Assessing the impact of environmental pressures on seagrass Blue Carbon stocks in the British Isles

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I, Alix Evelyn Green confirm that the work presented in this thesis is my own. Where information has been derived from other sources, I confirm that this has been indicated in the thesis.

Abstract

The requirements of nations to respond to the Paris Climate Agreement by outlining National Determined Contributions (NDC's) to reduce their emissions is placing an increased global focus on the spatial extent, loss and restoration of seagrass meadows. Despite such interest, local carbon storage trends and the spatial extent of seagrass remains poorly mapped globally, and knowledge of historical loss is limited. In the British Isles this information is largely absent. The primary aim of this work was to provide a foundation of knowledge on seagrass Blue Carbon and the status of seagrass in the British Isles, to 1) better inform local conservation and management, and 2) further advance the field's understanding of trends in sediment carbon storage.

The work raised questions about the globally accepted standards for Blue Carbon research, particularly in extrapolating estimates from short (<40cm) to long (>100cm) cores. This underestimated carbon stocks by >40% in one site. Across 13 studied seagrass meadows, seagrass carbon stocks were similar, apart from at one anomalous site, and differences could not be explained by sediment silt content or aboveground biomass. Despite local similarities, on a European scale the average recorded carbon stocks were high, representing the second most carbon dense sediment per hectare of any documented European country. I found that seagrass sediments disturbed by anchoring and mooring activates had significantly less sediment carbon than undisturbed seagrass sediment. Finally, I documented 8,493 ha of recently mapped seagrass in the British Isles. With high certainty, 41% of British seagrass has been lost since 1936, and historic seagrass losses could be as high as 92%. The results are discussed in terms of conservation and management of seagrass, particularly pertaining to Blue Carbon provisions.

Impact Statement

The intention of this PhD was to conduct research to provide evidence in support of marine conservation and policy of seagrass ecosystems. Chapters within this thesis test assumptions related to global-scale seagrass carbon storage trends while providing regional estimates of carbon storage in the UK. Further, it makes contributions to improve understanding of the importance of seagrass ecosystems to Blue Carbon (i.e., carbon captured by the world's coastal ecosystems).

Global estimates of seagrass carbon stocks, used to promote seagrass Blue Carbon, are disproportionately provided by extrapolating data obtained from sediment cores <30cm to a standardised depth of 100cm. By investigating this methodology (Chapter 4), I revealed that errors in this approach can lead to underestimating the seagrass carbon stocks by >40%. These findings provide evidence that such overgeneralising has the potential to undermine Blue Carbon estimates. Further, Chapter 5 (Green *et al.*, 2018) highlights inconsistencies in global carbon stock estimates, with evidence that overgeneralising seagrasses ability to store carbon can be misleading. The international community heavily relies on discussing Blue Carbon potential from globally averaged data, which are strongly skewed by ~30% of the data coming from *Posidonia oceanica*, a species that stores ~40% more carbon than many other seagrass species. Using this global data to estimate UK seagrass carbon storage would have led to a 40% over-estimation of carbon stock. These results should act as a warning for seagrass conservationists not to rely on overgeneralised Blue Carbon estimates. For future research and conservation endeavours to be a success, we must be transparent about expected results, and this thesis could act as a catalyst to reinforce and promote this. Requests from DEFRA and Natural England for data from Green *et al.*, 2018, highlights its significance to this research to the policy community.

The work included in Chapters 5, 6 and 7, has gained significant interest from government and non-government bodies that work in seagrass conservation and management. The work of Chapter 7 represents amongst the first attempts to document areal extent and historic loss of seagrass at a country-level scale. Especially important is the attempt to do this in a way that is not subject to *shifting baseline syndrome*. The results have been used to promote Sky Ocean Rescue and Swansea Universities seagrass restoration project. Further, this work has been featured across several news stations including the BBC, the Guardian, the Times and Sky News. It has also received significant interest from the UNEP World Conservation Monitoring Centre, where I was asked to advise on seagrass Blue Carbon potential.

In support of the UK's Marine Conservation Zone consultation for Studland Bay, I submitted the results from Chapter 4 into the public consultation, highlighting that the total estimated carbon in the top 100cm of its seagrass meadow holds a value of £129,695[1], comparable to the yearly recreational activities that could, in part, be effected by reduction in anchoring (£93,100) (Defra, 2018). Chapter 6, which looked at the impact of anchoring on seagrass sediment carbon content, has direct management recommendations for Studland Bay and I have been in consultation with local NGOs advising on potential management practices for the site based on this research.

[1] Since this submission I have revised the monetary value of carbon to reflect actual traded natural carbon on the voluntary market (see Chapter 4).

Acknowledgements

Initial thanks must go to NERC for funding this work. I would like to acknowledge and thank my supervisor's Dr Peter Jones and Dr Mike Chadwick for their support and guidance throughout this process. Special thanks to Dr Richard Unsworth for providing me additional supervisory support for the work contained in Chapter 7. The work in this thesis would not have been possible without the incredible UCL Geography lab staff, in particular I would like to thank Ian Patmore for his ingenuity and problem-solving genius, and for teaching me how to make things. Finally, without Mark Parry and the Community Seagrass Initiative, my fieldwork reach would have been far less impactful, thank you.

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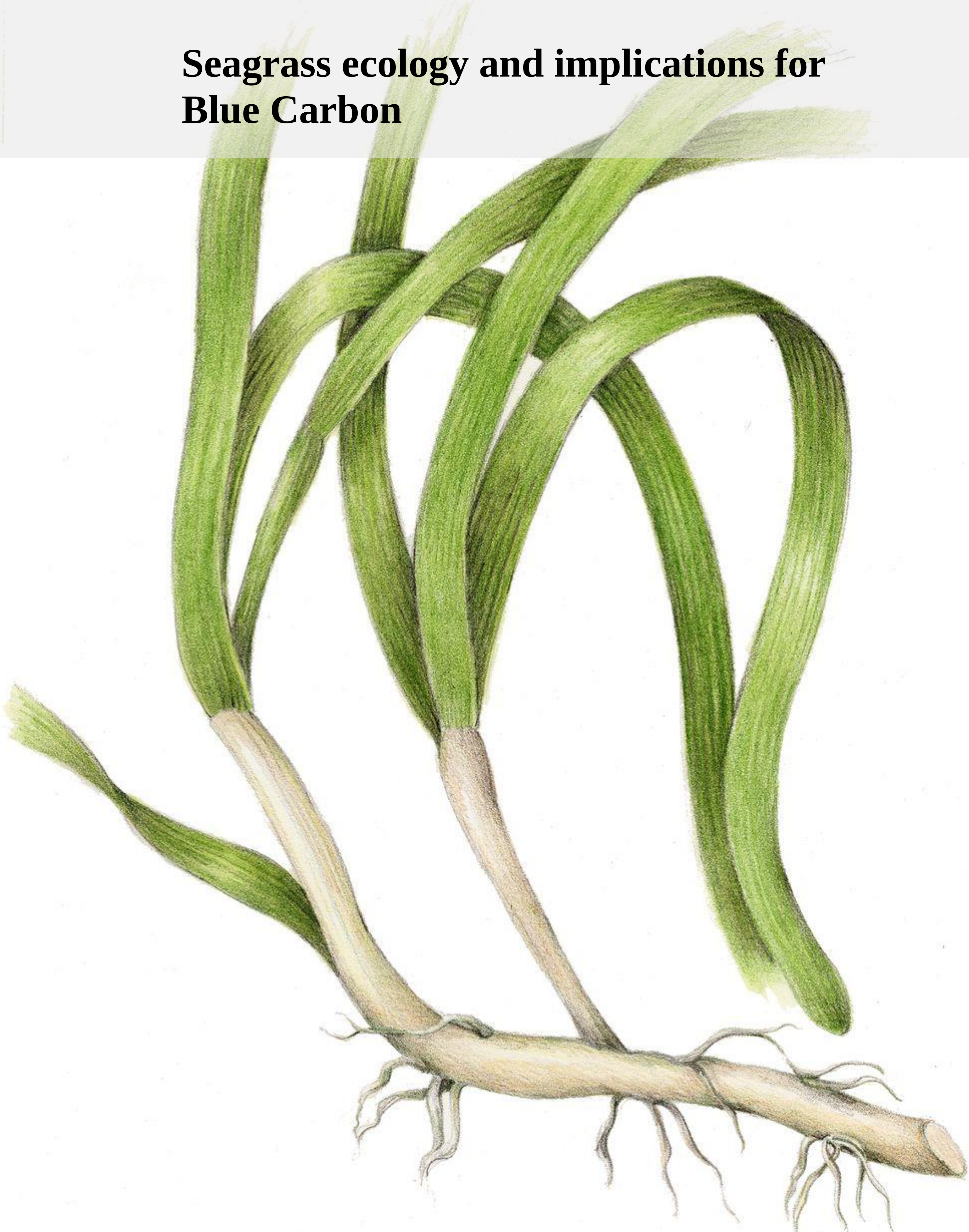
To Ed. For the red pen and arguments during my MSc and for not needing them during my PhD. For telling me when I'm working too hard, and not working hard enough. For reminding me of the achievement at the end and for always having my back. And to me. For my unyielding drive and determination. For designing, implementing and fulfilling a PhD that I have loved, and which provides impactful data for our British seagrass community.

Contents

Chapter 1

Seagrass ecology and implications for Blue Carbon



Seagrass

Ecological description of seagrass

Seagrasses are a mixed group of the only marine flowering plants, or *Angiospermae*, existing fully submerged in marine and estuarine environments (Green and Short, 2003). Evolving around 100 million years ago, seagrasses are now one of the most widespread coastal ecosystems, found fringing sheltered coastlines in all but the most polar seas (Hartog, 1970; Green and Short, 2003) (Fig. 1). To date 60 species of seagrass have been described, with 12 genera found across five families (*Hydrocharitaceae, Cymodoceaceae, Posidoniaceae, Zosteraceae, Ruppiaceae*) (Hartog, 1970; Green and Short, 2003).

Figure 1. Global distribution of seagrass (shaded green). From: UNEP-Grind Arendal.

There is a wide range of morphological diversity among species (Fig. 2). The smallest (*Halophila ovials*) has paddle-like leaves growing only 1cm in length, whilst the largest (*Zostera caulescens*) has long blade-like leaves that can grow over 7m in length (Aioi *et al.*, 1998; Green and Short, 2003). Typically, seagrasses inhabit sheltered coastal waters on sandy and muddy substrates, up to depths of around 10m (Green and Short, 2003). However, some occupy rocky beds, where plants extrude between rocks, and some specialist species are found at depths of up to 58m. (Green and Short, 2003).

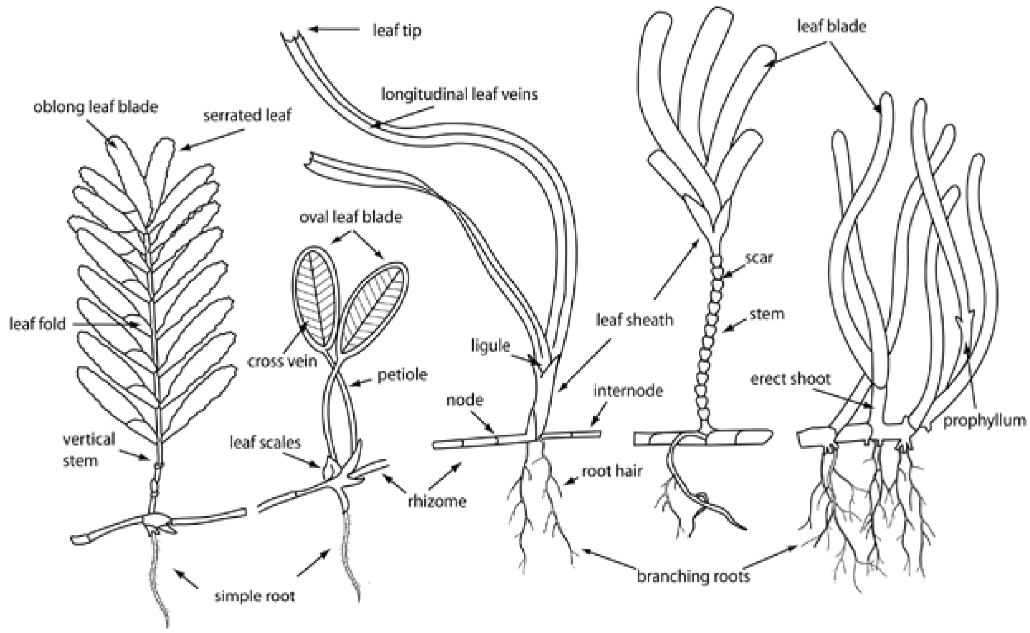


Figure 2. Morphological features of the main seagrass taxonomic group (Mckenzie, 2008)

Seagrass plants are rhizotomous or clonal in growth; modular rhizomes branch away into the sediments, sending up new shoots that are genetic clones of the original plant (Fig. 2). Some meadows can expand as dense uninterrupted beds over 10,000km^2, others, often stunted by natural or anthropogenic pressures, form patchily (Hemminga and Duarte 2000b). Because of their rhizomatous growth, meadows are typically limited in genetic diversity (Arnaud-Haond *et al*. 2012; Hemminga and Duarte 2000a; Hemminga & Duarte 2000b). This homogeneity can affect long term adaptability and resilience, which makes them susceptible to disease (Sinclair *et al.* 2016). As *Angiospermae* they can flower and produce seeds and, although rare, this practice helps to increase genetic diversity (Marbà and Walker 1999; Diaz-Almela *et al.* 2007).

Meadows expand sub- and intertidally and can be monospecific or contain multiple species (Hemminga and Duarte 2000a; Hogarth 2007; Arber 2010). Where multiple species occur, one commonly dominates, meaning species evenness is typically low (Hemminga and Duarte 2000a). Like most plants, seagrass expansion is restricted by light and nutrient availability, with photosynthesis requiring average underwater irradiance of ~11% (Duarte, 1990, 2002), some of the highest levels of light of any marine flora (Orth *et al.* 2006). They are, therefore, highly sensitive and vulnerable to natural and anthropogenic disturbance that affects turbidity or limits light (Orth *et al.* 2006; Cullen-Unsworth *et al.* 2014).

Seagrass ecosystem services and threats

Seagrasses are a foundation species, regulating population and community dynamics and supporting ecosystem processes (Hughes *et al.*, 2009). Seagrass can modify the surrounding environment and the meadows they create form important ecosystems that influence the structure and function of associated estuarine and oceanic communities (Heck *et al.*, 2008; Fourqurean *et al.*, 2012). The structural complexity created by these plants modifies physical habitat features, which subsequently alters water flow and sediment movement. This creates an environment for fauna to find refuge, allows processing of sediment and nutrients and protects coastlines from erosion (Fig. 3).

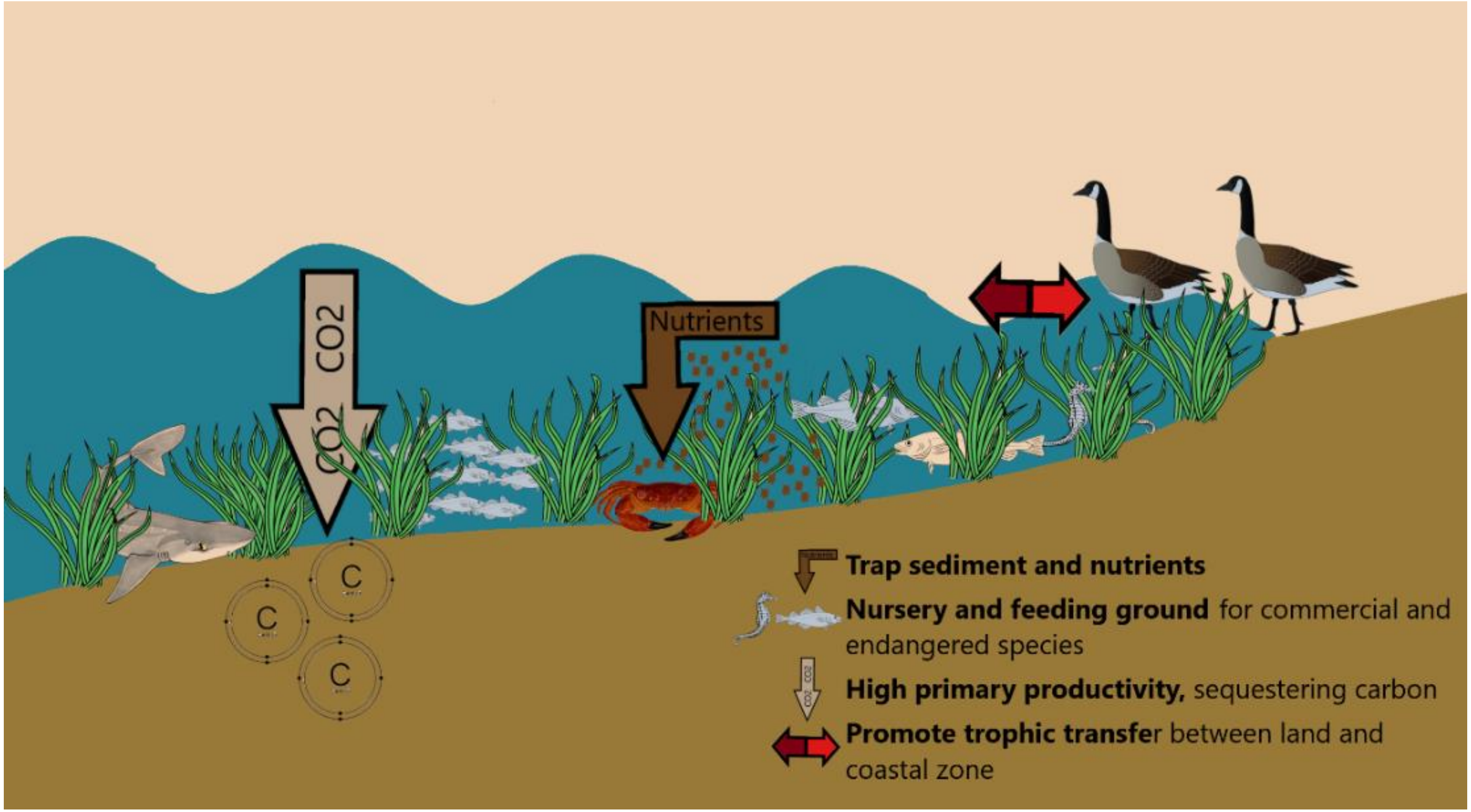


Figure 3. Seagrass ecosystem services in temperate regions. Image made by author in Paint 3D.

Habitat formation and trophic transfer

Seagrass meadows are varied in structure making them suitable habitat for many types of species, increasing species richness. In terms of overall animal densities, a single acre of seagrass can support up to 40,000 fish and over 50 million invertebrates (Miththapala, 2008). Seagrasses complex above ground foliage supports this biodiversity by creating safe refuge for small and juvenile epibenthic species and offering substrate for epifaunal species to attach to (Edgar and Shaw, 1995; Nakamura and Sano, 2004). Further, meadows' complex root systems provide stable habitat for infaunal species (Hemminga and Duarte 2000a; Hemminga and Duarte 2000b). The collection of invertebrates found on the leaves of seagrass, along with the smaller crustaceans and fish that use their canopy for shelter, attracts larger creatures who use the ecosystem to hunt (Jackson *et al.*, 2001). Other species feed directly on the grass itself; turtles, manatees, dugongs and a plethora of waterfowl are meadow grazers and seagrass forms part of their primary habitat (Fox, 1996; Coles *et al.*, 2011; Bertelli and Unsworth, 2014). Seahorses and other *Syngnathidae* also inhabit seagrass and evidence of this has been exploited by conservationists as lobbying power for seagrass protection, to varied success (Shokri *et al.*, 2009; Garrick-Maidment *et al.*, 2010). This mix of residential and migratory species, and the creation of a haven for spawning and juvenile fish, supports both artisanal and commercial fisheries and provides a large subsidy to near and distant marine and terrestrial environments (Heck *et al.*, 2008). There is direct evidence of species migration to coral reefs and pelagic fisheries (Mcmahon *et al.* 2012.). In the British Isles nine species of commercially important fish are found in abundance in seagrass habitats, providing significant long-term benefit to British fisheries (Coles *et al.*, 2011; Bertelli and Unsworth, 2014).

Sediment stabilisation and carbon sequestration

Seagrass meadows stabilise marine substrates. They have deep root structures that extend vertically, anchoring them unto the seabed. Roots bind sediments in underground 'mats', preventing resuspension and erosion (Short and Wyllie-Echeverria 1996; Hogarth 2007; Miththapala, 2008; Dahl *et al.*, 2016). Their meadow canopies further protect the coastline from erosion by buffering the effects of wave action on the coastal fringe (Duarte *et al*., 2005; Short *et al*., 2011). By slowing the water column sediments are forced to settle and, by binding these, seagrass systems slowly raise the ocean floor, combating sea-level rise (Duarte *et al.,* 2013b). This increased sediment deposition plays a key role in nutrient cycling, and acts as a filter to the wider ocean (Short *et al.,* 2007). Lack of movement within meadows slows re-suspension giving seagrass and its associated epiphyte communities time to absorb the nutrients, contaminants and detrital carbon from the water column, enriching nutrient loading into the system (Duarte *et al.,* 2005; Short, 2007). This process also controls the amount of sediment that intrudes onto neighbouring ecosystems, such as coral reefs, and diffuses freshwater discharge, to the benefit of neighbouring ecosystems (Harborne *et al.* 2006).

Settled sediments contain organic carbon from terrestrial and aquatic systems and are buried, binding this allochthonous carbon from multiple sources (Hemminga and Duarte 2000b; Marba *et al*. 2015; Short *et al*. 2007). Seagrasses also fix autochthonous carbon that is absorbed when they photosynthesise. As productive autotrophic species, seagrasses fix carbon dioxide (CO_2) as organic matter in excess of their needs (Duarte and Cebrih 1996). The excess carbon is transported from the leaves into the roots and rhizomes and is excreted into the surrounding sediments. Here, anaerobic soils with low microbial activity and little disturbance bind carbon on millennial timescales

(Mateo *et al.* 1997). This capacity to sequester carbon has propelled seagrass into focus for potential climate change mitigation strategies (see later).

Declines and threats to seagrass

Like almost all other marine and terrestrial habitats, seagrass meadows are declining at accelerating rates in all areas of the globe (Duarte *et al.* 2013a). An estimated 29% of seagrass has been lost globally since the 1980s (Short *et al.*, 2010; Waycott *et al.*, 2009). The rate of loss is estimated at being between 1.5% and 7% per year, globally (Waycott *et al.* 2009). Lack of baseline data on the global extent of seagrass means the degree of known declines are likely under-representative (Duarte 2002; Duarte *et al*. 2013b). Estimates of areal extent of seagrass range from 18 to 60 million ha (McLeod *et al.* 2011; Fourqurean *et al*. 2012). However, a recent attempt to rationalise and update the existing datasets for seagrass global extent suggests the actual figure is around 29 million ha (McKenzie *et al*, 2019 in review). The range of these estimates reflects huge gaps in knowledge. Even in developed countries such as those of the British Isles, where monitoring and mapping of the natural environment is commonplace, accurate estimates of seagrass extent do not exist.

Inhabiting shallow coastal waters that fringe human-occupied coastlines, seagrass is profoundly affected by human activities (Orth *et al.* 2006). Natural threats also occur, but anthropogenic degradation greatly alters their ability to cope with natural disturbances. One of the most substantial threats and, in some areas, the primary cause of seagrass loss is linked to turbidity and reduction of water clarity (Short and Wyllie-Echeverria 1996; Duarte *et al.* 2013). That indirect terrestrial activity is often the source of reduced irradiance complicates conservation strategies (Short and Wyllie-Echeverria, 1996). Rapid use expansion in coastal areas, such as development, off- and near-shore mining, agricultural land use and deforestation directly impacts silt loads and water clarity (Bach, 1998). With burgeoning populations, these activities are likely to intensify, especially in developing areas. Local terrestrial practices, such as farming run-off and land use, are rarely considered when implementing protected areas (Giakoumi *et al.*, 2015), but watershed intrusion directly links these practices to coastal habitat health. Alongside these indirect threats, dredging, land filling and reclamation, dock and jetty construction, fishing activities, aquaculture and propeller and anchor damage can all directly damage and reduce the health of seagrass meadows (Short *et al*., 2007; Waycott *et al.*, 2009).

Alongside anthropogenic disturbances seagrasses are susceptible to damage by storms and their genetic uniformity makes them particularly vulnerable to disease. In the 1930's 'Wasting Disease' spread across the North Atlantic, decimating seagrass populations (Green and Short, 2003). The pathogen *Labyrinthula zosterae* presents as small black patches on the leaves, which eventually become completely black and die (Short *et al.*, 1988). Reports claimed an almost total disappearance of seagrass within two years of the first records of *Labyrinthula* on *Zostera* leaves (Short *et al.*, 1988; Muehlstein, 1989). However, much of the documentation over this period is anecdotal and without accurate data of where seagrass occurred prior to this event, it is difficult to understand the full extent of these disease-driven declines. Regardless, supporting regrowth has been problematic as often meadows that show the strongest recovery are found in sheltered bays, also favoured by yachters, where anchoring and mooring can damage seagrass (Green and Short 2003). Further, replanting seagrass meadows has not had the successes that were hoped for, although success stories are beginning to emerge (Matheson, 2015; NOAA, 2019).

Zostera species in the British Isles

The British Isles contains two species of seagrass; *Zostera marina* and *Zostera noltii*, which form the focus of this thesis. Both species are from the family *Zosteraceae* and are a monocotyledonous angiosperm in the Class *Alismatidae*, Order *Najadales* (Wilkinson and Wood, 2003). They are commonly known as eelgrass, or dwarf eelgrass in the case of *Z. noltii*, and have previously been referred to as grass wrack or widgeon grass in the British Isles (Butcher, 1934). The plants have distinct morphological differences (Tubbs and Tubbs, 1983a). *Zostera marina* is predominantly sublittoral, typically found between low tide and up to 10m depth, although occasionally penetrating the intertidal fringe (Hartog, 1970; Wilkinson and Wood, 2003). It forms ribbon-shaped leaves that typically grow between 20-60cm long, but have been recorded up to 3m in length (Butcher, 1934; Hartog, 1970; Fonseca and Cahalan, 1992; Paul *et al.,* 2011). This subtidal species forms rhizomes up to 10mm wide that extend as deep as 12cm into the sediment (Tubbs and Tubbs, 1983). *Zostera noltii* is much smaller and occurs intertidally, with leaves typically measuring 6-12mm in length and 1mm in width (Wilkinson and Wood, 2003). Its rhizomes similarly, are much smaller, with widths up to 2.5mm, buried up to 8cm into the substrate (Tubbs and Tubbs, 1983).

Sublittoral *Zostera* meadows are some of the most productive shallow water coastal ecosystems (Davidson and Hughes, 1998). They can be annual or perennial but are predominantly perennial in Britain (Foden and Brazier, 2007). Leaf growth occurs in spring and summer, and the leaves that shed in autumn are commonly replaced with smaller winter leaves (Davidson and Hughes, 1998; Foden and Brazier, 2007). Although technically flowering plants, in Britain *Zostera* spp. tend to maintain populations through non-sexual rhizotomous reproduction, with sexual production taking a limited role (Davidson and Hughes, 1998; Wilkinson and Wood, 2003). When it occurs, seed dispersal is in late summer or early autumn (Probert and Brenchley, 1999; Potouroglou *et al.,* 2014).

In Britain, knowledge of where seagrass occurs is, at best, patchy. It is found in muddy or sandy environments in estuaries, lagoons and along the coast both tidally and intertidally (Butcher, 1934). British seagrass is considered to have a much reduced extent compared to the rest of Europe (Foden and Brazier, 2007) and, despite the protection afforded to it by a number of legislative forces (see later), British seagrasses are considered to be in a perilous state and largely degraded (Jones and Unsworth, 2016).

Conservation efforts

Conservation efforts for seagrass are not straight forward. Moderating activities related to human development is challenging. Further, seagrass meadows often occur in areas with high recreational use, complicating implementation of conservation protocols.

Recent studies have suggested prioritising sites not affected by terrestrial practices, in favour of those with simpler to manage threats, such as trawling or boat anchoring (Giakoumi *et al.*, 2015). However, even these seemingly easier to manage restrictions invite their own challenges. Arguments for conserving seagrass meadows in Britain have frequently taken a flagship species approach (Simberloff, 1998; Garrick-Maidment *et al.,* 2010), which often fails to entice resources users and can even invoke hostility between users and conservationists (Shokri *et al.,* 2009). The result of this in the British Isles is hostile resistance to protecting seagrass from damages caused, for instance, by anchoring. Despite many sites being protected by numerous legislative forces (see below) almost none have restrictions related to anchoring or mooring in place.

Both *Zostera* spp. are protected under the EU Habitats Directive (92/43/EEC) as features included in five Annex 1 habitats (Table 1). Comparatively, *Posidonia oceanica*, found predominantly in the Mediterranean, is a priority species under Annex 1 and receives direct protection as a result. If the status of seagrass is as dire as assumed in the British Isles, there would be reason to argue for protecting British *Zostera* spp. in the same way. In its current state the directive does not directly protect *Zostera* within these habitats but accords for designation as Special Areas of Conservation (SAC) for these features (Jackson *et al.*, 2016). Special Area of Conservation designation requires "necessary conservation measures to be applied for the maintenance or restoration, at a favourable conservation status" (EU Commission, 2012; p.1). The status and trends of the species and habitats for which sites are protected must be reported on every six years (EU Commission, 2012). Many seagrass meadows are within sites protected by SAC status; though, seagrass is often not a named feature of designation and, therefore, receives little direct conservation management (see Chapter 2). In such cases reporting on seagrass status every six years is not required.

Both species are also indicators of Good Ecological Status under the EU Water Framework Directive (WFD) (Foden and Brazier, 2007), which requires yearly monitoring of selected sites by the Environment Agency (EA). In addition, *Zostera* spp. gain protection, indirectly, from several other EU Directives, due to their need for good water quality. Under the EU Nitrates Directive (91/676/EEC) declines in *Zostera* is accepted as supporting evidence for eutrophication (EU Commission, 1999). The

Urban Wastewater Treatment Directive (91/271/EEC) aims to reduce harmful runoff to aquatic environments and *Zostera* is an important habitat for wildfowl, so indirectly gains protection under the Birds Directive (79/409/EEC) (Jackson *et al.*, 2016).

Table 1. Annex I habitat for which seagrass is a named feature (EU Commission, 2007).

	Annex I habitat
1110	Sandbanks which are slightly covered by sea water all the time
1130	Estuaries
1140	Mudflats and sandflats not covered by seawater at low tide
1150	Coastal lagoons
1160	Large shallow inlets and bays

Seagrasses can also be a named feature of Marine Conservation Zones (MCZ) in the UK, which are designated under section 116(1) of the Marine and Coastal Access Act 2009 (MCCA), and form the UK's contribution to an international network of protected sites in the northeast Atlantic (DEFRA, 2013, 2019). Generally MCZ's have the conservation objective of each of its designated features being maintained or recovered to a favourable condition (DEFRA, 2013). However, guidelines are commonly vague on how returning or maintaining favourable conditions should be achieved. The vagueness of many of these legislative forces mean that management practices that directly support rejuvenation of seagrass meadows are rarely implemented.

Despite these legislative measures, there is insufficient monitoring or mapping of seagrass meadows of the British Isles, and there are huge gaps in knowledge of where seagrass occurs. One of the main global challenges of seagrass conservation is that the location and status of many seagrass meadows is unknown (Unsworth *et al.*, 2018). In Britain the first step in protecting this resource is surely understanding how much occurs and where it is located.

Climate change and natural carbon sinks

In October 2018 the Intergovernmental Panel on Climate Change (IPCC) reported that global warming is likely to surpass the safe limit of 1.5^{o}C between 2030 and 2052 (Masson-Delmotte *et al.*, 2018), spurring international alarm that we have '12 years to limit climate change catastrophe' (Watts, 2018). Human modification of the environment has been influencing the global climate for thousands of years (Ruddiman, 2003) and since the industrial revolution the climate has warmed by approximately 1.0^{o}C (Masson-Delmotte *et al.*, 2018). In May 2019 a record 414.7 parts per million (ppm) atmospheric CO_2 was recorded, 30% more than pre-industrial levels (280ppm) (NOAA). The last time atmospheric CO_2 concentrations were between 300 and 400ppm was 3 million years ago, when the global temperature was 3^{o}C warmer than pre-industrial, and sea levels were up to 25m higher than today (Jones, 2017). Despite these figures, and this knowledge, greenhouse gas (GHG) emissions continue to rise. In 2018 global CO_2 emissions rose by 2.7% to a record total of 33.1Gt CO_2 (Le Quéré *et al.*, 2018). Without curtailing these trends, the earth could warm as much as 6.8^{o}C by the end of the 21st century (Betts *et al.*, 2011), with catastrophic consequences for humanity. All emissions pathways that limit warming to 1.5^{o}C require carbon dioxide removal from the atmosphere in the order of 100-1000 Gt CO_2 by the end of the century (Masson-Delmotte *et al.*, 2018). The technologies needed to achieve this are still in research and development and it is increasingly recognised that natural CO_2 sinks must play a role in this removal.

Carbon is regulated by three key natural reservoirs: the atmosphere, the oceans, and plant and soils from terrestrial ecosystems. While CO_2 in the atmosphere is part responsible for global warming, terrestrial and ocean ecosystems can reduce this by actively removing CO_2 directly from the atmosphere and fixing it in long-term stores. Although our understanding of the importance of terrestrial sinks is widely understood, the importance of marine vegetated coastal ecosystems in mitigating climate change is less well explored. Despite their global area being orders of magnitude smaller than terrestrial forests the three Blue Carbon habitats (Fig. 4), mangroves, salt marshes, and seagrass, have a much greater proportionate contribution to long-term carbon storage (McLeod *et al.*, 2011). This is, in part, because they can absorb carbon up to 40 times faster (Fig. 5) (McLeod *et al.*, 2011). Despite this, terrestrial stores remain far better documented (Duarte et al. 2008), and in both a historic and contemporary context more research, conservation and policy is focused on their usefulness in mitigating climate change (Waycott *et al.*, 2009; Orth *et al.*, 2006; Duarte, Dennison, Orth, *et al.*, 2008).

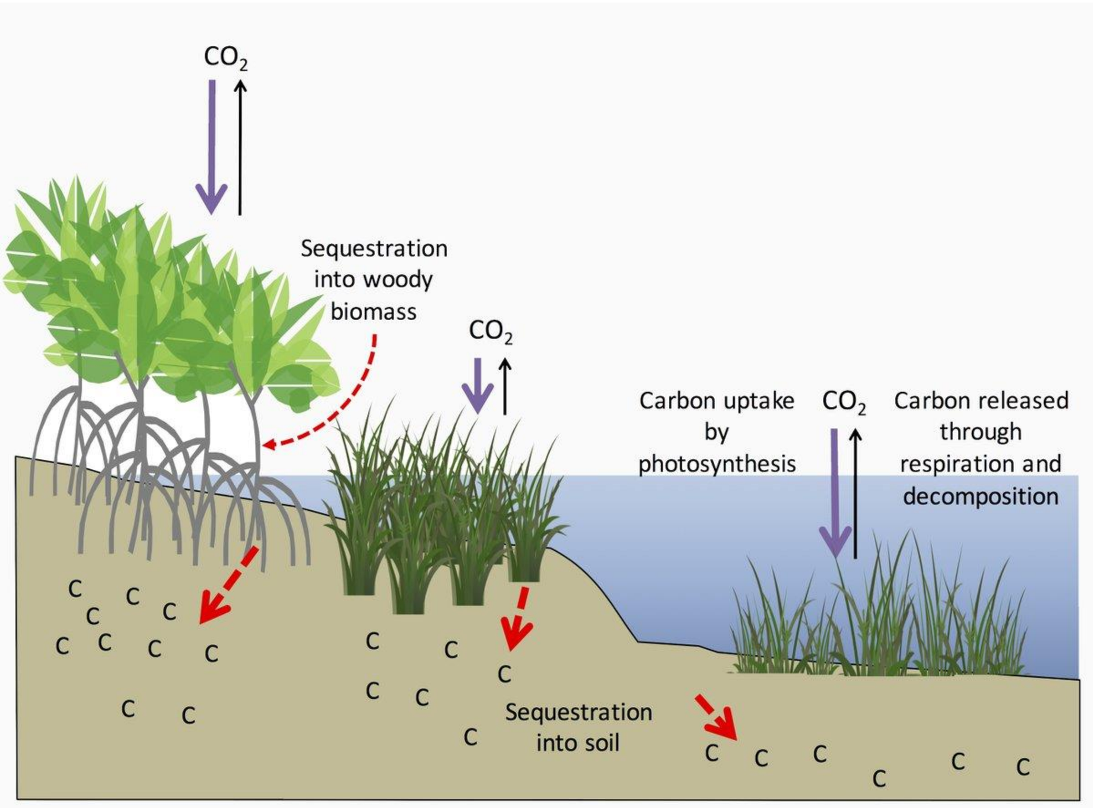


Figure 4. Blue carbon habitats (from left to right: mangroves, tidal marshes and seagrasses). Carbon is taken up via photosynthesis (purple arrow) and is respired back (black arrow) or sequestered into woody biomass and soil (red dash arrow). (Blue Carbon initiative).

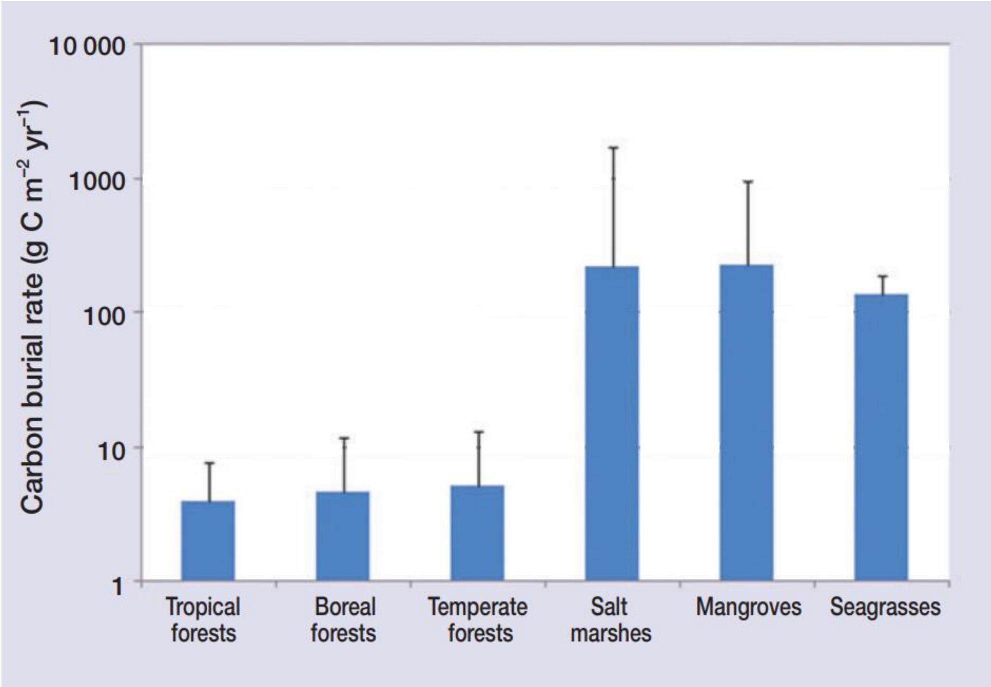


Figure 5. Annual mean carbon sequestration rates per unit area of marine ecosystems vs. terrestrial forests (McLeod *et al*, 2011).

Blue Carbon

The term 'Blue Carbon' was originally coined in a 2009 rapid response assessment, conducted as an inter-agency collaboration between the UN Environment Programme (UNEP), the Food and Agriculture Organisation (FAO), and the UN Educational, Scientific and Cultural Organisation (UNESCO) by GRID-Arendal (Nellemann, 2009). Since then, the term has been utilised to increase awareness of the important contributions coastal ecosystems make to sequestering carbon dioxide from the atmosphere, with seminal papers (McLeod *et al.* 2011; Fourqurean *et al.* 2012, Pendelton *et al.*, 2012) increasing the global awareness of the role of key 'Blue Carbon' habitats.

The ocean plays a significant role in processing CO_2, (Nellemann, 2009). Entering the ocean via a gas exchange CO_2 is processed by either the solubility pump, the downwelling of carbon rich water to the deep seas, or the biological pump, the uptake of CO_2 by marine plants through photosynthesis (Keeling, 1973; Volk and Hoffert, 1985; McLeod *et al.*, 2011). A fraction of the CO_2 that is taken up by marine photosynthesis is buried in the oceans' sediments (Volk and Hoffert, 1985; McLeod *et al.*, 2011). Over 55% of this carbon is fixed via the three Blue Carbon habitats, which collectiely provide significant bio-sequestration capacity, despite occupying only 0.2% of the ocean floor (Nellemann, 2009; Herr, 2012). It it is estimated that the worlds Blue Carbon habiatats absorb an estiamted 235-240 Tg[2] of CO_2 every year, the upper limit of which is 100Tg more than the UK's CO_2 emissions for 2017 (Nellemann *et al.*, 2009; Eaton, 2019).

Blue Carbon habitats are highly productive. They fix CO_2 from the atmosphere and water column in excess of their needs, and store the residual in living above- and below-ground biomass, or sediments (Nellemann *et al.*, 2009) (Fig. 4). Collectively Blue Carbon sinks can be up to three times more effective than terrestrial, and where the latter can bind carbon for decades the former can for millennia (Macreadie *et al.* 2014a; Mateo *et al.* 1997).

Seagrass Blue Carbon

Of all the Blue Carbon habitats seagrass stores the least amount (Fig. 5), though, its share is proportionally significant to its areal extent. Covering less than 0.2% of the ocean floor it accounts for up to 10% of the total carbon buried in ocean sediments (Nellemann *et al.* 2009). Over 90% of the carbon stored by seagrass is stored in their

[2] Common units for carbon sequestration = 1 Gt (gigaton) = 1 Pt (Petagram) = 1,000 Tg (Teragram) = 1,000 Mt (Megaton) = 1,000,000,000 Mg (Megagram) = 1 t (ton)

sediments (Fourqurean *et al.*, 2012) (Fig. 5). On average 50% of this is derived from photosynthesis by the seagrass itself (autochthonous), and 50% from other sources of carbon that become trapped in their canopies and are absorbed into their anoxic sediments (allochthonous) (Kennedy *et al.*, 2010). A global assessment of 946 sampling locations estimated that on average 2.51 ± 0.49 Mg C ha is stored in the living biomass (roots and rhizomes) of seagrass compared to 194.2 ± 20.2 Mg C ha in its sediments (Fourqurean *et al.*, 2012). They estimate 19.9 Pg C is stored in the top 1m of the worlds' seagrass sediments, equivalent to the global fossil fuel and cement production in 2014 (Fourqurean *et al.*, 2012; Kennedy *et al.*, 2010; Kerr, 2017).

Although these global averages are useful, they contain substantial biases. The Mediterranean and North Atlantic comprise almost 60% of the total data (with only one data point from the British Isles), with over 30% coming from direct measurements of *Posidonia oceanica* (Fourqurean *et al.*, 2012). This species stores up to 70 times more carbon than tropical forests (Laffoley 2009) but its average carbon store per hectare is over six times that of the global median for all other seagrass species (Fourqurean *et al.*, 2012). The Northwest, Southeast, and Western Pacific contain one data point between them (Table 2). Further, 82% of all these data are from direct measurements of short sediment cores (<30cm) extrapolated to 100cm (Fourqurean *et al.*, 2012), based on rarely tested assumptions of the relationship between carbon and sediment depth.

With such limited available data, this study has been essential in promoting the advancement of seagrass carbon research. The challenge is that limited data means these estimates are biased regionally, and by species, and generalise storage capture trends within sites (Lavery *et al.*, 2013). The urge to rely on generalised data, such as these, to champion seagrass carbon storage can lead to questions of reliability, undermining attempts to support conservation (Johannessen and Macdonald, 2016).

Table 2. Regional variations of carbon stores in seagrass ecosystems. n = number of record (adapted from Fourqurean *et al.*, 2012).

Region	**Living seagrass biomass (MgC ha^{-1})**		**Soil C_{org} (MgC ha^{-1})**	
	n	**Mean ± 95%CI**	**n**	**Mean ± 95%CI**
Northesat Pacific	5	0.97 ± 1.02	1	64.4
Southeast Pacific	0	ND	0	ND
North Atlantic	50	0.85 ± 0.19	24	48.7 ± 14.5
Tropical W. Atlantic	44	0.84 ± 0.17	13	150.9 ± 26.3
Mediterranean	57	7.29 ± 1.52	29	372.4 ± 74.5
South Atlantic	5	1.06 ± 0.51	5	137.0 ± 56.8
Indo-Pacific	47	0.61 ± 0.26	8	23.6 ± 83
Western Pacific	0	ND	0	ND
South Australia	40	2.23 ± 0.63	9	268.3 ± 101.7
Global mean	***251***	***2.51 ± 0.49***	***89***	***194.2 ± 20.2***

Since the categorisation of seagrass into bioregions (Short *et al.*, 2007), it has been tempting to discuss findings within a regional model. However, variations in carbon storage among habitats formed of the same species, in the same region, are known, and often regions contain outliers that are inconsistent with the norm (Lavery *et al.*, 2013; Nordlund *et al.*, 2016). In reality, very little is known about the variation in carbon storage and sources among species, and between the same species whose habitat varies (Green and Short, 2003; Berkström *et al.*, 2012; Lavery *et al.*, 2013; Nordlund *et al.*, 2016; Röhr *et al.*, 2016;).

Policy implications

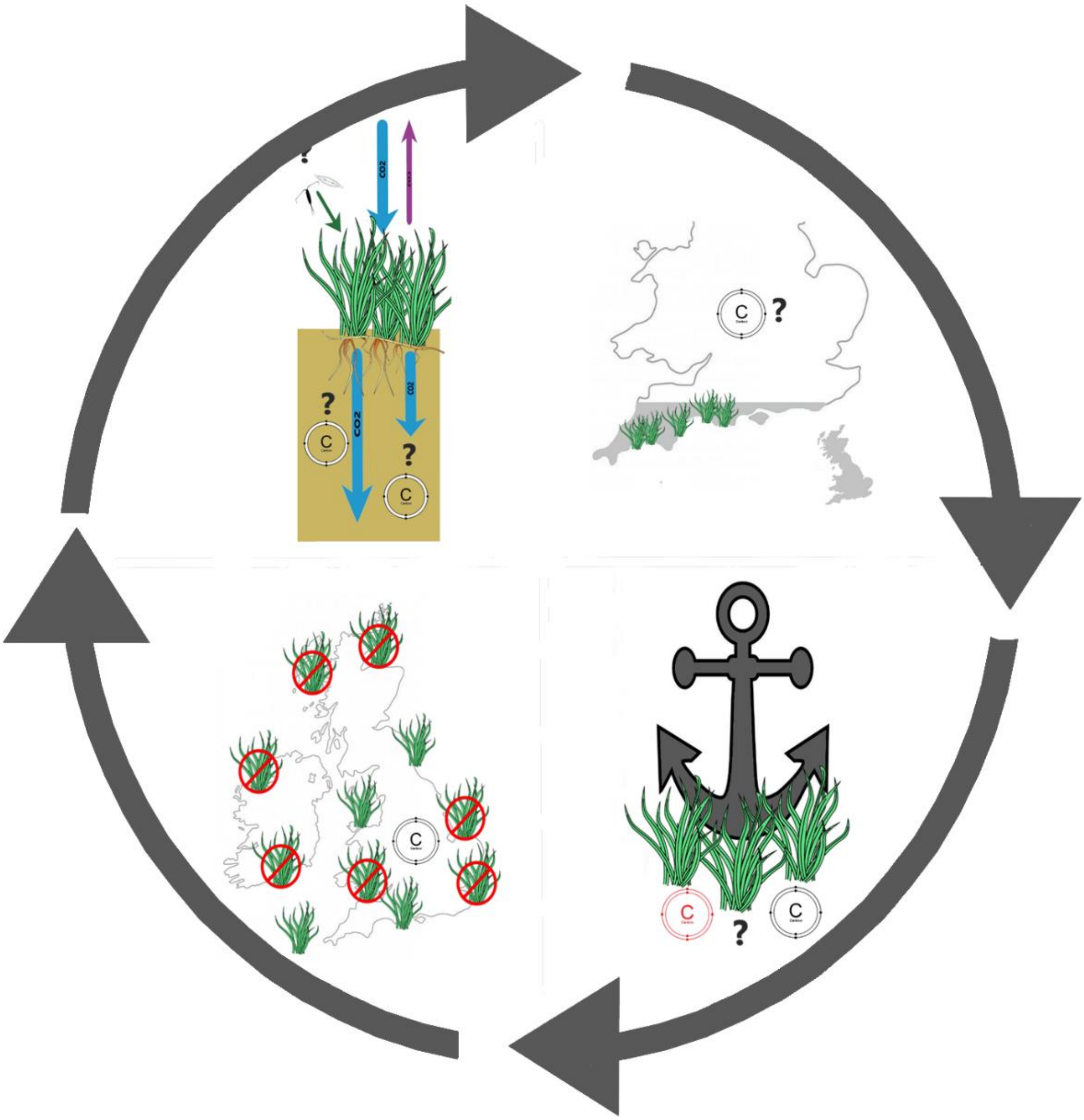
That terrestrial carbon stores could help mitigate climate change has been recognised since the Kyoto Protocol in 1997 (Read *et al.*, 2001). Since 2008, the UN Reduction of Emissions from Deforestation and forest Degradation programme (UN-REDD) has been working to restore, protect, and replant forests to increase natural terrestrial reservoirs. More recently, the International Union for the Conservation of Nature (IUCN) has instigated the Bonn Challenge, pledging to restore 350 million hectares of degraded and deforested land by 2030. Despite recognition in the 1980's that coastal habitats were significant global carbon sinks (Smith 1981), there remains no equivalent recognition of their potential to help mitigate global warming. However, interest is growing and scientists are working to promote seagrass Blue Carbon and its potential role (Nellemann, 2009; Fourqurean *et al.*, 2012; Duarte *et al.*, 2013a; Luisetti *et al.*, 2013).

Under the Paris agreement, countries pledged to outline National Determined Contributions (NDC's) to reduce their emissions, and seagrass is increasingly being named as a nature-based solution within these (Martin *et al.*, 2016). In some countries there have even been moves towards incorporating Blue Carbon storage into domestic climate policy, going so far as to discuss the inclusion of Blue Carbon stocks within GHG Inventories (Bell-James, 2016).

Primary limitations to these attempts include a distinct lack of regional data on seagrass carbon storage capacity, poor understanding of the varying dynamics of carbon storage in seagrass systems, and a paucity of data on the extent, health, and status of the world's seagrass systems. Clearly the next step toward successful integration of Blue Carbon policy is deeper knowledge on the dynamics of seagrass carbon storage, robust estimates of regional seagrass carbon storage and the pressures that threaten them, and knowledge of regional areal extent of seagrass.

Chapter 2

Thesis aims and objectives



The overarching objective of this thesis is to provide an improved foundation of knowledge on seagrass Blue Carbon for the British Isles, from which to better inform local conservation and management. To fulfil this objective, the dynamics and variation in sediment carbon storage from local *Zostera* meadows were assessed in several locations. Subsequently, the impacts that certain common anthropogenic practices have on these stores were evaluated for a single meadow to provide data in support of direct management practices. Finally, an up-to-date analysis on the spatial extent of seagrass in the British Isles is provided, along with estimates of historic loss. The results are placed in the context of conservation and provision of Blue Carbon storage, especially considering the impact of habitat loss on Blue Carbon storage in the British Isles.

This thesis comprises eight chapters, four of which are primary data chapters which encompass the tested hypotheses that work to fulfil the overarching objective. The other four provide an introduction and overview to this work, the common methods used throughout the data chapters, and deliver concluding statements. The following text provides a brief description of each chapter and a conceptual map of how the aims of this work is achieved (Fig. 1).

Chapter 1 - Seagrass ecology and implications for Blue Carbon

This chapter provides an ecological description of seagrass and highlights its importance through overview of its ecosystem services and status. It particularly highlights its importance as a Blue Carbon habitat and its contribution to global carbon budgets, framed in the context of climate change mitigation. It aims to be a broad background to the study topic and a critical review of literature.

Chapter 2 – Thesis aims and objectives

This chapter provides overview of the entire thesis and each chapter within, and acts as a reference to signpost major advancements and results in the subsequent chapters. Specific hypothesis tested in each of the data chapters are presented.

Chapter 3 – Materials and methods

This chapter outlines the study locations and common methods utilised within this thesis. This includes the development of novel coring techniques for underwater sediment extraction using SCUBA gear, and carbon stock assessment techniques. Subsequent chapters build on and refer to the materials presented in this chapter.

Chapter 4 - Carbon dynamics of *Zostera marina* sediments in the Fleet lagoon reveals inconsistencies in globally extrapolated data

This chapter describes a study of the carbon dynamics in the Fleet lagoon. The chapter provides novel data on the sedimentation rate and age of accumulated carbon, as well as the sources of organic carbon within the Fleet's seagrass sediment. Further, it builds on the current dearth of global knowledge on how carbon storage changes along a sediment depth profile.

The central aim is to analyse shallow vs. deep sediment profiles to test assumptions of sediment carbon relationship with depth. The chapter also aims to provide context for global estimates while presenting details of this unique site within the UK.

The tested hypothesis of this chapter was:

Hypothesis: Extrapolating carbon stocks from a 30cm depth profile to a 100cm depth profile underestimates the total carbon stored within the Fleets seagrass meadow.

This was tested by comparing sediment carbon stocks from 30cm cores extrapolated to 100cm, and 100cm cores.

Chapter 5 – Variability of British Isles seagrass sediment carbon: implications for Blue Carbon estimates and marine conservation management

This chapter provides estimates of organic carbon density from 13 seagrass meadows to assess how British Isles seagrass meadows vary in their carbon storage capacity and whether these follow comparative regional trends. The objective was to obtain local estimates for carbon storage in British Isles seagrass meadows to: 1) understand the variability of sediment carbon storage; 2) assess the impact of habitat variability on sediment carbon storage and; 3) compare local carbon storage trends with global and regional data. The data was used to help elucidate the significance of the British Isles seagrass sediments in terms of Blue Carbon value. The tested hypotheses of this chapter are:

Hypothesis 1: There are significant differences in British Isles seagrass sediment organic carbon density.

This was tested by extracting sediment cores and analysing carbon content from 13 seagrass meadows along the southwest coast of England.

Hypothesis 2: Aboveground biomass and sediment silt content significantly impact total carbon storage.

This was tested by documenting number of seagrass plants per m^2 and sediment silt content from the 13 surveyed meadows and conducting regression analyses to determine relationship.

Hypothesis 3: Local seagrass sediment carbon data reveals inconsistencies in regional seagrass sediment carbon estimates with implication for Blue Carbon schemes and seagrass conservation.

This was tested by comparing average carbon stocks from regionally extrapolated data and comparing stock estimates from primary data and data taken from the literature.

The data from this chapter was edited for publication and has been published in *Plos One*: Green, A., Chadwick, M. A. and Jones, P. J. S. (2018) 'Variability of UK seagrass sediment carbon: Implications for Blue Carbon estimates and marine conservation management', Plos One, 13(9) (Appendix 1).

