Blue Carbon – Mapping Risks and Opportunities

First published 17th January 2022

Natural England Research Report ME5440

Natural England Research Report ME5440

Blue Carbon – Mapping Risks and Opportunities

Ginny Swaile, Maija Marsh, Jessica Elias, Susan Burton, Dylan Todd, Peter Walker, Laura Gannon, Jasmine Elliott, Lucy Smibert, George Perry and Melanie Hartley

NATURAL ENGLAND

Published January 2022

Project details

This report should be cited as: G. Swale, M.K. Marsh, J.L. Elias, S.M. Burton, D. Todd, P. Walker, L. Gannon, J.M. Elliott, L. Smibert, G. Perry and M. Hartley (2022) Blue carbon – mapping risks and opportunities. Natural England Research Report ME5440 to Defra

Natural England Project Manager

Dr Maija Marsh (Maija.marsh@naturalengland.org.uk)

Keywords

Blue carbon, marine habitats, stocks, accumulation, mapping, anthropogenic activities, nutrients

Acknowledgements

We are grateful to the external experts who reviewed and provided comments on the report.

This project was funded by the Department for Environment Food and Rural Affairs (Defra), Marine and Fisheries Directorate, and commissioned by the Marine Biodiversity Impact Evidence Group (IEG). The IEG was established in March 2014 to co-ordinate evidence collection concerning impacts of human activities. Membership of the IEG at the time of publication is Defra, Welsh Government, Natural England, Cefas EA, the IFCAs, JNCC, NRW and the MMO.

Executive Summary

There is considerable focus around climate change and the actions required to meet international agreements, mainly focussed on reducing emissions of greenhouse gases and increasing carbon accumulation and storage. The marine environment plays a central role in climate regulation and the carbon cycle. It acts as a large carbon sink, with benthic habitats providing long term storage of carbon when natural processes are unfettered. The term 'blue carbon' refers to the carbon captured by coastal and marine ecosystems and stored in the vegetative biomass and sediments. However, human activities are known to disturb and reduce the quality and extent of marine habitats which in turn limits their ability to accumulate and store carbon and can even result in elevated carbon emissions.

This desk-based study sought to map the extent and distribution of blue carbon habitats in English waters and estimate the associated carbon accumulation and storage rates. The potential for recovery and restoration of coastal blue carbon habitats was explored alongside the key pressures that are driving habitat loss and inhibiting recovery, to identify opportunities for enhancing carbon storage.

The results show that the coastal blue carbon habitats (saltmarsh, intertidal mud and seagrass) are the richest in terms of carbon accumulation rates and storage per unit area, but that the largest stocks are held in the subtidal sediments, due to their vast habitat extents. The widespread nature and variability in the limited data available makes improving the evidence base around carbon stocks essential for evidence-led prioritisation of areas for protection. There has been significant loss of coastal habitats over the last century as a result of pollution, sea level rise, disease, urbanisation and industrial development around estuaries and the coastal hinterland, with only 10% of the historic extent remaining. However, there is potential for restoration and recovery of these habitats in terms of suitable areas based on physical parameters: up to twice the extent of saltmarsh and up to four times the extent of seagrass. Whilst this is still a fraction of historic extents and the maximum restoration potential is not practically achievable for some coastal blue carbon habitats due to location-specific constraints, it would make a significant contribution to improving resilience of the marine ecosystem alongside accumulation and storage of additional blue carbon.

Blue carbon habitats are currently subject to a range of anthropogenic pressures from land and marine activities; elevated nutrient levels caused by runoff from land and physical disturbance by fishing, boating (anchoring and mooring) and offshore wind were mapped where data were available and the overlap between these activities and existing and potential blue carbon habitats were assessed. Areas where the pressures were greatest were identified and should be subject to more detailed analysis to scope management requirements. Natural England has developed a modelling approach which will be used to take this further. It is recommended that for catchments where nutrients have been identified as a significant pressure and cause of unfavourable condition in MPAs, plans to reduce nutrient loadings would have significant synergy with objectives to improve soil health and mitigate climate change on land and should be explored as strategic nature-based solutions. Although a large proportion of blue carbon habitats are now within MPAs and a

programme to put in place management of fishing activity across all sites is in development, it is recognised that this does not automatically confer protection of the carbon stocks. The feature-based approach means that regulation and management is only applicable to the areas of an MPA where designated features are located. The whole site approach, which is to be taken with the forthcoming pilot of Highly Protected Marine Areas, is discussed as a more effective way of maximising ecological recovery and optimising ecosystem services such as carbon storage.

This study represents initial analysis of existing data and a scoping exercise for risks and opportunities as well as proof of concepts for analysing the impacts of activities and the likely benefits of different management scenarios. Recommendations for further work are provided.

Background

The UK Government has put in place ambitious targets around climate change, including achieving net zero greenhouse gas (GHG) emissions by 2050. Accounting for the carbon gains and losses is a critical part of meeting the domestic target and our international obligations and the Net Zero Strategy: build back greener, references the potential contribution of blue carbon habitats[1]. Considerable research and action on land is being directed at increasing carbon accumulation and storage, with strategies for peatland restoration and tree planting receiving substantial funding.

Whilst terrestrial habitats are included in in the UK GHG Inventory and the UK's Nationally Determined Contribution (NDC) 'blue carbon', a term for carbon captured by coastal and marine ecosystems and stored in the vegetative biomass and sediments, currently cannot be included for two reasons: Firstly, coastal wetland habitats (including saltmarsh and seagrass) are not included due to lack of required data to meet the standards set out in the 2013 Intergovernmental Panel on Climate Change (IPCC) Wetlands Supplement. The importance of marine habitats has however been acknowledged in the UK NDC's information to facilitate clarity, transparency and understanding, as well as in the UK's Adaptation Communication and work is underway to gather the relevant information so that saltmarsh and seagrass can potentially be included in future. Secondly, international accounting methods under the IPCC do not yet exist for offshore habitats such as marine sediments as globally, as evidence is limited.

Offshore wind energy production and carbon captured from industry and then stored in the marine area is being accounted for in the UK GHG inventory alongside other human activities such as vessel emissions, but not the emissions that may result from their installation. Defra have an ambitious programme in place to identify strategic solutions that minimise the negative impacts and facilitate the sustainable and coordinated expansion of offshore wind whilst ensuring deployment contributes to delivering clean, healthy, productive and biologically diverse seas.

---

[1] net-zero-strategy-beis.pdf (publishing.service.gov.uk) [Accessed December 2021]

The marine environment plays a critical role in climate regulation, and habitats such as saltmarsh and seagrass sequester a significant amount of carbon. Carbon is also stored in inter/subtidal sediments. However, the marine area is becoming increasingly busy and competition for space is high. Some marine activities have the potential to limit the sequestration of carbon by marine ecosystems through damage to key habitats and to cause carbon to be released when sediments are disturbed. It is therefore important to quantify and understand the drivers of habitat loss which contribute to carbon emissions, to support action to prevent further loss or damage to marine habitats. It is important to quantify the carbon sequestered by these habitats, to assess the historic damage and potential for enhancing and expanding them and to protect the existing carbon stored in the sediments, regardless of the fact that blue carbon habitats are currently excluded from net zero reporting. In many cases, actions to increase carbon storage will also achieve other objectives such as enhancing biodiversity, climate change adaptation and resilience of marine habitats, increasing fish stocks, providing nature-based experiences for people, and improving the overall health and productivity of our seas.

A strategic approach is needed to ensure that the potential to store more carbon is realised alongside the viability of key marine industries and uses. Nature-based solutions designed to protect, restore and manage blue carbon habitats in a sustainable way present a holistic mechanism for delivering actions for climate mitigation alongside a raft of other objectives.

This report aims to provide a starting point and a methodology for the examination of blue carbon storage and accumulation in the marine environment in English waters. The project maps the current distribution and extent of blue carbon habitats that are known to have significant capacity for sequestering and/or storing carbon, either alone or as part of a habitat mosaic. This includes saltmarsh, seagrass, kelp, native European flat oysters (*Ostrea edulis*), and intertidal and subtidal sediments. The potential for restoration and creation of these habitats is also explored and mapped where data were available. Best available evidence on the capacity of these habitats to sequester and store carbon has been applied as a range of values to the areas mapped (Parker *et al.,* 2021, Gregg *et al.,* 2021).

The blue carbon habitats mapped are subject to myriad anthropogenic pressures which restrict their extent and impact on quality, resulting in reduced carbon accumulation and storage compared to their natural state. The key pressures affecting blue carbon habitats in English waters are described and the extent to which blue carbon habitats overlap with these pressures is explored.

The project draws on existing literature and tools to analyse the potential impacts and spatial footprint of activities which might be having a negative impact on blue carbon. A modelling approach has been developed to test scenarios for managing marine activities and its potential use for blue carbon has been assessed. How this new information can be built into advice to regulators and developers/investors is explored in the discussion and recommendations sections.

Contents

List of Figures

List of Tables

1. Mapping key blue carbon habitats in English waters

1.1 Introduction

The key blue carbon habitats within English waters[2] considered in this project are saltmarsh, seagrass, intertidal sediments, subtidal sediments, kelp, and native European flat oyster (*Ostrea edulis*) reefs. It should be noted that most of the carbon in our seas is bound up as dissolved inorganic carbon in the water column or carbonate in the sediment (Thomson *et al.,* 2017; Armstrong *et al.,* 2020). However, this study focuses on the organic carbon secured as carbon stocks within blue carbon habitats and the ability of these habitats to sequester and accumulate carbon within sediments.

Saltmarsh habitats, which form on the landward edges of intertidal sand and mudflats, receive carbon from both marine and terrestrial sources (Gregg *et al.,* 2021). Saltmarsh is considered as one of the most important blue carbon habitats due to its ability accumulate carbon and store it in plant biomass and the surrounding sediments. High variability in the carbon storage and accumulation between saltmarsh sites within the UK has been recorded, with more mature, natural, undisturbed, and botanically diverse saltmarsh habitats with non-sandy soil characteristics likely to perform more effectively as carbon sinks (Parker *et al.,* 2021; Gregg *et al.,* 2021).

Like saltmarsh, seagrass not only traps and stores organic carbon generated through photosynthesis, but also traps and buries allochthonous carbon, which is carbon originating elsewhere (Legge *et al.,* 2020). Typically, 50% of the sediment carbon in seagrass habitats is allochthonous (Kennedy *et al.,* 2010). By buffering the strength of waves and currents, seagrass habitats provide physical structure on a somewhat structureless sediment, which reduces the amount of sediment that is resuspended whilst trapping suspended particles. Seagrass habitats enhance biodiversity and provide vital nursery habitats for commercial and non-commercial fish species (Duffy, 2006; Unsworth *et al.,* 2018). In the UK, this includes species such as pollock, sole, mullet, plaice, skates, rays, lobster, crawfish and crab (Ashely *et al.,* 2020).

Not all of the carbon sequestered and trapped within the seagrass habitat is retained. Seagrass habitats export a proportion of their carbon to species which feed on the leaves, to the seabed as

[2] 'English waters' is the area of sea adjacent to the English coastline that is within the exclusive economic zone and the UK sector of the continental shelf (up to 200 nautical miles). This excludes the waters of any devolved administration.

dead leaves, and to the deep sea. Storms can enhance the export of seagrass carbon stocks, which suggests that this process is intermittent rather than continuous (Duarte and Krause-Jensen., 2017).

The vast majority of the seabed in the English waters is composed of sediments. The character of intertidal and subtidal sediments is determined by the source and strength of the waves and currents. Coarse, mobile sediments tend to persist in areas where the seabed is exposed to regular disturbance by waves and strong currents, which sweep clear any finer sediments. In more sheltered locations, such as shallow bays and in deeper offshore waters below the wave base, mud and muddy sands are more prevalent (Parker *et al.,* 2021 and references therein). Here the water is less dynamic, allowing the mud and muddy sands, rich in carbon-based organic matter, to settle out of the water column. The remains of marine animals and plants also contribute to this layer. Over time these are buried by subsequent sediment accumulation, forming blue carbon stocks. Recently, the potential importance of intertidal and subtidal sediment habitats as long-term reservoirs of carbon has been highlighted (Parker *et al.,* 2021 and references therein), however there are significant spatial heterogeneities in the distribution of sediments and their associated carbon content across the United Kingdom's Exclusive Economic Zone (EEZ) (Smeaton *et al.,* 2021). When disturbed, marine sediments can become mixed and resuspended, exposing them to oxidation of organic compounds, leading to breakdown or transformation of organic matter and release of carbon dioxide (Atwood, 2020; Smeaton *et al.,* 2021).

As one of the world's most productive habitats, kelp is regarded as a major source of blue carbon, although the vast majority (>80%) of material is exported from the kelp forest to other marine habitats (Krumhansl and Scheibling, 2012). Kelp-derived detritus may be transported many kilometres, eventually accumulating as carbon stocks within other blue carbon habitats including seagrass, saltmarsh, and subtidal sediments (Krause-Jensen and Duarte, 2016).

The role of European flat oyster beds in carbon storage is complex. Oyster respiration and growth releases carbon, while shell calcification leads to both the release and storage of carbon as calcium carbonate. Carbon can be retained within the ecosystem through the burial and entrapment of excreted, carbon-rich shells and other material under newly settled oysters as the reef grows (Emerson and Archer, 1990; Fariñas-Franco *et al.,* 2018). In addition to their role as potential carbon sinks, oyster beds can also have other ecosystem benefits and support other blue carbon habitats. For example, Fodrie *et al.* (2017) found that nearly half of all the fringing-saltmarsh oyster reefs constructed in the Rachel Carson Reserve in USA facilitated the localised, seaward expansion of saltmarsh. This is thought to have occurred because oyster reefs serve as natural breakwaters, dampening wave energy and increasing sediment deposition and stabilisation.

This chapter investigates the current spatial extent and distribution of saltmarsh, seagrass, intertidal sediments, subtidal sediments, kelp, and European flat oyster (*Ostrea edulis*) in English waters, using the most up-to-date evidence and spatial data available. The carbon stock and accumulation information of these habitats were compiled, building on work undertaken by Parker *et al.* (2021) and Gregg *et al.* (2021).

The extent of the blue carbon habitats within the Marine Protected Area (MPA) network is also considered, as although there are no specific duties in nature conservation legislation to protect carbon stores, management measures can be applied (and are already in place in many inshore MPAs) to protect the blue carbon habitats where these are designated features of the site.

1.2 Methodology

1.2.1 Defining habitat boundaries for blue carbon habitats

The habitat boundaries used in the mapping of blue carbon habitats in English waters are based on European Nature Information System (EUNIS) habitat classification. The habitats and their EUNIS codes included in this study were as follows:

- Saltmarsh (including saline reed beds) - EUNIS codes A2.5
- Littoral sand and muddy sand - EUNIS code A2.2
- Littoral mud - EUNIS code A2.3
- Sublittoral sand - EUNIS code A5.2
- Sublittoral mud - EUNIS code A5.3
- Seagrass (littoral and sublittoral combined) - EUNIS code A2.61 or A5.53 (HOCI_17)[3]
- Kelp - only records with a specific kelp biotope were included EUNIS biotopes (see Appendix 1 for further detail)
- European flat oyster - EUNIS code A5.4 (HOCI_14)

Broad Scale Habitat (BSH) polygons and Habitat of Conservation Importance (HOCI) polygons from Natural England's Marine Evidence Geodatabase (clipped to English waters[4]) were used to derive the extent of each habitat. For the kelp data, only records where there was a definitive biotope code for kelp occurrence on rock were used and therefore their occurrence is likely to be greatly underestimated.

Any sediment polygons also identifying as seagrass beds (HOCI_17) were removed from the littoral and sublittoral sediment data to ensure no double counting of these two habitats. Due to the sensitivities around the protection and commercial importance of *Ostrea edulis* beds, extent data was calculated for European flat oysters but not displayed within the maps created for the report.

The extents used in this study are derived from EUNIS level 3 layers. It should be noted that EUNIS is based on Folk classification sediment types (mud and sandy mud for littoral, sublittoral and deep-sea muds defined as approximately 50% fines/sand). This does not account directly for sediment permeability transitions which drive particulate organic carbon (POC) storage and fate which occur at ~ 10% fines alone (Parker *et al.,* 2012; Silburn *et al.,* 2018). The carbon stock and accumulation values (Figure 3) for subtidal sediment take the ~10% fines boundaries into account. When scaled up to the total stock and accumulation rates, the extents for the EUNIS habitat boundaries are used.

---

[3] See Appendix 1 for further detail on how the seagrass extent and distribution were agreed and verified in collaboration with the Environment Agency.
[4] See Appendix 1 (p.64) for the exact definition of "English waters" adopted for this study.

This leads to some inaccuracy in blue carbon state assessments, however EUNIS is the most accepted habitat classification system at present.

Appendix 1 provides details of the spatial analysis methodologies used, data sources, caveats, and limitations of this data.

1.2.2 Carbon stocks and accumulation rates in English waters

As data on sediment carbon stock and accumulation rates of UK habitats are scarce, the best available evidence was used in this study to compile an updated summary of stocks and accumulation rates. The sediment carbon stock and accumulation rate values for seagrass, saltmarsh, intertidal and subtidal sediments, and the biomass stock of kelp habitats used in this study were largely adopted from Parker *et al.* (2021) and Gregg *et al.* (2021). Parker *et al.* (2021) and Gregg *et al.* (2021) provided the most up to date figures through a review of published literature, compiling a database of over 500 records extracted from 114 publications. Parker *et al.* (2021) focused on literature reporting measurements from the UK, but also included some global-scale reviews to consider contrasting carbon accumulation rates. Further screening was undertaken to consider how different types of management may be influencing carbon stores and accumulation rates. Parker *et al.* (2021) should be referred to for more details on the exact definitions for carbon stocks and accumulation rates, and for how the data were analysed. Additional evidence not included in the summary estimates by Parker *et al.* (2021) was added in the carbon stock and accumulation rate ranges and averages where applicable.

Parker *et al.* (2021) outlines the main sources of error and uncertainty associated with the sediment carbon stock and accumulation rate data, and the derived average estimates. The littoral sand and sublittoral muddy sand layer used in this study includes a component of the carbon stock and carbon accumulation rate values that Parker *et al.* (2021) assigned to littoral muddy sediment and sublittoral muddy sediment. This is because the sediment permeability transition, which drives POC, occur at approximately 10% fines while EUNIS is based on Folk classification sediment types where the boundary is approximately 50% fines/sands. The total carbon stock and accumulation rate values assigned to littoral sand and sublittoral muddy sand should therefore be regarded as underestimates of their true values.

In addition, the report uses the value for kelp flux from Parker *et al.* (2021). There were no data on carbon accumulation rates in seagrass beds in English waters. Saltmarsh, seagrass, and kelp biomass stock values were taken from Gregg *et al.* (2021) and references therein.

Most of the previous studies on the carbon storage and accumulation potential of oysters have focused on American species (Armstrong *et al.,* 2020), and direct estimates for European flat oyster beds in the UK were not available. Even if European flat oysters had similar carbon producing characteristics as American oysters, oyster densities tend to be far smaller in the UK, which greatly reduces their carbon storage and accumulation potential (Armstrong *et al.,* 2020). For this study, the sedimentary carbon standing stocks for the European flat oyster were adopted from Armstrong *et al.* (2020) and included the sedimentary carbon stock values estimated by Burrows *et al.* (2014). Accumulation rates were not included due to the lack of available evidence.

The average and range of sediment carbon stocks prescribed to each of the blue carbon habitats were based on the data reported in the source material. To calculate the total carbon stock and accumulation rates associated with blue carbon habitats within English waters, the total extent was multiplied by the corresponding carbon stock and accumulation averages and ranges for each blue carbon habitat. Confidence levels of the carbon stock and accumulation rates for each habitat were assessed based on the geographical variation and number of records within UK waters from which the average stocks and accumulation rates were derived.

Finally, estimates of sediment carbon stocks within and outside MPAs were derived for each blue carbon habitat. The protected site designations included in these calculations were Special Areas of Conservation (SACs), Special Protection Areas (SPAs) and Marine Conservation Zones (MCZs).

1.3 Results

1.3.1 Habitat maps

Figure 1 shows the current spatial extent and distribution of blue carbon habitats within English waters. European flat oyster habitat has not been mapped due to the small extent and vulnerability of this habitat to extraction. Subtidal sand and subtidal mud are prevalent across the English waters, whereas the extent and distribution of coastal habitats such as saltmarsh and seagrass are small and patchy, and therefore best illustrated at a finer scale. Figure 2 shows a finer-scale example of the current spatial distribution and extent of saltmarsh and seagrass habitats in the south coast of England.

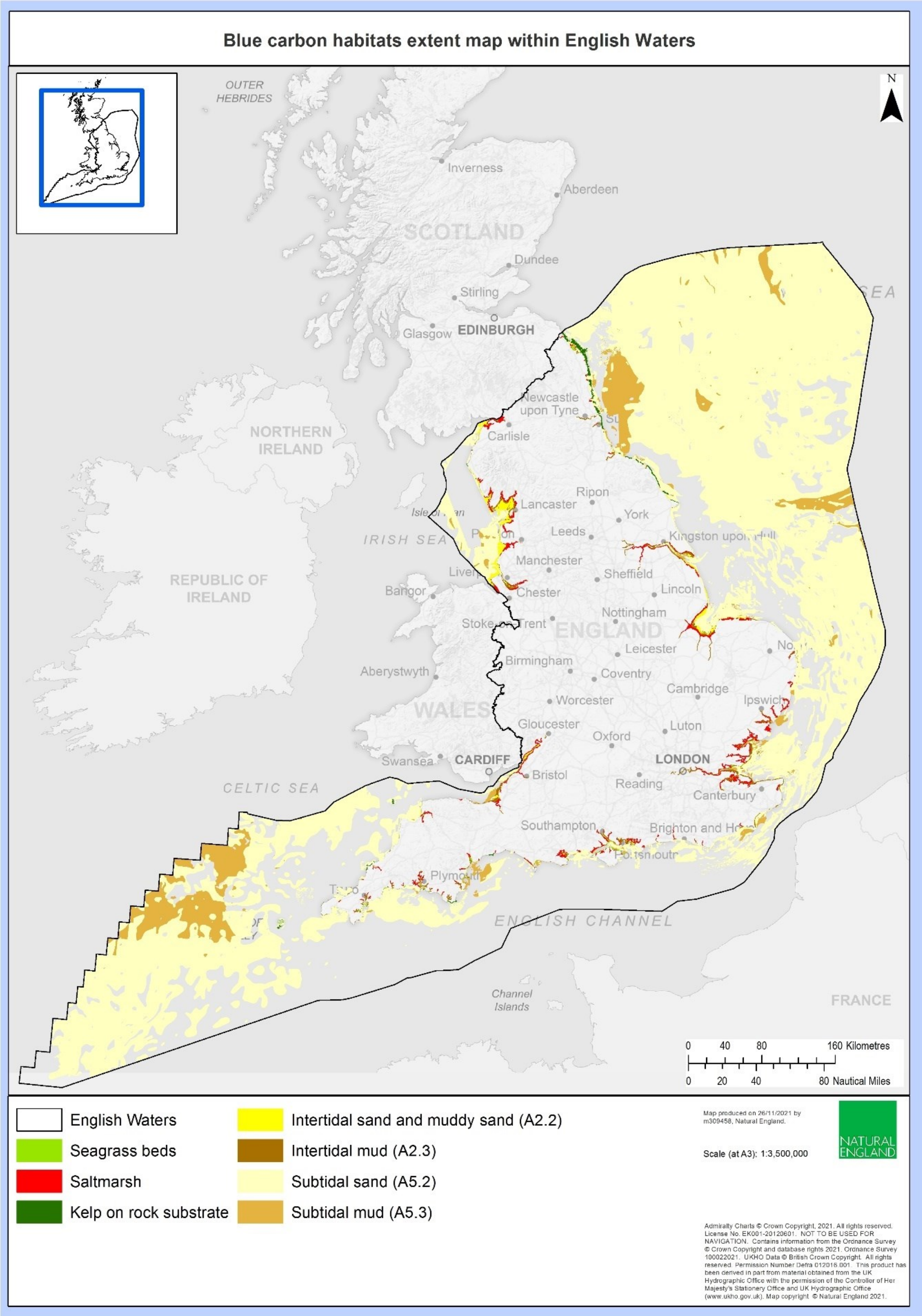
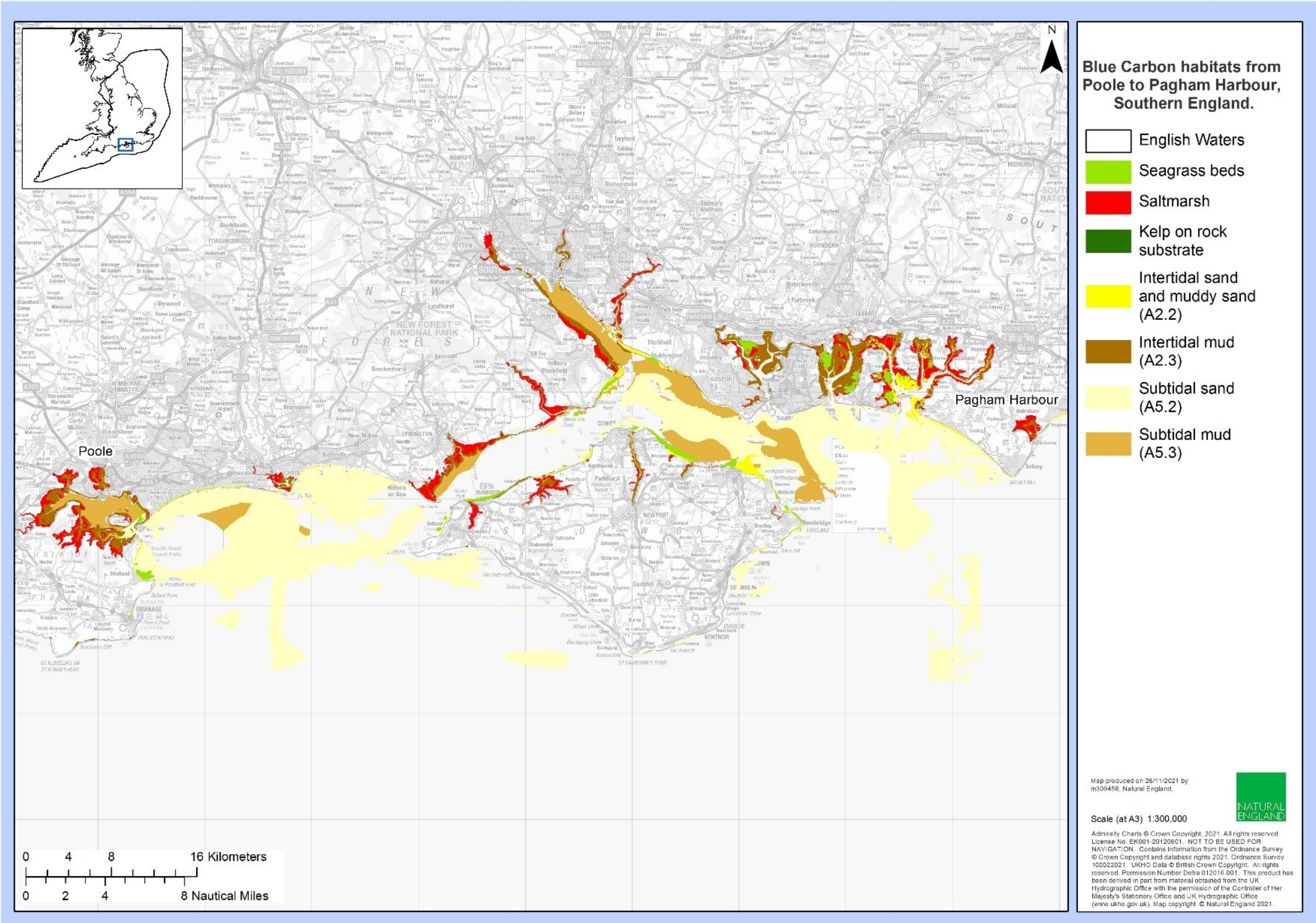


Figure 1. The extent and distribution of blue carbon habitats in English waters. Please note that European flat oyster habitat is not included in the maps, see 1.2.1 and Appendix 1 for further detail.



