Marsh Resiliency Strategies in the Face of Sea-Level Rise: Pilot Project Opportunities for Fraser River Delta Tidal Marshes

by

Alexander Caldicott

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Name:	**Alexander Caldicott**
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Examining Committee:	**Chair:** Dr. Ken Ashley Supervisor and Chair
	Dr. Ken Ashley Senior Supervisor Faculty, BCIT
	Dr. Eric Anderson Internal Examiner Faculty, BCIT
	Dr. Ruth Joy Internal Examiner Faculty, SFU
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Abstract

Coastal wetlands are naturally resilient to changing sea levels; however, as rates of sea-level rise increase, the interaction between changing sea-level and ongoing human impacts will be a major driver in future coastal tidal marsh stability. My goal is to provide decision makers with recommendations to increase the resilience of the Fraser River delta front tidal marsh communities over the twenty-first century. I conducted a literature review to (1) examine the current knowledge base regarding effects of sea-level rise on tidal marshes and (2) identify current ecosystem-based adaptation strategies for increasing tidal marsh resilience to sea-level rise. Based on this review, recommendations are made for strategies that could be used to increase tidal marsh resilience in the Fraser River delta. Recommendations include (1) initiating delta-wide marsh accretion modeling to assess tidal marsh vulnerability under possible sea-level rise scenarios and (2) implementing sediment augmentation pilot projects for both direct (e.g., layered sediment lifts) and indirect (e.g., mud motor) sediment augmentation strategies to test ecosystem based adaptive management strategies as part of an adaptive management framework.

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Chapter 1.

Introduction

Sea-level rise is expected to have wide-ranging effects on both coastal ecosystems and coastal human communities in the twenty-first century (Oppenheimer et al., 2019, Church et al., 2013, IPCC, 2018). Coastal wetlands are naturally resilient to changing sea-levels due to feedbacks between inundation period, sediment supply, and vegetation growth, and thus have been able to accrete sediment to match historic rates of sea-level rise (Cahoon, 2009; Kirwan et al., 2016; Moller and Christie, 2018; Perillo, 2019). Despite natural resilience to changing sea-levels, there are many locations where coastal marshes have deteriorated (e.g., coastal Louisiana). This is due in part to higher than average rates of relative sea-level rise, high rates of subsidence, and decreased rates of sediment delivery (Crosby et al., 2016). At the interface between human communities and coastal ecosystems, where human influence has disturbed the natural feedbacks necessary to maintain stability, tidal marshes are among the most vulnerable ecosystems to projected sea-level rise (Morris et al., 2002; Hill et al., 2013; Pratolongo, et al 2019).

In the Fraser River delta, coastal brackish and salt marshes are highly productive ecosystems, providing important refuge and feeding grounds for diverse wildlife populations. They offer a critical migratory stopover for a multitude of bird species and provide critical refuge for salmon transitioning from freshwater to saltwater environments (Church and Hales, 2007; Williams et al., 2009). Tidal marshes at the delta front also provide important flood protection services through dissipation of wave energy against dikes and terrestrial foreshore environments. Increasing recognition has been given to soft engineering flood protection measures that may increase shoreline stability and attenuate wave energy (Narayan et al., 2016).

There is concern that the tidal marshes of the Fraser River delta front may not be resilient to rising sea levels over the long term (Eric Balke, South Coast Conservation Land Management Program Coordinator [SCCLMP], personal communication). As coastal communities along the Fraser River delta front continue to develop flood adaptation strategies for long-term sea-level rise, now is the time to look at opportunities that could potentially benefit both coastal ecosystems and coastal communities in the face of sea level rise. The SCCLMP and the British Columbia Provincial Ministry of Forests, Lands, Natural Resource Operations, and Rural Development (FLNRORD) are working with local governments to promote tidal marsh resilience as a beneficial coastal flood protection strategy for adjacent communities developing ongoing and future upgrades to coastal flood protection infrastructure.

Identifying how Fraser River delta tidal marshes are changing in response to sea-level rise and corresponding strategies for enhancing their resilience are complex challenges. Pilot projects are preliminary, small scale studies that provide an opportunity to evaluate the feasibility (e.g., time, cost, and potential pitfalls) of novel sea-level rise resilience strategies and can offer an opportunity to improve upon designs prior to large scale measures (Thabane et al. 2010). If developed at appropriate scales and with appropriate experimental design and monitoring requirements, pilot projects can provide insight into the outcomes of full-scale projects.

1.1. Goals & Tasks of this Project

My goal was to provide the SCCLMP and FLNRORD with recommendations for increasing the resilience of the Fraser River delta front tidal marsh communities over the twenty-first century. I completed a literature review to 1) critically examine the current knowledge base regarding effects of sea-level rise on tidal marshes, and 2) identify current ecosystem-based adaptation strategies for increasing tidal marsh resilience to sea-level rise. Based on this review, I have made recommendations for strategies that can be explored to mitigate marsh risks and increase tidal marsh resilience to sea-level rise in the Fraser River delta over the twenty-first century.

Primary tasks included the following:

1. Review potential and ongoing effects of sea-level rise on tidal marshes;
2. Review Fraser River delta biogeomorphic processes and potential stressors to delta front tidal marshes in the face of sea-level rise;
3. Review current strategies for increasing tidal marsh resiliency to sea-level rise (at a global scale); and,

4. Provide recommendations for potentially increasing tidal marsh resilience to sea-level rise over the next century and opportunities for pilot projects to test and evaluate adaptive management strategies at the Fraser River delta front.

This report is divided into five chapters. Chapter 1 (this chapter) includes goals and tasks for this project. Chapter 2 describes sea-level rise and associated effects on tidal marshes. Chapter 3 provides an overview of the Fraser River delta and potential tidal marsh response. Chapter 4 provides a review of strategies for increasing tidal marsh resiliency in the face of sea-level rise and Chapter 5 includes recommendations and potential strategies that could be implemented in the Fraser River delta to increase tidal marsh resilience to sea-level rise over the long-term.

Chapter 2.

Sea-Level Rise & Tidal Marsh Response

2.1. Sea-Level Rise

Since the mid-1800's, recorded global mean sea-levels have risen by approximately 20 cm (Le Cozannet et al., 2014) with the majority of sea-level rise occurring in the last few decades. Estimates from 2018 reveal that global mean sea-level was approximately 8 cm above 1993 averages and increased at a rate of approximately 3.2 mm/year over the period of 1993 to 2015 (Oppenheimer et al., 2019). Long term records of global mean temperatures indicate that there was substantial increase in warming during the latter part of the twentieth century with human-induced warming reaching approximately 1 °C above pre-industrial levels in 2017 (Pratolongo et al., 2019). The Intergovernmental Panel on Climate Change (IPCC) Fifth Assessment Report (AR5) projects global mean sea levels to increase 0.26 m to 0.98 m by the year 2100 (Church et al., 2013). Depending on future emission pathways, anthropogenic emissions are expected to contribute to continued and increasing rates of sea-level rise well beyond the year 2100 (IPCC, 2018). Model-based projections of global mean sea-level rise (GMSL; relative to 1986–2005) for the year 2100 have ranged between 0.26 m to 0.77 m (under a 1.5°C warming and representative concentration pathway [RCP] 2.6 scenario); however, there has been little consensus from modelling efforts (Hoegh-Guldberg et al. 2018). Under a 2°C warming scenario, global mean sea-level rise is projected to be approximately 0.04 to 0.16 m higher than a 1.5°C warming scenario. In addition, potential irreversible instability and/or loss of ice sheets in Antarctica and Greenland could result in much higher projections of sea-level rise over thousands of years (IPCC, 2018).

Representative Concentration Pathways (RCPs) are greenhouse gas concentration trajectories used by the IPCC during the Fifth Assessment Report (2014). Four pathways were selected to describe potential future climate conditions based on projected greenhouse gas emissions (RCP2.6, RCP4.5, RCP6 and RCP8.5; values are estimates of radiative forcing with units W/m^2). The lowest scenario RCP2.6 assumes a peak in emissions between 2010 and 2020 with emissions declining afterwards. In the highest emissions scenario (RCP8.5) emissions are expected to continue to rise over the twenty-first Century (Figure 1, below; IPCC, 2014; Oppenheimer et al., 2019).

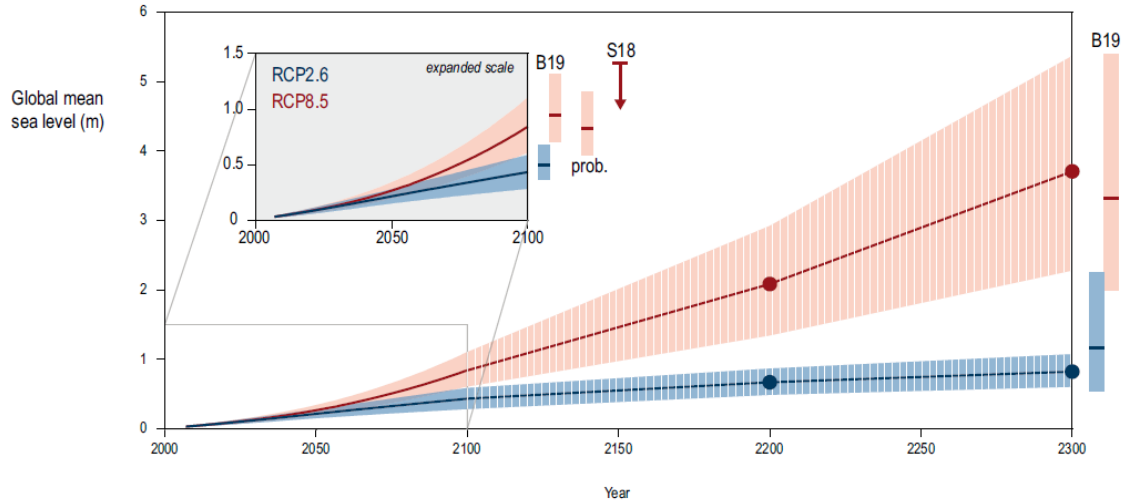

Figure 1: Projected sea-level rise until 2300 for RCP pathways 2.6 and 8.5. From Oppenheimer et al., 2019.

Future estimates of global mean sea-levels (GMSL) are highly dependent on which RCP emission scenario is followed. Under all scenarios, sea-level rise is projected to increase; however, GMSL at the end of the twenty-first century is projected to be lower under RCP2.6 (0.29 m to 0.59 m) compared to RCP8.5 (0.61 to 1.10 m). Under a high emissions scenario, rates of sea-level rise are projected to be 10 to 20 mm/year by the end of the century. Sea-levels are expected to continue to rise for several centuries under all emissions scenarios due to deep ocean heat uptake and Greenland and Antarctic ice sheet loss (with Antarctica potentially adding 28 cm of sea-level rise under high emission scenarios; Oppenhiemer, 2019).

Changes to global mean sea-levels do not necessarily mean that coastal zones around the world are affected in the same way. Regional and local coastal sea-levels will differ due a variety of factors and may be greatest in coastal areas where rates of isostatic changes, compaction and anthropogenic subsidence are high. Recent observations from tide gauges and satellite altimetry have shown this variability (Hamlington et al., 2018). Local relative sea level trends measured by tide gauges since 1992 are presented in a map produced by the US National Oceanic and Atmospheric Administration (NOAA, 1992-2018).

In deltas around the world, anthropogenic subsidence can be a primary factor in current high rates of local sea-level rise. This is often due to these areas having flat, low lying topography and densely populated urban centres. Globally, modern anthropogenic subsidence rates in major deltas range from 6 mm/year to 100 mm/year (Oppenheimer et al., 2019).

Relative rates of sea-level rise have been estimated on the west coast of north America over the twentieth century. For western Canada and northwestern United States, Mazzotti et al. (2008) calculated sea-level trends using tide gauge records. Results from this study indicate a regionally average rate of absolute sea-level rise of 1.8 mm/year which was close to the global average of 1.7 mm/year (as calculated by the IPCC Fourth Assessment; Church et al., 2013). Sea level trends as measured by NOAA indicate a relative sea level trend of 0.53 mm/year (±0.21 mm/year) for Vancouver over the period of 1909 to 2018. Within the Fraser delta, relative sea-level rise is also affected by local subsidence rates which are estimated to be approximately 2 mm/year on average throughout the Delta (Lambert et al., 2008; Mazzotti et al, 2009).

In British Columbia, the provincial government has provided guidelines for flood risk management; however, local governments are responsible for developing their own risk and adaptive management options (Barron et al., 2012). Current recommendations for flood construction levels include 0.5 m of sea-level rise allowance for the year 2050, 1.0 m for the year 2100, and 2.0 m by the year 2200 (Arlington Group, 2013). Sea-level rise of 1 m over the next century includes ongoing subsidence of the Fraser River delta (Figure 2).

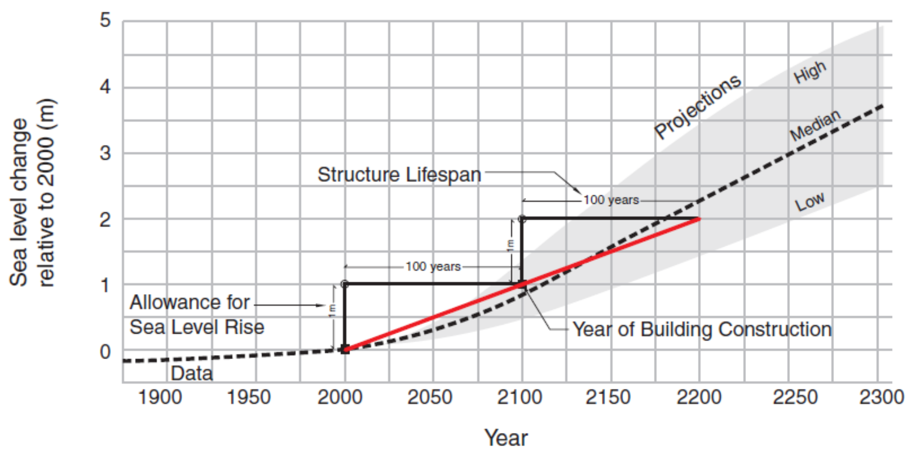

Figure 2: Recommended flood construction allowance for sea-level rise. From Ausenco Sandwell, 2011.

2.2. Tidal Marsh Ecology

Specific marsh characteristics are determined by a variety of factors including topography, sediment supply, nutrient supply, tidal range, water quality, wave energy and species of vegetation (Church and Hales, 2007). Macrophyte zonation occurs due to individual species tolerance to abiotic factors including soil moisture content, redox state, nutrient limitation, pH and salinity as well as biotic factors such as competition. Abiotic factors are typically correlated with elevation and hydroperiod (Moffett et al., 2015, Pratolongo et al., 2019). Changes in these inputs over time can potentially lead to changes in ecosystem geomorphology and functioning (Moller and Christie, 2018).

