Variability of Blue Carbon Stocks in Restored Saltmarsh Wetlands
Bay of Plenty, Aotearoa New Zealand

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Abstract

As anthropogenic pressures including urbanisation and climate change continue to pressure the natural environment, there is a great need to focus on solutions that both increase ecosystem resilience and mitigate climate change. Coastal wetlands are a “powerhouse” environment that have the potential to achieve this and much more. Among the variety of coastal wetlands, saltmarsh habitats are one of the most degraded environments globally, with a mere <10% estimated to remain in Aotearoa New Zealand. Many saltmarshes have been lost due to factors such as, conversion to pasture, and/or environmental degradation. As part of the conversation around saltmarsh restoration, nature-based solutions are a tool that is being increasingly considered when it comes to climate change mitigation and adaptation, due to the large potential saltmarshes possess to sequester CO_2 alongside adapting to sea level rise such as by providing a space for flood water and for vegetation to slow down storm surges, alongside other environmental and societal benefits.

This study investigates the blue carbon dynamics in restored and control saltmarsh habitats in the Bay of Plenty Aotearoa New Zealand, focusing on bulk density, concentration of carbon (%), Total Organic Carbon (TOC), Sediment Accumulation Rates (SAR), and Carbon Accumulation Rates (CAR). We focus on changes in blue carbon with time since saltmarsh restoration. Bulk density analysis revealed minimal variability among sites, with Matua Saltmarsh exhibiting the largest standard error (0.765 ± 0.057g cm^3) and Wainui River Saltmarsh the lowest mean bulk density (0.633 ± 0.032g cm^3). Carbon concentration was generally highly variable across sites, with Matua Saltmarsh exhibiting the highest mean concentration (5.83 ± 1.151%) and Wainui Repo Whenua the lowest (2.91 ± 0.26%). Total organic carbon stock decreased with depth across sites, with Matua Saltmarsh having the highest total organic carbon stock (1.21 ± 0.005g cm^3 or

121.42 ± 0.534t ha^{-1}) and Te Awa o Ngātoroirangi the lowest (0.69 ± 0.004g cm^3 or 68.66 ± 0.401t ha^{-1}). Carbon accumulation rates generally increased with restoration age, with Wainui River Saltmarsh exhibiting the highest sediment accumulation rate (11.57mm yr^{-1}) and Matua Saltmarsh the highest carbon accumulation rate (161.87g m^{-2} yr^{-1} over the last 20 years). While no distinct trends in blue carbon stocks and concentration emerged between the restored and control environments, it is notable that restored saltmarshes consistently exhibited substantial carbon stocks compared to controls and global standards. Moreover, there were higher mean TOC figures within the first ~30 years post-restoration. These results align with expectations from previous studies, emphasizing the efficacy of saltmarsh restoration efforts in enhancing future carbon storage potential in Aotearoa.

These findings underscore the need for comprehensive investigations into the roles of vegetation, sedimentation rates, microbial processes, alongside catchment hydrological conditions history. This thesis also highlights that long-term monitoring beyond the initial 20-30 years post-restoration is crucial to comprehensively understand the trajectory of blue carbon dynamics. In addition, comparative studies accounting for diverse restoration techniques and assessments of broader ecosystem services delivered by saltmarsh restoration are essential. Addressing these factors will deepen our understanding of the specific mechanisms influencing blue carbon dynamics in Aotearoa's restored saltmarshes.

Dedication

"Nature is the source of all true knowledge" – Leonardo da Vinci. To my ancestors but most specifically my Ouma, Nelly, who truly knew what it meant to have nothing, her deep appreciation and love for her garden fostered a love for nature I didn't even realise I possessed.

This thesis also stands as a testament to every child who has been underestimated due to their neurodivergence. I was once told I'd never read. Yet, standing here two decades later, having earned an undergraduate and postgraduate degree, and now completing a thesis in Environmental Science, fills me with immense pride. ADHD isn't a limitation; it represents a distinct perspective, a unique lens of inquiry and curiosity. When harnessed correctly, it fosters profound appreciation and insight into what ever fills you with joy and excitement. This is for the little girl who felt stupid when she couldn't get her b's and d's correct, who always believed she couldn't succeed with anything involving numbers, for the girl who just overall felt emotions on a whole different spectrum. With love, kindness, discipline, and a never dying tenacity, you can do absolutely anything you set your mind to.

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Chapter 1
Introduction

1.1 Problem Statement – Climate Change and Management Strategies

Environmental management has traditionally been convened by command-and-control (hard) hard engineering and predict-and-provide (soft) management techniques. Hard engineering techniques are human-centred solutions that focus on solving environmental issues with convenience, cost, and competence as the guiding values, and often involve significant capital investments and unforeseeable legacy costs (Gregory *et al.*, 2011). On the other hand, soft management techniques also known as nature-based solutions (NBS) or green management solutions, prioritize actions that work synergistically with the natural environment, resulting in as little disruption as possible, with core principles prioritising the inherent natural functions and ecosystem services present within an environment. These solutions are often economically feasible, significantly more integrative in approach, and consider social sustainability (Fenemor *et al.*, 2011). Conventional management techniques using hard engineering are testimony to the disconnected relationship present between humans and the environment, the lack of understanding and appreciation for intricate ecological interdependencies and services, leading to significant losses in biodiversity and ecosystem stability, making NBS a promising management to mitigate and adapt to climate change (Eggermont, 2015).

NBS can be utilised to reduce carbon emissions by implementing management techniques that incorporate ecosystems services such as carbon burial, referring to the process of storing carbon in long-term sinks. Reforestation, wetland restoration, and sediment management are examples of NBS that enhance carbon burial and reduce carbon emissions. The most common carbon burial NBS is agroforestry reforestation through the planting of monocultures such as pines

(*Pinus* spp.*)* which increases carbon sequestration through photosynthesis and stores carbon in the biomass and sediment.

The European Environment Agency conducted a summary of carbon stocks across terrestrial and marine environments and found that forest ecosystems had a range of 30-110t ha^{-1} (tonnes of carbon per hectare), with the median value being 74t ha^{-1}. Wetland sediments were found to have much higher carbon stocks with reed marshes storing 150-330t ha^{-1}, and saltmarshes 200-400t ha^{-1} (Hendriks *et al.* 2020). This highlights the potential of NBS's including restored wetlands for climate change mitigation through the improvement of ecosystems that are known to be major hot spots for carbon sequestration and storage. This thesis aims to fill knowledge gaps regarding the role of restored saltmarsh environments in Aotearoa New Zealand (Aotearoa) as significant reservoirs of carbon stocks since restoration.

Ecological restoration focused on vegetation offers numerous benefits in terms of carbon storage, making it a powerful tool in the fight against climate change. Trees and plants in these restored habitats absorb carbon dioxide during photosynthesis, converting it into biomass, which is stored within their roots, trunks, and leaves. Consequently, carbon is sequestered from the atmosphere, helping to reduce greenhouse gas (GHG) concentrations. Moreover, restoration efforts often focus on improving sediment health and organic matter content, further increasing carbon storage potential. By protecting existing carbon-rich areas, like old-growth forests and peatlands from further degradation, ecological restoration ensures the preservation of vast carbon reservoirs. Additionally, restored ecosystems tend to be more biodiverse and resilient, allowing them to maintain their carbon storage capacities over the long term. Overall, ecological restoration plays a critical role in mitigating climate change by sequestering carbon, promoting sustainable land management, and safeguarding valuable carbon-rich habitats (Simonson, 2021) .

The Anthropocene is an epoch characterised by significant increases in human activity. Anthropogenic activities, such as burning fossil fuels and land use changes, have significantly increased atmospheric GHG concentrations, particularly carbon dioxide (CO_2). Atmospheric CO_2 concentrations have risen from around 280 parts per million (ppm) before the Industrial Revolution (around 1784) to over 419 ppm (National Aeronautics and Space Administration, (NASA), 2023). The increase in atmospheric CO_2 concentrations has contributed to a range of climate changes, including rising global temperatures and more frequent and severe extreme weather events.

As climate change becomes an undeniable reality for all of humanity, urgent action is needed to reduce carbon emissions and transition to a low-carbon economy to mitigate its impacts. The significance of ecological restoration in addressing climate change is increasingly evident, with restoration projects playing a crucial role in enhancing ecosystem resilience, sequestering carbon, and fostering biodiversity (Duarte *et al.*, 2020). Traditional environmental management involves assessing environmental impacts and determining if solutions are needed to avoid, mitigate, remedy, or offset pressures on the system. However, these standards often do not fully capture the complexities of ecosystems and their various services. Thus, there is a need for standards that champion ecological services and environmental success as a core outcome while still achieving cultural, social, and economic prosperity in the long term. Coastal wetlands have been identified as an important habitat for restoration including due to their role in sequestering carbon, although coastal wetlands are heavily degraded, lost, and impacted by anthropogenic affects worldwide.

1.2 Coastal Wetlands

Coastal wetlands are those wetlands located in coastal zones where land meets the sea and are influenced by the ebb and flow of tides. Coastal wetlands can be classified into several types, including saltmarshes, mangrove forests, and seagrass meadows. Coastal wetland ecosystems provide a range of ecological functions and services, including protecting shorelines from erosion, filtering pollutants and nutrients, provide unique habitat for a diverse array of species, and play a crucial role in mitigating the impacts of climate change as an important carbon sink (Gerbeaux & Hume, 2022). Despite their importance, coastal wetlands are under threat from human activities such as coastal development, land reclamation, pollution, and climate change (Barbier, 2019).

This thesis focuses largely on saltmarshes, which are intertidal habitats characterized by the presence of halophytic (salt-tolerant) plants and that are heavily influenced by regular tidal inundation. They play a vital role in coastal protection, acting as a buffer against storm surges and wave energy, while also providing important habitats for a diverse range of plant and animal species, including migratory birds and fish (Barbier, 2019; Ramsar Convention, 2010). Coastal saltmarshes are typically vegetated by grasses, sedges, and other halophytic plants that are adapted to thrive in saline conditions (Johnson & Gerbeaux, 2004; Johnson & Brooke, 1989). Species such as oioi /jointed wire rush (*Apodasmia similis*), sea rush (*Juncus kraussii var australiensis*), wīwī /knobby clubrush (*Ficinia nodosa*), harakeke/flax (*Phormium tenax*), mākaka/ saltmarsh ribbonwood (*Plagianthus divaricatus*) are some of the plants commonly found in Aotearoa’s saltmarshes (Haacks & Thannheiser, 2003).

Saltmarshes and the wider coastal wetland environment have been proven to possess huge economic value in terms of ecological services provided. For example, de Groot *et al.* (2012) reviewed 320 academic publications investigating the value of our 10 major global biomes in terms of ecosystem services (Figure 1.1). A few of the services that were investigated included, climate regulation, waste treatment, nutrient cycling, food, recreation and nursery services, overall, for regulating services coastal wetland environments possess the highest monetary value at USD$171,515 per hectare per year, while temperate forests were only valued at USD$2529 per hectare per year (based off 2007 price levels).

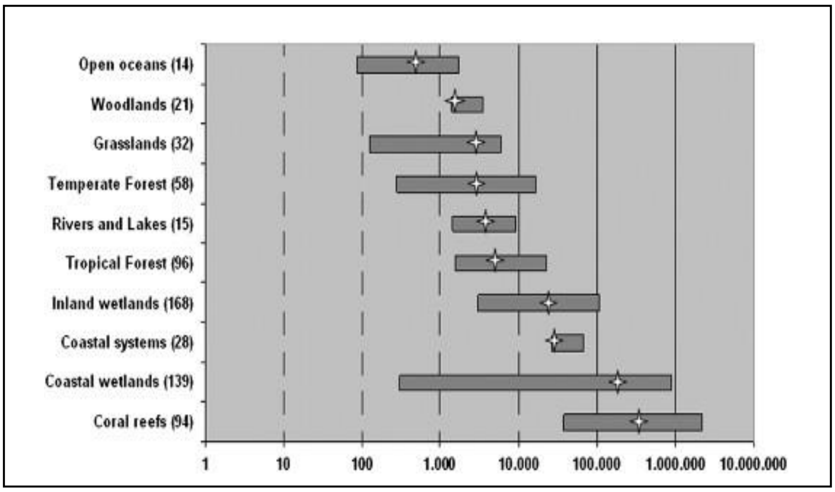

Figure 1.1: Modified from Groot et al. (2012). Range and average of total monetary value of bundle of ecosystem services per biome (in Int. $/ha/yr 2007/PPP-corrected). The total number of values per biome is given between brackets; the average of the value-range.

