Carbon storage in Northern Ireland's aquatic ecosystems: an evidence synthesis to support policy development.

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Abstract

Blue carbon is defined as carbon that is naturally sequestered and stored in the world's aquatic ecosystems. Governments around the world are currently seeking to develop a range of tools to help meet their commitments to reduce greenhouse gas emissions and potentially reverse anthropogenic climate change. As such, there is growing interest from policy makers in natural processes which may be managed to maximise carbon sequestration. Northern Ireland is a constituent region of the United Kingdom that encompasses 14,330 km^2 of land and 6000 km^2 of shelf sea on the North easter portion of the island of Ireland. This review and evidence synthesis outlines the biogeochemical processes involved in carbon sequestration and storage in marine ecosystems and seeks to provide initial estimates of carbon sequestration and burial in Northern Ireland's marine habitats. Overall Northern Ireland's marine space encompasses 3812 km^2 of shallow subtidal habitats, 1670 km^2 of deep offshore habitats in the Irish Sea and 518 km^2 of sea loughs. These habitats store between 14 and 22 Teragrams of Carbon, according to current estimates. However, there is considerable uncertainty regarding these estimates of Northern Ireland's blue carbon stocks. As such, a detailed mapping exercise is needed to support the development of a regional blue carbon inventory. This is of particular importance, given that the deep offshore habitats in the Irish Sea which are estimated to support much of the regions carbon storage are heavily fished (primarily for Nephrops norvegicus). This evidence synthesis identifies uncertainty in the current scientific literature around impacts of fishing on seabed carbon storage. This highlights a clear need to obtain region-specific information on fishing impacts in the Irish Sea to support future policy development.

Glossary of Relevant terms

Blue Carbon. Carbon sequestered by the world's aquatic ecosystems including carbon fixed by algae, seagrasses, macroalgae, mangroves, salt marshes and other plants in coastal wetlands, and organic matter stored in marine sediments and the biomass of long-lived fauna.

Carbon Sequestration. The biological, chemical or physical processes through which gaseous carbon (such as carbon dioxide or methane) is removed from the atmosphere, converted to a solid or liquid form and stored for long time period, ranging from decades to geological timescales.

Diagenesis. The physical and chemical changes that occur during the conversion of sediment to sedimentary rock. The early diagenesis of the sediment encompasses all actions between the deposition of sedimentary material until its consolidation into rock. This includes mechanical processes such as resuspension and compaction; chemical processes such as dissolution, precipitation, cementation and organo-mineral sorption; and biological processes such as bioturbation, faunal ingestion and excretion and microbial degradation.

Resuspension. The movement of sedimentary material back into suspension in the overlying water column by physical or biological disturbance.

Remineralisation. The transformation of organic matter back into its simplest inorganic form, for example conversion of glucose and other sugars back to carbon dioxide by respiration.

Introduction

Since the beginning of the industrial revolution over two hundred years ago, atmospheric carbon dioxide concentrations have increased from ~280 ppm to ~415 ppm in December 2021 (https://gml.noaa.gov/ccgg/trends/). The insulation effect of this extra carbon dioxide within the atmosphere has increased global temperatures by almost 2 °C, compared with pre-industrial times (IPCC 2021). However, the role of carbon in our environment extends far beyond its influence on global temperatures. Carbon represents the fundamental unit for energy transfer within an ecosystem. Carbon dioxide (and other inorganic carbon forms) is fixed into complex organic compounds through autotrophic processes such as photosynthesis and chemosynthesis. Subsequently organic carbon is passed through the food web as heterotrophic consumers (microorganisms, fungi and animals) feed, and remineralized to carbon dioxide by respiration. This recycling between plants and animals was first recognised by Joseph Priestly in the 18th century and the concept of carbon cycle was subsequently developed by the German chemist Justus Liebig in his Eulogy on Priestly for the Royal Society in 1840. This is elegantly summarised by Baas Becking in terms of chemical energy, stored within biomass, which is then released as kinetic energy to support the activities of life (Becking 2016). In this context carbon dioxide in the atmosphere is much like the carbonate rock in the ground, with an energy level close to zero. Through photosynthesis, the energy from solar radiation fixes the carbon dioxide as a sugar molecule with an energy level of 39.9 kJ g^{-1} (478.23 kJ / mole). This sugar provides a potential energy source to support an organism's functions, which can subsequently be transferred through the food web. It is the locking away of this energy rich organic carbon in soils / sediments and organismal tissues, and their preservation over geological time period that provides the fossil fuels that have supported industrial activity over the past 200 years (Aller and Mackin 1984, Hedges and Keil 1995, Cai 2011, Bauer et al. 2013a, Becking 2016, Bradley et al. 2022).

Interest in natural carbon sequestration processes have grown as society seeks to mitigate the atmospheric carbon dioxide concentrations that drive climate change (Macreadie et al. 2019, Atwood et al. 2020, Macreadie et al. 2021). Aquatic ecosystems play a key role in the carbon cycle and make significant contributions to the fixation of carbon within plant and algal biomass, facilitating the burial of organic matter within aquatic sediments. Over the past 200 years, the oceans have absorbed around 40 % of the carbon dioxide emissions generated by industrial societies, providing a key source of inorganic carbon to fuel photosynthesis by the phytoplankton, algae and aquatic plants that form the base of the marine food-web (Bauer et al. 2013a, Gruber et al. 2019). As a consequence, up between 0.2 and 0.4 Pg of organic carbon is ultimately buried annually within marine sediments (Bauer et al. 2013a). Likewise in freshwater systems, the annual input of carbon from the terrestrial environment ranges between 2.7 and 3.3 Pg C yr^{-1} in the form of leaf litter and other terrestrial organic matter. Of this approximately 0.9 Pg C yr^{-1} ultimately reaches the marine environment, accounting for approximately 40 % of the 2.2 Pg organic carbon that accumulates in the oceans every year (Battin et al. 2009, Aufdenkampe et al. 2011). In terms of burial around 0.6 Pg C yr^{-1} is estimated to accumulate in freshwater sediments (Battin et al. 2009, Aufdenkampe et al. 2011). Globally, the economic value of aquatic carbon burial in freshwater sediments ranges between £2.3 and £8.7 billion per year, whilst in marine sediments it is between £1.8 and £5.8 billion per year. These figures are based on projected carbon values between 2015 and 2030 (Nordhaus 2017), and assume a US$ to GB£ conversion of 0.77. Whilst poorly constrained they highlight the potential value of enhanced aquatic carbon storage as nation states seek to meet their climate change commitments under the UN Climate Change Treaty (Luisetti et al. 2019).

Blue carbon research has tended to focus upon carbon storage in vegetated marine habitats, such as mangroves, saltmarshes and seagrass beds (Atwood et al. 2015, Atwood et al. 2020). This was primarily due to their ability to fix CO_2 and directly trap high concentrations of organic carbon for storage in the underlying sediments (Duarte et al. 2013). Whilst these habitats are highlight productive in terms of carbon fixation and sequestration, they account for only 1 million km^2 of the sea floor (approximately 0.2 % of the global sea floor) and contain only a small proportion of the ocean's total organic carbon stocks (Atwood et al. 2020). In addition, the global area of vegetated marine habitats is decreasing by around 1 % per year due to anthropogenic pressures such as the development of coastal and marine infrastructure, disturbance from extractive activities like fisheries and climate change (Cai 2011, Bauer et al. 2013b, Atwood et al. 2015). As such, there is a clear need to expand the definition of blue carbon habitats to include not non-vegetated marine sediments and the sedimentary environments found in freshwaters.

