What's in our toolbox: Exploring and unlocking Canada's blue carbon potential

By

Debra Sinarta

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I would like to acknowledge that Dalhousie University sits on the ancestral and unceded territory of the Mi'kmaq. We are all Treaty people.

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Abstract

Despite growing recognition of the ability of blue carbon ecosystems to sequester and store atmospheric carbon dioxide (CO_2) for several decades, their value is often overlooked in marine management decisions and climate change policies. Integrating blue carbon into climate mitigation and marine planning requires quantification of habitat extent and carbon dynamics. Canada has the world's longest coastline and supports an extensive range of productive carbon-sequestering marine ecosystems, yet blue carbon inventories have not been established in of most of these areas. Substantial geospatial data and assessments of carbon stocks and sequestration are necessary to produce inventories, yet their availability remains limited in Canada. This study explores the information required to establish blue carbon inventories and uses IPCC's three-tiered assessment structure to begin assembling inventories in two Canadian case study regions: Owls Head Provincial Park (OHPP), Nova Scotia, and the British Columbia (BC) Northern Shelf Bioregion (NSB) Marine Protected Area (MPA) Network. Through these assessments, we demonstrate how carbon inventories can be established in situations with varying levels of data and resource availability, while developing a preliminary estimate of carbon storage in the areas. This research indicates that, while existing data and information has enabled baselines estimates for several blue carbon ecosystems, significant knowledge gaps and limitations remain. On this basis, we provide recommendations for research priorities, and insights into integrating blue carbon inventories into the management of Canada's coastlines.

Keywords: Blue Carbon, Blue carbon inventories, Atlantic Canada, Pacific Canada, Northern Shelf Bioregion, Marine protected area network, Owls Head Provincial Park

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List of Abbreviations

BCE	Blue carbon ecosystem
BCMCA	British Columbia Marine Conservation Analysis
C	Carbon
CDM	Clean Development Mechanism
CO_2	Carbon dioxide
CO_3	Carbonate
DFO	Department of Fisheries and Oceans
DIC	Dissolved inorganic carbon
DOC	Dissolved organic carbon
ESS	Ecologically Significant Species
HCO_3	Bicarbonate
InVEST	Integrated Valuation of Ecosystem Services and Trade-offs
IPCC	The Intergovernmental Panel on Climate Change
GHG	Greenhouse gas
LOI	Lost on Ignition
MCT	Marine Conservation Target
MPA	Marine Protected Area
NDC	Nationally Determined Contribution
NPP	Net primary productivity
NSB	Northern Shelf Bioregion
NbCS	Nature-based Climate Solutions
OBIA	Object-based image analysis
OC	Organic carbon
OHPP	Owls Head Provincial Park
PES	Payments for ecosystem services
POC	Particulate organic carbon
SSP	Shared Socio-economic Pathway
UAV	Unoccupied aerial vehicles
UNFCCC	United Nations Framework Convention on Climate Change
VER	Verified Emission Reduction

Chapter 1: Introduction

1.1 Background

The Intergovernmental Panel on Climate Change (IPCC) warns that, under the Shared Socio-economic Pathway (SSP) 2-4.5 scenario, projections point towards a global average temperature rise of 2.7°C (IPCC, 2022). To maintain Earth's average temperature rise beneath the 1.5°C 2015 Paris Climate Agreement target, signatory states must reduce greenhouse gas (GHG) emissions and implement mitigation strategies (UNFCC, 2018). Amongst these climate mitigation strategies is the emerging approach of Nature-based Climate Solutions (NbCS) (Nesshöver et al., 2017). NbCS are actions that promote working with and enhancing nature to help mitigate climate change (IUCN, 2020). In marine and coastal environments, NbCS support climate adaptation functions such as flood control (Hossain, 2010; Oppenheimer et al., 2022), protection from erosion (Gedan et al., 2011; Shepard et al., 2011), and enhanced soil accretion (Duarte et al., 2013; McKee & Cherry, 2009). Additionally, coastal and marine NbCS support mitigation strategies including carbon sequestration and storage (Griscom et al., 2017; Spalding et al., 2014).

The ocean has a high capacity to absorb and store carbon dioxide (CO_2) from the atmosphere. Research suggests that ocean absorbs approximately 25% of the CO_2 produced by humans, and thus can serve as a partial buffer against rising carbon emissions (Bakker et al., 2016; Gruber et al., 2019). In the marine environment, carbon is predominantly found in the form of dissolved inorganic carbon (DIC) (e.g., dissolved CO_2, carbonate (CO_3), bicarbonate (HCO_3)) (Ciais et al., 2014). Additionally, organic carbon (OC), particulate or dissolved, can be found in living plants and marine organisms or decaying matter (Thompson et al., 2017). Both organic and inorganic carbon can be stored either short-term (days to years) or long-term (hundred years to millennia); for this report, sequestration refers to the long-term storage of carbon for at least one hundred years to millennia.

The role of blue carbon in climate change mitigation has been a growing area of research in the past decade (Duarte de Paula Costa & Macreadie, 2022). The term "blue carbon" refers to the OC that is sequestered and stored long-term in marine and coastal plants, animals, and sediments (Barbier et al., 2011; Hutto, Brown, et al., 2021; Lovelock & Duarte, 2019; Nellemann et al., 2009). Vegetated coastal ecosystems, including salt marshes, seagrass meadows, and mangroves, are well-recognized for their ability to store and sequester carbon (Table 1)

(Lovelock & Duarte, 2019; Nellemann et al., 2009). These blue carbon habitats have higher rates of carbon sequestration per unit area relative to terrestrial habitats (Mcleod et al., 2011). For example, while sequestration rates of tropical forests average 4.0 (±0.5) g C m^{-2} yr^{-1}, tidal marshes can average over 200 (±24) g C m^{-2} yr^{-1} (Mcleod et al., 2011). In addition to carbon sequestration, blue carbon ecosystems (BCEs) also provide a range of other valuable benefits such as coastal protection (Spalding et al., 2014; Zedler & Kercher, 2005), enhanced socio-cultural value (Vierros et al., 2020), improved water quality (Barbier et al., 2011; Reynolds et al., 2016), and increased marine biodiversity (Smale et al., 2018).

Table 1. *Global extent and net primary productivity estimates of coastal blue carbon ecosystems.*

Ecosystem	Global Extent	Net Primary Productivity (NPP)
Salt marshes	22,000 km^2 -54,950.89 km^2 (Mcowen, 2017)	9.4 Mg C ha^{-1} yr^{-1} (Belowground) (Alongi, 2020)
Mangrove forests	138, 000 km^2- 152,361 km^2 (Inoue, 2019)	11.1 Mg C ha^{-1} yr^{-1} (Alongi, 2014)
Seagrass meadows	160,387 km^2 - 266,562 km^2 (McKenzie et al., 2020)	4 Mg C ha^{-1} yr^{-1} (Duarte et al., 2010)

Improvements of carbon sequestration and storage potential estimates have supported the inclusion of blue carbon in international climate policies and agreements, such as Nationally Determined Contributions (NDCs) (Herr et al., 2017; Herr & Landis, 2016). While recognition of blue carbon in international climate policy is progressing, blue carbon remains absent from many national emission reduction frameworks and marine management decisions (Dunn et al., 2022; Wedding et al., 2021). Additionally, blue carbon assessments still contain many knowledge gaps in their methodology, coverage, and analysis (Cavanagh et al., 2021). Increasing our understanding of carbon sequestration efficiency and storage in BCEs is a necessary step to exploring the potential of blue carbon as a natural climate mitigation option. Furthermore, before blue carbon can be integrated into national and sub-national levels management schemes, the challenges that impede its application must be addressed.

1.2. Management challenges

1.2.1 Policy implications

Canada has the world's longest coastline, that also supports an extensive range of productive carbon sequestering ecosystems. However, despite growing research, blue carbon value and functionality are rarely incorporated into national marine and climate policies (Hutto, Brown, et al., 2021; Wilson et al., 2020). Canada's recognition of blue carbon sinks in GHG accounting only began in 2020 with the federal *Strategic Assessment of Climate Change* (Government of Canada, 2020). Still, there are no federal or provincial regulations that explicitly include blue carbon into national GHG inventories. Moreover, there is a lack of implementation of blue carbon in planning and management decisions. Currently, for example, only about 1.5% of global BCEs are protected by marine protected areas (MPAs) (Sala et al., 2021), highlighting missed conservation opportunities for carbon sinks. Communicating the importance and value of BCEs to stakeholders, potential funders, and the public is therefore of importance. However, the economic valuation of ecosystem services, such as carbon sequestration, remains a heavily debated topic as many limitations still surround the validity, accuracy, and applicability of valuation schemes (Herr & Landis, 2016; Himes-Cornell et al., 2018). Detailed information relevant to BCE extent, habitat and region-specific sequestration and storage capacities is needed to support both the economic valuation and conservation of blue carbon habitats.

1.2.2 Threats to blue carbon ecosystems

BCEs are under significant pressure from anthropogenic activities and have experienced extensive habitat loss in recent decades (Pendleton et al., 2012; Worm et al., 2006). Within the last 50-100 years, approximately 25-50% of vegetated coastal ecosystems have disappeared worldwide (Mcleod et al., 2011). In British Columbia, around 70% of tidal wetland habitats have been lost during the colonial period due to urban and agricultural development (Government of British Columbia, 1978). Similarly, on the Canadian East Coast, half of salt marshes across Nova Scotia have been lost since the 1700s, mainly due to dyking (Government of Nova Scotia, n.d.). The degradation of these vital habitats limits their ability to sequester and store carbon (Wedding et al., 2021; Young et al., 2021), and can result in the potential release of large amounts of previously sequestered carbon (Macreadie et al., 2021; Pendleton et al., 2012). The global loss of mangrove, salt marsh, and seagrass ecosytems, for instance, can release an estimated 0.15-1.02 billion tons of CO_2, annually (Pendleton et al., 2012). Conversely, the conservation of these ecosystems can help optimize their climate mitigation potential by ensuring that carbon remains stored long-term (Mcleod et al., 2011). Therefore, identifying and monitoring threats to BCEs can better describe how disturbances influence carbon sequestration and storage capacity.

1.2.3 Knowledge gaps

Blue carbon inventories form the foundation of marine policies and management decisions aiming to maximize blue carbon potential. A blue carbon inventory refers to a catalogue of information on the location, extent, sequestration rates, and carbon storage of BCEs (Carlson, 2020). The strength of the inventory will ultimately depend on completeness and quality of this data (Holmquist et al., 2018). Unfortunately, current approaches to mapping and measuring carbon related processes are unrefined and often expensive to conduct (Postlethwaite et al., 2018). Consequently, geospatial data and carbon storage estimates in Canada remain sparse and are limited to select habitat types and spatial scales. More specifically, the available blue carbon data for British Columbia are largely outdated or incomplete (Carlson, 2020). And although Nova Scotia's inventory of salt marsh habitats is more comprehensive than the one in British Columbia, relevant carbon storage and sequestration information is still missing and other BCEs are largely ignored (Carlson, 2020). Expanding methods of identifying blue carbon areas and assessing carbon-related processes are necessary steps in establishing comprehensive blue carbon inventories for Canada's Atlantic and Pacific coasts.

1.3 Research questions and objectives

This research project aims to explore the current approaches for building blue carbon inventories and demonstrate how inventories can be established at varying degrees of data availability. This study aims to grow blue carbon inventories for Canada's Atlantic and Pacific coasts using Owls Head Provincial Park (OHPP), Nova Scotia, and the British Columbia Northern Shelf Bioregion (NSB) MPA Network as case study regions. Overall, this research has the following objectives:

1. Explore current approaches and limitations for establishing blue carbon inventories

2. Demonstrate ways to contribute to blue carbon inventories with varying degrees of available information
3. Recommend improvements for how future assessments can fill in data gaps and support more comprehensive blue carbon inventories

To achieve these objectives, this project reviewed literature on the approaches used to map BCEs, assess carbon sequestration and storage, and value blue carbon. Additionally, an application of IPCC's three-tiered blue carbon inventory approach was used to evaluate how carbon-related processes can be assessed based on the available data within the case study regions. Based on findings from the literature review and tiered analysis, knowledge gaps and limitations were identified and compiled to recommend improvements for future blue carbon inventories.

Chapter 2: Exploring our toolbox—A review of blue carbon habitats and inventory tools

2.1 Blue Carbon

2.1.1 Carbon sequestration and storage

BCEs make significant contributions to global carbon sequestration (Mcleod et al., 2011; Suyadi et al., 2020). Carbon sequestration generally refers to the process of plants capturing atmospheric CO_2 and fixing it into organic carbon, such as biomass, through photosynthesis (Moomaw et al., 2018; Tang et al., 2018). Coastal blue carbon habitats sequester carbon from the atmosphere and ocean water and store the fixed organic carbon in above ground biomass of the plants (e.g., stems, leaves), below ground root systems, and non-living biomass (e.g., soils, sediments) (Figure 1) (Howard et al., 2017; Zhang et al., 2017). Additionally, carbon from multiple sources and pathways can be stored in seafloor sediments— most notably from macroalgae and burial of marine biota such as whales and fish (Figure 1) (Bax et al., 2022; Bayley et al., 2021; Tang et al., 2018). The enhanced carbon sequestration and storage abilities of these ecosystems can be attributed to several properties. Submerged or partly submerged vegetation can reduce wave energy (Koch et al., 2009), promoting sedimentation (Gacia & Duarte, 2001) and trapping of suspended sediments (Barbier et al., 2011). Moreover, high net primary productivity and enhanced vertical sediment accretion in BCEs contribute to high OC deposits (Kennedy et al., 2010). Still, as sequestration rates and storage capacities have considerable variation, it is important to understand the unique flows of carbon across habitats to produce robust blue carbon assessments.

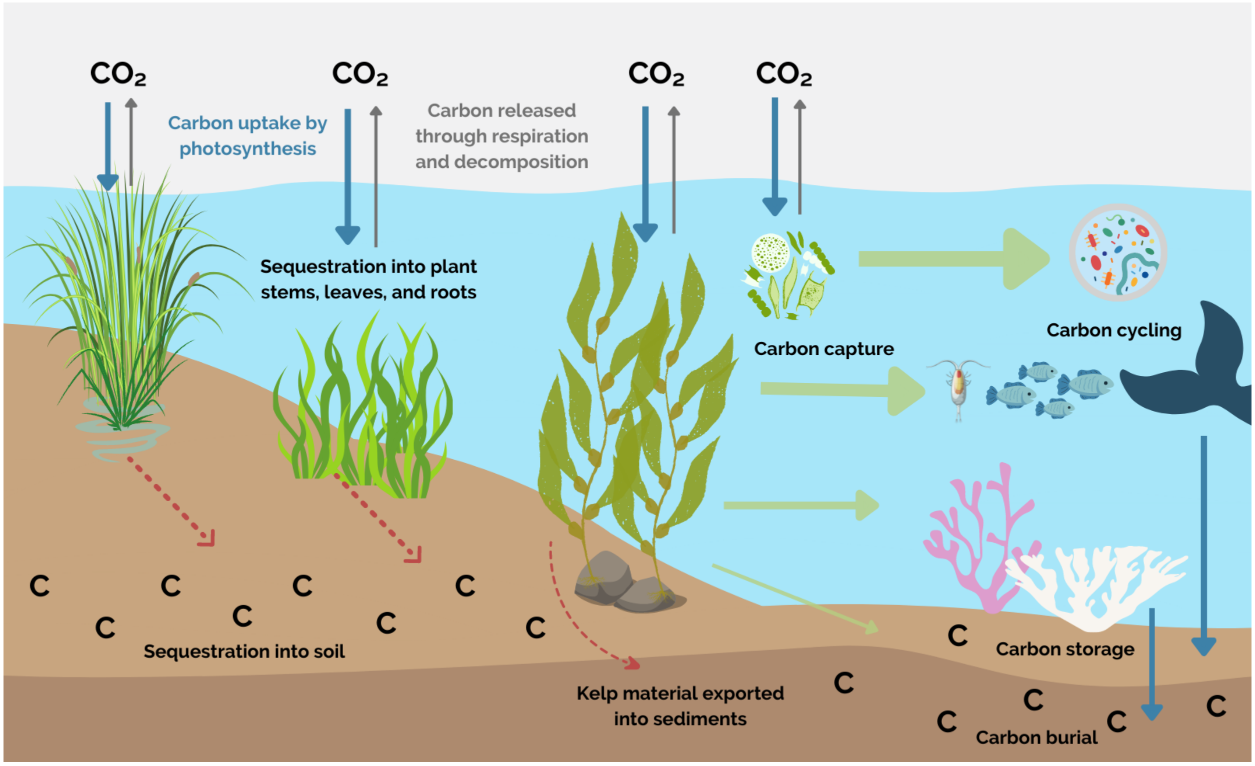

Figure 1. *Carbon sequestration process in blue carbon ecosystems (left to right: salt marshes, seagrasses, macroalgae, phytoplankton, and marine biota).*

2.1.2 Blue carbon ecosystems

Blue carbon ecosystems can be distinguished by three main characteristics: 1) significant capacity to remove CO_2 (Santos et al., 2021); 2) ability to hold and store carbon for centuries if left undisturbed (Moomaw et al., 2018); and 3) amenability to management actions (e.g., restoration, reducing anthropogenic nutrient inputs) that can enhance carbon storage (Macreadie et al., 2017). Vegetated coastal ecosystems including salt marshes, seagrass meadows, and mangroves are well-recognized for their ability to store and sequester carbon (Lovelock & Duarte, 2019; Nellemann et al., 2009). Despite only making up approximately 0.07-0.22% of Earth's surface (Spivak et al., 2019), these ecosystems capture and store up to 70% of the carbon permanently stored in marine systems (Nellemann et al., 2009). In addition, kelp forests and seabed sediments are gaining recognition for their potential contributions to global carbon sequestration (Atwood et al., 2020; Nellemann et al., 2009). Moreover, although other blue carbon stores such as phytoplankton, marine fauna (e.g., whales), and coral reefs exist, they do not have the recognized capacity to fix CO_2 long term (Lovelock & Duarte, 2019). The following section will explore BCEs further, focusing on the carbon sequestering ecosystems found along Canada's coastlines.