1.3 Coastal Wetland Loss and Potential for Restoration

Wetland ecosystems have experienced significant declines in extent and health globally, primarily driven by the conversion of land to grasslands for agricultural purposes. In the case of saltmarshes, there is a lack of a comprehensive and accurate global inventory documenting the extent and distribution, making it challenging to estimate their specific decline on a worldwide or national scale (Adam, 2002). However, some global estimates include an overall predicted 50% loss of saltmarshes across globally with other regions experiencing higher loss rates including 65% loss of Atlantic tidal/saltmarshes in Canada, a 75% loss of coastal wetlands in the Swan plain and Coastal New South Wales in Australia, and in Aotearoa an estimated >90% loss (Aber *et al.*, 2012; Barbier *et al.*, 2011; Doney *et al.*, 2012; Mitsch & Gosselink, 2007). Coastal saltmarshes are one of the most impacted environments in Aotearoa, including due to our farming history and that approximately 65% of the population and their associated major infrastructure is located within 5 km of the coast (Reisinger *et al.*, 2015).

The <10% of wetlands that remain are often highly degraded. It is evident that there is huge potential to restore coastal wetlands which typically involves a combination of ecological and hydrological restoration techniques. The specific methods employed can vary depending on the characteristics of the wetland and the goals of the restoration project, where common restoration practices may include.

I. Tidal reconnection by restoring tidal flow to the wetland by removing or modifying barriers such as dikes, culverts, or tide gates. Tidal reconnection aims to restore historic hydrology and helps to reintroduce nutrient-rich water, sediments, and marine organisms into the wetland, promoting its ecological functions (Zedler & Kercher, 2005).

II. Vegetation planting —replanting native vegetation is often a critical component of wetland restoration. Native species can help stabilize the sediment, enhance habitat for wildlife, and contribute to the overall functioning of the wetland ecosystem. Planting can be done using direct seeding, transplants, or a combination of both (Zedler & Callaway, 2002).

III. Sediment addition has been found successful in some cases to restore wetland structure by adding sediment to degraded or subsided saltmarsh areas to help raise the elevation to appropriate levels relative to the tide, and support plant growth. Sediment can be obtained from offshore or upland sources and placed strategically to achieve the desired elevation and hydrological conditions (Morris *et al.*, 2002).

IV. Control of invasive species can be a key tool in wetland restoration. Invasive plant species can disrupt the natural composition and functioning of saltmarsh ecosystems, removal or control of invasive species through manual methods, herbicides, or biological control can be necessary to promote the establishment and growth of native vegetation (Billah *et al.*, 2022).

In addition to the above, two major factors following restoration that affect the ongoing health of a restored saltmarsh are ongoing disturbances and coastal squeeze. The reality with ecological restoration in heavily disturbed environments is that historic states may never be perfectly recreated. However, restoring a fraction of the critical ecosystem services these environments possess is key to mitigating and managing climate change.

1.4 Blue Carbon and Coastal Saltmarshes

Coastal ecosystems such as mangroves, seagrass beds, and saltmarshes have the ability to capture and store significant amounts of CO^2 from the atmosphere through photosynthesis and subsequent sedimentation. Carbon that is stored in these coastal and marine ecosystems is referred to as “blue carbon”. Since then, the term has gained widespread use in scientific literature, conservation efforts, and policy discussions surrounding coastal ecosystem management and climate change mitigation.

There are two ways in which carbon can enter a coastal wetland environment. Allochthonous carbon comes from various external sources such as terrestrial vegetation, leaf litter, woody debris, or dissolved organic matter transported by rivers, streams, or groundwater. This carbon input can be derived from adjacent forests, wetlands, or other land areas surrounding the aquatic ecosystem autochthonous carbon is produced within a specific ecosystem or habitat through primary production, primarily by autotrophic organisms such as plants, algae, and other photosynthetic organisms (Saintilan *et al.*, 2013). It is the carbon that originates within the ecosystem itself, typically through the process of photosynthesis. The balance between autochthonous and allochthonous carbon inputs is essential in understanding blue carbon dynamics and functioning of coastal wetland ecosystems. Autochthonous carbon represents the internal carbon cycling and productivity of the ecosystem, while allochthonous carbon provides external inputs that can influence the ecosystem's structure and functioning (Howard *et al.*, 2014).

Saltmarshes are more heavily influenced by allochthonous sources because organic carbon derived from local sources is likely to be fresher and more easily decomposed compared to material from other locations (Saintilan *et al.*, 2013; Van de Broek *et al.*, 2018). In estuaries, the majority of non-local organic carbon is expected to come from land, which is often either resistant to decomposition or partially already broken down. It is believed that there is typically an 80:20 ratio allochthonous to autochthonous carbon in deeper sediment layers (> 1m), while there is a 50:50 ratio near the surface (Williamson & Gattuso, 2022).

Carbon is distributed within different parts of an estuarine ecosystem through various processes and compartments. In the water column, carbon exists in the form of dissolved inorganic carbon (DIC) and dissolved organic carbon (DOC). DIC primarily originates from atmospheric carbon dioxide dissolution, while DOC consists of organic compounds derived from terrestrial sources, phytoplankton exudates, and microbial activity. Particulate organic carbon (POC) is another pool found in the water column, composed of suspended particles, such as detritus and plankton. Carbon can also be located within the sediment, where it is stored as organic matter. This organic carbon can be derived from autochthonous sources (e.g., primary production within the estuary) or allochthonous sources (e.g., terrestrial plant material entering the estuary through rivers). Sediments also serve as an important burial site for carbon, with some fraction becoming sequestered for long periods. Additionally, estuarine vegetation plays a crucial role in carbon storage. These plants photosynthesize, taking up carbon dioxide from the atmosphere and incorporating it into their tissues. As a result, estuarine ecosystems are dynamic carbon reservoirs, with carbon being located in different compartments, including the water column, sediments, and vegetation, influencing the overall carbon cycling and storage within these vital transitional zones.

In a recent case study by Bulmer *et al.* (2020), organic carbon stocks across coastal wetland habitats in Tairua estuary were investigated. The study found that saltmarshes contained 90 t of carbon ha-^{1}, mangroves 46t ha^{-1} , seagrass 27t ha^{-1}, and unvegetated habitats contained 26t ha^{-1}. Saltmarshes hold an exciting potential for carbon storage compared to other habitats, the results in Bulmer *et al.* (2020) are also comparable to many other studies that similarly found that saltmarshes organic carbon stocks are some of the highest across coastal habitats. The findings in Bulmer *et al.* (2020) are similar to Burden *et al.* (2013; 2019) who found that carbon stocks in natural saltmarshes were as high as 93t ha^{-1}. Natural Research England (2021) undertook a review of carbon storage and sequestration by habitat, surmising and comparing research to date. They found that vegetated undisturbed saltmarshes stored between 40–60t ha^{-1} in *Juncus gerardii* and *J. maritimus* plant communities, while sediment organic carbon stocks were between 29t ha^{-1} in sandy sediment, and 43t ha^{-1} in non-sandy sediments to 10cm depth.

Saltmarshes possess lower carbon turnover rates or rather efficient carbon sequestration rates compared to terrestrial habitats, meaning these environments lock in carbon for longer in comparison (de Groot *et al.*, 2012). This is due to the high levels of productivity within these systems caused by the interactions of flora and fauna which influence rates of photosynthesis and biomass production. The interactions between bioturbators, complex root systems and anoxic sediment layers stabilize organic matter, preventing it from decomposing and releasing carbon into the atmosphere (Tobias & Neubauer, 2019). Carbon found within these waterlogged anoxic layers decomposes at a much slower rate due to the lack of oxygen, and thus is released back into the atmosphere at a much slower rate, often remaining in these sediment layers for centuries. Studies have credited saltmarshes for their natural ability to store carbon with estimates that suggest carbon accumulation in coastal wetlands is 30-50 times higher per unit area compared to forest ecosystems (Bridgham *et al.*, 2006; Mcleod, 2011; Ouyang, 2013).

Restored saltmarsh habitats vary significantly in terms of carbon dynamics compared to stable natural saltmarsh environments, which tend to remain steady in both their rate of carbon sequestration and carbon stocks. Restored saltmarsh environments generally take longer to accumulate carbon compared to undisturbed environments due to the time it takes for initial colonization and establishment of natural flora and fauna and their associated ecosystem services. Restored saltmarsh habitats may have higher carbon stocks compared to natural saltmarshes. This is primarily due to the history of the restored environment and its relative surroundings. For example, the habitat may have previously been grazed agricultural land with high nutrient inputs, or the restored wetland could be surrounded by primarily urban-zoned land with high levels of nutrient-rich runoff (Rogers *et al.*, 2018).

Burden *et al.* (2013) found that saltmarsh habitats between 0-20 years post restoration displayed elevated rates of carbon stocks, however, this was highly dependent on the surrounding environments and their carbon input. The carbon stocks across the restored saltmarshes in question were between 68-75t ha^{-1} with habitats between 16-20 years of restoration possessing the largest carbon stock, while natural saltmarshes had carbon stocks between 41-93t ha^{-1}. Burden *et al.* (2019) theorise that restored saltmarshes require approximately 100 years of recovery to function equivalently to a natural undisturbed saltmarsh. Future blue carbon research and implementation needs to consider the effectiveness and potential limitations of blue carbon in restored environments particularly in terms of the variabilities present in carbon burial rates across habitats and geographical location and the overall errors in determining these rates. The lateral transport of carbon across habitats and in catchments, the dissolution and formation of carbonates, the level of vulnerability imposed by future climate change and non-climatic factors, and the overall cost-effectiveness and scalability in terms of the overall benefits these environments may possess in mitigating climate change (Williamson & Gattuso, 2022).

1.5 Cultural Significance

The tangata whenua (people of the land) of Aotearoa have beliefs that are deeply rooted in kaitiakitanga which can be translated to stewardship or guardianship. It is important to understand that for Māori the land, the sky, the water, the spiritual realm, and living people are interwoven through whakapapa (genealogy) to live a life that is harmonious with the natural world and its rhythms (Hayes, 1998).

Wetlands hold profound significance within Māori culture, serving as cherished sites of heritage and connection. These landscapes, often encompassing areas like Kūkūwai Whakapoukōrero, are vital repositories of cultural knowledge and practices. Beyond their spiritual and historical importance, wetlands are also source of practical resources for tangata whenua such as kai moana (sea food) and kai awa (freshwater food), as well as materials for traditional weaving practices. Wetlands also play a crucial role in mātauranga (knowledge, wisdom, understanding, skill), allowing for the sharing and continuation of knowledge through hands-on engagement. Additionally, wetlands are spaces where the practice of rongoā (traditional Māori herbal medicine) thrives, showcasing the intersection of ecological understanding and cultural wisdom. In this intricate web of heritage and utility, wetlands stand as testament to the interdependence of Māori people and the environment, nurturing traditions, sustenance, and holistic well-being (Cultural Impacts Assessment; Kaituna River Re-Diversion, 2014).

1.6 Research Significance and Study Objectives

Blue carbon research is gaining momentum worldwide as scientists, practitioners, and policymakers strive to harness the potential of this NBS in combating climate change. However, in Aotearoa there is much to be done to develop a nationally focused understanding of the blue carbon already stored in our coastal environments, and the potential to grow blue carbon sequestration. To help understand this, this thesis focuses on assessing blue carbon stocks in restored saltmarsh wetlands at various time-frames post-restoration. We also consider how the blue carbon stocks in these restored systems compare to saltmarsh wetlands in a "natural" state.

The findings from this research can provide valuable insights on the role of restored saltmarsh in blue carbon stocks and help understand how long it takes for their functionality to be restored. It emphasizes the importance of comprehending the functionality of restored saltmarshes, as they not only offer significant cultural services but also contribute to the regeneration of essential environmental services. Understanding opportunities for growth of carbon sequestration in Aotearoa is pivotal in making progress towards achieving our goals for net-zero greenhouse house emissions by 2050 (New Zealand Parliament, 2019).

1.7 Study Sites

Aotearoa is thought to have one of the highest rates and extents of wetland loss compared to global trends (Mitsch & Gosselink, 2000), where approximately 1% of palustrine wetlands remain today, 249,484 ha for freshwater wetlands and 292 ha for inland saline wetlands (Ausseil *et al.*, 2008). Ninety percent of Aotearoa's wetlands have been lost in the last 150 years due to drainage and conversion to agricultural land (Denyer & Peters, 2020). In the Bay of Planty (BOP), the Kaituna, Waihī and Rangitaiki plains previously consisted of 40,000 ha of wetlands; today less than 1% of this area remains intact. The BOP possesses a range of restored coastal wetlands at various stages, particularly at Maketū, where native revegetation planting began in 1991 and has been a common practice in restoration efforts since.

The project design, methods and sampling regime were co-developed with Te Rūnanga o Ngāti Whakaue ki Maketū with Rewi Boy Corbett and the Bay of Plenty Regional Council (BOPRC). This included Rewi sharing his mātauranga into the process for selecting our study sites in Maketū. Our primary requirement was to select saltmarsh wetlands that were at different stages of restoration. Overall, six saltmarsh wetland habitats were selected, three within the wider Tauranga region and three within Maketū (Figure 1.2). The three sites in Maketū were in Te Awa o Ngātoroirangi (37˚45'55"S, 176˚26'37"E), Kūkūwai (which translates to wetland) Whakapoukōrero (37˚45'58"S, 176˚26'47"E) (Figure 1.3), and Te Pā Ika Wetland (37˚45'05"S, 176˚ 25"E) (Figure 1.4). Te Awa o Ngātoroirangi (MK1) is a relatively undisturbed but fragmented saltmarsh environment that displays clear patterns in saltmarsh zonation and is being treated as a control environment in this work. Te Pā Ika Wetland (MK2) has been subject to heavy anthropogenic pressures (Section 2.1.1) and has been restored within the last 5 years from pasture to reclaimed saltmarsh, Kūkūwai Whakapoukōrero (MK3) has been undergoing restoration since the mid 2000's, from pasture to a mixture of brackish and freshwater wetland.

