Drivers of blue carbon dynamics in saltmarsh ecosystems

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Abstract

Political and economic interest in blue carbon as a natural climate solution has grown exponentially in recent years. In particular, the effective management of coastal wetlands, including saltmarshes, via conservation and restoration, is an attractive climate change mitigation tool. However, to become a scalable natural climate solution, a better understanding of the drivers governing blue carbon dynamics is required. Therefore, this thesis sought to identify drivers of blue carbon dynamics to guide effective coastal management. Work to investigate the influence of sea-level rise on carbon accumulation and stocks, and to determine major fine-scale drivers responsible for carbon stock variability between restored saltmarshes was conducted. Societal perceptions of blue carbon and coastal wetlands were also explored to address the growing need for trans-disciplinary approaches to ensure the longevity of blue carbon management.

Sea-level rise was a dominant control on saltmarsh carbon accumulation rates and drove spatial and temporal patterns across the 20th century. However, evidence from a regional study showed declines in carbon stock when saltmarshes were unable to keep pace with more recent rates of sea-level rise. Further research is required as global sea-level rise continues to accelerate. Work to identify fine-scale drivers of carbon stocks in restored saltmarshes found sites elevated above mean high water neap tidal levels had significantly higher carbon stocks, likely driven by suitable conditions for mature vegetation communities, including key plant species. Together, these findings can inform future restoration schemes that aim to maximise blue carbon stocks. Finally, a limited public awareness of blue carbon and a low connection with coastal wetlands was found, with implications for societal support of coastal wetland management. Targeted education and outreach campaigns, based on sociodemographic characteristics, are recommended.

Overall, this thesis has contributed new knowledge to progress blue carbon science and guide effective saltmarsh management for climate change mitigation.

“In the end, we will conserve only what we love; we will love only what we understand and we will understand only what we are taught”

Baba Dioum

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There are so many amazing friends of mine that I would like to thank for their continued support and the pride they take in telling people what I do. I'm so very lucky. I was initially nervous about starting my PhD as I was told it can be a very lonely experience, but I have found it to

be the complete opposite. I met one of my very best friends, **Lauren Rawlins**, as part of this journey. We have seemed to be at the same part of the rollercoaster throughout and I love how our friendship is all about building each other up. I have been just as excited about her achievements as I have my own. A special thanks also goes to **Sophie Williams** who was my ‘PhD buddy’ when I first started. Now she is a really good friend and I always look forward to our girls evenings with Lauren. To the rest of my wonderful PhD cohort, I have never met a more fun and supportive bunch: **Katey Valentine, George Day, Alex Burkitt, Ruochan Ma, Abby Mycroft** and **Nat Anderson**. I can’t wait for us all to let our hair down once this has finished.

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Author's Declaration

I declare that this thesis is presentation of original work completed by myself as a PhD student in the Environment & Geography Department at the University of York. This work has not previously been presented for an award at this, or any other, University. All sources are acknowledged as references. This research was funded by the Natural Environment Research Council (NERC) Adapting to the Challenges of Changing Environment (ACCE) Doctoral Training Partnership (funding code: NE/L002450/1).

Chapter 5 has been published in an international peer-reviewed journal. All co-authors have contributed to work associated with this chapter, and a statement of contributions has been included below to credit their contributions. The published manuscript from Chapter 5 has been re-worked to be presented in a consistent style within the thesis.

The following publication has arisen from this work:

- **McMahon, L.**, Ladd, C., Burden, A., Garrett, E., Redeker, K.R., Lawrence, P. and Gehrels, R. 2023. Maximising Blue Carbon Stocks through Saltmarsh Restoration. *Frontiers in Marine Science, 10*, p.557. doi: 10.3389/fmars.2023.1106607.

During my time as a PhD student, I was fortunate to be involved with research collaborations outside of my core thesis chapters. This was mostly with the Natural Environment Research Council funded Carbon Storage in Intertidal Environments (C-SIDE) project (grant NE/R010846/1). The following publications resulted from work related to my participation with fieldwork, data processing (identifying and classifying saltmarsh vegetation communities from citizen science photos), manuscript writing and editing for the C-SIDE project:

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- Smeaton, C., Burden, A., Ruranska, P., Ladd, C.J., Garbutt, A., Jones, L., **McMahon, L.,** Miller, L.C., Skov, M.W. and Austin, W.E. 2022. Using citizen science to estimate surficial soil Blue Carbon stocks in Great British saltmarshes. *Frontiers in Marine Science, 9*, p.1461. doi: 10.3389/fmars.2022.959459.

Signed:

Date: 28/04/2023

Statement of Contributions

Contributions from co-authors and the supervisory team are outlined below for the four data chapters:

- **Chapter 3:** This chapter has been prepared ready for manuscript submission. The intended journal for submission is *Global and Planetary Change*. Idea conceptualisation and experimental design was carried out by myself and Roland Gehrels. Previous work conducted by Roland Gehrels (University of York) provided materials for lab work. Lab work, data analysis and lead writing was carried out by myself. Ed Garrett (University of York) and Sophie Williams (Durham University) produced age-depth models from which Fiona Hibbert (University of York) was able to calculate rates of relative sea-level rise for each site. Roland Gehrels, Ed Garrett, Fiona Hibbert and Sophie Williams contributed to manuscript editing and revisions. Ed Garret and Fiona Hibbert wrote the methodology section on age-depth modelling and rates of relative sea-level rise, respectively.

- **Chapter 4:** Initial work conducted by Mark Wood (Massachusetts Department of Environmental Protection), Jody Goodman (University of Plymouth) and Roland Gehrels (University of York) between 1986 and 2003 provided the scientific basis for this chapter to build upon. Idea conceptualisation and experimental design was carried out by myself and Roland Gehrels. Fieldwork was conducted by myself, Roland Gehrels and Mark Wood. Lab work, data analysis and chapter writing was conducted by myself. Due to COVID-19 delays to lab work, an MSc dissertation student (Rhys Grabham) also contributed to lab work for this chapter, supported by myself. Roland Gehrels and Ed Garrett (University of York) provided comments and revisions to the writing of this chapter.

- **Chapter 5:** This chapter has been published (McMahon *et al.,* 2023; see above for full reference). Idea conceptualisation, experimental design, lab work, data analysis and lead writing was carried out by myself. Fieldwork was completed by myself and Cai Ladd. All co-authors: Cai Ladd (University of Glasgow), Annette Burden (Centre for Ecology and Hydrology), Ed Garrett (University of York), Kelly Redeker (University of York), Peter Lawrence (University of Cumbria), and Roland Gehrels (University of York) contributed to discussions on research approach, and manuscript editing and revisions.

- **Chapter 6:** This chapter will form a manuscript to be submitted for publication in the journal *Ocean and Coastal Management*. Idea conceptualisation, survey design, data analysis and lead writing was carried out by myself. Emma McKinley (Cardiff University) and Mark Atkinson (Department for Environment, Food & Rural Affairs) contributed to survey design, incorporation of the survey into the wider Ocean Literacy survey and the editing and revision of multiple versions of the chapter. Ed Garrett (University of York), Roland Gehrels (University of York) and Annette Burden (Centre for Ecology & Hydrology) contributed to edits of the final chapter version.

Chapter 1 of 7

Introduction

1.1 Motivation

The climate and biodiversity crises are arguably the greatest challenges facing societies around the world today. There is a growing realisation that these crises are interlinked, with calls for solutions to treat both crises as interlinked in order to effectively address them (Seddon *et al.,* 2021; WWF, 2022). The sustainable management and use of nature for tackling social and environmental challenges, hereafter referred to as nature-based solutions (NbS), has gained popularity in recent years as an effective approach to ameliorate the twin crises of climate change and biodiversity loss through the protection, conservation, restoration and management of natural ecosystems (Osaka *et al.,* 2021; Seddon *et al.,* 2021).

Numerous approaches fall under the umbrella term of NbS including natural climate solutions, the current discourse of which is relatively new and focuses primarily on climate change mitigation (Griscom *et al.,* 2017; Osaka *et al.,* 2021; Seddon *et al.,* 2021). Natural climate solutions (NCS) relate to conservation and management actions that simultaneously reduce greenhouse gas emissions, including CO_2, from natural ecosystems while increasing their carbon storage potential (Griscom *et al.,* 2017). NCS, representing around 30 % of the cost-effective mitigation of climate change, can be a powerful tool in enabling nations to deliver on the Paris Climate Agreement to stabilise global warming to 1.5 °C above pre-industrial levels, if adopted alongside measures to reduce CO_2 emissions (Griscom *et al.,* 2017). Traditionally, terrestrial forest ecosystems have been the focal point of NCS and early efforts to mitigate climate change through biosequestration, with the role of ocean and coastal ecosystems largely neglected (Nellemann *et al.,* 2009; Macreadie *et al.,* 2021).

Although several earlier studies point to the role of marine vegetation in carbon capture and burial (e.g., Duarte *et al.,* 2005), the term and wider recognition of 'blue carbon', referring to the organic carbon captured and stored by marine and coastal habitats, was articulated by an United Nations Educational, Scientific and Cultural Organization (UNESCO) report in 2009 (Nellemann *et al.,* 2009). In this report coastal wetlands, including saltmarshes, mangroves, and seagrasses, were identified as the main habitats responsible for the capture and storage of blue carbon (Nellemann *et al.,* 2009). Other influential papers built upon this report to demonstrate the carbon sequestration potential of coastal wetlands; they occupy around 0.2 % of the ocean's surface area but contribute 50 % of stored carbon in marine sediments with a sequestration capacity orders of magnitude above that of terrestrial forests per unit area (Mcleod *et al.,* 2011; Duarte *et al.,* 2013). Since then, scientific interest and research efforts in blue carbon, demonstrated by the number of relevant publications, has grown exponentially, underpinned by a desire to better understand the role of coastal wetland conservation and restoration in climate change mitigation (Fig. 1.1; Duarte de Paula Costa & Macreadie, 2022).

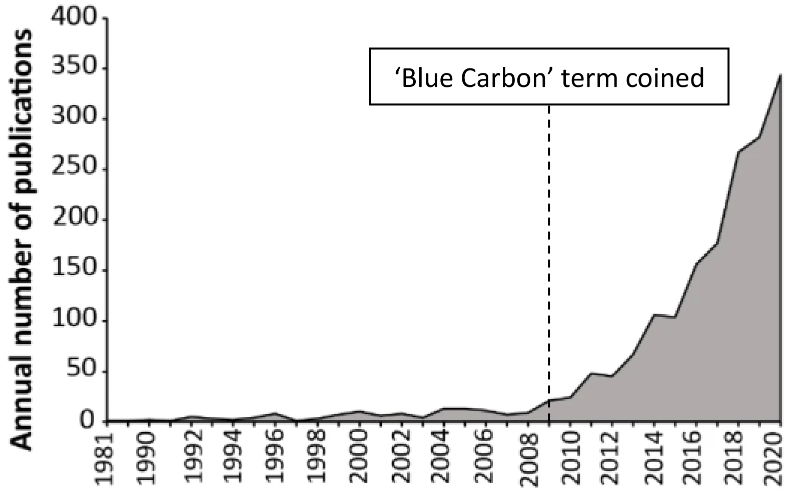


Figure 1.1. Annual number of scientific publications on blue carbon science, between 1981 and 2020. The dashed line shows the year the term 'blue carbon' was coined (Nellemann *et al.,* 2009), before which the identification of blue carbon publications was more challenging. Overall, an annual percentage growth in publications showed a rate greater than 20 % yr^{-1} during this time period. Image adapted Duarte de Paula Costa & Macreadie (2022).

A wealth of research over the past decade (Fig. 1.1) has demonstrated the globally relevant role of coastal wetlands in climate change mitigation (Macreadie *et al.,* 2021), of which saltmarshes are considered the most widespread and important blue carbon habitat outside of the tropics (Mcleod *et al.,* 2011). Blue carbon habitats are highly efficient carbon sinks characterised by high productivity (Tang *et al.,* 2018). The production of vast amounts of autochthonous carbon via photosynthesis is complemented by a high trapping capacity of tidally imported allochthonous carbon derived from external ecosystems (Duarte *et al.,* 2013). Furthermore, the depositional environment of blue carbon habitats at the interface between land and ocean enables them to accrete sediment and carbon without reaching saturation (Mcleod *et al.,* 2011; Macreadie *et al.,* 2017). The high climate change mitigation potential of coastal wetlands thus results from their capacity to produce, capture and store carbon over climatically relevant timescales up to millennia (Macreadie *et al.,* 2021). Yet, coastal wetlands are among the most threatened habitats in the world and have experienced large-scale declines globally (Lovelock *et al.,* 2017). For example, Duarte *et al.* (2008) estimated saltmarsh habitat loss to occur at rates of 1 – 2 % annually, with more recent estimates showing a global loss rate of 0.28 % annually (Campbell *et al.,* 2022). Such degradation and loss threatens the potential release of ancient stored carbon and the conversion of these habitats from carbon sinks into carbon sources (Pendleton *et al.,* 2012; Macreadie *et al.,* 2013; Howard *et al.,* 2017).

The effective management of blue carbon habitats can therefore enhance carbon production and capture while avoiding CO_2 emissions related to habitat degradation and loss (Duarte *et al.,* 2013; Macreadie *et al.,* 2017b; Kelleway *et al.,* 2020). Core to the concept of blue carbon strategies is the effective management of coastal wetlands, via conservation and restoration, as an approach to mitigate against climate change (Duarte *et al.,* 2013). Coastal restoration and conservation efforts have increased in recent years, complemented by the United Nations (UN) Decade of Ocean Science for Sustainable Development and Decade on Ecosystem Restoration, both of which run between 2021 and 2030 (Waltham *et al.,* 2020). Given that only around 1.5 % of blue carbon habitats are included within the boundaries of marine protected areas (MPA) globally, there is abundant opportunity for their conservation and restoration (Zhao *et al.,* 2020; Macreadie *et al.,* 2021). Indeed, recent evidence and advice for UK policy has been the inclusion of blue carbon habitats within marine protected areas to improve the climate resilience of the marine environment (UK Government, 2023). This is reflective of the growing political interest in blue carbon strategies, with global and national efforts to include them in climate policies, evidenced by the 46 countries which, as of 2021, mention coastal and marine ecosystems as mitigation solutions within their Nationally Determined Contributions to the Paris Agreement (Lecerf *et al.,* 2021).

A major attraction of effectively managing blue carbon habitats is also rooted in the potential to mitigate climate change while achieving a multitude of co-benefits from coastal protection to biodiversity enhancement and the maintenance of water quality (Nellemann *et al.,* 2009; Mcleod *et al.,* 2011; Duarte *et al.,* 2013; Macreadie *et al.,* 2019; Friess *et al.,* 2020). The prospect of aligning climate and other goals increases the appeal of, and motivation behind, rapid deployment of blue carbon strategies, which also support a diversity of sustainable development goals (Anderson *et al.,* 2019; Macreadie *et al.,* 2021).

The concept of blue carbon has thus captivated political and scientific dialogues in recent years. In particular, the conservation and restoration of coastal wetlands, including saltmarshes, plays a prominent role in global and national discussions on natural climate solutions. As a result of global interest, the scientific community has been motivated to address knowledge gaps and uncertainties in blue carbon science required to inform policy and management actions (Macreadie *et al.,* 2019). This thesis is a culmination of interdisciplinary chapters that each contribute to the wider blue carbon knowledge base, and help to support coastal wetland management decisions, notably for saltmarsh ecosystems.

1.2 Thesis aims and objectives

The overarching aim of this thesis is to advance knowledge on the role of saltmarsh ecosystems in climate change mitigation, in order to guide effective coastal management. The subsidiary aims are to: (i) identify environmental characteristics that drive spatial and temporal variability in saltmarsh blue carbon stocks and accumulation rates, and; (ii) understand current levels of public awareness and perceptions of blue carbon and coastal wetlands. The inclusion of a social science chapter reflects the growing need for trans-disciplinary approaches to ensure the longevity of blue carbon management.

This thesis therefore contributes interdisciplinary knowledge towards questions in blue carbon science through the successful completion of the following objectives:

1) **To ascertain whether rates of relative sea-level rise, a global driver of saltmarsh stability and function, are a dominant control on blue carbon accumulation rates in saltmarshes.** Under relative sea-level rise, space available for mineral and organic matter accumulation, known as accommodation space, is created. The hypothesis that '*the creation of vertical accommodation space, as a direct result of relative sea-level rise, enhances blue carbon accumulation rates'* will be tested on a global scale to account for known spatial and temporal variability in sea-level change. High-resolution records of relative sea-level rise, obtained using proxy reconstructions, will be analysed against carbon accumulation rate over decadal-centennial and multi-centennial timescales.

2) **To investigate whether saltmarsh resilience to relative sea-level rise influences blue carbon stocks.** This objective will be tested on a spatial scale between sites in Maine, USA, and on a temporal scale, between two time periods representing more rapid and stable sea-level rise. Saltmarsh resilience, or accretionary balance, will be measured by comparing rates of vertical accretion with rates of relative sea-level rise and analysed against carbon stock across both time periods at each site.

3) **To determine the major environmental drivers of blue carbon stocks for restored saltmarshes.** To obtain an understanding of major environmental drivers, surficial carbon stock in restored saltmarshes will be analysed against site-specific

environmental characteristics. The observed variability in carbon stock between restored sites, explained by major drivers, will then be used to make recommendations for future restoration schemes. The Blackwater Estuary, UK, will be used as a case study.

4) **To quantify public perceptions, understanding and awareness of blue carbon and associated habitats (saltmarshes and seagrasses), while identifying social and demographic backgrounds that influence perceptions.** This objective will report on the first assessment of public perceptions, awareness, and knowledge of blue carbon and related habitats, using Great Britain as a case study. Data collected via a large-scale online survey will also be analysed against respondents' social and demographic background to provide an insight into drivers of perception.

1.3 Thesis structure

This thesis has a hybrid structure with four main data chapters. Three data chapters have either been submitted for publication, or have been prepared for submission (see Author's Declaration and Contributions section), and one data chapter is currently presented in a traditional thesis chapter format. As such, there is bound to be a degree of repetition across chapter introduction and methodology sections. The broad-scale approach of this thesis combines research from a global, national, regional and local perspective. Work performed to address the previously stated objectives is described in the main data chapters, supported by an introductory literature review and a conclusion which summarises the main findings from each chapter and places the results in a broader context:

Chapter 2 is a literature review which presents a summary of current scientific understanding of blue carbon including environmental drivers for spatial and temporal variation, and the position of blue carbon within international policy discussions focused on nature-based solutions to climate change. This chapter highlights knowledge gaps in current blue carbon science.

Chapter 3 investigates whether relative sea-level rise can enhance blue carbon accumulation in saltmarshes, testing the theory of vertical accommodation space. To determine whether this relationship is applicable on a global scale, seven saltmarsh sites, spanning three continents (North America, Europe, and Australasia) and five countries, subject to differing rates of relative sea-level rise, were studied.

Chapter 4 investigates whether blue carbon stocks are influenced by sea-level rise and saltmarsh accretionary balance, by comparing two distinct time periods representing lower and higher rates of relative sea-level rise. This chapter also assesses the effectiveness of marker horizons in saltmarsh vertical accretion studies in excess of 30 years.

Chapter 5 is based on a paper published in the journal *Frontiers in Marine Science, Marine Ecosystem Ecology* section (McMahon *et al.,* 2023) and identifies major environmental drivers responsible for the variability of blue carbon stocks in restored saltmarshes. Knowledge of such drivers is required to maximise the climate change mitigation potential of saltmarsh restoration schemes, and this chapter focuses on saltmarshes restored by managed realignment. Adjacent natural saltmarshes were used as references.

Chapter 6 examines public perceptions, knowledge and awareness of blue carbon in Great Britain. The data collection survey associated with this chapter was incorporated into the wider Ocean Literacy survey and campaign commissioned by the Department for Environment, Food & Rural Affairs (Defra). The manuscript of this chapter cannot be submitted for publication until Defra's dataset has been made publicly available.

Chapter 7 summarises the main findings from the preceding chapters and makes recommendations for future blue carbon management. Opportunities for future work are also highlighted.

Chapter 2 of 7

Literature review

2.1 The climate crisis and natural climate solutions

This year has seen the publication of another major global synthesis report by the Intergovernmental Panel on Climate Change (IPCC), detailing the far-reaching impacts of anthropogenic-driven climate change (IPCC, 2023). Global evidence of the climate crisis is unequivocal – in the marine environment, warm-water corals, for example, are projected to see losses between 70 – 90 %, even with global warming limited to 1.5 °C (Hoegh-Guldberg *et al.,* 2018). Currently, average global temperature has increased by at least 1.1 °C since pre-industrial times (IPCC, 2021). Furthermore, the implementation of Nationally Determined Contributions (NDCs) – the country-level plans to address climate change as part of the Paris Agreement – announced in 2021 are insufficient for ambitions to maintain average global warming under 1.5 °C during the 21st century (IPCC, 2022). The escalating risk of climate change impacts, even under low emission trajectories, necessitates a dramatic scaling up of global mitigation measures (Magnan *et al.,* 2021).

Climate change mitigation measures require natural climate solutions to be adopted alongside reductions in CO_2 emissions (Griscom *et al.,* 2017). The simultaneous adoption of both measures would address the time lags that would occur if emission reductions were adopted alone and enable the removal of excess atmospheric CO_2 (Macreadie *et al.,* 2021; Magnan *et al.,* 2021; O'Leary *et al.,* 2022). Natural climate solutions form a branch of nature-based solutions that primarily focus on the mitigation of climate change via the effective management of natural ecosystems (Griscom *et al.,* 2017; Osaka *et al.,* 2021; Seddon *et al.,* 2021). The significant role of the ocean in climate change mitigation has emerged in recent years, particularly in relation to blue carbon, that is the carbon captured and stored by coastal wetlands, notably saltmarshes, mangrove forests and seagrass meadows, as a natural climate solution (Gallo *et al.,* 2017; Macreadie *et al.,* 2021; Jacquemont *et al.,* 2022).

In addition to climate change mitigation, the appeal of natural climate solutions involves a multitude of other co-benefits, a perk well-observed from coastal wetland management (Barbier *et al.,* 2011; Friess *et al.,* 2020). However, the role of coastal wetlands in climate change mitigation is less well understood than the provision of other ecosystem services, such as coastal protection. For example, the recent IPCC synthesis report stated medium confidence in the capacity of coastal wetlands to reduce emissions and increase carbon capture and storage through effective coastal management (IPCC, 2023). In comparison, very high confidence was awarded to the protection coastal wetlands provide against coastal erosion and flooding (IPCC, 2023). Confidence levels increase as scientific evidence becomes more robust and higher levels of scientific agreement are achieved (Kause *et al.,* 2022). Given the relatively recent evolution of blue carbon science, highlighted in **Chapter 1** (Duarte de Paula Costa & Macreadie, 2022), there is an urgent need to refine knowledge on the role of coastal wetlands in climate change mitigation (Macreadie *et al.,* 2019).

2.2 Blue carbon: the role of coastal wetlands in mitigating climate change

The recent recognition of blue carbon, a term coined in 2009, stems from research that highlights the disproportionate capacity of coastal wetlands to sequester carbon and function as highly efficient carbon sinks (Nellemann *et al.,* 2009; McLeod *et al.*, 2011). For example, although coastal wetlands only occupy 0.2 % of the ocean's surface area, these ecosystems contribute 50 % of carbon burial in marine sediments (Duarte *et al.,* 2005; Duarte *et al.*, 2013).

Coastal wetlands have a much smaller aboveground biomass than terrestrial forests, yet the carbon burial and storage capacity of coastal wetlands far exceeds that of terrestrial forests, with estimates up to 30 – 50 fold greater per unit area (McLeod *et al.*, 2011; Duarte *et al.*, 2013; Tang *et al.*, 2018; Fig. 2.1). Blue carbon is predominantly stored in the sediments of coastal wetlands rather than biomass, the latter of which is the predominant carbon store for terrestrial forests (Taillardat *et al.*, 2018). This distinction likely explains the high carbon sink capacity of coastal wetlands which will be covered in further detail within this chapter. As a result, saltmarshes, mangrove forests and seagrass meadows represent a globally significant climate regulation service (Duarte *et al.*, 2013).

While coastal wetlands function as one of the most efficient carbon sinks per unit area, these ecosystems occupy a narrow fringe along global coastlines and have a combined global extent equivalent to less than 3% than that of terrestrial forests (Barbier *et al.*, 2011; Duarte *et al.*, 2013; Saintilan *et al.*, 2014). The restricted spatial coverage of saltmarshes, mangrove forests and seagrass meadows limits their potential for climate mitigation at the global scale (Taillardat *et al.*, 2018), and calls for the urgent conservation of these ecosystems.

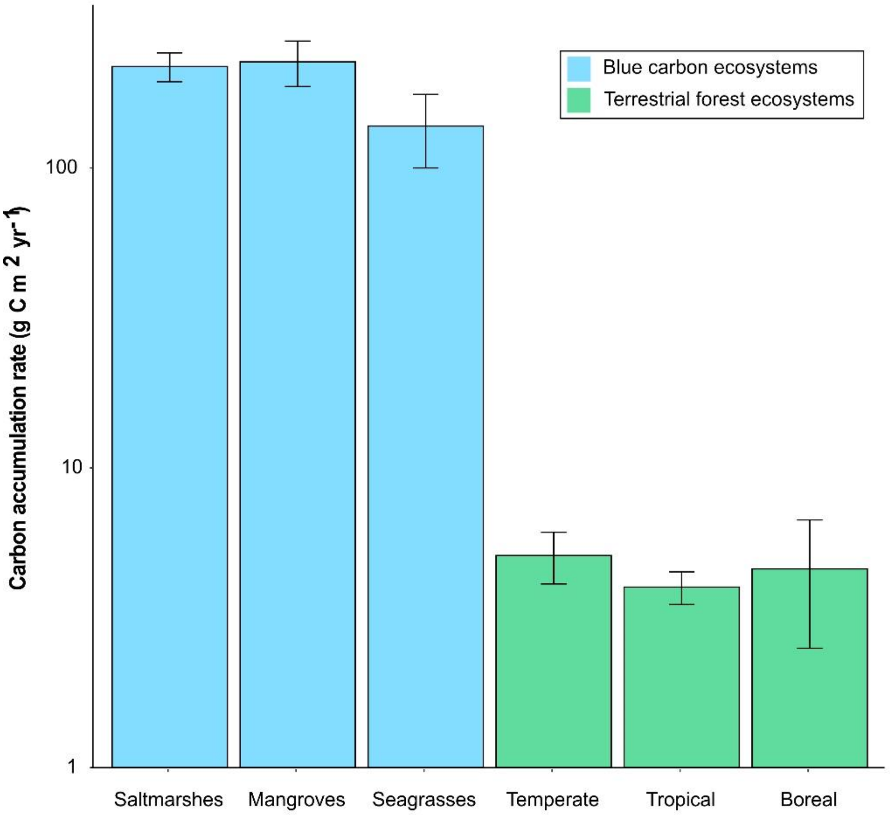


Fig. 2.1. Mean carbon burial rate (g C m^{-2} yr^{-1}) ± standard error for blue carbon (blue) and terrestrial forest ecosystems (green). Note the logarithmic scale of the Y-axis. Data extracted from Mcleod *et al.* (2011).

2.2.1 Blue carbon cycles

Coastal wetlands are net autotrophic ecosystems, meaning they produce more energy than they utilise and are characterised by high productivity which generates the capacity of coastal wetlands to produce excess organic carbon and act as blue carbon sinks (Mcleod *et al.,* 2011; Hopkinson *et al.,* 2012; Duarte *et al.,* 2013; Tang *et al.,* 2018). Over half of organic matter production in saltmarsh plants occurs in roots and rhizomes that are buried within sediment (Duarte *et al.*, 2013). The capacity of coastal wetlands to act as blue carbon sinks is intensified by their ability to trap particles (Kelleway *et al.*, 2016a). Trapped particles include autochthonous carbon, derived from within the ecosystem (e.g., plant litter) and allochthonous carbon, derived from external sources, such as tidally imported suspended sediment (Xiong *et al.,* 2018).

Carbon stocks in saltmarsh ecosystems are known to be significantly greater than in adjacent un-vegetated mudflats (Liao *et al.*, 2007). This is due to significant carbon inputs to sediment stocks from vegetation in the form of aboveground litter (i.e. leaves and propagules) and belowground root turnover (Ouyang & Lee, 2014; Radabaugh *et al.*, 2017; Radabaugh *et al.*, 2018). Saltmarsh vegetation is also fundamental for the accumulation of laterally imported, allochthonous carbon from suspended sediment supply brought in by the tides (Duarte *et al.*, 2013; Saintilan *et al.*, 2013). Firstly, dense vegetation attenuates waves and slows tidal flow which facilitates the settling of sediment, often rich in organic matter (Morgan *et al.,* 2009; Mudd *et al.,* 2010). Carbon buried as a result of sedimentation can either form labile, easily broken-down, or refractory, decay-resistant, carbon pools (Unger *et al.*, 2016). Total decay of labile carbon is predicted to occur on a timescale between several years to decades, although under anaerobic conditions, labile carbon may be preserved for longer time periods (Mudd *et al.*, 2009; Unger *et al.*, 2016). Refractory carbon, leftover after the break-down of labile fractions, forms a large proportion of buried carbon and is incorporated into long-term sediment stocks (Dahl *et al.*, 2016). Secondly, the structural complexity of saltmarsh vegetation enables effective trapping of suspended sediment and associated organic carbon (McLeod *et al.*, 2011).

As a result of the hydrodynamic environment in which saltmarshes reside and the trapping efficiency of vegetation, saltmarshes are able to retain sediment, and associated carbon, delivered on the tides from neighbouring ecosystems (marine and terrestrial) (Howard *et al.*, 2014; Kelleway *et al.*, 2016a). Saltmarshes are thus carbon sinks for a much greater area than their ecosystem boundaries alone, through the accumulation of carbon from internal and external sources (McLeod *et al.*, 2011). Multiple studies have demonstrated the ability of saltmarshes to function as a sink for carbon derived outside of the ecosystem using stable carbon-isotopic analysis, a method intended to distinguish between autochthonous and allochthonous carbon (Kelleway *et al.*, 2016a). Although some research suggests autochthonous inputs to be the greatest contributor to blue carbon stocks (Saintilan *et al.*, 2013), other research is more speculative as to whether sediment carbon stocks are derived primarily from autochthonous or allochthonous sources (Xiong *et al.,* 2018). It is clear, however, that saltmarsh vegetation enhances blue carbon accumulation through both autochthonous and allochthonous sources (Xiong *et al.,* 2018).

In coastal wetlands, captured blue carbon can be stored in one of three ways: (1) in living biomass, both aboveground in leaves and stems, and belowground in roots; (2) in non-living biomass in the form of plant litter; and, (3) in underlying sediments (McLeod *et al.*, 2011). Blue carbon within living and non-living biomass is stored over shorter timescales, such as years, or decades (Duarte *et al.,* 2005; McLeod *et al.*, 2011) in comparison to blue carbon within underlying sediments which can be stored over longer, millennial timescales (Duarte *et al.,* 2005). It is speculated that between 50% and 90% of carbon within coastal wetlands is stored in the sediment, with saltmarshes at the upper end of the estimate (Nellemann *et al.,* 2009; Pendleton *et al.*, 2012). Belowground blue carbon stocks remain the least studied of all carbon pools (Howard *et al.,* 2017), and this thesis will focus on belowground blue carbon dynamics in saltmarshes.

The anoxic, or at least low-oxygen, condition of coastal wetland sediments, as a result of regular inundation, inhibits microbial action and slows decay (Kristensen *et al.*, 2008), which favours long-term, relatively stable carbon stores (Howard *et al.*, 2017). The waterlogged condition of the sediment also reduces the risk of sudden loss associated with forest fires, which are capable of decimating carbon stocks stored within biomass in terrestrial forests, and account for significant global carbon dioxide emissions (Pearson *et al.,* 2017; Taillardat *et al.*, 2018). Carbon deposits in the sediment are somewhat protected from erosion due to the entangled network of belowground roots and the dense aboveground canopy of saltmarsh vegetation (Duarte *et al.*, 2013). However, saltmarsh erosion is an unavoidable process, exacerbated by anthropogenic disturbance (e.g. climate-driven sea-level rise), thus sediment carbon stocks are never fully protected from erosion.

Highly effective blue carbon accumulation rates are therefore attributed to strong photosynthetic rates (Tang *et al.*, 2018) and the depositional environment of the saltmarsh (Kelleway *et al.*, 2016a). Coastal wetlands also function as highly efficient blue carbon sinks due to the low decomposition rates associated with anoxic sediment, where blue carbon predominantly resides (Tang *et al.*, 2018). Therefore, the disproportionate accumulation of blue carbon in coastal wetlands (Fig. 2.1.) is owed to high productivity, sediment conditions and the delivery of allochthonous carbon inputs, in the form of suspended sediment, from tidal inundation (Duarte *et al.,* 2005). Blue carbon accumulation is thus the net result of multiple complex processes (Hopkinson *et al.,* 2012; Tang *et al.*, 2018), making stocks and sequestration rates vulnerable to environmental change.

2.2.2 Blue carbon degradation

The valuable nature of blue carbon alongside the provision of multiple co-benefits and the anthropogenic threats facing coastal wetlands, including saltmarshes, makes it imperative to improve scientific understanding of blue carbon sinks and the potential impact of future environmental change (McLeod *et al.*, 2011).

The fate of carbon inputs has already been discussed in relation to storage within the sediment, which acts to counter the atmospheric CO_2 pool (Macreadie *et al.*, 2017b). However, there are multiple pathways for the carbon accumulated in coastal wetland sediments. Arguably the most important fate of accumulated carbon is break-down through microbial mineralisation (Couwenberg, 2010; Macreadie *et al.*, 2017b). Carbon mineralisation is an ongoing process, often inhibited in waterlogged, anaerobic sediments of coastal wetlands (Fenner *et al.,* 2005). Sediment exposure to oxygen as a result of saltmarsh disturbance (e.g. seaward erosion), enhances microbial activity and subsequent processes that oxidise carbon and release CO_2, which in some cases can lead to major emissions (Howard *et al.*, 2017). Carbon mineralisation is of widespread concern due to the ongoing anthropogenic degradation of coastal wetlands which threatens a shift in ecosystem functioning from net carbon sinks to sources (McLeod *et al.*, 2011; Pendleton *et al.*, 2012). Studies referring to habitat degradation and destruction tend to consider CO_2 release as the outcome of lost blue carbon, but other fates include consumption by grazers, and exportation to neighbouring ecosystems, or deeper water, where carbon may be reburied within sediment (Macreadie *et al.,* 2013).

Damage to the vegetation communities of coastal wetlands can also affect blue carbon accumulation rates and sink capacity in two main ways. Firstly, removal of above-ground vegetation by livestock grazing reduces carbon-trapping efficiency and consequently limits laterally-imported allochthonous carbon inputs (Duarte *et al.*, 2013; Macreadie *et al.,* 2013). However, losses of carbon inputs owed to the removal of above-ground vegetation is speculated to have negligible impact as plant growth rate is rapid in coastal wetlands (Macreadie *et al.*, 2014). Secondly, reductions in plant biomass, both above and below-ground, diminishes photosynthetic capacity and therefore limits autochthonous contributions to carbon accumulation (Macreadie *et al.*, 2017b). Moreover, sedimentary carbon stocks risk subjection to microbial mineralisation following removal of below-ground plant material (Macreadie *et al.*, 2014).

As mentioned in **Chapter 1**, the effective management of blue carbon habitats, including saltmarshes, is a recognised solution to enhance carbon production and capture while avoiding CO_2 emissions related to habitat degradation and loss (Duarte *et al.,* 2013; Macreadie *et al.,* 2017b; Kelleway *et al.,* 2020). Therefore, the effective management of saltmarshes via conservation and restoration is at the core of blue carbon strategies and is necessary to enable saltmarshes to fulfil their role in climate change mitigation.

2.2.3 Saltmarsh blue carbon discrepancies

Global estimates of saltmarsh blue carbon are subject to large uncertainties. For example, Ouyang & Lee (2014) reported the global average carbon accumulation rate (CAR) for saltmarsh sediments as 245 ± 26 g C m^{-2} yr^{-1}, based on a compilation of 143 sites across the world. A more recent study estimated global saltmarsh CAR at a lower rate of 168 ± 7 g C m^{-2} yr^{-1}, which the authors attributed to the inclusion of an increased dataset (Wang *et al.,* 2021). Indeed, earlier studies estimating global saltmarsh CAR have been criticised for focusing on a limited number of saltmarshes with insufficient data collected from regions south as South America and Africa (Chastain *et al.,* 2018). Estimates of CAR also require accurate dating methods to be meaningful. Furthermore, global assessments of carbon stocks and CAR are hindered by uncertainties in global saltmarsh extent that are complicated by ongoing saltmarsh decline (Kelleway *et al.,* 2016b).

The methodology used for blue carbon measurements are inconsistent among researchers and can make for difficult comparisons between studies and habitats (Howard *et al.*, 2017). The *International Blue Carbon Initiative* released a manual for measuring, assessing and analysing coastal blue carbon with a goal to standardise protocols for more robust blue carbon data (Howard *et al.,* 2014). In addition to inconsistent methodology, blue carbon terminology is often used interchangeably within scientific literature and can result in a misinterpretation of reported data. For example, carbon sequestration refers to the photosynthetic long-term removal of atmosphere CO_2 whereas carbon accumulation encompasses the magnitude of carbon inputs, acknowledging that allochthonous inputs that do not derive from in-situ sequestration may be a contributor to observed rates (Van de Broek *et al.,* 2018). Blue carbon stocks and rates of carbon accumulation also tend to not account for carbon lost through decomposition and coastal erosion and therefore only represent apparent accumulation rates – this is more challenging to address and requires further consideration.

The methods for calculating carbon stocks in this thesis will follow the protocol outlined by the *International Blue Carbon Initiative* manual. A consistent approach in measuring and reporting blue carbon dynamics is vital for its inclusion in climate change mitigation policy and for the effective management of coastal wetlands for blue carbon.

2.2.4 Blue carbon relevance to policy

A comprehensive overview of blue carbon policy goes beyond the scope of this thesis, which focuses on the drivers of blue carbon dynamics, but key policy areas will be highlighted. Within the past decade, there has been an increase in global and national efforts to include coastal wetlands in climate policies due to their capacity for climate change mitigation (Sutton-Grier & Moore, 2016).

In 2013, the IPCC released the Wetlands Supplement to the 2006 guidelines for national greenhouse gas inventories, acknowledging the role of blue carbon and encouraging the inclusion of coastal wetland management within national inventories (Hiraishi *et al.,* 2014). Prior reports excluded coastal wetlands as a direct result of the insufficient scientific knowledge of these ecosystems (Moomaw *et al.,* 2018). As of 2022, marine and coastal habitats are not included in the UK's greenhouse gas inventory, despite a basis for the inclusion of saltmarshes and seagrass meadows (Climate Change Committee, 2022). The restoration of these habitats could help the UK reach the legally binding target of Net Zero greenhouse gas emissions by 2050 (Climate Change Committee, 2022). As part of the IPCC's most recent sixth assessment cycle, there are numerous mentions on the contribution of blue carbon habitat management to climate change mitigation, particularly through the Special Report on the Ocean and Cryosphere in a Changing Climate, alongside the Special Report on Global Warming of 1.5° C (IPCC, 2018; IPCC, 2019).

As of 2021, 71 countries mention marine and coastal nature-based solutions in their NDCs to the Paris Agreement; 46 of those countries included marine and coastal NbS as climate change mitigation measures (Lecerf, 2021). The next round of updated NDCs are due to be submitted by parties in 2025.

Carbon financing mechanisms also exist for blue carbon under the voluntary carbon market, although such schemes are in their infancy compared to the well-established carbon offset schemes for terrestrial forests (Vanderklift *et al.,* 2019; Kelleway *et al.,* 2020; Sapkota & White, 2020). Voluntary carbon markets represent an increasingly popular method, predominantly for private companies, to reduce carbon footprints and demonstrate social responsibility (Mack *et al.,* 2015). Through the purchase of carbon credits, finance is made readily available for the conservation and restoration of coastal wetlands, thus contributing to climate change mitigation (Vanderklift *et al.,* 2019). The UK is currently developing a pilot saltmarsh carbon code to enable saltmarsh carbon to be marketed and traded as carbon offsets, with ambitions to develop a UK Blue Carbon Code.

The implementation of large-scale blue carbon restoration schemes financed via carbon offsets through the voluntary carbon market is becoming more viable, with successful projects already observed (Wylie *et al.,* 2016). However, few countries have policy frameworks that support the financing of climate change mitigation through coastal wetland management (Kelleway *et al.,* 2020). At the time of publication, Wylie *et al.* (2016) noted limitations for the financing of blue carbon in the lack of accounting for sediment carbon pools, and the impacts of climate change and sea-level rise on the long-term success of blue carbon schemes.

2.3 The effective management of saltmarshes for climate change mitigation

Lovelock & Duarte (2019) noted that term 'blue carbon' can have multiple meanings: firstly, the carbon captured and stored by marine and coastal habitats, and secondly, how marine and coastal habitats can be managed to reduce CO_2 emissions and contribute to climate change mitigation. The blue carbon concept thus encompasses both meanings and refers to the effective management of coastal wetlands, via conservation and restoration, to enhance climate change mitigation (Macreadie *et al.,* 2019). Saltmarshes are considered the most widespread and important blue carbon habitat outside of the tropics (Mcleod *et al.,* 2011), and are the focal ecosystem in this thesis.

2.3.1 Saltmarsh ecosystems: ecology, services and threats

Saltmarshes are intertidal ecosystems composed of salt-tolerant plants (halophytes) that dominate low-energy, saline wetlands in temperate climatic zones (Adam, 1990; Barbier *et al.,* 2011; Sarika & Zikos, 2021). Low-energy, sheltered environments, such as estuaries and shallow bays, provide suitable conditions for saltmarshes to accumulate fine sediments (Fagherazzi *et al.,* 2012). Furthermore, at the interface between land and sea, saltmarshes are exposed to regular cycles of tidal inundation, which also provides the main mechanism for sediment delivery (Fagherazzi *et al.,* 2012; Fig. 2.2). The development of saltmarshes and their ability to accrete, a vital adaptive capacity to sea-level rise, are governed by complex interactions between sediment availability, vegetation, morphology, elevation in the tidal frame and hydrodynamics (Fagherazzi *et al.,* 2012; Kirwan *et al.,* 2016; Pratolongo *et al.,* 2019). Saltmarsh vegetation communities exhibit clear zonation patterns, primarily driven by environmental gradients that are imposed by elevation e.g., species tolerance to the frequency and duration of tidal inundation (Pratolongo *et al.,* 2019). As such, saltmarshes are inextricably connected to sea-level change and tidal oscillations (Fagherazzi *et al.,* 2012).

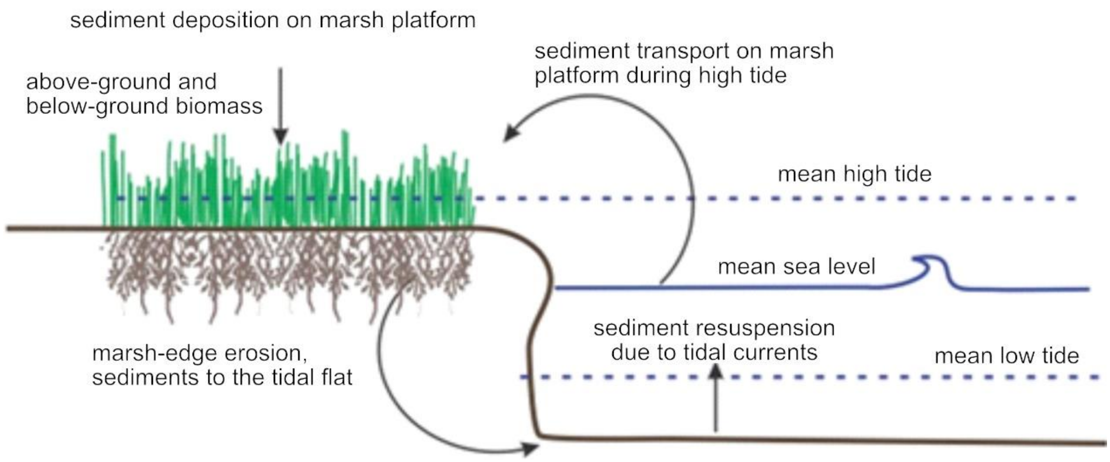


Figure 2.2. Mechanisms and sediment fluxes that control saltmarsh vertical and lateral dynamics. Image reproduced from Pratolongo *et al.* (2019).

Saltmarshes are highly valuable ecosystems delivering a multitude of societal benefits. Among these services is the attenuation of waves, even under storm surge conditions, and the slowing of tidal flow by saltmarsh vegetation (Morgan *et al.,* 2009; Möller *et al.,* 2014). This capability acts to protect coastlines and encourages the retention of sediments and nutrients, preventing an excessive influx into coastal waters as well as more sensitive ecosystems, such as seagrass meadows (Barbier *et al.,* 2011). Saltmarshes also help to protect coastlines against erosion as their root structures stabilise coastal sediments (Shepard *et al.,* 2011; Bouma *et al.,* 2014). The structural complexity of saltmarshes create suitable nursery areas for estuarine fish species, whilst providing foraging habitat for numerous wading bird species (Whitfield, 2017; Enners *et al.,* 2019). Finally, saltmarshes are becoming increasingly renowned for their disproportionate ability to mitigate climate change through the capture and storage of blue carbon (Lovelock & Duarte, 2019; Macreadie *et al.,* 2019).

Despite their high value, coastal wetlands are anticipated to have lost approximately 50 % of their global extent during the 20th century (Li *et al.,* 2018). Saltmarshes are susceptible to climate change impacts and their persistence is particularly threatened by accelerating rates of sea-level rise (Crosby *et al.,* 2016; Saintilan *et al.,* 2022). Mangrove encroachment into saltmarshes in subtropical regions has also been observed as a consequence of poleward expansion by mangroves due to warmer temperatures (Saintilan *et al.,* 2014; Kelleway *et al.,* 2016b). However, saltmarsh vulnerability to the impacts of climate change are exacerbated by non-climatic drivers, with individual pressures acting synergistically to cause impacts greater than their sums (Duarte *et al.,* 2008; Deegan *et al.,* 2012). Although, non-climatic pressures such as land reclamation, including the draining of saltmarshes for agricultural conversion, coastal development and reductions in sediment supply as a result of dam constructions are considered to be the primary cause of observed declines in saltmarsh extent (Adam, 2002; Kroeger *et al.,* 2017; Bindoff *et al.,* 2019).

Recent global restoration efforts have not only been motivated by an awareness of anthropogenic-driven declines in saltmarsh ecosystems, but also by the increased recognition of the societal benefits they provide (Li *et al.,* 2018; Murray *et al.,* 2022). Furthermore, the co-occurrence of the UN Decade on Ecosystem Restoration and the UN Ocean Decade, the latter of which aims to reverse deteriorations in ocean health, provides greater support for future saltmarsh restoration efforts (Waltham *et al.,* 2020).

2.3.2 Saltmarsh restoration and blue carbon

A comprehensive overview of saltmarsh restoration methods can be seen in the following publication: '*Saltmarsh restoration handbook*' for the UK and Ireland (Hudson *et al.,* 2021). However, the predominant method of saltmarsh restoration in the UK is managed realignment – the deliberate breaching of coastal defences to restore tidal hydrology to the land behind defences (Luisetti *et al.,* 2011).

Coastal habitat restoration for blue carbon benefits is becoming an important tool in climate change mitigation (Duarte *et al.,* 2020; Wylie *et al.,* 2016). Although climate change mitigation is often reported as a co-benefit of saltmarsh restoration schemes, it has never been the reason for restoration, nor yet a measure of a scheme's success (Burden *et al.,* 2013; Austin *et al.,* 2022). In the UK, all current managed realignment schemes have either restored saltmarsh habitat for coastal protection to reduce costs associated with hard-engineering approaches to flooding, or to enhance biodiversity in line with the EU Habitats Directive to compensate for historic losses (Shepherd *et al.,* 2007; Hudson *et al.,* 2021). Saltmarsh restoration is thus an attractive option to achieve a multitude of co-benefits.

The carbon accumulation and storage potential of restored saltmarshes has previously been highlighted as an important, yet understudied area (Tang *et al.,* 2018). Recent work has made contributions to address this knowledge gap reporting on carbon accumulation rates and carbon stocks of restored saltmarshes (Wollenberg *et al.,* 2018; Burden *et al.,* 2019; Poppe & Rybczyk, 2021; Mossman *et al.,* 2022). Following saltmarsh restoration, carbon accumulation rates are initially very high and can far exceed those reported for nearby natural saltmarshes which are well-established (Wollenberg *et al.,* 2018). This is often linked to the low elevation of restored sites where tidal inundation is more frequent and the opportunity for carbon accumulation is greater, predicted by sediment accretion (Pontee, 2014; Poppe & Rybczyk, 2021). Evidence of this relationship is more dramatic for restored sites in hypertidal systems, with a mean tidal range exceeding 6 m (Rtimi *et al.,* 2021), particularly when characterised by high suspended sediment loads. Initial carbon accumulation rates for saltmarshes restored in hypertidal systems (e.g., Bay of Fundy, Canada, and Steart Marshes, UK), were an order of magnitude above those reported for other restored saltmarshes (Wollenberg *et al.,* 2018; Mossman *et al.,* 2022). The bulk of carbon accumulation at both sites was presumed to be allochthonous, given the rapid sediment accretion, with important implications for carbon financing in saltmarsh restoration (Wollenberg *et al.,* 2018; Mossman *et al.,* 2022). Allochthonous material can be the most importance source of carbon for newly restored sites which initially have a relatively low plant colonisation (Drexler *et al.,* 2020).

Studies measuring carbon accumulation rates across a chronosequence of restored saltmarshes suggest that initial rates are expected to decline as saltmarsh develops on a trajectory towards natural saltmarshes in the area (Burden *et al.,* 2019). This reflects a decline in accretion rates when saltmarshes reach equilibrium elevations with RSLR (Morris *et al.,* 2002). However, the timescale for the decline in accretion rates and carbon accumulation rates are likely to be highly site-specific, in large part driven by the availability of suspended sediment and sufficient accommodation space.

Conversely, carbon stocks in restored saltmarshes remain lower than natural saltmarshes, with little empirical evidence to demonstrate they become functionally equivalent (Moreno-Mateos *et al.,* 2012; Burden *et al.,* 2013; Burden *et al.,* 2019). Managed realigned saltmarshes often lack the topographic diversity required to become functionally equivalent to natural marshes and more closely resemble the topography of low-lying agricultural land they were restored from (Lawrence *et al.,* 2018). Furthermore, managed realigned saltmarshes are not anticipated to be on a trajectory to develop the topographic complexity observed in mature, natural saltmarshes with implications for the development of plant communities and subsequently, carbon stock (Lawrence *et al.,* 2018). Indeed, a meta-analysis of restored wetlands showed that even after a century, the biological structure, represented by plant assemblages, and biogeochemical functioning, represented by soil carbon storage, of restored sites was ~ 25 % lower than natural wetlands (Moreno-Mateos *et al.,* 2012). In restored saltmarshes, the plant community composition can take decades to resemble those of nearby natural marshes, if at all (Mossman et al., 2012a). Likewise, carbon stock has been shown to be more similar to nearby agricultural fields after 15 years, with a predicted timescale of 100 years to reach equivalency with natural saltmarshes (Burden *et al.,* 2013).

As a result, restoration is often prioritised as a secondary strategy behind the preservation and conservation of natural saltmarshes given the timescale for significant carbon stocks to develop in restored saltmarshes (Burden *et al.,* 2013; Macreadie *et al.,* 2017b; Burden *et al.,* 2019). However, there is also high variability in the timescale reported for saltmarshes to achieve equivalent carbon stocks to nearby natural saltmarshes across sub-decadal to decadal and even centennial timescales. For example, carbon stocks at a restored saltmarsh in northwestern USA were similar to neighbouring natural saltmarshes after only four years post-restoration (Poppe & Rybczyk, 2021). It is important to note that carbon stocks for restored saltmarshes are often measured in surficial soil (~ top 30 cm) in an attempt to capture only recent saltmarsh layers. Comparatively, natural blue carbon habitats have thick sediment layers that can extend several metres deep (Pendleton *et al.,* 2012; Lavery *et al.,* 2013).

Current research regarding blue carbon stocks in restored saltmarshes tends to focus on the comparison of functional equivalence with natural saltmarshes, with any variability between restored saltmarshes often attributed to age differences since restoration (Abbott *et al.,* 2019; Burden *et al.,* 2019; Santini *et al.,* 2019). However, a multitude of environmental drivers are considered to influence carbon stock variability in natural saltmarshes. This highlights a current research gap as to the major drivers that can explain variability between restored saltmarshes. Such knowledge can be utilised to predict blue carbon hotspots and priority areas for maximising carbon stocks through saltmarsh restoration.