Flow of water through tidal marsh plays an important role in the formation, growth and function of the marsh. Water provides nutrients and sediments as well as energy to the system, and continued persistence of the marsh depends on both the quantities of materials and the hydrodynamic energy transported to the system via water. While flow of water is critical to tidal marsh development, marshes contribute to their own nutrient and sediment budgets through growth and decay of plant and invertebrate biomass and can affect hydrodynamic energy transfer through the wetland (Moller and Christie, 2018).

Sediment supply allows the marsh to build vertically and horizontally. Sediment inputs can keep the marsh in equilibrium with potentially erosion as well as any changes in sea-level. For a marsh to prograde or aggrade, there needs to be a net positive sediment input to counteract sediment being removed by waves and tides (erosion) as well as longer term issues such as subsidence and sea-level rise (Perillo, 2019).

Tidal marshes build vertically through sediment and plant organic matter accumulation. At high tides, when the surface of the marsh is flooded, sediment deposition occurs. As such, flooding depth and duration are important factors controlling sedimentation rates (Figure 3;

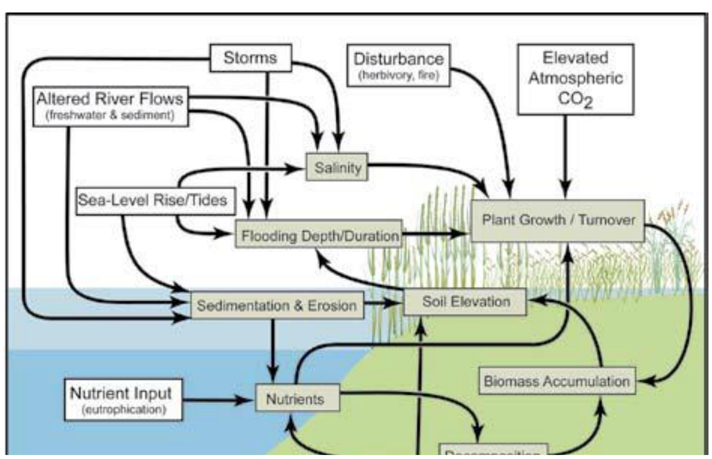

Figure 3: Conceptual model of processes influencing wetland development and accretion. From Cahoon, 2009.

Cahoon, 2009). Since tidal marshes exist on a sloped plain, the frequency and duration of tidal inundation decline with elevation. Under simple sediment accretion models, net sediment accretion increases salt marsh elevation and reduces hydroperiod, ultimately leading to an equilibrium marsh elevation (Pratolongo et al., 2019). Tidal marsh vegetation can also influence sediment deposition potential by capturing and binding sediments in leaves, stems and roots (Stumpf, 1983). Models that consider these accretionary effects of vegetation predict that sedimentation and accretion rates are influenced by factors including areas near channels that are more frequently flooded and areas where dense vegetation occurs allowing for increased capture of suspended sediments (Fagherazzi et al., 2013; Kirwan et al., 2016). Lateral marsh migration typically occurs due to wave erosion which is one of the main sources of marsh loss throughout the world (Pratolongo et al., 2019).

2.3. Tidal Marsh Response to Sea-Level Rise

As described above, tidal marshes build vertically through complex processes involving accumulation of sediments and organic matter. A key process in tidal marsh evolution involves response to relative changes in sea-level. Historically, tidal marshes elevations have been considered at equilibrium with sea-level rise with long-term accretion rates matching rates of sea-level rise (Hill et al., 2013). The capacity of tidal marshes to maintain stability with changing sea levels is complex, however, and is affected by local factors including slope, soil erodibility and vertical accretion dynamics. Accretion dynamics are also affected by other human and climate related drivers (e.g., changes in river discharge and precipitation; Cahoon, 2009). Increased atmospheric CO_2 could also lead to an increase in net primary production and carbon sequestration if other factors do not limit plant growth (Rozema et al., 1991).

In the absence of negative anthropogenic effects and under historic rates of sea-level rise, many coastal marshes can maintain their position within a tidal range given enough inorganic and organic sediment supply (Morris et al., 2002). One of the major potential stressors to current marsh stability is projected increased rates of sea-level rise over the later half of the century. Tidal marshes will maintain their structure and function only if they can accrete sediment to maintain elevation relative to the rate of sea-level rise. Whether or not surface elevation of the tidal marsh can keep pace with sea-level is key to determining tidal marsh vulnerability. As sea-levels rise, initial deepening of the tidal marsh platform may be mitigated by increased flood period. Extended flood periods can provide more opportunity for sediment deposition and the marsh may be able to approach a new equilibrium elevation. However, extended inundation periods may reduce macrophyte productivity, ultimately reducing the ability of the marsh to trap sediments. If the tidal marsh cannot accrete sediment to keep pace with sea level rise, the marsh must either migrate to higher ground or become submerged for longer periods and ultimately transition to mudflat (Morris et al., 2002). As marshes accrete sediment and increase in elevation, they may migrate landward depending on factors such as slope, sediment supply and rate of sea level rise; however, dikes or other anthropogenic hard structures prevent landward migration.

Several models attempt to predict tidal marsh vulnerability to sea-level rise. Early studies predicted large losses of coastal wetlands. More advanced models take into account ecogeomorphic feedbacks between plant growth and geomorphology (Kirwan et al., 2016). These models indicate that tidal marsh vulnerability is dependent on biophysical feedbacks that accelerate accretionary processes under a rising sea level. Kirwan et al. (2016) argue that marsh vulnerability to sea-level rise can be exaggerated by not considering biophysical feedback processes and potential for migration inland. Analysis of five dynamic accretion models indicate that marshes with adequate sediment supply may survive in place under conditions of high rates of sea level rise (10 mm/year to 50 mm/year) and that threshold sea-level rise rates for marsh survival depend on suspended sediment concentrations and local tide range (Kirwan et al., 2016).

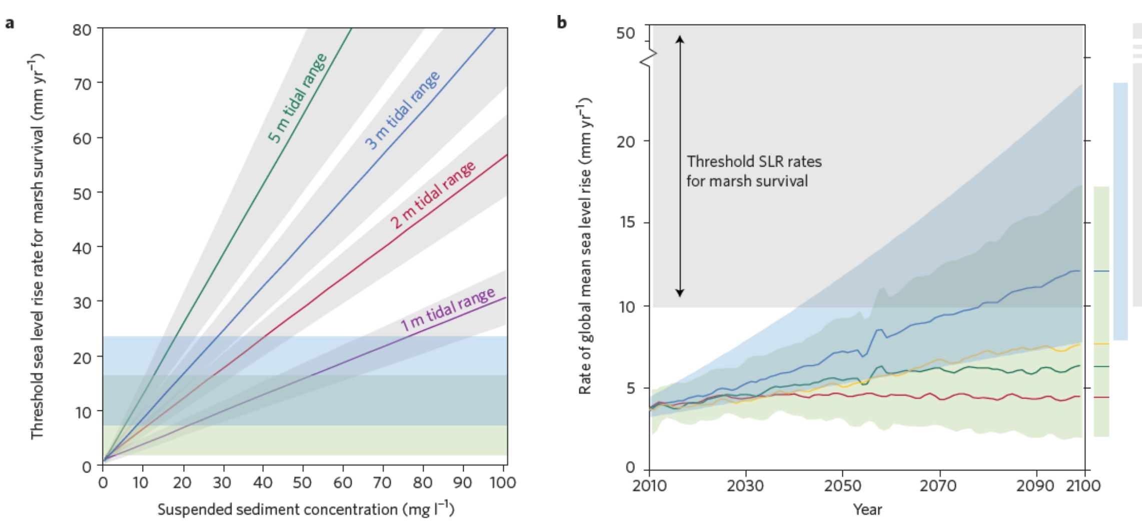

Figure 4: Maximum rates of sea level rise for marsh survival. From Kirwan et al. (2016).

Marshes that failed to survive even low rates of sea-level rise (i.e., few mm per year) occurred in areas with low suspended sediment concentrations (<10 mg/L) or in areas where tidal ranges were low (few tenths of a metre; Figure 4). In addition, mean rate of elevation increase for high marshes was 3.0 mm/year while the mean rate of elevation increase for low marshes was 6.9 mm/year, suggesting that studies that focus on high marsh accretion rates may overestimate vulnerability to sea-level rise. This result also shows that the low marsh tends to accrete sediment at a much higher rate than the high marsh due to extended periods of tidal and suspended sediment exposure. It should be noted that most of the marshes included in the study are located along the Atlantic Coast of North America with dominant macrophyte species including *Spartina patens* and *Spartina alterniflora*. In Atlantic and Gulf Coast marshes, increased periods of inundation tend to increase productivity of dominant vegetation (*Spartina alterniflora*) which may lead to higher sediment and organic matter accretion rates. In the brackish marshes of the Fraser River delta, at least one of the dominant low marsh species (*Bolboschoenus maritimus*) tend to reduce growth with increased inundation and may be less resilient to accelerating sea-level rise (Hill et al., 2013).

Zhu et al. (2020) examined tidal marsh resilience at the marsh-tidal flat interface. Field and model results indicate that small increases in water depth and wave forcing due to sea-level rise can cause a decrease in marsh resilience to lateral erosion at the seaward boundary. This may be due, in part, to reduced ability of vegetation to re-establish at eroding marsh edges. Results demonstrated that seed persistence on the tidal flat surface was reduced substantially under increased wave disturbance. Tidal marshes that can accrete vertically with increases in sea-level rise may still be susceptible to erosion at the seaward boundary, especially if there is no migration potential for the marsh (Zhu et al., 2020).

Moving forward, it will be challenging to assess how well ecogeomorphic feedbacks will reduce tidal marsh vulnerability to sea-level rise as anthropogenic influences often interfere with these processes, including effects from nutrient inputs, sediment supply rates, and potentially subsidence rates (Pratolongo et al., 2019).

2.3.1. Coastal Squeeze

In addition to vertical accretion, coastal wetlands may be able to migrate landward in response to sea-level rise; however, if inland migration is prevented due steep slopes or physical barriers (e.g., dikes), the marsh will be unable to expand (Pratolongo et al., 2019). If a physical barrier prevents migration, the marsh will either become submerged for more extended periods of time or build vertically to match sea-level rise and become more susceptible to wave erosion at the seaward edge. If there is an obstruction preventing landward migration and the marsh is able to keep pace with sea-level rise with no erosion at the seaward edge, then the marsh would likely remain in place. Lateral erosion coupled with no migration potential can result in the marsh becoming narrower over time, a result known as coastal squeeze (Figure 5; Cahoon, 2009).

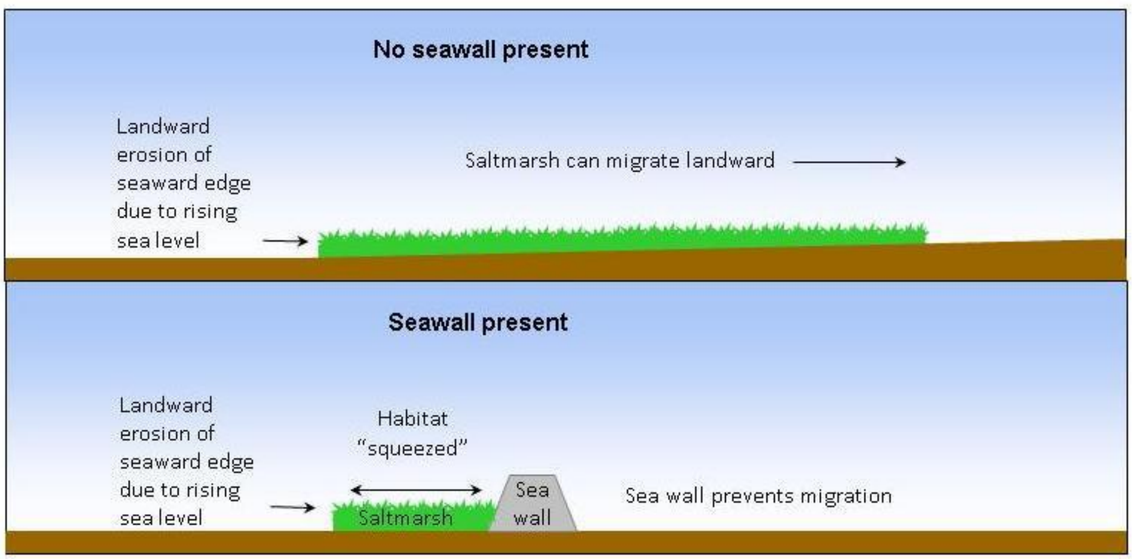

Figure 5: Illustration showing the concept of coastal squeeze. From Pontee, 2013.

Chapter 3.

Fraser River & Delta

The Fraser River is the largest river along the west coast of Canada (both for volume of water and sediment discharged; Milliman, 1980). The river drains a watershed area of approximately 230,000 km^2 and is the longest river in British Columbia, flowing approximately 1,400 km from the Rocky Mountains and draining into the Strait of Georgia. Mean long-term discharge calculated at the Hope hydrological station was 2,748 m^3/s for the period of 1912 to 2001 (Mikhailov et al., 2007). From the fall to early spring period, discharges are relatively low (mean long-term minimum discharge of 687 m^3/s at the Hope station) while late spring and early summer periods are dominated by a snowmelt freshet (mean long-term maximum discharge of 8,705 m^3/s at the Hope station). During the high-flow freshet period, about 72.5% of river runoff occurs (Mikhailov et al., 2007).

Approximately 11,000 years ago, the Fraser River delta began to form as the Fraser Lowlands deglaciated, and sediments were deposited by the Fraser River. During this time, isostatic rebound of the Fraser Lowland caused relative sea levels to fall about 12 m below present levels. Sea levels subsequently rose to within 2 m of present levels around 5,000 years ago and have been relatively stable since then (Williams, 1988). The eastern edge of the delta stabilized around the same time (5,000 years ago) and ongoing deposition has expanded the delta westward. The delta currently extends 15 km south and 23 km west from the city of New Westminster with an estimated area of 975 km^2 (Atkins et al., 2016; Figure 6).

Within the delta, the Fraser River splits into four distributary channels that enter the Strait of Georgia along the western delta front, including (from north to south) the North, Middle and Main Arms and Canoe Pass. Sea Island at the north end of the delta is dissected by the North and Middle Arm. Lulu Island is located between the Middle Arm and the Main Arm and Westham Island is separated from the mainland by the Main Arm and Canoe Pass. Approximately 80% to 85% of Fraser River flows run through the Main Arm with the rest of the flow distributed among the remaining North and Middle Arms (Williams, 1988).

The shoreward perimeter of the delta is approximately 40 km long with 27 km of shoreline facing westward towards the Strait of Georgia and 13 km facing south towards Boundary Bay (Clague et al., 1991). To the west, the delta is fringed by gently sloping tidal flats extending from the diked edge of the delta to subtidal areas and forming Sturgeon Bank and Roberts Bank. The intertidal area is approximately 158 km^2 and includes sand flats, mud flats, eelgrass and marsh communities (Hutchinson, 1988).