Northern Ireland hosts approximately 2.8 % of the UK's populations but accounts for 4.7 % of its annual greenhouse gas emissions (McNickle et al. 2021). As such, the development of effective strategies to offset and mitigate Northern Ireland's greenhouse emissions are critical to the UK's strategy to achieve a 'net zero' economy by 2030. Northern Ireland is a 'wet place' with four percent (539 km^2) of Northern Ireland's total land area covered by freshwater lakes. Lough Neagh alone has a surface area of 396 km^2 alone and catchment extending across 3 % of Northern Ireland's total surface area. The region's coastal and shelf seas encompass 6000 km^2 of seafloor (0.088 % of the total UK EEZ), of which 1670 km^2 are deep offshore habitats in the Irish Sea and 518 km^2 are coastal sea loughs (DAERA 2018). As such, the development of an effective strategy to maximise carbon sequestration and storage in Northern Ireland's aquatic systems could provide an important tool in offsetting the regions greenhouse gas emission.

The potential for aquatic carbon storage to potentially offset national carbon emissions has been highlighted in recent discussions around by the Conference of the Parties (CoP) of the United Nations Framework Convention on Climate Change (Christianson et al. 2022), and the carbon storage potential is under evaluation as an indicator of good environmental status within the UK Marine Strategy and Oslo-Paris Treat (OSPAR) Commission's Quality Status report of the environmental status of the North East Atlantic. This demonstrates the need to supply environmental managers and policy makers with a reference toolkit to support decision making around carbon sequestration and storage in marine and aquatic systems. Here we provide a synthesis of the mechanisms that underpin carbon sequestration storage in marine and aquatic ecosystems, identify the critical processes that underpin carbon preservation and storage, and review current knowledge gaps with regard to the management of Northern Ireland's aquatic systems to maximise carbon storage.

Northern Ireland as a component in the Global Carbon Cycle

The carbon cycle provides a useful model through which we can understand the mechanisms through which the anthropogenic release of carbon dioxide into the environment can be balanced against carbon sequestration processes (Berner 1978, Berner and Raiswell 1983, Hedges and Keil 1995, LaRowe et al. 2020). The carbon cycle links carbon accumulation and burial across the terrestrial, freshwater and marine systems are clearly outlined in the form of a mass balance (Battin et al. 2009). The key problem, however, is that the individual processes that govern accumulation of carbon in the terrestrial, freshwater and marine domains all remain poorly constrained. As such, the authors highlight the need for policy makers and environmental managers to understand the coupling between land and water, and between the carbon cycle and other biogeochemical processes is essential if we are to effectively mitigate societies' carbon dioxide emissions at both regional and global scales. With this in mind, it is useful to consider an inventory of the earth's carbon sinks (Table 1). This pre-industrial carbon stock-take highlights a number of key figures. Firstly, the bulk of all carbon present on planet earth is bound within stable geological pools, with the fossil hydrocarbon pool alone dwarfing all available carbon sources present in the atmosphere and active biogeochemical components. The active biogeochemical pools of carbon represent around 0.1 % of the earth's carbon stocks, but this increases year on year through the anthropogenic activities such as fossil fuel use and cement production which liberate carbon bound in fossil hydrocarbons and inorganic carbonates respectively (Hedges and Keil 1995).

Table 1. Major global carbon reservoirs

Reservoir	Quantity C (10^{12} t C)	Reference
Geological Pools	**Total: 75,000**	
Inorganic Carbonates	60,000	Berner (1989)
Organic rocks and fossil hydrocarbons (Kerogen, coal etc.).	15,000	Berner (1989)
Biogeochemical Pools	**Total: 43.06 (100 %)**	
Inorganic	*39.76 (91.19%)*	
Marine DIC	38 (88.25 %)	Olson et al., (1985)
Soil Carbonates	1.1 (2.55 %)	Olson et al., (1985)
Atmospheric CO_2	0.66 (1.53 %)	Olson et al., (1985)
Organic	3.3 (7.66 %)	
Soil	1.6 (3.72 %)	Olson et al., (1985)
Plant Tissue	0.95 (2.21 %)	Olson et al., (1985)
Seawater DOC	0.60 (1.39 %)	Williams and Druffel (1987)
Surficial Marine Sediments	0.15 (0.35 %)	Emerson and Hedges (1988)

All values corrected to levels before anthropogenic effects.
Adapted from Hedges and Keil (1995).

In global terms, Northern Ireland accounts for less than 0.01 % of the earth's total land mass and the regions territorial seas account for 0.001 % of global seafloor area (DAERA 2018). As such, it could be argued that the potential impacts of carbon sequestration in Northern Irelands' blue carbon habitats are unlikely to have a significant influence on the global climate in isolation. However, as a component part of one of the world's seven leading economies Northern Irish greenhouse emissions are of global significant. As previously stated the per capita greenhouse gas emissions of the Northern Irish population are almost double the UK average (McNickle et al. 2021). Within the context of the UK's commitment to achieve net-zero Greenhouse gas emissions by 2050, a reduction in Northern Irish emissions is, therefore, likely to be prioritised as an equitable solution by the UK central government and the other devolved nations (Scotland and Wales).

Northern Ireland's overall contribution to anthropogenic greenhouse gas emissions may be relatively small in global but its shared commitment to the UK's policy of achieving net-zero emissions by 2050 places a key priority in the development of the region's environmental policy (https://www.gov.uk/government/publications/net-zero-strategy). Within this context, understanding the potential reservoirs for carbon storage in Northern Ireland's aquatic ecosystems provides one of the first steps towards a regional carbon / greenhouse gas budget and management plan.

Factors governing carbon storage in Northern Irish Marine Systems

The storage of carbon within marine ecosystems, and in particular marine sediments, has been extensively studied and reviewed over many decades (LaRowe et al. 2020). The concept of 'blue carbon' developed from this work in the late 2000s as a tool to explain the potential for carbon sequestration and storage within vegetated marine habitats (e.g. seagrass beds, saltmarshes and kelp forests) (Duarte et al. 2013, Atwood et al. 2015). Since then the blue carbon concept has been further expanded to encompass carbon storage across all marine sedimentary ecosystems (Macreadie et al. 2021). Within the marine environment, there are range potential carbon storage reservoirs range from estuarine and intertidal sedimentary sediments, the traditional blue carbon habitats such as seagrass beds, salt marshes and mangroves, shellfish beds, coastal and shelf sea sediments and a range of deep-sea environments. Whilst the figures are poorly constrained, a number of attempts have been made to quantify carbon storage across these systems. The most recent effort by Atwood et al (Atwood et al. 2020) suggests that continental shelf sediments may account for up to 11.5 % of the total carbon stored at the seabed (Table 2).

Table 2 Global carbon stock values for top 1 m of marine sediments and proportion of the global seabed carbon stock (adapted from Atwood et al., 2020). ***Coastal values refer to estuaries, intertidal and vegetated blue carbon habitats*****.**

	Area (km²)	Total sediment C stocks (Pg)	Proportion of global sediment stocks.
Oceanic Provinces			
Continental Shelf (<200m)	14,250,873	256 – 274	11.5 %
Coastal (< 200 m)	4,894,100	19 – 20	0.8 %
Continental Slope (> 200 m)	19,693,306	164 – 175	7.3 %
Abyssal (>2000 m)	306,595,886	1777 – 1898	79.4 %
Hadal (> 4000m)	3,437,928	23 – 24	1 %
MPAs	18,164,927	92 – 97	4 %
All MPAs	8,498,959	47 – 50	2 %
Highly Protected MPAs			

Whilst these numbers provide an indication of the global extent of seabed carbon stores and the current levels of protection afforded to seabed carbon by marine protected areas, the authors acknowledge that they are coarse estimates which do not account for the active the range of physical and biogeochemical processes which may influence carbon residence times at the seabed.