2.1.3 Blue carbon ecosystems in Canada

Blue carbon research, to date, has emphasized mangrove forests and tidal marshes; blue carbon habitats in temperate regions, including Canada, are therefore often underrepresented (Lovelock & Duarte, 2019; Mcleod et al., 2011). Seagrass and salt marsh ecosystems are two well-recognized coastal blue carbon habitats in Canada, and thus, will be focal habitats in this research (Nellemann et al., 2009; Rogers et al., 2019). In addition, this study will also investigate the two major oceanic carbon sequestration processes: macroalgae and seabed sediments.

2.1.3.1 Seagrass

Seagrasses are the only flowering plants in the marine environment (Figure 2). Their estimated global coverage extends around 160,387 km^2 - 266,562 km^2, occupying largely shallow coastlines of tropical and temperate regions (Fourqurean et al., 2012; McKenzie et al., 2020). There are approximately 60-70 species of seagrasses worldwide that form complex habitats in intertidal and subtidal zones, some down to 12 m depth (DFO, 2009). Seagrasses provide vital ecosystem services including sediment stabilization, shoreline protection, critical habitats for aquatic species, nutrient cycling, and carbon storage (Barbier et al., 2011; Duffy et al., 2015; Orth et al., 2006); thus, their protection and management are worldwide priorities (DFO, 2009; Unsworth et al., 2019).

Although seagrasses only make up around 0.1 to 0.2% of total global seafloor area, they capture up to 18% of carbon permanently stored within marine ecosystems (Duarte et al., 2013; Fourqurean et al., 2012). This high carbon capture rate and long-term storage potential is supported by high primary productivity, low herbivory rates and high particle capture abilities of seagrasses (Borum et al., 2005; Duarte et al., 2013). Current seagrass carbon inventories, however, are largely based on data derived from tropical and subtropical regions and rely on extrapolating global scale measurements (Douglas et al., 2022; Miyajima et al., 2017). This regional bias could ultimately result in information that is unrepresentative of dominant temperate species and produce inaccurate global carbon budget estimates. For instance, core samples of temperate eelgrass meadows reveal average carbon storage and sequestration rates that are considerably lower than global estimates that largely capture tropical seagrass species (4.6 to 93.0 g OC m^{-2} yr^{-1}) (Postlethwaite et al., 2018; Prentice et al., 2020). Thus, more robust, and site-specific assessments are needed to fill this discrepancy.

Figure 2. *Eelgrass meadow at Owls Head Provincial Park (left) and underwater photo of eelgrass meadow* (Bostrom, n.d.; Winkler, 2022).

In Canada, eelgrass (*Zostera marina*) is the dominant species of seagrass that occupies the shallow coastal waters of all three coastlines (Green & Short, 2003). Eelgrass meadows are generally monospecific with occasional mixing with other seagrass such as widgeon-grass (*Ruppia maritima*) in the Atlantic or Japanese eelgrass (*Zostera japonica*), an invasive species, in the Canadian Pacific (Shafer et al., 2014; Wong et al., 2013). Bioregions across Canada are characterized by a unique range of abiotic and biotic factors such as salinity, sea temperature, wave energy, sediment type, and anthropogenic stressors (Namba & Nakaoka, 2018; Röhr et al., 2018). These variations influence the distribution, species composition, resilience, and morphology of eelgrass ecosystems, and therefore, are important to consider when developing management and conservation strategies (DFO, 2009; Wong, 2018).

The health and resilience of eelgrass ecosystems are increasingly threatened by anthropogenic climate change, and activities including nutrient loading, coastal development, commercial fishing, invasive species, and aquaculture (Murphy et al., 2019; Orth et al., 2006; Wilson & Lotze, 2019). Although *Z. marina* is recognized as an Ecologically Significant Species (ESS) by the Department of Fisheries and Oceans (DFO), protection is not guaranteed, and explicit management efforts remain limited. Protected and continuous seagrass meadows are known to have higher carbon storage levels and sequestration potential in comparison to patchy and degraded seascapes (Asplund et al., 2021; Macreadie et al., 2015). Thus, preserving the connectivity of and documenting threats to eelgrass ecosystems across Canada should be a conservation priority. Moreover, despite being a widely distributed and abundant habitat type across Canada's bioregions, there are no published assessments of seagrass distribution or extent (McKenzie et al., 2020). The collection of comprehensive data on seagrass extent and carbon dynamics are vital steps to enhancing Canada's blue carbon inventory.

2.1.3.2 Salt Marsh

Salt marsh ecosystems have a global coverage of around 51,000 km^2, occupying deltas, estuaries, and low-lying coastal areas of sub-tropical and temperate regions (Figure 3) (Callaway et al., 2012; Siikamäki et al., 2013). They are characterized by mineral soil, fluctuating water levels, and sparse vegetation including goose grasses (*Puccinellia maritime)*, cordgrass (*Spartina patens*), sedge (*Cyperaceae*), and low shrubs (Alberta Environment and Sustainable Resource Development, 2015; Roberts & Robertson, 1986; Selby et al., 2022). These plant community assemblages are controlled by tide heights, flooding regimes, salinity levels, and soil drainage (Bowron et al., 2012). Positioned uniquely along the transitional zone between land and sea, salt marshes provide valuable ecosystem services including coastal flood protection, nutrient cycling, nursery habitats, and carbon sequestration (Barbier et al., 2011; Himes-Cornell, Grose, et al., 2018; Möller et al., 2014; Shepard et al., 2011). Increasing our understanding of these ecosystem services is necessary to design management and conservation strategies that appropriately recognize natural capital.

Across vegetated coastal blue carbon habitats, salt marshes have the highest mean carbon storage per unit area (58.78 ± 19.30 Mg C ha^{-1}) (Douglas et al., 2022). Their high carbon sequestration efficiency can be attributed to high primary productivity, strong sediment deposition, low decomposition rates, and continuous burial rate of vegetation (Callaway et al., 2012). Unfortunately, this sequestration potential can be hindered by anthropogenic activities such as coastal development, agriculture, dyking and draining of areas, shoreline armouring, and sea level rise (Mcleod et al., 2011; Pendleton et al., 2012; Roughan et al., 2018). Still, current knowledge gaps stall the conservation of salt marsh habitats, specifically: lack of tidal marsh sampling, limited estimates of areal extent, insufficient carbon dating methods, ambiguous wetland classification schemes, and imprecise carbon accumulation calculations (Chastain et al., 2021; Duarte et al., 2013; Howard et al., 2017; Ouyang & Lee, 2014). Current global estimates for salt marsh extent have around 20-fold uncertainty (22,000-to 400,000 km^2 range); discrepancies in these estimates are partly due to the bias towards regions such as Europe and underrepresentation of other regions including Canada (Chastain et al., 2021). For instance, an assessment of a salt marsh ecosystem in British Columbia revealed carbon stocks and accumulation rates 45% lower than global estimates (Gailis et al., 2021). Therefore, collecting more site-specific carbon stock estimates will better reflect local-scale dynamics and ultimately support more accurate carbon accounting.

Figure 3. *Salt marsh at Boundary Bay, British Columbia (left) and salt marsh in Cole Harbour, Nova Scotia (right)* (Murray, 2013; Taylor, n.d.).

2.1.3.3 Macroalgae forests

Macroalgae ecosystems are an emerging area of interest within blue carbon research and have increasingly been recognized for their contribution to carbon sequestration (Hill et al., 2015; Krause-Jensen et al., 2018). Macroalgal species, such as kelp, are photosynthetic, fast-growing marine macrophytes that are predominantly found on rocky shorelines (Duarte, 1995). Macroalgae have high primary productivity rates, slow decomposition rates, and slow organic matter turnover rates— all characteristics that support their ability to sequester carbon (Duarte, 1995; Krause-Jensen et al., 2018). However, for macroalgae to contribute to long-term removal and storage, the absorbed carbon must ultimately end up buried in marine sediment (Santos et al., 2021).

Macroalgae export approximately 43% of their production as particulate organic carbon (POC) or DOC (Duarte & Cebrián, 1996). The carbon absorbed by kelp will re-enter the carbon cycle through fronds and tissues breaking off or end up buried in deep-sea sediments either near to the source or several kilometers away (Krause-Jensen et al., 2018; Santos et al., 2021). Thus, although kelp forests are unable to grow their own organic-rich sediments, they can still act as carbon "donors" through the export and burial of detritus into ocean sediments (Duggins et al., 1989; Hill et al., 2015; Krause-Jensen et al., 2018). New methods are needed to trace donor and sink macroalgal habitats and measure burial rates before integrating their contributions into existing climate policies. And while it is important not to over-promise carbon sequestration potential until knowledge gaps are filled, it is valuable to explore the potential of macroalgae in future blue carbon frameworks.

Figure 4. *Bull kelp off the coast of Vancouver Island (left) and Giant kelp forest in Haida Gwaii (right)* (Miller, n.d.; Norman, n.d.).

On the West Coast of Canada, bull kelp (*Nereocystis luetkeana)* and giant kelp (*Macrocystis pyrifera*) are the dominant kelp species (Figure 4) (Coyer et al., 2001; Druehl, 1968). Atlantic Canada, on the other hand, is dominated by laminarian kelp such as oarweed (*Laminaria digitata*), sugar kelp (*Saccharina latissima*), and winged kelp (*Alaria esculenta*) (Wilson et al., 2015). These macroalgae communities carry significant commercial, ecological, and socio-cultural value. For instance, the coast of British Columbia hosts at least 190 km^2 of kelp forests that are estimated to sequester at least 7,755 tons of CO_2 per year (Lang-Wong et al., 2022). Yet, kelp populations on both East and West Coasts have shown signs of decline (Filbee-Dexter et al., 2016; Krumhansl et al., 2016; Starko et al., 2019; Teagle & Smale, 2018) and continue to be threatened by stressors such sediment loading, as well as significant shifts in sea surface temperature, salinity levels, and wave exposure (Markel & Shurin, 2015; Watson & Estes, 2011). Understanding of the local variabilities of and pressures on kelp ecosystems is needed to successfully inventory the extent and biomass of these species.

Despite historical data on kelp extent in Canada, existing shoreline and aerial kelp surveys only cover a fraction of Canada's Pacific and Atlantic coastlines, and long-term studies that track kelp forest extent and biomass are rare (Merzouk & Johnson, 2011; Sutherland et al., 2008). Several current knowledge gaps include overcoming remote sensing challenges to map kelp forest extent, tracing carbon pathways from source to sediment sinks, understanding how oceanographic variables influence sequestration capacity, and measuring carbon fluxes between the atmosphere and the ocean (Gallagher et al., 2022). Close monitoring and having a well-established inventory of kelp forest extent and abundance, and regional-level carbon storage measurements can guide decision-makers towards kelp conservation and reforestation efforts.

2.1.3.4 Seabed ecosystems

One of Earth's largest carbon reservoirs is found in marine sediments (Atwood et al., 2020) (Figure 5). Current measurements estimate that the top meter of ocean sediment stores around 2,322 petagrams of carbon (Pg C), which exceeds carbon storage estimates for terrestrial soils (Atwood et al., 2020). The importance of sequestration pathways to seafloor biodiversity and sediment burial is becoming more evident (Atwood et al., 2020; Barnes et al., 2018; Lee et al., 2019; Sala et al., 2021). However, while our understanding of vegetated coastal blue carbon ecosystems is advancing, our understanding of carbon stocks in marine sediment is very limited (Atwood et al., 2020; Laffoley, 2020). Despite generally low carbon concentrations found in the deep-sea, the vast geographical extent of deep-sea ecosystems (approximately 85% of all seafloor area) (Harris et al., 2014) explains why around 84% of total carbon stored in marine sediment is found there (Estes et al., 2019; Lee et al., 2019). Seafloor biodiversity, including that found along seamounts, can also act as blue carbon pathways by fixing carbon from the water column to build their body tissue and skeleton, and eventually being buried in benthic sediments

(Barnes & Sands, 2017; Douglas et al., 2022; Nolan et al., 2017). (Morley et al., 2022). A greater knowledge of burial rates and related source-to-sink pathways is still necessary to fully understanding the storage capacity of seabed ecosystems.

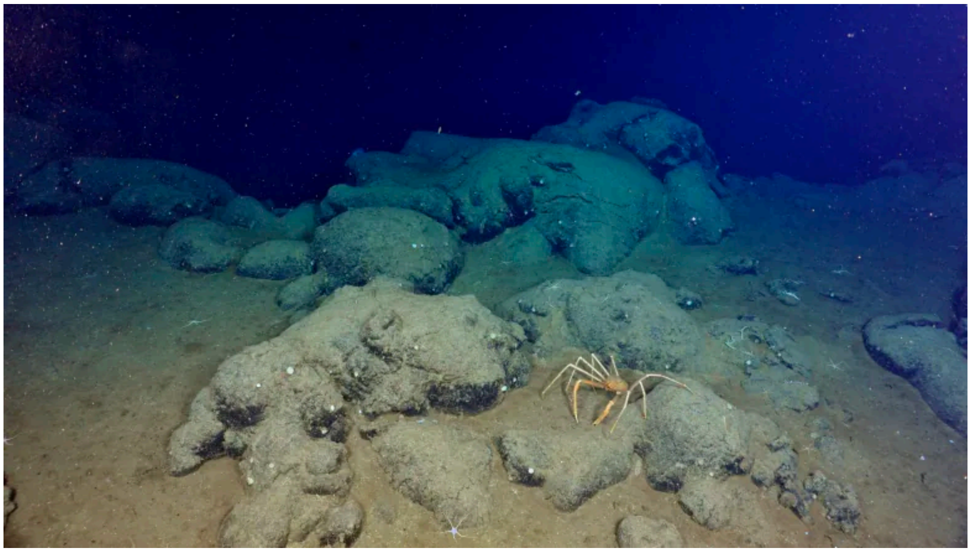

Figure 5. *Deep ocean floor off the west coast of Vancouver Island* (Ocean Networks Canada/CSSF, 2021).

If left undisturbed, carbon stocks in the seabed can be stored indefinitely. Unfortunately, destructive anthropogenic activities (e.g., deep-sea mining, oil and gas exploration, bottom trawling) can release large quantities of stored carbon into the water column and remineralize it to CO_2 (Bianchi et al., 2016). Sala et al. (2021) estimate that 0.6 to 1.5 gigatons of carbon per year is released from the seabed due to destructive activities; this is equivalent to global emission released from the entire aviation sector (918 million tonnes in 2019). To help protect carbon stocks from future threats, it is important to consider developing management plans that protect and preserve unique benthic habitats and species. Currently, only approximately 2% (48 Pg C) of carbon stocks in marine sediments occurs in MPAs that prohibit extractive commercial activities (Sala et al., 2021). Although conservation tools, such as MPAs, will not eliminate all potential disturbances to carbon stocks in marine sediments, they can help limit and alleviate the impacts of existing threats. Moreover, quantifying seabed carbon and assessing the impacts of human activities on the seafloor can support conservation of these carbon stocks.

2.2 Establishing Blue Carbon Inventories

Establishing blue carbon inventories is an essential step in scaling up the management of carbon sequestering ecosystems and developing concrete policies that consider the value of blue carbon. Blue carbon inventories can be established at various spatial scales (e.g., site-specific, regional, national, and global) and generally require the following information: 1) spatial distribution and extent of ecosystems; 2) assessment of carbon storage and sequestration; and 3) calculation of fluxes, net sequestration, or potential release of carbon (Carlson, 2020; Howard et al., 2014). This inventory can then be used in management decisions and policies related to the valuation and protection of carbon stocks. This section aims to review the current range of methods and approaches used to establish blue carbon inventories through mapping, assessing, and valuing carbon storage and sequestration. This will outline the current trends and limitations of blue carbon accounting.

2.2.1 Spatial data, mapping, and analysis

Spatial data are critical components of conservation and management decisions (Howard et al., 2014; Mustard et al., 2012). The strength of blue carbon inventories largely depends on the accuracy and reliability of spatial information such as ecosystem extent, habitat distribution, and species location (Carlson, 2020). Currently, however, the total extent and area of BCEs on Canada's coastlines remains unknown (Murphy et al., 2021), Moreover, mapping BCEs can be costly, time-consuming, and some blue carbon habitats are hard to access when using traditional field survey methods (Howard et al., 2014). Therefore, reliable methods of acquiring spatial information are essential to support long-term mapping and monitoring of blue carbon habitats.

2.2.1.1 Remote sensing technologies

Remote sensing has made significant contributions to blue carbon research through producing data necessary to map and monitor BCEs in a time-efficient, repeatable, accurate, and scalable way (Hossain et al., 2015; Howard et al., 2014; Johnston, 2019; Pham et al., 2019). Remote sensing can be used to monitor habitat change (Lanceman et al., 2022), identify blue carbon conservation priority areas (Mcleod et al., 2011), and when integrated with field surveys, can be used to estimate carbon stocks (Kwan et al., 2022) (Table A1, Appendix A). Additionally, the combination of geospatial information and biodiversity observations (e.g., species presence and species richness) can produce useful tools, such as species distribution models (SDMs). SDMs can be effective for monitoring blue carbon habitat loss (Zhong et al., 2021) and predicting distribution changes under climate scenarios (Jayathilake & Costello, 2020).

Unoccupied aerial vehicles (UAVs), such as drones, are now a common method of mapping BCEs. UAVs can collect data at finer spatial resolutions than commercial satellites and can surveys of challenging or hard to access ecosystems such as kelp forests, narrow wetlands, and small seagrass patches (Duffy et al., 2018; Murfitt et al., 2017; Ozesmi & Bauer, 2002). Additionally, UAVs are adjustable for collecting optimal mapping of BCEs under different environmental conditions (O'Neill & Costa, 2013). For example, under ideal environmental conditions (e.g., cloud cover, tidal height, sun angle, wind speed), UAVs were able to map eelgrass sites of unideal conditions (e.g., sparge eelgrass coverage, high cloud cover), with high confidence (Nahirnick et al., 2019) (Table 2). Similarly, aerial photographs can be an accessible approach to generate foundational spatial datasets. For instance, St-Pierre & Gagnon's (2020) survey of subtidal kelp demonstrates how supervised classifications of aerial imagery can produce maps with overall accuracies of up to 90% (Table 2).