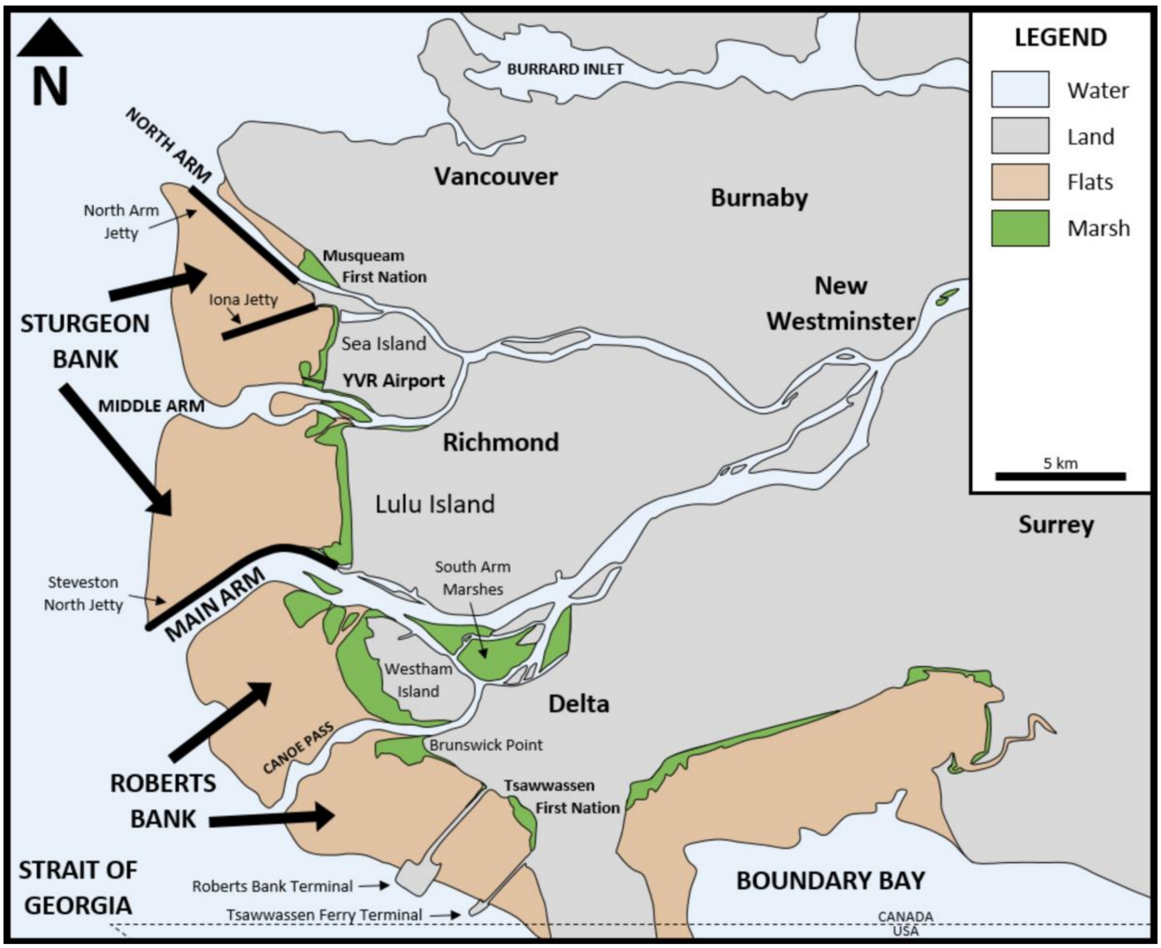

Figure 6: Map of the Fraser River delta including its leading-edge ecosystems. Modified from Balke (2017).

Tides in the Strait of Georgia are mixed semidiurnal with an average tide height of approximately 3.0 m and a typical range of 5.0 m. Tides generally flood to the northwest and ebb to the southeast (Barrie and Currie, 2000; Mikhailov et al., 2007; Atkins et al., 2016). Tidal forcing tends to deflect the discharge plume to the north (Atkins et al., 2016). Depending on tidal and fluvial conditions, salt wedge influence can be observed inland to New Westminster (Atkins et al. 2016). Mean wind direction is from the east while maximum wind speeds typically come from the west resulting in larger waves impacting the delta front (Williams et al., 2009). Average height of wind generated waves is approximately 0.6 m with maximum heights of 2.9 m observed during winter storms (Atkins et al., 2016).

3.1. Sediment Transport

The Fraser River discharges sediments through the Fraser delta and into the Strait of Georgia with an average annual sediment load of approximately 12,300,000 m^3/year and 450,000 tonnes of organic matter (Schaefer, 2004). Annual sediment load consists of 65% silt/clay and 35% sand, notable among large deltas for the high portion of sand (NRC, 2008). As sediment is discharged through the delta and into the Strait of Georgia, it forms a plume, typically consisting of fine-grained sediments; however, both river flow and sediment transport vary greatly depending on the time of year. An estimated 80% of annual sediment loads are discharged during freshet (late spring/ early summer) with most sediments discharged through the Main Arm (Williams, 1988). During the three-month freshet period, coarse bedload materials are resuspended, and more than half of the discharged sediments are sand (Millman, 1980). During other periods (and during freshet high tides), a salt wedge can infiltrate upstream and trap bedload sediments in the estuary (Barrie and Currie, 2000). The salt wedge can migrate inland as far as New Westminster (Atkins et al. 2016), and the Fraser River can be tidally influenced as far upstream as Sumas River (Clague et al., 1983).

The Fraser River delivers a substantial amount of sediment to the Fraser delta and estuary. The sandy depositional regime of the Fraser River is different than many mud-dominant river systems. Coarse sands and silts form and stabilize channel bottoms and settle along the delta front during high flows (Millman, 1980, Williams et al., 2009). Historically, longshore drift of depositional sands built up the tidal flats along the delta front; however, natural processes have been interrupted by ongoing human activities (Millman, 1980). Silt and clay load generally pass through the lower reaches of the Fraser River in suspension, yet contributing greatly to physical and ecological development of the delta (Attard et al., 2014). The southern delta front (including Boundary Bay) is separated from the western delta front by Point Roberts and has been without direct supply of Fraser River sediment since Point Roberts became connected to the rest of the delta (Williams, 1988).

Suspended sediment deposition and distribution within the Fraser River discharge plume are affected by tidal, wind and fluvial processes. Typically, suspended sediments in the Fraser River plume are pulled north by Coriolis effects on dominant flood tides and even during ebb tides (Barrie and Currie, 2000; Figure 7). Sediment grain size deposited along the intertidal fore slope of the Fraser delta generally increases with increasing water depth as silts and clays are carried into the tidal marshes through suspension in the water column (Williams, 1998). Along the intertidal areas of the delta front, the main sediment mobilizing force is caused by waves. High tides and strong westerly winds can result in increased flooding and larger waves. On Sturgeon Bank, dunes have formed where wave energy is strongest, while at Roberts Bank, these effects are reduced due to jetties, as described below (Williams et al., 2009).

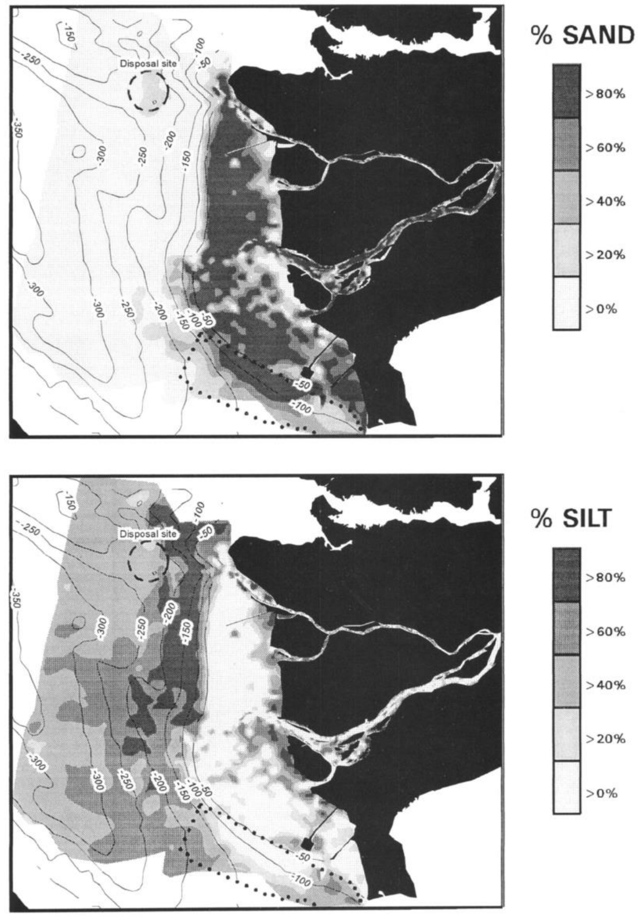

Figure 7: Percentage of sand and silt in Fraser River delta surficial sediments. From Barrie & Currie (2000)

3.2. Human Impact on the Delta & Sediment Distribution

As with many other major deltas around the world, the Fraser delta has long been a focal point for human settlement and is exposed to a variety of human development pressures. Natural deltaic processes have been limited since the 1800's due to human activity such as dredging for shipping and navigation access, channel diversion and channelization (e.g., jetties) for land use, and construction of armouring and flood management structures (e.g., dikes; Figure 8).

Sediment trend analysis completed by McLaren and Tuominen (1998) suggests that sand deposition on the intertidal flats is no longer caused by natural deltaic processes. Evidence of net erosion and reduced sand and silt deposition noted during that study were attributed primarily to channelization and removal of sands through dredging.

3.2.1. Training Structures & Flood Protection Diking

Church and Hales (2007) and Atkins et al. (2016) provide a timetable of river training structures near the Fraser delta front from the 1800's to the 2000's that affect the sedimentary regime within the Delta. A summary of major training structures is provided below:

- The North Arm Jetty (constructed from 1914 to 1917) deflects flows away from the north end of Sturgeon Bank.

- The 9 km long Steveston North Jetty was constructed from 1912 to 1932 and is still relied upon today for navigation, shipping, and industry. This rock jetty runs along the northern bank of the Main Arm and controls the position of the arm, directs river flows and sediment away from the south end of Sturgeon Bank and out to the seaward edge of the tidal flats. In 1978 several gaps were installed in the jetty to allow fish, water and some fine sediments to be directed towards Sturgeon Bank. In 2019, additional breaches were constructed by the Raincoast Conservation Foundation to further increase salmonid access from the Main Arm to Sturgeon Bank.

- The Steveston South Jetty was initially constructed in the early 1930's and reconstructed in 1954 to restrict Main Arm drainage to the south.

- The Woodward Training Jetty and Dam was constructed in the 1920's and 1930's to control the south side of the Main Arm, promote scour and reduce dredging requirements. Training works in this area sand bars on the downstream side and have contributed to the formation of the South Arm Marshes. Recently, Ducks Unlimited Canada (DUC) has been modelling effects of potential breaches in the dam and may proceed within the next few years.

Only the Middle Arm of the Fraser River has relatively unobstructed discharge to Sturgeon Bank; however, it accounts for little overall sediment discharge. In addition to training structures, several other structures have been built in the delta that alter the natural sediment distribution regime including the Iona causeway (constructed in 1961), Roberts Bank Terminal (developed and expanded over the period of 1970 to 1997) and Tsawwassen Ferry Terminal (developed over the period of 1959 to 1960), as well as Roberts Bank Terminal 2 which is currently proposed to expand existing port facilities.

Figure 8: Extent of dikes and jetties in the Fraser River Delta in 2020. Satellite Imagery provided by NASA Earth Observatory (2011) clearly displaying sediment plume discharge from the Main Arm of the Fraser River.

Construction for flood protection diking has also been ongoing since 1906 (Atkins et al, 2016). The majority of current flood protection works were constructed between 1968 and 1995. As with other training structures in the Fraser River delta, flood protection diking along the banks of the Fraser River prevent flooding and flood sediments from dispersing through the estuary, ultimately limiting the number of pathways for sediment to reach the delta front.

3.2.2. Dredging

Dredging in the lower Fraser River has occurred since the 1800's when navigation channels were first established. From the early 1900's to 1998, the Department of Public Works and Government Services Canada maintained dredging programs on the Fraser River for navigation and shipping. The Vancouver Fraser Port Authority took over the dredging program the following year and focused dredging efforts on maintaining port operation (Broś, 2007).

Dredging is a necessary component of maintaining existing navigation channels between New Westminster and the mouth of the Fraser River. Dredging also helps to maintain water levels such that dike overflooding does not occur. Prior to 1998, non-navigational dredging was also done to support the aggregate industry (i.e., sand and gravel construction materials); however, borrow dredging no longer occurs in the lower Fraser River (Atkins et al., 2016). Several million tonnes of sand are dredged annually from the various arms of the Fraser River (FREMP, 2006).

The Fraser River Estuary Management Program (FREMP) was established to coordinate and streamline environmental management in the Fraser River Estuary among federal, provincial and municipal partners. In 1998, FREMP developed a sediment budget to ensure sustainable sediment removal in the lower Fraser River. The budget was followed by the development of Dredge Management Guidelines in 2001, a review of the budget model in 2002, and the development of an Estuary Management Plan to provide a link between the navigation system and relationship to the surrounding environment. Dredge Management Guidelines were incorporated into the Management Plan in 2005. The sediment budget was designed such that the average amount of sediment removed over a 5- to 10-year period did not cause a net change in the riverbed with an original dredging target of 70% of the inflow. The sediment budget only considers sediments consisting of coarse sand with a grain size between 0.177 mm and 2.0 mm. The following is a summary of dredging strategies for lower Fraser River channel segments as described in the Environmental Management Strategy for Dredging in the Fraser Estuary (2006).

- For the tidal segments of the North and Middle arms, dredging is not carried out regularly. Dredging for boat access and log booming grounds occurs on occasion upstream of Sea Island and the North Arm Jetty. Upstream, from the eastern end of Sea Island to the New Westminster Quay, the North Arm channel is infrequently dredged for channel navigation, barge loading facilities, and sawmill sites on a five-year cycle.

- The Sand Heads channel, which extends from the Steveston North Jetty at the mouth of the Fraser River, is critical for larger ship access into the Fraser River. This area requires ongoing channel dredging (more than once a year). The navigation channel is dredged to accommodate 11.5 m draft vessels at low tide.

- The South Arm Tidal Channel (Main Arm) is frequently maintained for navigation along the north bank. Infrequent dredging for small craft harbour access also occurs in localized areas for vessel draft accommodation.