Blue Carbon habitats from Poole to Pagham Harbour, Southern England.
English Waters
Seagrass beds
Saltmarsh
Kelp on rock substrate
Intertidal sand and muddy sand (A2.2)
Intertidal mud (A2.3)
Subtidal sand (A5.2)
Subtidal mud (A5.3)
N
Poole
Pagham Harbour
0 4 8 16 Kilometers
0 2 4 8 Nautical Miles
Map produced on 26/11/2021 by m309458, Natural England.
Scale (at A3) 1:300,000
Admiralty Charts © Crown Copyright, 2021. All rights reserved. License No. EK001-20120601. NOT TO BE USED FOR NAVIGATION. Contains information from the Ordnance Survey © Crown Copyright and database rights 2021. Ordnance Survey 100022021. UKHO Data © British Crown Copyright. All rights reserved. Permission Number Defra 012016.001. This product has been derived in part from material obtained from the UK Hydrographic Office with the permission of the Controller of Her Majesty's Stationery Office and UK Hydrographic Office (www.ukho.gov.uk). Map copyright © Natural England 2021.

Figure 2. The extent and distribution of blue carbon habitats in the south coast of England. Please note that European flat oyster habitat is not included in the maps, see 1.2.1 and Appendix 1 for further detail.

1.3.2 Carbon stocks and accumulation rates for blue carbon habitats in English waters

Table 1 summarises the best available evidence of habitat sediment organic carbon stocks in the UK. The estimates of the total habitat extent were calculated from the habitat polygons shown in Figure 1.

Table 1 shows that the coastal habitats are a richer store of sedimentary carbon and have a higher average carbon accumulation rate per unit area compared to offshore sediments. The current measurements of average sedimentary carbon stocks rank saltmarsh as by far the most carbon rich of the blue carbon habitats considered here. The muddy sediments have a higher carbon storage value per area than sandy sediments, both within intertidal and subtidal habitats (Table 1). Although seagrass has recently gained attention as an important blue carbon habitat globally, it ranked lower than intertidal mud for its average sediment carbon stock and carbon accumulation rates and has a very small spatial extent. However, the subtidal seagrass measurements in English waters are mainly from subtidal beds in sandy sites and therefore have lower carbon values than muddy sediments.

Even though subtidal sediments have one of the lowest average sediment carbon stocks compared to all other blue carbon habitats shown in Table 1, the large habitat extent of subtidal sand makes it by far the largest total sediment carbon stock in English waters. The carbon stock value attributed to subtidal sand is likely to be larger than shown here, due to the inaccuracies in blue carbon state assessments based on EUNIS classification system (see Parker *et al.,* 2021 for further details). Subtidal mud has the second largest total carbon stock, again due to its large habitat extent. The average carbon stock for European flat oysters (calculated as soil standing stock in the top 10cm) was very small due to the limited presence of the habitat.

The wide ranges in carbon estimates for all blue carbon habitats indicate high variability between datasets. The sediment carbon stock and accumulation figures assigned for all intertidal sediments and seagrass were given low confidence ratings because they were derived from a small number of studies with limited geographic distribution.

Table 1. Habitat sediment organic carbon stocks in English waters. The estimates derived by Parker et al. (2021) were adopted in this study as they provided the best available evidence at the time. Sediment carbon stocks are reported to a sediment depth of 1m as per IPCC guidance. Kelp was excluded because the carbon accumulation in kelp habitat is transported elsewhere. Please note that the average sediment organic carbon stock estimates are not suitable for carbon accounting, as at present the existing methods of measuring and monitoring carbon do not account for blue carbon in coastal and marine soils and biomass.

Habitat	Average sediment organic carbon stock (kg C m^{-2}) (± SE)	Range (kg C m^{-2})	Total habitat extent (km^2)	Total sediment stock (million tonnes C) (Range)	Confidence
Saltmarsh* (incl. saline reed beds)	36.8 (±0.5) (n=82)	10-70	379.5	14.0 (3.8-26.6)	Medium Focussed on Welsh and Essex records
Intertidal mud (A2.3)	19.9 (±4.0) (n=8)	5.4-35.6	707.3	14.1 (3.8-25.2)	Low Focussed on a few locations on the East coast of England
Seagrass	13.7 (±0.2) (n=14)	5.9-38.0	30.5	0.4 (0.2-1.2)	Low Two studies focussed on the SW of England
Intertidal sand and muddy sand (A2.2)	6.5 (±4.0) (n=4)	1.3-18.6	966.0	6.3 (1.3-18.0)	Low Focussed on a few locations on the East coast of England
Subtidal mud (A5.3)	5.5 (±0.5) (n=33)	0.6-12.3	15607.1	85.8 (9.4-192.0)	Medium Good information but requires improved records of deeper sediment
Subtidal sand (A5.2)	1.7 (±0.1) (n=90)	0.4-7.6	121525.1	206.6 (48.6-923.6)	Medium
European flat oyster	0.13**	NA	0.06	0.000008 (NA)	Low
Kelp	NA	NA		NA	Not applicable
Total sediment carbon stock				327.2 (67.0-1186.4)	

* Includes natural and restored habitats

** Sediment standing stock – top 10cm

Table 2 shows the estimated values of average carbon accumulation rates within the key blue carbon habitats together with the associated ranges within estimates taken from published literature. As in Table 1, the values are largely adopted from the review of Parker *et al.* (2021), however, the estimates of extent are updated for this study. The confidence in carbon accumulation rate estimates were low for most habitats apart from saltmarsh (Table 2). The results for carbon accumulation rates follow the same pattern as described for the sediment carbon stocks: Saltmarsh had the highest value for average rate of carbon accumulation rate per given area, but subtidal mud had the largest total accumulation, due to their total extents being considerably larger than saltmarsh.

Table 2. Annual total carbon accumulation in English seas, adapted from Parker et al. (2021) and the references within. Seagrass carbon accumulation rates are based on temperate north-west Atlantic sites as no data exists for the UK. Also please note that European flat oysters were not included as current evidence suggests carbon accumulation to be minimal (see section 1.2.2 for further detail).

Habitat	Average carbon accumulation rates (g C m^2 yr^1) (± SE)	Range of carbon accumulation (g C m^2 yr^1)	Habitat extent (km^2)	Total carbon accumulation rates (million tonnes C yr^1) (Range)	Confidence
Saltmarsh* (incl. saline reed beds)	136.2 ±15.1 (n=7)	66.0-195.5	379.5	0.05 (0.02-0.07)	Medium
Intertidal mud (A2.3)	83.5 ±10.2 (n=2)	73.3-93.7	707.3	0.06 (0.05-0.07)	Low Less than 5 measurements
Subtidal mud (A5.3)	29.5 ±29.3 (n=2)	0.2-58.7	15607.1	0.5 (0.003-0.9)	Low Less than 5 measurements
Kelp**	0.3 ±0.017 (n=4)	0.2-0.4	131.4	0.00004 (0.00003-0.00005)	Low
Seagrass	86 ±19 (n=7)	8-230	30.5	NA***	Sediment C accumulation rates - Novak *et al.,* 2020 (non-UK summary estimate for temperate NW Atlantic sites)
European flat oyster	NA	NA		NA	

* Includes natural and restored habitats

** Flux of kelp transported away from the habitat

*** Total accumulation rate not calculated for seagrass as data for carbon accumulation are for NW Atlantic sites and therefore not suitable for direct multiplication with English seagrass habitat extent

Table 3 provides estimates of carbon storage for the vegetative biomass of the blue carbon habitats rather than the associated sediment. Biomass carbon stocks were highest overall for saltmarsh habitats. However, considerably less information is available on measured biomass carbon stocks and are likely to be highly regionally dependent. Therefore, confidence assessments for biomass carbon stocks are low for all of the habitats included.

Table 3. Carbon storage values for the plant biomass of the blue carbon habitat itself (not sediment) in English waters. The values were adopted from Gregg et al. (2021). Please note that the average carbon stock estimates are not suitable for carbon accounting, as at present the existing methods of measuring and monitoring carbon do not account for blue carbon in coastal and marine soils and biomass.

Habitat	Average carbon stock (kg C m^{-2})	Range of values (kg C m^{-2})	Habitat extent (km^2)	Total biomass carbon stock (million tonnes C) (Range)	Confidence
Saltmarsh (incl. saline reed beds)	1.3	N/A	379.5	0.49 (NA)	Low
Seagrass	0.03	0.007-0.037	30.5	0.0009 (0.0002-0.001)	Low
Kelp	0.25	0.2-0.3	131.4	0.033 (0.026-0.039)	Low

Table 4 shows the extent of blue carbon habitats within the existing MPA network. Apart from subtidal mud and subtidal sand, 90% - 96% of the mapped coastal blue carbon habitats are situated within the MPA network. Whilst this may infer legal protection of the carbon stocks, that is only the case where the blue carbon habitats are themselves a designated feature of a site or a supporting habitat for a designated feature, for example, intertidal mud for wading birds. 20% of sublittoral mud and 44% of sublittoral sand are located inside MPAs. Further measures of protection are required to ensure the vast resource of carbon stocks contained across the large extent of sublittoral mud and sand is adequately protected.

Table 4. Blue carbon habitats within the Marine Protected Area (MPA) network. Please note that average sediment organic carbon stock estimates are not suitable for carbon accounting, as at present the existing methods of measuring and monitoring carbon do not account for blue carbon in coastal and marine soils and biomass.

Habitat	Average sediment organic carbon stock (kg C m^{-2})	Habitat extent within MPAs (km^2)	Habitat extent outside MPAs (km^2)	Total sediment stock within MPAs (million tonnes C)	% of habitat extent within MPAs
Saltmarsh*	36.8	346.7	32.8	12.8	91
Intertidal mud (A2.3)	19.9	639.7	67.6	12.7	90
Seagrass	13.7	29.4	1.1	0.4	96
Intertidal sand and muddy sand (A2.2)	6.5	908.5	57.4	5.9	94
Subtidal mud (A5.3)	5.5	3095.9	12511.2	17.0	20
Subtidal sand (A5.2)	1.7	53480.5	68044.6	90.9	44
European flat oyster	0.13	0.060472	0	0.000008	100
Total blue carbon sediment stock				140.0	43

***** Includes saline reed beds

1.4. Discussion

The spatial extent and distribution of the key blue carbon habitats was mapped within English waters, based on the most up-to-date data layers in Natural England's Marine Evidence Geodatabase (see Appendix 1 for further detail on data sources). The average sediment carbon stocks and accumulation rates per area for saltmarsh, seagrass, intertidal sediments, subtidal sediments, kelp, and European flat oyster (*Ostrea edulis*) reefs (carbon stock only) within English seas were compiled, bringing together the best available evidence and building on work undertaken by Parker *et al.* (2021) and Gregg *et al.* (2021). Based on the updated estimates of habitat extents together with the sediment carbon stock and accumulation rate values, the potential range of total carbon stock and accumulation rates were estimated for the key blue carbon habitats across English waters.

The results highlight the importance of existing blue carbon habitats in carbon storage and accumulation. Although saltmarsh is the most effective at sequestering and storing carbon, the large extent of subtidal sand and mud provides the largest total sediment carbon stocks and accumulation rates within English waters (Tables 1 and 2). This agrees with the study by Burrows *et al.* (2014) on the carbon budgets and potential blue carbon stores in Scotland's marine environment, where coastal and offshore sediments were found to be the main repositories for carbon. Carbon accumulation within subtidal sediments have been shown to be highly variable in English waters, with organic carbon accumulation hotspots found in distinct seabed areas such as estuaries and coastal muds and large accumulations of inorganic carbon in the South West of England (Smeaton *et al*., 2021). Within these hotspots, the muds offer potentially valuable opportunities for targeted future management and protection of sedimentary carbon stocks and should thus be considered as a priority for management (Smeaton *et al*., 2021).

Despite international recognition of the importance of seagrass as a blue carbon habitat (Parker *et al.,* 2021), the average carbon stock within seagrass habitats was considerably lower than that of saltmarsh and the total carbon stocks were the lowest for all habitats included in the calculations, due to the low habitat extent (Table 1). However, as with saltmarsh, other functions of seagrass, including carbon accumulation, capture of allochthonous carbon, and stabilisation of underlying sediments are highly important aspects of the carbon cycle. Furthermore, the value of seagrass habitats in providing other ecosystem and societal benefits alongside climate change mitigation should not be underestimated.

The estimates of average carbon biomass stock of kelp were the highest of all vegetative blue carbon habitats (Table 3). However, kelp habitats are not thought to store carbon in soft sediments (Alongi, 2018), and the accumulated carbon within plant biomass is transported away from kelp habitats to other blue carbon habitats such as subtidal sediments. Carbon fluxes from macroalgae as well as other marine habitats such as oyster beds require further research. Furthermore, the role that blue carbon habitats play in capturing and storing carbon from terrestrial ecosystems will allow for better understanding of the role of blue carbon habitats in the wider carbon cycle.

Although the up-to-date habitat extent layer used in this study provides more comprehensive values than was previously used by Parker *et al.* (2021), some uncertainties in the data and gaps in the mapped habitats remain. For example, the kelp habitat extent presented here is likely to be an underestimate, because only the data recorded as the kelp biotope were included even though it is likely that kelp is also present within other habitat types, such as infralittoral rock. The distribution of kelp habitats is therefore likely to be wider than the current distribution shown in Figure 2, which is concentrated in the South-West and North-East of England.

Furthermore, the littoral sand (EUNIS code A2.2) and sublittoral muddy sand (EUNIS code A2.3) layers used in this study include a component of the carbon stock and carbon accumulation rate values that Parker *et al.* (2021) assigned to littoral and sublittoral muddy sediments. This is because the sediment permeability transition, which drives particulate organic carbon (POC) storage and accumulation, occur at approximately 10% fines, while EUNIS is based on Folk classification sediment types, where the boundary is approximately 50% fines/sands. The total carbon stock and accumulation rate values assigned to littoral sand and sublittoral muddy sand should therefore be regarded as underestimates of their true values. To improve the estimates, the littoral and sublittoral sediment layers need to be better defined to consider their POC content. The EUNIS folk classification from which these biotopes have been identified needs to be refined to consider the boundaries that best fit the sediment fine content associated with Particulate Organic Carbon

(Parker *et al.,* 2021), to enable more appropriate carbon stock and accumulation rates to be assigned to these habitat layers.

The role of factors such as species composition and sediment depth in the variability of carbon stock estimates also requires better understanding. For example, saline reed beds were included within the saltmarsh habitat, as the mapping layers available were not always distinguishable between these habitats. However, as very little information exists on saline reedbed carbon as a standalone habitat, it is not currently possible to compare the differences in carbon stores of the two habitats. Furthermore, in Parker *et al.* (2021), carbon stocks and accumulation rates for sublittoral sediments were estimated to 1 metre below $1m^2$ depth, based on IPCC Wetlands guidance. Few other studies have estimated carbon stocks to greater than 1 metre, with most studies focussing on measurements to less than 10 cm depth.

Overall, most of the sedimentary carbon stocks and accumulation rates reported in this study are based on relatively few data points. For seagrass, data on carbon stocks in English waters are highly variable (Gregg *et al*., 2021), whereas the total carbon accumulation estimates were excluded from this study because the existing data are from on Mediterranean species such as *Posidonia oceanica* and therefore not suitable for direct estimates for seagrass meadows in English waters (Parker *et al.,* 2021). Even the saltmarsh data, which were assigned a medium confidence level, were limited to samples collected from the Welsh and Essex estuaries. It is becoming increasingly evident that factors such as habitat condition, sediment characteristics, and anthropogenic impacts can influence carbon storage and accumulation capability and potential of blue carbon habitats, creating significant variability in carbon estimates between locations at different spatial scales (Gregg *et al.,* 2021). For example, Lima *et al.* (2020) found considerable variability in carbon stocks between the six seagrass sampling sites in the Solent and concluded that sediment characteristics (such as sediment bulk density, % mud content and the degree of sorting) are particularly important indicators of carbon storage within seagrass beds. Further ground truthing and collection of more carbon samples are therefore required to understand the variability within and between blue carbon habitats and to increase confidence in the values at different scales. For saltmarsh, further data on stocks and accumulation rates are currently being collected and analysed as part of the NERC CSide (Carbon Storage in Intertidal Environments) project[5].

One of the important findings of this study is that the majority of the coastal blue carbon habitats within English waters are located within the existing MPA network (Table 4). This report provides a starting point and methodology for examination of the extent of current and potential blue carbon habitats within MPAs and the potential pressures affecting them, and further work is required to establish a more accurate evidence base. Burrows *et al.* (2017) provide a detailed study on the blue carbon resources within the inshore MPAs in Scotland, which assesses both organic and inorganic carbon stocks within a comprehensive list of blue carbon habitats, including maerl beds and biogenic reefs. Different MPAs are ranked for their potential importance to blue carbon accumulation and storage, and threats to blue carbon stocks within the Scottish inshore MPA from physical

[5] https://www.c-side.org/ [Accessed December 2021]

disturbance, moorings, coastal development and renewable energy are also investigated (Burrows *et al.,* 2017).

Regardless of the need for further quantification of blue carbon assets within the MPA network in English waters, the role of the MPA network is likely to become increasingly important in safeguarding blue carbon stocks. A programme to assess and, where necessary, manage fishing activity to protect designated features within MPAs is well underway in the inshore area and has begun also in the offshore area. As an independent coastal state, the Government now has powers to put in place management measures throughout UK waters.

However, it is important to note that the legislation that is in place to protect MPA features may not adequately safeguard blue carbon stocks. This is because blue carbon habitats within an MPA boundary may not be designated features (or supporting habitat) of the MPA, or the management measures may not yet be in place to ensure their protection. Even where the blue carbon habitats are designated features and management measures are in place, the management of activities within MPAs is specifically aimed at preventing damage to and removing barriers to achieving favourable condition of the individual features, which may not always equate to the management measures needed to reduce pressures on carbon stocks or maximise carbon accumulation potential.

As a much smaller proportion of sublittoral mud and sublittoral sand falls within the MPA network, it is likely that further areas of protection are required, to ensure that carbon stocks contained across the large extent of sublittoral mud and sand is adequately protected. The potential influence of anthropogenic pressures on habitat condition and carbon stocks will be further explored in Section 3 of this study.

2. Scoping the potential for restoration and creation of coastal blue carbon habitats

2.1 Introduction

Many blue carbon habitats have suffered considerable degradation and decrease in extent and distribution over the last century (Gregg *et al.,* 2021 and references therein). Seagrass used to cover most sandy-mud sediment habitats close to English coasts prior to the significant losses that occurred due to wasting disease in the 1920-30s, which have been estimated to be greater than 90% loss across Atlantic coastlines (Garrard and Beaumont., 2014). In a recent review, Green *et al.* (2021) estimated a loss of at least 44% of United Kingdom's seagrasses since 1936, and of this 39% has been lost since the 1980s. Even greater proportions of saltmarsh and littoral sediment have been lost through historic land claim for agriculture and ongoing coastal development, sea level rise and coastal erosion (Burden *et al.,* 2020). European flat oysters are believed to have declined by approximately 95% in British waters over the last century (Preston *et al.,* 2020).

The extent and quality of blue carbon habitats could be enhanced via three distinct mechanisms:

- creation of new habitat, for example mudflat and saltmarsh in managed realignment sites where the coastline is modified to inundate or expose new areas to salt water
- active restoration - interventions to re-establish a habitat where it has been lost, or improve the quality of existing degraded habitat - such as planting saltmarsh plants and seagrass, laying European flat oysters or removal of sea defences in locations where these habitats used to thrive,
- Natural recovery - removal of pressures which have caused the loss of extent or condition of a habitat or are preventing it returning to allow natural processes such as recolonisation to work.

It is important to note that for active restoration to be effective, pressures may also need to be removed where they would impact on the mechanism of intervention or the structure and functioning of the restored habitat. For example, planting seagrass in an area which is regularly dredged or trawled would not be effective. This section explores the potential areas for coastal blue carbon habitats to be created, restored or allowed recover naturally and the potential gains in carbon stocks and accumulation rates that could be achieved. It is not intended to be definitive but rather a starting point from which to look in more detail at where efforts and funding may be best targeted to achieve gains in blue carbon.

2.2 Methodology

Restoration potential maps for saltmarsh/mudflat, seagrass and European flat oysters were obtained from existing sources (MMO for saltmarsh/mudflat, and Environment Agency for seagrass and European flat Oysters). These maps provide a national 'high level' indication of areas where saltmarsh, intertidal and subtidal seagrass, and European flat oyster habitats could potentially be created and restored, based on some key physical attributes.

The restoration potential map for saltmarsh, which was originally commissioned by the Marine Management Organisation (MMO), considers areas where suitable habitat could be created through managed realignment within currently defended floodplain areas (MMO, 2019). In some cases, managed realignment within English coastline is expected to lead to the creation of a mixture of mudflat and saltmarsh habitats.

For seagrass, wave energy, current energy, depth, and salinity data were merged, and thresholds that seagrass can tolerate were selected to identify areas that could be suitable for restoration.

For European flat oysters, EUNIS biotope A5.4 Subtidal Mixed Sediment was overlaid with current energy, and thresholds that the European flat oyster can tolerate were selected.

Further details on the maps can be found in the documentation accompanying the data download links (see 'Data sources' in Appendix 1).

2.3 Results

Comparison of the current and potential habitat extents suggest restoration and creation of habitats could result in four times the current extent of seagrass habitat and twice the extent of littoral sediment and saltmarsh habitats (Table 5). However, it should be noted that the restoration potential is based on physical attributes only. The maximum potential habitat extent and carbon stock estimates should therefore be treated with caution, as habitat restoration (or creation) may not be practical in all areas that are suitable in terms of their physical characteristics.

As the current extent of European flat oyster beds are based on only a few locations (data not shown), the potential to increase the extent of the existing habitat is considerable (Table 5). However, the current carbon stock estimates for native oysters suggest that the maximum potential extent of this habitat would only provide a relatively small increase in carbon stocks in comparison to the other blue carbon habitats.

Table 5. Comparison of the current and potential blue carbon habitat extents, sediment carbon stocks and accumulation rates within English waters. Please note that the average sediment organic carbon stock estimates are not suitable for carbon accounting, as at present the existing methods of measuring and monitoring carbon do not account for blue carbon in coastal and marine soils and biomass.

Habitat	Average sediment organic carbon stock (kg C m^{-2})	Total habitat extent (km^2)	Total sediment carbon stock (million tonnes)	Potential habitat extent (km^2)	Maximum potential sediment carbon stock (million tonnes)*
Saltmarsh (incl. saline reed beds)**	36.8	379.5	13.97	2580.14	25.68-94.95***
Intertidal mud (A2.3)	19.9	707.3	14.08		
Seagrass	13.7	30.5	0.04	447.77	6.13
European flat oyster biomass	0.130	0.06	0.000008	337.99	0.0439

* Caution should be applied when considering the upper limits of potential total carbon stock figures

** Includes natural and restored habitats

*** Because the data layer used here does not distinguish between saltmarsh and mudflat, and as it is uncertain whether the habitat created will become mudflat or saltmarsh, a range of values are provided with the minimum value representing 100% mudflat and the maximum value representing 100% saltmarsh. See Appendix 1 for further detail on data sources and analyses

Figure 3 provides an overview of the potential locations for restoration of saltmarsh and mudflats, seagrass habitats and European flat oyster beds. The areas shown are not definitive locations, and not all areas will create these habitats while other areas of opportunity are likely to exist. Figures 4 and 5 show in more detail where these locations are in the south coast of England, between Poole and Pagham harbours. These figures are intended for illustration purposes only, as a detailed analysis of restoration potential at a local scale is out of scope for this study. Although no detailed regional analysis of the location of potential restoration sites was undertaken, Figure 4 does suggest potential to restore large areas of saltmarsh and mudflats along the East coast of England. The distribution of seagrass could be expanded greatly across the South coast of England (Figure 5).

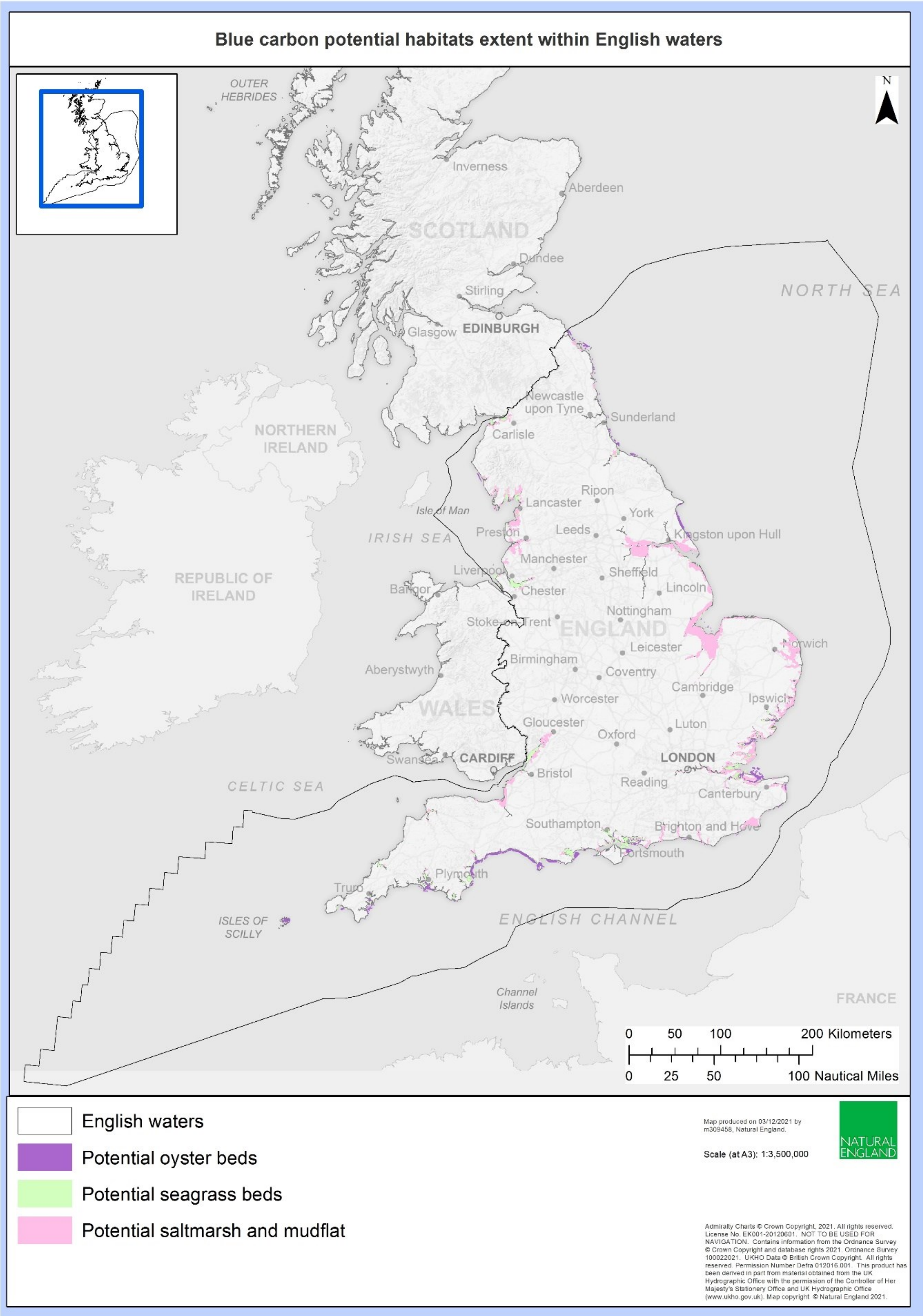


Figure 3. Potential sites for the restoration of European flat oyster and seagrass and the recreation of saltmarsh or mudflats. Data source: Environment Agency.