2.3.3 Marine social science and effective coastal management

The blue carbon concept has been described as multifaceted and interdisciplinary in nature, characterised by collaborations between scientists, conservationists and policy makers (Lovelock & Duarte, 2019). Yet, the incorporation of social research in blue carbon is severely lacking, despite being considered key to the successful adoption and longevity of blue carbon as a natural climate solution (Macreadie *et al.,* 2022). More generally, people-centric research has been identified as a priority to advance the implementation and management of nature-based solutions in marine and coastal habitats as social science has been previously underrepresented in marine and coastal management (O'Leary *et al.,* 2022).

Within marine social science, public perceptions and knowledge have been identified as a knowledge gap and area of research priority (McKinley *et al.,* 2020a). O'Leary *et al.* (2022) also noted that the effective management of marine and coastal nature-based solutions requires greater public awareness of the value of marine and coastal habitats, including coastal wetlands. Currently, the UK public exhibit a low connection with saltmarsh habitat, perhaps due to the perceived lack of charisma and attractiveness of coastal wetlands when compared to other marine habitats, such as coral reefs (Duarte *et al.,* 2008; McKinley *et al.,* 2020b). Public awareness of the societal benefits delivered by marine habitats, such as the climate change mitigation capacity of coastal wetlands, can promote public interest in their conservation, restoration, and overall management (Nordlund *et al.,* 2018).

Greater prominence given to the concept of ocean literacy, simply defined as an understanding of the ocean's influence on you – and your influence on the ocean, is also encouraged to achieve marine and coastal management goals (McKinley *et al.,* 2023; O'Leary *et al.,* 2022). In a similar vein, aspirations of the UN Ocean Decade include a stronger integration of natural and social sciences, while coastal habitat restoration under the UN Decade on Ecosystem Restoration requires trans-disciplinary approaches (Claudet, 2021; Waltham *et al.,* 2020).

2.4 Environmental drivers of saltmarsh blue carbon accumulation and stocks

Global blue carbon stocks and accumulation rates in coastal wetlands are reportedly high, particularly in comparison to terrestrial forests per unit area, yet there is great variability on local to global scales, both in a spatial and temporal context (Duarte *et al.,* 2013; Ouyang & Lee, 2014; Ewers Lewis *et al.,* 2018). Coastal wetlands, including saltmarshes, are amenable to management, and the identification of key environmental drivers of blue carbon variability can be used to inform management activities (Macreadie *et al.,* 2017b; Lovelock & Duarte, 2019; Ewers Lewis *et al.,* 2020). Furthermore, the identification of major drivers that best account for blue carbon variability in saltmarsh ecosystems represents a key objective in blue carbon science (Ewers Lewis *et al.,* 2020). Environmental drivers of saltmarsh blue carbon stocks and accumulation rates can broadly be categorised into climatic, ecological, tidal/ geomorphic and sedimentary characteristics.

2.4.1 Climatic drivers

Climatic drivers are considered to have the greatest influence on saltmarsh blue carbon variability at a global scale (Ewers Lewis *et al.,* 2020). Specifically, relative sea-level rise (RSLR) and mean annual temperature when modelled together explain a large proportion of variability in saltmarsh carbon accumulation rates (Wang *et al.,* 2021). Higher temperatures under global warming are thought to enhance decomposition rates, yet this could be offset by enhanced plant productivity and associated carbon gains (Kirwan *et al.,* 2014; Wang *et al.,* 2019; Table 2.1). In addition to warmer temperatures, elevated atmospheric CO_2 levels has also been linked to increases in plant primary productivity which in turn can promote carbon accumulation rates and the adaptive capacity of saltmarshes to respond to RSLR via elevation gain (Kirwan *et al.,* 2014; Reef *et al.,* 2017; Lovelock & Reef, 2020). A summary of climate change impacts on saltmarsh blue carbon, adapted from Macreadie *et al.,* 2019, can be seen in Table 2.1.

Table 2.1. The influence of climate change factors on saltmarsh blue carbon stocks and accumulation rates. Positive effects are highlighted by italics, negative by bold type, and effects that have the potential to be either positive or negative remain in roman text. Table adapted from Macreadie et al. (2019).

	Sea-level rise	Extreme storms	Temperature increase	Elevated CO2
Influence on saltmarsh blue carbon	*Increased areal extent and carbon stocks* **Coastal squeeze could prevent saltmarsh transgression causing loss of extent and reduced carbon stocks** *Increasing accommodation space enhances blue carbon accumulation*	**Loss of saltmarsh area and subsequently carbon stocks** *Enhanced sediment accretion, and elevation gain, increases carbon stocks and, reducing effects of sea-level rise*	Increased temperatures may increase decomposition of soil organic matter, but offset by increased plant productivity Poleward expansion of mangroves will replace saltmarshes, but evidence shows this will increase carbon storage	*Increased plant productivity due to availability of CO_2 for photosynthesis*

2.4.1.1 Relative sea-level rise, vertical accommodation space and blue carbon accumulation

RSLR is predicted to have variable impacts on saltmarsh blue carbon, depending on the ecosystems ability to keep pace with increasing global rates of RSLR (Lovelock & Reef, 2020). Global rates of RSLR are accelerating – estimates of global mean sea level have demonstrated an increasing trend of 1.56 ± 0.33 mm yr^{-1} between 1900 and 2018, driven by two key contributors: ocean thermal expansion and mass water input from ice melt (Nicholls & Cazenave, 2010; Frederikse *et al.,* 2020). It is predicted that in alternative scenarios where anthropogenic influence was removed, global sea-level rise would be less than 58% of its observed value in the 20th century (Kopp *et al.,* 2016). The anoxic condition of saltmarsh sediment reduces decomposition and enables mineral and organic material to accrete both vertically and laterally, providing saltmarshes an adaptive capacity to keep pace with rates of RSLR (Kirwan & Blum, 2011; Wollenberg *et al.,* 2018). The ability of saltmarshes to keep pace with RSLR depends on sufficient sediment supply to support vertical accretion gain, and the availability of lateral space for saltmarshes to transgress without constraint by coastal development, known as coastal squeeze (Lovelock *et al.,* 2015; Borchert *et al.,* 2018).

The impacts of climate change and RSLR have dominated research and conservation policy for saltmarsh ecosystems over the past 3 decades (Crosby *et al.,* 2016). Model simulations of saltmarsh accretion and projected RSLR demonstrate marsh resilience up to a certain, site-specific threshold of RSLR, beyond which inundation rates are too high and saltmarsh plant mortality and extensive habitat loss occurs (Kirwan & Mudd, 2012; Kirwan & Megonigal, 2013; Thorne *et al.,* 2018). A potential loss of 78 % of global coastal wetlands (saltmarshes and mangroves, in this instance) has been predicted by 2100 if RSLR reaches 1.1 m (Spencer *et al.,* 2016). Although, global model projections are often criticised for failing to account for the dynamic nature of saltmarshes to migrate inland via transgression, or biophysical feedbacks that enable vertical accretion to (Kirwan *et al.,* 2016; Schuerch *et al.,* 2018). The resilience of saltmarshes to RSLR is still under debate (Langston *et al.,* 2021), but accelerating rates of RSLR are likely to influence saltmarsh structure, function and the provision of ecosystem services, including the capacity for climate change mitigation (Craft *et al.,* 2009).

Accelerating RSLR has been demonstrated to promote a restructure of saltmarsh vegetation, causing a shift from communities associated with higher marsh elevations to favour pioneer plant species (Gonneea *et al.,* 2019). In Cape Cod, USA, an increase in saltmarsh accretion and biomass productivity enhanced carbon accumulation following a drop in the saltmarsh high marsh zone to elevations suitable for low marsh plant species (Gonneea *et al.,* 2019). A positive feedback effect between RSLR and saltmarsh vertical accretion can create optimal conditions for enhanced blue carbon storage (Gonneea *et al.,* 2019). Vertical accretion in saltmarshes, as a result of RSLR, is underpinned by the concept of vertical accommodation space, which refers to the vertical space available for sediments to accumulate, for example, during tidal inundation (Schuerch *et al.,* 2018; Rogers *et al.,* 2019a; Fig. 2.3). Saltmarshes experience rapid accumulation of both mineral and organic material under RSLR due to an increase in inundation frequency that is linked to increases in sedimentation (Pethick, 1981; Rogers *et al.,* 2019a). Enhanced sediment accretion rates under RSLR also slow decomposition rates by rapidly burying deposited material which creates anoxic conditions, promoting long-term blue carbon storage (Rogers *et al.,* 2019a).

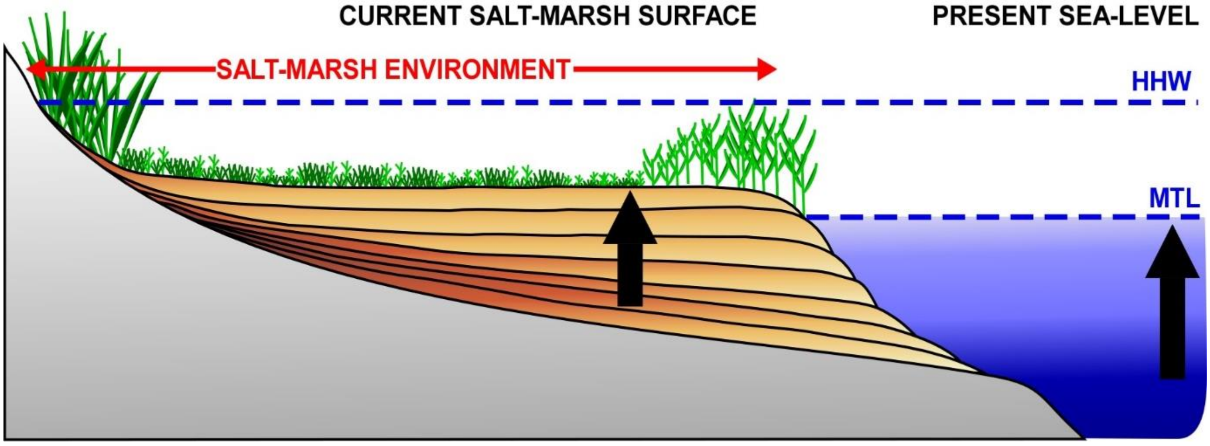


Figure 2.3. Saltmarsh vertical accommodation space increases in response to increases in both mean tidal level (MTL) and highest high water (HHW).

Based on a global analysis with data collated from 345 saltmarsh sites, marshes on coastlines subject to rapid RSLR were found to have an estimated 1.7 – 3.7 times the amount of blue carbon stored in the upper 20 cm of sediments, and between 4.9 – 9.1 times the amount at 50 – 100 cm, when compared to marshes in areas of reasonably stable RSLR over the past millennia (Rogers *et al.,* 2019a). These differences were attributed to rates of RSLR whereby the creation of vertical accommodation space (Fig. 2.3) is greater under rising sea levels (Rogers *et al.,* 2019a). Consequently, RSLR is considered a major control on blue carbon accumulation and storage in saltmarshes (Rogers *et al.,* 2019a).

Several other recent research papers have found vertical accommodation space, as a direct consequence of RSLR, to enhance saltmarsh blue carbon stocks and accumulation rates. For example, sediment vertical accretion rates and carbon accumulation rates increased 10-fold over a 71 year period in the Celestun lagoon, Mexico, attributed to increased inundation under RSLR (Carnero-bravo *et al.,* 2018). Similarly, enhanced sediment accretion under RSLR has been observed to enhance blue carbon accumulation rates in other coastal wetlands – subtidal seagrass meadows and mangrove forests (Watanabe *et al.,* 2019). Some modelling studies even project net gains in blue carbon due to increased coastal wetland extent under projected future RSLR (Lovelock & Reef, 2020; Wang *et al.,* 2021). Although, significant reductions in blue carbon accumulation can also occur as a result of future saltmarsh loss under coastal squeeze in projected RSLR scenarios (Lovelock & Reef, 2020).

While blue carbon accumulation rates in a tropical saltmarsh in Mexico were observed to increase over a centennial timescale due to RSLR and higher sediment accretion rates, carbon stocks were reduced (Ruiz-fernández *et al.,* 2018). This was potentially attributed to the increased marine conditions with a greater input of mineral sediment and a reduction in terrestrial organic material (Ruiz-fernández *et al.,* 2018).

Until recently, there was a scarcity of field-research that examined the relationship between RSLR and saltmarsh blue carbon accumulation. However, given climate change impacts vary geographically (Macreadie *et al.,* 2019), the use of site-specific, high-resolution sea-level reconstructions paired with carbon accumulation rates for saltmarshes across different geographic regions, and subject to different sea-level histories, is needed to advance understanding of the relationship between RSLR and blue carbon accumulation. Overall, the impact of RSLR on blue carbon accumulation rates remains poorly understood, despite recent advances (Watanabe *et al.,* 2019). The impact of climate change on blue carbon accumulation has been deemed a high priority research area to progress blue carbon science, with RSLR considered one of the most influential drivers for future blue carbon stocks and accumulation rates (Macreadie *et al.,* 2019).

Studies evaluating drivers of variability in blue carbon stocks and accumulation rates in saltmarshes tend to focus on global (Rogers *et al.,* 2019a), and even regional scales (Ford *et al.,* 2019), yet drivers at finer spatial scales, relevant to management activity, are less well understood (Broek *et al.,* 2016). Fine-scale drivers explaining variability of blue carbon stock variability includes ecological, tidal/ geomorphic, and sedimentary characteristics.

2.4.2 Ecological characteristics

Plant community composition is an important indicator for blue carbon capacity in saltmarshes and has been demonstrated to be a major driver of variability in sediment carbon stocks (Macreadie *et al.,* 2017a; Ford *et al.,* 2019; Rogers *et al.,* 2019b). In particular, fundamental differences in productivity and structure among saltmarsh plant communities exert a primary control on blue carbon stocks and accumulation through primary productivity and trapping efficiency (Saintilan *et al.,* 2013). A large proportion of primary production occurs in the belowground biomass (roots and rhizomes) of saltmarsh plants that are buried within the sediment (Chmura *et al.,* 2003; Saintilan *et al.,* 2013). As such, belowground biomass likely plays an important role in the production of autochthonous carbon. Plant species richness is thought to increase root biomass, due to increased functional diversity, and enhance sediment stabilisation, thereby enhancing carbon stocks which are better preserved against erosion (Ford *et al.,* 2016). Numerous studies have now associated greater carbon stock for communities associated with higher elevations, such as mid and high marsh zones, where plant community compositions are predicted to be more diverse (Sharpe & Baldwin, 2009;

Ford *et al.,* 2016; Smeaton *et al.,* 2022).

In addition to carbon accumulation and stocks, species-level differences in saltmarsh plants have also been observed to drive variability in the capacity to retain allochthonously sourced carbon (Ouyang & Lee, 2014; Sousa *et al.,* 2010). The ability to trap and retain particles is mediated by the structural complexity of above-ground biomass (Mudd *et al.,* 2010; Kelleway *et al.,* 2017). In turn, above-ground biomass in saltmarshes is best described by vegetation height, species and vegetation density (Owers *et al.,* 2018). For example, dense saltmarsh vegetation is capable of attenuating wave energy and consequently, encouraging the settling of suspended particles, which can be rich in allochthonously sourced carbon (Mudd *et al.,* 2010). Furthermore, tall, dense plant assemblages (e.g. *Spartina alterniflora*, a low-marsh species native to the east coast of North America) are considered to promote the retention of autochthonous carbon, such as plant litter, which would otherwise be exported on the tides (Kelleway *et al.,* 2017). Accounting for vegetation community composition can improve understanding of spatial variability in saltmarsh blue carbon stocks, while knowledge of structural and functional roles of plant communities can be used to inform the maximisation of blue carbon stocks.

2.4.3 Tidal and geomorphic setting

Allochthonous carbon can make a considerable contribution to saltmarsh carbon stocks, particularly when saltmarsh plants have a high trapping efficiency (Duarte *et al.,* 2013; Van de Broek *et al.,* 2018). However, the supply of allochthonous carbon to saltmarshes itself is largely regulated by environmental setting, i.e. position in the tidal frame, and alongside sediment accretion rates, is linked to the local tidal regime (Harvey *et al.,* 2019; Hudson *et al.,* 2021). Saltmarsh elevation can impact the duration and frequency of tidal inundation – at lower elevations, a greater duration of tidal inundation increases the time for allochthonous carbon to be deposited or captured by vegetation (Kelleway *et al.,* 2017). Macreadie *et al.* (2017b) recommended restoring hydrology, via freshwater flows and tidal exchange, as a management strategy to enhance blue carbon accumulation and stocks. Two main saltmarsh restoration methods, managed realignment and regulated tidal exchange, restore hydrology by either breaching coastal defences and enabling the land behind to flood, or by managing tidal regimes through engineered structures, such as tide-gates (MacDonald *et al.,* 2020; Hudson *et al.,* 2021).

Saltmarshes located within estuaries are subject to strong environmental and ecological gradients (Broek *et al.,* 2016). In addition to their vertical elevation within the tidal frame, also known as relative tidal height, the position of saltmarshes within an estuary can also influence blue carbon stocks (Ewers Lewis *et al.,* 2020). Carbon stocks in freshwater marshes were four-fold greater than those of saltmarshes along an increasing estuarine salinity gradient in the Netherlands (Van de Broek *et al.,* 2016). Similar patterns have been observed in southeastern Australia where saltmarshes in marine settings generally have half the stock of blue carbon as those in fluvial environments (Saintilan *et al.,* 2013; Kelleway *et al.,* 2016a; Macreadie *et al.,* 2017a). In fluvial settings, the deposition of terrestrially derived allochthonous carbon is greater and tends to be more refractory than marine sources, such as algal detritus, which are more prone to remineralisation (Kelleway et al., 2016a; Macreadie *et al.,* 2017a; Macreadie *et al.,* 2017b).

2.4.4 Sedimentary characteristics

Sediment grain size has also proven to be a key predictor in explaining spatial variability of saltmarsh blue carbon stocks; carbon stocks are better preserved in fine-grained sediments, such as clay and silt (Ford *et al.,* 2019; Kelleway *et al.*, 2016a). In comparison to coarse-grained sediments, such as sand, fine-grained sediments have a greater specific surface area which reduces oxygen exchange and carbon remineralisation rates, promoting long-term carbon storage (Bock & Mayer, 2000; Serrano *et al.,* 2016b). Furthermore, clay has also be shown to be more resistant to erosion than sandy soils, with consequences for the preservation of carbon (Ford *et al.,* 2016).

2.5 Knowledge gaps

As a research field, blue carbon has had a relatively recent evolution. Increased interest in blue carbon from political and economic fields has created an urgent need to refine knowledge on the role of coastal wetlands, including saltmarshes, in climate change mitigation. In particular, identifying major environmental characteristics that drive variability in blue carbon stocks and accumulation rates has been considered a key objective to progress blue carbon science.

Climate change impacts on blue carbon accumulation rates has been deemed a priority research area and relative sea-level rise is considered to be one of the most influential drivers. However, the impact of RSLR on blue carbon accumulation rates for saltmarshes remains poorly understood, as does the resilience of saltmarshes and associated carbon stocks under accelerating rates of RSLR. The influence of RSLR on carbon accumulation rates needs to be understood across varied geographic regions in saltmarshes subject to different sea-level histories.

Current understanding of environmental drivers that act on saltmarsh blue carbon dynamics at finer spatial scales is not as advanced as drivers acting across regional and global scales. Yet, finer spatial scales are more amenable to management activity. Furthermore, the variability of carbon stocks in restored saltmarshes has previously been related to time since restoration, with little knowledge of key environmental drivers. As blue carbon is likely to become an important facet of future saltmarsh restoration schemes, the identification of key fine-scale drivers of variability can be used to locate priority areas to maximise carbon stocks.

Finally, the focus of progressing blue carbon science thus far has focused on natural sciences relevant to better understanding carbon stocks and accumulation rates. Whilst this area still requires great consideration and future work to address knowledge gaps, there has been a severe lack of social research on blue carbon. To better implement the effective management of saltmarshes as natural climate solutions, societal support for blue carbon and coastal wetlands needs to be understood from a diversity of perspectives.

In order to address these knowledge gaps and priority research areas, the following objectives have been identified. For more information on the broader aims of the thesis and greater detail on the objectives outlined below, see **Chapter 1**.

1) To ascertain whether rates of relative sea-level rise are a dominant control on blue carbon accumulation rates in saltmarshes.

2) To investigate whether saltmarsh resilience to relative sea-level rise influences blue carbon stocks.

3) To determine the major environmental drivers of blue carbon stocks for restored saltmarshes.

4) To quantify public perceptions, understanding and awareness of blue carbon and associated habitats (saltmarshes and seagrasses), while identifying social and demographic backgrounds that influence perceptions.

Chapter 3 of 7

The influence of relative sea-level rise on blue carbon accumulation rates in saltmarshes

Abstract

The impacts of climate change and associated relative sea-level rise on blue carbon accumulation rates have been highlighted as a priority research areas for blue carbon science. Whilst relative sea-level rise is believed to threaten the stability of saltmarshes, with negative consequences on their capacity as carbon sinks, it has also been proposed as a mechanism to enhance blue carbon accumulation rates. The latter is attributed to the creation of vertical accommodation space, the space available for mineral and organic matter accumulation. Research efforts to further understand the mechanism by which relative sea-level rise influences carbon accumulation rates have been made over temporal and spatial scales, yet there remains a scarcity of site-specific, high-resolution research, which this study addresses. We find the rate of relative sea-level rise was a dominant control driving temporal variability in saltmarsh carbon accumulation rates, notably during the 20th century when proxy sea-level records are more reliable than over longer, multi-centennial timescales. Large-scale spatial patterns were also observed. Rates of sediment accretion and carbon accumulation were significantly higher across sites in Australia and New Zealand than those in the North Atlantic, linked to a greater acceleration in 20th century rates of relative sea-level rise in the Southern Hemisphere. This holds important implications given recent predictions of the globally relevant, enhanced carbon accumulation capacities of coastal wetlands located in the Southern Hemisphere under future sea-level rise. Overall, our findings add to the growing evidence base that suggest the creation of vertical accommodation space, as a direct result of relative sea-level rise, enhances blue carbon accumulation rates. Further research recommendations are also provided to better understand carbon accumulation rates over decadal-centennial and multi-centennial timescales.

3.1 Introduction

Blue carbon refers to organic carbon that is captured and stored by marine and coastal ecosystems, notably coastal wetlands: saltmarshes, mangrove forests and seagrass meadows. Together, these ecosystems occupy 0.2% of the ocean’s surface area, but are responsible for 50% of carbon buried in marine sediments (Duarte *et al.,* 2013). This is in part due to the high productivity of vegetation that dominates coastal wetland ecosystems, and the storage of carbon in belowground, waterlogged sediment which inhibits microbial action and enhances long-term preservation (Mcleod *et al.,* 2011; Hopkinson *et al.,* 2012; Tang *et al.,* 2018). These conditions enable coastal wetlands to rank among the most efficient, natural carbon sinks in the world (Mcleod *et al.,* 2011).

The role coastal wetlands, including saltmarshes, play in climate change mitigation and their potential inclusion in emission reduction policy (Kelleway *et al.,* 2020), has garnered increased interest in their effective management among scientists, policy makers and coastal managers over the past decade (Bouillon *et al.,* 2008; Fourqurean *et al.,* 2012; Macreadie *et al.,* 2017b). To successfully manage these ecosystems and their climate regulation services, it is imperative to understand environmental drivers that govern spatial and temporal blue carbon dynamics; this includes the impact of climate change on blue carbon accumulation rates and drivers of carbon burial in coastal wetlands, both of which have been identified as unanswered questions and areas of priority for future blue carbon research (Macreadie *et al.,* 2019).

The acceleration of climate-driven relative sea-level rise (RSLR) threatens the stability of saltmarsh extent, structure, and productivity with negative impacts on their capacity as blue carbon sinks (Chmura, 2013; Watson *et al.,* 2017). However, healthy saltmarsh ecosystems display vertical resilience to RSLR through the build-up of an external supply of mineral sediments and organic matter, and the subsequent promotion of increased belowground biomass production, to enhance elevation (Morris *et al.,* 2002; Fagherazzi et al., 2012; Kirwan & Megonigal, 2013; Wang *et al.,* 2019). For example, the capture and storage of autochthonous carbon via photosynthesis and allochthonous carbon through the trapping of organic carbon delivered on the tides, contributes to the accumulation of organic matter, which in addition to the input of mineral sediment, enables saltmarshes to vertically accrete and maintain an elevation above sea level, providing the rate of RSLR is not too great (Mcleod *et al.,* 2011; Wollenberg *et al.,* 2018). Horton *et al.* (2018) suggested that when subjected to RSLR rates greater than ca. 7 mm yr^{-1} saltmarshes are likely to retreat, but in the vertical dimension, saltmarshes may be able to survive RSLR rates of up to 10 mm yr^{-1} (Kirwan *et al.* 2016).

Accelerated RSLR creates accommodation space (Rogers *et al.,* 2019a) and increases inundation period, which is positively correlated with sedimentation (Pethic, 1981). The creation of vertical accommodation space, known as the space available for mineral and organic matter accumulation (Rogers *et al.,* 2019a), has enabled saltmarsh sediment accretion rates to keep pace with rates of relative sea-level rise (Kirwan & Megonigal, 2013; Kirwan *et al.,* 2016; Schuerch *et al.,* 2018). Sediment accumulation rates are modulated by saltmarsh macrophytes (Morris *et al.,* 2002), whether by direct organic sedimentation, particle capture by vegetation, or enhanced settling resultant from reduced tidal flow through vegetation canopy (Mudd *et al.,* 2010). These biophysical feedbacks governing sediment accumulation in saltmarshes directly control blue carbon burial rates (Watanabe *et al.,* 2019), and thus RSLR is considered among the most influential factors on blue carbon stocks and accumulation rates (Macreadie *et al.,* 2019).

Saltmarshes in locations around the globe subject to rapid rates of RSLR have reported blue carbon stocks up to 3.7 and 9.1 times higher at soil depths in the upper 20 cm and 50 to 100 cm, respectively, when compared to locations in areas of more stable sea-level history (Rogers *et al.,* 2019a). This disparity is owed to enhanced vertical accommodation space created by RSLR (Rogers *et al.,* 2019a). Previous modelling studies have shown a similar mechanism for blue carbon accumulation whereby rates increase with RSLR, up to a certain threshold (Mudd *et al.,* 2009; Kirwan & Mudd, 2012). These findings complement more recent literature and field-based research on a local, or national, scale (Ruiz-Fernández *et al.,* 2018; Breithaupt *et al.,* 2020; Rogers *et al.,* 2019a; Wang *et al.,* 2019) and on a global scale (Wang *et al.,* 2021) In particular, the acceleration of saltmarsh blue carbon accumulation rates in the past century has been attributed to RSLR (Breithaupt *et al.,* 2020; McTigue *et al.,* 2019), which reflects the accelerating trend in the rate of global sea-level rise from the 20th century (~1.7 mm yr^{-1}) to the 21st century (~3.4 mm yr^{-1}; Kopp *et al.,* 2016).

Until recently, the relationship between temporal changes in relative sea level and blue carbon accumulation rates was poorly understood, owing to a scarcity of field-based research (Ruiz-Fernández *et al.,* 2018). Numerous studies have since explored this relationship (Breithaupt *et al.,* 2020; Rogers *et al.,* 2019a; Wang *et al.,* 2021; Weston *et al.,* 2023), yet there remains a scarcity of field research that pairs site-specific, high-resolution sea-level reconstructions with blue carbon accumulation rates in saltmarshes across a global scale. Whilst global average sea-level rise is on an upward trend (Frederikse *et al.* 2020; Walker *et al.,* 2021), spatio-temporal variability in sea-level change exists, and patterns are anticipated to oscillate across multi-decadal time scales (Nicholls & Cazenave, 2010). The impacts of RSLR on blue carbon accumulation rates in saltmarshes is thus likely to vary across past sea-level history and geographic location (Macreadie *et al.,* 2019; Rogers *et al.,* 2019a; Wang *et al.,* 2021), and requires a comparison on decadal-centennial time scales for locations spanning latitudinal and longitudinal gradients, subject to differing rates of RSLR.

Here, we examine the relationship between RSLR and blue carbon accumulation rates at saltmarsh locations spanning three continents (North America, Europe, and Australasia) and five countries. Our goal was to test the following hypothesis over decadal-centennial and multi-centennial timescales: '*the creation of vertical accommodation space, as a direct result of RSLR, enhances blue carbon accumulation rates'*.

3.2 Materials and methods

3.2.1 Study locations

Seven saltmarsh sites from around the world were investigated; four in the Northern Hemisphere and three in the Southern Hemisphere (Fig. 3.1; Table 1).

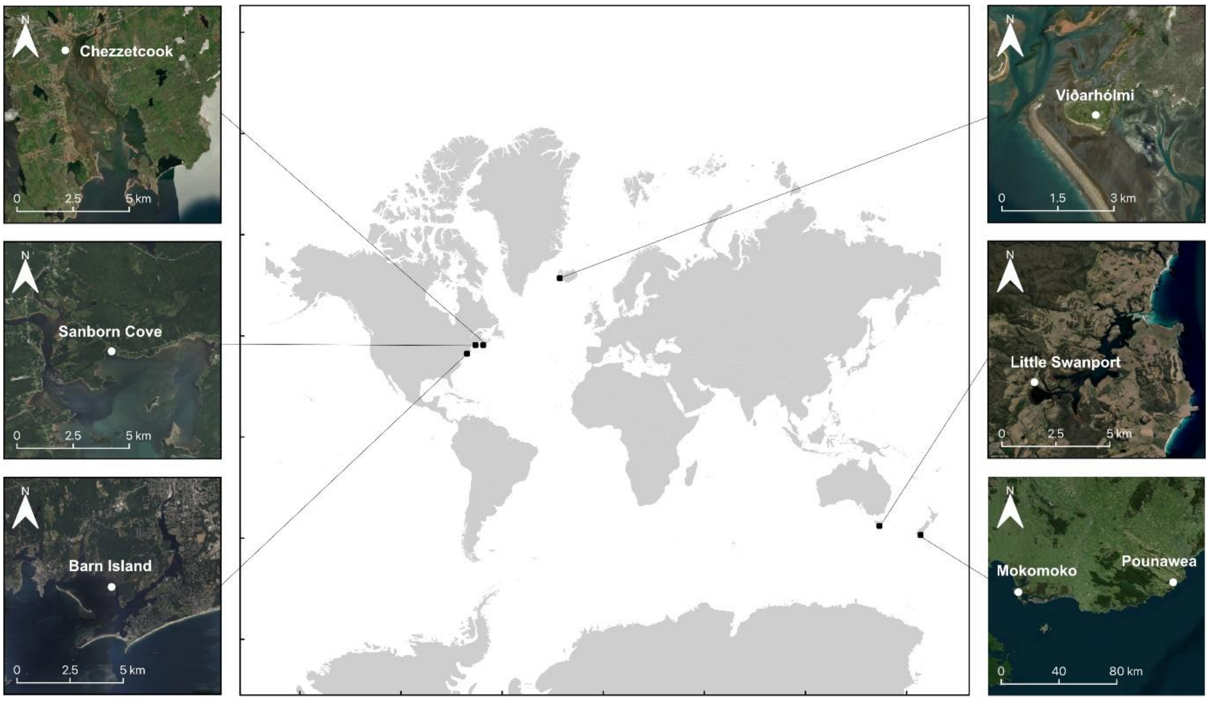


Figure 3.1. Location of study sites with four locations in the Northern Hemisphere and three locations in the Southern Hemisphere. Note: the two New Zealand sites (Mokomoko and Pounawea; bottom right inset map) are approximately 110 km apart.

We selected these sites due to the availability of high-resolution (sub- to multi-decadal) sea-level reconstructions from previous work (Table 3.1), and the availability of sediment cores to measure CAR for this study.

Table 3.1. Core depth and corresponding time span for sediment cores collected during prior research on sea-level change at each study site. The same sediment cores were used to measure carbon accumulation rates in this study.

Location	Country	Core depth (m)	Time span (years CE)	Prior research
Viðarhólmi	Iceland	0.49	1596 – 1994	Saher *et al.,* 2015
Chezzetcook	Canada	1.10	1459 – 2003	Gehrels *et al.,* 2020
Sanborn cove	USA	0.40	1757 – 2006	Gehrels *et al.,* 2020
Barn Island	USA	0.50	1506 – 2008	Gehrels *et al.,* 2020
Little Swanport	Australia	0.41	1820 – 2001	Gehrels *et al.,* 2012
Mokomoko	NZ	0.51	1773 – 2004	Garrett *et al.,* 2022
Pounawea	NZ	0.39	1598 – 2003	Garrett *et al.,* 2022

To produce a detailed, high-resolution dataset from which to evaluate the relationship between relative sea level (RSL) and carbon accumulation rates (CAR), the published sea-level reconstructions were re-analysed and subjected to Bayesian age-depth and Gaussian process modelling to produce age (year CE) estimates, sediment accretion rates (SAR) and the rate of RSLR at the same resolution (1 cm) as measurements of CAR for each location.

3.2.2 Data collection

3.2.2.1 Age-depth modelling and sediment accretion rate

Updated radiocarbon calibration curves (Hogg *et al.,* 2020; Reimer *et al.,* 2020) and software necessitate updated age-depth models for some of the published sea-level reconstructions. Such revised age-depth models have been published for Mokomoko, Pounawea (Garrett *et al.,* 2022) and Little Swanport (Williams *et al.,* in press). To derive age-depth models for the remaining sites, we combine all available chronohorizons for each core in a Bayesian framework either using the *rplum* package for sites with ^{210}Pb data, or the *rbacon* package for sites lacking ^{210}Pb data (Blaauw *et al.,* 2022). The use of *rplum* allows the integration of ^{210}Pb data with other chronohorizons (e.g. ^{137}Cs, ^{14}C, pollen and stable lead isotope) without the requirement to remodel the outputs from traditional ^{210}Pb-only depositional models (Aquino-

López *et al.,* 2018; Aquino-López *et al.,* 2020). We use *rplum* and *rbacon* to derive age estimates and sediment accretion rates, each with 2σ uncertainties, for every centimetre of depth.

3.2.2.2 Rates of relative sea-level rise

Gaussian process regression (Rasmussen and Williams, 2006; Cahill *et al.,* 2015) was used to account for both temporal and vertical uncertainties in the sea-level data. The sea-level dataset consists of N sea-level index points ($d_{1 \ldots N}$), which are assumed to be related to sea level (f(x)) by:

$$d_i = \int_{\chi} \omega_i(x) f(x) dx$$

We assume that f(x) is described by a Gaussian process function with some mean function μ, and covariance function, k.

$$f(x) \sim GP(\mu(x), k(x, x'))$$

A linear mean and Matérn covariance function were assumed. Hyperparameters for each function (i.e., intercept and gradient, and amplitude, length-scale and order parameter, respectively) were determined from the data by maximising the likelihood of a given set of observations (i.e., minimizing the negative log likelihood) using the bound-constrained limited memory BFGS optimization ('L-BFGS-B') algorithm (this iteratively searches parameter space to find values for the hyperparameters that minimize the objective function). The Gaussian process model is then constructed using these optimal hyperparameters, and evaluated at 1-year intervals. From this ensemble, the mean and standard deviation were determined to give the posterior mean sea level and confidence intervals. The rate of sea-level change is the first derivative of the ensemble mean, and is evaluated at the same 1 cm depth intervals as rates of sediment accretion and carbon accumulation data.

In addition to rates of RSLR at each sampled depth, a singular rate of RSLR was calculated across the whole core depth (Table 3.1) and for specific time intervals at each location (e.g., 18^{th}, 19^{th} and 20^{th} centuries; Table 3.2; Appendix A1 and A2). Rates of RSLR for these specific time intervals were calculated using sea-level index points and their associated 1σ age and vertical uncertainties. For each time interval, a linear best fit from 10,000 realisations on the data were used to produce mean rates of RSLR with 95 % confidence intervals.

3.2.2.3 Sedimentary analysis

Where sufficient core material was available, sampling was conducted at 1 cm resolution for all cores. At each sampling point, the whole width of the sediment core was removed and homogenised to ensure samples encapsulated heterogeneity. Saltmarsh sediment samples were oven-dried at 60 °C for a minimum of 24 hrs. Dry bulk density (DBD) was calculated by dividing the dry weight of soil by sample volume. Dried samples were then sieved (1 mm mesh size) to remove large inorganic particles and thick roots, homogenised to a fine powder using a ball mill (Retsch MM200; 250 µm, 2 mins), and weighed into silver capsules. Total organic carbon (TOC) was measured using a Carbon and Nitrogen elemental analyser (Flash 2000, Thermo Fisher Scientific). The instrument was calibrated using aspartic acid as a standard. Prior to TOC analysis, samples were treated with 10 % HCl acid to remove carbonates, and oven-dried overnight at 60 °C. A replicate was taken every 5 samples to assess the reliability of %TOC data.

3.2.2.4 Calculation of carbon accumulation rates

Organic carbon accumulation rates were calculated using depth intervals of subsamples, DBD, TOC and sediment accretion rates. Soil carbon density was calculated using the approach outlined in the Coastal Blue Carbon Manual (The International Blue Carbon Initiative; Howard *et al.,* 2014):

$$Soil\ carbon\ density\ (g\ C\ cm^3) = DBD\ (g\ cm^{-3}) \times \frac{TOC\ (\%)}{100}$$

CAR (converted to g C m^{-2} yr^{-1}) were then calculated by multiplying soil carbon density by sediment accretion rate, the latter of which was derived as an output from age-depth models.

3.2.3 Data analysis

Statistical analysis was conducted using R software (version 4.1.2). Linear mixed models (“lme4” package; Bates *et al.,* 2015), fit assuming a Gaussian error distribution, were used to evaluate whether rates of RSLR influence CAR across spatial and temporal scales. Linear mixed models were carried out on multi-centennial records and for data on the 20th century. Saltmarsh site (Table 1) was included as a random factor in models as sampling did not encapsulate all environmental influences on CAR at each location. P-values were obtained using the “lmerTest” package (Kuznetsova *et al.,* 2017). General linear models (“MASS” package; Venables & Ripley, 2002) were then used to observe location-specific relationships between rates of RSLR and CAR for multi-centennial timescales to ensure such relationships were not overlooked.

Model residuals were checked for normality, homoscedasticity, and bias by unduly influential observations using diagnostic plots, supported by the Shapiro-Wilk test for normality, and Breusch-Pagan test for heteroscedasticity. When necessary, the response variable, CAR, was log-transformed. The proportion of variance in CAR explained by general linear models were calculated as follows: [1 – Residual deviance/ Null deviance]. For linear mixed models, the proportion of variance in CAR explained by the model was calculated using the 'r.squaredGLMM' function ("MuMIn" package; Bartón, 2003).

3.3 Results

Rates of RSLR, SAR and CAR are reported for each site below. To determine the influence of sea-level change on CAR, the rate of RSLR was analysed against CAR (i) across the whole sequence spanning multi-centennial timescales, and (ii) during the 20th century which represents a period of global sea-level rise.

3.3.1 Rate of relative sea-level rise

3.3.1.1 Multi-centennial timescales

All locations across the Northern and Southern Hemisphere showed an increase in RSL over the study period, but variability in rates of RSLR over multi-decadal and centennial timescales were experienced at all locations (Appendix A1; A2).

The three sites in North America (Chezzetcook, Sanborn Cove and Barn Island) showed a relatively similar sea-level history (Appendix A1). A rapid rise in sea level was experienced in both the 18th and 20th century, with a period of more stable sea-level rise in the 19th century, as indicated by oscillations in centennial rates of RSLR (Table 3.2; Appendix A1; Gehrels *et al.,* 2020). Viðarhólmi, in Iceland, experienced most relative sea-level rise over multi-decadal timescales between the late 18th and early 19th century, and from the mid-20th century, the latter of which was the greatest rate of RSLR across the study period (Appendix A1; Table 3.2; Saher *et al.,* 2015).

The remaining three sites (Little Swanport, Mokomoko and Pounawea) are all located in the Southern Hemisphere (Fig. 3.1). Mokomoko and Pounawea, both located in New Zealand, are the closest locations in this study at approximately 110 km apart (Fig. 3.1). At all three sites, RSL rose gradually in the 19th century, before rapidly accelerating in the early 20th century, peaking mid-century (Appendix A2; Table 3.2; Garrett *et al.,* 2022). The sequence at Pounawea extends back to the late 16th/ early 17th century when RSL was stable (Appendix A2).

Table 3.2. Mean rate of relative sea-level rise for each site across the whole core depth sampled ('Overall') and specified time periods. Note that the record end for each location differs (Table 1). For a visualisation of rates across specified time periods see Appendix A1 for Northern Hemisphere locations and Appendix A2 for Southern Hemisphere locations.

Location	**Mean rate of relative sea-level rise (mm yr^{-1})**			
	Overall	1700 - 1799	1800 - 1899	1900 – record end
Viðarhólmi	1.58	0.62	0.96	1.22
Chezzetcook	1.88	1.90	1.33	2.29
Sanborn Cove	1.62	3.45	1.02	3.28
Barn Island	1.67	4.24	0.50	2.60
Little Swanport	3.08	NA	1.44	3.46
Mokomoko	3.08	NA	1.44	3.46
Pounawea	0.93	NA	-0.11	3.47

3.3.1.2 20th Century

Although the exact timing and amplitude differs between sites, a rapid rise in RSL was experienced during the 20th century at all locations in the Northern Hemisphere (Appendix A1).

The 20th century pattern of relative sea-level change is also consistent at sites in Australia and New Zealand and marked by a rapid acceleration in the rate of RSLR during the early 20th century, peaking around mid-century, before a decline during the latter half of the century (Appendix A2). The overall rate of RSLR during the 20th century at locations in the Southern Hemisphere exceeded the rate experienced by locations in the Northern Hemisphere (Table 3.2).

3.3.2 Sediment accretion and carbon accumulation rate

3.3.2.1 Multi-centennial timescales

Over multi-centennial scales, SAR ranged between 1.2 and 2.9 mm yr^{-1}, and CAR ranged between 47.7 and 136.3 g C m^{-2} yr^{-1} (Appendix A3). Across all sites except for Sanborn Cove, mean SAR and CAR were greater in the 20th century when compared to rates over the full sequence (Appendix A3). For the majority of locations, rates of RSLR were also greater in the 20th century than across the full sequence (Table 3.2).

Spatial comparisons of both SAR and CAR should be treated with caution across multi-centennial sequences in this study because calculations derived from each location were averaged over different timescales (Table 3.1; Appendix A3).

3.3.2.2 20th Century

During the 20th century, mean SAR ranged between 1.5 and 3.2 mm yr^{-1} (Appendix A3). Mean SAR in the Southern Hemisphere (2.6 ± 0.5 mm yr^{-1}) was higher than in the Northern Hemisphere (1.9 ± 0.5 mm yr^{-1}).
Mean CAR across all sites in the 20th century was 98.4 ± 33.0 g C m^{-2} yr^{-1} ($n = 145$). Locations in the Southern Hemisphere had a mean CAR (127.9 ± 26.7 g C m^{-2} yr^{-1}) greater than locations in the Northern Hemisphere (76.3 ± 13.1 g C m^{-2} yr^{-1}) during this time period. At all locations in the Southern Hemisphere, the variability in CAR was greater than at locations in the Northern Hemisphere (Fig. 3.2), indicative of the variability in RSLR rates (Appendix A2).

CAR differed significantly among sites ($F = 15.2$; $p < 0.001$; Fig. 2). Viðarhólmi and Barn Island, sites with the lowest mean CAR, had significantly lower mean CAR when compared to all locations in the Southern Hemisphere (Fig. 3.2; Appendix A3; Appendix A4). Mean CAR at Little Swanport was the highest of all sites and was also significantly greater than at Chezzetcook and Sanborn Cove (Fig. 3.2; Appendix A3; Appendix A4). CAR for locations within the Northern Hemisphere did not differ significantly from one another (Fig. 3.2). In the Southern Hemisphere, mean CAR at Little Swanport was significantly greater than at Mokomoko and Pounawea, but the latter two sites did not differ from one another (Fig. 3.2; Appendix A4).

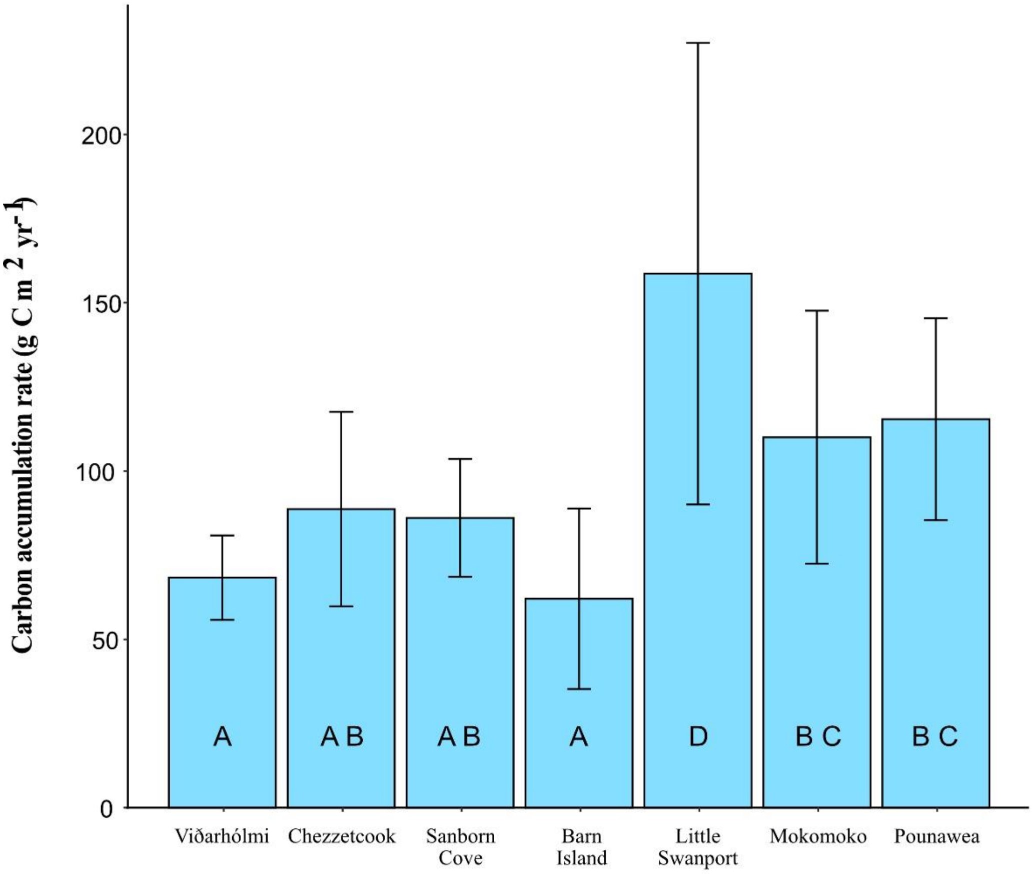


Figure 3.2. Mean carbon accumulation rate ± standard deviation at each site over the 20th Century. The first four sites (Viðarhólmi, Chezzetcook, Sanborn Cove and Barn Island) are in the Northern Hemisphere and the latter three (Little Swanport, Mokomoko and Pounawea) are in the Southern Hemisphere. Rates that do not share a letter are significantly different (ANOVA: $F = 15.2$; $p < 0.001$; Tukey post-hoc $p < 0.05$).

3.3.3 Relative sea-level rise as a driver of saltmarsh carbon accumulation

3.3.3.1 Multi-centennial timescales

Overall, the LMM for all sites across multi-centennial timescales did not show the rate of RSLR to be a significant driver of CAR. However, GLMs revealed the strength of the relationship between CAR and the rate of RSLR to be highly variable among sites. No significant relationship was identified between CAR and the rate of RSLR at Viðarhólmi, Chezzetcook, Sanborn Cove, Mokomoko and Pounawea ($p > 0.05$) and these locations will not be explored further. Significant relationships between CAR and the rate of RSLR were identified at two of the seven sites (Table 3.3).

Table 3.3. General Linear Models of carbon accumulation rate against the rate of relative sea-level rise (RSLR). Statistics include the model's AIC value, the proportion of variance in carbon accumulation rate explained by the model (% D), the test statistic (t), and the probability of deviation from the null hypothesis (p).

Location	**Model containing the rate of RSLR**			
	AIC	% D	t	p
Barn Island (USA)	47.9	32	4.8	<0.001
Little Swanport (Australia)	60.9	18	2.7	0.01

The rate of RSLR was a positive, significant predictor of CAR at Barn Island and Little Swanport (Table 3.3): CAR was enhanced during periods of higher rates of RSLR and followed a similar trend to the rates of RSLR experienced at the site (Fig. 3.3). The rate of RSLR explained a greater amount of variance in CAR at Barn Island (32 %) than at Little Swanport (18 %; Table 3.3).

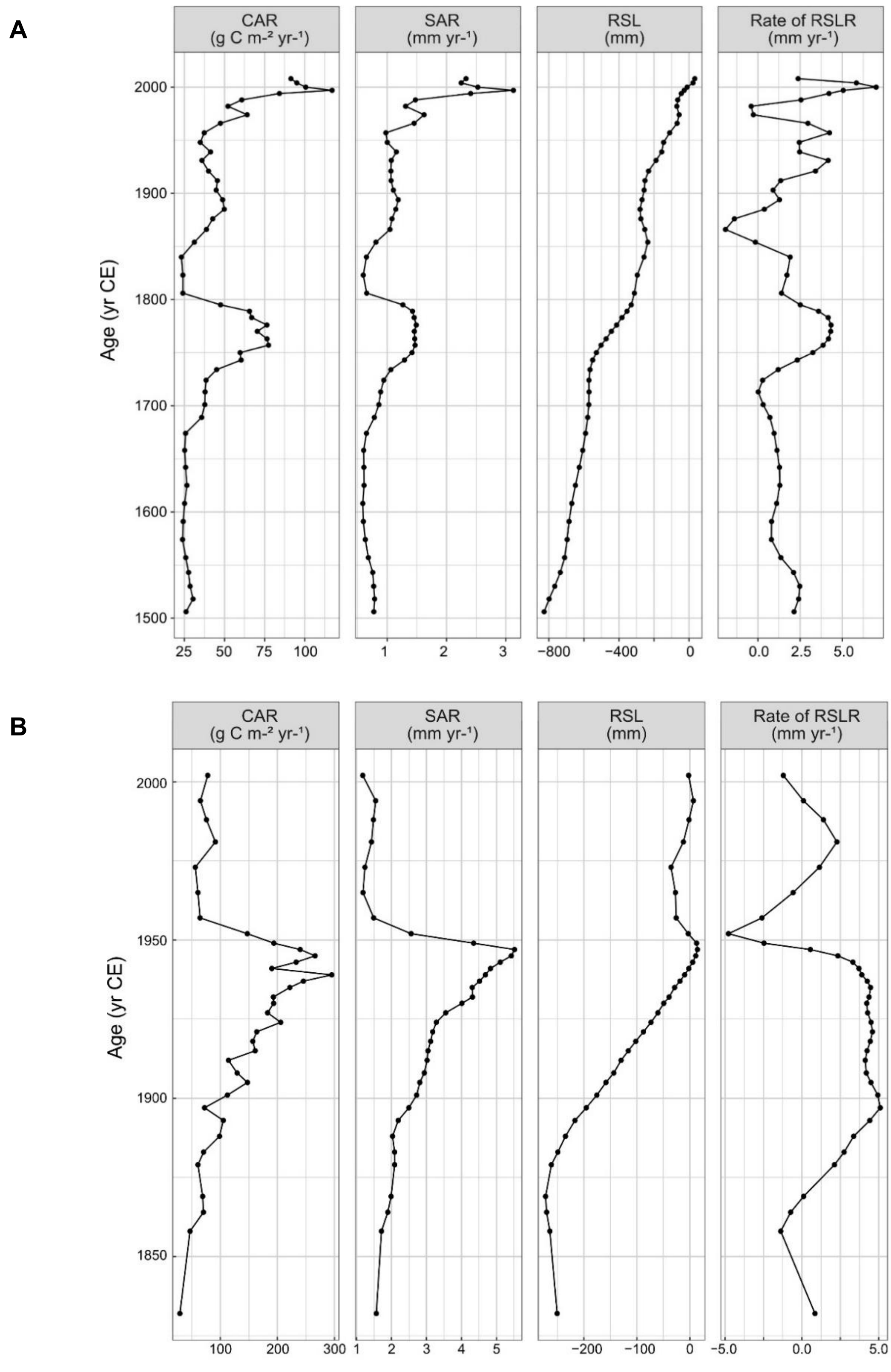


Figure 3.3. Down-core profile for Barn Island (A) and Little Swanport (B) showing carbon accumulation rate (CAR), sediment accretion rate (SAR), relative sea level (RSL) and the rate of relative sea-level rise (RSLR) at 1 cm intervals. The top of the profile indicates the sediment surface.

3.3.3.2 20th Century

The linear mixed model showed the rate of RSLR to have a positive, significant effect on CAR ($t = 3.7$; $p < 0.001$) during the 20th century, and explained 45 % of the variability in CAR across all sites (Fig. 3.4).