BOPRC was a key stakeholder in this project, through their support and funding three additional sites to Maketū were able to be sampled. Upon consultation with BOPRC, Wainui River Saltmarsh (37˚38'06"S, 175˚57'35"E), Wainui Repo Whenua Saltmarsh (37˚37'56"S, 175˚57'43"E) (Figure 1.5), and Matua saltmarsh (37˚40'17"S, 176˚07'50"E) (Figure 1.6) were selected. Wainui River Saltmarsh (Sargent Drive (SD) - SD4) has had no known changes to land use, Wainui Repo Whenua Saltmarsh (Sargent Drive (SD) - SD5) has recently (2021) been subject to a large saltmarsh reclamation project and was previously grazed pasture, Matua Saltmarsh (MS6) has been moderately affected by a rise of urbanisation in the surrounding area and in the last 20 years there have been subject to various planting projects aiming to increase the functionality and health of the saltmarsh.

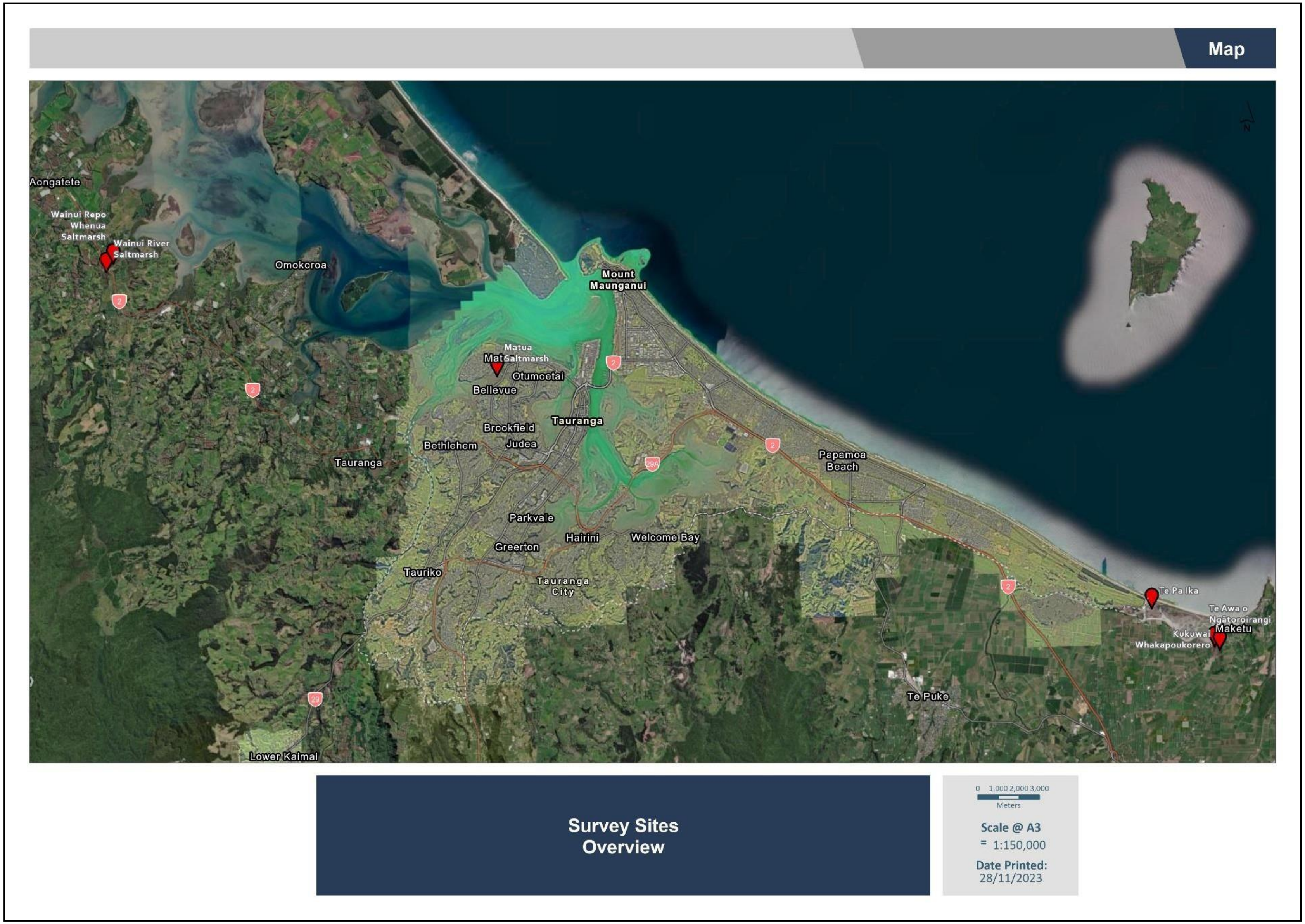

Figure 1.2: Landscape scale map displaying all six selected study sites across Tauranga.

Figure 1.3: Map displaying Te Awa o Ngātoroirangi and Kūkūwai Whakapoukōrero.

Figure 1.4: Map displaying Te Pā Ika Wetland, also visible is the Kaituna flood gate.

Figure 1.5: Map displaying Wainui River Saltmarsh and Wainui Repo Whenua Saltmarsh.

Figure 1.6: Map displaying Matua Saltmarsh.

Chapter 2
Materials and Methods

2.1 Site Selection

Site selection within Maketū was undertaken in collaboration with Te Rūnanga o Ngāti Whakaue. Tangata whenua have been key contributors in the restoration of Maketū wetlands. Rewi Boy Corbett played an essential role with his intergenerational knowledge, wisdom, and deep understanding of how the whenua has been physically transformed over the years. To better understand the whenua, we conducted a reconnaissance hikoi (walk) with Rewi Boy, he shared knowledge and expertise on how Maketū has transformed over his and his father's lifetime to help frame this research and the sampling.

2.1.1 Te Awa o Ngātoroirangi (Maketū Estuary) and the Kaituna River Diversion

The Maketū Estuary once teemed with abundant kai moana (seafood) and kai awa (river food), sustaining tangata whenua and their families for ages. This encompassed a diverse array of marine and freshwater species like kūtai, pāua, koterotero, tuangi, eels, kina, and fish such as snapper, hāpuku, and kahawai. Sharing and preparing kai moana fosters community bonds and passes cultural wisdom through generations. People journeyed to Maketū due to its rich kai (food) specifically kai moana (seafood) abundance, attributed to the environment's quality and shared tikanga (traditions and customs).

At Maketū Estuary's northern coast, the Kaituna River starts, once the "umbilical cord" connecting tribes. However, disruption began in 1926 when dairy, sheep, and fruit farmers pushed for Kaituna River diversion, seeking flood protection for 7,690 ha of low-lying land. Decision-making excluded Ngāti Whakaue, who opposed the 'Kaituna Cut'. In 1956, the river redirected towards Tauranga Harbour, altering this natural union. The diversion profoundly impacted the ecosystem and local well-being, especially Maketū's tangata whenua, with reduced food and waters facing eutrophication (Cultural Impacts Assessment; Kaituna River Re-Diversion, 2014).

This hydrological shift led to a loss of 160 ha of saltmarsh and estuarine benthic decline. In response, the Maketū Action Group formed in 1979. Their 1984 Parliamentary Petition led to Cabinet Paper approval and Restoration Strategy for Maketū and Te Arawa. Governance challenges hindered the project, only restoring 4% of river flow. Numerical modelling in 2001 and 2012 plans aimed to restore Kaituna River-Maketū estuary hydrology, including flood gates, salinity blocks, widened channels, and coastal wetland restoration. Re-diversion gate construction began in 2018, reopening in February 2020, restoring 20% of flows (Bay of Plenty Regional Council Toi Moana, 2023). The Kaituna River re-diversion underscores large-scale waterway alteration consequences, emphasizing ecological, cultural, and social factors in environmental decisions.

2.2 Sampling Design

In collaboration with BOPRC sampling was conducted in SD4, SD5, and MS6 from the 23rd of August until the 26th of August 2022. Field work was conducted across all three of the sites in Maketū occurred between the 7th of September until the 9th of September 2022, each day of field work in Maketū was initiated with a karakia (prayer) and a personal account of the selected site from Rewi Boy Corbett. In recognition of the profound connection between culture, tradition, and the natural surroundings of Maketū, the utmost importance was placed on acknowledging the land, people, and ancestors before commencing any field work, as an embodiment of kaitiakitanga.

2.2.1 Sediment Sampling

Sediment sampling was conducted within 3 to 4 hours either side of low tide, stratified sampling methods were used to select two sites at each of our six chosen saltmarshes, across all sites a lower (L) saltmarsh and mid (M) saltmarsh location was chosen to take a single sediment core sample (12 cores total). To maximise our chances of achieving a comprehensive representation of each environment, sampling in the saltmarsh zones in which carbon is known to best be sequestered was imperative due to our limited number of samples at each site (Elsey-Quirk, *et al.*, 2011). The lower and mid saltmarsh zones were determined by examining physical features within the wetland such as proximity to a water channel and vegetation composition (King *et al.*, 1990; Haacks & Thannheiser, 2003; Johnson & Brooke, 1989), samples sites were also dependent on having enough proximate space to later lay the 20m x 50m vegetation transect. Traditionally, vegetation strata form perpendicular to the dominant water channel, however, when identifying the lower and mid saltmarsh zones at each site a standardised method could not be applied across all sites. Due to our more recently restored saltmarshes displayed homogenous landscape characteristics that lacked zonation at present (MK2, MK3, SD4), determining sample location was established using best judgement of where we imagined zonation will occur in future. A Garmin GPSMAP 65s Multiband GNSS Outdoor Handheld GPS was used to input and store the location of each sample, three recordings were taken at a single site to ensure accuracy.

To collect sediment, a 3.8cm diameter steel piston corer was use with the aim of reaching a depth of 100cm. However, it is uncommon to reach 100cm depth when coring in coastal wetlands, thus three cores were taken at each sample site and the best core was chosen based on length and compaction of sediment. Sediment cores that were not selected were placed directly back into the earth from where they were extracted, this was a method that practiced kaitiakitanga and was suggested by Rewi Boy Corbett. The piston corer required two people to apply weight downwards once the corer felt as if it reached its maximum depth. The corer was removed at an approximate 45˚ angle, once the end of the corer was visible it was covered by hand to prevent sample loss, once aboveground the sediment core sample was transferred into a half-cut PVC pipe and then wrapped with food grade plastic cling wrap and labelled appropriately (Howard *et al.*, 2014) (Bouma, 1978).

The location of sampling and the sediment cores were photographed to aid in data analysis; photographic records of the sediment core samples are useful to visually represent differences in sediment layers. This can provide information on oxic (lighter oxygen rich sediment) and anoxic (darker oxygen depleted sediment) layers along the depth gradient and could later provide further information around carbon pooling in relation to depth.

Sediment cores were divided into depth fragments on the same day of collection, this was done both in the field and in the laboratory. We adopted more recent methods in core division by depth, instead of dividing the top 10cm by 1cm fragments we chose to separate the top 10cm by 2cm fragments, providing more sample for analyses later (Bulmer *et al.*, 2020). After the top 10cm the sediment core was cut every 4cm then 1cm until the end of the sediment core where the final 2cm of the core were divided into 1cm fragments (Figure 2.1). Cores were carefully cut into depth sections and the instrument was cleaned between each section to ensure cross contamination did not occur (IUCN, 2019).

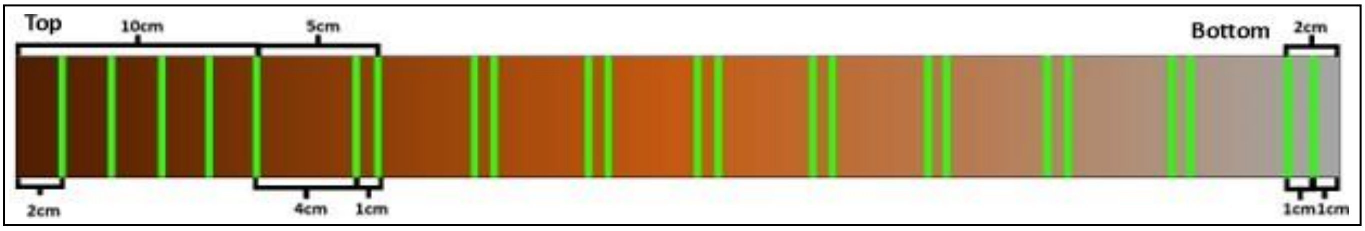

Figure 2.1: Protocol followed when segmenting each sediment core into subsamples, the number of subsamples varies depending on the size of the core taken but the same methods were applied when segmenting each core into subsamples.