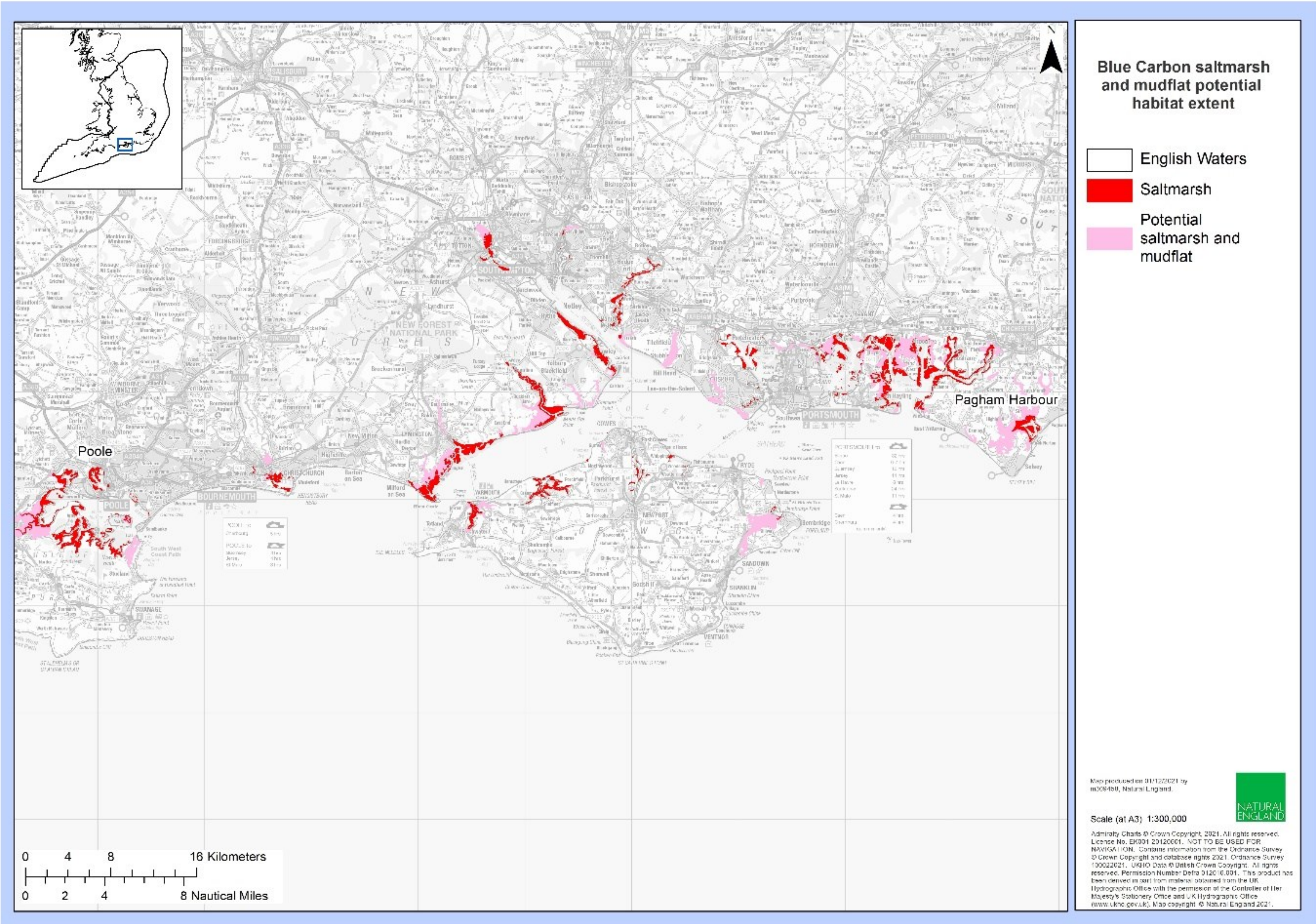


Figure 4. Existing saltmarsh and potential saltmarsh and intertidal mudflat habitat in the south coast of England. See Appendix 1 for further information on data sources.

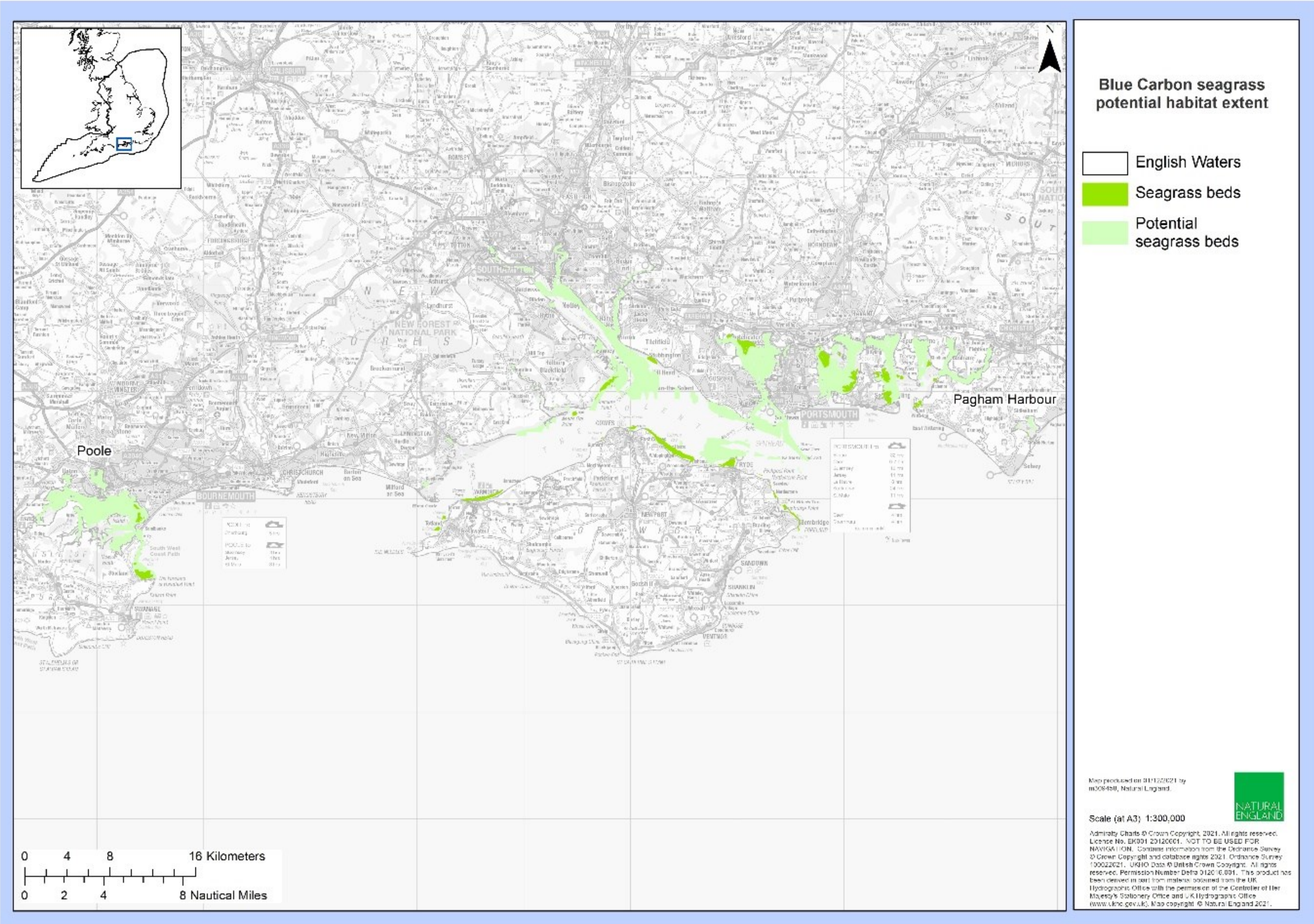


Figure 5. Current and potential seagrass habitat in the south coast of England. See Appendix 1 for further information on data sources.

2.4 Discussion

In this section, we have combined the data on sediment carbon stocks with the maximum extents to which blue carbon habitats could be restored, to assess the potential carbon stocks associated with the fully restored blue carbon habitats (Table 5). The current figures suggest that there is potential to increase carbon stocks across the coasts of England, in particular through the recreation of saltmarsh and mudflats. It should however be noted that the maximum potential carbon stock estimates (Table 5) are based on carbon stocks within the existing blue carbon habitats and therefore assume equal build-up of carbon in all sites, even between natural and restored habitats. In reality, this relationship is likely to be more complex and variable. For example, estimates of long-term burial rates in blue carbon habitats are varied (Craft, 2007) and may be affected by both physical (temperature, sea level, salinity, sedimentation, nutrient status, pollution) and biological (species composition, herbivory) factors (McLeod *et al.,* 2011). Burden *et al.* (2019) reported that carbon accumulation in newly restored saltmarsh was more rapid in the first 20 years (1.04 t C ha^{-1} yr^{-1}), slowing to (0.65 t C ha^{-1} yr^{-1}) thereafter. However, Burden *et al.* (2013) estimated that it would take approximately 100 years for restored saltmarshes to attain equivalent carbon stocks to natural saltmarsh, which agrees with previous estimates. This highlights the importance of protecting existing blue carbon habitats as these hold the greatest carbon stocks, as well as creating new habitat with higher initial accumulation rates.

Restoring and recreating functioning blue carbon habitat will depend on identifying suitable habitat. The potential habitat maps for saltmarsh/ mudflat, seagrass, and European flat oysters were based on a few environmental parameters that may restrict where these habitats thrive. More refined models are now being created to assist identification of suitable habitat locations that will improve the likelihood of successful restoration (Brew, in prep; Early *et al.,* 2020). The model created by Early *et al.* (2020) considers how environmental variables may influence the distribution of seagrass habitat in Plymouth Sound and the Solent. A need to consider habitat suitability on a site-by-site basis was identified, as different variables appear to be more influential depending on the site. Furthermore, the newly published Restoration Handbook series, commissioned by the cross-agency Restoring Meadow, Marsh and Reef (ReMeMaRe) initiative[6], provide excellent practical guidance on restoring and recreating coastal and estuarine habitats. The published restoration handbooks cover blue carbon habitats including saltmarsh (Hudson *et al.,* 2021), seagrass (Gamble *et al.,* 2021), and native oysters (Preston *et al.,* 2020), and provide practical advice on all aspects of habitat restoration, from site selection and licencing processes to restoration methods, monitoring, and community engagement during and after restoration initiatives.

To maximise the climate change mitigation potential of blue carbon habitats, it is crucial to apply landscape-scale approaches in habitat creation and restoration programmes, rather than considering sites in isolation. Changing climatic conditions including sea level rise, more frequent storm events,

---

[6] https://esca.international/reach/rememare-and-latest-updates [Accessed December 2021]

and increasing sea temperatures will lead to losses of habitats in some locations, whereas elsewhere habitat may be gained. The connectivity of habitats therefore needs to be fully understood in habitat creation and restoration programmes, as ensuring the ability of blue carbon habitats to naturally respond to changing conditions will provide further adaptation and resilience to climate change. Restoring and recreating functioning blue carbon habitat will also depend on removing other pressures that are hindering this process. Section 3 of this study considers pressures that require better management actions to maximise the restoration potential of blue carbon habitats in English waters.

3. Blue carbon - mapping risks and opportunities

The previous sections have outlined how coastal and marine habitats can effectively accumulate carbon and store it in their subsoil and vegetative parts. We have estimated ranges for the current carbon stocks within the key blue carbon habitats and explored how the carbon stocks could be increased through habitat creation and restoration within suitable locations. Many coastal and marine habitats have experienced significant deteriorations in their extent and condition, which could be linked to pollution and a range of physical impacts caused by human activities. Understanding the drivers of these losses is key to protecting the existing carbon stock as well as mechanisms for increasing carbon stocks and alleviating any pressures which are still operating will be a key element of facilitating the restoration of lost and damaged habitat.

There are a number of pressures which can limit the sequestration and accumulation functions of blue carbon habitats and cause physical degradation and eventual loss of the habitat (Pendleton *et al*., 2012). Abrasion and penetration (which physically disturb the surface of the habitat) may directly cause sediment loss into the water column and some of the carbon contained in the mobilised sediment may be oxidised and released to the atmosphere (Crooks *et al*., 2011). Chemical pressures which restrict the growth of saltmarsh and seagrass plants or reef-forming organisms may reduce their ability to sequester and accumulate carbon. Smothering can similarly restrict growth or kill off plants altogether, resulting in loss of sequestration and mobilisation of sediments previously stabilised by root mats (Crooks *et al*., 2011). The pressures are considered in more detail in respect to the different blue carbon habitats below.

This section investigates the incidence of some key anthropogenic activities (Orth *et al*., 2006) that could cause the pressures which can damage or disturb blue carbon habitats, both on the existing extent and on the suitable areas where blue carbon habitats could be enhanced or expanded. The spatial overlap of inshore blue carbon habitats with areas where the key anthropogenic activities occur are mapped, where data are available. The anthropogenic pressures and activities considered in this report are eutrophication, bottom trawling, anchoring and mooring, and offshore wind farm operations, and these are illustrated with figures and tables where data are available. Other anthropogenic activities likely to have a significant impact on blue carbon habitats such as coastal development, flood prevention and defence, and dredging were out of scope for this proof-of-concept study.

3.1 Introduction

3.1.1 Eutrophication

Increased urbanisation and intensification of agriculture in the last century has doubled the global inputs of fixed nitrogen to water (Vitousek *et al.,* 1997). Many parts of England's coastal waters have been subject to highly elevated nutrient levels and some have become eutrophic. Combined efforts on improving wastewater treatment facilities and reducing agricultural inputs to freshwater have reduced overall catchment nutrient levels. However, nutrient levels remain high in many estuarine and coastal areas.

Eutrophic estuaries are characterised by regular excessive phytoplankton blooms and/or the growth of dense opportunistic macroalgae on mudflats which renders the sediment anoxic, smothers infauna and can form a physical barrier to feeding waders. In addition to impacts on littoral sediment, elevated nutrients can be detrimental to other blue carbon habitats such as saltmarsh, seagrass, and oyster beds. Nutrient enrichment can reduce the below ground root biomass of saltmarsh plants and alters soil structure, increasing risk of erosion (Deegan *et al.,* 2012; Penk, 2020; Newton and Thornber, 2013; Wasson *et al.,* 2017). Decomposing macroalgae can also smother saltmarsh vegetation, leading to reduced plant growth and even physical losses through erosion (Newton and Thornber, 2013). Nutrient enrichment has contributed to the recent saltmarsh losses in Poole Harbour, across the Solent MPA as well as the Isle of Wight (Bardsley *et al.,* 2020). The impact of eutrophication on the biodiversity of saltmarsh habitats can also reduce their ability to accumulate and store carbon (Geoghegan *et al.,* 2018; Bulseco *et al.,* 2019).

The Water Environment (Water Framework Directive) (England and Wales) Regulations 2017 set out objectives for water quality including nutrient standards, which are required for water bodies to attain good ecological status. To meet the target for 'Good' quality in transitional and coastal waters, dissolved inorganic nitrogen (DIN) should be maintained at or reduced to levels where the spatial extent, biomass, and entrainment of opportunistic macroalgae overlying littoral sediment are limited and do not exceed the thresholds identified by UK Technical Advisory Group (UKTAG)[7]. Similarly, phytoplankton levels should not exceed the threshold identified by UKTAG for 'Good' quality. Environment Agency monitoring results for each component are combined and reported as a classification showing the overall status of a waterbody. Classifications for DIN and macroalgae have been used in this study as a broad indication of eutrophication pressure.

3.1.2 Bottom trawling

The use of bottom towed fishing gear is the most widespread activity causing sediment disturbance in English waters (Luisetti *et al.,* 2019). The abrasion and penetration pressures caused by bottom trawling affect 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 (Duplisea *et al.,* 2001). While the long-term impacts of trawling on blue carbon stocks are poorly understood, a study in the north-western Mediterranean Sea showed trawling to affect deep sea sediments to a depth of 10 cm, with a 52% reduction in organic carbon storage, slower carbon turnover, and reduced abundance and biodiversity of meiofauna (Pusceddu *et al.,* 2014).

[7] http://www.wfduk.org/resources/tags/transitional-coastal-water-159 [Accessed December 2021]

All sediment habitats are subjected to abrasion and penetration pressures from the use of bottom towed gear in English waters however, the most intensively trawled areas that support greater numbers of fish tend to be the subtidal mud and muddy sand habitats that are also the most carbon rich sediment habitats, which makes these blue carbon stores particularly vulnerable to bottom trawling. Bottom trawl fishing intensity has seen a rapid global expansion since the 1950s (van de Velde *et al.,* 2018), with the same areas of subtidal sediment habitat affected 1-3 times per year on average in the shallow Southern North Sea (Eigaard *et al.,* 2016; van de Velde *et al.,* 2018)). Some evidence suggests that repeat trawls of an area of seabed have little effect on carbon loss or may even lead to small gains in carbon concentration (Hale *et al.,* 2017). However, a recent study on the global scale impacts of fishing activity by commercial trawlers on sediment carbon stocks, using certain modelled assumptions, suggests that after the initial decline in emissions, after nine years of continuous trawling the emissions were predicted to stabilise at c. 40% of the first year's emissions and will continue at this rate for c. 400 years until all of the sediments in the top metre are depleted (Sala *et al.,* 2021).

There is some ongoing work investigating the impacts of trawl fishing activity on sediment carbon stocks in UK offshore waters (Cefas, unpublished data), so this report is complimentary in looking at the smaller fleets in inshore waters. Fishing effort data are not widely available for inshore vessels smaller than 12m in length as these vessels are not required to use the Vessel Monitoring Systems (VMS) used to monitor the activity of larger fishing vessels under UK law. A variety of data are available, including sightings data by enforcement patrols, and landings and port data, but these datasets are not collated or ubiquitous enough to provide a comprehensive picture of activity. This study explores the potential impact of bottom trawling by the inshore fleet on the blue carbon habitats present within the Sussex Inshore Fisheries and Conservation Authority (Sussex IFCA) district in the south coast of England. In this pilot study, the most recent trawling activity data from the Sussex IFCA (Nelson, 2020) were used to investigate the incidence of use of bottom towed gear on blue carbon habitats. Work is ongoing to collate and analyse the various datasets available on the activity of fishing vessels under 12m length in inshore waters and further analysis will be carried out for other districts once sufficient data are available. In future it is hoped that the smaller fleets will use the inshore VMS and provide accurate and comprehensive data on activity.

3.1.3 Anchoring and mooring

Anchoring and mooring activities tend to be focused in sheltered, shallow locations in close proximity to the shore and intertidal coastal zones, which are also the areas that are most suitable for seagrass habitats (Cullen-Unsworth *et al.,* 2014). Anchoring and mooring activities have been shown to result in various disturbances to seagrass habitats that can expose the carbon rich sediments, including abrasion and scouring caused by anchors and mooring chains (Jackson *et al.,* 2013; d'Avack *et al.,* 2014). For example, localised damage occurs when anchors drag along the seagrass bed, and when the chains attached to traditional swing moorings rotate with the tidal current or wind movements, uprooting seagrass plants and resulting in fragmentation and clear reduction in seagrass cover around the fixed point of the mooring (Jackson *et al.,* 2013; Ouisse *et al.,* 2020 and references therein). Once the sediments are no longer bound by the mat of rhizome roots, they may be resuspended by waves and tides. This combination of human and natural disturbance can rapidly reduce the carbon stocks, as scars expand and fragment seagrass habitats. As well as physically impacting the seagrass beds, the disturbed sediment and organic matter re-enters the water column, increasing turbidity levels which reduces the light available for photosynthesis by the remaining seagrass (Unsworth *et al.,* 2017).

Even though the physical disturbance on blue carbon habitats caused by anchoring and mooring tends to be localised, the damage to habitat extent and condition could have significant impacts on carbon accumulation and storage of blue carbon habitats such as seagrass, which are relatively scarce in English waters. This study investigates the spatial overlap of existing blue carbon habitats with anchoring and mooring activities. For further detail on the rationale for the mapping work and the approach taken, see Appendix 1.

3.1.4 Offshore wind farms and associated cabling

Offshore wind farms (OWF) are now a common sight in English waters, and the UK is considered as one of the best locations for wind power in the world. As offshore wind is the most cost-effective way of achieving UK's Net Zero ambitions, delivering 40GW of offshore wind by 2030 with further targets through to 2050 to contribute to decarbonising the energy system is an essential part of achieving this target[8] The deployment of offshore wind is rapidly increasing in line with Government's ambition for the sector. This is fundamentally changing our seascapes, and the cumulative impacts of offshore wind farm installation, operation, decommissioning and repowering on the underlying blue carbon habitats are yet to be established. However, both turbines and cables have the potential to impact blue carbon habitats, both offshore and inshore where export cables make landfall. This report explores the overlap of the existing and planned OWFs (and the associated cabling) with blue carbon habitats within England's Marine Plan areas.

3.2 Methodology

Detailed notes on the mapping methodology and data layers used to explore the potential impact of some key anthropogenic pressures and activities on blue carbon habitats are given in Appendix 1.

To investigate the incidence of eutrophication pressure on blue carbon habitats, Environment Agency WFD water body classification map layers (based on data collated in 2019) for DIN and opportunistic macroalgae were overlaid onto the blue carbon habitat (existing and potential) map layers. The classification layers only cover waterbodies in transitional and coastal waters as defined by the WFD and do not cover areas above mean high water, or those more than one nautical mile from the coast. Although the WFD water body classification map layers do not represent a comprehensive picture of the nutrient status of all coastal blue carbon habitats, but they are the most appropriate datasets available and indicative of the quality in the areas of intertidal and

[8] https://www.gov.uk/government/news/new-plans-to-make-uk-world-leader-in-green-energy [Accessed December 2021]

shallow subtidal habitats most sensitive to nutrient-related pressures. The majority of seagrass, kelp, European flat oysters, and littoral mud and muddy sand habitats explored in this study were within the areas assessed for DIN. However, only 23% of saltmarsh was within the assessment area, mostly due to the proportion of this habitat which is above mean high water. A low percentage of sublittoral mud and muddy sand layers was also classified because the assessment area only covers the most inshore (to 1nm) extent of these habitats, so the majority lie outside the scope of WFD. The UK Marine Strategy Assessment gives a broad overview of the nutrient status of coastal waters against the standards for Good Environmental Status – this is a much less regular assessment, and the latest publication is from 2016 shows all English waters as not having a problem except for a few areas highlighted which correspond to areas covered by WFD more recently and in more detail. For this reason, the GES assessment data has not been used here as it doesn't add further info.

Macroalgae is only assessed in areas of littoral sediment where DIN levels are moderate, so that classification layer represents an even smaller extent. It is nevertheless useful as an additional indicator for the intertidal habitats, and where it is present it represents an impact rather than just nutrient pressure as it has manifested as an ecological imbalance and additional physical pressure in the form of smothering. The extent to which each habitat fell within these WFD classification areas was calculated and presented as a proportion of the assessed habitat area rather than the total extent. English waters are divided into 11 marine plan areas[9] for marine planning purposes and these are primarily based on biogeographical regions. The nutrient pressure information is also broken down on a marine plan area level to give a regional breakdown relevant to decision making.

To explore the overlap and potential interaction between trawling activity and blue carbon habitats, the south coast of England was selected as a case study area due to the availability of trawling activity data. Trawling activity data layers obtained from Sussex IFCA were overlaid with the blue carbon habitat extent and distribution maps. The trawling activity data combines both single and paired trawling activities within 1km x 1km grid cells across the Sussex IFCA district between 2015 and 2019. A detailed explanation for the single and paired trawling data can be found in Nelson (2020). The extent to which each blue carbon habitat fell within the Sussex IFCA trawling effort polygons was calculated.

To investigate the interaction and potential impact of anchoring and mooring activities on blue carbon habitats, commercial anchoring and mooring data layers and recreational anchoring and mooring activities from NE's Marine Evidence Geodatabase were overlaid with blue carbon habitat layers. Finally, for activities associated with OWF construction, operation, and decommissioning (e.g., cabling activities), OWF data layers obtained from The Crown Estate were overlaid with blue carbon habitat layers.

With the exception of eutrophication and the Sussex IFCA fishing activity pilot study, quantifying the potential scale of impact of the key anthropogenic activities on blue carbon habitats and the

[9]https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/325688/marine_plan_areas.pdf [Accessed December 2021]

associated carbon stocks and accumulation rates was beyond the scope of this study as the available data were not sufficient to allow objective or robust analysis. The other spatial comparisons still provide usual visualisations of areas of blue carbon habitat which could be potentially impacted by anthropogenic activities.

3.3 Results

3.3.1 Eutrophication

Figure 6 provides an overview of the DIN concentrations assessed in English waters and the distribution of blue carbon habitats that have been considered for this study. DIN is elevated (Moderate classification status) around large parts of the coast, particularly within the estuaries and shallow inlets and bays which coincides with the locations of many of the coastal blue carbon habitats.

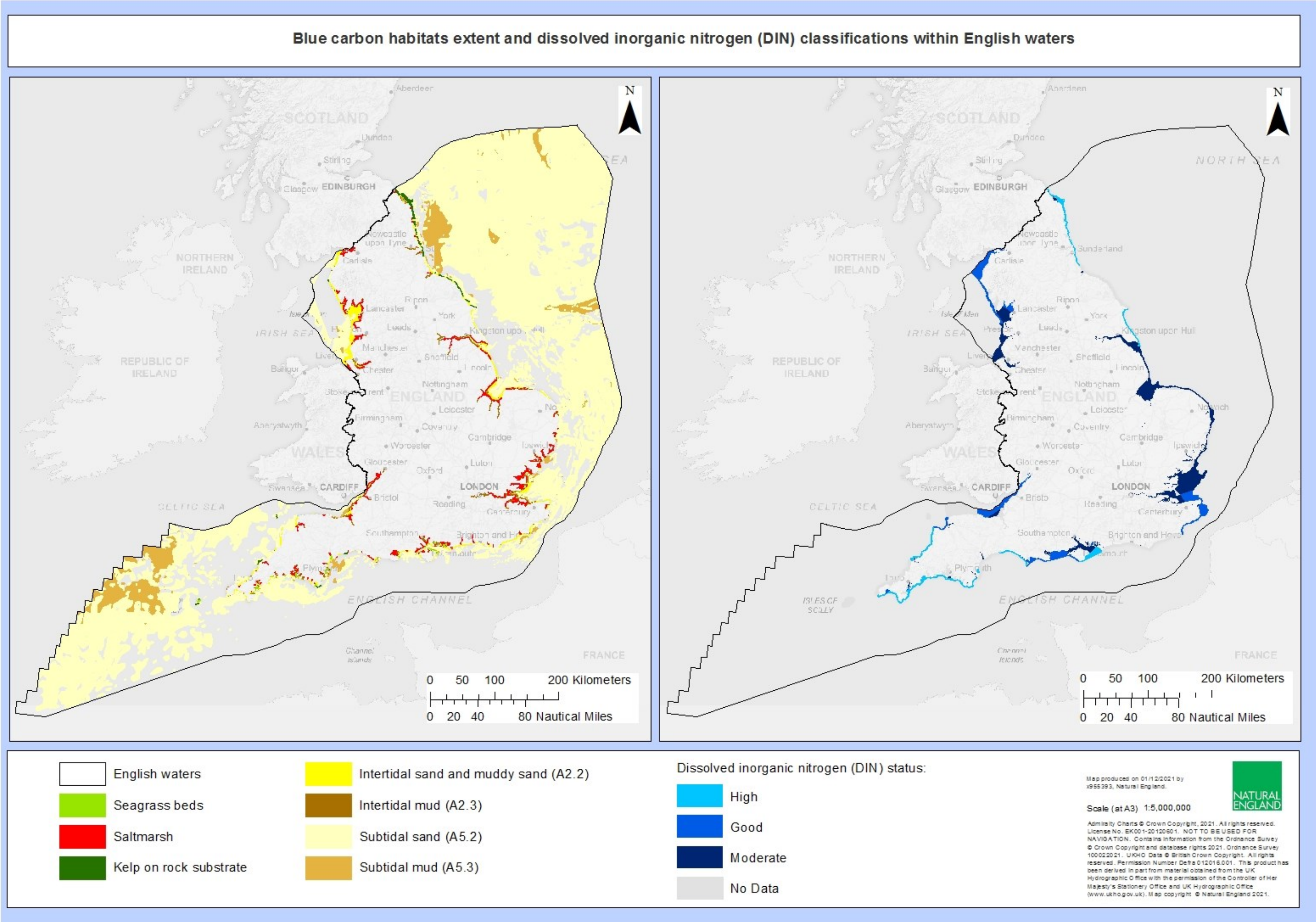


Figure 6. Blue carbon habitat extents and dissolved inorganic nitrogen (DIN) classifications within English waters.