Chapter 6 - Anchoring and mooring reduces carbon storage within *Zostera marina* sediments in Studland Bay, UK

This chapter describes a study of the impacts of mooring and anchoring in the seagrass meadow at Studland Bay. It aims to document the impact of these activities to help inform management practices and conservation for Studland Bay and other seagrass meadows where anchoring and mooring activities occur. The tested hypothesis of this chapter was:

Hypothesis: Sediments from within the seagrass meadow at Studland Bay contain more organic carbon than in-meadow bare patches created by mooring and anchoring, and adjacent bare patches that never knowingly contained seagrass.

This was tested by comparing the carbon content of extracted sediment from four habitat conditions: seagrass meadow; anchor scars; mooring scars and; bare sand.

The data from this chapter has been edited for publication and is under review at *Aquatic Conservation: Marine and Freshwater Ecosytems*.

Chapter 7 - Historic analysis exposes catastrophic seagrass loss in the British Isles, with implications for climate change mitigation

This chapter provides the most up-to-date and accurate analysis of seagrass areal extent in the British Isles. It aims to estimate the current areal extent and percentage loss of seagrass throughout the British Isles. These estimates and loss figures are used to explore the value of seagrass Blue Carbon in Britain, and the consequences of historic seagrass loss.

The data was used to test the following hypotheses:

Hypothesis 1: The current estimates of seagrass areal extent in the British Isles are out of date and inaccurate.

This was tested by collating data from two large public data sets and contacting stakeholders who work with seagrass to collect as much contemporary data on seagrass areal extent in the British Isles as possible. This was supported by a systematic search of literature to obtain additional published areal extent data.

***Hypothesis 2*:** There has been a substantial reduction in the spatial extent of seagrass in the British Isles with significant consequences to the Blue Carbon capacity of this resource.

This was tested by comparing historic (older than 20years) data with contemporary data collected to test ***Hypothesis 1***. This was also supported by a systematic literature search and the production of simple models, which are supported by habitat suitability studies, to provide low, medium, and high confidence estimates of historic seagrass loss.

The data from this chapter has been edited for publication and is currently under review at *Global Change Biology*. Results from this work also appear in various press accounts including the following: BBC News, Sept 2019: ''Wonder plant' will tackle climate change, conservationists say'; The Guardian, Sept 2019: 'The UK's biggest seagrass restoration scheme'; Sky News, Sept 2019: 'British seagrass could help tackle climate change'; The Times, Sept 2019: 'A new seagrass restoration scheme could be used to fight UK emissions'.

Chapter 8 – General discussion and conclusions

This chapter summarises the main findings from the data chapters contained within this thesis and provides general conclusions. Special emphasis is given to conservation and management of British seagrass meadows, in particular the usefulness of seagrass conservation and restoration as a supporting climate change mitigation. Implications for globally extrapolated data are also explored. Finally, this chapter provides critical recommendations for future research projects with the hope that this will support the continued development of seagrass science and conservation within the British Isles.

Chapter 4
Carbon dynamics of *Zostera marina* sediments in the Fleet lagoon reveals inconsistencies in globally extrapolated data

Chapter 5
Variability of British Isles seagrass sediment carbon: implications for Blue Carbon estimates and conservation

Chapter 6
Anchoring and mooring reduces carbon storage within *Zostera marina* sediments in Studland Bay, UK

Chapter 7
Historic analysis exposes catastrophic seagrass loss in British Isles, with implications for climate change mitigation

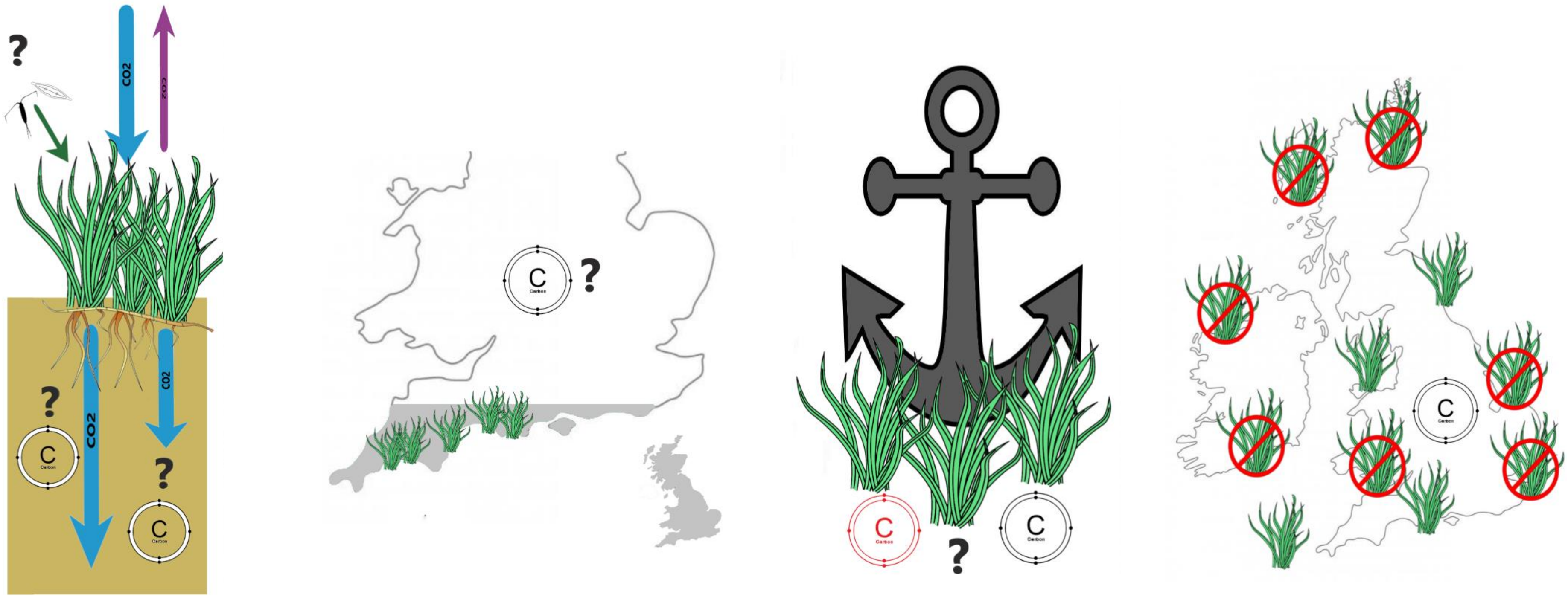
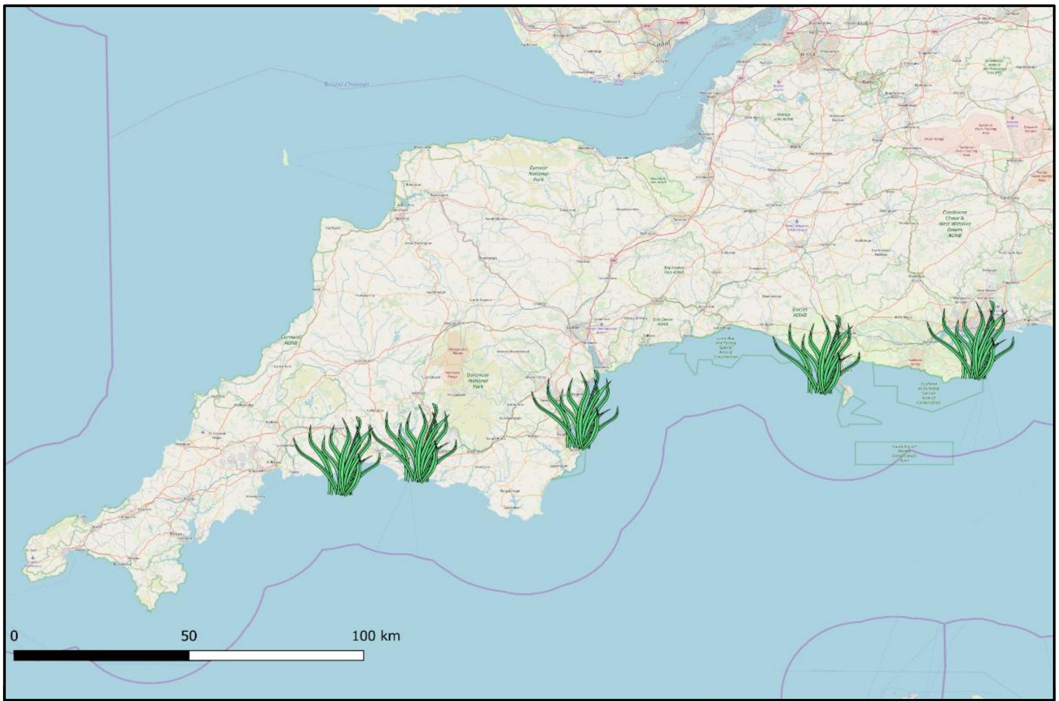


Figure 1. Conceptual diagram outlining the key themes of each empirical chapter from, understanding the dynamics and variances in carbon storage in British Isles seagrass sediments, to documenting how certain anthropogenic practices impact this storage, and documenting contemporary seagrass areal extent and historic loss of seagrass in the British isles, with overall implications for seagrass Blue Carbon storage in the British Isles. Image made by author in 3D Paint

Chapter 3

Materials and methods



Study sites

Cornwall, Devon and Dorset, along the southwest coast of England contain a large proportion of seagrass meadows, which have benefited from comparatively high levels of documentation (Tutin, 1938; Wilson, 1948; Webster *et al.*, 1998; Frost *et al.,* 1999; Scarlett *et al.,* 1999; Bunker *et al.*, 2004; Tweedley *et al.*, 2008; Deamicis, 2012; Daoudi, 2013), compared to the rest of the UK. The meadows that occur along this coastline exhibit varied habitat features. Meadows are found in enclosed lagoons, along estuaries, along developed and rural coastlines, and in the centre of busy harbours, ranging from intertidal to depths of up to 10m. The southwest coast of England, therefore, provides a unique opportunity to assess several seagrass habitats within proximity to one another.

In 2014 the Community Seagrass Initiative (CSI) set up a citizen science project on the back of a £500k Heritage Lottery Fund grant. Their aim was to increase awareness of seagrass in the southwest of England whilst providing research on the ecosystem health of seagrass habitats. Their study sites stretched 191 miles from Looe in Cornwall, to Weymouth in Dorset (CSI, 2019). A collaboration with CSI meant I was able to gain access to the diversity of sites within their research area, many of which required boat access. In addition to these sites, Studland Bay was included in this thesis because it is one of the most contested seagrass meadows in the UK, vying for multiple use from recreational boaters and conservationist alike. In fact, its exclusion from CSI's sites was due to its high profile and the challenging stakeholder relationships that occur here.

In total 13 seagrass meadows are studied within this thesis (Table 1; Fig. 1). The majority of these are located in Plymouth Sound (n=6) (Table 1; Fig. 2), and Torbay (n=4) (Table 1; Fig. 3). Plymouth Sound is a 6,000 ha water body that includes around 3,000 ha of tidal rivers, estuaries, mudflats, lagoons and sand flats (JNCC, 2018). The water body is a recognised Special Area of Conservation (SAC) within which the seagrass meadows are technically protected as an Annex II habitat under 1140 Annex I habitat, ‘mudflats and sandflats not covered by seawater on low tide’ (JNCC, 2018). However, they are not a named feature, and feature 1140, although recognised as a qualifying feature, was not a named reason for SAC site selection (JNCC, 2018). The protection this status affords the seagrass meadows is negligible.

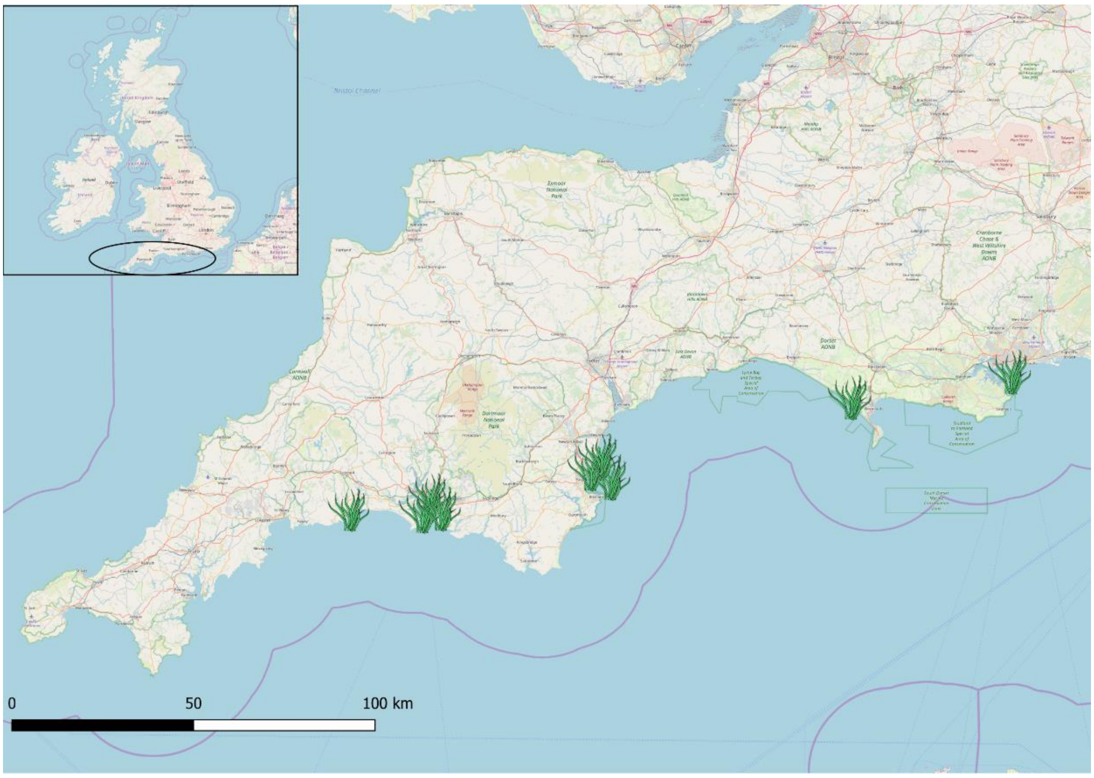


Figure 1. Seagrass meadows studied in the thesis (left to right): Looe, Cornwall; meadows of Plymouth sound and Torbay, Devon and; the Fleet and Studland Bay, Dorset.

Table 1. Details of all study sites included in this thesis, including general site information.

Site	Meadow Location	Exposure	Meadow formation	Ave. meadow depth (m)	Area (ha)	N	W
Cornwall							
Looe	Open coastline	Exposed	Very patchy	6.8	56.52	50° 21' 11.52"	4° 26' 30.48"
Plymouth Sound, Devon							
Cawsands	Open bay	Partly sheltered	Very patchy	6.9	11.77	50° 19' 52.32"	4° 11' 53.52"
Firestone Bay	Open coastline	Exposed	Patchy	5.1	0.76	50° 21' 37.8"	4° 9' 37.44"
Drakes Island	Island coastline	Partly sheltered	Dense	5.9	4.25	50° 21' 25.56"	4° 9' 10.08"
Jennycliff Bay	Open coastline	Exposed	Patchy	5.0	11.77	50° 20' 27.96'	4° 7' 49.08"
Yealm CC	Estuary mouth	Sheltered	Dense	6.3	0.14	50° 18' 36.36"	4° 3' 58.68"
Tomb Rock	Bay	Sheltered	Sparse	5.0	0.15		
Torbay, Devon							
Elbery Cove	Sheltered bay	Sheltered	Sparse	3.4	29.31	50° 24' 17.64'	3° 32' 41.28"
Torre Abbey	Open bay	Very exposed	Very patchy	5.0	104.11	50° 27' 38.52"	3° 32' 1.32"
Fishcombe Cove	Sheltered bay	Very sheltered	Very patchy	3.3	0.23	50° 24' 11.52"	3° 31' 17.76"
Hopes Cove	Bay	Partly sheltered	Gradient	7.7	2.73	50° 27' 52.56"	3° 29' 16.44"
Dorset							
Fleet	Enclosed lagoon	Extremely sheltered	Dense	0.8	274.68	50° 37' 72.20"	2° 33' 43.30"
Studland Bay	Sheltered bay	Very sheltered	Dense	2.5	53.37	50° 38' 34.20"	1° 56' 38.30"

The city of Plymouth, with a population of 234,982, flanks the water. The bay is a natural harbour and includes a naval dockyard and three commercial harbours which are utilised by passenger ferries, fishing and leisure boats and privately owned shipping and leisure craft (CHC, 2019). The harbour contains six seagrass meadows that were studied within this thesis (Fig. 2). Three (Drakes Island, Firestone Bay and Jennycliffe Bay) are in the inner harbour and are reasonably exposed to the heavy boat traffic that frequents the area. Cawsands, the largest site, is located alongside an underdeveloped coastline with reasonable protection from the ocean. Two meadows are at the mouth of the river Yealm (Tomb Rock and Yealm) with good protection from the ocean.

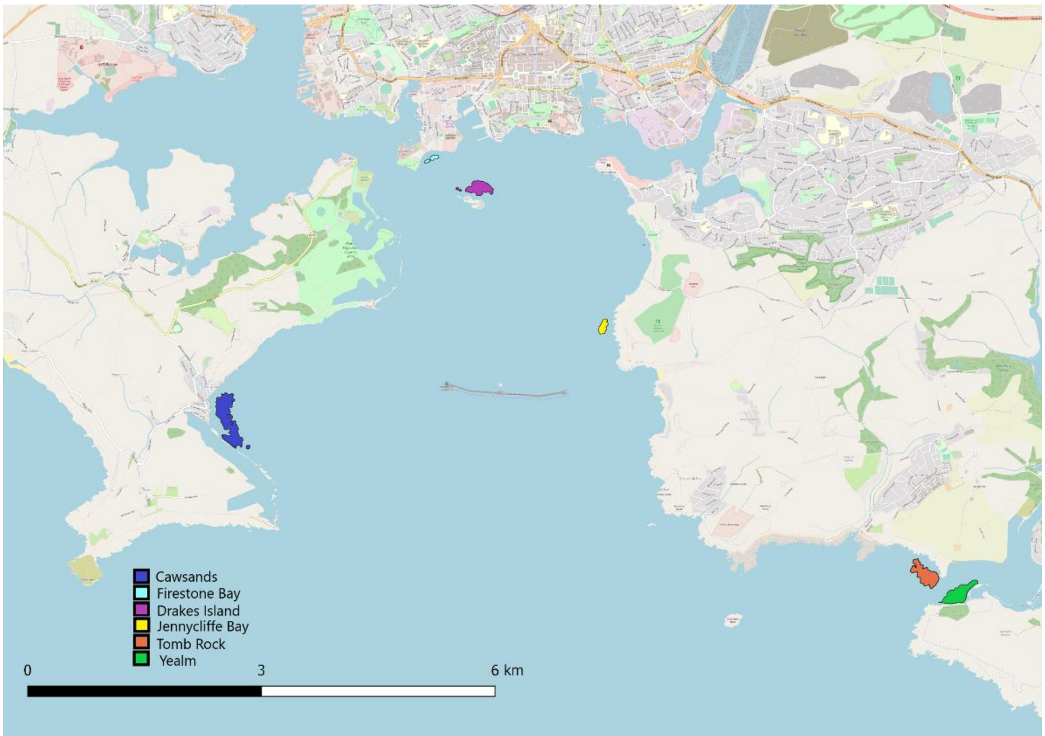


Figure 2. Seagrass meadows from Plymouth Sound, Devon included in this work.

A further four sites included in this work are located in Torbay (Table 1; Fig. 3). Torbay is included in the SAC designation of Lyme Bay, which extends from Torbay to Lyme Regis in Dorset. Neither mudflats nor seagrass are a named feature of the designation. Torbay is 6,287ha and is flanked by Torquay town with a population of 65,245. Torbay's harbours predominantly cater for private and commercial leisure boaters, including pleasure cruises. The sites are all reasonably sheltered and their main differences are size.

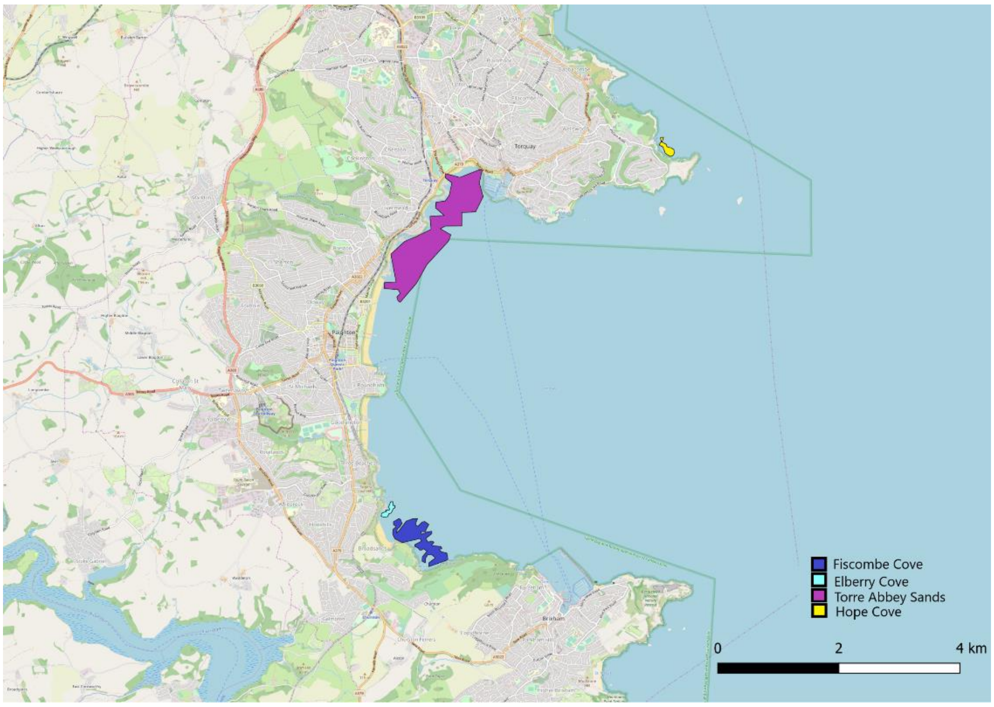


Figure 3. Seagrass sites from Torbay, Devon included in this work.

Three other sites sit outside of these areas; Looe, the Fleet and Studland Bay. The seagrass meadow in Looe, Cornwall (Fig. 4), is located just off the coastline, flanked by the small coastal town with a population of just over 5,000 people. Looe has been an MCZ since 2013 and its seagrass meadows are named as designated features (NE, 2017).

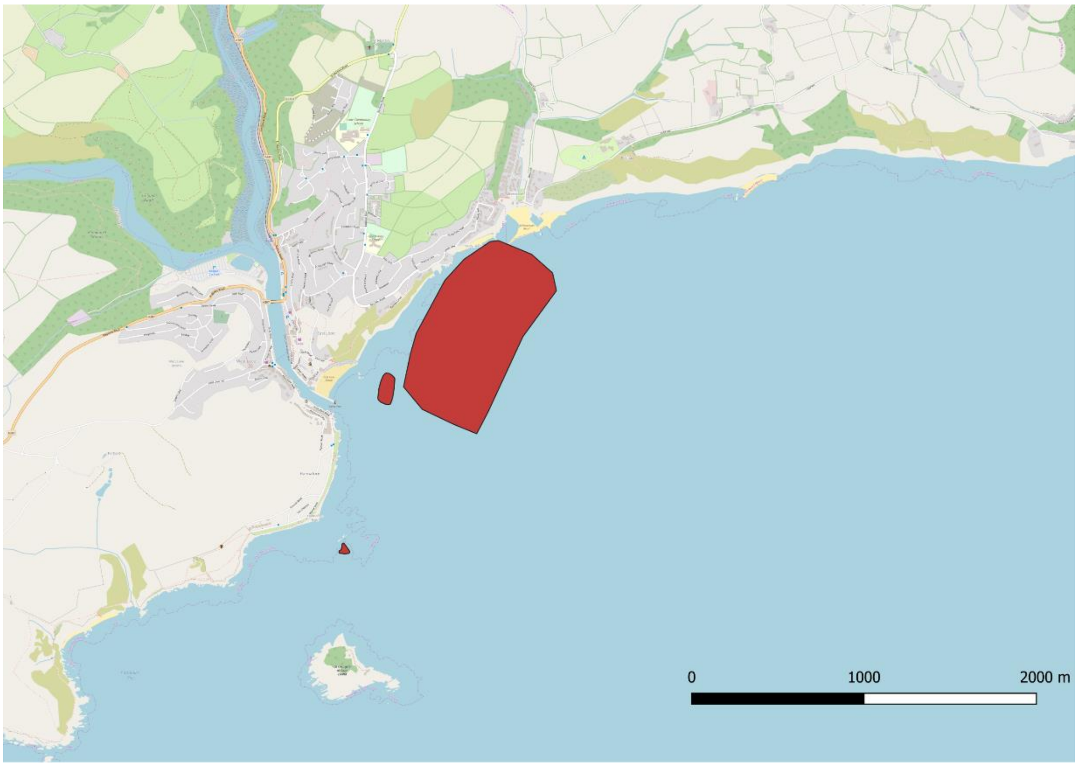


Figure 4. Seagrass meadows highlighted in red in Looe, Corwnall

The Fleet is a 480ha enclosed lagoon (Johnston and Gilliland, 2000), almost entirely cut off from the sea. The Fleet is an SAC but seagrass is not a named Annex II feature (Defra, 2018). The Fleet also gains protection as a Site of Special Scientific Interest (SSSI) a RAMSAR site (Wetlands), a Special Protected Area (SPA) and a UNESCO world Heritage site. It has been property of the Ilchester Estate for 400 years (Johnston and Gilliland, 2000) and does not have the same type of boating pressure as the other sites. Its main pressures come from the fact that it is surrounded by farmland and has a large swannery in its easterly bounds.

Studland Bay in Dorset is a 4,000ha bay surrounded by a national nature reserve managed by the National Trust. It is, therefore, protected from typical pollution encroachment from farmland or that of a town/city. Its main pressure comes from recreational boating activity, as it is a popular anchorage for yachts coming out of Poole harbour. In May 2019 it was designated as an MCZ, with seagrass named as a feature requiring efforts to restore it to favourable condition. What measures will be put in place to do so are as yet unknown.

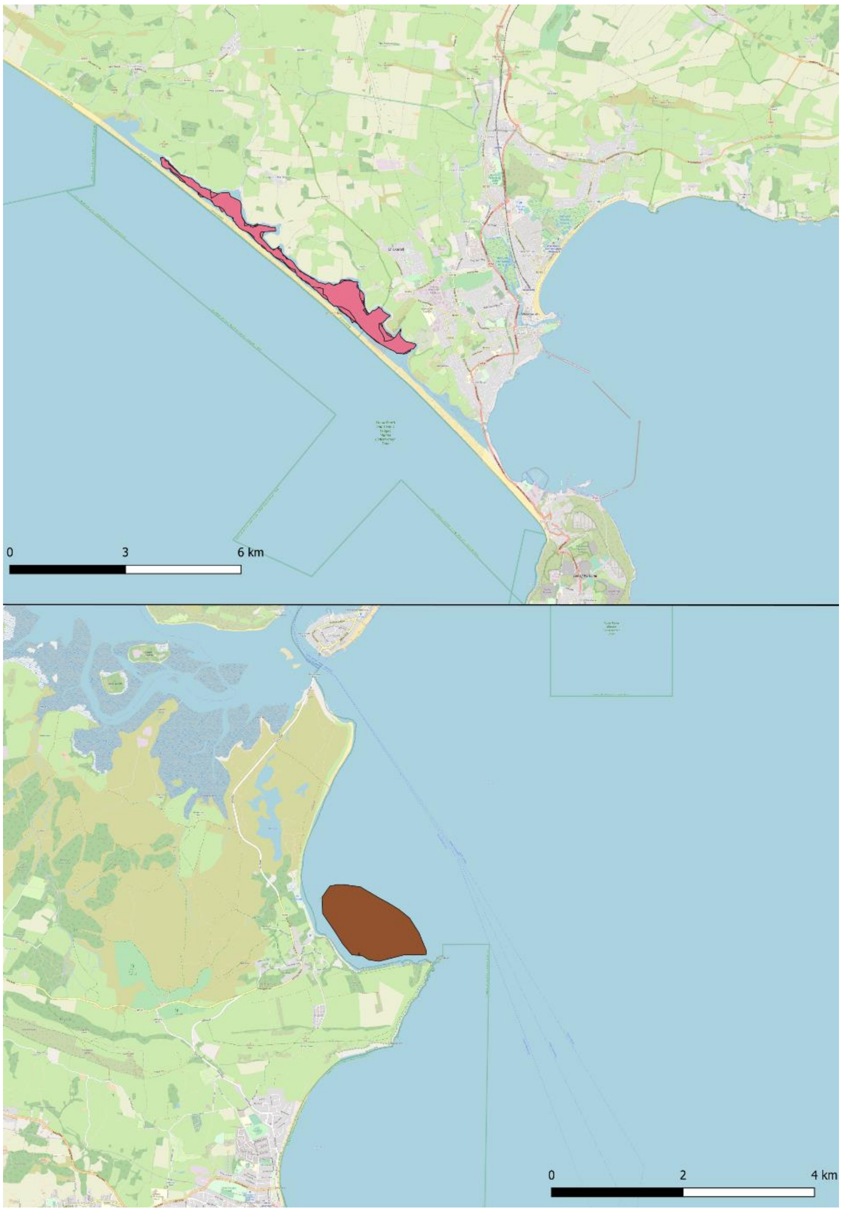


Figure 5. Seagrass meadows at the Fleet (pink, top) and Studland Bay (Brown, bottom) in Dorset.

The Fleet and Studland Bay offered excellent case study opportunities (Chapters 3 and 5 respectively), benefiting from easy site access and exhibiting unique conditions. The Fleet is an enclosed lagoon containing one of the largest seagrass meadows in the UK. It is easily accessible from the shoreline and benefits from being almost entirely excluded from the sea, meaning there are no waves, and is also flanked by a large shingle barrier, meaning protection from coastal winds. It offered an excellent opportunity to trial deep (>100cm) coring techniques, with the hope to roll out the methodology to other sites. Studland Bay is a popular anchorage, with numerous scars within its meadow from both moorings and anchors and offered a unique opportunity to test in-situ the impact of seafloor scoring on carbon storage. Both sites were included, with the remaining 11 sites, in a wider study that assessed the seagrass sediment carbon storage along the southwest coast England (Chapter 4).

Coring techniques

Two distinct methodologies were utilised to extract sediment from the study sites within this thesis, depending on whether short cores (30-40cm) or long cores (100-120cm) were being extracted. The deployment of short cores was reasonably straight forward. Polyvinyl chloride (PVC) tubes of 7cm diameter were cut down to 40cm long cores in the UCL lab. These were pre-sawn and re-sealed using Duct and electric tape. The bottom edges were filed down to create a sharp edge to improve ease of cutting through roots and rhizomes. All sites, apart from the Fleet, required the cores to be deployed underwater using scuba gear. Cores were labelled every 10cm, so it was clear how deep they had been inserted into the sediment whilst underwater (Fig. 6). As a PADI Divemaster I am fully trained on diver safety and adhered to strict regulations to ensure the safety of myself and my diving buddy (Table 2).

Table 2. Equipment and procedures that were used to ensure diver safety

Safety equipment	Use
Safety buoy	Alert boats to diver presence
Dive computer	Keep track of bottom time a decompression limits
Whistle, light, knife	In case of emergency
Dry suit, gloves, hood and under suit	To avoid hypothermia
Oxygen tank and first aid kit	Either located on boat or shore in case of emergency
Charged telephone	Either located on boat or shore in case of emergency
Warm / dry gear	To keep warm between dives
Diving safety procedures	
Familiarise yourself with nearest hyperbaric chamber and transport to access	
Familiarise yourself with local emergency numbers	
Ensure equipment is serviced regularly	
Always dive with a buddy	

When using a boat always ensure someone remains on boat for cover and emergency assistance
Never conduct dives where exit to surface is obstructed (caves etc.)
Always remain within 5m of DECO time
Never dive deeper than 15m
Emergency exit route to shore mapped in case of emergency
Check tides and currents before each dive
Familiarisation with local fauna that could cause injury (stings, bites etc.) and with first aid procedures
Check for bad weather before each dive

Diving in the UK, even in summer months, requires the use of a dry suit, which greatly increases diver buoyancy. To overcome the issues with diver buoyancy when pushing cores into the sediment I ensured I was overweight with lead weights, within the limits of safety. A coring apparatus was designed with the UCL Geography lab staff to assist this process (Fig. 6). The device was made from materials found around the lab. It consisted of two cut pieces of sturdy plastic, one which fit into the centre of the core, to give stability, which was then attached to a thinner wider piece that provided a platform that could withstand sheer stress without damaging the core integrity (Fig. 6). These were attached to a metal pole and two lifting weights weighing 3kg each were inserted over the pole, to be used as a weight to hammer the core into the sediment (fig. 6). In more solid sediment a small sledgehammer was also used to help this process. Once cores were submerged to a minimum of 35cm into the sediment, or as far as they could be hammered in, a rubber bung was inserted into the top of the core and the core was gently extracted from the seafloor. A rubber bung was then placed in the bottom of the core and the cores were stored upright in a mesh bag. Due to the weight of the sediment cores a partially inflated lift-bag was used to keep the cores upright and to reduce the pressure on the divers carrying a heavy weight underwater.

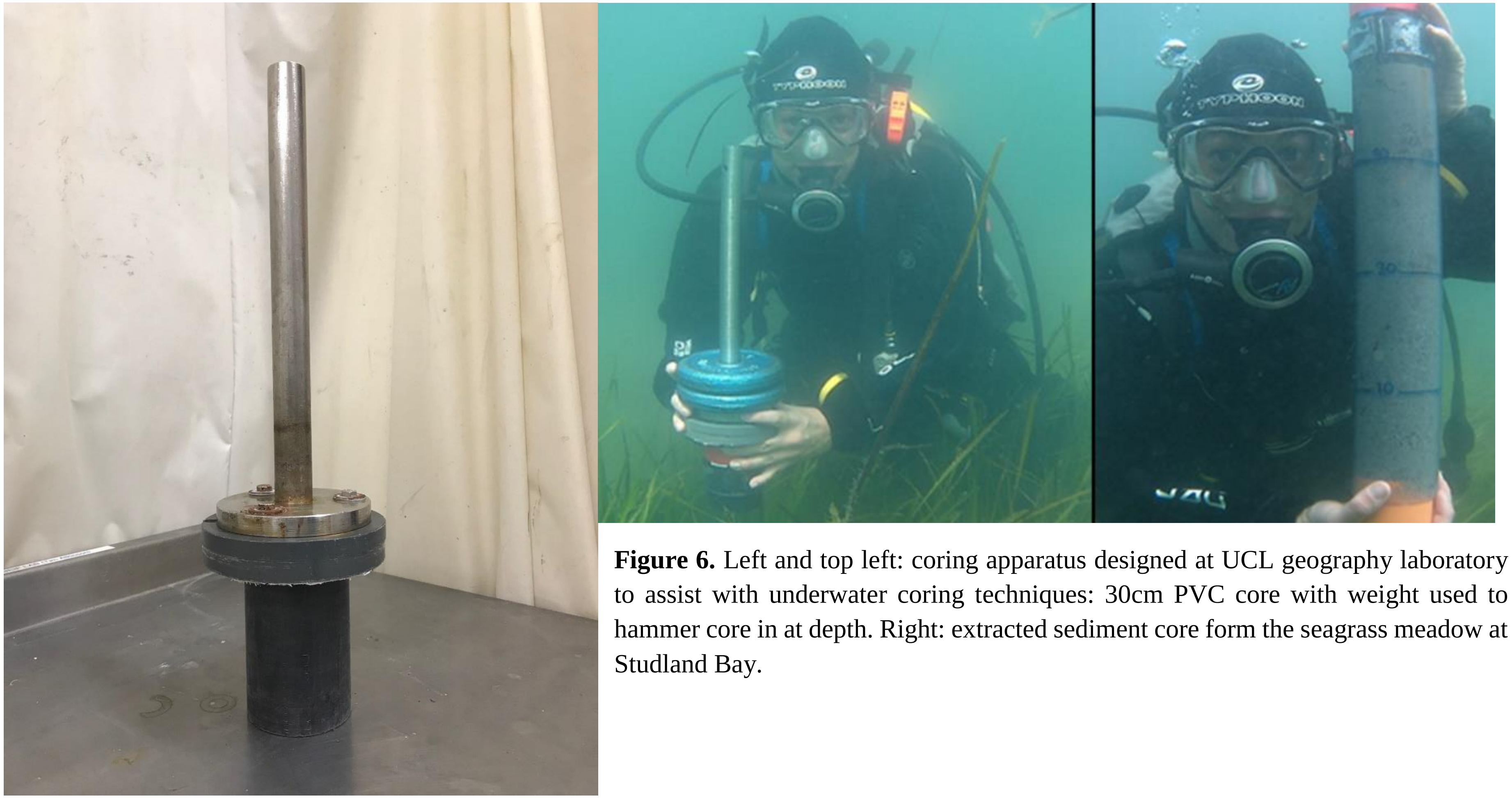


Figure 6. Left and top left: coring apparatus designed at UCL geography laboratory to assist with underwater coring techniques: 30cm PVC core with weight used to hammer core in at depth. Right: extracted sediment core form the seagrass meadow at Studland Bay.

This thesis had the ambition to extract long (>100cm) cores from research sites. For this type of coring I used the Livingston Piston coring technique (Nesje, 1992). This technique is typically utilised in lake and marine sediments. However, the UCL laboratory did not have experience using this method in substrate that has complex detritus, such as the root and rhizome systems found under seagrass meadows. Hammering the shorter cores into the seagrass meadows exerted a substantial amount of sheer stress on to the short PVC cores, at times causing them to rupture. Hammering long (>100cm) cores into the sediment would increase this stress and I decided another stronger material should be used. I decided to use aluminium pipe, which could not be pre-split before insertion. Therefore, I had to develop a methodology that allowed me to split the cores post exhumation. To slice the cores, I made a slicing machine that mimicked the action of a metal pipe cutter, cutting horizontally along the length of the pipe rather than vertically. This was made with materials found around the lab, with cutting wheels inserted into a metal encasement, and rolling ball bearing units acting to guide the cores into place (Fig. 7).

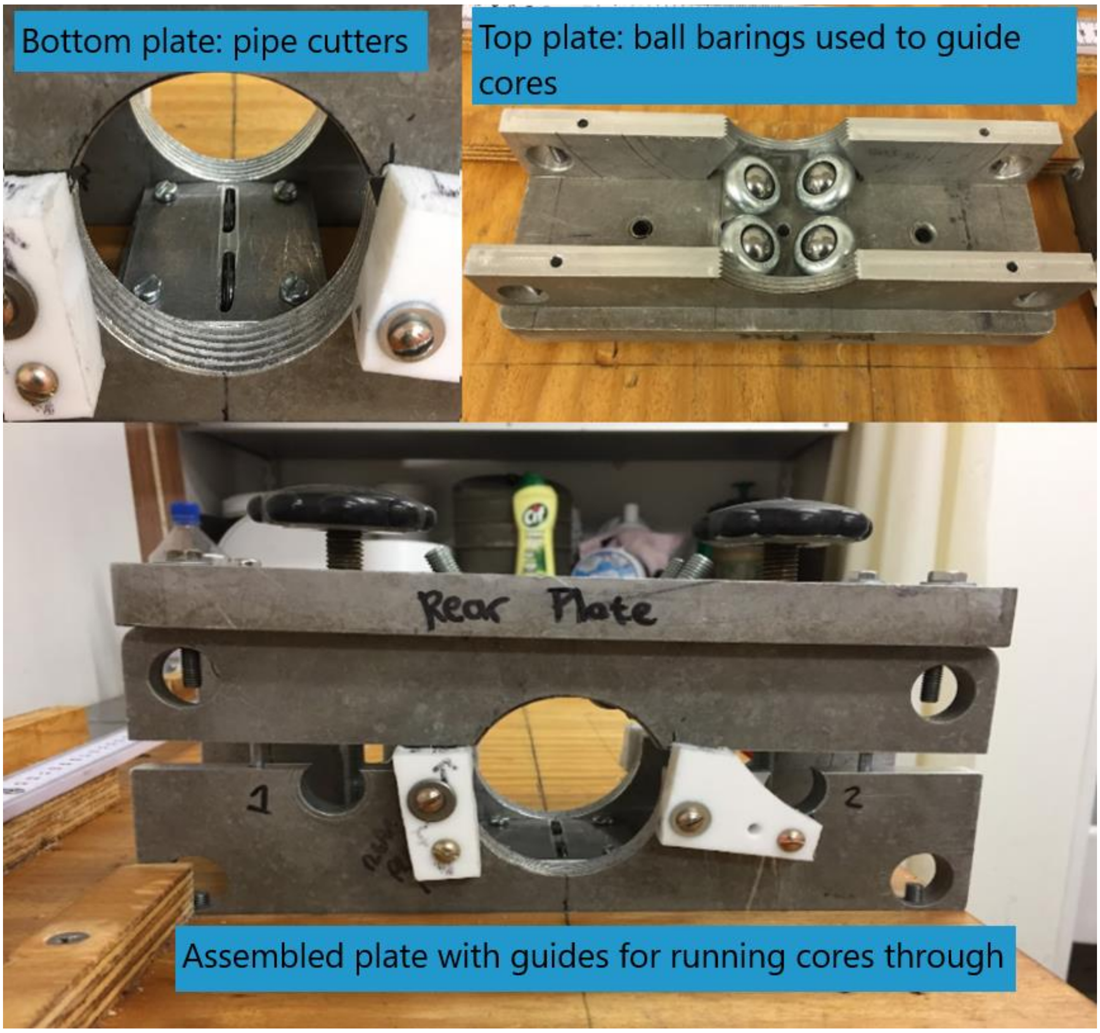


Figure 7. Core slicer made at UCL Geography laboratory to slice 120cm aluminium cores. Top left: cutter dismantled from a standard pipe cutter. Top right: top unit with ball bearings screwed into it to act as a guide for the core. Bottom: slicer fully assembled.

The slicer worked by inserting an aluminium core through the metal guide and running it backwards and forwards, slightly tightening the two metal plates together each time (Fig. 8). To ease use in the field, cores were put through the slicer to cause an indentation which would only need to be rolled over a few more times to cut the core. The slicer could only cut the cores up to a few cm from either end. Cores were, therefore, pre sawn at the top to ease slicing in field. The sawn ends were taped up to avoid small detritus getting wedged in the slits. It was decided that slicing the bottom ends would jeopardise the integrity of the cores, so these were cut in field.

Figure 8. Fully assembled core slicer made at UCL Geography laboratory to slice 120cm aluminium cores: a) pre-cutting cores in lab; b) core processing in the field; c) whole and cut core in slicer

Permission to collect all the material within this thesis was granted by the Marine Management Organisation by providing ‘notice of intention to carry on an activity under The Marine Licensing (Exempted Activities) Order 2011 (MMO, 2011) (as amended) “the Exemptions Order”’ (EXE/2016/00148). Since the Fleet is property of the Ilchester Estate further permission was provided by the Fleet Warden and by Natural England.

Carbon extrapolation

Since its presentation in 1974 (Walter, 1974) loss on ignition (LOI) has been widely used as a method to estimate the amount of organic matter (OM) and carbonate mineral content in soil samples (Santisteban *et al.*, 2003). The relationship between LOI at 550^{o}C and OM is presented as a % of the total weight of the sediment and calculated as follows:

$$\%OM = \frac{(dry\ weight - \ 550oC\ combusted\ weight)}{dry\ weigh \times 100)}$$

Because there exists a linear relationship between OM and organic carbon (OC), the relationship between LOI at 550^{o}C and OC content has been accepted as standard (Santisteban *et al.*, 2003). However, this method is semi-quantitative and relies on an empirically derived relationship between OC and OM, the strength of which varies with material (Santisteban *et al.*, 2003). In some cases, LOI has been reported to overestimate OC content (Leong and Tanner, 1999). The most accurate method to analyse OC is through dry combustion in an Elemental Analyser (EA) (Howard *et al.*, 2014), however, the costs involved are often prohibitive.

A study analysing 1,748 samples of seagrass sediment that had been analysed for %OM (using LOI) and %OC (using an EA) demonstrated that the relationship between the two is strong (Fourqurean *et al.*, 2012) (Fig. 9). The regression analysis for sediments including those with over 20% OC content (R^2=0.96) is stronger than for samples with <20% OC content (R^2=0.87), which make up most of the data (n=1,667). Regardless, the strength of both has resulted in the acceptance of OM as a proxy of OC, as per the IUCN Blue Carbon methods guidelines (Howard *et al.*, 2014).

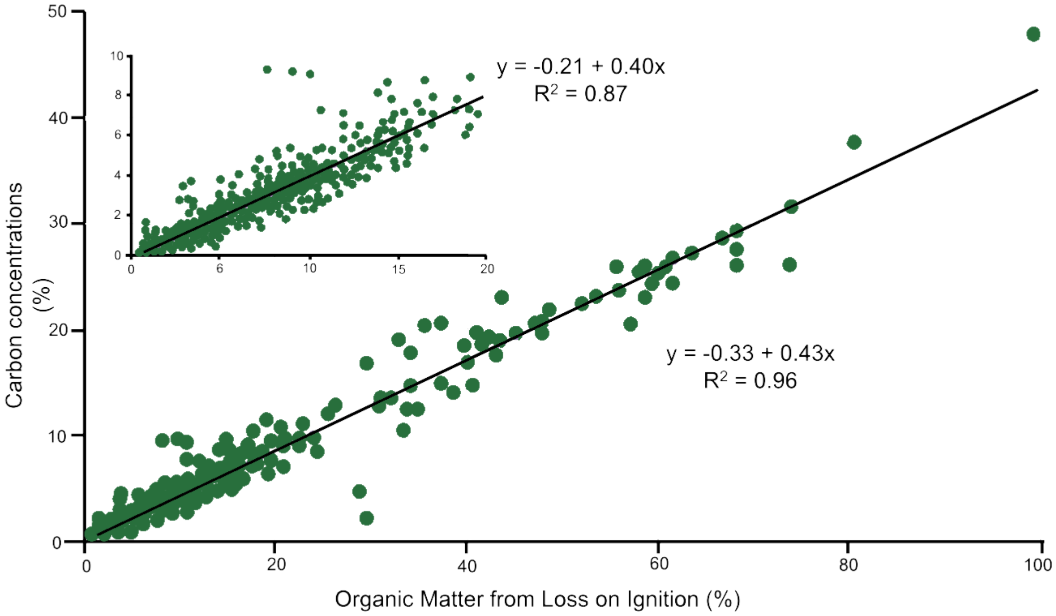


Figure 9. The relationship between %OC measured by an elemental analyser and LOI at 550oC from 1,748 seagrass sediment samples across the globe. Insert graph is for samples where %OC is less than 20% (n=1,677). (Data from Fourqurean et al., 2012 graph edited from Howard et al., 2014).

Two equations were developed to determine the relationship between LOI and OC for sediments samples where LOI>20%:

$$\%OC \ = \ -0.33 + 0.43\ (\%\ LOI)$$

and for sediment samples where LOI<20%

$$\%OC \ = \ -0.21 + 0.40\ (\%\ LOI)$$

Although these equations are deemed suitable for use under IUCN guidelines, their reliability can further be improved by sending a limited number of samples to be analysed by an EA and generating a linear equation from the results. This was done for 10% of the sediment samples in Chapter 4 and the results were used to convert samples for Chapter 3 and 5.

Sediment was analysed in a Flash EA in the Bloomsbury Environmental Isotope Facility (BEIF), University College London. For preparation for analysis in the Flash EA samples were dried, weighed and sieved to remove large items such as roots and shells. Due to the high levels of carbonates 1 g of ground sample was acidified with HCL diluted to 1N until outgassing stopped. This removes any inorganic carbon present in the sample, ensuring that analysis only represents organic carbon. Samples were left overnight and re-submerged with HCL to check for more outgassing. They were then centrifuged, and the supernatant acid was removed with a pipette. Samples were then washed with deionised water, centrifuged, and the supernatant water removed with a pipette three times per sample. The remaining sample was re-dried before analysis. The absolute effectiveness of the detector was determined using calibrated sources and samples of known activity.

A positive relationship ($R^2 = 0.38$) between %OM and %OC was found among the analysed sediments (Fig. 10). This relationship was not as strong as the relationship derived from the global literature ($R^2 = 0.96$) (Fourqurean *et al.*, 2012). To assess the reliability of the developed equation a selection of %OM results were put through both equations and differences were statistically analysed. The differences were not significant, so our linear equation was applied to the %OM samples to determine %OC:

$$\%\mathrm{OC} = 0.3708\%\mathrm{LOI} + 0.3732$$

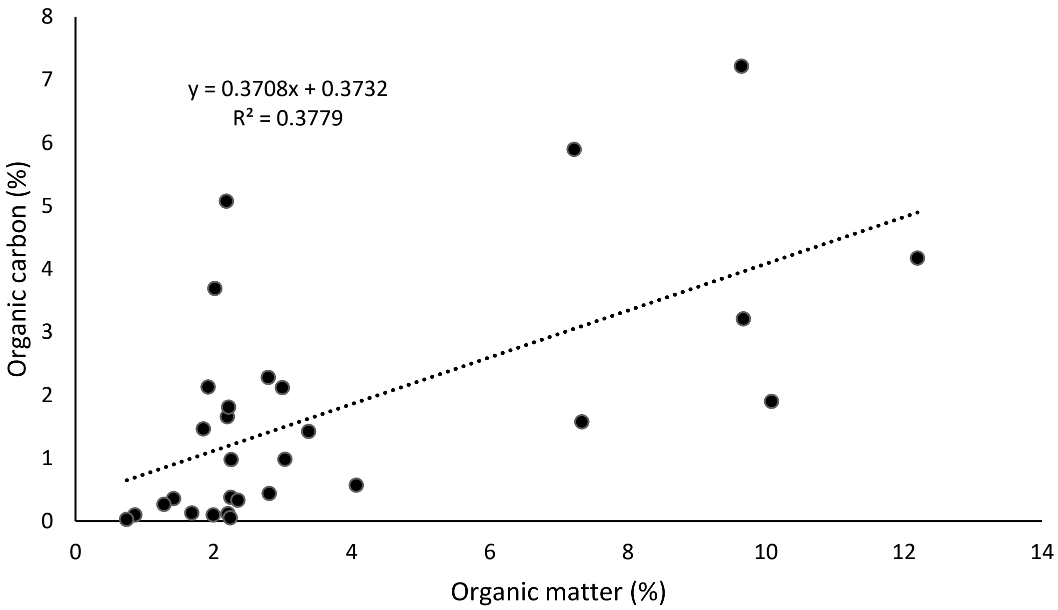


Figure 10. Relationship between organic matter (%OM) derived from loss on ignition and organic carbon (%OC), derived from isotopic analysis in a *Flash EA* (BIEF lab, UCL). Equation used to correct samples from %OM to %OC.

Estimating organic carbon stocks

Carbon stocks of entire systems are estimated in Chapters 3 and 4 of this thesis. For the purpose of this thesis, a carbon stock is made up of the total amount of carbon stored in the carbon reservoirs of a seagrass meadow of a known area (Howard *et al.*, 2014). 'Stored' carbon typically refers to carbon that is locked into a reservoir, away from the atmosphere, long-term (i.e. >100 years). The vast majority of seagrass carbon is stored within its sediments and the aboveground biomass is typically negligible, since leaves are either transported away from meadows through tidal movements, or decompose very quickly (Howard *et al.*, 2014). Therefore, the carbon stock here refers to the total amount of carbon found within the sediments of a meadow of a known size.

Stocks for each meadow were estimated either over a 30cm or 100cm core sample. Where cores could not be extracted to the desired depth, missing slices were estimated using the relationships between depth, soil weight from a known volume (dry bulk density, hereafter, DBD) and OC, to determine OC at 3cm intervals up to 30cm or 100cm (Fourqurean *et al.*, 2012). Less than 5% of core slices were estimated in this way. To allow for global comparisons, shorter cores were extrapolated to 100cm, assuming a simple, linear extrapolation. Unless stated otherwise unit area estimates of stocks are given down to 1m depth. Carbon stocks were estimated using the IUCN protocols (Howard *et al.*, 2014):

Step 1: For DBD sediment from each sample was measured in a $2cm^3$ crucible and dried to constant weight in the oven in the UCL Geography laboratory at 105^oC for >12hours (Santisteban, 2004). DBD was then calculated from the mass of a dried sample and its original volume for each cores slice:

$$DBD\ (g\ cm3) = mass\ of\ dry\ soil\ (g) \div original\ volume\ sampled\ (cm3)$$

Step 2: Soil carbon density (SCD) was calculated from the DBD and total OC content:

$$SCD\ (g\ cm3) = DBD\ (g\ cm3) \times \%OC \div 100$$

Step 3: Total amount of carbon was calculated for each core slice (TCS) by multiplying SCD by the thickness of the core slice:

$$TCS\ (g\ cm3) = SCD(g\ cm3) \times 3cm$$

Each slice within the core was then summed over the total core depth to determine total carbon within each core (TCC):

$$TCC = TCSi\ (g\ cm3) + TCSii\ (g\ cm3) + TCSn \ldots)$$

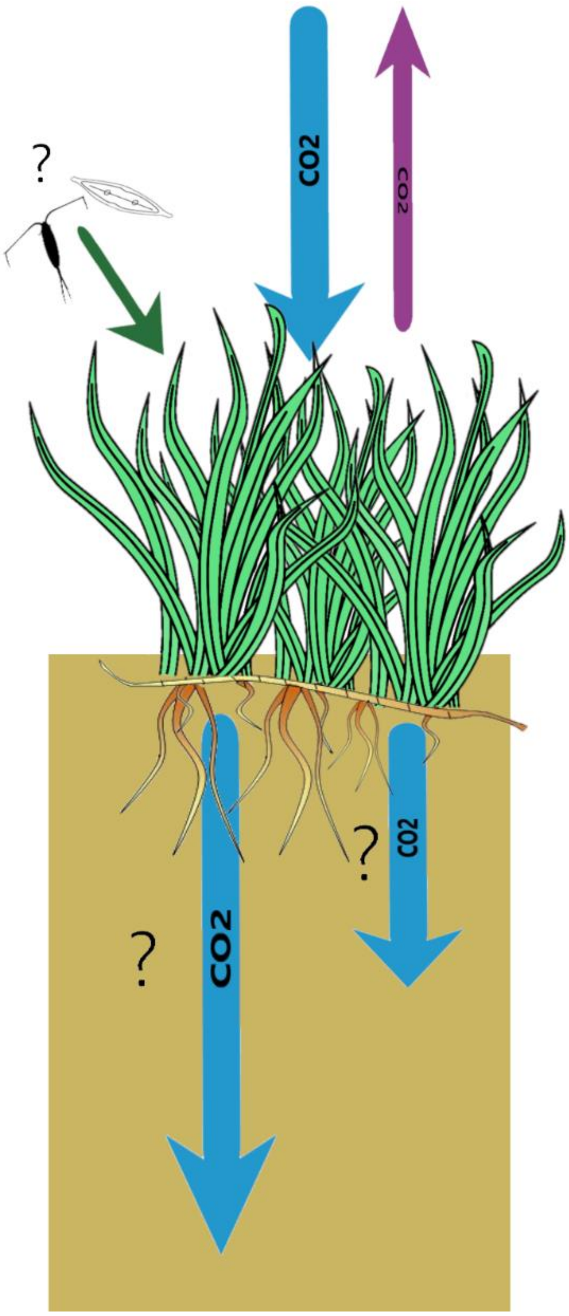
Step 4: Values were converted into Mg C/Hectare^{-1}, the units commonly used in carbon stock assessment using the following conversion units:

$$1{,}000{,}000\ g\ per\ Mg\ (megagram) and\ 100{,}000{,}000 cm3\ per\ hectare$$

Step 5: This was repeated for each core in the known area and average carbon in the total sampling depth was determined and presented with standard deviation.