Dredged material is either disposed of at the Sand Heads ocean disposal site or sent to transfer pits. Disposal-at-sea permitting is required to dispose of dredged materials and, under the Dredge Material Management Program, the Port Authority prefers to limit ocean disposal in order to maximize beneficial use of dredge material. Dredged material can be used for construction, land reclamation and habitat creation. According to the FREMP (2007) sediment budget annual report, average volume of sand dredgeate was approximately 2,000,000 m^3 per year over the period of 1997 to 2007. In 2007, approximately 20% of the dredgeate material was disposed of at sea (FREMP, 2007).

3.3. Elevation & Subsidence

Several isostatic factors affect the elevation of the delta including land mass rebound (from ice age depression), subsidence, sediment density and tectonic plate movements. It is estimated that the rate of land mass rebound is approximately 0.25 mm/year (Thomson et al. 2008). Subsidence can be caused by sediment accumulation and consolidation over time, with these factors estimated to cause 1-2 mm/year of subsidence annually in the Fraser delta (Williams et al., 2009). Major construction projects may also contribute to anthropogenic subsidence rates; however, rapid subsidence caused by projects such as the BC Ferries terminal have been observed to slow substantially within a period of 20 years (Mazzotti et al., 2009). Subsidence is common among major deltas; however, the addition of new river sediments can compensate for subsidence if human intervention does not divert sediment away from the delta (typically through dredging and training structures; Bornhold 2008; Thomson et al. 2008). The resulting implication of ongoing subsidence is that it can amplify the effects of relative sea-level rise.

3.4. Flood Adaptation Planning

In densely populated coastal communities, flood management strategies will require unprecedented levels of investment to combat twenty-first century sea-level rise and storm events. Sea-level rise adaptation strategies are typically grouped into four methods including “Protect”, “Accommodate”, “Retreat” and “Avoid”. Protection is the most conventional strategy commonly employed in densely populated coastal communities, which entails reactive measures to reduce flood risks using hard engineering structures such as dikes, seawalls, levees, groynes and dams. Flood protection measures that use this strategy disrupt natural delta sedimentation regimes and are often considered short-term solutions under projected long-term sea-level rise (Arlington Group, 2013; Oppenhiemer et al, 2019).

Dikes protect much of the Fraser River delta including sea dikes along the delta foreshore of south Vancouver, the Tsawwassen First Nation, and Cities of Richmond, Delta and Surrey. In general, lower Fraser River dikes do not meet current provincial flood control standards, and most dikes were built to design criteria established in the 1960’s and 1970’s (NHC 2015). In 2011, the province of BC developed *Climate Change Adaptation Guidelines for Sea Dikes and Coastal Flood Hazard Land Use* (Ausenco Sandwell, 2011) to establish sea-level rise flood protection requirements to assist local governments and land use managers in developing long-term flood protection strategies. Over the twenty-first century there will be increased risk of flooding (due to sea-level rise and storm surge) and the majority of existing dikes will require enhanced dike crest elevations and widths (Arlington Group, 2013). The Ausenco Sandwell (2011) report estimated that costs for developing flood protection measures in the Lower Mainland to adapt for projected sea-level rise, including land acquisition and other works would be approximately $2.8 billion.

Ecosystem-based adaptation strategies are getting more attention as they provide opportunities for both ecosystem conservation and flood protection (through wave attenuation and shoreline stabilization; Arlington Group 2013). Waves lose energy and wave height when passing through vegetated foreshores. Mudflat and tidal marsh communities can reduce wave energy through bottom friction, depth induced wave breaking and wave attenuation processes. A recent global meta-analysis on the protection benefits from coastal ecosystems showed that salt marshes reduce wave heights by 62% to 79% (Narayan et al., 2016). Reductions in wave energy and height lead to reduced wave run-up and dike overtopping, which can limit breaching and the amount of inner slope erosion (Vuik et al., 2016).

Over the next century, cost implications and maintenance concerns may lead to increased implementation of alternate flood protection measures (i.e., “Accommodate” and “Retreat” strategies); however, it is expected that many high-density coastal communities around the world, including the Fraser River delta, are likely to continue using “Protect” strategies in the short term (Oppenhiemer, 2019). Where fixed hard shoreline protection structures (i.e., dikes) remain in place or expand for future flood protection, tidal marshes are, in many cases, prevented from upland migration and may eventually be drowned as a result of “coastal squeeze” (discussed in Chapter 3; Glick et al., 2007; Lovelock et al., 2015; Mills et al., 2016).

3.5. Fraser Delta Tidal Marshes

Throughout the development of the Fraser River delta, sediment deposition has led to the development of extensive sand and mud flats. Estuarine conditions in these areas provide a highly productive environment for the growth of algae, diatoms, eelgrass and one of the largest areas of intertidal marshes on the west coast of North America (Church and Hales, 2007; Williams et al., 2009). These ecosystems provide habitat for a diversity of invertebrate species and are considered critical habitat for a variety of fish and mammals, as well as wintering and migratory bird species. The Fraser River is also one of the most important salmon-spawning rivers in the world and the estuary provides critical rearing grounds for juvenile salmonids during transition from fresh to saltwater (Levings, 2016). In addition, the mudflat-marsh-biofilm complex provides important feeding grounds for hundreds of shorebird species on their annual migration along the Pacific Flyway (Church and Hales, 2007; Williams et al., 2009).

In order to better conserve these critically important fish, wildlife and habitat values, the majority of the tidal ecosystems in the Fraser River estuary have been designated as Wildlife Management Areas (WMAs) under the provincial *Wildlife Act.*

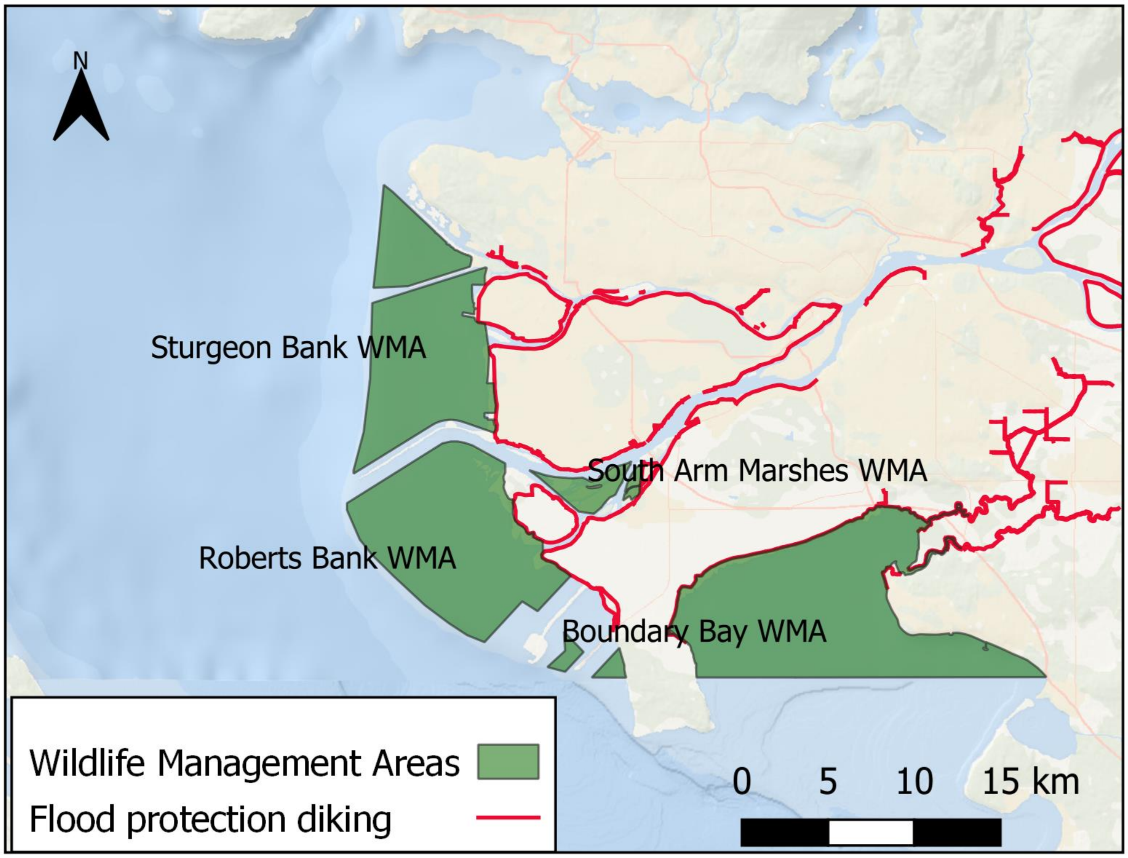

Figure 9: Fraser River delta front Wildlife Management Areas

Wildlife Management Areas are the primary designation of BC provincial Conservation Lands. The primary purpose of these areas is to conserve and manage important habitat for the benefit of regionally or internationally significant fish and wildlife species. Activities are constrained within these conservation areas and written permission is required for any use of the land or resources. WMAs provide the Ministry with important tools for regulating the land, consolidating land management and planning processes, increasing opportunities for scientific research and resources and helping provide public awareness for the importance of these areas.

Sturgeon Bank, Roberts Bank, South Arm Marshes, and Boundary Bay WMAs have been established along the Fraser Delta front and cover an area of over 26,000 hectares (Figure 9; Table 1). The Boundary Bay, Sturgeon Bank and South Arm

Marshes WMAs are internationally designated as Western Hemisphere Shorebird Reserve Network sites.

Table 1: Fraser River delta front Wildlife Management Areas

Wildlife Management Area	Date Designated	Purpose	Size (hectares)
South Arm Marshes	1991	Management of critical habitat for fish, waterfowl, shorebirds, raptors, songbirds and small mammals	937 ha
Boundary Bay	1995	Conservation of critical, internationally significant habitat for year-round, migrating and wintering waterfowl populations, along with important fish and marine mammal habitat	11,470 ha
Sturgeon Bank	1998	Conservation of critical, internationally significant habitat for year-round, migrating and wintering waterfowl populations, along with important fish habitat	5,152 ha
Roberts Bank	2011	Management of critical habitat for fish, waterfowl, shorebirds, raptors, and other species	8,770 ha

Tidal marshes at the Fraser Delta front are characterized as brackish or salt marsh based on plant species composition, and their relative locations are dependent on the salinity regime which is influenced by tidal and river water flow regimes (Hutchinson, 1988). Delta front marshes at Sturgeon and Roberts Banks, including Sea Island, Lulu Island, Westham Island and Brunswick Point are considered brackish in nature while tidal marshes at Boundary Bay support salt marsh as this area has been separated from direct Fraser River influence. Salt marshes also occur at the Iona intercauseway on Sturgeon Bank and between Brunswick Point and the Tsawwassen ferry terminal. The brackish marshes have higher plant species diversity than the salt marshes, and typical species in the brackish communities include three-square bulrush (*Schoenplectus pungens*), seacoast bulrush (*Bolboschoenus maritimus*) and Lyngbye's sedge (*Carex lygnbyei*). The salt marshes support species tolerant of greater salinity including pickleweed (*Sarcocornia pacifica*), orache (*Atriplex patula*) and saltgrass (*Distichlis spicata*; Williams et al., 2009).

As described above, marsh zonation occurs primarily due to gradients of limiting factors including flooding regimes, salinity gradients and substrate characteristics (Hutchinson, 1988). As such, marsh zonation occurs both with proximity to freshwater discharge and with changes in elevation. The marsh platform is divided into zones based on tidal thresholds and plants that exist at lower elevations are exposed to increased periods of inundation and higher salinities. The brackish marshes of Roberts Bank and Sturgeon Bank extend from a lower elevation of approximately 3.2 m chart datum to the higher high-water mark at 4.8 m chart datum at the base of the sea dikes (Williams et al., 2009). Three-square bulrush occurs at the leading edge (lowest elevation) where the marsh may be submerged 60% of the time (Hutchinson, 1988). It forms dense stands on silty-sandy substrates and is an early colonizing species capable of withstanding higher wave energy. The three-square bulrush community transitions into low to middle marsh seacoast bulrush stands or Lyngbye's sedge depending on salinity exposure (Balke 2017). Marsh substrates are primarily silts with some fine sands and clay. Mean grain sizes decrease with marsh elevation, indicating lower tidal energy in the high marsh. At low tide, soil water content is highest in the high and middle marsh due to the higher proportion of fine sand in low marsh substrate and higher rate of drainage of this substrate (Hutchinson 1982).

Long-term accretion rates in the tidal marshes along Lulu Island range from 2.6 mm/year to 8.5 mm/year based on Cs-137 fallout (Williams and Hamilton, 1995). Highest sedimentation rates (8.5 mm/year) were found in the middle marsh zone. The upper part of the lower marsh zone showed accretion rates of 6.3 mm/year and 6.1 mm/year. Rates were lowest at lower low marsh (2.6 mm/ year and 3.7 mm/year) likely due to sparse and patchy areas at the leading edge (Williams and Hamilton, 1995). In several locations, a minor scarp has been observed just below the Mean Lower High-Water tidal threshold (3.6 m above chart datum; Burgess, 1970, Hutchinson 1982). Hutchinson describes the minor scarp as a boundary between compacted, fine-grained sediments with distinct drainage patterns above and a zone of partially stabilized mud and silt, below.

In addition to the brackish and salt marsh communities, extensive eelgrass beds occur at intertidal and shallow subtidal elevations in the Fraser delta. At higher elevations, the dominant eelgrass species is the introduced *Zostera japonica* while at lower elevations, *Z. marina* dominates (Williams et al., 2009). English cordgrass *(Spartina anglica*) is an aggressive invasive species that has been observed in several areas including Boundary Bay, Brunswick Point and along the Deltaport Causeway (Balke, 2017).

3.5.1. Potential Fraser Delta Tidal Marsh Response to Sea-Level Rise

As described above, sediment supply is a primary component controlling vertical accretion dynamics and is a major factor in tidal marsh resilience to sea levels rise. In the Fraser River delta, sediment supplies have been altered through a number of anthropogenic means; river training structures, causeway and jetty construction, navigation dredging, and flood protection diking have likely reduced the amount of sediments reaching the delta front marshes. Williams and Hamilton (1995) attributed a 51% reduction in sediment supply between the periods of 1954 to 1964 and 1964 to 1991 at Roberts Bank to a reduced concentration of suspended sediments caused by the development of engineering structures and dredging activities. Recent 2019 breaches in the Steveston Jetty are anticipated to help increase the amount of fine sediment reaching the south end of Sturgeon Bank; however, this is one of several barriers for sediment supply to delta front tidal marshes. The Fraser River discharges a large quantity of sediment every year, yet the amount of sediments reaching the delta front tidal marshes may not be adequate to match rates of sea-level change, especially through the later part of the century.

