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The Marine Coastal Zone and Blue Carbon

Chapter Findings

- Canada's Atlantic, Arctic, and Pacific coasts require regionally specific approaches to NBCS application due, in part, to variations in climate that affect coastal freezing and thawing. Modern geological conditions and history make some coastlines less vulnerable to climate change impacts, such as sea-level rise.
- Additional mapping of areal extent and measurements of specific carbon budgets along Canada's coasts are required to produce more robust estimates of blue carbon sequestration potential.
- The restoration or avoided conversion of tidal wetlands provides numerous co-benefits, but the economic feasibility of these interventions and the need to determine the impact it has on development may limit their potential. Regulatory controls of coastal zones can vary substantially among jurisdictions, and local social acceptability is likewise variable.
- Although limited, research on restoration of Canadian salt marshes indicates that, immediately after the return of tides, rates of carbon storage may be even higher than those in undisturbed marshes.
- There are considerable knowledge gaps in the understanding of coastal carbon sink potential, including the impacts of NBCSs on cultural land uses — most notably, Indigenous land-use and coastal-water practices.

NBCSs relating to marine coastal ecosystems sequester what is widely known as *blue carbon*[25] and traditionally focus on carbon sequestration in mangroves,[26] salt marshes, and seagrass meadows (Nellemann *et al.*, 2009). Salt marshes are defined as coastal ecosystems "mainly occupied by halophytic[27] vegetation and exposed to low hydrodynamic conditions and tidal flooding" (Simas *et al.*, 2001). A seagrass meadow is "a coastal wetland vegetated by seagrass species (rooted, flowering plants), permanently or tidally covered by brackish/saline water" (IPCC, 2013). Carbon stored by macroalgae, such as kelp, may be a form of blue carbon (Krause-Jensen *et al.*, 2018), but the long-term storage potential and manageability is uncertain (Troell *et al.*, 2022). In the Panel's view, limited data on Canadian kelp forests make assessing NBCSs related to these zones unfeasible at this time (Box 6.1), and recent research suggests that some of

25 Blue carbon is defined by IPCC (2022) as "biologically-driven carbon fluxes and storage in marine systems that are amenable to management."

26 Mangrove ecosystems are excluded from discussion in this chapter, as they are not found in Canada (Nellemann *et al.*, 2009).

27 Plants which survive in high salinity soil or water.

these ecosystems may be a source of CO_2 when the entire system is considered (Krause-Jensen & Duarte, 2016; Gallagher *et al.*, 2022).

Most data on carbon stocks and fluxes in North American salt marshes and seagrass meadows come from the contiguous United States, underrepresenting ecosystems in higher latitudes where carbon stocks and rates of carbon sequestration are, in some places, substantially lower than global averages (Ouyang & Lee, 2014; Postlethwaite *et al.*, 2018; Windham-Myers *et al.*, 2018; Prentice *et al.*, 2020; Gailis *et al.*, 2021). Canada has more than 240,000 km of marine coastline (longer than any other country) (StatCan, 2016), which contains coastal ecosystems that sequester carbon while providing other co-benefits (Section 6.6.1). However, there is much uncertainty about the amount of carbon sequestered and its vulnerability to release in response to anthropogenic impacts and changing environmental conditions.

6.1 Opportunities for Enhancing Carbon Sequestration in Marine Coastal Areas

Tidal salt marshes and seagrass meadows store and release carbon through several processes. Organic matter from roots, rhizomes,[28] and aboveground growth is buried in coastal sediments. The decomposition processes that release CO_2 back to the atmosphere are relatively slow. As in other wetlands, the decomposition of organic matter is inhibited by a lack of oxygen due to water saturation, facilitating carbon accumulation and storage (Reddy & Patrick, 1975; Brinson *et al.*, 1981). Per unit area, both tidal salt marsh and seagrass meadow carbon stocks can be substantial, and rates of sequestration are greater than terrestrial forest or peatland soils (Mcleod *et al.*, 2011).

> "Per unit area, both tidal salt marsh and seagrass meadow carbon stocks can be substantial, and rates of sequestration are greater than terrestrial forest or peatland soils."

The accumulation of organic-rich soil tracks the rate of sea-level rise (Rogers *et al.*, 2022); where sea level has been continuously rising, such as on Canada's Atlantic coast, some marshes have been accumulating soil for thousands of years (e.g. Shaw & Ceman, 1999; Kemp *et al.*, 2018). In contrast, much of Canada's northern coastline is experiencing a declining sea level, as lands are still regaining the elevation lost by glacial depression in the last ice age (Pendea & Chmura, 2012). Although these tidal marshes accumulate organic matter, their lifespan is limited, as land uplift brings them out of the tidal frame (Pendea *et al.*, 2010; Pendea & Chmura, 2012). Where tidal marshes

28 Plant stems which grow below the surface of the soil.

transition into freshwater wetlands (i.e., fens and bogs), carbon is preserved (Pendea & Chmura, 2012), but there are currently no studies available that examine the fate of the blue carbon in other situations. On Canada's west coast, neotectonic processes (i.e., motions in the Earth's crust) cause rates of relative sea-level rise to be lower; thus, carbon accumulation rates on the British Columbia coast are lesser than on the coast of Eastern Canada (Chmura *et al.*, 2003; Mazzotti *et al.*, 2008; Montillet *et al.*, 2018; Gailis *et al.*, 2021).

As with tidal salt marshes, seagrass meadows have the potential to accrete vertically for long periods of time and accumulate carbon via their high rates of primary productivity, low rates of decomposition, and ability to trap carbon that originates from non-seagrass sources (Fourqurean *et al.*, 2012; Duarte *et al.*, 2013; Prentice *et al.*, 2020). Although seasonal ice can scour and remove aboveground biomass from eelgrass meadows, plants often dedicate more of their production to their underground structures in these conditions, likely maintaining or even increasing their belowground carbon stock (Robertson & Mann, 1984; Murphy *et al.*, 2021).

Box 6.1 Kelp Forests in Canada

Kelp is common on all three of Canada's coastlines, and global studies of blue carbon suggest it can play a significant role in carbon sequestration. Worldwide, kelp and other macroalgae are estimated to cover approximately 3.5 million km^2 (Krause-Jensen & Duarte, 2016), yet this figure is derived without comprehensive estimates of the overall distribution of kelp forests in Canadian waters. Studies have assessed distribution and trends over time for specific species and regions (Filbee-Dexter *et al.*, 2019) using methods such as aerial surveys (Rogers-Bennett & Catton, 2019), satellite imagery (McPherson *et al.*, 2021), and comparisons with historical benchmarks based on early navigational charts (Costa *et al.*, 2020). A lack of comprehensive data on the extent of these coastal ecosystems makes estimating their aggregate carbon sequestration potential problematic. Documenting long-term carbon sinks associated with kelp forests is challenging; much of the carbon collects in either coastal sediments distant from its point of origin or in the deep ocean, outside Canadian jurisdiction. Additional information on kelp forests would also be helpful to communities managing them for multiple purposes. For example, carbon storage is one of the many traditional, cultural, and ecosystem co-benefits of kelp restoration and cultivation in Gwaii Haanas (Parks Canada, 2021b).

