\Delta Cb_t + \Delta Cf_t$). The ecosystem service loss is represented by ΔCb_t. By definition ΔCf_t can only be negative or zero, thus we are in the presence of a negative externality that should be accounted for in the management of coastal and shelf sea stores. If ΔCf_t is caused by an unmanageable pressure such as climate change (e.g. storminess – extreme event; or sea level rise), then, technically that externality should be considered as a societal loss of force majeure type (society as whole then may be asked to pay, e.g. through a tax or international agreements, to reduce pollution/emissions).

3. Results and discussion

The results of the biogeochemistry analysis are presented in Fig. 2. The infographic shows that there are large carbon stores in the UK's coastal and shelf sea areas. Fig. 2 clearly shows that in the UK, the shelf sea sediment store is larger than the coastal store (i.e. saltmarshes and seagrasses combined). However, it also shows that in both future scenarios considered, BAU and CG&CC, there is the potential for loss of carbon stocks and storage services. This suggests that both coastal and shelf sea sediment carbon stores are subject to different human pressures showing a different type of vulnerability for each store. In fact, saltmarshes and seagrass store more carbon per unit area, but these areas are small relative to shelf sediments (Fig. 2). Therefore, both coastal and shelf sea sediment stores, when disturbed, can release substantial amount of previously stored carbon. Since this would apply at the global level too, both carbon stores should be internationally protected in order to help mitigating the effects of climate change. From an economic point of view, over just 25 years, in the CG&CC scenario, global economic damage costs of disturbances to the UK's coastal and shelf sea carbon stores that induce the release of previously stored carbon into the atmosphere could be up to a present value of US\$12.5 billion (Table 1).

In Table 1 we report the present value ($PVCB$) in 2016 of a change in the amount of the carbon sequestration and storage benefit lost within a

Table 1
Present Value (US$) of the economic loss of the carbon sequestration and storage benefit due to climate change effects and anthropocentric disturbances to coastal and shelf sea carbon stores in the BAU and CG&CC scenarios over 25 years (2016–2040). The PVCB is reported in its two parts: the economic value of ΔCb_t, the cost to society of the lost C sequestration and storage service due to spatial extent lost; and the economic value of ΔCf_t, the cost to society of C released.

Blue Carbon compartment	Annual tC storage lost due to spatial extent loss[a]	Net tonnes of C released per year due to other disturbances[a]	Other disturbances	The cost to society of the lost C sequestration and storage service due to spatial extent lost (US$ billion)			The cost to society of C released (US$ billion)[b]		
				Abatement cost: all relevant year values (BEIS, 2017)[*]	Social Cost of Carbon: US$50 (Tol, 2005)	Social Cost of Carbon: all relevant year values (Nordhaus, 2017)	Abatement cost: all relevant year values (BEIS, 2017)[*]	Social Cost of Carbon: US$50 (Tol, 2005)	Social Cost of Carbon: all relevant year values (Nordhaus, 2017)
Business as usual (BAU)									
Saltmarsh	369[b]	n/a	n/a	0.002	0.0003	0.0009			
Seagrass	25	n/a	n/a	0.00015	0.00002	0.00006			
Shelf sea sediments	n/a	975,856	0.02 tC/ha/year released from one trawl pass				6	0.8	2.5
TOT Present Value loss value				0.00215	0.00032	0.00096	6	0.8	2.5
Continuous growth and climate change (CG&CC)									
Saltmarsh	369[b]	91,771[c]	300 tC/ha released due to disturbance of top 1 m	0.002	0.0003	0.0009	0.5	0.075	0.2
Seagrass	50	n/a	n/a	0.0003	0.00004	0.0001			
Shelf sea sediments	n/a	1,951,712	0.04 tC/ha/year released from two trawl passes				12	1.6	5
TOT Present Value loss value				0.0023	0.00034	0.001	12.5	1.7	5.2

[a] It is assumed that the annual loss rate in habitat area is constant. This annual loss in habitat area is the equivalent value of a given proportion of the original spatial extent of the habitat (see Fig. 2). For example, the existing size of saltmarsh is 42,712 hectares. Under the BAU scenario, it is assumed that 1% of this area will be lost annually. An estimated value of sediment resuspension due to bottom trawling of 18.84 kgC/ha/year per trawl pass was used for the whole of the shelf sediment area of 51,797,389 ha. See method and supplementary material for details on the calculations of carbon estimates.