Table 6 shows the extent of blue carbon habitats which overlap with the three recorded classification status levels for DIN: High status (representing the best water quality); Good status (where the levels are still within the standards for good ecological status); and Moderate status (below the thresholds for Good ecological status and favourable condition for nutrients in MPAs). Data are presented for the habitat extents which are within the areas assessed under the WFD. The majority (97.0 km^2) of assessed kelp habitat was within a High DIN status classification (i.e., where DIN concentrations were low), reflecting the records of this habitat being mainly along the more exposed north east and south west coasts. Only a very small proportion (0.2%) was found to be within the Moderate classifications for DIN. In contrast, 83% of assessed saltmarsh extent and 80% of assessed seagrass lay within moderate DIN status areas. Of the subtidal mud and subtidal sand assessed, 46% and 57% of the extents were within moderate DIN status respectively. 78% of the assessed extent intertidal mud and 68% of assessed intertidal sand and muddy sand () also lay within moderate DIN areas. 68% of the assessed extent for European flat oysters was within moderate DIN classification areas.

Table 6. Extent of blue carbon habitats found within each status category for DIN shown as a percentage of the habitat within the area assessed under WFD. Please note that the average sediment organic carbon stock estimates are not suitable for carbon accounting, as at present the existing methods of measuring and monitoring carbon do not account for blue carbon in coastal and marine soils and biomass.

Habitat	Average sediment organic carbon stock (kg C m^{-2})	Extent within High DIN status (2019) (km^2)	Extent within Good DIN status (2019) (km^2)	Extent within Moderate DIN status (2019) (km^2) *(% within this WFD assessment)*	Total extent included in EA WFD assessment for DIN status km^2 *(% of total habitat extent)*
Saltmarsh* ***(incl. saline reed beds)***	36.8	0.03	15.9	78.8 (83)	94.8 (25)
Intertidal mud (A2.3)	19.9	0.8	120.5	442.0 (78)	563.3 (80)
Seagrass	13.7	1.8	3.5	21.9 (80)	27.2 (89)
Intertidal sand and muddy sand (A2.2)	6.5	20.7	240.7	548.8 (68)	810.2 (84)
Subtidal mud (A5.3)	5.5	39.4	548.9	505.6 (46)	1093.9 (20)
Subtidal sand (A5.2)	1.7	465.0	1006.9	1967.1 (57)	3439.0 (3)
Kelp on rock substrate	0.25	97.0	0.2	0.2 (0.2)	97.4 (74)
European flat oyster	0.13	0	0.02	0.04 (68)	0.06 (100)

*Includes natural and restored saltmarsh habitats

Table 7 shows the extent of intertidal blue carbon habitats which overlap with High, Good and Moderate status macroalgal classification. Data are presented for the habitat extents which are within the areas assessed under the WFD. For saltmarsh, 13% lay within Moderate macroalgal classification. Extents within Moderate macroalgae status for intertidal mud and intertidal sand and muddy sands were 15% and 7%, respectively. European flat oyster habitats had no recorded overlap with the moderate macroalgae classification. Seagrass had the highest percentage of extent within a Moderate macroalgae classification (40%).

Table 7. Extent of blue carbon habitats found within each status category for macroalgae shown as a percentage of the habitat within the area assessed under WFD. Please note that the average sediment organic carbon stock estimates are not suitable for carbon accounting, as at present the existing methods of measuring and monitoring carbon do not account for blue carbon in coastal and marine soils and biomass.

Habitat	Average sediment organic carbon stock (kg C m^{-2})	Extent within High macroalgae status (2019) (km^2)	Extent within Good macroalgae status (2019) (km^2)	Extent within Moderate macroalgae status (2019) (km^2) *(% within this WFD assessment)*	Total extent assessed by EA 2019 waterbody assessment *(% of habitat within the WFD macroalgae assessment)*
Saltmarsh* ***(incl. saline reed beds)***	36.8	38.1	14.6	7.7 (13)	60.4 (16)
Intertidal mud (A2.3)	19.9	190.9	91.2	49.3 (15)	331.3 (47)
Seagrass	13.7	4.9	7.6	8.5 (40)	21.0 (69)
Intertidal sand and muddy sand (A2.2)	6.5	221.6	30.3	18.4 (7)	270.3 (28)
European flat oyster	0.13	0.03	0.01	0 (0)	0.04 (68)

*Natural and restored saltmarsh habitats included

Table 8 shows the overlap of Moderate DIN and Moderate macroalgae classification with the potential habitat extents for mudflat/saltmarsh, seagrass, and the European flat oyster. The overlap of combined potential saltmarsh/mudflat habitat was almost the same as the existing saltmarsh, with 86% in Moderate DIN classification and 17% within Moderate macroalgae status.

Unsurprisingly, the blue carbon habitats most at risk from eutrophication impacts are those typically found in and around estuaries and sheltered coastal inlets and bays where tidal flushing is reduced and the influence of land-based nutrients from point and diffuse sources is strongest. This indicates that eutrophication pressure is high and widespread for existing littoral mud and muddy sand sediments as well as saltmarsh and seagrass habitats, and a key pressure to alleviate if the quality of existing habitat and the distribution of European flat oyster is to be expanded and further habitat extent restored.

Table 8. Extent of potential blue carbon habitats with eutrophication pressure - where DIN is elevated (Moderate status) and where macroalgal status is classed as moderate.

Potential habitat	Total extent of potential habitat Km^2	Extent within Moderate DIN Status (2019) Km^2 *(% within this WFD assessment)*	Total extent assessed by EA 2019 waterbody assessment for DIN status *(% of total habitat extent assessed)*	Extent within Moderate Macroalgae Status 2019 (km^2) *(% within this WFD assessment)*	Total extent assessed by EA 2019 waterbody assessment for macroalgae status *(% of total habitat extent assessed)*
Saltmarsh/ mudflat	2580.1	3.4 (86)	3.9 (0.1)	0.3 (17)	2.0 (0.1)
Seagrass	447.8	289.9 (70)	413.9 (92)	69.1 (23)	299.6 (67)
European flat oyster	338.1	95.2 (36)	266.2 (79)	9.0 (10)	103.2 (31)

Table 9 shows the distribution of habitats within Moderate DIN status across the inshore marine plan areas. This analysis helps to understand the relative proportion of existing stocks and the level of risk posed by eutrophication to the existing blue carbon habitats in each plan area.

Table 9. Extent of blue carbon habitats at risk from eutrophication by regional inshore marine plan areas, as indicated by where DIN is elevated. Saltmarsh is not included due to the low proportion of habitat assessed.

	Habitat extent (km^2) that was within moderate DIN Status (2019) ***(% of habitat extent that was assessed for WFD DIN status that was within moderate status for DIN) by inshore marine plan area***						
Habitat	**North East**	**East**	**South East**	**Southern**	**South West**	**North West**	**% of habitat within WFD classification assessment for DIN**
Intertidal mud (A2.3)	2.9 (98)	127.7 (100)	174. (99)	50.1 (100)	10.4 (21)	73.3 (57)	80
Seagrass	7.2 (100)	0 (0)	3.1 (94)	10.6 (75)	0.4 (8)	0.5 (100)	89
Intertidal sand and muddy sand (A2.2)	17.6 (58)	155.8 (100)	155.8 (100)	155.8 (100)	8.6 (13)	340.9 (67)	84
Kelp on rock substrate	0.13 (0.1)	0	0 (0)	0 (0)	0.04 (4.5)	0 (0)	74
European flat oyster	0 (0)	0 (0)	0 (0)	0.01 (100)	0.03 (59)	0 (0)	100

Littoral sediments make up the majority of the habitat extents within Moderate DIN classification. These were mainly located in the North West, East inshore and South East marine plan areas with a significant extent also in the Southern marine plan area. In all four plan areas the total extent of area assessed was in Moderate DIN classification.

The largest extent of seagrass under nutrient pressure was in the Southern inshore plan area, with 75% of the 10.6km^2 assessed for WFD being within Moderate DIN classification. The North East and South East inshore plan areas also had smaller but significant seagrass habitat extents within the Moderate DIN classification. The South West marine plan area had hardly any extent mapped within the Moderate classification and this is reflective of the small area that was assessed for WFD in that plan area.

All European flat oyster beds in Moderate DIN status classification areas were in the Southern and South West marine plan areas.

3.3.2 Bottom trawling

Figure 7 shows the areas where bottom trawling activity overlaps with blue carbon habitats within the Sussex IFCA district. The vast majority of the of trawling activity (26%) occurred within subtidal sand, which has the lowest carbon stock per area across the key blue carbon habitats. 41% of the subtidal mud (1.2km^2) that exists across the Sussex IFCA district coincides with area subjected to bottom trawling, although the overall potential impact on sediment carbon stocks is relatively small due to the small extent of this sediment type in the area (Table 9). The third largest overlap of extent with trawling effort was for the potential seagrass habitat, which highlights the importance of comprehensive site-specific review of all factors, including existing anthropogenic pressures, when choosing locations for seagrass restoration (Gamble *et al.,* 2021). Very small and discrete area of saltmarsh in Chichester Harbour also coincides with trawling activity, which may be an error caused by the resolution of the activity data within the habitat boundary. It is however possible that some trawling activity could occur on the fringes of the habitat at high tides when saltmarshes are submerged. Use of small-scale bottom towed gear with saltmarsh habitat has also been reported in Poole Harbour (Sue Burton, NE, pers. comm.).

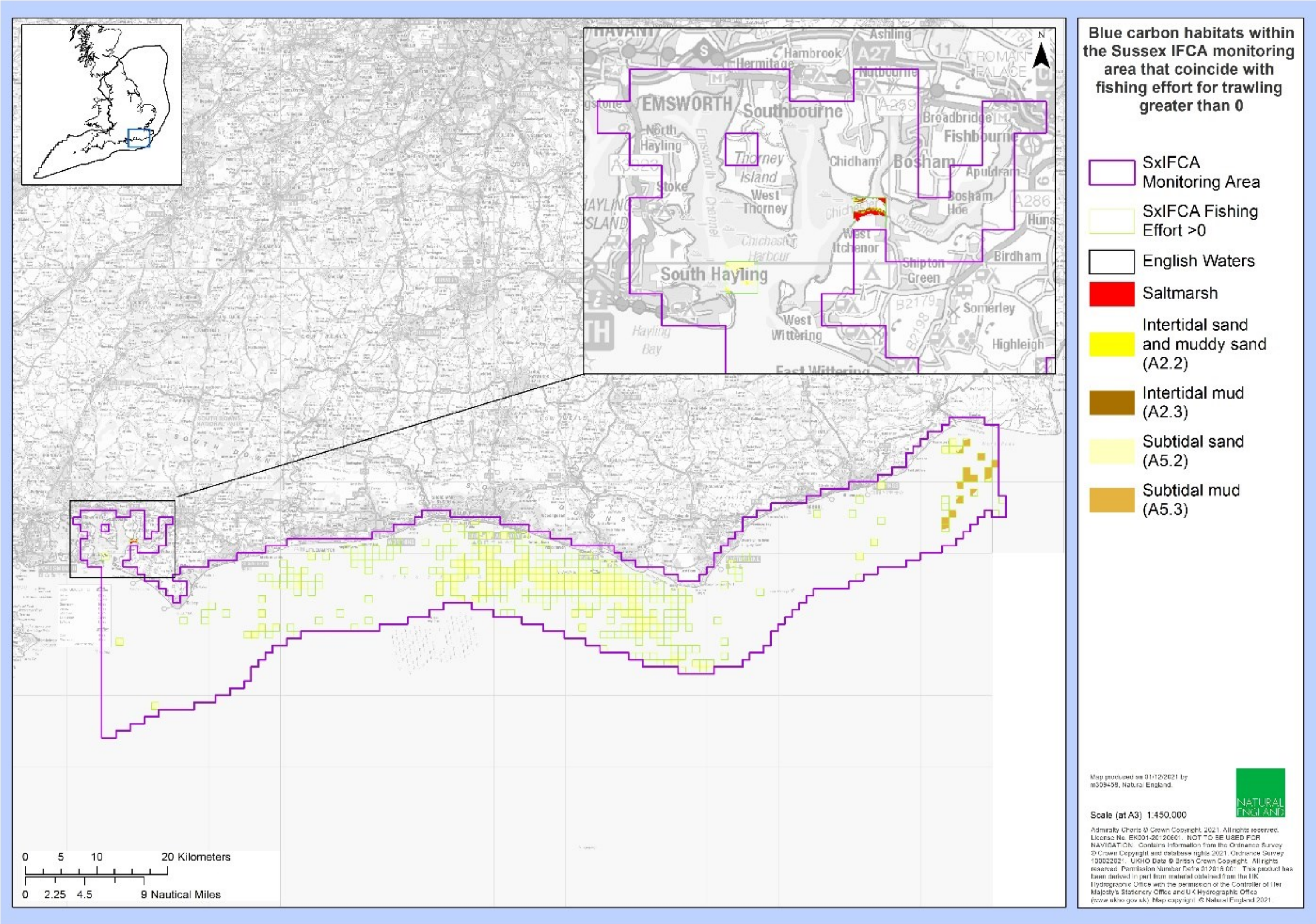


Figure 7. Areas where trawling activity overlaps with blue carbon habitats within the Sussex IFCA district. Source: SxIFCA; Nelson (2020).

Table 10. Total extent of blue carbon habitats within Sussex IFCA district, and blue carbon habitat extents overlaid with trawling effort data collected within the same area. Trawling effort data source: SxIFCA.

Blue Carbon Habitat	Total extent within Sussex IFCA district (km^2)	Total extent overlapping with trawl fishing effort (km^2)	% Habitat within trawling activity layer
Saltmarsh	7.3	0.1	1.4
Seagrass beds	1.7	0	0
Potential seagrass beds	14.6	0.6	4.3
Intertidal sand and muddy sand (A2.2)	18.1	0.1	0.6
Intertidal mud (A2.3)	10.5	0.1	0.8
Subtidal sand (A5.2)	522.4	135.0	25.9
Subtidal mud (A5.3)	2.9	1.2	41.0
Oyster beds	0.0	0	0
Kelp on rock substrate	0.0	0	0
Total including other habitats	1989.5	326.1	

3.3.3 Anchoring and mooring

Figures 8 and 9 illustrate how the areas where commercial and recreational anchoring and mooring occur tend to be in the sheltered inshore locations along the English coasts. These areas are also often where blue carbon habitats such as saltmarsh, seagrass, kelp, and the more stable muddy sediments tend to exist, as outlined in the previous sections. The overlap of anchoring and mooring activities with seagrass habitats can be seen most clearly for the Isles of Scilly, although the overlap along the south coast is also noticeable (Figures 10 and 11).

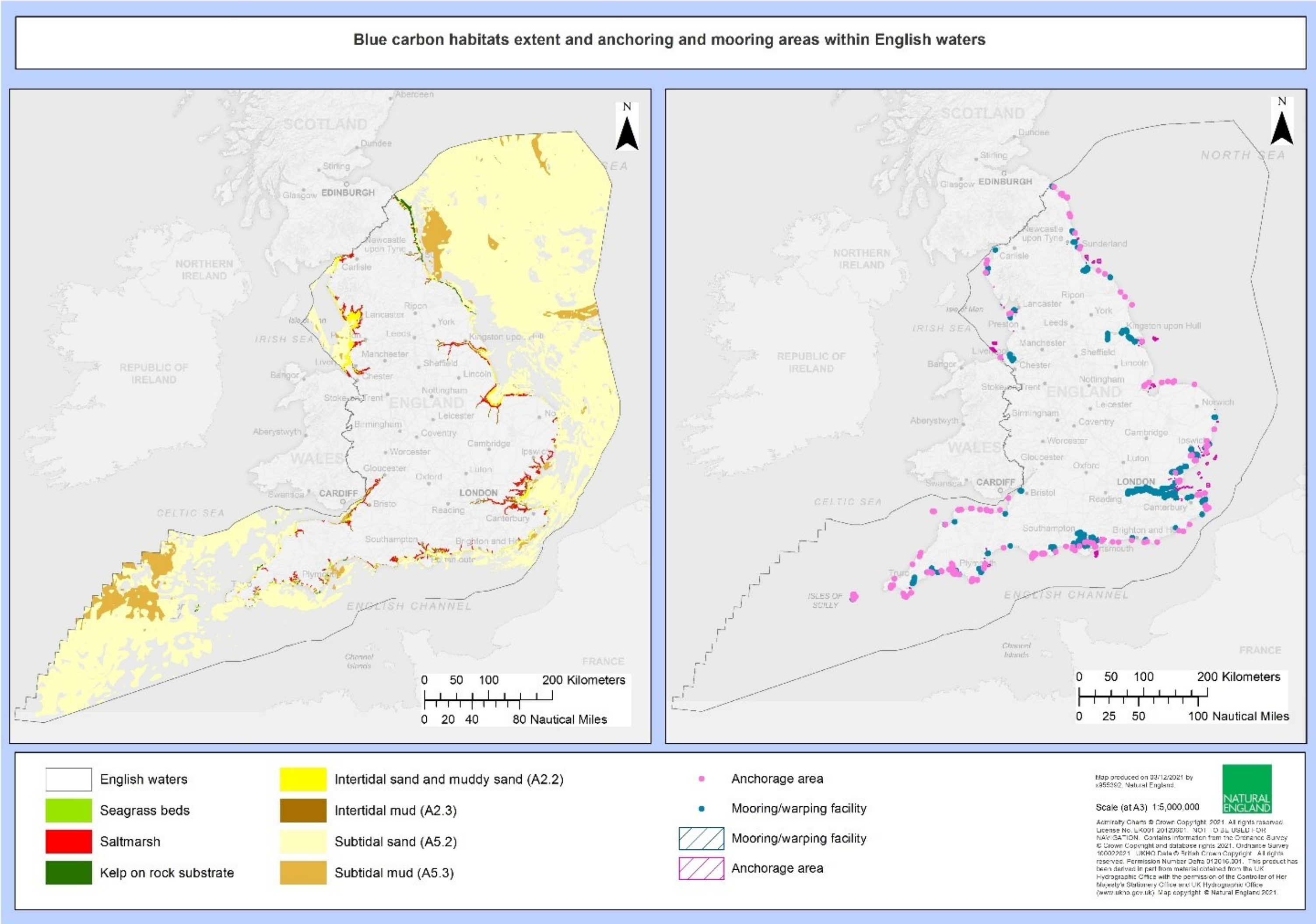


Figure 8. Commercial anchoring and mooring areas with blue carbon habitats in English waters. See Appendix 1 for data source.

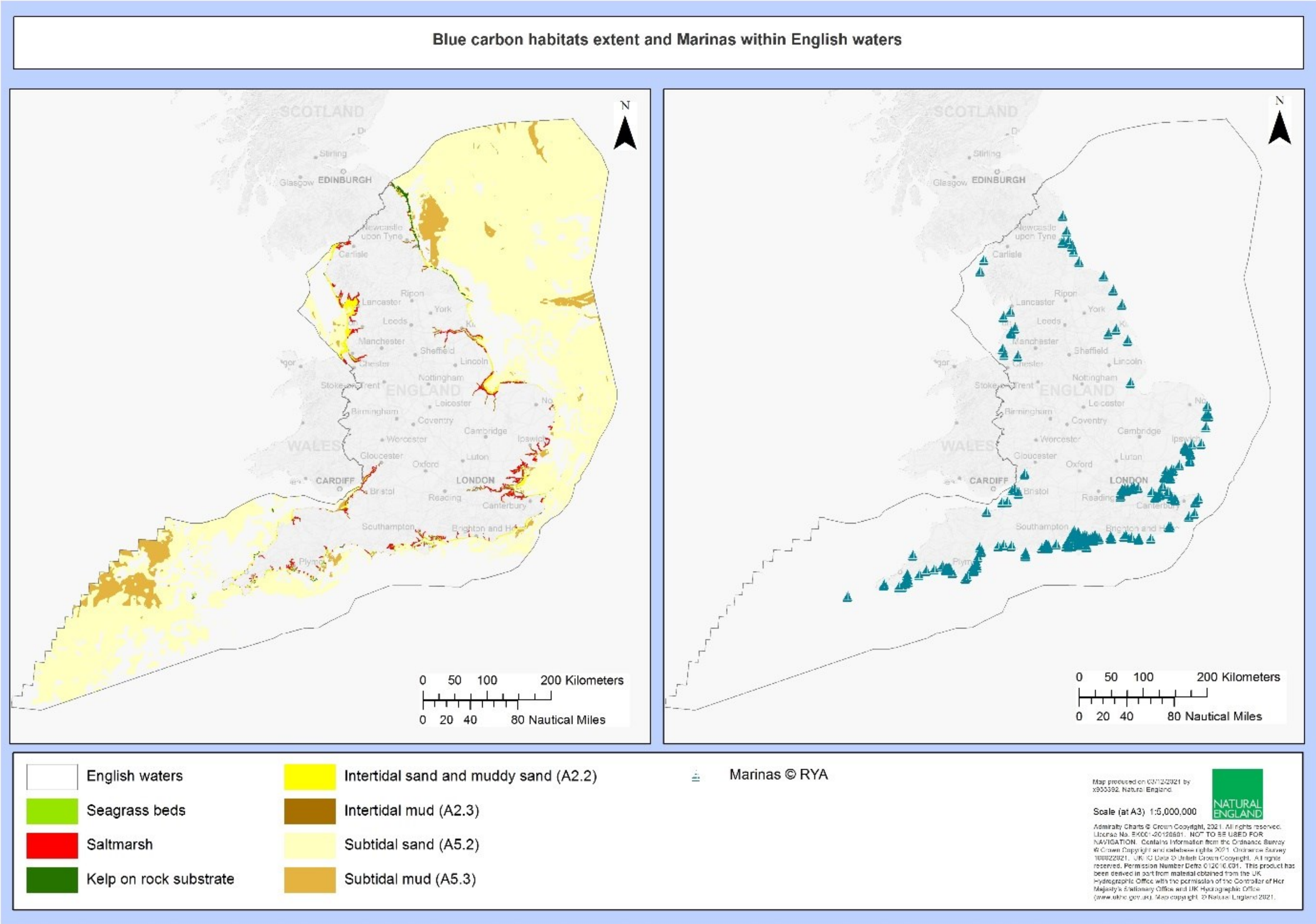


Figure 9. Recreational anchoring and mooring areas with blue carbon habitats in English waters. See Appendix 1 for data source.

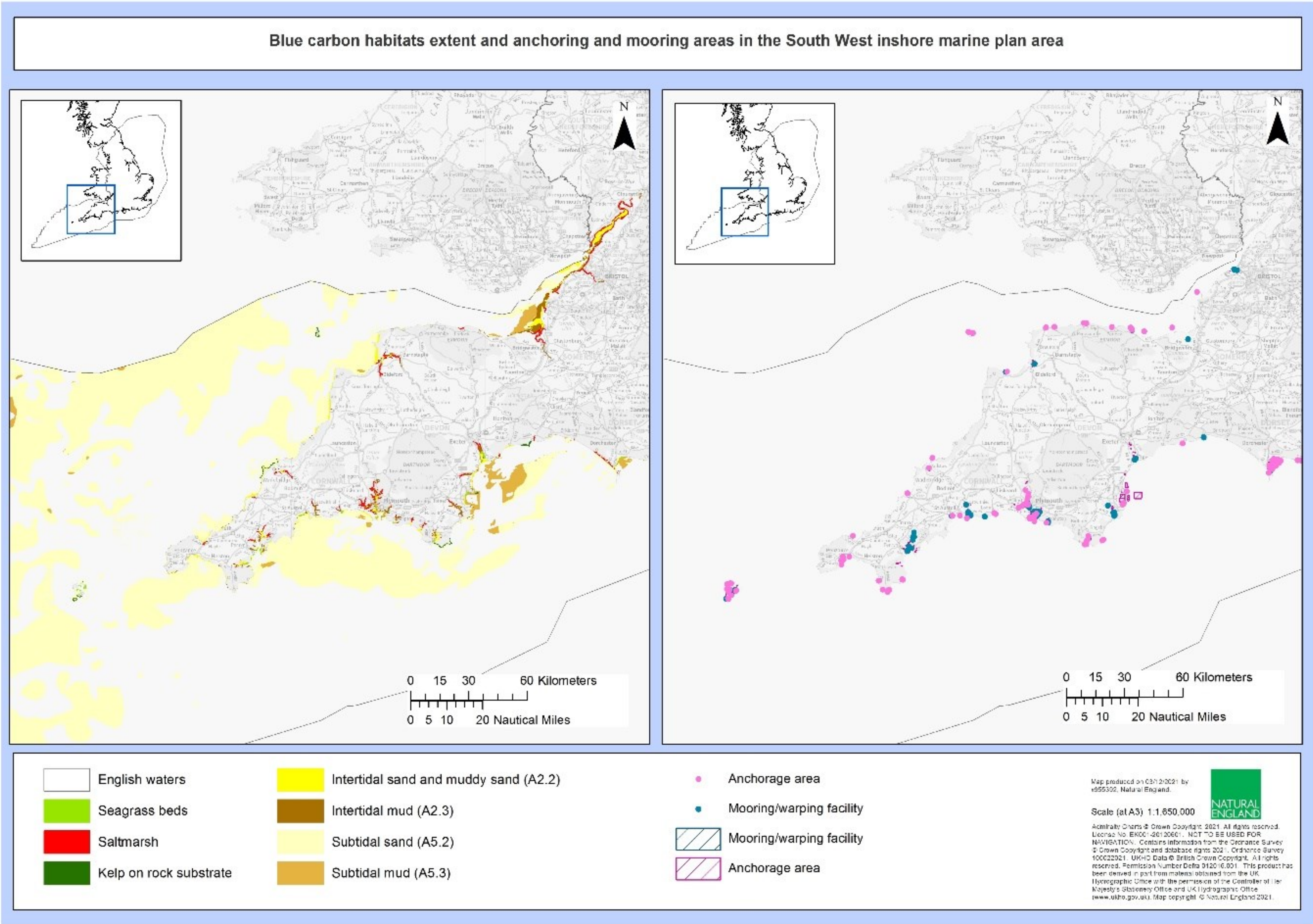


Figure 10. Commercial anchoring and mooring areas with blue carbon habitats in the South West marine plan area. See Appendix 1 for data source.