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Kelp forests and the carbon they contain are vulnerable to a range of anthropogenic, ecological, and climatic stresses, including storms and wave events that result in large losses of kelp density, biomass, and cover — these can impact between 40% and 100% of an area (Reed *et al.*, 2008; Krumhansl & Scheibling, 2012). Local management actions for conserving kelp forests, reducing eutrophication, increasing underwater light penetration, managing harvests, limiting bottom trawling, and reintroducing apex predators such as sea otters could help avert kelp loss and enhance carbon sequestration (Wilmers *et al.*, 2012; Filbee-Dexter & Wernberg, 2018; Gregr *et al.*, 2020).

Climate change is expected to reduce the resilience of kelp forests and beds, leading to large losses in kelp biomass due to warmer ocean temperatures, changes in nutrient dynamics, and increased storm frequency and intensity (Gerard, 1997; Steneck *et al.*, 2002; Springer *et al.*, 2010; Wernberg *et al.*, 2010). However, it may also facilitate the northward expansion of kelp ecosystems in the rocky substrates of the Arctic due to reduced ice cover, increased availability of light, permafrost thaw, and warmer temperatures (Krause-Jensen & Duarte, 2014; Filbee-Dexter *et al.*, 2019). Krause-Jensen and Duarte (2014) suggest that vegetated marine ecosystems possibly expanding in the Arctic could contribute to carbon sequestration. However, relatively little is known about the extent and diversity of Arctic kelp communities (Filbee-Dexter *et al.*, 2019), and more research is needed to better estimate this potential expansion, and its implications for carbon sequestration, in response to rapidly changing conditions.

Coastal ecosystems and their ability to sequester carbon are often impacted by anthropogenic stresses, including impacts from waterborne pollution such as agricultural runoff, aquaculture, or hydrologic alterations (Short & Wyllie-Echeverria, 1996; CEC, 2016a). Carbon storage rates in tidal marshes depend on a balance of factors such as plant growth, belowground carbon accumulation, and decomposition. When plant growth is affected by stress, there is decreased carbon storage as well as a loss of soil volume, ultimately reducing soil elevation to a level below what is needed for marsh vegetation to survive extended periods of tidal flooding (CEC, 2016a). Preserving or enhancing carbon storage, consequently, often relies on expanding protection for these ecosystems, or restoring them to ensure ongoing sedimentation and burial of organic material (Macreadie *et al.*, 2021).

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As such, NBCSs for tidal saltwater wetlands are similar to those for freshwater wetlands in that they involve managing, protecting, or restoring these ecosystems and their capacity to sequester carbon, while minimizing other GHG emissions, including CH_4 and N_2O. Relevant interventions include tidal wetland restoration, tidal wetland conservation, and avoided seagrass conversion (Table 6.1). The avoided conversion of tidal wetlands to other land uses can also prevent or reduce GHG emissions, though such interventions may not satisfy the criterion of additionality in regions with no-net-loss policies, such as eastern Canadian provinces, where protection is legislated (Section 6.5.2).

NBCSs for seagrass meadows include the avoided conversion or restoration of these ecosystems. Seagrass habitats have been destroyed as a result of coastal development and are sensitive to anthropogenic impacts such as high nutrient loading, which contributes to eutrophication, and water quality changes associated with sediment discharge, which can block seagrasses with soil or sand and lead to insufficient light conditions for photosynthesis (Orth *et al.*, 2006).

Table 6.1 NBCSs for Tidal Wetlands and Seagrass Meadows

Definition of NBCS	Mechanism
The **conservation of tidal wetlands** through regulation, policy, or economic incentives protects tidal wetlands from potential anthropogenic disturbances or development.	The primary source of carbon stored in tidal marshes is from plant growth, although marshes also trap particulate organic matter transported in tidal floodwaters. Wetland drainage rapidly releases this carbon to the atmosphere; conservation can avoid these emissions (Macreadie *et al.*, 2021).
In situations where tidal wetlands have already been affected, whether through conversion to agricultural lands or development, **the restoration of hydrological and biological regimes** through reflooding and the removal of dikes can eventually re-establish carbon sequestration.	The removal of hydrological restrictions enables the restoration of salt marshes and allows natural processes to restore native vegetation, aiding carbon sequestration (Bowron *et al.*, 2012; Wollenberg *et al.*, 2018; Drexler *et al.*, 2019).

Definition of NBCS	Mechanism
Tidal wetlands can be created or reinforced where they previously did not exist. Not all “living shorelines” are blue carbon-focused, but instead involve stabilization of upland embankments.	Living shorelines use the planting of vegetation to control coastal erosion and create tidal wetlands. Designs can include changing coastal rock structure to reduce wave energy (Bilkovic & Mitchell, 2013; Davis *et al.*, 2015). Additional research is required to determine the carbon sequestration of living shorelines and the potential decline in carbon stored with the age of the wetland (Davis *et al.*, 2015).
Conservation of seagrass meadows can avoid carbon release into the coastal ocean and emissions associated with the erosion of these meadows.	Seagrass ecosystems are vulnerable to waterborne stressors and other impacts (CEC, 2016a). The avoided conversion of seagrass meadows through the creation of protected areas or programs can potentially reduce anthropogenic disturbances of seagrass.
Where seagrass vegetation has been disturbed or removed, **restoration** and replanting may be possible.	Restoration efforts for seagrasses require targeted seeding or planting of seagrass shoots that consider habitat and planting strategies to increase successful restoration of vegetation (Marion & Orth, 2010).

6.2 Indigenous Coastal Land Management

Marine coastal areas have provided critical resources for Indigenous communities for millennia. Indigenous communities have applied Traditional Knowledge and practices to maintain or increase the area of tidal marshes, and as a result, the amount of carbon stored. Several treaties extend into waters, and many Indigenous Peoples do not assign different value to land and water, which could advance protected areas and legislation (e.g., Saugeen Ojibway Nation, 2022). Landscape modifications to create or enhance resource-rich areas have been undertaken by Indigenous communities along coasts across the country, affirming their longstanding occupancy and stewardship of the land (Sayles & Mulrennan, 2019).