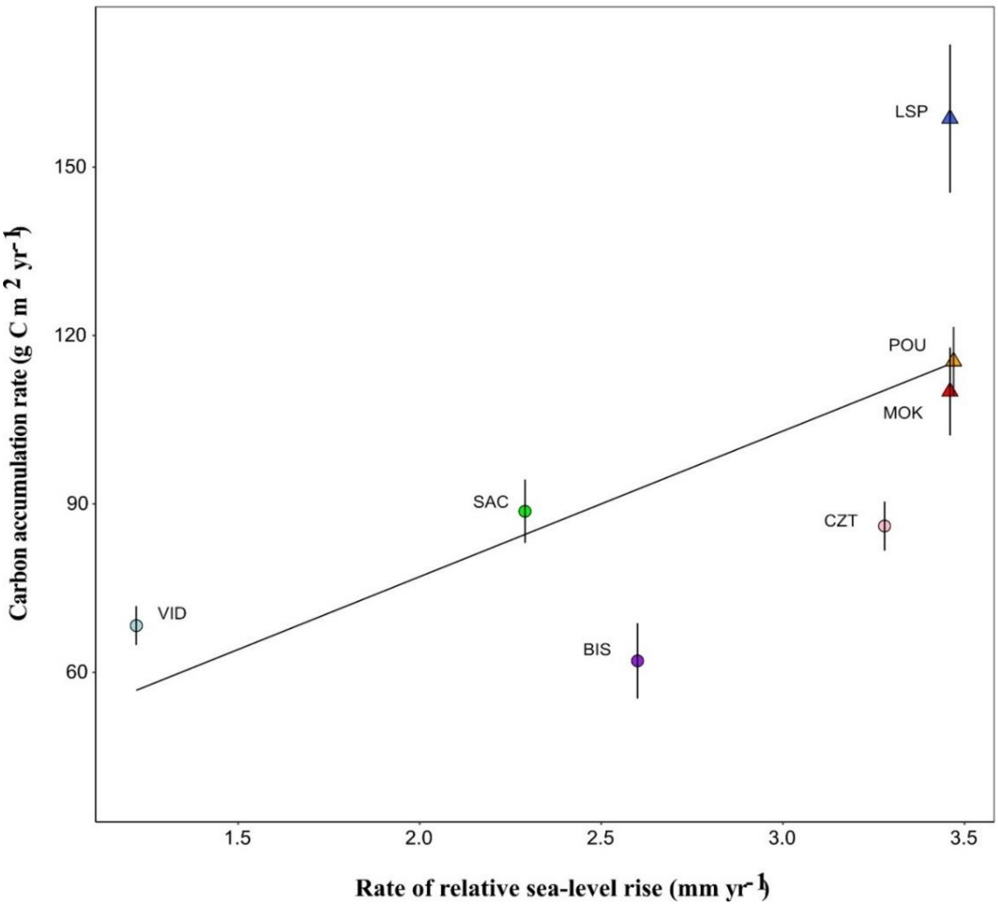


Figure 3.4. Site-averaged carbon accumulation rate (g C cm^{-2} yr^{-1}) ± SE against site-averaged rate of relative sea-level rise (mm yr^{-1}) across the 20th century. Locations in the Northern Hemisphere are displayed with a circle and locations in the Southern Hemisphere with a triangle (VID = Viðarhólmi; CZT = Chezzetcook; SAC = Sanborn Cove; BIS = Barn Island; LSP = Little Swanport; MOK = Mokomoko; POU = Pounawea).

SAR and CAR were positively correlated ($r = 0.87$). SAR was also significantly higher under higher rates of RSLR ($t = 3.8$; $p < 0.001$), with 42 % of the variability in SAR explained by the linear mixed model. During the 20th century, mean SAR and CAR were observed to be greater for locations in the Southern Hemisphere (Fig. 3.2; Appendix A3) where mean rates of RSLR were generally higher over the same time period (Table 3.2).

3.4 Discussion

Here, using location-specific, high-resolution records for saltmarshes spanning latitudinal gradients and subject to different sea-level histories, we demonstrate that the rate of relative sea-level rise was largely responsible for driving temporal variability in carbon accumulation rate. This relationship is particularly noticeable for records in the 20th century, and whilst there was evidence of the influence of RSLR on CAR over longer, multi-centennial timescales, it is highly location-specific.

The mean rate of sediment accretion and carbon accumulation across all sites in this study, on both the multi-centennial and 20th century timescales, are consistent with previously reported rates for saltmarshes at similar latitudinal bands (Wang *et al.,* 2021; Weston *et al.,* 2023). However, globally averaged rates for saltmarsh CAR are much higher than our observations (Chmura *et al.,* 2003; Ouyang & Lee, 2014). An exception is the recent, and only slightly higher, estimate of global saltmarsh CAR by Wang *et al.* (2021), in which the disparity with previous global estimates was owed to the use of an increased dataset in their study.

In corroboration with numerous earlier studies on regional, national and global scales, we find strong evidence that vertical accommodation space, as a direct result of relative sea-level rise, enhances blue carbon accumulation rates in saltmarshes, notably for CAR observed over the 20th century (Rogers *et al.,* 2019a; Breithaupt *et al.,* 2020; Ruiz-Fernández *et al.,* 2020; Wang *et al.,* 2019; Wang *et al.,* 2021; Weston *et al.,* 2023). Rates of RSLR are found to significantly regulate rates of carbon accumulation and support for this concept is strengthened through the use of locations spanning latitudinal gradients and subject to different sea-level histories.

We also observe enhanced rates of sediment accretion during periods of higher RSLR, linked to the creation of vertical accommodation space (Rogers *et al.,* 2019a). Such accretion is known to be modulated by the accumulation of both mineral and organic sediment (Wang *et al.,* 2019). However, numerous studies indicate that RSLR has a greater influence on organic rather than mineral sedimentation (Morris *et al.,* 2016; Wang *et al.,* 2019; Weston *et al.,* 2023). Indeed, sediment accretion in saltmarshes along the eastern US coast, driven by RSLR, was dominated by organic material (Weston *et al.,* 2023). This is an important finding given that global sediment supply is in decline (Weston, 2014; Ladd *et al.,* 2019). Whilst saltmarshes, and other blue carbon habitats, can capture allochthonous carbon delivered on tides, the delivery of which is generally enhanced under greater inundation periods, RSLR has also been reported to stimulate primary productivity and the production of autochthonous carbon (Morris *et al.,* 2013; Wollenberg *et al.,* 2018). Both mechanisms contribute to enhanced carbon accumulation rates under RSLR and increased sedimentation that enables saltmarsh stability under RSLR (Morris *et al.,* 2013; Wang *et al.,* 2019). Determining the mineral vs organic contribution to sediment accretion, as well as the organic carbon sources (i.e. auto- or allochthonous), would generate additional detail to the relationship between RSLR and CAR in saltmarshes.

Although rates of RSLR were shown to be a significant driver of carbon accumulation rates in the 20th century, the variability in CAR explained by the linear mixed model was marginally lower than reported for other studies (e.g., Wang *et al.,* 2021). We attribute this disparity to the inclusion of mean annual temperature, another climatic driver known to influence CAR, in models from these studies. Numerous other studies also highlight the importance of temperature in driving carbon accumulation rates. For example, warming increases primary productivity, contributing to both vertical accretion and carbon accumulation rates (Kirwan & Mudd, 2012). Conversely, on a broader geographic scale, lower temperatures, reflecting decreased rates of organic matter decomposition relative to production, enhance sediment accretion and carbon accumulation rates (Spivak *et al.,* 2019; Weston *et al.,* 2023). Suspended sediment supply also represents an important non-climatic driver of CAR over spatial and temporal scales, and contributes to saltmarsh resilience to RSLR via sediment accretion (Kirwan *et al.,* 2016; Weston *et al.,* 2023). The inclusion of such climatic and non-climatic drivers, including temperature and sediment supply, in future studies on decadal-centennial timescales, alongside rates of RSLR, may explain greater variability in observed saltmarsh CAR.

Contrary to Wang *et al.* (2021), we did observe significant differences in CAR between saltmarshes located in the Northern and Southern Hemisphere over the 20th century; rates of sediment accretion and carbon accumulation were higher for saltmarshes in Australia and New Zealand than those in the North Atlantic. This was linked to greater acceleration in the rate of RSLR at the Southern Hemisphere sites compared to those in the Northern Hemisphere over the same time period, a trend also identified by previous sea-level reconstructions (Gehrels *et al.* 2012; Dangendorf *et al.* 2019).

Current understanding of the impacts of sea-level rise on coastal wetlands tends to come from studies concentrated in the Northern Hemisphere, notably Europe and North America (Rogers *et al.,* 2019a), perhaps due to the geographical bias of tide-gauge stations here and a poorer coverage of longer-term tide-gauge records in the Southern Hemisphere (Hogarth, 2014). However, coastal wetlands in the Southern Hemisphere are considered to have significant and long-term climate change mitigation potential in the face of predicted global sea-level rise. For example, Rogers *et al.* (2019a) noted that regions where accommodation space has previously been constrained by relatively stable sea level over the past millennia, which they reported to have occurred in much of the Southern Hemisphere, have the potential for significant increases in CAR under RSLR. Furthermore, using projections under the Intergovernmental Panel on Climate Change (IPCC) moderate and high emission scenario pathways, Wang *et al.* (2021) predicted Australia to be a hot-spot for blue carbon accumulation in the 21st century. This was attributed to an expansion in coastal wetland extent driven by RSLR (Wang *et al.,* 2021). A greater research focus on the dynamics between RSLR and CAR for coastal wetlands located in the Southern Hemisphere is thus needed.

Although evident, support for the concept that RSLR enhances blue carbon accumulation in saltmarshes was less convincing over longer multi-centennial timescales in our study. Proxy sea-level records have uncertainties associated with chronology and relative sea level which produce reconstructions less precise, but covering a longer-time period than instrumental records such as tide-gauges (Cahill *et al.,* 2015). Such uncertainties tend to increase with sediment core depth, presenting a potential limitation in using proxy sea-level records in high-resolution analysis, such as in our study. Proxy and instrumental sea-level datasets have been shown to be comparable over the 20th century (Gehrels & Woodworth, 2013), where we observed strong support for rates of RSLR regulating CAR. Other factors previously discussed, such as temperature and sediment supply, may also represent more important driving forces for carbon accumulation over multi-centennial timescales. However, two out of the seven locations, spanning the past 500 (Barn Island) and 181 years (Little Swanport), showed CAR were driven by rates of RSLR. At both locations, sediment accretion rate also increased with RSLR, further supporting the hypothesis that the creation of vertical accommodation space under enhanced RSLR drives carbon accumulation.

The organic carbon content of saltmarsh soil is a balance between organic matter inputs and carbon loss, mostly through decay and supplemented by exportation (Macreadie e*t al.,* 2013; Macreadie *et al.,* 2017b; Weston *et al.,* 2023). Only a small amount of organic material will be preserved over centennial to millennial timescales, whilst the remaining labile fractions are lost to remineralisation (Davis *et al.,* 2015). With soil age and depth, the contribution of labile carbon decreases and, as a result, the apparent carbon accumulation rate also decreases (Davis *et al.,* 2015; Weston *et al.,* 2023). This may result in an overestimation of CAR. A two-pool modelling approach that adjusts carbon accumulation rates to account for faster-cycling organic carbon pools in shallower soil may provide a suitable solution, and has already been successfully applied in multi-centennial carbon accumulation studies in blue carbon ecosystems (Belshe *et al.,* 2019).

3.4.1 Conclusion

Overall, we provide strong evidence to support the theory that relative sea-level rise regulates carbon accumulation rates in saltmarshes, due to the creation of vertical accommodation space for sediment accretion. This was observed at locations around the world using high-resolution records, and was particularly notable over the 20th century. Whilst there was some supporting evidence over multi-centennial timescales, the relationship between RSLR and CAR was highly location-specific and may in part be attributed to the uncertainty in proxy records over such timescales.

We suggest that further research into the identification of mineral vs organic sediment accretion, sources of carbon and the inclusion of a greater number of drivers alongside the rate of RSLR, such as temperature and sediment supply, will help to create a clearer picture of carbon accumulation rates over decadal-centennial and multi-centennial timescales. The use of two-pool modelling is also encouraged to allow for the distinction between labile and refractory carbon in CAR. Furthermore, a greater research focus is required for saltmarshes in the Southern Hemisphere, given predictions of their globally relevant, enhanced CAR capacities under future sea-level rise. Finally, while we have added to the growing evidence base that RSLR can enhance blue carbon accumulation rates, it is important to note that the overall strength in coastal wetland CAR has declined due to losses in habitat extent worldwide (Duarte, 2017), urging support for ongoing coastal wetland conservation and restoration efforts worldwide.

Chapter 4 of 7

The role of sediment accretion in developing saltmarsh carbon stocks under relative sea-level rise: a case study from Maine, USA

This is an experimental chapter which uses previously established marker horizons to measure saltmarsh vertical accretion rates, and carbon stocks at sites in Maine, USA, over two time periods (1986 – 2003; and, 2003 – 2019). These periods represent relatively slower and more rapid rates of relative sea-level rise (RSLR) over the last 33 years, respectively, and allow a unique opportunity to test the overarching hypothesis that saltmarsh carbon stocks will be greater under faster rates of relative sea-level rise. The coastline of Maine provides a natural laboratory for studies relating to the influence of RSLR and tidal range on saltmarshes and will be discussed in more detail later in the chapter.

Fieldwork for this chapter took place in July 2019. This research expands upon previous work by Wood *et al.* (1989), who placed brick dust marker horizons on the surface of saltmarshes along the coast of Maine in 1986 for the purpose of monitoring sedimentation, and Goodman *et al.* (2007) who relocated a number of horizon layers 17 years later, in 2003. This chapter presents findings on vertical accretion and carbon stocks in relation to RSLR, and discusses the use of marker horizons in studies on coastal wetlands.

4.1 Introduction

Coastal wetlands, including saltmarshes, mangroves and seagrasses, are highly productive habitats capable of exceptionally high carbon burial rates (Duarte *et al.,* 2013). The vast majority of captured carbon is then preserved in sediments for long timescales, up to millennia (McLeod *et al.,* 2011). High carbon stocks associated with coastal wetlands, or blue carbon habitats, are in part attributed to sediments that can extend several metres deep (Howard *et al.,* 2014), and continue to accumulate carbon without reaching saturation (Macreadie *et al.,* 2017b).

Both organic material and inorganic, mineral material contribute to sediment accumulation in coastal wetlands (Woodroffe *et al.,* 2016). In areas of relatively low sediment supply, autochthonous inputs are greater; this includes the incorporation of root biomass and aboveground plant litter into the sediment, both of which are rich in organic carbon (Howard *et al.,* 2014). Allochthonous carbon can also accumulate in sediments owed to vegetation canopies trapping particles and organic matter delivered on tides and/ or rivers (Kristensen *et al.,* 2008; Kennedy *et al.,* 2010). The high trapping efficiency of coastal wetland vegetation also encourages the deposition of inorganic, mineral sediment which contributes to sediment accumulation (Fagherazzi *et al.,* 2012), although does not directly enhance organic carbon stocks. Carbon accumulation in coastal wetlands, whether produced in-situ (autochthonous) or originating from another habitat (allochthonous), is therefore a function of sedimentation rates (Saintilan *et al.,* 2013), which in turn supports vertical accretion and resilience to RSLR (Morris *et al.,* 2016). Accurate measurements of sedimentation are therefore key to saltmarsh studies interested in the impact of RSLR and the development of blue carbon stocks.

The use of marker horizons has been widespread in studies of sedimentation in coastal wetlands, including the blue carbon habitats of saltmarshes and mangroves (van Wijnen & Bakker, 2001; McKee, 2011; Saintilan *et al.,* 2013; Swales & Lovelock, 2020). This direct method of measuring vertical accretion involves the use of material, such as feldspar, sand, or brick dust, applied to the soil to create a reference layer, known as a marker horizon (Nolte *et al.,* 2013; Villa & Bernal, 2018). Over time, and generally at regular intervals, soil cores are extracted and the accumulation of mineral and organic sediment is measured relative to the upper surface of the marker horizon (Nolte *et al.,* 2013; Villa & Bernal, 2018; Swales & Lovelock, 2020).

Studies of vertical accretion using marker horizons have the advantage of relatively low material costs and sampling effort (Cahoon & Turner, 1989), as well as the ability to provide high-resolution measurements in dynamic coastal environments (Swales & Lovelock, 2020). However, limitations in this method have been reported, particularly in areas where bioturbation is prevalent (Krauss *et al.,* 2010; Swales & Lovelock, 2020). As a result, marker horizons tend to be restricted to measurements of vertical accretion over shorter timescales from months to decades (Nolte *et al.,* 2013).

Whilst saltmarsh accretion rates are frequently compared to rates of RSLR to determine saltmarsh resilience, or survival (van Wijnen & Bakker, 2001; Goodman *et al.,* 2007), it also provides an important indication of blue carbon stocks (Saintilan *et al.,* 2013; Rogers *et al.,* 2019a). This study reports on vertical accretion rates for saltmarshes in Maine, USA, using brick dust marker horizons established in 1986 (Wood *et al.,* 1989), and previously measured in 2003 (Goodman *et al.,* 2007). These measurements, plus additional measurements from 2019, were used to explore the relationship between RSLR, sediment accretion, and the development of blue carbon stocks in saltmarshes along the coast of Maine.

The coastline of Maine is subject to significant rates of RSLR (Moser, 2006), which have been more rapid in recent years (Ray & Talke, 2019). However, rates of RSLR and tidal range vary greatly along the coast, as shown by well documented tide gauges (Holgate *et al.,* 2013; Fig. 4.2). Rates of relative sea-level rise are faster in northeastern Maine compared to the south-west and the disparity has increased in recent decades (Holgate *et al.,* 2013; Fig. 4.3). Likewise, mean tidal range increases steadily from the south-west of coastal Maine (Portland tide gauge: 2.8 m) to the north-east (Eastport tide gauge: 5.6 m; Fig. 2; NOAA, 2022), towards the Bay of Fundy, located on the northeastern end of the Gulf of Maine, an area known to have among the highest tidal ranges in the world of up to 16m in the Minas Basin (Archer, 2013; Ashall *et al.,* 2016). The coast of Maine therefore provides a natural laboratory for studying the influence of RSLR and tidal range on sedimentation rates and blue carbon stocks in saltmarshes.

Differing rates of RSLR and tidal range will have impacts on marsh accretion rates as sedimentation in coastal wetlands is positively correlated with inundation frequency and duration (Pethick, 1981). Rogers *et al.* (2019a) demonstrated that higher accumulation rates of mineral and organic material, and associated carbon, in coastal wetlands were the result of enhanced inundation periods due to RSLR. If rates of RSLR are too rapid, however, accretion rates may not be able to keep pace, leaving saltmarshes, and their capacity for carbon storage, vulnerable (Kirwan & Mudd, 2012; Thorne *et al.,* 2018).

This study sets out to measure vertical accretion rates, rates of RSLR and blue carbon stocks for saltmarsh sites across Maine, USA, during two distinct time periods: between 1989 and 2003, the 17 year period from when brick dust plots were originally established (Woods *et al.,* 1989) and then relocated by Goodman *et al.* (2007); and between 2003 and 2019, a 16 year period which represents a rapid increase in RSLR compared to previous decades. This study also aims to assess the effectiveness of marker horizons for studies in excess of 30 years. In turn, this research is underpinned by three hypotheses: (1) that saltmarsh accretion rates are higher in the more recent time period due to increased rates of RSLR; (2) that blue carbon stock is greater when saltmarshes have a greater accretionary balance with rates of relative sea-level rise; and, (3) that sites that experience higher tidal range will have greater carbon stock.

4.2 Methods

4.2.1 Study background

In 1986, circular brick dust layers (diameter: 330 mm; maximum particle size: 1.70 mm), containing metal discs (diameter: 44.5 mm; thickness: 1.6 mm) in the centre, were placed on the surface of 26 saltmarsh sites along the coast of Maine (Wood *et al.,* 1989). The purpose was to act as marker horizons to measure future accretion rates. Each site had a minimum of 6 brick dust plots located in representative areas of the low and mid marsh, beside a metal stake (Wood *et al.,* 1989; Fig. 4.1). Sediment accretion rates were then observed at 24 sites by Wood *et al.* (1989) after a one-year period in 1987.

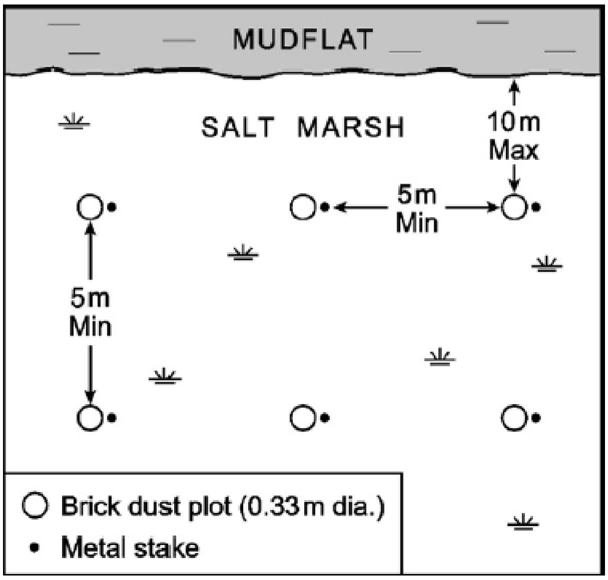


Figure 4.1. Exemplar sampling design showing 3 replicate plots in the low and mid-marsh, with 6 sampling plots in total where brick dust layers were emplaced (Wood *et al.,* 1989).

Brick dust marker horizon plots were relocated at 11 sites in 2003, aided by metal detection and detailed site maps (Goodman *et al.*, 2007). From these, annual sedimentation rates were reported in relation to relative sea-level rise. In turn, these rates were used to determine whether marshes were in a positive or negative accretion balance. Of the eleven sites, seven had a positive balance in relation to relative sea-level rise (Goodman *et al.,* 2007). The majority of studied marshes were therefore keeping pace with sea-level rise, but predictions by Goodman *et al.* (2007) suggested that at rates exceeding ~ 4 mm yr^{-1}, many of the marshes would be in an accretion deficit.

This study seeks to expand upon previous studies by comparing accretion rates from 1986 – 2003 with rates in 2003 – 2019. Accretion rates are then compared to RSLR, and the influence this has had on carbon stocks that have accumulated over the 33 year study period is explored.

4.2.2 Study area

Maine is situated on the east coast of the USA and the approx. 5,970 km long coastline is bordered by the Gulf of Maine, an inlet from the Atlantic Ocean (Jacobson *et al.,* 1987). Saltmarshes are well established and, as of 1987, account for approximately 20 % of the coastline (Jacobson *et al.,* 1987), although Kelley *et al.* (1988) noted that marshes are more common in areas with a lower tidal range and rates of RSLR. Saltmarshes along Maine's convoluted coastline are, therefore, subject to differing rates of RSLR, and meso- to macrotidal ranges, both of which are well documented by tide-gauge stations (Kelley *et al.,* 1988; Goodman *et al.,* 2007; Fig. 4.2). The distinct vegetation zones found in saltmarshes along Maine's coast are reflective of marshes across New England. The low marsh is dominated by *Spartina alterniflora*, whilst *Spartina patens* dominates the mid-marsh zone (MacKenzie & Dionne, 2008). A saltmarsh rush, *Juncus gerardii*, usually dominates the uppermost, or high, marsh zone (Gehrels, 1994). However, in this study we only focus on the low and mid-marsh zones, due to the location of marker horizons established in 1986 (Wood *et al.,* 1989).

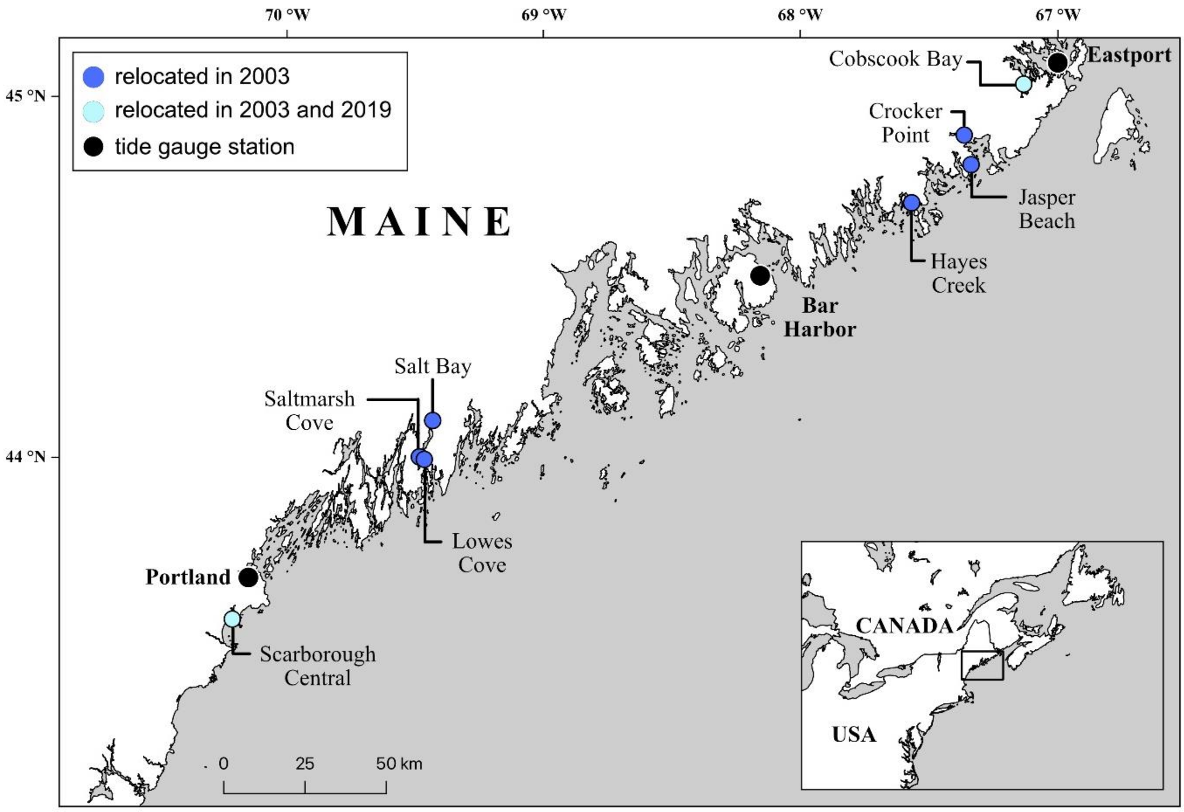


Figure 4.2. Location of saltmarsh study sites along the coast of Maine where marker horizons were relocated in 2003 (dark blue circles) and 2019 (light blue circles). Tide gauge stations with long-term rates of relative sea-level rise from 1986 are also shown (black circles).

Eight mature saltmarsh sites along the coast of Maine, where marker horizons were successfully relocated in 2003, were chosen for this study: Scarborough Central (SC), Saltmarsh cove (SMC), Salt Bay (SB), Lowes Cove (LC), Hay Creek (HC), Jasper Beach (JB), Crocker Point (CP), and Cobscook Bay (CB; Fig. 4.2). Although brick dust plots were successfully relocated in three other sites in 2003, these sites were not included in the study.

Attempts were made to relocate brick dust marker horizons at all sites, using metal detectors and detailed site maps, in addition to GPS coordinates taken nearby to plots in 2003. When relocation was successful, sediment cores were then extracted. Figure 4.3 provides a visualisation of a relocated brick dust marker horizon in a sediment core, where the depth between the core surface (representing 2019) and marker horizon (representing 1986) can be observed. When relocation was not possible, efforts were still made to take sediment cores without marker horizons. At these sites, a sampling transect was set up perpendicular to the marsh edge, with two sampling plots: one in the low-marsh, and another in the mid-marsh. A second transect was set up, with a minimum distance of 5 m between transects, to provide sediment sample replicates. This was reflective of the original sampling design by Wood *et al.* (1989) (Fig. 4.1), but replicates were restricted due to transport considerations. When either low-marsh, or mid-marsh, zones were not present at a site, replicates were only taken from present vegetation zones. More detail is provided later in this chapter. All fieldwork was carried out in July 2019.

Figure 4.3. Relocated brick dust marker horizon (red) in a sediment core extracted using a gouge auger (site: Scarborough Central). Relatively sharp upper and lower boundaries of the brick dust can be seen. The depth between the marsh surface (2019) and the top of the brick dust layer (1986) provided a sediment accretion rate, which was calculated across an average of 10 measurements.

4.2.3 Data collection

4.2.3.1 Rate of sea-level rise and tidal range data

Annual rates of RSLR were calculated using tide gauge data from the following stations along the coast of Maine: Portland, Bar Harbor, and Eastport (Holgate *et al.,* 2013; Permanent Service for Mean Sea Level (PSMSL) 2022; Fig. 4.2). Other tide gauge stations exist but do not provide long-term rates of RSLR that cover the study period between 1986 and 2019. Mean sea level data from each station is provided in relation to a local reference datum, and data was gathered for 2 time periods: 1986 – 2003, and 2003 – 2019. Linear regressions (*R version 4.1.2*) were used to generate rates of relative sea-level rise (mm yr^{-1}) for each time period (Fig. 4.4).

With the exception of Scarborough Central, all other study sites fell between Portland tide gauge station in the south-west, Bar Harbor station, and Eastport station in the north-east (Fig. 4.2). Rates of RSLR for those study sites were linearly interpolated based on distance between the two closest stations. For Scarborough Central, rates of RSLR gathered from Portland tide gauge station were used without interpolation due to close proximity, approximately 12 km away (Fig. 4.2).

Data for tidal range was obtained from multiple water level and tide gauge stations along the coast (National Oceanic and Atmospheric Administration (NOAA), 2022). Stations were selected based on proximity to study sites, and data monitoring post-2003. Tidal range is shown to increase from the most south-west station [Wells] to the most north-east station [Eastport] (Table 4.1). For each study site, tidal range was interpolated based on distance between the two closest stations.

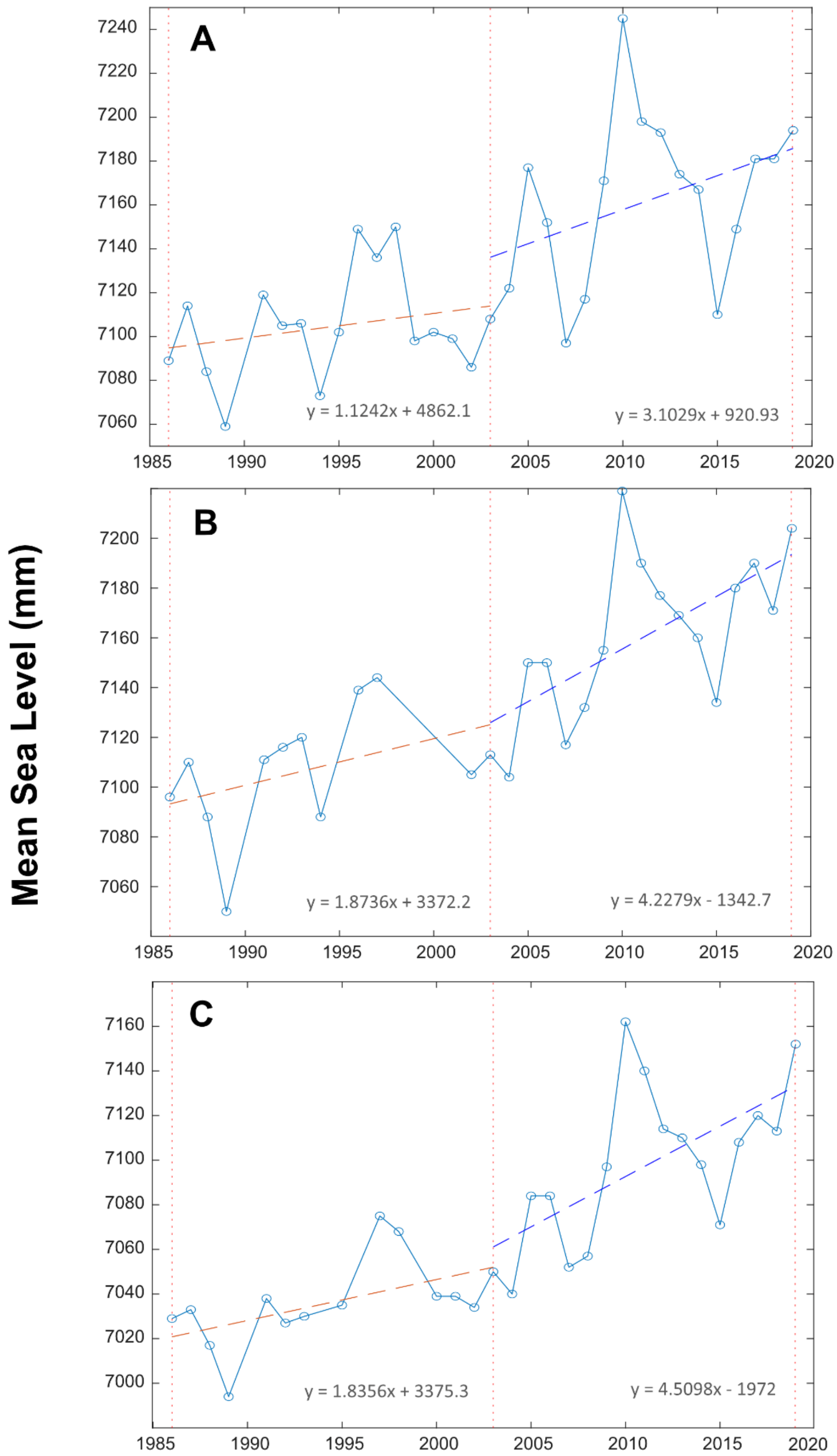


Figure 4.4. Mean sea level (mm), in relation to a local datum, from tide gauge stations at Portland (A), Bar Harbor (B), and Eastport (C) over the duration of our study period. Rates of relative sea-level rise with linear regression equations are shown for the period between 1986 – 2003 (orange dashed line), and 2003 – 2019 (blue dashed line). Data obtained from PSMSL, 2022 (Holgate *et al.,* 2013).

Table 4.1. Water level and tide gauge stations located along the coast of Maine with corresponding tidal range (m). Stations are presented in order from south-west to north-east. [Data obtained from NOAA, 2022].

Station	Tidal range (m)
Wells	2.68
Portland	2.78
Wiscasset	2.89
Bar Harbor	3.22
Cutler Farris Wharf	4.18
Gravelley Point	5.46
Eastport	5.59

4.2.3.2 Sediment accretion rates and sediment cores

When brick dust marker horizons were relocated, sediment cores (width: 5 cm) were extracted from the marsh surface to the marker horizon using a gouge auger (Fig. 4.3). Ten measurements were taken from the marsh surface to the top of the marker horizon in the sediment core and averaged to provide amount of sediment accumulated (mm) for the plot. Accretion rate per year was then calculated as follows:

$$\text{Sediment accretion rate (mm yr}^{-1}\text{)} = \frac{sediment\ accumulated\ (mm)}{time\ period\ (yr)}$$

where:

time period (yr) = number of years since the brick dust was laid (e.g. 2019 – 1986 = 33)

Sediment accretion rates reported by Goodman *et al.* (2007), which were measured from small sediment plugs, were used to identify the depth of the 2003 horizon (Appendix B1). In turn, this provided two sub-samples from each sediment core with a relocated marker horizon, representing time periods between 1986 – 2003 (slower rates of RSLR), and 2003 – 2019 (faster rates of RSLR). Saltmarsh accretion balance was then calculated for each sub-sample by taking the rate of RSLR away from average sediment accretion rates for each time period. At sites where marker horizons were not relocated, short sediment cores (width: 5 cm; depth: 20 cm) were extracted at sampling points along transects. Based on sediment accretion rates reported by Goodman *et al.* (2007) between 1986 – 2003, and measurements taken from this study, these sediment cores likely cover the late 20th century to present day (2019), although the specific timeframe cannot be specified. In order to generate two sub-samples representing a period of slower and more rapid rates of RSLR, there was an intention to use known rates of sediment accretion from relocated marker horizons in this study to identify the 2003 horizon. However, there was no obvious pattern to allow an accurate calculation. Instead, extrapolations, based on a constant sediment accretion rate with those observed from relocated plots between 1986 – 2003 (Goodman *et al.,* 2007), were used to estimate the 2003 horizon, and in turn, the 1986 horizon. This was achieved by matching sampling points in the same marsh zone and site with marker horizon plots that were relocated in 2003 by Goodman *et al.* (2007). Whilst I do not believe that the sites would experience a constant sediment accretion rate across both time periods, this provided two sub-samples which cover unknown durations, but the lower sub-sample represents a period of slower RSLR than the upper sub-sample. For the sites where marker horizons were not relocated, the number of replicates vary across study sites based on the number of relocated plots found at each site in 2003.

All sediment cores were kept in cold stores at various institutions across Maine before being transported back to the University of York, UK, for sedimentary analysis. Logistics placed limitations on the number of sediment core replicates that were able to be taken during fieldwork.

4.2.3.3 Measurement of sediment characteristics

Two sub-samples from each core (representing the periods of slower and faster sea-level rise) were oven-dried (60 °C, minimum 24 hrs), and dry bulk density was calculated by dividing the dry weight of sediment by sample volume. Dried samples were then ground, and sieved (1 mm mesh size) to remove large inorganic particles. In preparation for total organic carbon (TOC) analysis, samples were then homogenised to a fine powder using a ball mill (Retsch MM200; 250µm, 2 mins), treated with 10 % HCl acid to remove carbonates, and oven-dried (60 °C, overnight). TOC was measured using a Carbon and Nitrogen elemental analyser (Flash 2000, Thermo Fisher Scientific). The instrument was calibrated using aspartic acid as a standard.

4.2.3.4 Calculation of organic carbon stock

Organic carbon sediment stocks were calculated using dry bulk density (g cm^3), depth interval of the sub-sample (cm), and total organic carbon (%). Calculations were based on the approach outlined by the Coastal Blue Carbon Manual (The International Blue Carbon Initiative; Howard *et al.,* 2014). Soil organic carbon density was determined using the following calculation:

$$\text{Soil organic carbon density } (\text{g C cm}^{-3}) = \text{Dry bulk density } (\text{g cm}^{-3}) * \frac{\text{organic carbon } (\%)}{100}$$

Organic carbon content for each sample was then calculated by multiplying soil organic carbon density by the depth interval of the sub-sample (Appendix B1; B2). From this, organic carbon stocks (reported as Mg C ha^{-1}) were generated for each sample.

4.2.4 Data analysis

Statistical analyses were conducted using Rstudio software (*R version 4.1.2*). Sites with relocated marker horizons (n = 6) were analysed in a separate dataset to sites with extrapolated marker horizons (n = 30). The small sample size of the relocated marker horizon dataset requires that analysis results are treated with caution, as the power of statistical tests will be reduced (Button *et al.,* 2013). However, studies of low statistical power are mostly considered to reduce the chance of detecting a true effect, as type II errors have an inverse relationship with statistical power (Button *et al.,* 2013).

Differences in carbon stock between sites with extrapolated marker horizons were tested using a One-way ANOVA, and significant differences were identified with a post-hoc Tukey test. The assumptions of normality and homoscedasticity were met. Linear regression models were used to determine the relationship between carbon stock and a set of predictor variables (sediment accretion rate; rate of RSLR; and tidal range) for the two time periods (representing slower and faster rates of RSLR). The response variable (carbon stock) was normally distributed. Model residuals were checked for normality, homoscedasticity and leverage using diagnostic plots, supported by the Shapiro-Wilk test for normality, and Breusch-Pagan test for heteroscedasticity. To meet the assumption of homoscedasticity for the extrapolated model, the response variable (carbon stock) was log transformed. Backwards-forwards stepwise reduction was used to create an optimal model. Prior to analysis, predictor variables were tested for correlation with other predictors, and with the response variable (organic carbon stock), using Pearson's coefficient (r). Correlation was considered high when $r \geq 0.6$. Where predictor variables displayed high intercorrelation, only the variable with the strongest univariate correlation to the response variable was included in the model. The "relaimpo" package (Grömping, 2006) was used to identify the deviance (%) explained by the model, and independent effects (%) that each predictor variable contributed.

4.3 Results

Here, the number of marker horizon plots relocated during this study and in previous studies, since 1986, are reported. The remainder of the results are then discussed in two separate sections – firstly, for sites with relocated marker horizon plots; and secondly, for sites where relocated was not possible and extrapolated sediment accretion values were used. Sediment accretion rates and carbon stocks are presented for saltmarsh study sites across Maine (Fig. 4.2). Carbon stock is then analysed against sediment accretion rate, tidal range and rate of RSLR between two time periods representing slower and faster rates of RSLR. For relocated marker horizons, the exact duration these time periods represent are known. Finally, the relationship between saltmarsh accretionary balance with RSLR and carbon stock is also explored.

4.3.1 Marker horizon relocation

In the original study in 1986, a minimum of 6 brick dust marker horizon plots were established across 26 saltmarsh sites (Fig. 4.5). In this study, only three marker horizon plots, across two sites, were relocated in 2019 (Fig. 4.5). This is compared to fieldwork in 2003 where marker horizons were relocated in 11 sites (Goodman *et al.,* 2007), and in 1987, a year after being established, where relocations occurred in 24 sites (Wood *et al.,* 1989; Fig. 4.5). The number of sites at which plots were relocated showed a very strong, negative correlation with time since establishment ($r = -0.97$). Of the three marker horizons relocated in 2019, two were in Scarborough Central, south-west Maine, and one in Cobscook Bay, in the north-east (Fig. 4.2).

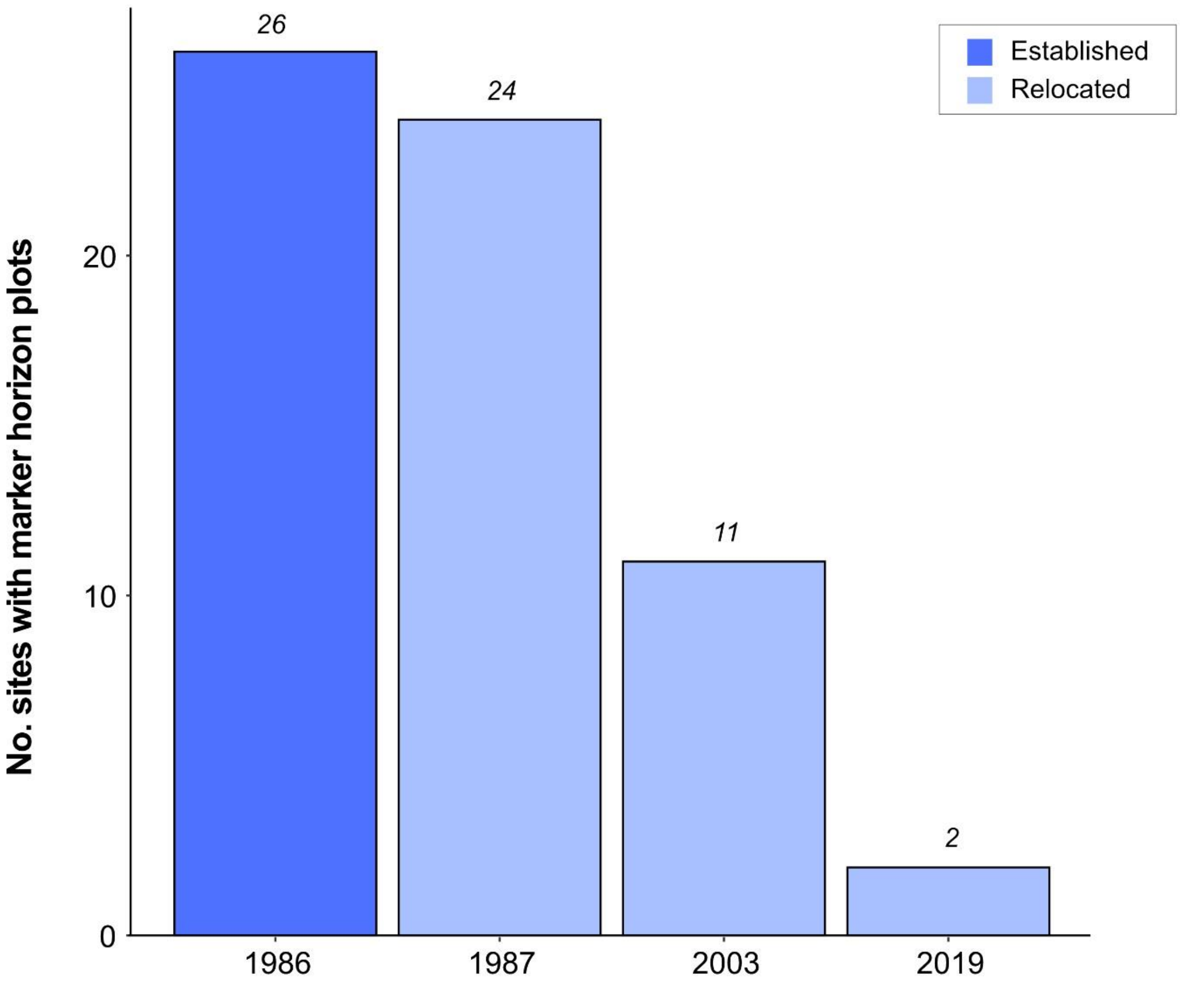


Figure 4.5. Number of saltmarsh sites along the coast of Maine where marker horizons were established (dark blue: 1986), or relocated (light blue: 1987; 2003; 2019). Data was extracted from Wood *et al.* (1989) for 1986 and 1987, Goodman *et al.* (2007) for 2003, and this study for 2019.

4.3.2 Sites with relocated marker horizons

4.3.2.1 Rates of relative sea-level rise and tidal range

Study sites where marker horizons were relocated, Scarborough Central and Cobscook Bay, were at an equivalent distance of ~ 20 km to their closest tide gauge station (Portland and Eastport, respectively), both of which were to the north-east of the sites (Fig. 4.2). For the sake of consistency, these sites used rates of RSLR from Portland and Eastport tide gauge stations without interpolation, as Scarborough Central did not have a tide gauge station to the south to interpolate between.

Rates of RSLR show an increase since marker horizons were established in 1986 (Table 4.2). At both sites, rates were higher in the most recent time period from 2003 – 2019 (Table 4.2), but, overall, the rate of RSLR has been higher at Cobscook Bay in north-east Maine, represented by the Eastport tide gauge station, across all time periods when compared to Scarborough Central in south-west Maine, represented by the Portland tide gauge station (Fig. 4.2; Table 4.2).

Table 4.2. Rates of relative sea-level rise (mm yr^{-1}) obtained via a linear regression on mean sea level data from Portland and Eastport tide gauge stations (Permanent Service for Mean Sea Level (PSMSL); Holgate et al., 2013). Rates are shown across the two time periods when marker horizons were relocated at Scarborough Central [Portland] and Cobscook Bay [Eastport] (1986 – 2003; and, 2003 – 2019).

	Rate of relative sea-level rise (mm yr^{-1})	
Time period	*Scarborough Central* [Portland]	*Cobscook Bay* [Eastport]
1986 – 2003	1.12	1.84
2003 – 2019	3.10	4.51

In the 17 year period between 1986 and 2003, the rate of RSLR at Cobscook Bay (1.84 mm yr^{-1}) was 0.72 mm greater than the rate at Scarborough Central (1.12 mm yr^{-1}; Table 4.2). In the period between 2003 and 2019, when rates of RSLR were higher across the coast, Cobscook Bay had a rate 1.41 mm higher than at Scarborough Central (Table 4.2). This is a two-fold increase in the disparity between rates of RSLR for the sites between 1986 and 2003, when rates were lower (Table 4.2).

Tidal range at Cobscook Bay (5.47 m) was two-fold greater than at Scarborough Central (2.75 m) over the duration of the study period. Tidal range was interpolated between Wells and Portland for Scarborough Central; and, between Gravelly Point and Eastport for Cobscook Bay (Table 4.1).

4.3.2.2 Sediment accretion rates and carbon stocks

Over the 33 year study period, the plot at Cobscook Bay had the greatest sediment accretion rate (3.82 mm yr^{-1}), compared to the two relocated plots at Scarborough Central (3.18 [plot 1] and 2.33 [plot 2] mm yr^{-1}). At Scarborough Central, relocated plots had a greater sediment accretion rate in the most recent time period (2003 – 2019), when rates of relative sea-level rise were higher (Table 4.2; Table 4.3). In contrast, the sedimentation rate at Cobscook Bay was lower under greater rates of RSLR (Table 4.2; Table 4.3).

All relocated plots at Scarborough Central (1.59 [plot 1] and 1.00 [plot 2] mm yr^{-1}) and Cobscook Bay (2.45 mm yr^{-1}) were in a positive accretion balance with the rate of RSLR in the period between 1986 – 2003 (Table 4.2; Table 4.3). Although the accretion balance at plot 1 in Scarborough Central was lower in the period between 2003 – 2019 when rates of RSLR were higher (Table 4.2), it remained in a positive balance (0.59 mm yr^{-1}; Table 4.3). In contrast, plot 2 was in a slight negative accretion balance (-0.54 mm yr^{-1}) with the rate of RSLR during the later time period (Table 4.3). Portland tide gauge station is approximately 20 km to the north of Scarborough Central, and rates of RSLR are known to increase up the coastline. It is possible that plot 2 was in a positive accretion balance with the rate of RSLR experienced at Scarborough Central, rather than Portland, but there is no available mean sea level data from tide gauge stations further south over the study period to interpolate from. Furthermore, if sediment accretion rates were averaged between both relocated plots, the resultant rate would be in a positive balance with the rate of RSLR across both time periods. Regardless, both plots at Scarborough Central had a higher sediment accretion rate in the period between 2003 and 2019 when rates of RSLR were higher (Table 4.2; Table 4.3). However, the plot at Cobscook Bay had a lower sediment accretion rate in the period between 2003 and 2019, and was in a negative accretion balance (-1.20 mm yr^{-1}; Table 4.3). Under interpolated rates of RSLR between Bar Harbor and Eastport tide gauges (2003 – 2019: 4.48 mm yr^{-1}), the plot would still remain in a negative accretion balance (-1.17 mm yr^{-1}).

Mean carbon stock at Cobscook Bay (20.27 ± 2.16 Mg C ha^{-1}) was higher than the mean carbon stock at Scarborough Central (18.60 ± 3.77 Mg C ha^{-1}) across the whole study period (1986 – 2019). However, there is high variability in carbon stock between the two plots at Scarborough Central (Table 4.3), and when considered separately, mean carbon stock at plot 1 (21.26 ± 3.87 Mg C ha^{-1}) was higher than at Cobscook Bay. Furthermore, carbon stock varied with time period and site (Table 4.1). Between 1986 – 2003, carbon stock was highest at Cobscook Bay (21.80 Mg C ha^{-1}). Whereas between 2003 – 2019, carbon stock was highest at plot 1 in Scarborough Central (24.00 Mg C ha^{-1}; Table 4.3).

Table 4.3. Relocated marker horizon plots (n = 3) at Scarborough Central and Cobscook Bay showing sediment characteristics for the two time periods of the study, and accretion balance with rates of RSLR.

Site	Time period	Sediment accretion rate (mm yr^{-1})	Dry bulk density (g cm^{3})	Carbon stock (Mg C ha^{-1})	Accretion balance (mm yr^{-1})
Scarborough Central_1	1986 – 2003	2.71	0.46	18.52	1.59
	2003 – 2019	3.69	0.51	24.00	0.59
Scarborough Central_2	1986 – 2003	2.12	0.57	14.04	1.00
	2003 – 2019	2.56	0.43	17.83	-0.54
Cobscook Bay	1986 – 2003	4.29	0.35	21.80	2.45
	2003 – 2019	3.31	0.33	18.74	-1.20

The linear regression model retaining sediment accretion rate, tidal range, and rate of RSLR explained 98 % of the observed variation in carbon stocks (adjusted R^2 = 0.95; F = 32.07; p = 0.03), for which sediment accretion rate explained the majority (80 %), followed by tidal range and rate of RSLR (14 % and 6 %, respectively). Sediment accretion rate had the strongest univariate correlation with carbon stock (r = 0.86), and was a positive, significant predictor (p = 0.01): carbon stocks were generally higher when sediment accretion rates were greater (Fig. 4.6). Mean sediment accretion rates across the whole study period were also higher at Cobscook Bay. It is important to note the variation of accretion rates and carbon stock seen in plots within the same site (Scarborough Central), and between sites when comparing the two time periods (Table 4.3).