At the UK level the potential for carbon storage marine sediments has been received growing attention in recent years, with a recent review and meta-analysis of the evidence for blue carbon storage in the coastal and shelf sea sediments under the DEFRA Secretary of State's jurisdiction (primarily England and Wales) completed by Parker et al. (Parker et al. 2021). These authors provide a valuable summary of the sediment carbon stocks across a range of coastal and shelf sea blue carbon habitats (summarised in Table 3). However, they caution that few studies reported POC stocks to 1 m depth, as recommended by the IPCC, and as such there is potential for overestimation of the total carbon stocks reported. This is particularly problematic in subtidal environments where the bulk of sediment geochemistry datasets are focussed on the upper 10 cm of the sediment. As such, methodological standardisation is required, to develop a convention for future measurements.

Table 3. Sediment carbon stock estimates for blue carbon habitats in England and Wales (as summarised by Parker et al., 2021). Sediment carbon stocks are standardised to a sediment depth of 1 m, providing a total sediment volume of 1m^3, as per IPCC wetlands guidance (IPCC, 2014).

Blue Carbon Habitat	Carbon Stock Range (kg C m^{-2})
Saltmarsh	10 – 70
Non-vegetated Intertidal Mud	5 – 35
Intertidal Sand	2.5 – 10.5
Seagrass Meadow	13.5 – 13.9
Subtidal Sand	1.6 – 2.0
Subtidal Mud	6.0 – 6.8

Parker et al. (Parker et al. 2021) highlight a lack of evidence on carbon accumulation rates across the habitats listed, with no information currently available to assess carbon accumulation in UK seagrass beds, intertidal sands or offshore slope areas. In the absence of UK specific estimates, for seagrass, carbon accumulation rates are derived from estimates of carbon accumulation by UK seagrass species (*Zostera spp.*) in northern temperature environments. As such, the estimates of carbon accumulation in UK seagrass meadows range between 67 and 105 g C m-2 yr^{-1} (based upon a review by (Novak et al. 2020)). In the remaining habitats Parker et al (2021) report that carbon accumulation in UK saltmarshes range between 66 and 196 g C m^{-2} yr^{-1}, whilst mean values of 83.5 g C m^{-2} yr^{-1} and 29.5 g C m^{-2} yr^{-1} are reported for intertidal and subtidal muds. Overall, the evidence base around carbon accumulation is reported as poor, with the ranges of values for seagrass meadows and saltmarshes suggesting that the mean values are unlikely to be representative of a particular site. As such, site specific data on sediment carbon stocks and carbon accumulation rates represent key evidence gaps in assessing the carbon storage potential of blue carbon habitats both at the UK and devolved regional levels.

As Parker et al. (2021) point out, we need to develop standardised methods for blue carbon monitoring. A recent, comprehensive review by Graves et al. (Graves et al. 2022) , goes further in outlining a toolbox of methods which could be used to support the quantification and monitoring of sediment carbon in marine systems. This includes methods aimed at identifying the sediment carbon stock, accumulation, provenance / source and potential sensitivity to disturbance, building upon the handbook for coastal blue carbon assessment methods (Howard et al.). This provides a starting point for the development of UK-wide monitoring programmes; work that is currently being facilitated through the UK Blue Carbon Evidence Partnership and a DEFRA-funded method standardisation workshop. Effective monitoring, however, requires the prioritisation of cost-effective methodologies, whose data quality can be assured (preferably through existing quality assurance / quality control schemes such as the European QUASIMEME scheme for marine monitoring laboratories. As such, coordination of blue carbon monitoring requires a coordinated approach at the UK level.

Combining estimates of sediment carbon stocks and accumulation rates with information on the areal extent of each habitat may allow estimates of Northern Irelands marine carbon storage potential to be made. However, as discussed above, site specific estimates of carbon stocks are ultimately needed to constrain the potential errors in regional carbon stock assessments. The mapping of seabed carbon stocks has been received increased interest over the past ten years, with efforts made within the NERC-funded UK Shelf Sea Biogeochemistry programme to achieve assess seabed carbon storage in the UK EEZ and develop an economic valuation of this ecosystem service (Diesing et al. 2017, Luisetti et al. 2019). Diesing et al. (2017) provide a regional assessment of seabed carbon stocks in the northwestern European continental shelf, using machine learning approaches (Random Forest prediction algorithms) to model seabed carbon stocks, parameterised by empirical measurements of sediment carbon concentrations, sediment granulometry (mud, sand and gravel content), and oceanographic conditions Broadly, the model is useful in highlighting the spatial heterogeneity of sediment carbon stores within the NW European shelf. However, the spatial scale of the model, and the inherent uncertainty in the parameterising data mean that its use for regional scale assessment is limited.

Luisetti et al. (Luisetti et al. 2019) build on this model and combining it with estimates of carbon stocks in coastal habitats (seagrass, saltmarsh and intertidal sediments) to provide a first attempt to value the carbon stored in the marine sediments of the UK EEZ. The valuation is built upon estimates of the cost to society associated with potential carbon release from each blue carbon habitat, under difference climate change scenarios. The approach take is illustrative, compounding uncertainty in the amount of carbon stored in UK blue carbon habitats with uncertainty about the spatial extent of these habitats and, uncertainty about the economic valuation of carbon storage as an ecosystem service. Nevertheless, it provides a useful indicator of the potential damage caused by carbon release, which ranges between £ 1.2 billion and £ 9,4 billion, over 25 years. Accounting for the equal distribution of the costs across the UK population this could account for between £ 45 million and £ 357 million of additional costs for Northern Ireland over the same period. Consequently, there is an economic imperative to accurately quantify the regions potential blue carbon stores, to ensure that they are reflected in any future UK-wide valuation of ecosystem-scale carbon sequestration.

Accurately mapping blue carbon habitats could provide a valuable investment towards achieving regional aspects of the UK's climate change commitments and ensure that there is an equitable distribution of costs across the UK population. Thus, regional scale mapping of Northern Ireland's marine habitats with a focus on blue carbon is recommended. At present, there is relatively little literature available on the spatial extent of the main blue carbon habitats within Northern Ireland's regional seas and coastal environments. In May 2021, Ulster Wildlife published a feasibility study on the potential for the restoration of three specific coastal blue carbon habitats in Northern Ireland (Strong et al. 2021). The authors focused upon assessing the spatial extent of four specific blue carbon habitats: kelp forests, seagrass meadows; saltmarsh and, native oyster (Ostrea edulis) and mussel (Mytilus edulis) bed. This represents a significant body of work to both map existing blue carbon habitats and assess their potential for expansion under favourable marine spatial management conditions. The caveat with the authors' approach, however, is whilst they provide empirical data on the current extent to these habitats in Northern Ireland (table 4), their habitat suitability models are limited by the availability of empirical data on seabed type and oceanographic conditions for parameterisation. Nevertheless, the study highlights the relatively small area of Northern Ireland's marine space which is currently occupied by priority blue carbon habitats. Of these habitats, approximately 26 % of the total area of 1398 km^2 is afforded some protection within the regions marine protected area network.

Table 4. Extent of blue carbon habitats identified by Strong et al. (2021) in Northern Ireland's regional seas, their presence within the regions marine protected area network and % coverage of the regional sea-floor area (6000 km^2).