One land management practice is the building of dikes to maintain quality hunting sites. Seasonal goose hunts are important to the James Bay Cree, not just as a means to obtain food, but also as an activity with considerable social and cultural significance (Sayles, 2015). The bay’s tidal marshes, which are important feeding sites for geese, can rapidly change. Glacial rebound on the coast of James Bay lifts existing tidal marshes above the reach of tidal flooding, where they drain or become nutrient-poor freshwater wetlands (Pendea & Chmura, 2012). In efforts to maintain the existing marsh and hunting locations, Wemindji Cree on the east coast of James Bay build dikes to delay wetland succession

(Sayles & Mulrennan, 2010). Although these marsh areas and soil carbon stocks have yet to be documented, the Panel considers that this likely enhances the blue carbon sink. The proposed creation of an IPCA in James Bay (Box 6.2) could provide opportunities to measure and assess the impact of management actions on carbon stocks; however, IPCAs are not the only way Indigenous Peoples enact their authority (Section 2.4).

Indigenous people have also actively domesticated landscapes in the tidal salt marshes and applied a range of methods to manage the quality and quantity of plant resources (Turner *et al.*, 2013). The Nuu-chah-nulth, Kwakwaka'wakw, and other First Nations along the Pacific coast have a long-held tradition of creating *estuarine gardens* by mounding soils above the low elevation tidal marsh, which allows the seaward expansion of the high salt marsh (Turner *et al.*, 2013). Although plants are harvested from this marsh, increased soil carbon storage may also result; Gailis *et al.* (2021) found that carbon stocks in the high marsh were more than twice that measured in the low marsh. Similarly, sediment in the coastal zone was altered with rock to create intertidal clam habitats known as *clam gardens* (Groesbeck *et al.*, 2014). Indigenous Nations interested in renewing marsh garden cultivation could contribute to research on how marsh cultivation may have a co-benefit related to carbon storage.

Management of coastal ecosystems involves local adaptations to fluctuations in the environment (Sayles & Mulrennan, 2019). Nature-based approaches have been considered by the Squamish, Semiahmoo, and Tsawwassen Nations as ways to adapt to sea-level rise (PICS, 2020a). In these instances, it seems that marsh restoration or expansion would be the primary objective, and carbon storage would be a co-benefit. Further examination is needed to determine to what extent Indigenous management and knowledge of ecological dynamics can enhance tidal marsh soil carbon stocks.

6.3 Magnitude of Sequestration and Emissions Reduction Potential

Most carbon in salt marshes and seagrass beds is stored in soils rather than in aboveground biomass (Chmura *et al.*, 2003; Fourqurean *et al.*, 2012; Moomaw *et al.*, 2018). When these ecosystems and their sediments are disturbed through anthropogenic impacts or changes in environmental conditions, a portion of the carbon they contain (ranging from 25–100%) can be released back to the atmosphere as organic material decomposes (Pendleton *et al.*, 2012). Actions that reduce or avoid disturbances can thus reduce or prevent these emissions. Ensuring that emissions reductions linked to deliberate management actions truly result in additional sequestration requires analysis using a projected baseline (Section 2.3.2), one that factors in current (or expected) rates of wetland conversion and other relevant trends. Alternatively, actions that increase the area(s) of these ecosystems or their rates of carbon accumulation, through restoration or improved management techniques, can increase total carbon sequestered. In either case, estimating sequestration benefits requires knowledge of the carbon fluxes and soil carbon accumulation rates in these ecosystems, as well as fluxes of N_2O and CH_4 — two GHGs more potent than CO_2 — that can be emitted from salt marsh soils (Magenheimer *et al.*, 1996; Moseman-Valtierra *et al.*, 2011; Poffenbarger *et al.*, 2011; Chmura *et al.*, 2016; Roughan *et al.*, 2018).

Estimates of carbon in salt marshes and seagrass meadows should ensure it is not double counted by including carbon transported from other ecosystems (i.e., allochthonous carbon). For example, data from the Pacific coast of North America suggest that the majority of carbon that accumulates in seagrass meadow sediments originates from non-seagrass sources (Prentice *et al.*, 2020). Methodological approaches on the Pacific coast attempt to account for this, in part, by considering large woody debris (Gailis *et al.*, 2021). Policy frameworks can limit the allocation of offset credits for allochthonous carbon due to the risk of double counting (Emmer *et al.*, 2015; Macreadie *et al.*, 2019). However, detailed information on the source of the stored carbon in many of these ecosystems remains unknown.

6.3.1 Carbon Flux Estimates for the Coastal Zone

Salt marshes and seagrass ecosystems are highly productive

Globally, tidal salt marshes and seagrass ecosystems are estimated to sequester CO_2 at rates of 7.98 t CO_2/ha/yr and 1.58 t CO_2/ha/yr, respectively (IPCC, 2014b; EPA, 2017).[29] Seagrass beds have lower carbon accumulation rates than tidal marshes; however, some regions cover larger areas, and these can have higher carbon sequestration capacity in aggregate (Pacala *et al.*, 2001). Carbon fluxes from seagrass meadows in British Columbia are estimated as an average of (± SE) 0.65 ± 0.12 t CO_2/ha/yr and range between 4.6–33.1 g OC/m^2/y (0.17 and 1.21 t CO_2e/ha/yr)[30] (Postlethwaite *et al.*, 2018; Prentice *et al.*, 2020); this is somewhat lower than global estimates, which include species not found on Canada's coastlines.

Restoration of tidal wetlands has also been shown to result in the resumption of active carbon sequestration at rates similar to, or higher than, those of undisturbed wetlands (e.g., Wollenberg *et al.*, 2018; Drexler *et al.*, 2019; Arias-Ortiz

29 These flux rates were used to estimate the national carbon mitigation potential associated with blue carbon NBCSs in the United States in NASEM (2019).

30 Organic carbon (OC) is used here as reported in the primary research.

et al., 2021). Tidal marsh restoration sites have shown higher rates of carbon accumulation than neighbouring natural marsh sites of the Stillaguamish (Poppe & Rybczyk, 2021) and Snohomish (Crooks *et al.*, 2014) estuaries in Washington state in the northwestern United States — although the brackish marshes in the Snohomish estuary are expected to have substantial CH_4 emissions, which may offset their carbon storage benefits. In the Stillaguamish estuary, annual carbon accumulation rates averaged 0.123 ± 0.03 t C/ha/yr (0.45 ± 0.11 t CO_2e/ha/yr) for natural and 0.23 ± 0.046 t C/ha/yr (0.84 ± 0.17 t CO_2e/ha/yr) for a restored marsh four years after flooding with salt water (Poppe & Rybczyk, 2021).