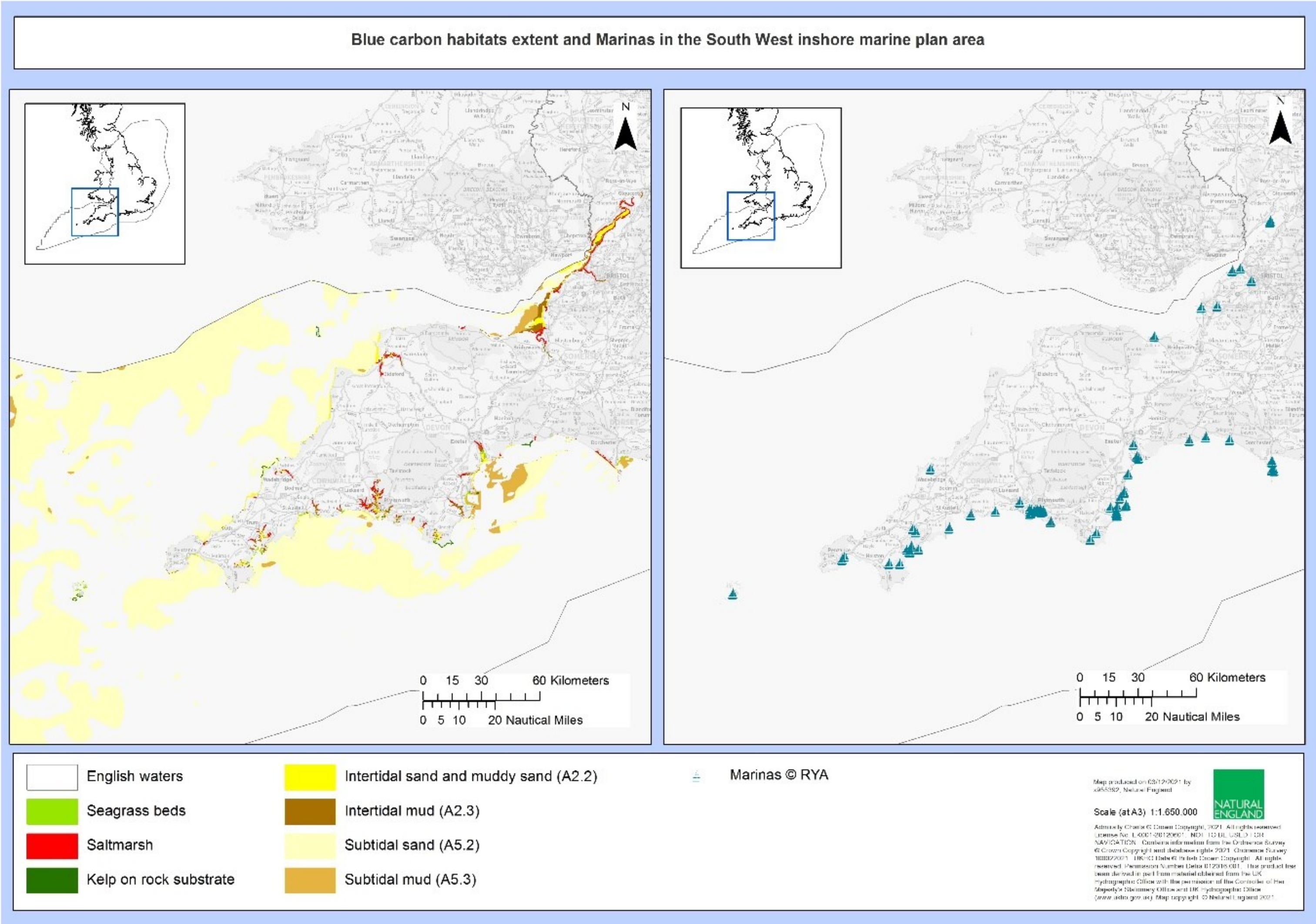


Figure 11. Commercial anchoring and mooring areas with blue carbon habitats in the South West marine plan area. See Appendix 1 for data source.

3.3.4 Offshore wind farms and associated cabling

Figures 12-16 provide an overview of the operational and planned offshore wind farms within the marine plan areas, alongside the current extent and distribution of blue carbon habitats within English waters. The darker shades of blue show the current (active) agreements, whereas the lighter shades of blue highlight the significantly increased footprint planned for OWF in the future, especially in the North Sea and the Irish sea. The offshore blue carbon stocks within the footprint of the wind farms are likely to be affected, and the cabling activities are likely to cause disturbance to the inshore and intertidal blue carbon habitats within their footprint.

In the North West marine plan area, the overlap of current and proposed wind farm developments with large extents of blue carbon habitats, especially subtidal mud, is apparent (Figure 12). As the Irish Sea is less physically dynamic than the North Sea and Eastern English Channel, muddy sediments where carbon is more likely to settle and accumulate are prevalent. Offshore wind farms such as Walney, Ormonde and West of Duddon Sands are all positioned in areas of subtidal mud.

The majority of operational OWFs across the North Sea and Eastern English Channel are within subtidal sand (Figures 13 and 14). Thanet OWF, located off the north east coast of Kent, overlaps with areas of seabed characterised by subtidal mud (Figure 15). The cabling activities overlap with intertidal and nearshore blue carbon habitats such as intertidal muds and saltmarsh, especially within the East and the South East marine plan areas (Figures 15 and 16).

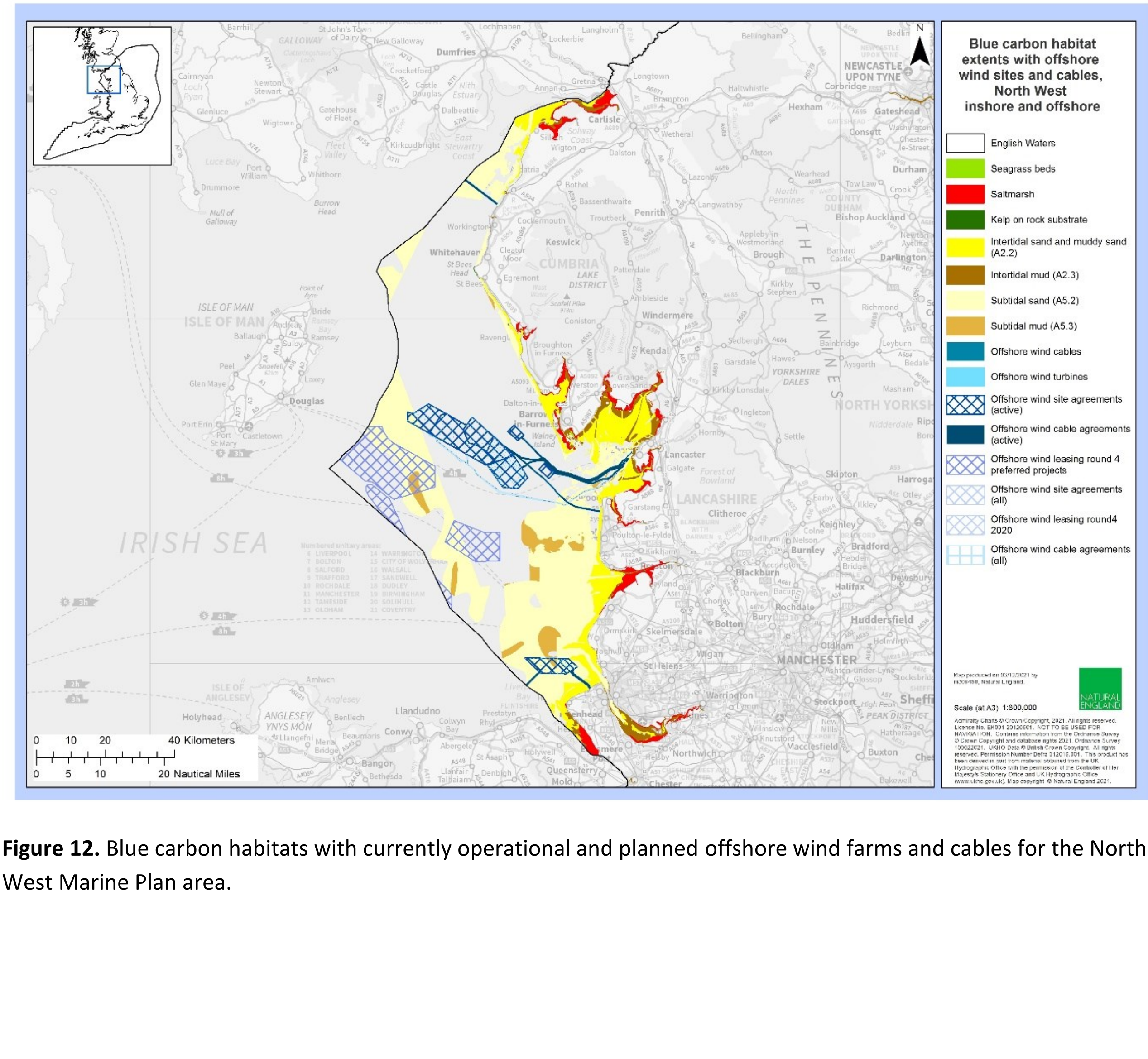


Figure 12. Blue carbon habitats with currently operational and planned offshore wind farms and cables for the North West Marine Plan area.

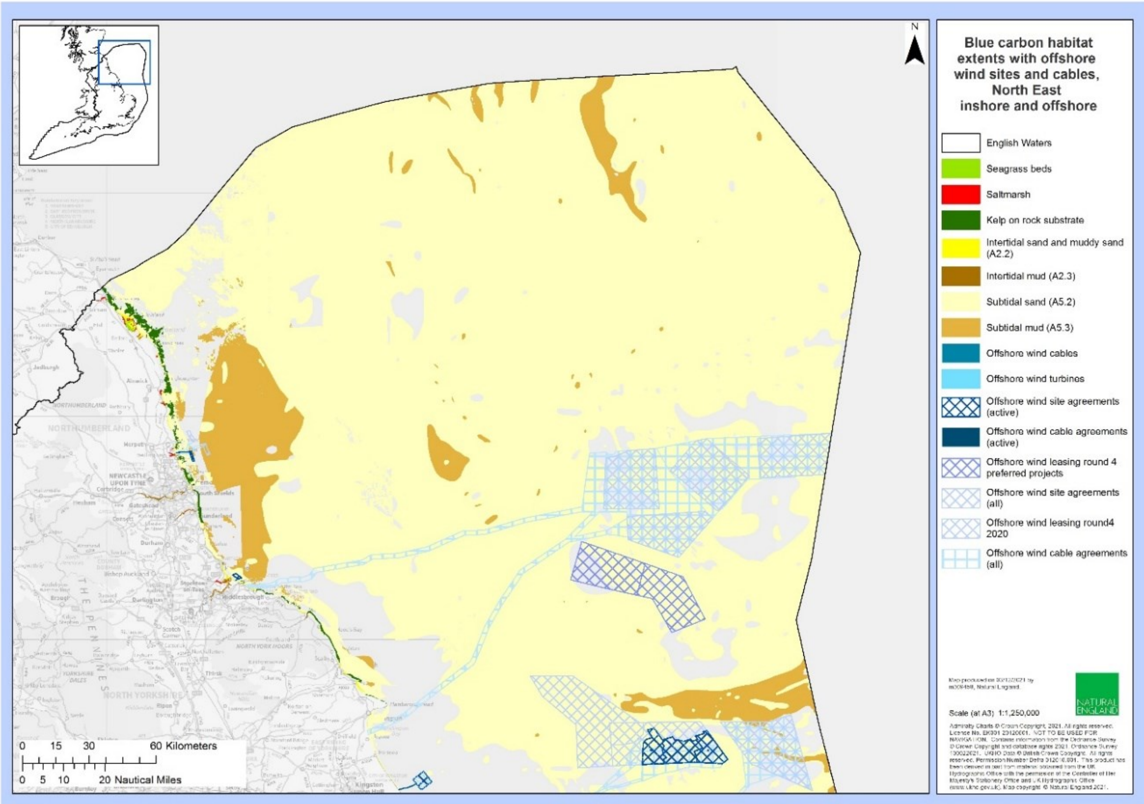


Figure 13. Blue carbon habitats with currently operational and planned offshore wind farms and cables for the North East Marine Plan area.

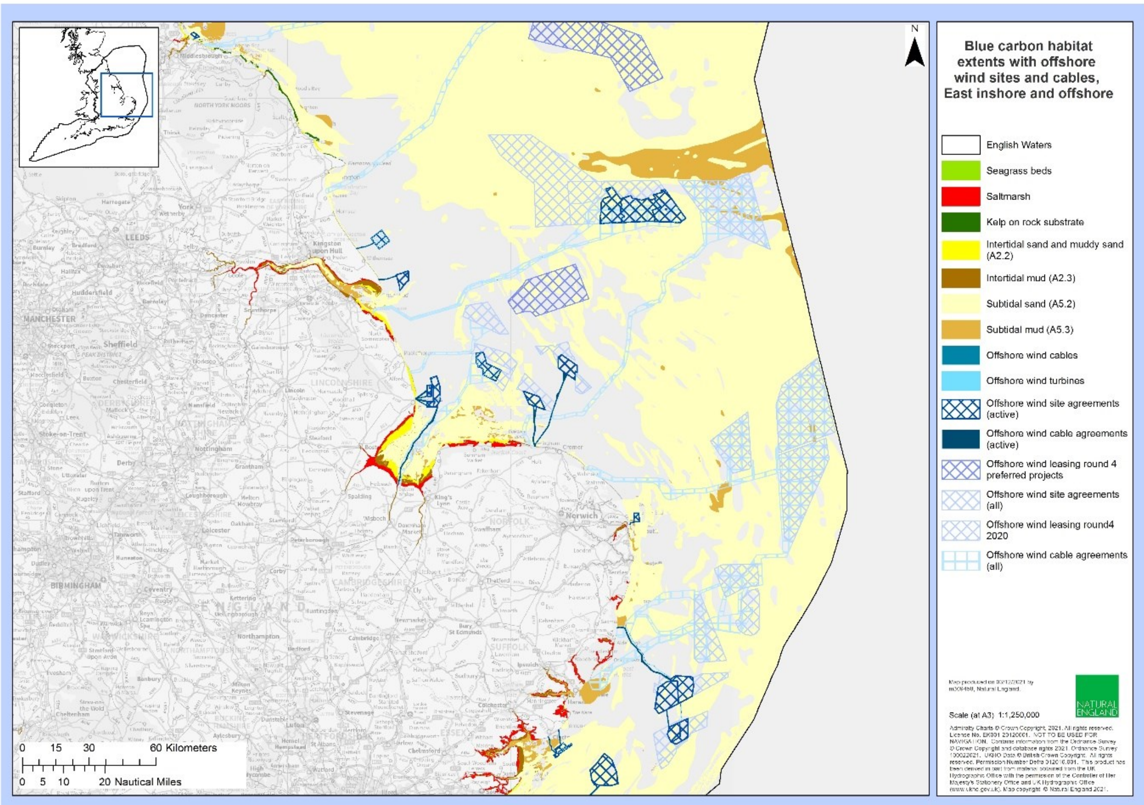


Figure 14. Blue carbon habitats with currently operational and planned offshore wind farms and cables for the East Marine Plan area.

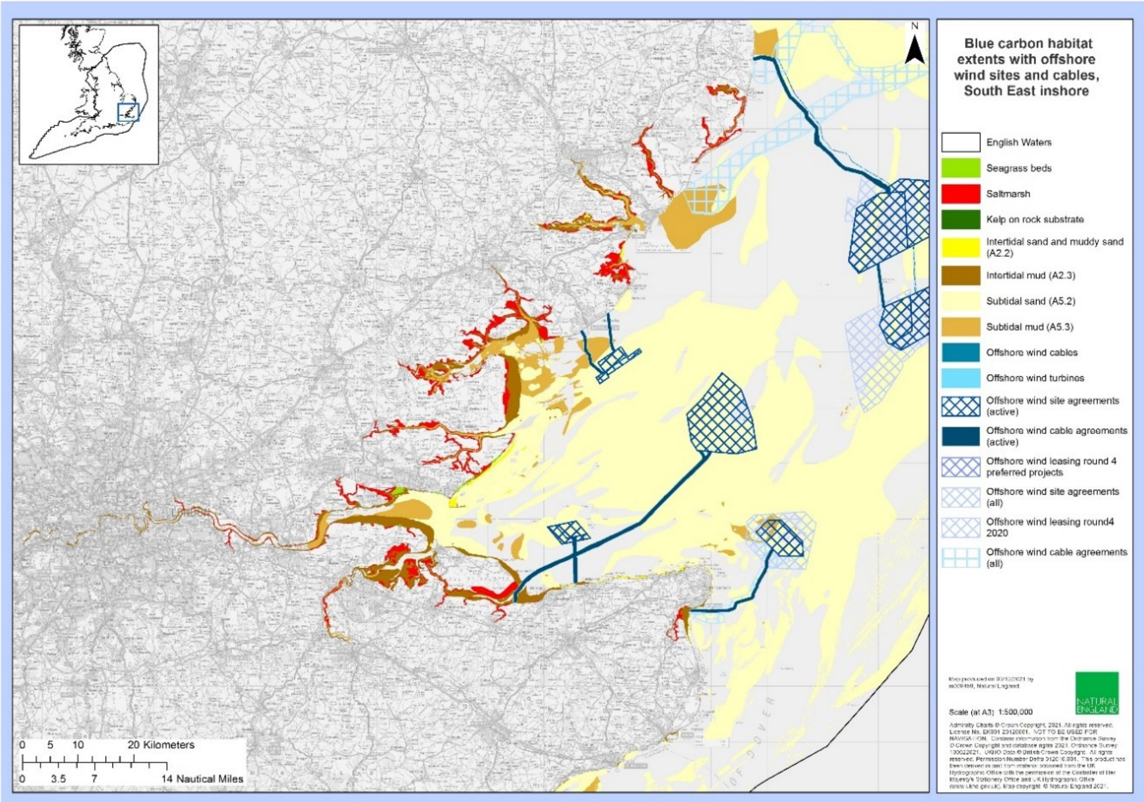


Figure 15. Blue carbon habitats with currently operational and planned offshore wind farms and cables for the South East Marine Plan area.

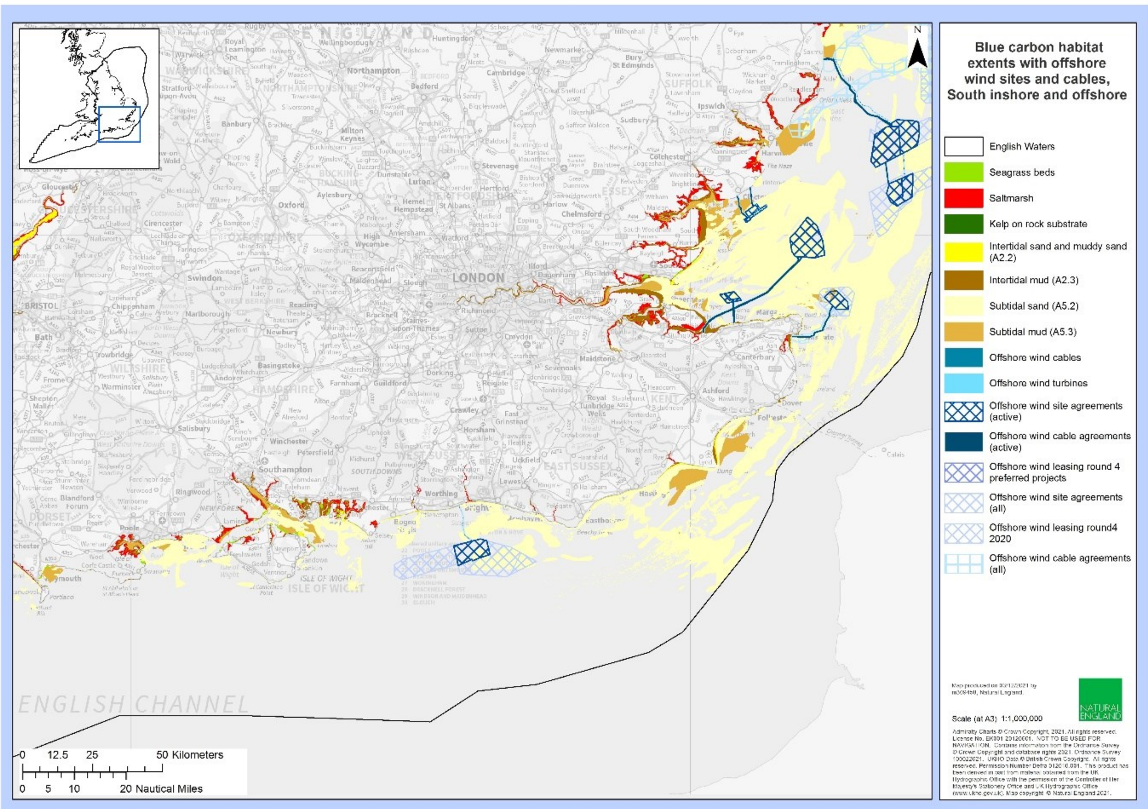


Figure 16. Blue carbon habitats with currently operational and future offshore wind farms and cables for the South Marine Plan area.

3.4 Discussion

In this section, the exposure of blue carbon habitats to some key anthropogenic activities has been explored by comparing the overlap of current and potential extent of blue carbon habitats with spatial data on eutrophication (Figure 6), bottom trawling (Figure 7), anchoring and mooring (Figures 8-11), and offshore windfarm infrastructure, including associated cabling (Figures 12 - 16).

3.4.1 Eutrophication

The mapping of WFD DIN and macroalgae classifications provided a broad overview of the areas currently subject to nutrient pressure and the extent of blue carbon habitats which are currently located in these areas that could be at risk. Whilst this gives an overview of the areas likely to be most affected, it is recognised that most of the large estuaries, inlets and bays are complex systems and the spatial scale of WFD classification can result in smaller areas that remain eutrophic being masked by measurements being averaged over a large geographical area or, conversely, smaller areas of improvement being concealed. To assess where elevated nutrients may be impacting habitats, appropriate scaling is required to reveal acute water quality pressures at a local level. Elevated nutrient levels are already cited as a contributary cause of unfavourable condition on many Marine Protected Areas and coastal Sites of Special Scientific Interest, and there is a body of historic work by Environment Agency and Natural England identifying relevant point and diffuse sources and potential management required. EA (2016) described the effects of eutrophication at Lindisfarne NNR, noting the littoral sediment was impacted by extensive mats of green opportunistic macroalgae; with dead macroalgae smothering saltmarsh and threatening to lead to the deterioration of seagrass. Modelling in Poole Harbour has resulted in a bespoke nutrient loading target aimed at restoring the site features to favourable condition. In the Solent, elevated nutrient levels and the dominance of macroalgae on the mudflats, seagrass and saltmarsh features underpin their unfavourable condition (Bardsley, 2020).

Restoration of blue carbon habitats is a complex process which may be affected by various environmental variables and other factors. Early *et al.* (2020) attempted to predict the key environmental variables that determine seagrass habitat in the Solent and the Plymouth Sound. Although identifying there may be a low tolerance to high levels of seasonal deposition of nitrogen in the Solent, the strongest predictor was found to be bathymetry, but also highlighted the crucial importance physical sediment characteristics on seagrass distribution.

Nitrogen and carbon cycles in estuaries are closely interlinked and plants, and animals and microbes require a fixed ratio of the two elements. An improved understanding is required of this relationship and how it may impair the ability of coastal habitats to function as carbon stores. However, in eutrophic waters the carbon storage and accumulation rates of seagrass and saltmarsh habitats can be severely compromised, not just through loss of habitats, but by restricting plant growth to the point of die-off and breakdown where they become sources rather than sinks of greenhouse gases (Crooks, *et al.,* 2011). The anoxic impact from smothering on sediment habitats may also affect the process of carbon burial by benthic infauna where the redox layer is very small or absent.

It would be useful to look in more detail at the areas of high nutrient pressure, building on the work already ongoing by Natural England and Environment Agency to meet biodiversity objectives, to align these with a strategy to restore coastal habitats through collaborative partnerships to deliver a range of objectives from ecosystem recovery to improving carbon accumulation and storage[10]. It would also be preferable to expand the eutrophication analysis to cover the marine areas beyond the WFD assessment, and to include analysis of phytoplankton data to improve our understanding of the part elevated nutrients plays in impacting estuarine systems, although the data available for phytoplankton are more limited. Other variables associated with increased nutrient loading, such as increased sedimentation and turbidity, should also be considered in any future work.

3.4.2 Bottom trawling

Disturbance of blue carbon sediments through bottom trawl fishing activity can result in the remineralisation of the organic carbon stored within subtidal sediments, leading to an increase in ocean acidity and a reduced potential of seas to absorb atmospheric carbon dioxide (Lovelock *et al.,* 2017). Although the use of bottom towed gear is the most widespread activity causing sediment disturbance in English waters, inshore fishing activity data for vessels smaller than 12m in length is scarce. To explore the use of bottom trawl fishing activity data to investigate the potential risk to blue carbon habitats in inshore waters, this study focused on the south coast of England where a dataset on vessel activity was available from the Sussex IFCA.

The spatial comparison of blue carbon habitats and bottom trawling activity in the Sussex IFCA district indicates that the main blue carbon habitat subjected to abrasion and penetration pressures from bottom trawling activity is subtidal sand (Figure 7, Table 8). The overlap of trawling activity with over 40% of the subtidal mud habitat present within the area implies that fishing activity could be more focused within this habitat, however the total extent of subtidal mud within the studied area is small (Table 8). Intertidal sediments and very small area of saltmarsh also overlapped with the bottom trawl activity data, suggesting that bottom trawling could potentially have a localised impact on intertidal habitats at high tides. However, as the Sussex IFCA activity data were presented within 1km x 1km grid cells, and it is unclear where within the squares the activity has occurred or to what intensity.

The results of this study suggest that there could be a local scale impact of bottom trawling on blue carbon stocks within the Sussex IFCA district. However, the recent introduction of the Sussex nearshore trawling byelaw will help protect and enhance sediment carbon stocks associated with the blue carbon habitats previously subjected to trawling as long as the byelaw remains in place and is adhered to (Sussex IFCA, 2019).

Fishing activity impacts need to be assessed at a wider scale to better understand the risks to blue carbon habitats within the coastal waters of England. Work is ongoing to collate and analyse the various datasets available on the activity of fishing vessels under 12m length in inshore waters, and further analysis should be carried out for other IFCA districts once sufficient data are available. Once these data are available, a wider analysis including all English inshore waters can be overlaid with the blue carbon extent and distribution maps, and the extent of each blue carbon habitats impacted by trawling activity can be quantified.

---

[10] 20201208_Blue Recovery Fund_Feb2021.pdf (ecsa.international) [Accessed December 2021]

3.4.3 Anchoring and mooring

The disturbance caused by commercial and recreational anchoring and mooring activities on vulnerable blue carbon habitats, especially seagrass, is becoming increasingly well documented in the UK and worldwide. The average impact area from moorings within seagrass beds has been calculated as $122m^2$ (Unsworth *et al.,* 2017) and 1- $4m^2$ (Collins *et al.*2010) for mooring and anchoring scars, respectively. These data suggest that the localised impact of traditional swing moorings is larger than that created of anchors. However, the impact of anchoring is likely to be unpredictable and variable spatially and over time because unlike moorings, anchors are not fixed to a location.

Even though the impacts of anchoring and mooring activities on seagrass beds tend to be localised, the damage to habitat extent and condition could have considerable impacts on carbon accumulation and storage. In Australia, carbon stocks were found to be five times lower in mooring scars (1.3 Kg C m^{-2}) than in the surrounding undisturbed seagrass bed (6.4 Kg C m^{-2}) (Serrano *et al.,* 2016), and with the average mooring scar $122m^2$ (Unsworth *et al.,* 2017), mooring has the potential to reduce the carbon accumulation rates of seagrass habitats. The impact an anchor has on seagrass can depend on the type and size of anchor. The number of shoots uprooted during the complete anchoring process range from 1.8 (Hall and Danforth) to 5.5 (Folding Grapnel) (Milazzo *et al.,* 2004).

Once damaged, seagrass beds are expensive to restore and often take long periods of time to recover. In many cases, the attempts at restoration have been unsuccessful (Fonseca, 1994). Protecting existing seagrass habitat should therefore be the priority, to ensure that carbon stocks and ongoing accumulation are secured. The EU Life Recreation ReMEDIES[11] partnership project aims to protect and enhance seagrass habitats within five SACs in the south coast of England through restoration and reduction of anchoring and mooring impacts by encouraging local boat owners to change their behaviours and to adopt new, environmentally friendly mooring systems that avoid damage to seagrass habitats. Simple, cost-effective modifications designed to reduce the impact of mooring chains on seagrass beds have been shown to significantly increase shoot density and blade length of seagrass, improving habitat condition and extent (Luff *et al.,* 2019). In Studland Bay MCZ, MMO's new voluntary approach for the management of anchoring includes a no anchor zone within the protected seagrass bed, which is expected to reduce disturbance and improve the condition of this blue carbon habitat[12].

The spatial overlap of blue carbon habitats with both commercial and recreational anchoring and mooring activity data within English inshore waters (Figures 8 and 9) and within the South West marine plan area (Figures 9 and 10) suggests that mooring and anchoring pressures do not only potentially affect seagrass habitat which overall is relatively scarce in the English waters, but these activities occur across all English inshore regions and spatially overlap also with other blue carbon habitats including saltmarsh, kelp, and intertidal and subtidal sediments. The potential impacts of anchoring and mooring activities on other blue carbon habitats are not well documented and should be subject to further research.