Chapter 4

Carbon dynamics of *Zostera marina* sediments in the Fleet lagoon reveals inconsistencies in globally extrapolated data



Introduction

Seagrasses are foundation species, creating habitat for numerous fauna and flora and adapting the environments they inhabit (Thomson *et al.*, 2015). They are among the most productive and complex of ocean systems (McRoy and McMillan, 1977) and early evidence highlighted the role they play in carbon storage and fluxes (Smith, 1981). Despite this, seagrass science and conservation is lacking compared to other coastal habitats (Duarte *et al.*, 2008) and attention has only recently focused on them as useful to climate mitigation strategies. Of all the Blue Carbon habitats (mangroves, saltmarsh and seagrass), seagrass remains the most underrepresented in research and conservation effort (Duarte *et al.*, 2008), despite knowledge that it can store up to two times more carbon in its sediments than terrestrial forests (Fourqurean *et al.*, 2012).

Seagrass sediment is among the most carbon dense in the world, with 90% of the carbon stored by seagrass habitats found within their sediments (Fourqurean *et al.*, 2012). Despite occupying less than 0.2% of the ocean floor they are annually responsible for more than 20% of the total carbon buried by the oceans sediments (Duarte *et al.*, 2005; Kennedy *et al.*, 2010). Seagrasses also absorb carbon at a faster rate than many terrestrial stores and once stored it is more stable (Mateo *et al.*, 2006; Macreadie *et al.*, 2014a). Seagrasses ability to store carbon in their sediments has been heralded as a way to increase awareness, protection and restoration of seagrass, whilst supporting climate change mitigation (Laffoley, 2009; Nellemann, 2009; McLeod *et al.*, 2011). Unfortunately, seagrass is being lost at rapid rates globally with greater than 29% lost since the 1980s (Waycott *et al.*, 2009; Short *et al.*, 2010).

Despite their low spatial coverage, seagrasses account for 1% of the global net primary production of the oceans (Duarte and Cebrián, 1996). Their gross primary production is normally significantly higher than respiration, producing positive mean net community production (27.2 ± 5.8 mmol O_2 m^{-2} d^{-1}), making them autotrophic and, therefore, CO_2 sinks (Duarte *et al.*, 2010). In addition, low herbivory rates, poor tissue nutrient concentrations and slow seagrass decomposition add to their ability to store CO_2 (Duarte and Cebrián, 1996; Duarte *et al.*, 2013a; Howard *et al.*, 2014). Much of the biomass produced by seagrass is in extensive root systems, facilitating autochthonous sediment carbon storage, where anaerobic conditions reduce decomposition rates (Duarte and Cebrián, 1996; Gacia *et al.*, 1999; Duarte *et al.*, 2013a; Howard *et al.*, 2014).

Seagrasses exist in hydrodynamically active settings, where waves, tides and currents transport carbon created in one location to another, supporting allochthonous sequestration (Howard *et al.*, 2014). Their canopies can slow the water flow from 2- to over 10-fold compared to outside the meadow (Gambi *et al.* 1990; Hendriks *et al.*, 2008). This increases sediment deposition, directly forcing particles in the water column to settle, and reduces resuspension, giving settled particles time to be incorporated into the sediment (Hendriks *et al.*, 2008; Duarte *et al.*, 2013a). The typically low turnover of seagrass sediments and associated anoxic conditions mean, if left undisturbed, meadows can bind carbon for thousands of years (Duarte, 1990; Mateo *et al.*, 1997; Duarte *et al.*, 2013a). Sediment $\delta^{13}C$ values indicate that on average 50% of seagrass carbon is autochthonous and 50% is allochthonous (Kennedy *et al.*, 2010), making seagrass particularly useful, since it can bury carbon from other systems which would otherwise be re-released back into the atmosphere.

Up to 829.2 Mg C ha and on average 194.2 ± 55.9 Mg C ha have been reported to be stored in the top 100 cm of seagrass sediments, globally (Fourqurean *et al.*, 2012). Based on estimates of global seagrass coverage of between 300,000 and 600,000 km^2, an estimated 19.9 Pg of carbon could thus be stored in the top 100 cm of seagrass sediments world-wide (Fourqurean *et al.*, 2012). These data, although important in championing this vital service provided by seagrass meadows, are limited. Over 30% comes from Mediterranean records, where *Posidonia oceanica* is the predominant species. This species is unusual in its carbon storage capacity and the Mediterranean stores ~40% more carbon than its next closest region (Mediterranean = 372.4 Mg C ha^{-1} vs. South Australia = 268.3 Mg C ha^{-1}), as a result (Fourqurean *et al.*, 2012). Further, these estimates heavily rely on data collected from <30 cm sediment cores, extrapolated to represent a 100 cm sediment profile. In fact, only 18% of the cores included in these estimates are from 100cm depth profiles. The others are extrapolated based on relationships between depth, dry bulk density (DBD) and organic carbon (OC) to estimate DBD and OC at the desired depth intervals up to 100cm for each core (Fourqurean *et al.*, 2012). With such few data on 100cm cores it is difficult to know whether estimating carbon stores in this way leads to an over- or under- estimation.

Despite the growing interest in Blue Carbon and the potential role of seagrass for this vital ecosystem service, few studies have closely examined carbon storage depth profiles. Additionally, no studies in the British Isles have looked to examine widely held assumptions about carbon storage relationship between 30 cm and 100 cm depths. This study seeks to examine relationships of sediment characteristic across depth profiles in the Fleet Lagoon on the southwest coast of England. The Fleet lagoon is considered to be one of the finest, largest and best studied lagoons of its type in the British Isles (Johnston and Gilliland, 2000), but comparatively little work has focused on its seagrass meadow. The site offers an excellent case study opportunity with conditions that allow for extraction of deep (100cm) and shallow (30cm) cores.

Therefore, the aims of this chapter are to investigate sediment conditions and carbon storage dynamics of the seagrass sediments found within the Fleet lagoon. Further, the study aims to analyse shallow vs. deep sediment profiles to test assumptions of sediment carbon relationship with depth. The chapter will test the following hypothesis:

Hypothesis: extrapolating carbon stocks from a 30cm depth profile to a 100cm depth profile underestimates the total carbon stored within the Fleets seagrass meadow.

It will also determine rates of sediment accumulation and age of carbon stores, as well as the potential sources of carbon within these sediments.

Methods

Study site

The Fleet lagoon covers 480ha (Johnston and Gilliland, 2000), forming a unique environmental habitat benefiting from shelter and low levels of flushing, which imparts an unusual tidal and salinity regime (Robinson, 1982). The lagoon runs 12.5km, west to east, along the Dorset coast, west of Weymouth and the Isle of Portland (Fig. 1). It is flanked by Chesil Beach, a large shingle barrier to the English Channel. It is shallow, ranging between 0.3 and 3m deep, and ranges from 200m to 40m in width (Robinson, 1982; Johnston and Gilliland, 2000; Bennett *et al.*, 2008). Chesil beach itself is internationally renowned, one of the most famous landforms in coastal Britain, and of scientific significance (Bennett *et al.*, 2008). The Fleet is almost entirely enclosed, the western end at Abbotsbury forming an embayment, with a small channel to the east at Smallmouth connecting it to Portland Harbour and the sea (Fig. 1). It is a natural lagoon with a rural shoreline and a small rural catchment draining 28km^2 via seven streams (Johnston and Gilliland, 2000). It has been property of the Ilchester estate for 400 years, which has provided it with some levels of environmental protection, and allowed it to remain the only major shingle structure along the British coastline to be free from development (Carr and Blackley, 1973; Johnston and Gilliland, 2000). It represents one of the most protected coastal areas in Britain. It is a Site of Special Scientific Interest (SSSI) a RAMSAR site (Wetlands), a Special Protected Area (SPA) and a UNESCO world Heritage site.

Figure 1. The Fleet lagoon and its seagrass meadow (shaded pink). Circle indicates connection to the sea via small channel into Portland Harbour. Study sites locations (left to right) Langton Herring, Gore Cove and, Butterstreet Cove denoted with green seagrass icon. Insert is location along the Dorset coast. Map made by author in QGIS.

The system's substrate is a mix of silt, sand, pebbles and peat that provides favourable environmental conditions for the plentiful seagrass meadow found within the lagoon, which forms the predominant habitat (May, 1980). *Zostera marina* is the dominant species with *Z. noltii* occurring in mixed sections in certain areas. The lagoon also provides habitat for tussleweed (*Ruppia cirrhosa* and *Ruppia maritima*), as well as the rare foxtail stonewort (*Lamprothamnium papulosum*) (Langston *et al.*, 2003). The specialised conditions that allow these plants to flourish provide habitat for specialist lagoonal species, including starlet sea anemone (*Nematostella vectensis*), lagoon sandworm (*Armandia cirrhosa*), and lagoon sand shrimp (*Gammarus insensibilis*) (Langston *et al.*, 2003). Marine species are also found in exceptional abundance, including snakelocks anemone (*Anemonia viridis*) and cushion star (*Asterina gibbosa*) (Langston *et al.*, 2003). The lagoon supports economically important fish species, including adult grey mullet and eels, and juvenile bass, as well as non-economic smelt, three spined stickelbacks, deep-snouted pipefish and mud gobies (Johnston and Gilliland, 2000). Further, the seagrass meadows of the Fleet support numerous wildfowl, including a Swannery at Abbotsbuty that houses a herd of mute swan that have been farmed there since the 1300's (Robinson, 1982; Langston *et al.*, 2003). The swans consume seagrass in spring and early summer and later migrant birds such as wigeon, pochard, brent geese, and coot feed on seagrass and algae in autumn through to winter (Johnston and Gilliland, 2000).

The Fleet is subject to eutrophication which has led to its designation as a Polluted Water under the EU Nitrates Directive (91/676/EC), and its catchment area as Nitrate Vulnerable (Langston *et al.*, 2003). Nutrient loading is predominantly found in the western lagoon, especially the Abbotsbury embayment, near to the swannery (Langston *et al.*, 2003). Agricultural run-off and high density of bird-life are the primary sources of nutrient inputs, and are the probable causes for reduced water quality, coupled with low flushing and exchange rate of this water body (Langston *et al.*, 2003). The seagrass meadow found in the Fleet has reduced in size over time, but still covers 275ha of the lagoon (Natural England, 2017). Although the exact causes of this are unknown it is likely related to nutrient enrichment; globally a leading factor for damaging seagrasses (Short and Wyllie-Echeverria, 1996).

Field methods

Three sites within the Fleet lagoon were selected for the current study, in the upper, middle and lower section of the lagoon (Fig. 1). Sites were selected to ensure an even spatial coverage within the meadow. Sample collection was completed in the summer months, when seagrass growth is at its fullest. At all sites one stainless steel core (120cm long 50mm diameter) was inserted into the sediment to a depth of 110cm.

Additionally, three PVC cores (40cm long, 70mm diameter) were manually inserted to depths of 40cm. At Gore Cove a further 50cm core was inserted to a depth of 40cm for ^{210}Pb analysis, to determine sediment accumulation rates. This was extracted at Gore Cove because it is in the middle of the meadow. Cores were extracted and stacked upright on the boat before being transported back to shore for immediate slicing and storing. Metal cores were sliced using a slicing mechanism made at UCL Geography laboratory (see Chapter 2). Polyvinyl chloride (PVC) cores were pre-sliced and taped together and were re-opened to process the sediment (see Chapter 2). Cores were sectioned in 3cm intervals, apart from the ^{210}Pb core, which was sectioned in 0.5cm intervals. Slices were bagged and labelled and kept in a cool box, with ice, before transferring back to UCL and frozen within 48 hours.

Laboratory analysis

In the laboratory, core samples were thawed and divided into two sub-samples. One set was analysed for DBD and percent organic matter (%OM) using Loss on Ignition (LOI; see Chapter 2) and another was freeze-dried for isotopic analysis in a FlasEA (BEIF Lab, UCL) to determine carbon δ^{13}C signature.

Plankton, epiphytes, macroalgae and terrestrial organic matter that accumulate in seagrass systems have different, identifiable stable isotope signatures from seagrass tissues (Moncreiff and Sullivan, 2001). By measuring the isotopic composition of seagrass sediment the balance between allochthonous and autochthonous material can be determined (Kennedy *et al.*, 2010). Generally less depleted δ^{13}C suggests greater seagrass contribution (Gacia *et al.,* 2002). Samples were analysed in IsoSource 1.3, an isotope mixing model (Phillips and Gregg, 2003).

The %OM values were corrected to %OC using the equation generated by the author (Chapter 3). Organic carbon stocks were calculated as per the Blue Carbon protocols (see Chapter 3 (Howard *et al.*, 2014)).

The 50cm core was separated for ^{210}Pb analysis to determine recent (~100 years) sedimentation rates. The ^{210}Pb samples were freeze-dried, homogenised and analysed for ^{210}Pb and ^{226}Ra to determine at what depth equilibrium is reached. Caesium^{137} activity was analysed to determine artificial fallout radionuclides from the 1963 fallout. All radionuclides were measured by direct gamma assay in the Environmental Radiometric Facility at University College London, using ORTEC HPGe GWL series well-type coaxial low background intrinsic germanium detector. Lead^{210} was determined by gamma emission and ^{226}Ra by gamma rays emitted by its daughter isotope ^{214}Pb following storage for 3 weeks to ensure radioactive equilibrium. The absolute effectiveness of the detector was determined using calibrated sources and sediment samples of known activity. Carbon accumulation rates were estimated by multiplying the depth integrated carbon stocks by seagrass area and sediment accumulation rate from the ^{210}Pb analysis.

Statistical analysis

Kruskal Wallace rank sum test was used to denote differences between sites. Pairwise comparisons were made using Wilcoxon rank sum test. A simple bootstrapping technique (resampling with replacement; Manly, 2007) was used to establish variation in %OC at depth. Values for each depth were randomly selected 1000 times and averaged to provide best estimate of sediment %OC along 3cm interval depth for the entire core profile. Regression analyses were completed to determine relationships between depth and DBD, OM and OC. Relationships were presented as a value of R^2.

Results

Sediment characteristics

The DBD in Fleet seagrass sediments ranged from 0.09 g cm^3 at Gore Cove and 0.82 g cm^3 at Butterstreet Cove, with an average of 0.40 g cm^3 ± 0.11 g cm^3 (Table 1). The average among sites ranged from 0.38 g cm^3 ± 0.09 g cm^3 at Langton Herring to 0.42 g cm^3 ± 0.12 g cm^3 at Gore Cove (Table 1; Fig. 2). Average DBD for the 30cm cores ranged from 0.33 g cm^3 ± 0.13 g cm^3 at Butterstreet Cove to 0.35 g cm^3 ± 0.11 g cm^3 at Gore Cove with an overall average of 0.34 g cm^3 ± 0.13 g cm^3. There were no significant differences in DBD between any of the 30cm core slices.

Average DBD for the 100cm cores ranged from 0.41 g cm^3 ± 0.09 g cm^3 at Langton Herring to 0.47 g cm^3 ± 0.11 g cm^3 at Gore Cove, with an overall average of is 0.45 g cm^3 ± 0.10 g cm^3 (Table 1; Fig. 2). There was no significant difference in DBD between any of the 100cm core slices.

Dry bulk density was significantly lower in all the 30cm cores compared to 100cm cores ($p<0.05$). There is no relationship between DBD and depth for either the 30cm ($R^2 = 0.02$) or 100cm ($R^2 = 0.01$) cores (Fig. 3; a & b).

Table 1. Sediment characteristics of 30cm and 100cm cores from three sites in the Fleet seagrass meadow

	DBD ($g\ cm^3$)	**%OM**	**%OC**	**SCD ($mg\ C\ cm^2$)**	**C_{stock} (Mg C ha)**	**C_{stock} extrap. to 100cm (Mg C ha)**
All data	*0.40 ± 0.11*	*10.78 ± 3.54*	*4.40 ± 1.44*	*N/A*	*N/A*	*N/A*
			30cm Cores			
Langton Herring	0.34 ± 0.08	9.03 ± 3.47	3.59 ± 1.42	12.76 ± 0.74	34.17 ± 4.53	113.90 ± 15.10
Gore Cove	0.35 ± 0.11	9.32 ± 2.40	3.83 ± 0.89	12.22 ± 2.11	38.56 ± 1.92	128.19 ± 6.41
Butterstreet cove	0.33 ± 0.13	9.40 ± 3.03	3.86 ± 1.12	11.22 ± 1.61	37.29 ± 5.06	124.64 ± 16.87
Meadow mean	*0.34 ± 0.10*	*9.39 ± 2.96*	*3.81 ± 1.16*	*12.07 ± 1.49*	*36.67 ± 3.84*	*122.25 ± 12.79*
			100cm cores			
Langton Herring	0.41 ± 0.09	13.80 ± 2.20	5.53 ± 0.81	20.09 ± 7.30	196.23	N/A
Gore Cove	0.47 ± 0.11	11.19 ± 4.90	4.50 ± 1.85	17.53 ± 7.46	221.60	N/A
Butterstreet cove	0.46 ± 0.09	10.57 ± 2.00	4.53 ± 1.52	18.91 ± 7.95	219.12	N/A
Meadow mean	*0.45 ± 0.10*	*11.86 ± 3.59*	*4.85 ± 1.53*	*18.84 ± 7.57*	*208.98 ± 11.67*	*N/A*

Data are core means ± standard deviation or site overall meadow mean ± standard deviation. DBD = $g\ cm^3$ dry bulk density; %OM = % organic matter; %OC = % organic carbon; SCD = $mg\ C\ cm^2$ soil carbon density; C_{stock} Mg C ha = megagrams of C per hectare; C_{stock} extrap. = extrapolated. N/A = not applicable. 100cm cores were extrapolated as per Chapter 3.

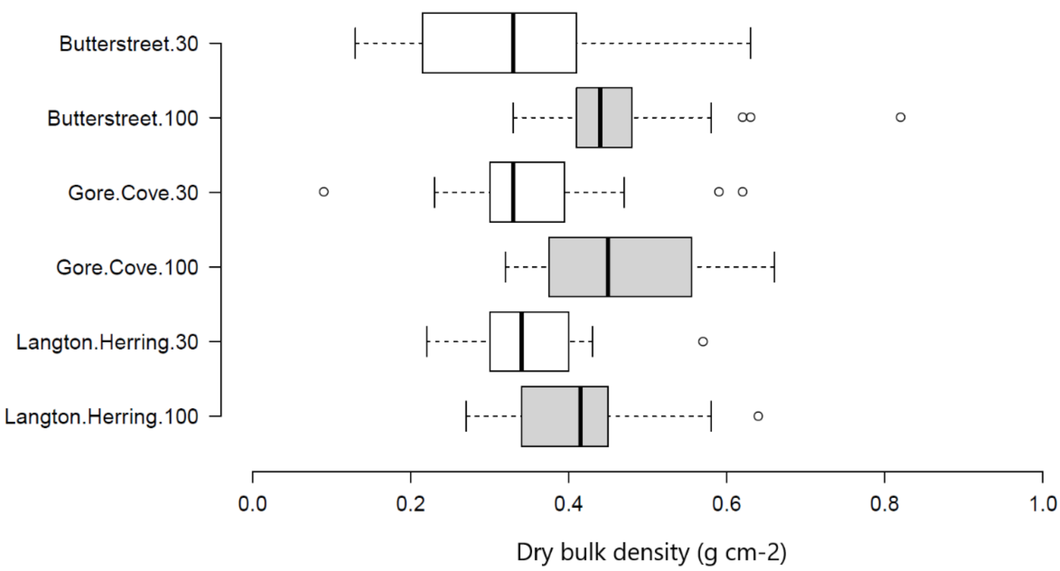


Figure 2. Dry bulk density across three sites in the Fleet lagoon from 30 and 100cm depths.

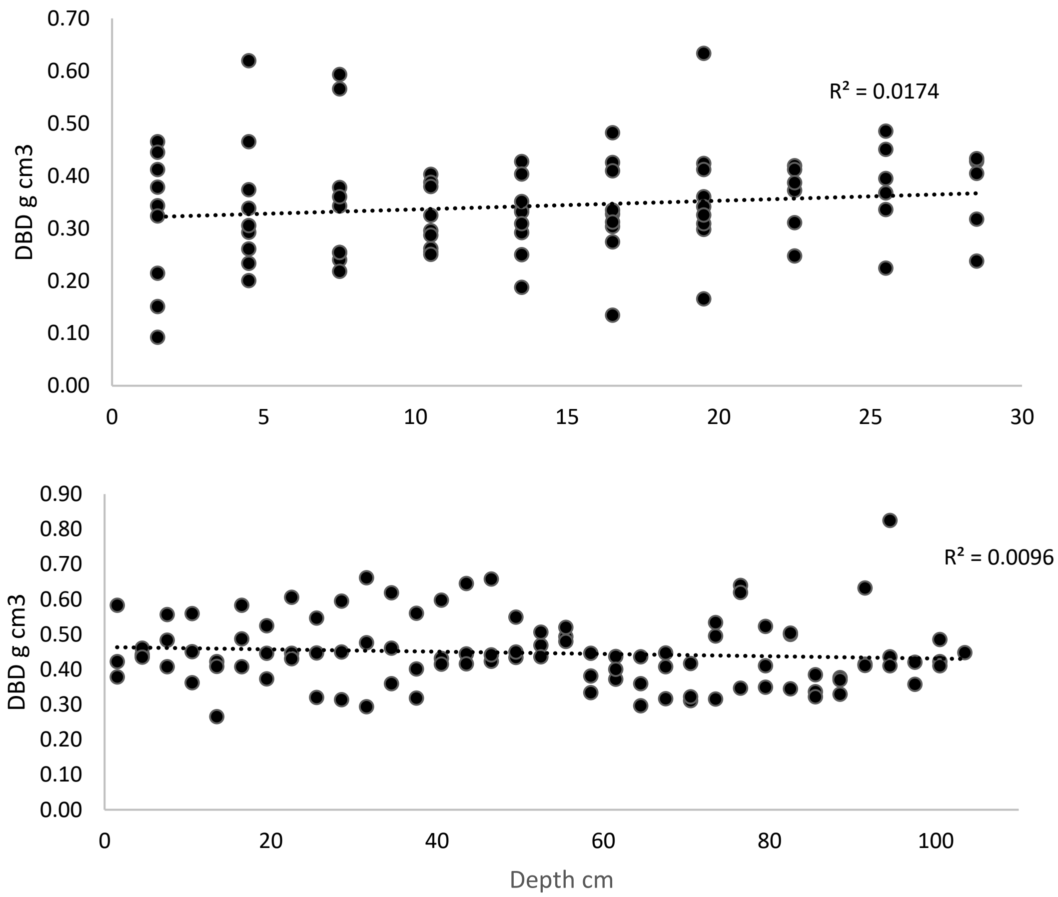


Figure 3. Regression analysis of dry bulk density (g cm^3) against depth for 30cm cores and 100cm cores (see difference in x axis).

Percent OM in Fleet seagrass sediments ranged from 2.49% at Gore Cove to 23.47% at Langton Herring, with an average of 10.78% ± 3.54%. Overall site averages ranged between 10.08% ± 2.53% at Butterstreet Cove to 11.84% ± 3.56% at Langton Herring (Table 1; Fig. 4). Average %OM for the 30cm cores ranged from 9.03% ± 3.47% Langton Herring and 9.40% ± 3.03% at Butterstreet Cove, with an overall average of 9.39% ± 2.96%. There were no significant differences between any of the 30cm cores. Average %OM for the 100cm cores ranged from 10.57% ± 2.00% at Butterstreet Cove and 13.80% ± 2.20% at Langton Herring, with an overall average of 11.86% ± 3.59%.

The average OM content in sediment from Langton Herring was significantly higher than any other 100cm or 30cm cores (p<0.05). The 100cm core from Butterstreet Cove had significantly higher %OM content than the 30cm cores from Gore Cove (p=0.04) and Langton Herring (p=0.03) but not Butterstreet Cove. The 100cm core from Gore Cove was not significantly different from any of the 30cm cores. In the 30cm cores there is no relationship between %OM and depth (R^2 = 0.01) (Fig. 5). In the 100cm cores there is a stable trend in increasing OM% with depth to 55cm, after which values fluctuate and become more sporadic. There is a weak positive relationship between %OM and depth in the 100cm cores (R^2 = 0.14) (Fig. 5).

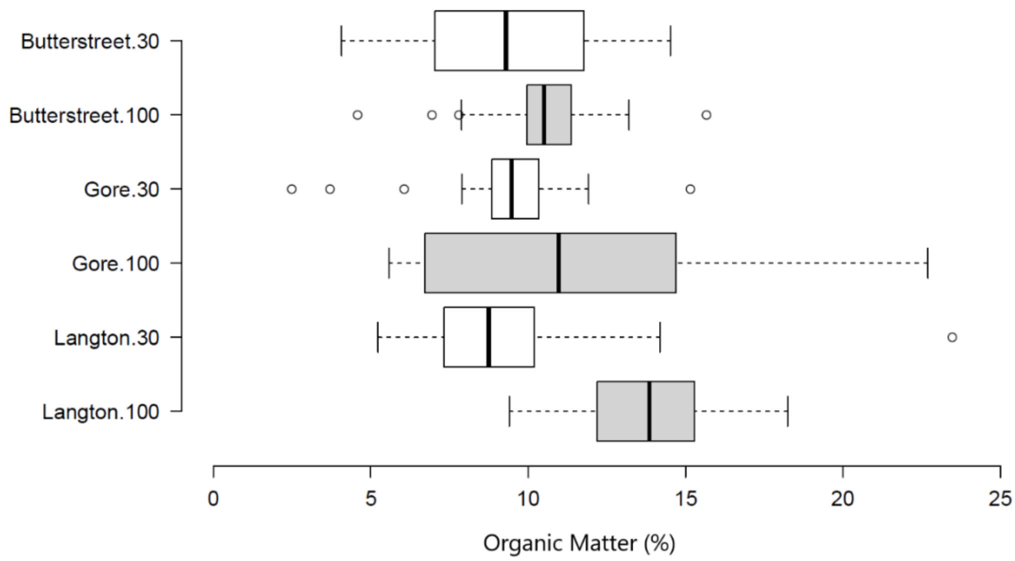


Figure 4. Sediment organic matter content across three sites in the Fleet lagoon from 30 and 100cm depths.

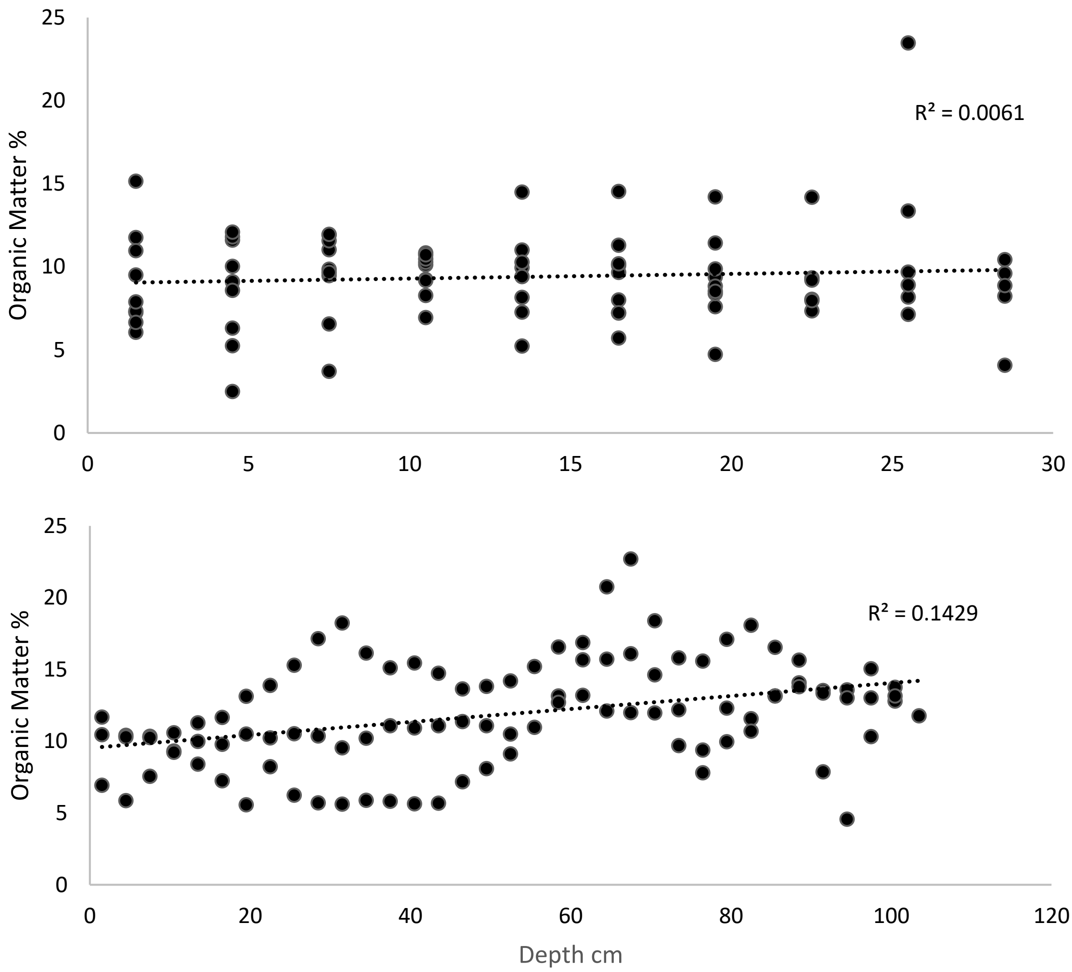


Figure 5. Regression analysis of average organic matter content (%) against depth for 30cm cores and 100cm cores.

Percent OC ranged from 1.30% at Gore Cove to 11.75% at Butterstreet Cove, with an average of 4.40% ± 1.44%. Overall site average ranged from 4.19% ± 1.54% at Gore Cove to 4.79% ± 1.34% at Langton Herring. Average %OC for the 30cm cores ranged from 3.59% ± 1.42% at Langton Herring to 3.86% ± 1.12% at Butterstreet Cove, with an overall average of 3.81% ± 1.2% (Table 1; Fig. 6). There were no significant differences between any of the 30cm cores.

Average %OC for the 100cm cores ranged from 4.53% ± 1.52% at Butterstreet Cove to 5.53% ± 0.81% at Langton Herring, with an overall average of 4.85% ± 1.53% (Table 1; Fig. 6). The 100cm core at Langton Herring contained significantly more OC than all other 100cm and 30cm cores ($p<0.05$). The 100cm core from Butterstreet Cove was also significantly higher than the 30cm cores from Langton Herring ($p<0.05$) and Gore Cove ($p<0.05$). There were no significant differences between any of the other cores.

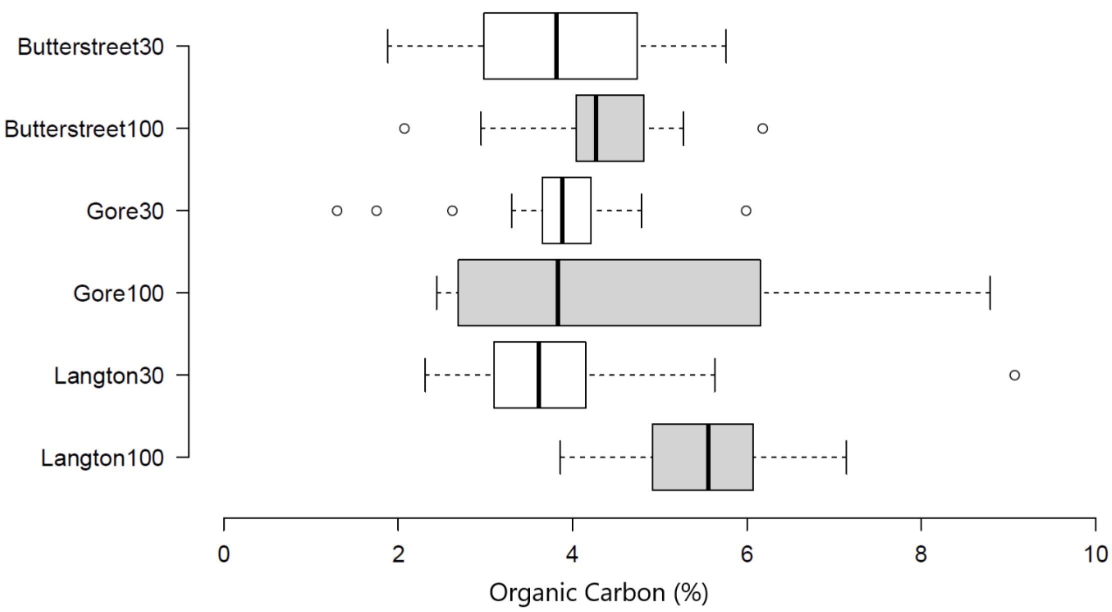


Figure 6. Sediment organic carbon content across three sites in the Fleet lagoon from 30 and 100cm depths.

The %OC depth profiles for 30cm cores displayed varying trends. Langton Herring displayed a slight increase in %OC with two anomalous readings at 13cm and 25cm (Fig. 7). Butterstreet Cove displayed a slight decrease in %OC at depth, and Gore Cove had an overall static profile (Fig 7). Little variation in %OC at depth was noted across all 30cm cores (R^2 = 0.058) (Fig. 8). The %OC depth profiles for 100cm cores also displayed varying trends. Langton Herring showed increasing %OC to 30cm and then a wave of increasing and decreasing %OC every 20-30cm (Fig. 7). Butterstreet Cove showed relatively stable %OM until 55-60cm where an anomalous recording occurred, followed by a slightly more erratic pattern (Fig. 7). Gore Cove showed steadily increasing %OC to 70cm, followed by declines, before stabilising towards the end of the core.

Across all 100cm cores %OC is relatively stable until 50cm, where it becomes more erratic (Fig. 7). The regression analysis confirms a weak correlation of increasing %OC at depth (R^2 = 0.328) (Fig. 8). Bootstrapping data also revealed an overall trend of stably increasing %OC with depth up to a depth of 56cm (Fig. 9). Beyond this %OC becomes more unstable with higher and low values occurring in a rapid series of peaks and troughs.

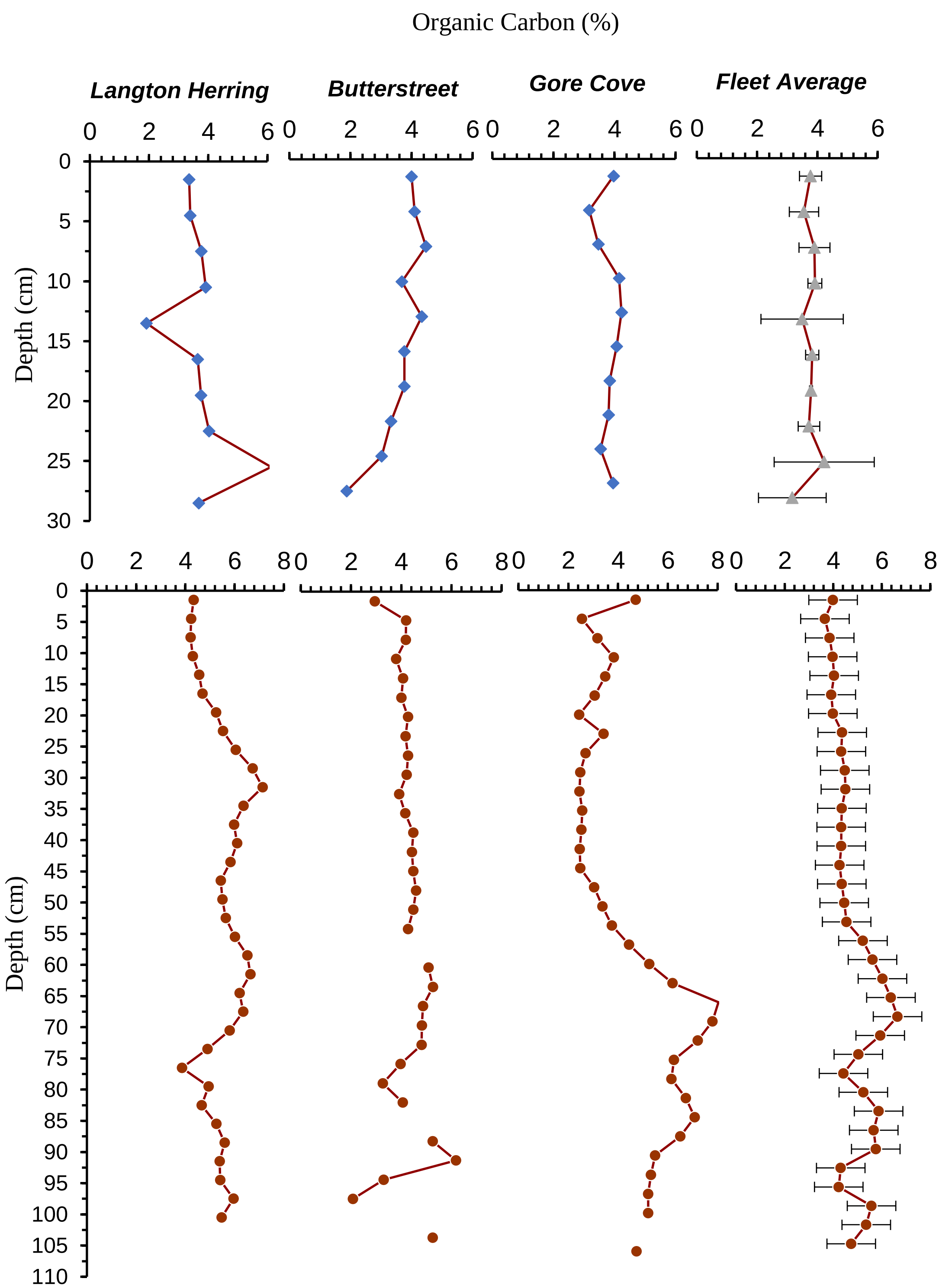


Figure 7. Top graph: Mean organic carbon depth profiles from three cores from three sites in the Fleet lagoon, and site mean plus standard deviation for 30cm cores. Bottom graph: organic carbon depth profiles from one core from three sites in the Fleet lagoon, and site mean plus standard deviation for 100cm cores.

Average soil carbon density (SCD) in the 30cm cores ranged from 11.22mg C cm^3 ± 1.61 C cm^3 at Butterstreet Cove to 12.76 C cm^3 ± 0.74 C cm^3 at Langton Herring, with an overall average of 12.07 C cm^3 ± 1.49 C cm^3 (Table 1). Average SCD in the 100cm cores ranged from 17.53 C cm^3 ± 7.46 C cm^3 at Gore Cove to 20.09 C cm^3 ± 7.30 C cm^3 at Langton Herring, with an overall average of 18.84 cm^3 ± 7.57 cm^3 (Table 1).

The carbon stocks from the short cores ranged from 34.17 Mg C ha ± 4.53 Mg C ha at Langton Herring to 38.56 Mg C ha ± 1.92 Mg C ha at Gore Cove, with an overall site average of 27.67 Mg C ha ± 3.84 Mg C ha over a 30cm depth profile (Table 1). The carbon stock from the long cores ranged from 196.23 Mg C ha at Langton Herring to 221.60 Mg C ha at Gore Cove, with an average of 208.98 Mg C ha ± 11.67 Mg C ha over a 100cm depth profile (Table 1). Extrapolated from 30cm to 100cm, the short cores estimated an average 122.25 Mg C ha ± 12.79 Mg C ha in the top 100cm of sediment in the seagrass meadow in the Fleet lagoon, almost half that of the long cores (Table 1).

Carbon isotopes and radionuclide dating

The carbon isotope signatures ($\delta^{13}C$ ‰) were similar across the depth profile of the 100cm core taken from Gore Cove (Fig. 10). The $\delta^{13}C$ ‰ values ranged from -18.70 to -16.08. The mean value was -17.30 and the median value was -17.20. There were no significant differences in $\delta^{13}C$ ‰ values along the depth profile. As with the trends in %OM and %OC, for the 100cm cores different trends occurred in the upper and lower 50cm of the core. The $\delta^{13}C$ ‰ between 0-50cm are more stable than 50-100cm, varying between -17.79 and -16.75 (Fig. 10). The $\delta^{13}C$ ‰ values in the lower 50cm of the core are more erratic and show greater variation (Fig 10).

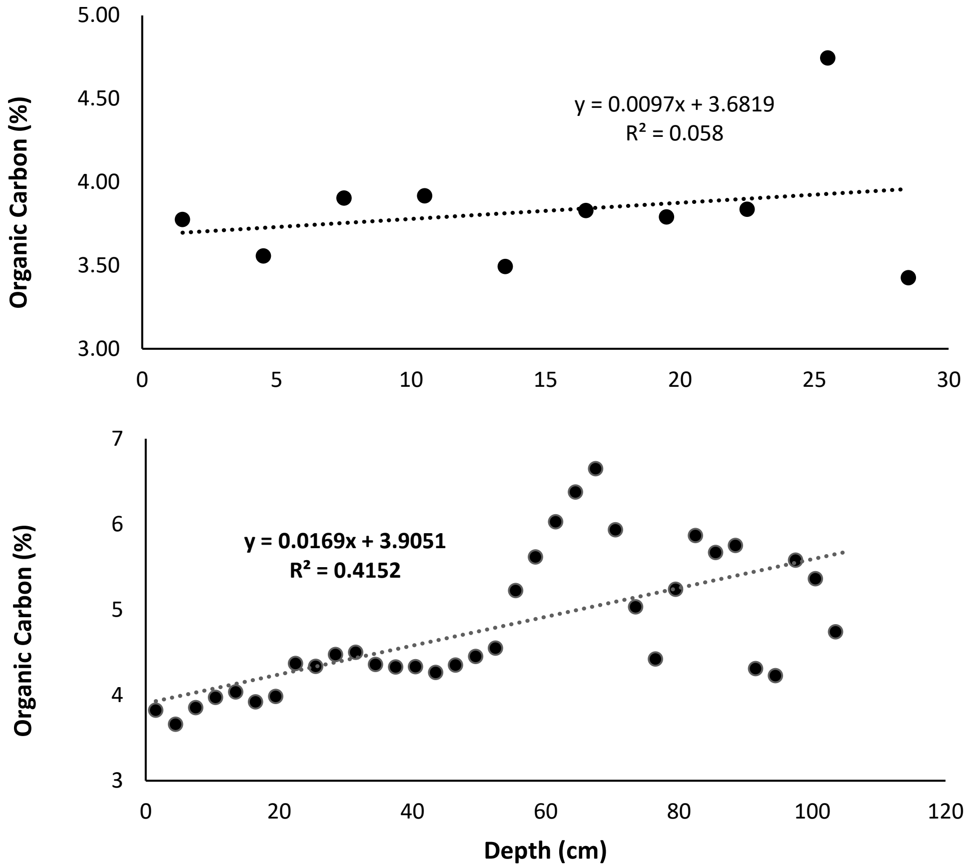


Figure 8. Regression analysis of average organic matter content (%) against depth for 30cm cores and 100cm cores.

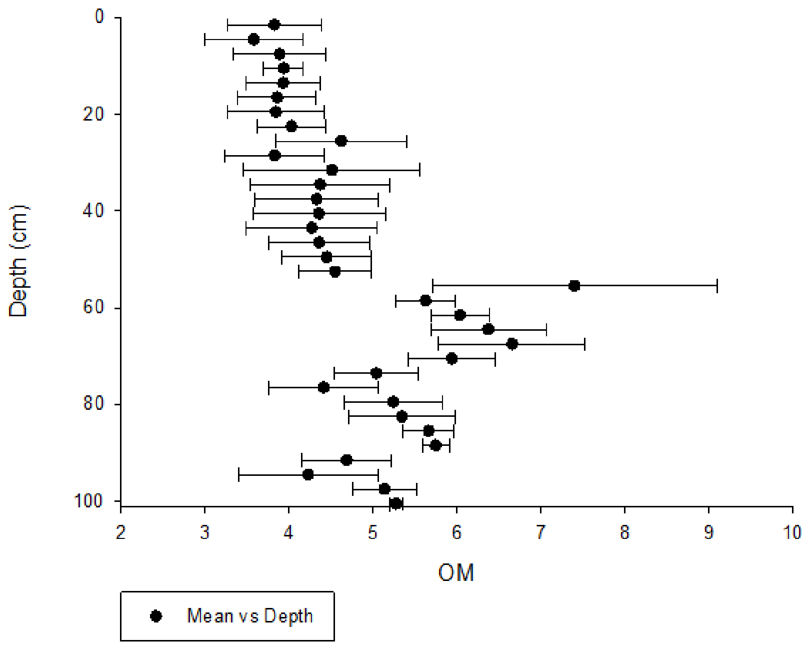


Figure 9. Depth profiles from bootstrapped depth data from sediment samples from the fleet lagoon. Data are means with standard error.

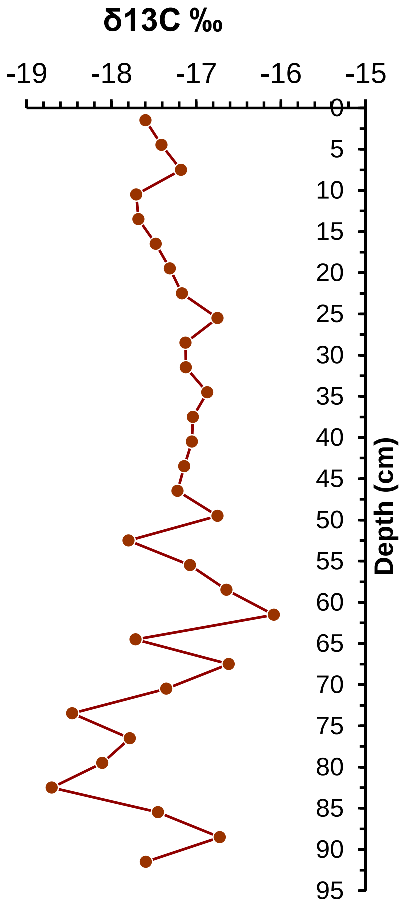


Figure 10. carbon isotope signatures ($\delta^{13}C$ ‰) along depth profile of Gore Cove 100cm core.

Unsupported ^{210}Pb horizons (occurrence of ^{210}Pb from decayed ^{222}Rn) were reached at 4.25cm depth. Artificial fallout concentrations of ^{241}Am, originating from nuclear testing, were not detectable in the sediment samples. Artificial fallout concentrations of ^{137}Cs (from the 1963 fallout) were detectable at 0.25cm, 0.75cm, 1.75cm and 2.75cm intervals and decreased with depth (Table 2). Sedimentation rate was low, ranging from 0.024 cm yr^{-1} ± 0.024 cm yr^{-1} at 3.25cm depth to 0.086 cm yr^{-1} ± 0.081 cm yr^{-1} at 1.75cm depth. Average sedimentation rate was 0.044 cm yr^{-1} ± 0.020 cm yr^{-1}. Based on the dating model, it takes approximately 10 years to accumulate 0.25cm of sediment within this system. The oldest date the model predicted was 1929 which is reached at 3.75cm (Table 2).

Table 2. Radionuclide inventories of supported and unsupported ^{210}Pb concentrations and artificial fallout concentrations of ^{137}Cs and ^{241}Am, chronology and sedimentation rate for seagrass sediments in the Fleet lagoon.

		^{210}pb concentrations		Artificial fallout concentrations		Chronology		Sedimentation rate	
Depth	Dry mass	Total supported	Total unsupported	^{137}Cs	^{241}Am	Date	Age		
cm	g cm^2	Bq Kg^{-1}	Bq Kg^{-1}	Bq Kg^{-1}	Bq Kg^{-1}	AD	yr	g cm2 yr	cm yr
0.25	0.116	60.73 ± 5.92	41.27 ± 6.19	2.95 ± 0.9	N/D	2013	5 ± 2	0.0250	0.046 ± 0.011
0.75	0.4055	40.2 ± 5.16	27.13 ± 5.28	1.94 ± 0.55	N/D	2002	16 ± 4	0.0268	0.039 ± 0.014
1.25	0.8049	24.26 ± 5.38	11.7 ± 5.54	N/D	N/D	1990	28 ± 8	0.0428	0.050 ± 0.031
1.75	1.2593	15.73 ± 3.85	4.87 ± 3.94	0.91 ± 0.44	N/D	1982	36 ± 11	0.0801	0.086 ± 0.081
2.25	1.7365	22.64 ± 4.19	10.25 ± 4.36	N/D	N/D	1972	46 ± 16	0.0276	0.029 ± 0.022
2.75	2.2199	13.93 ± 3.4	2.33 ± 3.5	0.75 ± 0.36	N/D	1961	57 ± 23	0.0472	0.049 ± 0.045
3.25	2.705	15.74 ± 3.69	6.16 ± 3.8	N/D	N/D	1950	68 ± 28	0.0232	0.024 ± 0.024
3.75	3.1934	15.78 ± 4.91	3.16 ± 5.06	N/D	N/D	1929	89 ± 34	0.0238	0.025 ± 0.025
4.25	3.6504	16.83 ± 2.76	3.22 ± 2.9	N/D	N/D	N/D	N/D	N/D	N/D

Data are means ± standard deviation. Bq Kg-1 = Kg Becquerel (unit of radioactivity); ^{210}pb = Lead-210; ^{137}Cs = Caesium-137; ^{241}Am = Americium-241; N/D = no data.

Discussion

This study provides the first estimates of carbon density in an English seagrass meadow, and joins many other studies that have recently attempted to highlight the roles seagrasses play in carbon storage worldwide (Gacia *et al.*, 2002; Fourqurean *et al.*, 2012; Lavery *et al.*, 2013; Röhr *et al.*, 2016; Serrano *et al.*, 2016a; Githaiga *et al.*, 2017; Potouroglou, 2017). The results support the growing understanding that there is substantial variation in the sediment characteristics of seagrass meadows, both globally and locally. Overall higher concentrations of carbon occur in the deeper sediments, challenging assumptions of sediment carbon relationship with depth. These confirm the hypothesis that extrapolating carbon stocks from a 30cm depth profile to a 100cm depth profile underestimates the total carbon stored within the Fleets seagrass meadow by over 40%.

Interestingly the %OM and %OC values from 100cm cores suggest a change in carbon accretion at ~50 cm. This pattern corresponds with $\delta^{13}C$ ‰ depth profiles, which also show stability in the top ~50cm followed by more sporadic, fluctuating values from ~50-100cm. The values suggest two very different conditions of carbon accumulation, which may be revealed through analysing the carbon inputs and modelling corresponding conditions from ^{210}Pb age profiles.

Comparisons of sediment characteristics with regional and global data

The average DBD at the Fleet lagoon (0.40 ± 0.11 g cm^3) was substantially lower than the global average of 1.03 ± 0.02 g cm^3 (Fourqurean *et al.*, 2012) and also the averages from comparable Danish (1.25 ± 0.02 g cm^3) and Finish (1.35 ± 0.01 g cm^3) *Z. marina* meadows (Röhr *et al.*, 2016). It was also substantially lower than mixed *Zostera* spp.

meadows in Scotland (1.62 ± 0.02 g cm^3) (Potouroglou, 2017). The sediment was noticeably wet and silty. High %OM and %OC has been linked to high sediment silt content in other sites (Röhr *et al.*, 2016) and could be part of the reason high levels were recorded here.

The average OM% within the Fleets sediment was high (10.78 ± 3.54 %) compared to both the global average (5.7 ± 0.3%) and Scottish (between 0.97 ± 0.11 % and 4.26 ± 0.21 %) (Potouroglou, 2017), Danish (3.90 ± 1.50 %) and Finish (1.40 ± 0.33%) averages (Röhr *et al.*, 2016). The average carbon content from the 100cm cores (208.98 ± 11.67 Mg C ha) was slightly higher than the global average (194.20 ± 20.20 Mg C ha), which include cores extrapolated from 30cm (Fourqurean *et al.*, 2012) (Fig. 11). This was considerably higher than the average from the North Atlantic region (48.70 ± 14.50 Mg C ha) (Fig. 11) and only lower than averages from South Australia (268.2 ± 101.7) and the Mediterranean (374.4 ± 74.5), regions that are dominated by two slow-growing species known to be exceptional in their carbon storage capacity: *Posidonia australis* and *Posidonia oceanica* respectively (Duarte, Middelburg and Caraco, 2005). Comparatively, the estimated carbon stocks from the 30cm cores extrapolated to 100cm (122.25 ± 12.79) falls below the global average but remains substantially higher than the North Atlantic region (Fig. 11).

At this stage it is not possible to say whether the Fleet is an outlier in its carbon storage capacity in the region, or the North Atlantic averages are vastly under-representative of UK seagrass meadows. The only other data on British Isles seagrass sediment carbon storage is from subtidal *Zostera noltii* meadows from Scotland, which include 50cm depth profiles that ranged from 22.70 Mg C ha to 107.90 Mg C ha (Potouroglou, 2016). Extrapolating to 100cm, the upper range of this is about 50 Mg C ha lower than the Fleet, but also much higher than the average for the North Atlantic region. Of course, the sediment values may demonstrate similar erratic variations and trends in carbon beyond 50cm depth. It is reasonably common for seagrass meadows found within similar conditions among a much smaller spatial range to contain one or two sites whose sediment carbon is drastically greater than its neighbours (Röhr *et al.*, 2016). It may be that other *Z. marina* meadows in the south of England are more analogous to the North Atlantic estimates. The number of data points in this global study for the North Atlantic region (n=24) are greater than all but the Mediterranean (n=29). This suggests that either the Fleet represents such an anomalous site, or generalisation across regions is an inaccurate representation of Blue Carbon values. The true significance of these findings can only be fully assessed by comparing the Fleet to other seagrass meadows in proximity to it.

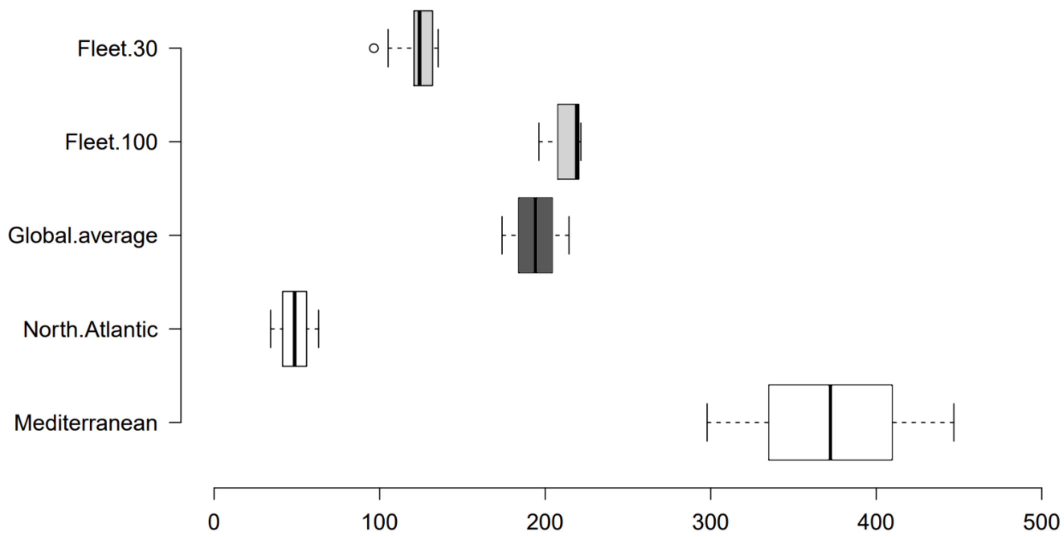


Figure 11. Carbon stock estimates from the seagrass meadow at the fleet lagoon using 30cm cores extrapolated to 100cm, and 100cm cores (light grey). Global average (dark grey) and North Atlantic and Mediterranean comparisons (white) are also included. (extrapolated from (Fourqurean *et al.*, 2012)).