Satellite imagery is another widely used data source for mapping blue carbon habitats. Some satellite datasets (e.g., Landsat, MODIS, SRTM) produce large scale global coverage that can be especially useful for classifying and monitoring ecosystem dynamics over time (O'Neill & Costa, 2013; Pe'eri et al., 2016; Schroeder et al., 2019; St-Pierre & Gagnon, 2020). Satellite imagery can also provide multi-sensor, high spatial resolution information that is useful for mapping habitat extent (Beland et al., 2016; Nijland et al., 2019). For instance, O'Neill and Costa (2013) utilized IKONOS satellite imagery, a four-band multispectral satellite sensor, to accurately map and monitor the distribution of eelgrass ecosystems that will support conservation efforts in the Gulf Islands National Park Reserve, British Columbia (Table 2).

Another common approach to producing robust datasets is to use both active (e.g., Radio Detection and Ranging (RADAR); Light Detection and Ranging (LiDAR); Sound Navigation Ranging (SONAR)) and passive (e.g., spectrometer, radiometer, imaging radiometer) remote sensing systems, and multiple sensors (Howard et al., 2014). Collin et al (2009), for example, demonstrate how coastal salt marsh habitat and land cover maps can be created using a combination of single topographic and bathymetric LiDAR surveys (Table 2). Remote sensing imagery can be further combined with field observations and local knowledge. Wilson et al.

(2019), for instance, took an integrated approach by using remote sensing data and local ecological knowledge systems to produce eelgrass and macroalgae habitat maps of the southern shore of Nova Scotia (Table 2).

Table 2. *Remote sensing and analysis methods used to map blue carbon habitats in Canada.*

Mapping method	Study year	Habitat	Region	Analysis method	Reference
High spatial resolution airborne (AISA) and satellite (IKONOS) imagery	July-August 2008	Eelgrass	Gulf Islands National Park Reserve, Canada	Maximum likelihood classification	O'Neill & Costa, 2013
SPOT 6/7 satellite imagery; field surveys	July 2015	Eelgrass and benthic	Nova Scotia, Canada	Maximum likelihood classification; ISOCLUST	Wilson et al, 2019
False color near-infrared aerial imagery	2001 to 2005	Eelgrass	Oregon, USA	Hybrid image classification	Young et al, 2010
Hyperspectral data collected from remote sensing platforms (e.g., airborne or satellite) or in situ (e.g., buoys)	1996-2012	Eelgrass and macroalgae	New Hampshire, USA	ISODATA/ CLUSTER classification	Pe'eri et al., 2016
Unoccupied Aerial Systems (UAS)	June-August 2017	Eelgrass	British Columbia, Canada	Object-based image analysis (OBIA); eCognition Developer	Nahirnick et al., 2018
Unoccupied Aerial Vehicle (UAV); subtidal field surveys	June 2016	Seagrass	Clayoquot Sound, British Columbia	Manual segmentation on ArcGIS	Postlethwaite et al., 2018

__Table 2__ continued

Mapping method	Study year	Habitat	Region	Analysis method	Reference
WorldView 3 satellite imagery; kayak field surveys	August 2016	Macroalgae	Cowichan Bay, British Columbia (case study)	Object-based image analysis (OBIA); eCognition Developer	Schroeder et al., 2019
Digital aerial photographs (acquired on board a helicopter); satellite (SPOT 7)	August 2016	Macroalgae	Mingan Archipelago ,Canada	Unsupervised pixel, supervised classification, and visual classification	St-Pierre & Gagnon, 2020
Scanning Hydrographic Operational Airborne LiDAR Survey (SHOALS)	July 2006	Salt marsh	Gulf of St. Lawrence, Canada	Maximum likelihood algorithm	Collin et al., 2010

2.2.1.2 Analysis and classification methods

Analysis and classification methods used for remote sensing and spatial data have greatly advanced in recent years, making it easier to access and analyze remote sensing data. Pham et al.'s (2019) review of blue carbon remote sensing analyses revealed that the most common classification methods are supervised classification, unsupervised classification, object-based image analysis (OBIA), and sub-pixel classification. The use of software such as eCognition Developer and Image Classification Wizard (ArcGIS Pro), as seen in studies by Nahirnick et al., (2018) and Schroeder et al., (2019), have significantly improved the ease and accuracy of image classification processes. Classification algorithms, such as maximum likelihood classification and linear discrimination analysis, are other common approaches that can be integrated or used independently to generate blue carbon habitat maps (Collin et al., 2010; O'Neill & Costa, 2013; Wilson et al., 2019). Young et al. (2017), for example, developed a hybrid technique of unsupervised and pixel-based classifications to process aerial images of eelgrass habitats. Additionally, there is a growing body of research aiming to improve the accuracy and accessibility of classification algorithms through various techniques such as deep learning methods (Pham et al., 2019). This growing knowledge base can significantly improve the accuracy of detecting and monitoring changes in the extent or distribution of BCEs.

2.2.1.3 Field-based methods

Field sampling and snorkel or dive surveys are traditional, albeit labor-intensive, methods of mapping and assessing blue carbon habitats (Gagnon et al., 2004; van Rein et al., 2009). Field surveys can support remotely sensed data with ground referenced data, also known as ground truthed data (Howard et al., 2014). The integration of field and remotely sensed data is an important validation tool and key component in understanding the spatial and temporal distribution of carbon stocks (Nagai et al., 2020). While remote sensing may be more efficient than traditional field methods, remotely sensed data and analysis methods without ground truthing can misclassify habitats and land cover and fail to accurately define smaller habitat features (Manson et al., 2001). In contrast, survey plots can measure parameters such as species composition, biomass, and canopy height and cover (Howard et al., 2014). This field data is valuable for validating and calibrating remote sensing data and can help establish more detailed blue carbon inventories (Roelfsema & Phinn, 2013; Satyanarayana et al., 2011). Moreover, the collection of ground truthed data provides opportunities to involve local communities and enhance capacity building (Altamirano et al., 2010; Howard et al., 2014). Building community relationships through field work can be valuable for fostering long-term support for blue carbon projects.

2.2.1.4 Limitations

Despite the advances in remote sensing technologies over the past years, each individual approach also involves limitations that need to be considered. A large disadvantage to passive remote sensing techniques, for example, is its reliance on sunlight for high-quality imagery (Howard et al., 2014). Factors such as cloud cover and light availability can greatly limit data quality and availability, especially in regions such as the tropics where these conditions are common. For example, Nahirnick et al.'s (2018) survey of eelgrass meadows identifies environmental variables such as sun glare, water turbidity, and cloud cover, that can limit an instrument's ability to capture clear imagery. These remote sensing and mapping challenges vary across habitat types. In the case of mapping macroalgae, for instance, much kelp biomass is found in low-lying, or understory regions that make it especially difficult to map from aerial imagery (Schroeder et al., 2019). Therefore, mapping macroalgal communities may require hyperspectral remote sensors to adequately capture the complexity of kelp assemblages and minimize environmental interferences (e.g., glare, water column depth) (Adão et al., 2017). With all remote sensing methods, ground truthing is also required to validate and improve accuracy of remotely sensed data (Nagai et al., 2020). In addition to these technical barriers, remote sensing sensors can be expensive, and the classification process can be a very time-consuming one that often requires specialist knowledge (Young et al., 2010).

Field collection methods are limited by site accessibility, resource availability, and desired map product or application of data (Andréfouët, 2008). While field-based surveys can produce valuable information on species composition and biomass, they also are time-consuming, labor-intensive, and have limited spatial coverage (Krumhansl et al., 2016). Wetland habitats, for instance, are often inaccessible for field sampling and are generally very time-consuming to survey (Mahdavi et al., 2018). Moreover, submerged blue carbon habitats, such as seagrass, are challenging survey due to limitations such as environmental constraints (e.g., water column depth, clarity) and technical capacities (e.g., underwater equipment, and safety) (Roelfsema et al., 2015). Nevertheless, the limitations to field-based and remote sensing methods will vary with its application to specific BCEs and thus, methods could consider incorporating multiple mapping techniques to support spatial inventories.

2.2.2 Assessing blue carbon storage and sequestration

Measuring and establishing an inventory of carbon stocks and sequestration is necessary to translate blue carbon value into concrete policies and conservation efforts, such carbon markets (Carlson, 2020). Carbon markets are trading systems that facilitate selling and buying of carbon credits (Friess et al., 2022; Sapkota & White, 2020). Compliance markets, such as the Clean Development Mechanism (CDM), is the primary tool for signatory states of the Kyoto Protocol to meet emission reduction goals through carbon projects (Ayostina et al., 2022). Alternatively, voluntary carbon markets, such as carbon offset credits or Verified Emission Reductions (VERs), are purchased under independent trading schemes and open for any country or company (private, local, regional) interested in blue carbon projects without further obligations (Kuwae et al., 2022).

To guide replicable and comparable assessment approaches, the IPCC has established a three-tiered method to quantify current and future blue carbon stocks and sequestration (Table 3) (Howard et al., 2014). Each tier was created according to the level of detail, degree of uncertainty, and accuracy of the carbon stock assessment. Data and resource availability and purpose of the analysis will ultimately determine the appropriate level of assessment for a specific project.

Table 3. *IPCC analysis tiers that can be used to develop a blue carbon inventory adapted from Howard et al., 2014 and Hutto et al., 2021.*

Tier	Data Requirements	Description	Purpose
1	Global IPCC default values	Least accurate and lowest levels of certainty Based on IPCC default activity data and emissions factors The range of error +/-50% for above ground pools and +/-90% for below ground pools	Provides a rough estimate of the amount of carbon stored and annually sequestered for a given site Creates foundational knowledge and awareness of blue carbon
2	Country-specific data for key factors	Includes aspects of site-specific or country-specific data Increased accuracy and resolution from Tier 1 assessment	Provides increased clarity on the amount of carbon stored and sequestered for a given area Supports restoration and conservation projects.
3	Site-specific carbon stock inventory	Requires direct, site-specific data for carbon stocks in each component of the ecosystem Repeated measurements of carbon stocks required over time Can be provided through direct measurements or modeling.	Entry into the carbon market Produce highly accurate information on the amount of carbon stored and sequestered for a given site

Tier 1 assessments are global and based on the IPCC default estimates of carbon stocks and emission factors and calculated by multiplying the ecosystem area extent by the global mean carbon stock value. Compared to the other tiers, this is the least costly assessment method and can provide rough, high-level estimates of carbon storage and sequestration. Although this tier can serve as useful foundational knowledge, this level of assessment will produce carbon inventories with the least accuracy and greatest range of error.

A Tier 2 assessment utilizes region-specific carbon stock data for blue carbon ecosystems, rather than global estimates. Using regional estimates will increase the accuracy and resolution of national carbon inventories. For example, a recent survey of kelp beds off the coast of BC used carbon storage ratios and conversion factors of kelp species specific to the region to calculate carbon storage and sequestration (Lang-Wong et al., 2022). While this level of assessment still carries large margins of error, it can benefit marine managers to better understand and communicate blue carbon value to inform policy and planning.

Tier 3 assessments use site-specific and repeated measurements of carbon stocks that can provide estimates of carbon fluxes within the area through long-term measurements. Postlethwaite et al.'s (2018) assessment of seagrass carbon storage and accumulation rates using sediment cores is an example of the site-specific quantification of carbon dynamics. Although this level of assessment will produce the most accurate calculations of carbon storage, it remains the most resource and time intensive approach.

2.2.3 Blue carbon valuation

The integration of blue carbon value into management decisions and policies is gaining the interest of governments and industries around the world. Opportunities to value carbon value can be expressed through various methods such as climate financing, payments for ecosystem services (PES), market-based, and conservation finance (Börger et al., 2014; Cavanagh et al., 2016). These valuation opportunities aim to highlight the invisible benefits of carbon sequestration and are often expressed through various mechanisms including climate financing, market-based schemes, cost-based methods, and conservation finance (Table 4) (Börger et al., 2014; Cavanagh et al., 2016). Climate policy instruments and voluntary carbon markets are market-based approaches that attempt to estimate the economic value of climate regulation services such as carbon sequestration (Santos et al., 2018). The United Nations Framework Convention on Climate Change (UNFCCC) treaty, for instance, uses credit training and offset mechanisms to allow signatory countries to meet their emission reduction obligations (Ayostina et al., 2022).

Cost-based valuation schemes, such as the Social Cost of Carbon (SCC), are also common alternatives to market-based approaches; SCC estimates the monetary value of damages induced from emitting an additional metric tonne of CO_2 in the atmosphere (Nordhaus, 2017). Likewise, models such as the Integrated Valuation of Ecosystem Services and Trade-offs (InVEST) Coastal Blue Carbon Model can be a useful tool to estimate blue carbon value based on the carbon market and climate change induced damage (Wedding et al., 2021). Moreover, blue carbon value can be indirectly expressed through innovative conservation financing methods, such as opportunity cost, that reflect the willingness to invest in protecting, managing, and restoring BCEs (Kuwae et al., 2022). Avoided or opportunity cost refers to the economic value of the sequestration services that will be lost in the absence of protection (Murray et al., 2010). This value is often represented as the cost of avoided habitat-conversion projects or the gross financial returns of conservation measures (e.g., establishing marine protected areas) (Bertram et al., 2021; Murray et al., 2010). Moreover, demonstrating the economic value of blue carbon can also be a powerful tool for marine managers to help justify restoration and conservation projects or participate in the carbon market and increase financing options.

Table 4. *Examples of blue carbon economic assessments through various valuation methods (market based, InVEST, social cost of carbon (SCC), and avoided cost).*

Habitat and Region	Valuation Method	Value of Ecosystem Service	Reference
Mangroves (Auckland, New Zealand)	Market-based approach	31.5 million US$ total or 331 US$/ha	Suyadi et al., 2020
Seagrass (South Atlantic Spanish Coast)	InVEST	580 €/ha.	González-García et al. 2022
Salt marsh (Mainland Portugal)	Social Cost of Carbon (SCC)	218.412 € to 328.856 €/t CO_2 (SCC price)	Santos et al., 2018

Table 4 *continued*

Habitat and Region	Valuation Method	Value of Ecosystem Service	Reference
Mangroves (Borneo, Indonesia)	Avoided cost	\$4 to \$10 ton^{-1} CO_2 (Cost avoided emissions)	Siikamäki et al., 2013

2.2.3.1 Limitations to blue carbon valuation

Although a variety of valuation methods are available through international polices and financing approaches, limitations still surround the validity, accuracy, and applicability of blue carbon valuation schemes. A shared limitation across all valuation methods is data availability and accuracy. Current inventories of BCEs in Canada are not equally available across all regions and ecosystem types. Sparse data availability particularly hinders methods, like InVEST, that depend on accurate and comprehensive ecosystem service measures. More robust, and accurate region-specific data collection is needed to support future valuation assessments.

The valuation of ecosystem services is not a new concept, and many studies are over a decade old (Himes-Cornell et al., 2018b). However, valuation methods themselves largely remain debated for their ability to promote or protect ecosystems, resulting in their exclusion from many policy decisions and conservation efforts (Herr & Landis, 2016; Himes-Cornell et al., 2018). For instance, PES programs may be impeded by lack of incentives and willingness for sectors to participate (Chan et al., 2017). Additionally, opportunity costs are challenging to assess as calculations are based on potential loss and unpredictable future climate damages (Nijnik & Miller, 2017). Ensuring that information is updated and consistent with current projections and carbon market prices is crucial to support the validity and reliability of valuation estimations. Moreover, blue carbon valuation often isolates sequestration services from other supporting services the ecosystems provide (Himes-Cornell et al., 2018b). Assessing the synergistic ecological interactions within blue carbon ecosystems can provide a more comprehensive valuation of ecosystem.

Chapter 3: Methodology

3.1 Case study sites

3.1.1 British Columbia (BC) Northern Shelf Bioregion (NSB) MPA Network

The Northern Shelf Bioregion (NSB) is one of thirteen bioregions in Canada's waters which inform national marine planning initiatives (DFO, 2009). The NSB spans from the northern tip of Vancouver Island across to the Alaskan border (Figure 6). The Northern Shelf covers approximately 102,000 km^2 of marine area and supports diverse aquatic species and important blue carbon habitats including eelgrass, kelp forests, and seabed ecosystems (MPA Network BC Northern Shelf, 2022).

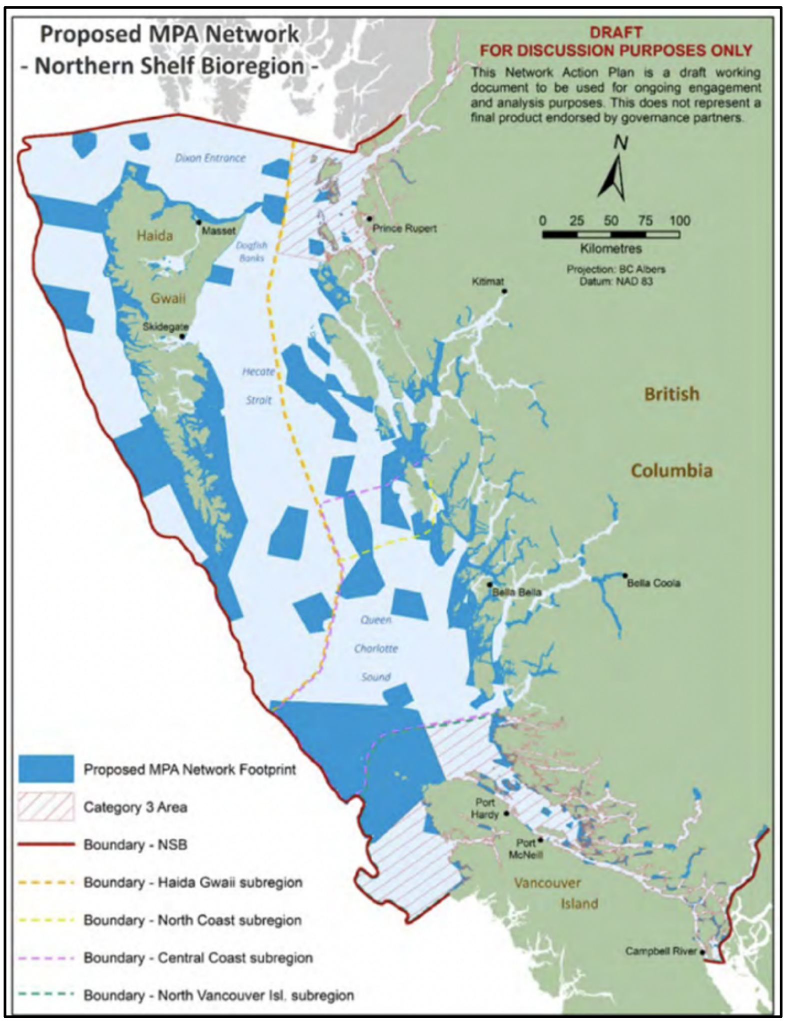

Figure 6. *Draft map of proposed Network with sub-region boundaries* (MPA Network BC Northern Shelf, 2022).