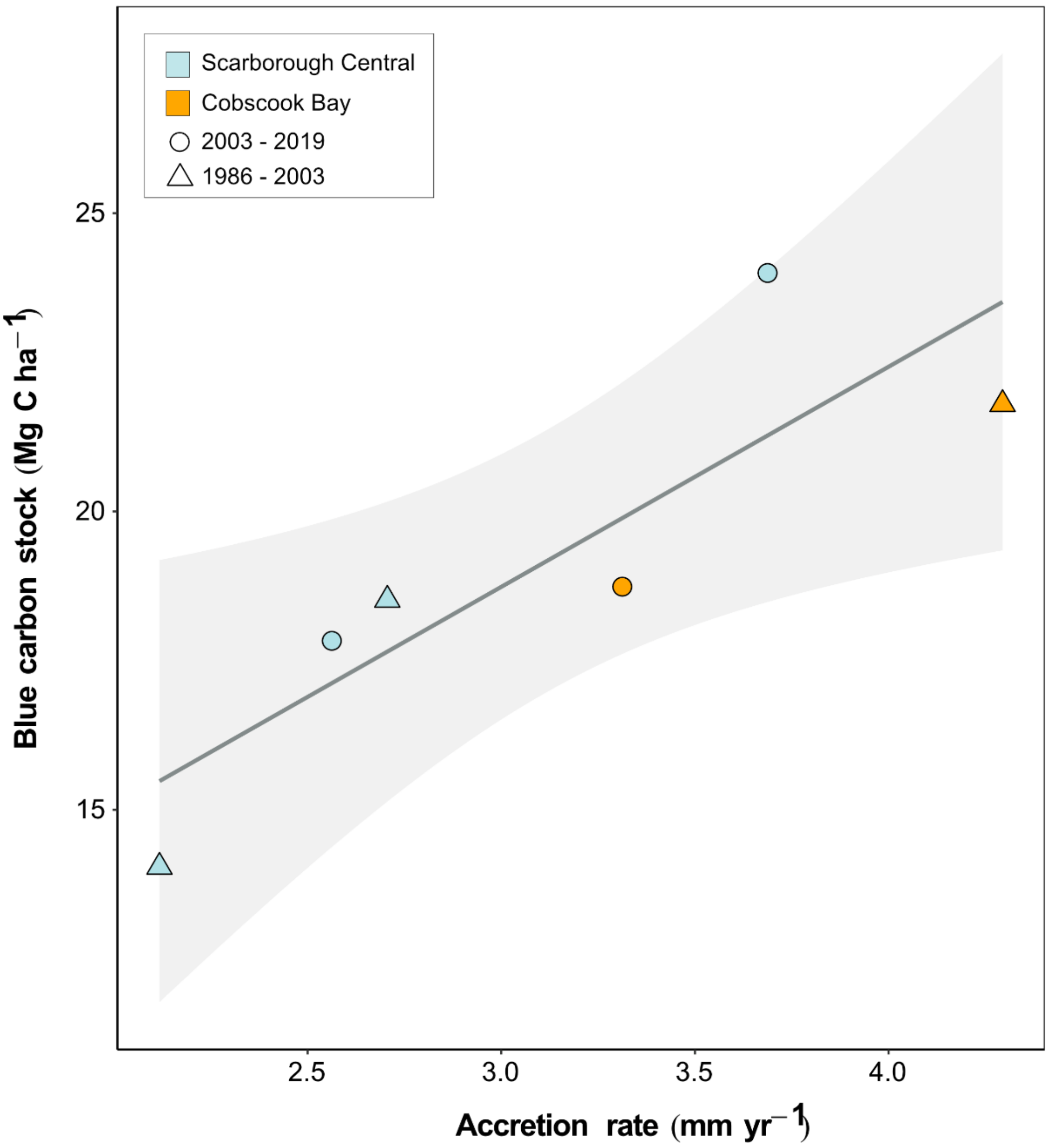


Figure 4.6. Blue carbon stock and accretion rate for relocated marker horizons at Scarborough Central (blue) and Cobscook Bay (orange) during the time periods between 1986 – 2003 (triangle) and 2003 – 2019 (circle), with a linear regression line and 95 % confidence interval (grey bar).

Tidal range was a borderline significant predictor of carbon stock ($p = 0.04$), and showed a weak, positive correlation ($r = 0.25$). Mean carbon stock in the soil profile between 1986 and 2019 was marginally higher at Cobscook Bay (20.27 Mg C ha^{-1}), where tidal range was two-fold greater than Scarborough Central (mean carbon stock = 18.60 Mg C ha^{-1}). Rate of RSLR was not a significant predictor of carbon stock ($r = 0.32$; $p = 0.16$) for sites with relocated marker horizons.

Rate of RSLR and accretionary balance had a strong, negative correlation (-0.82); the ability of saltmarshes to keep pace with RSLR through accretionary balance decreased over the study period as rates of RSLR increased (Table 4.3; Fig. 4.7). However, the rate of RSLR was positively correlated with sediment accretion rate (r = 0.62) and carbon stock (r = 0.65) for relocated marker horizon plots at Scarborough Central. Conversely, based on the observation between two points, Cobscook Bay experienced the opposite trend: values for sediment accretion rate, carbon stock and accretionary balance were lower in the most recent time period (2003 – 2019) when rates of RSLR were higher.

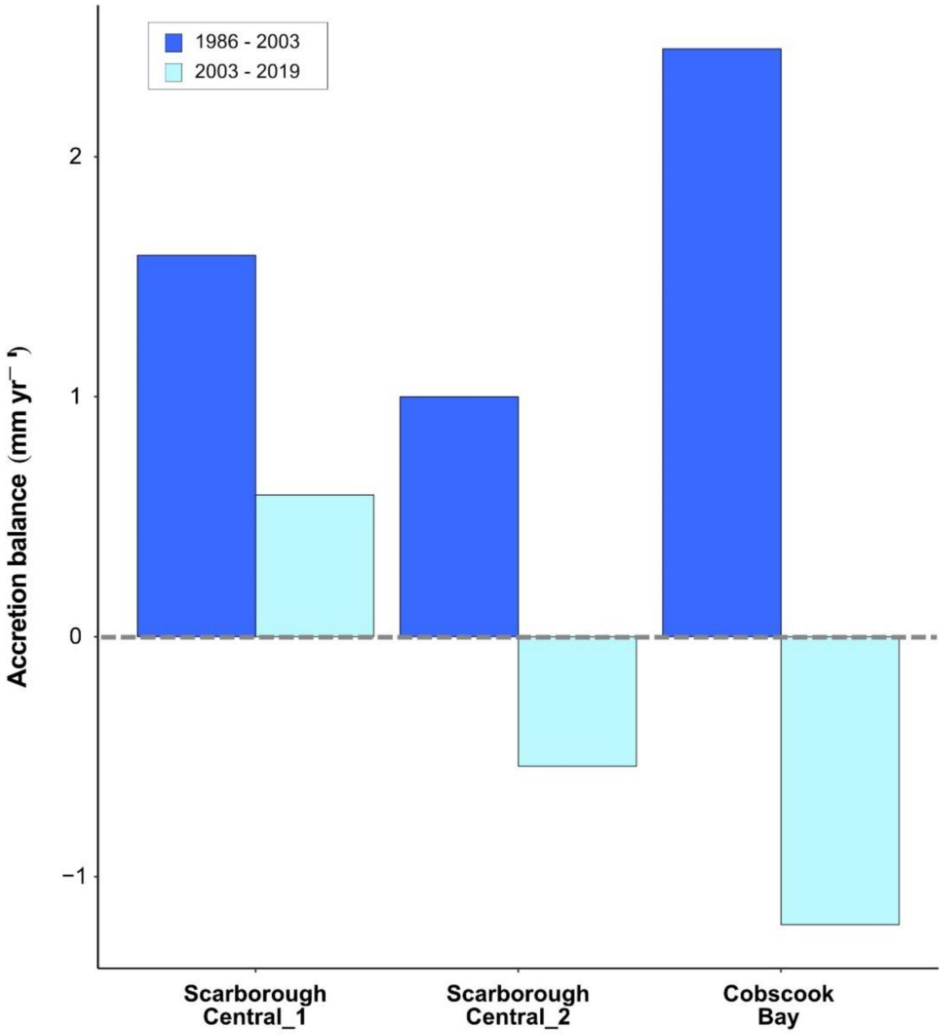


Figure 4.7. Saltmarsh accretionary balance with rates of rates of relative sea-level rise in the period between 1986 – 2003 (dark blue) and 2003 – 2019 (light blue). Equilibrium with RSLR is shown by the dashed grey line; plots from relocated marker horizons above this line are in a positive accretionary balance with rates of RSLR and plots below this line are in a negative accretionary balance.

Thus, greater carbon stocks are found under increased sediment accretion rates for relocated marker horizon plots at both sites. However, enhanced sediment accretion rates only resulted from an increase in the rate of RSLR for plots at Scarborough Central. At Cobscook Bay, sediment accretion rate, and associated carbon stock, was higher when the rate of RSLR was lower (between 1986 and 2003), and the saltmarsh was in a positive accretionary balance with RSLR.

4.3.3 Sites with extrapolated marker horizons

4.3.3.1 Rates of relative sea-level rise and tidal range

Rates of RSLR for all saltmarsh sites across Maine have increased over the study period (Table 4.4). The rate of RSLR, and tidal range, increases from more southwestern sites (Saltmarsh cove, Salt bay, Lowes Cove) to northeastern sites (Hay Creek, Jasper Beach, Crocker Point; Table 4.4; Fig. 4.2). The difference in rates of RSLR between sites in these two geographical areas became more pronounced in the most recent time period (2003 – 2019; Table 4.4). However, due to the lack of relocated marker horizons at these sites, the alignment of these time periods with the sub-samples analysed in this section is uncertain.

Table 4.4. Tidal range (m) and rates of relative sea-level rise (mm yr^{-1}) obtained via a linear regression on mean sea level data from Portland, Bar Harbor, and Eastport tide gauge stations (Permanent Service for Mean Sea Level (PSMSL); Holgate et al., 2013). Rates are shown across two time periods (1986 – 2003 and 2003 – 2019). Rates of RSLR and the tidal range (NOAA, 2022) were interpolated for each study site based on distance between the tide gauge stations.

Site	*Rate of relative sea-level rise (mm yr^{-1})*		Tidal range (m)
	1986 - 2003	**2003 – 2019**	
Saltmarsh Cove[a]	1.39	3.51	2.95
Salt Bay[a]	1.41	3.55	2.96
Lowes Cove[a]	1.39	3.52	2.95
Hay Creek[b]	1.86	4.36	3.75
Jasper Beach[b]	1.85	4.40	3.96
Crocker Point[b]	1.85	4.41	4.67

[a] Rates of RSLR were interpolated between Portland and Bar Harbor tide gauge stations [southwestern sites].
[b] Rates of RSLR were interpolated between Bar Harbor and Eastport tide gauge stations [northeastern sites].

4.3.3.2 Sediment accretion rates and carbon stock

As saltmarsh plant zone had a minimal correlation with carbon stock ($r = -0.04$), mean carbon stock between sites is comparable, regardless of whether plots were extracted from the low or mid-marsh. Mean carbon stock was calculated across the two sub-samples so each core had one value spanning the lower and faster periods of RSLR. There was a significant difference in mean carbon stock between sites with extrapolated marker horizons ($n = 30$; $F = 12.07$; $p < 0.001$; Fig. 4.8). Salt Bay had the highest mean carbon stock (22.52 ± 7.2 Mg C ha^{-1}), followed by Hay Creek (21.94 ± 1.8 Mg C ha^{-1}) and Saltmarsh Cove (19.22 ± 5.2 Mg C ha^{-1}; Fig. 4). Jasper Beach, Crocker Point and Lowes Cove had a similar mean carbon stock (7.67 ± 3.3; 7.32 ± 2.6; 7.13 ± 0.5 Mg C ha^{-1}, respectively), which was significantly lower than at Salt Bay, Hay Creek and Saltmarsh Cove ($p < 0.05$; Fig. 4.8). High variability in carbon stocks was observed within sites (Fig. 4.8).

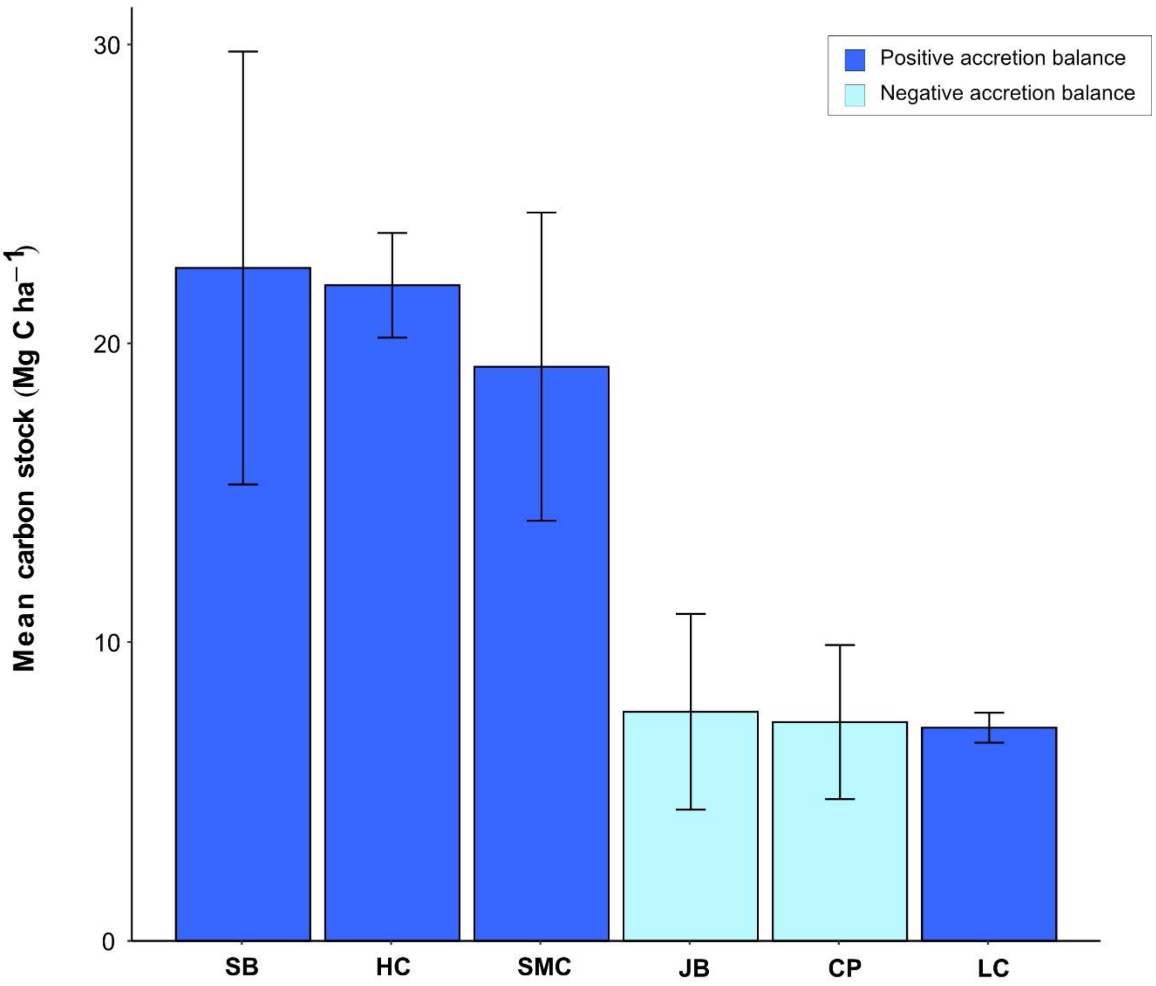


Figure 4.8. Mean carbon stock at saltmarsh sites across the whole core. Site accretionary balance (positive: dark blue; negative: light blue) is also shown based on calculations from Goodman *et al.* (2007) covering the period of lower rates of RSLR. [SB = Salt Bay; HC = Hay Creek; SMC = Saltmarsh Cove; JB = Jasper Beach; CP = Crocker Point; and, LC = Lowes Cove]. Sites are displayed in decreasing value of mean carbon stock.

For extrapolated sites, accretion rate and rate of RSLR explained 89 % of the observed variability in carbon stock between sites and time periods (adjusted $R^2 = 0.89$; $F = 75.77$; $p < 0.001$). Of these, accretion rate was the only significant predictor of carbon stock ($p < 0.001$), and was responsible for 95 % of the variation explained by the optimal model. Accretion rate had a positive relationship with carbon stock (Fig. 4.9).

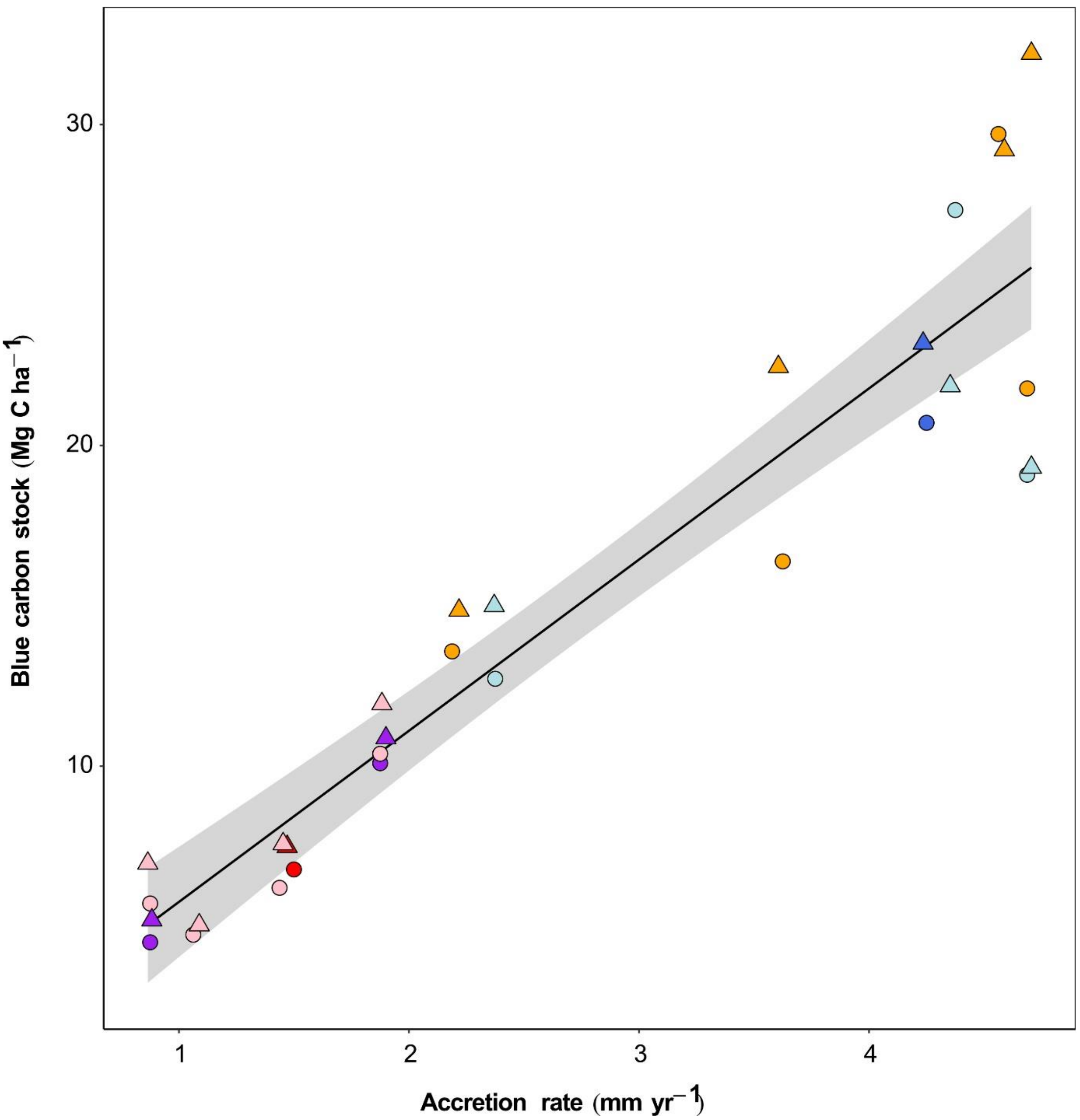


Figure 4.9. Blue carbon stock and accretion rate for extrapolated plots at Saltmarsh Cove (light blue), Salt Bay (orange), Lowes Cove (red), Hay Creek (dark blue), Jasper Beach (purple), and Crocker Point (pink), with a regression line and 95 % confidence interval (grey bar) during periods of slower (circle) and faster (triangle) rates of RSLR.

Both accretion rates and carbon stocks are highly site-specific, and sites with a higher average accretion rate were associated with higher carbon stocks (Fig. 4.9; Table 4.5). Rate of RSLR was not a significant predictor of, and had a weak correlation with, carbon stock (r = -0.25).

Sites with the greatest overall mean carbon stock were those that were in a greater accretionary balance with RSLR during periods when rates were slower, based on calculations by Goodman *et al.* (2007) (Fig. 4.8; Table 4.5). Lowes Cove had the lowest mean carbon stock and was only marginally keeping pace with the rate of RSLR during the time period when rates were lower (Fig. 4.8; Table 4.5). Accretionary balance cannot be calculated for the time period when rates of RSLR were faster.

Table 4.5. Average sediment characteristics for each site with extrapolated marker horizon plots. Accretion balance (+ positive; **-** negative) is shown in relation to rates of RSLR between 1986 and 2003 obtained from Goodman et al. (2007) for the same sites. *Sediment accretion rate for the period of faster RSLR was based on a constant sediment accretion rate from the period of slower RSLR calculated by Goodman et al. (2007) – see Appendix B2.

Site	Time Period (RSLR)	Sediment accretion rate (mm yr^{-1})*	Dry bulk density (g cm^3)	Carbon stock (Mg C ha^{-1})	Accretion balance (mm yr^{-1})
Saltmarsh Cove	slower	3.8	0.5	18.7	+2.4
	faster	3.8	0.4	19.7	Na
Salt Bay	slower	3.7	0.3	24.7	+2.4
	faster	3.7	0.3	20.4	Na
Lowes Cove	slower	1.5	0.4	7.5	+0.1
	faster	1.5	0.4	6.8	Na
Hay Creek	slower	4.2	0.4	23.2	+2.4
	faster	4.2	0.8	20.7	Na
Jasper Beach	slower	1.4	0.2	8.0	-0.5
	faster	1.4	0.2	7.3	Na
Crocker Point	slower	1.3	0.5	7.9	-0.5
	faster	1.3	0.5	6.8	Na

Tidal range showed a relatively high correlation with accretion rate ($r = -0.67$) and was therefore not included in the model. However, tidal range also had a relatively strong negative correlation with carbon stock ($r = -0.65$), suggesting that sites with a lower tidal range were generally associated with a higher sediment accretion rate and carbon stock.

It is likely that sites with a higher accretionary balance with RSLR are thus associated with greater carbon stocks. Sites with a lower tidal range were generally associated with greater sediment accretion rates, and the rate of RSLR was not shown to be a significant driver of carbon stocks. It is important to note, however, that a constant sediment accretion rate assumed across both time periods (representing a slower and faster rate of RSLR) is unlikely to reflect accurate responses of carbon stock to rates of RSLR.

4.4 Discussion

Due to the limited number of relocated marker horizons, this study can only present a snapshot into the response of saltmarsh carbon stock to varying rates of relative sea-level rise. Variability from plots within the same site were observed, and a larger sample size is thus required to make definitive conclusions. Results from extrapolated plots, where marker horizons were not relocated, are intended to be informative, and should be considered with caution. Nevertheless, these findings provide an insight into the fate of saltmarshes, and the provision of ecosystem services (e.g. carbon storage), under increasing rates of relative sea-level rise. Such insights are of high importance as rises in global mean sea level have been accelerating since the 1960s (Dangendorf *et al.,* 2019). In this section, the difficulty in relocating marker horizons for long-term studies, and the relationship between sediment accretion rates, carbon stock and rates of RSLR for saltmarshes across Maine are discussed.

4.4.1 Marker horizon recovery

In this study, only three brick dust marker horizons were successfully relocated across two sites (Scarborough Central and Cobscook Bay) since plots were established in 1986, 33 years prior. Whilst the main advantages of using marker horizons in sediment accretion studies relate to the low expense and sampling effort required for high resolution measurements (Cahoon & Turner, 1989; Nolte *et al.,* 2013; Swales & Lovelock, 2020), the maximum time the method can be used is dependent on the durability of the marker horizon (Breithaupt *et al.,* 2018). In turn, durability is influenced by environmental conditions, such as hydrodynamic setting and sediment composition (Callaway *et al.,* 2013).

Brick dust marker horizons are unlikely to be effective in areas with high levels of bioturbation, in highly erosive sites, or when established in coarse sediment, such as sand, where fine brick dust can sink into deeper sediments (Thomas & Ridd, 2004; Callaway *et al.,* 2013; Swales & Lovelock, 2020). Bioturbation by the green crab (*Carcinus maenas*) has been found to be a major cause of the loss of eelgrass meadows in Maine (Neckles, 2015), and this species, alongside other bioturbators, is present in high abundance across Maine's saltmarshes (Aman & Grimes, 2016; Raposa *et al.,* 2020). Despite this, in 2003, Goodman *et al.* (2007) reported sharp upper and lower boundaries of brick dust plots, suggesting mixing of sediments due to bioturbation is not an issue in sites with relocated marker horizons. However, 17 years after establishment, marker horizons were only relocated in 11 out of the original 26 sites in Maine (Goodman *et al.,* 2007; Fig. 4.5). In sites where plots were not able to be relocated, Goodman *et al.* (2007) reported clear signs of transgression and erosion, with the encroachment of low marsh vegetation communities into mid marsh zones, and the replacement of low marsh by mudflat. In highly erosive sites, marker horizons are likely to be washed away in a short timeframe following establishment (Callaway *et al.,* 2013). Erosional processes can hinder the accurate retrieval of marker horizons, particularly when studies are conducted over longer timescales. The two marsh sites where brick dust plots were relocated in 2019 were observed to have remained relatively stable between 1986 and 2003; there was limited evidence of erosion and vegetation communities where plots were established were still dominated by the same plant species (Goodman *et al.,* 2007).

Even under ideal conditions when saltmarshes are composed of fine sediments, and where bioturbation and erosion is minimal, marker horizons will still become dispersed and more problematic to relocate over time (Callaway *et al.,* 2013). This trend is reflected in this study as the successful relocation of brick dust declined with time since establishment, and the number of relocations were high within the first year (Wood *et al.,* 1989; Goodman *et al.,* 2007; Fig. 4.5). It is likely the 33 year period of this study is too long a timeframe to rely on brick dust marker horizon retrieval. The majority of coastal wetland studies using marker horizons report accretion rates on a scale of months to years (van Wijnen & Bakker, 2001; McKee, 2011; Nolte *et al.,* 2013; Saintilan *et al.,* 2013; Swales & Lovelock, 2020), as opposed to decades. Instead, sedimentation plates and erosion pins may be preferable options for measuring accretion on timescales up to decades, and longer term accretion rates are best estimated using sediment dating techniques such as Lead (Pb) and radiocarbon (Cahoon & Turner, 1989; Nolte *et al.,* 2013).

4.4.2 Sediment accretion rate, relative sea-level rise and carbon stock

Relocated marker horizons at Scarborough Central showed higher carbon stocks and sediment accretion rates in the period between 2003 and 2019, when rates of relative sea-level rise were higher. Feedbacks between increased inundation under RSLR and enhanced sediment deposition provide saltmarshes with an adaptive capacity to keep pace with increasing RSLR rates (Crosby *et al.,* 2016). Accommodation space created by RSLR enables greater sediment accretion rates under this feedback, and includes both mineral and organic matter accumulation (Connor *et al.,* 2001; Rogers *et al.,* 2019a). Whilst increases in sea-level rise can lead to enhanced sediment delivery, potentially rich in allochthonously sourced organic carbon, it can also stimulate plant productivity and the in-situ production of organic carbon (Morris *et al.,* 2013). At Scarborough Central, enhanced carbon stocks are thus controlled by sediment accretion rates as a direct response to increases in the rate of relative sea-level rise, in agreement with previous studies (Gonneea *et al.,* 2019; Rogers *et al.,* 2019a). However, the relationship from the relocated plot in Cobscook Bay was more complex. During the earlier time period (1986 – 2003), the sediment accretion rate and carbon stock was greater than the more recent time period (2003 – 2019), despite a lower rate of RSLR. Furthermore, during 1986 – 2003, the plot at Cobscook Bay had a higher rate of RSLR and accretion, as well as a carbon stock that was two-fold greater than the relocated plots at Scarborough Central. In contrast, the accretion rate and carbon stock in one of the Scarborough Central plots was greater than that of Cobscook Bay, in the more recent time period despite a lower localised rate of RSLR.

These findings support analytical and numerical models that show carbon stocks are nonlinearly related to the rate of RSLR; stocks increase with RSLR until it reaches a threshold rate, beyond which vegetation drowns, sediment accretion rates are unable to keep the marsh in equilibrium with RSLR, and consequently, the accumulation of carbon stocks decrease (Mudd *et al.,* 2009). Goodman *et al.* (2007) predicted 4 mm yr^{-1} to be the threshold rate of RSLR, based on the maximum observed sediment accretion rates for all relocated plots along the coast of Maine between 1986 and 2003. The accretionary balance of saltmarshes was estimated to be in a deficit when 4 mm yr^{-1} of RSLR were to be exceeded (Goodman *et al.,* 2007). In the period between 2003 and 2019, localised rates of RSLR were 3.10 mm yr^{-1} at Scarborough Central, and 4.51 mm yr^{-1} at Cobscook Bay (Holgate *et al.,* 2013), and the plot at Cobscook Bay was shown to be in a negative accretion balance. It is possible that the saltmarsh at Cobscook Bay has become overwhelmed by, and unable to keep pace with, the increasing rates of relative sea-level rise in the past couple of decades. A numerical model developed by Kirwan & Mudd (2012) anticipated an increase in organic matter content of saltmarshes under rising rates of RSLR until rates reached 4.5 mm yr^{-1}, and this rate of RSLR has been surpassed at Cobscook Bay in the more recent time period (rate of RSLR: 4.51 mm yr^{-1}). This is due to carbon-climate feedback effects where marsh submergence and drowning under high RSLR rates cause carbon burial rates to decline (DeLaune & White, 2012; Kirwan & Mudd, 2012). Furthermore, saltmarsh loss, such as that experienced through submergence, can contribute to the global carbon cycle as stored carbon may be released into the atmosphere as CO_2 emissions following degradation (Macreadie *et al.,* 2013). Whilst these findings from relocated marker horizons at Scarborough Central and Cobscook Bay appear to support recent models in the theory that carbon stock accumulation increases linearly with enhanced rates of RSLR until a threshold of ~ 4.5 mm yr^{-1} (Mudd *et al.,* 2009; Kirwan & Mudd, 2012), more work is required at these saltmarshes along the coast of Maine. This study does not have a sufficient number of samples to make definitive claims, and high variability has been observed between plots in the same site.

Across all sites (incl. both relocated and extrapolated plots), and time periods, accretion rate was the strongest driver of carbon stock. However, the extrapolated sites with the greatest sediment accretion rates and carbon stock between 1986 and 2003 were not driven by higher rates of RSLR, as the majority of these sites were in the south-west where the rate of RSLR and the tidal range is lower (Holgate *et al.,* 2013). A large scale study in Great Britain, which surveyed saltmarshes across 25 estuaries, found sediment supply to be a stronger indicator of saltmarsh lateral dynamics than sea-level rise (Ladd *et al.,* 2019). Saltmarshes that experienced long-term erosion were subject to reduced sediment fluxes, and were also associated with reduced vertical accretion balance in relation to rates of RSLR (Ladd *et al.,* 2019). Saltmarsh accretion in areas of low sediment supply rely primarily on in-situ belowground biomass accumulation, and are more likely to be overwhelmed by increases in RSLR and inundation (Crosby *et al.,* 2016). Goodman *et al.* (2007) considered inorganic, mineral sediment to be the main contributor to saltmarsh accretion rates in Maine, and it is possible that allochthonous sources are an important component of organic carbon stocks at these sites. However, without the use of compound-specific isotope analysis, we are unable to discern whether accretion rates and carbon stock retrieved from above marker horizons are from autochthonous or allochthonous contributions (Saintilan *et al.,* 2013). If saltmarsh sites that were in a low, or negative, accretionary balance during periods when the rate of RSLR was lower were sediment-starved, it could explain the general trend of lower accretion rates and carbon stocks at extrapolated sites that experienced higher rates of RSLR. Sediment accumulation is often correlated with local hydrodynamics, such as sea-level rise (Ma *et al.,* 2014), but is dependent on the availability of sediment for deposition (Rogers *et al.,* 2019a). Sediment supply can be affected by a multitude of factors, most notably river damming (Vörösmarty *et al.,* 2003; Ma *et al.,* 2014). Differences in plant community composition can also have an influence over the trapping efficiency of sediment and organic material in saltmarshes (Ford *et al.,* 2019).

In this study, the relationship between carbon stock and increased rates of RSLR was unable to be accurately tested for extrapolated sites as sediment accretion rates in the most recent time period were calculated based on a constant sediment accretion rate from the earlier time period. This is unlikely to be a true reflection of the response of saltmarsh sediment accretion rates to increases in the rate of RSLR. Increases in RSLR can enhance carbon stocks through the creation of accommodation space, and increased sediment deposition (Rogers *et al.,* 2019a). If sediment supply does represent a more important driver of sediment accretion and carbon stocks than in-situ production at these sites, it is unlikely that the three sites in a negative, or marginally positive, accretionary balance with rates of RSLR in the earlier time period, as calculated by Goodman *et al.* (2007), would have kept pace with RSLR, or accumulated greater carbon stocks, in more recent decades. Global declines in sediment supply to coastlines (Ma *et al.,* 2014; Ladd *et al.,* 2019) has already led to observed losses in saltmarsh extent along the east coast of the U.S. (Weston, 2014). Extrapolated sites with low sediment accretion rates recorded in the earlier time period (Goodman *et al.,* 2007) may have experienced erosion and transgression under higher rates of RSLR in recent decades, due to possible insufficient sediment supply, and this could also explain why marker horizons were not able to be relocated in 2019. Without the use of Surface Elevation Tables, or Sedimentation Erosion Tables, the sediment accumulated above marker horizons does not take erosional processes, elevation in the tidal frame, or subsidence into account (Callaway *et al.,* 2013). It is also worth noting that three of the extrapolated sites would have experienced rates of RSLR above 4 mm yr^{-1} in more recent decades, with potentially negative consequences for saltmarsh accretionary balance (Goodman *et al.,* 2007; Kirwan & Mudd, 2012).

Tidal range for all saltmarshes remained consistent over the 33 year study period, and was notably higher for sites on the north-east of Maine's coastline (NOAA, 2022). The relationship between tidal range and carbon stock differed between relocated and extrapolated marker horizons. For relocated sites, both tidal range and mean carbon stock across the whole study period (1986 – 2019) was highest at Cobscook Bay, despite declines in sediment accretion rates and carbon stock in the most recent time period. Although the correlation between tidal range and carbon stock at relocated sites was weak, other studies have shown sediment supply, and in turn, carbon stock, is greater for saltmarshes that experience a greater tidal range – Cobscook Bay is in a macrotidal environment whilst Scarborough Central is in a mesotidal environment (Wollenberg *et al.,* 2018). Sea-level rise is often thought to amplify tidal range (Khojasteh *et al.,* 2021), and Cobscook Bay has already shown signs of being overwhelmed by increases in the rate of RSLR over recent decades, so it is likely the relationship between tidal range and carbon stock at relocated sites will become negative in future decades. A negative relationship was observed for extrapolated sites, whereby carbon stock was greater at sites that experienced a lower tidal range. Kirwan *et al.* (2010)

demonstrated that the maximum rate of RSLR saltmarshes can persist is dependent on sediment supply and tidal range. Macrotidal saltmarshes are thought to be more sensitive to sediment supply than marshes with lower tidal ranges (Kirwan *et al.,* 2010). Again, this could point to a reduced supply of sediment for saltmarshes in a low, or negative, accretionary balance with RSLR, the majority of which are sites with higher tidal ranges. Saltmarshes already limited by reduced sediment supply have lower opportunities for allochthonous carbon inputs, which may represent the greatest contributor to carbon stocks for sites in macrotidal estuaries (Van de Broek *et al.,* 2018). Furthermore, low sediment supply, coupled with macrotidal environments, threatens saltmarsh persistence, diminishing the capacity for new carbon to be produced (Schile *et al.,* 2014).

4.4.3 Conclusions and further study

The use of marker horizons for studies on vertical accretion in coastal wetlands are generally characteristic of shorter timescales from months to years, and their effectiveness depends on retrieval timescale, in addition to various environmental conditions. This study has shown the retrieval success of marker horizons in saltmarshes across Maine is likely a function of time since establishment and alternative approaches, such as sedimentation plates and erosion pins, are recommended for studies that wish to observe sediment accretion rate and carbon stock on decadal timeframes.

Modelling studies have been paramount in predicting how saltmarsh carbon stocks respond positively to increases in the rate of relative sea-level rise, whether through the enhanced delivery of allochthonous carbon sources, or stimulated plant productivity; as well as identifying thresholds when saltmarshes, and their ability to capture and store carbon, may become overwhelmed by accelerating RSLR. In this study, the threshold rate of RSLR was observed to be 4.5 mm yr^{-1} for sites where marker horizons were relocated, in agreement with previous studies (e.g. Kirwan & Mudd, 2012). Beyond this threshold, sediment accretion rate and carbon stock began to decline. However, such thresholds are highly site-specific, as observed across extrapolated sites. For example, some sites were in a negative, or very low, accretionary balance with RSLR between in the earlier time period, when rates were much lower than recent decades. This could potentially result from reduced localised sediment supply at these sites, lessening the possibility for allochthonous inputs and healthy ecosystem functioning, particularly for saltmarshes in macrotidal environments. Studies into vertical accretion of saltmarshes and the resilience of carbon stocks under increasing rates of relative sea-level rise would therefore benefit from the consideration of localised sediment supply, and the use of isotopic analysis to determine contributions from allochthonous and autochthonous sources to carbon stock. Such studies would paint a more coherent picture of carbon stock response to sea-level rise, and are of particular interest as rates of RSLR continue to increase, whilst sediment supply to coastlines is on a declining trend. This combination threatens the future stability of saltmarshes and the benefits they provide, including carbon capture and storage. To summarise, carbon stocks in saltmarshes along the coast of Maine were greater under enhanced rates of sediment accretion, but this in turn was not linearly related with rates of relative sea-level rise. Carbon stocks are likely to be influenced by saltmarsh ability to keep pace with rates of relative sea-level rise.

This study therefore provides an initial exploration of recent saltmarsh carbon stocks under different rates of relative sea-level rise and sediment accretion on a regional scale along the coastline of Maine, with further investigation required to quantitatively confirm the findings produced from this work.

Chapter 5 of 7

Maximising blue carbon stocks through saltmarsh restoration

Abstract

Political discourse around coastal wetland restoration and blue carbon management strategies has increased in the past decade, yet carbon storage has neither been a reason for restoration, nor a criterion to measure the success of current saltmarsh restoration schemes in the UK. To maximise climate change mitigation through saltmarsh restoration, knowledge on the key drivers of carbon stock variability is required. We use restored saltmarshes of similar age, paired with adjacent natural marshes as references, to identify drivers of carbon stocks following managed realignment within an estuary in southeastern England. From surficial soil cores (top 30 cm), we measured carbon stock alongside environmental characteristics. Carbon stock between natural and restored sites were similar after ~ 30 years when restored sites were above mean high water neap (MHWN) tidal levels. Elevated marsh platforms likely provide suitable conditions for the development of mature plant communities associated with greater capture and production of organic carbon. The restored site at Tollesbury (Essex, UK) had a 2-fold lower carbon stock than other restored sites in the estuary. We attribute this to the site's low position in the tidal frame, below MHWN tidal levels, coupled with low sediment supply and the dominance of pioneer plant communities. As blue carbon is anticipated to become an important facet of saltmarsh restoration, we recommend that sites above MHWN tidal levels are selected for managed realignment or that preference is given to coastlines with a high sediment supply that may rapidly elevate realignment sites above MHWN. Alternatively, elevation could be artificially raised prior to realignment. Restoration schemes aiming to maximise climate change mitigation should also encourage the establishment of key plant species (e.g., *Atriplex portulacoides* in our study) to enhance carbon stocks. However, the overall goal of restoration ought to be carefully considered as trade-offs in ecosystem services may ensue if restoration for climate change mitigation alone is pursued.

5.1 Introduction

The capture and storage of blue carbon (organic carbon found within marine and coastal ecosystems) is increasingly recognised by governments, industries, and scientists around the world as a nature-based solution to help mitigate climate change (Lovelock & Duarte, 2019; Macreadie *et al.,* 2019; Bertram *et al.,* 2021). Coastal wetlands, including saltmarshes, store more carbon per unit area than terrestrial forests, and represent a globally significant carbon sink (Duarte *et al.,* 2005; Nellemann *et al.,* 2009; Mcleod *et al.,* 2011; Temmink *et al.,* 2022). In addition to their climate change mitigation capacities (via carbon capture and subsequent burial), coastal wetlands provide numerous other ecosystem services and benefits, including fisheries support, coastal flood protection, and biodiversity enhancement (Barbier *et al.,* 2011; Möller *et al.,* 2014; Unsworth *et al.,* 2019). Coastal wetlands are also amongst the most threatened habitats in the world. Although global assessments of saltmarsh extent change are lacking, saltmarsh habitat loss occurs by approximately 1 – 2 % per year (Duarte *et al.,* 2008), with large-scale losses to sea-level rise anticipated (Saintilan *et al.,* 2022). Recent restoration efforts have successfully offset a large proportion of tidal wetland loss globally (Murray *et al.,* 2022). However, in the UK, the area of saltmarsh created via restoration is considered to be behind the pace required to compensate for current and historic losses in extent (Rupp-Armstrong & Nicholls, 2007; Lawrence, 2018).

Saltmarsh loss and degradation not only reduces future carbon sequestration potential (Howard *et al.,* 2017; Lovelock *et al.,* 2017), but can result in the ecosystem switching from a carbon sink to a carbon source (Macreadie *et al.,* 2013). This occurs when soil disturbance leads to the remineralisation of particulate organic matter, whereby carbon stored in biomass and soil is oxidised and released back into the atmosphere as CO_2 (Pendleton *et al.,* 2012; Lovelock *et al.,* 2017). The conservation and restoration of blue carbon habitats can therefore enhance CO_2 capture, as well as avoid CO_2 emissions from habitat degradation (Duarte *et al.,* 2013; Macreadie *et al.,* 2013; Kelleway *et al.,* 2020). Saltmarshes are therefore part of the solution to limiting global warming to below the 1.5 °C threshold set out in the Paris Agreement, and enhancing the removal of excess CO_2 from the atmosphere (Chausson *et al.,* 2020; Macreadie *et al.,* 2021). Political discourse around coastal wetland restoration and blue carbon strategies has exponentially increased in the past decade, with 46 countries now mentioning coastal and marine ecosystems as mitigation solutions within their Nationally Determined Contributions – the country-level plans to address climate change as part of the Paris Agreement (Lecerf *et al.,* 2021). To maximise climate change mitigation through coastal wetland restoration, an understanding of the environmental drivers that influence carbon storage in restored coastal wetlands is required.

Nations around the globe are investing in coastal wetland restoration and there is international interest in the inclusion of coastal wetlands in climate change mitigation policy through carbon financing (Wylie *et al.,* 2016; Vanderklift *et al.,* 2019). Numerous approaches to restoring saltmarsh habitat exist (MacDonald *et al.,* 2020; Hudson *et al.,* 2021); however, the predominant method of saltmarsh restoration in the UK and Europe is managed realignment (MR), defined as the deliberate breaching of coastal defences to reinstate tidal inundation and recreate intertidal habitat (Luisetti *et al.,* 2011; Burden *et al.,* 2019). The capacity of healthy saltmarshes to keep pace with sea-level rise (Kirwan *et al.,* 2016) and to naturally dissipate wave energy (Möller *et al.,* 2014), makes their restoration an attractive option in reducing maintenance costs of coastal defences and improving flood management (Shepherd *et al.,* 2007). In line with the EU Habitats Directive, MR also compensates for historic losses of intertidal habitat and supports biodiversity (Hudson *et al.,* 2021). Of the 48 MR schemes in the UK where saltmarsh habitat has been restored, cost-effective flood protection, compensation for natural habitat loss, and habitat creation were listed as the primary reasons for restoration (ABPmer, 2022).

Although saltmarsh carbon storage is now frequently highlighted as a co-benefit of restoration, blue carbon has neither been a reason for restoration, nor a criterion against which to measure the success of current MR schemes in the UK (Burden *et al.,* 2013; Austin *et al.,* 2022). Despite this, coastal wetland restoration for blue carbon benefits has become a recognised tool in climate change mitigation (Wylie *et al.,* 2016; Duarte *et al.,* 2020). The increasing interest in using nature-based solutions for climate change mitigation, complemented by the United Nations Decade on Ecosystem Restoration (2021 – 2030), provides an opportunity for blue carbon and the maximisation of climate change mitigation benefits to be an important facet of future MR schemes (Seddon *et al.,* 2020; Waltham *et al.,* 2020; Austin *et al.,* 2022). There have already been calls for blue carbon habitat restoration to be a key focus of the UN Decade on Ecosystem Restoration (Macreadie *et al.,* 2021).

Initial carbon accumulation rates in MR saltmarshes can be high. The large accommodation space created by MR rapidly fills with riverine and/or marine sediment (if available) that can be rich in allochthonous carbon (Wollenberg *et al.,* 2018; Drexler *et al.,* 2020). However, the timescales involved in the development of carbon stocks equivalent to adjacent natural sites is unclear. Previous studies have reported carbon stocks in restored saltmarshes become comparable to natural marshes over decadal to centennial timescales, with estimates up to 100 years (Burden *et al.,* 2013; Burden *et al.,* 2019). Conservation of existing carbon stocks in natural marshes is therefore often prioritised as a climate change mitigation strategy over restoration (Macreadie *et al.,* 2017b). Recent evidence from the USA, however, found carbon stocks in the top 30 cm of a MR site were only marginally lower than adjacent natural sites in the 4 years following restoration (4.43 and 5.95 kg C m^{-2} respectively; Poppe & Rybczyk, 2021). The various timescales reported for the development of carbon stocks in restored saltmarshes are highly site-specific - an important consideration for MR as a technique to enhance climate change mitigation. For example, Nightingale *et al.* (2022) recommended choosing optimal regional and ecological factors that enhance carbon accumulation rates in realigned saltmarshes, such as higher latitudes, when focusing on the climate change mitigation potential of future MR schemes.

Whilst advances have been made on a global (Rogers *et al.,* 2019a) and regional (Ford *et al.,* 2019) scale, few studies have examined saltmarsh blue carbon variability at the estuarine scale (Broek *et al.,* 2016) and even fewer for MR sites. Identifying local scale controls on blue carbon variability within MR sites will ascertain how the increasing number of MR schemes globally contribute to addressing the Climate Crisis. Drivers of blue carbon stock in saltmarshes include environmental characteristics such as elevation and local hydrology, that influence the supply of sediment and allochthonous carbon into a site (Wollenberg *et al.,* 2018; Rogers *et al.,* 2019a; Mossman *et al.,* 2022), sedimentary characteristics such as particle size that affect the preservation of carbon stocks (Kelleway *et al.,* 2016a; Ford *et al.,* 2019), and vegetation characteristics that include vegetation community composition and diversity, which affect both the supply of autochthonous carbon through primary production as well as the trapping of allochthonous carbon (Ouyang & Lee, 2014).

To our knowledge, no study has yet compared a range of environmental drivers against blue carbon stocks in MR saltmarshes. Here, we investigate which drivers enhance blue carbon capture of saltmarsh MR sites within an estuary in southeastern England to inform future schemes seeking to maximise the potential blue carbon benefits of MR. The objectives of this study are to: (1) quantify blue carbon stocks for the top 30 cm of soil for saltmarshes with different management histories (managed realigned and adjacent natural sites) within the same estuary; (2) evaluate the relationship between environmental variables and blue carbon stock to determine the greatest drivers of spatial variation, and; (3) provide management recommendations to maximise blue carbon stocks in future MR schemes.

5.2 Materials and methods

5.2.1 Study sites and sampling design

We studied six saltmarshes (three managed realigned sites and three adjacent natural sites) in the Blackwater Estuary, Essex, UK (Fig. 5.1). The Blackwater Estuary is of high biological and conservation importance, as reflected in numerous conservation designations: Site of Special Scientific Interest, Ramsar site, Special Area of Conservation, and Special Protection Area. Alongside neighbouring estuaries (Crouch, Roach, and Colne), the area forms part of a designated inshore marine conservation zone. The estuary is also home to the first MR scheme in the UK at Northey Island, which was undertaken in 1991 as a pilot project to improve coastal defences (Doody, 2013; Ladd, 2021).

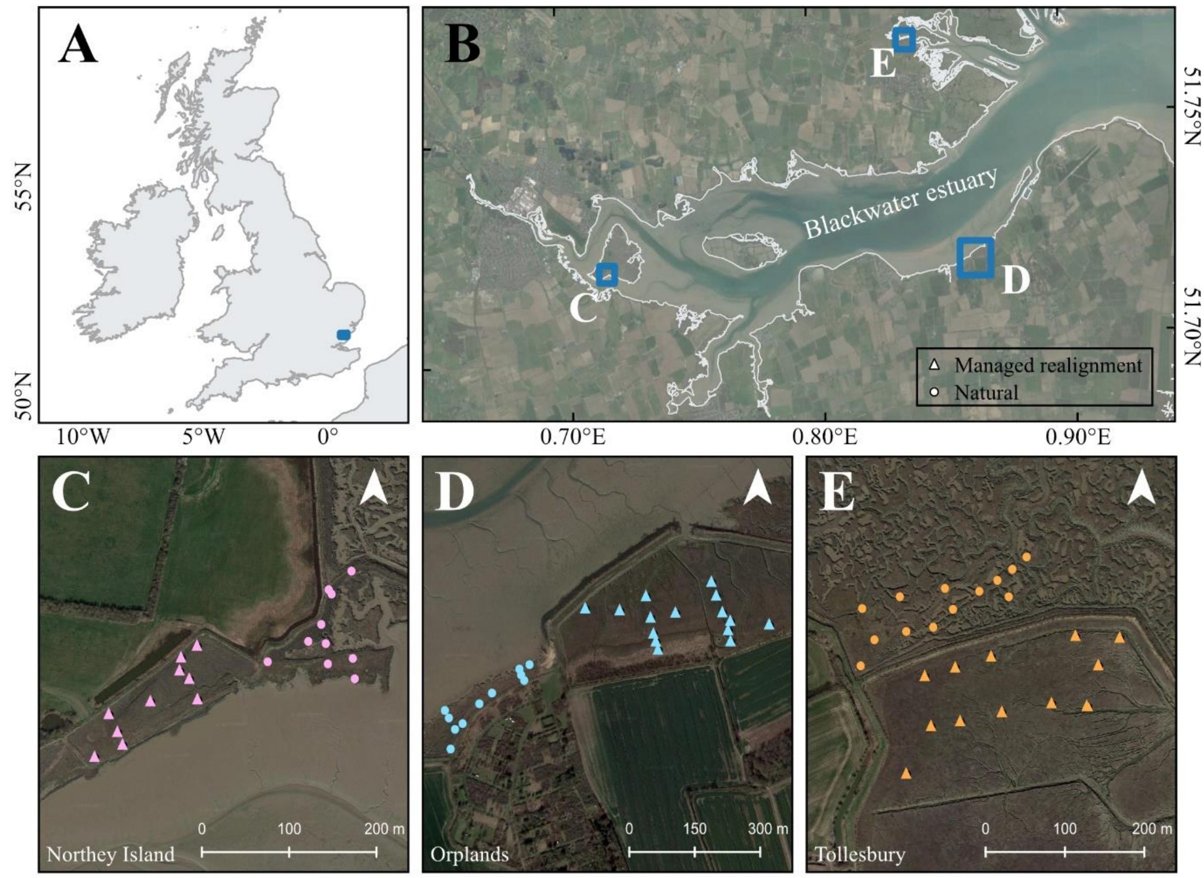


Figure 5.1. Location of Blackwater Estuary, Essex within southeastern England (**A**) and saltmarsh study sites within the Blackwater Estuary (**B**). Sampling transects and plot locations are shown for managed realigned (triangle) and natural (circle) sites at Northey Island (**C**), Orplands (**D**), and Tollesbury (**E**).

Three managed realigned sites in the Blackwater Estuary were chosen for this study: Northey Island, Orplands and Tollesbury (Table 5.1), all of which were restored primarily for the purposes of improving flood protection and creating habitat (ABPmer, 2022). Sites were selected due to similar timing of realignment in the early to mid-1990s (Table 5.1) and the presence of adjacent natural saltmarshes that could act as controls for the study. All realignment sites were restored by breaches to existing coastal defences (Table 5.1). A single breach was implemented at Northey Island and Tollesbury MR sites during construction, whereas multiple breaches were implemented at Orplands MR (ABPmer, 2022; Table 5.1). The Blackwater estuary is macrotidal (tidal range: 5.2 – 5.8m) and ebb-dominant, with a net export of sediment and material from the mouth of the estuary (Ladd *et al.,* 2019).

Table 5.1. Implementation year, size, and breach width of managed realignment site schemes.