6.3.2 Variability and Uncertainty in Carbon Flux and Stock Measurements

Any estimate of carbon stocks and fluxes in these ecosystems is subject to multiple sources of uncertainty, including measurement challenges, variability in processes within ecosystems, spatial heterogeneity, and challenges in assessing areal extent. Methods of measuring soil carbon density and accumulation rates in marine coastal wetlands vary widely and can influence overall assessments of stocks and fluxes (Kennedy *et al.*, 2014). Additional uncertainties include challenges with determining carbon sources and accurate quantification of the contributions of GHG fluxes to the total carbon budgets.

Soil carbon densities and depth of tidal wetland deposits are highly variable by region, complicating efforts to estimate carbon stocks

Soil carbon density (SCD), a component needed to calculate carbon stocks, can be variable. Averaging soil carbon stocks over large areas can consequently be misleading, as there is substantial variation across different regions, depending on local ecological and environmental characteristics. In Atlantic Canada, tidal marsh SCD varies from 0.008 grams of carbon per cubic centimetre (g C/cm^3) to 0.067 g C/cm^3 and averages 0.027 ± 0.002 g C/cm^3 for all sites (Chmura *et al.*, 2003). The SCD in salt marshes in Pacific Canada averaged 0.026 g C/cm^3 (Chastain *et al.*, 2021; Gailis *et al.*, 2021). Gailis *et al.* (2021) noted significant differences between high-marsh (0.042 ± 0.013 g C/cm^3) and low-marsh (0.018 ± 0.008 g C/cm^3) SCD. On Canada's eastern coastline, most SCD measurements have been limited to the marsh zones dominated by *Spartina patens* and *Spartina alterniflora* (Chmura *et al.*, 2003). In addition, less information is available on zones at higher elevations and more data are needed to understand regional differences, including dynamics related to density and types of vegetation on different coasts.

Carbon stocks depend on the depth of salt marsh soil. Chmura *et al.* (2003) estimated the global carbon stock of 250 t C/ha assuming a 50 cm depth. Canada's Pacific marshes, however, are shallow, with basal depths ranging from 17–29 cm. Thus, in Boundary Bay on the Pacific coast, carbon stocks in tidal wetlands have been measured as 67 ± 9 t C/ha for Clayoquot Sound (Chastain *et al.*, 2021), and 83 ± 30 and 39 ± 24 t C/ha for high and low marsh, respectively (Gailis *et al.*, 2021). On the east coast of Canada, tidal salt marsh soil depths may range from less than 1 m to 7 m (Shaw & Ceman, 1999; Chmura *et al.*, 2001; van Ardenne *et al.*, 2018). These differences emphasize the importance of considering local contexts when estimating carbon. Stocks could vary with ecosystem characteristics such as vegetation type, soil elevation, and soil flooding status.

Other geographic and environmental factors also contribute to regional variation (Gailis *et al.*, 2021). A relationship between SCD and average annual air temperatures has been noted in eastern North American salt marshes, with warmer surface air temperatures correlating to higher SCD (Chmura *et al.*, 2003). Carbon content in seagrass meadows varies in relation to factors such as water depth, wave height, water motion, and exposure, which impact the carbon content of their sediments as well as accumulation rates (Samper-Villarreal *et al.*, 2016; Dahl *et al.*, 2018; Prentice *et al.*, 2020). These differences make the application of global averages problematic and can lead to an overestimation of blue carbon stocks in regions where specific characteristics of the blue carbon ecosystem have not been measured (Ricart *et al.*, 2015; Postlethwaite *et al.*, 2018).

Rates of soil carbon accumulation also vary widely. Around the Bay of Fundy, rates of carbon accumulation range from 0.72 to 9.28 g C/m^2/yr (2.64 to 34.1 t CO_2e/ha/yr) (Chmura *et al.*, 2003). Carbon accumulation rates in Pacific Canada show similarly high variability, ranging from 0.20 to 4.54 g C/m^2/yr (0.72 to 16.7 t CO_2e/ha/yr) in Boundary Bay and averaging (± SE) of 6.75 ± 1.83 t CO_2e/ha/yr in Clayoquot Sound (Chastain *et al.*, 2021). Canadian measurements tend to be consistent with the IPCC 2013 global estimates of carbon accumulation rates but show high variability within marshes (Chastain *et al.*, 2021). Studies in Clayoquot Sound and Boundary Bay both show that high-marsh locations have higher rates of sediment carbon sequestration than low-marsh ones; it was surmised these high rates of accumulation were due to deeper-rooting plants as well as increased production of belowground biomass in high-marsh relative to low-marsh zones (Connor *et al.*, 2001; Gailis *et al.*, 2021). Moreover, low-marsh deposits are less stable and less likely to hold large amounts of carbon (Gailis *et al.*, 2021); the variability apparent in these marsh systems suggests that accurate determination of carbon stocks and rates requires a sampling design that accounts for spatial variability.

Soil depth measurements are key to estimating coastal carbon stocks

Measurements of SCD are often reported to a depth of 0.5 m (e.g., Chmura *et al.*, 2003) and rarely below 1 m. Estimates of salt marsh carbon storage by the IPCC assume soil depths of 1 m (Kennedy *et al.*, 2014). As mentioned above, while average soil depths in Atlantic Canada are likely to be close to 1 m, many Pacific coast marsh and seagrass sediments are substantially shallower than 50 cm due to the specific nature of the depositional environments in the region (Postlethwaite *et al.*, 2018; Chastain *et al.*, 2021; Gailis *et al.*, 2021). National calculations of carbon sequestration potential (such as those in Drever *et al.* (2021)) may overestimate carbon stored in west coast tidal salt marshes, given their shallower sediments.

Information on carbon stocks in Canadian seagrass meadows is limited

Limited information about the area of Canadian seagrass ecosystems (McKenzie *et al.*, 2020) has resulted in few measurements of bulk carbon density of sediments in seagrass meadows in Canada relative to the length of coastline. All of the sites are vegetated by eelgrass (*Zostera marina*). At sites with the same species in North America and Europe, meadows have an average organic carbon stock of 88.2 (50.2 to 380.07) t C/ha (Prentice *et al.*, 2020). Measurements on the Pacific coast of Canada, however, find average carbon stocks of 13.43 ± 4.82 t C/ha (Postlethwaite *et al.*, 2018) to 20.5 ± 12.85 t C/ha (Prentice *et al.*, 2020), which are substantially lower than global estimates.