The development of an MPA Network within NSB is currently underway through a collaborative effort between the Government of Canada, the Province of British Columbia, and seventeen First Nations (MPA Network BC Northern Shelf, 2022). The proposed Network will encompass 30,493 km^2 of the NSB and stretch across 29,385 km of coastline (MPA Network BC Northern Shelf, 2022). MPAs and MPA networks are gaining recognition as a valuable tool to protect blue carbon ecosystems from anthropogenic threats (Barbier et al., 2011; Liquete et al., 2013). The NSB MPA Network has the potential to play a vital role in achieving Canada's climate mitigation goals through the conservation of carbon sequestering habitats and the application of climate resilience principles into its planning and management.

3.1.2 Owls Head Provincial Park (OHPP)

The ambiguity of coastal ecosystems often excludes them from the scope of either marine or land habitat inventories (Murphy et al., 2019). This may partially explain why carbon storage estimates rarely exist at this transition zone between land and water (Canuel et al., 2012). Therefore, OHPP can constitute a unique case study to explore blue carbon potential of the coastal zone and contribute to Canada's inventory of wetland and seagrass ecosystems. OHPP is located on the Eastern Shore of Nova Scotia/Mi'kma'ki on the Atlantic coast (Figure 7). This coastal headland is home to an array of biodiverse coastal ecosystems and endangered species and features expansive wetlands and eelgrass meadows that support important carbon sequestering ecosystem services. After the delisting of its pending protected area status and subsequent public resistance, the government of Nova Scotia announced its recent designation under the *Provincial Parks Act* in July 2022 (Province of Nova Scotia, 2022). This designation commits to the protection of over 266 hectares of its Crown lands from development and is a valuable step towards the conservation and restoration of blue carbon ecosystems (Province of Nova Scotia, 2022).

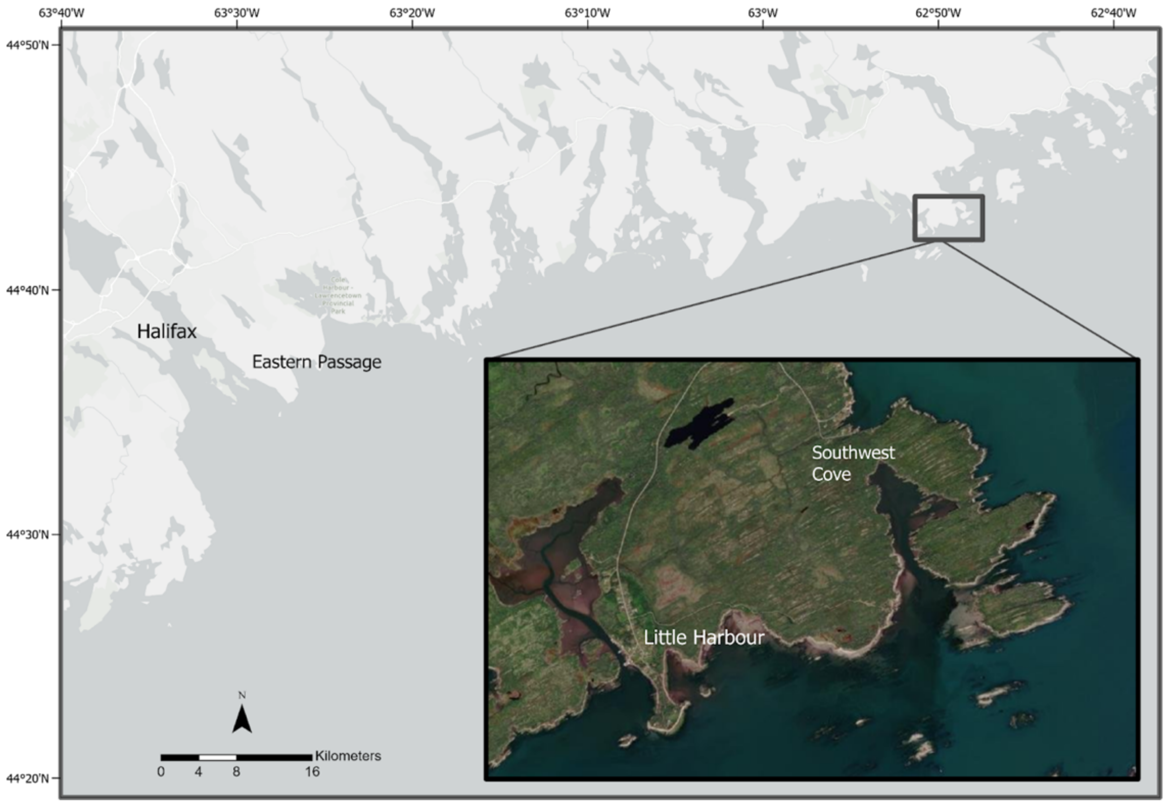

Figure 7. *Location of Owls Head Provincial Park on the Eastern Shore of Nova Scotia (Basemap from Esri, 2022).*

3.2 Applying IPCC's 3 Tier Analysis

This study uses the IPCC's three-tiered blue carbon approach to contribute to blue carbon inventories for the two study areas in the Atlantic and Pacific Ocean, based on the status of available spatial data and information for each region (Table 5). The NSB MPA Network, Pacific Canada, case studies applied IPCC's Tier 1 and Tier 2 analyses to demonstrate how existing information can support multiple tiers of blue carbon assessments. While a Tier 3 analysis did not fall within the scope of this project, the OHPP case study demonstrated approaches to collecting spatial data and field samples, and compiled the necessary tools to support a site-specific assessment.

Table 5. *Summary of available data and information used to conduct blue carbon assessments based on IPCC's three-tiered analysis method.*

	Data available	**Source references**	**Analysis method (IPCC's 3-Tiers)**
Seabed sediments, Haida Gwaii MCT Areas	Global spatial dataset of mean seabed carbon stock	Atwood et al., 2020	**Tier 1:** Area extent and zonal statistics
	Geomorphic units	DFO, 2022	
	Biodiversity hotspot maps (Nearshore habitat richness, fish and invertebrate diversity, and fish and invertebrate biomass)	DFO, 2021	
Kelp forest, Caamano Sound MCT Areas	Giant and bull kelp bed polygons	BC Marine Conservation Analysis (BCMCA), 2008, 2012	**Tier 2:** Area extent and region-specific conversion factors
	Plant density and biomass	Sutherland et al., 2008	
	British Columbia wet-to-dry conversion factors	Wickham et al., 2019	
	Global estimates of macroalgae DOC and POC burial/sequestration	Karuse-Jensen & Duarte, 2016	
Salt marsh and eelgrass, OHPP	Drone imagery		**Tier 3:** Area extent of habitats and lab analysis of core samples
	Sediment samples		

A blue carbon assessment of the NSB MPA Network was conducted, using Haida Gwaii Marine Conservation Targets Areas (MCT Areas) and Caamano Sound MCT Areas as focal sites. Currently, there are no calculated measurements of seabed carbon storage within the NSB. However, publicly available geomorphic units for the benthic zone and a global spatial dataset of mean carbon stocks was able to support a Tier 1 analysis of seabed carbon stock across the NSB and within Haida Gwaii MCT Areas (Figure 8).

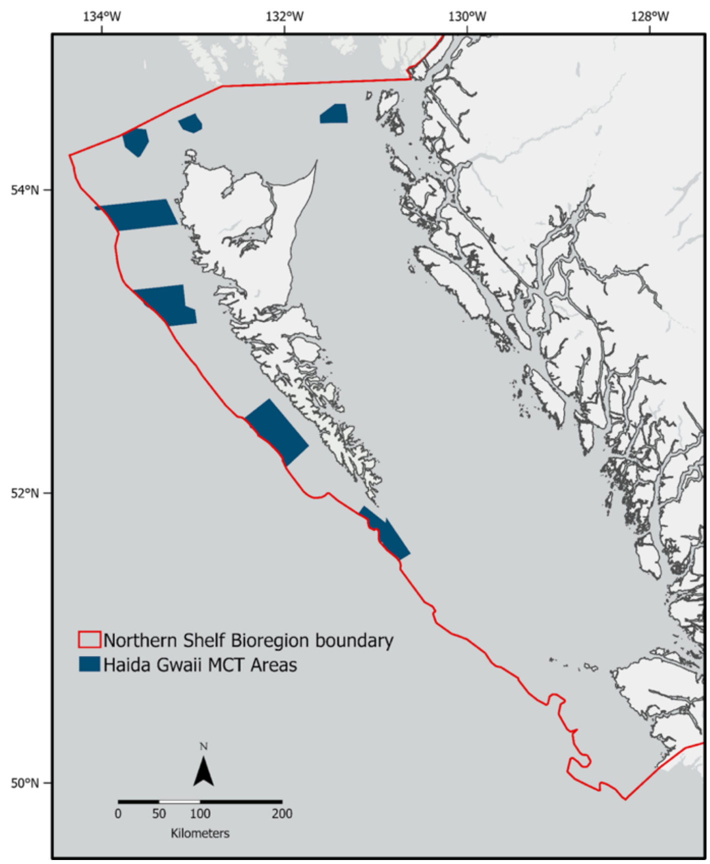

Figure 8. *Haida Gwaii Marine Conservation Target Areas.*

Considerable kelp forest mapping efforts in British Columbia (BC Marine Conservation Analysis (BCMCA), 2008; Costa et al., 2020; Schroeder et al., 2019; Sutherland et al., 2008) have contributed to the generous availability of historical and current spatial datasets. Moreover, research on kelp carbon fluxes (Wheeler & Druehl, 1986) and wet-dry mass calibrations (Wickham et al., 2019) have been conducted on species relevant to the region. Together, these provided the necessary information to support a Tier 2 analysis of carbon storage potential of kelp ecosystems in Caamano Sound MCT Areas (Figure 9).

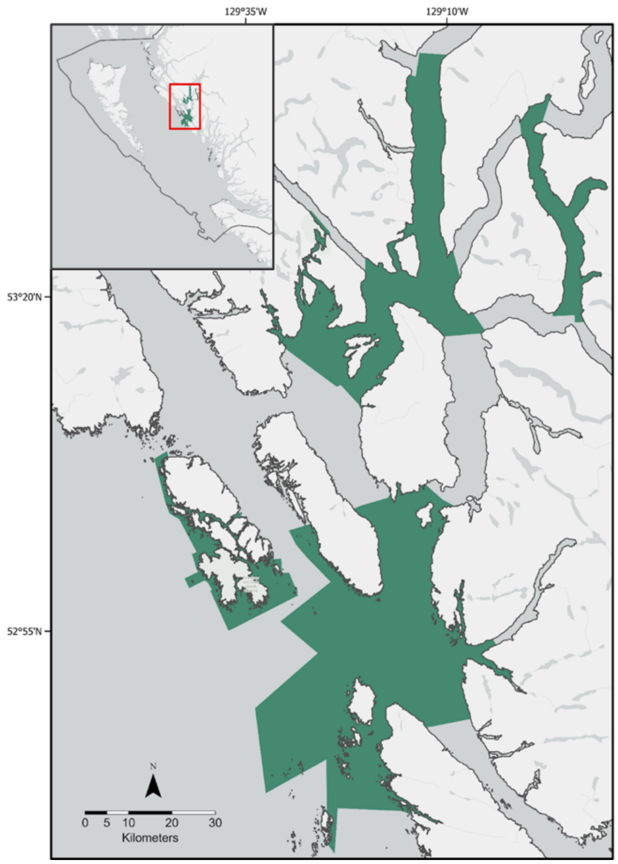

Figure 9. *Caamano Sound Marine Conservation Target Areas.*

Currently, there are no available spatial datasets for the location or extent of eelgrass and wetland ecosystems in OHPP. Additionally, there are no measurements of carbon storage specific to this site. In the absence of these necessary tools, spatial data and field samples of eelgrass and salt marsh ecosystems were collected as first steps to support a Tier 3 analysis in the future.

3.2.1 Tier 1 Analysis: seabed carbon storage of the Haida Gwaii MCT Areas, NSB

3.2.1.1 Spatial data and information

The IPCC's Tier 1 analysis approach was based on Bax et al. 's (2022) blue carbon assessment of the Falkland Islands. Geomorphic spatial units published by DFO (DFO, 2022) were used to define the benthic terrain of NSB (Figure 10a). These geomorphic units delineate areas of the Canadian Pacific continental shelf and slope according to unique geomorphological structures with distinct biological assemblages. They were defined by DFO using a benthic terrain modeller tool with broad and fine-scale benthic positioning index. Furthermore, DFO's published hotspot map of nearshore habitat richness (eelgrass, canopy-forming kelp, and estuaries), fish and invertebrate (e.g., sponges, coral) diversity, and biomass of fish and invertebrates (DFO, 2021) (Figure 10b) was overlayed to outline the potential sequesters that may contribute to the seabed carbon storage.

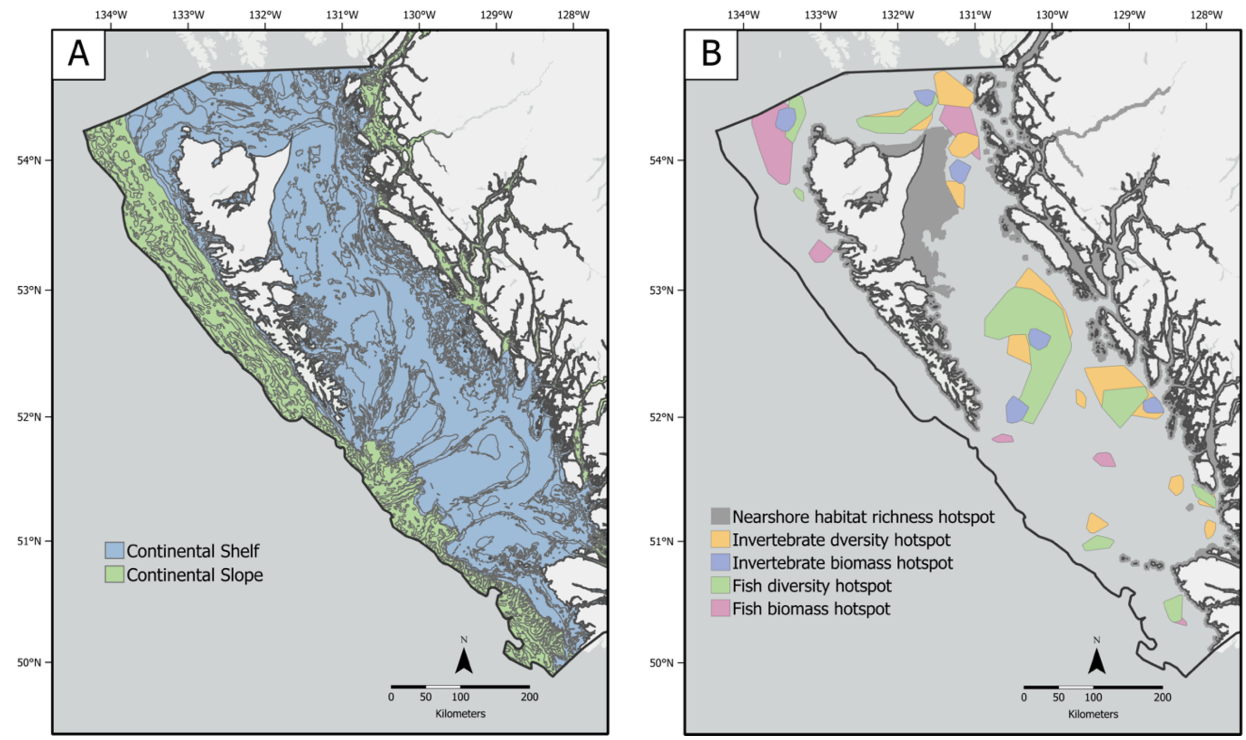

Figure 10. *(a) Geomorphic units outlining the NSB continental shelf and slope (DFO, 2022) and (b) diversity, species richness, and biomass hotspot polygons (DFO, 2021).*

A first-order estimation of the seabed carbon storage potential was derived using a spatial layer of global mean sediment carbon stocks published by Atwood et al. (2020) (Figure B1, Appendix B). Atwood et al.'s dataset was developed by compiling 11,578 sediment core samples of hadal, continental slope, continental shelf, and abyss/basin regions that have been collected by other studies. For the Tier 1 analysis, the dataset was reprojected from its original 1-km to a 500 x 500-meter resolution and resampled using bilinear interpolation. This spatial layer was used to estimate the mean carbon pool found in the first 1 metre of ocean sediment in each spatial pixel for the respective extents.

3.2.1.2 Analysis

The global mean carbon data layer was clipped to the geomorphic units and Haida Gwaii MCT Areas and the mean values of carbon stock were derived using zonal statistics in ESRI ArcGIS Pro. The calculated values were then multiplied by the area of each geomorphic unit and Haida Gwaii MCT Zone to produce a Tier 1 estimate of the total sediment organic carbon pool.

3.2.2 Tier 2 Analysis: Kelp ecosystems of the Caamano Sound MCT Areas, NSB

3.2.2.1 Spatial data and information

Bull kelp (*Nereocystis luetkeana*) (Figure 11a) and giant kelp (*Macrocystis pyrifera*) (Figure 11b) extent in Caamano Sound MCT Areas was calculated using the most recent kelp polygons published by British Columbia Marine Conservation Analysis (BCMCA) (BCMCA, 2008; BCMCA, 2012). The BCMCA kelp polygons were developed through a compilation of survey techniques including aerial overflights and field surveys (BCMCA, n.d).