Flood protection diking limits the ability for the delta front tidal marshes to migrate landward as sea-levels rise and under long-term adaptation planning, managed retreat is not currently prioritized in most areas of the delta. The Fraser Basin Council is currently developing Phase 2 of the Lower Mainland Flood Management Strategy initiative which is aimed at developing a regional strategy for reducing flood risk and increasing lower Fraser community resilience. As part of Phase 2 assessments, mitigation options are currently being examined for hard and soft engineered flood mitigation structures. Coastal flood adaptation strategies are being implemented by the Cities of Surrey, Richmond and Delta and include exploration of ecosystem-based adaptation and hybrid strategies (e.g., living dikes).

For the tidal marshes of the South Arm Marshes Wildlife Management Area, dikes back the marshes on several of the main islands; however, those dikes are not likely to be maintained for future flood protection. Restoration and tidal marsh expansion opportunities exist in areas where dikes can be breached. There is limited space for these tidal marshes to migrate; however, they are likely exposed to higher quantities of fine sediments than tidal marshes at the delta front and may be more capable of vertical accretion.

Over the period of 2003 to 2006, a multi-disciplinary assessment of effects of sea-level rise on Roberts Bank was completed in conjunction with IPCC Fourth Assessment findings (Kirwan and Murray, 2008, Hill et al., 2013). The project incorporated pre-existing Light Detecting and Ranging (LiDAR) data and aerial photography to examine geomorphological characteristics of the tidal flats, field data for several parameters (i.e., grain size, bed elevation, sediment shear strength, eelgrass density and composition, meiofauna density, and chlorophyll-a) and limited data on wave height and direction, current speed, and suspended sediment concentrations. A three-dimensional model was developed to adjust accretion rates with water depth and productivity under various sea-level rise scenarios (Kirwan and Murray, 2008). Results predicted marsh platform deepening and vegetation zone changes are likely to occur mainly in the latter half of the century (between 2050 and 2100). High marsh vegetation was reduced due to limited ability to migrate landward (coastal squeeze), transition zone vegetation migrated into high marsh areas, and much of low marsh vegetation was unable to keep up with sea-level rise (Figure 10). Kirwan and Murray (2008) predicted that while these tidal marshes will be able to mitigate effects of sea-level rise up to a threshold rate, Roberts Bank marsh erosion will occur due to coastal squeeze and increased wave energy with “low to moderate confidence that drastic changes will not occur before 2050”.

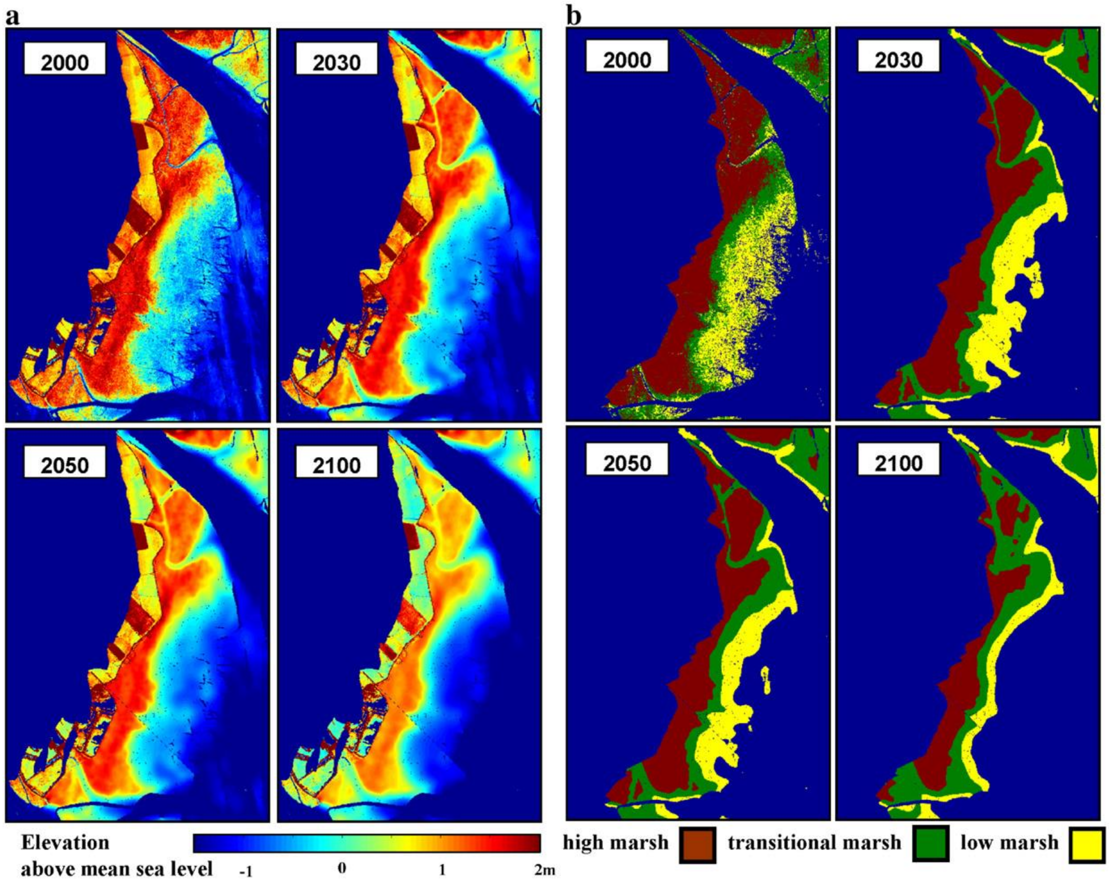

Figure 10: Westham Island bed elevations and ecosystem zones in response to a high sea-level rise scenario. From Kirwan and Murray, 2007.

3.5.2. Other Climate Related Stressors

In addition to direct threats on Fraser delta tidal marshes due to sea-level rise and related anthropogenic stressors (e.g., sediment supply disruption and coastal squeeze), climate change may further influence tidal marsh resilience. Climate change may affect tidal marsh communities through other drivers such as changes in precipitation leading to altered river flow volume, salinity, freshet timing and sediment discharge patterns (Cahoon 2009). In addition, sea-level rise can potentially increase tidal range, tidal velocities, or tidal asymmetry, consequently altering water circulation, salinity regimes and mixing patterns (Cahoon, 2009). Changes in freshwater inputs, sedimentation and erosion could lead to altered marsh structure and function (Sklar and Browder, 1998). Potential increases in magnitude and frequency of storms due to increased ocean surface temperatures could result in greater rates of erosion, resuspension, and sedimentation of nearshore sediments (Cahoon, 2009).

Climate change may also affect plant organic matter accumulation. Increased carbon dioxide concentrations in the atmosphere, increased temperatures, altered precipitation patterns, nutrient loading and other factors may reduce marsh root and stem growth, which may limit the ability of the marsh community to accrete sediments and lead to a decrease in tidal marsh resilience to sea-level rise (Cahoon, 2009). Changing temperature and sea levels affect *Spartina alterniflora* growth allocation (seed production vs. belowground rhizome production), which may affect accretion rates and resilience to sea-level rise depending on how the marsh gains elevation (either through mineral sediments or peat-based processes; Crosby et al. 2016). Increased temperatures may also lead to changes in species distributions, and thus restoration projects that require revegetation and planting may need to consider alternate species or genotypes that are better suited to new environmental conditions (Adam, 2018). How these climate change related factors may affect tidal marshes both on a global and regional scale is a question that remains to be answered.

3.5.3. Marsh Recession

Within the Fraser River estuary, over 70% of tidal marshes have been either isolated or have disappeared and between 1989 and 2011, about one third of the tidal marshes at Sturgeon Bank receded (Balke, 2017). Since 2015, the Sturgeon Bank Marsh Recession Project (SBMRP) has investigated potential causes of marsh recession. No individual hypothesis has explained the marsh recession to date; however, sea-level rise may be partially responsible and is likely to increase tidal marsh vulnerability in the future (Balke, 2017).

Chapter 4.

Tidal Marsh Resilience Strategies

Resilience is defined as a measure of the "ability of a system to absorb disturbance" and "to persist and adapt over time and under changing conditions" (Holling 1973, Gunderson 2000). It is a critical component of tidal marsh ecosystems as it represents the ability of the community to maintain structure and function under changing conditions and may prevent transition to alternate states (Standish et al. 2014). Under changing conditions, feedback mechanisms within tidal marshes can lead to increased stability (negative feedbacks) or lead to faster rates of change (positive feedbacks) which can ultimately create shifts to alternate states (Scheffer 2009). Equilibrium dynamics as defined by Connell and Sousa (1983) and DeAngelis and Waterhouse (1987) refer to disturbances shifting ecosystems away from a stable state (equilibrium) while recovery brings the system back towards the equilibrium. Both human activities and changing climate can affect tidal marsh recovery under stress and can potentially lead to irreversible transitions (e.g., low marsh transitioning to mudflat). Multiple stressors may act together to increase the risk of these transitions and are associated with reduced ecological resilience.

On a global scale, coastal wetlands have been reduced by half since pre-industry as a result of impacts from both climatic and non-climatic drivers (including urbanization, drainage, sediment supply alteration, and flooding; Kirwan et al., 2016). There are many coastal regions around the world where effects of sea-level rise on tidal marshes have already been observed and in some instances, management efforts have been ongoing for several decades. These regions provide opportunities to examine historic and current tidal marsh management strategies, their relative success, and how these strategies could potentially be applied to other areas, including the Fraser River delta.

Review Framework

This literature review involved an examination of current practices. The focus of the review was to examine ecosystem based tidal marsh resilience strategies and other soft engineering strategies with a primary goal of identifying key strategies for possible application in the Fraser River delta.

Academic databases and search engines were used to identify relevant material from a wide variety of journals. Research included both peer-reviewed and grey literature (e.g., government publications). The review was developed as a funnel down approach to first broadly identify where tidal marsh communities occur and where they are the most vulnerable to ongoing sea-level rise (with a primary focus on areas adjacent to high density coastal communities). This strategy assisted in identifying potential locations where existing management strategies occur. The review was exploratory with key sources and existing meta-analyses often leading to related articles. Available tidal marsh management literature was limited for many countries, and thus I focused on areas with regional and local management plans. The research was thematically organized based on individual management strategies. Existing information was summarized regarding how management strategies are being used, how effective they are, and what knowledge gaps exists (where possible).

4.1. Tidal Marsh Vulnerability & Management Strategies

Despite occurring globally, tidal marshes are limited in total surface area. Non-arctic tidal marshes are estimated to cover an area of 45,000 km^2 and occur in almost 100 countries (Greenberg et al., 2009, McCowan et al., 2017).

Tidal marshes occur along protected coastlines behind barrier islands and in estuaries and deltas where sediment supplies can support plant growth. The largest concentration of tidal marshes occurs in temperate estuaries along the South Atlantic and Gulf Coasts of North America, and China. Other areas where large tidal marsh communities occur include coastal estuaries in Argentina, Uruguay, San Francisco Bay on the Pacific Coast of the United States, and European coasts along the North Sea and the Baltic Sea. In subtropical regions, mangrove swamps typically dominate (Figure 11, Table 2, Greenberg et al., 2009).

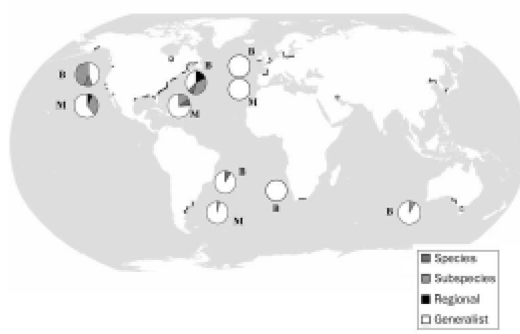

Figure 11: Global distribution of tidal marshes. From Chapman, 1977.

Table 2: Estimated surface area and dominant vegetation for tidal marshes. From Greenberg et al., 2009.

Continent	Coastline	Extension (square kilometers)	Dominant vegetation
North America	North Atlantic	500	*Spartina, Phragmites*
	South Atlantic, Gulf Coast	15,000	*Spartina, Juncus*
	Pacific	440	*Salicornia, Spartina*
Eastern South America	Atlantic	2300	*Spartina, Juncus*
Europe	All	1400	*Salicornia, Spartina,* grasses
Asia (Japan, Korea, China)	Pacific	25,000	Chenopods, *Phragmites*
Australia, New Zealand, Tasmania[a]	Southern (temperate)	772	*Sarcocornia,* other chenopod shrubs
South Africa	Southern	70	*Sarcocornia, Spartina,* grasses

As described in Chapter 3, climate change induced sea-level rise varies regionally. Regional and local influences affect relative sea level, including effects from subsidence, human activities (e.g., groundwater or oil extraction), and tectonic activity (Adam, 2018). Tidal marsh eco-morphodynamic feedbacks, especially the ability for tidal marshes to trap and accrete sediments, allow for some resilience to sea-level rise; however, human related stressors (e.g., fragmentation and landward migration restrictions) can limit tidal marsh ability to adapt (Oppenheimer et al., 2019). In areas where ecosystem processes are maintained, these feedbacks have allowed tidal marshes and other coastal ecosystems (e.g., mangrove systems) to build at the same rates or greater than rates of sea-level rise (Kirwan et al, 2016); however, threshold rates vary from site to site. Process-based models predict that some tidal marshes will be able to match relatively high rates of sea-level rise (1 cm to 5 cm per year) before drowning. Other marsh communities (e.g., with minimal tidal activity, reduced sediment supply, limited landward migration potential or increased anthropogenic subsidence) may drown at relatively low rates of sea-level rise (less than 1 cm per year; Kirwan et al., 2016). Expected effects due to sea-level rise over the next century include tidal marsh contraction, reduced functionality and biodiversity, as well as lateral and inland migration of marsh communities. Where barriers to migration occur, the other effects will be larger (Oppenheimer et al., 2019).

Relatively slow rates of sea-level rise at the beginning of this century provide opportunities for management actions that can be implemented to increase resilience of tidal marsh communities (Thorne et al., 2018). While many coastal cities continue to use hard protection measures (e.g., dikes, sea walls) as a primary means of flood protection, a growing number of coastal communities are implementing ecosystem-based adaptation and looking for innovative restoration opportunities. In the last few decades, there has been increased recognition of the value in protecting coastal ecosystems for the various ecosystem services they offer. Ecosystem-based adaptation strategies attempt to conserve and restore coastal ecosystems as a measure to protection the coastline through reduction of wave energy and storm surge, and by stabilizing coastal sediments and decreasing rates of erosion (Oppenheimer et al., 2019).