Blue Carbon Habitat	Areal Extent (km^2)	Areal Extent in MPA network (km^2)	Coverage of NI regional sea area (%)
Kelp	1041.7	213.0 (20 %)	17.3
Ostrea edulis	167.9	41.0 (24 %)	2.79
Mytilus edulis	140.2	97.6 (70 %)	2.33
Saltmarsh	31.1	8.5 (27 %)	0.52
Seagrass	17.2	11.1 (65 %)	0.28

From a carbon management perspective there is an evidence gap pertaining to carbon storage across Northern Ireland's blue carbon habitats, including non-vegetated intertidal and subtidal sediments (as highlighted in Parker et al., 2021). Northern Ireland's seafloor is primarily composed of shallow, shelf seas, with deep offshore muddy sediments accounting for 28 % of the region's seafloor area (located in the Western Irish Sea). These offshore muddy sediments support a regionally significant fishery for *Nephrops norvegicus* in the western Irish Sea. Based on the compiled data from the draft Northern Ireland Marine Plan (DAERA 2018), Parker et al.

(Parker et al. 2021) and Strong et al. (Strong et al. 2021) we can derive estimates for seabed carbon storage in Northern Irish marine systems in support of a regional blue carbon assessment (Table 5). Based on these data we apply the environmental carbon valuations outlined by (Nordhaus 2017) to estimate both current and future economic values of Northern Ireland's marine carbon stocks (table 6). Based on these values we estimate that the overall marine carbon stock for Northern Ireland may be valued between £2 and £3.2 billion by 2030. Although poorly constrained, these figures highlight the potential importance of Northern Ireland's subtidal sediments as the largest potential carbon stores in the marine environment. To achieve an accurate carbon budget, however, requires mapping of the spatial variability of carbon storage within Northern Ireland's marine sediments.

Table 5. Estimated total carbon stocks of Northern Ireland's blue carbon habitats. Kelp stock refers to the living biomass of the plants, whilst all other estimates relate are for the upper 1m of sediment below the seabed. Sediment carbon stocks are standardised to a sediment depth of 1 m, providing a total sediment volume of 1m^3, as per IPCC wetlands guidance (IPCC 2014)

Habitat	Estimated Area km^2	Estimated Carbon Stock (Tg C)
Kelp	1041.7	0.304 – 0.340 (Biomass Standing Stock)
Ostrea edulis	167.9	1.007 – 1.142*
Mytilus edulis	140.2	0.841 – 0.953*
Saltmarsh	31.1	0.311 – 2.177
Seagrass	17.2	0.232 – 0.239
Subtidal Sands	2963	4.740 – 5.926
Deep Offshore Mud	1670	6.680 – 11.356

* For *Ostrea edulis* and *Mytilus edulis* beds we assume sediment carbon stocks to be similar to subtidal muddy sediments, given the enrichment of the sediment from shellfish pseudo-faeces

Table 6. Estimated economic valuations of Northern Ireland's blue carbon habitats based on the data summarised in table 5, and economic valuations of carbon sequestration (per metric ton) from Nordhaus (2017), with a fixed US$ TO GB£ conversion of 0.77.

Habitat	2024 Carbon Stock Valuation (£ million)	2030 Carbon Stock Valuation (£ million)
Kelp	37.39 - 42.28	44.33 - 49.58
Ostrea edulis	125.21 - 141.99	146.84 - 166.52
Mytilus edulis	104.57 - 118.49	122.69 - 138.96
Saltmarsh	38.67 - 270.69	45.35 - 317.44
Seagrass	28.85 - 29.72	33.83 - 34.85
Subtidal Sands	589.37 - 736.84	681.16 - 863.09
Deep Offshore Mud	830.59 - 1412.00	974.04 - 1655.87
Total	1755.05 - 2752.01	2058.18 - 3227.32

Seabed biogeochemical processes

Carbon preservation and storage in marine sediments is governed by a suite of biogeochemical processes within the sediment. To understand the potential long-term fate of organic matter reaching the seafloor, and its spatial and temporal variability, requires elucidation of these processes. In marine sediments, the microbial community of bacteria, archaea and unicellular organisms represent the biogeochemical engine which drives the diagenesis and remineralisation of carbon (Witte et al. 2003, van Nugteren et al. 2009b, Woulds et al. 2009, Hunter et al. 2012, Woulds et al. 2016, Hunter et al. 2019). The microbial community supports a range of both aerobic and an anaerobic metabolic pathways by which sedimentary carbon is transformed and remineralised back to inorganic carbon species, such as carbon dioxide, which are then released into the overlying water column or atmosphere (Jørgensen and Boudreau 2001). The key limitation on microbial activity is oxygen availability and penetration into the sediment, which ultimately governs the shift from metabolic pathways with high energy output such as aerobic respiration, to less efficient anaerobic pathways such as sulphur reduction, methanogenesis, sulphur reduction and fermentation (see Figure 1 and Figure 2). Whilst microbial activity represents the engine of carbon cycling in marine sediments, these processes are mediated by the faunal community resident within marine sediments (van Nugteren et al. 2009a, Hunter et al. 2012, Hunter et al. 2013, Hunter et al. 2019). Key processes including bioturbation and bioirrigation of the sediment extend oxygen penetration into the sediment (Pascal et al. 2019). In addition, the feeding behaviour of deposit feeding animals subducts fresh organic matter from the surface to deeper sediment layers (Blair et al. 1996, Levin et al. 1999), whilst the transit of organic matter through their digestive tracks breaks organic matter particles down into small pieces (van Nugteren et al. 2009a). This increases the surface area to volume ratio of the organic matter particles, making them more amenable to microbial degradation (Witte et al. 2003). As such, an understanding of the benthic community composition and estimates of benthic community respiration are needed to understand the residence time for carbon in marine and aquatic sediments.

Figure 1. Schematic of the diagenetic zones and trends in pore water dissolved nutrient profiles during the early diagenesis of carbon in aquatic sediments. Oxygen availability is the key environmental parameter that controls the metabolic pathways for respiration in aquatic sediments, causing a shift to less energetically efficient metabolic pathways as oxygen stress increases with sediment depth (adapted from (James 2005)