[*] Mean conversion rate in 2016: 1 US$ = 0.74£

[b] These values take into account the storage lost from both the plant material and the sediment material in saltmarshes.

[c] These values are for year 1 only. Net tC/tCO_2e released gradually increases over time because of the loss in spatial extent of the habitat and the loss of the top 1 m of sediment; *i.e.* the habitat releases more than it can store due to the combined effects of the spatial and the release due to the damage to the top 1 m of the habitat.

Table 2
Goods/Benefits provided by saltmarshes, seagrasses, and shelf seas – adapted from Turner et al. (2015).

Goods/Benefits provided	Saltmarshes	Seagrasses	Shelf sea sediments
Provisioning services			
	Food (through fish nurseries)	Food (through fish nurseries) Fertiliser and biofuels Medicines and blue biotechnology	
Regulating services			
	Healthy climate (through carbon sequestration and storage) Prevention of coastal erosion Sea defence Waste burial/removal/neutralisation	Healthy climate (through carbon sequestration and storage) Prevention of coastal erosion Waste burial/removal/neutralisation	Healthy climate (through carbon sequestration and storage) Waste burial/removal/neutralisation
Cultural services			
	Tourism and nature watching Spiritual and cultural well-being Aesthetic benefits Education, research Human health benefits	Education, research	Education, research

time horizon of 25 years. We show this present value in its two parts: the economic value of ΔCb_t, the cost to society of the lost C sequestration and storage service due to spatial extent lost; and the economic value of ΔCf_t, the cost to society of C released. In both cases we use the three different values of carbon described at the beginning of Section 2.3.

In the CG&CC scenario, the cost to society of the carbon released ranges between: US$1.7 billion using the social cost of carbon approach (Tol, 2005); US$5.2 billion using Nordhaus' mixed approach of social cost of carbon and abatement cost (Nordhaus, 2017); and US$12.5 billion using the UK's abatement cost approach (BEIS, 2017). As shown in Table 1, the cost to society of the lost C sequestration and storage service due to habitat spatial extent lost is much lower than the cost of released carbon upon habitat disturbance. However, for policy purposes, if the loss of other important ecosystem services and related goods/benefits provided by the lost ecosystems (see Table 2) were also considered, the economic losses related to the loss and/or disturbance of saltmarshes, seagrasses, and shelf sea sediments may be higher than those presented here.

A restoration scenario would compensate for the current annual loss of saltmarshes through economic development. However, since the functioning of re-created saltmarshes reaches its full capacity after decades, in our calculations we find that a Net Present Value of the carbon sequestration and storage benefit over a 25-year time horizon (2016–2040) would, depending on the carbon value adopted, be equal to: US$1.6 million using the social cost of carbon approach (Tol, 2005); US$6 million using Nordhaus' mixed approach of social cost of carbon and abatement cost (Nordhaus, 2017); and some US$10 million using the UK's abatement cost approach (BEIS, 2017).

Although our analysis is for the UK shelf, as noted earlier, this is a large shelf and subject to similar biogeochemistry and environmental pressures to other shelves. We therefore think that our conclusions about the importance of carbon storage in shelf systems is applicable to other shelf systems.

The damage to coastal and marine habitats, including shelf sediments, are therefore likely to have consequences for the global climate as severe as deforestation. However, shelf carbon storage service provision could be guaranteed through the local management of coastal and shelf sea ecosystems supported by economic mechanisms such as the 'polluter pays' principle, coupled with a currently lacking appropriate international governance process and legislative framework. While it may be possible to manage pressure to maintain sedimentary carbon storage, trade-offs with other global social welfare benefits such as food security may arise. New management strategies aimed at reducing trawling intensity, and its consequent impact on benthic habitats and fish stocks, will need to take into account the effect this might have on the fishing industry. When compared to the costs to society related to the damages inflicted to coastal ecosystems such as saltmarshes, our analysis on the value of restoration supports the hypothesis that conservation would be a more economical and effective, as well as more sustainable, management action. However, as observed by Lawrence et al. (2018), restored saltmarshes still provide important ecosystem services compared to the agricultural land for which most intertidal habitats have been sacrificed for in the past.