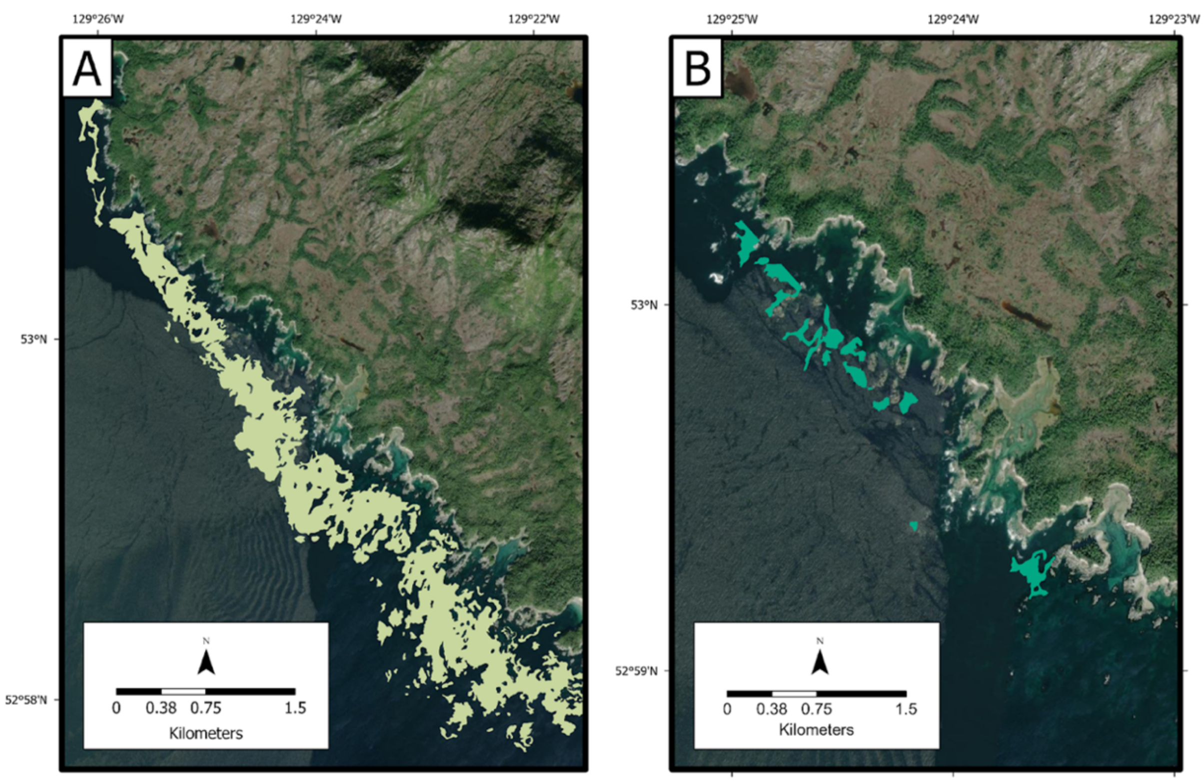

Figure 11. *Snapshot of (a) bull kelp bed polygons published and (b) giant kelp bed polygons (BCMCA, 2008; BCMCA, 2012).*

3.2.2.1 Analysis

Carbon capture potential of kelp beds within the Caamano Sound MCT Areas was assessed using an approach derived from Hutto et al., (2021). Kelp density and biomass estimates were based on field survey data and conversion formulas from British Columbia's 2007 Kelp Inventory, with a total of 396 Nereocystis plants and 882 Macrocystis fronds sampled in August 2007 (Sutherland et al., 2008). We used British Columbia specific wet-to-dry conversion rates (Wickham et al., 2019) of 0.045 and 0.115 for giant kelp and bull kelp, respectively, to calculate dry weight. Given that approximately 30% (19-31%) of kelp dry weight is carbon (Ahn et al., 1998; Rosell & Srivastava, 2004) we multiply total dry weight by 0.3 to calculate total carbon standing stock in kilograms of carbon (kg C). To calculate total net primary productivity (NPP), we multiplied total carbon standing stock by 1.3 (Hutto et al., 2021b; Krause-Jensen et al., 2018) to account for losses via dissolved organic carbon (DOC) and blade erosion. Additionally, Krause-Jensen & Duarte (2016) identify that the percentage of NPP sequestered long-term in kelp beds, deep seabed, or shelf sediments, is estimated to be 10.92%. Moreover, as 1 ton of carbon is equivalent to 3.67 tons of CO_2, the total CO_2 naturally sequestered was calculated from equation 1 (Eq 1).

$$\textbf{Total CO}_2\textbf{ naturally sequestered (tCO}_2\textbf{/yr}^{-1}\textbf{)} = \text{NPP} * 0.11 * 3.67 \qquad \textbf{(1)}$$

Eq 1. *Equation to calculate the total amount of CO_2 naturally sequestered* (Krause-Jensen & Duarte, 2016).

3.2.3 Tier 3 Analysis: Eelgrass and wetlands, OHPP

3.2.3.1 Spatial data

We collected aerial photos of the OHPP land and intertidal region using a DJI Inspire 2 drone equipped with a Zenmuse X4S camera on May 13, 2022. All required drone flight certifications were acquired, and flight routes were mapped out using DroneDeploy software prior to the flight. The flight mission was planned and conducted under minimal to zero wind, slack low tide conditions, and overcast weather to reduce glare. The eelgrass meadows were partly submerged underwater during the image collection period. The drone flight path was operating at approximately 120 metres above ground level and camera was pointed down at a 90-degree angle during the entire flight.

3.2.3.2 Field sampling data

A series of four eelgrass and six salt marsh sediment cores were collected in the area of OHPP and surroundings, between July 2022 and September 2022 (Figure 12). Eelgrass sediment samples were collected during low tide, using a Polyvinyl chloride (PVC) Universal Corer. The corer was hammered upright into the seafloor sediment until the handle reached the sediment or the corer did not move any further. Once inserted, we measured the distance from the sediment to the top of the corer to establish core compaction. After removal, the coring device was capped and transported to the shore keeping the device upright. Onshore, we placed the corer onto an extruding rod and used 10mm spacers to slice the core at controlled intervals.

Wetland samples were collected using a Russian Peat Corer in 10cm x 10cm areas cleared of vegetation. The peat corer was inserted until the top of the sample chamber was flush with the ground. We rotated the barrel clockwise, 180^{o} and used the handle to close the sampling chamber. Once the corer was extracted, we placed the device on the ground and rotated the fin to reveal the semi-core. To collect our subsamples, we laid a measuring tape along the length of the core and used a knife to slice the core at intervals of 0-2cm, 2-5cm, 5-10cm, and 10cm onward up to 50cm. Once all the sediment was removed from the device, we reinserted the corer and followed the same procedure to continue collecting samples at deeper intervals. Core subsamples were frozen in Ziplock bags until analysis.

Figure 12. *Salt marsh (pink) and eelgrass (yellow) core sampling sites in OHPP.*

3.2.3.3 Image processing

The Agisoft Metashape Profession software was used to create an orthophoto mosaic of the collected images and correct for the shift in perspective and topography (AgiSoft, 2018). The drone imagery captured is +/- 10-30 cm of accuracy to maps, making it more accurate than satellite imagery. The workflow of generating the orthophoto mosaic was generated using the following process: (1) inputting drone imagery; (2) aligning photos through tie point matching; (3) building dense clouds and generating a 3D polygon mesh. Dense point clouds were also cleaned up and points with low confidence intervals were removed; (4) generating an orthophoto mapping projection from the 3D polygon mesh; and (5) exporting the orthophoto mosaic as a GeoTIFF file (Balletti et al., 2014). Prior to building the orthomosaic, collected photograph lighting and colour was enhanced and a digital elevation model (DEM) was also calculated.

The Image Classification Wizard tool on ArcGIS Pro was used to conduct unsupervised OBIA to segment and classify the imagery following methods used by… The imagery was segmented using an unsupervised algorithm to separate the image into separable spectral clusters. The algorithm parameters were set to the following values: spectral detail of 20, spatial detail of 5, and minimum segment size in pixels of 20. Following this process, the segmented images were classified using an unsurprised ISO Cluster classifier to identify the pixel clusters into eight habitat classes. Through visual interpretation, referencing classification guides (ESRD, 2015), on-site observations and pictures (Figure B2, Appendix B), and citizen science data from iNaturalist (iNaturalist, n.d.) (Figure B3, Appendix B), individual pixels were assigned into one of the following classes: eelgrass, salt marsh, wetland, water, sand, tree/large shrubs, bare ground/rock, and seaweed, bare rock/ground.

3.2.3.4 Preparing for lab analysis

Lab analysis methods were based on the "Coastal Blue Carbon Manual" published by the Blue Carbon Initiative (Howard et al., 2014). First, frozen core samples were catalogued and weighed to calculate wet volume based on the internal diameter of the coring device (63.5 mm). The samples were then thawed and placed in a 60 ºC oven for at least 24 hours or until dry. Once fully dried, the mass of the samples was weighed again to calculate dry bulk density, then sifted through a 2mm mesh sieve to remove solid matter or large clumps. The following steps are still underway: analyses of organic carbon content using an elemental analyzer or Lost on Ignition

(LOI) methods, and calculation of total soil carbon stocks within case study area. These results will go towards supporting future analysis to determine the total carbon stocks found in eelgrass and salt marsh ecosystems of the OHPP.

3.2.3.5 Application to Tier 3 assessment

Although a Tier 3 analysis was not completed within the timeframe of this study, the collected data support a site-specific assessment of carbon stocks (belowground, soil/sediment) in OHPP moving forward. The collected drone aerial images and classified blue carbon habitats can be used to determine the distribution and areal extent of eelgrass beds and salt marsh habitats. Moreover, once organic carbon content is determined from the lab analysis, total carbon content of each core can be calculated. To estimate the total amount of carbon stored within these ecosystems, the average carbon value for each core sample can be multiplied by the area of each habitat.

Chapter 4: Results

4.1 Tier 1 Analysis: seabed carbon storage of the NSB and Haida Gwaii MCT Areas

The geomorphic units of published by DFO served as the geomorphological spatial data for NSB, while Atwood et al. 's (2020) global dataset provided global carbon stock. Together, this information allowed produced a Tier 1 estimate of seabed carbon storage of the NSB's continental shelf (Figure 13a) and slope (Figure 13b) regions, as well as Haida Gwaii MCT Areas (Figure 14).

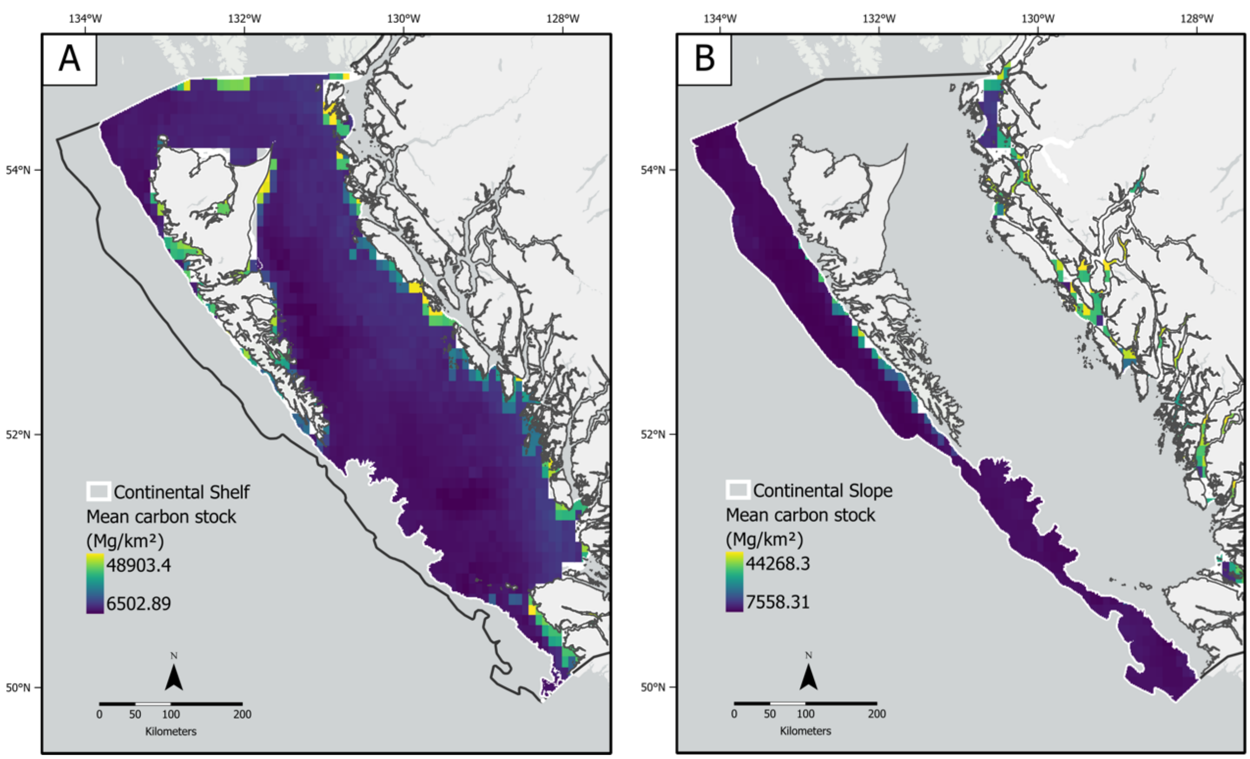

Figure 13. *Global mean seabed carbon stock spatial dataset published by Atwood et al. (2020) clipped to NSB (a) continental shelf and (b) continental slope geomorphic units.*

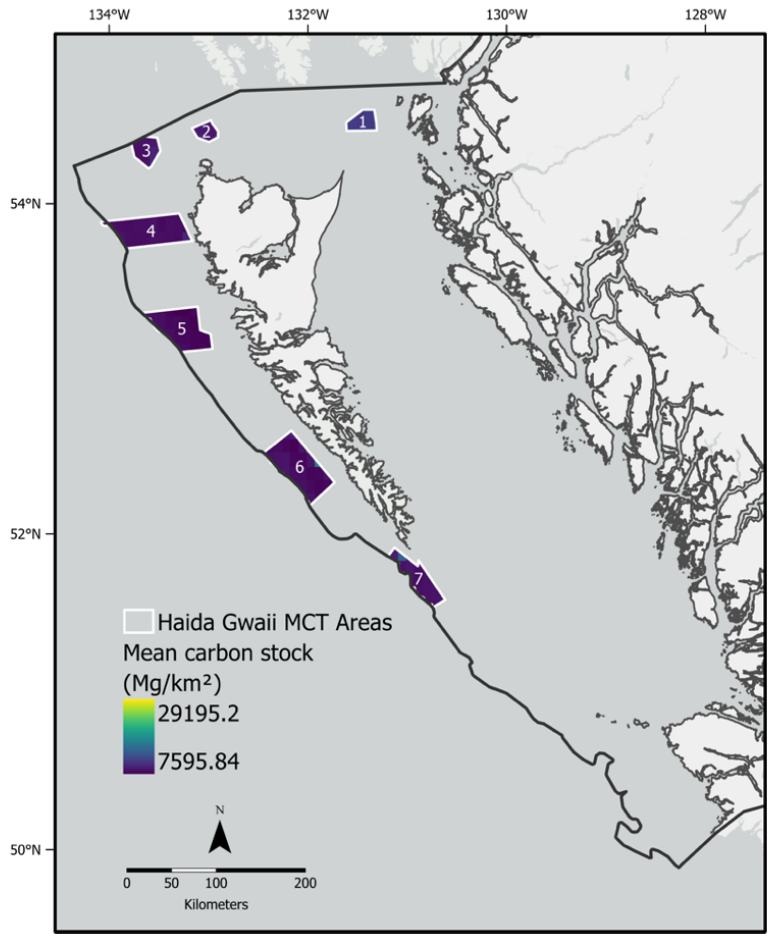

Figure 14. *Global mean seabed carbon stock spatial dataset published by Atwood et al. (2020) clipped to Haida Gwaii MCT Areas.*

The mean carbon stock of the NSB continental shelf was estimated to be 14,358 megagrams of carbon (Mg C) per square kilometer (Figure 15) (Table C1, Appendix C). Meanwhile, the continental slope reveled a higher mean carbon stock of 21,540 Mg C per square kilometer (Figure 15). The values derived for Haida Gwaii MCT Areas indicated that zone 1 has the highest mean carbon stock per square kilometer of 11,327 Mg C/km^2, followed by zone 2 at 9,168 Mg C/km^2, and zone 7 at 9,002 Mg C/km^2 (Figure 15). The remaining zones have estimates ranging between 8,038 to 8,908 Mg C/km^2 (Figure 15).

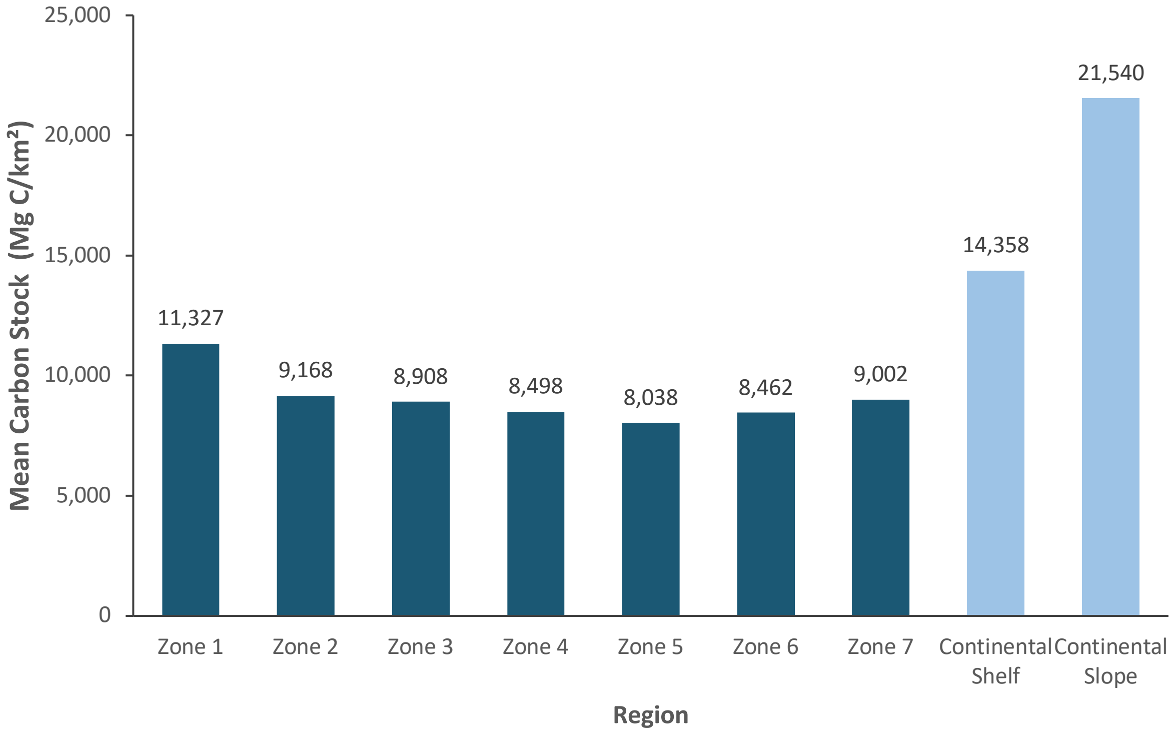

Figure 15. *Mean carbon stock (Mg C/km^2) for Haida Gwaii MCT Area zones (dark blue) and geomorphic units.*

Our calculations indicate a higher total carbon stock within the first metre of sediments for the continental shelf, 98,064,113 Mg C, than the continental slope, 65,141,607 Mg C (Figure 16). Together, this yields a total organic carbon pool of 163,206,720 Mg C across the two NSB geomorphic units (Table C1, Appendix C). The Haida Gwaii MCT Areas yield a total carbon pool size of approximately 3,208,492 Mg C within the first metre of seabed sediment (Figure 17) (Table C2, Appendix C).