Isotopic conditions of the Fleets seagrass sediments

Unfortunately, I was unable to analyse $\delta^{13}C$ ‰ values of *Zostera* tissue or other potential carbon sources at the site. However, a seminal paper analysed pared carbon isotope signatures of seagrass and its sediments from 207 sites across 88 locations globally (Kennedy *et al.*, 2010). From 17 sites they found that the global mean $\delta^{13}C$‰ of *Z. marina* tissue was -10.9 ± 0.6 (Kennedy *et al.*, 2010). A more recent study also found comparable signatures for *Z. marina* leaves and rhizomes in Denmark and Finland (-10.3 ± 0.4) (Röhr *et al.*, 2016). In the global study the average sediment $\delta^{13}C$‰ within *Z. marina* meadows was -18.4 + 0.5 (Kennedy *et al.*, 2010), and in the more recent study this figure was -19.9 + 0.3 for Denmark and -16.2 + 0.2 for Finland (Röhr *et al.*, 2016). Both these papers found that *Z. marina* on average contributed half of the carbon found within these sediments, the remining coming from external sources. The values found within Fleets sediments (-17.41 + 0.55) are isotopically similar, which would suggest a similar pattern.

By subtracting the known average seagrass tissue $\delta^{13}C$ ‰ from the literature from analysed sediment $\delta^{13}C$‰, the contribution of seagrass to the sediment carbon pool can be estimated ($\delta^{13}C$ ‰$_{seagrass-sediment}$) (Kennedy *et al.*, 2010c). The $\delta^{13}C$‰$_{seagrass-sediment}$ value for the fleet was 6.40. This positive value suggests that non-seagrass matter makes a strong contribution to the carbon that is accumulated in these sediments, since other potential OC sources have a more negative $\delta^{13}C$ ‰ (Kennedy *et al.*, 2010c). For example, recognised average values from phytoplankton as suspended particulate matter (SPOM), taken from a review of the literature base, are -20‰ (Goericke and Fry, 1994). Similarly, recognised mean values for terrestrial matter, derived from 570 species-site combinations globally, are −28.46 ± 2.52‰

(Diefendorf *et al.*, 2010). That *Zostera* tissue is isotopically heavy relative to these other sources means that their occurrence in the sediment profile would lighten the isotopic value (Kennedy *et al.*, 2010), as has been observed here.

I ran the IsoSource 1.3 isotope mixing model (Phillips and Gregg, 2003) using the sediment $\delta^{13}C$ ‰ values from this chapter, the average *Z. marina* $\delta^{13}C$ ‰ values from the literature (Kennedy *et al.*, 2010), and the above values of SPOM and terrestrial matter with an increment of 1% and tolerence of 0.1 (Phillips and Gregg, 2003). The model indicated that on average *Z. marina* was the major contibutor, contributing between 32% and 55% with an average of 47% ± 10% to the sediment carbon pool. Suspended particulate matter contributed between 28% and 48%, with an average of 36% ± 21%, and terrestrial organic matter contributed between 13% and 23%, with an average of 18% ± 11%. Between ~50-100cm the $\delta^{13}C$‰ become far more erratic with lighter and heavier isotopic peaks occurring. The lighter isotopic peaks (<-18) have a higher contribution of terrestrial organic matter (>20%) and a lower contribution of *Z. marina* (<40%). The heavier values in this period have less terrestrial contribution (<16%) and a higher overall *Z. marina* (>55%) and SPOM (~30%) contribution. These fluctuations in carbon source may represent different, less stable environmental conditions of the Fleet lagoon at the time this carbon was laid down.

This is a simple way of understanding the contribution of carbon to these pools, since it relies on data from the literature. However, there is a clear indication that seagrass is only a proportion of the carbon source within this site, which may account for the seemingly high %OC sediment profiles. It is not overly surprising that carbon stored in the Fleet's seagrass meadow follows the common trend of containing 50% allochthonous and 50% autochthonous carbon (Kennedy e*t al.* 2014). The low flushing conditions and slow water movement mean that external carbon that enters the system would have time to settle and become absorbed into the sediment.

Radionuclide values of the Fleets sediment

The radionuclide analysis confirms that sedimentation is low, but variable. The highest sediment accretion (0.086 ± 0.081) occurred at 1.75cm, around 1982. This is around the time when large floods hit the region. Eyewitnesses reported huge waves breaking over Chesil Bank and waves of 60ft hitting nearby Portland (Sutton, 1973). This unusual weather would account for sediment accretion double the average of the remaining data.

A simple regression analysis was conducted on age vs. depth (R^2=0.99) to model the age of sediment down the core (Fig. 12). The produced linear equation based on the accretion rates noted in the top 4cm was applied at depth intervals of 0.5cm:

$$y\ yr = 22.263x\ cm - 1.2451$$

Error increases with depth, but assuming sedimentation rate remains reasonably constant, the age of the deepest sediments at 100cm are estimated to be -207 BCE, with a range of between -1,142 BCE and 747 CE (Fig. 12). Even at the lower end of this value the carbon that has been laid down at these depths has been stored on millennial timescales, as is often assumed (Mateo *et al.,* 2006; Macreadie *et al.*, 2014). Having data to support the longevity of carbon stores in specific meadows reiterates the importance of these sites for long term carbon capture and storage.

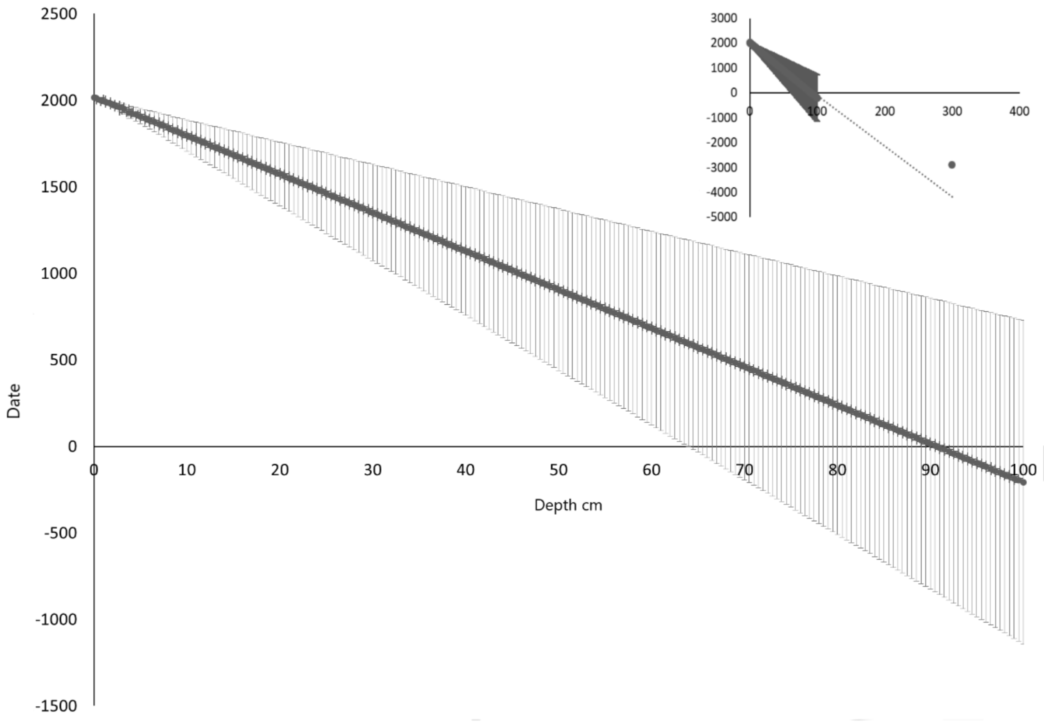


Figure 12. Projected age of the Fleet's seagrass sediment with associated error. Inset: projection with ^{14}C value.

This is a crude method of determining age, though, radiocarbon (^{14}C) dating at 3m, where peat starts to replace sand in the Fleet lagoon, have been dated -2,890 ± 70 BCE (May, 1980). Our model would project a far greater age at this depth, -4,660 BCE, though, with a lower range of -1,320 BCE the ^{14}C date is within its range. The suggestion is, therefore, that sedimentation rate has not been stable. The history of the formation of the Fleet would support this. Between around 3050 and 2050 BCE peat deposition occurred. This would have required high levels of shelter, suggesting it was a closed freshwater lagoon where sedimentation would likely have been lower than it is today (May, 1980). Chesil Beach began to form in its current state from 50 BCE, after which a more stable system began to emerge (May, 1980). Assuming sedimentation rate was stable up to the Fleet's current, enclosed, formation the point at which $\delta^{13}C$ ‰ and %OC values become more erratic (~50-55cm) would have been around 916 CE. Before this period the site was exposed to raising sea levels and subject to changes in form, representing a much more dynamic system (May, 1980). This may well account for the fluctuations noted in these values between 50-100cm depth. The more dynamic form of Chesil Bank may have resulted in variations in terrestrial vs. marine derived external carbon sources for this period.

Carbon stocks

The depth integrated carbon stock from 30cm cores extrapolated to 100cm (122.25 Mg C ha ± 12.79 Mg C ha) was substantially lower than the depth integrated carbon stock from 100cm cores (208.98 Mg C ha ± 11.67 Mg C ha). Based on these data the total carbon stored in the top 100cm of the Fleet lagoon if extrapolated from 30cm cores is estimated at 33,578Mg C, whereas the actual figure measured from the 100cm cores is 57,403Mg C. The extrapolated figure is almost half the total amount of carbon found in the Fleet lagoon, which is a significant under-representation.

This disparity arose because of a general increase in the %OC with depth in the deeper cores. This finding runs counter to the general expectation in the literature that %OC decreases with depth . In fact, an overall trend of increasing %OC at depth was present. The Fourqurean (2012) paper assumes that there are depth-dependent declines in seagrass carbon stocks among all its data. This calls into question the reliability of the global estimates where 82% of the data is extrapolated in this way (Fourqurean *et al.*, 2012). It may be that the Fleet lagoon is a unique system, and these findings are not typical for most seagrass meadows. However, as research effort and output increase, we are finding that the premise of a 'typical' seagrass meadow is itself unreliable. A plethora of studies have found that contemporary regional data does not fit the regional projections of studied sites (Lavery *et al.*, 2013; Röhr *et al.*, 2016; Githaiga *et al.*, 2017). More data on seagrass meadows within the same region as the Fleet will help elucidate these findings.

Value of the Fleet's carbon stores

There is increasing interest in placing a monetary value on ecosystems, in an attempt to communicate value to a non-ecological audience. There are many ways to value carbon in this way. The EU Emissions Trading Scheme currently values the carbon emitted by certain sectors at £24/t (Decc, 2011), and to ensure we remain below 1.5^{o}C the Grantham Institute suggest this value should rise to £160/t over the next 30 years (Burke *et al.,* 2019). The Stern Review suggests that for ecological systems the true social cost, which attempts to value the externalities of carbon, should be >£80/t (Stern, 2006). The actual traded value of 'blue' or 'green' carbon on the voluntary market, is actually much lower than any of these, currently around £7/t. Using this value the total carbon stored in the top 100cm of the Fleets' seagrass sediment equates to £401,821worth of carbon. Extrapolated from the shorter cores this value would be substantially lower £235,046). Taking the true societal cost of carbon, from the Stern Review, the value of carbon stored in the top 100cm of the Fleet's seagrass sediment would equate to £4.6 million.

There is a drive to increase awareness and conservation of seagrass meadows by promoting them as Blue Carbon habitats and valuing their carbon storage capacity. It is vitally important that research works to ensure these values are accurate so the full impact of the worth of seagrass meadows are fully appreciated.

There are wide variations in sediment accumulation rates among seagrass meadows, with rates as low as 0.015 cm y^{-1} and as high as 0.99 cm y^{-1} recorded among sites (Lavery *et al.*, 2013). Based on this range, the sediment accumulation rates at the Fleet seagrass meadow were low (0.044 cm y^{-1}). The meadow, therefore, accumulates 25 Mg C y^{-1}. Based on the same carbon trading figures used earlier, this is the equivalent of £599 of carbon per year. This may not seem a significant figure, but when you consider that the Fleet is just one of many seagrass meadows within the British Isles, it gives an indication of the potential support these habitats could give to climate change mitigation. Of course, the value of this site extends beyond its carbon storage capacity. The host of marine and bird life it supports, along with the unique characteristics of the Fleet lagoon and Chesil Bank provide other important provisioning and cultural services (Costanza *et al.*, 1997). This added knowledge expands the already recognised importance of this site.

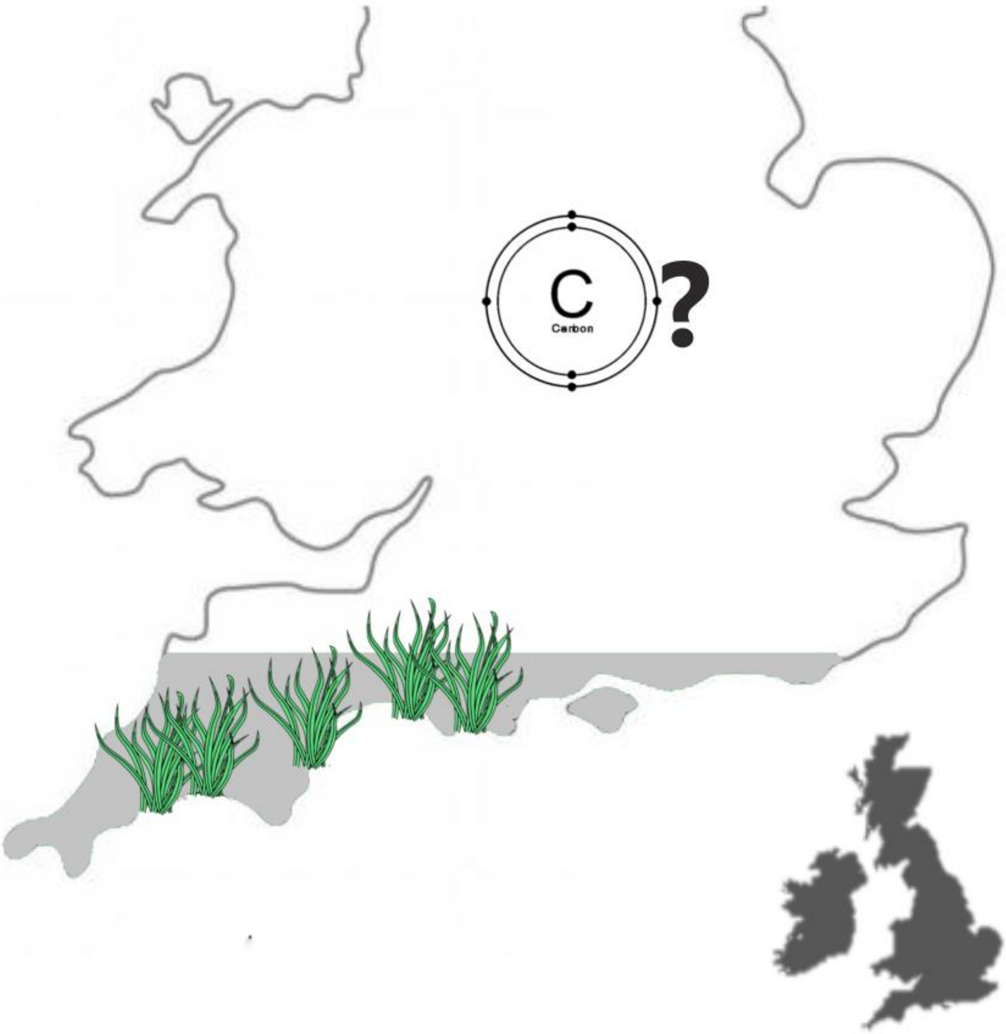
Conclusions

This Chapter described a baseline study that aimed to understand the dynamics of carbon storage in a seagrass meadow along the southwest coast of England. It has added to the knowledge that there are vast differences in the sediment dynamics of seagrass meadows, since the findings are contrasting to averages projected for the region. It supports the premise that seagrass meadows can store carbon on millenary timescales, since modelling from radionuclide analysis suggests sediment at 100cm depth contains carbon over 2,000 years old. It also supports findings that the sources of carbon within seagrass meadows are often equally derived from external sources and from seagrass itself. Importantly, it raises concerns about the globally accepted methods used to extrapolate carbon stocks from short cores to depths of up to 100cm. In the case of the Fleet, this method underestimates the amount of carbon stored by almost half, which would undervalue the entombed carbon by a significant amount.

The seagrass meadow in the Fleet lagoon contains a total of 57,403Mg C in the top 100cm of its sediments and is accumulating 2,497 Mg C y^{-1}. This equates to £1.4 million worth of stored carbon and £599 of carbon accumulated per year. These data provide an important first step in understanding the significance of the seagrass meadows of the British Isles. The following chapter looks to add to this understanding, by assessing the carbon stored within a further 12 seagrass meadows along the southwest coast of the England, to elucidate trends and variances in seagrass carbon storage.

Chapter 5

Variability of British Isles seagrass sediment carbon: implications for Blue Carbon estimates and marine conservation management



Introduction[3]

Seagrass meadows provide a multitude of ecosystem services, including a capacity to sequester CO_2 within their sediments (Smith, 1981). Along with mangroves and salt marshes, the organic carbon absorbed in these coastal ecosystems has been termed 'Blue Carbon' and has generated considerable interest in recent years, in part because preservation and restoration of these habitats can help mitigate climate change (Nellemann *et al.*, 2009). Unfortunately, seagrasses are declining with estimates that between 25% and 49% of British seagrass coverage has been lost in the last 35 years (Hiscock, Sewell and Oakley, 2005). This loss not only removes the sequestration potential of these habitats but can also remineralise sedimentary carbon that has accumilated over time, leading to a reduction of nursery and feeding habitat for commercially important and endangered speices (Jackson *et al.*, 2001), increases in sediment and coastal erosion (Fonseca and Cahalan, 1992) and reductions in coastline nutrient cycling (Hemminga and Duarte, 2000b; Orth *et al.*, 2006; Kennedy *et al.*, 2010b).

Zostera marina, the British Isles dominant seagrass species, is a temperate seagrass found throughout Europe, the USA and the northwest Pacific. Globally, seagrass is declining by approximately 1.4% per year, with large scale declines in some locations, particularly within Europe and east coast USA, due to wasting disease (Short *et al.*, 2010). The accountable pathogen, *Labyrinthula zosterae*, attacks the plants

[3] The data from this chapter has been published in PlosOne (Appendix 1). When citing this work, please cite the paper: Green A., Chadwick M. A., Jones P. J. S. (2018). Variability of UK seagrass sediment carbon: Implications for blue carbon estimates and marine conservation management. Plos One. 13 (9), e0204431. doi: /10.1371/journal.pone.0204431.

chloroplasts at first discolouring leaves, then leaving brown and black patches, before killing the plant altogether. It reportedly wiped out 90% of *Zostera* spp. in the early 1900's (Short *et al.*, 1988; Muehlstein, 1989). Much of the evidence of wasting disease in the British Isles is anecdotal, and with no complete historic inventory of seagrass meadows mapping accurate changes over time is challenging at best. Prior to the outbreak of wasting disease in the 1930s, *Z. marina* would have been found in the majority of subtidal mudflats in Britain, which was once considered 'clothed' in seagrass (Davidson and Hughes, 1998). Following the outbreak of wasting disease, seagrass was restricted to only the most sheltered sites, such as lagoons, and is now considered nationally scarce (Davidson and Hughes, 1998). Meadows that do persist are reportedly in a 'perilous state', damaged and degraded, and healthy beds are now a rarity (Jones and Unsworth, 2016).

Despite recognition by the EU Water Framework Directive of seagrass as bioindicators for ecosystem health (Foden and Brazier, 2007), research related to British seagrass habitats is lacking relative to other regions (e.g., Mediterranean and Australia (Fourqurean *et al.*, 2012)). More specifically, to the best of the authors knowledge, there are no published estimates for the carbon stored in the *Z. marina* habitats of the British Isles. The only other estimates I found were data from a PhD thesis of *Z. noltti* from Scotland (Potouroglou, 2017). This is surprising considering the proliferation of Blue Carbon research in recent years, with key papers (Foden and Brazier, 2007; McLeod *et al.*, 2011; Fourqurean *et al.*, 2012; Pendleton *et al.*, 2012; Duarte, Kennedy, Marba, *et al.*, 2013) highlighting the vital role seagrasses play in absorbing CO_2. Occupying less than 0.2% of the ocean floor, seagrass habitats are estimated to be responsible for approximately 10% of the yearly ocean carbon burial (Fourqurean *et al.*, 2012; Duarte *et al.*, 2013a; Duarte *et al.*, 2013b), a disproportionately large storage potential relative to their global extent (Laffoley, 2009). Seagrasses produce aboveground foliage forming canopies in the water column, which slow water, forcing sediment to settle and become trapped within the canopy layer. In this way particles from the water column are absorbed into their sediments as allochthonous carbon, where the overwhelming majority of the carbon stored by these habitats is located (Kennedy *et al.*, 2010). As highly productive plants, seagrass also stores autochthonous carbon through photosynthesising in excess of their need, burying the superfluous carbon in their sediments (Duarte and Cebrih 1996).

Seagrass ecosystems likely represent a '*globally significant carbon stock*', with estimates suggesting that 19.9 Pg carbon is stored in the top 100cm of the worlds' seagrass sediments, equivalent to the global fossil fuel and cement production in 2014 (Fourqurean *et al.*, 2012; Kennedy et al., 2010; Kerr, 2017). The Fourqurean paper (2012) has done much to increase awareness and has propelled seagrass into Blue Carbon research focus. However, values are derived from regional estimates, with between 1 and 29 data points and Mediterranean and Australian habitats comprise 42% of the total data points from this study (Fourqurean *et al.*, 2012). Further, the North Atlantic averages are from 24 samples, none of which are from British waters (Fourqurean *et al.*, 2012).

With such limited available data, these studies have been useful in promoting the advancement of seagrass carbon research. The challenge is that limited data means these estimates are biased regionally, and by species, so tend to generalise storage capture trends (Lavery *et al.*, 2013). Species with the highest known carbon storage capacity (i.e. *Posidona oceanica*) dominate the literature, which has been evidenced to skew regional and global extrapolations (Lavery *et al.*, 2013). Variations in carbon storage among species, and among habitats formed of the same species, are known

(Lavery *et al.*, 2013; Nordlund *et al.*, 2016), but the characteristics that affect this, and the impact of habitat distinction are less well understood (Lavery *et al.*, 2013; Nordlund *et al.*, 2016; Röhr *et al.*, 2016).

Direct measurements from regions and species that are under-represented will help to improve global knowledge and develop more reliable estimates of the carbon storage capacity and potential of seagrasses. For countries where Blue Carbon research has developed further, there has been a move towards incorporating it into domestic climate policy, going so far as to discuss the inclusion of Blue Carbon stocks within Greenhouse Gas (GHG) inventories (Bell-James, 2016). If efforts to integrate Blue Carbon into policy are to succeed robust estimates of regional carbon storage across the varied seagrass habitats are needed.

This chapter provides estimates of organic carbon (OC) density from 13 seagrass meadows to assess how British seagrass meadows vary in their carbon storage ability and whether these follow comparative regional trends. The objective was to obtain local estimates for carbon storage seagrass meadows of the British Isles to: 1) understand the variability of sediment carbon storage; 2) assess the impact of habitat variability on sediment carbon storage; 3) compare local carbon storage trends with global and regional data. The data was used to help elucidate the significance of the British Isles seagrass sediments in terms of Blue Carbon value. The tested hypotheses of this chapter are:

Hypothesis 1: There is significant variation in British Isles seagrass sediment carbon density.

Hypothesis 2: Aboveground biomass and sediment silt content significantly impact total carbon storage among British Isles seagrass meadows.

Hypothesis 3: Local seagrass sediment carbon data reveals inconsistencies in regional seagrass sediment carbon estimates, with implication for Blue Carbon schemes and seagrass conservation.

Methods

Study sites

The southwest coast of England contains a large density of seagrass meadows exhibiting varied habitat features and, therefore, provides an excellent opportunity to study several contrasting systems within proximity to one another. Thirteen seagrass meadows (Fig. 1, Table 1), considered representative of sub-tidal seagrass meadows found across the British Isles varying in size, degree of shelter and formation, were selected for this study (Chapter 3). In addition, sites represented varying degrees of marine protection, ranged from 0.02ha to 275ha and varied in aboveground density. Sites were located on the same latitudinal gradient between. 50° 18' 36.36" and 50° 38' 34.20"N.

Figure 1. Location of seagrass meadows along the southwest coast of the UK that were included in this study.

Table 1. Characteristics of the 13 surveyed seagrass meadows along the southwest coast of England.

Site	Protected status	Exposure	Meadow formation	Area (ha)	N	W
Cornwall						
Looe*	MCZ	Exposed	Very patchy	57	50° 21' 11.52"	4° 26' 30.48"
Plymouth, Devon						
Cawsands*	SAC	Partly sheltered	Very patchy	12	50° 19' 52.32"	4° 11' 53.52"
Firestone Bay*	SAC	Sheltered	Patchy	0.76	50° 21' 37.8"	4° 9' 37.44"
Drakes Island*	SAC	Partly sheltered	Dense	4	50° 21' 25.56"	4° 9' 10.08"
Jennycliff Bay*	SAC	Exposed	Patchy	12	50° 20' 27.96'	4° 7' 49.08"
Yealm CC*	SAC	Sheltered	Dense	0.14	50° 18' 36.36"	4° 3' 58.68"
Tomb Rock*	SAC	Sheltered	Sparse	0.15		
Torbay, Devon						
Elbery Cove*	MCZ	Sheltered	Sparse	29	50° 24' 17.64'	3° 32' 41.28"
Torre Abbey*	MCZ	Very exposed	Very patchy	104	50° 27' 38.52"	3° 32' 1.32"
Fishcombe Cove*	MCZ	Very sheltered	Very patchy	0.23	50° 24' 11.52"	3° 31' 17.76"
Hopes Cove*	SAC	Partly sheltered	Gradient	3	50° 27' 52.56"	3° 29' 16.44"
Dorset						
Fleet	SAC, SSSI, RAMSAR SPA, UNESCO	Very sheltered	Dense	275	50° 37' 72.20"	2° 33' 43.30"
Studland Bay	No protection	Very sheltered	Dense	53	50° 38' 34.20"	1° 56' 38.30"

*Abbreviations are as follows: MCZ = marine conservation zones, SAC = special area of conservation, SSSI = special scientific site of interest, RAMSAR = convention on wetland of international importance, SPA = special protected area, UNESCO = world heritage. Area values provided by CSI (Community Seagrass Initiative). CSI sites accessed during their summer exhibition highlighted with *.*

Field Methods

The majority of sites were accessed in the summer of 2016 on an exhibition run by the Community Seagrass Initiative (CSI). The exhibition afforded access to all their sites, though sediment conditions meant that I was only able to extract sediment from 11 of these (Table 1). At each site, two divers were dropped from a 40ft dive boat roughly in the centre of the bed and sampling locations, at least 20m apart, were randomly selected. At each sampling location one PVC sediment core was manually inserted 34-40cm into the sediment at sea depths of 3-8m using SCUBA gear. Additional sample collection from the Fleet and Studland Bay (not included in CSI's expedition) also occurred in the summer months. The data collected at the Fleet for Chapter 4 was also included in this study. At Studland Bay sampling locations were accessed using a small inflatable dingy with a motor. This provided divers with safety cover.

It was the intention of this study to extract long (100cm) sediment cores as well as short (30cm) ones. Extraction of long cores requires a diver to be in the water and at least two people on the surface guiding the core into the sediment. I was not able to do this on the CSI expedition because the additional time taken to extract deep cores would disrupt their research schedule. I tried, and failed, to extract long cores at Studland Bay because the substrate made it too difficult to insert the cores. To be able to extract long cores from the site included in this study, a boat and winch would be required, which were outside the budget constraints of this PhD.

In addition to the sediment cores, three 50cm^2 quadrats were randomly placed around each core and plant densities were estimated by counting the number of plants within the quadrant. Meadow exposure and bed formation were visually assessed during site visits. Cores were returned to shore, sliced into 3cm sections, bagged and frozen in the Plymouth National Marine Aquarium, Torbay Association of Inshore Fisheries, and

Conservation Authorities (IFCA) and Weymouth Wildlife Centre freezers, to await transfer back to the laboratory for analysis.

Size of meadow (ha) was provided by the CSI, apart from the Fleet, which was provided by Natural England, via the OSPAR dataset, and Studland Bay, which was estimated on Q-GIS using Google imagery.

Laboratory analysis

In the laboratory, samples were thawed and divided into two sub-samples. One sub-sample was analysed for dry bulk density (DBD) and percent organic matter (%OC) using Loss on Ignition (LOI: see Chapter 2), and the other was freeze-dried for grain size analysis and total organic carbon content (%OC) using an elemental analyser. The regression analysis determined by the author (Chapter 3) was used to correct %OM to %OC.

Sediment grain size was determined from freeze dried samples from one core for each meadow, which was assumed to be broadly representative of the entire site. Sediment samples were dry sieved through a sieving tower for 10 minutes. Seven sieves were used; 2mm, 1mm, 0.5mm, 0.25mm, 0.15mm, 0.125mm and 0.054mm. Total mass of sample and mass of retained soil in each sieve was recorded. Sediment silt content was calculated as the percentage of sediment retained below 54µm (0.054mm). Sediment characteristics were further analysed using GRADISTATv8 software (Blott and Pye, 2001).

Statistical comparisons for carbon stock (C_{stock}), DBD and plant density were conducted to determine site-specific differences. Test for normality and homogeneity of variance established if ANOVA or Kruskal Wallace test should be performed.

Results

The mean DBD in the studied seagrass sediments ranged from 0.34 ± 0.10 g cm^3 (Fleet) to 1.19 g cm^3 ± 0.09 g cm^3 (Studland Bay) with an average of 0.96 ± 0.22 g cm^3 (Table 2). The Fleet had significantly lower DBD than any other site ($p < 0.05$), with no significant difference in DBD among any of the other sites (Fig. 2).

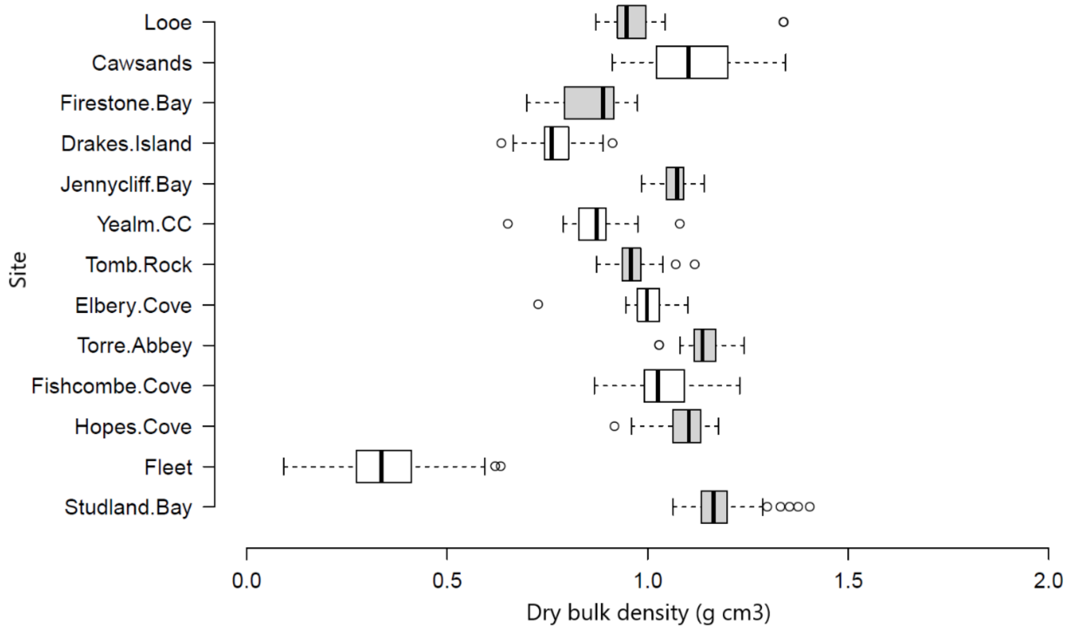


Figure 2. Dry bulk density (g cm^3) per study site. Values are means with standard deviation.

Mean %OM content ranged from 1.40% ± 0.67% (Studland Bay) to 12.32% ± 5.39% (Drakes Island) with an average of 3.61% ± 3.31% and a median of 2.47%. The Fleet and Drakes Island %OM were markedly higher (Table 2; Fig. 3) and differed significantly to all other sites ($p<0.05$). There was no significant difference between the Fleet and Drakes Island, and no significant difference between all other sites.

Table 2. Sediment characteristics and aboveground biomass from the 13 surveyed seagrass meadows in the southwest coast of England.

Site	Sediment silt content %	DBD ($g\ cm^3$)	%OM	%OC	SCD ($mg\ C\ cm^2$)	C_{stock} 30cm (Mg C ha)	Plant density ($plants/50cm^2$)
Cornwall							
Looe	20.03 ± 1.25	0.98 ± 0.10	2.3 ± 0.76	1.20 ± 0.31	11.08 ± 0.49	33.30 ± 1.47	8.53 ± 6.27
Plymouth Sound, Devon							
Cawsands	12.72 ± 1.77	1.11 ± 0.12	2.47 ± 0.74	1.25 ± 0.32	14.21 ± 1.08	42.07 ± 3.08	6.08 ± 5.76
Firestone Bay	13.34 ± 2.91	0.86 ± 0.08	3.47 ± 0.55	1.62 ± 0.31	14.19 ± 0.67	40.99 ± 3.38	4.05 ± 5.88
Drakes Island	5.51 ± 1.43	0.77 ± 0.07	12.32 ± 5.39	4.94 ± 2.00	37.76 ± 6.75	114.02 ± 21.45	10.42 ± 8.40
Jennycliff Bay	2.44 ± 0.66	1.07 ± 0.04	2.51 ± 0.43	1.30 ± 0.16	13.89 ± 0.65	39.07 ± 5.35	2.84 ± 4.75
Yealm CC	14.55 ± 1.70	0.87 ± 0.07	2.68 ± 0.48	1.37 ± 0.18	11.83 ± 0.21	35.39 ± 0.70	6.7 ± 7.01
Tomb Rock	8.85 ± 1.29	0.96 ± 0.05	1.85 ± 0.48	1.04 ± 0.21	10.15 ± 0.40	29.40 ± 0.65	4.21 ± 4.51
Torbay, Devon							
Elbery Cove	21.99 ± 2.46	1.05 ± 0.28	2.59 ± 0.47	1.33 ± 0.18	13.84 ± 0.56	41.74 ± 2.28	10.63 ± 9.45
Torre Abbey	12.02 ± 2.50	1.14 ± 0.09	1.97 ± 0.10	1.10 ± 0.04	12.56 ± 0.50	37.76 ± 1.50	5.52 ± 5.10
Fishcombe Cove	4.81 ± 1.79	1.04 ± 0.10	2.43 ± 0.65	1.28 ± 0.24	13.08 ± 0.72	38.94 ± 2.44	5.71 ± 7.64
Hopes Cove	14.71 ± 1.83	1.09 ± 0.07	1.56 ± 1.84	0.95 ± 0.68	10.73 ± 3.91	30.08 ± 8.89	7.61 ± 5.98
Dorset							
The Fleet	29.92 ± 5.30	0.34 ± 0.10	9.39 ± 2.95	3.82 ± 1.14	12.07 ± 1.49	37.76 ± 3.84	N/A
Studland Bay	1.99 ± 0.66	1.19 ± 0.09	1.40 ± 0.67	0.86 ± 0.27	10.13 ± 1.80	37.76 ± 5.39	53.53 ± 10.45

Data are site means ± standard deviation. % silt content; DBD = g dry bulk density; %OM = % organic matter; %OC = % organic carbon; SCD = soil carbon density mg C / cm^2; C sock Mg C ha = megagrams of C per hectare; plant density = no. plants per $50cm^2$

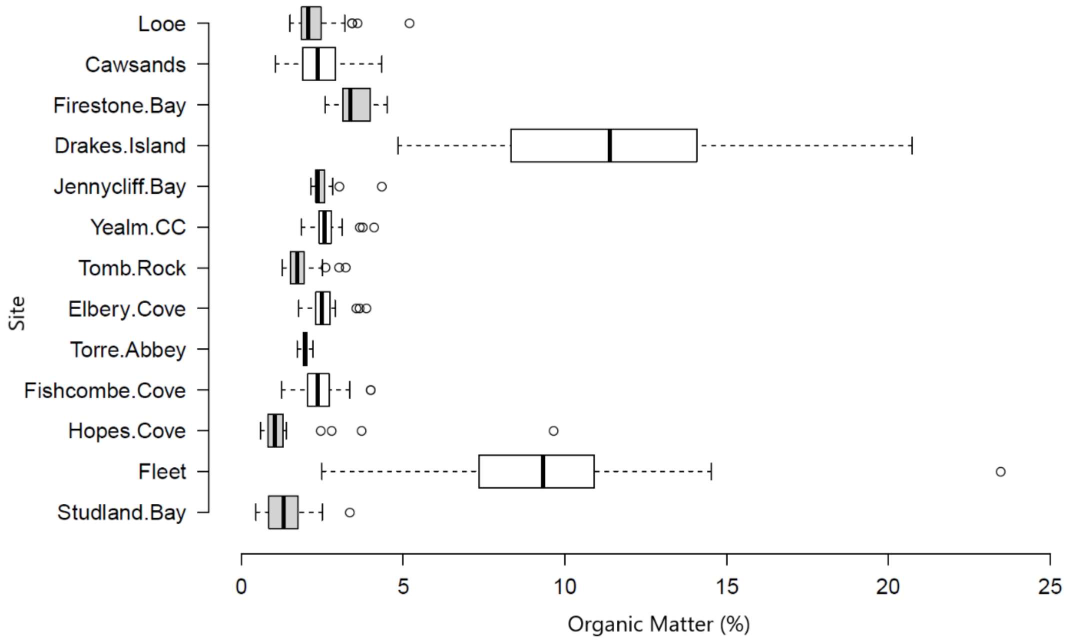


Figure 3. Organic matter (%) by site. Values are means with standard deviation.

Mean %OC content ranged from 0.86% ± 0.27% (Studland Bay) to 4.94% ± 2.00% (Drakes Island). Mean %OC was 1.70% ± 1.23% and median %OC was 1.28% (fig. 4). Sediment profiles showed no change at depth (Fig. 6). As with %OM, %OC at the Fleet (3.82% ± 1.14%) and Drakes Island (4.94% ± 2.00%) were significantly higher than all other sites ($p<0.05$) (Fig. 4). There was no significant difference between any of the other sites.

Mean soil carbon density (SCD) ranged from 10.13 ± 1.80 mg C cm^2 (Studland Bay) to 37.76 ± 6.75 mg C cm^2 (Drakes Island) (Table 2) with an overall average of 14.27 ± 7.21 mg C cm^2. Drakes Island was markedly higher than all other sites and was the only site to be significantly different ($p<0.05$). Integrated over a depth profile of 30cm, the C_{stock} of the studied seagrass meadows ranged from 29.40 ± 0.65 Mg C ha (Tomb Rock) to 114.02 ± 21.45 Mg C ha (Drakes Island), more than twice the value of the next highest C_{stock} (42.07 ± 3.08 Mg C ha at Cawsands), with an average of 41.54 ± 4.54 Mg C ha (Table 2).

Despite the high %OC at the Fleet, the low DBD meant that its C_{stock} was below average among the sites (37.76 ± 3.84 Mg C ha). Removing Drakes Island from the data reduces the range substantially with an average of 37.02 ± 4.22 Mg C ha. To allow for global comparisons C_{stock} was extrapolated to 100cm as per the IPCC guidelines for coastal wetlands (Santisteban *et al*, 2003; Howard *et al.*, 2014). The 100cm depth integrated C_{stock} among sites ranged from 98.01 ± 2.15Mg C ha (Tomb Rock) to 380.0 ± 71.51 Mg C ha (Drakes Island), with an average of 140.98 ± 73.32 Mg C ha. In both cases (C_{stock} 30cm and 100cm) there was a significant difference between the total C_{stock} of Drakes Island compared with all other sites. There were no significant differences between any other sites.

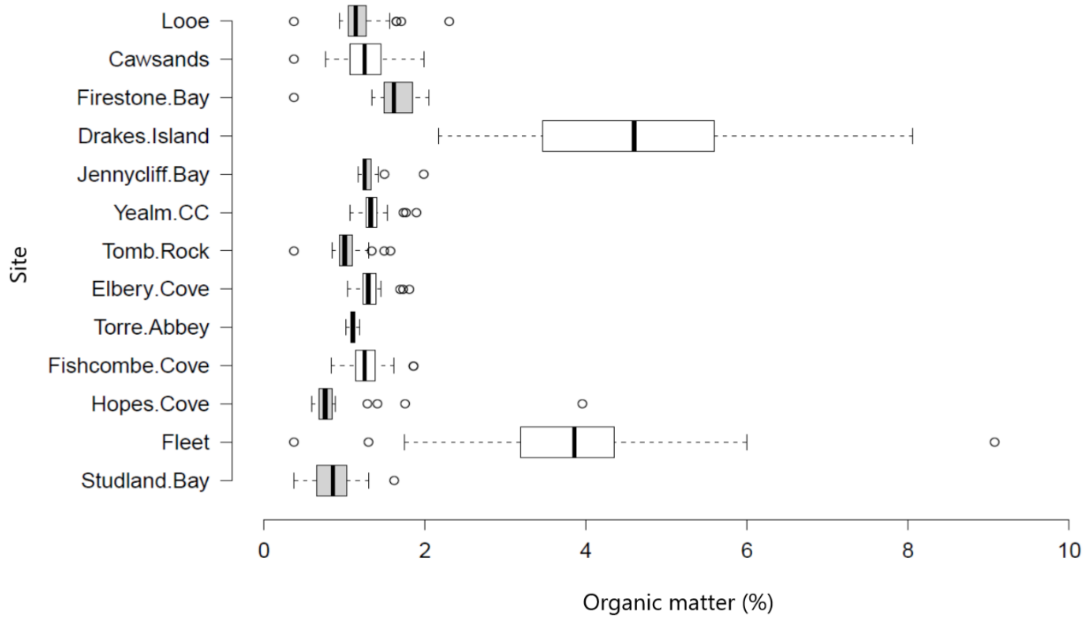


Figure 4. Organic carbon (%) content by site. Values are means with standard deviation.

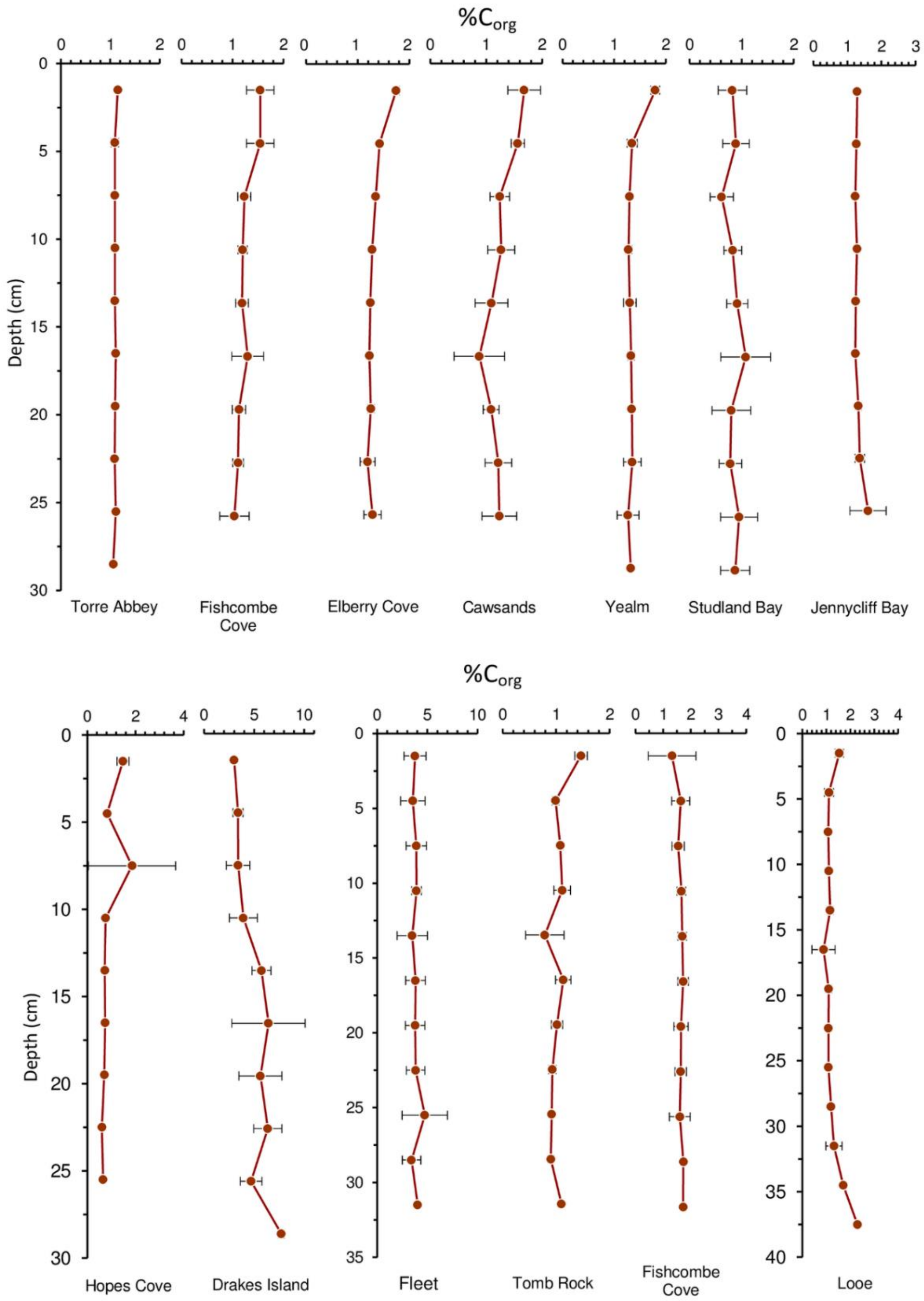


Figure 6. Organic carbon depth profiles of the top 25cm to 40cm of sediment cores from 13 sites along the southwest coast of England. Organic carbon is expressed as a mean percentage of the dry weight, with standard deviation. Note the variations in x and y axis.

Sediment characteristics also varied between sites, ranging from sand to sandy silt. Sediment silt content ranged from 1.99 % ± 0.66% (Studland Bay) to 29.92% ± 5.30% (the Fleet). Only Studland Bay and Drakes Island were statistically different from one another ($p<0.05$). The Folk and Ward description of sorting (Blott and Pye, 2001) ranged from Moderately Well Sorted to Very Poorly Sorted among sites (Table 2). The Fleet was the least well sorted (Very Poor), Cawsands and the Yealm were also Poorly Sorted. The remaining sites were Moderately and Moderately Well sorted.

Plant density ranged from 2.84 ± 4.75 plants per $50cm^2$ (Jennycliff Bay) to 53.53 ± 10.45 plants per $50cm^2$ (Studland) (Table 2). Studland Bay had statistically higher aboveground biomass than any other site ($p<0.05$), there was no significant difference between any other site. This contradicts the visual inspection of the sites and may show that taking biomass from the centre of the meadow is misrepresentative of the whole site. Many sites recorded high standard deviation compared to average plant count, highlighting the patchiness of some sites. Patchiness was particularly pronounced at Fishcombe Cove (5.71 ± 7.64 per $50cm^2$) and Jennycliff Bay (2.84 ± 4.75 per $50cm^2$). Standard deviation was high across all sites apart from Studland Bay (53.53 ± $10.46cm^2$), which was the most consistently dense meadow.

In general, the surveyed meadows ranged from dense uninterrupted beds (Fleet, Studland, Drakes Island) to open sand with small patches of seagrass cover (Cawsand, Firestone Bay). The Fleet and Studland Bay both contain large bare patches within their dense beds, the Fleet for reasons currently unknown and Studland Bay because it is a popular anchorage and contains numerous anchor scars. Site exposure differed among sites. The Fleet is a lagoon, flanked by Chesil Bank and connected to the sea by a narrow channel to the south that leads into Portland Harbour. In comparison, the meadow at Torre Abbey lies in the middle of a large bay, ~500m from shore, with frequent through traffic from the port, and no protection from oncoming weather.

Meadow size varied from 275ha (the Fleet) to 0.14ha (Yealm) with most sites smaller than 60ha (Table 1). Sea depth of site ranged from 2.5m (Studland Bay) to 7.7m (Hopes Cove). Average site depth was 5.10 ± 1.60m. The environmental data showed very weak regression relationships between most parameters and C_{stocks}: C_{stock} and plant density (R^2=0.003); C_{stock} and average site depth (R^2=0.034); C_{stock} and sediment silt content (R^2=0.064); C_{stock} and size (R^2=0.021) and; C_{stock} and dry bulk density (R^2=0.012). A weak correlation was noted between C_{stock} and %OM (R^2=0.372).

Discussion

This study is the first to estimate carbon storage from a range of British Isles *Z. marina* meadows. It confirms the first hypothesis that there is significant variation in British Isles seagrass sediment carbon density, despite contrasting habitat features. It also rejects the second hypothesis that aboveground biomass and sediment silt content significantly impact total carbon storage. These results, therefore, contradict a growing body of literature that has found variations in the carbon storage of seagrass meadows among habitats formed of the same species (Lavery *et al.*, 2013; Röhr *et al.*, 2016; Githaiga *et al.*, 2017). Although documenting large variation, these studies were unable to provide an adequate understanding of factors influencing OC accumulation and storage. These results suggest that habitat conditions do not meaningfully influence the C_{stock} within the studied seagrass meadows. The mechanisms which influence sediment carbon accumulation in seagrass meadows, therefore, remain unclear.

Drakes Island appears to be exceptional in its carbon storage ability in the region. The 100cm depth integrated C_{stock} at Drakes Island is nearly three times higher (380.07 ± 17.51 Mg C ha) than the average of all other sites (140.98 ± 73.32 Mg C ha). All other sites contained similar C_{stocks}, ranging from 98.01 ± 2.15 Mg C ha to 140.24 ± 10.27 Mg C ha. Other studies have found that accumulation of fine-grained sediments within seagrass beds significantly influences seagrass carbon storage (Röhr *et al.*, 2016; Miyajima, 2017). The relationship between sediment silt content and C_{stock} among these sites was weak, suggesting this was not an influencing factor among sites. Drakes Island had one of the lowest sediment silt contents (5.51 ± 1.43%) and the site with the highest silt content (Fleet 29.92 ± 5.30) did not have particularly high C_{stock}, although its %OC (3.82 ± 1.14%) was high and the low C_{stock} is likely due to the low dry bulk density at the site (0.34 ± 010). Aboveground biomass is also attributed to higher C_{stocks} among seagrass meadows (Samper-Villarreal *et al.*, 2016), though this was not evident in the data (R^2=0.003). Studland Bay had by far the highest average plant count (53.53 ± 10.45 per 50cm^2) (Table 2), but an average C_{stock} (37.76 ± 5.39). Plant count at Drakes Island was reasonably high (10.42 ± 8.40 per 50cm^2) but standard deviation was also high, suggesting a less uniform cover of dense growth, confirmed by visual inspection. Patchiness within sites was generally high, indicating potentially poor ecosystem health (Jones and Unsworth, 2016). Fishcombe Cove (5.71 ± 7.64), Jennycliff Bay (2.84 ± 4.75), Firestone Bay (4.05 ± 5.88) and Yealm (6.70 ± 7.01) all displayed vast variations among surveyed quadrats but overall no relationship was noted between plant count or patchiness, and C_{stock}.

That the expected trends are not identified within these results should not render them insignificant. It is likely that the high OC content found at the Fleet is in part attributable to the high sediment silt content. More intricate factors are likely involved that allow Drakes Island to store more carbon where its sediment is less suited and restrict Studland's sequestration capacity where its canopy is more favourable. This study was unable to assess the sources of carbon within the seagrass meadows, which can be an important influencing factor determining C_{stocks} (Röhr *et al.*, 2016). Sources of carbon contributed to up to 73% of the difference between carbon storage in *Z. marina* habitats in the Nordics (Röhr *et al.*, 2016). On average 50% of sedimentary OC is derived from allochthonous sources (Kennedy *et al.*, 2010b), and it may be that the ratio of carbon contribution (*Z. marina* : external sources) is an influencing factor. Further analysis should be considered to understand the relationships between C_{stock}, silt content and aboveground biomass among these sites.

Seagrass systems typically have very little sediment turnover (Burdige, 2007). Carbon diagenesis causes a gradual breakdown of labile and later increasingly stable carbon (Burdige, 2007). The result is an assumed decrease in organic matter at depth (Serrano *et al.*, 2012). The sediment profiles at these sites did not fit this trend. However, as is noted in Chapter 4 it is possible that the shallow 30cm cores are not sufficiently deep to note changes in OC with depth. To really understand the trends of OC with depth deep cores (≥100cm) should be assessed at all the sites, which would require substantial funding.

Carbon stock comparisons

The results support the third hypothesis that local seagrass sediment carbon data reveals inconsistencies in regional seagrass sediment carbon estimates with implication for Blue Carbon schemes and seagrass conservation. Mean sediment C_{stock} for the top 100cm of sediment (140 ± 73.32 Mg C ha) was just short of the global average of 194.2 ± 20.2 Mg C ha (Fourqurean *et al.*, 2012) (Fig. 2). The range of C_{stock} between sites was large (98.01 – 380 Mg C ha), but greatly reduced when Drakes Island (380.07 ± 71.51 Mg C ha), was removed (98.01 - 140.24 Mg C ha). Four sites fell below the globally documented range of 115.5 – 829.2 Mg C ha (from 41 100cm cores), though when you include the global extrapolated data from cores at least 20cm deep (extrapolated to 100cm), the range widens from 9.1 - 829.2 Mg C ha (Fourqurean *et al.*, 2012). In these cases, carbon values tend to be lower, so deeper cores at the surveyed sites may well reveal higher carbon stores. If the differences in trends noted in Chapter 4 occur here, the actual stock figures could be up to twice as high.

All the surveyed sites contain average C_{stock} well above the average for North Atlantic seagrass meadows (48.7 ± 14.5 Mg C ha) (Fig. 3) and increased the number of data points from 24 to 37 (Fourqurean *et al.*, 2012). Surprisingly, Drakes Island is comparable to the Mediterranean averages, dominated by *P. oceanica* (Fig. 3). It is not uncommon for sites to exhibit carbon stores well above those within its region (Röhr *et al.*, 2016). Understanding the mechanisms behind the carbon stores at Drakes Island might help to deepen our understanding of seagrass carbon storage.