---

[11] https://saveourseabed.co.uk/ [Accessed December 2021]
[12] https://www.gov.uk/government/publications/managing-marine-non-licensable-activities-studland-bay-next-steps [Accessed December 2021]

3.4.4 Offshore wind farms

This report has illustrated a clear overlap of key blue carbon habitats with the existing and planned OWF installations. The habitats with the highest carbon stocks particularly at risk are likely to be subtidal muddy sediments in the North West Marine Plan area where the footprint of existing and planned OWFs overlap with the habitat, and intertidal mud and saltmarsh, where the wind farm cables intersect the land-sea boundary especially within the East and South East inshore marine plan areas (Figures 12-16). As very little information is currently available on the physical impacts that offshore wind farm construction, operation and decommissioning or repowering may have on blue carbon habitats, it remains unclear whether the turbines and cables are a significant threat to the extent and condition of blue carbon habitats. It is also possible that the OWF may protect carbon stocks and ongoing carbon accumulation from other activities such as trawling, as even though there is no legal restriction on fishing in windfarms, fishers tend to avoid them due to conflicts between gear and infrastructure.

Although long-term data monitoring of OWF is lacking, some evidence suggests that offshore wind turbines may act as artificial reefs to bivalves such as blue mussels, creating hotspots of biological activity and changed ecosystem dynamics (Maar *et al.,* 2009) and potentially contributing to accumulation and storage of blue carbon, the role of bivalves in blue carbon cycle is controversial and evidence is lacking.

3.4.5 Linking potential anthropogenic activities and pressures to habitat sensitivity

This study has investigated the potential linkages between the human – induced pressures and activities with blue carbon habitats. To quantify the impacts, benthic sensitivity models can be applied to each blue carbon habitat and pressure combination. Benthic sensitivity model has been developed at Natural England, and the proof-of-concept study to explore its use to investigate impacts of pressures on blue carbon habitats can be found in Appendix 2 of this report.

4. General discussion

This study highlights the paucity of carbon data in English waters and the need for further data collection and analysis to quantify carbon stocks and fluxes in the marine environment. There is sufficient information in the literature to form broad estimates of carbon stocks for coastal and marine habitats (compiled by Parker *et al.,* 2021 and Gregg *et al.,* 2021) and to compare sequestration, sediment carbon accumulation and storage, and the interconnectedness of different blue carbon habitats. However, these data are of low confidence for most marine habitats in English waters, largely due to the small number of highly variable datasets collected from a limited number of sites or locations. The existing data have shown that coastal habitats hold the largest carbon stocks in English waters, with saltmarsh being by far the richest carbon store per unit area and with a higher carbon accumulation rate than any other blue carbon habitat (Parker *et al.,* 2021; Gregg *et al.,* 2021). Notwithstanding the evidence gaps, it is therefore clear that the coastal habitats are key carbon stores and have the potential to continue to build carbon stock via sequestration and sedimentation. Damage to and losses of these habitats can lead to a release of stored carbon, a proportion of which is ultimately emitted to the atmosphere. In this study, a clear priority which emerges is the protection of existing habitats which store and accumulate carbon, to maintain existing stocks and prevent further releases. In recent decades, the major losses in saltmarsh in England have occurred in the South East marine plan area, and the biggest potential to increase the extent of saltmarsh and littoral sediment habitats is in the East marine plan area. Efforts should be focussed in these plan areas to address the pressures on saltmarsh, including loss of extent due to development, sea level rise leading to coastal squeeze, eutrophication, interruptions of sediment supply through dredging, and erosion caused by boat wash and potentially even trawling on the fringes where the saltmarsh is submerged by the recent higher than expected tides.

Whilst the coastal habitats sequester carbon and represent the richest stores per unit area, the largest total stocks are those accumulated and stored in subtidal sediments. Understanding the origin of this stock is important. The physical and biological carbon pumps drive the transfer of carbon from the atmosphere via photosynthesis by phytoplankton into the deep sediments. Allochthonous carbon originating from the detritus from coastal habitats including the shells of marine animals and plant material such as seagrass and kelp also gets transported to and buried in subtidal sediments. This carbon pump plays a significant role in regulating atmospheric CO_2 levels (Passow and Carlson, 2012) and protecting the processes in the water column that result in these seafloor stocks is as critical as protecting the stocks themselves. Subtidal sediments are sensitive to some of the same pressures as intertidal sedimentary habitats, such as abrasion and penetration, which mobilise the sediment and can lead to the release and loss of carbon to the atmosphere. The activities causing these pressures in the offshore environment were not explored in this study, however work in parallel led by Cefas aims to map relevant activities offshore to examine the potential impact on offshore sedimentary carbon stocks. Protection of these vast stocks is a priority to ensure they continue as a repository for carbon lost from land and that sequestered by, but transported out of, coastal habitats.

Comparisons of current inshore blue carbon habitat extent and distribution with areas where these habitats could be restored/created indicate the considerable potential for increasing the extent of some blue carbon habitats in English waters. Several authors have highlighted the importance of prioritising protection of existing habitats over restoration due to the long timescales required for carbon stocks to recover (Thomson *et al.,* 2017). Macreadie *et al.* (2017) concluded that more attention needs be placed on optimising the efficiency of existing blue carbon systems rather than achieving carbon offsets through blue carbon habitat restoration. Several studies have shown lower carbon stocks in restored saltmarsh habitats than within natural saltmarshes, up to several decades after initial restoration interventions had taken place (Gregg *et al.,* 2021), which is unsurprising given the stocks in existing saltmarshes are the result of centuries of accretion. Recent research suggests that accumulation rates in new saltmarshes can in fact be rapid and initially higher than established saltmarshes for up to 20 years, after which the accumulation rate reduces to a steady state (Burden *et al.,* 2019). This reinforces the need to protect historic stocks of carbon held in existing saltmarsh so that carbon is in effect 'banked' in perpetuity and in optimal conditions will continue to accumulate. Expanding the extent of saltmarsh habitat by restoring and creating new areas will bring added value to historic stocks through accumulation of even more carbon which will contribute to enhanced stocks over time.

In addition to the ability of a habitat to accumulate and store carbon, an important consideration in prioritising restoration action is the flow and fate of carbon and the ecological and physical interactions between blue carbon habitats. The individual blue carbon habitats examined in this study do not function in isolation but as mosaics with interdependencies and ecosystem linkages relevant to climate regulation and carbon storage that make the ecosystem much more than the sum of its parts. For example, even though the existing seagrass and oyster habitats do not contribute significantly to the carbon stocks in English waters due to their relatively small extents, both habitat types have key roles in the carbon cycle and in stabilising the underlying sediments. The contribution of these blue carbon habitats should therefore not be overlooked, especially given the considerable potential for restoration to increase their extent. Globally, seagrasses occupy less than 0.2% of the seabed, but they are estimated to sequester 10% of the yearly ocean organic carbon (Fourqurean *et al.,* 2012). Although the biomass carbon stock of seagrass (held in the vegetation) is lower than the sediment carbon stock of littoral mud, restoring seagrass on littoral mud will lock in and protect the existing sediment stock and continue to sequester and trap sedimentary carbon, thus enhancing the total carbon storage value. The loss of sediment and detritus from vegetated blue carbon habitats will increase the carbon store value of surrounding sediment including the littoral mud (Thomson *et al.,* 2017; Duarte and Krause-Jenson, 2017), providing it is not subject to disturbance which resuspends the carbon rich sediment. European flat oyster beds and other reef types can also play an important role in protecting adjacent habitats such as saltmarshes. Reefs form a physical barrier to waves and their actions filtering water, lowering turbidity, and improving light levels are believed to have had an essential role in the past in maintaining healthy conditions within which other habitats such as seagrass can recover (Green *et al.,* 2021). Restoring European flat oyster reefs as part of habitat mosaic could magnify the carbon accumulation.

As well as being a standing biomass carbon stock, kelp habitats are thought to play a role as carbon donors to adjacent areas when the standing stock of plants breaks down and gets transported elsewhere by currents. Querios *et al.* (2019 found a positive, year-round carbon flux between kelp beds (detritus) and nearby subtidal sediments off Plymouth. The extent to which kelp detritus supports other global blue carbon habitats on the coast, in shallow shelf seas and the deep ocean are yet to be fully quantified but the evidence available so far suggests it could be significant (Krumhansl and Schreibling, 2012). Kelp habitats are also important nursery and feeding grounds for invertebrates and fish. However, their role in supporting a higher biomass of animals further up the food chain is not sufficiently quantified in English waters to allow estimates of potential carbon gains. As with all the coastal blue carbon habitats, the contribution of kelp to overall carbon stocks should not be viewed in isolation but as part of a functioning coastal ecosystem which optimises blue carbon potential.

In the same way the blue carbon habitats are connected to one another by natural processes, so they are linked to land via the water environment, exposing them to a multitude of anthropogenic pressures relating to land use, land management and modification of watercourses. It is estimated that around 2.7 billion tonnes of terrestrial carbon enter freshwaters globally every year, 50% of which returns to the atmosphere as CO_2 (Gregg *et al.,* 2021). Drainage of peatlands, modification of river channels and loss on functioning floodplains has contributed to the increased loss of carbon and nutrients from terrestrial systems in England. Chemical pollution from industrial discharges and shipping was the dominant pressure on the estuarine environment during the last century but inputs are now regulated. Nutrient loading is the remaining key pressure which has contributed to the continued decline in quality and loss of estuarine habitats. Estuaries have undergone major ecological shifts from supporting extensive seagrass to being dominated by opportunistic macroalgae. Estuaries with large historical losses of seagrass, likely caused by several pressures including the extensive outbreak of wasting disease in the early 20th century, have continued to experience further losses in recent decades (Green *et al.,* 2021). Other land-based pressures that have contributed to the seagrass loss include increased sedimentation. This ecological shift has been observed to occur in coastal waters throughout the world where they have become eutrophic (Green *et al.,* 2021).

Measures to reduce nutrient loading in catchments where there is evidence of ecological impact have been ongoing for over two decades and significant reductions have been made via the Periodic Review of water company investments process and by better regulation of farming practices, designation of nitrate vulnerable zones, and application of catchment sensitive farming initiatives. However, this has not been sufficient to reduce nutrient inputs to the levels needed to achieve good ecological status in the English transitional and coastal waters and achieve favourable condition of MPAs. There are various other initiatives, funds and a plethora of legislation around managing land to help mitigate climate change which could be synergistic with actions to reduce nutrient inputs from land. It is crucial to join these up at the policy level in a way that helps to address the impacts on marine habitats. Further action is needed in the shape of catchment scale plans for nutrient and carbon management (that may be Diffuse Water Pollution Plans where applicable). Macreadie *et al.* (2017) found a reduction in nutrient inputs to be the key to improving the accumulation and storage of blue carbon. The importance of managing carbon stocks at a landscape scale was also highlighted, with the flux of carbon influenced by enhanced carbon export from soils through agricultural run-off (Regnier *et al.,* 2013). In 2017, agriculture contributed to 10% of the total UK greenhouse gas emissions: 70% of total nitrous oxide emissions, 50% of the total methane emissions and 1% of total carbon dioxide emissions (Defra, 2019). Improved management of agricultural soils and creation of buffers around arable areas will not only improve their carbon stock and reduce the greenhouse gases emitted, but also facilitate the reduction of nutrient inputs into watercourses and the negative effect this may have on estuarine blue carbon habitats. Land use change of this nature is beginning to be used to offset point source inputs from development in order to achieve nutrient neutrality in MPAs which are in unfavourable condition. Investment into further changes in land use and agricultural practices on a landscape or catchment scale, such as improved soil management and reinstatement of woodland habitat and wetlands around watercourses, would benefit terrestrial carbon and nitrogen management strategies, contribute to nature recovery on land, and ultimately improve how estuaries and coastal waters function with associated increase in blue carbon.

English waters have been modified by human activities over hundreds of years, as our use of the seas has intensified for transport, oil and gas extraction, communications, military operations, food, aggregate and more recently energy production, industrial development, urbanisation and recreation. The direct physical imprint of seabed disturbance, chiefly through the sediment abrasion and penetration caused by bottom trawling, has rapidly increased globally since the 1950s and fishing activities that impact seabed sediments have continued ever since (Smeaton *et al.*, 2021; van de Velde *et al.,* 2018). Dredging activities associated with aggregate production and maintenance of harbours and waterways have also greatly intensified in recent centuries and like bottom trawling cause physical disturbance to the seabed. The carbon stocks these sediments contain are remobilised and become available in the water column, some of which could be released back into the atmosphere as CO_2. The analysis of overlap of fishing effort data from the Sussex Inshore Fisheries and Conservation Authority with the blue carbon habitats shows that large areas of the inshore habitats are subject to trawling activity, in particular subtidal sand. There was also an overlap with potential seagrass habitats, indicating that this could be a cause of loss and inhibiting recovery. Analysis of other activities showed additional pressures from commercial and recreational anchoring and mooring and offshore find farm infrastructure in similar locations as eutrophication pressure. Although the cumulative impacts were not studied directly here, this geographic overlap highlights the complexity of management considerations that are required and the need for a strategic approach rather than siloed sectoral measures.

In the past ten years there has been an extensive programme of designation of Marine Conservation Zones in England which, with the existing Special Protection Areas and Special Areas of Conservation, largely completes the Marine Protected Area network. A total of 178 MPAs are now in place, 157 sites covering 51% of inshore waters and 40 offshore covering 37% of this region; in total this covers 92,633km^2. Activities which could damage the interest features of these sites or hinder their conservation objectives are regulated mainly via marine planning and licensing, and a programme of assessing the risk to MPA features from commercial fishing activity has been ongoing for several years. As a result of the assessments, byelaws are being introduced by the Marine Management Organisation and the Inshore Fisheries and Conservation Authorities to manage the activities, so they do not cause deterioration of the features or hinder achievement of the site conservation objectives. There are also examples of more holistic approaches being taken, for example the Sussex IFCA employed a natural capital approach to evaluating the wider benefits of management actions and have put in place a byelaw prohibiting the use of towed gear (trawling) in a large area of their district (not MPA) in order to recover kelp and associated seabed habitats.

As an independent coastal state, the Fisheries Act 2020 contains new powers enabling the MMO to implement management measures across English waters (not just in MPAs). A climate change objective will also be embedded in the new Fisheries Management Plans. The MMO is developing a programme to assess sites and implement byelaws, where necessary, to manage fishing activity in all English offshore MPAs. It should however be noted that ecological feature-based conservation, as employed in UK MPAs, means that protection and management measures are applicable only to areas within sites which are occupied by interest features. The MPA network extent does not confer protection of 40% of the seabed or the associated carbon stocks where features are not related to blue carbon habitats, and activities which disturb the seabed and affect blue carbon habitats may therefore still occur within MPAs.

The Government is committed to designating a number of pilot Highly Protected Marine Areas (HPMAs) based on criteria including biological diversity, naturalness, recovery potential, and importance for long term carbon storage. HPMAs will allow the marine ecosystem to recover to a more natural state, increasing its resilience and potentially enabling it to adapt to climate change impacts. Protection of HPMAs will take a whole site approach, protecting all species and habitats within the site as well as the associated processes, including in the water column. Evaluation of changes to biodiversity and the provision of ecosystem services during the pilot phase of HPMAs will improve the evidence base on the recovery of marine ecosystems. This study highlights the multiple pressures on blue carbon habitats, their ecosystem linkages, and potential to recover if pressures were removed. We show that a large proportion of the existing blue carbon habitat extent is within the existing MPA network and describe the shortfalls of a feature-based system which was not designed to protect carbon stocks. The benefits of the whole site approach in enhancing blue carbon are clear, as this would protect the majority of the existing coastal carbon stock. Considering more holistic management of MPAs and recovery of the wider seas to conserve and enhance blue carbon habitats alongside the piloting of HPMAs would be prudent, given the urgency of the climate crisis.

In summary, improving the quality of existing habitats to maximise their capacity for carbon accumulation and restoring mosaics of habitats together rather than singe habitats in isolation will optimise carbon storage and minimise loss, while creating greater resilience to climate change. Whilst physical intervention to create new habitat is possible and desirable on the landward boundary, for example through managed realignment to create intertidal mud, saltmarsh, saline lagoons and reedbeds, wherever possible the aim should be to create or restore self-sustaining systems where natural processes dictate the resulting habitat and ongoing intervention is not required. For blue carbon habitats that are subtidal and further away from the coast, physical intervention to create habitat becomes less viable and in the main, and removing anthropogenic pressures is the key action required to allow natural processes to resume and the blue carbon habitats to recover naturally. Where the habitat is so degraded that the substrate is not suitable or seed stock is unlikely to arrive/settle, it might be advantageous to initiate recovery through one-off or short-term interventions such as the introduction of sediment, the planting of seagrass or the seeding of oysters, but this should be subject to careful site-specific assessment and planning. There are several resources in development which guide the restoration process if intervention is required (for example Preston *et al.,* 2020). However, restoration efforts are only likely to succeed once the relevant pressures have been removed or reduced to a level where they no longer inhibit the recovery and functioning of the habitats. Consideration of how climate change effects such as temperature changes will influence the distribution of habitats is also crucial, and in some cases restoration of previous extents may no longer be possible.

5. Recommendations for further work

- Further data collection is required to improve the evidence base and reduce the uncertainty around variability in the carbon accumulation and stocks associated with different blue carbon habitats in English waters. The largest evidence gap for management purposes is subtidal sediments, as better understanding of subtidal sediment carbon stocks would directly inform where management would provide the greatest benefits for blue carbon. Improved evidence on carbon accumulation rates and stocks for habitats such as seagrass is also crucial to meet the data standards required to contribute towards our future pathway to more comprehensive carbon accounting for net zero.

- Understanding the cumulative impacts of pressures on all blue carbon habitats is important, and the effects of other activities affecting blue carbon, such as dredging, also need to be quantified. Natural England is currently developing a benthic sensitivity modelling approach to tackle this for inshore and coastal waters, see Appendix 2 for further details. Joining up the inshore and offshore activity analysis and modelling scenarios is key for achieving a comprehensive understanding of impact on all blue carbon habitats in English waters.

- Although the wider ecosystem impacts could be considerable, the implications of different fisheries management scenarios to the wider food chain and carbon cycle in the marine environment are yet to be investigated.

- Protection and restoration of saltmarsh is a priority, as it is the blue carbon habitat with the highest carbon accumulation rate and stock per unit area. Protecting existing habitats should be prioritised over restoration efforts, as the carbon stocks within natural habitats tend to be larger (per unit area) than those for restored habitats. Restoring seagrass and reducing impacts on existing seagrass habitats is also important, not only for the carbon gains but also to achieve wider ecosystem benefits.

- To maximise the climate change mitigation potential of blue carbon habitats, it is crucial to apply landscape scale approaches to restoration and recreation programmes, rather than considering sites in isolation. Restoring and recreating functioning blue carbon habitats will also depend on removing other pressures and activities hindering this process, such as elevated nutrients and anthropogenic activities that have a potential of causing direct physical habitat disturbance.

- Quantifying the role of blue carbon habitats in capturing carbon from elsewhere, including from terrestrial habitats, as well as other marine habitats is fundamental, as understanding this interaction will strengthen landscape scale approaches and help identify co-benefits of restoration and protection.

- The crossover between blue carbon habitats and MPA features needs to be better understood. Identifying this crossover would enable a more accurate assessment of what existing management is in place for the inshore blue carbon habitats and what the ‘carbon gap’ is.

- Nutrient and carbon management plans need to be developed for catchments identified at risk of eutrophication, and a policy join-up is crucial to allow these plans to be effectively executed. This would enable a connected and holistic strategic approach to management of the terrestrial environment and the coastal waters.

6. References

Adams, C.A., Andrews, J.E. and Jickells, T. (2012). Nitrous oxide and methane fluxes vs. carbon, nitrogen and phosphorous burial in new intertidal and saltmarsh sediments. Science of the Total Environment, 434, 240–251. https://doi.org/10.1016/j.scitotenv.2011.11.058.

Alongi, D.M. (2018). Kelp Forests. Blue Carbon: Coastal Sequestration for Climate Change Mitigation. D. M. Alongi. Cham, Springer International Publishing, 53-57.

Alonso, I., Weston, K., Gregg, R., and Morecroft, M. (2012). Carbon storage by habitat: review of the evidence of the impacts of management decisions and condition of carbon stores and sources. Natural England Research Report NERR043.

Armstrong, S., Hull, S., Pearson, Z., Wilson, R. and Kay, S. (2020). Estimating the Carbon Sink Potential of the Welsh Marine Environment. NRW, Cardiff, 74p.

Ashely, M., Rees, S.E., Mullier, T., Reed, B., Cartwright, A., Holmes, L. and Shee, M. (2020). Isles of Scilly Natural Capital Asset and Risk Register to Inform Management of Isles of Scilly Fisheries Resources.

Atwood, T.B., Witt, A., Mayorga, J., Hammill, E. and Sala, E. (2020). Global Patterns in Marine Sediment Carbon Stocks. Frontiers of Marine Science, 7:165. doi: 10.3389/fmars.2020.00165.

Bardsley, L., Brooksbank, J., Giacomelli G., Marlow, A., and Webster, E. (2020). Review of Chichester Harbour sites: intertidal, subtidal and bird features. Natural England Research Report, Number 090.

Beaumont, N.J., Jones, L., Garbutt, A., Hansom, J.D. and Toberman, M. (2014). The value of carbon sequestration and storage in coastal habitats. Estuarine and Coastal Shelf Science, 137, 32–40.

Bijoor, S. (2020). Mainstreaming blue carbon to finance coastal resilience. https://www.conservationfinancenetwork.org/2020/01/27/mainstreaming-blue-carbon-to-finance-coastal-resilience [Accessed Oct 2021].

Brew, D. (in prep). Coastal habitats potential mapping. Royal Haskoning DHV Report for Natural England.

Burden, A., Garbutt, R.A., Evans, C.D., Jones, D.L. and Cooper, D.M. (2013). Carbon sequestration and biogeochemical cycling in a saltmarsh subject to coastal managed realignment. Estuarine, Coastal and Shelf Science, 120, 12-20.

Burden, A., Garbutt, A. and Evans, C.D. (2019). Effect of restoration on saltmarsh carbon accumulation in Eastern England. Biology Letters, 15, 0–3.

Burden, A., Smeaton, C., Garbutt, A.S., Jones, L., Lewis H.D. and Rees. S.M. (2020) Impacts of climate change on coastal habitats relevant to the coastal and marine environment around the UK. MCCIP Science Review, 228–255.

Burrows, M.T., Kamenos, N.A., Hughes, D.J., Stahl, H., Howe, J.A. and Tett, P. (2014). Assessment of carbon budgets and potential blue carbon stores in Scotland’s coastal and marine environment. Project Report. Scottish Natural Heritage Commissioned Report No. 761. 90 pp.

Burrows, M.T., Hughes, D.J., Austin, W.E.N., Smeaton, C., Hicks, N., Howe, J.A., Allen, C., Taylor, P. and Vare, L.L. (2017). Assessment of Blue Carbon Resources in Scotland's Inshore Marine Protected Area Network. Scottish Natural Heritage Commissioned Report No. 957.

Bulseco, AN, Giblin, AE, Tucker, J., Murphy, A.E., Sanderman, J., Hiller-Bittrolff, K. and Bowen, J.L. (2019). Nitrate addition stimulates microbial decomposition of organic matter in salt marsh sediments. Global Change Biology, 25, 3224– 3241. https://doi.org/10.1111/gcb.14726.

Callaway, J.C., DeLaune, R. D. and Patrick, W. H. (1996). Chernobyl ^{137}Cs used to determine sediment accretion rates at selected northern European coastal wetlands. Limnology and Oceanography, 41, 3:444–450. https://doi.org/10.4319/lo.1996.41.3.0444.

Cannell, M.G.R., Milne, R., Hargreaves, K.J., Brown, T.A.W., Cruickshank, M.M., Bradley, R.I., Spencer, T., Hope, D., Billett, M.F., Adger, W.N. and Subak, S. (1999). National inventories of terrestrial carbon sources and sinks: The U.K. Experience. Climatic Change, 42, 3:505–530. https://doi.org/10.1023/A:1005425807434.

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R. and Lynch, J.C. (2003). Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles, 17, 4. https://doi.org/10.1029/2002gb001917.

Collins, K.J., Suonpää, A.M. and Mallinson, J.J. (2010). The impacts of anchoring and mooring in seagrass, Studland Bay, Dorset, UK. Underwater Technology, 29, 3: 117–123.

Craft, C. (2007). Freshwater input structures soil properties, vertical accretion, and nutrient accumulation of Georgia and US tidal marshes. Limnology and Oceanography, 52, 1220– 30.

Crooks, S., Herr, D., Laffoley, D., Tamelander, J. and Vandever, J. (2011). Regulating Climate Change Through Restoration and Management of Coastal Wetlands and Near-shore Marine Ecosystems: Mitigation Potential and Policy Opportunities. World Bank, IUCN, ESA PWA, Washington, Gland, San Francisco.

Cullen-Unsworth, L.C., Mtwana, L., Paddock, J., Baker, S., Mckenzie, L.J. and Unsworth, R.K.F. (2014). Seagrass meadows globally as a coupled social – ecological system: Implications for human wellbeing. Marine Pollution Bulletin. 83, 387-397. https://doi.org/10.1016/j.marpolbul.2013.06.001.

d'Avack, E.A.S., Tillin, H., Jackson, E.L. and Tyler-Walters, H. (2014). Assessing the sensitivity of seagrass bed biotopes to pressures associated with marine activities. JNCC Report No. 505. Peterborough, Joint Nature Conservation Committee.

Deegan, L., Johnson, D., Warren, R., Peterson, B.J., Fleeger, J.W., Fagherazzi, S. and Wolheim, W.M. (2012). Coastal eutrophication as a driver of salt marsh loss. Nature, 490, 388–392. https://doi.org/10.1038/nature11533.

Defra (2019). Agricultural statistics and climate change 9th Edition.

de Haas, H., Boer. W. and Van Weering, T.C.E. (1997). Recent sedimentation and organic carbon burial in a shelf sea: The North Sea. Marine Geology, 144, 131–146. https://doi.org/10.1016/S0025-3227(97)00082-0.

Duarte, C.M. and Krause-Jensen, D. (2017). Export from seagrass meadows contributes to marine carbon sequestration Marine Science. Frontiers of Marine Science, 4, 1-7. https://doi.org/10.3389/fmars.2017.00013.

Duffy, J.E. (2006). Biodiversity and the functioning of seagrass ecosystems. Marine Ecology Progress Series, 311, 233–250.

Duplisea, D.E., Jennings, S., Malcolm, S.J., Parker, R. and Sivyer, D.B. (2001). Modelling potential impacts of bottom trawl fisheries on soft sediment biogeochemistry in the North Sea†. Geochemical Transactions, 2, 112. https://doi.org/10.1186/1467-4866-2-112.

Early, R.I., Duffy, J.P., Ashton, I.G.C, Maclean, I.M.D. and Laing, C.G. (2020). Modelling potential areas for Seagrass restoration within Plymouth Sound & Estuaries SAC and Solent Maritime SAC as part of the LIFE fund Recreation ReMEDIES project. UofE Consulting Ltd commissioned for Natural England. Natural England Commissioned Reports.

Eigaard, O., Bastardie, F., Breen, M., Dinesen, G.E., Hintzen, N.T., Laffargue, P., Mortensen, L.O., Nielsen, R.J., Nilsson, H.C., O'Neil, F.G., Polet, H., Reid, D.G., Sala, A., Sköld, M., Smith, C., Sørensen, T.K., Tully, O., Zengin, M. and Rijnsdrop, D.A. (2016). Estimating seabed pressure from demersal trawls, seines, and dredges based on gear design and dimensions. ICES Journal of Marine Science, 73(Supplement 1), i27–i43 https://doi.org/10.1093/icesjms/fsv099.

Emerson, S.R. and Archer, D. (1990). Calcium carbonate preservation in the ocean. Philosophical Transactions of the Royal Society of London, Series A, Mathematical and Physical Sciences, 331, 1616:29-40.

Fariñas-Franco, J.M., Pearce, B., Mair, J.M., Harries, D.B., MacPherson, R.C., Porter, J.S., Reimer, P. and Sanderson, W.G. (2018). Missing native oyster (*Ostrea edulis* L.) beds in a European Marine Protected Area: Should there be widespread restorative management? Biological Conservation, 221, 293-311.

Fodrie, F.J., Rodriguez, A.B., Gittman, R.K., Grabowski, J.H., Lindquist, N.L., Peterson, C.H., Piehler, M.F. and Ridge, J.T. (2017). Oyster reefs as carbon sources and sinks. Proceedings of the Royal Society of London, B, 284, 20170891.