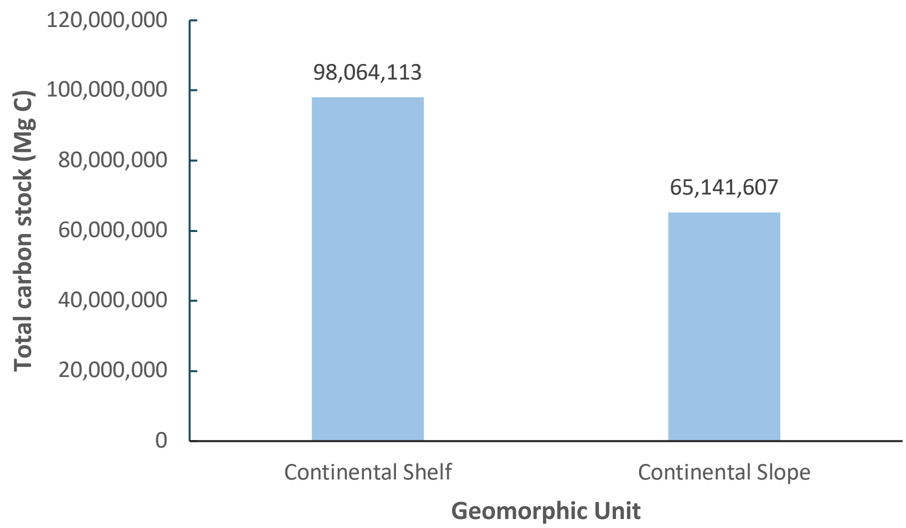

Figure 16. *Total carbon stock (Mg C) for NSB geomorphic units.*

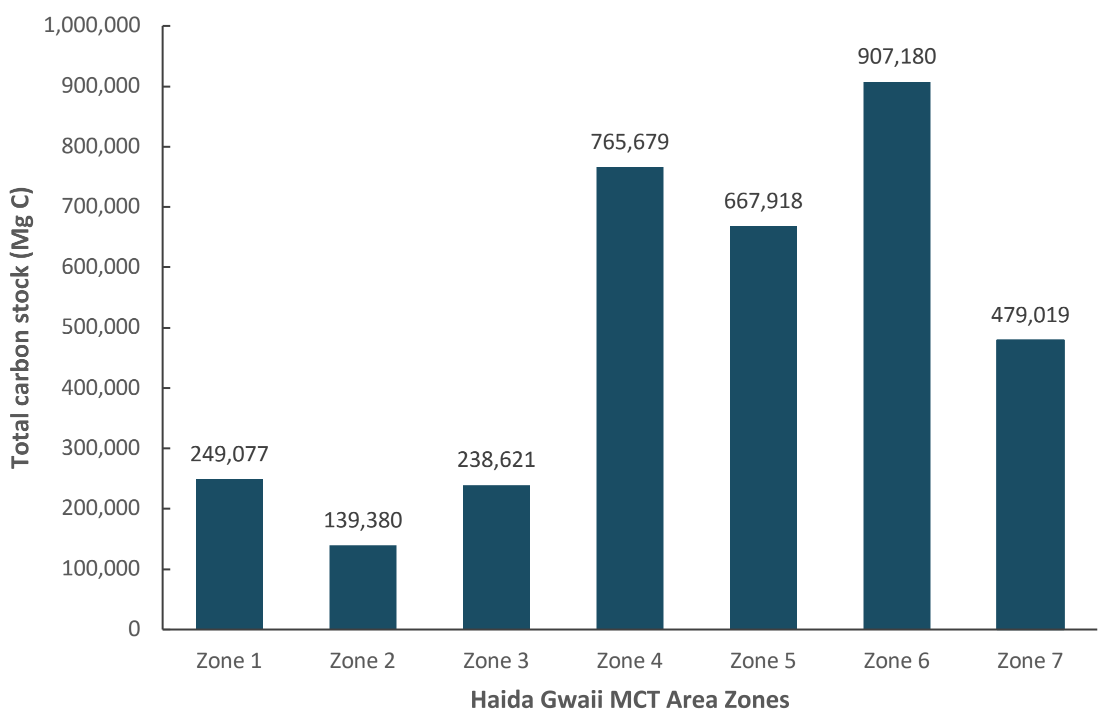

Figure 17. *Total carbon stock (Mg C) for Haida Gwaii MCT Area zones.*

Overlayed with DFO's diversity, biomass, and species richness hotspot polygons (DFO, 2021), several hotspots representing sequestering species are found within Haida Gwaii MCT Zones. Zone 1 is a hotspot for invertebrate and fish biomass and diversity; Zone 3 contains fish biomass and diversity hotspots; Zone 4 is a fish biomass and diversity hotspot; and Zone 5 intersects with a fish biomass hotspot (Figure 18).

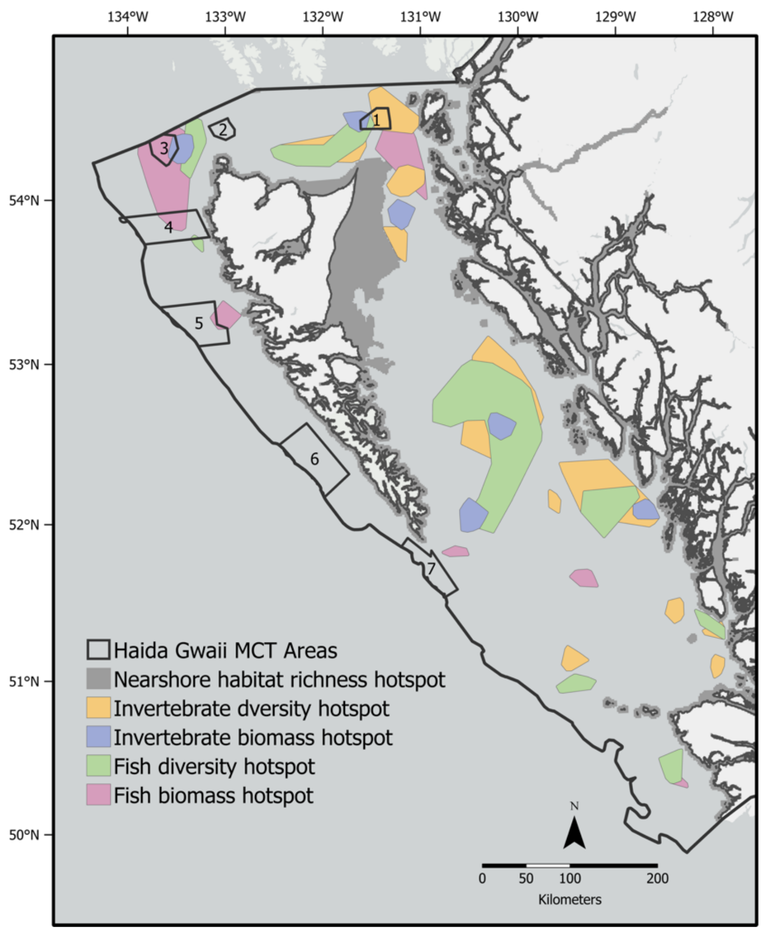

Figure 18. *DFO diversity, biomass, and species richness hotspots overlayed onto Haida Gwaii MCT Areas.*

4.2 Tier 2 Analysis: Kelp ecosystems of Caamano Sound MCT Areas

The most recently published BCMCA data showed canopy-forming kelp beds found in two zones of Caamano Sound MCT Areas: Aristazabal Rennison (Figure 19a) and Trutch Island Group (Figure 19b). Based on this dataset, bull kelp distribution was estimated to be around 14 km^2 and giant kelp to cover around 0.95 km^2 (Table 6).

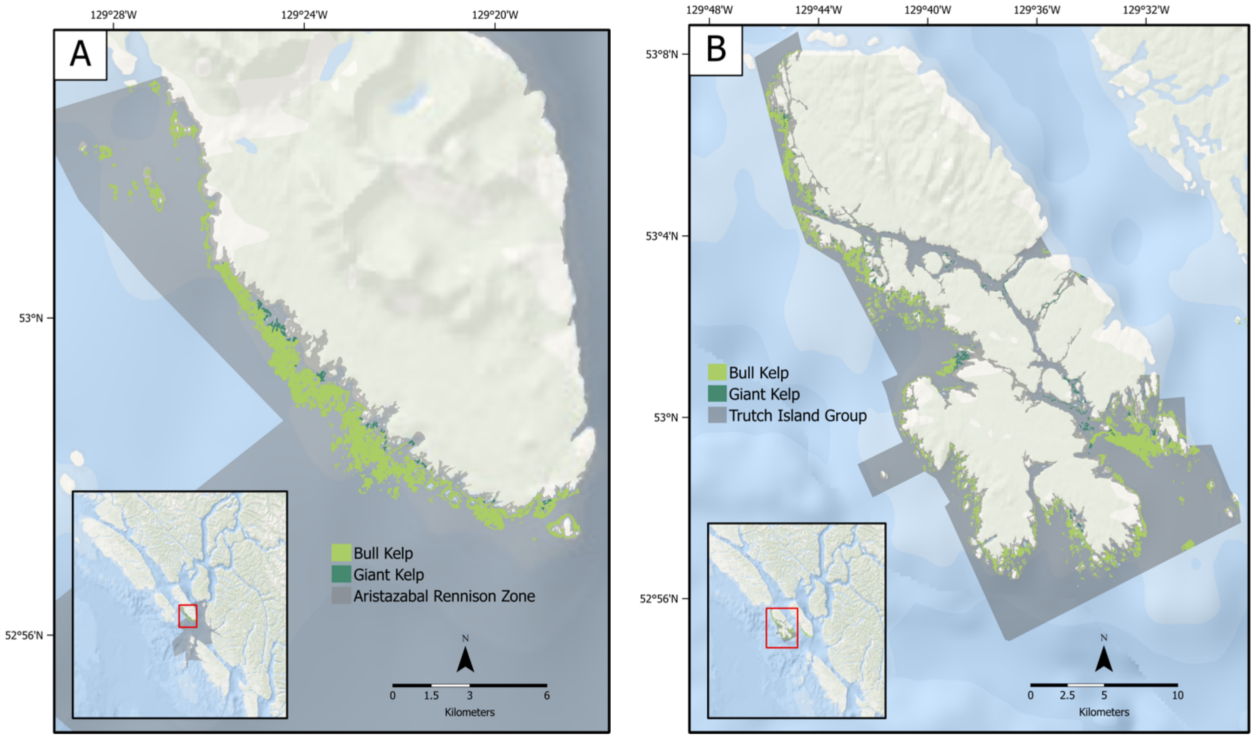

Figure 19*. Distribution of canopy-forming kelp beds within (a) Aristazabl Rennison and (b) Trutch Island Groups of the Caamano Sound MCTs.*

Based on previous estimates of kelp biomass (Sutherland et al., 2008), a total of 36,017 megagrams in wet weight and total dry weight of 4,599 megagrams was estimated (Table 6). This gives an estimated standing stock of 1,003 megagrams of carbon. Using the estimates of total carbon standing stock and a conversion factor of 0.3 to account for losses via DOC and POC (Rosell & Srivastava, 1985; Ahn et al., 1998), a total net primary productivity (NPP) of 1,303 megagrams of carbon per year was estimated for both giant and bull kelp. Additionally, with approximately 10.92% of kelp NPP exported to deep-sea ecosystems (Eq. 1) (Hutto et al., 2021; Krause-Jensen & Duarte, 2016) giant and bull kelp forests in the Caamano Sound MCT Areas of the NSB can be assumed to naturally sequester approximately 526 megagrams of CO_2 every year. This annual sequestration rate is about the equivalent of removing 113 gasoline-powered passenger vehicles from the road for one year (EPA, 2022).

Table 6. *Kelp carbon storage estimates in the Caamano Sound MCT Areas based on kelp dataset published by BCMCA.*

Caamano Sound MCT Areas	**Total Area (km^2)**	**Wet Weight total (Mg)**	**Dry Weight total (Mg)**	**Carbon (Mg)**	**Total NPP (Mg C/yr^{-1})**	**Total CO2 Naturally sequestered (Mg CO_2/yr^{-1})**
Bull Kelp	14	32,669	4,214	887	1,153	465
Giant Kelp	0.95	3,348	385	116	150	61
Total	**14.95**	**36,017**	**4,599**	**1,003**	**1,303**	**526**

4.3 Tier 3 Analysis: Eelgrass and wetlands, OHPP

4.3.1 Spatial data

Aerial drone surveys collected at OHPP served as a first attempt at mapping eelgrass and salt marsh habitats. The collected drone imagery was stitched to produce the following orthophoto mosaic of the coastal zone of OHPP pictured in Figure 20.

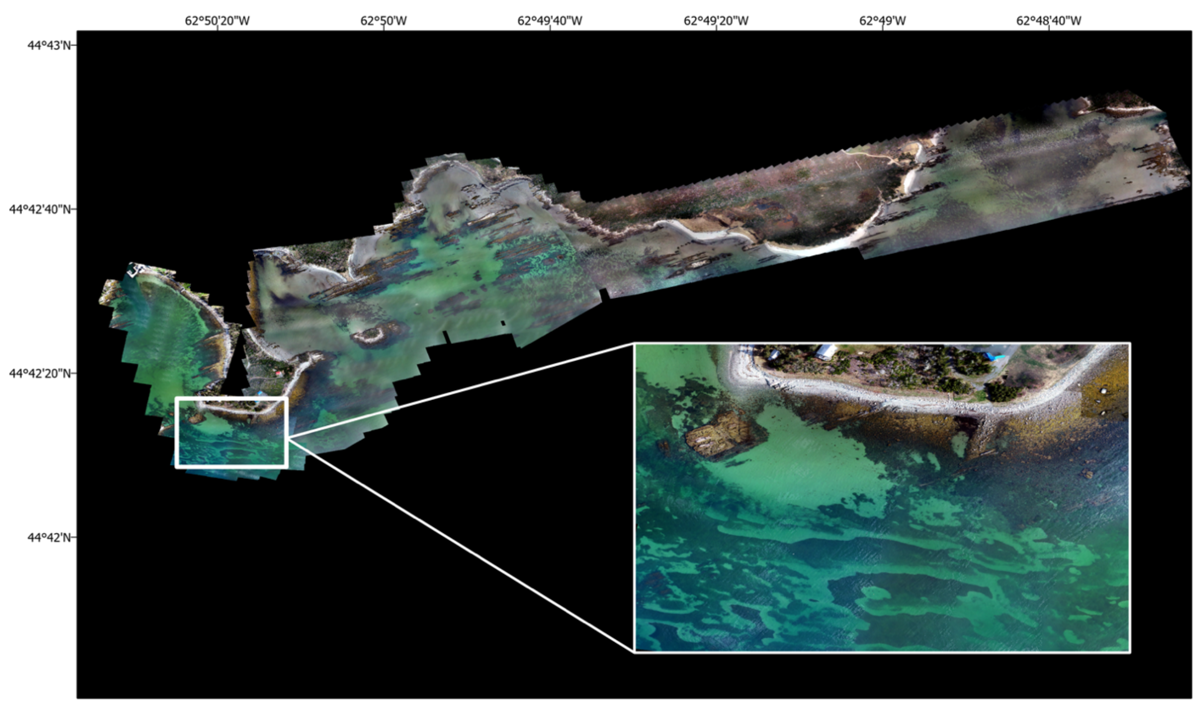

Figure 20. *Orthophoto mosaic of OHPP drone images developed using Agisoft Metashape Profession software.*

Unsupervised OBIA classification of drone images resulted in characterization of eight classes: eelgrass, salt marsh, wetland, seaweed, bare ground/rock, water, sand, trees/large shrubs, bare ground/rock, and seaweed, bare ground/rock throughout the images (Figure 21). The classification was solely based on the segmented pixels and assigned a class based on the dominant habitat type in each pixel cluster. Several segmented clusters were indistinguishable based on the drone images, and thus were given multiple potential classifications such as "seaweed, bare ground/rock".