The disparities between these results and the average for the North Atlantic further highlight the dangers of using global and regional data as a proxy for local seagrass carbon storage. There is a growing desire to use seagrass Blue Carbon as a mechanism to increase seagrass protection worldwide. Blue Carbon research has come under recent scrutiny (Johannessen and Macdonald, 2016) and to maintain robustness we must be transparent about the services provided by local habitats, and refrain from overgeneralising. The C_{stock} values documented for the British Isles seagrass meadows fall within the upper range of those recorded in the rest of Europe. Across Europe, estimates of *Z. marina* C_{stock} vary considerably, ranging from 500 ± 50.00 g C m^2 to 4,324.50 ± 1,188.00 g C m^2 in the top 25cm of sediment (Dahl *et al.*, 2016; Jankowska *et al.*, 2016; Röhr *et al.*, 2016) (Table 3). With an average C_{stock} of 3,372.47 ± 1,625.79 g C m^2 the south coast of England is second only to Denmark. The variation between regions is considerable. This data and Denmark contain anomalous sites with significantly higher C_{stocks} than the rest of their region; 8,649.93 ± 2,330.02 (Drakes Island) and 26,138 ± 385.00 (Thurøbund) respectively, but also consistently higher C_{stocks} across all sites when compared to the rest of Europe.

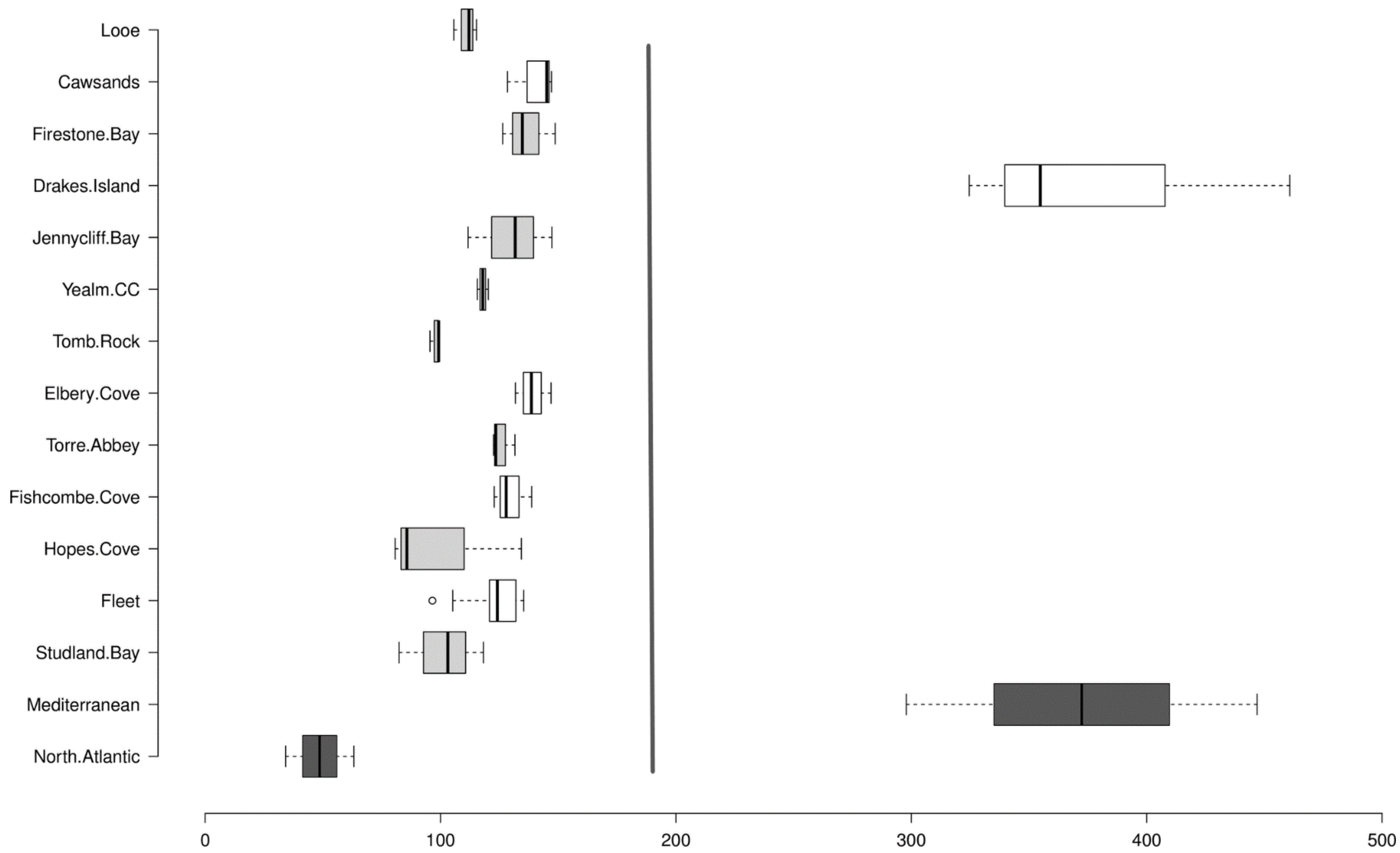


Figure 3. Average C_{stock} of 13 seagrass meadows along the southwest coast of England with regional comparisons (dark grey Mediterranean and North Atlantic) and global average (grey line) extrapolated from (Fourqurean *et al.*, 2012).

As with Drakes Island, no obvious explanation for the Danish sites' high carbon content was given, above its location in a 'relatively sheltered site' and large amounts of organic sediments (Röhr *et al.*, 2016). This study found greater variations in the C_{stocks} of seagrass sediments than our study noting that seagrass production, root : shoot ratio and contribution of *Z. marina* to the carbon pool explained 67% of the variation. Similar analysis at the sites included in this study would make an interesting comparison here. The large variation among regions, demonstrated by these studies, further highlights the risk in using global and regional data as a proxy for estimating local Blue Carbon values. It also confirms that even within species, there is considerable variation in seagrass carbon storage capacity and suggests that abiotic factors are more important than biological. Although the drivers remain unclear, the carbon stored in the seagrass meadows along the southwest coast of England represents one of the largest known stocks within Europe and, therefore, represents important sites for further study and conservation.

Table 3. Mean C_{stocks} in European *Z. marina* meadows

Country	Region	C_{stocl} (g C m^2)	Depth (cm)	Sampling locations (no.)	Reference
Denmark	*North Sea*	4324 ± 1188	25	10	Röhr et al, 2016
UK	*English Channel*	3371 ± 1625	25	13	Present study
Sweden	*Baltic Sea*	2000 ± 2121	25	5	Dahl et al, 2016
Portugal	*North Atlantic*	1000 ± 120	25	2	Dahl et al, 2016
Finland	*Baltic Sea*	627 ± 25	25	10	Röhr et al, 2016
Bulgaria	*Black Sea*	500 ± 50	25	2	Dahl et al, 2016
Poland	*Baltic sea*	148 ± 90	10	3	Jankowska et al, 2016

That seagrass meadows can also be a source of CO_2 and atmospheric methane (CH_4) has largely been neglected in the literature (Garcias-Bonet and Duarte, 2017; Howard *et al.*, 2018). A recent study suggests that seagrass could be contributing up to 30% more to the global CH_4 emissions than previously thought, and calls for these emissions to be included in seagrass carbon calculations (Johannessen and Macdonald, 2016). There is also a lack, at the root of Blue Carbon science, of adequate understanding of how OC accumulated in soils can be remineralised to CO_2 and re-released back into the water column, where it has the potential to enter the atmosphere (Howard *et al*., 2018). A recent paper suggests the dissolution of calcium carbonate from the inorganic carbon pool has the potential to undermine the carbon sequestration capacity of seagrass meadows, in some cases perhaps shifting habitats to carbon sources (Howard *et al.*, 2018). These mechanisms need more exploration and will vary regionally. Regardless, they call into question the reliability of global seagrass carbon sequestration estimates. Unfortunately, these considerations are outside the scope of this PhD, the core aim of which is to provide the first estimates of carbon standing stock in British Isles seagrass sediments. It is argued that this data is much needed, especially within the current climate of forwarding marine conservation goals. Thus, stock calculations alone provide vital, much needed, information on this under-studied habitat. Hopefully, future studies can investigate the flux of carbon, and further add to the data pool both locally and globally.

Valuing British Isles seagrass carbon stocks

There were marked differences in the sizes of seagrass meadows in the surveyed sites, and by association, the carbon pools within these (Table 4). The estimated total carbon pool in the top 100cm of the surveyed sites ranged from 14.52 Mg carbon at Tomb Rock to 33,578.31 Mg carbon at the Fleet. Despite the high C_{stock} found within Drakes

Island, the site itself is very small (4.25ha) and contains only an estimated 1,616.67 Mg carbon within the top 100cm of its sediments. The estimated carbon pool in the top 100cm of the 13 surveyed sites along the southwest coast of the England was 66,337 Mg carbon, or the equivalent of 10,512 UK peoples CO^2 emissions per year. This is clearly not a significant number in terms of the UK's GHG emissions. However, for an area covering half the size of Richmond Park (London's largest park) this figure is significant relative to its size. The Fleet is a large seagrass meadow and contains 10% of the annual CO^2 emissions of its closest town (Weymouth). The seagrass beds within this study make up a fraction of those found in the British Isles. A number of studies have estimated the areal extent of seagrass meadows in the British Isles, although the actual extent remains uncertain (Davidson and Hughes, 1998; Maddock, 2008; Luisetti *et al*., 2013; Garrard and Beaumont, 2014). The total mapped area of *Z. marina* is 4,887ha (Luisetti *et al.,* 2013), though not all seagrass beds have been mapped. This figure is derived from some Special Areas of Conservation (SAC) and additional data from published studies only (Garrard and Beaumont, 2014). Other estimates have suggested that up to 22,066ha of seagrass can be found in the UK, but these are from some published data and point data which has been extrapolated to represent areal cover, and the paper doesn't provide detail on how this figure was arrived at (Jackson *et al.,* 2012). A generally accepted estimate of seagrass extent seems to fall between 5,000 and 10,000ha (Davidson and Hughes, 1998; Maddock, 2008; Luisetti *et al.,* 2013; Garrard and Beaumont, 2014). Taking the average from this study the estimated total standing stock of carbon in British Isles seagrass meadows is, therefore, between 108,427 and 221,870 Mg carbon. This is substantially higher than the Garrard and Beaumont (2014) estimates which, using *Z. marina* carbon stocks from European sites, estimated that British seagrass meadows had the potential to store between 8,050-16,100 Mg carbon. To fully grasp the significance of these stocks, a full inventory of the British Isles seagrass habitats should be completed (Chapter 7) and sediment cores from a wider range of meadows analysed. Further, the sequestration rate of these beds should be analysed to understand how much carbon per year these sites are able to sequester. Using the traded value of ecological carbon on the voluntary market (£7/t) the estimated value for this seagrass sedimentary carbon stock is between £750,000 and £1.5 million. Using the Stern Review's societal value of carbon (>£80), the upper rage of this stock could be valued at £17.8 million (Stern, 2006).

Table 4. Mean C_{stock} and equivalent monetary value of the 13 surveyed seagrass meadows along the southwest coast of the UK

Site	C_{stock} 100cm (Mg C ha)	C_{stock} 25cm (g C m^2)	Size (ha)	Total C (Mg C ha)	Monetary value
Cornwall					
Looe	111 ± 5	2,644 ± 146	56.52	6,274	£43,918.00
Plymouth Sound, Devon					
Cawsands	140 ± 10	3,437 ± 229	11.77	1,651	£11,557.00
Firestone Bay	137 ± 11	3,253 ± 271	0.76	104	£728.00
Drakes Island	380 ± 18	8,649 ± 2,330	4.25	1,615	£11,305.00
Jennycliff Bay	130 ± 18	3,273 ± 95	11.77	192	£1,344.00
Yealm CC	118 ± 2	2,883 ± 10	0.14	16	£112.00
Tomb Rock	98 ± 2	2,397 ± 69	0.15	15	£105.00
Torbay, Devon					
Elbery Cove	139 ± 8	3,344 ± 204	29.31	4,078	£28,546.00
Torre Abbey	126 ± 5	2,995 ± 120	104.11	13,106	£91,742.00
Fishcombe Cove	130 ± 8	3,175 ± 143	0.23	30	£210.00
Hopes Cove	100 ± 30	2,539 ± 812	2.73	274	£1,918.00
Dorset					
Fleet	122 ± 13	2,850 ± 376	274.68	33,578	£235,046.00
Studland Bay	101 ± 18	2,390 ± 432	53.37	5,404	£37,828.00

C_{stock} Mg C ha = mean megagrams of C per hectare over 100cm profile ± standard deviation; C_{stock} g C m2 = mean grams C per M^2 over 25cm profile ± standard deviation; Size = meadow size; total C = total C in top 100cm Mg C ha. Monetary value = £/t on voluntary market (£7/t)

Conservation implications

This study adds to the growing literature base that highlights the importance of British Isles seagrass habitats (Jackson *et al.*, 2001; Bertelli and Unsworth, 2014; Jones and Unsworth, 2016). Despite the growing knowledge that *Z. marina* beds in Britain are nursery grounds for economically important fish species (Bertelli and Unsworth, 2014), and that they are mostly in a poor ecological condition (Jones and Unsworth, 2016), conservation of these habitats is lacking.

At the time of writing (2017) Studland Bay was the only site without any legislative protection, although it has since been designated as a Marine Conservation Zone (MCZ) in 2019. The remaining sites are protected either as Special Areas of Conservation (SACs) or as MCZs, apart from the Fleet, which is a SAC, a Site of Special Scientific Interest (SSSI), a RAMSAR site (Wetlands), a Special Protected Area (SPA) and a UNESCO World Heritage Site (Table 1). Despite these designations, there are no restrictions on dropping anchor at any of the SAC or MCZ sites. Studland Bay, Fishcombe Cove and Cawsands are favoured anchorage sites for yachters and have several anchor scars within their meadows. The impact of anchoring activities on seagrass beds is contested, especially in Britain where the yachting community are greatly opposed to any anchorage restrictions. However, a recent paper (Serrano *et al.*, 2016) has unequivocally demonstrated that direct scouring of the bed by anchors, and the subsequent resuspension and loss of fine-grained sediments as a consequence, has resulted in a loss of OC content in disturbed areas. Scars showed evidence of intensive sediment mixing, which lead to the OC stocks being significantly lower than sediments under undisturbed seagrass (Serrano *et al.*, 2016). In British Isles seagrass meadows, moorings, which are also present at Studland Bay, have also been shown to negatively impact seagrass cover with one mooring chain potentially responsible for the loss of up to 122m^2 of local seagrass (Unsworth *et al.*, 2017).

The designation of Studland Bay as an MCZ in May 2019 occurred despite forceful opposition from the yachting community (DEFRA, 2016). Attempts to introduce conservation methods, such as installation of ecologically friendly moorings, have largely focused on the occurrence of charismatic seahorse species in the bay (Garrick-Maidment *et al.*, 2010). This flagship approach often fails to entice the diversity of stakeholders needed to ensure effective conservation (Simberloff, 1998) and attempts here have further polarised views between conservation and yachting communities.

To help broaden the arguments for conservation I submitted data to the MCZ consultation of Studland Bay, highlighting that the total estimated carbon in the top 100cm of its seagrass meadow holds a value of £129,695 (since this submission I have revised the estimated monetary values). This was comparable to the yearly recreational activities that could, in part, be effected by reduction in anchoring (£93,100) (Defra, 2018). Whether this was used in the monetary evaluations at the site or was part of the evidence that weighed in favour of designation is unknown. In reality, the MCZ designation does not disclose how the seagrass meadow should be 'returned to favourable condition' (DEFRA, 2019b). Building an evidence base that shows how mooring activities impact the services provided by the seagrass meadow (Chapter 6) may assist the formulation and implementation of management practices.

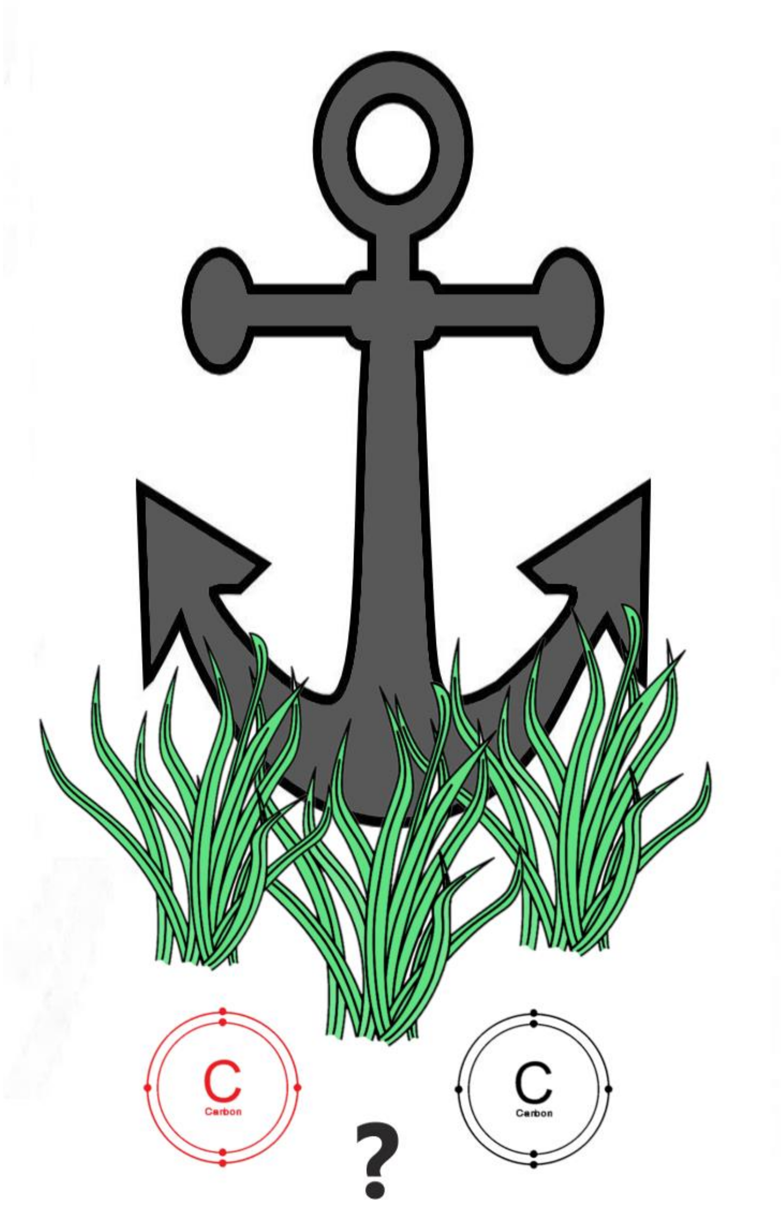
Conclusions

This study provides the first data on *Zostera marina* sediment carbon storage in the British Isles and offers a more accurate estimation of seagrass Blue Carbon stocks in British waters. The work brings 13 more seagrass meadows into the global and regional dataset and, like many other studies, highlights uncertainties surrounding the variances in sediment carbon storage. The results show considerable uniformity, which is unusual, and, in line with other research, indicate an incomplete understanding of the factors that influence this. Considered alone, the uniformity of the sites within this study suggests abiotic factors are not a strong driver of sediment carbon variability. However, when estimates of carbon storage from other European *Z. marina* meadows are considered, it seems clear they are the primary cause of variance. Although unable to identify the drivers for this, the seagrass meadows along the southwest coast of the UK contain C_{stocks} that are significant in a European context and are, therefore, important both ecologically and in terms of ecosystem services to the region. I would argue that, for Blue Carbon purposes at least, grouping seagrass into bioregions is not a useful way to discuss similarities or differences, as even the same species within the North Atlantic bioregion vastly contradict each other.

Studies like this provide an essential snapshot of the complex processes that influence carbon sequestration. Detailed analysis of sedimentary structure, hydrodynamic regime and seagrass canopy structure is vital if we are to better understand the causes of variation. Without this detail, global estimates will remain unreliable. Only by documenting inter-habitat variability will we be able to extrapolate the importance of seagrass ecosystems in a meaningful way, and thereby justify and promote measures for their improved protection.

Chapter 6

Anchoring and mooring reduce carbon storage within *Zostera marina* sediments in Studland Bay, UK



Introduction

Seagrasses are marine flowering plants, the only marine *Angiosperm*, found along shallow, sheltered coastlines in all but the most polar seas (Green and Short, 2003). As *Angiospermae* they require high levels of water irradiance to allow for photosynthesis. They are typically found between depths of 0-10m (Green and Short, 2003), occupying the shoreline and thus competing for space with coastal activities, such as fishing and boating. They are predominately rhizomatous in growth, with often a single genetic plant extending along a rhizome to form large, dense meadows (Hartog, 1970). Because of this, growth can be slow, and they can similarly be slow to recover from disturbance, especially that which causes in-meadow blow-outs; bare or sparsely vegetated open areas in a meadow caused by physical disturbance from continuous wave action or anchoring/mooring activity (Macia and Robinson, 2005).

Seagrass meadows are threatened globally; it is estimated that the spatial extent of seagrass has reduced by at least 29% since the 1980's (Waycott *et al.*, 2009; Short *et al.*, 2010). Their characteristics mean that coastal activities threaten populations at wide spatial scales. Coastal development, land reclamation and nutrient and pollutant encroachment can decimate entire meadows (Orth *et al.*, 2006; Waycott *et al.*, 2009). More locally, they are threatened by direct physical disturbance caused by recreational and small-scale fishing boat activity, which are significant global nature-based industries (Walker *et al.*, 1989; Diedrich *et al.*, 2013). The shallow sheltered coastlines they predominately occupy provide favourable conditions for recreational boaters, who often set up moorings or drop anchors within seagrass meadows, damaging them and the multitude of services they provide (Hastings *et al.*, 1995; Serrano *et al.*, 2016; Unsworth *et al.*, 2017).

Anchoring and mooring can impact seagrass at an individual and population level (Montefalcone *et al.*, 2008). Heavy mooring chains, dragging anchors, and, in shallow environments, boat propellers can directly scour meadows, ripping seagrass from the ocean floor and uprooting the roots and rhizomes that anchor them to the sandy sediment. These types of activities can cause in-meadow blow-outs and small-scale but persistent damage, evidence of which is widespread (Walker *et al.*, 1989; Demers *et al.*, 2013; Diedrich *et al.*, 2013; Deter *et al.*, 2017). At the population level such boat-related activities can increase sediment resuspension, increasing turbidity (i.e., reducing light availability and thus photosynthesis capacity). Moreover, boat presence is often associated with meadow-wide increased nutrient and pollution encroachment (Marba *et al.*, 2002; Montefalcone *et al.*, 2008). These anthropogenic disturbances can lead to long-term loss of seagrass and macroalgae, and altered carbon, nitrogen and phosphorous storage in associated sediments (Bourque *et al.*, 2015; Macreadie *et al.*, 2015). In fact, long-term disturbance to seagrass meadows has been shown to release sediment-stored carbon into the water column, which likely accumulated over hundreds to thousands of years (Macreadie *et al.*, 2015), although no data exists for this in the British Isles.

The British Isles predominant seagrass species is *Zostera marina*. This subtidal species is normally found up to 10m depth, within the boundaries of light limitations of Britain’s coastal water bodies. The impact of boating activities on British seagrass meadows has not had much attention. One study has shown that damage can cause sustained local losses (Unsworth *et al.*, 2017). Despite their ability to flower, British *Zostera* species do not support blow-out recruitment by localised flowering (Davidson and Hughes, 1998; Wilkinson and Wood, 2003), meaning disturbance can be persistent.

Despite protection form a number of legislative forces (see Chapter 2), British seagrass meadows are in a perilous state, degraded by poor water quality and direct disturbance from agriculture, coastal development, effluent encroachment and recreational boating and fishing activities (Jones and Unsworth, 2016). Seagrasses are often a named feature of Marine Conservation Zones (MCZ), which are designated under section 116(1) of the Marine and Coastal Access Act 2009, and form the UK's contribution to an international network of protected sites in the northeast Atlantic (DEFRA, 2013, 2019a). Generally MCZ's have the conservation objective of each of their designated features being maintained or recovered to a favourable condition (DEFRA, 2013). However, guidelines are commonly vague on how returning or maintaining favourable conditions should be achieved.

Research on the carbon storage capacity of British Isles seagrass habitats has only recently been explored (Green et al., 2018; Chapter 5) and understanding what activities threaten this important service can help inform management strategies for protecting this natural carbon store, and aid the recovery of meadows. Despite the recognition that seagrass meadows are negatively impacted by boating (Unsworth et al., 2017), and that similar disturbances can release stored carbon into the atmosphere, very few studies have documented this in situ. Those that have are conflicting in their findings, either noting vast reductions in carbon accumulation, (Serrano *et al.*, 2016a), or little difference between experimentally disturbed and undisturbed meadows (Macreadie *et al.*, 2014b).

Studland Bay, on the south coast of England, contains a large seagrass meadow, which is a favoured anchorage for yachters coming out of Poole harbour. Originally proposed as one of the 127 MCZ designations across England in 2011, it was pushed out of trance one and two, due mainly to the objections of the local yachting community (DEFRA, 2016). In May 2019 Studland Bay was finally designated as an MCZ, despite over 15 years of successful lobbying from the yachting community. The aim of this study is to document the difference in carbon storage from disturbed and undisturbed seagrass sediment samples in Studland Bay to assess the impact anchoring and mooring activities have on carbon storage. This was done by testing the following hypothesis:

Hypothesis: Sediments from within the seagrass meadow at Studland Bay contain more organic carbon than in-meadow bare patches created by mooring and anchoring, and adjacent bare patches that never knowingly contained seagrass.

The chapter aims to document the impact of these activities to help inform management practices and conservation for Studland Bay and other seagrass meadows where anchoring and mooring activities occur.

Methods

Study site

Studland Bay (Fig. 1) (50˚38’34.20’’2˚N; 1˚56’38.30’’W) is a small sheltered bay covering 4,000ha located on the Dorset coast in the eastern English Channel (DEFRA, 2018). It is flanked by a sharp cliff that cuts the bay off from south-westerly weather, making it a favoured anchoring location for small boats. The protected bay has a sandy shallow seabed upon which is a 54 ha seagrass meadow (DEFRA, 2018). The meadow is recognised as one of the healthier and more abundant seagrass meadows in Britain (Bull *et al.*, 2012; Jones and Unsworth, 2016), though it contains a number of blow-outs and scours formed by boating activities (Fig. 1).

Famously, it is one of the only known breeding ground for the two rare seahorse species, the short snouted seahorse (*Hippocampus hippocampus)* and the long-snouted seahorse (*Hippocampus guttulatus*) found in the British Isles. It also supports a number of other commercially important fish species, including bass, bream and flatfish, and endangered species such as the undulate ray (DEFRA, 2018). Finally, it is recognised by Natural England as one of the best recovered sites since the decimation of the British Isles seagrass by wasting disease in the 1930s (Bull, Kenyon and Cook, 2012).

Studland Bay is also one of the most highly contested seagrass meadows in England. The yachting community are stanchly against any restrictions to anchoring at the site, and seahorse lobbyists have aggressively called for complete anchor bans for many years (Garrick-Maidment *et al.*, 2010). The opposing views have created such a turbulent relationship between the two groups that any attempts at resolution are met with animosity (Green *et al.*, 2018). As a popular anchorage, the meadow contains a number of block and chain moorings, a mooring type used widely across Europe (DEFRA, 2018; Glasby and West, 2018). However, in the summer months the sheer extent of visitors means that boaters drop anchor in their hundreds (Collins, Suonpää and Mallinson, 2010). The site is also subject to low levels of fishing activity, mainly in the form of potting, and occasional trawling and dredging (DEFRA, 2018).

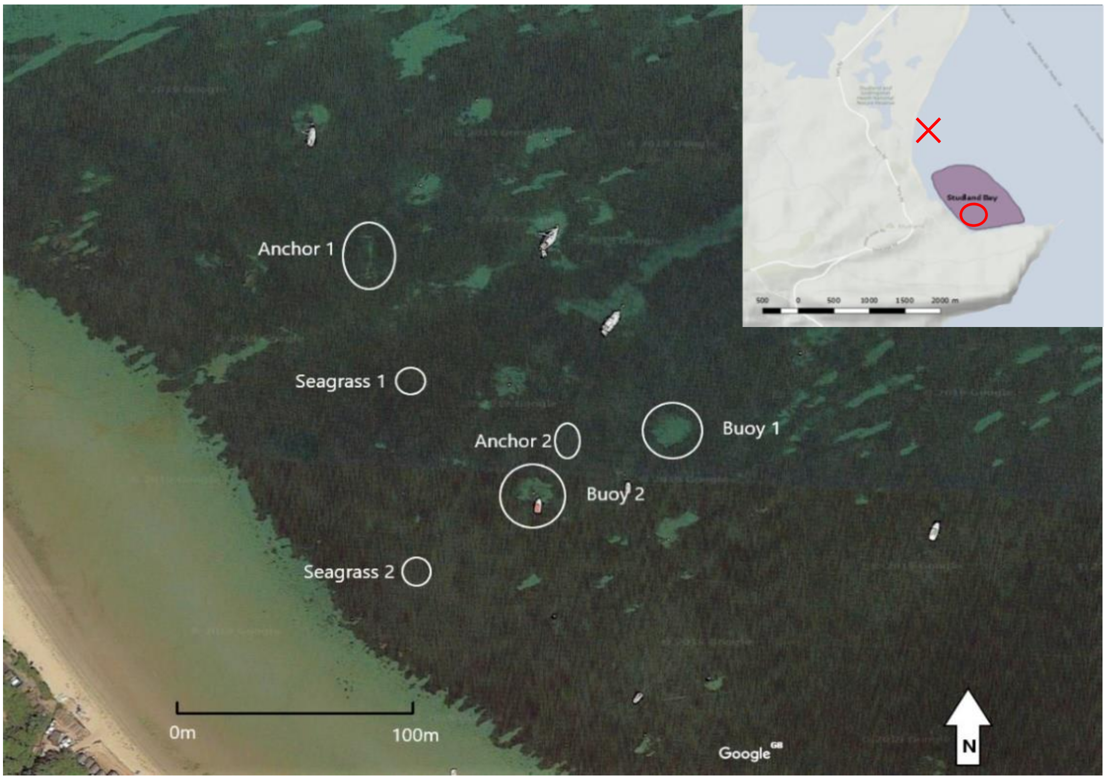


Figure 1. Seagrass meadow (purple) in Studland Bay with in-meadow survey location denoted with red circle and location of samples taken outside of the seagrass meadows denoted with red cross. Main image: sampling locations for the in-meadow conditions included in this study; anchor scar; buoy scar and seagrass.

Field Methods

Sampling was conducted at Studland Bay in the summer months of 2018, prior to its designation as an MCZ. Eight sediment cores were extracted from four different conditions: two in the undisturbed meadow, two around permanent moorings (buoy), two in anchor scars, and two in the north of Studland Bay where seagrass has never knowingly been present (Fig. 1). Sediments were sampled using SCUBA gear and manually inserting PVC cores (30cm long, 7cm diameter) into the sediment at a water depth between 3m and 5m. Cores were pre-cut and secured together with strong tape (Chapter 2). Once collected, cores were returned to shore, kept cool and upright in an ice-fed cool box and returned to the UCL lab for processing. Samples were frozen within 48 hours.

Laboratory analysis

In the laboratory, cores were opened lengthways, and the sediment was sliced into 3cm sections. Sediment from each slice was dried to determine dry bulk density (DBD) and percent organic matter (%OM) by Loss on Ignition (LOI) (Chapter 2). The relationship between %OM and organic carbon (%OC) was determined by applying the equation from elemental analysis from Chapter 2 to %OM samples (see chapter 2 & 5).

Data analysis

Due to the logistical constraints of underwater sampling at this site, I was only able to collect two replicates per treatment. I therefore used non-parametric techniques to evaluate differences, As the cores were divided in to 3cm sections a total of 12 samples per treatment were used to evaluate differences between treatments. A Kruskal Wallis test was used to confirm difference between treatments and pairwise comparisons were analysed using Wilcox rank sum test.

Results

Dry bulk density ranged from 1.10 g cm^3 and 1.51 g cm^3 with an average of 1.34 ± 0.13 g cm^3 (Table 1; Fig. 2). Dry bulk density was significantly higher in the sediments taken from anchor and buoy scars compared to the samples taken from beneath seagrass and the bare sand ($p<0.05$). There was no significant difference between anchor or buoy sediment samples, or between seagrass and bare sediment samples.

Table 1. Average dry bulk density g cm^3 (DBD), % organic matter content (%OM) and % organic carbon content (%OC) from four environmental conditions in Studland Bay ± standard deviation.

	DBD g cm^3	**OM** %	**OC** %
Anchor Scar	1.43 ± 0.05	0.52 ± 0.06	0.57 ± 0.02
Mooring Scar	1.42 ± 0.43	0.84 ± 0.43	0.68 ± 0.16
Seagrass	1.17 ± 0.65	1.61 ± 0.65	0.97 ± 0.24
Bare sand	1.39 ± 0.29	0.83 ± 0.67	0.68 ± 0.45
Studland Bay overall mean	*1.32 ± 0.13*	*0.95 ± 0.64*	*0.72* ± 0.23

Organic matter (%) content ranged from 0.40% to 3.35% with an average of 0.95% ± 0.64% (Table 1). Sediments within undisturbed seagrass had the highest %OM content (1.61% ± 0.65%), significantly higher than all other sites ($p<0.05$) (Fig. 3). Sediments taken from anchor scars had the lowest %OM content (0.52% ± 0.06%) (Fig. 3). Sediment from within mooring scars and from bare sand had the same %OM content (0.84% ± 0.43% and 0.83% ± 0.67% respectively) (Table 1; Fig. 3).

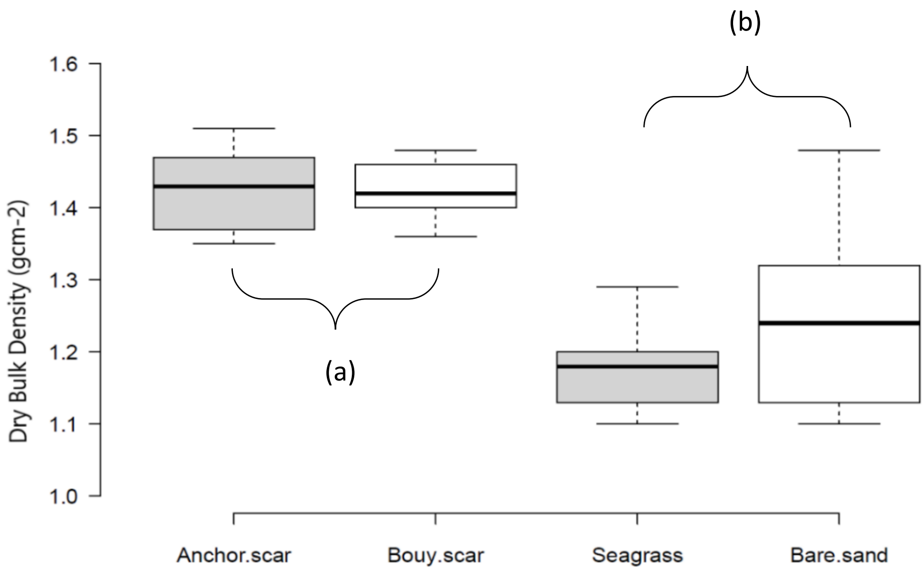


Figure 2. Sediment dry bulk density (g cm^2) from four environmental condition in Studland Bay. Letters indicate statistical grouping. Box plots represent all data collected across depth n=12.

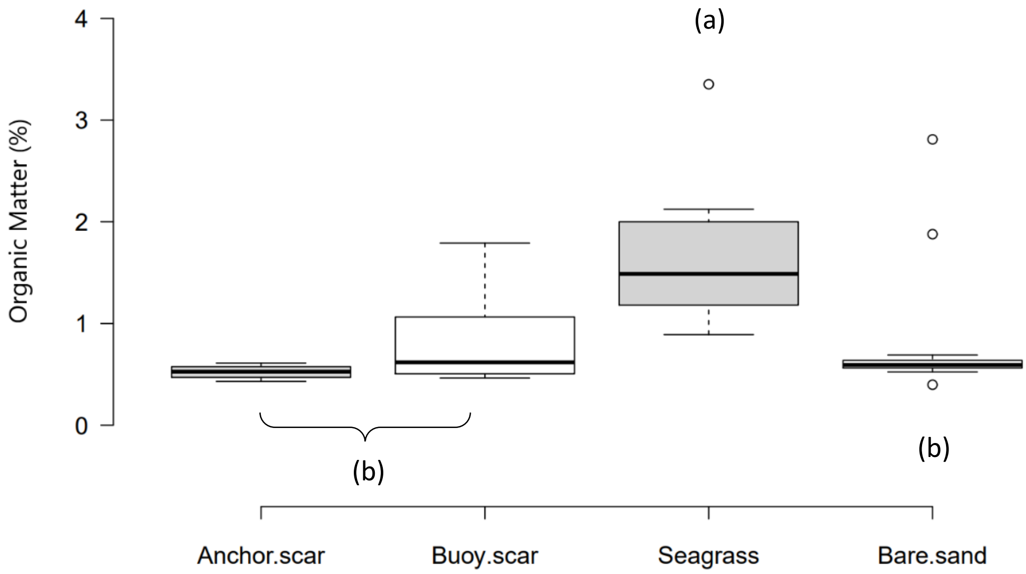


Figure 3. Sediment organic matter content (%) from four environmental condition in Studland Bay. Letters indicate statistical grouping. Box plots represent all data collected across depth n=12.

Organic carbon content ranged from 0.57% ± 0.02% to 0.97% ± 0.24% with an average of 0.72% ± 0.23% (Table 1). Undisturbed seagrass sediments had the highest %OC content (0.97% ± 0.24%), significantly higher than all other sites (Fig. 4). Sediment from the anchor scars had the lowest %OC content (0.57% ± 0.02%). This was significantly lower than seagrass and bare sediment (p<0.05) but not significantly different to buoy scars. Sediment from buoy scars and bare sand also had the same %OC content (0.68% ± 0.16% and 0.68% ± 0.45% respectively). No changes in %OC were noted along depth profiles for any of the environmental condition (Fig. 5).

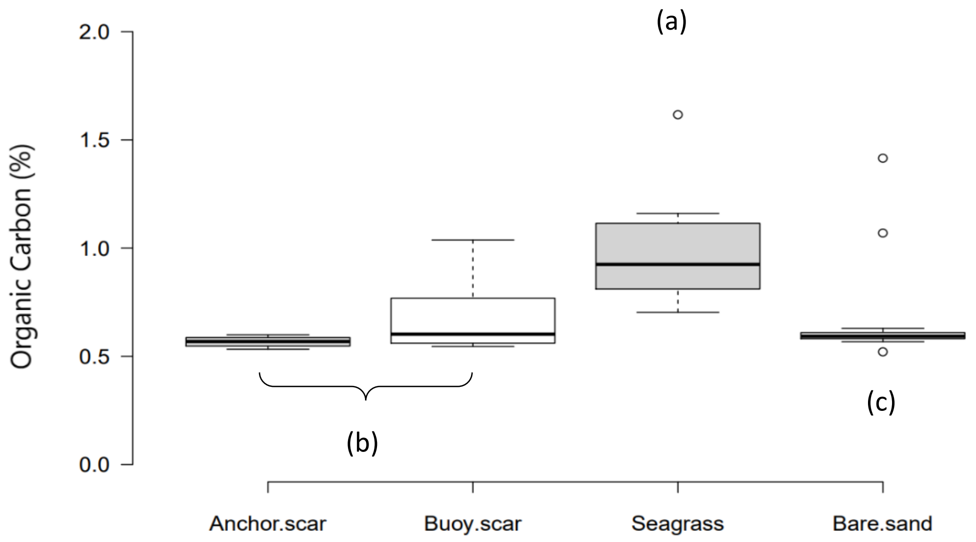
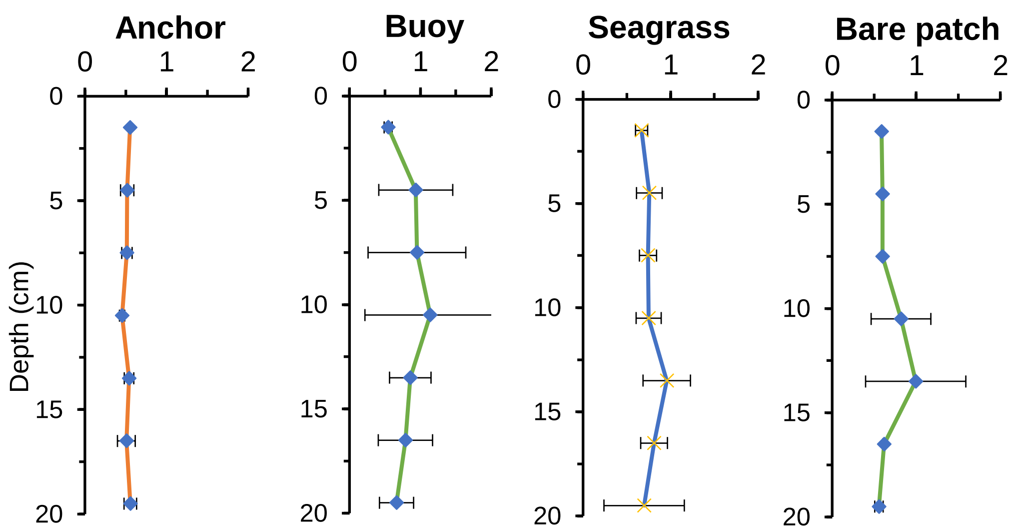


Figure 4. Sediment organic carbon content (%) from four environmental condition in Studland Bay. Letters indicate statistical grouping. Box plots represent all data



collected across depth n=12.

Figure 5. Mean ± standard deviation organic carbon depth profiles from four environmental conditions in Studland Bay.

Discussion

This is the first study to document the impacts of anchor and mooring damage on the carbon storage capacity of a seagrass meadow in the British Isles, adding to the growing body of literature that documents the impacts of anchoring and mooring scars on seagrass sediment carbon (Macreadie *et al.*, 2014b; Serrano *et al.*, 2016). Importantly, both %OM and %OC content was significantly higher in the sediments of undisturbed seagrass sediments than in sediments disturbed by anchoring or mooring and bare sediments, which likely never contained seagrass plants. This supports the hypothesis that sediments from within the seagrass meadow at Studland Bay contain more organic carbon than in-meadow bare patches created by mooring and anchoring, and adjacent bare patches that never knowingly contained seagrass. That anchor scars contained significantly less carbon than bare sand means that disturbance in this way can create sediment conditions less carbon rich than those where seagrass has never knowingly been present.

These results support findings that sediment destabilisation can reduce carbon content, up to 75% in some meadows (Serrano *et al.*, 2016; Glasby and West, 2018). However, the impacts are likely species and/or habitat dependant. A simulation for a *Zostera nigracaulis* meadow in Australia found that removal of above and below ground biomass down to 10 cm (e.g., simulating high intensity disturbance from boat or anchor damage) had no effect on carbon content of the sediment at these depths (Macreadie *et al.*, 2014b). Whether this was due to methodological issues that decreased simulation authenticity, such as size of disturbance, or un-natural re-encroachment of sediment, is unclear. Clearly more research is necessary to fully understand the impacts of small-scale anthropogenic disturbances on Blue Carbon storage among various seagrass species and habitat formations.

Interestingly no significant difference between %OM or %OC from either mooring or anchor scars was found, suggesting that both activities had the same negative impact on carbon storage for Studland Bay. The current mooring system in Studland Bay is block-and-chain mooring, commonly used throughout Europe (Defra, 2018; Glasby and West, 2018). There is extensive evidence to suggest that this type of mooring, which typically have heavy metal chains that drag along the seafloor, causes extensive damage to seagrass meadows (Walker *et al.*, 1989; Hastings *et al.,* 1995; Demers *et al.,* 2013; Glasby and West, 2018). They have been shown to cause greater damage than other mooring types (Hastings *et al.,* 1995), in some cases causing patches up to 300m^2 (Walker *et al.*, 1989; Demers *et al.,* 2013). It seems likely, therefore, that in their current state the use of moorings at Studland Bay, although restricting seagrass damage to specific locations, has a similar environmental impact as anchoring in-meadow. Although other, less impactful, mooring types are available (i.e. eco-moorings), these are considerably more expensive (RYH, 2019).

A substantial amount of carbon is currently stored in Studland's seagrass meadow and the continued degradation by direct scouring by moorings or anchors should certainly be avoided. In an analysis of carbon stocks in 13 seagrass meadows along the southwest coast of the UK, Studland Bay contained the 5th largest stock of carbon, totalling 5,404 Mg, or the monetary equivalent of £129,695 of carbon in the top 100cm of its sediments (Green *et al.,* 2018; Chapter 5). Fragmentation of this site releases the carbon that is stored within these sediments and reduces the ability for more carbon to be sequestered. There is also evidence that this habitat fragmentation can dramatically reduce ecosystem integrity in seagrass meadows (Wilcox and Murphy, 1985; Walker *et al.*, 1989; Hovel and Lipcius, 2001). Given the slow recovery of these meadows, and the general poor condition of British Isles seagrass (Jones and Unsworth, 2016), loss of this important store should spur more effective management activities.

It is important to recognise that there are multiple stressors which can affect seagrass meadows and this study focused on impacts of recreational boating activity only. Decreased meadow health and extent are not restricted to the fragmentation caused by the anchoring and mooring activities evaluated in this study. For example, high boat presence can increase eutrophication due to pollutants and increased wave creation; stunting seagrass growth, and reducing faunal assemblages, which reduces a key service provided by seagrass (Koch, 2002; Silberberger *et al.*, 2016). Evidence suggests that protection from physical disturbance can only be fully effective if organic and nutrient inputs to the site are also removed, which can be achieved by reducing the number of visitors (Marba *et al.*, 2002). Studland Bay has highly elevated N:P rations which suggests a high nutrient imbalance and limitation of P, as well as high C:P rations, indicating poor growing environments (Jones and Unsworth, 2016). Considering how to mitigate the impacts of boating should therefore look beyond the physical disturbance caused and consider the wider system.

Marine Conservation Zone designation

Studland Bay was designated an MCZ in May 2019, despite recognition that its designation could reduce the amount of anchoring and mooring allowed at the site, which, in part, supports £93,000 of recreational spending in Studland Bay per year (DEFRA, 2018). The designation requires the intertidal coarse sediment, long-snouted seahorse and subtidal sands to be maintained in a favourable condition and for seagrass beds to be recovered to favourable condition (DEFRA, 2019a). Despite this site-specific legislation for Studland Bay, the regulations do not go into detail about how the seagrass meadow is to be returned to favourable condition, simply stating that "some activities may need additional management" (DEFRA, 2019a; p. 2). The actual management regulations might include voluntary measures, use of existing licensing frameworks or specific by-laws and orders, which would have to be designed via public consultation (DEFRA, 2019a). Recommendations in 2018 included three options: 1) replacing block-and-chain moorings with eco-moorings, but allowing anchoring to continue; 2) introducing no-anchoring zones and installing a total of 100 eco-moorings within the meadow and; 3) removal of all moorings and complete restriction of anchoring within the meadow (Defra, 2018). The role of an MCZ is to "protect a range of nationally important, rare or threatened habitats and species" (Defra, 2018) and based on this, and the requirement to return the meadow to favourable condition, the final option would seem the most suitable. However, whatever measures are put in place must go through public consultation. That the yachting community have, until now, successfully countered 15 years' worth of lobbying to protect this site under UK law, means that any significant restrictions to anchoring and mooring in the site are likely to be met with fierce opposition.

The carbon storage potential of the meadow at Studland was not obviously considered during consultation, though the author did submit data to the public consultation. Conservationists have been arguing for anchoring restrictions for nearly a decade, fixating on the flagship seahorse species as the primary reason for site conservation (Garrick-Maidment *et al.*, 2010). Arguably, part of the reason for such adamant resistance to any marine protection or introduction of ecologically friendly moorings has partly been because of this focus. The calls to protect these species have largely fallen on deaf ears. The attempts have created a turbulent relationship between the conservation and yachting community so that any efforts to approach mutual resolution have been met with animosity. The flagship approach is one that often fails to entice the diversity of stakeholders needed to ensure resilient conservation (Simberloff, 1998), and an ecosystem approach, whereby the multitude of services afforded by this meadow are documented and communicated, should be considered. This evidence adds another reason to protect this site and should be used to support direct management recommendations.

Future research and management recommendations

From a purely ecological perspective, a complete ban on mooring and anchoring within the seagrass meadow at Studland Bay (i.e. allowing meadows to re-wild (Lorimer *et al.*, 2015)) would create favourable conditions to ensure recovery. However, to assume that marine conservation is purely about ecological favourability is naïve, especially at a site as highly contested as Studland Bay. Lack of communication and damage assessment on the extent of anchor damage within a seagrass meadow in Richardson Bay, San Francisco, has caused years of conflict between government agencies and the boating community (Kelly *et al.,* 2019). Rather than deploy a management plan based on assumptions it is hoped this work will be a basis from which to conduct other studies to understand the best possible management plan, that considers the social, ecological and geophysical factors of the site (Kininmonth *et al.*, 2014). This site already poses management challenges and by collecting evidence in support of specific actions the robustness of recommendations can be heightened.

Evidence from the Mediterranean has shown that removing moorings can result in increased anchoring and subsequent scarring of meadows, despite anchoring restrictions in place (La Manna *et al.*, 2015). Unless a full-time surveillance vessel can be deployed, which is unlikely, completely removing moorings will more likely increase meadow fragmentation. The overwhelming evidence is that anchoring within seagrass meadows, regardless of the type of anchor or material of tie-off, detrimentally impacts shoot density, rhizome barring and sediment stabilisation across a multitude of different species (Milazzo *et al.*, 2004; Montefalcone *et al.*, 2008; Okudan *et al.*, 2011; Kininmonth *et al.*, 2014; Abadie *et al.*, 2016; Serrano *et al.*, 2016), in turn increasing blow-outs and reducing carbon storage capacity. The benefit of moorings is that they are restricted to a single designated area so if used in isolation can keep damage to a minimum. Unfortunately, I was unable to test the impacts of different mooring styles on carbon storage since Studland Bay only contains traditional block-and-chain moorings (Fig. 6 A). This is also true for other disturbance studies on British seagrass meadows (Unsworth *et al.*, 2017). However, in Jarvis Bay, Australia, researchers found that helical anchors (Fig. 6 B) caused minimum impact to seagrass meadows, with seagrass densities remaining similar to within-meadow reference areas (Demers *et al.*, 2013). Conversely, other mooring types, including block-and-chain, caused up to 254m^2 scoured patches per mooring (Demers *et al.*, 2013)

The Royal Yachting Association (RYA) recommends two types of environmentally friendly moorings: a helical/helix anchor with an elastic rope (Fig. 6 B), and a block anchor with extra floats to keep the mooring chain off the seafloor (Fig. 6 C). Helix moorings can withstand over 20,000 lb of breakout force, and cause little to no disturbance, being drilled directly into the seafloor (HMS, 2019; Demers *et al.*, 2013). To achieve the equivalent holding capacity with a cement block you would need a block twice that weight (HMS, 2019), which, apart from being completely unfeasible, would also cause high levels of disturbance. The RYA suggest the use of an elastic rope with the helix system, though there is evidence to suggest that rope has the same deleterious impact as swinging chains within seagrass meadows (Milazzo *et al.*, 2004).

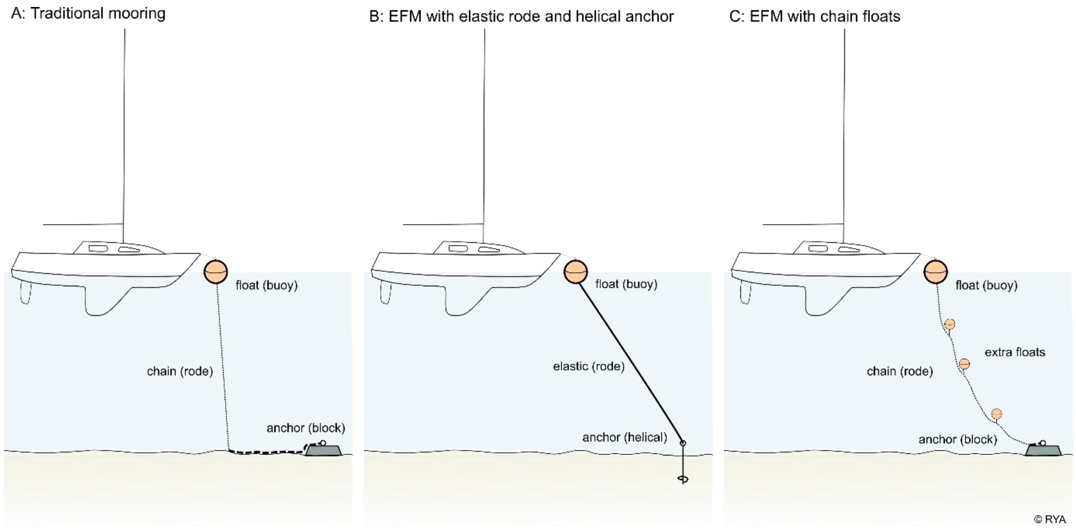


Figure 6. Traditional mooring vs. types of environmentally friendly mooring as suggested by the Royal Yachting Association. Photo credit: RYA (RYA, 2019).

The Community Seagrass Initiative (CSI) and National Marine Aquarium have trailed a hybrid between the two moorings suggested by the RYA, using a helix screw with a long chain that is held off the ground by a series of floats, in Torbay (Fig. 7). Based on the success of this mooring trial, I would recommend a small number of such moorings be trialled in Studland Bay. The trial, before full scale deployment, is an important step in this process. Shared management protocols, although convenient, can be ineffective since success of one system over another is highly dependent on the environmental conditions of the meadow, most notably in this case wave energy exposure (Hastings *et al.*, 1995). Based on a successful trial I would recommend replacing the current moorings with these and deploying more eco-moorings to allow for a total anchoring ban at the site.

Figure 7. Float system attached to helical anchor on an ecologically friendly mooring trial in Torbay Harbour. Photo credit: Mark Parry CSI.

To allow for a complete anchor ban within Studland's seagrass meadow, DEFRA has suggested 100 in-meadow moorings be deployed (DEFRA, 2018). This would equate to two moorings every hectare, which is quite dense. High mooring density can have catastrophic impacts on seagrass meadows (Hastings *et al.*, 1995) so the decision on how many moorings should be deployed should be taken with care. Arguably the optimum number should be determined by assessing the physical disturbance of each mooring, but also by understanding the other impacts boating activity has on the seagrass meadow at Studland Bay. For example, a greater understanding of the boating-related, or other, activities that might account for the high N:P and C:P ratios found at the site (Jones and Unsworth, 2016) might inform optimum visitor numbers to allow the meadow to return to favourable condition.