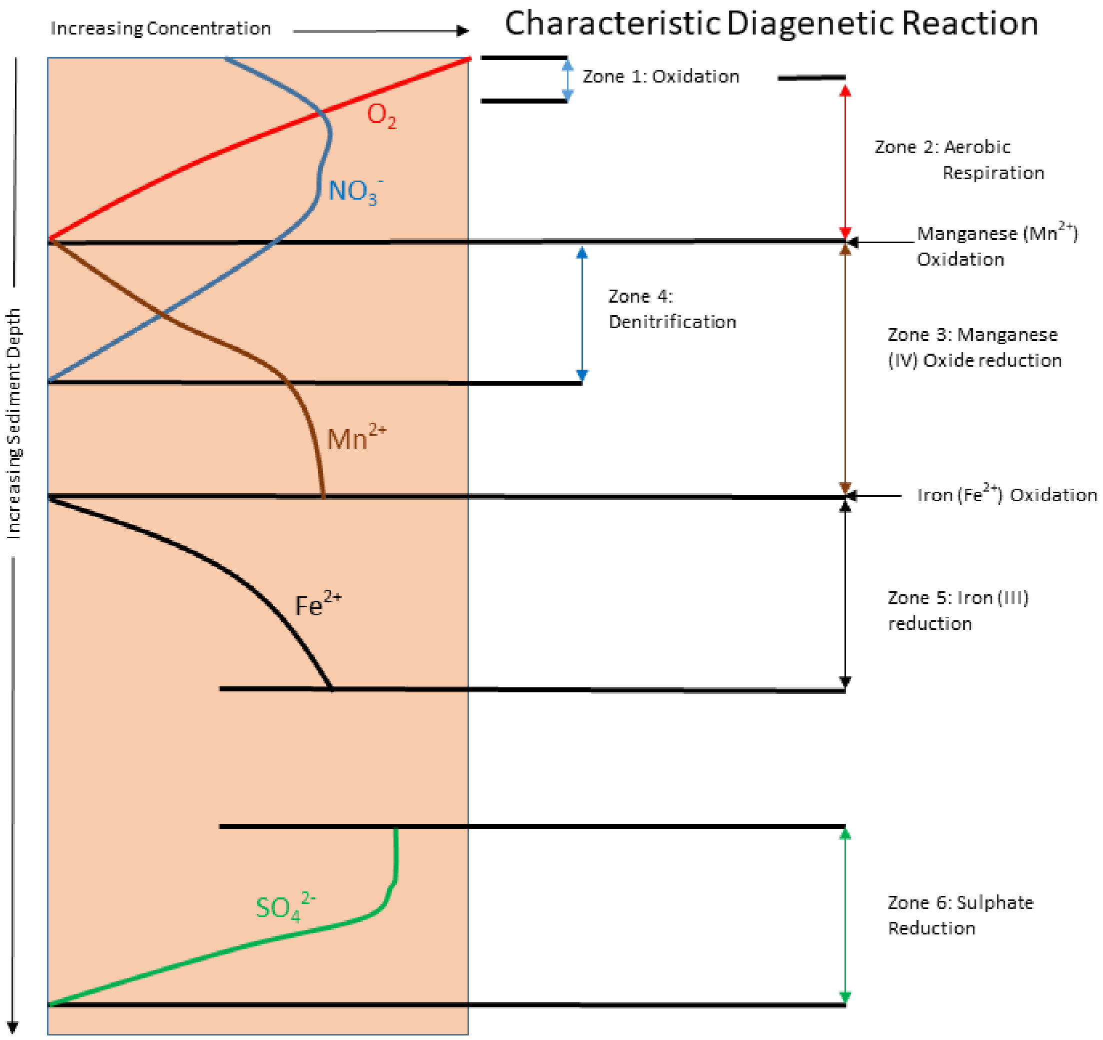


Figure 2. Sequence of Redox reactions for organic matter mineralisation in marine sediments and the amount of energy they release in kJ per mole of organic carbon (Adapted from (Jørgensen and Boudreau 2001)

Metabolic Pathway	Oxidant	Principal By-product	Energy released (kJ mol^{-1})
Aerobic Pathways			
Oxic Respiration	Dissolved Oxygen (O_2)	Carbon Dioxide (CO_2)	3190
Anaerobic Pathways			
Denitrification	Dissolved Nitrate (NO_3^-)	Nitrogen Gas (N_2)	3030
Manganese (IV) Reduction	Manganese oxide (MnO_2)	Dissolved Manganese (Mn^{2+})	2920 to 3090
Iron (III) Reduction	Iron Oxide (Fe_2O_3)	Dissolved Iron (Fe^{2+})	1330 to 1410
Sulphur Reduction	Dissolved sulphate (SO_4^{2-})	Sulphide ions [which may combine with H+ ions to form Hydrogen sulphide (H_2S), or with dissolved iron to form Iron pyrite (FeS_2)]	380
Methanogenesis	Acetic Acid or Carbon dioxide	Methane	350
Fermentation	N/A	Ethanol	59.5
Lactate Fermentation	N/A	Lactic Acid	59.5

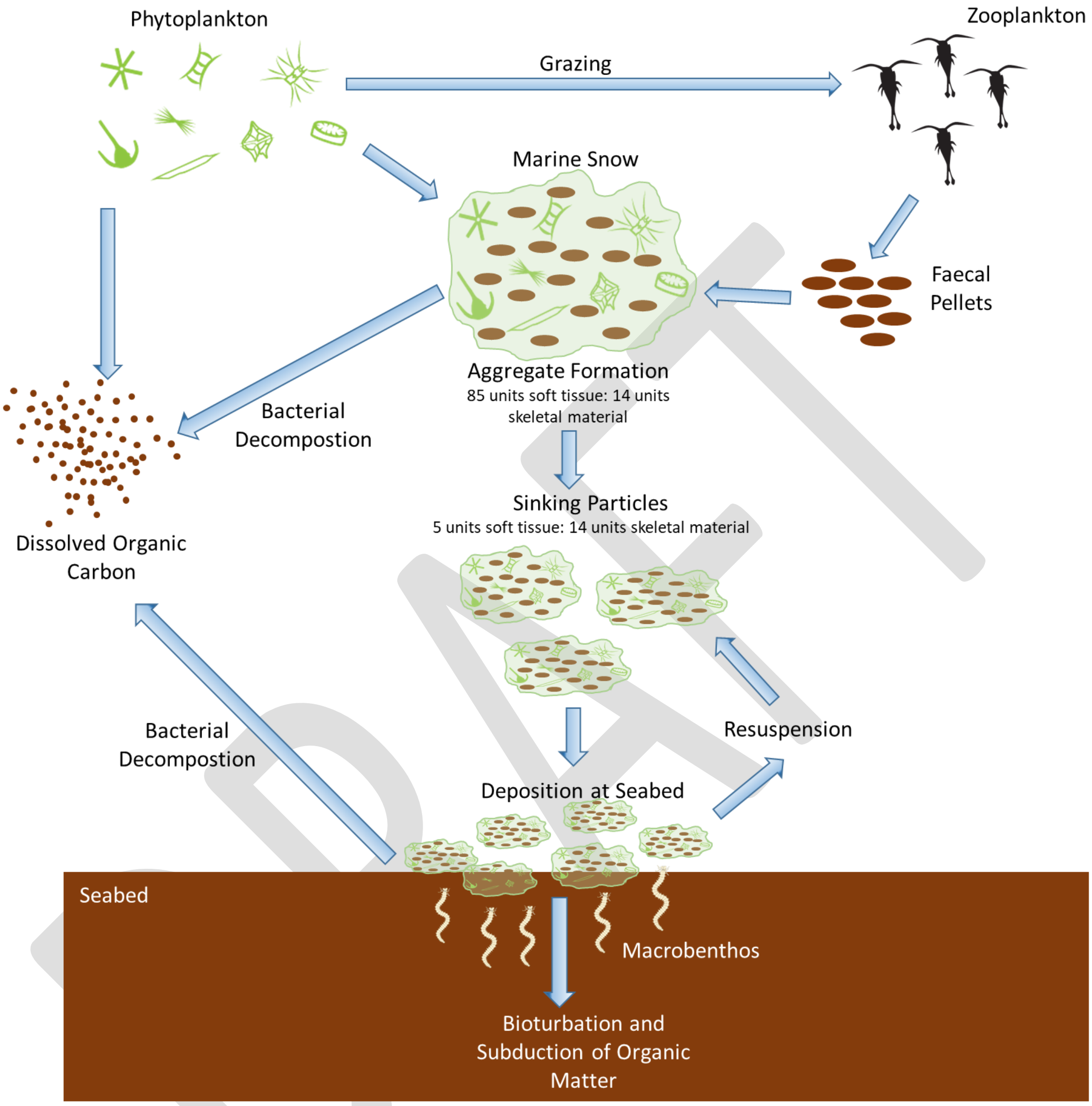
Decreasing Energetic Efficiency

To directly quantify carbon residence time, sediment community oxygen consumption (SCOC) is an effective proxy for the rate at which carbon is remineralised to carbon dioxide within marine sediments. SCOC is a composite measurement of both animal and microbial respiration within a known area of the sea bed (Jørgensen et al. 2022). Unlike seabed carbon stocks, which have received considerable investigation over recent years (Graves et al. 2022), the processes associated with measuring carbon loss from the sediment have not been considered within a monitoring context. However, there already exists an international database of SCOC datasets which is derived from 230 papers published on the topic between 1968 and 2019 (Stratmann et al. 2019). This database provides an insight into how SCOC changes over latitudinal, temperature and depth gradients and allows an assessment of how both in situ and ex situ methods may influence the data reported. Using these data, we can derive estimates for SCOC in the North and Celtic Seas regions between 30.4 and 2260.94 mg O_2 m^{-2} d^{-1}. Based on these figures we can calculate the carbon dioxide emissions from Northern Irish marine sediments are likely to fall in a range between 7.99 and 594.04 mg C m^{-2} d^{-1}, assuming a respiratory quotient of 0.7 (Tanioka and Matsumoto 2020). This method can be combined with outer proxies for carbon residence to support the development of a mass-balanced estimate of carbon storage in marine and aquatic sediments, provided the sources and input of organic matter to the seabed can also be identified (Graves et al. 2022).