North America: Atlantic and Gulf Coasts

On the south Atlantic Coast and the Gulf Coast of North America rates of sea-level rise have been higher than the global average and are contributing to substantial ongoing wetland loss. The Mississippi River Delta wetlands have been reduced by 25% over the last several centuries due to reduced sediment load caused by dam construction in the Mississippi Basin which restricts sediment supply and freshwater inflow (Ray, 2007; Elsey-Quirk et al., 2019). Blum and Roberts (2009) estimate that at least 10,000 km^2 of the flood plain will be submerged by 2100 as a result of ongoing subsidence and sea-level rise in the absence of increased sediment inputs. Thorne et al., (2018) note that resilience in the Mississippi Delta may be high compared to other areas due to opportunities for marsh migration inland.

Thin layer sediment augmentation measures (described further in Section 4.2.3 and Section 4.3) have been a primary means of marsh restoration, marsh creation and beach nourishment in this region. Living shorelines (described further in Section 4.2.1) are also being used as an alternative to hard shoreline protection structures in many areas (e.g., Galveston and Mississippi watersheds; EPA, 2015). Recently, the Coastal Protection and Restoration Authority (CPRA) of Louisiana has updated a Coastal Master Plan which includes various restoration projects including managed realignment and sediment diversion projects for implementation over the next 50 years with an estimated cost of over $50 billion over this period (CPRA, 2017). The plan includes $18 billion for marsh creation projects using dredged materials and $5 billion for sediment diversion projects (Elsey-Quirk et al., 2019).

In North Atlantic United States, thin layer sediment augmentation methods have been used to assist in restoration of salt marshes at several locations. Some examples of locations where this technique has been used include Chesapeake Bay Blackwater National Wildlife Refuge, Maryland (in 2002), Prime Hook, Delaware (2014-2016)

Avalon, New Jersey (2014-2016), John H. Chafee National Wildlife Refuge, Rhode Island (2016-2017), and Pepper Creek, Delaware (in 2013).

In Atlantic Canada, sea-level rise has been higher than the global mean due to effects of regional subsidence of the Earth's crust (glacial isostatic adjustment). In Nova Scotia, for example, subsidence has been estimated at approximately 20 cm/century with some areas (e.g., Halifax, Yarmouth, and North Sydney) experiencing over 30 cm/century (Arlington Group, 2013).

As part of the "Making Room for Wetlands" project at the Missaguash River salt marsh in Fort Lawrence, Nova Scotia, pilot project sites are being established for managed realignment. Dikes are being breached and moved inland, transforming former farmland, and allowing the marsh to expand into areas where they once existed along the coast. The costs of maintaining dikes in these high-risk areas was determined to be prohibitive, especially with projected erosion concerns in the face of sea-level rise. By realigning the dikes and allowing the marshes to expand as a living shoreline, they can provide flood defence ecosystem services through wave attenuation and erosion control (CBC, 2020).

North America: Pacific

A recent study by Thorne et al. (2018) modelled coastal wetland resilience to sea-level rise across 14 estuaries on the Pacific coast. These estuaries are relatively small compared to other regions in the world and typically occupy narrow riverine valleys. Thorne et al. (2018) concluded that these tidal marsh communities are vulnerable to projected twenty-first century sea-level rise, primarily due to limited migration potential caused by urban encroachment or steep terrain. Areas with low accretion potential, including Morro Bay (Central California) and Bandon marsh (Oregon Coast) have limited resilience to projected sea-level rise without direct intervention (Thorne et al., 2018).

At the Seal Beach National Wildlife Refuge in California, a thin-layer sediment augmentation pilot project was conducted for the first time on the Pacific Coast. Project scope included pre-construction monitoring followed by five years of annual monitoring (USFWS, 2018) and sediment was applied to the 4-ha project in April 2016. The goal of the project was to test benefits of the technique as a sea-level rise adaptation strategy and as a conservation strategy for listed and sensitive species dependent on the Pacific cordgrass (*Spartina foliosa*) ecosystem.

San Francisco Bay is the second largest estuary in the United States, the largest on the Pacific Coast and contains approximately 90% of California's wetlands (Hine, 2015). Accelerated rates of sea-level rise over the next century (projections of up to 165 cm by 2110 under high scenarios) in combination with reduced sediment supplies are expected to affect coastal ecosystems in the Baylands as well as over $62 billion in development (Goals Project, 2015, Hine, 2015).

Currently, tidal marshes in San Francisco Bay are accreting sediment to keep pace with sea-level rise; however, recent marsh accretion modelling projections indicate that there will be an increase of mid-marsh between 2010 and 2030 with a reduction of high-marsh and upland ecosystems throughout the estuary. Between 2030 and 2050, projections show an increase in low marsh and decrease in high marsh. Beyond that period, model projections vary drastically depending on sea-level rise scenarios with worst case (high sea-level rise/ low sediment) including conversion of over 90 percent of mid marsh and high marsh to low marsh, mudflat and subtidal zones (Goals Project, 2015). Under the U.S. Climate Resilience Toolkit, the Future San Francisco Bay Tidal Marshes Tool provides a useful way to visualize projected changes to tidal marshes under various scenarios (Veloz et al., 2020).

The Goals Project (Goals Project, 2015) has examined several strategies for increasing San Francisco Bay tidal marsh resilience in the face of sea-level rise including protecting upland tidal marsh migration space in less developed areas and management of transition zones, realigning levees in combination with marsh restoration, erosion control features, increasing supply of fine sediment (through direct placement, thin layer sediment applications and/or changes to watershed management), and increasing sediment trapping efficiency (e.g., through planting). Choosing which strategies to implement and where will require extensive analysis of several factors including trade-offs between competing uses, near and long-term benefits, and which ecosystem services to protect. With sea-levels projected to rise more rapidly by mid-century, the Goals Project recommends restoring diked tidal marshes before 2030 to develop natural processes within the marshes and allow them the accrete vertically more effectively (Goals Project, 2015). Currently, more than 8,000 ha of tidal wetlands in the

San Francisco Bay area have been reintroduced to tidal processes as conservation groups have obtained land and breeched historic levees. The goal of these projects is to restore natural processes to tidal marsh ecosystems and increase coastal resilience to impacts of climate change and sea-level rise (U.S. Climate Resilience Toolkit).

Western Europe

The Online Managed Realignment Guide (OMReG) provides a database, interactive maps, and project details for coastal adaptation projects implemented primarily in Western Europe and the United Kingdom. Projects include various managed realignment strategies, beneficial use of dredge material (thin layer sediment augmentation techniques), regulated tidal exchange, and other approaches.

In the twenty-first Century, managed realignment has become more common as a coastal adaptation strategy in the United Kingdom (e.g., Blackwater and Humber estuaries), Belgium and the Netherlands (e.g., Scheldt estuary) with primary goals including both coastline stability and mitigation for intertidal marshes. According to the *National Adaptation Programme strategic plan for coastal realignment*, the UK plans to realign 550 km of coastline by 2030 (Esteves, 2014).

While not used in tidal marsh ecosystems, beach nourishment has been implemented often in the Netherlands since 1990 as a coastal defence strategy. On the south coast of the Netherlands a pilot project was implemented in 2011 to test the effectiveness of mega-nourishment. This method, called the Sand Motor or Sand Engine technique required an initial deposit of 20 million cubic meters of sand that is expected to, by natural processes, distribute across the beach and dunes over a 20-year period (Stive et al., 2013). Initial post construction assessments indicate that the sand is spreading along the coast and is still considered a potentially viable option for coastal management (Taal et al., 2016).

China

Seawalls have been constructed along 60% of mainland China’s coastline and rapid urbanization has led to a loss of 50% of coastal wetlands over the period of 1950 to 2000. In 2000, a National Wetland Conservation Action Plan was developed, and in

2004, a National Wetland Protection Plan was approved; however, implementation of marsh management strategies have been difficult due to fragmented control by multiple agencies, and inadequate regulations to protect wetlands (Ma et al., 2014).

A focal point for tidal marsh response to sea-level rise has been the Yangtze Delta, Shanghai, China. Tian et al. (2010) modelled potential sea-level rise related effects on tidal marshes and predict that with a 0.88 m increase in sea level, approximately 40% of the coastline within the Chongming Dongtan Nature Reserve will be submerged by the year 2100. Medium and long-term projections indicate substantial reductions in the marsh community due to ongoing subsidence, decreased sediment supply from the Yangtze River (due to construction of dams) and seawalls preventing inland migration (Ge et al., 2016). Wang et al. (2014) recommended several mitigation measures to increase tidal marsh resilience in the nature reserve. Mitigation options include sediment management changes and erosion reduction (via high profile groynes), dredged sediment augmentation strategies, and policies to reduce anthropogenic subsidence (caused by groundwater pumping and large infrastructure development).

4.2. Ecosystem Based Adaptive Management Options

Based on existing literature and large-scale flood management programs, the prominent strategies for increasing tidal marsh resilience to sea-level rise have been identified and are discussed below.

4.2.1. Living Shorelines

Living shorelines are protected coastal areas designed with natural materials. The method is a form of soft infrastructure that can increase resilience of coastal ecosystems while also providing a cost-effective, low maintenance flood management technique. Shoreline restoration that involves maintaining or expanding tidal marsh areas is likely to improve the resilience of tidal marshes to the effects of sea-level rise (Ganju, 2019). Long-term tidal marsh resilience to sea-level rise will be site dependent and marshes may require opportunity for landward migration (Miller et al., 2016, Mitchell and Bilkovic, 2019). Other methods can also be used in conjunction with living shorelines to increase tidal marsh resilience to sea-level rise. The methods discussed below incorporate the concept of living shorelines into their overall design.

4.2.2. Managed Retreat & Realignment

Managed retreat is a proactive method of coastal management designed to move current and planned coastal developments away from short and long-term coastal risks (e.g., erosion, tides, and flooding storm surge). Management realignment is a form of managed retreat where hard coastal protection structures are selectively removed to allow coastal ecosystems to re-establish, often in reclaimed areas (e.g., agricultural land; Neal et al., 2017).

Managed retreat methods have two main objectives with order of priority dependent on specific site objectives. These objectives are to defend the coastline against effects of sea-level rise and to increase or maintain tidal marsh ecosystem function (French, 2006). Managed retreat attempts to mimic what would happen for natural systems under increases in sea-level with tidal marsh communities migrating landward over time. Depending on the elevation profile, managed realignment may cause immediate flooding of areas behind foreshore dikes. As such, for managed realignment to be most successful detailed site investigations are required to determine if sites are appropriate for tidal marsh expansion (French, 2006). An example of this method is provided in Figure 12, below.

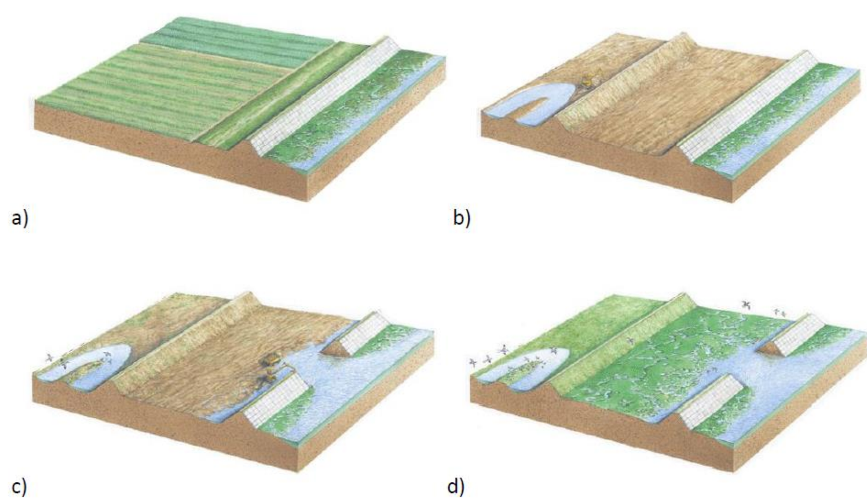

Figure 12: Example of the stages of managed realignment: a) Existing hard defenses; b) New flood bank built and contoured; c) Old defenses breached; d) intertidal areas develop and absorbed wave energy. From Transcoastal Adaptation Resources.

Implementing managed retreat can be a complex issue as it involves a variety of social and political challenges. Managed retreat involving removal of hard defences is more likely to be implemented in locations where erosion and flooding risks are relatively acceptable (i.e., more rural or low-density areas). Under threats from long-term sea-level rise, this method, if successful, is likely cost-effective for potentially maintaining tidal marsh resilience and coastal defences over the long-term. In areas, such as the Fraser River delta, where opportunities for landward migration are limited, the costs associated with ongoing sediment placement can be justified, especially where dredge material is continually sourced from the area (Ganju, 2019).

In locations with high levels of coastal development, where unused land is scarce (such as in Belgium and the Netherlands), managed realignment for tidal marsh creation has, in some cases, been implemented through controlled tidal restoration projects. These strategies maintain flood defence lines while allowing tidal exchange behind seawalls through tidal gates, spillways or sluices (Esteves, 2014).

4.2.3. Thin Layer Sediment Augmentation

Thin layer sediment placement of dredged river materials is a strategy designed to potentially slow wetland loss by reducing subsidence rates and increasing sediment accretion. The strategy involves spraying sediment slurries over the marsh surface (Figure 13). In the United States, millions of dollars have been spent on sediment nourishment techniques over the last few decades (Ganju, 2019). In 2007, the US Army Corps of Engineers released a technical review of thin layer placement of dredged material on coastal wetlands (Ray, 2007). The majority of examples were from locations within Louisiana with assessment of wetland response as far back as 1987. The method of thin layer sediment placement was developed in Louisiana (modified from existing hydraulic dredging methods) and has been increasingly used in various locations along the Gulf and Atlantic Coasts (Ray, 2007).

A more recent document (2019) provided by the U.S. Army Corps of Engineers (USACE) discusses the technical definition of thin layer placement activities and provides additional application examples from the north Atlantic region and the Pacific coast of the U.S. In their review, USACE synthesizes several definitions of thin layer placement from the literature into one clarification:

> "Purposeful placement of thin layers of sediment (e.g., dredged material) in an environmentally acceptable manner to achieve a target elevation or thickness. Thin layer placement projects may include efforts to support infrastructure and/or create, maintain, enhance, or restore ecological function" (USACE, 2019).

General lessons learned from recent USACE experience with thin-layer placement projects in the North Atlantic region include the following:

1. Thin layer placement (TLP) techniques are ideal for sites where substantial marsh elevation has been lost (either from subsidence or sea-level rise) and where sediment inputs are likely to be insufficient to nourish the marsh over time;

2. Adequate characterization of dredged material and exposure site are critical for success (e.g., bathymetry, topography, water levels, tides, sediment grain size, texture, and contamination);

3. Key features of the marsh should be protected where possible (e.g., tide channels); and,

4. Vegetation typically respond well to applications in the 15 cm to 30 cm range and recolonization without planting is possible in this range. Recovery periods typically range from 3 – 5 years.

Figure 13: Thin layer placement of sediment. Photo credit: Tim Welp; USACE, 2019.