6.3.3 CH_4 and N_2O Fluxes

Quantification of the CH_4 and N_2O fluxes of tidal wetlands is needed to determine a wetland's overall contribution to mitigation of climate change. Salt marshes have been reported to be a sink of N_2O (Moseman-Valtierra *et al.*, 2011; Chmura *et al.*, 2016) and, where salinities are >18 part per trillion (ppt), also a sink of CH_4 (Poffenbarger *et al.*, 2011). There have been no GHG studies reported for seagrass meadows in temperate or higher latitude regions in Canada.

When tidal marshes serve as sinks for CH_4 and N_2O, they have even greater value as NBCSs. However, anthropogenic activities in watersheds can change marshes from sinks to sources. Roughan *et al.* (2018) found N_2O emissions from Prince Edward Island tidal salt marshes in watersheds that had intensive agriculture, where fertilizer runoff causes eutrophication of the coastal waters (Section 4.6.1). The control area used in this study had no N_2O emissions. Agricultural soils are also recognized as sources of N_2O emissions. A study examining the impact of reflooding of agricultural lands created by draining salt marshes showed that N_2O

emissions reduced to near zero (Wollenberg *et al.*, 2018). This demonstrates that the return of tidal flooding to drained marshes does not just reinitiate the marsh CO_2 sequestration, but further mitigates climate change by reducing emissions of the more potent GHG, N_2O, in formerly agricultural soils.

Measurements of GHG fluxes in salt marshes have been restricted to Canada's Atlantic coast. Most measurements have been taken in the *Spartina patens* high marsh, as this comprises the greatest area of the majority marshes on the eastern coast of Canada (Comer-Warner *et al.*, 2022). Studies that included sampling in other vegetation zones, however, showed a significant difference in CH_4 emissions (Alongi, 2018; Roughan *et al.*, 2018; Comer-Warner *et al.*, 2022), suggesting the need for more extensive sampling within marshes. Due to substantial differences in vegetation, results from the east coast cannot be extrapolated to tidal marshes on Canada's west or northern coasts. Thus, considerable research is still needed.

6.3.4 Estimating National NBCS Mitigation Potential in the Coastal Zone

There are challenges in estimating areal extent

In Canada, the areal extent of tidal wetlands and seagrass meadows has been mapped as 54,600 ha and 64,500 ha respectively, but this area does not include wetlands in certain locations, including James Bay and southern Hudson Bay (CEC, 2016a). Drever *et al.* (2021) estimated that the area of seagrass meadows is larger at 190,000 ha, and the authors also suggested this number is an underestimate. The Panel has limited confidence in the current evidence for estimating seagrass meadows in Canada. Due to lack of comprehensive mapping, there are no estimations of tidal salt marsh area on the coasts of the Hudson and James Bays, Newfoundland, and parts of Quebec (particularly the northern shore of the St. Lawrence) (CEC, 2016a). Research on the Pacific coast suggests that the extent of marshes has been overestimated. For example, original provincial maps of Boundary Bay, the largest salt marsh in southwestern Canada, indicated its area was 1,207 ha (CEC, 2016a), but recent research reveals the extent of the marsh is closer to 275 ha (Gailis *et al.*, 2021). In the view of the Panel, further mapping to explore and rectify these kinds of discrepancies could be augmented with LIDAR.

Estimating the carbon sequestration potential of NBCSs on a regional or national scale requires calculating the area over which blue carbon NBCSs are found. However, this is also subject to methodological challenges. Limited mapping of salt marsh area may sometimes result in poor modelling of the magnitude of carbon stocks that could be lost because of wetland drainage and erosion (Chmura, 2013). Remote-sensing techniques may be imprecise in measuring small-scale changes and therefore underestimate wetland losses through drainage or erosion (Schepers *et al*., 2017; Windham-Myers *et al*., 2018).

There may be considerable potential for restoring the active CO_2 sink of tidal wetlands by reflooding historically drained and diked salt marshes. Using data from CEC (2016a) and van Proosdij *et al.* (2018), Drever *et al.* (2021) estimated that "the area of undeveloped [dikeland] that could be reflooded without damaging buildings or infrastructure [is] approximately 15,000 ha in New Brunswick [...]; 16,139 ha in Nova Scotia [...]; and 12,990 ha in Quebec [...]." There has been extensive diking of wetlands along the coast of British Columbia, but the area has yet to be estimated.

Canadian seagrass ecosystems are also inadequately mapped (McKenzie *et al*., 2020). Seagrass mapping is lacking in many parts of Atlantic Canada, but modelled estimates have been made for the Pacific coast (Murphy *et al*., 2021). The modelling approach in British Columbia uses mapping data to identify seagrass presence and then converts these line data into polygons that are overlain on bathymetric maps (topographic maps of the sea floor) (Howes *et al*., 2001; Short *et al*., 2016). All area within the polygon between the coastline and a water depth of 3 or 5 m (depending on location) is estimated as seagrass area, regardless of patchiness, substrate type, or environmental condition (Howes *et al*., 2001; Gregr *et al*., 2013; Short *et al*., 2016). Mapping challenges for these zones include the need to account for variation in seagrass patchiness, shape, composition, abundance, biomass, and complexity; the growth of seagrass in deep and turbid water; and challenges assessing the difference in low to moderate seagrass density from the substrate (McKenzie *et al*., 2020). Remote, inaccessible locations, as well as the variability of turbidity and water depth in space and time, make observations challenging (McKenzie *et al*., 2020).

Blue carbon sequestration potential associated with restoration may be overestimated

The most recent estimates for the magnitude of carbon sequestration potential associated with coastal ecosystem conservation and restoration in Canada come from Drever *et al.* (2021). Table 6.2 provides a summary of their findings for tidal wetlands and seagrass meadows, as well as the Panel's assessment of the quality and applicability of the evidence, and of the assumptions underlying the estimated values. In Drever *et al.* (2021), restoration only addressed the drained marshes of the Bay of Fundy, as data on carbon-storage potential with restoration elsewhere in Canada were not measured. Drever *et al.* (2021) did not report potential carbon sequestration from tidal wetland conservation, as tidal wetlands have high protection status in most coastal provinces (Section 6.4.2).