Fonseca, M.S. (1994). A guide to transplanting seagrasses in the Gulf of Mexico. Texas A&M University Sea Grant College Program, TAMU-SG-94-601.

Ford, H., Garbutt, A., Duggan-Edwards, M., Pagès, J.F., Harvey, R., Ladd, C. and Skov, M.W. (2019). Large-scale predictions of salt-marsh carbon stock based on simple observations of plant community and soil type. Biogeosciences 16, 425–436.

Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M.A., Apostolaki, E.T., Kendrick, G.A., Krause-Jensen, D., McGlathery, K.J. and Serrano, O. (2012). Seagrass ecosystems as a globally significant carbon stock. Nature Geoscience, 5, 7: 505–509.

Gamble C., Debney, A., Glover, A., Bertelli, C., Green, B., Hendy, I., Lilley, R., Nuuttila, H., Potouroglou, M., Ragazzola, F., Unsworth, R. and Preston, J. (eds) (2021). Seagrass Restoration Handbook. Zoological Society of London, UK., London, UK.

Garrard, S.L. and Beaumont, N.J. (2014). The effect of ocean acidification on carbon storage and sequestration in seagrass beds; a global and UK context. Marine Pollution Bulletin, 86, 138–146. doi: 10.1016/j.marpolbul.2014.07.032.

Geoghegan, E.K., Caplan, J.S., Leech, F.N., Weber, P.E., Bauer, C.E. and Mozdzer, T.J. (2018). Nitrogen enrichment alters carbon fluxes in a New England salt marsh, Ecosystem Health and Sustainability, 4:11, 277-287, DOI: 10.1080/20964129.2018.1532772.

Green, A.E., Unsworth, R.K.F., Chadwick, M.A. and Jones, P.J.S. (2021). Historical Analysis Exposes Catastrophic Seagrass Loss for the United Kingdom. Frontiers in Plant Science, 12: 629962. doi: 10.3389/fpls.2021.629962.

Gregg, R., Elias, J.L., Morecroft, M.D., Alonso, I.E., Crosher, I.E. and Muto, P. (2021). Carbon Sequestration and storage by habitat: a review of the evidence (second edition) Natural England Research Report NERR094. Natural England, York.

Hale, R., Godbold, J.A., Sciberras, M., Dwight, J., Wood, C., Hiddink, J.G. and Solan, M. (2017). Mediation of macronutrients and carbon by post-disturbance shelf sea sediment communities. Biogeochemistry, 135**,** 121–133 https://doi.org/10.1007/s10533-017-0350-9.

Hudson, R., Kenworthy, J. and Best, M. (eds) (2021). Saltmarsh Restoration Handbook: UK and Ireland. Environment Agency, Bristol, UK.

Jackson, E.L., Griffiths, C.A. and Durkin, O. (2013). A guide to assessing and managing anthropogenic impact on marine angiosperm habitat - Part 1: Literature review. Natural England Commissioned Reports, Number 111.

Kennedy, H., Beggins, J., Duarte, C.M., Fourqurean, J.W., Holmer and M., Marba, Â.N. (2010). Seagrass sediments as a global carbon sink: Isotopic constraints. Global Biogeochemical Cycles, 24, 4: https://doi.org/10.1029/2010GB003848.

Krause-Jensen, D. and Duarte, C. (2016). Substantial role of macroalgae in marine carbon sequestration. Nature Geosciences, 9, 737–742 (2016). https://doi.org/10.1038/ngeo2790.

Krumhansl, K.A. and Scheibling, R.E. (2012). Production and fate of kelp detritus. Marine Ecology Progress Series, 467, 281-302. https://doi.org/10.3354/meps09940.

Legge, O., Johnson, M., Hicks, N., Jickells, T., Diesing, M., Aldridge, J., Andrews, J., Artioli, Y., Bakker, D.C.E., Burrows, M.T., Carr, N., Cripps, G., Felgate, S.L., Fernand, L., Greenwood, N., Hartman, S., Kröger, S., Lessin, G., Mahaffey, C., Mayor, D.J., Parker, R., Queirós, A.M., Shutler, J.D., Silva, T., Stahl, H., Tinker, J., Underwood, G.J.C., Van Der Molen, J., Wakelin, S., Weston, K. and Williamson, P. (2020). Carbon on the Northwest European Shelf: Contemporary Budget and Future Influences. Frontiers in Marine Science, 7:143. Doi: 10.3389/fmars.2020.00143.

Lima, M.D.A.C., Ward, R.D. and Joyce, C.B. (2020). Environmental drivers of sediment carbon storage in temperate seagrass meadows. Hydrobiologia, 847, 1773–1792. doi: 10.1007/s10750-019-04153-5.

Lovelock, C.E., Atwood, T., Baldock, J., Duarte, C.M., Hickey, S., Lavery, P.S., Masque, P., Macreadie, P.I., Ricart, A.M., Serrano, S. and Steven, S. (2017). Assessing the risk of carbon dioxide emissions from blue carbon ecosystems. Front Ecol Environ, 15(5): 257– 265, doi:10.1002/fee.1491.

Luff, A.L., Sheehan, E. V., Parry, M. and Higgs, N.D. (2019). A simple mooring modification reduces impacts on seagrass meadows. Scientific Reports, 9, 1: 1–10.

Luisetti, T., Turner, R.K., Andrews J.E., Jickells, T.D., Kröger, S., Diesing, M., Paltriguera, L., Johnson, M.T., Parker, E.R., Bakker, D.C.E. and Weston, K. (2019). Quantifying and valuing carbon flows and stores in coastal and shelf ecosystems in the UK. Ecosystem Services, 35, 67-76.

Maar, M., Bolding, K., Petersen, J.K., Hansen, J.L.S. and Timmermann, K. (2009). Local effects of blue mussels around turbine foundations in an ecosystem model of Nysted off-shore wind farm, Denmark. Journal of Sea Research, 2–3: 159-174.

Macreadie, P., Nielson, D.A., Kelleway, J.J., Atwood, T.B., Seymour, J.R., Petrou, K., Connolly, R.M., Tomson, A.C.G., Trevathan-Tackett, S.M. and Ralph, P.J. (2017). Can we manage coastal ecosystems to sequester more blue carbon? Frontiers in Ecology and the Environment, 15, 4:206–213, doi:10.1002/fee.1484.

McLeod, E., Chmura, G.L., Bouillon, S., Salm, R., Björk, M., Duarte, C.M., Lovelock, C.E., Schlesinger, W.H. and Silliman, B.R. (2011). A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO_2. Frontiers in Ecology and the Environment, 9(10): 552-560.

Milazzo, M., Badalamenti, F., National, I., Ceccherelli, G. and Chemello, R. (2004). Boat anchoring on Posidonia oceanica beds in a marine protected area (Italy, western Mediterranean): effect of anchor types in different anchoring stages. Journal of Experimental Marine Biology and Ecology, 299, 51–62.

MMO (2019). Identifying sites suitable for marine habitat restoration or creation. A report produced for the Marine Management Organisation by ABPmer and AER, MMO Project No: 1135, February 2019, 93pp.

Nelson, K. (2020). Sussex IFCA Fishing Effort Report 2015- 2019. Sussex Inshore Fisheries and Conservation Authority. https://secure.toolkitfiles.co.uk/clients/34087/sitedata/files/Research/SxIFCA-fishing-effort-2015-to-2019.pdf [Accessed December 2021].

Newton, C. and Thornber, C. (2013) Ecological Impacts of Macroalgal Blooms on Salt Marsh Communities. Estuaries and Coasts, 36, 365–376. https://doi.org/10.1007/s12237-012-9565-0.

Novak, A. B., Pelletier, M. C., Colarusso, P., Simpson, J., Gutierrez, M. N., Arias-Ortiz, A., Charpentier, M., Masque, P. and Vella, P. (2020). Factors influencing carbon stocks and accumulation rates in eelgrass meadows across New England, USA. *Estuaries and Coasts*, 1-16. doi: 10.1007/s12237-020-00754-9.

Orth, R.J., Carruthers, T.J.B., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., Short, F.T., Waycott, M. and Williams, S.L. (2006). A global crisis for seagrass ecosystems. Bioscience 56, 987–996.

Ouisse, V., Marchand-Jouravleff, I., Fiandrino, A., Feunteun, E. and Ysnel, F. (2020). Swinging boat moorings: Spatial heterogeneous damage to eelgrass beds in a tidal ecosystem. Estuarine, Coastal and Shelf Science, 235, 1-9.

Parker, R., Weston, K., Barry, J. and C. Powell (2012). Recommended biogeochemical targets and indicators for MSFD monitoring and management of the UK seabed. Contract report ME5301.

Parker, R., Benson, L., Graves, L., Kröger, S. and Vieira, R. (2021). Carbon stocks and accumulation analysis for Secretary of State (SoS) plan area: Cefas Project Report for Defra, 40 pp.

Passow, U. and Carlson, C.A (2012). The biological pump in a high CO_2 world. Marine Ecology Progress Series, 470, 249-272.

Pendleton, L., Donato, D., Murray, B., Crooks, S., Jenkins, W., Sifleet, S., Craft, C., Fourqurean, J., Kauffman, J., Marbà, N., Megonigal, P., Pidgeon, E., Herr, D., Gordon, D. and Baldera, A., (2012) Estimating Global "Blue Carbon" Emissions from Conversion and Degradation of Vegetated Coastal Ecosystems PLoS ONE 7(9): e43542 https://doi.org/10.1371/journal.pone.0043542.

Penk, M.R., Perrin, P.M. and Waldren, S. (2020). Above- to Belowground Vegetation Biomass Ratio in Temperate North-East Atlantic Saltmarshes Increases Strongly with Soil Nitrogen Gradient. Ecosystems 23, 648–661. https://doi.org/10.1007/s10021-019-00428-z.

Preston J., Gamble, C., Debney, A., Helmer, L., Hancock, B. and zu Ermgassen, P.S.E. (eds) (2020). European Native Oyster Habitat Restoration Handbook. The Zoological Society of London, UK., London, UK.

Pusceddu A., Bianchelli S., Martín J., Puig P., Palanques A., Masqué P. and Danovaro R. (2014). Chronic and intensive bottom trawling impairs deep-sea biodiversity and ecosystem functioning. Proceedings of the National Academy of Sciences of the United States of America, 17; 111(24): 8861-8866. doi: 10.1073/pnas.1405454111.

Queirós, A.M., Stephens, N., Widdicombe, S., Tait, K., McCoy, S.J., Ingels, J., Rühl, S., Airs, R., Beesley, A., Carnovale, G., Cazenave, P., Dashfield, S., Hua, E., Jones, M., Lindeque, P., McNeill, C.L., Nunes, J., Parry, H., Pascoe, C., Widdicombe, C., Smyth, T., Atkinson, A., Krause-Jensen, D. and Somerfield, P.J. (2019). Connected macroalgal-sediment systems: blue carbon and food webs in the deep coastal ocean. Ecological Monographs, 89, 1–21.

Regnier, P., Friedlingstein, P., Cias, P. and Mackenzie, F.T. (2013). Anthropogenic perturbation of the carbon fluxes from land to ocean. Nature Geoscience, 6, 597-607.

Röhr, M.E., Holmer, M., Baum, J.K., Björk, M., Chin, D., Chalifour, L., Cimon, S., Cusson, M., Dahl, M., Deyanova, D., Duffy, J.E., Eklöf, J.S., Geyer, J.K., Griffin, J.N., Gullström, M., Hereu, C.M., Hori, M., Hovel, K.A., Hughes, A.R., Jorgensen, P., Kiriakopolos, S., Moksnes, P.O., Nakaoka, M., O'Connor, M.I., Peterson, B., Reiss, K., Reynolds, P.L., Rossi, F., Ruesink, J., Santos, R., Stachowicz, J.J., Tomas, F., Lee, K.S., Unsworth, R.K.F. and Boström, C. (2018). Blue Carbon Storage Capacity of Temperate Eelgrass (Zostera marina) Meadows. Global Biogeochemical Cycles, 32, 1457–1475.

Sala, E., Mayorga, J., Bradley, D., Cabral, R.B., Atwood, T.B., Auber, A., Cheung, W., Costello, C., Ferretti, F., Friedlander, A.M., Gaines, S.D., Garilao, C., Goodell, W., Halpern, B.S., Hinson, A., Kaschner, K., Kesner-Reyes, K., Leprieur, F., McGowan, J., Morgan, L.E., Mouillot, D., Palacios-Abrantes, J., Possingham, H.P., Rechberger, K.D., Worm, B. and Lubchenco, J. (2021). Protecting the global ocean for biodiversity, food and climate. Nature 592, 397–402. https://doi.org/10.1038/s41586-021-03371-z

Serrano, O., Lavery, P., Kendrick, G.A., Hickey, S., Masqué, P., Arias-Ortiz, A., Steven, A., Duarte, C.M. (2016). Impact of mooring activities on carbon stocks in seagrass meadows. Scientific Reports, 6, 23193.

Silburn, B., Parker, R., Mason, C. and Parker, R. (2018). Rapid fines assessment: A quantitative volumetric method to assess fines content in marine soft sediments. Limnology and Oceanography: Methods, 16, 376-389. Doi: 10.1002/lom3.10252.

Smale, D., Burrows, M., Moore, P., O'Connor, N. and Hawkins, S. (2013). Threats and knowledge gaps for ecosystem services provided by kelp forests: A northeast Atlantic perspective. Ecology and Evolution, 3.

Smale, D.A., Burrows, M.T., Evans, A.J., King, N., Sayer, M.D.J, Yunnie, A.L.E. and Moore, P.J. (2016). Linking environmental variables with regional scale variability in ecological structure and standing stock of carbon within UK kelp forests. Marine Ecology Progress Series, 542, 79–95.

Smeaton, C., Hunt, C.A., Turrell, W.R. and Austin, W.E.N. (2021). Marine sedimentary carbon stocks of the United Kingdom Exclusive Economic Zone. Frontiers in Earth Science, 9, 1-21. Doi:10.3389/feart.2021.593324

Sussex Inshore Fisheries and Conservation Authority (2019). Marine and Coastal Access Act 2009 (c.23) Nearshore Trawling Byelaw. Nearshore-Trawling-Byelaw.pdf (toolkitfiles.co.uk) [Accessed December 2021].

Thomson, K., Miller, K., Johnston, P. and Santillo, D. (2017). Storage of carbon by marine ecosystems and their contribution to climate change mitigation. Greenpeace Laboratories Technical Report.

Thornton, D.C.O., Dong, L.F., Underwood, G.J.C. and Nedwell, D.B. (2002). Factors affecting microphytobenthic biomass, species composition and production in the Colne Estuary (UK). Aquatic Microbial Ecology, 27, 285–300.

Unsworth, R.K.F., Williams, B., Jones, B.L. and Cullen-Unsworth, L.C. (2017). Rocking the boat: Damage to eelgrass by swinging boat moorings. Frontiers in Plant Science, 8, 1309.

Unsworth, R.K.F., Mtwana Nordlund, L. and Cullen-Unsworth, L.C. (2018). Seagrass meadows support global fisheries. Conservation Letters, 12566.

Van de Velde, S., Van Lancker, V., Hidalgo-Martinez, S., Berelson, W.M. and Meysman, F.J.R. (2018). Anthropogenic disturbance keeps the coastal seafloor biogeochemistry in a transient state. Scientific reports, 8, 5582, DOI:10.1038/s41598-018-23925-y.

Vitousek, P.M., Aber, J.D., Howarth, R.W., Likens, G.E., Matson, P.A., Schindler, D.W., Schlesinger, W.H. and Tilman, D.G (1997). Human alteration of the global nitrogen cycle: sources and consequences. Ecological Applications, 7, 737-750.

Wasson, K., Jeppesen, R., Endris, C., Perry, D.C., Woolfolk, A., Beheshti, K., Rodriguez, M., Eby, R., Watson, E.B., Rahman, F., Haskins, J. and Hughes, B.B. (2017). Eutrophication decreases salt marsh resilience through proliferation of algal mats. Biological conservation, 212, 1-11. https://doi.org/10.1016/j.biocon.2017.05.019.

Appendix 1: Blue Carbon mapping – technical notes and data analysis methodologies

Introduction

This technical appendix covers the mapping and data analysis carried out to support the Natural England Blue Carbon report. It provides an overview of the methodologies used, data sources, caveats and limitations and should be read and considered alongside the main report when viewing the tables and figures. Each section of the main report has been addressed below in turn.

Given the nature of the report being a proof of concept, mapping and data analysis was carried out using data layers already available for use, from a variety of sources. The collection of new data was outside the scope of the project, but where data gaps exist or further information would have been useful (or maybe useful in the future) this has been highlighted.

Data sources

A variety of data layers were used throughout the mapping and analysis section of the report. These are outlined in the below table and then referred to in the methodology. Note data collation/download happened in January 2021 so datasets outlined below may have been updated since.

Dataset	Source	Notes
Natural England Marine evidence base	Various	A collation of datasets to provide a comprehensive collection of marine habitat and species information for English waters.
Delineation of English Waters	Various	The polygon delineating English waters in the figures was used to clip habitat maps and generate area figures. Created using a mixture of OS boundary-line for MHW, and various maritime limits from the UKHO Marine data portal. https://datahub.admiralty.co.uk/portal/apps/sites/#/marine-data-portal
National saltmarsh map	Environment Agency	Available to download from https://data.gov.uk/dataset/0e9982d3-1fef-47de-9af0-4b1398330d88/saltmarsh-extent-zonation
Potential seagrass layer	Environment Agency	Available to download from https://data.gov.uk/dataset/5b943c08-288f-4d47-a924-a51adda6d288/seagrass-potential
Potential Oyster layer	Environment Agency	Available to download from https://data.gov.uk/dataset/31530300-0f98-42ac-9b68-b6c980f5383c/native-oyster-bed-potential

Potential saltmarsh and mudflat	Marine Management Organisation	Available to download from https://environment.data.gov.uk/DefraDataDownload/?mapService=MMO/MMO1135PotentialHabitatCreationSites&Mode=spatial
Marine Plan Areas	Marine Management Organisation	Available to download from https://environment.data.gov.uk/DefraDataDownload/?mapService=MMO/MarinePlanAreas&Mode=spatial
Water Framework Directive Estuarine and Coastal Waterbodies	Environment Agency	Available from: https://data.gov.uk/dataset/3a75ec5f-a361-475c-80e3-52d93bbc5dbe/wfd-transitional-and-coastal-waterbodies-cycle-2
Water Framework Directive Dissolved Organic Nitrate (DIN)	Environment Agency Water Quality Archive	https://environment.data.gov.uk/water-quality/view/landing
Opportunistic macroalgae datasets	Environment Agency	Obtained from Environment Agency 2021
Sussex IFCA trawling	Sussex IFCA	Report at: https://secure.toolkitfiles.co.uk/clients/34087/sitedata/files/Research/SxIFCA-fishing-effort-2015-to-2019.pdf Data provided directly via email.
Transport and routes	Marine Management Organisation	Anchoring point and polygon dataset provided from MMO data services. Listed on MMO master data register as URI 100063 (polygon) and URI 100065 (point). https://assets.publishing.service.gov.uk/government/uploads/system/uploads/attachment_data/file/503696/master-data-register.csv/preview
Natural England Marine Evidence Base – Collated Activity Layers	Various UK Hydrographic Office	Filter applied for Anchoring and Mooring/Warping. UKHO Data © British Crown Copyright. All rights reserved. Permission Number Defra012018.001. This product has been derived in part from material obtained from the UK Hydrographic Office with the permission of the Controller of Her Majesty's Stationery Office and UK Hydrographic Office (www.ukho.gov.uk). NOT TO BE USED FOR NAVIGATION
Marinas	Royal Yachting Association	UK Coastal Atlas of Recreational Boating 2.1 reproduced under licence from the Royal Yachting Association
Wind cable agreements 2020	The Crown Estate	Available to download from https://opendata-thecrownestate.opendata.arcgis.com/datasets/thecrownestat

		e::offshore-wind-cable-agreements-england-wales-ni-the-crown-estate/data
Wind site agreements 2021	The Crown Estate	Available to download from https://opendata-thecrownestate.opendata.arcgis.com/datasets/thecrownestate::offshore-wind-site-agreements-england-wales-ni-the-crown-estate/data
Offshore wind round 4 preferred projects	The Crown Estate	Available to download from https://opendata-thecrownestate.opendata.arcgis.com/datasets/thecrownestate::offshore-wind-leasing-round-4-preferred-projects-england-wales-and-ni-the-crown-estate-1
Anthropogenic hard protection layers		MBIEG (2020). Mapping Anthropogenic Hard Protection in the Marine Environment. A report produced by Intertek Energy and Water for Defra on behalf of the Marine Biodiversity Impacts Evidence Group. *Note: 2 versions of this dataset exist. The Open versions of the data were used for map presentation, and non-open data was used for calculations to provide more accurate figures.*

General mapping principles

- The mapping analysis was carried out using Geographical Information System (GIS) ArcMap 10.2.2 software with data management in file geodatabases.
- All calculations of area figures were carried out in the European Terrestrial Reference System 89 (ETRS89 LAEA) coordinate system and are usually given in Km^2 but units should be stated.
- All maps displayed throughout the report are in the British National Grid projection.

Study Area

The project Scope was English Inshore and Offshore waters, the boundary of which is shown as 'English Waters' on the report figures. These were delineated using the English continental shelf limit, and various boundaries between English and adjacent waters as published on the Admiralty data portal by the UKHO Law of the Sea unit. This boundary was then used to clip the blue carbon habitats to the study area within the GIS.

The landward boundary where required used the OS Boundary-Line MHW line and national land boundaries. However for the majority of the study this was not needed, in order to avoid removing any saltmarsh habitat above MHW.

Initial mapping of Blue carbon habitat boundaries

The first stage of the mapping analysis was to generate data layers for each Blue Carbon (BC) habitat. The foundation for this was the Natural England (NE) marine evidence base (updated in November 2020) which contains all the point and polygon data available to Natural England and is mapped in spatial format for viewing in a Geographical Information System (GIS). This evidence base consists of data which Natural England has gathered through marine survey work, datasets which have been shared with NE by partner organisations, and third-party data. These various data sources are then combined by NE to create a habitat map using the most up to date evidence available, which is updated every 6 months and used for habitat mapping and data analysis. Data within the NE evidence base is attributed with habitat information using the European Union Nature Information System (EUNIS[13]) and these codes were then used to subsequent analysis as detailed below.

The BC habitats within scope for the work were identified and data layers (file geodatabase feature classes) created for each one based on the following criteria:

BC Habitat	Habitat data used	EUNIS code used	Data notes
Saltmarsh	Saltmarsh	A2.5	Data set used: Input_BSH_Polys_WGS84_Internal. Includes saline reedbeds – data does not exist at the appropriate resolution to separate reedbeds out on a national scale. Obtained from the Environment Agency and incorporated into the NE evidence base.
Intertidal (sand) sediments	Littoral sand and muddy sand	A2.2	Data set used: Input_BSH_Polys_WGS84_Internal.
Intertidal (mud) sediments	Littoral mud	A2.3	Data set used: Input_BSH_Polys_WGS84_Internal.
Subtidal (sand) sediments	Sublittoral Sand	A5.2	Data set used: Input_BSH_Polys_WGS84_Internal.
Subtidal (mud) sediments	Sublittoral mud	A5.3	Data set used: Input_BSH_Polys_WGS84_Internal.
Seagrass	Seagrass bed polygons	A2.61 and A5.53 (HOCI_17)	Data set used: Input_HOCI_Polys_WGS84_Internal. Combined littoral and sublittoral seagrass meadows. The seagrass data layer was checked and verified in collaboration with EA habitat and blue carbon specialists to reach an agreed extent and distribution of current seagrass beds in English waters.
European flat oyster	Oyster beds habitat map	A5.4 (HOCI_14)	Data set used: Input_HOCI_Polys_WGS84_Internal

[13] https://www.eea.europa.eu/data-and-maps/data/eunis-habitat-classification [Accessed March 2021]

Kelp habitat	Specific kelp biotopes (see list)	Various	Data set used: Input_BSH_Polys_WGS84_Internal. Only where specific kelp biotopes present and excluding infralittoral rock where kelp isn't specifically stated to occur.

Blue Carbon habitat caveats

Sediment

Following discussions between NE and CEFAS, it was agreed for this study that only the following sediment habitats were to be mapped and to be treated separately, disregarding coarse and mixed sediments:

Littoral sand and muddy sand	**A2.2**
Littoral mud	**A2.3**
Sublittoral Sand	**A5.2**
Sublittoral mud	**A5.3**

Refer to main report section 1.3 for caveat and explanation around matching BC sediment habitats to the EUNIS habitat classification.

Kelp

Only records with a specific biotope code assigned for kelp occurrence on rock have been included. There are other records in the NE evidence base that could be assumed to be suitable kelp habitat or have kelp growing (such as certain infralittoral rock biotopes), but not enough information was available to subdivide the biotope classification down to a kelp level. The decision was made to only map definite kelp biotopes and remove other infralittoral rock biotopes. This is one uncertainty surrounding kelp. Another source of uncertainty is that specific kelp biotopes have likely only been mapped in areas around the coast where comprehensive intertidal survey or shallow subtidal survey has taken place. Therefore, there is almost certainly an underestimate of kelp habitat area and it is recommended that future work would look at other evidence sources / areas of research into kelp extent.

The kelp biotopes included in this study are in the table below:

JNCC_0405 Biotope Code	EUNIS_0711_L6 Biotope Code	JNCC_0405 Biotope Code	EUNIS_0711_L6 Biotope Code
IR.HIR.KFaR.Ala.Myt	A3.1111	IR.MIR.KR.LhypTX.Pk	A3.2132
IR.HIR.KFaR.Ala.Ldig	A3.1112	IR.MIR.KR.Lhyp	A3.214
IR.HIR.KFaR.AlaAnCrSp	A3.112	IR.MIR.KR.Lhyp.Ft	A3.2141
IR.HIR.KFaR.LhypFa	A3.113	IR.MIR.KR.Lhyp.Pk	A3.2142
IR.HIR.KFaR.LhypPar	A3.114	IR.MIR.KR.Lhyp.GzFt	A3.2143
IR.HIR.KFaR.LhypR	A3.115	IR.MIR.KR.Lhyp.GzPk	A3.2144
IR.HIR.KFaR.LhypR.Ft	A3.1151	IR.MIR.KR.Lhyp.Sab	A3.2145
IR.HIR.KFaR.LhypR.Pk	A3.1152	IR.MIR.KR.LhypVt	A3.216

IR.HIR.KFaR.LhypR.Loch	A3.1153	IR.MIR.KT.LdigT	A3.221
IR.HIR.KFaR.FoR	A3.116	IR.MIR.KT.XKT	A3.222
IR.HIR.KFaR.FoR.Dic	A3.1161	IR.MIR.KT.XKTX	A3.223
IR.HIR.KFaR.LhypRVt	A3.117	IR.MIR.KT.SlatT	A3.224
IR.HIR.KSed.Sac	A3.121	IR.LIR.K.LhypLoch	A3.311
IR.HIR.KSed.SlatSac	A3.122	IR.LIR.K.LhypSlat	A3.312
IR.HIR.KSed.SlatChoR	A3.123	IR.LIR.K.LhypSlat.Ft	A3.3121
IR.HIR.KSed.DesFilR	A3.124	IR.LIR.K.LhypSlat.Pk	A3.3122
IR.HIR.KSed.XKScrR	A3.125	IR.LIR.K.LhypSlat.Gz	A3.3123
IR.MIR.KR.Ldig	A3.211	IR.LIR.K.Slat.Ldig	A3.3131
IR.MIR.KR.Ldig.Ldig	A3.2111	IR.LIR.K.Slat.Ft	A3.3132
IR.MIR.KR.Ldig.Bo	A3.2112	IR.LIR.K.Slat.Pk	A3.3133
IR.MIR.KR.Ldig.Pid	A3.2113	IR.LIR.K.Slat.Gz	A3.3134
IR.MIR.KR.LhypT	A3.212	IR.LIR.K.LhypCape	A3.314
IR.MIR.KR.LhypT.Ft	A3.2121	IR.LIR.K.Sar	A3.315
IR.MIR.KR.LhypT.Pk	A3.2122	IR.LIR.KVS.Cod	A3.321
IR.MIR.KR.LhypTX	A3.213	IR.LIR.KVS.SlatPsaVS	A3.322
IR.MIR.KR.LhypTX.Ft	A3.2131	IR.LIR.KVS.SlatPhyVS	A3.323

Seagrass beds

The nature of the NE Marine Evidence base means that overlapping polygons can exist, where a Habitat of Conservation Importance (HOCI) occurs in the same location as a Broadscale Habitat (BSH). For example, a seagrass bed in the HOCI polygonal data layer can sit over the top of a sublittoral sediment BSH polygon.