Figure 2.3: Map displaying the exact location of where core sampling and vegetation surveys occurred at Te Awa o Ngātoroirangi and Kūkūwai Whakapoukōrero.

Figure 2.4: Map displaying the exact location of where core sampling and vegetation surveys occurred at Te Pā Ika Wetland.

Figure 2.5: Map displaying the exact location of where core sampling and vegetation surveys occurred at Wainui River Saltmarsh and Wainui Repo Whenua Saltmarsh.

Figure 2.6: Map displaying the exact location of where core sampling and vegetation surveys occurred at Matua Saltmarsh.

2.3 Laboratory Methods

2.3.1 Bulk Density

All sediment samples were frozen and stored until sediment processing could occur, the total wet weight of each core depth fragment was weighed and recorded, following this the sediment samples were transferred into individually labelled tin confoil cups (70mm x 20mm) to be dried at 60˚C for 7 days until a constant weight was reached (Figure 2.8). Dry core fragment samples were then reweighed and placed into individual labelled 70ml canister pottles for storage.

Bulk density which is an indicator of sediment compaction (Nyle, 2008) can then be calculated with the following formulae:

$$1.)\ W_{Fdry} = \frac{W_{Sdry} \times W_{Fwet}}{WSwet} \qquad 2.)\ Bulk\ density = \frac{W_{Fdry}}{V_F}$$

Where W_{Sdry} is the dry weight of the subsample (g), W_{Fwet} is the wet weight of the core fraction (g), W_{Swet} is the wet weight of the subsample (g), and V_F is the volume of the fraction (cm^3).

Approximately 2g of all the dried sediment and root/plant sample was then pulverised in a RETSCH MM 400 Mixer Mill at 27.5 Hz for 1:20 min to homogenise the sample and reduce grain size, sample was placed back into a labelled 70ml canister pottle for storage.

2.3.2 Carbon Concentration

Acidification

All sediment subsamples were acidified to remove sources of carbonate (decalcification) through matter such as shell hash. Prior to elemental analysis each pulverised sediment subsample was weighed out to 40mg (± 0.5 mg) and placed into a 10mm x 10mm tin capsules, acidification methods followed Verardo *et al.* (1990). Subsamples were initially treated with 50µL ~6 % sulphurous acid (H_2SO_3) and 50µL deionised water and then transferred to a 60˚C hotplate, once dry subsamples were incrementally treated with 50µL sulphurous acid and then dried between treatments. Treatment occurred until effervescence ceased or until a total of 200µL of acid had been added to the subsample.

Subsamples were carefully examined to ensure no sample had been lost during acidification treatments, if the tin cup the subsample was in appeared to have been corroded by the treatments or there was any sign of sample loss then the subsample would need to be completely redone. The tin cups all became brittle after treatment and so all subsamples were placed into a second tin cup before being pinched and rolled for storage and later analysis.

Organic Carbon Content

Acidified 40mg (± 0.5mg) subsamples were analysed on an Elementar Vario El Cube CN analyser for carbon and nitrogen content. The elemental analyser would undergo multiple blank and known acetanilide 10mg (± 10% tolerance) tests to initially test calibration and standards before analysis of subsamples could commence. Once standards were stable and within a 10% tolerance of 10.360% Nitrogen and 71.090% Carbon analysis could begin, with acetanilide tests being run every 20 subsamples. Subsamples weighing 40mg (± 0.5mg) and were run on the method of 35mg over 120 seconds of oxygen dosing, these parameters were established by running multiple tests of varying subsample weight and methods until the method and weight that most efficiently produced stable readings was chosen.

Total Organic Carbon Content

Once the %C of the subsample was obtained through elemental analysis, the total organic carbon content of each sediment depth fraction was calculated using the following formulae:

$$1.)\ C_{ss} = \frac{C_{sc} \times W_{Sdry}}{W_{sc}} \qquad\qquad 2.)\ C_f = \frac{C_{ss} \times W_f}{W_{Swet}}$$

Where C_{sc} is carbon content of the analysed sample (mg), W_{sdry} is the weight of the total dried subsample (mg), W_{sc} is the weight of sediment analysed in the cup (mg), C_{ss} is the carbon content of the dried subsample (mg), W_f is the wet weight of the core fraction (g), W_{swet} is the wet weight of the subsample that was dried (g), and C_f is the carbon content of the core fraction (mg).

2.4 Total Sediment Carbon Stock

Using the sediment depth, subsample depth and interval, dry bulk density and the % organic carbon total sediment carbon stock can be calculated using the following formulae:

1.) Sediment carbon density (g cm^3) = dry bulk density (g cm^3) * (% C_{org}/100)

The amount of carbon in each of the various fragments of the core can then be calculated by multiplying each sediment carbon density in each core fragment by the thickness of the sample fragment (cm), the following formulae can be applied:

2.) Amount carbon in core section (g cm^2) = Sediment carbon density (g cm^3) * thickness interval (cm)

To then calculate the amount of carbon in each core the following formulae can be used, it is critical that the total core length is known for this step:

3.) Core #1 summed = Amount carbon in core fragment A (g cm^2) + Amount carbon in core fragment B (g cm^2) + Amount carbon in core fragment C (g cm^2) + etc. (including every fragment within the core)

Next the amount of carbon per core is then converted from grams of carbon per cm^2 to tonnes (t) of carbon per hectare (t ha^{-1}), this calculation will need to be repeated for each core:

4.) Total core carbon (t ha^{-1}) = Summed core carbon (g cm^2) * (1 t/1 000 000 g) * (100 000 000 cm^2/1 hectare)

To determine variability and asses any potential errors the standard deviation across each core will be calculated using the following equation:

5.) Core Standard Deviation (σ) = $[\frac{(X_1-X)2+(X_2-X)2+\cdots(X_n-X)2}{(N-1)}]^{\frac{1}{2}}$

Where X is the average carbon in a core, X_1 is the individual result for core #1, MgC/hectare is represented by X_n =is the individual result for core #2, X_N etc., and N is the total number of results.

To then understand how much what the carbon stock is in each study site the average core carbon stock is used and multiplied by the size of the study site, it is critical to include the total final depth of each core as the final unit for sediment carbon stock (t ha^{-1}) in each site will be over the specific depth interval reached.

6.) Total organic carbon in a project area (t ha^{-1}) = (average core carbon from Stratum A[1] (t ha^{-1}) * area Site 1 (hectares)) + (average core carbon from Stratum B (t ha^{-1}) * area Site 1 (hectares) + etc.

[1] Where Stratum A is the low marsh core data and where Stratum B is the mid marsh core data.

Total variability was determined to represent any potential uncertainties in data across study sites, the standard deviation for each site is calculated using the total organic carbon in each project area and the size of each site (hectares). Final carbon stock values will be presented in tonnes per hectare (t ha^{-1}) and represented along with their relative ± uncertainty (standard deviation) (Howard *et al.*, 2014; Radabaugh *et al.*, 2018).

Isotopic analysis for Carbon dating

Sixty samples between 1-6 grams were sent to Environmental Radioactivity Laboratory (ESR) at the Institute of Environmental Science and Research Limited for alpha spectrometry analysis. The deepest core from each site was selected and depth fragments 0-2cm, 4-6cm, 8-10cm, 14-15cm, 19-20cm, 24-25cm, 29-30cm, 34-35cm, 39-40cm, and 44-45cm were analysed for Po-210 and Pb-210 (10 fragments per core).

Additionally, 12 samples between 3-20 grams were sent for gamma spectrometry analysis, the deepest core from each site was selected and depth fragments 4-6cm and 8-10cm were analysed. Samples were analysed for Pb-210, Ra-228, Ra-226 and Cs-137 simultaneously, Ra-226 and Ra-228 give a baseline for supported Pb-210 (via Ra-226) and may provide additional information. Cs-137 is a marker used for dating, the Ra-226 and Ra-228 data provided through gamma spectrometry can be used to measure the supported Pb-210 activity from alpha spectrometry and should further support our isotopic results.

2.5 Statistical Analysis

PRIMER7 (Anderson *et al.*, 2008) will be used to conduct a one-way PERMANOVA tests for significant (p <0.05) differences between mean carbon stocks across sites. And a two-way PERMANOVA or pairwise post-hoc tests will be used to test for significant (p <0.05) differences between depths and sediment carbon stocks between the sites (Bulmer *et al.*, 2020).

Chapter 3
Results

The following chapter presents key findings surrounding environmental characteristics, sediment carbon percentage, carbon stocks at the core (g cm^3) and site level (t ha^{-1}), and radioisotope carbon dating.

3.1 Environmental Characteristics

3.1.1 General Site Characteristics

In summary, six coastal saltmarsh wetlands were studied and were selected largely based on the occurrence of vegetation and hydrological restoration. Wainui Repo Whenua Wetland (~1 year) has most recently had restoration through a combination of planting, hydrological changes through re-wetting, and restoring tidal influence, Te Pā Ika Wetland (~5 years) and Kūkūwai Whakapoukōrero (~20 years) were restored through similar methods. Matua Saltmarsh (30 years) has had restoration through community-based plantings in the late 1990's, Te Awa o Ngātoroirangi and Wainui River Saltmarsh were the two control coastal saltmarsh wetlands for this study.

The study sites in order of restoration occurrence are visually presented from most recently restored to control sites (Figure 3.1). The first image is site wide and clearly shows the state of each of the sites, evidently the level of vegetation present in Wainui Repo Whenua Wetland (~1 year) compared to that of the two control sites Te awa o Ngātoroirangi and Wainui River Saltmarsh, the second image provides further context to the vegetation coverage and diversity. The first sediment core which was extracted from the low marsh environments is attached, due to error there is no image available for the low sediment core from Wainui River Saltmarsh. The last images are the sediment cores extracted from the mid-marsh at each site, all cores are arranged with the surface level on the left, it is notable that core depths varied by strata and site.

3.1.2 Sediment Core Characteristics and Bulk Density

The deepest sediment core samples were extracted from Wainui Repo Whenua Wetland (~1 year) and Matua Saltmarsh (~30 years) as a maximum depth of 69cm, while the cores that reached the shallowest depths were extracted from Te Pā Ika Wetland (~5 years) and Kūkūwai Whakapoukōrero (~20 years) at a maximum depth of 27cm. Due to the varying depths reached across the sediment core samples the number of sub-samples extracted subsequently varied, Wainui Repo Whenua Wetland (~1 year) had the highest total number of sub-samples at 55, while only 35 sub-samples were able to be analysed from Te Pā Ika (~5 years) across both sediment cores.

Wainui Repo Whenua Wetland (~1 year) and Te Pā Ika (~5 years) both had the highest bulk density of 0.907g cm^3, however Wainui Repo Whenua Wetland (~1 year) had a lower standard error of ±0.034 between the two sites. Wainui River Saltmarsh (control) had the lowest mean bulk density at 0.633 ± 0.032g cm^3. It is notable that the bulk density for Matua Saltmarsh (~30 years) had the largest standard error 0.765 ± 0.057g cm^3 (Table 3.1). Overall, there was little variability between bulk density across the sites and all bulk density values suggest that the sediment grain size is finer and comparable with clay (<1.10g cm^3).

Figure 3.1: Pictogram illustrating the sites surveyed in ascending order of date at which restoration (most recent to control).

Table 3.1: Additional data on site characteristics, core subsample information, and bulk density, ±SE.

	Wainui Repo Whenua Wetland (SD5) - 3.4ha	**Te Pā Ika Wetland (MK2) - 12.35ha**	**Kūkūwai Whakapoukōrero (MK3) - 35.68ha**	**Matua Saltmarsh (MS6) - 20.43ha**	**Te Awa o Ngātoroirangi (MK1) - 15.55ha**	**Wainui River Saltmarsh (SD4) - 15.75ha**
Age	~1 year since restoration	~5 years since restoration	~20 years since restoration	~30 years since restoration	Control	Control
Restoration	Ground works to restore hydrological flow and planting undertaken	Ground works to restore hydrological flow and planting undertaken	Ground works to restore hydrological flow and planting undertaken	Planting	N/A	N/A
Environmental characteristics	Previously paddock, surrounded by agricultural land, and adjacent to State Highway 2 and Wainui River	Previously paddock, surrounded by estuary, and adjacent to Ford Road	Previously paddock, surrounded by agricultural land, and adjacent to Maketū Road	Surrounded by urban residential development, boardwalks, railway, and adjacent to Ngatai Road	Surrounded by estuary, agricultural draining outlet adjacent to saltmarsh	Surrounded by agricultural land, and adjacent to State Highway 2 and Wainui River
Max Core Depth reached (cm)	69	48	49	69	51	46
Min Core Depth reached (cm)	57	27	27	42	49	41
Total sub-samples	55	35	42	49	44	38
Mean Bulk Density ($g\ cm^3$)	0.907 ± 0.034	0.907 ± 0.051	0.843 ± 0.054	0.765 ± 0.057	0.805 ± 0.054	0.633 ± 0.032
Highest Bulk Density ($g\ cm^3$)	1.458	1.614	1.762	1.393	1.324	1.159
Lowest Bulk Density ($g\ cm^3$)	0.310	0.315	0.287	0.142	0.209	0.146

3.2 Carbon Stock

3.2.1 Carbon Concentration

Carbon concentration is presented as mean carbon percentage calculated at the site level from each of the sediment core sub-samples. Matua Saltmarsh (~30 years) contained the highest mean concentration of carbon at 5.83 ± 1.151%, notably this is the highest standard error across concentrations, resulting in an upper limit of 6.38% and a lower limit of 4.08%. Wainui Repo Whenua (~1 year) displayed the lowest mean carbon concentration of 2.91 ± 0.26%, however this site had a maximum carbon concentration of 20.36% across depth sub-samples, this is the second highest maximum recording with Matua Saltmarsh (~30 years) having the highest sub-sample maximum of 21.13%. The minimum carbon concentration across sub-samples was present at Kūkūwai Whakapoukōrero (~20 years) at 0.22% (Table 3.2). Overall, there were no significant linear trends across carbon concentration, sites and temporal variance.