4.2.4. Living Dikes

The living dike concept is an innovative ecosystem-based adaptation strategy designed to maintain current flood protection boundaries and increase resilience of tidal marshes to the effects of sea-level rise. A similar concept, described as a horizontal levee has been included in the Baylands Ecosystem Habitat Goals 2015 Update (Goals Project, 2015).

This concept combines the use of hard protection structures (e.g., dikes) with thin layer sediment augmentation strategies. Thin sediment lifts are applied across the tidal marsh and over top of existing hard structures to mimic a more natural shoreline and provide opportunity for the tidal marsh community to migrate inland and onto the dike over time (Figure 14). Sediment lifts necessary to form a living dike would require a steepening of the intertidal area, however this change in slope is considered to be relatively small (SNC-Lavalin Inc., 2018).

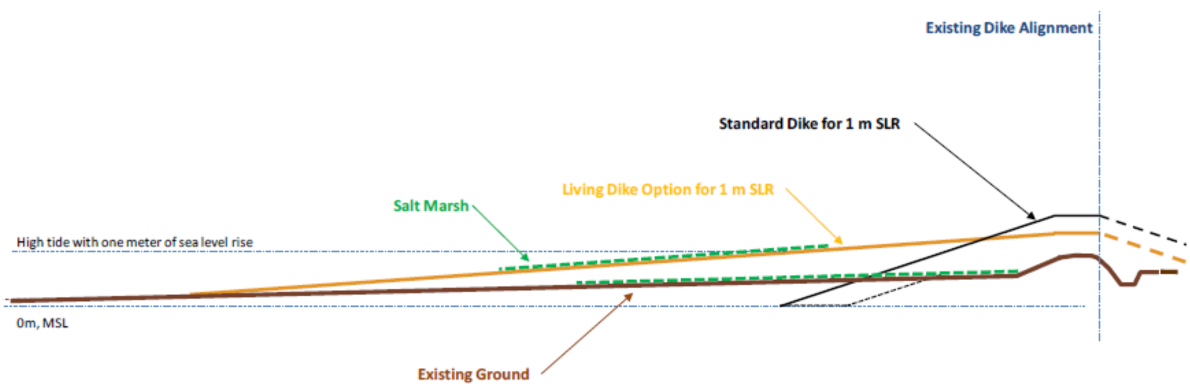

Figure 14: Schematic illustrating the living dike concept and standard sea dike (SNC-Lavalin Inc., 2018)

Three potential sediment delivery methods were included in the Design Basis study (prepared by SNC-Lavalin) for implementation in Boundary Bay, including: 1) delivery of sediment to shallow sub-tidal waters with the intention that sediments would be transported to the site via wave and current processes, 2) delivery of sediment to the lower intertidal zone for transport to the site via wave and current processes, and 3) delivery of sediment directly to site through standard high pressure spray techniques. (SNC-Lavalin Inc., 2018).

4.2.5. Sediment Pipeline

As part of the Louisiana Coastal Protection and Restoration Authority Coastal Master Plan goals for reducing land loss, construction of a long-distance sediment pipeline (LDSP) began in 2013 (CPRA, 2017). The LDSP has transferred approximately 7.6 million cubic metres of Mississippi River sediments over a distance of up to 16 km to provide additional sediments to several existing and new marsh restoration projects as well as for beach nourishment projects. The pipeline is designed to remain in place as a permanent corridor for future projects (Figure 15, CPRA, 2017).

Figure 15: Dredge boat and sediment pipeline; Louisiana CPRA, 2017.

4.2.6. Sediment Diversion

Within the Mississippi Delta, natural overbank flooding is limited due, in part, to construction of canals, oil and gas extraction, logging, and spoil banks. These human modifications reduce sediment supply, alter salinity and nutrient loading regimes and, in conjunction with ongoing subsidence and sea-level rise, have resulted in large losses of Mississippi River Delta Wetlands and additional losses are expected over the next 100 years (Elsey-Quirk et al., 2019). As described above, the Louisiana Coastal Protection and Restoration Authority (CPRA) has developed plans for extensive restoration projects, including several sediment diversion projects. Diversion works will involve engineered outlets through existing levees to periodically or continuously deliver freshwater, nutrients and sediment, create new wetlands and prevent ongoing degradation to existing wetlands (CRPA, 2017).

4.2.7. Mud Motor

The mud motor concept is a strategy for increasing tidal marsh resilience by increasing sediment supply to tidal marshes. The strategy involves depositing dredged sediment in key locations (e.g., tidal channels) to naturally disperse into nearby marshes through existing currents with the goal of accelerating vertical (and potentially lateral) marsh growth. This strategy aims to increase sedimentation while maintaining desired gradients and limiting direct disturbance to the marsh. The method is different than a sand engine in that sediment is meant to be supplied at regular intervals instead of a single large deposit.

A pilot project using the mud motor concept was conducted near the Port of Harlingen, in the Wadden Sea over a three-year period from September 2015 to August 2018 (Baptist et al., 2019). During the pilot, a total of 470,516 m^3 of dredged material was disposed of at stations selected for proximity to the tidal marshes and dredge vessel access. Dredge operations were completed daily over two winter seasons through bottom door disposal, depending on available tide windows and weather conditions.

Results suggest that transported sediments were largely affected by wind, wave and freshwater circulation patterns and that there were more hydrodynamic stresses than expected. Marsh elevations showed net accretions averaging 4.9 cm with both high spatial variability in accretion rates and high short-term fluctuations in thickness of sedimentation, suggesting that the method may have been more successful at a location with physical settings that allow for containment of the sediment load.

Lessons learned from the pilot project study indicated that for the mud motor to be used effectively, several factors should be considered including a thorough understanding of what factors are limiting marsh growth (e.g., energy exposure, sediment starvation, limited seed supply). Prior to conducting this type of project, sediment transport modelling and tracer studies would be beneficial to understand sediment transport rates and direction.

4.3. Sediment Augmentation: Potential Effects & Considerations

Sediment augmentation measures are primarily designed to increase the amount of mineral sediments provided to sediment starved, eroding, and subsiding tidal marshes in order to increase accretion rates and allow the marshes to maintain or increase elevation with the ultimate goal of increasing resilience to stresses from both sea-level rise and subsidence.

There are various examples of the positive benefits of sediment additions in deteriorating *Spartina alterniflora* salt marshes on the South Atlantic coast of the US including increased elevation and reduced flooding, increased soil aeration and bulk density, and increased nutrient concentrations (Nyman et al., 1990; Elsey-Quirk et al., 2019). Increased surface elevation reduces hydroperiod. Marshes that are experiencing overextended hydroperiods may become waterlogged and result in root oxygen deficiencies, soil phytotoxin accumulations (e.g., hydrogen sulfide) and increases in salinity beyond tolerance (leading to osmotic stresses, toxic ion effects and nutrient uptake inhibition), all of which can reduce plant productivity (Mendelssohn and Kuhn, 2003; Mendelssohn and Batzer, 2006).

Layer thickness, physical properties of dredge sediment, hydrologic forces on unconsolidated materials, and other considerations can also affect project success. This section provides a qualitative analysis of physical and biological effects of sediment augmentation techniques. Observations from thin layer sediment augmentation restoration projects were reviewed and are described below. Summaries of results for several individual studies are provided in Appendix 1. The majority of literature relating to tidal marsh response to sediment augmentation measures is from the South Atlantic Coast and typically examines *Spartina alterniflora* dominant salt marshes; however, sediment augmentation projects are occurring with increased frequency on the Pacific Coast.

4.3.1. Application Thickness

Appropriate thickness of sediment lifts will likely be dependent on elevation goals, material being deposited and compression of sediments. However, target elevations should be based on marsh zone capacity and elevation deficits, and some understanding of how layer depth will affect tidal marsh response. After placement, unconsolidated materials will likely be affected by physical factors and understanding how materials may move over time will increase the chances of meeting project goals.

Application thickness is likely one of the most important factors in how marsh vegetation will respond to this type of intervention (Cahoon and Cowan, 1988; Tong et al., 2013).

In general, application of too much sediment may result in smothering of existing plants and/or changes in marsh inundation zonation, potentially leading to alternate ecosystem states (e.g., high marsh or terrestrial migration into low and middle marsh zones). Application of too little sediment may not provide enough benefit to counter stresses from sea-level rise. Applying sediment in layers that can increase plant productivity and increase marsh elevation over the long term will have the highest potential for increasing marsh resilience (Figure 16; Elsey-Quirk et al., 2019).

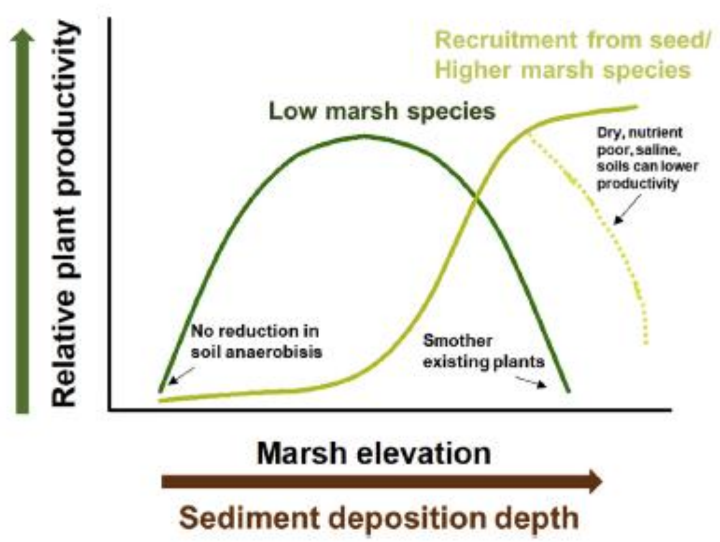

Figure 16: Tidal marsh productivity in response to sediment deposition depth. From Elsey-Quirk et al., 2019.

Cahoon and Cowan (1988) qualitatively observed post augmentation effects of high-pressure spray disposal techniques at two marsh locations in Louisiana. Vegetation was initially smothered with sediment application of 10 cm to 15 cm at one location and 18 cm to 38 cm at another. However, both sites were able to recolonize with native vegetation after 14 months. Ford et al. (1999) found that marsh plants that were initially smothered by sediment application were able to recover within one year with a 2.3 cm application of dredged material. Within a year of application, *S. alterniflora* percent cover had increased from pre-application conditions. Other examples have found comparable results, including increases in total *S. alterniflora* biomass in degrading marshes with sediment additions of >15 cm (Mendelssohn and Kuhn, 2003), 5-12 cm (Slocum et al., 2005), 14-20 cm (Schrift et al., 2008) and others (Appendix 1, Table 3, below). For these types of marshes, there may be a sediment depth threshold at approximately 30 cm (Elsey-Quirk et al., 2019). Sediment that smothers existing vegetation may still be able to colonize through seed establishment; however, if sediment additions are above the optimal range, changes in vegetation can occur (Leonard et al., 2002).

Several small scale mesocosm studies have examined the effects of adding sediment to *Spartina alterniflora* marshes with varying levels of application thickness (Reimold et al., 1978, DeLaune et al., 1990, Pezeshki et al., 1992). Reimold et al. (1978) observed that *S. alterniflora* was able to grow through up to 23 cm of sediment regardless of grainsize while 30 cm of sediment reduced plant biomass and the marsh did not recover when sediment layers of 60 cm or more were applied. Reimold noted that shoots were better able to emerge from sandier material however, overall biomass was greater in silty substrates, potentially due to increased nutrient retention in silty material (Reimold et al., 1978). Tong et al. (2013) analyzed the effects of thin layer sediment application at a *S. alterniflora* marsh in Louisiana and found that greatest recovery rates of vegetation, soil and benthic fauna occurred in an elevation range of 8-15 cm above ambient healthy marsh. Tong et al. (2013) also observed that above and below ground biomass did not recover at the same rates and seven years following sediment application (in areas of up to 15 cm application thickness) above ground biomass had recovered significantly faster than below ground biomass. Areas with greater application thickness had impaired rates of recovery (Tong et al., 2013).

Observations from these studies suggest that when vegetation is not completely smothered (under smaller lifts), the plant communities tend to respond faster, while sediment applications greater than 15 cm tend to recolonize via new plants instead of growing through the recent sediment additions (McAtee, 2018). Larger applications may also create sediment conditions that become increasingly dry, nutrient poor and more saline (Slocum et al., 2005; Elsey-Quirk et al., 2019).

Recent research by Hine (2015) on the potential for use of thin layer sediment application to increase tidal marsh resiliency at a marsh in San Francisco Bay included a review of thin layer sediment augmentation studies and limitations, in conjunction with marsh accretion modelling (Marsh Equilibrium Model) to determine how much sediment should be applied and within what timeframe. Hine estimated that under a 100 cm/century sea-level rise scenario, thin layer sediment application would be required by 2060 with an application thickness of 16 cm. Based on this sea-level rise scenario, a dredge sediment application of this thickness is estimated to increase tidal marsh resiliency for at least 40 years before another 16 cm application would be necessary. If sediment applications were delayed, thicker lifts would be necessary to attempt to maintain ecosystem function. As identified above and within Hine's assessment, thicker applications may reduce marsh function for longer periods of time. Under a higher sea-level rise scenario, initial application would be required around the year 2045 (Hine, 2015).

Table 3: Examples of thin layer sediment methods and application depths (Ray, 2007; USACE, 2019)

Location	Application Method	Depth (cm)	Citation
Barataria Basin, LA	Manual spreading	2-5	DeLaune et al., 1990
Bayou Lafourche, LA	Low pressure discharge	13-36	Schrift et al., 2008
Blackwater NWR, MD	High pressure discharge	ND	Nemerson 2007
Coos Bay, OR	Mechanical spreading	ND	Cornu and Sadros 2002
Delaware Day, NJ	High pressure discharge	ND	Weinstein and Weishar 2002
Dog Lake, LA	Various	10-15 and 18-38	Cahoon and Cowan, 1988
Glynn County, GA	Manual (Small enclosures)	8, 15, 23, 30, 61 and 91	Remold et al., 1978
Gulf Rock, NC	Various	10	Wilber 1992
Leeville, LA	High pressure discharge	ND	Streever, 2000
Masonboro Island, NC	Mud motor	0-10	Leonard et al., 2002
Narrow River, RI	Mechanical spreading	10-15	USFWS 2014
Pepper Creek, DE	High pressure discharge	0-20	Wilson 2013
Sachuest Point, RI	Mechanical spreading	2.5-30	Center for Ecosystem Restoration 2015
Seal Beach, CA	High pressure discharge	25	USFWS 2018
Venice, LA	High pressure discharge	2.3	Ford et al., 1999
Venice, LA	High pressure discharge	<15, 15-30, >30	Mendelssohn and Kuhn, 2003
Venice, LA	High pressure discharge	<15, 15-30, >30	Slocum et al., 2005

Vermillion Parish, LA	Low pressure discharge	0-20	Graham and Mendelssohn, 2013

4.3.2. Sediment Transport

Ganju (2019) places critical importance on understanding sediment transport mechanisms and developing models to understand three-dimensional sediment dynamics in the early phases of sediment nourishment planning. Ganju argues that focusing on only vertical or lateral aspects of sediment fluxes may lead to poor outcomes while a more complete understanding will help determine the viability and lifespan of these types of projects. As an example, sediment applications at Blackwater National Wildlife Refuge in Maryland, USA, should ultimately be offset by underlying export in less than six months (Ganju, 2019).