Managed Realignment	Implementation year	Size (ha)	Breach width (m)
Northey Island	1991	0.8	20
Orplands	1995	38	50 and 40
Tollesbury	1995	21	60

To gain insights into site-specific environmental setting, a range of tidal, vegetation and soil characteristics shown to influence blue carbon stocks were sampled at each site along three transects: two perpendicular to the marsh edge and one passing diagonally across (Fig. 5.1). Multiple sampling plots (1m × 1m quadrat) were placed along each transect to capture heterogeneity in saltmarsh vegetation zones and elevation. The number of sampling plots varied between 10 and 14. All fieldwork was carried out in July 2019.

5.2.2 Data collection

5.2.2.1 Tidal characteristics

Site-specific hydrology can provide insights on potential allochthonous carbon supply, as well as the development of vegetation communities and plant zones. Elevation was recorded at each sampling plot using a Trimble Catalyst differential global positioning system with an accuracy of <1 cm. To standardise sampling plot elevation to the respective height in the tidal frame and to allow for comparisons across sites, relative tidal height was calculated, where 0 = mean high water neap (MHWN) and 1 = mean high water spring (MHWS) tidal levels (Mossman *et al.,* 2012a; Mossman *et al.,* 2020):

$$\text{Relative Tidal Height} = \text{Elevation (m ODN)} - \frac{\text{MHWN}}{(\text{MHWS} - \text{MHWN})}$$

MHWN and MHWS tidal data were collated for each site from Mossman *et al.* (2012b). Tidal data was available for Orplands, Tollesbury (natural) and Tollesbury (MR) sites. There was no available data for Northey Island, so data was used from the next closest site, Steeple, less than 5 km away.

5.2.2.2 Vegetation characteristics

Vegetation characteristics were recorded as a proxy for potential autochthonous and allochthonous (via trapping) carbon supply (Ford *et al.,* 2019). Above-ground vegetation characteristics, which are best described by species identity, vegetation height and plant density (Owers et al., 2018) were measured within each sampling plot (1m × 1m quadrat). Average vegetation height was taken at 10 representative positions within each plot. Plant cuttings were removed from a 25 cm^2 area within each plot and oven-dried (60°C, 72 hrs) for above-ground biomass calculations. Saltmarsh plants were identified to species level and percentage cover of individual species was recorded to give an indication of community composition. Each quadrat/ sampling point was then categorised following National Vegetation Classification (NVC) and corresponding marsh zone (Rodwell *et al.,* 2000; Appendix C2). Species richness and Shannon-Wiener's Diversity Index (H') were also calculated per plot:

$$H' = \sum_{i=1}^{s} [(p_i) * \ln(p_i)]$$

where:

p_i = the proportion individuals found in the *i*th species
ln = the natural logarithm
s = the number of species in the community

5.2.2.3 Soil characteristics

Soil cores (diameter: 5 cm; depth: 30 cm) were extracted using a gouge auger and sub-sampled in-situ at 0 – 10 cm, 10 – 20 cm and 20 – 30 cm depth intervals per sampling point. A depth of 30 cm was chosen in line with other studies that have quantified carbon stocks in MR sites (Burden *et al.,* 2019). Regular monitoring at Tollesbury MR has shown mean annual sedimentation rates were 2.3 cm between 1995 and 2001 (Garbutt *et al.,* 2006). When the sampling for this study took place in 2019, it would have been 24 years post-breach at Orplands and Tollesbury MRs, and 28 years post-breach at Northey Island MR. The top 30 cm of soil for all MR sites in this study therefore likely represents post-realignment accumulation. Attempts to visually identify boundaries in the facies between pre- and post-restoration deposits per soil core were inconclusive.

Subsamples from each core were oven-dried (60 °C, minimum 24 hrs). At temperatures exceeding 60 °C, fractions of soil organic matter may oxidise and cause an under-estimation of organic carbon (Howard *et al.,* 2014). Dry bulk density was calculated by dividing the dry weight of soil by sample volume. Dried samples were then ground, sieved (1 mm mesh size) to remove large inorganic particles, and sub-sampled to provide material for total organic carbon and particle size analysis.

Total organic carbon (TOC) was measured using a Carbon and Nitrogen elemental analyser (Flash 2000, Thermo Fisher Scientific). The instrument was calibrated using aspartic acid as a standard. Prior to TOC analysis, samples were homogenised to a fine powder using a ball mill (Retsch MM200; 250µm, 2 mins). Between 10 – 15 mg of sample was placed into silver capsules and treated with 10% HCl acid to remove carbonates ($CaCO_3$) (Smeaton *et al.,* 2022). Following acidification, samples were not observed to effervesce, suggesting that inorganic carbon represented a minimal component of total sediment weight (Howard *et al.,* 2014). The acidified samples were then dried overnight at 50°C and sealed (Smeaton *et al.,* 2022). A replicate was taken every 5 samples to assess the reliability of %TOC data.

Prior to particle size analysis, dried soil samples (~2 g) were digested in 10ml 30% hydrogen peroxide (H_2O_2) and heated on a hotplate (50°C). To remove all residual organic material, H_2O_2 was continuously added until reactions had ceased. Samples were then heated to reduce liquid content and left to cool prior to centrifuging (3 times: 3,500 rpm; 8 minutes). To prevent flocculation, 2ml sodium hexametaphosphate $[(NaPO_3)_6]$ was added to samples. Particle size was measured using a laser granulometer (Malvern Mastersizer Hydro 2000), and test sand was used to calibrate the instrument. Outputs from the laser granulometer were given as percentages of different size categories, and then grouped into 3 main categories (clay, silt, and sand) using the Wentworth (1922) classification scheme. Clay particles were not detected in any samples during particle size analysis. A sand:silt ratio was then calculated for data analysis:

$$\text{Sand: silt ratio} = \frac{\text{Percentage of soil sample composed of sand particles (\%)}}{\text{Percentage of soil sample composed of silt particles (\%)}}$$

5.2.2.4 Calculation of organic carbon stock

Organic carbon soil stocks were calculated using soil core depth (30 cm), depth interval of subsamples (10 cm), dry bulk density (DBD) and total organic carbon (TOC). Soil organic carbon density (SOC) of each soil core was calculated using the approach outlined in the Coastal Blue Carbon Manual (The International Blue Carbon Initiative; Howard *et al.,* 2014) as follows:

$$\mathrm{SOC}\ (\mathrm{g\ C\ cm^3}) = \mathrm{Subsample\ DBD}\ (\mathrm{g\ cm^{-3}}) \times \frac{\mathrm{subsample\ TOC}\ (\%)}{100}$$

Organic carbon content (g C cm^3) for each subsample was multiplied by the depth interval of the subsample (10 cm) and summed across the three subsamples (0 – 10 cm, 10 – 20 cm, and 20 – 30 cm). Organic carbon stocks were converted to Mg C ha^{-1} to 30 cm for each sampling point. Mean organic carbon stock and standard deviation for the top 30 cm for each site was then calculated using each sampling point within a site.

5.2.3 Data analysis

Statistical analysis was conducted using R software (version 4.1.2). Site differences between each variable were tested using ANOVA. If present, significantly different means were identified using post-hoc Tukey tests. For categorical variables, including those with unequal sample sizes, non-parametric Kruskal-Wallis and post-hoc Dunn tests were used instead. When variables did not meet the assumption of normality, log and exponential transformations were applied and retested using Shapiro-Wilk and diagnostic plots. All models met the assumptions of normality, homoscedasticity, and leverage.

Multivariate Generalised Linear Models ("MASS" package; Venables & Ripley, 2002) were used to determine the relationship between organic carbon stock and a set of predictor variables. *A priori* knowledge about the data was used to determine the error distribution and link function for models. Model residuals were checked for normality, homoscedasticity, and bias by unduly influential observations using diagnostic plots, supported by the Shapiro-Wilk test for normality, and Breusch-Pagan test for heteroscedasticity. A log link function was used with a gaussian error family to reduce high leverage values observed in model residuals. When the response variable (carbon stock) was positively skewed, a Gamma error distribution and log link function was used to meet model assumptions. When heteroscedasticity was present and/ or model residuals were non-normally distributed after changing the error family or link function, the dependent variable (carbon stock) was log transformed. Diagnostic plots and Akaike Information Criterion (AIC) were used to confirm error family and link function choices during model selection. Models with the lowest AIC value were deemed best fit for the data. Model residuals were also tested for serial autocorrelation to determine whether a mixed effects model with a random component was necessary. Auto-Correlation Function plots and Durbin-Watson tests showed model residuals did not contain serial autocorrelation and therefore Multivariate Generalised Linear Models were an appropriate method for analysis.

Predictor variables were tested for correlation using Pearson's coefficient (r) and Variance Inflation Factors (VIF). Correlation was considered high if $r \geq 0.6$ and/ or VIF ≥ 3 (Zuur *et al.,* 2010). Predictor variables that displayed high levels of collinearity were not included in the same model (Appendix C1). The stepAIC function ("MASS" package; Venables & Ripley, 2002), based on Akaike information criterion (AIC), was used in model selection to produce an optimal model. Analysis of variance was used to test the probability of decreased deviance between the optimal and full model. Finally, for each optimal model, the deviance explained by the model was calculated [1 – Residual deviance/ Null deviance]. Where multiple predictors were retained in the optimal model, hierarchical partitioning using the "hier.part" package (Mac Nally & Walsh, 2004) was used to identify independent effects (%) that each predictor contributed to the optimal model. All figures were created using the "ggplot2" package (Wickham, 2016).

5.3 Results

Variation in carbon stock and environmental variables were compared between sites, and management history (managed realigned and natural saltmarshes). Environmental variables were then analysed against carbon stock to determine the key drivers of variation.

5.3.1 Variation among sites

5.3.1.1 Carbon stock

Mean carbon stock within saltmarsh soil (top 30 cm) of all six sites was 78.9 ± 27.5 Mg C ha^{-1} (n = 70). Natural sites had a mean carbon stock (87.7 ± 8.4 Mg C ha^{-1}) greater than MR sites (72.1 ± 25.8 Mg C ha^{-1}). Among all six sites, carbon stock ranged between 43.3 and 94.5 Mg C ha^{-1} (Table 5.2). Mean carbon stock was highest in the natural site at Orplands (Table 5.2; Fig. 5.2). Tollesbury MR had the lowest mean carbon stock which was 36.3 Mg C ha^{-1} less than the MR site at Orplands, and 50 Mg C ha^{-1} less than the MR site at Northey Island (Table 5.2; Fig. 5.2). The natural site at Tollesbury had a lower carbon stock than the MR and natural sites at Northey Island and Orplands (Table 5.2; Fig. 2).

Table 5.2. Mean carbon stock (Mg C ha^{-1}) in the top 30 cm of saltmarsh soil at managed realigned (MR) and natural (N) sites ± standard deviation. Sites are presented in decreasing order of mean carbon stock.

Site	Mean carbon stock (Mg C ha^{-1})
Orplands N	94.5 ± 23.8
Northey Island MR	93.3 ± 36.4
Northey Island N	90.2 ± 19.6
Orplands MR	79.6 ± 21.2
Tollesbury N	78.3 ± 13.6
Tollesbury MR	43.3 ± 11.2

Among all sites, carbon stock was significantly different ($N = 70$; $F = 8.796$; $p < 0.001$). Mean carbon stock at Tollesbury MR was significantly lower than the MR sites at Northey Island and Orplands ($p < 0.001$ and $p = 0.001$ respectively), and the natural sites at Northey Island ($p < 0.001$), Orplands ($p < 0.001$), and Tollesbury ($p = 0.02$). When excluding Tollesbury MR, there was no significant difference in mean carbon stock between sites ($N = 58$; $F = 1.286$; $p = 0.29$; Fig. 5.2) and mean carbon stocks between natural and MR sites were similar (87.7 ± 8.4 and 86.5 ± 9.7 Mg C ha^{-1}, respectively).

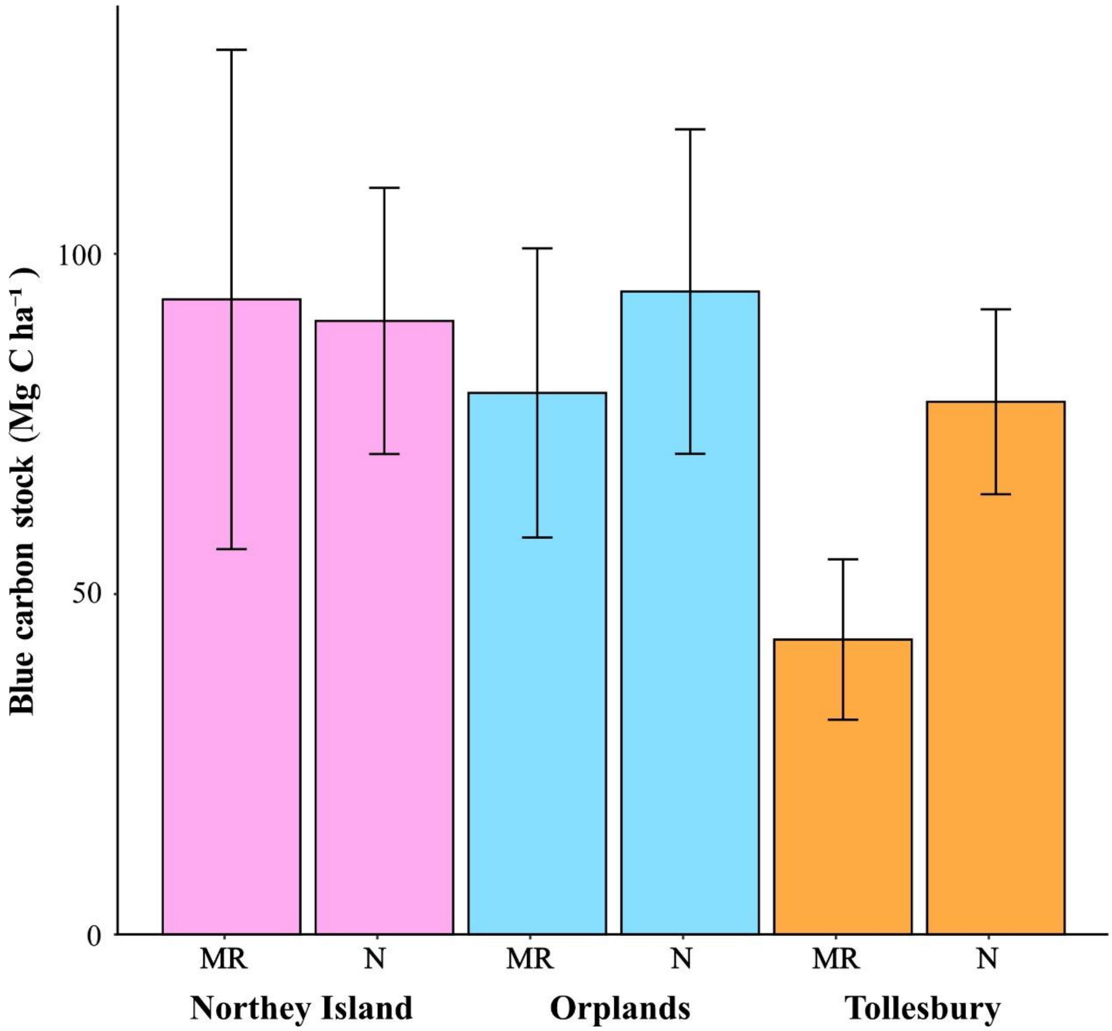


Figure 5.2. Mean carbon stock ± standard deviation in the top 30cm of soil for managed realigned (MR) and natural (N) saltmarsh sites at Northey Island (pink), Orplands (blue) and Tollesbury (orange) in the Blackwater Estuary, Essex, UK.

5.3.1.2 Environmental variables

Full statistical results for ANOVA and Kruskal-Wallis tests, and subsequent post-hoc tests, for environmental variables can be found in Appendix C3 (a – h). Our main findings are outlined below.

The mean dry bulk density (DBD) of all samples in this study was 0.64 ± 0.19 g cm^{-3}. However, the DBD at Northey Island and Orplands MR sites (0.82 ± 0.23 g cm^3 and 0.79 ± 0.20 g cm^{-3}, respectively; Table 5.3) was significantly higher than the three natural sites. DBD did not differ significantly between Tollesbury MR and any other site, nor between natural sites. Soil samples from all sites were predominantly composed of silt particles, according to classifications from the Wentworth scale. The percentage of silt recorded in samples ranged from 63.1 – 97.5 %, with the remainder of samples comprised of sand particles. However, the natural site at Orplands had a significantly higher sand:silt ratio (Table 5.3), and thus a significantly higher proportion of sand particles than all other sites. The MR and natural sites at Northey Island and Tollesbury had a similar sand:silt ratio (ranging between 0.12 ± 0.07 and 0.14 ± 0.11; Table 5.3). The sand:silt ratio at Orplands MR was 2-fold lower than the site with the next lowest value, whilst the natural site at Orplands was over 2 times greater than the next highest site (Table 5.3). Dry bulk density and particle size were not highly correlated with each other or any other variable (Appendix C1).

The mean relative tidal height of saltmarshes, where 0 represents MHWN and 1 represents MHWS tidal level, of natural sites was 0.89 ± 0.12. This was over 2-fold higher than the value obtained for MR sites (0.40 ± 0.22). Of our sites, only Tollesbury MR had a mean relative tidal height below the MHWN tidal level (- 0.14 ± 0.10; Table 5.3), and this was significantly lower in the tidal frame than all other sites. The natural site at Orplands was the only site with a mean relative tidal height above MHWS tidal level (1.07 ± 0.09; Table 5.3) and was significantly higher than all other sites. Relative tidal height was positively correlated with marsh plant zone ($r = 0.61$; $t = 6.32$; $p < 0.001$; Appendix C1) and NVC community ($r = 0.62$; $t = 6.40$; $p < 0.001$; Appendix C1).

With the exception of Tollesbury MR, there was no significant difference in marsh zone or NVC community between sites or management history. Marsh zone at Tollesbury MR was significantly different to all other sites, and NVC community was significantly different between Tollesbury MR and all other sites except for Orplands MR. The most dominant vegetation species recorded at all natural sites and Northey Island MR was *Atriplex portulacoides* (NVC code: SM14), representative of the mid-low marsh zone (Table 5.3, Appendix C2). Another mid-low marsh species, *Puccinellia maritima* (NVC code: SM13), was the most dominant vegetation at Orplands MR (Table 5.3, Appendix C2). Tollesbury MR was the only site where the vegetation community composition was predominantly pioneer species (*Spartina anglica*, NVC code: SM8; Table 5.3, Appendix C2). Marsh zone and NVC community were highly positively correlated with each other ($r = 0.86$; $t = 14.10$; $p < 0.001$), and with relative tidal height.

The vegetation community at Tollesbury MR, described by the Shannon-Weiner Diversity Index, was the least diverse (Table 5.3) and significantly lower in diversity than all other sites, except for Northey Island MR. The natural site at Orplands had the highest vegetation diversity

(Table 5.3) which was significantly higher than all MR sites. Mean species richness recorded at Tollesbury MR (1.75 ± 0.62; Table 5.3) was significantly lower than all three natural sites. Tollesbury MR was the only MR site to differ significantly to natural sites, and otherwise, there was no significant difference between natural sites or MR sites. The MR site at Northey Island had the next lowest mean species richness (3.70 ± 1.49; Table 5.3), yet this was still more than 2-fold higher than Tollesbury MR. Shannon-Weiner Diversity Index and species richness were highly correlated parameters ($r = 0.85$; $t = 13.50$; $p < 0.001$; Appendix C1).

Above-ground biomass at Tollesbury MR (17.47 ± 8.24 g m^{-2}; Table 5.3), the only site predominantly composed of pioneer species, was significantly lower than all other sites. The greatest difference was observed between Tollesbury MR and the natural sites, particularly at Northey Island and Orplands which had the greatest above-ground biomass (66.53 ± 40.69 and 62.00 ± 37.76 g m^{-2}, respectively; Table 5.3). The MR sites at Northey Island and Orplands did not differ significantly to any of the three natural sites, although above-ground biomass was higher for all three natural sites when compared to MR sites. Above-ground biomass showed little correlation with any other variable (Appendix C1).

Table 5.3. Mean environmental variables at each natural (N) and managed realignment (MR) site ± standard deviation. Sites are presented in decreasing order of mean carbon stock (see Table 2). Most dominant vegetation species recorded at each sampling point was used to assign marsh zone and NVC community (Appendix C2). A dominant vegetation species and marsh zone was assigned to each site based on the most frequently recorded species for each sampling point within a site.

Study site	**Dry bulk density ($g\ cm^{-3}$)**	**Sand:silt ratio**	**Relative tidal height**	**Dominant marsh zone**
Orplands (N)	0.49 ± 0.09	0.32 ± 0.19	1.07 ± 0.09	Mid-low
Northey Island (MR)	0.82 ± 0.23	0.13 ± 0.10	0.62 ± 0.23	Mid-low
Northey Island (N)	0.55 ± 0.18	0.12 ± 0.07	0.79 ± 0.15	Mid-low
Orplands (MR)	0.79 ± 0.20	0.06 ± 0.03	0.72 ± 0.32	Mid-low
Tollesbury (N)	0.57 ± 0.11	0.13 ± 0.09	0.81 ± 0.11	Mid-low
Tollesbury (MR)	0.61 ± 0.09	0.14 ± 0.11	-0.14 ± 0.10	Pioneer
Study site	**Dominant plant species***	**Shannon's Diversity Index**	**Species richness**	**Above-ground biomass ($g\ m^{-2}$)**
Orplands (N)	*Atriplex portulacoides*	1.24 ± 0.39	6.18 ± 1.66	62.00 ± 37.76
Northey Island (MR)	*Atriplex portulacoides*	0.73 ± 0.50	3.70 ± 1.49	38.86 ± 17.58
Northey Island (N)	*Atriplex portulacoides*	1.04 ± 0.55	5.80 ± 2.35	66.53 ± 40.69
Orplands (MR)	*Puccinellia maritima*	0.76 ± 0.34	4.29 ± 1.49	43.58 ± 23.12
Tollesbury (N)	*Atriplex portulacoides*	1.14 ± 0.27	4.85 ± 1.57	51.10 ± 30.14
Tollesbury (MR)	*Spartina anglica*	0.28 ± 0.27	1.75 ± 0.62	17.47 ± 8.24

5.3.2 Drivers of carbon stock

Four models were run to determine the key environmental drivers of carbon stock for all sites. First, drivers of carbon stock were examined across all sites. Given that Tollesbury MR had a significantly lower carbon stock than other MR sites, and so may account for the majority of variance in carbon stock, we also investigated whether drivers were consistent across all sites excluding Tollesbury MR. Drivers of carbon stock were then compared between natural and MR sites, with the exclusion of Tollesbury MR. A summary of model outputs can be seen below in Table 5.4.

Table 5.4. Multivariate General and Generalised Linear Models of carbon stock vs environmental predictors. Optimal model produced during model selection through the stepAIC function. Statistics include the direction of the relationship (positive +, negative -), probability of deviation from null hypothesis (p value), and the percentage deviance explained. Where optimal models retained more than 1 variable, individual contributions to deviance explained (%) are shown. Bold type indicates significance below the 0.05 level. Generalised linear model error family and link function for each model are as follows: Model 1 (gaussian|log); Model 2 (Gamma|log); Model 3 (Gamma|log); Model 4 (gaussian|log).

Optimal model	AIC	p value	Deviance explained (%)
Model One (all sites)			37
Elevation (+)	34.38	**< 0.001**	
Model Two (all sites excl. Tollesbury managed realignment)			9
Sand:silt ratio (+)	520.75	**0.03**	
Model Three (natural sites)			33
Dry bulk density (-)		**0.07**	31
Sand:silt ratio (+)	-15.40	**0.01**	69
Model Four (managed realigned sites excl. Tollesbury)			43
Marsh zone (+)		**0.03**	46
Shannon's Diversity Index (-)	216.49	**0.03**	41
Dry bulk density (-)		0.05	13

5.3.2.1 All sites

For all sites and management history, only relative tidal height was found to be a positive significant predictor of carbon stock (AIC = 34.38; $p < 0.001$; Table 5.4): carbon stock was lower when sampling points were lower in the tidal frame (Fig. 5.3). The optimal model retained only relative tidal height as a variable, and explained 37% of the observed variation in carbon stock (Table 5.4). In Tollesbury MR, all sampling points bar one were below Mean High Water Neap (Fig. 5.3). Carbon stock for sites higher in the tidal frame showed no apparent relationship with relative tidal height (Fig. 5.3). For this reason, another model was run for all sites excluding Tollesbury MR to see if other important relationships were masked.

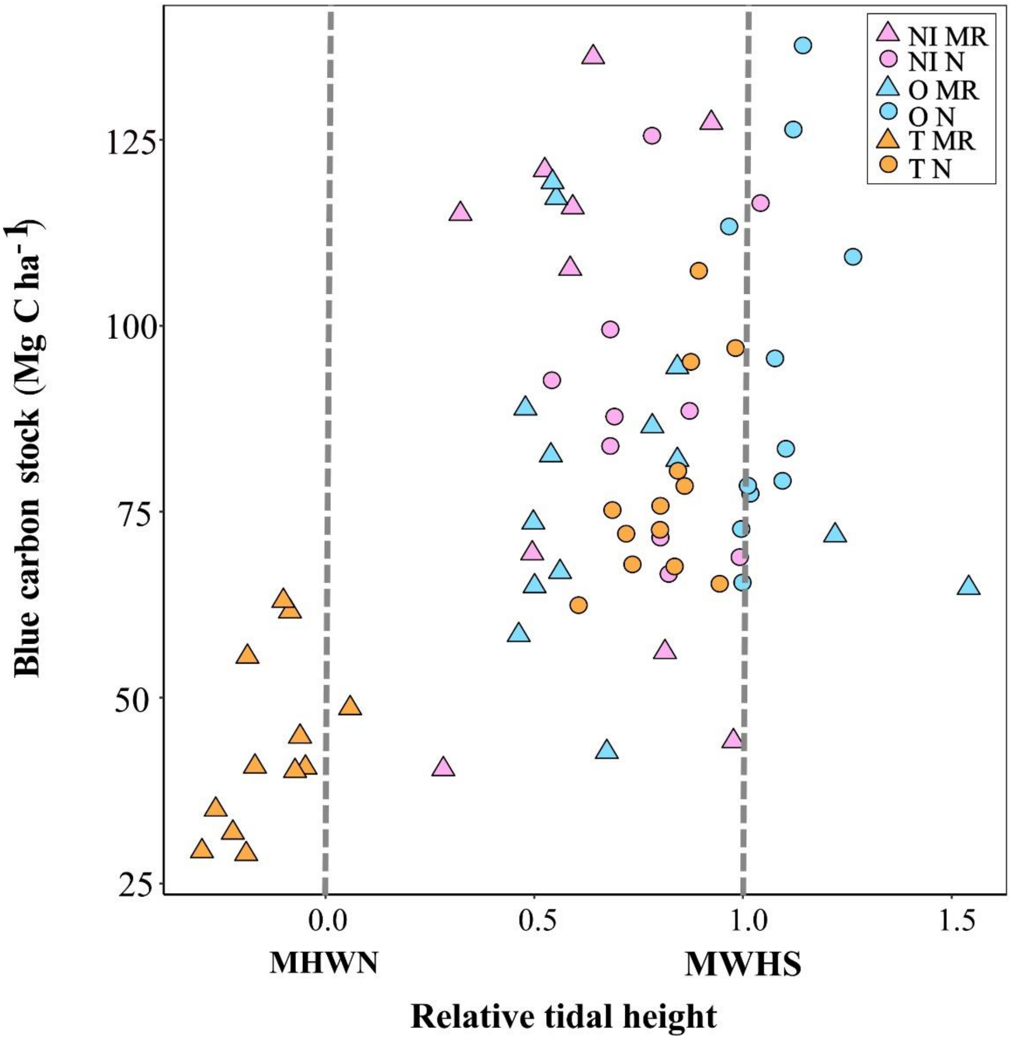


Figure 5.3. Carbon stock against relative tidal height for managed realigned (MR; triangle) and natural (N; circle) sites [NI = Northey Island, O = Orplands, T = Tollesbury]. Saltmarshes generally occur between mean high water neap (MHWN) and mean high water spring (MHWS) tidal levels (dashed grey lines; Lawrence *et al.,* 2018).

Relative tidal height was positively correlated with marsh zone and NVC community (Appendix C1), so that pioneer species (e.g. *Salicornia europaea* and *Spartina anglica*) were found lower in the tidal frame than vegetation communities associated with the mid-low marsh zone (Appendix C2). The highest carbon stocks were therefore found in the mid-low marsh zone. Tollesbury MR was the only site without mid-low plant communities recorded in sampling. Instead, vegetation communities recorded at all sampling points in Tollesbury MR were exclusively pioneer species *Spartina anglica* and *Salicornia europaea*.

5.3.2.2 Excluding Tollesbury MR

To determine drivers of carbon stock for sites above MHWN, Tollesbury MR was excluded in a second model. Relative tidal height was dropped as a predictor of carbon stock variability, whilst sand:silt ratio was retained and had a marginally significant and positive relationship with carbon stock (AIC = 520.75; $p = 0.03$). ratio. Samples with a higher percentage of sand particles generally had higher carbon stock. All samples across sites included in this model were predominantly composed of silt (between 63 to 97 %), with the remainder composed of sand particles. The optimal model, retaining only sand:silt ratio, explained very little of the variation in carbon stock (9 %), meaning 91 % of the observed variation in carbon stock for all sites, excluding Tollesbury MR, was unaccounted for.

5.3.2.3 Natural vs managed realignment sites (excl. Tollesbury MR)

Excluding Tollesbury MR, we also investigated whether predictors of carbon stock differed between natural and MR sites. For natural sites, dry bulk density and sand:silt ratio were retained in the optimal model (AIC = -15.40). Of the retained variables, only sand:silt ratio was identified as a significant predictor, showing a positive relationship with carbon stock ($p = 0.01$). Whilst the sand:silt ratio at Northey Island and Tollesbury natural sites were similar, and the soil in all 3 natural sites was predominantly composed of silt particles, the mean sand:silt ratio at Orplands was over 2-fold greater (Table 5.3; Fig. 5.4) than the other natural sites. The mean percentage of silt in the samples from Northey Island and Tollesbury natural sites was 89 % and 88 %, respectively, with the remainder composed of sand particles, whereas the mean percentage of silt for the natural site at Orplands was 77 %.

For natural sites, sand:silt ratio was positively correlated with relative tidal height, whereby samples with a lower proportion of sand particles in the predominantly silty sediment were found lower in the tidal frame. The natural site at Orplands was the only site with a mean relative tidal height above MHWS tidal levels (Table 5.3; Fig. 5.3).

The optimal model explained 33% of the total variation in carbon stock, of which hierarchical partitioning showed that sand:silt ratio accounted for the most variation (69 %). DBD accounted for the remaining explained variation (31 %) but was an insignificant predictor of carbon stock.

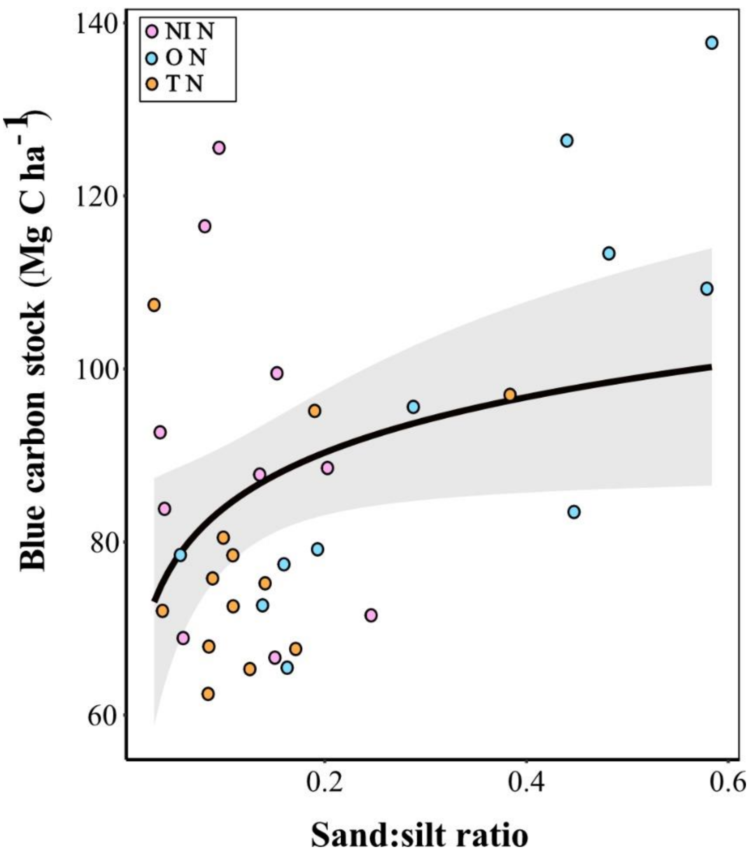


Figure 5.4. Carbon stock against Sand:silt ratio for natural (N) sites at Northey Island (NI; pink), Orplands (O; blue), and Tollesbury (T; orange) with a logarithmic curved line and 95 % confidence interval (grey bar). The higher the sand:silt ratio, the greater the proportion of sand particles in the predominantly silty sediment.

Across MR sites (excluding Tollesbury), marsh zone ($p = 0.03$) and Shannon's Diversity Index ($p = 0.03$) were significant predictors of carbon stock in the optimal model (AIC = 216.49). Soil carbon stock was higher in sampling points with a less diverse vegetation community composition, as described by Shannon's Diversity Index (Fig. 5.5b). Conversely, vegetation communities associated with the mid-low marsh had higher carbon stock than pioneer zones (Fig. 5.5a). However, associated soil carbon stock within both the mid-low and pioneer marsh zones varied depending on NVC communities (Fig. 5.5a).

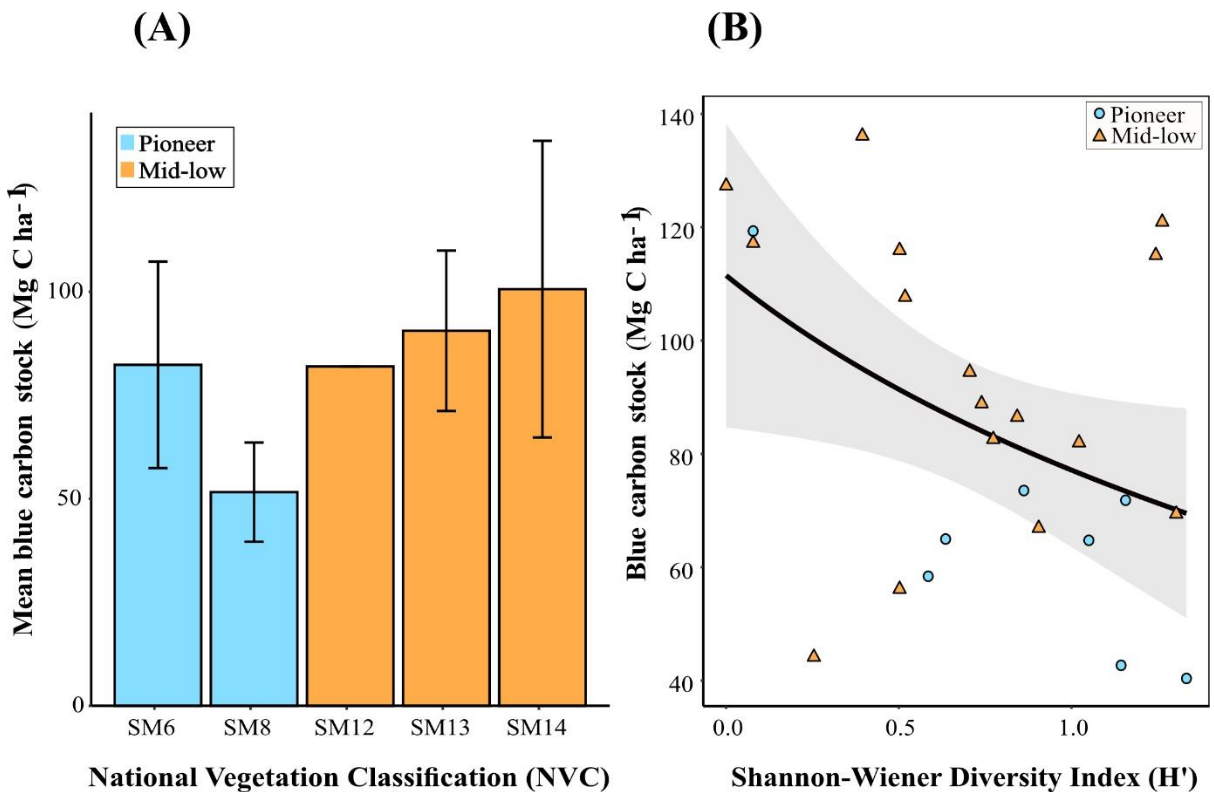


Figure 5.5 (A) Mean carbon stock ± standard deviation for each dominant National Vegetation Classification (NVC) community found in the managed realigned sites at Northey Island and Orplands (pioneer communities = blue; mid-low communities = orange). Number of observations varied across NVC groupings: SM6 (n = 4), SM8 (n = 4), SM12 (n = 1), SM13 (n = 8), and SM14 (n = 7). See Appendix C2 for NVC categories and corresponding marsh zone; and **(B)** Carbon stock against Shannon-Wiener Diversity Index for Northey Island and Orplands managed realigned sites, with a logarithmic curved line and 95 % confidence interval (grey bar). Data points correspond to marsh zones (pioneer = blue circle; and mid-low = orange triangle). The higher the H' value, the more diverse the vegetation community.

At Northey Island and Orplands MR sites, the following NVC communities were recorded: SM6, SM8, SM12, SM13, and SM14 (Fig. 5.5a; Appendix C2). Of these, SM6 and SM8 communities dominated by *Spartina anglica* and *Salicornia europaea*, respectively, are pioneer communities. The remaining communities (SM12, SM13, and SM14) are associated with the mid-low marsh zone (Fig. 5.5a; Appendix C2). The mean carbon stock in pioneer communities (66.98 ± 24.45 Mg C ha^{-1}) was lower than in vegetation communities associated with the mid-low marsh (94.44 ± 26.92 Mg C ha^{-1}) at these sites (Fig. 5.5a; Appendix C2). SM14 communities, dominated by *Atriplex portulacoides* (Appendix C2), had the lowest mean diversity index value (Fig. 5.5b) of all communities present in Northey Island and Orplands MR sites, but were associated with the greatest mean soil carbon stock (100.64 ± 35.82 Mg C ha^{-1}; Fig. 5.5a). Above-ground biomass showed a marginally significant correlation with NVC community ($r = 0.42$; $t = 2.14$; $p = 0.04$) for the MR model excluding Tollesbury (Table 5.4), but was dropped from the optimal model with no loss in explanatory power.

Dry bulk density was also retained in the optimal model but had a marginally insignificant, negative relationship with carbon stock ($p = 0.05$). The optimal model explained 43 % of the variation in carbon stock. Biological characteristics relating to plant community composition and diversity were responsible for 87 % of the variation (marsh zone: 46 %; Shannon's Diversity Index: 41 %), whilst dry bulk density explained 13 %.

5.4 Discussion

This study demonstrates that blue carbon stocks in managed realigned sites exhibit low spatial variability at an estuarine scale, with important implications for the inclusion of restoration schemes in climate change mitigation strategies. However, carbon stocks are significantly higher in MR sites elevated above MHWN tidal levels, which are associated with more mature saltmarsh plant communities. Sedimentary drivers appear more influential in explaining the carbon stock variability between natural saltmarshes. These findings provide a useful insight into environmental drivers that control blue carbon stocks in both natural and MR saltmarshes, which can be used to inform future restoration schemes that aim to maximise climate change mitigation.

We found that surficial (top 30 cm) soil carbon stocks in our MR sites become equivalent to natural saltmarsh stocks after 25 – 29 years post-breaching, but importantly only when sites were above MHWN tidal levels. A threshold relationship existed where mean carbon stocks almost doubled beyond MHWN elevations. The timescale to develop equivalent carbon stocks to natural saltmarshes is faster than reported in other studies for MR sites (Burden *et al.,* 2013; Burden *et al.,* 2019) and restored wetlands in general (Moreno-Mateos *et al.,* 2012). We attribute this to the vegetation community composition, an important biological indicator of the blue carbon capacity of saltmarshes (Rogers *et al.,* 2019b). Even the presence of a single key species is known to enhance soil carbon stocks in restored marshes (Zhao *et al.,* 2018). Specifically, at MR sites above MHWN tidal levels, we observed the successful colonisation and establishment of a considerable proportion of late-successional plant species from the local species pool, and site dominance by key mid-low plant species, such as *Atriplex portulacoides*, in comparison to the dominance of pioneer communities that often characterise MR sites (Garbutt *et al.,* 2006; Mossman *et al.,* 2012a; Brooks *et al.,* 2015). Key plant species (e.g., *Atriplex portulacoides*) therefore appear to rapidly enhance the carbon storage value of MR sites once suitable inundation characteristics are reached.

The lowest carbon stocks were found at Tollesbury MR, the only site with a mean relative tidal height below MHWN tidal levels (Fig. 3) at the lower limit of pioneer marsh establishment (Adam, 2002; Balke *et al.,* 2016; Lawrence *et al.,* 2018; Hudson *et al.,* 2021). Over a decade prior to our research, Adams *et al.* (2012) also observed Tollesbury MR to be dominated by pioneer communities of *Spartina anglica* and *Salicornia europaea*, indicative of the site's low position in the tidal frame. Across Great British saltmarshes, Smeaton *et al.* (2022) reported lower soil organic carbon stocks for pioneer communities when compared to communities associated with higher elevations, such as mid-low and high marsh zones. For MR sites higher in the tidal frame than MHWN tidal levels, saltmarsh plant species from the local species pool are expected to quickly colonise following pioneer establishment (Adams *et al.,* 2012) – on timescales as short as 4 years (Davy *et al.,* 2011; Mossman *et al.,* 2012c). Even 25 years post-restoration, Tollesbury MR remains below MHWN tidal levels. Time since restoration therefore is not necessarily a strong predictor of carbon stock in MR sites (Adams *et al.,* 2012; Abbott *et al.,* 2019; Burden *et al.,* 2019). The blue carbon potential of proposed MR schemes thus need to account for local scale controls of carbon variability in MR sites, which is of particular importance when including coastal wetland restoration in climate change mitigation strategies through robust blue carbon estimates (Hayes *et al.,* 2017).

Unless action is taken to increase the 'elevation capital' (Cahoon *et al.,* 2019; Langston *et al.,* 2021) of planned MR sites prior to breaching, such as through the use of dredged estuarine material (e.g. French, 2006; Kadiri *et al.,* 2011; Stagg & Mendelssohn, 2010), MR sites are often lower in the tidal frame than natural saltmarshes due to historical subsidence and inhibited sediment supply from the tide (Oosterlee *et al.,* 2018). The relationship between elevation in the tidal frame and redox potential is well established for MR sites across the UK (Davy *et al.,* 2011; Mossman *et al.,* 2012a; Mossman *et al.,* 2020). More frequent and prolonged inundation of low elevation sites, coupled with poor drainage of anoxic sediments, means reduced gas exchange, and subsequent low sediment redox potential can characterise MR sites (Castillo *et al.,* 2000; Garbutt *et al.,* 2006; Davy *et al.,* 2011; Mossman *et al.,* 2012a; Mossman *et al.,* 2020). Such conditions can inhibit the colonisation of plant communities associated with mid-low and high marsh zones, favouring more tolerant pioneer communities (French, 2006; Wolters *et al.,* 2008; Mossman *et al.,* 2012c), as is the case for Tollesbury MR. If the elevation of Tollesbury MR is unable to increase, whether naturally through sediment trapping and accretion or artificially, pioneer communities are likely to persist (Garbutt *et al.,* 2006). Sediment additions that raise elevations of MR sites comparable with mid to high marsh zones have been successful in restoring ecosystem functionality, and may also encourage long-term resilience to accelerated sea-level rise, particularly in regions with low sediment supply (Stagg & Mendelssohn, 2010; Langston *et al.,* 2021).

The capacity of a newly restored saltmarsh to increase elevation and facilitate vegetation development is usually a key aspect of restoration (Davy *et al.,* 2011), yet often relies on a suitable elevation of the tidal flat for pioneer vegetation to establish and for the positive feedback of elevation gain to commence (D'Alpaos, 2011). For MR sites low in the tidal frame, considerable sediment accretion is needed to raise the surface above MHWN tidal levels to elevations suited for vegetation colonisation (Garbutt *et al.,* 2006; Davy *et al.,* 2011). When hydrology is restored through breaching at MR sites, the resulting accommodation space creates a settling basin for sediments. Rapid elevation gain can ensue provided a sufficient sediment supply exists (Liu *et al.*, 2021) and that shallow compaction does not offset elevation gain (Saintilan et al., 2022). Soil carbon stocks in the early stages of MR are also developed if the deposited sediment is rich in organic material (Friess *et al.,* 2012; Hansen *et al.,* 2017; Wollenberg *et al.,* 2018). Sediment accumulation helps to ameliorate redox conditions via elevation gain and allows the colonisation of mid-low and high marsh species associated with a higher carbon stock potential (Wollenberg *et al.,* 2018; Mossman *et al.,* 2020; Smeaton *et al.,* 2022). Sediment supply, alongside other parameters, such as breach width, may also account for some of the unexplained variance in the data.

Saltmarshes in southeastern England (the study region for this research) appear to have limited sediment supply (Ladd *et al.,* 2019). In the Blackwater Estuary, suspended sediment concentrations (~ 50 – 150 mg/l), and thus sediment supply, are relatively low in comparison to other UK estuaries such as the Parrett Estuary in southwestern England which has a suspended sediment concentration in the range of 1,000 – 10,000 mg/l (Manning *et al.,* 2010; Spearman, 2011; Mossman *et al.,* 2022). Tollesbury MR was implemented at initial elevations lower in the tidal frame than Northey Island and Orplands MR (Wolters *et al.,* 2005). The lower sediment supply experienced in the region (Ladd *et al.,* 2019), exacerbated by the construction of a counter sea wall which restricted tidal inundation and further dampened sediment supply (Garbutt *et al.,* 2008), likely explains why Tollesbury MR remains in a low-elevation state despite 25 years since restoration. When saltmarsh restoration occurs at extremely low elevations in the tidal frame, such as Tollesbury MR, sediment supply should thus be a key consideration in forecasting the likely development of the MR site and carbon stocks. Furthermore, reliable sediment supply is required to ensure the restoration success of saltmarshes under projected sea-level rise (Liu *et al.,* 2021), regardless of initial elevation, which is vital in securing the longevity of vegetation communities and associated carbon storage.

Studies on natural blue carbon habitats (saltmarshes, seagrasses, and mangroves) report high spatial heterogeneity at the estuarine level (Broek *et al.,* 2016; Ricart *et al.,* 2020; Suello *et al.,* 2022). Conversely, we observed relatively low spatial variability in carbon stock among natural and MR saltmarshes in the Blackwater Estuary, with the exception of Tollesbury MR. We highlight the variability seen between Tollesbury MR and the other MR sites in the estuary is mostly linked to the vegetation community composition, whereas sediment properties are a more influential driver for natural saltmarshes.

Numerous studies have reported higher carbon stocks associated with fine-grained sediments, such as clay and silt, due to their enhanced preservation capacity over coarser sediments (Kelleway *et al.,* 2016a; Zhao *et al.,* 2018; Ford et al., 2019; Rogers *et al.,* 2019b). However, we found an opposite relationship where a greater proportion of sand particles in the soil were associated with higher carbon stocks in natural saltmarshes. It is important to note that soil samples across all sites were predominantly composed of silt, reflective of the wider Essex region (Ford *et al.,* 2016). Sand particles are coarser than silt particles, with higher hydraulic conductivity and drainage capacity (Crooks *et al.,* 2002). These factors are associated with better aerated sediment and are likely to enhance redox potential (Mossman *et al.,* 2012a). This could potentially create optimal conditions for greater above-ground biomass whereby greater carbon sequestration is supported mostly by autochthonous contributions (Kelleway *et al.,* 2016a). Indeed, we found the highest carbon stocks in the natural saltmarsh at Orplands, which not only had the greatest proportion of sand particles, but also exhibited among the greatest above-ground biomass and was the only site above MHWS tidal levels. The high elevation of this site suggests a lower opportunity for allochthonous carbon inputs, coupled with a lower capacity for tidal export of autochthonous carbon, due to reduced inundation frequency at high elevations (Kelleway *et al.,* 2016a). Similarly, in seagrass habitats, the expected positive relationship between fine-grained sediment and carbon stocks was poor in meadows of higher biomass when autochthonous contributions were high (Serrano *et al.,* 2016a).

The highest carbon stocks in MR sites were associated with mid-low plant communities, specifically *Atriplex portulacoides*, which was the dominant plant community at Northey Island, the MR site with the greatest carbon stock and a vegetation assemblage most similar to natural saltmarshes in the estuary. These findings corroborate a previous study (Adams *et al.,* 2012) which attributed the variability in carbon stock among MR sites in the Blackwater Estuary to the vegetation community composition. Communities of *A. portulacoides* form dominant, almost monospecific stands (Penk *et al.,* 2020), explaining why higher carbon stocks were associated with lower plant diversities for Northey Island and Orplands MR sites. Long-term monitoring at a restored saltmarsh site showed dominant plant species controlled and enhanced biomass, whilst reducing overall species richness, supporting the concept that greater diversity does not necessarily equate to enhanced ecosystem functioning (Doherty *et al.,* 2011). Furthermore, the shrub *A. portulacoides* has a complex branched and deep root system (Decuyper *et al.,* 2014), characteristics associated with greater belowground productivity and root biomass (Adams *et al.,* 2012) which contribute to enhanced carbon stocks (Chmura *et al.,* 2003; Ford *et al.,* 2016). For example, *Puccinellia maritima*, a shallow-rooted grass which forms mid-low plant communities together with other simple-rooted saltmarsh vegetation (Ford et al., 2019), was the dominant species at Orplands MR which had a lower carbon stock than Northey Island MR. As such, the role of belowground root biomass may represent an important environmental driver of spatial variability in carbon stocks between MR saltmarshes and should be explored in further studies. Over half of organic matter production in saltmarsh plants occurs in roots and rhizomes that are buried within the soil (Duarte *et al.,* 2013) and lower dry bulk densities have been associated with high biomass and abundance of roots in saltmarsh plant species (Santini *et al.,* 2019), hence the inverse relationship when carbon stock is higher at lower bulk densities observed across coastal blue carbon habitats (Dahl *et al.,* 2016; Barry *et al.,* 2018; Gao *et al.,* 2019; Lima *et al.,* 2020). In turn, root biomass is known to contribute significantly to carbon stocks (Jones & Donnelly, 2004; Ford *et al.,* 2019; Santini *et al.,* 2019; Chen *et al.,* 2022) and has been shown to be significantly lower in pioneer communities than for plant communities at higher elevations across UK saltmarshes (Harvey *et al.,* 2019).

It has been reported that plant communities in restored marshes differ from those found in natural saltmarshes, even after decades (Garbutt & Wolters, 2008; Moreno-Mateos *et al.,* 2012; Mossman *et al.,* 2012a). Such differences are thought to hinder the ability of restored marshes to reach functional equivalence with natural marshes (Wolters *et al.,* 2005; Mossman *et al.,* 2012a; Burden *et al.,* 2019). However, we found that although the vegetation communities at MR sites had a lower species diversity, species richness, and above-ground biomass than natural sites, these biological parameters did not differ significantly between natural and MR saltmarshes, nor did carbon stock, with the exception of Tollesbury MR. Our findings support those of Santini *et al.* (2019) who reported soil carbon stocks for the top 30 cm were similar between natural and regenerated saltmarshes in New South Wales, Australia, after 20 years, despite above-ground biomass being higher in natural marshes. The presence of key plant species, or dominant species within a community (e.g., *Atriplex portulacoides* in this study), is highly influential for productivity, and consequently carbon stocks, in restored marshes (Doherty *et al.,* 2011), and may hold more value than developing equivalent vegetation communities to nearby natural saltmarshes, depending on the overall goal of restoration schemes. However, *A. portulacoides* is restricted to areas of high redox potential and elevation with well-drained soils (Garbutt *et al.,* 2008; Davy *et al.,* 2011; Cott *et al.,* 2013; Mossman *et al.,* 2020) as it lacks the structural adaptations for flooding (Armstrong *et al.,* 1985), explaining its absence from Tollesbury MR which sits too low in the tidal frame. Experimental manipulations of topography at MR sites in southeastern England found *A. portulacoides* to respond well to raised elevations (Mossman *et al.,* 2020). Slight increases in elevation were sufficient to ameliorate redox conditions for the colonisation and survival of mid-marsh communities (Mossman *et al.,* 2020). Enhanced soil drainage, whether from increased elevation, or the creation of small creeks (Wolters *et al.,* 2005; Lawrence *et al.,* 2018), will improve redox conditions (Mossman *et al.,* 2020). In turn, primary productivity will increase (Stagg & Mendelssohn, 2010), boosting autochthonous carbon inputs, and providing a source of plant litter that contributes to carbon stocks (Kadiri *et al.,* 2011). Such interventions (e.g. enhanced elevation and/ or the creation of creeks/ shallow lagoons) have been shown to be cost-effective and efficient in altering plant communities in MR sites (Lawrence *et al.,* 2022). In addition, wave attenuation for coastal protection and the provision of heterogeneous habitat for biodiversity enhancement, the primary goals of managed realignment in the UK, benefit from dense vegetation canopies which are more associated with mid-low and high marsh plant communities (Möller & Spencer, 2002; Levin & Talley, 2002). *A. portulacoides* is a prominent species for saltmarshes in southern UK (Mossman *et al.,* 2012a), where our study sites are located. It represents a key species for MR schemes in the region to enhance carbon storage and supports the theory that ecosystem functioning may not be solely driven by diversity, but instead dominant species (Keer & Zedler, 2002). *A. portulacoides* is also an important species for promoting saltmarsh stabilisation and protection as the root systems provide high resistance to erosion (Chen *et al.,* 2019). However, Ford *et al.* (2016) demonstrated that saltmarsh plant diversity enhances sediment stability and erosion control, particularly in areas of sandy soils which are more prone to erosion, therefore trade-offs in ecosystem services must be considered in respect to the aim of restoration.