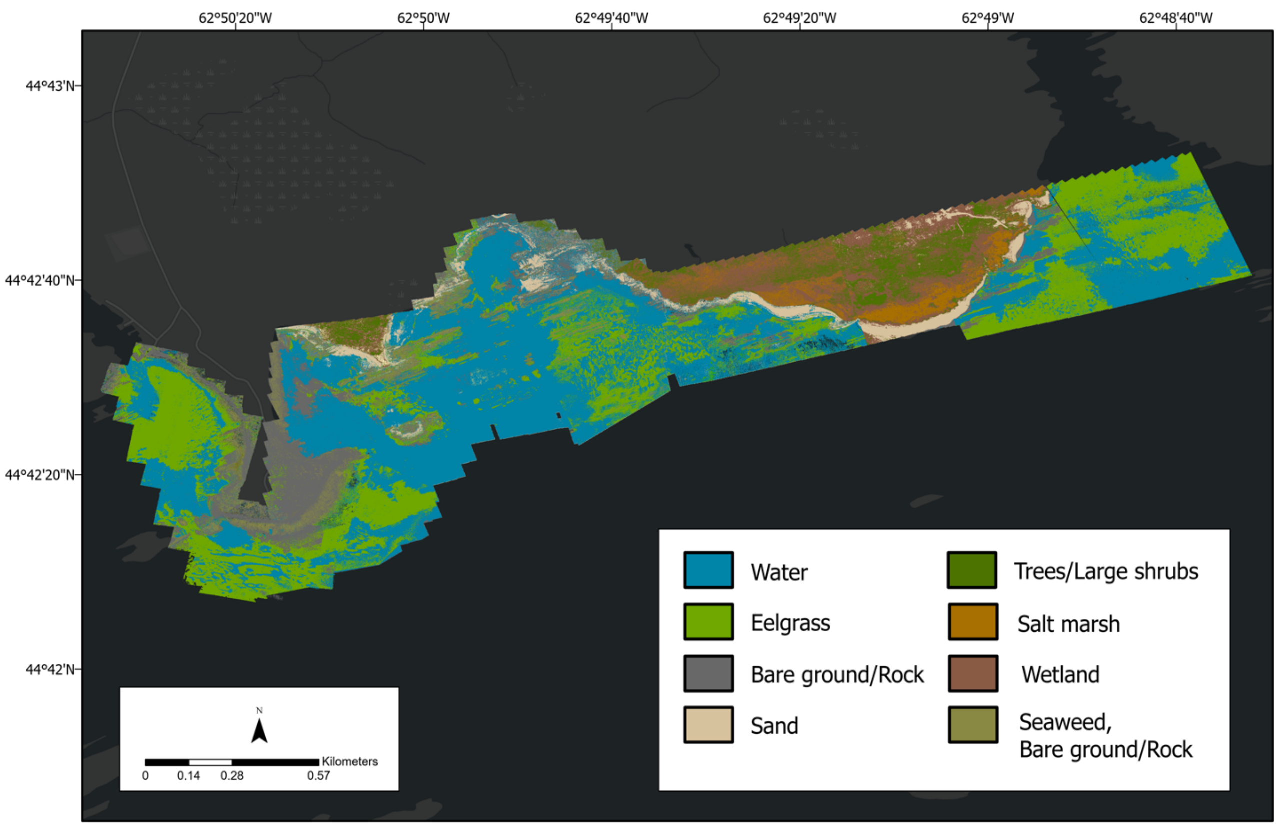

Figure 21. *OHPP drone images classified through unsupervised OBIA using Image Classification Wizard on ArcGIS Pro.*

The final habitat map of OHPP illustrates eelgrass beds of various sizes along the inner-subtidal area of the coastline. Based on the segmented and classified objects, and referencing the drone imagery, we outline the distribution of eelgrass beds and identify the presence of salt marsh habitats across the coast of OHPP. We identified the largest eelgrass patch on the western side of Scanlan Point (Figure 22a) and another large patch near Sand Bar Beach (Figure 22b).

Smaller eelgrass patches were identified within both Five Island Cove (Figure 22c) and Back Cove (Figure 22d). Salt marsh and wetland habitats were identified along the western coast of OHPP (Figure 23) and confirmed on the ground.

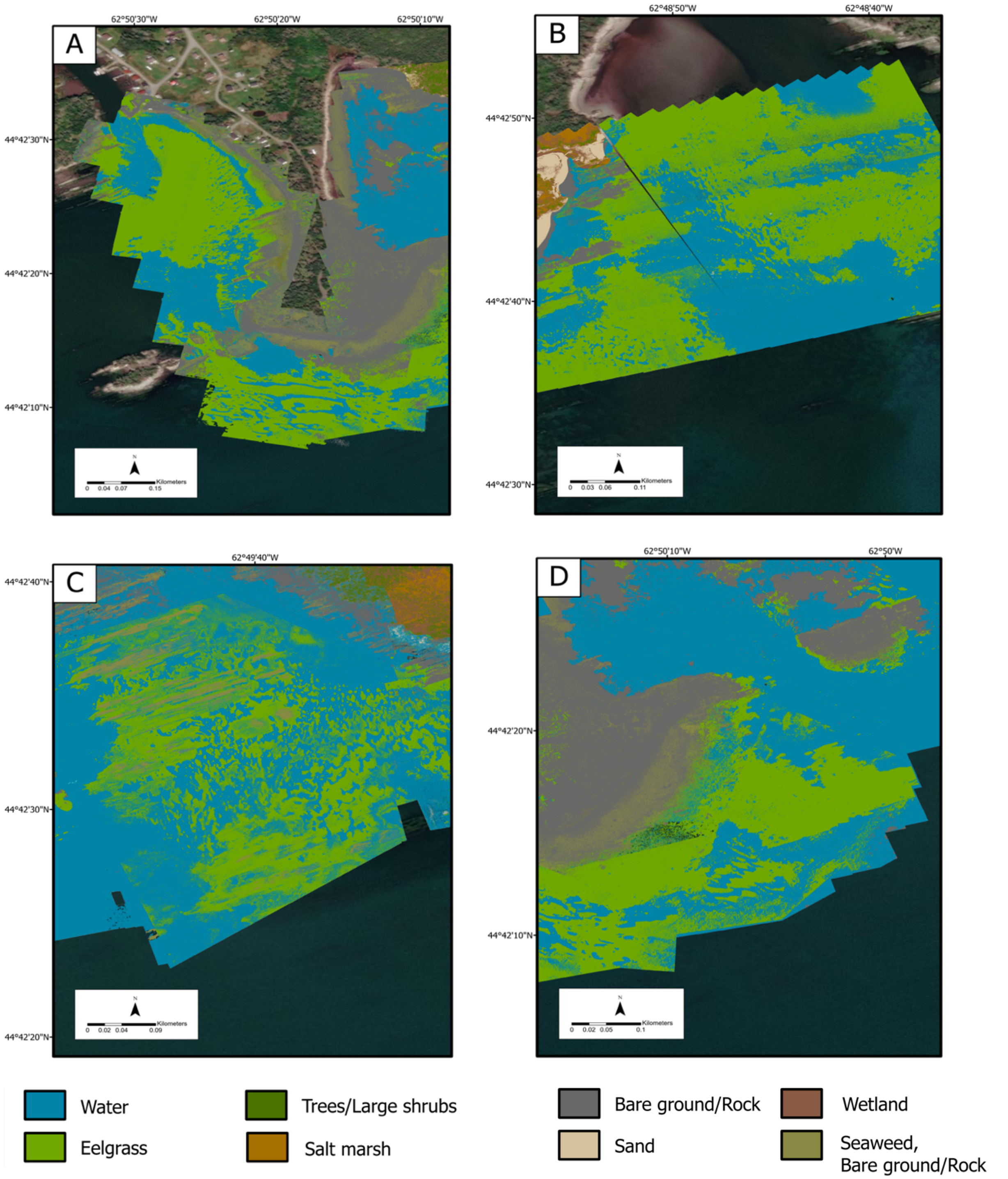

Figure 22. *Large eelgrass patches (green) located on (a) the western side of Scanlan Point (b) near Sand Bar Beach, and other patches within (c) Five Island and (d) Back Cove.*

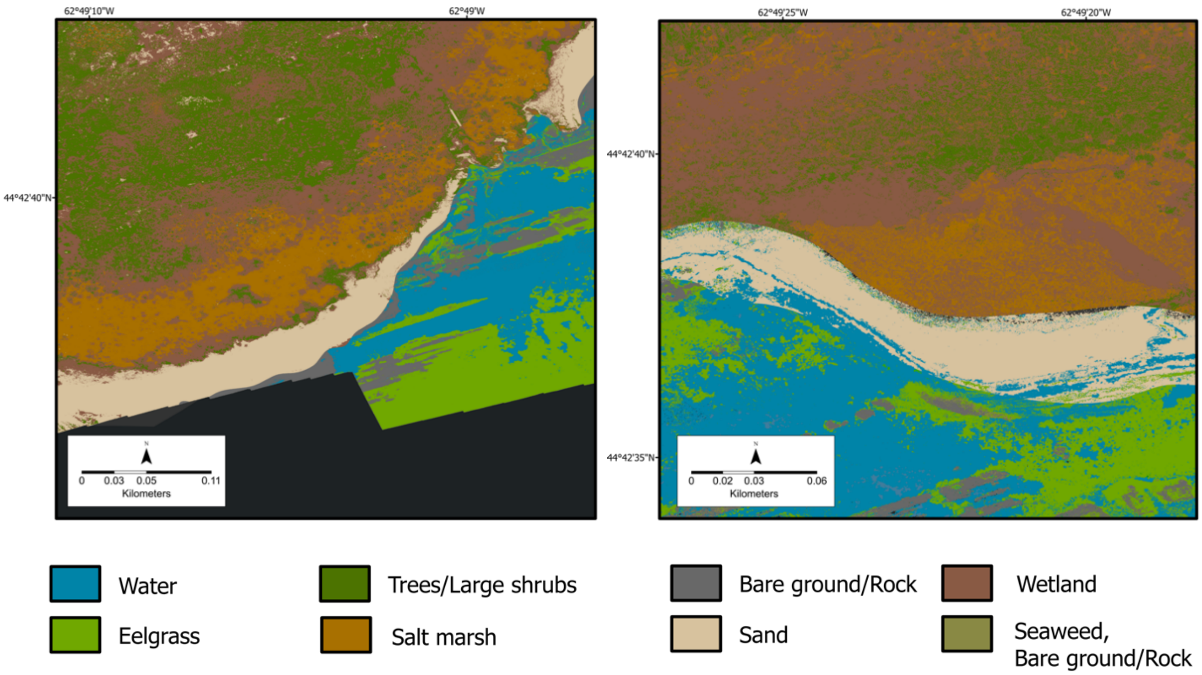

Figure 23. *Salt marsh and wetland ecosystems on the western side of OHPP.*

4.3.2 Application for future Tier 3 assessment

These results represent the first attempt at collecting spatial and field samples of eelgrass and salt marsh habitats of OHPP. The collected core samples are currently undergoing lab analysis to produce site-specific calculations of organic carbon content and to establish baseline carbon storage estimates for Nova Scotia. While the analysis could not be completed for the timeline of this study, results will contribute to the blue carbon inventory of this region by supporting a site-specific, Tier 3 assessment of eelgrass and salt marsh habitats as suggested in this work.

Chapter 5: Discussion and Recommendations

This project aimed to explore the fundamental data needed to establish blue carbon inventories under various levels of data availability. Given the limited spatial data and carbon storage measurements of BCEs in Canada, this study evaluated the opportunities and limitations in our current capacity for developing a blue carbon inventory. Through our case study applications of IPCC's three-tiered assessment, we demonstrate that, while preliminary estimates of carbon storage and sequestration can be calculated with the available data and information, the accuracy of these results are still hindered by the knowledge gaps of spatial data and carbon measurements. Nevertheless, baseline estimates can be leveraged by marine managers to communicate the value of protecting blue carbon habitats and processes, but care needs to be taken not to utilize these estimates in situations calling for more precise data such as carbon markets. Moreover, this study highlights limitations throughout the mapping and analyses process of establishing a blue carbon inventory. Future research can be expanded to refine spatial data collection methods, conduct more region- and site-specific assessments, increase knowledge and information sharing of BCEs, and develop guidelines on applying blue carbon inventories to marine management and policy.

5.1 Tier 1 Analysis: seabed carbon storage of the NSB and Haida Gwaii MCT Areas

Our assessment of seabed carbon storage demonstrates the feasibility of applying mean global blue carbon estimates, while exploring its limitations. However, while this Tier 1 assessment can produce baseline measurements, deriving values from global estimates can have high uncertainties and large margins of error. Atwood et al. 's (2020) utilizes a wide range of data and methods in a standardized model. Variations in coring methods, bulk density measurements, and analytical methods will result in a errors that influence the calculations (Sanderman et al., 2018). Region-specific sampling of sediment carbon stocks and accumulation rates –both field-based sampling and remote sensing– are essential in these cases to address the large range of uncertainties from global models. Moreover, aligning seabed samples with robust geospatial data can ensure that all necessary components of blue carbon inventories are accounted for at finer spatial and temporal scales. In addition, increasing seabed blue carbon assessments is necessary to facilitate cross-comparisons of seabed carbon stocks across distinct regions and seafloor type.

While seafloor geospatial datasets helped to delineate areas of our study region based on geomorphic characteristics, additional knowledge of the functional diversity, species assemblages, and the distribution of sequestering species is necessary to understand the seabed carbon sequestration pathways. The Haida Gwaii MCT zones identified with the highest mean carbon stock generally overlapped with diversity, species richness, and biomass hotspots. This supports the hypothesis that carbon accumulation in the seabed increases with diversity and the number of present functional groups and species assemblages (Barnes & Sands, 2017; Morley et al., 2022). Expanding our observational data on the presence and distribution of carbon rich taxa can validate carbon storage estimates and increase our knowledge of blue carbon connectivity between sequestering species and seabed carbon stocks. Furthermore, investigating the relationship between seabed carbon burial rates and substrata type, microbes, and environmental factors such as changes with seasons can give greater insights into the various fluxes that impact seabed carbon stocks. Future research should focus on developing approaches to identify species assemblages and trace carbon exported and sequestered in deep sea sediments through methods such as environmental DNA (eDNA) biomonitoring (Anglès d'Auriac et al., 2021; Ortega et al., 2019).

Despite the availability of data on NSB's benthic geography and biodiversity hotspots, this information has rarely been discussed in the context of carbon storage and sequestration potential. Our Tier 1 assessment, while preliminary, highlights areas across the NSB with considerable organic carbon storage potential. Moreover, our quantification of carbon storage within proposed conservation areas can be used by marine managers to communicate the value of MPAs regarding their climate mitigation potential. Seabed sediments along the continental shelf and slope are generally the most sensitive to human disturbances (Cavanagh et al., 2021). Future studies should consider evaluating the vulnerability of seabed carbon stocks to human disturbance through threat monitoring and quantifying the magnitude of sediment disruption on current carbon stocks. Additionally, measurements over multiple years would more accurately assess the amount of carbon entering long-term seabed storage and provide more precise baselines to assess the impacts of human disturbance on carbon storage (McLeod et al., 2011). This addition to blue carbon inventories can also be used to advocate for the preservation of current and future carbon storage and sequestration processes from escalating pressures, and further highlight the value of conservation measures.

5.2 Tier 2 Analysis: Kelp ecosystems of the Caamano Sound MCT Areas, NSB

With the abundance of existing kelp-related research on the Pacific coast of Canada, we were able to initiate the first steps towards establishing blue carbon inventory through a Tier 2 assessment of for kelp. However, our assessment focuses solely on Caamano Sound MCT Areas, accounting for only a small area of the NSB. While there is a subset of spatial data available for this region, not all areas are mapped extensively. For these reasons, the estimates suggest an underestimation of carbon sequestration potential of this area. Future mapping efforts should target data sparse regions and aim to supplement large-scale remote sensing mapping efforts with ground-truthing data to increase confidence and accuracy. Moreover, further efforts should aim to apply more accessible methods that are feasible for a wide range of users to adopt. Breaking down this barrier to data collection can streamline knowledge sharing and expand the capacity to support blue carbon accounting.

Detailed information relevant to blue carbon accounting can guide marine managers in prioritizing, conserving, and restoring blue carbon habitats. The carbon sequestration potential of kelp highlighted by this assessment is merely one of a wider range of ecosystem services they provide (Duarte, 1995; Krause-Jensen & Duarte, 2016). The sequestration benefits of kelp ecosystems, alongside additional services and socio-economic benefits can strengthen the case for protection even further and introduce another pathway through which conservation measures, like MPAs, can contribute to climate mitigation (Jacquemont et al., 2022). And while this study emphasizes ecosystem-based conservation, quantified estimates of carbon storage and sequestration can slo support restoration efforts to combat the decline of kelp populations in the West Coast (Filbee-Dexter et al., 2016; Krumhansl et al., 2016). Consistent mapping efforts and long-term monitoring of pressures to kelp ecosystems can ensure changes in ecosystem dynamics are accounted for in future carbon inventories. Moreover, the NSB has an active record of collaborative marine planning efforts through initiatives such as Marine Plan Partnership and the NSB MPA Network (MPA Network BC Northern Shelf, 2022). Moving forward, these collaborative networks could be leveraged as an avenue to exchange relevant research and pursue joint efforts to expand management efforts and build on the existing initiatives.

Although an assessment was conduct with the current available data, there are still several assumptions and limitations. Firstly, while some studies suggest that macroalgae have significant sequestration contributions through dissolved carbon export (Filbee-Dexter &

Wernberg, 2020; Ortega et al., 2009), other researchers express skepticism due to the uncertainties and the lack of empirical data supporting these assessments (Gallagher et al., 2022). A common challenge in integrating kelp in blue carbon assessments is the likelihood for kelp-derived blue carbon to be measured multiple times, as macroalgae detritus often end up in the sediments of other ecosystems (Boyer & Fong, 2005; Wernberg et al., 2006). Overall, new methods are needed to trace donor and sink macroalgal habitats and measure burial rates before fully integrating their contributions into climate policies.

Secondly, although kelp mapping and monitoring efforts are evidently active and available for the NSB, current information systems are scattered and rarely carry the complete subset of data to support the requirements of blue carbon inventories— kelp extent, biomass, and density. Our estimate of kelp-derived carbon sequestration is based on a compilation of data (e.g., biomass-from-canopy cover relationship, estimation of NPP) collected by various researchers, different approaches, and all for a broader range of purposes; this introduces multiple sources of uncertainty. For instance, although our estimates use a proposed multiplier of 1.3 to measure total NPP from biomass measurements, this number realistically can range from 1.0 up to 5.056, suggesting that our estimates are likely to be conservative (Filbee-Dexter & Wernberg, 2020). Therefore, creating centralized datasets that encompass kelp extent, density, and biomass at a site-specific level can produce more accurate and precise carbon sequestration estimates.