Table 6.2 Marine Coastal Zone and Blue Carbon NBCS Sequestration Potential, as Estimated by Drever *et al*. (2021), and Panel Confidence

Type of NBCS	Present to 2030	2030 to 2050	Panel Confidence	Panel Notes
	Annual (Mt CO_2e/yr)	**Annual** (Mt CO_2e/yr)		
Tidal wetland restoration in NB and NS around the Bay of Fundy	1.5 (1.2 to 1.8)	1.2 (0.9 to 1.5)	High	The tidal wetland restoration estimate is based on restoring 4,413 ha/yr in dikelands in NB and NS.
Avoided seagrass conversion	<0.1*	<0.1*	Low	The areal extent of seagrass used for these calculations was 813,835 ha, greater than the confirmed 190,000 ha of seagrass area estimated by Drever *et al.* (2021). Seagrass carbon stocks used by Drever *et al.* (2021) were five times higher than published evidence from BC (Prentice *et al.*, 2020).
Seagrass meadow restoration	<0.1 (0.0 to 0.8)	0.1 (0.0 to 0.3)	Low	The uncertainty, of both the area of opportunity and the estimated sequestration rate per hectare, is high.

Data source: Drever *et al.* (2021)

The Panel has indicated its level of confidence in these estimates by providing ratings for both the GHG flux and area of opportunity used by Drever *et al.* (2021) to calculate the mitigation potential. See the Appendix for Panel Confidence scale. Numbers marked with an asterisk (*) are estimates modified from Drever *et al.* (2021) with a high uncertainty. Estimates were originally reported as Tg CO_2e/yr.

"The proposed measurement of seagrass meadow carbon stocks could be two to four times too high."

The estimates of the carbon sequestration potential of seagrass meadow restoration and avoided conversion made by Drever *et al.* (2021) relied on several assumptions, which the Panel believes contribute to an overestimation of the area of annual loss as well as carbon stocks. The total area of seagrass was derived by averaging the total area of seagrass in the United States with the area estimated for Canada, thereby basing calculations on ecosystems that are six times larger than the confirmed area of seagrass for Canada.

The estimates of carbon stocks used by Drever *et al.* (2021) (88.2 t OC/ha) do not include recent evidence from British Columbia, where carbon stocks are more than five times smaller (15.2 t OC/ha) (Prentice *et al.*, 2020). Furthermore, these measurements of carbon stocks are extrapolated to 1 m when no cores from Canada's Pacific coast extend to that depth. In the view of the Panel, the proposed measurement of seagrass meadow carbon stocks could be two to four times too high.

6.4 Stability and Permanence

Tidal wetlands and seagrass meadows can sequester carbon through soil accumulation. However, biophysical limitations on the sustainability of sequestration rates and associated carbon stocks include the scale of and resiliency to ecosystem disturbances, sea-level change, and other altered environmental conditions, including remineralization and sediment redistribution (Chastain *et al.*, 2021). Carbon stocks are vulnerable to release back to the atmosphere upon ecosystem disturbance and changed environmental conditions.

6.4.1 Permanence of Carbon Storage in Tidal Wetlands

Potential threats to tidal salt marshes include land development, lack of suspended sediment, excess nutrients, and coastal squeeze, where wetland area is constricted by vegetation succumbing to excessive flooding at the seaward side, and inland migration is prevented by infrastructure built at the upper edge (Torio & Chmura, 2013). The nature of restoration processes can influence the types of sediments deposited, thereby affecting the amount of stored carbon (Drexler *et al.*, 2020). The rate of carbon accumulation in a restored wetland may vary over time (Poppe & Rybczyk, 2021).

The rate of sea-level rise influences salt marsh carbon burial rates and potential

Climate change and increased rates of sea-level rise pose threats to the long-term stability of coastal wetlands and their carbon stocks. If the rate of sea-level rise stays below a threshold level, then marsh vegetation can persist, and soil will accumulate carbon and maintain elevation (CEC, 2016a). Rates of sediment accretion are spatially variable, and the upper range of accretion is estimated to be 5–6.7 mm/yr in marshes on the northwest Atlantic coast (Gonneea *et al.*, 2019; Holmquist *et al.*, 2021). If sea-level rise reaches predicted rates (e.g., Vermeer *et al.*, 2009), then plant production in marshes as well as carbon accumulation and marsh elevation — which depend on carbon accumulation and sediment deposition — could fail to keep pace, resulting in unvegetated deposits vulnerable to erosion (CEC, 2016a). The fate of carbon in submerged tidal wetlands is uncertain. Marsh peat found submerged on the continental shelf off the east coast of North America suggests that at least some portion of submerged carbon stocks can persist, but no research has directly investigated the fate of submerged marsh carbon stocks in Canadian waters (CEC, 2016a).

While the future stability of marshes could be threatened by increased rates of sea-level rise, macro-tidal marshes — such as those on the Bay of Fundy and St. Lawrence Estuary — appear to be resilient (Kirwan *et al.*, 2016). Moreover, predicted rates of sea-level rise in Canada are not as high as elsewhere due to the isostatic rebound in parts of eastern North America, which is 11 cm per century (James *et al.*, 2014; Daigle, 2020). Equally important may be the extent to which wetlands have been modified through land-use change, which mitigates or amplifies the impacts of climate change (e.g., through wetland restoration or drainage) (Zona *et al.*, 2009; Petrescu *et al.*, 2015). Alterations to surrounding hydrology, such as culverts or berms, can impede the drainage of tidal floodwaters, adding to stresses on vegetation.

Understanding other climate change impacts on tidal salt marshes, such as changes in temperature, requires consideration of site-specific conditions and effect interactions

Increasing temperatures will increase decomposition rates in Canadian marsh soils, but will also increase primary production, possibly boosting carbon stocks (Chmura *et al.*, 2003). Other environmental factors that could be affected include soil water-table level and air and soil temperature, all of which have an impact on CH_4 emissions; the greatest impact is soil salinity (Bridgham *et al.*, 2021). However, there is insufficient evidence on the effects of changing temperature and precipitation regimes on tidal saltwater wetlands to make generalized predictions about multiple ecosystem properties and regions (Feher *et al.*, 2017; Moomaw *et al.*, 2018).

Boreal and Arctic tidal wetlands are also impacted by coastal erosion and carbon transported from thawing permafrost (Windham-Myers *et al*., 2018). Along the Arctic coast and Gulf of St. Lawrence, climate impacts such as changing sea-ice cover affect the terrestrial processes impacting coastal erosion and the transport of carbon, water, and nutrients (Pickart *et al*., 2013; Windham-Myers *et al*., 2018). Rapid shifts in salinity and seasonality in boreal and Arctic estuaries make assessing the relationships among climatic drivers, wetland extent, and carbon accumulation rates difficult (Windham-Myers *et al*., 2018).