5.5 Conclusion and recommendations

Climate change mitigation has become a key goal for coastal wetland restoration projects, and blue carbon represents an important facet of future managed realignment schemes. We found carbon stocks in MR and natural saltmarsh sites in the Blackwater Estuary exhibited a relatively low spatial variability, with the exception of Tollesbury MR. Investigating a range of environmental drivers, we consider the comparatively low carbon stock at Tollesbury MR to be governed by local-scale controls including relative tidal height and vegetation community composition. The homogeneity of soil carbon stocks across MR sites has implications for estimating the climate change mitigation potential of proposed MR schemes.

Overall, we found relative tidal height to be a key driver of the capacity for MR sites to develop equivalent carbon stocks to natural marshes. We attributed this to sediment redox potentials and plant community composition, an important indicator for blue carbon capacity in saltmarshes (Rogers *et al.,* 2019b). The MR site at Tollesbury sits too low in the tidal frame (below MHWN tidal levels) and we consider the corresponding low redox potential to have created unsuitable conditions for mid-low marsh communities, and key species for carbon storage such as *A. portulacoides*, to establish. The dominant presence of pioneer communities at this site 25 years after restoration has negative consequences for saltmarsh development and carbon storage. Biological characteristics explained the majority of the observed variability in carbon stocks between MR sites, whilst sedimentary characteristics appear to be more influential for carbon stocks in natural saltmarshes, although we consider these to have been mediated by plant dynamics.

In agreement with the recently published '*Saltmarsh restoration handbook*' for the UK and Ireland (Hudson *et al.,* 2021), we propose that relative tidal height is one of the most important design considerations for MR schemes. Our findings suggest that the elevation of MR sites should surpass mean high water neap (MHWN) tidal levels for significant carbon stocks, comparable to that of natural saltmarshes, to develop. We recommend that future managed realigned schemes, particularly those that aim to maximise blue carbon stocks, should restore saltmarsh habitat above MHWN tidal levels, where possible, or select restoration sites that can be raised to MHWN elevations, whether naturally through sedimentary processes that rely on levels of high sediment supply or artificially (e.g., via dredged material). This will also provide benefits for coastal protection and biodiversity enhancement. Careful consideration of local sediment supply and elevation capital should also be taken into account to ensure the longevity of restored saltmarshes and their carbon stocks, particularly under projected sea-level rise. Finally, topographic manipulations may also provide an alternative to enhance drainage conditions and improve sediment redox potentials to create suitable conditions for the establishment of successional saltmarsh plant species. If climate change mitigation is the aim of restoration, schemes should pay closer attention to the establishment of key species (e.g. *Atriplex portulacoides* in our study area) and communities representative of the area, with measures to encourage their establishment.

Chapter 6 of 7

Public perceptions, knowledge and attitudes towards blue carbon in Great Britain

Abstract

The concept of blue carbon has captivated political and scientific dialogues in recent years, with the conservation and restoration of coastal wetlands playing a prominent role in global and national discussions on natural climate solutions. However, social research of blue carbon is lacking, despite a growing need for trans-disciplinary approaches to ensure the longevity of blue carbon management. This study contributes to calls for greater public perception research focused on coastal wetlands in temperate regions and their ecosystem services. Here we present the findings of a large-scale survey (n = 12,166), which is the first account of public perceptions, knowledge and attitudes towards blue carbon and coastal wetland habitats (saltmarshes and seagrass meadows) across Great Britain. Through this survey, we found that there is a limited public awareness of blue carbon and a poor perception of coastal wetlands as natural climate solutions, particularly when compared to terrestrial forests. Furthermore, we find that certain respondent characteristics, such as age, influence individual perceptions and awareness of blue carbon. Equipped with an improved understanding of public audiences, targeted, research driven educational and outreach campaigns can be implemented. Advances in general ocean literacy and blue carbon education can thus help gain the necessary societal support for the effective management of coastal wetlands and the societal benefits they provide.

6.1 Introduction

Blue carbon refers to the organic carbon captured and stored by marine and coastal ecosystems, notably coastal wetlands: mangroves, saltmarshes, and seagrass meadows (Nellemann *et al.,* 2009; Mcleod *et al.,* 2011). In recent years, the interest in blue carbon, and coastal wetland management, i.e. the conservation and restoration of mangrove, saltmarsh, and seagrass habitat, has expanded across research and political fields concerned with natural solutions to mitigate climate change (Hoegh-Guldberg *et al.,* 2019; Bertram & Merk, 2020; Duarte de Paula Costa & Macreadie, 2022). Indeed, in 2021 blue carbon played a key role in ocean-climate discussions at COP26 – the United Nations Framework Convention on Climate Change (UNFCCC) Conference of Parties hosted in the UK. Bryan *et al.* (2021) reported 31 side events at COP26 that focused on the ocean, and coastal and oceanic blue carbon, which reflects a surge of support to address the ocean-climate nexus observed across the whole conference (Fullam *et al.,* 2021).The dialogue around blue carbon as a nature-based climate solution extended to COP27, hosted in Egypt in 2022, with the launch of the 'High-Quality Blue Carbon Principles and Guidance' framework by a global coalition of ocean leaders including Conservation International and The Nature Conservancy (see The Nature Conservancy, 2022).

The current decade is pivotal for advances in blue carbon management and coastal wetland restoration under the United Nations Decades on Ecosystem Restoration, and of Ocean Science for Sustainable Development (hereafter, the UN Decade on Ecosystem Restoration and the UN Ocean Decade), both launched in 2021 (Waltham *et al.,* 2020; Macreadie *et al.,* 2021). Despite growing recognition of the vast societal benefits that coastal wetlands provide, these habitats are among the most endangered in the world and face increasing anthropogenic pressure, including climate change impacts (Gamble *et al.,* 2021; Hudson *et al.,* 2021). At least 44 % of seagrass loss in the UK has occurred since 1936, with potential losses over longer time periods estimated to be as high as 92 % (Green *et al.,* 2021). Such habitat loss must be prevented and, crucially, reversed if saltmarsh and seagrass habitats are to play a role in climate change mitigation. In the UK there are numerous public-facing campaigns across marine and environmental charities (e.g. Marine Conservation Society, Wildfowl & Wetlands Trust, World Wildlife Fund, and Surfers against Sewage) that relate to blue carbon and encourage the rewilding of marine environments to enhance their blue carbon function, which have been complemented by the Environment Agency's publication of the Seagrass and Saltmarsh Restoration Handbooks for the UK and Ireland (Gamble *et al.,* 2021; Hudson *et al.,* 2021). Furthermore, the Benyon Review, conducted in 2019 and 2020, recommended that the UK government include blue carbon habitats during the Highly Protected Marine Areas site selection process, to improve the climate resilience of the seas, contribute to overall ocean health, and to protect blue carbon habitats from future damage (UK Government, 2023). Despite the evident rising interest in coastal wetland restoration, conservation, and the role they play in climate change mitigation, understanding of public knowledge of seagrass and saltmarsh habitats, and the ecosystem services they provide, such as carbon sequestration, is limited both within the UK and around the world (Nordlund *et al.,* 2018; McKinley *et al.,* 2020b; Kim *et al.,* 2022; Wang *et al.,* 2022). Public understanding of the importance of blue carbon could help gain wider support for coastal wetland management and restoration initiatives.

While there has been a growing interest in the relationships and values that people hold with and towards the ocean, coasts and seas (McKinley *et al.,* 2020a; Bennett, 2019), public perceptions research in marine environments to date has been skewed towards more charismatic, or attractive, species and habitats, with calls for greater research emphasis on coastal wetlands in temperate regions (McKinley *et al.,* 2020b; Jefferson *et al.,* 2021). Public perceptions research acknowledges that a wide range of components, such as knowledge, interest, and awareness, can influence individual interactions with, and perceptions of, the marine and coastal environment and its management (Jefferson *et al.,* 2015; Bennett *et al.,* 2017; Jefferson *et al.,* 2021). Gaining an insight into societal knowledge, perceptions, and attitudes is an integral step towards the implementation of marine and coastal policy and the effective management of marine and coastal environments (Bennett *et al.,* 2017; Carpenter *et al.,* 2018). In turn, this underpins current global objectives relating to the conservation and restoration of coastal wetlands – both of which are central to the blue carbon concept (Lovelock & Duarte, 2019).

Historically underrepresented in marine and coastal policy management, it is recognised that in the absence of social sciences, natural sciences alone will be ineffective in achieving marine conservation goals (McKinley *et al.,* 2020a; Jefferson *et al.,* 2021). Indeed, aspirations of the UN Ocean Decade include a stronger integration of natural and social sciences and an improved societal relationship with the ocean, driven in part by giving prominence to the concept of ocean literacy as a mechanism for change (Claudet, 2021; McKinley *et al.,* 2023). Ocean literacy, albeit a rapidly evolving concept (see McKinley *et al.,* 2023 for more on this) can be simply defined as "an understanding of the ocean's influence on you – and your influence on the ocean" (NOAA, 2020). Furthermore, incorporating social research has been deemed key to the successful adoption and longevity of blue carbon as a natural climate solution (Macreadie *et al.,* 2022), supported by growing calls for trans-disciplinary approaches to support coastal habitat restoration under the UN Decade on Ecosystem Restoration (Waltham *et al.,* 2020). To deliver this, however, understanding public awareness of the role coastal wetlands play in climate change mitigation is required as the identification of societal benefits can increase the focus on the importance of these habitats, promoting interest in their conservation, restoration, and overall management (Nordlund *et al.,* 2018). In the past, natural climate solutions have overwhelmingly focused on terrestrial environments, notably forests, with coastal and marine opportunities widely overlooked (Nellemann *et al.,* 2009; Macreadie *et al.,* 2021). This may feed into the seemingly low public awareness of the climate change mitigation potential of blue carbon habitats compared to terrestrial habitats, as observed by a US study in which public support for afforestation and reforestation was greater than all other climate solution strategies, including soil carbon storage which encompasses coastal wetlands (Sweet *et al.,* 2021).

In order to generate public interest and support, current levels of public awareness and knowledge of blue carbon need to be understood from a diversity of perspectives. Very few studies have explored public perceptions towards blue carbon, with those that have reporting a low awareness and knowledge among respondents (Kim *et al.,* 2022; Wang *et al.,* 2022).

For example, the role of saltmarsh and seagrass habitat as a carbon sink was the least known ecosystem service reported by an on-site visitor survey at a Ramsar wetland in Tasmania, Australia (Wang *et al.,* 2022). Similarly, a study into public willingness to pay for saltmarsh and seagrass habitat restoration in South Korea reported a minimal awareness of blue carbon among respondents (Kim *et al.,* 2022), while studies from the Philippines and Wales, UK, indicate public recognition of blue carbon as a societal benefit to be low in comparison to other ecosystem services delivered by coastal wetlands (McKinley *et al.,* 2020b; Quevedo *et al.,* 2020b). Awareness of ecosystem services was found to be generally higher when benefits are visible or when local communities have direct experience of them e.g. fishing (McKinley *et al.,* 2020b; Quevedo *et al.,* 2020b). Despite the trend of low levels of societal awareness, some studies suggest the public may be a receptive audience for outreach and engagement surrounding blue carbon, finding considerable public interest in learning more about the topic (e.g. Kim *et al.,* 2022; Wang *et al.,* 2022). For example, research on ocean-based carbon dioxide removal suggests that public acceptability of blue carbon management, i.e. habitat conservation and restoration, tends to be much higher than alternatives, e.g., ocean iron fertilisation (Bertram & Merk, 2020).

In the UK, studies over the last 10 years have indicated there to be a disconnect between the public and many aspects of the marine environment (Jefferson *et al.,* 2014; Defra, 2021). Despite the presence of saltmarshes and seagrass meadows along estuaries and coastlines across the UK this trend is reflected in the public's low connection with saltmarshes and seagrass meadows, with a recent study finding the Welsh public to have limited understanding of saltmarshes (McKinley *et al.,* 2020b). This echoes earlier research which found seagrass to be considered the least interesting marine species in a UK-based study in favour of more charismatic species such as seals, puffins and seahorses (Jefferson *et al.,* 2014). The perceived lack of charisma and attractiveness of saltmarsh and seagrass habitats compared to coral reefs, for example (Duarte *et al.,* 2008), places them at a disadvantage for building public awareness and recognition of their societal benefits (Nordlund *et al.,* 2018). Public indifference and unfamiliarity, which correlate with public levels of concern (Gelcich *et al.,* 2014) are considered a significant threat towards both seagrass (Nordlund *et al.,* 2018; Elggren, 2019) and saltmarsh habitat (McKinley *et al.,* 2020b) in the UK and around the world.

Using Great Britain as a case study, this paper presents the first assessment of public perceptions, awareness, and knowledge of blue carbon and related ecosystems. This is of particular significance due to the growing political dialogue around blue carbon, and the prominence of natural climate solutions in global and national discussions (e.g. UNFCCC). In addition, there has been a recent growth in public facing environmental campaigns focused on coastal habitat restoration and climate change mitigation (e.g. the Surfers Against Sewage Ocean & Climate petition ahead of COP26). Yet, little is known about how the concept of blue carbon is perceived and valued by the public, nor how this influences coastal wetland management. The objectives of this study are to: (1) examine public awareness of terms relating to blue carbon and nature-based solutions; (2) explore public perceptions of saltmarsh and seagrass habitat across Great Britain, and the ecosystem services they provide, the latter of which is an attempt to see how blue carbon is valued among other societal benefits, and; (3) understand how the public value coastal wetlands for carbon capture and storage against more well-known natural climate solutions - terrestrial forests. In light of existing research summarised above, we hypothesise that public perceptions and awareness of blue carbon will be minimal, with respondents' reporting a higher knowledge of other societal benefits provided by saltmarsh and seagrass habitat, and a greater awareness of the role of terrestrial forests in carbon capture and storage. The relationship between public perceptions and sociodemographic characteristics will also be explored. Findings from this study will contribute to the advancement of the social dimensions of blue carbon research and will also be used to inform policy recommendations that ultimately benefit blue carbon management. In addition, we intend for this study to serve as a baseline assessment of public perceptions of blue carbon habitats and the wider concept of blue carbon, against which future surveys can be compared as blue carbon continues to gain traction across multiple stakeholder groups who have an interest in the marine environment and climate change mitigation, including the public.

6.2 Methods

The purpose of this study is to develop the first assessment of public perceptions, awareness, knowledge and attitudes towards blue carbon in Great Britain. In order to achieve this, we designed a series of relevant questions which were commissioned by the Department of Food, Environment, and Rural Affairs (Defra) and incorporated into their online Ocean Literacy survey (Defra, 2022) as part of the project 'Understanding Ocean Literacy and ocean climate-related behaviour change in the UK' in collaboration with the Ocean Conservation Trust, Natural Resources Wales and Marine Scotland (see McKinley & Burdon, 2020; Defra, 2022; McKinley *et al.,* 2003 for more information on this research and ocean literacy in general). The survey builds upon research undertaken in 2021 when the first Ocean Literacy survey was commissioned, but the 2022 survey included an additional set of questions focused on blue carbon in order to fill evidence gaps in UK public perceptions and awareness recognised by policy in Defra.

Due to the nature of the large-scale study covering Great Britain, an online survey was considered the most appropriate method for data collection. A range of question types were used including closed (e.g. 'Are you aware of any restoration efforts for saltmarshes in the UK?') and Likert scale based questions (e.g. 'Please indicate your familiarity with the term blue carbon'), as well as open questions that provided qualitative data on how respondents' perceive seagrass meadows, saltmarshes, and blue carbon. The survey comprised four relevant sections: (1) respondent familiarity with terms linked to blue carbon and nature-based solutions; (2) questions related to the importance of various ecosystem services, or benefits, as an attempt to see how blue carbon is valued among other societal benefits for coastal wetlands; (3) questions on comparing the effectiveness of carbon capture and storage between blue carbon habitats and terrestrial habitats, the latter of which are more well-known for climate change mitigation; and (4) a focus on perceptions and awareness towards seagrass meadows and saltmarshes – their aesthetic appeal, benefits and management – including awareness of restoration efforts across the UK. A final section involved questions on respondents' sociodemographic and background characteristics. Survey questions relevant to this study can be seen in Appendix D7.

The data collection process was administered via a survey fieldwork agency (BMG research) between March and April 2022. To ensure maximum accessibility, the survey could be completed on tablets and mobile devices in addition to desktop and laptop PCs. Prior to the distribution of the final survey, a small pilot study of 152 respondents was conducted to ensure data was captured correctly and the script was working as intended (Defra, 2022b). Minor amendments were made as a result of the pilot and are detailed in the technical report (Defra, 2022b). Through the use of respondent panels, a total sample of 12,166 adult respondents across a range of sociodemographic backgrounds from England, Wales, and Scotland was achieved (Table 6.1). The survey was translated into Welsh and respondents in Wales were provided with the option to complete the survey in English or Welsh. Respondents, who had to have been a permanent resident in England, Wales or Scotland for at least the last 5 years to participate, were selected using BMG's online panel blend approach to ensure the final sample of respondents was reflective of the wider population across Great Britain (Defra, 2022b). Quotas set on age (based on the most recent Office for National Statistics estimates) and whether respondent locations were classified as coastal or non-coastal ensured completed surveys were representative of the population. These quotas were closely monitored and under-represented groups were sent further invitations and reminders to complete the survey in order to achieve representation. Other sociodemographic variables including gender, ethnicity, index of multiple deprivation quantiles, and urban/ rural classifications were also monitored during data collection to ensure a spread of responses were received (Defra, 2022b).

6.2.1 Respondent profile

Along with standard sociodemographic questions including country of residence, age, gender, and employment status (Table 6.1), respondents were asked if they were a member of an environmental organisation, such as the Wildlife Trusts or RSPB, and whether they had a connection with marine industries either through personal employment, or that of a family member. Different categories of marine industry (e.g. Oil & gas, Recreation, Marine conservation etc.) were combined into one variable with a "Yes/ No" response. Due to the low percentage of respondents who selected part-time student (0.38 %) under employment status, one 'Student' category was made combining both part-time and full-time students (Table 6.1). Using respondents' postcodes, the distance in kilometres each respondent lived from the coast was calculated (Defra, 2022b). To provide additional insight into public perceptions and awareness of blue carbon and coastal wetlands, respondents were also asked how often they visited a range of marine environments, including seagrass meadows and saltmarshes, and how important they felt the marine environment was to them personally. Analysis was carried out to examine any potential impact of sociodemographic characteristics on respondents' perceptions and awareness.

6.2.2 Data analysis

Statistical analysis was conducted using R software (version 4.2.1). Prior to analysis, all variables were checked for outliers. To find out which combination of sociodemographic characteristics best explained responses to "Yes/ No" questions (e.g. 'Is carbon capture and storage a benefit provided by saltmarshes?' and 'Are you aware of any restoration efforts for seagrass meadows in the UK?'), a generalised linear models (GLM) with Binomial error distribution and cloglog (conditional log-log) link function were used ("MASS" package; Venables & Ripley, 2002). A binomial GLM is suitable for binary data (Zuur *et al.,* 2009), and the cloglog link function allows for asymmetry in the dataset e.g. when there are not equal numbers of "Yes" and "No" responses (Thomas *et al.,* 2017). Responses marked "Don't know" were reported but filtered out prior to data analysis. The stepAIC function ("MASS" package; Venables & Ripley, 2002), based on Akaike information criterion (AIC), was used in model selection to produce an optimal model which contained only the predictor variables that significantly influenced responses. Analysis of variance was used to test the probability of decreased deviance between the optimal and full model. Akaike Information Criterion (AIC) values were used to confirm model selection whereby models with the lowest AIC values were selected. Binomial GLMs were validated using diagnostic plots. Binned residual plots are a recommend diagnostic check for binomial models due to easy interpretation and sensitivity to model failures (Gelman *et al.,* 2000). Models were deemed a good fit for the data when binned residual plots showed 95 % of residuals fell within predictive error bounds and there was no obvious pattern (Gelman & Hill, 2006), and when residuals were not present beyond Cook's distance contour lines in a leverage plot (Thomas *et al.,* 2017). Model residuals were also tested for serial autocorrelation to determine whether a mixed effects model with a random component was necessary, but Auto-Correlation Function plots showed this was not necessary.

Ordinal regressions using Cumulative Link Models (clm) ("ordinal" package; Christensen, 2019) were used to analyse the influence of sociodemographic characteristics on response variables with multiple values coded as rank-ordered categories. This included respondent familiarity with blue carbon ('Please indicate how familiar you are with the term blue carbon') which had the following ordered levels: Have never heard of the term < Heard of but do not understand < Heard of and have some understanding < Know and understand. Ordinal regression is the most appropriate method of analysis for response data on an ordinal scale which exhibits a natural ordering. Model selection was confirmed using AIC values and Log-likelihood ratio tests. The assumption of proportional odds was tested using the Brant test ("Brant" package; Schlegel & Steenbergen, 2020). However, the proportional odds assumption is often violated and such tests are strongly affected by large sample sizes (O'Connell & Liu, 2011), as is this case in our study (n = 12,166). Thus despite violations of the proportional odds assumption, ordinal regression was still considered the most appropriate model choice. In addition, the Hessian number of ordinal regressions is a measure of how identifiable the model is and large values (e.g. > 10,000) indicate a poorly defined model. All ordinal regressions displayed a Hessian value < 10,000.

Where multiple predictors were retained in the optimal model for binomial GLMs and ordinal regressions, hierarchical partitioning using the "hier.part" package (Mac Nally & Walsh, 2004) was used to identify independent effects (%) that each predictor contributed to the optimal model. Predictor variables that had < 5 % independent effects were not discussed in model results. All figures were created using the "ggplot2" package (Wickham, 2016).

Qualitative analysis was used to explore respondents' comments to the question "what image, word, or phrase comes to mind when you think of the term 'blue carbon'?". Of the 3,727 respondent comments, 78 % (n = 2,892) that expressed a measurable response were analysed, with the remaining comments, in which respondents indicated they were unfamiliar with the term or did not know what it meant, excluded. Responses were grouped into broad categories that incorporate other similar responses.

6.3 Results

Sociodemographic and background characteristics are reported across the total sample of 12,166 respondents. Sociodemographic variables were then analysed against survey responses to determine the key drivers that influence public perceptions, awareness, and attitudes towards blue carbon and coastal wetlands (saltmarsh and seagrass habitat). For simplicity, 'coastal wetlands' will exclusively refer to saltmarsh and seagrass habitats from hereafter.

6.3.1 Respondent profile

A summary profile for the total sample of respondents (n = 12,166) across Great Britain can be seen in Table 6.1. The majority of respondents came from urban locations (81 %) as opposed to rural locations (19 %). For an insight into deprivation, an Index of Multiple Deprivation was measured based on the geographic area respondents reside, using the following domains: income, employment, education, skills and training, health and disability, crime, barriers to housing, services, and living environment. Respondent deprivation was evenly spread (between 9 to 11 %) across all deciles ranging from 1 (least deprived) to 10 (most deprived). Annual household income (before tax) was relatively evenly spread across 10 brackets, with the greatest proportion of respondents (between 10 to 18 %) in the lower five bracket ranges (£0 to £50,000), compared to between 1 and 7 % of respondents in the higher five bracket ranges (£50,001 to £150,001+).

Connections with marine industry were low among respondents – 93 % were not employed in marine industries, nor had a family member who was. From the respondents that were employed in a marine industry, or had a family member who was (6 %), the majority were employed in oil and gas followed by the Royal Navy or Royal Marines, with less than 10 % in other marine industries including fisheries and aquaculture, marine conservation or research, and marine policy or management; the remaining 1 % of respondents were unsure. Few respondents (6 %) were subscribed to an environmental organisation such as the RSPB or National Trust.

Table 6.1. Socio-demographic characteristics showing the number (N) and percentage (%) of the total sample of respondents (n = 12,166) who fall under each category. Where percentages do not total 100 %, the remainder is comprised of respondents who either preferred not to answer or selected "Don't know".

Demographics	**N**	**%**	**Demographics**	**N**	**%**
Country			**Gender**		
England	7060	58	Female	7125	58
Wales	2051	17	Male	4988	41
Scotland	3055	25			
Age			**Ethnicity**		
16 to 24	1793	15	Asian/ Asian British	476	4
25 to 34	2056	17	Black / Black British/ Caribbean/ African	173	1
35 to 44	1958	16	Mixed or multiple ethnic group	231	2
45 to 54	1957	16	White	10759	88
55 to 64	1969	16	Other ethnic group	451	4
65 to 74	1842	15			
75+	591	5			
Employment status			**Highest qualification**		
Employed full-time	4330	36	PhD/ Doctor	219	2
Employed part-time	1712	14	Postgraduate Masters	1002	8
Self-employed	665	5	Undergraduate Bachelors or equivalent	3338	27
Unemployed	449	4	Higher education	1498	12
Retired	2475	20	A-level or equivalent	2766	23
Not working – caring responsibilities	619	5	GCSE or equivalent	2570	21
Not working – long-term sick/ disabled	597	5	No qualifications	530	4
Student (incl. part-time)	1244	10	Other qualifications	87	1

Approximately 10 % of respondents had never visited the marine environment before, whilst just over half had visited in the 12 months prior to data collection (53 %). The remaining had not visited in the past year (35 %), and 2 % did not know. A following question for respondents who had visited the marine environment in the past 12 months (n = 6519) asked how often, on average, they had spent in a range of marine environments (Appendix D1), from more than once per day to never, including weekly and monthly visits, and an option for not in the last 12 months. Seagrass meadows and saltmarshes were the least visited of all marine environments, with the majority of respondents having never visited them before (54 % and 38 % respectively; Appendix D1). The next least visited marine environment was mudflats, a habitat associated with coastal wetlands (Appendix D1). A greater proportion of respondents answered “Don’t know” to having visited seagrass meadows and saltmarshes (4 % and 3 % respectively) in comparison to other marine environments (Appendix D1), suggesting a lower familiarity with coastal wetlands among the public. Most respondents perceived the marine environment to hold some importance to them personally (very important: 38 %, and important: 46 %), whilst 11 % were indifferent. The remaining 5 % of respondents felt that protecting the marine environment was either not very important, not important at all, or selected “Don’t know”.

Sociodemographic characteristics (country, gender, age, ethnicity, employment status, highest qualification, urban/ rural location, distance from the coast, deprivation decile, household income, connection with marine industry, and environmental organisation membership), perceptions of the importance of the marine environment, and time spent in seagrass and saltmarsh habitat in the last 12 months were analysed against public perceptions, knowledge and awareness towards blue carbon and coastal wetlands in Great Britain. Only the variables shown to have a significant influence on public perceptions, knowledge, and attitudes are discussed further.

6.3.2 Public awareness and perceptions towards blue carbon

6.3.2.1 Familiarity with blue carbon

Respondents were initially asked to indicate their level of familiarity with certain terms linked to blue carbon (Fig. 6.1). Only 4 % of respondents reported to know and understand the term blue carbon, whilst 60 % of the remaining had never heard of the term (Fig. 6.1). Respondents were more familiar with the broader terms of ecosystem services and nature-based solutions and to a lesser extent, carbon sequestration (Fig. 6.1). Familiarity with the term climate change was much higher - with 92 % of respondents indicating that they either know and understand, or have some understanding, of the term.

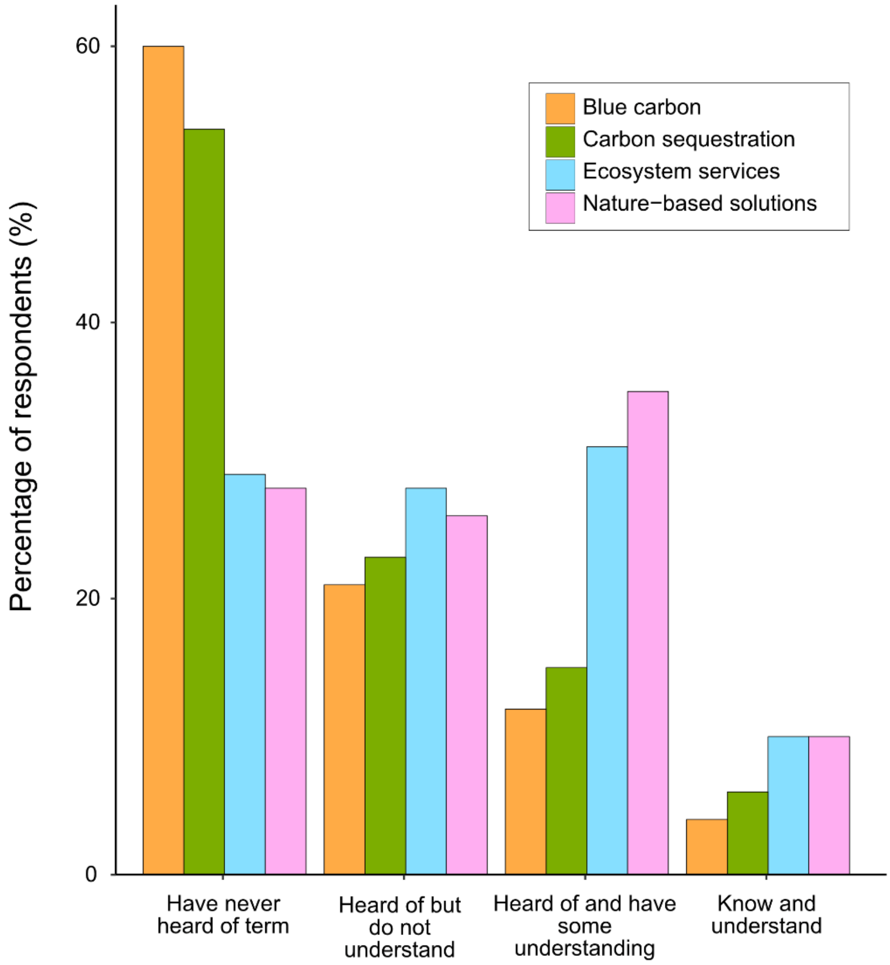


Figure 6.1. Respondents' self-reported level of familiarity with the following terms: Blue carbon (orange), Carbon sequestration (green), Ecosystem services (blue), and Nature-based solutions (pink).

Ordinal regression showed respondents who identified as female, who were less likely to indicate the marine environment was very important to them and those in the two least deprived deciles were significantly less likely to report a level of understanding for the term blue carbon (Appendix D3). In contrast, being under the age of 35 and having a connection with marine industry significantly increased the likelihood of being familiar with the term and acknowledging a higher level of understanding (Appendix D3); age and connection to marine industry were the most influential variables identified by hierarchical partitioning (Appendix D6). Respondents in higher income brackets (£60,000+), in addition to those with a household income between £15,001 and £30,000, significantly increased the odds of reporting a higher level of familiarity with the term (Appendix D3). Being a member of an environmental organisation or charity marginally increased the odds of reporting a higher level of understanding of the term blue carbon ($p = 0.04$), however, the variable showed a minimal contribution to the optimal model with an independent effect < 1 %. Respondents' ethnicity, educational attainment, distance from the coast, place of residence, and employment status did not influence their familiarity with the term.

Analysis found respondents' attitudes towards climate change and nature (Appendix D2) significantly influenced their familiarity with the term 'blue carbon'. For example, respondents who considered climate change to be the greatest threat to the natural world (identified by levels of disagreement to the first statement in Appendix D2), who believed climate action was urgently needed, and who strongly agreed that nature can help to reduce climate change impacts were significantly more likely to acknowledge a higher level of understanding with the term blue carbon. Main effect results from the ordinal regression which included respondents' attitudes towards climate change and nature as predictors showed each variable to be highly significant ($p < 0.001$).

When asked what image, phrase or word comes to mind when you think of the term 'blue carbon', responses were highly variable (see Appendix D4 for percentages and examples within each category). Three categories emerged as the most common themes among respondents ($n = 2,892$): chemistry and atmospheric conditions (11.8 %), worry and concern (9.1 %), and pollution (8.7 %) (Appendix D4). Although ocean and coastal carbon storage was the next most common theme (7.8 %) including responses such as "carbon locked in the sea" and "how the sea collects carbon from the atmosphere", only 0.4 % of responses in this category referenced coastal wetlands (mangroves, saltmarshes or seagrasses) specifically (Appendix D4). When responses mentioned marine habitats, a lower percentage of respondents referred to coastal wetlands (1.2 %) than other marine habitats (6.1 %), which included coral reefs, kelp forests, algae and seaweed (Appendix D4). Some respondents felt hopeful and/ or tranquil (6.9 %) when they thought of the term 'blue carbon', e.g., "hint of hope" and "calming", but this was outweighed by the percentage of respondents who associated the term with worry and/ or concern (9.1 %), e.g., "doomed" and "dangerous" (Appendix D4).

6.3.2.2 Carbon capture and storage as an ecosystem service

When asked whether a range of benefits, or ecosystem services, were provided by seagrass and saltmarsh habitats, 26 % of respondents answered "Don't know". After these responses were filtered out, the proportion of "Yes" and "No" answers was used to create a rank order of benefits, which was the same across both habitats, although total percentages differed slightly

(Fig. 6.2). In rank order, the creation of diverse habitats for wildlife; coastal protection; and pollution control/ water purification were perceived by the public to be the top 3 most important benefits provided by seagrass meadows and saltmarshes (Fig. 6.2). However, across all seven benefits, wildlife habitat and coastal protection were the only benefits to have more positive responses than negative – with between 38 % and 52 % positive responses, depending on the benefit and habitat (Fig. 6.2). Food (e.g. fisheries), mental health and wellbeing support, and recreation were perceived to be the least important benefits – with only 8 – 13 % of respondents answering positively across both habitats (Fig. 6.2). Following the top three, carbon capture and storage was considered the next most important benefit provided by seagrass meadows and saltmarshes with 32 % and 28 % positive responses, respectively. Nevertheless, the majority of respondents did not perceive this to be a benefit provided by these habitats and responded negatively (43 % and 46 % respectively; Fig. 6.2). A slightly lower percentage of respondents thus considered carbon capture and storage to be a benefit of saltmarshes when compared to seagrass meadows.

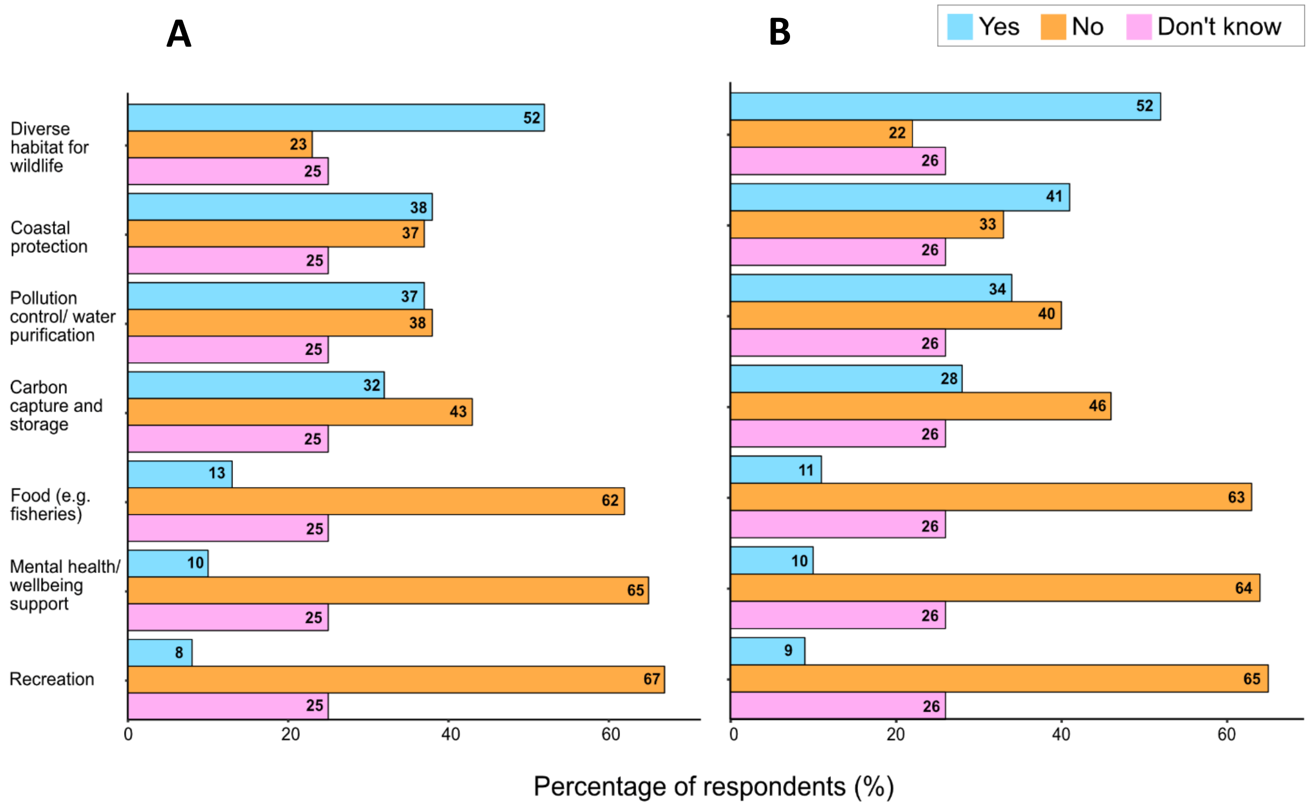


Figure 6.2. Public perceptions towards the ecosystem services, or benefits, provided by seagrass meadows (A) and saltmarshes (B), as indicated by the percentage of responses for "Yes" (blue) and "No" (orange). Note the high number of respondents who answered "Don't know" (pink).

A comparison of responses revealed that respondent gender, educational attainment, age, employment status and membership with an environmental organisation were sociodemographic characteristics that significantly influenced whether respondents perceived carbon capture and storage as a benefit of seagrass meadows and saltmarshes (Fig. 6.2), alongside respondents' perceived importance of the marine environment. Significantly more respondents who identified as male positively perceived carbon capture and storage as a benefit of seagrass meadows ($p < 0.001$; Fig. 6.3b) and saltmarshes ($p < 0.001$) than those who identified as female. Across all employment statuses (Table 6.1), students were significantly more likely to perceive carbon capture and storage as a benefit of saltmarshes ($p < 0.001$; Fig. 6.3d), whilst respondents who had no qualifications had significantly higher negative responses ($p = 0.007$) compared to those who had some form of qualification. Similarly, respondents with no qualifications ($p = 0.003$) and those with GCSE's or equivalent ($p = 0.01$) as their highest qualification were significantly more likely to respond "No" when asked if carbon capture and storage was a benefit provided by seagrass meadows. Respondent age was marginally significant as those in older demographic age groups were more likely to positively perceive carbon capture and storage as a benefit – with a higher proportion of respondents answering "Yes" for seagrass meadows when aged between 65 and 74 ($p = 0.04$) and for saltmarshes when aged 75 or older ($p = 0.02$).

Respondents who were not a member of, or subscribed to, an environmental organisation were significantly more likely to negatively perceive carbon capture and storage as a benefit of seagrass meadows ($p = 0.001$) than those who were a member; however, membership did not significantly influence responses for saltmarshes. Finally, when asked how important the marine environment was to them personally, respondents who were less likely to indicate the marine environment was very important to them were significantly more likely to respond negatively for seagrass meadows ($p < 0.001$; Fig. 6.3a) and saltmarshes ($p < 0.001$; Fig. 6.3c), although for respondents who reported the marine environment was not very important at all to them, this was only a marginal significant relationship, potentially due to the low percentage of respondents who held this perception of the marine environment (0.7 %).

Hierarchical partitioning identified respondents' perceived importance of the marine environment as the variable with the greatest influence over perceptions towards carbon capture and storage as a benefit of both habitats (Fig. 6.3a,c), followed by gender for seagrass meadows (Fig. 6.3b) and employment status for saltmarshes (Fig. 6.3d). Time spent in seagrass meadows and saltmarshes in the last 12 months did not influence whether responses were positive or negative; however, it did influence the percentage of respondents who did not know if carbon capture and storage was a benefit of both habitats. Respondents who had never visited seagrass meadows or saltmarshes contributed the most to “Don’t know” responses (68 % and 59% respectively) followed by those who had not visited in the last 12 months (23 % and 27 % respectively).

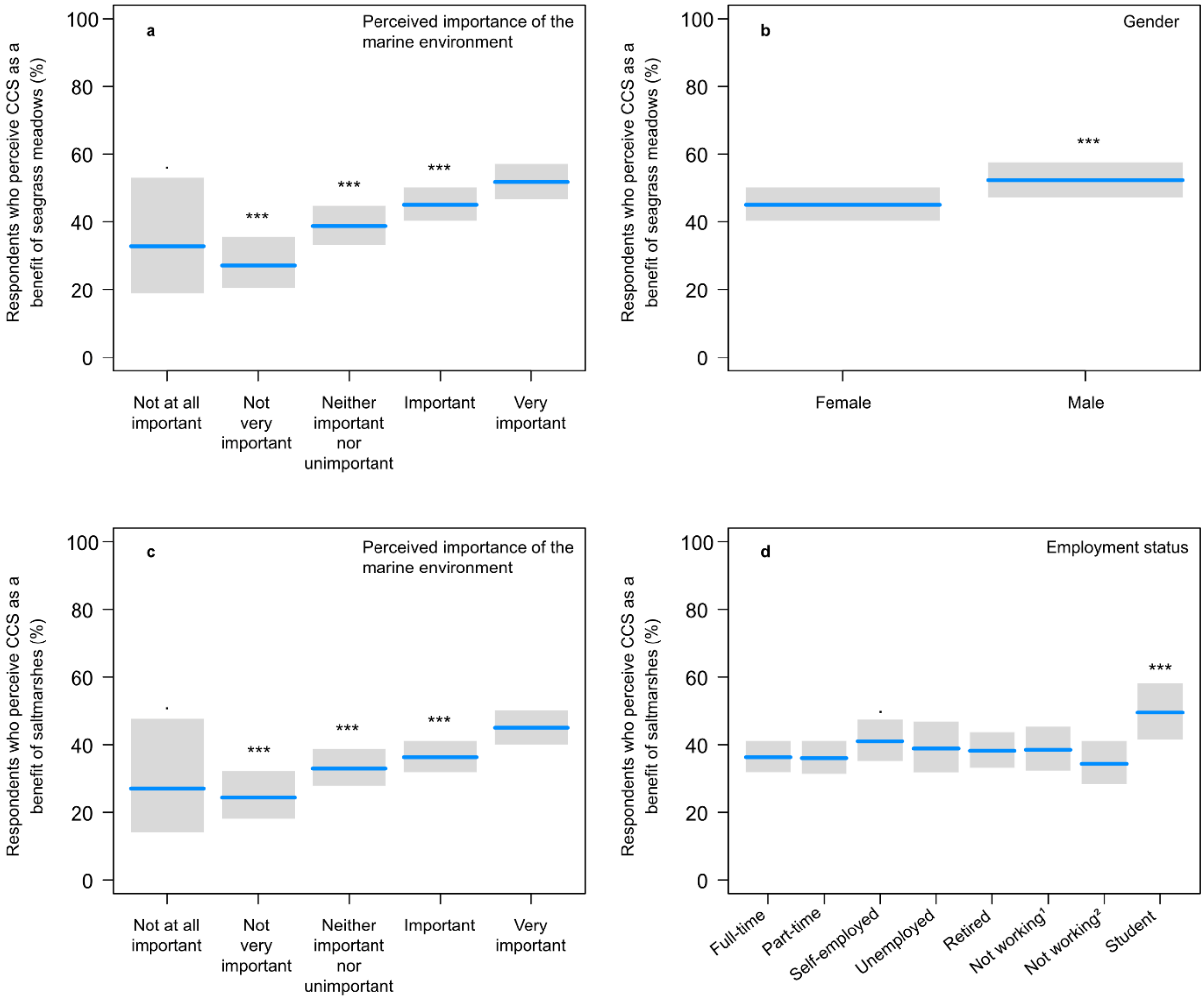


Figure 6.3. Percentage of respondents who positively perceived carbon capture and storage (CCS) as a benefit provided by seagrass meadows (a, b) and saltmarshes (c, d). Results from binomial GLMs are shown whereby blue lines represent model fitted values, and the grey bar shows the 95 % confidence intervals. The two most influential explanatory variables for the seagrass model were perceived importance of the marine environment (a; top left) and gender (b; top right), and for the saltmarsh model were perceived importance of the marine environment (c; bottom left) and employment status (d; bottom right; where “Not working[1]” represents respondents who are either looking after the house and/ or children or who have other caring responsibilities and “Not working[2]” represents respondents who are long-term sick or disabled). Most influential variables were obtained via hierarchical partitioning. Significance levels: 0 ‘***’ 0.05 ‘.’ .

To compare public perceptions of the importance of seagrass meadows and saltmarshes for climate change mitigation with other more traditionally recognised natural climate solutions (e.g., terrestrial forests), respondents were presented with a description and photograph of a range of natural habitats in both terrestrial and marine environments (Appendix D7: question NQ6) and asked two questions: which habitats are important for carbon capture and storage, and which can be found in the UK. Only 3 % of respondents selected tropical rainforest as being found in the UK; however, 62 % selected this habitat as the most important for carbon capture and storage (Fig. 6.4). Temperate forests, closely followed by seagrass meadows, were considered the next most important habitats (42 % and 40 % respectively), but in general, blue carbon habitats were not considered to be as important as terrestrial forests for carbon capture and storage (Fig. 6.4). Seagrass meadows were perceived to be more important for carbon capture and storage than saltmarshes and mangroves, yet fewer respondents thought seagrass meadows could be found in the UK than saltmarshes (Fig. 6.4). Saltmarshes had the greatest number of positive responses in relation to habitats found in the UK (75 %), other than grasslands (Fig. 6.4).

With the exception of mangrove and boreal forest, perceptions of whether a habitat was important for carbon capture and storage appeared to decrease as a greater proportion of respondents believed the habitat could be found in the UK (Fig. 6.4). Just over 1 % of respondents believed none of the habitats could be found in the UK. The percentage of respondents who selected “Don’t know” when asked which habitats were important for carbon capture and storage and which could be found in the UK (12 % and 7 %, respectively) was lower compared to “Don’t know” responses for questions related to the benefits provided by seagrass meadows and saltmarshes (Fig. 6.2). In addition, the percentage of positive responses amongst respondents was notably higher when asked ‘Are seagrass meadows important for carbon capture and storage?’ (40 %) compared to when respondents were asked whether carbon capture and storage was a benefit provided by seagrass meadows amongst a range of other benefits (32 %; Fig. 6.2). The number of positive responses for saltmarshes were similar across both questions (26 % and 28 %, respectively). It is important to note that in response to the question of which habitats are important for carbon capture and storage, positive responses (ranging between 12 % to 42 %; Fig. 6.4) were relatively low in comparison to negative responses (ranging between 46 % to 76 %) across all habitat types excluding tropical rainforests.

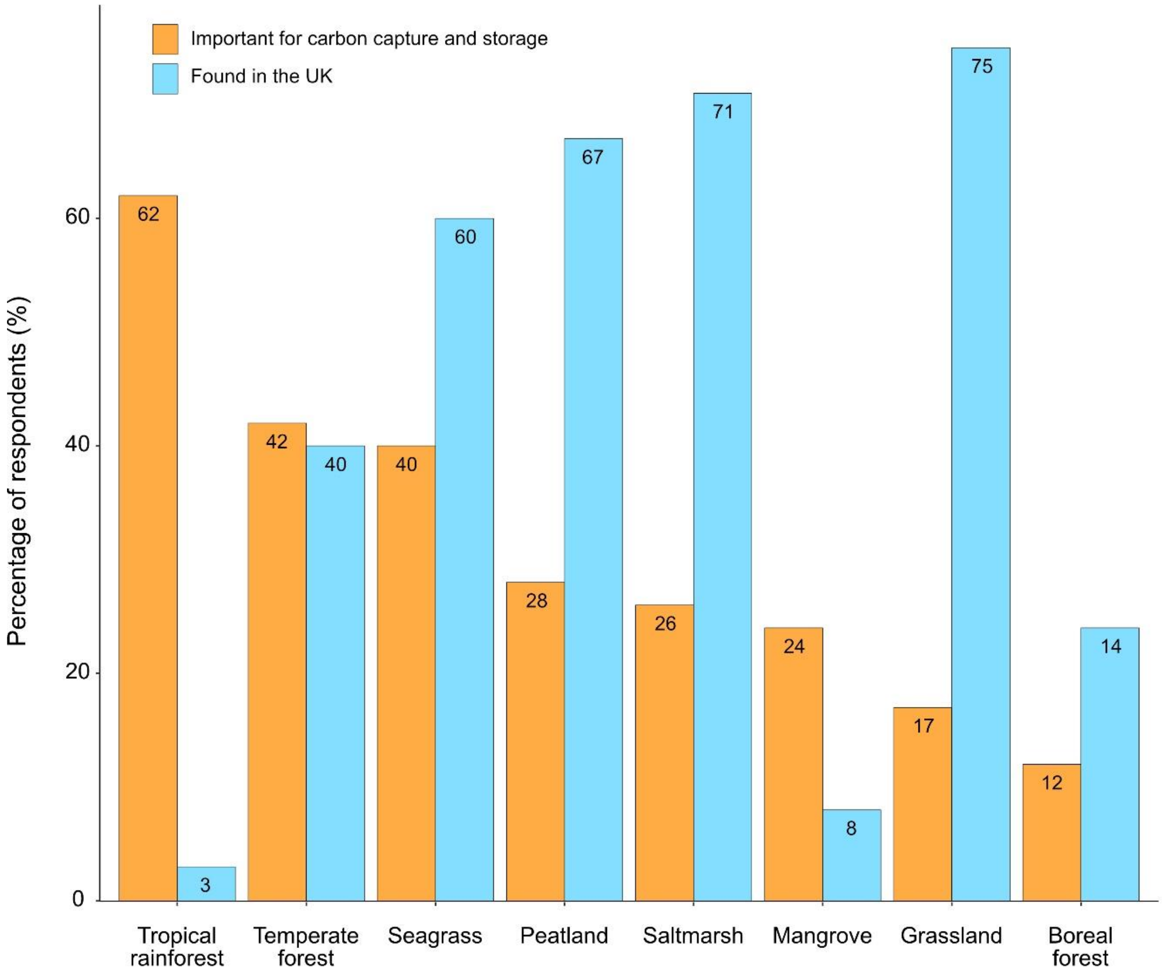


Figure 6.4. Public perceptions on whether habitats are important for carbon capture and storage (orange) and can be found in the UK (blue), as indicated by the percentage of "Yes" responses. Habitats are displayed in decreasing order of perceived importance for carbon capture and storage.

6.3.3 Public awareness of coastal wetland restoration

Public awareness of coastal wetland restoration efforts across the UK was low, but a higher proportion of participants were aware of restoration efforts for saltmarshes (17 %) than seagrass meadows (14 %). For both habitats, 7 % of respondents selected "Don't know". When respondents indicated awareness of restoration efforts, they were asked to provide examples of where they had seen efforts to restore. The majority of responses mentioned media sources such on the television (e.g. Countryfile) and radio, in documentaries, articles and the news, or via social media. The remaining responses mentioned specific environmental organisation such as Yorkshire Wildlife Trust and the Wildlife Trusts in general, Project

Seagrass, the WWF and the RSPB; or specific locations across the UK including Studland Bay (Dorset), Spurn point (Yorkshire), and the Essex, Welsh and Cornish coastlines. Some respondents were aware of restoration efforts but could not recall where from.

The odds of being aware of restoration efforts for seagrass meadows and saltmarshes in the UK were significantly increased if the respondent identified as male, was a member of an environmental organisation/ charity, and had some connection with marine industry whether through personal employment, or that of a family member (Appendix D5). In addition, respondents over the age of 55 were more likely to be aware of saltmarsh restoration efforts, but respondent age group did not influence awareness of seagrass restoration (Appendix D5). Respondents who were less likely to indicate the marine environment was very important to them, and whose highest qualification was below Postgraduate Masters level or equivalent, significantly decreased awareness of restoration efforts for both habitats (Appendix D5), with the exception of those who marked "Other qualifications", perhaps due to the small sample size they represent (1 %; Table 6.1). Furthermore, the country in which respondents reside significantly influenced awareness, although this was more influential for the saltmarsh model (Fig. 6.5b). Respondents in Wales and Scotland had lower levels of awareness of saltmarsh restoration efforts compared to those in England, whilst respondents in Scotland had lower levels of awareness of seagrass restoration efforts than those in England and Wales (Appendix D5). Out of all the sociodemographic variables that significantly influenced awareness of restoration efforts, hierarchical partitioning showed connection with marine industry, importance of the marine environment, and gender to be the most influential for seagrass restoration (Fig. 6.5a), whilst gender and importance of the marine environment were the most influential for awareness of saltmarsh restoration, followed relatively closely by connection with marine industry and whether respondents were a member of an environmental charity (Fig. 6.5b).