5.3 Tier 3 Analysis: Eelgrass and Saltmarsh, OHPP

During our study, we collected spatial and field data to support a future Tier 3 level assessment of blue carbon for eelgrass and saltmarsh ecosystems at OHPP on the Atlantic coast. This collection of information contributes to a regional blue carbon inventory and presents a clear opportunity to identify areas where further sampling and monitoring can improve this data. Ultimately, this data collection phase has the potential to turn into salient, site-specific data that can directly contribute towards Atlantic Canada's blue carbon inventory.

Through our image classification of the collected drone imagery, we were able to roughly label habitat classes of OHPP's coastline. Our habitat classifications, however, did not capture all physical variations within eelgrass meadows and wetland habitats and thus, results were collapsed into eight broader classes. Regions where eelgrass patches are expected to continue but were not classified may be due to misclassification or being too deep or sparse for the drone camera to detect, which is a common limitation to this mapping method (Postlethwaite et al., 2018). As a result, the habitat classes likely ended up underestimating the actual coverage, distribution, and extent of eelgrasses. Additionally, many of the images captured significant sun glare and reflections hindered the success of several image segmentation and classification stages and interfered with achieving optimal results from the image analysis software. Delineating eelgrass beds is challenging even in optimal conditions, as spectral signals of the water, sediments, and other aquatic vegetation often mix and intersperse with spectral signal of eelgrass beds within the images (Rowan et al., 2021). Thus, without additional sensors, the drone camera alone could not fully capture physical differences between eelgrass beds and other aquatic vegetation, and likely resulted in several misclassifications. The addition of adding a multi-spectral sensor to the drone camera or polarizing filter, while expensive, can help reduce glare for future mapping efforts and ease the obstructions in the classification process.

While studies have demonstrated how OBIA can model wetland soil properties and vegetation (Zhang et al., 2019), our classification could not segment pixel clusters to delineate terrestrial wetlands or vegetation types. Terrestrial habitats were categorized into wetland or trees/large shrubs to simplify. Future wetland habitat classification efforts should consider using remotely sensed data (e.g., Landsat, LiDAR, Wordview, hyperspectral imagery) to provide classifications of higher accuracy, as they have proven records of being able to successfully detect and delineate wetland habitats (Campbell & Wang, 2019; Collin et al., 2010; Zhang et al., 2019). In addition, using observations from citizen science data through iNaturalist, we were able to classify several eelgrass beds. However, additional in-situ surveys, specifically ground-truthing, are necessary to address the range of uncertainties through accuracy assessments and increase the confidence in our eelgrass and saltmarsh identifications. Moving forward, these field control points can produce more confident calculations on the area extent of eelgrass and saltmarsh habitats.

5.4 Summary of recommendations

Establishing blue carbon inventories with extensive spatial data and accurate estimates carbon-related comes with a wide suite of challenges. Similarly, while each tier of assessment offers its unique merits, it also carries its own sets of complexities. Addressing existing limitations and working towards refining current approaches can produce inventories that are exhaustive enough to apply to marine conservation and planning initiatives. Based on the findings of our study, the following recommendations are proposed to improve the quality of future assessments and pathways of applying blue carbon to marine management:

Mapping and spatial data:

1. Knowledge of the location and the extent of blue carbon habitats and processes forms the crucial foundation for blue carbon inventories. Further ground truthing data and field surveys can complement current remote sensing efforts and ultimately strengthen the accuracy and quality of spatial data.
2. Complete mapping coverage of entire regions, such as the NSB, can be highly expensive and infeasible to conduct in the absence of adequate funding and resources. Prioritizing data sparse areas and adopting more feasible methods to survey BCEs can enhance the availability of spatial information.
3. Growing blue carbon research and advancements in remote sensing and modeling have contributed to the body of spatial datasets required to evaluate blue carbon potential. Finding ways to facilitate sharing of spatial data between groups (e.g., government agencies, academic institutions, traditional ecological knowledge holders, non-profit organizations) could improve the accessibility of data and foster greater collaboration.

Assessing carbon-related processes:

1. While blue carbon estimates can be extrapolated from global estimates, the evident limitations hinder their applicability to management decisions and climate policy. Collecting region- and site-specific data could more accurately account for the regional variabilities in biotic and abiotic factors and carbon storage and sequestration potential.
2. Blue carbon assessments conducted in siloed projects make it challenging to compare findings and share information. Knowledge sharing across projects can provide insights towards recommended methods of assessing blue carbon habitats.
3. Thorough field surveys and sample collections can be extremely time-consuming and laborious. Given these restrictions, prioritizing samples in areas where substantial spatial

data exist are readily available can optimize time and resource limitations by growing and building-upon existing knowledge.

Valuation and applications to management:

1. Preliminary blue carbon assessments should be leveraged by marine managers as communication and education tools. Baseline quantifications of blue carbon storage and value can promote both the ecological and economic value of BCEs and the significance of management and conservation measures that aim to protect them.
2. While the federal government can initiate the establishment of a national blue carbon inventory, Pacific, Atlantic, and Arctic provinces and territories should conduct assessments at regional scales and develop coordinated strategies between sectors and regions.
3. Monitoring human impacts and threats to blue carbon habitats can provide valuable contributions to blue carbon inventories. Region-specific evaluations are key to developing appropriate management and conservation strategies.
4. Guidelines on the application of blue carbon inventories in marine planning should be established for marine planners and managers. Developing decision support tools and establishing region-specific blue carbon protocols can assist managers with conservation, restoration, and climate adaptation actions.
5. Blue carbon should be a metric for protected area site prioritization. Creating datasets and spatial planning tools that highlight areas of high carbon sequestration and storage potential can help realise the application of blue carbon science into marine planning.

Chapter 6: Conclusion

Blue carbon is an emerging concept that links science, policy, and management networks. As Canada works towards climate mitigation targets, the carbon storage and sequestration services provided by BCEs are a clear and valuable, yet currently underrated, piece of our national climate strategy. Evident gaps in our understanding of carbon-related processes and technical challenges gathering information on BCEs has to-date limited the availability of robust, accurate, and comprehensive blue carbon data. Our study shows that with our current toolkit, multi-tier assessments of blue carbon are possible for scenarios with varying data availability. Initial efforts towards growing blue carbon inventories can catalyze the conversation on carbon storage and sequestration potential and unlock a valuable foundation for future work. Moving forward, it is critical to continue to build upon the growing body of knowledge to address current gaps and challenges in current approaches and strive for high quality inventories. Refining our data collection methods, while still leveraging the knowledge base will support the inclusion of blue carbon in marine management and policy. Canada's blue carbon toolbox, while still missing pieces, is ever expanding and holds promise for the future.

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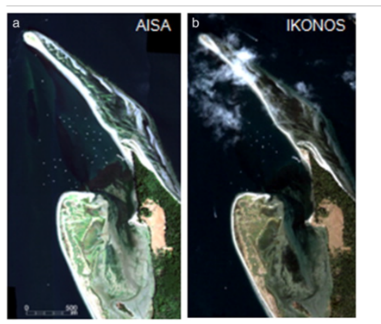
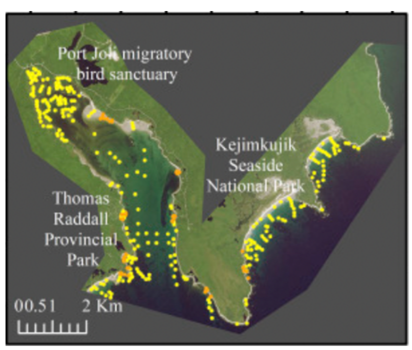
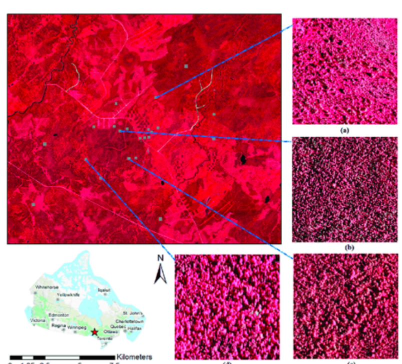
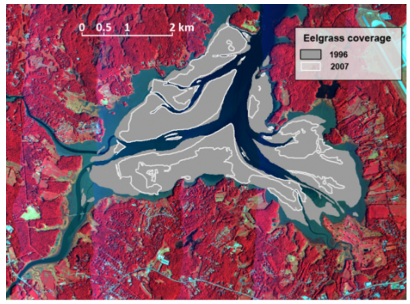
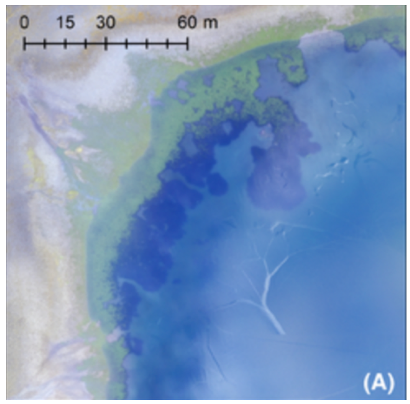
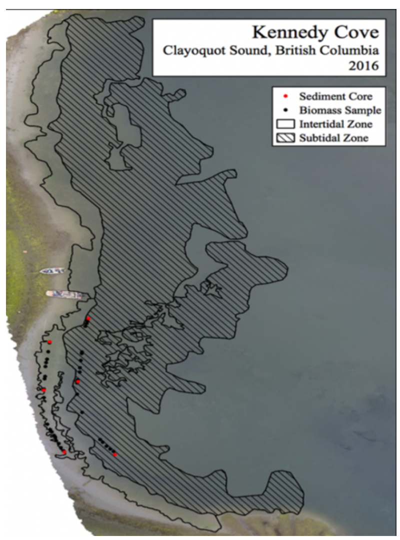
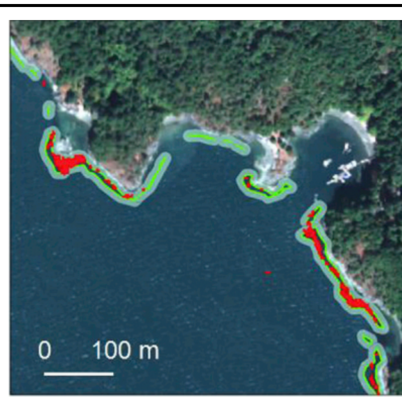
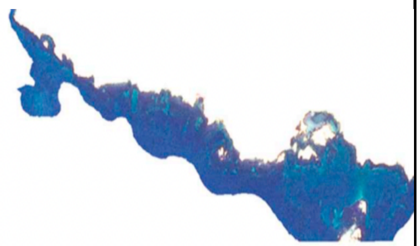
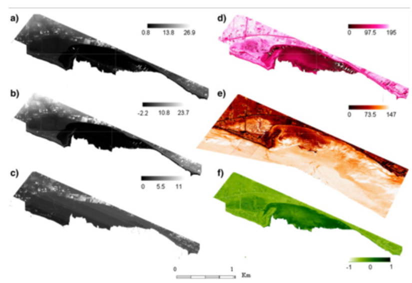
Appendix A: Supplementary Material, Chapter 2

Table A1. *Examples of data collected from various mapping methods and their application to blue carbon research.*

Mapping method	Data	Application to blue carbon	Reference
High spatial resolution airborne (AISA) and satellite (IKONOS) imagery		Mapping and monitoring eelgrass and benthic ecosystems	O'Neill & Costa, 2013
SPOT 6/7 satellite imagery; field surveys		Eelgrass distribution maps and classification framework	Wilson et al, 2019
False color near-infrared aerial imagery		Mapping and classifying eelgrass habitats	Young et al, 2010

Hyperspectral data collected from remote sensing platforms (e.g., airborne or satellite) or in situ (e.g., buoys)		Habitat classification to support management and monitoring of estuarine nutrient content	Pe'eri et al., 2016
Unoccupied Aerial Systems (UAS)		Identifying eelgrass beds at monitoring sites	Nahirnick et al., 2018
Unoccupied Aerial Vehicle (UAV); subtidal field surveys		Calculate carbon storage of eelgrass ecosystems; identify areas of high carbon stocks to support conservation	Postlethwait et al., 2018
WorldView 3 satellite imagery; kayak field surveys		Kelp mapping and biomass indicators	Schroeder et al., 2019

Digital aerial photographs (acquired on board a helicopter); satellite (SPOT 7)		Kelp habitat maps	St-Pierre & Gagnon, 2020
Scanning Hydrographic Operational Airborne LiDAR Survey (SHOALS)		Identify and classify saltmarsh habitats	Collin et al., 2010

Appendix B: Supplementary Material, Chapter 3

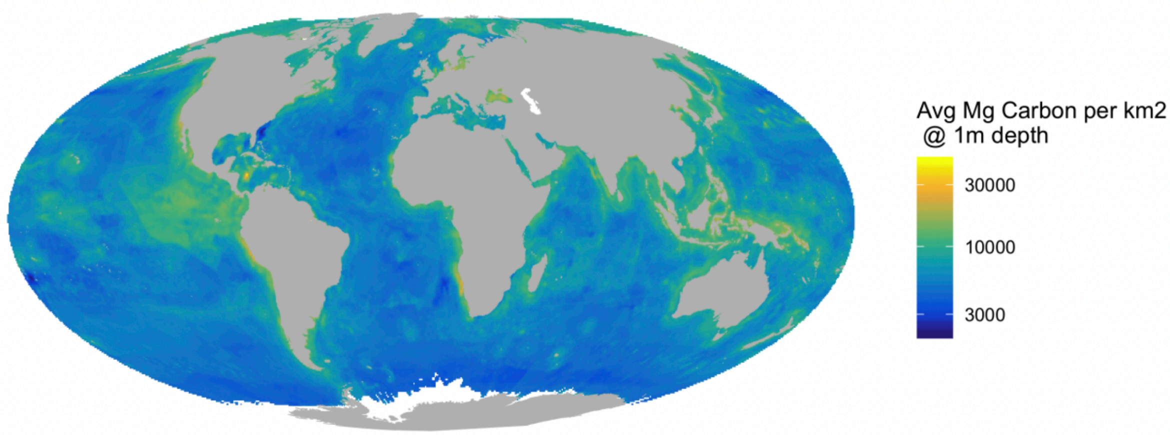


Figure B1. *Global mean sediment carbon stock spatial layer published by Atwood et al. (2020).*

Figure B2. *On-site field observations and pictures for salt marsh coring sites.*

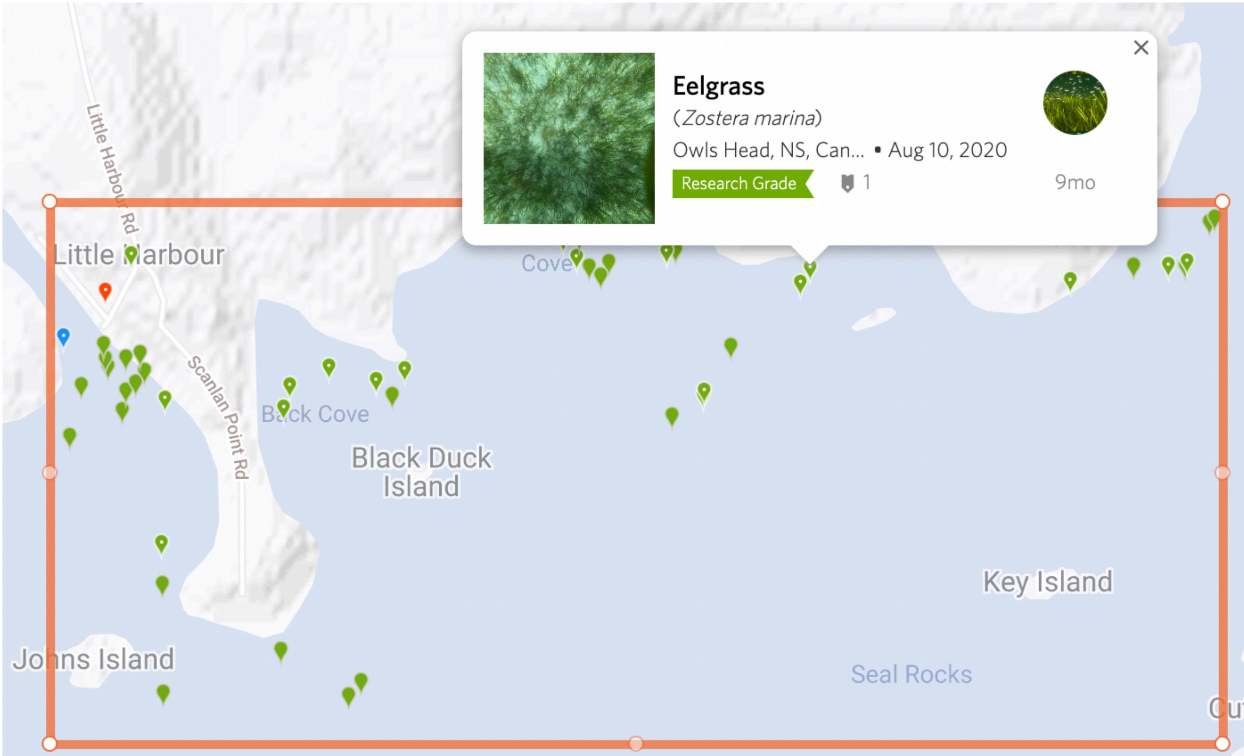


Figure B3. *Citizen science observations for eelgrass within OHPP (iNaturalist, n.d.).*

Appendix C: Supplementary Material, Chapter 4

Table C1*. Mean carbon stock and total storage estimates of the first metre of the continental shelf and slope seabed of NSB, calculated based on global estimates published by Atwood et al. (2020).*

	Continental Shelf	Continental Slope
Area (km^2)	68,301	30,243
Mean carbon stock (Mg/km^2), based on global mean estimates (Atwood et al., 2020)	14,358	21,540
Total carbon stock (Mg), based on global mean estimates (Atwood et al., 2020)	98,064,113	65,141,607

Table C2. *Mean carbon stock and total carbon stock estimates of the first metre of seabed across Haida Gwaii MCT Areas, based on global estimates published by Atwood et al. (2020).*

Haida Gwaii MCT Area Zones	Area (km^2)	Mean carbon stock ($Mg\ C/km^2$), based on global estimates (Atwood et al., 2020)	Total carbon stock (Mg C), based on global estimates (Atwood et al., 2020)
1	220	11,327	249,077
2	152	9,168	139,380
3	268	8,908	238.621
4	901	8,498	765,679
5	831	8,038	667,918
6	1072	8,462	907,180
7	532	9,002	479,019
Total	3976	9,058	3,208,492