Impact of water column processes

Development of a full mass balance for seabed carbon storage requires the carbon inputs to be quantified. In traditional blue carbon habitats such as saltmarsh and seagrass meadows, carbon dioxide is fixed by the in situ vegetation, and then preserved either within the subsurface biomass of the rhizome, or through burial of senescent leaf material (Duarte et al. 2013). Likewise in many 'non-vegetated' intertidal sediments, a secret garden of diatoms and other unicellular algae (the microphytobenthos), fix carbon dioxide through photosynthesis, and this organic matter is subsequently transferred deeper into the sediment through trophic transfer through the food-web and/or direct burial (MacIntyre et al. 1996, Hunter et al. 2019). In coastal and shelf seas much of the seabed lies below the photic zone (where enough light penetrates to support photosynthesis). As such, across most shelf seas the main input of organic matter (and carbon) to the seabed is from the overlying water column. Phytoplankton, unicellular algae floating close to the sea surface, fix carbon dioxide to build biomass, with major blooms in the early spring (March / April) and late summer (August). These algae, then either slowly sink towards the seafloor as the age and die or are eaten by zooplankton and/or fish and then sink to the seafloor bound in animal faecal pellets (Figure 3). Consequently, up to 50 % of the phytoplankton biomass generated near the sea surface will reach the sea floor as a phytodetritus or faecal material (Middelburg et al. 1997, Arndt et al. 2013).

Figure 3. Schematic of biological particle formation and deposition to the seabed, showing the mechanisms for and approximate composition of particulate organic matter sinking to the seafloor (James 2005).



Consequently, the supply of organic matter to the seafloor is intimately linked to ecosystem processes and food-web dynamics at play within the water column (Cavan and Hill 2021, Saba et al. 2021, Laurenceau-Cornec et al. 2023). Thus, changes in the phytoplankton, zooplankton and fish populations in our coastal and shelf seas may have serious implications for the supply of carbon to the seabed. Given the development of ecosystem-based modelling approaches to fisheries management, the dynamics of these processes should be possible to calculate for regions such as the Irish sea, building upon the food web models developed by, for example, the ICES workshop on Irish Sea ecosystems (WKIRISH) (ICES 2020).

Organic matter fluxes to the seafloor represent an important route through which carbon can be transferred from the sea surface for burial in marine sediments. However, burial is not the only route through which carbon may be stored in marine systems. Within the water column, there also exists a large pool of dissolved organic matter (Catalá et al. 2021). This is operationally defined as any organic matter that can pass through either a 0.45 or 0.7 µm filter (Church et al. 2002). It consists of a complex suite of biomolecules including carbohydrates, free amino acids, proteins, tannic acids, fulvic acids and humic substances, which can be loosely graded into three classes based on their reactivity (Repeta 2015, Tfaily et al. 2017, Rowe et al. 2018, Hansell and Orellana 2021). Labile DOM is defined as with turnover times ranging between hours and days; semi-labile DOM, that has turnover times across seasonal or annual time scales; non labile / refractory DOM, that is recycled on time scales ranging from centuries to millennia (Kirchman et al. 1991, Hansell et al. 2004, Hansell 2013, Kirchman and Gasol 2018, Hansell and Orellana 2021). Based on these definitions the pool of refractory DOM in the marine environment could be considered an important carbon store that mitigates against climate change. Globally, the marine DOM pool is enormous, accounting for up to 70 % of all organic carbon found in the oceans (Worden et al. 2015). DOM concentrations are spatially variable, ranging from < 0.4 mg C l^{-1} in the deep Pacific Ocean to concentrations in excess of 1 mg C l-1 in coastal seas (Hansell and Orellana 2021).

In marine systems, DOM is primarily produced through processes such as zooplankton feeding and viral lysis of planktonic cells (Nagata and Kirchman 1992, Nagata 2000, Middelboe and Jørgensen 2006, Suttle 2007, Kirchman and Gasol 2018). However, in coastal systems, contributions from riverine inputs and terrestrial organic matter can be important contributors to the marine DOM pool (Middelboe and Jørgensen 2006, Ask et al. 2009, Rowe et al. 2018). The functions of DOM in the marine environment are tightly coupled to the biological carbon pump, that controls the flux of particulate carbon from the sea surface to seabed (Riebesell et al. 2007, Riebesell et al. 2009, Carlson and Hansell 2015, Zhuang and Yang 2018, Catalá et al. 2021). This DOM represents a key energy source for the microbial community in marine systems, who in turn mediate the concentration and biogeochemical composition of the DOM pool (Worden et al. 2015, Mühlenbruch et al. 2018). This is elegantly outlined within the microbial carbon pump, leading to the accumulation and long-term storage of refractory DOM within the oceans (Lancelot et al. 1993, Hansell 2013, Hansell and Orellana 2021, Stukel et al. 2023). As such, the DOM pool within Northern Irish waters should be considered as a key link in any future regional carbon budget (Lønborg et al. 2024). In addition, as the UK Marine Strategy; and OSPAR’s Quality Status Reports on the marine environmental include DOM concentrations as indicators of trophic status of water bodies, monitoring and quantifying DOM concentrations can support regional assessments of water quality.

Impacts of disturbance and environmental change

Oxygen availability is critical to the degradation of organic matter and its mineralisation to carbon dioxide in marine sediments (see Figure 1 and 2). Activities that increase the exposure of the organic matter stored in marine sediment to increased oxygen concentrations are, therefore, predicted to reduce seabed carbon stocks (Berner 1978, Berner and Raiswell 1983, Hedges and Keil 1995, Middelburg et al. 1997). Based on this premise anthropogenic activities including fishing, aggregate extraction, dredging, underwater construction and anchoring of vessels and marine structures may all impact the potential for seabed carbon storage (Atwood et al. 2020, Cavan and Hill 2021) .

Fisheries have recently been highlighted as a profound threat to seabed carbon storage (Atwood et al. 2020, Sala et al. 2021). The mobile towed fishing gear typically used in bottom-contact fishing includes a range of trawls and dredges designed to collect either demersal fish or benthic crustaceans and / or molluscs (Black et al. 2022, Epstein et al. 2022). The capacity of bottom-contact fishing gear to resuspend sediments has been well established, and is function of basic physical principals (Churchill 1989). Alongside this, bottom-contact fishing is destructive of the communities of animals living on top of (epifaunal) and within (infaunal) the seabed. As such, the environmental impacts of fishing activities need to be considered in the context of preservation of seabed features and ecosystem services, such as carbon storage (Atwood et al. 2020, Black et al. 2022, Epstein and Roberts 2022) . In a recent review of the benthic impacts of towed fishing metiers on seabed carbon storage Epstein et al (2021) highlight the lack of consistent evidence regarding trawling impacts (table 7). Of the 38 studies reviewed by Epstein et al (Epstein et al. 2022), thirteen reported some negative effects of fishing pressure upon either sediment carbon content or remineralisation rates, whilst eight reported some form of positive impact (table 7). The majority of studies detected either no effects or mixed effects upon either sediment carbon content or remineralisation. This lack of consensus is likely driven by the environmental variability across the seafloor, and the complex suite of impacts of trawling, which drives not only sediment resuspension (Pusceddu et al. 2005, Pusceddu et al. 2015) , but also changes in sediment composition and porosity (Depestele et al. 2018), and the structure and functioning of the biological communities resident both on and within the seabed (Collie et al. 2000, Sciberras et al. 2018) . Consequently, assessments of the potential impacts of trawling on seabed carbon stocks need to account for regionally specific information on the seabed sedimentology, geochemistry and biological communities, to accurately assess potential risk.