To develop effective governance and management of ecosystems providing carbon storage benefits, robust evidence covering global biogeochemical cycles and in particular the fate of the mobilised organic carbon, the economic value of ecosystem stocks and flows affected by the direct and indirect impact of human activities, and the interplay between conservation of these ecosystems and co-located human activities, is necessary. In shelf seas, carbon cycling and storage services are often not geographically co-located, for example, CO_2 taken up to produce phytoplankton biomass in UK coastal waters will be transported by currents (Dyer and Moffat, 1998) and deposited in Dutch and Norwegian territorial waters (Fig. 1); this also needs to be recognised in the development of national and international management policies.

4. Conclusions

Offshore shelf sediments are a manageable carbon store, covering ~9% of global marine area, but not currently protected by international agreements to enable their conservation. Through a scenario analysis, we explored the economic value of the damage from human activities and climate change inflicted onto UK marine habitats, including shelf sea sediments. While the estimates of shelf sea carbon content of UK shelf sediments are quite well known, the estimates of sedimentation rates are less certain and the impact of trawling disturbance on this carbon store is also very uncertain.

In a scenario of increased human and climate pressures over a 25-year period, we estimate a present value of damage costs from carbon release ranging between US$1.7 billion using the social cost of carbon approach (Tol, 2005) and US$12.5 billion using the UK's abatement cost approach (BEIS, 2017), with an intermediate US$5.2 billion using Nordhaus' mixed approach of social cost of carbon and abatement cost (Nordhaus, 2017). When the costs of the damages to saltmarshes (i.e. loss of ecosystem service and CO_2 re-emission) are compared to the benefits of restoration aimed to compensate for the saltmarsh loss, our analysis shows that conservation would be more economical and effective than restoration.

These results highlight that offshore shelf sea sediments are important as carbon stores in the same way as tropical forests and blue carbon, and hence may play an important role in mitigating the effects of climate change. However, from a sustainability point of view, the trade-offs between conservation of the shelf sea sediments and the reduction in the provision of fish for food consumption has to be taken into account; the main challenge is finding the most appropriate incentive mechanism able to protect marine habitats whilst maintaining food security. Transboundary issues related to the location of storage and sequestration of carbon in the ocean will also have to be addressed within an appropriate governance framework in order to establish an equally appropriate carbon accounting framework.

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Declarations of interest

None.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoser.2018.10.013.