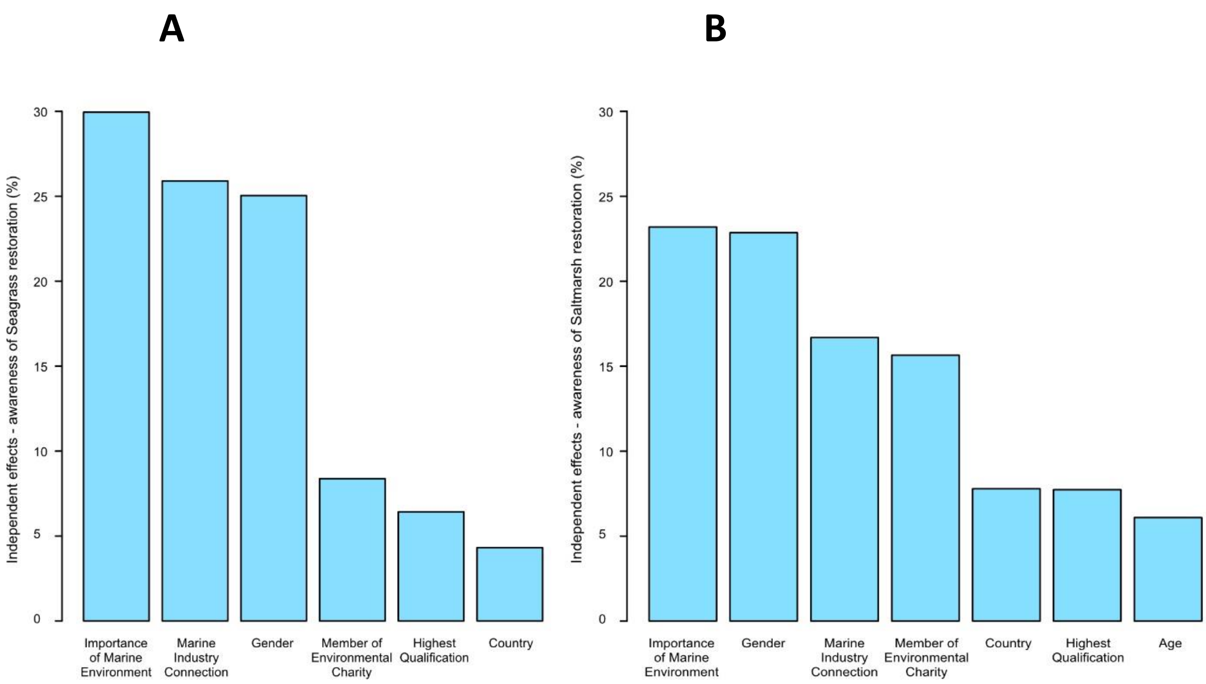


Figure 6.5. Independent effects (%) each sociodemographic variable contributed to explaining the optimal model. The optimal model only retained variables that significantly influenced awareness of seagrass (A) and saltmarsh (B) restoration. Independent effects summed across variables for each model will equal 100 %. Results are from hierarchical partitioning ("hier.part" package; Mac Nally & Walsh, 2004).

Time spent in seagrass and saltmarsh habitat (Appendix D1) was analysed against awareness of restoration efforts separately due to the reduced sample size – data were only gathered from respondents who had visited the marine environment in the last 12 months (53 %). Respondents who had spent less time in seagrass and saltmarsh habitat were significantly less likely to be aware of restoration efforts ($p < 0.001$ and $p < 0.001$, respectively).

Participants perceived the provision of habitat for supporting diverse wildlife, coastal protection and pollution control as the main reasons for restoring seagrass and saltmarsh habitat, although positive responses for pollution control were higher than coastal protection for seagrass meadows. Similarly to perceptions towards the most important benefits provided by these habitats, carbon capture and storage was deemed the next most important reason for restoration, with food (e.g. fisheries), mental health and well-being support, and recreation as the three least important reasons for restoration.

6.3.4 Public perceptions of seagrass meadows and saltmarshes

To determine how the public perceive the benefits and aesthetic appeal of seagrass meadows and saltmarshes and the extent to which the public are aware of their management, respondents were asked to indicate their level of agreement with a mixture of positive and negative statements (Fig. 6.6). In general, public perceptions towards seagrass and saltmarsh habitats were relatively positive, with an awareness that healthy habitats are more beneficial to people. Across both habitats, the statements with the highest levels of agreement were “Healthy seagrass meadows/ saltmarshes will provide more benefits to people than damaged seagrass meadows/ saltmarshes” closely followed by “Seagrass meadows/ Saltmarshes are undervalued and underappreciated habitats” and “Saltmarshes protect coastlines from sea-level rise and storms” (Fig. 6.6). In contrast, “Seagrass meadows/ Saltmarshes are of little benefit to people” and “Seagrass meadows/ Saltmarshes play no real role in carbon capture and storage” had the lowest levels of agreement; however, a greater number of respondents were either indifferent, did not know, or indicated a level of agreement when compared to the number of respondents who reported a level of disagreement with both statements (Fig. 6.6). A similar, but low, proportion of respondents felt seagrass meadows and saltmarshes were the least interesting marine habitat, indicated by 14 % and 17 % of respondents either agreeing or strongly agreeing with the statement, respectively (Fig. 6.6).

The greatest number of responses marked “Don’t know” or “Neither agree nor disagree” related to statements requiring knowledge on the management and protection of both habitats (Fig. 6.6). Respondent indifference was high across all statements with an average of 33 % “Neither agree nor disagree” responses per statement (Fig. 6.6), further highlighting a limited public connection with coastal wetlands.

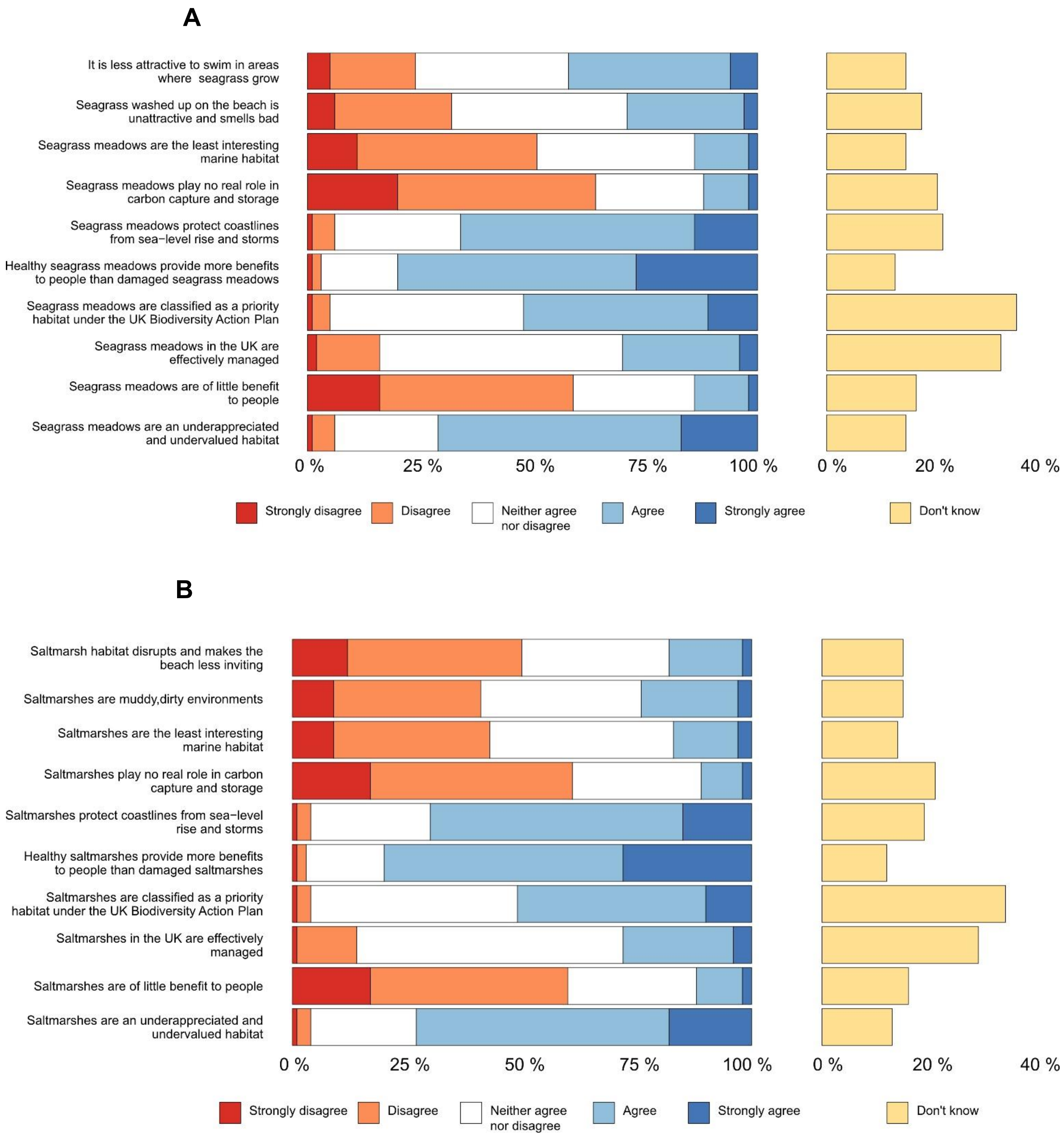


Figure 6.6. Respondent's level of agreement with a range of statements on seagrass meadows (A) and saltmarshes (B) relating to their management, benefits, and aesthetic appeal.

6.4 Discussion

Understanding public perceptions, awareness, and knowledge of blue carbon and relevant habitats (i.e. saltmarshes and seagrass meadows) is a critical step in effectively engaging society and gaining support for the management of coastal wetlands. This study is the first large-scale evaluation of public perceptions towards blue carbon habitats and their management in Great Britain, with implications for coastal wetland management and climate change mitigation policy. This study found public awareness of blue carbon to be generally low, indicated by low levels of familiarity with the term 'blue carbon' and poor perception of the role coastal wetlands play in climate change mitigation, particularly in relation to the provision of other societal benefits. However, perceptions towards saltmarshes and seagrass meadows were mostly positive. Additionally, these findings provide a useful insight into social and demographic characteristics that influence individual perceptions and awareness towards blue carbon, which can be used to promote targeted educational and outreach programmes.

6.4.1 Public awareness and perceptions of blue carbon and coastal wetlands

A key finding from our research is low levels of public awareness and knowledge of blue carbon and the role coastal wetlands play in climate change mitigation. For example, a large proportion of respondents associated the term 'blue carbon' with negative connotations such as worry and concern, or pollution. The majority of respondents also generally perceived terrestrial forests, namely tropical rainforests and temperate forests, to play a more important role in the capture and storage of carbon and thus climate change mitigation, than blue carbon habitats. This is in spite of research and an understanding within the scientific community that coastal wetlands are more efficient natural carbon sinks per unit area than terrestrial forests, both in terms of the longevity of blue carbon sinks and carbon capture rates (McLeod *et al.,* 2011; Bindoff *et al.,* 2019). The public's poor perception of blue carbon observed in our study, indicated by low awareness and knowledge of the carbon capture and storage capabilities of saltmarshes and seagrass meadows, reflects the findings from earlier studies from other regions (Quevedo *et al.,* 2020ab; Kim *et al.,* 2022; Wang *et al.,* 2022).

Heterogeneity in how people perceive the importance of ecosystem services provided by an environment can be attributed to their direct experience and utilisation of such resources or benefits (Jefferson *et al.,* 2015; Tauro *et al.,* 2018). For example, coastal communities in the Philippines who observed the protective function of mangrove forests during storm events (e.g. typhoons) had high levels of awareness for the role mangroves play in coastal protection (Delfino *et al.,* 2015; Quevedo *et al.,* 2020a). In contrast, awareness of the carbon sequestration capacity of mangrove forests and seagrass beds was low among coastal communities in the Philippines where the majority of respondents reported no awareness of this benefit (Quevedo *et al.,* 2020a; Quevedo *et al.,* 2020b). In general, people's connection with habitats and the ecosystem services they provide through direct experiences and use strongly influences their levels of awareness for different ecosystem services (Zhang *et al.,* 2016; Quevedo *et al.,* 2020ab). In the case of blue carbon habitats, this is perhaps one of the main challenges as people, on the whole, do not have direct experiences of carbon sequestration, while public awareness of regulating services in general has been found to be low (Zhang *et al.,* 2016).

Mirroring earlier UK based studies (e.g. Jefferson *et al.,* 2014; McKinley *et al.,* 2020b), this study found public connection with coastal wetlands to be limited. Saltmarshes and seagrass meadows were the least visited marine environment, with over half of respondents having never knowingly visited seagrass meadows before. In addition, when asked about the societal benefits provided by both habitats and reasons for their restoration, respondents reported high levels of uncertainty further indicating limited awareness and knowledge.

Despite low levels of connection with coastal wetlands and limited utilisation of the ecosystem services they provide, we found all services were considered somewhat beneficial by respondents, but noted variation in levels of perceived importance for different benefits. Variation in respondent perceptions likely results from the visibility of benefits rather than utilisation rates (McKinley *et al.,* 2020b). Perceived importance linked to the visibility of benefits may also explain why the same general order of importance was reported by respondents for saltmarshes and seagrass meadows and perhaps reflects current conservation designations in the UK, with saltmarsh and seagrass habitats often designated as nature protection sites for biodiversity and internationally important species. In addition to compensation for historic intertidal habitat loss and the provision of habitat to support biodiversity, provision of cost-effective coastal protection is a primary reason for saltmarsh restoration schemes in the UK (ABPmer, 2023), perhaps making their value in coastal protection more understandable to the wider public.

In general, regulatory services such as pollution control/ water purification and carbon sequestration, are less visible than provisioning services, such as biodiversity enhancement and coastal defence (Potts *et al.,* 2016). As such, we found wildlife habitat and coastal protection were the only services positively perceived by the majority of respondents as benefits provided by saltmarshes and seagrass meadows. Carbon capture and storage is less visible than other societal benefits (McKinley *et al.,* 2020b) and was not widely perceived to be among the important services provided by coastal wetlands in this study. Positively, 94 % of respondents agreed that climate action was urgently needed and two-thirds believed nature could help to reduce climate change impacts (Defra, 2022), but this did not translate into awareness of the role coastal wetlands play in climate change mitigation, nor as a natural climate solution. This suggests a greater need for education and improved communication around saltmarshes and seagrass meadows and their carbon storage capacity.

Climate change generally has high public recognition and often resonates as a threat to the ocean in the mind of the public (Potts *et al.,* 2016). However, whilst the majority of respondents in this study believed climate change was the greatest threat to the natural world, other impacts, notably pollution were considered of highest concern for the marine environment specifically (Defra, 2022). Numerous UK and European studies have also observed pollution, including industrial pollution, marine litter and plastic pollution, as the primary concern among threats to the marine environment (Potts *et al.,* 2016; Lotze *et al.,* 2018), or as the most important indicator of marine health (Jefferson *et al.,* 2014) by the public, largely owed to the associated visual consequences for society (Gelcich *et al.,* 2014; Stafford & Jones, 2019). The high visibility and greater public concern for pollution of the marine environment may also explain why a higher proportion of respondents perceived pollution control/ water purification as beneficial compared to carbon capture and storage, and why a common theme associated with the term 'blue carbon' related to pollution. Ecosystem services with broader societal relevance tend to be recognised as important by the public (Potts *et al.,* 2016) and it is possible the low perception of carbon capture and storage as beneficial could relate to a lack of awareness of how this relates to climate change mitigation, furthering calls for improved communication.

Cultural ecosystem services (mental health, wellbeing support and recreation) were seen as less valuable than other services, or benefits, provided by saltmarshes and seagrass meadows among public audiences. The visibility of ecosystem services and benefits provided by coastal wetlands can therefore influence perceptions around their importance, with a need for greater emphasis in the communication of less visible, but equally if not more important benefits, such as carbon capture and storage (McKinley *et al.,* 2020b).

6.4.2 Sociodemographic drivers of public awareness and perceptions

Societal perceptions and awareness are known to influence the implementation of marine and coastal governance and policy (Carpenter *et al.,* 2018), which is key to the effective management of coastal wetlands and thus the blue carbon concept (Lovelock & Duarte, 2019).

However, the interactions between social and demographic backgrounds and perceptions are often overlooked in public perceptions research on the marine and coastal environment (Jefferson *et al.,* 2021). We have identified several social and demographic characteristics that significantly influenced respondents' familiarity with the term blue carbon, perceptions towards carbon capture and storage as a benefit provided by saltmarsh and seagrass habitats, and awareness of coastal restoration efforts, building on earlier findings from McKinley *et al.* (2020b). For example, younger age groups (aged 35 and below) are more familiar with the term blue carbon with a higher level of understanding of the concept which may reflect the prevalence of current public-facing blue carbon campaigns and information on social media and the internet for which use is generally higher among younger adults (Office for National Statistics, 2020). However, carbon capture and storage is slightly more likely to be perceived as a benefit of saltmarshes and seagrass meadows by those in older demographic age groups (aged 65+).

Aside from social media and the internet, exposure to environmental information and campaigns, including features on blue carbon, is wider for members of environmental charities (Devenport *et al.,* 2021). For example, the climate change mitigation potential of seagrass meadows was highlighted in a blog post by Yorkshire Wildlife Trust (YWT, 2023), whilst the importance of estuarine habitat in the UK for wildlife was the cover feature for the 2019 winter edition of the Marine Conservation Society members' magazine. It is therefore unsurprising that members of environmental charities are more familiar with the concept of blue carbon, aware of coastal wetland restoration efforts across the UK, and are more likely to consider carbon capture and storage as a benefit provided by saltmarsh and seagrass habitat. This corroborates findings from numerous other studies that identify environmental charity membership to improve public awareness and perceptions towards the marine environment (Easman *et al.,* 2018; McKinley *et al.,* 2020b; Devenport *et al.,* 2021). Likewise, respondents who had a connection with marine industry were seen to have greater awareness and knowledge of blue carbon. Citizens worldwide, including the public in the UK, are considered to be relatively well-versed in threats to the marine environment (Lotze *et al.,* 2018), but less well informed about marine conservation and management strategies, under which the blue carbon concept falls, than those employed in marine environmental professions (Easman *et al.,* 2018). Respondents who have a connection with marine industry or an environmental charity membership are likely to have greater access to information relevant to blue carbon and marine environments in general (Devenport *et al.,* 2021). Indeed, in this survey, respondents' reported campaigns by environmental charities in the top five sources for their knowledge about the marine environment (Defra, 2022). Enabling widespread access to such information among the public is an important step towards promoting blue carbon literacy.

Gender is one of the most influential sociodemographic drivers in this study with respondents who identify as male reporting a higher awareness of blue carbon and the role coastal wetlands play in climate change mitigation than female respondents. Similar gender disparities were observed in a European study by Buckley *et al.* (2017) whereby men considered themselves better informed about the majority of marine climate change topics. Previous research shows women's self-perceived knowledge of climate change is typically underestimated in comparison to men (McCright, 2010; Selm *et al.,* 2019), although some studies suggest men have higher levels of environmental and climate change knowledge (Robelia & Murphy, 2012). Further work is required to understand the gender gap in self-reported perceptions and awareness of blue carbon and coastal wetlands. The effect of education levels emerges in this study as an influence differentiating public awareness of the carbon capture and storage capacity of coastal wetlands, in line with Quevedo *et al.* (2020a). Respondents who had attained qualifications above GCSE or equivalent levels were seen to have higher awareness of blue carbon. Furthermore, of all employment statuses, students were most aware of the climate change mitigation function of saltmarshes. Among the public, university students have been shown to possess higher levels of environmental knowledge (Kaplowitz & Levine, 2005), which may be indicative of university initiatives to foster pro-environmentally sustainable behaviour within their student communities (Cotton & Alcock, 2012).

In agreement with Potts *et al.* (2016), we did not find individuals' proximity to the coast, nor the country in which people live, to be a strong influence over public perceptions and awareness of blue carbon. However, whether people lived in England, Scotland, or Wales influenced their awareness of national coastal wetland restoration efforts which are a key staple in the blue carbon concept. English saltmarshes dominate the extent of saltmarsh habitat across Great Britain and current saltmarsh restoration projects in England are greater in number than the rest of the UK (Smeaton *et al.,* 2022; ABPmer, 2023), which may explain why awareness of saltmarsh restoration efforts are higher for people living in England compared to Scotland and Wales. In terms of national seagrass restoration efforts, respondents' living in England and Wales are more aware than those living in Scotland. This is potentially owed to the presence of large-scale restoration efforts in England and Wales that have been well publicised (e.g., the LIFE Recreation ReMEDIES project in southern England and the UK's first major seagrass restoration project in Wales led by Project Seagrass in collaboration with the World Wildlife Fund). Indeed, awareness of restoration efforts was mostly attributed to media coverage by respondents and there is evidence that increases in media coverage on environmental topics, such as climate change issues, enhances public awareness of such topics (Sampei, & Aoyagi-Usui, 2009).

Finally, the level of importance respondents perceived the marine environment to hold was seen to have the strongest effect on levels of awareness and knowledge of blue carbon. In the UK, there has historically been an overall level of disconnect between the public and the marine environment and, in particular, the societal benefits provided by the ocean, coasts and seas (McKinley *et al.,* 2020b). In the context of blue carbon and coastal wetlands, this study further highlights this given that the marine and terrestrial habitats considered most beneficial for carbon capture and storage by respondents were mostly skewed towards those believed to exist outside of the UK. It is important to note that only tropical rainforests were well recognised as important for carbon capture and storage by respondents. Media coverage of the Amazonian rainforest has been relatively widespread in Britain and worldwide due to the rate of deforestation and also more recently, the prevalence of wildfires, and in many cases, the importance of the rainforest is communicated as the 'Lungs of the Planet' (Ladle *et al.,* 2010). With growing focus on restoring coastal habitats, improved communication of the ecosystem services delivered by the marine environment, including climate change mitigation, via mass media campaigns, can enhance awareness of how nature contributes to societal development and foster an appreciation for, and desire to protect, habitats such as coastal wetlands among the public (Van Well *et al.,* 2022). Furthermore, blue carbon habitats present a natural solution to climate change and can help instil a sense of optimism which is perhaps more effective in inspiring and engaging public audiences rather than media coverage of ocean news that is often skewed towards the negative (Easman *et al.,* 2018; McAfee *et al.,* 2019). This is reflective of the change in narrative of ocean communication towards balanced, solution-driven ocean optimism to motivate action within society (Borja *et al.,* 2022; McKinley *et al.,* 2023).

6.4.3 Blue carbon education

In general, research on ocean perceptions note high levels of public interest in the marine environment alongside a desire to protect it (McKinley & Fletcher, 2012; Lotze *et al.,* 2018; Defra, 2022). However, low awareness of societal benefits and a subsequent underappreciation of the value of the marine environment is often reported, particularly in relation to coastal wetlands such as saltmarshes (McKinley *et al.,* 2020b). Indeed, over three-quarters of respondents in our study perceived the marine environment to hold a level of importance to them personally, whilst a similar number showed a keen interest in protecting the marine environment and support the creation of marine protected areas (Defra, 2022), despite low connections with saltmarsh and seagrass habitats and a limited awareness of blue carbon. The term 'blue carbon' only recently became a 'hot' topic within the scientific community in 2017 (Duarte de Paula Costa & Macreadie, 2022) and there has been a lag in blue carbon communication reaching public audiences. As such, there is a lack of public-facing information and education available on blue carbon. Previous studies (e.g. Kim *et al.,* 2022; Wang *et al.,* 2022) suggested public audiences may be receptive to outreach and environmental education campaigns focused on blue carbon with respondents reporting considerable interest in learning more about these topics. Whilst we did not ask respondents directly whether they were interested in learning more about blue carbon, we found a slight majority disagreed that saltmarshes and seagrass meadows were the least interesting marine habitat, supporting the notion that there has been a shift in how people view coastal wetlands over the past few decades away from habitats of perceived low value (Hudson *et al.,* 2021). There was also an awareness that healthy saltmarsh and seagrass habitats provide more societal benefits than damaged or degraded habitats and over 50 % of respondents disagreed that these habitats are of little benefit to people. This reflects the general view that the public in the UK and Great Britain are enthusiastic about marine conservation and protection (Hawkins *et al.,* 2016; Defra, 2022), suggesting the British public could be a receptive audience for environmental education on blue carbon and more general coastal wetland outreach. However, the majority of respondents in this study associated the term blue carbon under the theme chemistry and atmospheric conditions. As the public image of chemistry often has negative connotations (Hartings & Fahy, 2011), this perception of the term and the wider concept of blue carbon may present a barrier to public engagement.

Environmental education can have a meaningful impact in improving public awareness and encouraging a change in public attitudes and behaviours geared towards the preservation of the marine environment and the societal benefits it provides (Lotze *et al.,* 2018; Forleo *et al.,* 2021). There are already numerous calls to enhance public understanding of coastal wetlands and the role they play in climate change mitigation, alongside other societal benefits they provide, through targeted awareness raising campaigns that are effective and appropriate (McKinley *et al.,* 2020b; Quevdeo *et al.,* 2020b). In turn, environmental education can, alongside other drivers, encourage behaviour change that contributes to coastal wetland conservation (Ibrahim *et al.,* 2012), and improved public perceptions can support future policy changes that promote the effective management of coastal wetlands and climate change mitigation strategies (Jerath *et al.,* 2016). By better understanding public audiences, targeted, research-driven educational campaigns can be designed to foster enhanced levels of knowledge towards blue carbon and coastal wetlands.

While blue carbon initiatives currently attract private investment and there is growing interest in coastal wetland restoration from an economic and political lens, particularly with the development of carbon markets to offset emissions and the inclusion of coastal wetlands in international climate agreements (Macreadie *et al.,* 2021; see also Wylie *et al.,* 2016 for case studies of blue carbon projects and their carbon financing mechanisms), a gap clearly remains in terms of public perception and attitudes towards these ecosystems and the overall concept of blue carbon. With improved education around blue carbon, the interest in coastal wetland conservation and restoration could extend to the public and, in turn, public pressure to protect historically underappreciated and undervalued saltmarsh and seagrass habitats could have a large impact on future environmental decision making and priorities.

6.5 Conclusions and recommendations

The climate change mitigation potential of coastal wetlands, through the capture and storage of blue carbon, has gained significant political attention in recent years, with increased interest in their effective management (i.e. conservation and restoration). However, there has been a lag in blue carbon communication reaching public audiences with a lack of public-facing information and education available. Consequently, we found public awareness of the societal benefits saltmarshes and seagrass meadows provide, in particular the capture and storage of blue carbon, is limited and the public's low connection with these habitats is evident. Despite our findings, ecosystem services can be a useful tool in helping to articulate the importance of the marine environment, notably for previously underappreciated and undervalued habitats, such as coastal wetlands. Improved blue carbon education could thus foster an appreciation for, and desire to protect, coastal wetlands among the public.

As a result of this study, we make the following recommendations:

1) Targeted blue carbon and coastal wetland awareness raising campaigns that are effective and appropriate should be developed and delivered, potentially via mass media approaches, to improve access to information beyond communications that are generally restricted to members of environmental charities.

2) Public perceptions and awareness are influenced by sociodemographic characteristics and knowledge of the main drivers of perceptions can help to identify target audiences, ensuring the social and demographic backgrounds of these groups underpin educational blue carbon programmes and campaigns.

3) Ongoing marine social science research on blue carbon is needed to elucidate potential changes in public perceptions as the field continues to gain traction within economic and political lenses.

Such efforts would complement previous calls for greater research emphasis on public perceptions of coastal wetlands in temperate regions and improved communications of less visible ecosystem services (e.g., McKinley *et al.,* 2020b), and build on acknowledgements within the scientific blue carbon community of the need for trans-disciplinary approaches to support coastal habitat restoration and the successful adoption and longevity of blue carbon as a natural climate solution (Waltham *et al.,* 2020; Macreadie *et al.,* 2022). In turn, enhanced public perceptions, knowledge, and attitudes towards blue carbon can support future policy changes that promote the effective management of coastal wetlands and climate change mitigation strategies.

Chapter 7 of 7

Conclusions

7.1 Introduction

Scientific interest and political discourse around blue carbon has grown exponentially in recent years (Macreadie *et al.,* 2021; Duarte de Paula Costa & Macreadie, 2022). The rising interest in blue carbon is rooted in the carbon accumulation capacity of coastal wetlands at orders of magnitude above that of terrestrial forests per unit area, and the preservation of stored carbon over climatically-relevant timescales up to millennia (Mcleod *et al.,* 2011; Macreadie *et al.,* 2021). This has led to increased efforts to incorporate blue carbon habitats into global climate policies and carbon markets (Wylie *et al.,* 2016; Vanderklift *et al.,* 2019). However, as coastal wetlands rank among the most threatened habitats in the world, global interest in blue carbon habitats also concerns their potential conversion from carbon sinks to carbon sources through the release of ancient stored carbon (Macreadie *et al.,* 2013; Howard *et al.,* 2017; Lovelock *et al.,* 2017).

In line with their role as natural climate solutions, the effective management of blue carbon habitats can simultaneously enhance carbon capture and storage while avoiding CO_2 emissions related to habitat degradation and loss (Duarte *et al.,* 2013; Macreadie *et al.,* 2017b; Kelleway *et al.,* 2020). The effective management of coastal wetlands is, therefore, core to the concept of blue carbon strategies as an approach to mitigate against climate change (Duarte *et al.,* 2013). Given that saltmarshes are considered the most widespread and important blue carbon habitat outside of the tropics (Mcleod *et al.,* 2011), this thesis focused on saltmarsh ecosystems.

As a result of global interest in blue carbon beyond just the scientific community, the motivation to address knowledge gaps that can inform blue carbon management actions has built (Macreadie *et al.,* 2019). Current blue carbon estimates for natural and restored saltmarsh ecosystems reveal high variability across spatial and temporal scales (e.g., Ouyang & Lee, 2014; Ewers Lewis *et al.,* 2018; Rogers *et al.,* 2019a). Thus, the elucidation of environmental drivers that influence variability in blue carbon stocks and accumulation rates has been identified as a key objective in progressing blue carbon science (Ewers Lewis *et al.,* 2020). Furthermore, knowledge of major environmental drivers of blue carbon variability is imperative for the effective management of saltmarshes as a natural climate solution.

Whilst information on environmental drivers can guide blue carbon management actions, it is now recognised that the potential contribution of blue carbon to natural climate solutions is dependent on societal actions (Macreadie *et al.,* 2021). The successful adoption and longevity of blue carbon as a natural climate solution requires the incorporation of social science (Macreadie *et al.,* 2022), with societal perceptions of blue carbon presenting an important driver behind the success of effective coastal management. In order to generate public interest and support for the effective management of coastal wetlands, current levels of public awareness and knowledge of blue carbon need to be understood from a diversity of perspectives.

Therefore, the overarching aim of this thesis is to advance knowledge on the role of saltmarsh ecosystems in climate change mitigation, in order to guide effective coastal management. The subsidiary aims were to:

(i) *Identify environmental characteristics that drive spatial and temporal variability in saltmarsh blue carbon stocks and accumulation rates, and;*

(ii) *Understand current levels of public awareness and perceptions of blue carbon and coastal wetlands.*

The thesis aims were achieved through the following objectives, each relating to a specific data chapter:

1. To ascertain whether rates of relative sea-level rise are a dominant control on blue carbon accumulation rates in saltmarshes (**Chapter 3**).

2. To investigate whether saltmarsh resilience to relative sea-level rise influences blue carbon stocks (**Chapter 4**).

3. To determine the major environmental drivers of blue carbon stocks for restored saltmarshes (**Chapter 5**).

4. To quantify public perceptions, understanding and awareness of blue carbon and associated habitats (saltmarshes and seagrasses), while identifying social and demographic backgrounds that influence perceptions (**Chapter 6**).

This rest of this chapter summarises the main findings from the preceding chapters and highlights key recommendations for future blue carbon management and priority areas for future work.

7.2 Summary of thesis results

Subsidiary aim (i): *Identify environmental characteristics that drive spatial and temporal variability in saltmarsh blue carbon stocks and accumulation rates*

This subsidiary aim was addressed through the following three objectives covering a global, regional and local scale:

7.2.1 The influence of relative sea-level rise on blue carbon accumulation

Chapter 3 showed that temporal variability in carbon accumulation rates (CAR) were largely explained by rates of relative sea-level rise (RSLR). The hypothesis that '*the creation of vertical accommodation space, as a direct result of relative sea-level rise, enhances blue carbon accumulation rates in saltmarshes'* was tested in this chapter across sites in North America (Chezzetcook, Sanborn cove and Barn Island), Iceland (Viðarhólmi), Australia (Little Swanport) and New Zealand (Mokomoko and Pounawea). Rogers *et al.* (2019a) provided the basis for the hypothesis by reporting that gains in carbon accumulation and sediment accretion were proportional to the vertical accommodation space – the space available for mineral and organic matter accumulation – created by RSLR. The mechanism by which CAR increases with RSLR has been demonstrated by numerous modelling and field-based research on local, national and global scales (Kirwan & Mudd, 2012; Ruiz-Fernández *et al.,* 2018; Breithaupt *et al.,* 2020; Rogers *et al.,* 2019a; Wang *et al.,* 2019; Wang *et al.,* 2021; Weston *et al.,* 2023). However, this study presents the first evidence of this mechanism using field research that pairs location-specific, high-resolution sea-level reconstructions with CAR in saltmarshes on a global scale.

Over multi-centennial timescales, the rate of RSLR significantly influenced CAR at two out of seven locations, but no trend was observed when all sites were combined in a model and the relationship was highly site-specific. Other climatic factors known to influence CAR in saltmarshes, such as temperature (Rogers *et al.,* 2019a; Wang *et al.,* 2021), may have exerted a greater driving force over CAR variability over multi-centennial timescales. However, the lack of observed relationship between RSLR and CAR is potentially a result of the uncertainties associated with chronology and relative sea level when using proxy sea-level records at such high-resolution scales covering multi-centennial timescales (Cahill *et al.,* 2015). Proxy sea-level records have been demonstrated to be compatible with more concise, instrumental tide-gauge records over the 20th century (Gehrels *et al.,* 2013), so this study then focused on that time period.

Over the 20th century, RSLR presented a dominant control over CAR, with the model, containing all sites, explaining almost half of the observed variation in CAR. Rates of RSLR were also found to govern the variability of saltmarsh CAR on a spatial scale during the 20th century; rates of sediment accretion and CAR were significantly higher for sites in Australia and New Zealand than sites in the North Atlantic. This observation was attributed to a greater acceleration in rates of RSLR for locations in the Southern Hemisphere during the 20th century, a trend supported by previously published sea-level reconstructions (Gehrels *et al.* 2012; Dangendorf *et al.* 2019).

Overall, this chapter presents strong support for the hypothesis that '*the creation of vertical accommodation space, as a direct result of relative sea-level rise, enhances blue carbon accumulation rates in saltmarshes*', particularly for the 20th century dataset. Weaker support for the hypothesis over multi-centennial timescales may be related to the uncertainty in proxy sea-level records (Cahill *et al.,* 2015).

7.2.2 Blue carbon stocks and saltmarsh resilience to relative sea-level rise

Chapter 4 is an experimental chapter in which brick dust marker horizon plots, laid on the surface of saltmarshes in Maine, USA, in 1986, were used to measure saltmarsh vertical accretion rates and blue carbon stocks. This study expands upon work by Goodman *et al.* (2007) who relocated a number of marker horizon plots in 2003. Fieldwork for this study was conducted in 2019 and attempts were made to relocate marker horizon plots. In the limited number of sites where this was successful, sediment cores were extracted from the marsh surface to the top of the marker horizon. Sediment accretion rates reported by Goodman *et al.* (2007) were used to identify the 2003 horizon. Vertical accretion rate and carbon stock could then be compared over two distinct time periods: 1986 and 2003, and then 2003 to 2019, with the latter time period representing more rapid rates of RSLR compared to previous decades.

Saltmarsh accretionary balance, calculated by subtracting the rate of RSLR from average sediment accretion rates over the same time period, was used to indicate saltmarsh resilience. This study was underpinned by three hypotheses: (1) that sediment accretion rates are higher in the more recent time period due to increased rates of RSLR; (2) that carbon stock is greater when saltmarshes have a greater accretionary balance with rates of RSLR; and, (3) that sites that experience higher tidal range will have greater carbon stock.

In this study, only three marker horizons plots were relocated across two sites in 2019. In the original study in 1986, a minimum of six plots were established across 26 sites (Wood *et al.,* 1989), from which relocation was successful at 11 sites in 2003 (Goodman *et al.,* 2007). Even under ideal conditions (e.g., fine-sediment saltmarsh with limited influence of bioturbation and erosion), marker horizons become more problematic to relocate over time due to dispersion (Callaway *et al.,* 2013). The majority of previous studies reporting accretion rates from marker horizons do so on a scale of months to years (van Wijnen & Bakker, 2001; McKee, 2011; Nolte *et al.,* 2013; Saintilan *et al.,* 2013; Swales & Lovelock, 2020), with implications for their successful retrieval over longer timescales such as decades, as in our study. The two sites where marker horizons were relocated in our study are located in southwestern Maine (Scarborough Central) and the other in the north-east (Cobscook Bay). These sites are at either end of a natural gradient in the rate of RSLR and tidal range which both increase from the south-west to the north-east (Holgate *et al.,* 2013; NOAA, 2022).

Findings related to the first hypothesis were highly site-specific. At both sites, the rate of RSLR was higher in the more recent time period, although the disparity in rates between sites increased (Holgate *et al.,* 2013). At Scarborough Central, sediment accretion and carbon stock were greater when the rate of RSLR was higher, in line with the first hypothesis. In contrast, sediment accretion and carbon stock were lower at Cobscook Bay in the most recent time period when rates of RSLR were higher, rejecting the first hypothesis. These findings corroborate analytical and numerical models that show carbon stocks to increase with RSLR until a critical threshold is reached, causing the accumulation of carbon stocks to decline (Mudd et al., 2009; DeLaune & White 2012; Kirwan & Mudd, 2012). Previous studies predicted the threshold rate of RSLR to be between 4 and 4.5 mm yr-1 (Goodman *et al.,* 2007; Kirwan & Mudd, 2012), which was surpassed at Cobscook Bay in the more recent time period. This suggests that Cobscook Bay became overwhelmed by RSLR in the recent time period with insufficient sediment accretion rates to maintain equilibrium with RSLR (Mudd *et al.,* 2009). Carbon stock thus depends on saltmarsh ability to keep pace with rates of RSLR but more evidence is needed to support the second hypothesis that carbon stock is greater when saltmarshes have a greater accretionary balance with RSLR. Finally, despite a decline in sediment accretion and carbon stock in the more recent time period, overall mean carbon stock at Cobscook Bay, where tidal range was two-fold greater, was higher than Scarborough Central, in support of the third hypothesis. This could be attributed to a longer period of inundation and subsequent greater opportunity for the capture of allochthonous carbon (Wollenberg *et al.,* 2018). Due to the limited sample size, further investigation is required to quantitively confirm these findings.

7.2.3 Major environmental drivers of blue carbon stocks in restored saltmarshes

As well as measuring blue carbon stocks for restored saltmarshes in the Blackwater Estuary in Essex, UK, **Chapter 5** of this thesis also identified the main environmental characteristics that drove the observed variability in stocks between restored sites. Saltmarshes in this study were restored by managed realignment (MR), the predominant method of saltmarsh restoration in the UK.

Relative tidal height, i.e. the site's elevation in the tidal frame, was identified to be the most important driver governing carbon stock variability for restored saltmarshes. The MR site at Tollesbury, the only site with a relative tidal height below mean high water neap (MHWN) tidal levels, had a carbon stock significantly lower than all other sites (restored and natural) in the estuary. The remaining restored and natural sites had similar carbon stocks which did not differ significantly. Despite a similar time since restoration, the carbon stock at Tollesbury MR was also two-fold lower than the other two restored sites in the estuary. MHWN tidal levels represent the lower limit of pioneer marsh establishment (Balke *et al.,* 2016; Lawrence *et al.,* 2018; Hudson *et al.,* 2021). The low elevation of Tollesbury MR meant the site was dominated by pioneer plant communities that have been reported to have lower carbon stocks than communities associated with higher elevations in the mid-low and high marsh zones (Smeaton *et al.,* 2022), a finding consistent with our study. Restored saltmarshes at low elevations also experience more frequent and prolonged inundation periods which, coupled with poor drainage of anoxic sediments, create low sediment redox potential (Davy *et al.,* 2011; Mossman *et al.,* 2020). These conditions inhibit the colonisation of vegetation communities associated with higher elevation zones and greater carbon stock (Garbutt *et al.,* 2006).

Vegetation characteristics were also important for predicting carbon stock variability of restored sites above MHWN tidal levels (i.e., when Tollesbury MR was removed from analysis), in line with previous work (Adams *et al.,* 2012). Specifically, *Atriplex portulacoides* was identified as a key species for significantly enhancing carbon stocks at restored sites in this study. *A. portulacoides* forms dominant, almost mono-specific stands (Penk *et al.,* 2020), which holds management implications including potential decisions to either focus on the establishment of key species or the development of equivalent vegetation communities to nearby natural saltmarshes, depending on the overall goal of saltmarsh restoration.

It is important to note that sediment supply was not factored into this analysis, but may have contributed greater predictive power to models when explaining the observed variability in carbon stock for restored sites. For example, the Blackwater Estuary is characterised by relatively low sediment supply (Ladd *et al.,* 2019) and the construction of a counter sea wall at Tollesbury MR may have further restricted sediment supply (Garbutt *et al.,* 2008), in turn preventing the site from reaching elevations above MHWN tidal levels.

As coastal wetland restoration for blue carbon benefits has become a recognised tool for climate change mitigation (Wylie *et al.,* 2016; Duarte *et al.,* 2020), the findings from this chapter can be used to inform future restoration schemes that aim to maximise climate change mitigation (see section 7.3).

Subsidiary aim (ii): *Understand current levels of public awareness and perceptions of blue carbon and coastal wetlands*

This subsidiary aim was addressed through the following objective:

7.2.4 Public perceptions, understanding and awareness of blue carbon and coastal wetlands

There is an acknowledgement within the blue carbon community that trans-disciplinary approaches are necessary for the successful adoption and longevity of blue carbon as a natural climate solution (Waltham *et al.,* 2020; Macreadie *et al.,* 2022), yet social research on blue carbon is severely lacking. **Chapter 7** is the first large-scale (n = 12,166) evaluation of public perceptions, knowledge and attitudes towards blue carbon, using Great Britain as a case study. This study focused on the two coastal wetland ecosystems found in the UK: saltmarshes and seagrass meadows. Results showed a limited awareness of blue carbon among the public, and a poor perception of the role of coastal wetlands in climate change mitigation, particularly in relation to terrestrial forests.

A small proportion of respondents (4 %) reported that they knew and understood the term ‘blue carbon’, which may result from a lack of public-facing information available on blue carbon. Blue carbon was not widely perceived to be a societal benefit provided by saltmarshes and seagrass meadows, despite two-thirds of respondents believing nature could contribute to climate change mitigation (Defra, 2022). The perceived importance of societal benefits provided by coastal wetlands were linked to visibility whereby regulatory services, such as carbon sequestration, are less visible than provisioning services, such as biodiversity habitat and coastal protection (McKinley *et al.,* 2020; Potts *et al.,* 2016). Awareness of coastal restoration efforts for saltmarshes (17 %) and seagrass meadows (14 %) were low, and both habitats were the least visited of all marine habitats, with the majority of respondents having never visited them before. This reflects the public's low connection with coastal wetlands which has been previously reported in the UK (McKinley *et al.,* 2020b).

During the online survey, a range of social and demographic characteristics were obtained from respondents. These characteristics, often overlooked in public perceptions research on the marine and coastal environment (Jefferson *et al.,* 2021), were analysed against respondents' responses to provide an insight into drivers of perception of blue carbon. Respondent age, gender, educational attainment, environmental charity membership and connection with marine industry were all found to significantly influence perceptions and awareness of blue carbon. For example, environmental charity membership enhanced overall perceptions and awareness of blue carbon, in line with previous studies that showed such membership to improve perceptions and awareness of the marine environment in general (Easman *et al.,* 2018; McKinley *et al.,* 2020b; Devenport *et al.,* 2021). How important respondents perceived the marine environment to be was the most dominant driver of public perceptions on blue carbon and coastal wetlands. Positively, the majority of respondents believed the marine environment to hold some level of importance to them, suggesting the public could be a receptive audience for blue carbon outreach programmes. This supports the general view that the British public are enthusiastic about marine conservation (Hawkins *et al.,* 2016; Defra, 2022).

Findings from this study can support the development of relevant, targeted educational and outreach programmes to enhance blue carbon literacy and more generally, ocean literacy. In turn, public understanding of the importance of blue carbon could help gain wider support for coastal wetland management and restoration initiatives.

7.2.5 Synthesis

Overall, this thesis has contributed new knowledge to support the effective management of saltmarshes for climate change mitigation by identifying environmental drivers that operate across spatial and temporal scales to control saltmarsh blue carbon dynamics.

Sea-level rise was found to present a dominant control over carbon accumulation rates in saltmarshes in the North Atlantic and in Australia and New Zealand over centennial, and in some cases, multi-centennial timescales (**Chapter 3**). While relative sea-level rise (RSLR) was found to enhance blue carbon accumulation rates through the creation of vertical accommodation space in **Chapter 3**, this thesis also provided evidence that carbon stocks can decline if saltmarshes are unable to keep pace with rising sea levels, although further work is required to quantitatively confirm this finding (**Chapter 4**). An exploration of carbon stocks across saltmarshes on a regional gradient of RSLR in **Chapter 4** supported the concept of critical thresholds which have been proposed by numerical modelling. Here, blue carbon stocks and accumulation rates increase linearly under greater rates of relative sea-level rise until sediment accretion rates are no longer able to keep the saltmarsh in equilibrium, resulting in a decline in carbon accumulation rates and stocks (Mudd *et al.,* 2009; Kirwan & Mudd, 2012). It is also worth noting that temporal variations in RSLR observed in **Chapter 4** incorporated the recent acceleration in global rates of RSLR over the 21st century (Kopp *et al.,* 2016) which were not reflected across temporal trends in **Chapter 3**. This holds implications as global mean sea-level rise continues to accelerate with projections that the rate of sea-level rise will reach 15 mm yr^{-1} by 2100 (Oppenheimer *et al.,* 2019).

The respective height of restored saltmarshes in the tidal frame was proposed to be a key driver in the development of blue carbon stocks (**Chapter 5**). Relative tidal height has a strong association with sea-level rise and factors that influence the longevity of restored saltmarshes, and their carbon stocks, under projected levels of RSLR, such as local sediment supply, require closer attention. Ongoing research to understand the influence of relative sea-level rise on blue carbon dynamics in natural and restored saltmarshes, and how this varies across spatial and temporal scales, is therefore fundamental for the effective management of saltmarshes for climate change mitigation.

Findings from this thesis demonstrate that fine-scale drivers can also predict blue carbon stocks on a local (e.g. estuarine) scale, and that multiple drivers likely interact to shape blue carbon stocks (**Chapter 5**). For example, tidal and vegetation characteristics were observed to have the greatest influence over the development of carbon stocks in restored saltmarshes. Elevated marsh platforms likely provide suitable characteristics for the development of mature plant communities associated with the mid-low marsh zone which, in line with previous studies, were associated with greater carbon stocks than pioneer zone communities (**Chapter 5**). Conversely, marsh zone was not found to be a driver of carbon stock in **Chapter 4** as there was no significant difference in sediment carbon stock between low and mid marsh communities. Greater variability in carbon stocks between saltmarsh sites were also observed in **Chapter 4** while relatively low variability was found between sites in **Chapter 5**. The noted differences in spatial variability of carbon stock and the influence of saltmarsh zone could relate to the different spatial scales each chapter focuses on from local (**Chapter 5**) to regional (**Chapter 4**). Regional drivers of carbon stock, including RSLR and tidal range, may have a greater influence over carbon stock variability than more local scale controls such as relative tidal height and vegetation community composition that operate across finer spatial scales. Further work is required to tease out the influence of key environmental drivers explaining blue carbon variability across global, regional and local scales. Such knowledge in addition to the findings from **Chapter 5** can be used to identify priority areas for saltmarsh restoration where blue carbon is likely to be maximised. This research is timely given recent calls for blue carbon habitat restoration to become a focal point of the current UN Decade on Ecosystem Restoration (Macreadie *et al.,* 2021).

Advances in blue carbon science can be supported through synergies between both the UN Decade on Ecosystem Restoration and the UN Decade of Ocean Science for Sustainable Development (Macreadie *et al.,* 2021), the latter of which calls for a stronger integration of natural and social sciences (Claudet, 2021; McKinley *et al.,* 2023). This thesis is trans-disciplinary in nature and presents the first large-scale evaluation of social research on blue carbon. Baseline data on public perceptions, awareness and knowledge of blue carbon and coastal wetlands is discussed in **Chapter 6**. Societal engagement is highly influential for achieving marine conservation outcomes (Jefferson *et al.,* 2015) and as such, public perceptions of blue carbon and coastal wetlands in general, present an alternative driver of effective management. The baseline data collected as part of this thesis can inform measures to enhance blue carbon literacy. It is important to note that all drivers of blue carbon dynamics identified in this thesis, both environmental and social, are somewhat amenable to management, however further research on the social dimension of blue carbon is thoroughly required to progress blue carbon science.

Overall, the key take home messages from this thesis are as follows:

- **Relative sea-level rise is a key driver of carbon accumulation rates in saltmarshes.** The level of influence is likely to differ across spatial and temporal scales and is closely linked to saltmarsh resilience to increasing rates of RSLR.

- **The influence of key environmental characteristics that drive variability in saltmarsh blue carbon dynamics may differ across spatial scales (local, regional and global).**

- **Fine-scale controls (e.g., relative tidal height and vegetation community composition) can drive variability in blue carbon stocks between restored saltmarshes on an estuarine scale.** Identifying such drivers can be used to inform management recommendations and priority locations for future saltmarsh restoration schemes that aim to maximise climate change mitigation.

- **Advances in blue carbon literacy among the public is required.** Perceptions and awareness of blue carbon and coastal wetlands (saltmarshes and seagrasses) is low across Great Britain.

- **As a relatively young research field, further work is required to progress blue carbon science.**

7.3 Recommendations for future blue carbon management and priority research

From this thesis, the following recommendations for effective coastal management of blue carbon have been drawn, alongside priority areas identified for future research.

7.3.1 Blue carbon and relative sea-level rise

Given blue carbon dynamics under relative sea-level rise were mediated by sediment accretion in both **Chapter 3** and **Chapter 4**, monitoring of sediment supply and sediment accretion rates are recommended. This can improve predictions of saltmarsh resilience to relative sea-level rise and help recognise key locations for climate change mitigation, or locations that require greater protection. For example, Australia is predicted to become a hotspot for blue carbon accumulation in the 21st century due to the expansion of coastal wetlands under relative sea-level rise (Wang *et al.,* 2021). In contrast, in northeastern USA, saltmarshes are experiencing diminishing sediment supply, and are characterised by sediment accretion rates that are notably lower than current rates of relative sea-level rise with a reduction in organic matter accumulation (Carey *et al.,* 2017). In some cases, such as in the USA and the Netherlands, engineering solutions have been used successfully to promote saltmarsh resilience to RSLR through the management of sediment supply and tidal inundation (Stralberg *et al.,* 2011; Vandenbruwaene *et al.,* 2011). Although, given that RSLR is a complex and global issue, adaptation measures may need to be integrated into saltmarsh management (Macreadie *et al.,* 2017b).

Future studies should also attempt to better quantify the contribution of organic and mineral sediment accretion, and sources of carbon, to improve knowledge behind the mechanism by which saltmarsh blue carbon is enhanced as a result of vertical accommodation space. Further research is also required to quantitively confirm the hypothesis that '*blue carbon stock will be greater when saltmarshes have a greater accretionary balance with rates of relative sea-level rise*'.

As identified in **Chapter 3**, the reduction in labile carbon with soil age and depth diminishes apparent carbon accumulation rates (Davis *et al.,* 2015; Weston *et al.,* 2023), leading to the potential overestimation of carbon accumulation rates. Future research identifying temporal trends in carbon accumulation rates as a result of relative sea-level rise should adopt approaches to discriminate between labile and refractory carbon to address this limitation. Two-pool modelling, for example, accounts for faster-cycling organic carbon pools in estimations of carbon accumulation rates (Belshe *et al.,* 2019). This method will improve insights on the influence of relative sea-level rise on saltmarsh climate change mitigation, as carbon accumulation rates will be measured from refractory carbon that is incorporated into long-term storage (Dahl *et al.,* 2016).