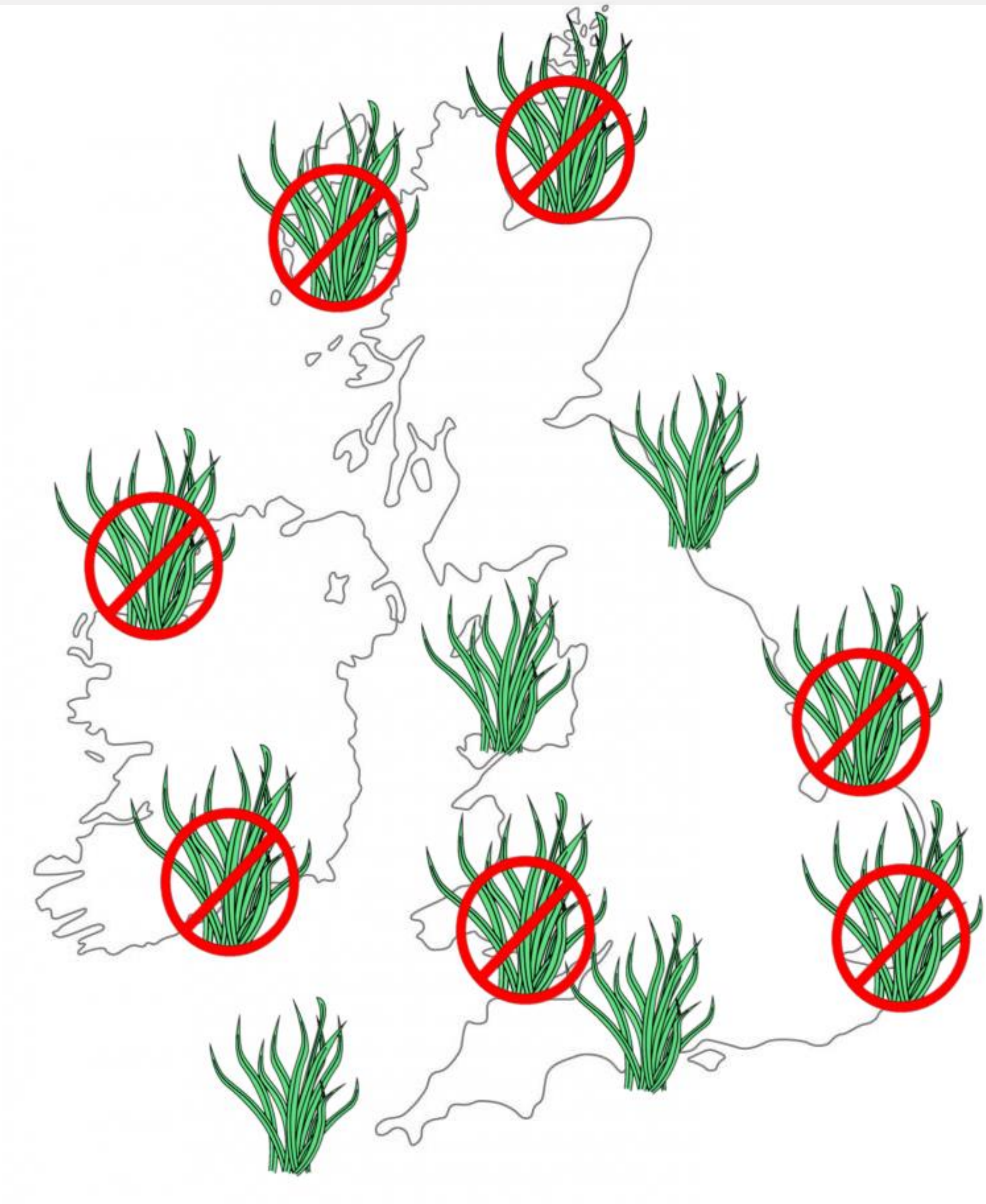
Conclusions

This study found that anchoring and mooring activities significantly reduce the amount of carbon stored in the seagrass sediment in Studland Bay. In fact, the sediments located in the vicinity of these activities contain significantly less carbon than sediments which never knowingly contained seagrass. This research is pertinent since Studland Bay has recently been designated an MCZ, with the intention to return the seagrass meadow to favourable condition. The strategy to achieve this has not yet been set and this research provides direct evidence of activities which are damaging to the seagrass meadow and one of its key services.

More studies on Studland Bay should be conducted to determine the optimum method of restoring this site, which accounts for both ecology and human use. Any management action will have to go through a consultation period, and this should include some attempt at knowledge-sharing. Initially a full assessment of the scale of the damage caused by moorings and anchors (i.e. area of blow-out) should be conducted and communicated clearly with stakeholders, so the reasons for desired management changes are understood. The implications of these losses should also be clearly demonstrated, including, but by no means limited to, a need to protect those environments that can absorb and store carbon in the face of climate change. Just as diversified incentives increase the resilience of marine protected areas (Jones, 2014), so too may they increase the likelihood of understanding, and therefore, compliance of the measures taken.

Chapter 7

Historic analysis exposes catastrophic seagrass loss in the British Isles, with implications for climate change



Introduction

Seagrass meadows are among the most productive coastal ecosystems, supporting diverse and abundant ocean life (Hemminga and Duarte, 2000b; Orth *et al.*, 2006). As foundation species (Thomson *et al.*, 2015) they provide a range of ecosystem services (Costanza *et al.*, 1997; Nordlund *et al.*, 2016) including creating habitat for diverse fauna, protecting the coastline from erosion by reducing wave energy and stabilising sediments, and sequestering carbon (Gambi *et al.*, 1990; Gacia *et al.*, 1999; Duffy, 2006; Unsworth *et al.*, 2010). Despite occupying 0.2% of the Earth's ocean floor, seagrasses represent one of the largest global carbon sinks storing an estimated 19.9Pt of carbon globally (Orth *et al.*, 2006; Fourqurean *et al.*, 2012; Duarte *et al*, 2013a; Duarte, 2013b). This is approximately equal to the CO_2 emissions from fossil fuel and cement production in 2014 (Kerr, 2017).

With our natural world becoming increasingly defined through the lens of natural capital (Guerry *et al.*, 2015), knowledge of the location, extent, and condition of seagrass has become increasingly important, particularly in light of the growing interest in Blue Carbon and its important role in mitigating climate change (Fourqurean *et al.*, 2012; Green *et al.*, 2018). Under the Paris agreement, countries pledged to outline National Determined Contributions (NDC's) to reduce their emissions (Martin *et al.*, 2016), and nature-based solutions are increasingly being adopted within these strategies. To date 28 countries refer to coastal wetlands in their mitigation strategies and 59 countries include coastal ecosystems, seven of these naming seagrass directly (Martin *et al.*, 2016). Regardless, to the best of our knowledge, there have been very few attempts to document national seagrass loss (e.g. Kenya; Harcourt *et al.*, 2019), and none to do so in consideration of shifting baselines.

One of the main global challenges of seagrass conservation is that the status of many seagrass meadows is unknown (Unsworth *et al.*, 2018). Knowing how much seagrass a country has is clearly an important step to knowing how to protect it. But knowing where seagrass was gives countries an opportunity to re-plant and restore seagrass in favourable areas. Seagrass restoration has historically been notoriously challenging (Rudge, 1970), however, *Zostera* spp. has seen some recent successes in the USA (NOAA, 2019) and New Zealand (Matheson, 2015). It is increasingly accepted that restoration of natural habitats must play a crucial role in global efforts to mitigate climate change (European Commission, 2009). That seagrasses can absorb more carbon up to 40 times faster than terrestrial forests (McLeod *et al.*, 2011) should make them a significant component of these attempts. Global loss of seagrass since the 1980's is thought to be at least 29% (Short *et al.,* 2010; Waycott *et al.*, 2009), and seagrass continues to be lost up to a rate of 1.4% a year (Short *et al.,* 2010). These losses must be stemmed if seagrass is to play a role in climate mitigation and understanding where loss has occurred is an important first step towards appropriate conservation planning.

Seagrasses are highly sensitive to degraded water quality and conditions which impose light limitations to photosynthesis (Orth *et al.*, 2006). Coastal development and nutrient enrichment have historically been responsible for worldwide declines, which threaten the substantial ecological services seagrass meadows provide (Fraser and Kendrick, 2017). Global seagrass declines only account for mapped populations and in many countries data on extent is limited. Even in developed countries, such as those of the British Isles, spatial data on seagrass extent is lacking. Given the paucity of seagrass mapping to date, the baseline from which global seagrass declines are calculated are almost certainly significant underestimations.

The most up-to-date estimate of seagrass coverage indicate that a minimum of 325,178 km^2, occurs globally, but these values do not include any data from the British Isles (Short, 2018; Unsworth *et al.,* 2018). Recent efforts have been made to demonstrate the substantial services afforded by British seagrass habitats through sediment stabilisation (Wilkie, 2011), supporting fisheries (Bertelli and Unsworth, 2014) and carbon sequestration (Green *et al.,* 2018). Understanding the significance of these services is challenging without robust estimates of the current and historic areal extent of British Isles seagrass meadows.

As with global seagrass observations, monitoring and mapping of British seagrass occurs with limited consistency. Where studies have occurred, the resulting data is largely in the grey literature or held disparately by local councils, national and devolved governments, and non-government organisations (NGO's). This has resulted in a lack of current and robust estimates on spatial coverage of seagrass. The need for these estimates are multiple. There is a scientific and biological prerequisite for actively recording and monitoring British environmental habitats, for which seagrass has seemingly fallen under the radar. The evidence for the services provided by seagrass, especially considering global efforts to provide monetary values for these, should make the accurate mapping of them a priority. There is a current effort to increase the knowledge of global seagrass cover with distribution maps held at the United Nations Environment Program - World Conservation Monitoring Centre (UNEP-WCMC and Short, 2018). The accuracy of these will be greatly improved if regional efforts look to support this work. Finally, recent studies highlighting the poor status of seagrass in the British Isles (Jones and Unsworth, 2016) have stressed this need.

Once considered a significant component of the natural heritage of British waters (Davidson and Hughes, 1998), seagrass is now accepted to be nationally scarce and sparsely distributed (EA, 2003; Hiscock, Sewell and Oakley, 2005). Conceptions of environmental degradation tend to shift depending on our temporal reference point. In the British Isles this *shifting baseline syndrome* (SBS) (Pauly, 1995) occurs when the earliest known data of areal extent is assumed as an unaffected baseline condition. This is further exacerbated by data being supported by qualitative accounts that refer to healthier conditions within a scientist's lifetime (e.g Butcher, 1932) (Pauly, 1995). With each generation the concept of a healthy ecosystem shifts, depending on their perceived baseline.

The earliest attempts to document seagrass extent already pointed to declines and the need for more data (Butcher, 1932, 1933). It is likely that Butcher's reports were already subject to SBS. Two periods of decline are emphasised throughout the literature: one immediately after WWI, and another during the northern Atlantic outbreak of wasting disease in the early 1930's (Butcher, 1934; Cottam, 1935). The wasting disease 'epidemic' has been perpetually attributed as the main cause of declines (Den Hartog, 1993; Garrard and Beaumont, 2014), without consideration for the pervasive environmental degradation that occurred in the centuries before. Regardless of the cause of these declines, more efforts are needed to evaluate the status and trends of these valuable marine habitats. To fully appreciate the extent of declines we must find a way to look beyond these earliest evaluations, which are almost certainly underplayed due to SBS.

The objectives of this chapter were accordingly to estimate 1) the current areal extent of seagrass in the British Isles; and 2) the percentage loss of seagrass throughout the British Isles in both recent histories, since the 1930s, and over longer time periods

(100s years). The paper places the results in the context of conservation and provision of ecosystem services, especially considering the impact of habitat loss on carbon storage. The data was used to test the following hypotheses:

Hypothesis 1: The current estimates of seagrass areal extent in the British Isles are out of date and inaccurate.

Hypothesis 2**:** There has been a substantial reduction in the spatial extent of seagrass in the British Isles with significant consequences to the Blue Carbon capacity of this resource.

Methods

For the purpose of this work I have categorised any data collected since 1998 as ‘contemporary’ (similar to current conditions), and any data older as ‘historical’ (not reflecting current conditions), since I cannot reliably confirm the presence of something that has not been mapped for over 20 years. To fulfil the first objective, multiple datasets were collated with other isolated data to determine the current mapped areal extent of seagrass in the British Isles. Due to the paucity of available data I have used three methods to assess seagrass loss with high, medium and low certainty. High certainty loss estimates were generated collating data older than 1998 and cross-checking them against available contemporary data to confirm loss of areal extent. Medium certainty loss includes sites where no contemporary data is available, i.e. sites that have not been revisited since 1998. In these cases, lack of data is counted as loss of seagrass. All these methods were supplemented by a systematic review to provide qualitative and quantitative data on seagrass loss. Low certainty loss estimates, not subject to SBS and data limitations, were modelled using best available data on historic seagrass extent and additional data regarding mudflat area of England, Scotland and Wales (mainland Britain) to model maximum seagrass extent and percentage loss in mainland Britain. These estimates excluded Ireland and Northern Ireland, because accurate data on mud- and sandflat area was not available.

‘Contemporary’ and ‘historical’ areal estimates

Two datasets were identified as containing records of *Zostera* spp. from multiple sites. The OSPAR Threatened and Declining Habitats (2017) dataset represents the current known areal extent of seagrass in the British Isles and includes records on *Zostera* spp. from between 1986 and 2015. Under the Water Framework Directive (WFD) the

Environment Agency (EA) is required to assess the condition of seagrass to help determine the biological condition of UK water bodies (Foden and Brazier, 2007). The outcome of this is another dataset that includes areal extent of *Zostera* meadows monitored under the WFD between 2007 and 2017. These data were analysed by region and date on QGIS (version 3.2.1) and were shown to contain substantial gaps. To supplement these, I contacted stakeholders from a multitude of organisations targeting local councils, national and devolved governments, government advisory organisations (such as Natural England and the regional Inshore Fisheries and Conservation Authorities), private environmental consultants, and scientists who work on seagrass in the British Isles. From these searches 14 additional contributors supplemented the OSPAR and WFD datasets, the collective of which makes up all the known available data, based on the searches I undertook (see Appendix 2). Since species identification was not provided across all data sets, they have not been included here. However, as an intertidal species there are far less technological constraints associated with surveying *Z. noltii*. Because of this it is expected to be in the majority, and it is further expected that some *Z. marina* meadows have gone unreported. This is especially true for the WFD dataset, which only includes *Z. noltii* in its assessments, due to ease of access. Because of these inconsistencies I have decided not to discriminate between species, since greater abundance of one species is likely due to mapping inconsistencies rather than variances in the conditions that allow one species to proliferate over another. It should also be noted that *Zostera angustifolia* was once considered its own species, although is now recognised as a phenotype of *Z. marina*. Because I do not distinguish between species, *Z. angustifolia* is treated in the same way as *Z. marina* and *Z. noltii*.

Data was provided in the form of individual observations (point data) and area estimates (polygon data). Polygon data was used to provide the area estimates contained herein. Spatial assessments were made using QGIS (version 3.2.1) and all data were analysed for duplicates or overlaps. Where they occurred, the most recent data was used, unless differences between years occurred that represented data collection restraints rather than area changes. For example, where data was present in the same location for years 2016, 2017 and 2018 but the 2018 data held substantially reduced area it was assumed that this was not representative of habitat degradation but of restrictions accessing the full extent of the meadow in that year. This was supported by several data fluctuating between three years – e.g. 2014 and 2016 showed the same area cover, but 2015 showed far reduced cover. The area of each polygon was calculated in m^2 using a cylindrical WGS-84 projection and converted to hectares (ha) for reporting purposes.

The contemporary data represents the minimum area of seagrass in the British Isles, as some meadows have certainly gone unreported. OSPAR data was used to provide high and medium certainty estimates of historic mapped areal extent. The maximum seagrass extent for each record within the dataset was checked against contemporary records and where contemporary records were found the difference between largest (oldest) and current meadow size was used to provide high certainty loss estimates. Where no contemporary record of the meadow was found, these were considered as spatial loss and included in medium certainty loss estimates. I acknowledge the approach, and the subsequent estimates of changes in coverage through time, is constrained by sampling efforts and data reporting of past research efforts, but this represents the best use of the best available data.

Systematic review of qualitative and quantitative data

Systematic reviews are used to encapsulate a broad range of literature on a discrete subject by aggregating large data and rigorously extracting relevant information (Minx *et al.*, 2017). I followed the distinct protocols required to achieve a systematic review (Minx *et al.*, 2017) by: 1) defining the discrete subject parameters and the timeframe of interest; 2) creating a search term to encapsulate all data that might be relevant to the subject; 3) inputting this into Web of Science (Thompson Reuters) to extract a literature database; 4) justifying and making a transparent selection of the literature and; 5) providing a synthesis of the relevant literature.

The purpose of this search was to extract qualitative and quantitative data on historic seagrass presence and trends of declines as far back as possible. To achieve this, I designed a search term that would capture all the literature on seagrass in the British Isles, including NOT terms to avoid irrelevant hits from places sharing part of their name with British Isles locations:

(Seagrass OR zostera OR "zostera marina" OR "z. marina" OR "z. noltii" OR "zostera noltii" OR "zostera angustifolia" OR "z. angustifolia" OR eelgrass OR "wigeon grass") AND (uk OR "united kingdom" OR "great britain" OR england OR wales OR scotland OR "northern Ireland" OR ireland OR "scilly isles" OR "isles of scilly" "isle of wight" OR guernsey OR jersey OR "isle of man") NOT ("New England" OR "New Jersey" OR "New South Wales")

The search term was entered into Web of Science, producing 160 papers. These were sifted by reading their abstract based on the following criteria: 1) the paper refers to an area within the British Isles and; a) seagrass is the primary focus or; b) seagrass is the secondary focus but the primary focus suggests significant contribution of seagrass to the paper e.g. feeding habitats for wintering wildfowl; coastal WFD assessments; ecosystem services of coastal habitats. If there was any doubt that British Isles seagrass formed a significant contribution to any of the papers, it was also included for review. Based on these criteria, 59 papers were considered relevant for this work. The database of papers was synthesised for qualitative and quantitative data on seagrass presence, loss, and trends of changing distribution. Because these data were being used to supplement quantitative data on areal extent only, data on density and condition of seagrass meadows was not captured.

Web of Science includes published, peer-reviewed articles as far back as 1990. Because of the distinct lack of published data on seagrass area cover in the British Isles, and a need to capture data as far back in time as possible, it was necessary to broaden the search to include published, unpublished and grey literature. These data were collected by extensive internet searches, through contacting stakeholders from government, private organisations and NGO's, and from scientists and the public who work in seagrass science, conservation and management throughout the British Isles. Papers were qualified and data extracted based on the same criteria as the systematic search. These searches unearthed a further 120 papers that were considered relevant to this work.

Modelling the percentage loss of seagrass

Because of the scarcity of historic empirical data, and the observation that many of the early qualitative reports are almost certainly subject to SBS, I used available data to model the maximum historic extent and low certainty loss estimates of seagrass in mainland Britain.

In 1932 Butcher reported on the distribution of *Zostera* spp. in the British Isles, including a spatial distribution map of mainland Britain (Butcher, 1932). In 1991 the Nature Conservancy Council (NCC) undertook a report on the 155 estuaries that exist in mainland Britain (Davidson *et al.,* 1991). Butcher's map corresponds very closely to the estuaries map presented in the NCC report (Fig. 1). Further, qualitative data suggest that before WWI seagrass would once have been found across a large proportion of subtidal mudflats and on the lower ranges of most intertidal mudflats throughout the British Isles, especially prevalent in estuaries (Butcher, 1932, 1934; Duncan and Cotton, 1933; Cottam, 1935). Mud- and sandflats, in particular those within estuarine are, therefore, a good proxy for modelling historic seagrass distribution.

I identified locations with best available data on seagrass area cover from either historic or contemporary estimates, where contemporary estimates represent meadows that are in reasonably good condition, and where total mudflat area for each site was available. This restricted the inclusion of sites to those designated as Special Protected Areas (SPA's) and Special Areas of Conservation (SAC's), since such sites have been accurately mapped by the Joint Nature Conservation Committee (JNCC) (JNCC, 2005, 2015, 2018) (Table 1).

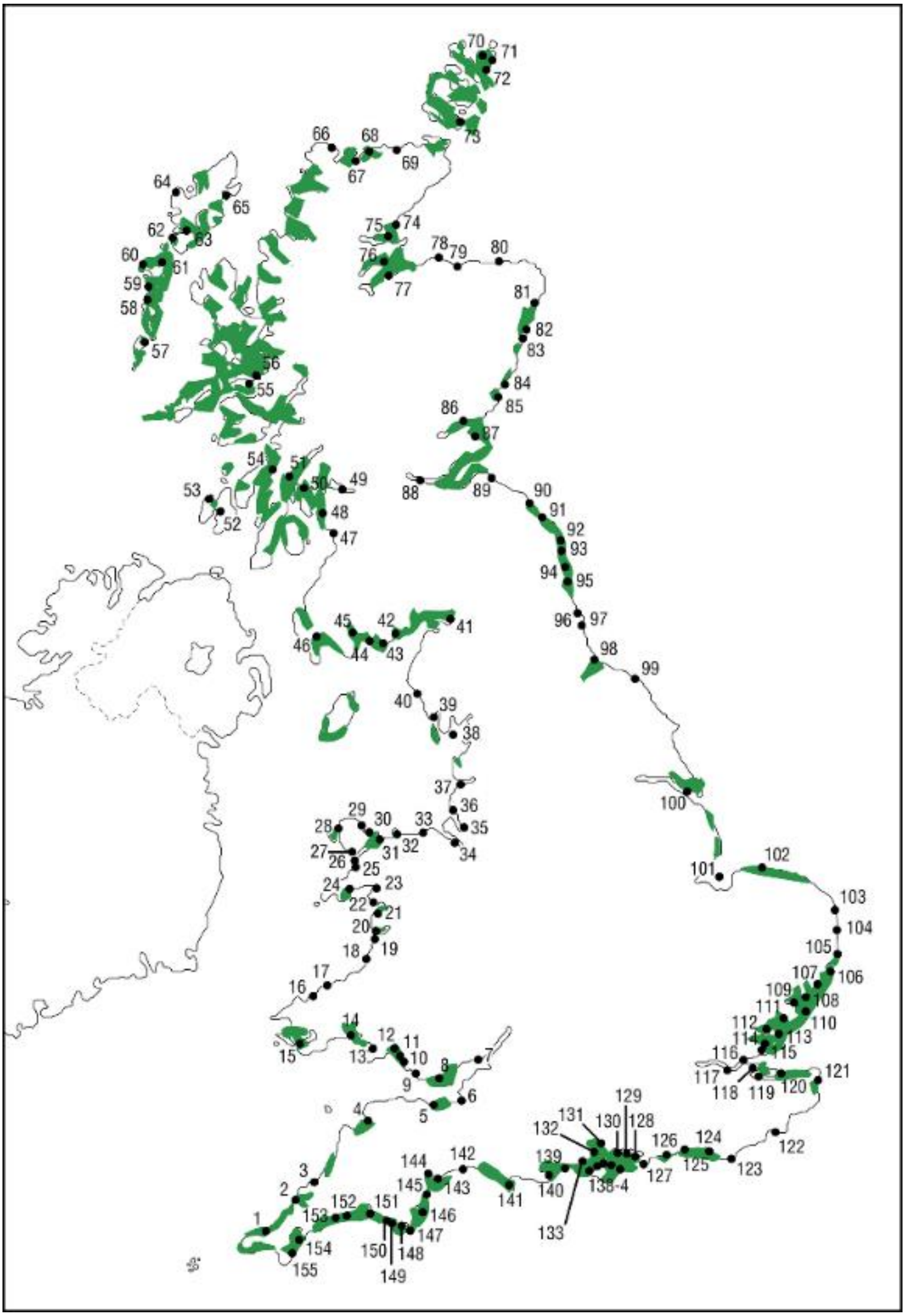


Figure 1. Butcher's (1932) estimate of seagrass area cover (shaded green) and numbers referring to estuary locations identified by Nature Conservancy Council (Davidson, 1991) (image created by UCL drawing office).

The chosen sites represent typical environmental variation for British seagrass, including intertidal and subtidal habitats within estuaries, rivers, lochs, spits, and along the coastline. Seagrass proportional coverage per hectare of mudflat was calculated for each site by dividing total seagrass area by total mudflat area (Table 1; Fig. 2i). Seagrass total coverage was estimated using bootstrapping techniques (Manly, 2007). The 10 sites were randomly selected with replacement 1000 times, as per the methodology, and multiplied by total mudflat area estimates from a) the OSPAR dataset (2017), which includes data in and outside of estuaries, and b) the NCC report (Davidson *et al.*, 1991), which includes data from estuaries only (Fig. 2a & b). The average of these 1000 estimates were used to estimate maximum extent of seagrass around mainland Britain (Fig. 2ai & bi). I used this simple bootstrapping procedure, rather than more typical parametric statistical methods, due to the paucity of available data. The data provides low certainty maximum seagrass coverage, i.e. extent, around mainland Britain.

Table 1. Seagrass meadow area data used in the model to calculate historic seagrass loss in the UK. Current and historic extent (where available) and mudflat area were used to determine average seagrass area per hectare (ha) of mud- and sandflat (m/s-flat).

Site name	Site Location	Current extent ha	Historic extent ha	M/s-flat area ha	Seagrass area/ha m/s-flat	Reference for M/s-flat area
Spurn Bight	Humber Estuary	0.59	550	4,842	0.11	JNCC, 2018
Lindisfarne	NE England	679	1,046	1,571	0.67	JNCC, 2015
Foulness/Maplin Sands	Thames Estuary	40	320	8,746	0.04	JNCC, 2005
Fal & Helford	Cornwall	104	208	645	0.32	JNCC, 2018
River Stour and Orwell	Thames Estuary	1.43	380	2,620	0.15	JNCC, 2018
Exe Estuary	Cornwall	142	**N.D.**	900	0.16	JNCC, 2018
Dornoch Firth	East Scotland	117	2,546	6,787	0.38	JNCC, 2018
Cromarty Firth	East Scotland	1200	3,241	3,766	0.86	JNCC, 2018
Moray Firth	East Scotland	**N.D.**	1,098	2,339	0.47	JNCC, 2018
Plymouth Sound	Devon	92	**N.D.**	2,555	0.04	JNCC, 2018

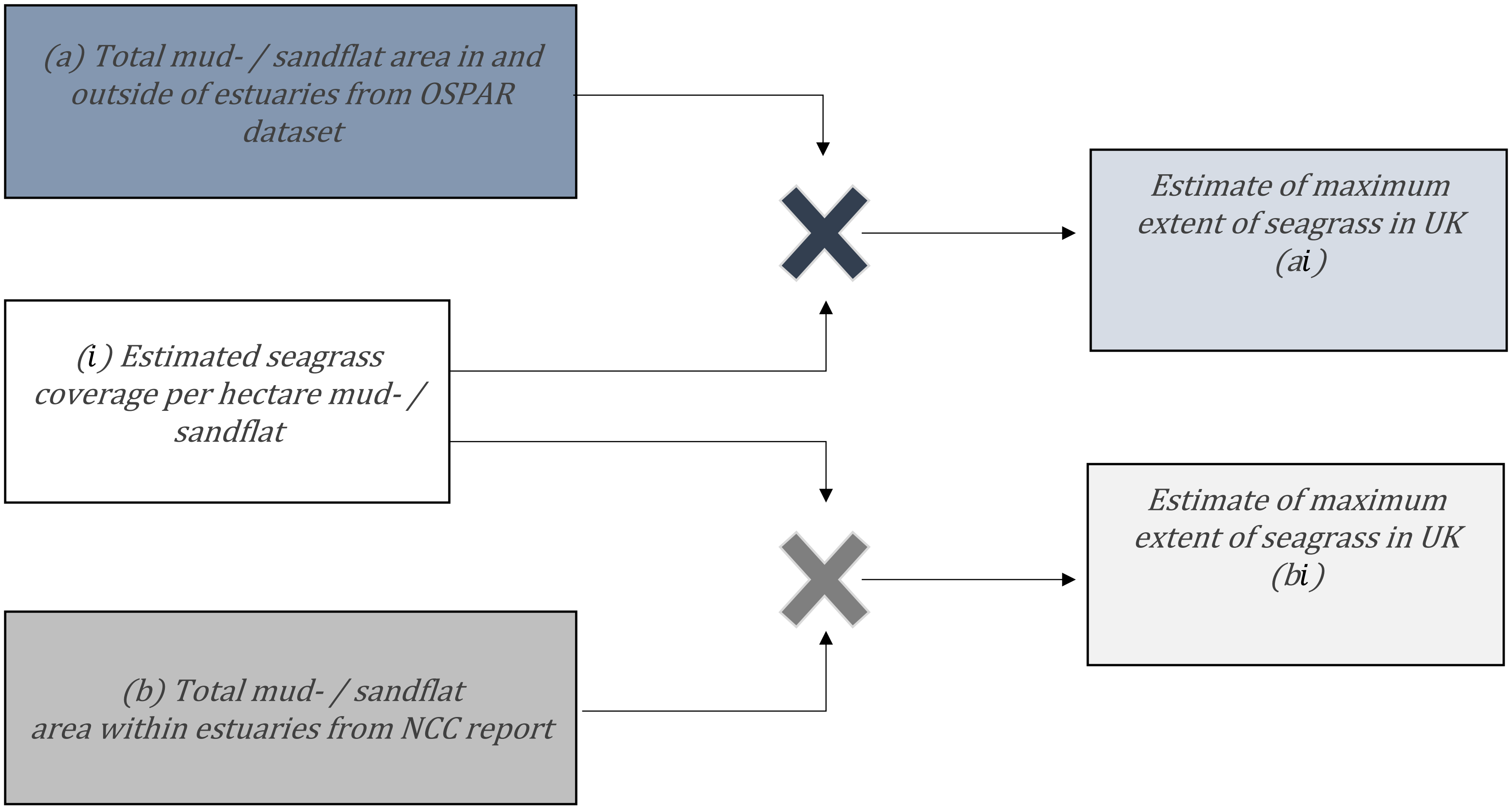


Figure 2. Calculations used to estimate maximum areal extent of seagrass in mainland Britain: $a \times i = ai; b \times i = bi$. i is the average seagrass area from 10 sites with good historic or contemporary estimates divided by known mud- / sandflat area.

Results

Contemporary and pre-1998 areal estimates of seagrass habitats in the British Isles

The full dataset for contemporary and pre-1998 areal estimates of seagrass habitats in the British Isles can be found in Appendix 3 and 4. The total mapped areal extent of contemporary seagrass records (post-1997) from the OSPAR dataset, the WFD dataset, and all other contributors includes 47 surveys spanning 20 years, 79% of which are from the last 10 years (see Appendix 3). In total the data confirms the presence of 8,493ha of seagrass in the British Isles (Table 3). Occurrence of seagrass is not uniform. Over half of all mapped seagrass occurs in the Scottish Highlands (24%), Devon (16%) and Northern Ireland (14%) (Table 2). Seagrass occurrence ranges from patches less than 1m^2 to meadows up to 1,200 ha. The average size of seagrass record is 2.64 ha ± 32.22 ha.

The OSPAR dataset, which represents the currently used known areal extent of seagrass in the British Isles, includes 13,753 ha of seagrass. Of this, 8,835 ha (64.2%) was historical (pre-1998) and 4,91 ha (35.76%) was contemporary (post-1997).

The total mapped historic extent of seagrass in the British Isles is 16,524 ha. The total documented seagrass loss of seagrass since 1936 is 6,697 ha. A further 1,364 ha of seagrass habitat has not been revisited since 1998 (table 3), a disproportionate amount of which is from Scotland (table 3). With high certainty the British Isles has, therefore, lost 41% of its seagrass since 1936, 34% since the 1980s. With medium certainty, including historic data with no recent observations, 50% has been lost since 1936, 42% since the 1980s. In Scotland high loss estimates are much higher. With high certainty

59% of seagrass has been lost, and with medium certainty 74% has been lost since 1936.

Table 2. Distribution of contemporary mapped seagrass area from the OSPAR and Water Framework Directive datasets, and other collected data sources since 1998. Data presents total known areal extent of seagrass in the British Isles by region, including relative contribution to the total mapped area.

Location	Area ha	% of total
Scottish Highlands	2,056	24.21
Devon	1,392	16.39
Northern Ireland	1,226	14.44
Hampshire and Isle of Wight	714	8.41
Northumbria	680	8.01
Ireland	584	6.88
South Wales	460	5.42
Dorset	372	4.38
Scilly Isles	196	2.31
North Wales	172	2.03
Suffolk, Essex, Kent	170	2.00
Cornwall	166	1.95
East Scotland	108	1.27
West Wales	90	1.06
Cumbria	65	0.77
Norfolk	42	0.49
Total	**8,493**	

Table 3. Estimated seagrass loss from high (known) and medium (unmapped) estimates across all regions and Scotland, calculated by analysing data older than 1998 from the OSPAR dataset. Known loss is from sites which have been revisited and data captured since 1998, unmapped is from sites that have not been revisited since 1998.

	High certainty, known seagrass loss	Total unmapped seagrass	Medium certainty seagrass loss
All regions	6,697	1,364	8,061
Scotland	4,790	1,358	6,148

Systematic review of qualitative and quantitative data

Of the 179 papers identified by the systematic review, 86 were relevant to this work. Of these, 50% were directly related to British Isles seagrass meadows and 50% included seagrass as a significant secondary focus (Fig. 3). The southwest coast (including the Isles of Scilly) contributed over a quarter of total publications (25.4%). The northeast contributed a further 21.3%, with a single site, Lindisfarne, contributing a total 8.2% of all publications. The vast majority of these (60%) were considering the wildfowl populations which feed on *Zostera* meadows. Wildfowl publications contributed 17% of total publications and are particularly prevalent in earlier years (1928-1988).

The first published account of seagrass in the British Isles was in 1831 (Winch, 1831), where it was included in a publication on the '*Flora of Northumberland and Durham'*. A peak in publications occurred around the time of the 1930's wasting disease, when naturalists became concerned with the substantial degradation of sites throughout the British Isles (Fig. 2). Publications were sporadic until 1990 (n=20) and since then have occurred more frequently as a series of peaks and troughs (n=66). No obvious trend of continued increasing development of seagrass science in the British Isles is apparent in the literature base over the last 30 years (R^2=0.04).

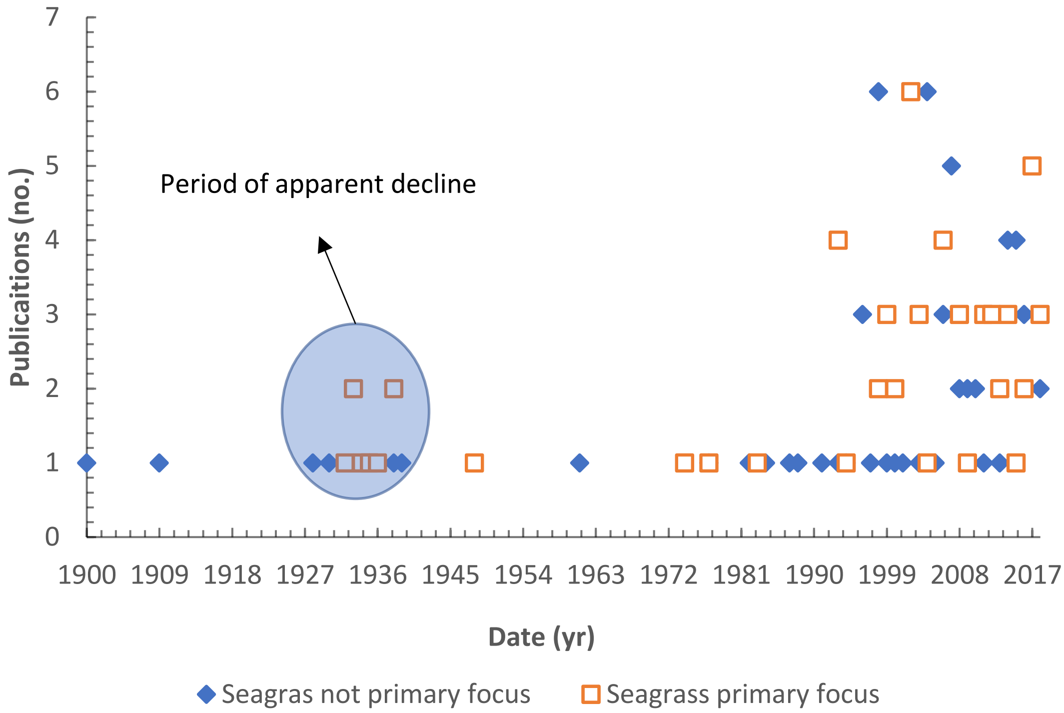


Figure 3. Number of publications (grey and peer reviewed) relating to British Isles seagrass habitats by date where seagrass is primary focus (orange square) and seagrass is significant secondary focus (blue diamond).

Attempts to map seagrass across the British Isles were made by Butcher in 1932 but without any defined methodology (Butcher, 1932). He considered the occurrence of seagrass regionally but did not provide area estimates. Seagrass seemingly occurred ubiquitously, 'apart from wave-swept, shingly and rocky shores to the west of the country' (Butcher, 1932). The abundance of seagrass in sheltered and protected areas on the east coast were noted, as were plentiful populations in the lochs of Ireland and the west of Scotland (Butcher, 1932). Aside from this early attempt, efforts were made to document the status of seagrass in Scotland in 1933 (Cleator, 1993), in Wales in 1998 (Kay, 1998), and in Cornwall and the Isles of Scilly in 2002 (Hocking and Tompsett, 2002a, 2002b). These reports provide presence and absence data and indicate widespread declines, but do not provide spatial estimates.

Synthesis of systematic review

The literature analysed through the systematic review show ubiquitous declines across almost every region of the British Isles. Areas where good historic data are available show declines of between 40% (Cornwall) and 100% (Suffolk) (Table 4).

Table 4. High certainty seagrass loss (by area and percent reduction) in regions where good historic data are available.

	Max extent (pre-1998)	**Contemporary area**	**High certainty loss since 1936**	
	ha	*ha*	*ha*	*%*
British Isles	16,524	8,335	6,826	41
Cornwall	271	166	167	62
Essex	450	170	280	62
Northumbria	1,595	679	916	57
NW England	224	65	159	71
Scilly Isles	325	196	129	40
Scotland	8,312	2,164	4,790	58
Suffolk	380	1	379	100

Cornwall: There are no county-wide quantitative historic data for seagrass areal extent in Cornwall, and I suspect much has gone unreported. Populations within the Fal and Helford estuaries are the best documented and once contained plentiful and abundant seagrass meadows, many of which were located at sites now completely devoid of the plant (Hocking & Tompsett, 2002a, 2002b). In 2004 Cornwall Wildlife Trust (CWT) documented a total of 271 ha between these two estuaries. Around 50% has been lost since then, the most recent estimates suggesting only 104 ha remains (Curtis, 2015). In total 62% of seagrass has been lost in Cornwall since 1936.

Devon: Devon contains some of the most extensive populations of seagrass throughout the British Isles. Historic accounts are rare, and although wasting disease reportedly 'wiped out' all the seagrass meadows in the major estuaries, populations are reported to have recovered relatively quickly (Butcher, 1932). Seagrass once occurred in the Axe estuary, which is now devoid of the plant.

***Dorset*:** The only historic accounts of seagrass meadows in Dorset occur at the Fleet lagoon. Historic reports suggest the meadow was 'under threat' from multiple sources (Ladle, 1984). Locals noted a change in species population, reduced seagrass abundance and an increase in algae cover over the years (Ladle, 1984). Despite these threats, a large meadow covering 220 ha of the 445 ha lagoon remained until 2004. Declines have been noted since then with fluctuations occurring yearly. Seagrass meadows remain at Studland Bay, Poole Harbour and Weymouth (totalling 152ha) though no historic records for these locations exist.

***Hampshire and the Isle of Wight*:** In the 1980s Portsmouth, Langstone and Chichester Harbours were described as rich in *Zostera*; 440 ha supported 75% of the Solent's Brent geese population (Tubbs and Tubbs, 1982). In recent years Portland Harbour has seen an increase in *Zostera* from 30 ha to 79 ha. Langstone and Chichester Harbours have both seen a 50% or greater reduction from 280 ha to 118 ha and 130 ha to 66 ha respectively. By the 1980s seagrass meadows outside of the harbours, which once prevailed throughout the Solent mudflats had completely vanished (Tubbs and Tubbs, 1982). Only two meadows persist here. Declines were directly linked to dock building and channel dredging (Tubbs and Tubbs, 1982).

The only historic reports of *Zostera* around the Isle of Wight suggest it occurred abundantly along the north of the island from Bembridge to Yarmouth. Populations still occur here, although at a reduced extent. Although once present, it had disappeared entirely from Cowes, Medina, Newton, Bembridge Harbour and the outer Yar estuary by the 1930s (Butcher, 1934).

Ireland and Northern Ireland**:** Butcher (1932) noted *Zostera* on every part of the Irish coast and later observed large areas of disappearance (Butcher, 1934). Significant reductions in seagrass extent were further noted between 1970 and 1990 (Portig, 1997, Mathers *et al.*, 1998). In 2017 an assessment of changes in seagrass extent was undertaken at 21 sites, noting a reduction of 234 ha within the last 6 years, about an 11% decline (Wilkes *et al.*, 2017).

Northumbria: The best-preserved record of seagrass found in the Northeast of England is around Lindisfarne, less than 20 miles from the Scottish border. Although declines in geese populations that feed on the meadow have been noted (Percival *et al.*, 1996) Lindisfarne represents one of the larger meadows in the British Isles (679ha), benefiting from isolation from human habitation and development, as well as legislative protection as an SPA. However, even this healthier meadow has seen reduction on its largest recorded extent of 1,046 ha (NE, 1997).

The search uncovered spatial estimates of seagrass cover from the early 1900's from meadows at Sprun Bight on the Humber estuary. The site reportedly contained at least 550 ha of seagrass in the 1930's (Butcher, 1934; Philip, 1936). Today, less than 1ha occurs, representing an almost 100% reduction of its original extent. The estuary has been subject to extensive modification since the 17th century (Batty, 1997). Reclamation, building of sea walls and harbour dredging, and eutrophication were responsible for a 30% reduction of salt marsh habitat (Batty, 1997). These modifications likely impacted the extent of seagrass meadows, though no record of this exists. These are the only two documented seagrass meadows in Northumbria and, collectively, they represent a 57% loss of seagrass extent.

***Northwest England*:** The only known meadows to occur in the Northwest of England are at Barrow-in-Furness in Morecambe Bay (Davidson and Hughes, 1998a; MMO, 2014). In 1996 224 ha of seagrass occurred here. Since then a gas pipeline was laid through the centre of the meadow (Davidson and Hughes, 1998a) and today 65 ha persists, representing a 71% loss.

Norfolk: Although no numeric estimates exist Norfolk was once considered to include ‘important’ seagrass meadows and now contains only remnants of the plant (Vickery *et al.*, 1994; Rowell and Robinson, 2004). Only sporadic and limited *Zostera* patches totalling 41ha remain in North Norfolk Area of Outstanding Natural Beauty (AONB), where it was once plentiful. Meadows also previously existed in Waveney and the inner Wash (Vickery *et al.*, 1994; Rowell and Robinson, 2004).

***Scotland*:** Despite the high relative abundance of seagrass found in Scotland, there is a dearth of historic data and published literature. Recent claims suggests that 20% of the seagrass that occurs in north-west Europe are found in Scottish waters (Foster and Davidson, 2018). However, this claim is unsubstantiated and the total mapped area from this study makes up only 27% of the total for the British Isles. The majority of the OSPAR data for Scotland is older than 1998. Some 7,243ha was mapped in 1986 in the Moray, Cromarty and Dornoch Firths alone (OSPAR, 2017). Today 1,317ha is mapped in the Dornoch and Cromarty Firths and no estimates exist for the Moray Firth. In total, with high certainty, there has been a 58% reduction in areal extent of Scotland’s seagrass meadows since 1936.

Scilly Isles**:** Benefiting from isolation from industrialisation and urbanisation, the seagrass meadows in the Scilly Isles are some of the most extensive and healthiest in the British Isles (Hocking and Tompsett, 2002b; Jackson *et al.,* 2011; Bull *et al.,* 2012; Bertelli and Unsworth, 2014). There are accounts of 100s of hectares of seagrass occurring in these waters in the early 1900's. A recent report documented 196ha (Jackson *et al.*, 2011), a 40% reduction on the 325 ha recorded in early 2000 (Hocking and Tompsett, 2002b).

Suffolk, Essex and Kent**:** In the Greater Thames Estuary large meadows of seagrass occurred at Leigh Marshes, Foulness/Maplin Sands, and the Stour and Orwell estuaries (Wyer *et al.,* 1977). In the 1960's and 1970's 844 ha of seagrass was recorded among these sites (Burton, 1961; Wyer *et al.,* 1977). The meadow at Foulness/Maplin Sands was once considered one of the largest *Z. noltii* beds in Europe, extending over 320 ha (Burton, 1961; Rudge, 1970; Wyer *et al.,* 1977). Today only 40ha occurs here, an 88% loss on original extent. The Stour and Orwell estuaries contained 380 ha of seagrass in the 1960's (Burton, 1961; Wyer *et al.,* 1977; Beardall *et al.,* 1991) and today hold a total of 0.5 ha, an almost 100% reduction. The largest meadow now remaining occurs at Leigh Marshes, where 121h a have been mapped, an increase on the 100 ha mapped in the 70's (Wyer *et al.,* 1977).

Estimates on the seagrass meadow extent at Leigh Marshes occur almost every decade throughout the last century, making it one of the best documented seagrass meadows in the British Isles. The site was decimated by wasting disease in the 1930's, but regained coverage to 50 ha in the late 1940s (Wyer *et al.,* 1977). In the early 1950's it declined to less than 9 ha due to flood damage (Davidson and Hughes, 1998a; Wyer *et al.,* 1977), but had recovered fully by 1977 (Wyer *et al.,* 1977), to the 100 ha referred to above. Today Leigh Marshes makes up the vast majority of the 170 ha of *Zostera* that occurs among the Greater Thames Estuary sites. In total these meadows have reduced by 80% since the 1930's, 62% from Essex and 100% from Suffolk.

Wales**:** There are few published records of historic seagrass cover in Wales and no historic spatial extent estimates. Observations from fishers in the 1930's and 1950's describe concern that seagrass had become so prolific along the Glamorgan coast as to hinder safe boat passage and deplete their shellfisheries (Butcher, 1932, 1934). Recent work suggests that intertidal meadows occurring at Milford Haven are beginning to recover from centuries of pollution and anthropogenic disturbance in the area (Bertelli *et al.,* 2018), although subtidal sites in the same area are suffering from light limitation, likely as a result of eutrophication (Jones *et al.,* 2018).

Modelling the percentage loss of seagrass

The proportion of seagrass area per hectare of mudflat ranged from between 4% to 86% with an average of 32% ± 27%. The 1991 NCC report (Davidson *et al.,* 1991) established that estuaries comprised a total of 530,000 ha of coastal waters in mainland Britain. Of these, half are in England and almost one third are found within Scottish waters (Davidson *et al*., 1991). Within these, intertidal mud- and sandflats make up about 43%, totalling 254,400 ha (Davidson *et al*., 1991). The OSPAR dataset does not include any data from Ireland and is lacking in Scottish datapoints. It reports 143,571 ha of mud- and sandflats in mainland Britain, including those outside of estuaries. Considering one third of estuaries are found in Scotland it is unsurprising that the figures from these two reports do not align. The total current mapped areal extent of seagrass in mainland Britain (from this chapter's data) is 6,682 ha.

Using the NCC data on total mud- and sandflat area, the estimated maximum seagrass extent for mainland Britain is 81,953 ha, with an upper 95% confidence interval ranging from 126,430 ha to 40,964 ha (Table 5; Fig. 6). Using the OSPAR data on total mud- and sandflat area the maximum seagrass extent for mainland Britain is 43,559 ha, with 95% confidence interval ranging from 72,647 ha and 24,267 ha (Table 5; Fig. 6). Statistical comparisons between the two seagrass coverage values were made by comparing 95% confidence intervals. Overlapping confidence intervals indicated no significant difference between area estimates. The modelled data suggests that, with low certainty, between 36,799 ha and 75,193 ha of seagrass has been lost from mainland Britain, this would represent an 84% and 92% decline respectively (Table 5).

Table 5. Modelled maximum seagrass areal extent, area and percentage loss in mainland Britain from the NCC report (Davidson *et al.*, 1991) and the OSPAR dataset. Average and 95% confidence interval displayed.

	NCC total mudflat area ha			**OSPAR total mudflat area ha**		
	254,400			143,571		
	Max. seagrass extent *ha*	**Seagrass loss *ha***	**decline *%***	**Max. seagrass extent *ha***	**Seagrass loss *ha***	**decline *%***
Average	81,953	75,193	92	43,559	36,799	84
Upper 95%	126,430	119,670	95	72,647	65,887	91
Lower 5%	40,965	34,205	83	24,267	17,507	72

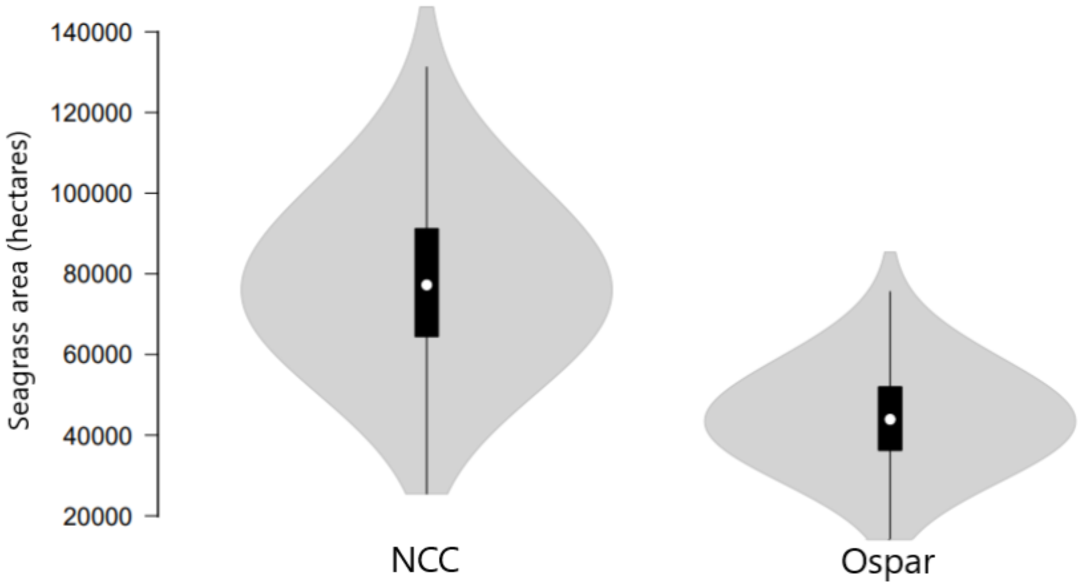


Figure 6. Modelled maximum seagrass areal extent in mainland Britain from the NCC report (Davidson et al., 1991) and the OSPAR dataset. Error bars indicate 95% confidence interval.

Discussion

To this best of my knowledge, this study is the first to systematically estimate the current and historic areal extent of seagrass in any country. I found a total of 8,493 ha of recently mapped seagrass, substantially less than the UNEP-WCMC figure of 13,753 ha and substantially more than the most recent estimates of 5,200 ha (Davidson and Hughes, 1998). These findings confirm the first hypothesis that: the current estimates of seagrass areal extent in the British Isles are out of data and inaccurate. With high certainty at least 41% of seagrass in the British Isles has been lost since 1936. Of this 34% has been lost since the 1980's, which is substantially more than the suspected global decline of 29% in the same period (Short *et al.*, 2010; Waycott *et al.*, 2009). With medium certainty 50% of seagrass has been lost since 1936, 42% since the 1980s. The modelled potential distribution of seagrass suggests with low certainty that up to 92% of seagrass has been lost from mainland British waters over the last 100+ years.

I provide estimated loss ranges because the paucity of survey data means it is impossible to know exactly how much seagrass has been lost from these waters. The high certainty estimates are almost certainly under-representative of the true scale of occurred losses. They only represent those meadows that have been documented and almost all of these would have undergone some level of degradation prior to their documentation. The model has limitations but without data it is an important step to understanding the wide-scale losses that have occurred. In consultation with ABPmer, the EA undertook a suitability model to assess where seagrass could occur in British waters. They documented 43,346 ha of suitable habitat in England alone. Based on the current areal extent of seagrass in England (3,796 ha) this would represent a 91% loss.

A similar suitability model was conducted for Wales, which indicated 4,541 ha of suitable habitat (Brown, 2015). Based on the current areal extent of seagrass in Wales (723 ha) this would represent an 84% loss. Together these models suggest a total of 47,888 ha of suitable seagrass habitat in England and Wales. This total is comparable to the lower estimate for the whole of mainland Britain (43,559 ha). Considering the suitability models and our lower estimate do not include any or many data points from Scotland, it would be reasonable to assume that the actual number is much closer to the higher estimate (81,953 ha).

The EA also undertook a modelling project to map the historic areal extent and loss of saltmarsh habitats in England. Digitally overlaying ordinance survey maps from 1860, they combined these with historic maps of saltmarsh extent, and estimated coastline flooding capacity using LIDAR data to calculate an historic areal extent estimate of 215,624 ha (Mike Best EA, 2019, personal communication). This represents an 85% reduction on current saltmarsh extent in England. Although the modelled estimates may seem high, they are seemingly not out of character for coastal degradation in the UK. If 85% of saltmarsh habitat has been lost in the UK, then the likelihood is that the environment which fringes it has also experienced widespread declines.

This study brings records together from disparate sources and provides the most up to date and accurate estimates of seagrass cover possible. The large-scale loss of seagrass described here redefines the severity and spatial extent of what is known about biodiversity loss in our coastal seas, setting a new baseline upon which future management and restoration can aspire to build. Given the need to restore and improve management of these ecosystems, in light of work highlighting their importance to British Isles fisheries (Bertelli and Unsworth, 2014) and carbon sequestration (Green *et al.*, 2018), and work highlighting the declining state of British Isles seagrass meadows (Jones and Unsworth, 2016), this work is much needed and timely in its arrival.

The rare accounts of documented areal extent of seagrass in the early 1900's provide an example of the changes that have likely occurred throughout the British Isles. Although these data are in isolation, the consistent declines noted in the literature, along with the documented 96% decline in average meadow size, confirms the trend of degradation pointed to by earlier studies (Butcher, 1932, 1933). Historic declines since the 1900's are mirrored by more recent declines noted in the last decade (Jackson *et al.,* 2016) and numerous incidences of small scale disturbances in recent years (Unsworth *et al.,* 2017).

Not all British seagrass has been lost and degraded. The research finds seagrass persisting at many sites across the British Isles, to varying degrees of extent, with occurrences of seagrass recovery at some sites. Although there is strong evidence to suggest that seagrass loss can lead to a state of negative feedback, where loss of seagrass reduce favourable (i.e. sheltered) conditions, preventing ecosystem recovery (Maxwell *et al.*, 2016), this has not been the case for all sites. Recoveries observed in Milford Haven, where historic pollution encroachment and oil spills had previously reduced seagrass (Bertelli *et al.,* 2018), indicate removing or reducing stressors can, in some locations, lead to habitat recovery. The recovery of Leigh Marshes in the Greater Thames Estuary further supports this. Its history suggests meadows are capable of fluctuating and can recover from dramatic losses. The ability of seagrass meadows to regain abundance is encouraging and should help spur conservation initiatives globally, especially current attempts to promote seagrass restoration.

Understanding the trends of seagrass decline in the British Isles

Rationalising the probable causes of such vast losses of seagrass in the British Isles is at best difficult, mostly because robust estimates regarding historic spatial extent of *Zostera* spp. are limited. Typically, the seagrass wasting disease *Labyrinthula* has been described as the primary cause of virtually all seagrass loss in Britain (Butcher, 1934; Cottam, 1935; Den Hartog, 1993; Garrard and Beaumont, 2014). This assumption stems from the discussion within Butchers 1932 report, where undoubtedly seagrass was lost due to disease. Butcher reported changes to habitat based on his own experiences of seagrass abundance, supported by those of individuals whose baselines only go as far back as their own inherited knowledge (Pauly, 1995). This case of SBS means the basis on which Butcher referred to healthy populations of seagrass is likely a gross underrepresentation of what once occurred in these waters and is an excellent example of how SBS impacts contemporary environmental knowledge. There has been no enquiry as to whether the seagrass habitats Butcher was assessing were already heavily degraded, with almost all the literature pointing to this period for the cause of seagrass degradation. Considering the early industrialisation of the British Isles, it is almost certain that the systems Butcher assessed in 1932 would have already undergone dramatic declines. It is likely that persistent gradual declines had been occurring for centuries before Butchers report and these have continued to the present day (e.g. Jackson *et al.,* 2016; Jones and Unsworth, 2016; Unsworth *et al*., 2017; Jones *et al.,* 2018).