References

Adams, C.A., Andrews, J.E., Jickells, T., 2012. Nitrous oxide and methane fluxes vs. carbon, nitrogen and phosphorous burial in new intertidal and saltmarsh sediments. Sci Total Environ. 434, 240–251.
Adams, C., 2008. Carbon burial and greenhouse gas fluxes of new intertidal and saltmarsh sediments. School of Environmental Sciences, University of East Anglia, Norwich (UK) PhD thesis.
Andrews, J.E., Jickells, T.D., Adams, C.A., Parkes, D.J., Kelly, S.D., 2011. Sediment record and storage of organic carbon and the nutrient elements (N, P, and Si) in estuaries and near coastal seas. In: Wolanskiand, E., McLusky, D.S. (Eds.), Treatise on Estuarine and Coastal Science (Vol. 4, pp. 9–38). Academic, Waltham.
Bauer, J.E., Cai, W.J., Raymond, P.A., Bianchi, T.S., Hopkinson, C.S., Regnier, P.A., 2013. The changing carbon cycle of the coastal ocean. Nature 504, 61–70. https://doi.org/10.1038/nature12857.
Beaumont, N.J., Jones, L., Garbutt, A., Hansom, J.D., Toberman, M., 2014. The value of carbon sequestration and storage in coastal habitats. Estuar. Coast. Shelf Sci. 137, 32–40.
BEIS, 2017. Guidance on estimating carbon values beyond 2050: an interim approach.
Burrows M.T., Kamenos N.A., Hughes D.J., Stahl H., Howe J.A., Tett P., 2014. Assessment of carbon budgets and potential blue carbon stores in Scotland's coastal and marine environment. Scottish Natural Heritage Commissioned Report No. 761.
Claus S., De Hauwere N., Vanhoorne B., Souza Dias F., Oset García P., Schepers L., Hernandez F., Mees J., 2018. MarineRegions.org. Accessed at http://www.marineregions.org/eez.php on 2018-11-14 (Flanders Marine Institute).
Cooper, K., Burdon, D., Atkins, J.P., Weiss, L., Somerfield, P., Elliott, M., Turner, K., Ware, S., Vivian, C., 2013. Can the benefits of physical seabed restoration justify the costs? An assessment of a disused aggregate extraction site off the Thames Estuary, UK. Marine Pollut. Bull. 33–45.
Costanza, R., de Groot, R., Braat, L., Kubiszewski, I., Fioramonti, L., Sutton, P., Farber, S., Grasso, M., 2017. Twenty years of ecosystem services: how far have we come and how far do we still need to go? Ecosyst. Serv. 28, 1–16.
da Silva Copertino, M., 2011. Add coastal vegetation to the climate critical list. Nature 473, 255.
Dasgupta, P.S., Ehrlich, P.R., 2013. Pervasive externalities at the population, consumption, and environment nexus. Science 340, 324–328.
Davidson, N.C., Buck, A.L., 1997. An Inventory of UK Estuaries – Volume 1. Introduction and Methodology. JNCC.
de Haas, H., Boer, W., van Weering, T.C.E., 1997. Recent sediment and organic carbon burial in a shelf sea; the North Sea. Mar. Geol. 144, 131–146.
Department of Agriculture Environment and Rural Affairs, 2003. Northern Ireland Habitat Action Plan – Seagrass Beds.
Dickie, I., Cryle, P., Maskell, L., 2014. UK National Ecosystem Assessment Follow-on. Work Package Report 1: Developing the evidence base for a Natural Capital Asset Check: What characteristics should we understand in order to improve environmental appraisal and natural income accounts?
Diesing, M., Kröger, S., Parker, R., Jenkins, C., Mason, C., Weston, K., 2017. Predicting the standing stock of organic carbon in surface sediments of the North–West European continental shelf. Biogeochemistry 135, 183–200.
Dounas, C., Davies, I., Triantafyllou, G., Koulouri, P., Petihakis, G., Arvanitidis, C., Sourlatzis, G., Eleftheriou, A., 2007. Large-scale impacts of bottom trawling on shelf primary productivity. Cont. Shelf Res. 27, 2198–2210.
Duplisea, D.E., Jennings, S., Malcolm, S.J., Parker, R., Sivyer, D.B., 2001. Modelling potential impacts of bottom trawl fisheries on soft sediment biogeochemistry in the North Sea. Geochem. Trans. 112–117.
Durrieu de Madron, X., Ferré, B., Le Corre, G., Grenz, C., Conan, P., Pujo-Pay, M., Bodiot, O., Buscail, R., 2005. Trawling-induced resuspension and dispersal of muddy sediments and dissolved elements. Cont. Shelf Res. 25, 2387–2409.
Dyer, K.R., Moffat, T.J., 1998. Fluxes of suspended matter in the East Anglian plume, Southern North Sea. Cont. Shelf Res. 18, 1311–1331.