Across the UK's EEZ, recent studies have sought to map seabed carbon storage and its potential economic value (Diesing et al. 2017, Luisetti et al. 2019)and assess its potential vulnerability to disturbance from bottom trawling (Black et al. 2022, Epstein et al. 2022, Epstein and Roberts 2022. Whilst these studies do not overcome the range of caveats outlined by Epstein et al. (2021), they demonstrate that the main areas of potential carbon storage in UK waters coincide to a large degree with active fishing grounds {Black, 2022 #677). There is, therefore, likely to be a significant economic impact to the UK economy associated with closing these areas to fishing. This is estimated to be in the region of between £55 and 212 million per year for the UK as a whole (Epstein and Roberts 2022). These studies are, however, conducted at spatial scales which are too broad to effectively support a regional based assessment for Northern Ireland. As such, there is insufficient data on Northern Ireland's seabed carbon stocks to assess their potential sensitivity to fisheries disturbance.

The discussion around fishing impacts on the seabed represents only one facet of a wider discussion about effective management of the seabed to support delivery of national climate change goals. As such, the development of effective marine spatial planning tools are essential to maximise seabed carbon storage in a multi-user marine space (Parker et al. 2021). Disturbance of the seabed other activities including sand and aggregate extractions, dredging for navigation purposes and the deployment of ships anchors will have significant impacts on the seabed (Barry et al. 2010; Dannheim et al. 2019; Dauvin, 2019; Watson et al. 2022). For example, Watson et al. (Watson et al. 2022) report that the anchor of a high-tonnage (>9000 Gross Tonnage) ship can excavate the seabed to a depth of over 80 cm, leaving a scar that is detectable four years after the anchorage event. As such, the development of climate smart marine spatial plans need to account for the full range of seabed users and their potential to impact seabed carbon storage (Barry et al. 2010, Dauvin 2019, Watson et al. 2022).

In light of the future importance of both marine windfarms to the future energy infrastructure of the UK, their potential impacts on seabed carbon stocks are likely to require consideration. These are potentially complex, with the initial construction phase associated with seabed disturbance and sediment resuspension (Dannheim et al. 2019). As such a full environmental impact assessment of windfarm construction should include a before, after, control, impact (BACI) study on seabed carbon stocks to ascertain the short-term impacts (refs). Once windfarms have been installed, however, there is likely to be pronounced changes in the benthic communities and biogeochemistry of the seabed underlying the turbine structure. This has been highlighted in the Belgian north sea, where the FACE-IT project has demonstrated increases in the flux of organic matter rich faecal material to the sea floor around wind turbines, as a consequence of the new habitat provided for sessile encrusting fauna such as bivalves and tunicates on the turbine structures (Coolen et al. 2018, De Borger et al. 2021, Gușatu et al. 2021). As a consequence, they report organic matter accumulation, and a shift from coarse sands to finer sediments beneath a wind farm (De Borger et al. 2021).

Table 6. Summary of studies investigating the impacts of mobile demersal fishing pressure on the seabed, measuring either changes in sediment organic carbon / organic matter content, sediment community oxygen consumption / remineralisation rates, or both. The last two columns indicate whether fishing pressure was associated with a lower (red), higher (green), mixed or non-significant (both grey) change in the parameter under investigation. Adapted from (Epstein et al. 2022)

Reference	Oceanic Region	Sediment	Depth (m)	Gear Type	Study Type	Impact Type	Sediment	Effect	Parameter
Adriano et al. 2005	Mediterranean	Sandy-mud	~ 1	Cam dredge	BA	Commercial Fishing	Surficial	1	Organic Carbon / Organic Matter Content
Atkinson et al. 2011	SE Atlantic	Muddy-sand	346-459	Otter-trawl	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Bhagirathan et al. 2010	N Indian	Mud	15-40	Otter-trawl	BA	Experimental	Surficial	-1	Organic Carbon / Organic Matter Content
Bown et al. 2005	NE Pacific	Muddy-sand	25-35	Otter-trawl	BACI	Experimental	< 5 cm	0	Organic Carbon / Organic Matter Content
Bown et al. 2005	NE Pacific	Muddy-sand	25-35	Otter-trawl	IC	Commercial Fishing	< 5 cm	0	Organic Carbon / Organic Matter Content
Dolmer et al. 2001	NE Atlantic	Muddy-sand	7	Mussel Dredge	IC	Experimental	Surficial	0	Organic Carbon / Organic Matter Content
Eleftheriou and Robertson, 1992	NE Atlantic	Sand	7	Scallop Dredge	BA	Experimental	< 10 cm	0	Organic Carbon / Organic Matter Content
Ferguson et al. 2020	SW Pacific	Muddy-sand	4	Otter-trawl	BACI	Experimental	Surficial	0	Organic Carbon / Organic Matter Content
Fiordelmondo et al. 2003	Mediterranean	Sand	~ 2	Clam Dredge	IC	Experimental	Surficial	-1	Organic Carbon / Organic Matter Content
Goldberg et al. 2014	NW Atlantic	Sand	03-05	Hydraulic Dredge	IC	Experimental	< 20 cm	0	Organic Carbon / Organic Matter Content
Hale et al. 2017	NE Atlantic	Mud and Sand	19-29	Otter-trawl	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Hale et al. 2017	NE Atlantic	Mud and Sand	19-29	Scallop Dredge	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Lamarque et al 2021	NE Atlantic	Sandy-mud	33-78	Mixed Trawls	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Lindeboom and de Groot 1998	NE Atlantic	Mud and Sand	30-75	Mixed Trawls	BACI	Experimental	< 10 cm	0	Organic Carbon / Organic Matter Content
Lindeboom and de Groot 1999	NE Atlantic	Mud and Sand	30-76	Mixed Trawls	IC	Commercial Fishing	< 10 cm	0	Organic Carbon / Organic Matter Content
Liu et al. 2011	W Pacific	Sandy-mud	20	Mixed Trawls	IC	Commercial Fishing	Surficial	1	Organic Carbon / Organic Matter Content
Martin et al. 2014	Mediterranean	Mud	453-591	Otter-trawl	IC	Commercial Fishing	< 50 cm	-1	Organic Carbon / Organic Matter Content
Mayer et al. 1991	NW Atlantic	Mud and Mixed	8-20	Otter Trawl	IC	Experimental	< 20 cm	0	Organic Carbon / Organic Matter Content
Mayer et al. 1991	NW Atlantic	Mud and Mixed	8-21	Scallop Dredge	IC	Experimental	< 20 cm	0	Organic Carbon / Organic Matter Content
McLaverty et al. 2020	NE Atlantic	Sandy-mud	3-11	Mussel Dredge	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Mercaldo-Allen et al. 2016	NW Atlantic	Fine Sand	3-5	Hydraulic Dredge	IC	Experimental	Surficial	1	Organic Carbon / Organic Matter Content
Meseck et al. 2014	NW Atlantic	Fine Sand	5-6	Hydraulic Dredge	BACI	Experimental	< 20 cm	0	Organic Carbon / Organic Matter Content
Morys et al. 2021	Baltic	Muddy-sand	12	Benthic Dredge	IC	Experimental	< 15 cm	-1	Organic Carbon / Organic Matter Content
Morys et al. 2021	Baltic	Muddy-sand	12	Benthic Dredge	IC	Experimental	< 15 cm	-1	Sediment Community Respiration / Remineralisation Rate
Palanques et al. 2014	Mediterranean	Mud	40-70	Otter Trawl	IC	Commercial Fishing	< 30 cm	1	Organic Carbon / Organic Matter Content
Paradis et al., 2019	Mediterranean	Mud	550	Otter Trawl	IC	Commercial Fishing	< 35 cm	-1	Organic Carbon / Organic Matter Content
Paradis et al., 2019	Mediterranean	Mud	550	Otter Trawl	IC	Commercial Fishing	< 35 cm	1	Sediment Community Respiration / Remineralisation Rate
Paradis et al. 2021	Mediterranean	Mud	425-494	Otter Trawl	IC	Commercial Fishing	< 10 cm	-1	Organic Carbon / Organic Matter Content