As the first country to industrialise in the 17th and 18th centuries, Britain had been undergoing dramatic land-use transformation long before Butcher assessed the status of seagrass. Industrialisation is intrinsically linked to environmental degradation. By the time Butcher had been writing, dramatic physical alterations to the British landscape had been occurring for over 300 years. A reference to seagrass in the Tyne estuary in the early 1800's (Winch, 1831) referred to a site that has since been reclaimed, now containing an industrial estate. Coastal reclamation, dredging and building of sea walls were prevalent in the 17th century, 200 years prior to this account (Batty, 1997), and are highly likely to have been as in conflict with seagrass as they still are today (Erftemeijer and Lewis, 2006). Importantly, the UK was at the forefront of the global metal industry, with metal mining prevalent throughout many parts of the UK during the 1700s and 1800s, with many of these mines (e.g. Wales) still producing extensive metal contaminated discharge into coastal and estuarine waters today. The negative impacts of a suite of heavy metals, causing toxic conditions for seagrasses, are well documented (Prange and Dennison, 2000; Macinnis-Ng and Ralph, 2002).

In addition to industrial development, the vast scale of loss of oysters around the UK cannot be ignored as a fundamental change to the environmental conditions. Locations such as the Firth of Forth have entirely lost up to 5,000 ha of oyster beds (Thurstan *et al.*, 2013). These oyster beds would have fundamentally influenced the volume of suspended particles in the water column and hence the water clarity, creating conditions suitable for photosynthetic production by seagrass. Similar estimates are available for areas such as the Solent, the Thames, The Clyde, the Humber and the Severn (Thurstan *et al.*, 2013).

The need for improved seagrass assessment

The present analysis highlights a lack of coherent and systematic monitoring and mapping programmes of seagrass meadows in the British Isles. That 64% of records in the OSPAR dataset are older than 20 years highlights the extreme lack of constant effort in seagrass mapping. Despite this fact, the OSPAR dataset is the baseline for estimates of British Isles seagrass extent included by the WCMC. Since this is the first attempt that I am aware of to provide an accurate and up to date map of seagrass occurrence and declines for an entire country, it is likely that these are not the only data included by the WCMC that are inaccurate. The paucity of current data means that estimates, even coarse ones, are a necessary step to evaluating the pressures imposed on this habitat, in keeping with the accepted approach of 'use available data' (Hiscock, 1997). However, data inconsistencies can make it challenging to talk meaningfully about global seagrass trends and arguably managers should only be working with temporal and spatial data that we are reasonably confident is accurate. Regional and local mapping of sites around the world is important in ensuring that current attempts to increase seagrass abundance through restoration, rehabilitation and conservation have any hope to succeed.

Data inconsistencies found within this work are most obvious in Scotland, where most data were collected over 20 years ago. The isolation from human population likely means many meadows remain intact, including some documented over 20 years ago. However, as is evident by the 58% losses in this region over the last 80 years, many have likely been impacted by the extensive aquaculture industry, with fish farming a known cause of seagrass loss in other parts of the world (Berry and Davison, 2001). The meadows where historic and contemporary data are available show mass declines. The once huge meadows in the Cromarty and Dornoch Firths have been reduced from 3,241 ha to 1,200 ha and 2,862 ha to 116 ha respectively. Regardless, huge swathes of Scottish waters have not been surveyed for over two decades and could represent a vital stronghold of this once ubiquitous British habitat. Their condition and extent should be assessed with urgency. Scotland is not the only region where survey efforts since 1998 have been insubstantial. Pre- and post-1998 maps show a reduced survey effort across all regions (Fig. 5).

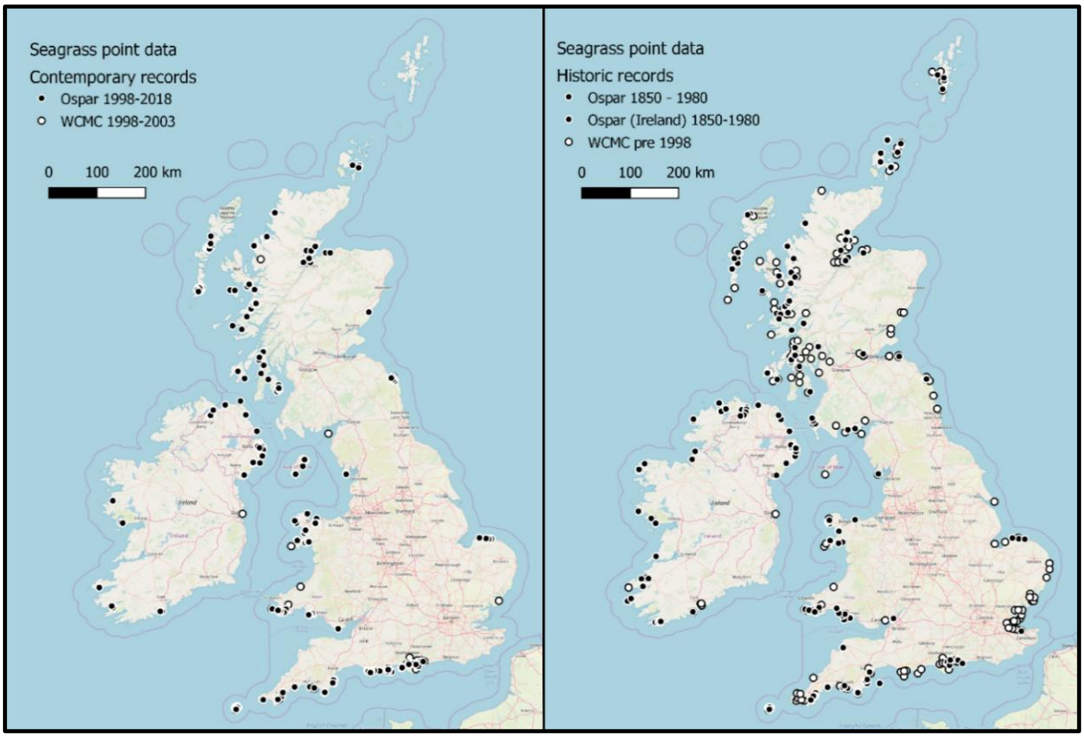


Figure 5. Seagrass point data from the OSPAR and UNEP-World Conservation Monitoring Centre datasets showing a) pre-1998 surveys and b) post-1998 surveys.

Impact of declines on the British Isles seagrass Blue Carbon capacity

In Chapter 3 sediment samples from 13 seagrass meadows along the southwest coast of England were analysed for organic carbon content. On average, the meadows contained 141 ± 73 Mg C ha within the top 100cm of seagrass sediment (Green *et al.*, 2019; Chapter 3). Based on these figures the estimated total carbon stored in the top 100cm of recently mapped seagrass of the British Isles is 1.2 Mt carbon. In mainland Britain this figure is 0.9 Mt of carbon (Table 5). Based on the upper (81,952 ha) and lower (40,965 ha) estimates of historic seagrass distribution, mainland British seagrass meadows could once have contained between 5.7 Mt and 11.5 Mt of carbon (Table 5).

The upper value of this is equivalent to 3% of the UKs CO_2 emissions in 2017 (Eaton, 2019). It is important to note that these data are from short cores (30cm), and the only data from long sediment cores (100cm) in the UK, from the Fleet lagoon (Chapter 4), suggest that extrapolating stocks in this way has the potential to underestimate the total stock by >40%.

This thesis only measured sedimentation rates for the Fleet lagoon, which has an unusual hydrodynamic regime, with extremely low tidal flushing (Chapter 4). Because of this it is unlikely to be representative of average seagrass sedimentation rate across the region. There are likely significant variations in the processes between sites, driven by a myriad of factors including location along the coastline and within an estuary, and proximity to rivers. The Fleet's sedimentation rate was low (0.044cm yr^{-1}), at the lower bound of sedimentation rates found in the literature (Duarte *et al.*, 2013a; Lavery *et al.*, 2013; Miyajima *et al.*, 2015; Röhr *et al.*, 2016). Reasonable and frequently used rates give low (0.044cm yr-1), medium (0.202cm yr-1), and high (0.42cm yr-1) bounds to frame carbon sequestration estimates (Duarte *et al.*, 2013a Macreadie; Lavery *et al.*, 2013; Miyajima *et al.*, 2015; Röhr *et al.*, 2016). Here, sequestration rates were estimated by dividing the total carbon estimates by the amount of time it takes to accumulate this stock using the sedimentation rates above, to provide estimates on average annual carbon accumulation of British Isles seagrass meadows (Lavery *et al.*, 2013; Röhr *et al.*, 2016) (Table 6). Assuming a medium sedimentation rate, the seagrass meadows of the British Isles are accumilating roughly 0.024 Mt C yr^{-1} (Table 6). Assuming this medium sedimentation rate, historic undisturbed seagrass meadows of the British Isles could have been absorbing 0.232 Mt C yr^{-1} (Table 6).

Table 6: Estimates of total carbon (Mt C) of modelled historic and contemporary seagrass distribution of mainland Britain, and of contemporary seagrass distribution of the British Isles, with low (0.044 cm yr^{-1}) medium (0.202cm yr^{-1}) and high (0.42cm yr^{-1}) estimates for carbon sedimentation per year (Mg C yr^{-1})

		Carbon stock	Sedimentation rates		
	Area ha	**Total carbon Mt**	**Low Mt C yr^{-1}**	**Medium Mg C yr^{-1}**	**High Mg C yr^{-1}**
Upper historic estimate	81,953	11.5	0.050	0.232	0.483
Lower historic estimate	40,965	5.7	0.025	0.115	0.239
Contemporary area British Isles	8,493	1.2	0.005	0.024	0.050
Contemporary area mainland Britain	6,760	0.9	0.004	0.018	0.038

Considering the need to include natural ecosystems in climate mitigation strategies, there is increasing interest in providing monetary valuations on carbon stock and sequestration estimates. The UK government has recently implemented a legal commitment to achieve Net-Zero greenhouse gas emissions by the year 2050. To reach this target will require major economic reforms, and substantial increases in natural carbon sequestration capacity. Putting a monetary value on carbon can help to portray value to a non-ecological audience. The proposed monetary value of carbon varies dramatically depending on the method used. The Stern Review suggests that for ecological systems the true social cost, which attempts to value the externalities of carbon, should be >£80/t (Stern, 2006). However, the traded value of ecological carbon on the voluntary market is actually much lower, currently around £7/t. Based on today's voluntary market price the value of the carbon stored in the top 100cm of recently mapped seagrass is £8.4 million with a yearly (medium) sequestration value (Table 6) of £0.14 million/ annum. Taking the Stern Reviews (Stern, 2006) societal value of carbon this would equate to £96 million with a sequestration value of 1.6 million/ annum (Table 7). Taking the upper (Table 6) range of the historic estimates of seagrass in mainland Britain, at today's voluntary market value, these would once have contained £80 million worth of carbon in their sediments, or £920million taking the Stern Review estimate. In this undisturbed state, these seagrass meadows could have been responsible for sequestering £1.6 million or £18.4 million worth of carbon every year, respectively (Table 7). These figures, although crude, offer a powerful indicative snapshot of what has been lost through long-term environmental degradation, and support the need to offer protection to those seagrass meadows that remain. Seagrass used to be ubiquitous along the shores of the British Isles, and if restored to even part its former extent, this ecosystem will provide valuable support to reaching a carbon neutral future.

Table 7. Estimates of total carbon (Mt C) and current and projected increases in carbon economic value (£million) of modelled historic and contemporary seagrass distribution of mainland Britain, and of contemporary seagrass distribution of the British Isles, medium (0.202cm yr^{-1}) estimates for carbon sedimentation per year (Mg C yr^{-1}) and associated current and projected increases in carbon economic value (£million).

			Total market value £million		**Total market value £million**		
	Area *ha*	**Total carbon** *Mt*	**Today's voluntary market £7/t**	**Stern Review >£80**	**Sequestration rate** *Mg C* yr^{-1}	**Today's voluntary market £7/t**	**Stern Review >£80**
Upper historic estimate	81,953	11.5	80.50	920.00	0.23	1.61	18.40
Lower historic estimate	40,965	5.7	39.90	456.00	0.12	0.84	9.60
Contemporary area British Isles	8,493	1.2	8.40	96.00	0.02	0.14	1.60
Contemporary area mainland Britain	6,760	0.9	6.30	72.00	0.02	0.14	1.60

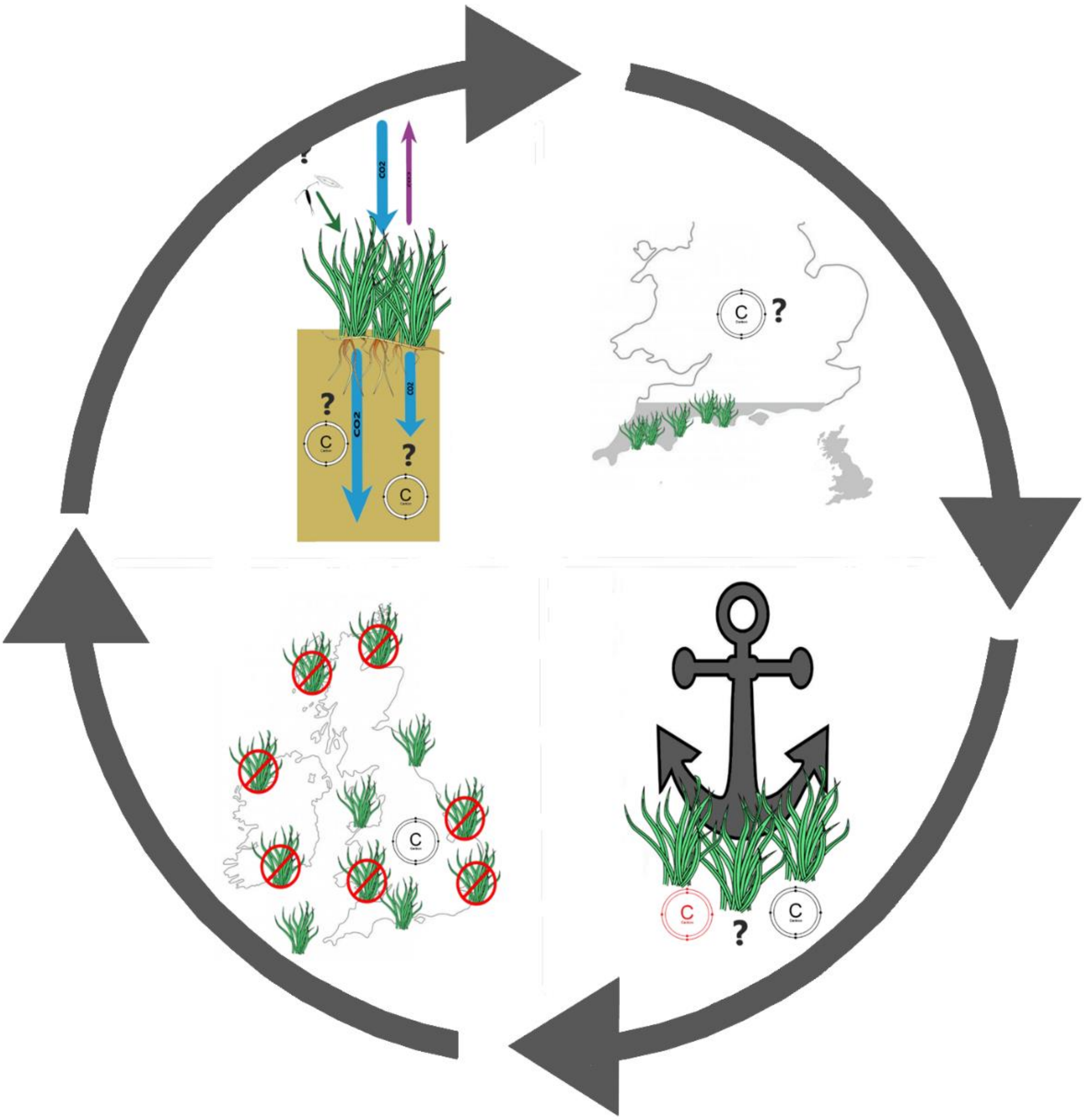
Conclusions

This study, to the best of my knowledge, provides the first attempt of any country to document the areal extent and historic loss of seagrass. It is particularly important considering the growing pressures of climate change, highlighting the result of uninhibited degradation of an ecosystem that could support emissions reduction. I document 8,493 ha of recently mapped (since 1997) seagrass in the British Isles. Using simple models to estimate seagrass declines across mainland Britain, triangulated against habitat suitability models, I provide evidence of catastrophic seagrass loss. With high certainty at least 41% of British Isles seagrass has been lost since 1936, 33% since the 1980’s. This loss may have been as high as 92%. This would equate to a loss of approximately 10Mt of stored carbon. Using average carbon accumulation values from the literature this amounts to a loss of over 0.02 Mt C yr^{-1}.

Although the British Isles have arguably been altering its natural habitats for longer than almost any other country, the trends and impacts of declines included in this chapter are likely occurring in many other developing and developed counties. It is hoped that this chapter will not only generate a better understanding of seagrass losses in the British Isles, but spur efforts to protect remaining seagrasses, restore historical losses and drive other countries to take stock of this vital coastal habitat to the same goal.

Chapter 8

Conclusions, future work and final reflections



Thesis conclusions

The research presented within this thesis has progressed the understanding of carbon storage dynamics and the status of seagrass in the British Isles, addressing significant knowledge gaps and highlighting the importance of British Isles seagrass meadows to regional and global Blue Carbon stocks (Fig. 1). By assessing the impact of short- and long-term environmental pressures on seagrass Blue Carbon stocks in the British Isles it has provided direct conservation and management suggestions for British Isles seagrass meadows. The aims of this thesis (Chapter 2) have been addressed by:

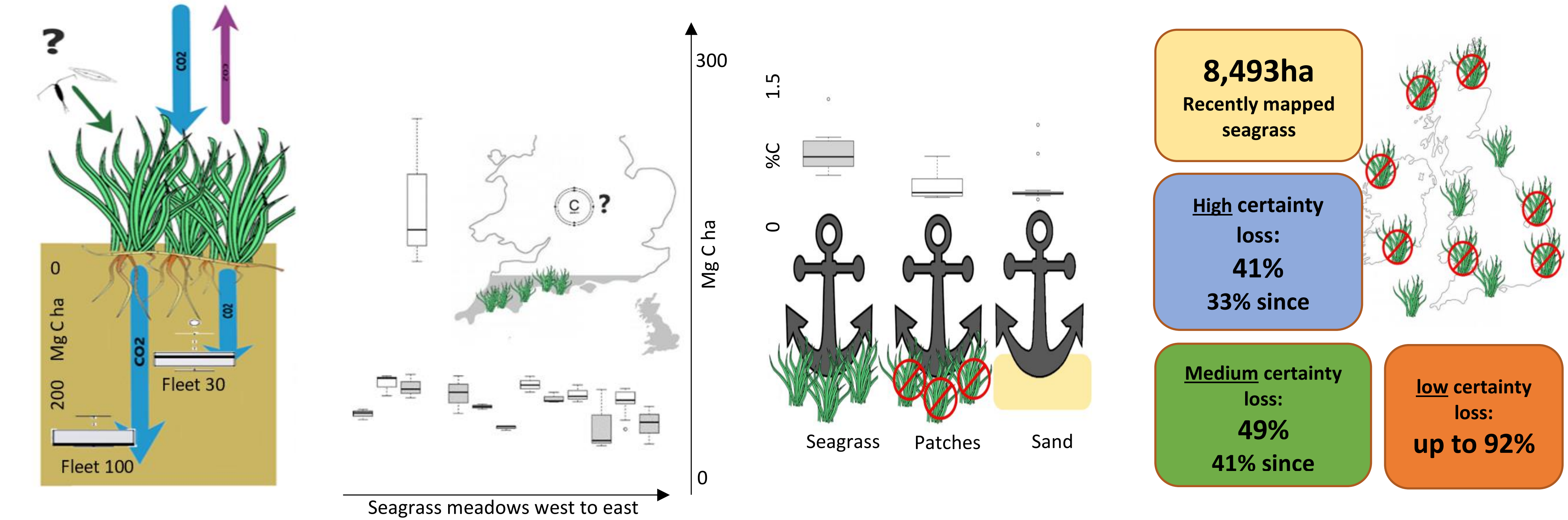
- Describing the carbon dynamics of a unique seagrass meadow, elucidating trends in storage, source and sequestration rates of carbon in comparison to globally accepted trends.
- Providing baseline data on the variability of carbon stocks from 13 meadows along the southwest coast of England, to deliver local estimates of Blue Carbon.
- Assessing short-term threats by testing, *in-situ*, the impacts of anchoring and mooring on seagrass carbon storage.
- Estimating the areal extent of seagrass within the British Isles, along with tiered high, medium, and low certainty percentage loss estimates. The aim of the low certainty estimates was to account, beyond *shifting baseline syndrome*, for the impacts of long-term environmental degradation on the seagrass meadows of the British Isles.

Chapter 4: Carbon dynamics of seagrass of the Fleet

Chapter 5: Variability of UK seagrass sediment carbon

Chapter 6: Anchoring and mooring reduce carbon storage

Chapter 7: Historic analysis exposes catastrophic seagrass loss in British Isles



Extrapolating carbon stocks from 30 to 100cm cores underestimated the stock by >40%.

There was a marked difference between stability in carbon accumulation in the lower and upper 50cm.

There was uniformity in carbon stocks across the 13 sites, apart from one site.

Sediment silt content and above-ground biomass could not explain differences.

All sites contained significantly more carbon than the average for the North Atlantic bioregion.

Carbon content was significantly higher in the sediments of undisturbed seagrass sediments than in sediments disturbed by anchoring or mooring and bare sediments, which never knowingly contained seagrass.

Helical mooring trial is recommended.

Catastrophic seagrass loss was documented across every region with good data in the British Isles.

Losses represent substantial monetary loss of carbon storage capacity, which could be rectified by restoration

Figure 1. Conceptual diagram outlining the key findings of each empirical chapter. Made by author in 3D Paint.

The primary aim of **Chapter 4** was to test widely held assumptions of carbon storage dynamics. Principally, it achieved this by testing the hypothesis that:

Extrapolating carbon stocks from a 30cm depth profile to a 100cm depth profile underestimates the total carbon stored within the Fleet's seagrass meadow.

This was achieved by analysing the relationship between carbon content and depth, the origin of carbon within the Fleet lagoon's seagrass meadow, and the rate of sequestration. The results confirmed the hypothesis, showing that, within this unique seagrass meadow, carbon stocks estimated by extrapolating data from a 30cm to a 100cm depth profile underestimated the carbon stock by >40%. This was due to the fact that organic carbon content showed an overall increase with depth, contradictorily to global assumptions of carbon dynamics at depth. The estimated carbon stock per hectare for this is greater than the global average and is almost four times richer in carbon than the average for the North Atlantic bioregion. Although this study focuses on one, atypical, seagrass meadow, it raises important questions regarding the use of generalised data in discussing carbon stocks, which are explored further throughout the thesis.

Interestingly, the carbon profile varied at depth, showing a stable, slightly increasing trend to ~55cm, followed by erratic and highly fluctuating carbon content up to 100cm. This is mimicked in the carbon signature values recorded and hinted at varied environmental conditions in carbon accumulation. The $\delta^{13}C$ signatures showed that seagrass contributed between 40-55% of the carbon pool, confirming global trends of seagrass carbon storage. $Lead^{210}$ analysis confirmed that sedimentation rates were low, falling at the lower bound of those recorded in the literature. An empirically based model suggests the carbon stored at 100cm was laid down between -1,142 BCE and

747 CE, supporting evidence that seagrass can accumulate carbon over millennial timescales. The model also explains the abrupt change in storage and source trends half-way down the core. The lower half would have been laid down before the Fleet formed in its current state and would have likely represented a far more unstable environment. Observed fluctuating values within the core data are likely representative of changing conditions across these times.

The primary aim of **Chapter 5** was to improve the baseline understanding of the carbon stocks found within the seagrass meadows of the British Isles, for local management and conservation, and for regional and global comparisons. It achieved this by testing three hypotheses:

1: *There is significant variation in British Isles seagrass sediment organic carbon density.*

2: *Aboveground biomass and sediment silt content significantly impact total carbon storage among British Isles seagrass meadows.*

3: *Local seagrass sediment carbon data reveals inconsistencies in regional seagrass sediment carbon estimates with implication for Blue Carbon schemes and seagrass conservation.*

The results showed surprising uniformity among sites, except for one site (Drakes Island) which contained carbon densities comparative to *Posidonia oceanica*, a species typically storing up to 70 times more carbon than other seagrass species. Unfortunately, I was unable to take long cores at any of the other sites, and it is possible that the overall stock is much higher. If the trends found for the Fleet are representative of the norm in the region, the stock estimates could be >40% higher. Using regression analysis, I found no relationship between plant count and dry bulk density, and depth, rejecting the second hypothesis. Importantly, the results showed that, although there was much uniformity among sites, they were significantly denser in carbon than the average for the North Atlantic region. Further, when compared to other *Z. marina* meadows within Europe, the stocks found along the southwest coast of England were high, second only to data from Denmark. This supports the third hypothesis, providing evidence to demonstrate the ecosystem service value of British Isles seagrass meadows.

The primary aim of **Chapter 6** was to provide evidence of short-term anthropogenic disturbance to seagrass carbon stocks in a site frequented by heavy pleasure-boat use. It tested the hypothesis that:

Sediments from within the seagrass meadow at Studland Bay contain more organic carbon than in-meadow bare patches created by mooring and anchoring, and adjacent bare patches that never knowingly contained seagrass.

This discrete study supported the hypothesis, finding that undisturbed sediments within the seagrass meadow contained significantly more carbon than sediments disturbed by anchoring and mooring, and sediments that never knowingly contained seagrass. Of interest is the finding that anchoring and mooring scars induced similar losses of carbon and returned the sediment to conditions similar to bare sand. In light of the recent Marine Conservation Zone designation at Studland Bay, it is recommended that ecologically friendly helical moorings be trialled instead of standard moorings, with the intention of wide-scale deployment in the meadow, supported by a ban on dropping anchor at the site. Whatever management practices are put in place, it is recommended that the reasons for management changes are carefully communicated to the local yachting community, including the findings reported here.

The primary aim of **Chapter 7** was to provide the most up-to-date and accurate estimate of seagrass areal extent in the British Isles, along with loss estimates. The tested hypotheses were:

1*: The current estimates of seagrass areal extent in the British Isles are out of date and inaccurate.*

***2:** There has been a substantial reduction in the spatial extent of seagrass in the British Isles with significant consequences to the Blue Carbon capacity of this resource.*

By compiling data from two large datasets and 14 additional providers I documented a total of 8,493 ha of seagrass in the British Isles, which was substantially more than the most recent published estimate of 5,200 ha, and substantially less than the OSPAR estimate of 13,752 ha (confirming the first hypothesis). The disparity is due to the fact that over 60% of the data from the OSPAR dataset is over 20 years old, which I excluded from my high certainty loss estimates. The decision to present high, medium and low certainty loss estimates is to account for the distinct lack of data, and so lack of certainty on the true picture of seagrass declines in the British Isles. The high certainty loss estimates showed that at least 41% of seagrass had been lost since 1936, 33% in the last 80 years. The medium certainty loss estimates showed that at least 50% of seagrass had been lost since 1936, 41% in the last 80 years. These figures fall substantially higher than the global estimation of 39% in the same period and are uniform across almost every region where good historic data is available.

The modelled data provided a snapshot of the impact of long-term environmental degradation on seagrass meadows throughout the British Isles. The purpose was to provide estimates that were not subject to *shifting baseline syndrome* and countered the perpetual blame of seagrass loss on the 1930s wasting disease. The results showed that up to 92% of seagrass has been lost historically across the British Isles, a result of long-term industrialisation and costal development, and associated pollution and encroachment on the coastal fringe. These figures, although based on a simple model, are supported by several habitat suitability studies, which point to similar pervasive declines.

The results from this work have far reaching impact. Firstly, they remind us that seagrass Blue Carbon science remains a relatively new field, and one that still has many uncertainties. The reasons for differences in carbon accumulation between sites have not been well accounted for in the literature. The findings of this work support the idea that there are vast variations among sites, even those formed of the same species, but were unable to elucidate the causes of these differences. Any additional data on local storage trends will support the move towards a more robust understanding of seagrass Blue Carbon storage and sequestration.

On a local level, the data within this thesis has provided multiple estimates of seagrass carbon storage, that have helped to raise awareness of seagrasses' ability to support climate mitigation in the British Isles. It has provided direct evidence of the anthropogenic disturbances that reduce carbon storage capacity of these sites. This is especially pertinent in relation to the recent designation of MCZ's in the UK. The UK government has now completed the designation process and is looking towards implementation of management and monitoring of these sites. It is hoped that the work in this thesis can provide direct evidence to support this implementation.

It is my opinion that the data collated for **Chapter 7** is the most impactful of this entire thesis. The results highlight a distinct lack of systematic and determined effort to monitor this important habitat. The monitoring effort between 1850-1998 is considerably higher across all regions, but particularly in Scotland, than from between 1998 and today. The fact that the countries of the British Isles did not, until this time, have a centralised understanding of where seagrass occurs in their coastal waters is quite astounding. The usefulness of this work to the wider community is supported by the fact that since presentation of this data at conferences, I have been contacted frequently by the British seagrass community for access to this data. It is hoped that this can form a strong basis on which to further improve the monitoring and mapping of seagrass throughout the British Isles.

Future work

I have learnt much over the four years of this PhD and looking back there are a few things I would have liked to do differently. I was able to join the Community Seagrass Initiatives fieldwork during the first summer of my PhD. This meant that I had a relatively short period of time, as an inexperienced PhD student, to design the coring apparatus and my data collection methodology. This resulted in my primary focus being on collecting sediment samples and, as a result, I overlooked the usefulness and importance of collecting seagrass leaves, roots and rhizomes, as well as other potential carbon sources from each site (Seston, algae, etc.). Returning to most of those sites required a significant amount of logistical support (i.e. money and time), and I was unable to retrospectively collect this data. By the time I decided I would like to have analysed carbon sources there was only one funding round at the NERC Life Sciences Mass Spectrometry Facility available to me, and I was unsuccessful in securing funds. If I had been able to trial some data before this, it would have helped my application process substantially and I may have been able to roll out carbon sources analysis across all my sites. However, the dataset contained within this thesis has far reaching impact and should support future work in this regard.

Alongside this, there is a substantial amount of further work I would have liked to complete during this PhD, had time and funding allowed. In fact, the nature of seagrass science in the UK means I could complete many more PhD's and subsequent years of research to help better answer the questions posed within this thesis. Fundamentally, to improve our knowledge of seagrass carbon storage and sequestration in the British Isles, a fuller inventory of carbon stock and sequestration rates should be taken. Ideally this would include taking long (>100cm) sediment cores from seagrass meadows along every coastal region in the British Isles, representing varied sub- and inter-tidal habitat conditions. This would offer a more accurate representation of average carbon storage, and therefore, stock estimates for the region, but would also provide extremely useful data on the relationship between carbon and depth, and the impact of varied habitat features on storage. This would require substantial funding, since extracting long cores from sub-tidal seagrass sediments is particularly challenging, in my experience. In an ideal world this work would also look at the sources of carbon found within these meadows, to examine their contribution to changes in storage and sequestration rates, as well as ^{210}Pb analysis to determine mean sequestration rates. This, alongside detailed analysis of other potential significant features such as sediment silt content/dry bulk density, above and belowground biomass and proximity to rivers or other sources of effluent, would hopefully illuminate causes of variation among sites.

One of the fundamental problems with seagrass Blue Carbon research at this time is that we do not have good data on seagrass respiration. Almost all estimates only consider what is already in the sediment, and what is sequestered into the sediment. I conducted some very basic preliminary data on whole system metabolism and methane emissions form the meadow at the Fleet but did not have the time to give the work the focus it needed. Further analysis of respiration of CO^2 and methane would be incredibly useful, both for the UK and globally.

Final thoughts

My reason for pursing a PhD was ultimately to prove myself as a scientist, capable of working within the interface between science and policy, alongside which I have a deeply rooted passion for the marine environment. Because of this it was important for me to design a PhD whose research would be of benefit to UK conservation and policy objectives. Seagrass was the obvious choice, since science and management is lacking globally, a trend which is mirrored here. What I didn't realise when making this choice was quite the potential to produce work that would be of so much use to the seagrass community, nor the importance of the timing of this piece of research. The UK has committed to achieve Net-Zero emissions by the year 2050 and there is a drive, like never before, to use our natural environment to support this goal. This is evidenced by the recent injection of funding into seagrass restoration via Sky Ocean Rescue in collaboration with Project Seagrass and Swansea University, whose primary promotional tactic is to discuss seagrass in terms of carbon storage potential. My data, showing up to 92% declines, has been used to support this. Many groups, including Sky Ocean Rescue, continue to use estimates of carbon storage from global trends that are skewed by *Posidonia oceanica,* which stores on average 40% more carbon than other seagrass species. The motivations for Sky's funding of this project are purely conservationist, desiring to be responsible for the first successful restoration scheme in the UK. Therefore, the use of these global figures are unlikely to have negative consequences here. However, if we want to upscale restoration efforts, for which there is much interest in the current climate, we need to present and discuss seagrass carbon storage values transparently. Results from this work have led to conversations with a wide range of people from government, NGO's and industry wanting to work with seagrass to help mitigate climate change. Overwhelming, these scientists and managers are being asked to provide firm and deliverable values on seagrass sequestration rates and outcomes of restoration projects, which the science cannot currently back up. To ensure momentum behind seagrass restoration does not wain as a result of undeliverable targets we need to first work on the science to ensure we have sound scientific backing for decisions made. The additional work highlighted above would help to achieve this but requires substantial financial input.

Another fundamental barrier to seagrass conservation in the UK, is the relationship between seagrass conservationists and the yachting community, epitomised by the hostile rapport between these two groups over conservation measures at Studland Bay. The conflicts that have occurred here have filtered through, at least in the southwest, whereby conservation groups struggle to instigate meaningful dialogue. Efforts need to turn to this to ensure conservation and restoration attempts happen with the support of this key stakeholder group, including the provision of supporting research findings. This could be further supported by a social science project that looks to understand the drivers for resistance to change within this group. Most importantly, though, conservation efforts need to take the time to understand the concerns and needs of the people that use the sites being protected.

It has been an absolute pleasure to complete the work found within this thesis. I am immensely grateful for the opportunity to follow my scientific interests, and design and pursue work that I feel is of real benefit to the seagrass community.

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Appendices

Appendix 1: Published data from Chapter 5

PLOS | ONE

RESEARCH ARTICLE

Variability of UK seagrass sediment carbon: Implications for blue carbon estimates and marine conservation management

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Abstract

Seagrass meadows provide a multitude of ecosystem services, including a capacity to sequester carbon dioxide (CO_2) within their sediments. Seagrass research in the UK is lacking and there is no published data on sediment carbon (C) within UK seagrass meadows. We sampled 13 *Zostera marina* meadows along the southwest coast of the UK to assess the variability in their sedimentary organic carbon (OC) stocks. The study sites were considered representative of sub-tidal *Z. marina* meadows in the UK, spanning a gradient of sheltered to exposed sites, varying in formation, size and density, but found along the same latitudinal gradient. OC stocks (C_{stocks}) integrated across 100cm depth profiles were similar among all sites (98.01 ± 2.15 to 140.24 ± 10.27 Mg C ha^{-1}), apart from at Drakes Island, which recorded an unusually high C_{stock} (380.07 ± 17.51 Mg C ha^{-1}) compared to the rest of the region. The total standing stock of C in the top 100cm of the surveyed seagrass meadows was 66,337 t C, or the equivalent of 10,512 individual UK people's CO^2 emissions per year. This figure is particularly significant relative to the seagrass area, which totalled 549.79 ha. Using estimates of seagrass cover throughout the UK and recent UK C trading values we approximate that the monetary value of the UK's seagrass standing C stock is between £2.6 million and £5.3 million. The C stock of the UK's seagrass meadows represent one of the largest documented C stocks within Europe and are, therefore, of important ecosystem service value. The research raises questions concerning the reliability of using global or regional data as a proxy for local seagrass C stock estimates and adds to a growing body of literature that is looking to understand the mechanisms of seagrass C storage. When taken with the fact that seagrass meadows are an important habitat for commercially important and endangered species in the UK, along with their declining health and cover, this research supports the need for more robust conservation strategies for UK seagrass habitats.

Introduction

Seagrass meadows provide a multitude of ecosystem services, including a capacity to sequester CO_2 within their sediments [1]. Along with mangroves and salt marshes, the organic C absorbed in these coastal ecosystems has been termed 'blue carbon' and has generated considerable interest in recent years, in part because preservation and restoration of these habitats can help mitigate climate change [2]. Unfortunately, seagrasses are declining with estimates that at least 49% of UK seagrass coverage has been lost in the last 35 years [3]. This loss not only removes the sequestration potential of these habitats but can also remineralise sedimentary C that has accumilated over time, leads to a reduction of nursery and feeding habitat for commercially important and endangered speices [4], increases sediment and coastal erosion [5]) and reduces coastline nutrient cycling [6–8].

Zostera marina, the UK's dominant seagrass species, is a temperate seagrass found throughout Europe, the USA and the northwest Pacific. Globally eelgrass is declining by approximately 1.4% per year, with large scale declines in some locations, particularly within Europe and east coast USA, due to wasting disease [9]. Much of the evidence of wasting disease in the UK is anecdotal, and with no complete historic inventory of UK seagrass meadows mapping accurate changes over time is challenging at best. Prior to the outbreak of wasting disease in the 1920s eelgrass would have been found in the majority of subtidal mudflats in Britain, which was once considered 'clothed' in eelgrass [10]. Following the outbreak of wasting disease, eelgrass was restricted to only the most sheltered sites, such as lagoons, and is now considered nationally scarce [10]. Meadows that do persist are reportedly in a 'perilous state', damaged and degraded, and healthy beds are now a rarity [11].

Despite recognition by the EU Water Framework Directive of seagrass as bioindicators for ecosystem health [12] research related to UK seagrass habitats is lacking relative to other regions (e.g., Med and Aus [13]). More specifically, there are no published estimates for the C stored in the UK's seagrass habitats. This is surprising considering the proliferation of blue C research in recent years, with key papers [12–16] highlighting the vital role seagrasses play in absorbing CO_2. Occupying less than 0.2% of the ocean floor, seagrass habitats are estimated to be responsible for approximately 10% of the yearly ocean C burial [13,17], a disproportionately large storage potential relative to their global extent [18]. Seagrasses produce aboveground foliage forming canopies in the water column, which slow water, forcing sediment to settle and become trapped within the canopy layer. In this way particles from the water column are absorbed into their sediments, where the overwhelming majority of the C stored by these habitats is located [8]. On average 2.51 ± 0.49 Mg C ha is stored in the living biomass (roots and rhizomes) of seagrass compared to 194.2 ± 20.2 Mg C ha in sediment [13]. This process means that seagrasses can store C through both photosynthesis (autochthonous) and through trapping particles containing C that has come from external sources (allochthonous) such as seston, algae or debris of terrestrial origin. A global assessment of studies suggest that up to 50% of the C stored by seagrass is allochthonous, making seagrasses particularly affective C sequesters since they bind C that could be released back into the ocean by other less stable sinks [8]. Seagrasses form understory mats, made up of dense root systems that stabilise sediments and bind C [17]. These mats can extend to over 10m and create anaerobic sediments that, if left undisturbed, can bind C for millennia [19,20]. In comparison, terrestrial soils whose productivity is often dependent on soil turnover, tend to bind C for decades only [21].

Seagrass ecosystems likely represent a '*globally significant carbon stock*', with estimates suggesting that 19.9 Pg C is stored in the top 1m of the worlds' seagrass sediments, equivalent to the global fossil fuel and cement production in 2014 [13,22]. The Fourqurean paper [13] has done much to increase awareness and has propelled seagrass into blue C research focus. However, values are derived from regional estimates with between 1 and 29 data points and Mediterranean and Australian habitats comprise 42% of the total data points from this study [13]. Further, the North Atlantic averages are from only 24 samples, none of which are from UK waters [13]. With such limited available data, these studies have been useful in promoting the advancement of seagrass C research. The challenge is that limited data means these estimates are biased regionally, and by species, so tend to generalise storage capture trends [23]. *Posidona* oceanica, a seagrass species found throughout the Mediterranean and known to be exemplar in its ability to store C, dominates the literature, which has been evidenced to skew regional and global extrapolations [23]. Variations in C storage among species, and among habitats formed of the same species, are known [23,24], but the characteristics that affect this, and the impact of habitat distinction are less well understood [23–25]. Direct measurements from regions and species that are under-represented will help to improve global knowledge and develop more reliable estimates of the C storage capacity and potential of seagrasses. For countries where blue C research has developed further there has been a move towards incorporating it into domestic climate policy, going so far as to discuss the inclusion of blue C stocks within Greenhouse Gas (GHG) inventories [26]. Clearly the next step toward successful integration of blue C policy is more robust estimates of C storage across the different blue C habitats.

This study aims to document the C storage in seagrass beds along the southwest coast of the UK. The study objectives were to: (1) compare sediment organic C (OC) of 13 seagrass meadows, on the same latitude and exhibiting varied habitat features; (2) establish the impact of habitat variance on sediment C storage; (3) estimate the average C stock (C_{stock}) per unit area to provide a comparison to global and regional data and; (4) estimate the total C stored within each habitat to understand the significance of the UK's seagrass habitats. The results will (a) provide a baseline assessment of the UK's seagrass C storage capacity; (b) build on the growing body of literature comparing seagrass C storage locally and globally; and (c) indicate the potential monetary significance of the UK's blue C storage for this habitat type.

Materials and methods

Thirteen study sites (Fig 1, Table 1) exhibiting varied habitat characteristics were selected for the current study. Sites were considered representative of sub-tidal seagrass meadows found across the British Isles, varying in size, degree of shelter, and formation (Table 1). In addition, sites represented varying degrees of marine protection, ranged from 0.02ha to 275ha and varied in aboveground density. Sites were located on the same latitudinal gradient between 50˚ 18' 36.36'' and 50˚ 38' 34.20''N.

Sample collection among all sites were completed in summer 2016 (Fig 1). At each site, three sediment cores were collected from sea depths of 3-8m using SCUBA gear, except at the Fleet, where samples were collected from depths of $<$0.6m using snorkelling gear. At each site, two divers were dropped from a dive boat, roughly in the centre of the bed, and sampling locations, at least 20m apart, were randomly selected. Permission to collect material was granted by the Marine Management Organisation by providing 'notice of intention to carry on an activity under The Marine Licensing (Exempted Activities) Order 2011 [27] (as amended) "the Exemptions Order"' (EXE/2016/00148). Since the Fleet is property of the Ilchester Estate further permission was provided by the Fleet Warden and by Natural England.

At each location one cylindrical PVC core (70mm diameter, 40cm long) was manually inserted into the sediment to a depth of 30-35cm. Cores were extracted and capped underwater and stored vertically in a lift bag for the remainder of the dive. Once returned to the boat, samples were kept vertically in a covered cool box until arrival on shore. On shore, cores were sliced into 3cm sections, bagged and frozen to await transfer back to the laboratory for analysis. In addition to the sediment cores, three 50cm^2 quadrats were randomly placed around the core and plant densities were estimated by counting the number of plants within the quadrant. The seagrass meadow at the Fleet was considerably larger than any other bed (Table 1) so three

Fig 1. Location of seagrass meadow sites along the southwest coast of the UK.

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sites were allocated for sampling. The strict protection surrounding the Fleet, and its shallow depth, meant that a kayak was used to reach core locations, so as not to disturb the seagrass. Meadow exposure and bed formation were visually assessed during site visits.

Laboratory analysis

On returning to the laboratory, samples were thawed and divided into two sub-samples. One sub-sample was used for Loss on Ignition (LOI) analysis and the other was freeze-dried for grain size analysis and total organic C content using an elemental analyser.

Organic C and carbonate analysis. Since its presentation in 1974 [28] LOI, the burning of sediments at 550˚C and 950˚C, has been widely used as a method to estimate the amount of organic matter (OM) and carbonate mineral content in soil samples [29]. The relationship between LOI at 550˚C and OM content, and LOI 950˚C and carbonate content is accepted as standard [29]. There exists a relationship between OM and OC, which has led to the OM found by burning sediment at 550˚C being used as a proxy for OC. However, this method is semi-quantitative and relies on an empirically derived relationship between OC and OM, the strength of which varies with material [30]. The most accurate method to analyse OC is

Table 1. Characteristics of surveyed seagrass meadows.

Site	Protected status	Exposure	Meadow formation	Area (ha)	N	W
Looe	MCZ	Exposed	Very patchy	56.52	50˚ 21’ 11.52’’	4˚ 26’ 30.48’’
Plymouth						
Cawsands	SAC	Partly sheltered	Very patchy	11.77	50˚ 19’ 52.32’’	4˚ 11’ 53.52’’
Firestone Bay	SAC	Sheltered	Patchy	0.76	50˚ 21’ 37.8’’	4˚ 9’ 37.44’’
Drakes Island	SAC	Partly sheltered	Dense	4.25	50˚ 21’ 25.56’’	4˚ 9’ 10.08’’
Jennycliff Bay	SAC	Exposed	Patchy	11.77	50˚ 20’ 27.96’	4˚ 7’ 49.08’’
Yealm CC	SAC	Sheltered	Dense	0.14	50˚ 18’ 36.36’’	4˚ 3’ 58.68’’
Tomb Rock	SAC	Sheltered	Sparse	0.15		
Torbay						
Elbery Cove	MCZ	Sheltered	Sparse	29.31	50˚ 24’ 17.64’	3˚ 32’ 41.28’’
Torre Abbey	MCZ	Very exposed	Very patchy	104.11	50˚ 27’ 38.52’’	3˚ 32’ 1.32’’
Fishcombe Cove	MCZ	Very sheltered	Very patchy	0.23	50˚ 24’ 11.52’’	3˚ 31’ 17.76’’
Hopes Cove	SAC	Partly sheltered	Gradient	2.73	50˚ 27’ 52.56’’	3˚ 29’ 16.44’’
Weymouth / Poole						
Fleet	SAC, SSSI, RAMSAR SPA, UNESCO	Very sheltered	Dense	274.68	50˚ 37’ 72.20’’	2˚ 33’ 43.30’’
Studland Bay	No protection	Very sheltered	Dense	53.37	50˚ 38’ 34.20’’	1˚ 56’ 38.30’’

Abbreviations are as follows: MCZ = marine conservation zones, SAC = special area of conservation, SSSI = special scientific site of interest, RAMSAR = convention on wetland of international importance, SPA = special protected area, UNESCO = world heritage. Area values provided by CSI (Community Seagrass Initiative)

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through dry combustion in an Elemental Analyser (EA) [30]. However, the costs involved are often prohibitive. A study analysing the global data set of seagrass C storage demonstrated that the relationship between OM and OC for seagrass sediments is strong, therefore, OM is accepted as a proxy for OC [13,30]. To improve the predictability of OM to OC two linear equations have been developed for samples with %OM higher or lower than 0.2% [13,30]:

$$\%LOI < 0.20\%\ OC = -0.21 + 0.40(\%LOI);$$

$$\%LOI > 0.20\%\ OC = -0.33 + 0.43(\%LOI).$$

Although these equations are deemed suitable for OC estimations under IPPC regulations, accuracy can be further improved by sending a limited number of samples to be analysed in an EA [13,30].

The dry mass of each sample was calculated by weighing wet sub-samples before and after drying at 105˚C for 18-24hrs [29]. The samples were then put in the furnace at 550˚C for two hours, re-weighed and returned for two hours at 950˚C [29]. OM content was calculated by subtracting the combusted sediment (550˚C) from the sediment dry weight. Carbonate content was calculated by subtracting the remaining combusted sediment (950˚C) from the sediment dry weight [29].

Using stratified random sampling 10% of dried samples were selected for analysis in a *Flash EA* (BEIF Lab; UCL, London). Large items, such as roots and shells, were removed by hand before the samples were homogenised. All samples contained significant levels of carbonates so were acidified to remove these before analysis. Sub-samples were submerged in HCL diluted to 1N and placed in an ultrasonic bath for 15 minutes [30]. Samples were then left overnight (>18hours). More acid was added the following day to check for further effervescence and once no new outgassing was observed samples were centrifuged and decanted from the acid. Samples were washed by adding deionised water, sonicated for 15 minutes and then centrifuged to separate the sample and the liquid for decanting. This was repeated three times before the samples were dried at 60˚C for >24hrs. The treated samples were then analysed using the *Flash EA* for %C [30].

The %C results represent %OC as the samples had any inorganic C removed prior to analysis. The relationship between %OM (LOI) and %OC was formulated by developing a linear equation for the analysed samples and applying this to the rest of the %OM results.

Grain size analysis. Sediment grain size was determined from freeze dried samples from one core for each meadow, which was assumed to be broadly representative of the entire site. Sediment samples were each dry sieved through a sieving tower for 10 minutes. Seven sieves were used; 2mm, 1mm, 0.5mm, 0.25mm, 0.15mm, 0.125mm and 0.054mm. Total mass of sample and mass of retained soil in each sieve was recorded. Sediment silt content was calculated as the percentage of sediment retained below 54μm. Sediment characteristics were further analysed using GRADISTATv8 software [31].

Estimating seagrass OC stocks. A C stock (C_{stock}) refers to the total amount of C within a habitat of a known size, normally comprising a number of C pools, i.e. reservoirs of C in soil, vegetation etc. [30]. Since the amount of C within the living biomass of seagrass is negligible [13] C_{stock} here refers to the total stock of OC within the sediments of each meadow of a known size.

C_{stock} for each meadow was estimated over a 30cm core sample. Where a 30cm sample was not achieved the missing slices were estimated using the relationships between depth, soil weight from a known volume (dry bulk density, hereafter, DBD) and OC, to determine OC at 3cm intervals up to 30cm [13], <5% of core slices were estimated in this way.

The C_{stock} of the top 30cm of the 13 studied seagrass meadows were calculated as follows. Soil DBD was calculated from the mass of a dried sample and its original volume (*DBD (g/ cm³)* = mass of dry soil (g) / original volume sampled (cm³)). Soil C density (SCD) was calculated from DBD and total OC content ($SCD = DBD^{m}(OC/100)$. Total C in each core slice (TC/S) was determined from the SCD and known sample volume ($TC/S = SCD^{m}3cm$), and, finally, each slice within the core was summed to give total C within a core ($TC/S_1 + TC/S_2 + TC/S_{n\ldots}$). Values were converted into Mg C/Hectare^{-1} and total C in the top 30cm of each meadow was determined by averaging the total core C and multiplying by area [30]. For global comparisons stock estimates were extrapolated to 100cm following the IPCC protocol [30] and then extrapolated for the whole of the UK to provide estimates of total standing stock. To compare to regional trends units were converted to g C m^2 and integrated values to 25cm were used.

Statistical comparisons for C_{stock}, DBD and plant density were conducted to determine site-specific differences. Test for normality and homogeneity of variance were done to establish if ANOVA or Kruskal Wallace test should be performed. All analysis was completed in SigmPLot 13.0.

Results

Seagrass meadow formation and sediment characteristics

The mean DBD in UK seagrass sediments ranged from 0.34 g cm^3 ± 0.10 (Fleet) to 1.19 g cm^3 ± 0.09 g cm^3 (Studland Bay) with an average of 0.96 g cm^3 ± 0.22 g cm^3 (Table 2). Mean %OM content ranged from 1.40% ± 0.67% (Studland Bay) to 12.32% ± 5.39% (Drakes Island) with an average of 3.61% ± 3.31% and a median of 2.47%. The Fleet and Drakes Island %OM were markedly higher (Table 2) and differed significantly to all other sites ($p < 0.001$). There was no significant difference between the Fleet and Drakes Island, and no significant difference between all other sites.

Table 2. Sediment characteristics and aboveground biomass.

Site	Sediment silt content %	DBD ($g\,cm^3$)	%OM	%OC	SCD (mg C cm^2)	C_{stock} 30cm (Mg C ha)	Plant density (plants/50cm^2)
Looe	20.03 ± 1.25	0.98 ± 0.10	2.3 ± 0.76	1.20 ± 0.31	11.08 ± 0.49	33.30 ± 1.47	8.53 ± 6.27
Plymouth Sound							
Cawsands	12.72 ± 1.77	1.11 ± 0.12	2.47 ± 0.74	1.25 ± 0.32	14.21 ± 1.08	42.07 ± 3.08	6.08 ± 5.76
Firestone Bay	13.34 ± 2.91	0.86 ± 0.08	3.47 ± 0.55	1.62 ± 0.31	14.19 ± 0.67	40.99 ± 3.38	4.05 ± 5.88
Drakes Island	5.51 ± 1.43	0.77 ± 0.07	12.32 ± 5.39	4.94 ± 2.00	37.76 ± 6.75	114.02 ± 21.45	10.42 ± 8.40
Jennycliff Bay	2.44 ± 0.66	1.07 ± 0.04	2.51 ± 0.43	1.30 ± 0.16	13.89 ± 0.65	39.07 ± 5.35	2.84 ± 4.75
Yealm CC	14.55 ± 1.70	0.87 ± 0.07	2.68 ± 0.48	1.37 ± 0.18	11.83 ± 0.21	35.39 ± 0.70	6.7 ± 7.01
Tomb Rock	8.85 ± 1.29	0.96 ± 0.05	1.85 ± 0.48	1.04 ± 0.21	10.15 ± 0.40	29.40 ± 0.65	4.21 ± 4.51
Torbay							
Elbery Cove	21.99 ± 2.46	1.05 ± 0.28	2.59 ± 0.47	1.33 ± 0.18	13.84 ± 0.56	41.74 ± 2.28	10.63 ± 9.45
Torre Abbey	12.02 ± 2.50	1.14 ± 0.09	1.97 ± 0.10	1.10 ± 0.04	12.56 ± 0.50	37.76 ± 1.50	5.52 ± 5.10
Fishcombe Cove	4.81 ± 1.79	1.04 ± 0.10	2.43 ± 0.65	1.28 ± 0.24	13.08 ± 0.72	38.94 ± 2.44	5.71 ± 7.64
Hopes Cove	14.71 ± 1.83	1.09 ± 0.07	1.56 ± 1.84	0.95 ± 0.68	10.73 ± 3.91	30.08 ± 8.89	7.61 ± 5.98
Weymouth/ Poole							
Fleet	29.92 ± 5.30	0.34 ± 0.10	9.39 ± 2.95	3.82 ± 1.14	12.07 ± 1.49	37.76 ± 3.84	n.a.
Studland Bay	1.99 ± 0.66	1.19 ± 0.09	1.40 ± 0.67	0.86 ± 0.27	10.13 ± 1.80	37.76 ± 5.39	53.53 ± 10.45

Data are site means ± standard deviation. % silt content; DBD = g dry bulk density; %OM = % organic matter; %OC = % organic carbon; SCD = soil carbon density mg C / cm^2; C sock Mg C ha = megagrams of C per hectare; plant density = no. plants per 50cm^2

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To determine the relationship between %OM and %OC 10% of samples were analysed in a *Flash EA* (BEIF Lab; UCL, London). A regression analysis determined the relationship as:

$$\%OC = 0.3708\%LOI + 0.3732$$

Our empirically derived relationship was not as strong as the equation derived from the global literature ($R^2 = 0.38$ vs $R^2 = 0.96$) [13]. To assess the reliability of the developed equation the %OM results were put through both equations and differences were statistically analysed. The differences were not significant ($p<0.001$), so analysis was based on our linear equation. Average %OC content ranged from 0.86 ± 0.27% (Studland Bay) to 4.94% ± 2.00% (Drakes Island). Mean %OC was 1.70% ± 1.23% and median %OC was 1.28%. Sediment profiles showed no change at depth (Fig 2). As with %OM, %OC at the Fleet (3.82% ± 1.14%) and Drakes Island (4.94% ± 2.00%) were significantly higher than all other sites ($p < 0.001$).