Eigaard, O., Bastardie, F., Breen, M., Dinesen, G., Hintzen, N., Laffargue, P., Nielsen, J., et al., 2016. Estimating seabed pressure from demersal trawls, seines, and dredges based on gear design and dimensions. ICES J. Marine Sci. 73 (Supplement 1), i27–i43.
Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marba, N., Holmer, M., Mateo, M.A., Apostolaki, E.T., Kendrick, G.A., Krause-Jensen, D., McGlathery, K.J., Serrano, O., 2012. Seagrass ecosystems as a globally significant carbon stock. Nat. Geosci. 5, 505–509.
French, P.W., 1997. Coastal and Estuarine Management. Routledge.
Hale, R., Godbold, J.A., Sciberras, M., Dwight, J., Wood, C., Hiddink, J.G., Solan, M., 2017. Mediation of macronutrients and carbon by post-disturbance shelf sea sediment communities. Biogeochemistry 135, 121–133.
Gallo, N.D., Victor, D.G., Levin, L.A., 2017. Ocean commitments under the Paris Agreement. Nat. Climate Change 833–838.
Griscom, B.W., Adams, J., Ellis, P.W., Houghton, R.A., Lomax, G., Miteva, D.A., Schlesinger, W.H., Shoch, D., Siikamäki, J.V., Smith, P., Woodbury, P., Zganjar, C., Blackman, A., Campari, J., Conant, R.T., Delgado, C., Elias, P., Gopalakrishna, T., Hamsik, M.R., Herrero, M., Kiesecker, J., Landis, E., Laestadius, L., Leavitt, S.M., Minnemeyer, S., Polasky, S., Potapov, P., Putz, F.E., Sanderman, J., Silvius, M., Wollenberg, E., Fargione, J., 2017. Natural climate solutions. PNAS 114, 11645–11650.
Harris, P.T., Macmillan-Lawler, M., Rupp, J., Baker, E.K., 2014. Geomorphology of the oceans. Mar. Geol. 352, 4–24.
Hiddink, J.G., Jennings, S., Sciberras, M., Szostek, C.L., Hughes, K.M., Ellis, N., Rijnsdorp, A.D., McConnaughey, R.A., Mazor, T., Hilborn, R., Collie, J.S., Pitcher, C.R., Amoroso, R.O., Parma, A.M., Suuronen, P., Kaiser, M.J., 2017. Global analysis of depletion and recovery of seabed biota after bottom trawling disturbance. PNAS 114, 8301–8306.
Hill, A.E., Brown, J., Fernand, L., Holt, J., Horsburgh, K.J., Proctor, R., Raine, R., Turrell, W.R., 2008. Thermohaline circulation of shallow tidal seas. Geophys. Res. Lett. 35.
Hiscock, K., Sewell, J., Oakley, J., 2005. Marine Health Check 2005. A Report to Gauge the Health of the UK's sea-life. WWF-UK, Godalming.
HM Treasury, 2018. The Green Book: Central Government Guidance on Appraisal and Evaluation.
Ivanović, A., Neilson, R.D., O'Neill, F.G., 2011. Modelling the physical impact of trawl components on the seabed and comparison with sea trials. Ocean Eng. 38, 925–933.
Jones, L., Angus, S., Cooper, A., Doody, P., Everard, M., Garbutt, A., Gilchrist, P., Hansom, J., Nicholls, R., Pye, K., Ravenscroft, N., Rees, S., Rhind, P., Whitehouse, A., 2011. Coastal margins. The UK National Ecosystem Assessment Technical Report. UK National Ecosystem Assessment. UNEP-WCMC, Cambridge.
Kaiser, M.J., Collie, J.S., Hall, J.S., Jennings, S., Poiner, I.R., 2002. Modification of marine habitats by trawling activities: prognosis and solutions. Fish Fish. 3, 114–136.
Koster, M., Meyer-Reil, L.A., 2001. Characterization of carbon and microbial biomass pools in Shallowater coastal sediments of the southern Baltic Sea (Nordrugensche Bodden). Mar. Ecol. Prog. Ser. 214, 25–41.
Laursen, A.E., Seitzinger, S.P., 2002. The role of denitrification in nitrogen removal and carbon mineralization in Mid-Atlantic Bight sediments. Cont. Shelf Res. 22, 1397–1416.
Lawrence, P.J., Smith, G.R., Sullivan, M.J.P., Mossman, H.L., 2018. Restored saltmarshes lack the topographic diversity found in natural habitat. Ecol. Eng. 58–66.
Lovelock, C.E., Fourqurean, J.W., Morris, James T., 2017. Modeled CO2 emissions from coastal wetland transitions to other land uses: tidal marshes, mangrove forests, and seagrass beds. Front. Marine Sci. 4.
Marine Information Service, 2016. EMODnet Digital Bathymetry (DTM 2016). Marine Information Service. https://dx.doi.org/10.12770/c7b53704-999d-4721-b1a3-04ec60c87238.
Mossman, H.L., Davy, A.J., Grant, A., 2012. Does managed coastal realignment create saltmarshes with 'equivalent biological characteristics' to natural reference sites? J. Appl. Ecol. 1446–1456.
Nordhaus, W.D., 2017. Revisiting the social cost of carbon. PNAS 114, 1518–1523.
Nottage, A.S., Robertson, P.A., 2005. The Salt Marsh Creation Handbook: A Project