6.4.2 Permanence in Seagrass Ecosystems

Climate change is expected to add new stresses to eelgrass ecosystems, which could impair carbon sequestration or cause carbon releases

The seagrass species found in Canada's coastal waters is *Zostera marina* (eelgrass). It has been designated an Ecologically Significant Species (ESS) by Fisheries and Oceans Canada (DFO, 2009). Climate changes and human impacts can impact seagrass health. Climatic changes affecting light availability are expected to impact eelgrass ecosystems along with human impacts (e.g., nutrient-loading, coastal development), increasing storm frequency and suspended sediments, which in turn increase water turbidity and smother plants (Curry *et al*., 2019; Murphy *et al*., 2021). Increased CO_2 may enhance eelgrass photosynthetic rate and productivity, but these ecosystems are still vulnerable to the ocean acidification caused as ocean waters take up CO_2 from the atmosphere; seagrass can help reduce CO_2 concentrations in the water (Koch *et al*., 2013; Waldbusser & Salisbury, 2014; Murphy *et al*., 2021). Climate change is also expected to have a strong impact on the input of freshwater and the timing of snow and ice melt in northern Canada (Bonsal *et al*., 2019), affecting seagrass meadows in James Bay and other areas of the Canadian Arctic. The protection and monitoring of seagrass in these coastal ecosystems is important due to its ecological significance (Box 6.2).

Box 6.2 James Bay Seagrass Meadows: An Indigenous-Led National Marine Conservation Area

The eelgrass meadows of James Bay were once estimated to be the most extensive in Canada (Lalumière *et al.*, 1994). Although comprehensive mapping is lacking, these meadows are assumed to have degraded to a fraction of their historical extent (Murphy *et al.*, 2021). While the primary factor responsible for this decline appears to be a change in local hydrology due to increased demands for hydropower and hydroelectric development (Murphy *et al.*, 2021), additional environmental factors can contribute to eelgrass degradation, including rising temperatures in recent decades.

In 2021, the Mushkegowuk Council and Parks Canada signed a memorandum of understanding to begin the designation of an area of more than 91,000 km^2 in western James Bay as an Indigenous-led National Marine Conservation Area (Parks Canada, 2021a). Parks Canada is setting up research study areas around James Bay; mapping will be a potential, valuable output of this research effort, one that could confirm the current extent of the eelgrass meadows. The creation of an Indigenous-led protected area can advance reconciliation and will be designed to maintain Mushkegowuk harvesting rights and practices, consistent with Treaty rights (Mushkegowuk Council, 2020) (Section 2.4).

6.5 Feasibility

Practices for conserving and restoring tidal wetlands and seagrass meadows are subject to a range of challenges and constraints, including cost, technical feasibility, and research gaps. Existing Canadian policies demonstrate, however, that governments have tools to overcome these barriers.

6.5.1 Coastal Zone NBCS Costs

Calculation of the net costs of salt marsh restoration must account for the cost of land acquisition, surveying, construction, adjustment, repair, and maintenance of dikes (Sherren *et al.*, 2019; Drever *et al.*, 2021). These costs vary depending on land characteristics and the interventions required (Haasnoot *et al.*, 2019). Experience from Atlantic Canada provides an indication of the potential range of these costs. Current annual maintenance and repair costs for dikes on the Bay of Fundy are $2 million/yr in Nova Scotia and $650,000/yr in New Brunswick, as cited in Drever *et al.* (2021). A study of all wetland types in Nova Scotia estimated the cost of recent wetland restoration projects to be between $30,000 and $100,000/ha (Gov. of NS, 2014). The net expense of tidal wetland restoration may not be this high, however, when considering the costs of dike management will increase in the face of rising sea levels (CEC, 2016a).

> "Restoration costs may be offset by avoided costs for existing infrastructure due to mitigated disaster risk, particularly flooding."

Such costs should be considered in relation to the value of carbon sequestered and other co-benefits. Restoration costs may be offset by avoided costs for existing infrastructure due to mitigated disaster risk, particularly flooding. Drever *et al.* (2021) estimated the difference between avoided maintenance costs (when dikes are removed) and wetland restoration costs to be $4,972/ha in Nova Scotia and New Brunswick. Wetland restoration still has a net cost, but that cost is considerably reduced after avoided expenditures on dikes are factored in.

Seagrass restoration potential is regionally variable, but costs have not been estimated

Seagrass restoration costs are likely to be high, although evidence is limited (Drever *et al.*, 2021). Seagrass restoration is possible on the Atlantic coast of Nova Scotia and in southern British Columbia, where land management can reduce threats to water quality (CEC, 2016a). However, regulatory jurisdiction over the coastal zone is complicated, potentially involving federal, provincial, municipal, and Indigenous governments. Additionally, costs will be impacted by regional variation in environmental conditions (e.g., water clarity, sediment, temperature, salinity) (CEC, 2016a) and human activities, which can all affect eelgrass survival and its restoration potential (Murphy *et al.*, 2021).

6.5.2 Policy and Regulatory Challenges

No-net-loss policies offset wetland development with restoration or creation, which influences their potential as an NBCS (both positively and negatively)

Existing policies and regulatory approaches provide examples of how wetland conservation and restoration actions can be implemented. For example, the *Nova Scotia Wetland Conservation Policy* focuses on no-net-loss in wetlands (Gov. of NS, 2011). It should be noted, however, that the loss and restoration of wetlands are not completely equal; the loss releases more CO_2 to the atmosphere than restoration can sequester. Therefore, the preservation of existing wetlands in the province tends to be more economically viable than the high costs of mitigation through restoration (Gallant *et al.*, 2020). The policy requires that construction on wetlands be offset through restoration or the creation of additional wetland area (Austen & Hanson, 2007).

Other provinces have similar policies. New Brunswick's *Wetlands Conservation Policy* (2002) considers salt marshes to be provincially significant, affording them the highest degree of protection (Gov. of NB, 2002). Prince Edward Island recognizes that wetlands serve multiple economic, social, and environmental functions; its policies aim to manage development to achieve no-net-loss of wetlands or wetland function (Gov. of PE, 2007). On the Pacific coast, British Columbia's *2015 Water Sustainability Act* protects wetlands from some human activities, but carbon is not mentioned (Gov. of BC, 2015). Complementary policy instruments are often used across the country to protect marsh habitat, such as the *1991 Federal Policy on Wetland Conservation*, the *1994 Migratory Birds Convention Act*, and the *1985 Fisheries Act* (GC, 1985, 1991, 1994).