Table 3.2: Summary of carbon concentrations (%) across sites, ±SE.

	Wainui Repo Whenua Wetland (SD5) - 3.4ha	**Te Pā Ika Wetland (MK2) - 12.35ha**	**Kūkūwai Whakapoukōrero (MK3) - 35.68ha**	**Matua Saltmarsh (MS6) - 20.43ha**	**Te Awa o Ngātoroirangi (MK1) - 15.55ha**	**Wainui River Saltmarsh (SD4) - 15.75ha**
Age	~1 year since restoration	~5 years since restoration	~20 years since restoration	~30 years since restoration	Control	Control
Total sub-samples	55	35	42	49	44	38
Carbon % to 50cm core depth	3.60 ± 0.30	4.22 ± 0.66	3.46 ± 0.77	6.38 ± 1.33	3.53 ± 0.79	3.70 ± 0.56
Carbon % to max core depth	2.91 ± 0.262	4.29 ± 0.619	3.46 ± 0.774	5.84 ± 1.151	3.42 ± 0.777	3.89 ± 0.566
Carbon % Upper Limit	2.98	4.58	4.38	6.38	4.19	4.36
Carbon % Lower Limit	2.46	3.35	2.83	4.08	2.64	3.23
Max Carbon%	20.36	13.64	14.48	21.13	14.46	11.97
Min Carbon%	0.50	0.64	0.22	0.25	0.34	0.88

3.2.2 Carbon Stock across Depths

It is essential to note that sediment cores reached varying depths across sites due to varying factors, for Figure 3.2 the total sediment organic carbon stock (TOC) is represented until the deepest mean depth reached across a site. For this reason, Te Awa o Ngātoroirangi (control, 54cm), Wainui Repo Whenua Wetland (~1 year, 69cm), and Matua Saltmarsh (~30 years, 69cm) observably have data past 50cm. Kukuwai Whakapoukōrero (~20 years), Wainui River Saltmarsh (control), and Te Pā Ika Wetland (~5 years) were standardised to 50cm. Overall, across sites, TOC (g cm^3) decreases with depth, particularly after 20cm (Figure 3.2). Wainui Repo Whenua (~1 year) initially has the highest TOC between 4 -10cm, with the highest stock present at 6cm (0.12g cm^3), the TOC then steadily decreases and then follows similar trends to Wainui River Saltmarsh (control) and Te Awa o Ngātoroirangi (control). Matua Saltmarsh (~30 years) has the next highest recording at 15cm with 0.138g cm^3, before rapidly dropping to 0.06g cm^3 between 15cm and 19cm. Te Pā Ika Wetland (~5 years) then peaks at 25cm with a recording of 0.13g cm^3 before it then steadily drops until 45cm where a reading of 0.10g cm^3 occurs, this follows trend to Wainui River Saltmarsh (control) that also experiences a sudden peak at 44cm with a reading of 0.07g cm^3. In summary TOC is higher in shallow depths compared to the TOC present at depths past 25cm, with the lowest TOC of 0.01g cm^3 recorded at 44cm in Kūkūwai Whakapoukōrero (~20 years). Statistical analysis of carbon stocks across all depths indicated significant difference between the 10cm and 44cm depth fragments ($p < 0.0019$, Table 3.5) suggesting that there are significant differences between TOC between depths and sub-sample fragments. It is notable that for Table 3.5 depths were standardised to 49cm in order to compare these three variables (depth, TOC, and site).

3.2.3 Carbon Stock and Accumulation Rates

Matua Saltmarsh (~30 years) had the highest total organic carbon stock at 1.21 ± 0.005g cm^3, this translates to 121.42 ± 0.534t ha^{-1} or 91.52 ± 0.75t ha^{-1} to a standard depth of 50cm . The lowest carbon stocks were present in Te Awa o Ngātoroirangi (control) at 0.69 ± 0.004g cm^3, translating to a maximum total OC of 68.66 ± 0.401t ha^{-1} and a standardised TOC of 66.83 ± 0.42t ha^{-1}. Wainui River Saltmarsh (control) had the highest carbon accumulation rates (CAR), with a sediment accumulation rate (SAR) of 11.57mm yr^{-1}, translating to an estimated 288.82g m^{-2} yr^{-1} of carbon over 20 years. It is notable that all sites were dated to 20 years except for Matua Saltmarsh (~30 years) which was dated to 30 years due to restoration through planting practices occurring in the 1990's, the SAR was 3.39mm yr^{-1} and the CAR was 107.91g m^{-2} yr^{-1} of carbon over 30 years or 161.87g m^{-2} yr^{-1} over the last 20 years. Wainui Repo Whenua (~1 year) had the lowest SAR 2.23mm yr^{-1}, while the CAR was 192.63 g m^{-2} yr^{-1} (Table 3.3). There appears to be a weak linear relationship between CAR and temporal gradient of restoration (Figure 3.4), CAR steadily increases with age except for Matua Saltmarsh (~30 years) and Te Awa o Ngātoroirangi (control). Suggesting carbon sequestration rates steadily increase in restored saltmarsh environments with time this is tightly interlinked to the Sediment Accumulation Rate (SAR) for each site (Figure 3.5).

3.2.4 Total Organic Carbon

TOC across sites was calculating by summing each of the sub-sample organic carbon stocks (t ha^{-1}) at maximum depth and then to a standardised 50cm depth to account for variability. Matua Saltmarsh (~30 years) had the largest TOC by maximum core depth of 121.42 ± 0.534t ha^{-1}, however once depths and TOC are standardised to 50cm depth Te Pā Ika Wetland (~5 years) displayed the highest TOC of 125.68 ± 0.61t ha^{-1}. Te Awa o Ngatoroirangi (control) had the lowest maximum core depth TOC of 68.66 ± 0.401t ha^{-1}, after standardisation to 50cm depth this figure only varied by 1.83t ha-1. The largest and smallest standardised TOC varied by 58.85t ha^{-1}. Three of the four restored sites have higher total maximum and standardised TOC results compared to both control sites with Kuku Whakapoukōrero Wetland (~20 years) being the only restored site with a total OC below Wainui River Saltmarsh (control), with only 5.39t ha^{-1} maximum TOC and 8.97t ha^{-1} standardised TOC separating this site from the Te Awa o Ngātoroirangi control site (Figure 3.3). However, a one-way PERMANOVA of TOC between sites suggests that the most significant difference between TOC across sites is evident between Te Awa o Ngātoroirangi (control) and Te Pā Ika Wetland (~5 years, $p < 0.0007$). It is also evident that Te Awa o Ngātoroirangi (control) displays a significant difference with Wainui Repo Whenua Wetland (~1 year, $p < 0.0054$, Table 3.4).

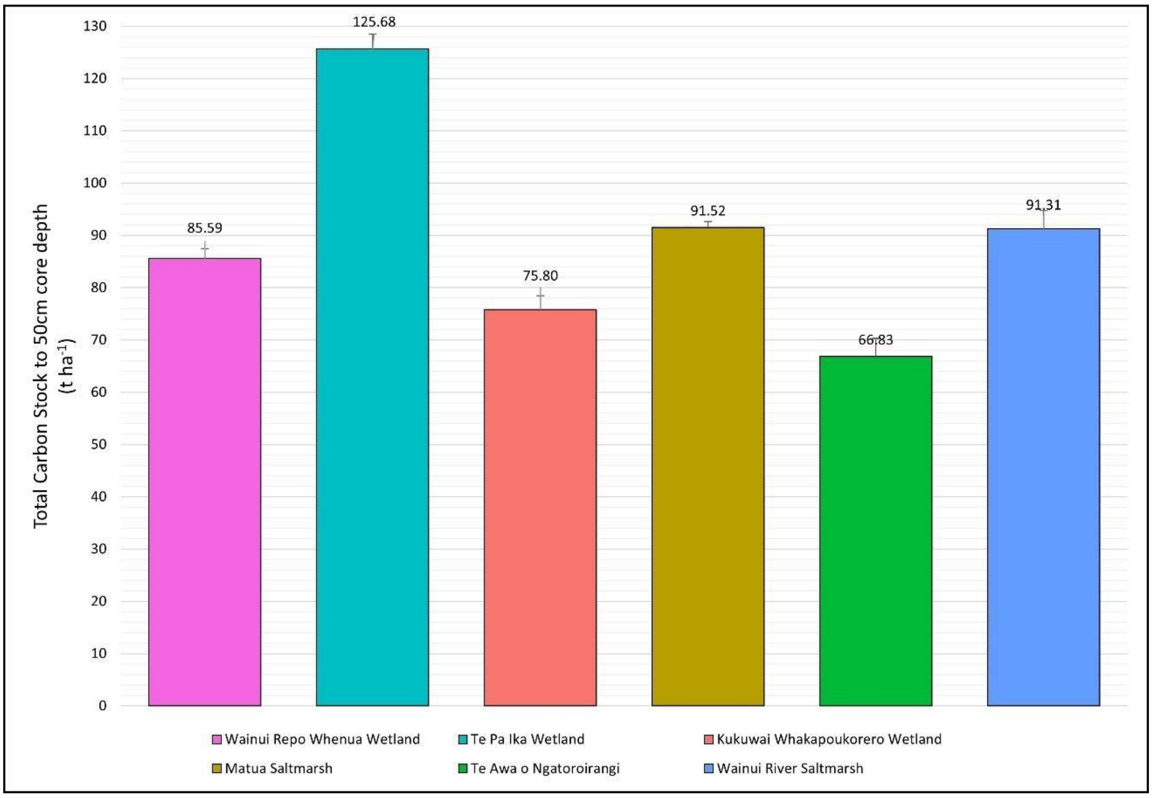

Figure 3.2: Total sediment carbon stocks across sites to a standardized depth of 50cm, ±SE.

Table 3.3: Summary of organic carbon stock across sites, as well as estimated radioisotope carbon dating data, ±SE

	Wainui Repo Whenua Wetland (SD5) - 3.4ha	**Te Pā Ika Wetland (MK2) - 12.35ha**	**Kūkūwai Whakapoukōrero (MK3) - 35.68ha**	**Matua Saltmarsh (MS6) - 20.43ha**	**Te Awa o Ngātoroirangi (MK1) - 15.55ha**	**Wainui River Saltmarsh (SD4) - 15.75ha**
Age	~1 year since restoration	~5 years since restoration	~20 years since restoration	~30 years since restoration	Control	Control
Total Carbon Stock to max core depth[2] (g cm^3)	0.97 ± 0.006	1.17 ± 0.006	0.74 ± 0.006	1.21 ± 0.005	0.69 ± 0.004	0.81 ± 0.002
Total Carbon Stock Upper Limit (g cm^3)	0.98	1.18	0.75	1.22	0.69	0.85
Total Carbon Stock Lower Limit (g cm^3)	0.97	1.16	0.74	1.21	0.68	0.84
Total Carbon Stock to max core depth[2] (Tonnes ha^{-1})	97.62 ± 0.673	116.88 ± 0.630	74.05 ± 0.587	121.42 ± 0.534	68.66 ± 0.401	80.90 ± 0.244
Total Carbon Stock to 50cm depth (Tonnes ha^{-1})	85.59 ± 0.76	125.68 ± 0.61	75.80 ± 0.58	91.52 ± 0.75	66.83 ± 0.42	91.31 ± 0.25
Estimated Carbon Stock across each site (Tonnes site area $^{-1}$ to 50cm)	291.01	1552.15	2704.38	1869.68	1039.17	1438.17
SAR over the last 20 years mm yr^{-1}	2.23	6.70	5.61	3.39[3]	5.31	11.57
Habitat CAR over last 20 years g C m-2 yr^{-1}	192.63	227.04	276.58	161.87 107.91[3]	217.95	288.82

[2] Note that max core depths vary across sites, refer to Table 3.2 for specifics.
[3] Matua saltmarsh (MS6) SAR calculated over the last 30 years as restoration (planting) occurred in the late 1990's.