Integrated over a depth profile of 30cm the C_{stock} of UK seagrass meadows ranged from 29.40 ± 0.65 Mg C ha (Tomb Rock) to 114.02 ± 21.45 Mg C ha (Drakes Island), more than twice the value of the next highest C_{stock} (42.07 ± 3.08 Mg C ha at Cawsands), with an average of 41.54 ± 4.54 Mg C ha (Table 2). Despite the high %OC at the Fleet the low DBD meant that its C_{stock} was below average among the sites (37.76 ± 3.84 Mg C ha). Removing Drakes Island from the data reduces the range substantially with an average of 37.02 ± 4.22 Mg C ha. To allow for global comparisons C_{stock} was extrapolated to 100cm as per the IPCC guidelines for coastal wetlands [29, 31]). The 100cm depth-integrated C_{stock} among sites ranged from 98.01 ± 2.15Mg C ha (Tomb Rock) to 380.0 ± 71.51 Mg C ha (Drakes Island), with an average of 140.98 ± 73.32 Mg C ha. In both cases (C_{stock} 30cm and 100cm) there was a significant difference between the total C_{stock} of Drakes Island compared with all other sites. There were no significant differences between any other sites.

Sediment characteristics also varied between sites, ranging from sand to sandy silt. Sediment silt content ranged from 1.99% ± 0.66% (Studland Bay) to 29.92% ± 5.30% (the Fleet).

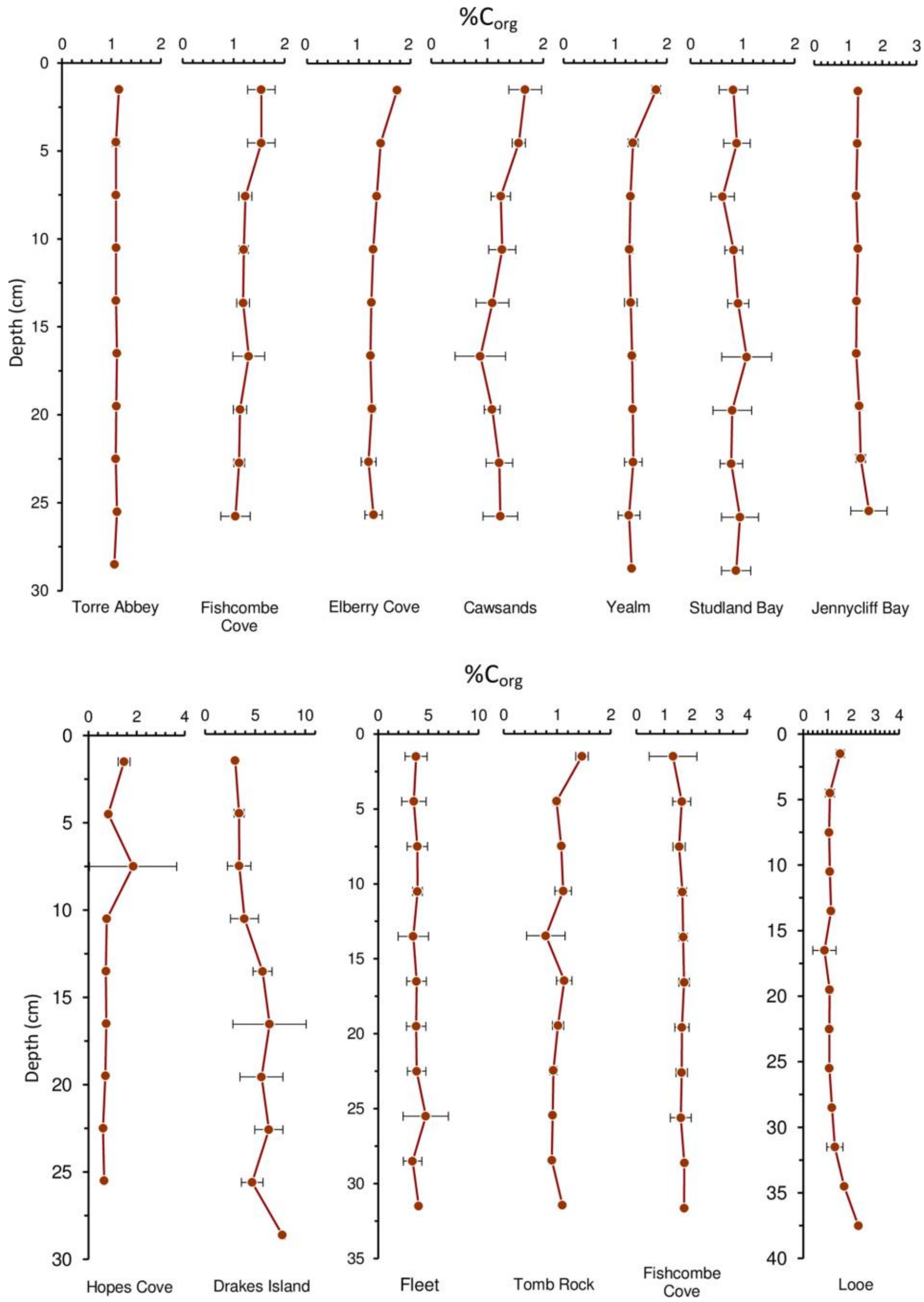
%Corg
Depth (cm)
Torre Abbey
Fishcombe Cove
Elberry Cove
Cawsands
Yealm
Studland Bay
Jennycliff Bay
%Corg
Depth (cm)
Hopes Cove
Drakes Island
Fleet
Tomb Rock
Fishcombe Cove
Looe

Fig 2. Depth profiles of the top 25-40cm of sediment cores from the average at each site. OC expressed as a percentage of the dry weight. Note the variations in x and y axis among some of the sites.

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Only Studland Bay and Drakes Island were statistically different from one another ($p < 0.001$). The Folk and Ward description of sorting [31] ranged from Moderately Well Sorted to Very Poorly Sorted among sites (Table 2). The Fleet was the least well sorted (Very Poor), Cawsands and the Yealm were also Poorly Sorted. The remaining sites were Moderately and Moderately Well sorted.

Plant density ranged from 2.84 ± 4.75 plants per 50cm^2 (Jennycliff Bay) to 53.53 ± 10.45 plants per 50cm^2 (Studland) (Table 2). Studland Bay had statistically higher aboveground biomass than any other site ($p < 0.001$), all other sites were the same. This contradicts the visual inspection of the sites and may show that taking biomass from the centre of the meadow is misrepresentative of the whole site. Many sites recorded high standard deviation compared to average plant count, highlighting the patchiness of some sites. Patchiness was particularly pronounced at Fishcombe Cove (5.71 ± 7.64 per50cm^2) and Jennycliff Bay (2.84 ± 4.75 per50cm^2). Standard deviation was high across all sites apart from Studland Bay (53.53 ± 10.46cm^2), which was the most consistently dense meadow.

In general, the surveyed meadows ranged from dense uninterrupted beds (Fleet, Studland, Drakes Island) to open sand with small patches of seagrass cover (Cawsand, Firestone bay). The Fleet and Studland Bay both contain large bare patches within their dense beds, the Fleet for reasons currently unknown and Studland Bay because it is a popular anchorage and contains numerous anchor scars. Site exposure differed among sites. The Fleet is a lagoon, flanked by Chesil Bank and connected to the sea by a narrow channel to the south that leads into Portland Harbour. In comparison, the meadow at Torre Abbey lies in the middle of a large bay, ~500m from shore, with frequent through traffic from the port, and no protection from oncoming weather.

Meadow size varied from 274.68ha (the Fleet) to 0.14ha (Yealm) with most sites smaller than 60ha (Table 1). Sea depth of site ranged from 2.5m (Studland Bay) to 7.7m (Hopes Cove). Average site depth was 5.10 ± 1.60m. The environmental data showed very weak regression relationships between all parameters and C_{stocks}: C_{stock} and plant density ($R^2 = 0.003$); C_{stock} and average site depth ($R^2 = 0.034$); C_{stock} and sediment silt content ($R^2 = 0.064$); C_{stock} and size ($R^2 = 0.021$); C_{stock} and dry bulk density ($R^2 = 0.012$) and; C_{stock} and %OM ($R^2 = 0.372$).

Discussion

This study is the first to estimate seagrass C storage in the UK. It demonstrates that despite contrasting habitat features there is little variation in the C_{stocks} among sub-tidal *Z. marina* habitats, existing on the same latitude along the southwest coast of the UK. These results contradict a growing body of literature that has found variations in the C storage of seagrass meadows among habitats formed of the same species [12,13,22,24,29,31]. Although documenting large variation these studies were unable to provide an adequate understanding of factors influencing OC accumulation and storage. Our results suggest that habitat conditions do not meaningfully influence the C_{stock} within the UK's seagrass meadows. The mechanisms which influence sediment C accumulation in seagrass meadows remain unclear.

Drakes Island appears to be exemplar in its C storage ability in the region. The 100cm depth integrated C_{stock} at Drakes Island is nearly three times higher (380.07 ± 17.51 Mg C ha) than the average of all other sites (140.98 ± 73.32 Mg C ha). All other sites contained similar C_{stocks} ranging from 98.01 ± 2.15 Mg C ha to 140.24 ± 10.27 Mg C ha. Other studies have found that accumulation of fine-grained sediments within seagrass beds significantly influences seagrass C storage [8,25,32]. The relationship between sediment silt content and C_{stock} among these sites was weak, suggesting this was not an influencing factor among sites. Drakes Island had one of the lowest sediment silt contents (5.51 ± 1.43%) and the site with the highest silt content (Fleet 29.92 ± 5.30) did not have particularly high C_{stock}, although its %OC (3.82 ± 1.14%) was high and the low C_{stock} is likely due to the low dry bulk density at the site (0.34 ± 010). Aboveground biomass is also attributed to higher C_{stocks} among seagrass meadows [33], though this was not evident in the data ($R^2 = 0.003$). Studland Bay had by far the highest average plant count (53.53 ± 10.45 per 50cm²) (Table 2), but an average C_{stock} (37.76 ± 5.39). Plant count at Drakes Island was reasonably high (10.42 ± 8.40 per 50cm²) but standard deviation was also high, suggesting a less uniform cover of dense growth. Patchiness within sites was generally high, indicating potentially poor ecosystem health [11]. Fishcombe Cove (5.71 ± 7.64), Jennycliff Bay (2.84 ± 4.75), Firestone Bay (4.05 ± 5.88) and Yealm (6.70 ± 7.01) all displayed vast variations among surveyed quadrats but overall no relationship was noted between plant count, or patchiness, and C_{stock}.

That the expected trends are not identified within these results should not render them insignificant. It is likely that the high OC content found at the Fleet is in part attributable to the high sediment silt content. More intricate factors are likely involved that allow Drakes Island to store more C where its sediment is less suited and restrict Studland's sequestration capacity where its canopy is more favourable. This study was unable to assess the sources of C within the seagrass meadows, which can be an important influencing factor determining C_{stocks} [25]. Sources of C contributed to up to 73% of the difference between C storage in *Z. marina* habitats in the Nordics [25]. On average 50% of sedimentary OC is derived from allochthonous sources [14,34], and it may be that the ratio of C contribution (*Z. marina*: external sources) is an influencing factor. Further analysis should be considered to understand the relationships between C_{stock}, silt content and aboveground biomass among these sites.

Seagrass systems typically have very little sediment turnover [35]. C diagenesis causes a gradual breakdown of labile and later increasingly stable C [35]. The result is normally a decrease in organic matter at depth. The sediment profiles at these sites did not fit this trend. It may be that the shallow 30cm cores are not sufficiently deep to describe the expected negative exponential profile that represents OC decomposition with age [20]. However, other studies have recorded this with similar core lengths [23]. Deep cores (1-2m) at key sites should be taken to fully examine this relationship.

C stock comparisons

The mean sediment C_{stock} for the top 100cm of sediment (140 ± 73.32 Mg C ha) was just short of the global average of 194.2 ± 20.2 Mg C ha [13] (Fig 2). The range of C_{stock} between sites was large (98.01–380 Mg C ha), but greatly reduced when Drakes Island (380.07 ± 71.51 Mg C ha), was removed (98.01–140.24 Mg C ha). Four sites fell below the globally documented range of 115.5–829.2 Mg C ha (from 41 100cm cores), though when you include the global extrapolated data from cores at least 20cm deep (extrapolated to 100cm), the range widens from 9.1–829.2 Mg C ha [13]. In these cases, values tend to be lower, so deeper cores at the surveyed sites may well reveal higher C stores. All the surveyed sites contain average C_{stock} well above the average for North Atlantic seagrass meadows (48.7 ± 14.5 Mg C ha) (Fig 3) and increased the number of data points from 24 to 37 [13]. Surprisingly, Drakes Island is comparable to the Mediterranean averages, dominated by *Posidonia oceanica* (Fig 3). It is not uncommon for sites to exhibit C stores well above those within its region [25]. Understanding the mechanisms behind the C stores at Drakes Island might help to deepen our understanding of seagrass C storage.

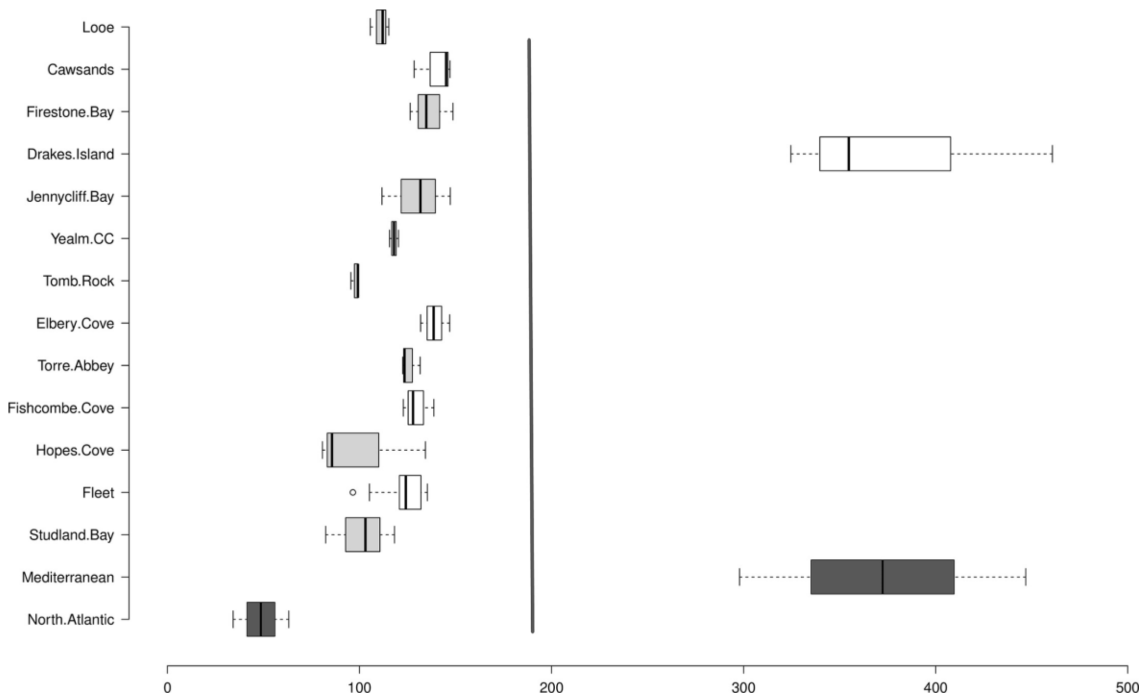

Fig 3. Average C_{stock} of 13 seagrass meadows along the southwest coast of the UK with regional comparisons (dark grey Mediterranean and North Atlantic) and global average (grey line) extrapolated from [12].

https://doi.org/10.1371/journal.pone.0204431.g003

Mediterranean cores contribute 22% of the total data for the global average, skewing the average substantially. The disparities between our results and the average for the North Atlantic further highlight the dangers of using global and regional data as a proxy for local seagrass C storage. There is a growing desire to use seagrass blue C as a mechanism to increase seagrass protection worldwide. Blue C research has come under recent scrutiny [36] and to maintain robustness we must be transparent about the services provided by local habitats, and refrain from overgeneralising. The C_{stock} values documented for the UK's seagrass meadows fall within the upper range of those recorded in the rest of Europe. Across Europe estimates of *Z. marina* C_{stock} vary considerably, ranging from 500 ± 50.00 g C m^2 to 4324.50 ± 1188.00 g C m^2

Table 3. Mean C_{stocks} in European *Z. marina* meadows in the literature and present study.

Country	Region	C_{stock} ± Stdev (g C m^2)	Depth (cm)	No. sampling locations	Reference
Denmark	Baltic Sea, North Sea	4324.50 ± 1188.00	25	10	(25)
UK	Southwest coast, English Channel	3371.47 ± 1625.79	25	13	Present study
Sweden	Southern Sweden, Baltic Sea	2000.00 ± 2121.32	25	5	(38)
Portugal	Southern Portugal, North Atlantic	1000.00 ± 120.00	25	2	(38)
Finland	Southern Finland, Baltic Sea	627.00 ± 25.00	25	10	(25)
Bulgaria	Eastern Bulgaria, Black Sea	500.00 ± 50.00	25	2	(38)
Poland	Puck Bay, Baltic sea	148.21 ± 90.31	10	3	(39)

https://doi.org/10.1371/journal.pone.0204431.t003

in the top 25cm of sediment [25,37,38] (Table 3). With an average C_{stock} of 3372.47 ± 1625.79 g C m^2 the UK is second only to Denmark. The variation between regions is considerable and both the UK and Denmark contain anomalous sites with significantly higher C_{stocks} than the rest of their region; 8649.93 ± 2330.02 (Drakes Island) and 26 138 ± 385.00 (Thurøbund) respectively, but also consistently higher C_{stocks} across all sites when compared to the rest of Europe.

As with Drakes Island no obvious explanation for the Danish sites high C content was given above its location in a 'relatively sheltered site' and large amounts of organic sediments [25]. This study found greater variations in the C_{stocks} of eelgrass sediments than our study noting that eelgrass production, root: shoot ratio and contribution of *Z. marina* to the C pool explained 67% of the variation. Similar analysis at the sites included in this study would make an interesting comparison here. The large variation among regions demonstrated by these studies further highlights the danger of using global and regional data as a proxy for estimating local blue C values. It also confirms that even within species there is considerable variation in seagrass C storage capacity and suggests that abiotic factors are more important than biological. Although the drivers remain unclear, the C stored in the seagrass meadows along the southwest coast of the UK represent one of the largest known stocks within Europe and, therefore, represent important sites for further study and conservation.

That seagrass meadows can also be a source of CO_2 and atmospheric methane (CH_4) has largely been neglected in the literature [39,40]. A recent study suggests that seagrass could be contributing up to 30% more to the global CH_4 emissions than previously thought, and calls for these emissions to be included in seagrass C calculations [36]. There also lacks at the root of blue C science adequate understanding of how OC accumulated in soils can be remineralised to CO_2 and re-released back into the water column, where it has the potential to enter the atmosphere [40]. A recent paper suggests the dissolution of calcium carbonate from the inorganic C pool has the potential to buffer the C sequestration capacity of seagrass meadows, in some cases perhaps shifting habitats to C sources [40]. These mechanisms need more exploration and will vary regionally. Regardless, they call into question the reliability of global seagrass C sequestration estimates. Unfortunately, these considerations are outside the scope of this study, the core aim of which to provide the first estimates of C standing stock in the UK's seagrass sediments. We argue that this data is much needed, especially within the current climate of forwarding marine conservation goals in the UK. Thus, stock calculations alone provide vital, much needed, information on this under-studied habitat. Hopefully, future studies can investigate the flux of C, and further add to the data pool both locally and globally.

Significance of C stocks for UK

There were marked differences in the sizes of seagrass beds in the surveyed sites, and by association, the C pools within each bed (Table 4). Total C pool in the top 100cm of the surveyed sites ranged from 14.52 t C at Tomb Rock to 33,578.31 t C at the Fleet. Despite the high C_{stock} found within Drakes Island, the site itself is very small (4.25ha) and contains only an estimated 1,616.67 t C within the top 100cm of its sediments. The estimated C pool in the top 100cm of the 13 surveyed sites along the southwest coast of the UK was 66,337 t C, or the equivalent of 10,512 individuals UK peoples CO^2 emissions per year. This is clearly not a significant number in terms of the UK's GHG emissions. However, for an area covering half the size of Richmond Park (London's largest park) this figure is significant relative to its size. The Fleet is a large seagrass bed and contains 10% of the annual CO^2 emissions of its closest town (Weymouth). The seagrass beds within this study make up a fraction of those found in the UK. A number of studies have estimated the cover of seagrass meadows in the UK, although the actual extent

Table 4. Mean C_{stock} and equivalent monetary value of seagrass meadows along the southwest coast of the UK.

Site	C_{stock} 100cm (Mg C ha)	C_{stock} 25cm (g C m^2)	Size (ha)	Total C (Mg C ha)	Monetary value
Looe	111.00 ± 4.91	2643.48 ± 146.31	56.52	6273.74	£150,570
Plymouth Sound					
Cawsands	140.24 ± 10.27	3436.78 ± 228.89	11.77	1650.65	£39,616
Firestone Bay	136.62 ± 11.26	3253.35 ± 271.38	0.76	103.83	£2,492
Drakes Island	380.07 ± 17.51	8649.93 ± 2330.02	4.25	1615.28	£38,767
Jennycliff Bay	130.25 ± 17.83	3273.08 ± 95.31	11.77	191.46	£4,595
Yealm CC	117.97 ± 2.34	2882.59 ± 10.05	0.14	16.16	£388
Tomb Rock	98.01 ± 2.15	2396.87 ± 69.82	0.15	14.52	£349
Torbay					
Elbery Cove	139.13 ± 7.60	3343.82 ± 204.30	29.31	4077.79	£97,867
Torre Abbey	125.87 ± 5.00	2995.01 ± 119.94	104.11	13105.65	£314,536
Fishcombe Cove	129.82 ± 8.12	3175.36 ± 143.14	0.23	29.86	£717
Hopes Cove	100.26 ± 29.62	2539.40 ± 812.30	2.73	273.71	£6,569
Weymouth/ Poole					
Fleet	122.25 ± 12.80	2849.96 ± 376.33	274.68	33578.38	£805,881
Studland Bay	101.25 ± 18.00	2389.48 ± 432.16	53.37	5403.96	£129,695

C_{stock} Mg C ha = mean megagrams of C per hectare over 100cm profile ± standard deviation; C_{stock} g C m2 = mean grams C per M^2 over 25cm profile ± standard deviation; Size = meadow size; total C = total C in top 100cm Mg C ha

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remains uncertain [41–44]. The total mapped area of *Z. marina* is 4887ha [43], though not all seagrass beds in the UK have been mapped. This figure is derived from some Special Areas of Conservation (SAC) and additional data from published studies only [44]. A reasonable estimated extent of seagrass seems to fall between this number and 10,000ha [41–44]. Taking the average from this study the estimated total standing stock of C in the UK's seagrass meadows is, therefore, between 108,427 and 221,870 t C. This is substantially higher than the Garrard & Beaumont [37] estimates which used *Z. marina* C stocks from European sites, estimating that the UK's seagrass meadows had the potential to store between 8050–16,100 t C. To fully grasp the significance of the UK's seagrass C stocks a full inventory of the UK's seagrass habitats should be completed and sediment cores from a wider range of meadows analysed. Further, the sequestration rate of these beds should be analysed to understand how much C per year these sites are able to sequester. Using the UK governments estimated traded central C value for 2017 of £24/t [45], the UK's seagrass sedimentary C stock has a monetary value of between £2.6 million and £5.3 million or an average of £3,360/ha in the top 100cm.

Conservation implications for C stocks in UK

This study adds to the growing literature base that highlights the importance of the UK's seagrass habitats [4,11,46]. Despite the growing knowledge that *Z. marina* beds in the UK are important nursery grounds for economically important fish species [46], and that they are mostly in a poor ecological condition [11], conservation of these habitats is lacking.

Studland Bay is the only site without any legislative protection, though it is being considered for designation as a Marine Conservation Zone (MCZ) this year (2018). The remaining sites are protected either as Special Areas of Conservation (SACs) or as MCZs, apart from the Fleet, which is a SAC, a Site of Special Scientific Interest (SSSI), a RAMSAR site (Wetlands), a Special Protected Area (SPA) and a UNESCO world heritage site (Table 1). Despite these designations there are no restrictions on dropping anchor at any of the SAC or MCZ sites.

Studland Bay, Fishcombe Cove and Cawsands are favoured anchorage sites for yachters and have several anchor scars within their meadows. The impact of anchoring activities on seagrass beds is contested, especially in the UK where the yachting community are greatly opposed to any anchorage restrictions. However, a recent paper [47] has unequivocally demonstrated that direct scouring of the bed by anchors, and the subsequent resuspension and loss of fine-grained sediments as a consequence, has resulted in a loss of OC content in disturbed areas. Scars showed evidence of intensive sediment mixing, which lead to the OC stocks being significantly lower than sediments under undisturbed seagrass [47]. In the UK, moorings, which are also present at Studland Bay, have also been shown to negatively impact seagrass cover with one mooring chain potentially responsible for the loss of up to $122m^2$ of local seagrass [48].

Studland Bay is one of the most highly contested seagrass sites in the UK, with forceful opinion on either side as to whether it should be designated as an MCZ. It provides a habitat for numerous commercially important (bass, bream, flatfish) and endangered (undulate ray) species, as well as being the only known breeding ground for both species of seahorses (*Hippocampus hippocampus* and *Hippocampus guttulatus*) found in the UK [49]. Further, it is recognised by Natural England as one of the best recovered sites since the decimation of the UK's seagrass meadows by wasting disease in the 1920s [49].

The seagrass bed in Studland Bay is a frequented anchorage for yachters coming out of Poole Harbour, who drop anchor in their hundreds during the summer months [50]. The anchor scars are visible from satellite images and cause obvious disruption to the otherwise dense meadow. The yachting community have successfully countered 15 years' worth of lobbying to protect this site under UK law. Initially included in the original proposal for 127 MCZ designations across England in 2011, Studland Bay was excluded from tranche one (2013) and two (2016), due mainly to the objections of local people and the yachting community [51]. It has been included for consideration in the third tranche, though it is likely to gain serious resistance from the local and yachting community. Arguably, part of the reason for their adamant resistance to marine protection or the introduction of ecologically friendly moorings has been the focus on a flagship species approach to conserving this habitat, i.e. efforts have fixated on highlighting the fact that the site is a breeding ground for two protected seahorse species [52]. The calls to protect these species have largely fallen on unsympathetic ears. The attempts have created a turbulent relationship between the conservation and yachting community so that now any efforts to approach a mutual resolution are met with animosity. The flagship species approach is one that often fails to entice the diversity of stakeholders needed to ensure effective conservation [53]. By widening the debate, to include a potentially growing C stock, a more positive dialogue may be allowed to develop. By taking a monetary approach to conserving this site there is reasonable argument in favour of protecting the C found within. The total estimated C in the top 100cm of the seagrass meadow at Studland Bay is 5,403 t which has a monetary value of £129,695. The estimated value of recreational and harbour activities that are argued to be affected by conservation management in Studland Bay totals £81,100 [49].

Conclusions

This study provides the first data on *Zostera marina* sediment C storage in the UK and offers a more accurate estimation of seagrass blue C stocks in UK waters. The work brings 13 more seagrass meadows into the global and regional dataset and, like many other studies, highlights uncertainties surrounding the variances in sediment C storage. The results show considerable uniformity, which is unusual, and, in line with other research, indicate an incomplete understanding of the factors that influence this [13,14,23,25,32]. Considered alone, the uniformity of the sites within this study suggests abiotic factors are not a strong driver of sediment C variability. However, when estimates of C storage from other European *Z. marina* meadows are considered it seems clear they are the primary cause of variance. Although unable to identify the drivers for this, the seagrass meadows along the southwest coast of the UK contain C_{stocks} that are significant in a European context and are, therefore, important both ecologically and in terms of ecosystem services to the region. We would argue that, for blue C purposes at least, grouping seagrass into bioregions is not a useful way to discuss similarities or differences, as even the same species within the North Atlantic bioregion vastly contradict each other.

Studies like this provide an essential snapshot of the complex processes that influence C sequestration. Detailed analysis of sedimentary structure, hydrodynamic regime, and seagrass canopy structure is vital if we are to better understand the causes of variation. Without this detail, global estimates will remain unreliable. Only by documenting inter-habitat variability will we be able to extrapolate the importance of seagrass ecosystems in a meaningful way, and thereby justify and promote measures for their improved protection.

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Appendix 2: Data providers from data collected in Chapter 7

List of all government, non-government, and private organisations, which contributed data to this work. This includes data contributors from the Ospar dataset, and additional contributors. * indicates providers outside of the OSPAR and EA dataset

Data Provider	**acronym**
Community of Arran Seabed Trust*	COAST
Community Seagrass Initiative*	CSI
Devon Biodiversity Records Centre	DBRC
Dorset Environmental Records Centre*	DERC
Department of Environment Northern Ireland	DOENI
Devon Wildlife Trust	DWT
Ecospan*	
Environment Agency	EA
English Nature	EN
Hampshire Wildlife Trust	HWT
Inshore Fisheries and Conservation Authority*	IFCA
Joint Nature Conservancy Committee	JNCC
Natural England	NE
Natural Resources Wales*	NRW
Poole Harbour Commissioners*	PHC
Regional Wildlife Trusts	WT
Scottish Natural Heritage	SNH

Appendix 3: Historic record (pre-1998) of seagrass occurrence included in this thesis

List of all data included in historic estimates of seagrass for Chapter 7 including: date of record, area, place name, data location, data owner, and area in ha. Contemp. = contemporary post-1998 area. High C. loss = high certainty loss, Medium C. loss = medium certainty loss (where the site has not been revisited since 1998). See acronyms in Appendix 2.

Date	Area	Place Name	Dataset	Data Owner	Historic area ha	Contemp. area ha	High C. loss ha	Medium C. loss ha
1986	Scotland	Cromarty Firth	Ospar	Fox et al 1986	3242	1200	2041.47	2041.47
1986	Scotland	Dornoch Firth	Ospar	Fox et al 1986	2862	116.98	2745.28	2745.28
1986	Scotland	Moray Firth	Ospar	Fox et al 1986	1140			1139.56
1995	Scotland	Islay	Ospar	JNCC	44			43.58
1995	Scotland	Loch Caolisport	Ospar	JNCC	4.38			4.38
1995	Scotland	Loch Carron	Ospar	JNCC	3.33			3.3264
1995	Scotland	Loch Creran	Ospar	JNCC	1.16			1.16
1997	Scotland	Shetland	Ospar	JNCC	6.67			6.67
1995	Scotland	Loch Sheildaig	Ospar	JNCC	11			11.18
1995	Scotland	Loch Sunart	Ospar	JNCC	3.33	0.03	3.3	3.3
1995	Scotland	Loch Sween	Ospar	JNCC	28			28.31
1995	Scotland	Loch Torridon	Ospar	JNCC	8.85			8.85
1990	Scotland	Lochgolihead	Ospar	JNCC	2.14			2.14
1995	Scotland	Longa Island	Ospar	JNCC	25			24.45
1995	Scotland	Oban	Ospar	JNCC	5.04			5.04
1995	Scotland	Skye	Ospar	JNCC	39			38.83
1991	Scotland	Dumfries area	Ospar	JNCC	37			36.55

1990	Scotland	Stranraer	Ospar	JNCC	3.86			3.86
1996	Devon	Cawsands	Ospar	JNCC	14	12.06	2.22	2.22
1996	Devon	Drakes Island	Ospar	JNCC	6.43	4.41	2.02	2.02
1996	Devon	Fowey	Ospar	JNCC	6.17			6.17
1996	Devon	Yealm	Ospar	JNCC	6.66	5.64	1.02	1.02
1996	Devon	Fal	Ospar	JNCC	86	38.8	46.81	46.81
1996	Dorset	Portland Harbour	Ospar	JNCC	6.3	1.59	4.71	4.71
1997	Northumbria	Lindisfarne	Ospar	NE	1046	678.96	367.04	367.04
2004	Cornwall	Fal and Helford Estuaries	Ospar	CWT	208	166	42	42
1998	Scilly Isles	Scilly Isles	Paper	NE	325	196	129	129
2004	Dorset	The Fleet	Ospar	DWT	233	62.5	170.5	170.5
1961	Essex	Foulness/Maplin Sands	Paper	Burton, 1961	320	40.19	279.81	279.81
1961	Suffolk	Stour and Orwell Rivers	Paper	Burton, 1961	380	1.43	378.57	378.57
1936	Northumbria	Spurn Bight	Paper	Philip, 1936	550	0.59	549.41	549.41

Appendix 4 – Contemporary (post-1997) of seagrass occurrence included in this thesis

List of all data included in historic estimates of seagrass for Chapter 7 including: date of record, area, place name, data location, data owner, and area in ha. Contemp. = contemporary post-1998 area. High C. loss = high certainty loss, Medium C. loss = medium certainty loss (where the site has not been revisited since 1998). See acronyms in Appendix 2.

Date	Area	Place Name	Database	Data Owner	Area ha
2017	Cornwall	Fal	WFD	EA	3.95
2017	Cornwall	Fal	WFD	EA	0.09
2015	Cornwall	Looe	Individual contribution	CSI	56.81
2015	Cornwall	Helford	Individual contribution	EcoSpan	23.63
2015	Cornwall	Cornwall	Individual contribution	EcoSpan	17.20
2015	Cornwall	Fal	Individual contribution	EcoSpan	16.94
2015	Cornwall	Fal	Individual contribution	EcoSpan	7.21
2015	Cornwall	Cornwall	Individual contribution	EcoSpan	5.89
2015	Cornwall	Fal	Individual contribution	EcoSpan	5.19
2015	Cornwall	Helford	Individual contribution	EcoSpan	4.67
2015	Cornwall	Helford	Individual contribution	EcoSpan	3.02
2015	Cornwall	Helford	Individual contribution	EcoSpan	2.61
2015	Cornwall	Fal	Individual contribution	EcoSpan	2.49
2015	Cornwall	Cornwall	Individual contribution	EcoSpan	2.37
2015	Cornwall	Helford	Individual contribution	EcoSpan	2.18
2015	Cornwall	Fal	Individual contribution	EcoSpan	2.02
2015	Cornwall	Fal	Individual contribution	EcoSpan	1.38
2015	Cornwall	Maenporth	Individual contribution	EcoSpan	1.36
2015	Cornwall	Looe	Individual contribution	CSI	1.35
2015	Cornwall	Helford	Individual contribution	EcoSpan	0.87
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.80

2015	Cornwall	Fal	Individual contribution	EcoSpan	0.79
2015	Cornwall	Helford	Individual contribution	EcoSpan	0.71
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.56
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.47
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.42
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.31
2015	Cornwall	Looe	Individual contribution	CSI	0.23
2015	Cornwall	Helford	Individual contribution	EcoSpan	0.13
2015	Cornwall	Helford	Individual contribution	EcoSpan	0.10
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.10
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.09
2015	Cornwall	Helford	Individual contribution	EcoSpan	0.04
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.03
2015	Cornwall	Fal	Individual contribution	EcoSpan	0.02
2015	Cornwall	Swanpool	Individual contribution	EcoSpan	0.01
2016	Cumbria	Barrow in furness	WFD	EA	36.88
2016	Cumbria	Barrow in furness	WFD	EA	20.66
2016	Cumbria	Barrow in furness	WFD	EA	4.06
2016	Cumbria	Barrow in furness	WFD	EA	1.65
2016	Cumbria	Barrow in furness	WFD	EA	1.17
2016	Cumbria	Barrow in furness	WFD	EA	0.59
2016	Cumbria	Barrow in furness	WFD	EA	0.02
2016	Cumbria	Barrow in furness	WFD	EA	0.02
2016	Cumbria	Barrow in furness	WFD	EA	0.01
2016	Cumbria	Barrow in furness	WFD	EA	0.01
2016	Cumbria	Barrow in furness	WFD	EA	0.01
2016	Cumbria	Barrow in furness	WFD	EA	0.01
2016	Cumbria	Barrow in furness	WFD	EA	0.01
2016	Cumbria	Barrow in furness	WFD	EA	0.00
2016	Cumbria	Barrow in furness	WFD	EA	0.00
2016	Cumbria	Barrow in furness	WFD	EA	0.00
2017	Devon	Exe Esturary	WFD	EA	95.65

2017	Devon	Plymouth Tamar	WFD	EA	30.98
2017	Devon	Exe Esturary	WFD	EA	27.67
2017	Devon	Salcombe	WFD	EA	11.61
2017	Devon	Exe Esturary	WFD	EA	10.13
2017	Devon	Plymouth Tamar	WFD	EA	9.79
2017	Devon	Exmouth	WFD	EA	7.66
2017	Devon	Plymouth Tamar	WFD	EA	7.66
2017	Devon	Salcombe	WFD	EA	3.69
2017	Devon	Plymouth Tamar	WFD	EA	0.52
2017	Devon	Exe Esturary	WFD	EA	0.46
2017	Devon	Salcombe	WFD	EA	0.31
2015	Devon	Dawlish	OSPAR	Devon Wildlife Trust	990.38
2004	Devon	Plymouth Sound	OSPAR	Devon Wildlife Trust; Natural England; Environment Agency; Devon Biodiversity Records Centre; Dorset Environmental Records Centre	6.84
2015	Devon	Torbay_Fishcombe	OSPAR	Devon Wildlife Trust	0.62
2015	Devon	Torbay_Brixam	OSPAR	Devon Wildlife Trust	0.51
2015	Devon	Torbay_Anstley	OSPAR	Devon Wildlife Trust	0.09
2015	Devon	Torbay_Torre Abbey	Individual contribution	CSI	104.37
2015	Devon	Torbay_Broadsands	Individual contribution	CSI	27.27
2015	Devon	Plymouth_Cawsand	Individual contribution	CSI	11.80
2014	Devon	Salcombe	Individual contribution	IFCA	6.70
2015	Devon	Plymouth_Tomb Rock	Individual contribution	CSI	6.58
2015	Devon	Plymouth_Cellars cove	Individual contribution	CSI	5.70
2015	Devon	Plymouth_Drakes Island	Individual contribution	CSI	4.26
2015	Devon	Torbay_Millstones	Individual contribution	CSI	4.05
2015	Devon	Plymouth_Red Cove	Individual contribution	CSI	3.76
2015	Devon	Torbay_Hopes Nose	Individual contribution	CSI	2.74
2015	Devon	Torbay_Broadsands	Individual contribution	CSI	2.07
2014	Devon	Salcombe	Individual contribution	IFCA	1.74
2015	Devon	Plymouth_Jennycliff	Individual contribution	CSI	1.47

2015	Devon	Plymouth_Red Cove	Individual contribution	CSI	1.18
2014	Devon	Salcombe	Individual contribution	IFCA	0.84
2015	Devon	Plymouth_Red Cove	Individual contribution	CSI	0.70
2015	Devon	Salcombe	Individual contribution	Natural England	0.58
2015	Devon	Plymouth_Firestone	Individual contribution	CSI	0.52
2014	Devon	Salcombe	Individual contribution	IFCA	0.43
2015	Devon	Plymouth_Firestone	Individual contribution	CSI	0.24
2015	Devon	Plymouth_Cawsand	Individual contribution	CSI	0.21
2015	Devon	Plymouth_Drakes Island	Individual contribution	CSI	0.15
2015	Devon	Salcombe	Individual contribution		0.07
2015	Devon	Torbay_Breakwater	Individual contribution	CSI	0.06
2015	Devon	Torbay_Breakwater	Individual contribution	CSI	0.04
2015	Devon	Plymouth_Jennycliff North	Individual contribution	CSI	0.00
2015	Dorset	Poole Harbour	WFD	EA	0.90
2015	Dorset	Poole Harbour	WFD	EA	0.46
2015	Dorset	Poole Harbour	WFD	EA	0.13
2015	Dorset	Poole Harbour	WFD	EA	0.08
2001	Dorset	Weymouth	Individual contribution	Dorset Environmental Records Centre	84.90
2004	Dorset	The Fleet	Individual contribution	Natural England	275.00
1999	Dorset	Studland Bay	Individual contribution	Dorset Environmental Records Centre	18.55
2002	Dorset	Poole Harbour	Individual contribution	Poole Harbour Commissioners	12.86
2002	Dorset	Studland Bay	Individual contribution	Poole Harbour Commissioners	12.74
2001	Dorset	Studland Bay	Individual contribution	Dorset Environmental Records Centre	7.74
2002	Dorset	Poole Harbour	Individual contribution	Poole Harbour Commissioners	4.25
2002	Dorset	Studland Bay	Individual contribution	Poole Harbour Commissioners	1.95
2004	Dorset	Swanage Pier	Individual contribution	Dorset Environmental Records Centre	1.77
1999	Dorset	Weymouth	Individual contribution	Dorset Environmental Records Centre	1.69
2002	Dorset	Studland Bay	Individual contribution	Poole Harbour Commissioners	1.37
1999	Dorset	Portland Harbour	Individual contribution	Dorset Environmental Records Centre	0.76
2002	Dorset	Poole Harbour	Individual contribution	Poole Harbour Commissioners	0.56
1999	Dorset	Portland Harbour	Individual contribution	Dorset Environmental Records Centre	0.51

1999	Dorset	Portland Harbour	Individual contribution	Dorset Environmental Records Centre	0.23
2002	Dorset	Poole Harbour	Individual contribution	Poole Harbour Commissioners	0.09
1999	Dorset	Portland Harbour	Individual contribution	Dorset Environmental Records Centre	0.09
2002	Dorset	Poole Harbour	Individual contribution	Poole Harbour Commissioners	0.03
2016	Hampshire and IoW	Pagham Harbour	WFD	EA	0.27
2016	Hampshire and IoW	Pagham Harbour	WFD	EA	0.03
2014	Hampshire and IoW	IoW_Ryde	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	84.56
2013	Hampshire and IoW	IoW_Osborne Bay	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	80.95
2013	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	70.15
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	68.94
2014	Hampshire and IoW	Calshot	Individual contribution	Hampshire Wildlife Trust; Southern IFCA	42.58
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	42.58
2010	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	38.16
2013	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	36.61
2011	Hampshire and IoW	IoW_Bouldnor	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	31.21
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	31.20
2014	Hampshire and IoW	Southampton Water	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	28.84
2010	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	27.76
2012	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	21.24
2009	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	17.99
2012	Hampshire and IoW	IoW_Yarmouth	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	9.63
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	8.75
2011	Hampshire and IoW	IoW_Totland	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	8.40
2012	Hampshire and IoW	IoW_Yarmouth	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	7.83
2012	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	7.60
2014	Hampshire and IoW	IoW_Seagrove	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	6.69
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	5.44
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	5.41
2008	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	4.67
2013	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	3.35
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	3.30
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	2.81

2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	2.20
2012	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	2.14
2014	Hampshire and IoW	IoW_Priory Bay	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	2.09
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	1.91
2013	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	1.69
2013	Hampshire and IoW	Bembridge	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	1.66
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	1.30
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	1.09
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.46
2008	Hampshire and IoW	IoW_Yarmouth	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.44
2013	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.34
2013	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.34
2008	Hampshire and IoW	IoW_Yarmouth	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.34
2008	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.32
2011	Hampshire and IoW	IoW_Bouldnor	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.31
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.18
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.16
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.15
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.12
2010	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.06
2013	Hampshire and IoW	Chichester Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.02
2008	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.02
2014	Hampshire and IoW	IoW_Seaview	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.00
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.00
2014	Hampshire and IoW	Portsmouth Harbour	Individual contribution	Southern IFCA, Hampshire Wildflie Trust	0.00
2017	Ireland	Tralee Bay	Paper	Wilkes, 2017	228.70
2017	Ireland	Cromane	Paper	Wilkes, 2017	182.50
2017	Ireland	Barrow Harbour	Paper	Wilkes, 2017	64.30
2017	Ireland	Ballysdare Estauary	Paper	Wilkes, 2017	41.72
2017	Ireland	Moy Estuary	Paper	Wilkes, 2017	23.93
2017	Ireland	Blacksod Bay	Paper	Wilkes, 2017	9.40
2017	Ireland	Drumcliffe Bay	Paper	Wilkes, 2017	9.13

2017	Ireland	Tramore Back Strand	Paper	Wilkes, 2017	8.20
2017	Ireland	Garavogue Estuary	Paper	Wilkes, 2017	6.41
2017	Ireland	Malahide Bay	Paper	Wilkes, 2017	4.80
2017	Ireland	Dublin Bay	Paper	Wilkes, 2017	1.83
2017	Ireland	Dungarvan Bay	Paper	Wilkes, 2017	1.23
2017	Ireland	Rogerstown Estuary	Paper	Wilkes, 2017	0.84
2017	Ireland	Killala Bay	Paper	Wilkes, 2017	0.64
2017	Ireland	Tolka Estuary	Paper	Wilkes, 2017	0.02
2017	Norfolk	Wells	WFD	EA	17.76
2017	Norfolk	Wells	WFD	EA	12.78
2017	Norfolk	Wells	WFD	EA	2.44
2017	Norfolk	Stiffkey	WFD	EA	2.11
2017	Norfolk	Wells	WFD	EA	1.83
2016	Norfolk	Burnham	WFD	EA	1.81
2017	Norfolk	Stiffkey	WFD	EA	0.85
2016	Norfolk	Burnham	WFD	EA	0.65
2017	Norfolk	Stiffkey	WFD	EA	0.62
2017	Norfolk	Stiffkey	WFD	EA	0.20
2017	Norfolk	Stiffkey	WFD	EA	0.19
2017	Norfolk	Stiffkey	WFD	EA	0.14
2016	Norfolk	Burnham	WFD	EA	0.13
2016	Norfolk	Burnham	WFD	EA	0.10
2017	Norfolk	Stiffkey	WFD	EA	0.04
2017	Norfolk	Stiffkey	WFD	EA	0.02
2017	Norfolk	Stiffkey	WFD	EA	0.01
2017	Norfolk	Stiffkey	WFD	EA	0.01
2017	Norfolk	Stiffkey	WFD	EA	0.01
2017	Norfolk	Stiffkey	WFD	EA	0.01
2017	Norfolk	Stiffkey	WFD	EA	0.00
2017	Norfolk	Stiffkey	WFD	EA	0.00
2017	Norfolk	Stiffkey	WFD	EA	0.00
2017	Norfolk	Stiffkey	WFD	EA	0.00

2016	Norfolk	Burnham	WFD	EA	0.00
2017	Norfolk	Stiffkey	WFD	EA	0.00
2004	North Wales	Anglesey	OSPAR	Countryside Council for Wales	72.92
2008	North Wales	Porth Dinllaen	OSPAR	Countryside Council for Wales	26.54
2013	North Wales	South of Caenarfon	OSPAR	Natural Resources Wales	23.31
2017	North Wales	Foryd Bay Nr Caernarfon	OSPAR	Natural Resources Wales	22.13
2012	North Wales	Bangor Flats	OSPAR	Natural Resources Wales	10.35
2003	North Wales	Anglesey	OSPAR	Scottish Natural Heritage	7.77
2008	North Wales	Porth Dinllaen	OSPAR	Countryside Council for Wales	5.23
2013	North Wales	Anglesey	OSPAR	Natural Resources Wales	2.23
2008	North Wales	Porth Dinllaen	OSPAR	Countryside Council for Wales	0.70
2008	North Wales	Caernarfon	OSPAR	Countryside Council for Wales	0.49
2017	Northern Ireland	Waterfoot	OSPAR	Wilkes, 2017	81.00
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	519.77
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	138.19
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	110.18
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	93.28
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	75.81
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	34.95
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	30.83
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	28.22
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	15.05

2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	13.26
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	12.54
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	10.26
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	5.73
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	5.19
2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	4.75
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	4.72
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	4.01
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	3.66
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	3.62
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	3.60
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	3.52
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	3.31
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	3.26
2012	Northern Ireland	Dundrum	OSPAR	Department of Environment Northern Ireland Marine Division	2.75
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	2.16

2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	2.01
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	1.72
2012	Northern Ireland	Dundrum	OSPAR	Department of Environment Northern Ireland Marine Division	1.21
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	1.10
2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	1.03
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	0.73
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	0.68
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	0.65
2012	Northern Ireland	Londonderry	OSPAR	Department of Environment Northern Ireland Marine Division	0.62
2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	0.53
2012	Northern Ireland	Dundrum	OSPAR	Department of Environment Northern Ireland Marine Division	0.52
2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	0.46
2012	Northern Ireland	Dundrum	OSPAR	Department of Environment Northern Ireland Marine Division	0.32
2012	Northern Ireland	Dundrum	OSPAR	Department of Environment Northern Ireland Marine Division	0.20
2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	0.19
2012	Northern Ireland	Dundrum	OSPAR	Department of Environment Northern Ireland Marine Division	0.12

2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	0.01
2012	Northern Ireland	Dundrum	OSPAR	Department of Environment Northern Ireland Marine Division	0.00
2012	Northern Ireland	Carlingford	OSPAR	Department of Environment Northern Ireland Marine Division	0.00
2012	Northern Ireland	Strangford Lough	OSPAR	Department of Environment Northern Ireland Marine Division	0.00
2016	Northumbria	Lindisfarne	WFD	EA	469.95
2016	Northumbria	Lindisfarne	WFD	EA	157.63
2016	Northumbria	Lindisfarne	WFD	EA	31.04
2016	Northumbria	Lindisfarne	WFD	EA	15.02
2016	Northumbria	Lindisfarne	WFD	EA	5.08
2016	Northumbria	Lindisfarne	WFD	EA	0.23
2016	Northumbria	Lindisfarne	WFD	EA	0.59
2016	Northumbria	Lindisfarne	WFD	EA	0.01
2011	Scilly Isles	Scilly Isles	Paper	Jackson, 2011	196.000
2008	Scottish Highlands	Cromarty Firth	Paper	Chapman	1200.00
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	58.89
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	55.63
2013	Scottish Highlands	Arran	Individual contribution	Scottish Natural Heritage	40.16
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	34.73
2013	Scottish Highlands	Arran	Individual contribution	Scottish Natural Heritage	25.71
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	17.13
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	14.60
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	14.38
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	14.31
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	13.97
2017	Scottish Highlands	Sound of Arisaig	Individual contribution	Scottish Natural Heritage	12.82
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	12.65
2013	Scottish Highlands	Arran	Individual contribution	Scottish Natural Heritage	11.11
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	8.41

2017	Scottish Highlands	Sound of Arisaig	Individual contribution	Scottish Natural Heritage	7.63
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	6.36
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	6.23
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	3.62
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	3.59
2013	Scottish Highlands	Arran	Individual contribution	Scottish Natural Heritage	3.58
2016	Scottish Highlands	Treshnish Isles	Individual contribution	Scottish Natural Heritage	3.05
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	3.02
2017	Scottish Highlands	Sound of Arisaig	Individual contribution	Scottish Natural Heritage	2.66
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	2.66
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	2.33
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	2.07
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	1.89
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	1.48
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	1.29
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	1.25
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	1.07
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	0.90
2017	Scottish Highlands	Sound of Arisaig	Individual contribution	Scottish Natural Heritage	0.79
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	0.61
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	0.55
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	0.51
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.35
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	0.13
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.10
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage.	0.10
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.09
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.09
2017	Scottish Highlands	Sound of Barra	Individual contribution	Scottish Natural Heritage	0.08
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.08
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.08
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.06

2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.05
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.04
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	0.04
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.03
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.03
2006	Scottish Highlands	Sunart	Individual contribution	Scottish Natural Heritage	0.03
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.02
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.02
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.02
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.02
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.01
2006	Scottish Highlands	Uist	Individual contribution	Scottish Natural Heritage	0.01
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00
2012	Scottish Highlands	Arran	Individual contribution	Community of Arran Seabed Trust (COAST)	0.00

2018	Scottish Highlands	Montrose Basin	Paper	Foster & Davidson	174.70
2018	East Scotland	Loch Ryan	Paper	Foster & Davidson	104.80
2018	Scottish Highlands	Eden Estuary	Paper	Foster & Davidson	55.88
2018	Scottish Highlands	Forth Estuary	Paper	Foster & Davidson	25.63
2018	Scottish Highlands	Forth Estuary	Paper	Foster & Davidson	21.85
2018	Scottish Highlands	Forth Estuary	Paper	Foster & Davidson	8.14
2011	East Scotland	Tay Estuary	Paper	Wilkie	3.00
2018	Scottish Highlands	Forth Estuary	Paper	Foster & Davidson	2.55
2018	Scottish Highlands	Forth Estuary	Paper	Foster & Davidson	2.03
2015	Scottish Highlands	Uist	OSPAR	Joint Nature Conservation Committee	55.17
2017	Scottish Highlands	Dornoch Firth and Morrich More	Individual contribution	Scottish Natural Heritage	63.55
2017	Scottish Highlands	Dornoch Firth and Morrich More	Individual contribution	Scottish Natural Heritage	37.80
2017	Scottish Highlands	Dornoch Firth and Morrich More	Individual contribution	Scottish Natural Heritage	10.38
2017	Scottish Highlands	Dornoch Firth and Morrich More	Individual contribution	Scottish Natural Heritage	5.25
2016	Essex	Leigh-on-Sea	WFD	EA	109.49
2016	Essex	Foulness/Maplin Sands	WFD	EA	40.19
2016	Essex	Leigh-on-Sea	WFD	EA	8.47
2015	Kent	Allhallows_Thames lower	WFD	EA	5.31
2016	Essex	Leigh-on-Sea	WFD	EA	2.89
2015	Essex	Thorpe Bay	WFD	EA	2.36
2016	Essex	BLACKWATER	WFD	EA	0.47
2014	Suffolk	Orwell	WFD	EA	0.44
2015	Essex	Thorpe Bay	WFD	EA	0.11
2014	Suffolk	Stour	WFD	EA	0.06
2015	Essex	Thorpe Bay	WFD	EA	0.05
2016	Essex	BLACKWATER	WFD	EA	0.00
2013	South Wales	St Lawrence Bay	OSPAR	Natural Resources Wales	6.48
2013	South Wales	Burry Point	OSPAR	Natural Resources Wales	1.51

2013	South Wales	Burry Point	OSPAR	Natural Resources Wales	1.26
2013	South Wales	Burry Point	OSPAR	Natural Resources Wales	1.16
2010	South Wales	Burry Point	OSPAR	Natural Resources Wales	0.72
2013	South Wales	Newport	OSPAR	Natural Resources Wales	0.27
2017	South Wales	Burry Point	Individual contribution	Natural Resources Wales	275.38
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	97.51
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	39.83
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	7.92
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	5.55
2016	South Wales	River Lougher nr Swansea	Individual contribution	Natural Resources Wales	4.73
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	4.18
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	2.56
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	2.28
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	2.04
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	1.44
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	1.31
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	1.10
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	0.95
2017	South Wales	Caldicot	Individual contribution	Natural Resources Wales	0.86
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	0.57
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	0.49
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	0.17
2017	South Wales	Milford Haven	Individual contribution	Natural Resources Wales	0.08
2017	West Wales	Caerdigan Bay	Individual contribution	Natural Resources Wales	90.26