Table 6 continued

Reference	Oceanic Region	Sediment	Depth (m)	Gear Type	Study Type	Impact Type	Sediment	Effect	Parameter
Polymenakou et al. 2005	Mediterranean	Sandy-mud	30-51	Otter Trawl	BA	Commercial Fishing	Surficial	1	Sediment Community Respiration / Remineralisation Rate
Pusceddu et al. 2005	Mediterranean	Sandy-Mud	30-80	Otter Trawl	BA	Commercial Fishing	< 10 cm	1	Organic Carbon / Organic Matter Content
Pusceddu et al. 2014	Mediterranean	Mud	454-556	Otter Trawl	IC	Commercial Fishing	< 10 cm	-1	Organic Carbon / Organic Matter Content
Pusceddu et al. 2014	Mediterranean	Mud	454-556	Otter Trawl	IC	Commercial Fishing	< 10 cm	-1	Sediment Community Respiration / Remineralisation Rate
Rajash et al. 2019	N Indian	Sad	5-35	Beam Trawl	BA	Experimental	Surficial	-1	Organic Carbon / Organic Matter Content
Ramalho et al. 2018	NE Atlantic	Muddy-sand	285-550	Otter Trawl	IC	Commercial Fishing	Surficial	-1	Organic Carbon / Organic Matter Content
Ramalho et al. 2018	NE Atlantic	Mud and Sand	285-550	Otter Trawl	LH	Commercial Fishing	< 5 cm	0	Organic Carbon / Organic Matter Content
Rosli et al. 2016	SW Pacific	Sandy-mud	670-1561	Otter Trawl	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Sciberras et al 2016	NE Atlantic	Mud and Sand	20-43	Otter Trawl	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Sciberras et al 2016	NE Atlantic	Mud and Sand	20-43	Scallop Dredge	LH	Commercial Fishing	Surficial	0	Organic Carbon / Organic Matter Content
Serpetti et al 2013	NE Atlantic	Muddy-sand	769-823	Mixed Trawls	IC	Commercial Fishing	< 10 cm	0	Organic Carbon / Organic Matter Content
Sheridan and Doerr 2005	NW Atlantic	Mud and Sand	5-20	Otter Trawl	IC	Commercial Fishing	< 5 cm	0	Organic Carbon / Organic Matter Content
Smith 2000	Mediterranean	Sandy-mud	~ 200	Otter Trawl	BACI	Commercial Fishing	< 5 cm	0	Organic Carbon / Organic Matter Content
Tiano et al. 2019b	NE Atlantic	Muddy-sand	34	Mixed Trawls	BA	Experimental	< 5 cm	0	Organic Carbon / Organic Matter Content
Tiano et al. 2019b	NE Atlantic	Muddy-sand	34	Mixed Trawls	BA	Experimental	< 5 cm	-1	Sediment Community Respiration / Remineralisation Rate
Trimmer et al 2005	NE Atlantic	Muddy-sand	20-80	Beam Trawl	LH	Commercial Fishing	< 10 cm	0	Organic Carbon / Organic Matter Content
van de Veldte et al. 2018	NE Atlantic	Mud	~ 7	Unknown	BA	Commercial Fishing	< 30 cm	1	Sediment Community Respiration / Remineralisation Rate
Wang et al. 2021	W Pacific	Mud and Sand	1-28	Mixed Trawls	Recovery	Commercial Fishing	Surficial	-1	Organic Carbon / Organic Matter Content
Watling et al 2001	NW Atlantic	Muddy-sand	15	Scallop Dredge	BA	Experimental	< 15 cm	0	Organic Carbon / Organic Matter Content

For ' Study Type': BA - Before-after fishing impact; IC = Impact-control site comparison; LH = Low to high impacted sites; BACI = Before-After Control-Impact study; Recovery = Change after removal of fishing pressure.

Synthesis

Based upon the evidence currently available, it is not currently possible to accurately assess the potential carbon stocks within Northern Ireland's marine sediments. The spatial extent of the 'traditional' blue carbon habitats, such as seagrass meadows and saltmarsh, have been investigated and are relatively limited in area defined (see table 4,). The extent of subtidal muds and sands are, however, remain poorly constrained. In addition, region-specific data on seabed carbon content for the Northern Irish sections of the Irish and Malin Seas is limited. As such, there is not currently sufficient data to accurately assess the regions seabed carbon stocks (table 5). Whilst we have a good understanding of the geochemical processes that govern seabed carbon sequestration and storage, efforts to map and quantify the seabed carbon stocks are currently in their infancy, with sediment carbon content data for the UK's EEZ limited (Diesing et al. 2017, Luisetti et al. 2019). As such, efforts to map the spatial and temporal changes in seabed carbon content within Northern Irish seas is needed to better constrain the stock estimates provided in Table 5.

The impacts of disturbance of seabed carbon stocks from anthropogenic activities such as fishing and anchorages has been recently highlighted as a potential threat to carbon sequestration and storage at the seabed (Atwood et al. 2020, Sala et al. 2021). According to simplified models of the reactivity of seabed organic carbon, disturbance and resuspension of sediments is predicted to increase carbon remineralisation by moving this carbon up into the water column. However, as Hedges and Keil (Hedges and Keil 1995) point out, the controls on sedimentary carbon reactivity are complex and often unpredictable. In northern temperate seas, pulses of more labile organic matter typically reach the seabed during the period after the spring phytoplankton bloom, stimulating increases in benthic community activity (Lampitt et al. 2001). Consequently, the preservation of organic matter within the sediment is tightly coupled to the community's response to organic matter inputs (Witte et al. 2003, van Nugteren et al. 2009a, van Nugteren et al. 2009b, Hunter et al. 2012, Hunter et al. 2013, Hunter et al. 2019) . As a consequence, further information is needed on the rates of sediment community activity and its relationship to organic matter content within shelf sea systems. This links directly to the potential impacts of commercial fishing, which both modifies the seabed community and physically disturbs the impacted sediments. The overall impacts of fishing pressure on seabed carbon content, however, are inconsistent, with some studies reporting either positive, negative, mixed or null effects (Table 7). The evidence base is currently insufficient to offer any definitive advice on potential impacts of fishing on seabed carbon storage. As such, there is a clear need for studies to be undertaken at local scales to assess how management measures such as fisheries closures may alter sediment carbon potential and it's relationships to localised biodiversity and other ecosystem functions such as inorganic nutrient cycling (Tiano et al. 2019a).

The marine space is critical to the UK's commitment to achieve net zero by 2050, both through carbon sequestration and storage and the expansion of marine renewable energy infrastructure. As such, we need to develop a clear evidence base to support marine spatial management decisions. However, most studies on blue carbon and seabed carbon storage are not at spatial scales that support regional policy development (refs). In the Northern Ireland context, there are few studies specific to the region and a clear need for evidence base development.

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