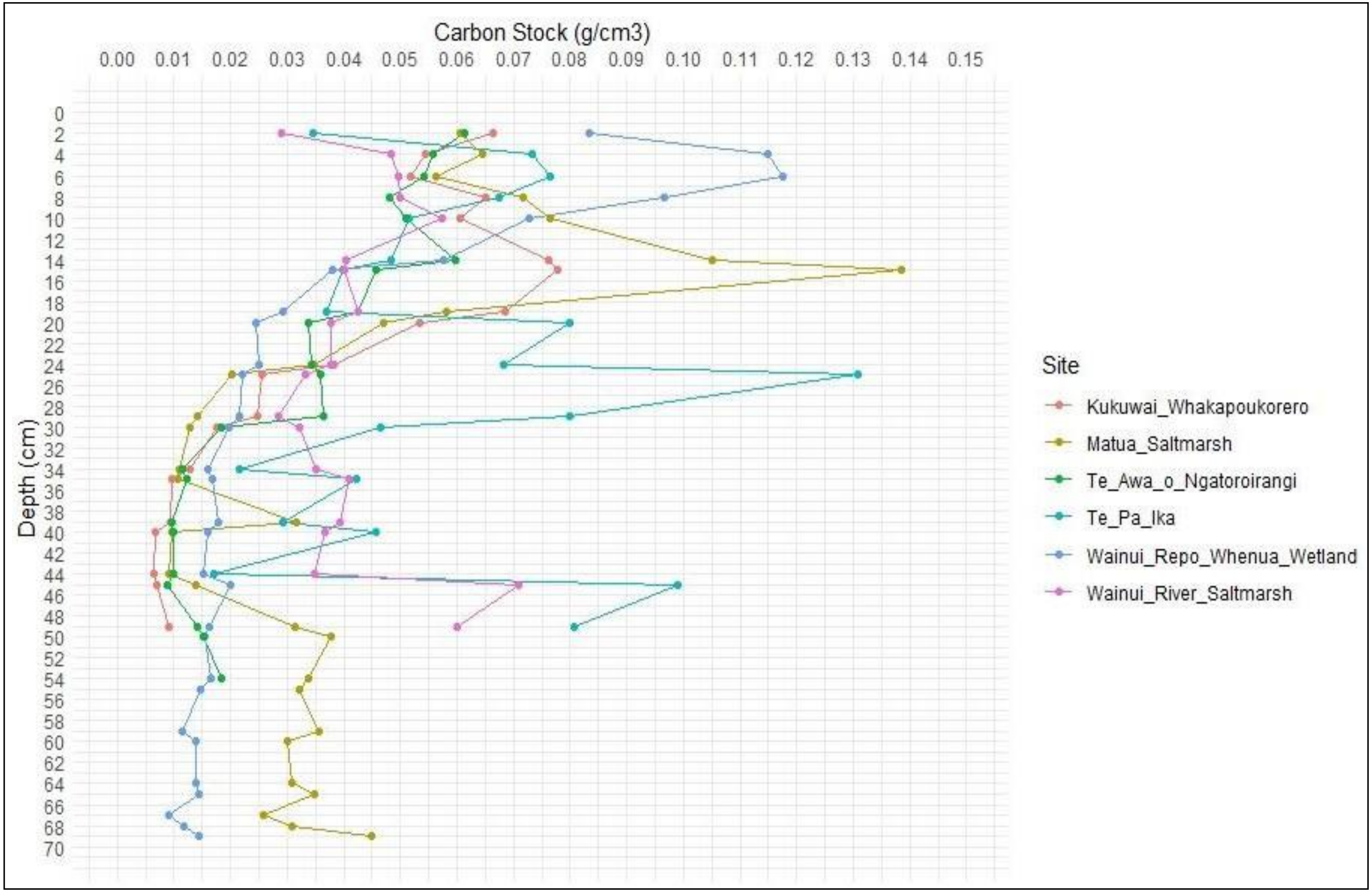

Figure 3.3: Mean sediment organic carbon stock (grams per cubic centimetre) across depth intervals across sites, each point represents a sample.

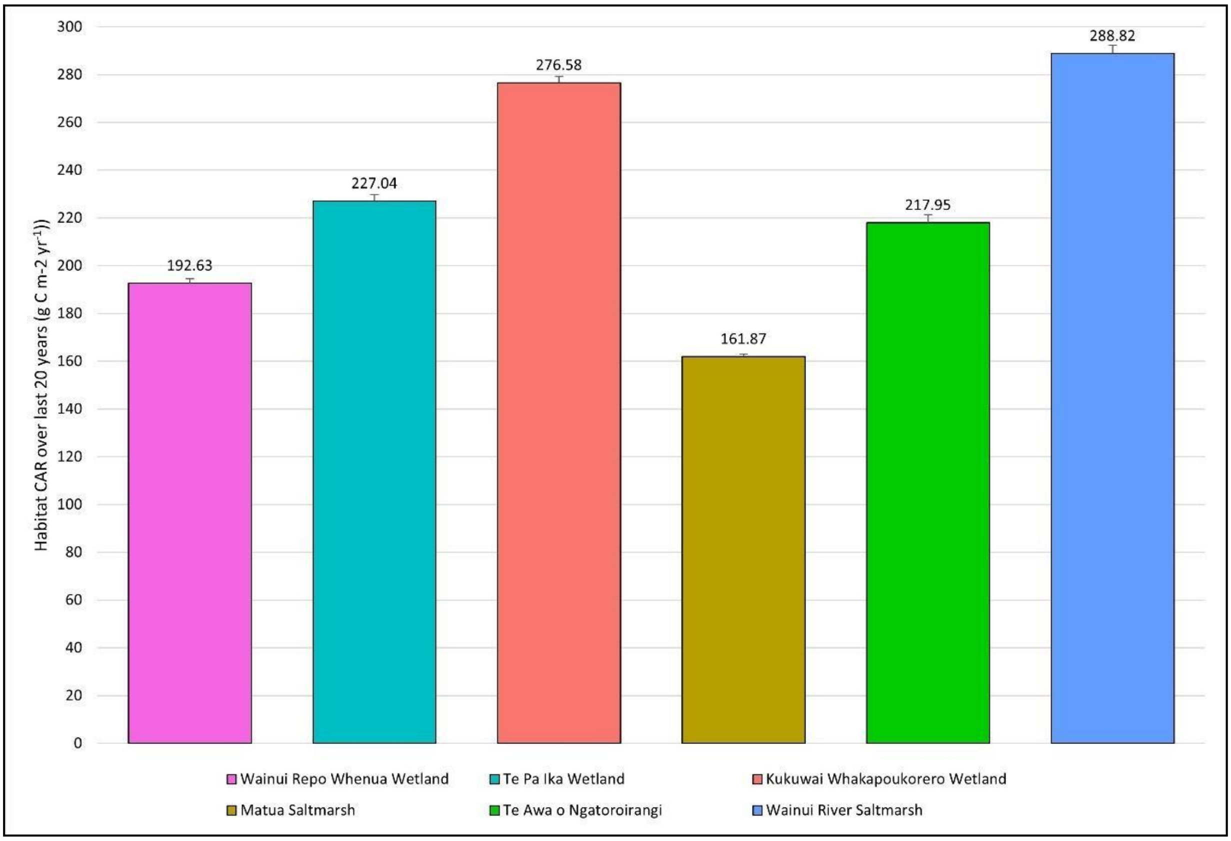

Figure 3.4: Carbon Accumulation Rates (CAR) across sites for the last 20 years.

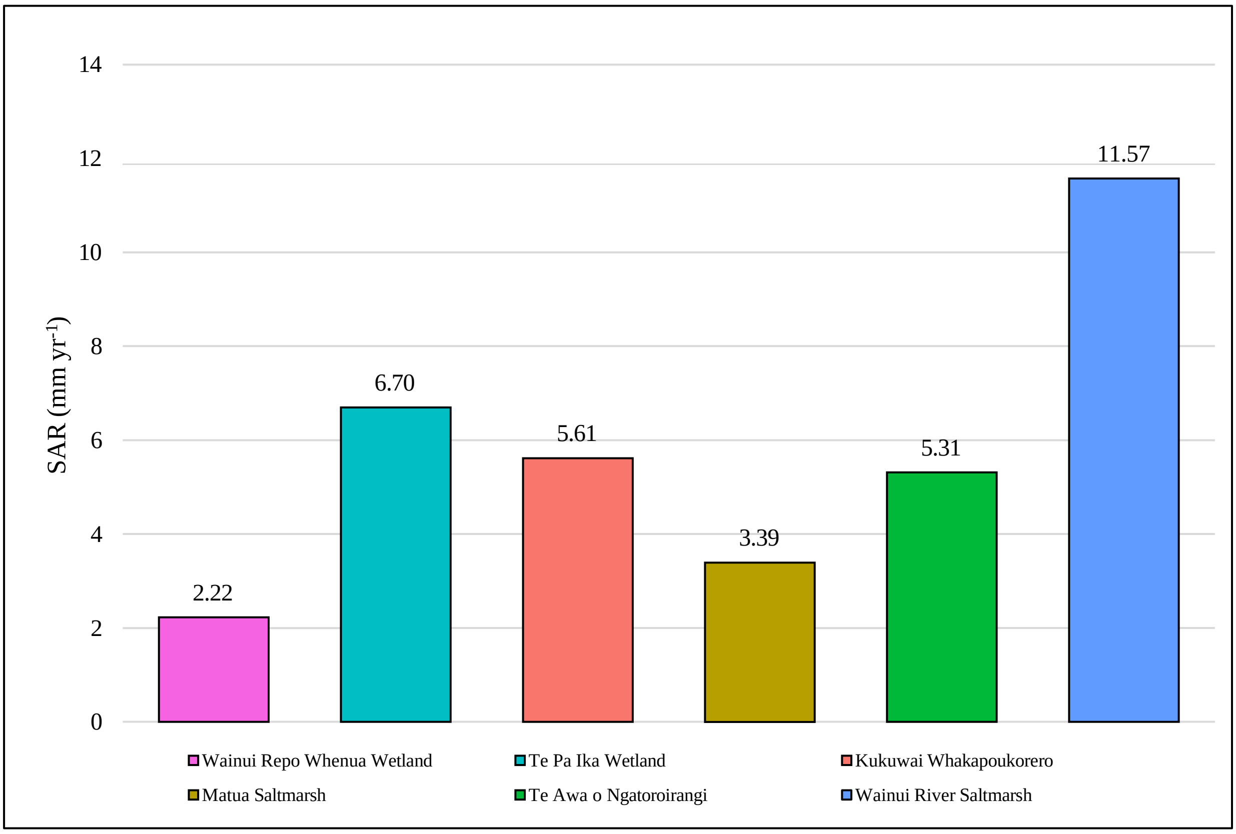

Figure 3.5: Sediment Accumulation Rate (SAR) across sites for the last 20 years.

Table 3.4: One-way pairwise PERMANOVA results, comparing total organic carbon stock (t ha^{-1}) across sites, displayed are the sites that displayed significant differences to each other ($p<0.05$).

Site	Site	t-value	P-value (p <0.05)
Te Awa o Ngātoroirangi	Te Pā Ika Wetland	3.717	0.0007*
Te Awa o Ngātoroirangi	Wainui River Saltmarsh	2.292	0.0263
Te Pā Ika Wetland	Kūkūwai Whakapoukōrero	2.470	0.0172
Te Pā Ika Wetland	Wainui River Saltmarsh	2.402	0.0172
Te Pā Ika Wetland	Wainui Repo Whenua Wetland	2.966	0.0054
Te Pā Ika Wetland	Matua Saltmarsh	2.160	0.0374

Table 3.5: Two-way pairwise PERMANOVA results, comparing carbon stocks (t ha^{-1}) to 50cm.

Depth from Surface (cm)	Depth from Surface (cm)	t-value	P-value (p <0.05)
2	44	3.465	0.0183
4	44	4.680	0.0049
6	44	4.473	0.0044
8	44	5.516	0.0034
10	44	6.820	0.0019*
14	39	3.671	0.0129
15	34	2.463	0.0171
19	44	3.561	0.0164
20	30	3.601	0.0117
24	30	4.704	0.0037
25	40	1.839	0.0342
34	44	2.653	0.0291
40	50	2.496	0.047

Chapter 4
Discussion

As the global urgency to combat climate change grows, the focus on blue carbon research has intensified, with scientists, practitioners, and policymakers globally recognizing the potential of NBS in mitigating GHG emissions. Among these solutions, saltmarsh ecosystems play a crucial role in sequestering carbon, making these environments a key player in climate change mitigation and adaptation efforts. These habitats have undergone significant change and are experiencing accelerating shifts due to both climate change impacts and anthropogenic activities. Despite their importance, there is a significant knowledge gap in understanding of blue carbon dynamics in restored saltmarsh habitats, particularly in Aotearoa.