For this study the seagrass extent was considered distinct from the intertidal and subtidal sediment layers. So, for instances where the seagrass occurred on top of these layers, we ensured that no double counting of polygon area occurred.

Natural England and the Environment Agency were already collaborating to undertake an audit of seagrass evidence in England (ongoing at time of writing). So to calculate and agree an area figure for seagrass for this report, an expert judgement approach was used to identify and include all seagrass polygons considered to contribute to the current area, while excluding historical or lower confidence data.

European flat oyster

There are no specific UK carbon accumulation rates available for oyster beds, but they were included in the mapping analysis as a proof of concept. Data for both current and potential oyster beds extent were available. Oysters are not displayed on the map figures in the report due to their presence on the NBN sensitive species list[14]. Oysters are a protected species due to commercial importance and therefore exact spatial information is not publicly released.

Blue Carbon extent calculations methodology

This methodology resulted in the initial BC habitat layers which were to be used for the rest of the analysis, as well as generating the area figures in table 5.

1. The input Broad Scale Habitats (BSH) polygons and Habitats of Conservation Importance (HOCI) polygons from the Marine Evidence geodatabase were clipped to the study area using an English territorial waters polygon.
2. All data was re-projected from WGS84 to ETRS89 to enable area calculations
3. The BC habitats from the clipped data were extracted based on the principles detailed above, and each habitat saved as new individual feature class.
   Some habitats were a direct export by filtering/selecting data based on the EUNIS_L3 code or FOCI_code in the attributes. For kelp, biotopes were selected based on joining a list of biotopes (identified in the table above) to the HAB_TYPE field.
4. Any duplicated sediment polygons also tagged for seagrass beds (HOCI_17) were removed.
5. The clip and erase tools were used to create feature classes for each BC habitat inside and outside the Marine Protected Area (MPA) boundaries. The MPA layer included all inshore and offshore sites within English waters. Cross border sites (e.g. with Wales or Scotland) were clipped to just include the English portion
6. The ‘summarize’ function was used for each of the feature class datasets created to obtain the total area figures for BC habitat extent.

Mapping potential extent of BC habitats

This section of the analysis focussed on Seagrass bed and Oyster bed habitats, as maps of habitat potential for these are available from the Environment Agency. Reference should be made to the dataset summaries and supporting documentation on data.gov.uk (see links in ‘data sources’ above) when considering the caveats and limitations regarding these layers, which are only briefly summarised below. Note it is clearly stated that these layers are only designed to provide national ‘high level indications’. It is outside the scope of this project to evaluate the pros and cons of the layers here, and they were used ‘as available’.

Seagrass

To allow analysis of future potential impacts on blue carbon habitats, potential habitat layers were used where available. The potential seagrass layer available from the Environment Agency was used which was derived using wave and current energy, elevation and salinity criteria. The criteria were based on the

---

[14] https://docs.nbnatlas.org/sensitive-species-list/ [Accessed December 2021]

preferences for the seagrasses *Zostera marina* and *Z. noltei as identified by the Marine Life Information Network (MarLIN).*

Main caveats of dataset:

- The layer is based on large scale models so may not be accurate at a local scale.
- Activities affecting potential were not factored into the layer (e.g. dredging, cables).
- Is only an initial attempt to identify areas where seagrass could be re-established at a national level, not all areas may meet the environmental conditions needed.
- There may be areas outside of the layer which are suitable which weren't selected by the modelling criteria.

Oyster

There were only 11 records available from 4 surveys for existing oyster bed habitat. The oldest data was from a 1985 – 1996 survey and it was therefore decided to use all 11 oyster records as the current best available evidence, with no historical data available.

The Environment Agency have also made available a layer for potential native oyster beds. This data layer is derived from seabed sediment and current energy criteria listed by MarLIN.

Caveats of dataset:

- Same as for potential seagrass layer

Saltmarsh and mudflat

For analysis of potential saltmarsh and mudflat habitat, a potential saltmarsh and mudflat layer created by the Marine Management Organisation for project MMO1135 was used. The layer considered managed realignment and regulated tidal exchange within the current floodplain when creating the layer.

Caveats:

- Saltmarsh and mudflat considered together
- May not be accurate at local scale and local knowledge would be needed to confirm site suitability.
- The dataset should be considered in its infancy – more trials, research and monitoring is needed to improve the accuracy of the dataset.

Potential habitat extent calculations methodology

Each potential habitat layer was first clipped to English waters prior to analysis. Extent values were then generated by adding an extent field within the attribute table which was populated using the calculate geometry function, selecting the area to be calculated in Km^2. The total extent was then obtained using the statistics function which provided a total extent value.

Carbon risk heat mapping

This section of analysis considered the spatial interactions between existing carbon stores (blue carbon habitat maps), potential carbon stores (blue carbon potential habitat maps) and various spatial datalayers available. It should be noted that these may not be the only factors deemed to be potentially having an impact on blue carbon stores but a reflection of the datalayers which were easily accessible to use as case studies for this project.

DIN and Macroalgae input layers

The Environment Agency (EA) provided a dataset for dissolved inorganic nitrogen (DIN) and opportunistic macroalgae status per EA water body for 2019 (the most recent data available) and a spatial dataset for the waterbodies within the Water Framework Directive. DIN and opportunistic macroalgae status was recorded on a scale of High, Good, Moderate or 'No Data' with Moderate status as least favourable.

To create a spatial data layer for DIN status and for opportunistic macroalgae status, the Water Framework Directive layer was projected to the geographical coordinate system ETRS 1989 to allow later extent calculations and was clipped to English waters. The status data was then joined to the clipped Water Framework Directive layer to create a geospatial layer for DIN status and a layer for opportunistic macroalgae status using the 'Add Join' tool.

Caveats:

- Lots of areas with no data available, particularly for opportunistic macroalgae in the SW

Current and potential BC habitats and water quality

To highlight where eutrophication could affect existing or potential new blue carbon habitats, each of the habitat layers (for potential, just saltmarsh & mudflat, seagrass and oysters were used) were overlaid with the DIN and opportunistic macroalgae layers. To quantify the amount of habitat affected, the extent of each habitat layer within each of the statuses (High, Good and Moderate) was calculated at a national scale for all English waters and was then further broken down by marine plan area.

To calculate the extent of each habitat layer within each water quality status for all English waters, a definition query was applied to the DIN/macroalgae layer to select the status (e.g. Moderate). The habitat layer was then clipped to the status layer to create a habitat layer for just the selected status. Extent values were then calculated based on this new clipped layer by adding an extent field within the attribute table which was populated using the calculate geometry function, selecting the area to be calculated in Km^2. The total extent was then obtained using the statistics function which provided a total extent value. Once an extent value for the habitat was generated for each water quality status at a national scale, the marine plan areas layer was used to generate extent values per plan area using the same method.

Analysis of habitat sensitivity to pressures

Extending the heat mapping further to overlay with data on activities which exert pressures which could result in carbon release or reduced carbon accumulation. Again, the caveat here is that this is a proof of concept using available data sources. It should not be inferred that these are the only activities exerting pressures on BC habitats, nor that by overlaying activity data with BC habitats in this study implies that all the activities are detrimental to the BC habitat at all locations where an overlap is identified.

Sussex IFCA trawling data proof of concept

For this proof of concept analysis a 1x1km grid of trawling effort within the Sussex IFCA (SxIFCA) district was applied to the blue carbon habitat extent data (created earlier in this project) as an absolute overlay. The fishing effort grid was created by SxIFCA by combining sightings data and patrol effort into a grid form, as outlined in Nelson (2020).

The 'combined trawling effort grid' supplied by the IFCA and used for the overlay is not illustrated in the SxIFCA report, but is a combination of both the single and pair trawling effort grids as both methods can potentially impact the seabed.

GI method:

1. Calculate areas of each BC habitat and the area of potential seagrass (from the EA dataset) within the SxIFCA district.
2. Overlay grid of combined trawling effort and calculate area per habitat (and potential seagrass) which overlaps with any grid squares where fishing effort >0.

Caveats

- The fishing effort grid has been generated using a combination of both fishery activity sightings, and patrol effort data per 1km^2 grid cell and therefore an element of modelling has been used to transfer the raw data into a grid form.
- Any overlap between the trawling data and the blue carbon habitats highlights where fishing pressure could be considered to be having a potential impact on BC habitat.
- An overlap only highlights the fact that a BC habitat occurs in the same grid cell that trawling effort occurs. It does not imply that there is a direct spatial interaction between the habitat and the activity, as it is not possible to identify where within the 1km cell the trawling occurred.
- The method used here is purely an overlay, and did not take into account any trawling 'intensity' value information supplied. Therefore, each 1km square is either trawled (effort >0) or it isn't.
- The sensitivity of each BC habitat, the impact from different types of fishing methods or number of passes for example was also not considered in the analysis.

Anchoring and Mooring data analysis

Like other 'proof of concept' sections of this BC analysis into the potential impact of pressures on BC habitats, the original intention was to generate a very broad-brush area value for potential impacts of anchoring and mooring on BC habitats. However, following a review of the available literature detailing how the area of seabed surrounding anchors and moorings can be impacted, it was apparent that there are many different variables and factors which can influence the shape and area of seabed potentially impacted by these. Other more comprehensive studies have looked into this specific area of research, and applying a significantly rigorous analysis of the existing data and information available was beyond the scope of this project. Instead of generating area values, a straightforward overlay exercise was carried out instead.

GI method:

The layers for anchoring and mooring were overlaid with the blue carbon habitats to provide a visualisation of areas which could be potentially impacted by commercial anchoring and mooring. Marinas data from the RYA was overlaid to provide a visualisation of areas which could potentially be impacted by recrecreational anchoring and mooring.

The 'Transport and Routes' point and polygon datasets were filtered for anchoring data by applying a definition query for data where the Feature was 'Anchor berth' or 'anchorage area'. Natural England's Activity layer was also filtered for data where the activity name was 'anchorage area', 'anchor berth', 'anchorage' or 'mooring/warping facility'. The 'Marinas' layer was filtered for non-inland marinas.

Caveats:

- Anchorage areas (shown as polygons) does not necessarily mean damage is being caused as anchors may not be dropped everywhere inside the area.
- A mooring area may not further damage the seabed once the pontoon or mooring structure for example is in place.

Available literature:

A brief summary of the literature considered and recommended as a starting point for possible future work is given here:

- Griffiths, C.A., Langmead, O.A., Readman, J.A.J., Tillin, H.M. (2016). Anchoring and mooring impacts in English and Welsh marine protected areas: Reviewing sensitivity, activity, risk and management. A report to Defra Impacts Evidence Group. *A comprehensive report considering activity footprint, exposure and sensitivity information. Also collated data on scale, frequency and intensity of anchoring and mooring within English and Welsh MPAs. This would be a good place to start in the future, maybe extending the data analysis outside of MPAs to focus on BC habitats.*

Additional reports considered and potential starting points for future work:

- Lee, J. 2018. Recreational anchoring and mooring in Marine Protected Areas (MPAS): Activity data collection, Defra. *A complementary study to Griffiths et al. (2017) collating additional activity data and reviewing / updating the assessment of levels of risk within MPAs*.

- Jackson, E.L., Griffiths, C.A., Collins, K., Durkin, O., McNie, F. (2018). A Guide to assessing and managing anthropogenic impact on marine angiosperm habitat. PART 2: A review of natural and anthropogenic pressures in Studland Bay. Natural England and MMO, Peterborough, UK. *Currently unpublished*. *Includes an area assessment of scars in one specific location.*

- Milazzo, M., Badalamenti, F., Ceccherelli, G., Chemello, R. (2004). Boat anchoring on *Posidonia oceanica* beds in a marine protected area (Italy, western Mediterranean): effect of anchor types in different anchoring stages. Journal of experimental marine biology and ecology, 299, 51-62. *An assessment of different anchor types in a single Italian MPA. Shows level of damage can vary depending on equipment used.*

- Amec Foster Wheeler Environment and Infrastructure UK Ltd (2017) Potential for eco-moorings as management option for Marine Protected Areas (MPAs). Report for CEFAS. *A review and assessment of eco-moorings as a management option for MPAs to avoid or limit physical pressures on marine habitats caused by traditional anchors or moorings.*

Offshore installations analysis

To highlight the overlap of blue carbon habitats with offshore wind farm and cable installations, an extent estimate for wind turbine footprints and cable footprints was calculated. The method used Crown Estate data and data available in the Natural England marine evidence base for anthropogenic hard protection (AHP). These datasets were then overlain with the blue carbon habitat layers to give a visual impression of the extent of active and potential future offshore wind development.

Extent calculations methodology:

1. The extent of the turbine footprints was estimated by generating a ratio using the available data and scaling this up for all active wind site agreements. An extent value was first calculated for turbine footprints where available in the AHP dataset and a second extent value was calculated for the active wind site agreement areas the turbines were located within. These two values were then used to generate a ratio (ratio = turbine footprint extent / wind site area extent).
2. The extent of all active offshore wind sites was then calculated and multiplied by the ratio to generate an extent value estimate of turbine footprints in all active wind site agreement areas.
3. A value for wind cable impact extent was calculated using the available AHP data for wind cables.

Caveats:

- The calculation of turbine footprint extent over the active windfarm sites is an estimation based on the ratio method outlined using limited data.
- AHP data is only available for some developments so is a large underestimate, especially for cables. Other caveats associated with the AHP dataset include data protection rules, narrow scope of data collation projects, data is not yet published, difficulties associated with obtaining the data and sources that aren't available publically. The final report[15] should be consulted for a full assessment of the data gathering methodology and associated caveats/limitations. The non-open version of the dataset was used for the extent calculations however an open version of the dataset was used for visualisation in the maps.
- The turbine footprint extent is an estimate but allows a more accurate extent value for actual area impacted than by using the outer boundary of the wind site agreement areas.

[15] http://sciencesearch.defra.gov.uk/Default.aspx?Module=More&Location=None&ProjectID=20442 [Accessed December 2021]

- The extent of impact due to cable installation is only calculated on the available information. This information may just provide a polygon indicating a cable corridor / route and this area will have been used for the extent calculation. No further analysis of information (if/where available) has been carried out on the actual footprint of the cable installation within the permitted corridor. The impact would also be affected by the method of cable installation, for example whether ploughed through a habitat, or tunnelled underneath. Therefore figures calculated here should be treated with caution as a proof of concept, and further in-depth analysis of available data aas well as uncertainties / data gaps is recommended.
- The impact of indirect pressures on BC habitats associated with offshore wind and energy installations is not considered, for example maintenance boat wash impacting seagrass beds.

Appendix 2: Modelling the sensitivity of blue carbon habitats to pressures from human activity

Natural England is developing a tool to present sensitivity of benthic habitats to anthropogenic induced pressures. The Natural England Sensitivity Tool (NEST) currently presents the sensitivity of benthic biotopes to pressures. However, investigations are underway to explore the possibility of using NEST to present sensitivity of other key ecosystem functions such as blue carbon stores and ecosystem services. This appendix presents part of this investigation, and the scale of the knowledge gap from habitats that can be linked to current sensitivity assessments in relation to those that will require further study.

Introduction to the benthic sensitivity model

The MarESA sensitivity assessments use an evidence-based framework to demonstrate the resistance (tolerance) and resilience (recoverability) of marine biotopes to human induced pressures. These biotopes are described at EUNIS levels 4-6 and classify substrate, key structural, functional and characterising species to an increasing level of detail in each EUNIS level, with 6 being the most detailed (Tyler-Walters *et al.,* 2018). The sensitivity of a biotope is defined using a categorical scoring approach to a benchmarked intensity of 39 standardised pressures, as defined by OSPAR (OSPAR, 2011). An activity in the marine environment (such as demersal trawling) can elicit several pressures on a biotope (e.g. abrasion, penetration). This sensitivity information and the relevance of an activity/pressure/biotope combination underpin Natural England's conservation advice database.

Previous work undertaken to address the impacts of fishing effort displacement throughout English waters has developed a model to link the sensitivity database with the NE and JNCC combined benthic habitat map (Haupt and Vaughan unpublished). Where benthic habitat data is of insufficient detail to assign a sensitivity assessment, a proxy assessment is assigned following the EUNIS hierarchy and applying the precautionary principle. This allows for the spatial presentation of the sensitivity of biotopes (below the MHWL) to pressures resulting from activities in the marine environment and aims to inform spatially explicit conservation advice. Both the sensitivity assessments and benthic habitat map are updated on a bi-annual cycle and represent the latest available evidence in marine mapping and biotope sensitivity.

As understanding the potential for marine activities to influence carbon accumulation and storage is of paramount importance, the potential for the further development of the benthic sensitivity model to spatially present sensitivity of blue carbon habitats to pressures was investigated.

Limitations of benthic sensitivity model in relation to ecosystem services

As stated above, the sensitivity assessments are based on the sensitivity of biotopes. At the broadscale habitat level used to assess blue carbon habitats in this report, there is a range of potential biotopes that may occur in within each habitat, which have a range of sensitivity ratings to marine pressures. Additionally, the sensitivity of the biotope may have been assessed on one or more components of the biotope relevant to the preservation of the biotope, but not necessarily relevant to carbon accumulation and/or carbon storage.

For example, EUNIS A5.361 Sea pens and burrowing megafauna in circalittoral fine mud are assessed as being highly sensitive to penetration or disturbance of the substratum subsurface. However, this assessment is based on the sensitivity of the epifaunal, suspension feeding, sea pens as significant reduction in the presence of sea pens would remove the biotope. This would represent a high risk to biodiversity loss, but not necessarily a high risk to the ecosystem functioning that would facilitate the accumulation and storage of carbon. This precludes the direct use of the benthic sensitivity model in its existing form.

The extension of sensitivity assessments to encompass ecosystem services, such as the climate regulating service of carbon accumulation, has been investigated within NE (Harvey-Fishenden and Vaughan, unpublished) and by JNCC to support ecosystem service modelling of offshore benthic habitats (Tillin *et al.,* 2020). In brief, JNCC have taken the approach of disaggregating species sensitivity from biotopes, examining the sensitivity of functional bio-assemblages to pressures, and modelling the effect of pressures on relevant ecosystem services that the functional bio-assemblages contribute to.

Although a less onerous task offshore due to the lower numbers of biotopes present, the concept of reassessing existing sensitivity assessments to relate to relevant ecosystem services has been demonstrated. The sensitivity of blue carbon habitats to human induced pressures is a rapidly developing field of study. By reviewing sensitivity assessments to relate to blue carbon accumulation and storage and presenting this spatially, Natural England along with colleagues and developers would have a framework with which to base advice on marine developments and fisheries management from the perspective of biodiversity (biotopes) and blue carbon.

Demonstration of approach to linking sensitivity assessments and blue carbon habitats

The reassessment of biotope sensitivity is a resource heavy activity and outside of the scope of the current project. However, to demonstrate the concept, a review of the sensitivity of seagrass biotopes as relevant to penetrating impacts of demersal trawling was undertaken.

Seagrass beds relevant to English waters are classified as A2.6111 : *Zostera noltii* beds in littoral muddy sand and A5.5331 : *Zostera marina/angustifolia* beds on lower shore or infralittoral clean or muddy sand. The sensitivity assessments for both biotopes are based on the sensitivity of the plant itself. As the carbon accumulation value of a seagrass bed is directly related to the capacity of the plant to photosynthesise and attenuate particle flow, a pressure that is rated as high sensitivity to the biotope will also be of high risk to carbon accumulation.

Penetration caused by demersal trawling has a high sensitivity rating to littoral and infralittoral seagrass beds, due to the action of this activity on the plant and the low resistance and resilience of the species to recover from this pressure. As such, the removal or damage of Seagrass beds by demersal trawling will impact on the carbon accumulation capacity of seagrass beds and also the stability of carbon stored in sediments as they will become more exposed to dispersal by wave action and nutrient cycling (Macreadie *et al* 2019, Gacia and Duarte 2001).

This proof of concept assessment would classify the carbon accumulation and storage capacity of seagrass beds as highly sensitive to penetration, although to formalise this assessment a rigorous assessment process with appropriate audit trail is required.

Future development of sensitivity assessments to understand risks to blue carbon and ecosystem services

Seagrass beds, although of high blue carbon value, represent a small area of blue carbon habitats in English waters. There is a strong need to examine the sensitivity of more prevalent blue carbon habitats to pressures to offer the best possible advice to mitigate the impacts of marine activities on blue carbon accumulation and storage. Taking the fishing pressure example used above, penetration or disturbance of the substratum subsurface due to demersal trawling, and presenting the biotopes underpinning existing sensitivity assessments in a conceptual figure (fig.1), the scale of the limitations in our current understanding can be discussed.

Figure 1 conceptually presents the blue carbon habitats that could potentially interact with demersal trawling, as defined in NE advice on operations database. Habitats are represented as boxes relative to their area present within English waters, below the MHWL. The EUNIS codes presented are biotopes extracted from the benthic sensitivity model that would require re-assessment in relation to blue carbon and relevant ecosystem services. The rudimentary assessment of high sensitivity for carbon accumulation and storage in seagrass beds represents 0.08% of blue carbon habitats identified in this project that may interact with demersal trawling activity. This demonstrates the urgent need to understand the sensitivity of benthic habitats in relation to carbon storage and ecosystem services. As only seagrass could be included in the proof of concept assessment, spatial representation was not meaningful at a national or regional level. However, the framework developed with the benthic habitat sensitivity model could be extended to incorporate blue carbon and ecosystem service sensitivity assessments, providing a spatially explicit visualisation of risks to the seabed to support marine spatial planning advice.

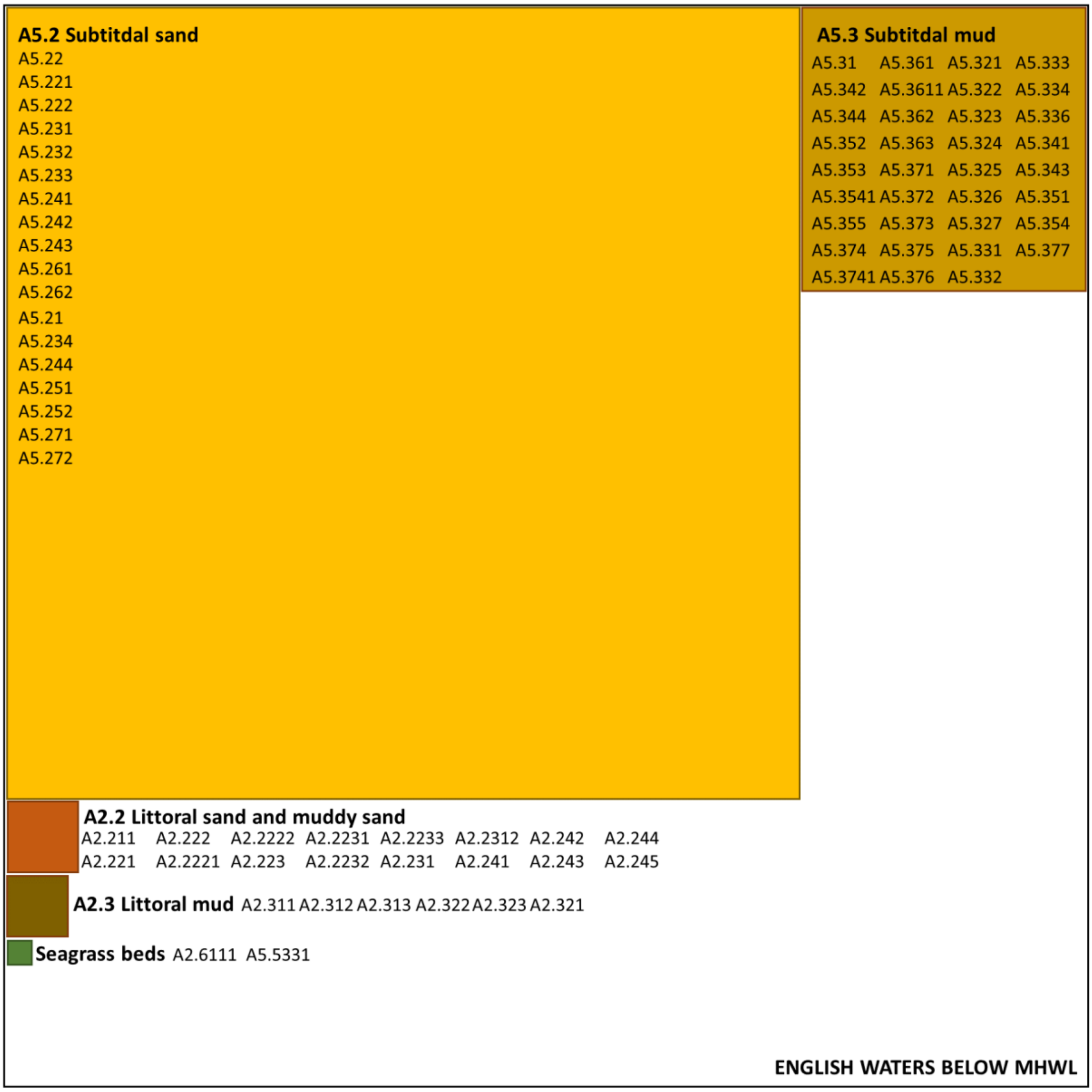


Figure 1. Conceptual figure representing blue carbon habitats that could potentially interact with demersal trawling activity. The size of boxes is relative to the area these habitats occupy within English waters (outer box). EUNIS codes are extracted from the benthic habitat sensitivity model n=77.

References

Gacia, E. and Duarte, C. M. (2001). Sediment retention by a Mediterranean *Posidonia oceanica* meadow: The balance between deposition and resuspension. Estuarine, Coastal and Shelf Science, 52, 505–514.

Harvey-Fischenden, A. and Vaughan D. (unpublished). Methodology for adding ecosystem services information to Fisheries displacement project sensitivity mapping. Natural England.

Haupt, P. and Vaughan D. (unpublished). Mapping benthic habitat sensitivity to fishing activities in English waters. Natural England.

Macreadie, P.I., Atwood, T.B., Seymour, J.R., Schmitz Fontes, M.L., Sanderman, J., Nielsen, D.A. and Connolly, R.M. (2019). Vulnerability of seagrass blue carbon to microbial attack following exposure to warming and oxygen. Science of the Total Environment, 686, 264–275.

OSPAR (2011). Intersessional Correspondence Group on Cumulative Effects – Amended 25th March 2011 https://data.jncc.gov.uk/data/1e136107-1396-4c67-b755-dc9f43bf3bb1/20110328-ICG-C-Pressures-list-v4.pdf [Accessed December 2021].

Tillin, H.M., Langmead, O., Pegg, S., Carr, S., Gibson- Hall, E., La Bianca, G., Luff, A., Keen, P.W., Wilding, C., Nicholson, J.C., Ivory, P. and van Rein, H. (2020). Development of JNCC Marine Ecosystem Services Optimisation models. JNCC Report No. 650. JNCC, Peterborough, ISSN 0963-8091.

Tyler-Walters, H., Tillin, H.M., d'Avack, E.A.S., Perry, F. and Stamp, T., (2018). Marine Evidence-based Sensitivity Assessment (MarESA) – A Guide. Marine Life Information Network (MarLIN). Marine Biological Association of the UK, Plymouth, pp. 91.

Natural England is here to secure a healthy natural environment for people to enjoy, where wildlife is protected and England's traditional landscapes are safeguarded for future generations.

Natural England publications are available as accessible pdfs from www.gov.uk/natural-england.

Should an alternative format of this publication be required, please contact our enquiries line for more information: 0300 060 3900 or email enquiries@naturalengland.org.uk.

NATURAL ENGLAND