In Atlantic Canada, no-net-loss policies mean that tidal wetland conservation has limited potential as an NBCS, because they effectively ensure that wetland conservation already occurs and nothing additional can be done. However, supplementary watershed management protections can be implemented, since no-net-loss policies do not always provide effective protection due to a lack of historical enforcement, appropriate land area, and limited capacity to recreate the qualities of pristine sites (Macreadie *et al.*, 2019). In places where legislation already exists, policy may be modified to incorporate carbon rather than creating new policy.

Conversely, the potential impact of the conservation of seagrass meadows is far greater. The designation of eelgrass as an ESS (Section 6.4.2) provides a strong basis for management actions (DFO, 2009, 2011; Murphy *et al.*, 2021). Moreover, seagrass habitat has been prioritized for conservation and inclusion in future marine protected areas in Canada; the Government of Canada aims to protect 30% of coastal and marine areas by 2030 (PMO, 2019).

Monitoring policies and enforcement for restoration and conservation are limited

Monitoring provides a baseline of conditions that can be compared with future conditions following the implementation of an NBCS, such as restoration of a wetland (Bowron *et al.*, 2014). Without a national research framework, monitoring and evaluation that account for the carbon in restored tidal salt marshes and seagrass meadows are limited to specific research sites (ECCC, 2020d). While Canada does not have an equivalent to the Long-Term Environmental Research sites in the United States, Parks Canada has permitted long-term research on salt marsh and seagrass beds in the Kouchibouguac, Pacific Rim, Gulf Islands, and Wapusk National Parks (CEC, 2016b). NBCS carbon accounting in these habitats would ideally consider environmental factors, such as double-counting carbon entering the ecosystem from other locations, as well as economic and policy concerns, such as ensuring sufficient funds to maintain a monitoring system. Even with a national framework, monitoring and evaluation would still depend on research from specific sites.

One monitoring challenge is jurisdiction, notably in cities or communities where municipal, county, and provincial/territorial interests potentially overlap, making policy development and implementation difficult (Seddon *et al.*, 2020a). For example, a living dike project in Boundary Bay required the collaboration of three jurisdictions — the City of Surrey, the City of Delta, and the Semiahmoo First Nation — to raise the elevation of a salt marsh along a 250-km stretch of coastline (Wood, 2020). Conflicting policy objectives and incentives among jurisdictions can be addressed through adaptive governance, which considers the complexities of the social-ecological system by incorporating a range of knowledges (Raadgever *et al.*, 2008; Morris & de Loë, 2016). In coastal zones, adaptive governance can provide an approach to managing jurisdictional complexities while considering the variable social awareness and acceptability of policy approaches (Schultz *et al.*, 2015).

Funding and enforcement of monitoring restoration projects over multiple years are key issues. The design and creation of NBCSs should consider the timeframe for funding and expectations for monitoring targets and maintenance (Kabisch *et al.*, 2016). The cost of carbon monitoring and accounting is a common barrier to participation in carbon-offset markets (Monahan *et al.*, 2020). Challenges to effective and consistent assessment of carbon within ecosystems should be considered in advance by the stakeholders who are planning the NBCS, along with the allocation of jurisdictional responsibilities and required funding.

6.6 Co-Benefits and Trade-Offs

6.6.1 Co-Benefits

The restoration and avoided conversion of tidal wetlands and seagrass meadows provide a wide range of ecosystem services, including protecting shorelines from erosion, stabilizing sediments by attenuating wave action, and protecting habitat for a variety of fauna and flora. Canadian salt marshes provide habitat for rare and endangered species (e.g. Mazerolle & Blaney, 2010); those habitats support artisanal harvests of waterfowl and vegetation important to Indigenous and recreational hunters and foragers (Chmura *et al.*, 2012; Dick *et al.*, 2022). Salt marshes help maintain commercial fisheries by providing nurseries for young fish and protection from larger predators (Barbier *et al.*, 2011). The uptake of nutrients and pollutants by salt marshes purifies water (Hung & Chmura, 2007), which benefits human health as well as adjacent ecosystems, such as seagrass meadows that would otherwise be vulnerable to pollutants (Barbier *et al.*, 2011). Coastal wetlands also provide social benefits associated with recreation and education (Gov. of NS, 2011; Chmura *et al.*, 2012). In general, the conservation and restoration of coastal ecosystems can increase the adaptive capacity of communities to cope with natural hazards and climate change, while also enhancing coastal livelihoods (Barbier *et al.*, 2011). Seagrass meadows also provide co-benefits in terms of shoreline protection and nutrient cycling (Murphy *et al.*, 2021); they can survive increased ocean acidification for long time periods, providing localized protection against this threat (Koweek *et al.*, 2018).

6.6.2 Trade-Offs

Competing interests and land-use values are potential barriers to wetland restoration or conservation in Canada's marine coastal areas. Demands stemming from development or the agricultural industry can make coastal areas valuable, increasing the costs of conservation. In Atlantic Canada, the maintenance of community status quo and limited local government budgets have been identified as two of the largest impediments to wetland conservation and restoration (Sherren *et al.*, 2019). The higher population density on the southern coast of British Columbia relative to the Atlantic coast impacts demands on land use and land value. There may also be concerns about the extent to which wetlands offer an equivalent level of protection from flooding compared to dikes or other infrastructure (Zhu *et al.*, 2020).

Salt marshes in Atlantic Canada historically drained and diked for agricultural use may be restorable if communities feel that residential, commercial, and transportation infrastructure can be adequately protected from disturbances (Sherren *et al.*, 2019). While tidal salt marshes can provide a similar level of coastal protection from disturbances, they may require a comparatively greater amount of land compared to infrastructure (Sutton-Grier *et al.*, 2015; Haasnoot *et al.*, 2019), but provide many more ecosystem services. The degree of coastal protection from disturbances provided by restored marshes will vary depending on geography, biomass productivity, and storm type and severity (Sutton-Grier *et al.*, 2015).

6.7 Conclusion

Tidal saltwater marshes and seagrass meadows are productive ecosystems that have the potential to maintain or improve carbon sequestration. Tidal salt marsh restoration, especially in sites on the Atlantic and Pacific coasts, has a high potential of mitigating climate change impacts. Assessing the value of NBCSs will require regionally specific approaches for each of the Atlantic, Arctic, and Pacific coasts due to variations in vegetation, climate, and sea-level change. Atlantic Canada currently has the highest feasibility for NBCS development, while Hudson Bay and the Pacific coast could permit regionally beneficial actions with additional understanding of local conditions. Further research is required to assess areas of opportunity to implement the restoration or avoided conversion of coastal ecosystems, while potential land use (including cultural uses) in jurisdictions needs to be considered, as well — most notably Indigenous land-use practices.