This study addresses this critical gap by investigating blue carbon stocks in restored saltmarsh habitats in the BOP. Saltmarsh restoration is increasingly recognized as a valuable strategy not only for preserving biodiversity and ecosystem services but also for contributing to environmental resilience. Our findings aim to provide additional information for guiding future restoration practices and policies surrounding saltmarsh habitats and blue carbon, broadening the current understanding of the role that saltmarshes possess in climate change mitigation in the unique context of Aotearoa.

4.1 Summary

A key finding in this study is the high level of variability in blue carbon stocks across the study sites. These multidimensional bionetworks encompass intricate ecosystems with services occurring both aboveground and belowground. While saltmarsh restoration may seem conceptually simple, providing ecosystem-level solutions is inherently complex. Restored habitats are dynamic, influenced by various factors, resulting in multiple possible outcomes over time and a restored saltmarsh may not exhibit all characteristics of a “pristine” control environment. However, it is clear that as saltmarsh habitats evolve after restoration efforts are commenced, there can be substantial changes to the system as it naturalizes (Staszak & Armitage, 2013). There are several key elements that significantly influence the evolving saltmarsh’s biotic and abiotic nature, including its history, catchment, elevation, hydrology, species composition, age since restoration, and level of inundation (Billah *et al.*, 2022). Although this study did not cover all these elements in depth, we can still derive meaningful insights into how blue carbon stocks in restored saltmarsh systems in the BOP evolve with time.

In our study that examined blue carbon stocks in both restored and control saltmarsh habitats, there was consistently high TOC in both types of habitats, within expected national and global ranges of TOC (Bulmer *et al.*, 2020). This indicates the potential of restored saltmarsh habitats in Aotearoa to serve as effective blue carbon sinks. Notably, TOC in the restored saltmarshes in this study exhibited elevated TOC levels compared to control habitats both compared to control sites in this study and compared to both restored and control sites in other literature, challenging the anticipated lower TOC in restored environments (McMahon *et al.*, 2023). Notably, the most recently restored site, Wainui Repo Whenua Wetland, displayed elevated TOC levels compared to the control environments despite its short timeframe since restoration (~1 year), which was in line with findings such as that of Burden *et* al (2019) and Noll *et* al (2019) who proved blue carbon accumulation rates to be most rapid in the first 20 years post restoration.

Despite variability and weak linear trends between TOC and CAR, our findings affirm that surveyed saltmarsh wetlands in the BOP, Aotearoa, consistently exhibit elevated TOC and CAR levels compared to restored saltmarshes globally and control sites. This suggests that restored saltmarsh environments in our study hold significant potential as blue carbon sinks, with restoration positively impacting carbon storage. The variability in our data highlights the importance of considering broader environmental factors, such as catchment characteristics, elevation, hydrology, age, history, sediment accumulation rates, and restoration techniques when evaluating blue carbon in the context of saltmarsh restoration.

4.2 Sediment Heterogeneity in Restored Saltmarsh Ecosystems

The following section discusses the sediment variability across our study sites, focusing on sediment bulk density in relation to blue carbon stocks, and compares our findings to previous studies. The expected trend across our study sites based on previous studies was that bulk density would decrease with time since restoration and with depth (USDA Natural Resources Conservation Service, 2008). Bulk density (a measure of sediment mass per unit volume) is influenced by factors such as particle size, composition and arrangement. For example, more finer particles like clay and silt generally lead to higher bulk densities compared to more coarser particles such as sand. This is because smaller grains can pack closer together with less pore space. Although bulk density can offer insights into sediment composition and texture, it is not a direct measure of grain size. Detailed grain size analysis is beyond the scope of this study although we recommend it in future work for more accurate understanding of the relationship between grain size and bulk density.

The trend of higher mean bulk densities at recently restored sites in our study suggests that these sites have more finer particle sediment (e.g., silt) compared to older less disturbed sites. This is in line with findings from Kadiri *et al.* (2011), French (2019) and Spencer *et al.* (2017). There is a non-significant negative trend across our six sites with bulk density generally decreasing with time since restoration. For example, this is evident for Wainui Repo Whenua Wetland (~1year) and Te Pā Ika Wetland (~5 years), both of which have higher mean bulk density compared to the least disturbed control habitat (Wainui River Saltmarsh). In addition, bulk density was generally lower at the surface and increased with depth in each core. This was particularly evident in Kūkūwai Whakapoukōrero (~20 years) which displayed a slightly higher maximum bulk density particularly in deeper levels within the sediment column compared to our other study sites, suggesting high levels of silt within these deeper levels of sediment at the site.

In saltmarsh environments, the relationship between bulk density and depth is not always negatively correlated; instead, it can be influenced by a variety of factors and bulk density may increase with depth due to processes such as sediment accumulation and compaction. As new layers of sediment accumulate over time, compaction can occur, leading to higher bulk density in deeper layers (Stagg & Mendelssohn, 2010). Additionally, the decomposition of organic matter near the surface and alterations in hydrological conditions, whether natural or influenced by restoration efforts, can impact sediment compaction, and contribute to an increase in bulk density with depth. Although the reasons for this are unclear due to the rage of possible differing influencing factors, some studies find opposite trends in bulk density with depth compared to our findings. For example, Noll *et al.*, (2019) tracked the restoration of a constructed tidal saltmarsh in North Carolina, USA, spanning five decades from the 1970s. Periodic measurements of a range of sediment properties including bulk density and organic carbon showed that bulk density decreased over the five decades and was higher in the top 0-10cm of cores than 10-30cm below.

Our study had a wide range of core lengths ranging from 27cm to 69cm. This variation can be attributed to a range of factors including differences in bulk density, sediment grain size, historic land use, and levels of inundation (Cai *et al.*, 2022). Environments characterized by visibly larger grain sizes, such as sand compared to silt, generally yielded shorter sediment cores, and we would expect these environments to have lower bulk densities. Based on visual observations of the general appearance of the sediment at each of the sites, especially those with relatively high sand content, we would have expected Te Pā Ika Wetland (~5 years) to have the lowest bulk density paired with the mean lowest core depth. However, the lowest mean bulk density was observed at our "pristine" control environment Wainui River Saltmarsh. This may have been due to less anthropogenic impacts compared to that of our modified restoration sites, steady levels of inundation, and that levels of elevation across the site are naturalised and optimal to the surrounding habitat.

The site exhibiting the highest silt content and the greatest maximum core length, Wainui Repo Whenua Wetland (~1 year), has an interesting dynamic in bulk density variations and grain size. The elevated bulk density observed at this site could be attributed to several potential processes stemming from recent disturbances to the sediment column. Changes in hydrological flow in efforts to re-establish regular tidal influence along with the mechanical and physical works associated with this, may have led to sediment compaction and higher bulk density (Spencer *et al.*, 2017). The site's history of agricultural land use could have left a lasting imprint on sediment characteristics both morphologically and biochemically, contributing to the observed fine grain size and higher bulk density (Koppenaal *et al.*, 2022). Additionally, the absence of detailed analysis regarding the correlation of vegetation root systems with sediment core extraction in this study leaves room for further exploration. The presence and interference of vegetation root systems can significantly impact sediment core length. Future research should consider incorporating this element of analysis to definitively assess the influence of root mass on sediment core length and draw more conclusive insights.

The overall low vegetation coverage observed at Wainui Repo Whenua Wetland (~1 year) likely played a role in determining the length of sediment cores. This aspect aligns with studies such as Kelleway *et al.* (2016), highlighting the intricate relationship between vegetation coverage, root systems, and sediment characteristics.

Figure 4.1: Left, Te Pā Ika Wetland (~5 years), and right, Wainui Repo Whenua Wetland (~1 year).

Table 4.1: Sediment textures and ideal bulk density ranges for plant growth, resource from the South Dakota Soil Health Coalition.

Sediment Texture	Ideal Bulk Densities for plant growth (g cm^3)	Bulk densities that restrict root growth (g cm^3)
Sand	<1.60	>1.80
Silt	<1.40	>1.65
Clay	<1.10	>1.47

The relationship between grain size and the duration since restoration has been extensively observed in ecological studies. For example, Lawrence *et al.* (2018) investigated sediment dynamics post-restoration and found that saltmarsh habitats undergoing restoration often begin with a higher proportion of fine-grained sediments (silt). Over time, there appears to be a transition towards a sediment composition that is increasingly dominated by coarser particles (sand).This shift towards coarser sediment composition with time since restoration is influenced by a range of ecological and hydrological factors that dictate sediment transport and deposition processes. Factors such as vegetation growth, water flow dynamics, and sediment supply can all contribute to this evolving sediment landscape. While detailed investigations of these factors were beyond the scope of this thesis, vegetation has obvious differences across our study sites (refer to Figure 3.1). Our sites that were either "pristine" or were over ~20 years since restoration had the highest average vegetation coverage in low and mid marsh habitats. These vegetated sites were generally dominated by oioi (*Apodasmia similis*) and/or wīwī (*Juncus kraussii*), while our most recently restored sites were generally dominated by bare ground, woody debris, and herbs such as batchelor's button (*Cotula coronopifolia)*.

As restored saltmarshes naturalize, the growth and maturation of vegetation plays a pivotal role in stabilizing sediments. The intricate and dynamic root systems of saltmarsh plants trap and bind sediments, reducing their susceptibility to erosion. Moreover, the type and density of vegetation can influence sediment accretion and determine the overall sediment texture over time such as where greater vegetation coverage leads to less erosion and increased SAR (Boorman *et al.*, 1998).

Water flow dynamics contributes significantly to the grain size increasing with time. The hydrological flows within saltmarsh habitats, influenced by tidal patterns and freshwater influx, dictates the movement of sediments. Generally, water flows in saltmarshes redistribute sediments by selectively transporting finer particles downstream while allowing coarser materials to settle. In their 2017 study, Spencer *et al.* explored the relationship between restored saltmarsh habitats and hydrological changes. Their findings unveiled a significant impact in regions experiencing fluctuations in tidal flows or adjustments in freshwater inputs due to restoration efforts. Notably, habitats subjected to new and modified hydrological flows often exhibited finer grain sizes. This phenomenon was observed, for instance, in areas influenced by tidal flood gates, exemplified in our study by our two most recently restored sites, Wainui Repo Whenua Wetland (~1 year) and Te Pā Ika Wetland (~5 years), which displayed the highest bulk density and presumed grain size. Sediment supply, another key determinant, is influenced by external sources within the catchment, such as freshwater inputs and tidal currents. Restoration activities can alter sediment transport pathways and introduce sediments of varying sizes from external sources. This influx of new sediments, combined with internal processes, contributes to the evolving composition of the substrate (French, 2019).

Restoration and acclimatisation of restored saltmarsh environments are not isolated events but ongoing processes shaped by a multitude of interacting forces. Our analysis across our six study sites at varying stages of restoration, supports this trend. We identified a weak linear relationship in mean bulk density with bulk density form recently restored sites to control sites, further supporting the notion that the composition of sediments within these habitats evolves over time, reflecting the intricate interplay of ecological and hydrological variables.

The sediment type including grain size and composition of a saltmarsh (restored or ”natural”) plays a crucial role in influencing carbon cycling within the habitat. Fine-grained sediments, such as silt and clay, generally have higher organic carbon content and provide favourable conditions for carbon sequestration. These sediments create anaerobic environments that limit microbial decomposition, promoting the accumulation of organic matter over time. On the other hand, coarse-grained sediments, like sand, may have lower organic carbon content but offer increased aeration, potentially influencing decomposition rates (Kelleway *et al.*, 2016). Additionally, sediment accumulation rates and the interaction between inundation frequency and sediment deposition further modulate carbon storage in saltmarsh ecosystems. Understanding these complex relationships is essential for comprehending the long-term efficacy of saltmarsh restoration efforts in mitigating climate change (Chmura, 2013).

4.3 Blue Carbon

4.3.1 Blue Carbon Concentration Trends in Restored Saltmarshes

The analysis of carbon concentration across surveyed sites unveils insights into the variability of carbon content across restored and control saltmarsh habitats, particularly concerning depth. Within this study, carbon concentrations followed anticipated trends, with the highest percentage found in the upper 30cm sediment cores, decreasing with depth. This pattern aligns with other work in Aotearoa by Bulmer *et al.* (2020), who emphasized the significance of the upper sediment layer in carbon storage within saltmarsh ecosystems. The heightened carbon concentrations in the top 30cm are likely due to a combination of factors. First, the surface layer experiences more active biological processes, including root activity, microbial decomposition, and organic matter turnover, contributing to elevated carbon concentrations (Howard *et al.*, 2017). Second, the upper sediment layer is influenced by plant productivity and detritus accumulation, further enhancing carbon storage near the surface. These intricate interactions among vegetation, microbial activity, and sediment processes underscore the dynamic nature of carbon concentration in saltmarsh ecosystems, emphasizing the necessity for detailed investigations into the factors influencing carbon dynamics across different sediment depths.