7.3.2 Maximising blue carbon stocks through saltmarsh restoration

Key practical recommendations for future saltmarsh restoration schemes were drawn from **Chapter 5.** Findings suggest that, where possible, sites above MHWN tidal levels should be selected for saltmarsh restoration schemes to enable significant carbon stocks to develop, equivalent to those of nearby natural saltmarshes. In line with recommendations from **Chapter 3** and **Chapter 4**, monitoring of local sediment supply is recommended, in this case to ensure the restoration success of saltmarshes under projected sea-level rise (Liu *et al.,* 2021). Preference should be given to the implementation of restoration sites in areas of known high sediment supply, particularly when initial elevations are below MHWN tidal levels. Alternatively, topographic manipulations (e.g., sediment additions) to enhance the elevation of restored sites could create suitable conditions for enhanced carbon stocks and encourage the establishment of key vegetation species (Mossman *et al.,* 2020; Lawrence *et al.,* 2022). Furthermore, monitoring pre- and post-restoration carbon stocks, alongside other environmental characteristics, is recommended to determine the success of saltmarsh restoration schemes in line with pre-specified goals of such schemes.

High levels of collinearity were observed between multiple predictor variables in **Chapter 5**, suggesting multiple drivers likely interact to shape blue carbon stocks. Structural equation modelling can be used to evaluate indirect effects that may be mediated by other environmental characteristics (Pagès *et al.,* 2019) and could help to improve predictions of carbon stock. Further work to identify key drivers of carbon stocks in restored saltmarshes could also utilise larger geographic scales that incorporate fine-scale drivers known to influence variability with regional gradients. This includes soil type and sediment supply (Ford *et al.,* 2016; Ladd *et al.,* 2019).

Environmental drivers identified from this thesis could be used in predictive mapping to identify potential ‘hotspots’ for maximising blue carbon stocks through saltmarsh restoration. For example, Ewers Lewis *et al.* (2020) modelled regional carbon stock in southeastern Australia using fine-scale ecological and geomorphological drivers, highlighting areas that represented important carbon sinks. This approach can be adapted to identify priority areas for saltmarsh restoration. Similar assessments that focus on environmental drivers for other ecosystems services (e.g., habitat provision for biodiversity and coastal protection) could be incorporated into a national framework identifying key locations for saltmarsh restoration efforts, depending on the overall goal of restoration.

Finally, further research opportunities and management actions should involve positive, open dialogues with landowners. Linking to **Chapter 6**, stakeholder support for saltmarsh restoration is vital for successful implementation, and involves numerous actors including the public, and particularly for managed realignments in the UK, private landowners and farmers (Liski *et al.,* 2019).

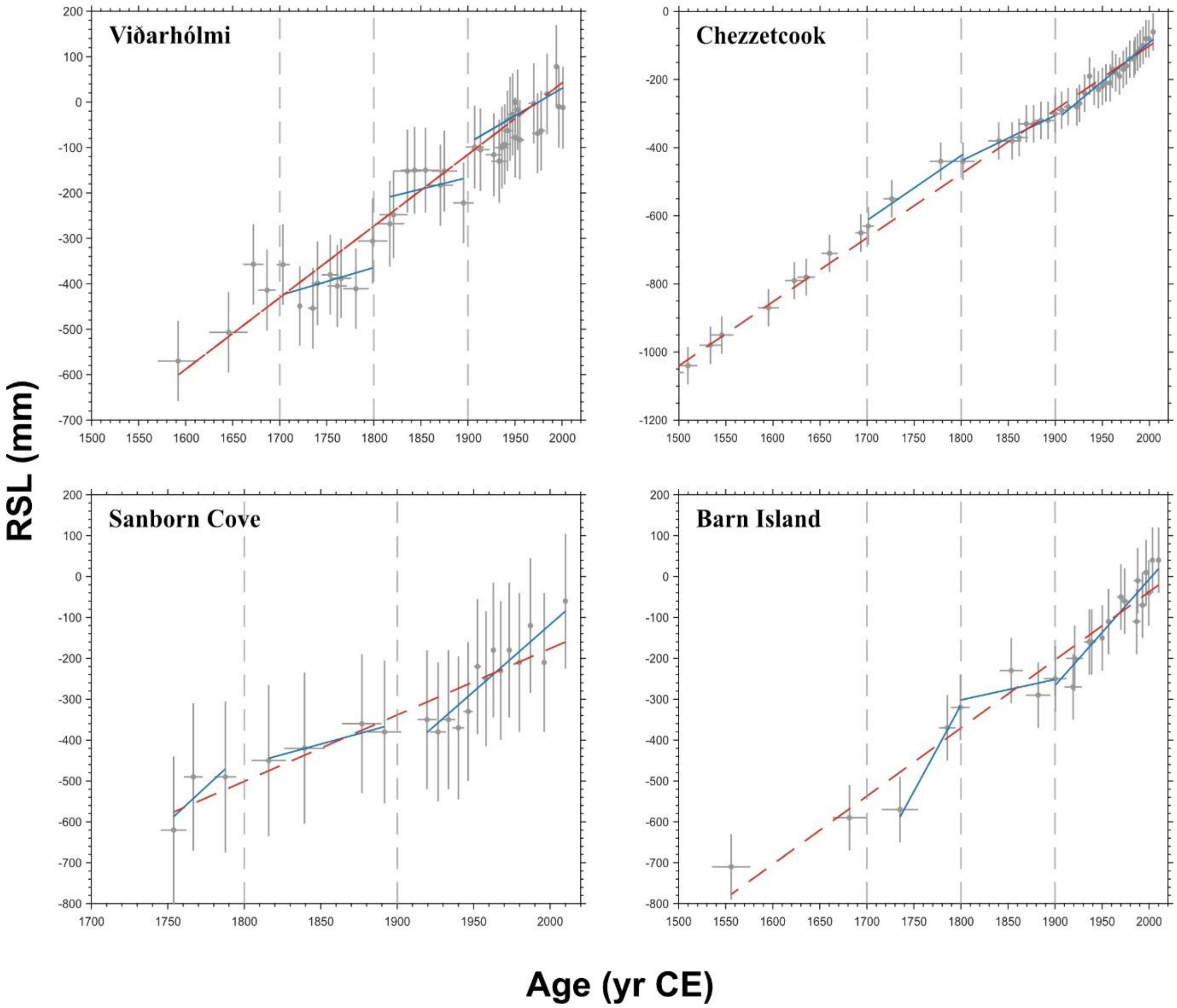
7.3.3 Societal engagement with blue carbon

Chapter 6 provided the baseline from which to improve current levels of public perceptions and awareness of blue carbon and to change the narrative for coastal wetlands. Target audiences for educational campaigns can be identified using social and demographic drivers that explained the variability in perceptions of blue carbon. For example, subscription to an environmental charity significantly increased respondents’ awareness of the blue carbon capacity of coastal wetlands. Promoting the wider reach and accessibility of blue carbon campaigns beyond the traditional route of information disseminated by environmental charities to their members is thus highly recommended.

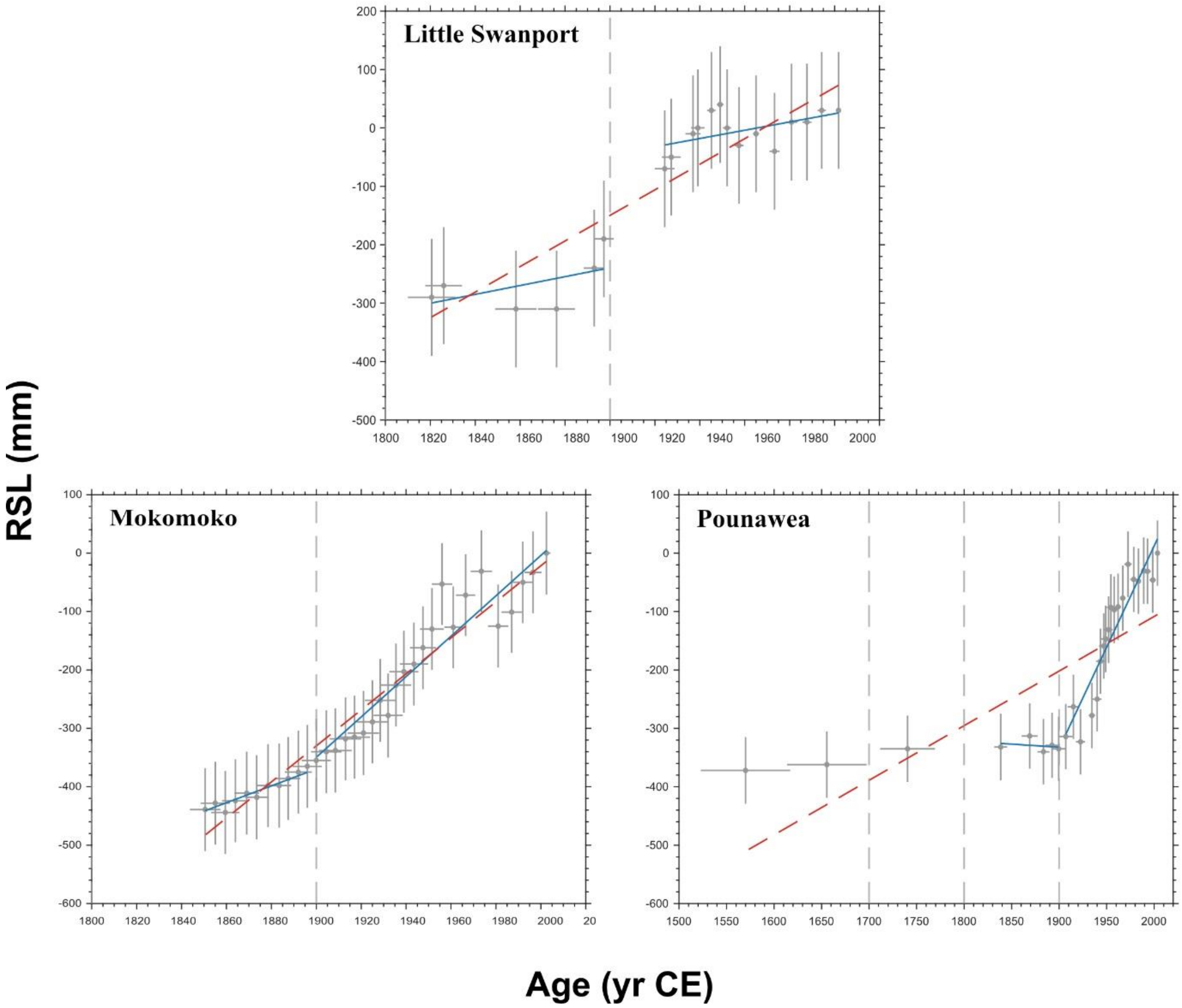
A concerted effort to incorporate the social dimension of blue carbon research is necessary to identify any trends in perceptions as the field continues to gain traction, and as a method to quantitatively assess the effectiveness of ongoing, and future, outreach and educational programmes. Improved blue carbon literacy among the public could help to build societal support for the effective management of coastal wetlands, such as saltmarshes.

Appendix A

Appendix for Chapter 3



A1. Relative sea level at four Northern Hemisphere locations ± 1σ age and vertical uncertainties for each sea-level index point. Rates of relative sea-level rise are shown across the whole core sequence (red dashed line), and during the 18th, 19th and 20th centuries (solid blue lines), where applicable.



A2. Relative sea level at three Southern Hemisphere locations ± 1σ age and vertical uncertainties for each sea-level index point. Rates of relative sea-level rise are shown across the whole core sequence (red dashed line), and during the 18th, 19th and 20th centuries (solid blue lines), where applicable.

A3. Sediment accretion rate and carbon accumulation rate ± standard deviation for all locations across the whole sequence, representing multi-centennial timescales, and in the 20th century. Superscripts denote whether the location is found within the Northern[a] or Southern[b] Hemisphere.

Location	SAR (mm yr^{-1})		CAR (g C m^{-2} yr^{-1})	
	Full sequence	20th century	Full sequence	20th century
Viðarhólmi[a]	1.3 ± 0.4	1.7 ± 0.3	52.3 ± 16.4	68.3 ± 12.5
Chezzetcook[a]	2.2 ± 0.6	2.6 ± 0.7	79.7 ± 24.0	88.7 ± 28.9
Sanborn Cove[a]	1.6 ± 0.3	1.5 ± 0.3	101.9 ± 29.9	86.1 ± 17.5
Barn Island[a]	1.2 ± 0.6	1.6 ± 0.7	47.7 ± 23.1	62.1 ± 26.8
Little Swanport[b]	2.9 ± 1.3	3.2 ± 1.4	136.3 ± 71.9	158.6 ± 68.6
Mokomoko[b]	2.2 ± 0.3	2.2 ± 0.4	100.0 ± 39.5	110.0 ± 37.6
Pounawea[b]	1.9 ± 1.0	2.4 ± 0.6	110.6 ± 88.5	115.4 ± 30.0

A4. Results from the Post-hoc Tukey test showing site differences in mean carbon accumulation rate (g C m^{-2} yr^{-1}). Values are only shown for sites that are significantly different and display the probability of deviation from the null hypothesis (p). Superscripts denote whether the location is found within the Northern[a] or Southern[b] Hemisphere [VID = Viðarhólmi; CZT = Chezzetcook; SAC = Sanborn Cove; BIS = Barn Island; LSP = Little Swanport; MOK = Mokomoko; POU = Pounawea].

	VID[a]	CZT[a]	SAC[a]	BIS[a]	LSP[b]	MOK[b]	POU[b]
VID	-	-	-	-	<0.001	0.04	0.01
CZT	-	-	-	-	<0.001	-	-
SAC	-	-	-	-	<0.001	-	-
BIS	-	-	-	-	<0.001	<0.01	<0.001
LSP	<0.001	<0.001	<0.001	<0.001	-	<0.001	<0.01
MOK	0.04	-	-	<0.01	<0.001	-	-
POU	0.01	-	-	<0.001	<0.01	-	-

Appendix B

Appendix for Chapter 4

B1. The sediment accretion (cm) across two time periods (1986 – 2003 and 2003 – 2019) for relocated marker horizon plots. The 2003 horizon, and sediment accretion value, between 1986 and 2003 for all plots was obtained from Goodman *et al.* (2007). The sediment accretion rate in the more recent time period (2003 – 2019) was calculated based on the depth from the marker horizon to the marsh surface in 2019, and the known 2003 horizon.

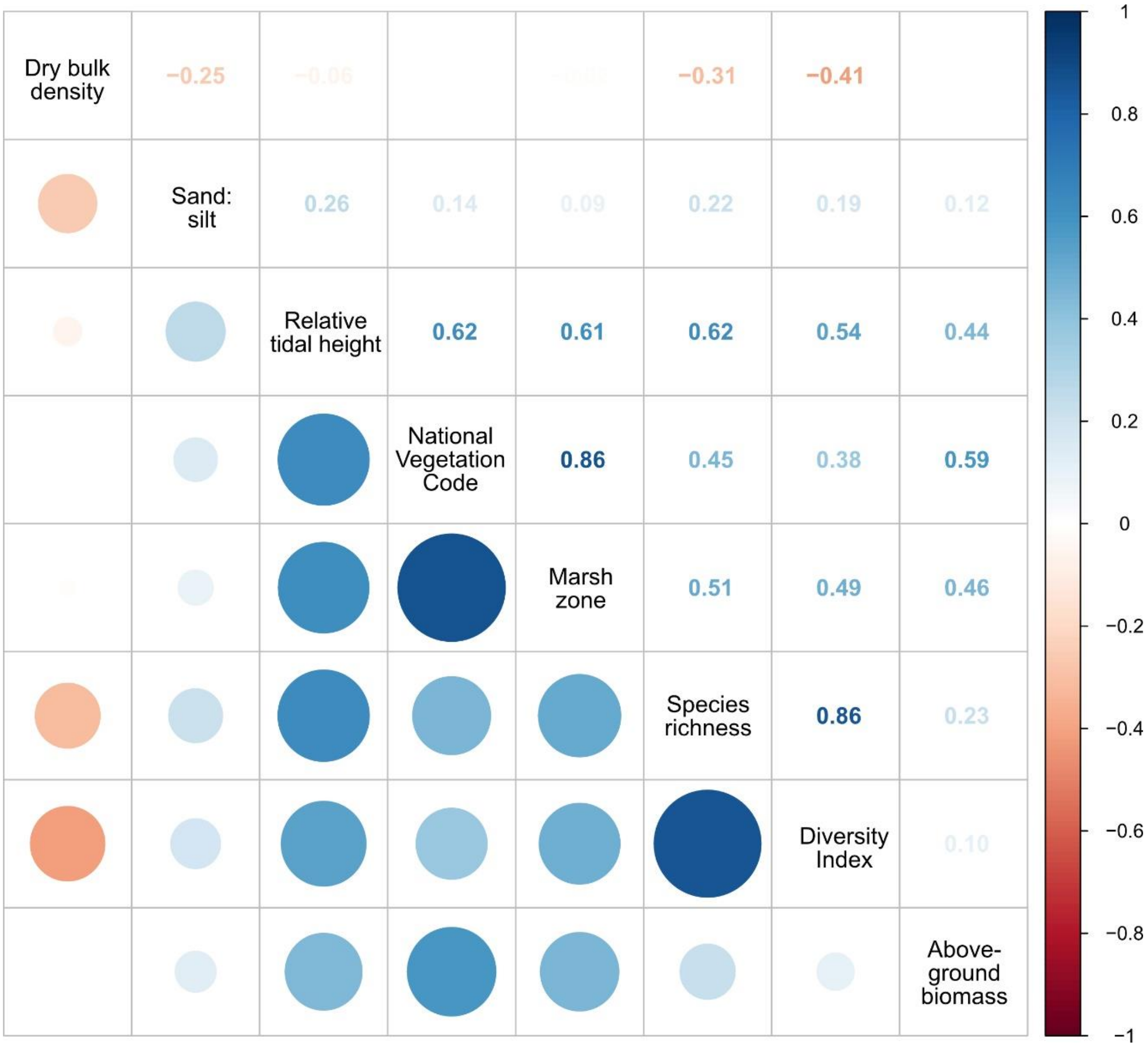
Site	**Plot**	**Sediment accretion (cm)**	
Relocated		1986 – 2003	2003 – 2019
Scarborough Central	1	4.6	5.9
Scarborough Central	2	3.6	4.1
Cobscook Bay	3	7.3	5.3

B2. The sediment accretion (cm) across two time unknown time periods representing slower and more rapid rates of relative sea-level rise (RSLR) for extrapolated marker horizon plots. Sediment accretion for the time period representing slower rates of RSLR were obtained by matching plots in the same marsh zone and site with marker horizon plots that were relocated in 2003 by Goodman *et al.* (2007). Although there is uncertainty, this sub-sample of the core likely overlaps some of the time period between 1986 – 2003. A constant sedimentation rate was then used to calculate sediment accretion in the more recent time period. This is slightly lower for all extrapolated sites due to the difference in timescales: 17 years (1986 – 2003) and 16 years (2003 – 2019). Sediment accretion rates in the time period represented by faster rates of RSLR do not accurately reflect saltmarsh response to enhanced rates of RSLR.

Site	**Plot**	**Sediment accretion (cm)**	
Interpolated		slower RSLR	faster RSLR
Saltmarsh Cove	2	8.0	7.5
Saltmarsh Cove	3	7.4	7.0
Saltmarsh Cove	4	4.0	3.8
Salt Bay	1	6.1	5.8
Salt Bay	2	3.8	3.5
Salt Bay	4	8.0	7.5
Salt Bay	5	7.8	7.3
Lowes Cove	1	2.5	2.4
Hay Creek	1	7.2	6.8
Jasper Beach	1	3.2	3.0
Jasper Beach	3	1.5	1.4
Crocker Point	1	1.5	1.4
Crocker Point	2	3.2	3.0
Crocker Point	4	1.9	1.7
Crocker Point	5	2.5	2.3

Appendix C

Appendix for Chapter 5



C1. Correlation plot for all environmental variables displaying Pearson’s correlation value, and direction of relationship (positive = blue; negative = red). The darker the colour shade, and larger the circle, the stronger the correlation. Strongly correlated variables (considered r => 0.6) were not included in models together.

C2. Saltmarsh plant communities categorised under the National Vegetation Classification (NVC) scheme. Communities in bold were recorded in this study.

Community	NVC	Simplified NVC	Marsh zone
Zostera communities	SM1	SM1	Littoral
Ruppia maritima saltmarsh community	SM2	SM2	
Spartina alterniflora saltmarsh	SM5	SM5	**Pioneer**
***Spartina anglica* saltmarsh**	**SM6**	**SM6**	
Annual *Salicornia* saltmarsh	**SM8**	**SM8**	
Suaeda maritima saltmarsh	SM9	SM9	
Transitional low-marsh vegetation with *Puccinellia maritima*, annual *Salicornia* species and *Suaeda maritima*	**SM10**	**SM10**	**Mid-low**
Coastal stands of rayed *Aster tripolium*	**SM12a**	**SM12**	
Puccinellia maritima* saltmarsh - *Puccinellia maritima	**SM13a**	**SM13**	
Puccinellia maritima saltmarsh - *Glaux maritima*	SM13b		
Puccinellia maritima* saltmarsh - *Limonium vulgare-Armeria maritima	**SM13c**		
Puccinellia maritima saltmarsh - *Plantago maritima-Armeria maritima*	SM13d		
Puccinellia maritima saltmarsh - *Puccinellia maritima*-turf fucoid	SM13e		
Puccinellia maritima saltmarsh - *Puccinellia maritima-Spartina anglica*	SM13f		
Atriplex portulacoides* saltmarsh - *Atriplex portulacoides	**SM14a**	**SM14**	
Atriplex portulacoides* saltmarsh - *Puccinellia maritima	**SM14c**		
Juncus maritimus-Triglochin maritima saltmarsh	SM15	SM15	Mid-upper
Festuca rubra saltmarsh - *Puccinellia maritima*	SM16a	SM16	
Festuca rubra saltmarsh - *Juncus gerardii*	SM16b		
Festuca rubra saltmarsh - *Agrostis stolonifera/Festuca rubra-Glaux* maritima	SM16c		
Festuca rubra saltmarsh - Tall *Festuca rubra*	SM16d		
Festuca rubra saltmarsh - *Leontodon autumnalis*	SM16e		
Festuca rubra saltmarsh - *Carex flacca*	SM16f		
Artemisia maritima saltmarsh	SM17	SM17	
Juncus maritimus saltmarsh community	SM18	SM18	
Blysmus rufus saltmarsh	SM19	SM19	
Eleocharis uniglumis saltmarsh	SM20	SM20	
Suaeda vera-Limonium binervosum saltmarsh community	SM21	SM21	
Atriplex portulacoides-Frankenia laevis saltmarsh community	SM22	SM22	
Spergularia marina-Puccinellia distans saltmarsh	SM23	SM23	
Elytrigia atherica saltmarsh	SM24	SM24	
Suaeda vera saltmarsh	SM25	SM25	
Inula crithmoides stands	SM26	SM26	
Ephemeral saltmarsh vegetation with *Sagina maritima*	SM27	SM27	
Elytrigia repens	SM28	SM28	

C3 (a - h). ANOVA and Kruskal-Wallis statistical outcomes for site differences in environmental variables, including post-hoc results (*n.s.* denotes a non-significant p-value and therefore no significant difference between sites). Variable means can be found in Table 5.3 (main text).

a. One-way ANOVA and post-hoc Tukey test results (p-value) for Dry bulk density.

Dry bulk density was significantly different between sites (F = 9.082; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	0.000064	*n.s.*	0.000056	*n.s.*	*n.s.*
NI_MR	0.000064	-	0.0018	*n.s.*	0.0054	*n.s.*
NI_N	*n.s.*	0.0018	-	0.0023	*n.s.*	*n.s.*
O_MR	0.000056	*n.s.*	0.0023	-	0.0069	*n.s.*
T_N	*n.s.*	0.0054	*n.s.*	0.0069	-	*n.s.*
T_MR	*n.s.*	*n.s.*	*n.s.*	*n.s.*	*n.s.*	-

b. One-way ANOVA and post-hoc Tukey test results (p-value) for silt: sand fraction.

Silt: sand fraction was significantly different between sites (F = 7.583; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	0.035	0.017	0.0000008	0.015	0.015
NI_MR	0.035	-	*n.s.*	*n.s.*	*n.s.*	*n.s.*
NI_N	0.017	*n.s.*	-	*n.s.*	*n.s.*	*n.s.*
O_MR	0.0000008	*n.s.*	*n.s.*	-	*n.s.*	*n.s.*
T_N	0.015	*n.s.*	*n.s.*	*n.s.*	-	*n.s.*
T_MR	0.015	*n.s.*	*n.s.*	*n.s.*	*n.s.*	-

c. One-way ANOVA and post-hoc Tukey test results (p-value) for Relative tidal height.

Relative tidal height was significantly different between sites (F = 24.01; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	0.000057	0.0131	0.0018	0.012	0
NI_MR	0.0000570	-	*n.s.*	*n.s.*	*n.s.*	0.000050
NI_N	0.013	*n.s.*	-	*n.s.*	*n.s.*	0.0000001
O_MR	0.0018	*n.s.*	*n.s.*	-	*n.s.*	0
T_N	0.012	*n.s.*	*n.s.*	*n.s.*	-	0
T_MR	0	0.0000500	0.0000001	0	0	-

d. Kruskal-Wallis and post-hoc Dunn test results (p-value) for Marsh zone.

Marsh zone was significantly different between sites (Chi squared = 42.872; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	*n.s.*	*n.s.*	*n.s.*	*n.s.*	0.0000031
NI_MR	*n.s.*	-	*n.s.*	*n.s.*	*n.s.*	0.0000062
NI_N	*n.s.*	*n.s.*	-	*n.s.*	*n.s.*	0.000097
O_MR	*n.s.*	*n.s.*	*n.s.*	-	*n.s.*	0.0022
T_N	*n.s.*	*n.s.*	*n.s.*	*n.s.*	-	0.00000092
T_MR	0.0000031	0.0000062	0.000097	0.0022	0.00000092	-

e. Kruskal-Wallis and post-hoc Dunn test results (p-value) for NVC community.

NVC community was significantly different between sites (Chi squared = 37.915; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	*n.s.*	*n.s.*	*n.s.*	*n.s.*	0.00022
NI_MR	*n.s.*	-	*n.s.*	*n.s.*	*n.s.*	0.000078
NI_N	*n.s.*	*n.s.*	-	*n.s.*	*n.s.*	0.00014
O_MR	*n.s.*	*n.s.*	*n.s.*	-	*n.s.*	*n.s.*
T_N	*n.s.*	*n.s.*	*n.s.*	*n.s.*	-	0.000070
T_MR	0.00022	0.000078	0.00014	*n.s.*	0.000070	-

f. One-way ANOVA and post-hoc Tukey results (p-value) for Vegetation diversity index.

Vegetation diversity index was significantly different between sites (F = 10.08; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	0.019	*n.s.*	0.032	*n.s.*	0.0000019
NI_MR	0.019	-	*n.s.*	*n.s.*	*n.s.*	*n.s.*
NI_N	*n.s.*	*n.s.*	-	*n.s.*	*n.s.*	0.00026
O_MR	0.032	*n.s.*	*n.s.*	-	*n.s.*	0.03
T_N	*n.s.*	*n.s.*	*n.s.*	*n.s.*	-	0.0000072
T_MR	0.0000019	*n.s.*	0.00026	0.03	0.0000072	-

g. Kruskal-Wallis and post-hoc Dunn results (p-value) for Species richness.

Species richness was significantly different between sites (Chi squared = 33.948; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	*n.s*	*n.s.*	*n.s*	*n.s.*	0.0000050
NI_MR	*n.s*	-	*n.s.*	*n.s.*	*n.s.*	*n.s.*
NI_N	*n.s.*	*n.s.*	-	*n.s.*	*n.s.*	0.000079
O_MR	*n.s*	*n.s.*	*n.s.*	-	*n.s.*	*n.s*
T_N	*n.s.*	*n.s.*	*n.s.*	*n.s.*	-	0.0021
T_MR	0.0000050	*n.s.*	0.000079	*n.s*	0.0021	-

h. One-way ANOVA and post-hoc Tukey results (p-value) for Above-ground biomass.

Above-ground biomass was significantly different between sites (F = 7.161; p < 0.001)						
	O_N	**NI_MR**	**NI_N**	**O_MR**	**T_N**	**T_MR**
O_N	-	*n.s.*	*n.s.*	*n.s.*	*n.s.*	0.000053
NI_MR	*n.s.*	-	*n.s.*	*n.s.*	*n.s.*	0.027
NI_N	*n.s.*	*n.s.*	-	*n.s.*	*n.s.*	0.000086
O_MR	*n.s.*	*n.s.*	*n.s.*	-	*n.s.*	0.0035
T_N	*n.s.*	*n.s.*	*n.s.*	*n.s.*	-	0.00064
T_MR	0.000053	0.027	0.000086	0.0035	0.00064	-

Appendix D

Appendix for Chapter 6

D1. How often, on average, respondents have visited the following marine environments in the last 12 months. Responses are from those who reported they had visited the marine environment in the last 12 months (n = 6519). Note the low visitation rates at seagrasses, saltmarshes, and mudflats.

	More than once per day %	**Everyday %**	**Several times a week %**	**Once a week %**	**Once or twice a month %**
Sandy beaches	1	2	5	6	15
Sand dunes	1	1	2	3	9
Shingle/stony beaches	1	2	4	4	11
Rocky shores	1	2	3	3	11
Coastal cliffs	0	1	3	3	10
Saltmarshes	0	1	1	1	3
Mudflats	0	1	1	1	3
Coastal/seaside towns	3	9	5	5	14
Open sea	1	2	2	2	5
Seagrass meadows	0	1	1	1	2

	Once every 2 – 3 months %	**Once or twice in the last 12 months %**	**Not in the last 12 months %**	**Never %**	**Don't know %**
Sandy beaches	19	41	8	2	0
Sand dunes	12	31	30	11	1
Shingle/stony beaches	15	36	21	6	0
Rocky shores	14	34	24	8	1
Coastal cliffs	13	34	26	9	0
Saltmarshes	4	11	37	38	3
Mudflats	5	11	37	38	2
Coastal/seaside towns	20	38	4	1	0
Open sea	7	19	42	20	1
Seagrass meadows	2	4	30	54	4

D2. Respondents level of agreement with a range of statements related to climate change and nature (n = 6,411).

	Climate change is not the greatest threat to our natural world	Climate action is urgently needed	Nature can help to reduce climate change impacts
	%	%	%
Strongly disagree	30	1	1
Disagree	39	1	4
Neither agree nor disagree	14	4	18
Agree	10	27	51
Strongly agree	6	67	23
Don’t know	1	0	3

D3. Main effects of significant variables retained in the optimal model (Ordinal regression) for the response variable ‘Please indicate how familiar you are with the term blue carbon’ (n = 12,166). Statistic is chi-squared value.

Variable	Statistic	Df	p-value
Age	204.70	6	< 2.2e-16 (***)
Gender	74.66	1	< 2.2e-16 (***)
Annual Household Income	137.96	9	< 2.2e-16 (***)
Multiple Deprivation Decile	44.37	9	1.20e-06 (***)
Connection to marine industry	117.83	1	< 2.2e-16 (***)
Perceived importance of marine environment	157.33	4	< 2.2e-16 (***)

D4. Categorisation of responses to the question 'what image, phrase or word comes to mind when you think of the term blue carbon' (n = 2,892) following qualitative content analysis. Unsure responses were removed prior to analysis. Categories are shown in decreasing order of the percentage of responses that fall under each category. Sub-categories are also shown within brackets.

Category	%	Examples
Chemistry and atmospheric conditions	1.8	"carbon", "emission", "gases", "CO2", "water quality", "organic", "chemistry"
Worry/ concern	9.1	"doomed", "concern", "very bad", "dangerous", "worrying", "bad for the environment"
Pollution	8.7	"underwater pollution", "plastic pollution", "toxic pollution", "harmful chemical in the sea"
Ocean and coastal carbon storage	7.8	"carbon locked in the sea", "how sea collects carbon from the atmosphere"
(carbon capture and storage by coastal wetlands)	(0.4)	("carbon absorbed by seagrass", "carbon sequestration by the world's oceanic and coastal ecosystems, notably coastal wetlands")
Sustainability	7.5	"important for our future", "sustainable", "eco-friendly", "net zero"
(carbon footprint)	(1.9)	("footprint", "carbon footprint")
Hopeful/ tranquil	6.9	"good for the planet", "lovely", "calming", "hint of hope", "exciting", "motivated"
Other	6.7	"paper", "bubbles", "cartoon", "material"
Ocean and/ or coastal	6.6	"ocean", "sea", "whales", "fish", "sealife"
Conservation and environment	6.2	"conservation", "marine conservation society campaign", "saving the sea", "environment", "habitat conservation"
Other marine habitat	6.1	"coral", "algae", "seaweed", "kelp forests", "reef"
Colourful	5.4	"blue", "the blue green colour of the ocean", "colours"
Carbon sink and storage unrelated to the marine environment	4.4	"carbon sink", "carbon capture", "carbon sequestration", "absorbed carbon"
Water	2.5	"water body", "water", "something to do with water"
Energy	2.4	"energy efficient", "using the sea for power", "alternative energy"
(natural/ clean)	0.6	("clean", "natural", "fresh")
(unnatural/ unclean)	0.5	("must be cleaned", "dirty", "doesn't sound like wildlife or natural")
Cars/ engineering	2.1	"electricity", "rolls royce engine", "cars", "fibre wrap"
Climate change	2.0	"climate change", "global warming", "ocean acidification"
Rocks and minerals	1.7	"rocks", "minerals", "diamond", "fossils"

Coastal wetlands	1.2	“mangroves”, “seagrass”, “saltmarshes”, “wetlands”
More knowledge needed	0.6	“new concept”, “people need to be more informed about it”, “needs to be better understood”
Neutral	0.3	“neutral”, “okay”, “fine”

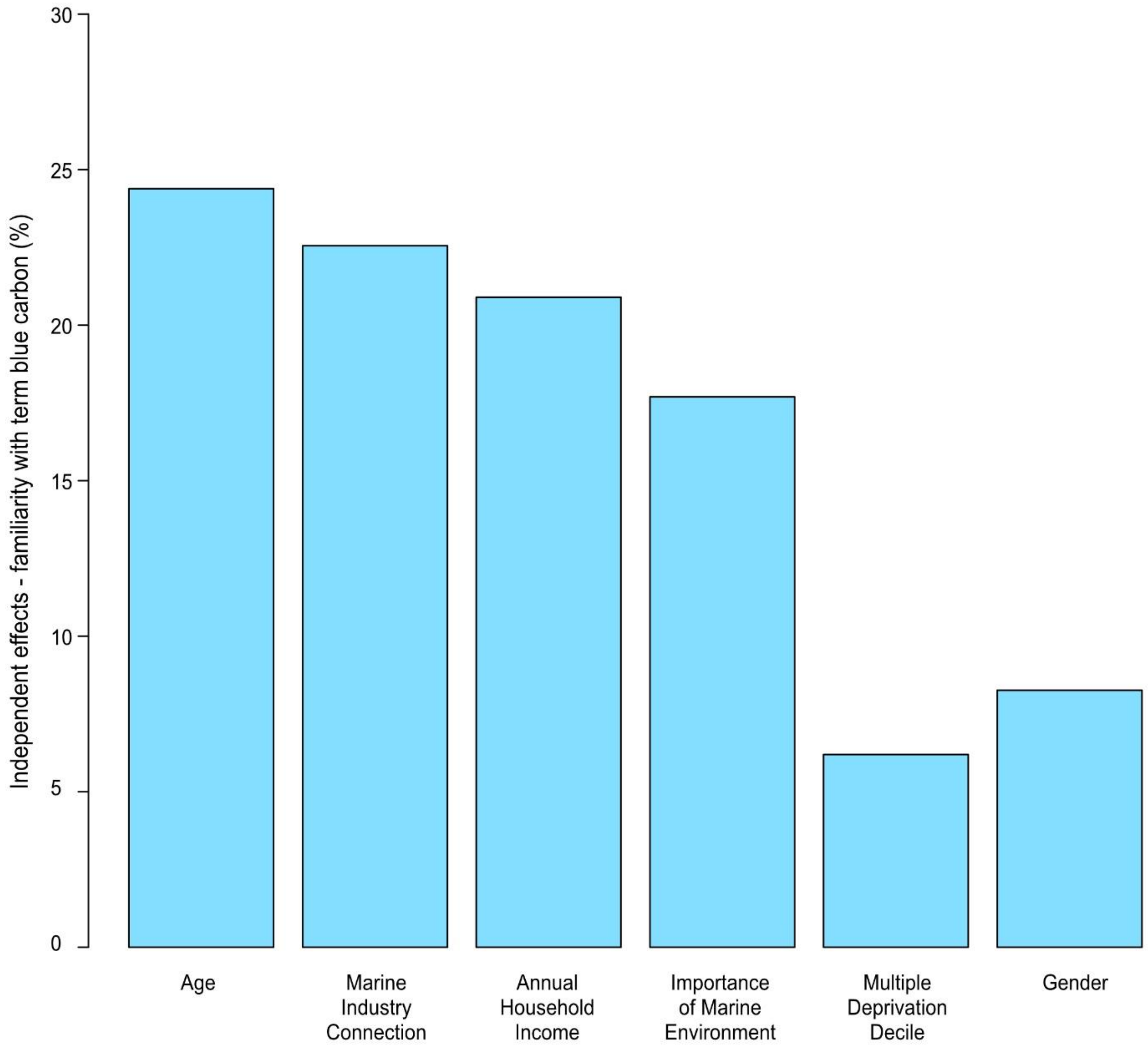
D5. Main effects of significant variables retained in the optimal model (binomial GLM) for the response variable ‘Are you aware of restoration efforts?’ for seagrass meadows (A) and saltmarshes (B) (n = 12,166).

A) Seagrass meadows

Variable	Deviance	Df	p-value
Country	25.74	2	2.58e-06 (***)
Gender	134.14	1	< 2.2e-16 (***)
Highest qualification	58.69	7	2.75e-10 (***)
Connection to marine industry	123.45	1	< 2.2e-16 (***)
Environmental charity membership	49.08	1	2.45e-12 (***)
Perceived importance of marine environment	177.54	4	< 2.2e-16 (***)

B) Saltmarshes

Variable	Deviance	Df	p-value
Country	49.52	2	1.76e-11 (***)
Age	53.79	6	8.14e-10 (***)
Gender	119.22	1	< 2.2e-16 (***)
Highest qualification	81.81	7	5.88e-15 (***)
Connection to marine industry	108.59	1	< 2.2e-16 (***)
Environmental charity membership	79.23	1	< 2.2e-16 (***)
Perceived importance of marine environment	155.62	4	< 2.2e-16 (***)



D6. Independent effects (%) each sociodemographic variable contributed to explaining the optimal model (response variable: familiarity with the term blue carbon). The optimal model only retained variables that significantly influenced respondents' familiarity with the term blue carbon. Variables that explained < 5 % independent effects were not included. Independent effects summed across variables for each model will equal 100 %. Results are from hierarchical partitioning ("hier.part" package).

D7. The rest of this appendix displays the relevant questions used in this study from Defra's wider ocean literacy questionnaire (for full questionnaire, see Defra, 2022b). I contributed to the design of particular questions across all survey sections, however I conceptualised and led the design of Section 4: Marine Environment Habitats. Each section starts on a new page.

SCREENING SECTION

ASK ALL

S3. Can you put your age into one of the following age bands?

SINGLE CODE

Please choose one age band only

1) 16 to 24

2) 25 to 34

3) 35 to 44

4) 45 to 54

5) 55 to 64

6) 65 to 74

7) 75+

Prefer not to say

SECTION 1: THE GLOBAL MARINE ENVIRONMENT

The following questions are about your views on the global marine environment. By the marine environment, we are referring to the wider ocean, seas, coasts, and estuaries and are including all aspects including marine wildlife and plants, marine habitats and ecosystems as well as marine heritage.

ASK ALL

NQ1. What are the first images, phrases, or words that come to mind when you think about each of the following marine terms?

1) Saltmarsh
2) Seagrass
3) Blue carbon

OPEN TEXT (Limit of 100 words)

Don't know

ASK ALL

Q7. Please indicate how familiar you are with each of these terms by selecting the relevant box

DYNAMIC GRID STATEMENTS. Single code. ROTATE ORDER
Please select one answer only.

6) Climate change
8) Nature based solutions
14) Carbon sequestration
16) Blue carbon

REPONSE CODES.

1) Know and understand
2) Heard of and have some understanding
3) Heard of but do not understand
4) Have never heard of the term

Don't know

SECTION 2: YOUR NATIONAL MARINE ENVIRONMENT

The following questions all refer to your views on the marine environment in [England; Scotland; Wales]. By the marine environment, we are referring to the wider ocean, seas, coasts, and estuaries and are including all aspects including marine wildlife and plants, marine habitats and ecosystems as well as marine heritage.

ASK ALL

Q8. How important is protecting the marine environment to you personally?

SINGLE CODE

Please select one answer only.

1) Very important
2) Important
3) Neither important nor unimportant
4) Not very important
5) Not at all important

Don't know

SECTION 3: YOU AND THE MARINE ENVIRONMENT

By the marine environment, we are referring to the wider ocean, seas, coasts, and estuaries and are including all aspects including marine wildlife and plants, marine habitats and ecosystems as well as marine heritage.

ASK ALL

Q15. Which of the following activities, if any, have you done to protect the marine environment in [England; Scotland; Wales]?

MULTICODE. ROTATE ORDER

Please select all that apply

11) I have subscribed to an environmental organisation (e.g. RSPB, National Trust)

If yes, which <u>environmental organisations have you subscribed to</u>?
………………………………………………..

OPEN TEXT (Limit to 50 words)

The next question is about time you spend at the coast (e.g. beaches, cliffs) and in open sea (e.g. swimming, sailing). By the marine environment, we are referring to the wider ocean, seas, coasts, and estuaries and are including all aspects including marine wildlife and plants, marine habitats and ecosystems as well as marine heritage.

Do include:
- visits of any duration (including short trips to the beach, dog walking, etc.)

However, do not include:
- time outside in the marine environment as part of your job
- time spent outside of [Wales; Scotland; England]

ASK ALL

Q25. Thinking about the last 12 months have you made any visits to a marine environment?

SINGLE CODE
Please select one option that best applies.

1) Yes
2) I have not made any visits in the last 12 months
3) I have never visited the marine environment

Don't know

ASK IF Q25 =1

Q26. Thinking about the last 12 months, how often on average, if at all, have you spent your leisure time in the following marine environments. This does not include indoor locations and places which you visit as part of your job: [More than once per day; every day; several times a week; once a week; once or twice a month; once every 2-3 months; once or twice per year; never]

DYNAMIC GRID SINGLE CODE ROTATE ORDER

1) Sandy beaches
2) Sand dunes
3) Shingle/stony beaches
4) Rocky shores
5) Coastal cliffs
6) Saltmarshes
7) Mudflats
8) Coastal / seaside towns
9) Open sea
10) Seagrass meadows

RESPONSE CODES

1) More than once per day
2) Every day
3) Several times a week
4) Once a week
5) Once or twice a month
6) Once every 2-3 months
7) Once or twice in the last 12 months
8) Not in the last 12 months
9) Never

Don’t know

SECTION 4: MARINE ENVIRONMENT HABITATS

The following series of questions focus in greater detail on marine habitats and the potential benefits they can provide.

ASK ALL

NQ5. The following are a range of POTENTIAL BENEFITS of saltmarshes and seagrass meadows. In your opinion, do you think the following benefits are provided by each habitat?

DYNAMIC GRID. MULTICODE. ROTATE ORDER

1) Saltmarshes
2) Seagrass meadows

DYNAMIC GRID. MULTICODE. ROTATE ORDER

1) Carbon capture and storage
2) Recreation
3) Mental health and wellbeing support
4) Diverse habitats for wildlife
5) Natural forms of coastal protection
6) Food e.g. fisheries
7) Pollution control and water purification

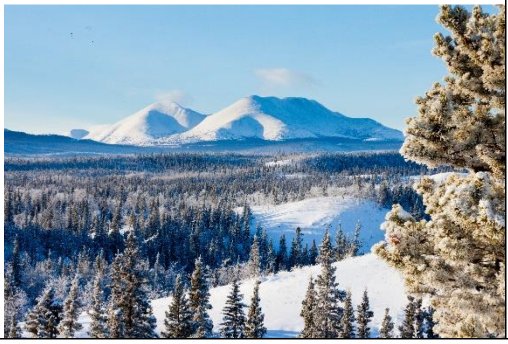
RESPONSE CODES

1) Yes
2) No

Don't know

NQ6. Below is a description and image of a range of natural habitats. Thinking about carbon capture and storage, in your opinion which of the following habitats provide this as a benefit?

MULTI CODE. ROTATE ORDER.

1. Boreal forest		Coniferous forests of pine, spruce and larch trees. Also known as snow forests or taiga. [ID: 167453798]

2. Mangroves		Salt-tolerant trees and shrubs that live along rivers, shores and estuaries. [ID: 1602988900]
3. Tropical rainforest		Forests that are found in the tropics in areas of high rainfall around the equator. [ID: 473899657]
4. Peatland		Wetland ecosystems, sometimes known as bogs, fens, and mires, found inland. [ID: 450567865]
5. Saltmarsh		Salt-tolerant plants such as herbs, grasses, or low shrubs found between land and sea. [Author's own image: Lucy McMahon]
6. Temperate forest		A mix of deciduous, broadleaved and evergreen trees found in temperate regions. [ID: 439886914]
7. Grassland		Large, open areas where the vegetation is mostly dominated by grass. Also known as savanna, prairies, steppes or pampas. [ID: 1088983649]

8. Seagrass		The only flowering plants able to live fully submerged in sea water. Found between land and sea, and under the sea. [ID: 2006114480]

RESPONSE CODES

1) Yes
2) No

Don't know

ASK ALL

NQ7. Which of the following habitats do you believe are found in the U.K.?

MULTICODE. ROTATE ORDER.

Please select all that apply.

1) Boreal forest
2) Mangroves
3) Tropical rainforest
4) Peatland
5) Saltmarsh
6) Temperate forest
7) Grassland
8) Seagrass

None of the above

Don't Know

ASK ALL

NQ8. Are you aware of any efforts to restore the following coastal habitats?

DYNAMIC GRID. SINGLE CODE. ROTATE ORDER.

1) Saltmarshes
2) Seagrass meadows

RESPONSE CODES

1) Yes
2) No

Don't know

ASK IF NQ8 RESPONSE = 1

NQ9. Where have you seen efforts to restore [saltmarsh or seagrass]? Provide examples.

Open text (Limit of 100 words)

ASK ALL

NQ10. The following are a range of potential reasons for restoring saltmarshes and seagrass meadows. In your opinion for each habitat, do you think the following benefits are important reasons to restore these habitats?

DYNAMIC GRID. MULTICODE. ROTATE ORDER

1) Saltmarshes
2) Seagrass meadows

DYNAMIC GRID. MULTICODE. ROTATE ORDER

1) Carbon capture and storage
2) Recreation
3) Mental health and wellbeing support
4) Diverse habitats for wildlife
5) Natural forms of coastal protection
6) Food e.g. fisheries
7) Pollution control and water purification

RESPONSE CODES

1) Yes
2) No

Don't know

ASK ALL

NQ11. The following are statements about saltmarshes. Please indicate to what extent you agree with each statement.

DYNAMIC GRID. SINGLE CODE. ROTATE STATEMENTS.

1) Saltmarshes are an underappreciated/ undervalued habitat
2) Saltmarshes are of little benefit to people.
3) Saltmarshes in the UK are effectively managed.
4) Saltmarshes are classified as a priority habitat under the UK Biodiversity Action Plan.
5) A healthy saltmarsh will provide more benefits to people than a damaged saltmarsh.
6) Saltmarshes protect coastlines from sea-level rise and storms.
7) Saltmarshes play no real role in carbon capture and storage.
8) Saltmarsh is the least interesting marine habitat.
9) Saltmarshes are muddy, dirty environments.
10) Saltmarsh habitat disrupts and makes the beach less inviting.

RESPONSE CODES.

Please select one answer only.

1) Strongly agree
2) Agree
3) Neither agree nor disagree
4) Disagree
5) Strongly disagree

Don't Know

ASK ALL

NQ12. The following are statements about seagrass meadows. Please indicate to what extent you agree with each statement.

DYNAMIC GRID. SINGLE CODE. ROTATE STATEMENTS.

1) Seagrass is an underappreciated/ undervalued habitat.
2) Seagrass habitat is of little benefit to people.
3) Seagrass habitats in the UK are effectively managed.
4) Seagrass is classified as a priority habitat under the UK Biodiversity Action Plan.
5) A healthy seagrass habitat will provide more benefits to people than a damaged seagrass habitat.
6) Seagrass protects coastlines from sea-level rise and storms.
7) Seagrass plays no real role in carbon capture and storage.
8) Seagrass is the least interesting marine habitat.
9) Seagrass washed up on the beach is unattractive and smells bad.
10) It is less attractive to swim in areas where seagrass grow.
11) Seagrass washed up on the beach is unattractive and smells bad.

RESPONSE CODES.

Please select one answer only.

1) Strongly agree
2) Agree
3) Neither agree nor disagree
4) Disagree
5) Strongly disagree

Don’t Know

SECTION 5: ABOUT YOU

This data will help us to understand diverse views and experiences with the marine environment in England, Scotland and Wales. It will give us valuable insight into how we can work to improve access and raise awareness of issues. You may find some of the questions sensitive, in which case you are free not to answer them. All information collected through this survey will be held securely and treated in the strictest confidence. No identifiable information will be shared with Defra, The Scottish Government, The Welsh Government, or anyone else. Results will be reported at an overall level so individual responses will not be identifiable.

ASK ALL

D2. What is your sex?

SINGLE CODE

Please select one answer only
Note: A question about gender will follow

1) Male
2) Female
3) Other (please specify)

Prefer not to say

ASK ALL

D3. Is the gender you identify with the same as your sex registered at birth?

SINGLE CODE

Please select one answer only
Note: This question is optional

1) Yes
2) No

Prefer not to say

ASK ALL

D4. Which one of the following best describes your ethnic group or background?

SINGLE CODE

Please select one answer only

White

1) [English/Welsh/Scottish/Northern Irish/British; Welsh/English/Scottish/Northern Irish/British]
2) Irish
3) Gypsy or Irish Traveller
4) Any other White background, please describe

Mixed/Multiple ethnic groups

5) White and Black Caribbean
6) White and Black African
7) White and Asian
8) Any other Mixed/Multiple ethnic backgrounds, please describe

Asian/Asian British

9) Indian
10) Pakistani
11) Bangladeshi
12) Chinese
13) Any other Asian background, please describe
Black / African / Caribbean / Black British
14) African
15) Caribbean
16) Any other Black/African/Caribbean background, please describe

Other ethnic group

17) Arab
18) Any other ethnic group, please describe

Prefer not to say

ASK ALL

D6. Which of the following best describes your total annual household income before tax?

SINGLE CODE

Please select one answer only

1) £0–15,000
2) £15,001–20,000
3) £20,001–30,000
4) £30,001–40,000
5) £40,001–50,000
6) £50,001–60,000
7) £60,001–80,000
8) £80,001–100,000
9) £100,001-£150,000
10) £150,001+

Prefer not to say

ASK ALL

D7. What is your highest level of qualification?

SINGLE CODE

Please select one answer only

1) PhD/Doctor
2) Masters
3) Bachelor’s Degree or equivalent (such as a NVQ level 5)
4) Higher education (such as a HND or a NVQ level 4)
5) A level or equivalent (such as a Welsh Baccalaureate, Scottish Highers, NVQ level 3, BTEC National)
6) GCSE or equivalent (such as O Level, NVQ level 2, BTEC First or an RSA Diploma)
7) No qualifications
8) Other qualifications (please specify)

Prefer not to say

ASK ALL

D8. Are you?

SINGLE CODE

Please select one option that best applies.

1) In full-time employment (31+ hours per week)
2) In part-time employment (Up to 30 hours per week)
3) Self-employed
4) Unemployed – less than 12 months
5) Unemployed (long term) – more than 12 months
6) Not working – retired
7) Not working – looking after house/children/other caring responsibilities
8) Not working – long term sick or disabled
9) Student – in full-time education
10) Student – in part-time education

Prefer not to say

ASK ALL

D9. Are you or any of your immediate family employed in one of the following industries in [England; Scotland; Wales]?

MULTICODE

Please select all that apply

1) Fishing and aquaculture (including processing etc.)
2) Oil & gas
3) Offshore wind or renewable energy
4) Ports and shipping
5) Marine recreation and tourism e.g. scuba diving, cruise tourism, recreational boating
6) Marine conservation
7) Marine research
8) Extraction of marine aggregates
9) Sub-marine cabling and other infrastructure
10) Marine policy making
11) Marine planning
12) Marine management
13) Royal Navy / Royal Marines
14) Merchant Navy
15) Maritime heritage
16) No

Prefer not to say

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