Managers Guide to the Creation of Salt Marsh and Intertidal Mudflat. (RSPB Management Guides).
Oberle, F.K.J., Storlazzi, C.D., Hanebuth, T.J.J., 2016. What a drag: quantifying the global impact of chronic bottom trawling on continental shelf sediment. J. Marine Systems 159, 109–119.
Parkes, D.J., 2003. Storage and cycling of organic carbon and nutrients in Holocene coastal sediments. School of Environmental Sciences, University of East Anglia, Norwich (UK) PhD thesis.
Pendleton, L., Donato, D.C., Murray, B.C., et al., 2012. Estimating global "Blue Carbon" emissions from conversion and degradation of vegetated coastal ecosystems. PLOS One 7 (9), e43542.
Pusceddu, A., Bianchelli, S., Martín, J., Puig, P., Palanques, A., Masqué, P., Danovaro, R., 2014. Chronic and intensive bottom trawling impairs deep-sea biodiversity and ecosystem functioning. PNAS 111, 8861–8866.
Rees, J.G., Ridgway, J., Ellis, S., O'B. Knox, Newsham, R.W.R., Parkes, A., 2000. Holocene sediment storage in the Humber Estuary. In: Sheenan, I., Andrews, J.E. (Eds.), Holocene Land-Ocean Interaction and Environmental Change around the North Sea. The Geological Society, Bath, pp. 119–144.
Rupp-Armstrong, S., Nicholls, R.J., 2007. Coastal and estuarine retreat: a comparison of the application of managed realignment in England and Germany. J. Coast. Res. 23, 1418–1430.
Stephens, D., Diesing, M., 2015. Towards quantitative spatial models of seabed sediment composition. PlosOne.
Sullivan, M.J.P., Davy, A.J., Grant, A., Mossman, H.L., 2017. Is saltmarsh restoration success constrained by matching natural environments or altered succession? A test using niche models. J. Appl. Ecol. 1207–1217.
Tol, R.S.J., 2005. The marginal damage costs of carbon dioxide emissions: an assessment of the uncertainties. Energy Policy 33, 2064–2074.
Turner, K., Schaafsma, M., Elliott, M., Burdon, D., Atkins, J., Jickells, T., Tett, P., Mee, L., van Leeuwen, S., Barnard, S., Luisetti, T., Paltriguera, L., Palmieri, G., Andrews, J., 2014. UK National Ecosystem Assessment Follow-on. Work Package Report 4: Coastal and Marine Ecosystem Services: Principles and Practice. UNEP-WCMC, LWEC, UK.
Turner, R.K., Schaafsma, M., Mee, L., Elliott, M., Burdon, D., Atkins, J.P., Jickells, T., 2015. Conceptual framework. In: Turner, R.K., Schaafsma, M. (Eds.), Coastal Zones Ecosystem Services: From Science to Values and Decision Making. Springer.
van der Molen, J., Aldridge, J.N., Coughlan, C., Parker, E.R., Stephens, D., Ruardij, P., 2013. Modelling marine ecosystem response to climate change and trawling in the North Sea. Biogeochemistry 113, 213–236.
van Katwijk, M.M., Thorhaug, A., Marbà, N., Orth, R.J., Duarte, C.M., Kendrick, G.A., et al., 2016. Global analysis of seagrass restoration: the importance of large-scale planting. J. Appl. Ecol. 53, 567–578.
Watson, S.C.L., Paterson, D.M., Queirós, A.M., Rees, A.P., Stephens, N., Widdicombe, S., Beaumont, N.J., 2016. A conceptual framework for assessing the ecosystem service of waste remediation: in the marine environment. Ecosyst. Serv. 20, 69–81.