The vegetation, elevation, and level of inundation present within a habitat must also be considered when assessing and sampling for carbon concentration within a saltmarsh. In this study, we made our best attempt to extract a core from a low-marsh and mid-marsh zone to better capture carbon dynamics across the entire site. However, delineating our study sites often posed difficulties as younger sites generally lacked definitive morphology and vegetation. In terms of carbon concentration between low and mid marsh zones, the low-marsh sediment cores had higher observed maximum and minimum carbon concentration (21.13%, 0.29%) compared to within the mid-marsh cores (18.57%, 0.22%). Elsey-Quirk *et al.* (2011) observed a similar trend of increasing carbon concentration from high to low marsh zones, with the highest concentration between the mid and low zones, likely attributed to vegetation turnover and detritus accumulation.

There was significant variation in carbon sequestration capacities across our study sites. Matua Saltmarsh (~30 years) demonstrated the highest average and absolute carbon concentrations among all sites, showing that this site is a significant carbon reservoir. In contrast, Wainui Repo Whenua (~1 year) had the lowest mean carbon concentration, despite exhibiting a notable maximum carbon concentration between 2-4cm within the low-marsh sediment core. Kūkūwai Whakapoukōrero (~20 years) presented the minimum carbon concentration. The absence of significant linear trends across carbon concentrations, sites, and temporal scales suggests that influencing factors are complex, emphasizing the need for a detailed assessment of carbon dynamics within each site's specific context.

4.3.2 Depth-Dependent Dynamics of Blue Carbon Stocks

Total sediment organic carbon (TOC) displayed consistent trends in vertical distribution across our study sites, aligning with established patterns observed in carbon concentrations within saltmarsh sediments. Typically, mean TOC values were higher in the upper 30cm of sediment and gradually decreased with depth, in accordance with the findings of similar studies (Howard *et al.*, 2017; Kirwan *et al.*, 2010). The most significant differences in TOC were observed between the 10cm and 44cm sub-samples due to significant differences in TOC likely caused by differing levels of aerobic activity, in line with previous studies (Yuan *et al.*, 2020). Differences in aerobic conditions across saltmarsh sediment layers play a crucial role in shaping the distribution of organic carbon and other key components.

Generally, sediment layers closer to the surface exhibit higher aerobic activity due to increased exposure to oxygen. This aerobic zone, typically extending to the upper 30cm of sediment, fosters microbial decomposition of organic matter and supports plant root systems. As sediment layers experience reduced oxygen availability with depth, this leads to a shift toward anaerobic conditions. This transition has significant implications for the decomposition processes, microbial communities, and biogeochemical cycling within the sediments (Wiegert *et al.*, 1981). The variations in aerobic activity contribute to distinct patterns in TOC distribution, with higher concentrations observed in aerobic zones and decreasing levels with depth. Understanding these aerobic differences in saltmarsh sediment layers is essential for comprehending the dynamics of carbon sequestration, nutrient cycling, and overall ecosystem functioning in these critical coastal environments (Howarth & Hobbie, 1982).

Two intriguing outliers, Te Pā Ika Wetland (~5 year) and Wainui River Saltmarsh (control), challenged this trend of lower TOC being expected in the deeper levels of sediment by exhibiting unexpected peaks in TOC between 44cm and 45cm, respectively. These atypical TOC peaks suggest a potential influence of historical land use, particularly in the case of Te Pā Ika Wetland (~5 year), indicating the presence of sediment heterogeneity across the vertical column. The unexpected TOC peaks identified in this study serve as potential indicators of the intricate relationship between historical organic matter, land use and contemporary restoration efforts in saltmarsh ecosystems. Future studies exploring these variances can contribute to better understanding the complex interplay between human activities, sediment dynamics, and carbon sequestration in saltmarshes. This knowledge can guide more informed restoration strategies, considering and leveraging historical legacies to achieve sustainable and effective saltmarsh restoration outcomes in alignment with blue carbon and climate change mitigation.

4.3.3 Variations in Blue Carbon Stocks: Unveiling Patterns

The weak negative linear relationship observed between temporal variance and TOC is primarily driven by the notably high TOC levels detected within Te Pā Ika Wetland (~5 year). In this study, TOC patterns do not exhibit substantial trends that allow for conclusive insights into how TOC behaves over time across restored habitats compared to controls. Nevertheless, both our standardized and maximum depth TOC values in both restored and control sites fall within the expected range for a saltmarsh habitat globally. In the context of Aotearoa, our TOC readings are consistently high, aligning with typical values observed in saltmarsh habitats.

It appears standard for saltmarsh habitats to have TOC figures between 20t ha^{-1} and 90t ha^{-1}. Bulmer *et al.*, (2020) concluded that the Tairua estuary saltmarshes contained an average of 90t ha^{-1}, which is similar to findings of Burden *et al.* (2013; 2019) who found TOC stocks as high as 93t ha^{-1} in Tollesbury, Essex, UK. Our average standardised TOC stock across all six sites was 89.45 ± 0.31t ha^{-1}. McMahon *et al.*, (2023) similarly investigated six saltmarshes, three restored, and three natural within the Blackwater Estuary, Essex, UK and found that the TOC within restored saltmarsh (78.9 ± 27.5t ha^{-1}) environments is generally lower than in control environments (87.7 ± 8.4t ha^{-1}). Our study had standardised TOC within a similar range, however, an interesting trend is that our restored saltmarshes (94.65 ± 0.39t ha^{-1}) had a higher TOC than our control saltmarshes (79.07 ± 0.22t ha^{-1}). Overall, our study sites (both restored and control) displayed higher TOC across the mean, maximum, and minimum recordings when compared to other relevant studies (McMahon *et al.*, 2023; Burden *et al.* 2013; 2019).

The elevated TOCs observed in this study compared to previous work can be attributed to a combination of ecological, environmental, and catchment-related factors inherent to the saltmarsh habitats under investigation. The consistently high TOC levels (especially notable in Te Pā Ika Wetland (~5 years), suggest robust carbon sequestration processes within these environments. The presence of vegetation, microbial activity, and organic matter turnover in the upper sediment layers can contribute significantly to higher TOC concentrations, particularly in the top 30cm of sediment, as demonstrated by similar findings in saltmarsh ecosystems (Howard *et al.*, 2017; Kirwan *et al.*, 2010). Moreover, the catchment inputs, including nutrient runoff, allochthonous organic matter, and sediment deposition from adjacent terrestrial and aquatic ecosystems, play a crucial role in shaping TOC levels in saltmarshes (Keller *et al.*, 2012; McLeod *et al.*, 2011 In reference to the hypothesized 80:20 ratio (allochthonous to autochthonous), it is plausible that our saltmarshes primarily derive carbon from allochthonous sources, indicating a significant influence from the surrounding catchments, as hypothesised by Williamson and Gattuso (2022). This ratio underscores the importance of external inputs in contributing to the blue carbon stocks within these saltmarsh environments, shedding light on the intricate interplay between autochthonous and allochthonous carbon sources.

The historical land use and restoration activities in the study sites, combined with the unique characteristics of each site, such as hydrology, vegetation composition, and disturbance history, may have interacted with catchment inputs to influence sediment dynamics, altering carbon storage patterns over time. Understanding the intricate relationships between catchment processes and saltmarsh TOC levels is vital for comprehensive ecosystem management and effective restoration strategies. Future research could delve deeper into these site-specific and catchment-related attributes to unravel the complex interplay between environmental conditions, ecological processes, and carbon sequestration.

4.3.4 Carbon Storage: Blue Carbon's Sequestration Potential

The observed variations in CAR and SAR across the surveyed saltmarsh sites is likely due to a combination of factors, including site-specific restoration practices, environmental conditions, and historical land use. The influence of elevation, and consequently tidal inundation, emerges as a pivotal factor impacting SAR (Blum *et al.*, 2021). Saltmarsh wetlands situated at lower elevations experience greater coastal inundation and tidal effects, resulting in heightened sediment movement and dispersal. Conversely, environments at higher elevations are less susceptible to tidal influence and inundation, leading to reduced sediment transport. This elevation-related dynamic underscores the critical link between SAR and CAR; higher SAR levels correlate with increased potential for carbon accumulation, emphasizing the intricate relationship between topography, hydrology, and sedimentation processes in shaping carbon sequestration outcomes in saltmarsh ecosystems (Chastain *et al.*, 2018). The correlation between the CAR and SAR present across our study sites appears to be proportional and complimentary to each other.

Overall, CAR displayed a non-significant positive linear relationship which suggest that it increases with time since restoration. Matua Saltmarsh was the “oldest” restoration site in this study, having undergone restoration in the 1990s, the 30-year CAR was much lower (107.91 g C m^{-2} yr^{-1}) than the 20-year CAR, to ensure better comparability between sites we decided to use the 20-year CAR for Matua Saltmarsh in further analysis. The relatively long time since restoration at Matua and the associated lower CAR (161.87 g C m^{-2} yr^{-1}) and SAR (3.39 mm yr^{-1}), emphasizes the influence of restoration duration on carbon and sediment accumulation. The lower CAR and SAR may also indicate that the environment has become limited in this ability to accumulate carbon and sediment and may be experiencing higher levels of environmental stressors such as poor water quality, altered hydrology, or disturbances that hinder the natural processes of carbon and sediment accumulation.

Wainui River Saltmarsh (control), displayed the highest CAR and SAR, suggesting efficient carbon sequestration and sediment accumulation, potentially linked to favourable hydrological conditions and vegetation cover, facilitating organic matter retention. These CAR and SAR figures are suggestive of the overall health of Wainui River Saltmarsh and support it as a control site in terms of efficient carbon cycling and accumulation. Wainui Repo Whenua Wetland (~1 year), despite having a lower SAR, demonstrates a comparatively high CAR, indicating that carbon sequestration is more pronounced than sediment accumulation. This difference could be due to specific restoration interventions, variations in vegetation dynamics, or most likely in the case of this environment significant hydrological and inundation factors influencing carbon dynamics independent of sediment accretion. The weak linear relationship observed between CAR and the temporal gradient of restoration, with exceptions in Matua Saltmarsh (~30 years) and Te Awa o Ngātoroirangi (control), underscores the interactions between restoration history, environmental factors, and carbon sequestration rates in saltmarsh ecosystems.

Conclusions and Future Applications

No distinct temporal trend emerged in the comparison of blue carbon stocks between restored and control environments although it was clear that restored environments exhibited substantial carbon storage in comparison to control environments. The restored saltmarshes in this study (~1-30 years since restoration) have higher mean TOC figures compared to our control environments. The TOC and CAR data conformed to expectations reflected in other studies, affirming that restored saltmarsh habitats demonstrate a notable capacity to store and accumulate significant carbon quantities within their initial 25 years before gradually stabilising (Noll *et al.*, 2019; Burden *et al.*, 2013). This highlights the effectiveness of saltmarsh restoration efforts in enhancing carbon storage, emphasizing the long-term potential of these restored habitats as robust carbon sinks. The higher TOC in the restored saltmarshes, even within the relatively short timeframe of 1 to 20 years, suggest a rapid establishment of carbon accumulation processes following restoration, further supporting the ecological benefits of saltmarsh restoration in the context of climate change mitigation.

The heightened TOC and CAR observed in our restored saltmarshes prompt an exploration of the underlying factors driving enhanced carbon storage such as the accelerated growth and subsequent decomposition of vegetation in restored areas, with specific attention to the types of plant species thriving in these environments (Cai *et al.*, 2023). Additionally, alterations in sedimentation patterns post-restoration may play a crucial role, emphasizing the need to examine sedimentation rates and their correlation with TOC and CAR (McMahon *et al.*, 2023). Microbial activity within saltmarsh sediments, influenced by restoration activities, presents another opportunity for further investigation, shedding light on microbial processes contributing to elevated carbon figures (Santini *et al.*, 2019). Hydrological conditions, including tidal influence and water circulation, also warrant scrutiny for their impact on carbon dynamics. A thorough examination of land-use history at restored sites is essential, offering insights into past disturbances that may influence carbon storage (Burden *et al.*, 2019).

Saltmarsh habitats in Aotearoa present significant restoration opportunities, given the historical disturbances and destruction they have endured (>90%). With their intrinsic capacity to serve as blue carbon sinks and function as NBS to mitigate rising GHG's and climate pressures, it is imperative that future research on restored saltmarsh wetlands in Aotearoa places a high priority on long-term monitoring and protection. This monitoring should extend beyond the initial 20-30 years post-restoration to comprehensively track the trajectory of blue carbon dynamics. Conducting comparative studies among diverse restoration techniques is essential, as is an assessment of the broader ecosystem services delivered by restoring saltmarshes. Addressing these factors in future research should aim to deepen our understanding of the mechanisms underpinning TOC and CAR figures specific to Aotearoa's restored saltmarshes.

In conclusion, it is my sincere hope that future policymakers in Aotearoa recognize the intrinsic ecological value embedded in saltmarsh environments. This acknowledgment should extend to their demonstrated role as blue carbon sinks, a capacity that remains noteworthy even during various stages of restoration. What is particularly exciting is the prospect that these environments have the potential to persistently store substantial amounts of blue carbon over time, surpassing the capabilities of control environments in some cases. By understanding and acknowledging the enduring ecological contributions of saltmarsh habitats, we can inform and shape future policies to ensure their preservation and effective management. This commitment is essential for the well-being of our people, wider habitats, and future generations.

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