In general, natural river sediment discharge regimes provide gradients of sediment particle sizes to delta foreshores and fine sediments are typically dispersed and settle further than coarse sands. These fine particles ultimately settle in tidal marshes and are important in tidal marsh biogeochemical processes. As such, sediment augmentation strategies using dredge material should try to mimic natural tidal marsh particle size regimes. There are examples from the literature of sediment augmentation restoration projects that have had issues with vegetation recovery when using sediments that are too coarse. Coarse materials (i.e., sand) have higher porosity and lower water and nutrient retention capacity than finer materials (Langis et al., 1991; Gibson et al., 1994; Elsey-Quirk et al., 2019). If sediments are not able to retain water long enough for the vegetation to extract nutrients, then these nutrients will be removed from the system on an ebb tide. If water remains in the sediment for too long, the system can become anoxic and build up hydrogen sulfide (Mendelssohn and Kuhn, 2003).

Consolidation and compression of sediment slurries will alter final elevation levels after augmentation and should be factored into treatment design when developing desired target elevations. The amount of consolidation will be site specific and depend on the nature of the substrate characteristics, and hydrodynamic and sedimentation processes (Wilber, 1993). At the Seal Beach National Wildlife Refuge thin-layer sediment augmentation site in California, mean elevations immediately following sediment augmentation were measured (using deep rod surface elevation tables [SETs]) to have increased by 216 mm. Site elevations were then observed to have decreased approximately 82 mm post sediment application (between April 2016 and January 2018).

According to the Seal Beach 2018 annual monitoring report, several challenges were described regarding the sediment augmentation process. First, sediment was not able to cover the full area of the site and was less uniform than anticipated due to sediment filling in ponds, creeks and low-lying areas first instead of matching contours. Another issue was that the sediment delivered to the site contained a higher percentage of course particles than the analysis showed for the dredge sites where the sediment had been taken from. Post augmentation surveys also indicated a decline in area and density of eelgrass compared to the reference area; however, there is currently no direct evidence to determine that this is a direct effect of sediment augmentation (USFWS, 2018). Eelgrass communities may be affected by extended periods of suspended sediments resulting from sediment augmentation measures.

Spartina foliosa has been returning to the Seal Beach project site; however, not as quickly as initially anticipated. This is likely due to the thickness of the applied layer of sediment and the coarse nature of the dredge material applied to the site (85% to 95% sand). As discussed above, coarse substrates drain quicker, potentially increasing salinity levels and removing nutrients from the sediment, which may lead to reduced vegetation germination and growth rates (McAtee, 2018). Finer silty sediments will likely accumulate at the site over time (via bioturbation and river sedimentation processes) and are expected to increase nutrient levels and organic matter content and ultimately more favorable conditions for the marsh species. The research team is now considering planting strategies. While replanting and potential fertilization methods can speed up marsh recovery, the most effective solution may be to amend augmented substrates with finer sediments (McAtee, 2018).

4.3.3. Invasive Species

Disturbances, including flattening or smothering existing tidal marsh communities with dredge sediment augmentation methods, facilitates invasive plant establishment. Tidal marshes are particularly susceptible to invasion, in part, as they are considered landscape sinks which accumulate materials (Zedler and Kercher, 2004). On the Pacific Northwest Coast, *S. alterniflora* has become highly invasive and while no known locations occur in B.C., it is prevalent in Washington State. Other invasive cordgrasses have been observed in B.C. including *S. anglica* which has been observed in the Fraser River Delta (at Boundary Bay and Roberts Bank). These cordgrasses spread through rhizomes and viable seeds and can form large monotypic stands in intertidal and low marsh communities, which can decrease habitat for shorebirds, nursery grounds for juvenile fish, and disrupt drainage patterns (BCMOE, 2019).

4.3.4. Application Techniques

For dredge sediment application techniques, bucket dredging (mechanical spreading) and low-pressure sprays are less desirable than high pressure spray techniques. Low pressure options distribute sediments in uneven layers and have limited range; whereas, high pressure sprays can create more well mixed, uniform, layers (Ray, 2007; USACE, 2019). Behavior of dredged material will, however, vary with grain size, organic matter content and bulk density. Hydraulically dredged sediment slurries tend to separate during placement with fine-grained sediments spreading further distances, making it difficult to predict thin-layer application thicknesses (USACE, 2019). Spray techniques are limited by distribution range. Typical distribution range is restricted to less than 100 m from the spray equipment. Pipelines can increase the range of sediment distribution areas; however, it may be more difficult to control evenness of sediment distribution. Distribution of fine sediments can be increased with tidal currents; though, this may also result in fine sediments dispersing beyond target areas (Cahoon and Cowan, 1988).

The use of sediment pipelines to distribute sediment slurries over far distances should consider physical effects of pipelines on tidal marshes. Pipelines, if laid directly on the marsh or mudflat, may flatten vegetation, alter drainage patterns and promote ponding. Floating platforms may be used to limit any direct contact that the pipeline has with the marsh or mudflat. Another option is to extend the pipeline along the crest of existing dikes and route the pipeline into the exposure sites at various points.

When planning thin layer sediment methods, marsh platform slope should be considered as sediment applications can create ponding that may decrease tidal marsh productivity (Ray, 2007). Sediment applications may also infill existing tidal channels and erosion control methods may be necessary to assist in controlling placement of material.

According the Wilber (1993) it is relatively easy to control the direction of spray which may allow for reduced infilling of tidal channels and other sensitive areas.

4.3.5. Physical & Chemical Testing

The Canadian Environmental Protection Act,1999 requires that dredged materials undergo chemical and physical testing and that appropriate permits are obtained before disposal at sea can occur. Under the Vancouver Fraser Port Authority Dredge Material Management Program, beneficial use of dredged material is preferred and includes options for construction use, land reclamation and habitat creation. Any projects that use dredged material for beneficial use such as with thin layer sediment augmentation would likely require a thorough characterization of the chemical and physical properties of the sediment before application.

4.3.6. Timing of Works

Planning sediment augmentations to coincide with Fraser River freshet (late spring/ early summer) may provide the best way to simulate natural, historic sedimentation periods. Other factors also need to be considered; however. Thin layer sediment augmentation techniques are expected to have direct and indirect effects on a variety of organisms. Various regulatory bodies have restrictions when works can be undertaken to protect species and their habitats. Under measures to protect fish and fish habitat, Fisheries and Oceans Canada requires that some works in and around water be undertaken during least risk timing windows to reduce the chances of causing death of fish or harmful alteration, disruption or destruction of fish habitat. For the Fraser River Estuary, the work window is from July 16 to February 28. Project timing should also be considerate of bird nesting and feeding (e.g., biofilm disruption) periods and species at risk.

Chapter 5.

Fraser River Delta Pilot Project Opportunities & Recommendations

Under all greenhouse gas (GHG) emissions scenarios, sea-levels are projected to continue to rise over the next several centuries (Oppenhiemer, 2019). While it is difficult to predict how sea levels are going to change over the next 100 years and beyond, it is clear that sea-level rise and increased rates of sea-level rise have the potential to reduce tidal marsh structure and function at the Fraser River delta front. Since 1989, all of the major brackish marshes have receded with approximately 250 ha of tidal marsh lost (E. Balke, unpublished data). The delta front tidal marshes provide habitat for a diversity of invertebrate species, are considered critical for a variety of mammals, wintering and migratory bird species, and juvenile salmonids (Levings, 2016). Additional reduction in tidal marsh resilience and further losses may lead to substantial negative effects on many species.

Relatively high suspended sediment concentrations in Fraser River discharge, and a relatively high tidal range give the Fraser River and delta characteristics that can reduce marsh vulnerability to sea-level rise (Attard et al., 2014; Atkins et al., 2016). However, multiple stressors exist that may increase tidal marsh vulnerability. As with many deltas around the world, anthropogenic influences in the Fraser River delta may reduced both sediment supply to the delta front and upland migration potential. River training structures, causeway and jetty construction, navigation dredging, and flood protection diking have potentially led to reductions in the amount of sediments reaching the delta front marshes. Flood protection diking runs along most of the Fraser River delta front and limits the ability for tidal marshes to migrate landward as sea-levels rise. Aside from Mud Bay, managed realignment strategies are not currently favoured under coastal flood adaptation strategies (CFAS, 2019).

With the critical importance that Fraser River delta tidal marshes have, and the potential risks of loss associated with sea-level rise, strategies for attempting to increase tidal marsh resilience should be implemented. A status quo approach would increase delta front tidal marsh vulnerability over time and limit our ability to apply successful adaptive management strategies when required.

Based on this review of ecosystem based coastal adaptation strategies, long term options for increasing tidal marsh resilience to sea-level rise are limited to managed retreat and realignment and various sediment augmentation options. These options can be enhanced through ongoing restoration and living shorelines projects.

Managed retreat and realignment strategies can be effective for increasing tidal marsh resilience. If upland conditions are suitable for marsh expansion, this can allow for natural response of tidal marsh communities to migrate landward over time. Public perception and social impacts, land use rights, loss of agricultural lands and other implications must be taken into consideration under these scenarios and require extensive community consultation. In many parts of the Fraser River delta, land behind the dikes is lower in elevation than tidal marshes located outside. As such, managed realignment will also require sediment additions. Under hold-the-line scenarios, constructing and maintaining hard defence structures to combat sea-level rise over the next few centuries will be costly (Ausenco Sandwell, 2011). Managed retreat may be a more favourable option in the future; although, tidal marshes may be in a more vulnerable state by then.

Sediment augmentation strategies have been in use for several decades with primary means of application through direct placement of thin layers of sediment. Improving sediment pathways to the marsh surface can increase vertical accretion rates. Fine sediments can be added directly to the marsh surface (thin layer sediment application techniques), within the water column (e.g., mud motor; Baptist et al., 2019) or both on the marsh surface or on adjacent tidal flats and within the water column. Both of direct and indirect techniques are likely to have advantages and disadvantages associated with their use. There is substantial literature on thin layer sediment applications and effects on salt marshes (especially from the South Atlantic coast). Layer thickness appears to be one of the most important factors for marsh recovery and layer depths in the range of 15 to 20 cm seem to be within the upper limits for both vegetation recovery (abundance and percent cover) and sustained elevation gains for cordgrass dominated marshes (Hine, 2015).

Mud motor (or water-column recharge) techniques attempts to mimic more natural freshet sedimentation regimes in areas where marshes have become sediment starved. If fine sediment slurries can be pumped into a target area during flood tides, the material can migrate into the marsh while potentially reducing direct disturbance to the marsh (e.g., smothering vegetation with spray techniques). With this technique it may be difficult to evenly distribute sediment over the entire site; however, if movement of fine silts and sands is primarily shoreward, low marsh zones with higher flood period will have increased opportunity for deposition and accretion. Low marsh zones typically accrete sediments faster than high marsh zones (Kirwan et al., 2016) suggesting that this approach may be suitable for increasing sediment supply. There is evidence to suggest that under flood conditions fine sands and silts will migrate landward (McLaren, 1996); however, sediment transport modelling will likely be required to better understand sediment dynamics and migration throughout the delta.

Sediment applications may require multiple additions to keep pace with sea-level rise over the next century. Unlike the Mississippi River delta, typical Fraser River delta dredged sediments are high in sand content. This may pose a challenge for providing adequate sediment slurry particle sizes for distribution amongst Fraser River delta front tidal marshes. If sediment augmentation measures were used in the Fraser River delta, sediment slurries for tidal marsh application would likely require additional dredge material of finer particle size to supplement typical dredgeate. Louisiana, a long-distance sediment pipeline was designed to remain in place as a permanent corridor for future sediment augmentation projects (CPRA, 2017). Similarly, a permanent or temporary sediment pipeline could be built in the Fraser River delta to increase range of sediment applications over shallow tidal flats. Planning for these types of strategies certainly requires consideration of long-term sediment supply and availability.

As sea-levels rise over the next century and beyond, it is clear that we need a better understanding of Fraser River delta tidal marsh ecogeomorphic processes and a way to accurately assess tidal marsh vulnerability under possible sea-level rise scenarios. In addition, pilot projects offer the opportunity to test ecosystem based adaptive management strategies and are needed to determine how successful they might be, what the potential effects are and how largescale applications could be implemented effectively. A better understanding of how sediment additions are retained, what the effects are on marsh vegetation and other species (including invasive species response) and how they can be effectively and efficiently applied is needed.

Based on this review, the following recommendations are made for potentially increasing tidal marsh resilience to sea-level rise over the next century:

1. ***Fraser River Delta-Wide Marsh Accretion Modeling***

   Marsh accretion modeling that simulates marsh elevation response to sea-level rise can help determine potential sediment volumes required to keep pace with sea-level rise and assist in determining tidal marsh areas that are at higher risk of drowning. Some examples of these models include Wetland Accretion Rate Model of Ecosystem Resilience (WARMER), Marsh98, Marsh Equilibrium Model (MEM) and Sea Level Affecting Marshes Model (SLAMM). Models should include dynamic accretion rates and marsh migration potential where possible to improve our understanding of potential tidal marsh response to sea-level rise. For this type of modelling to inform delta-wide tidal marsh management, site specific data is required, including marsh elevation change over time, accretion rates across elevation gradients, lateral migration of marsh boundaries, and anthropogenic barriers to marsh migration.

2. ***Sediment Augmentation Pilot Projects***

   While sediment augmentation strategies have shown promise for increasing tidal marsh resilience to sea-level rise, there are still many uncertainties with long-term response and how application measures can be implemented successfully over large areas. Most available literature on sediment augmentation techniques is from the South Atlantic coast of the USA in deteriorating *Spartina alterniflora* salt marshes and brackish and salt marshes within the Fraser River delta may respond differently to sediment applications.

   Results of pilot projects can help inform additional phases of work towards application at larger scales as part of an adaptive management framework. It may take years to decades to understand the effects of these strategies and how they can be implemented successfully at larger scales and therefore, pilot projects should be designed and implemented as soon as possible. If possible, both direct (layered lifts) and indirect (mud motor) sediment augmentation strategies should be tested as both strategies can provide additional insight. Sediment augmentation is expensive, so developing pilot project designs and monitoring programs that provide useful learning opportunities are important.

Currently, sediment augmentation pilot projects are being considered in both Boundary Bay and Sturgeon Bank and preliminary options are being developed. To mitigate the effects of coastal squeeze in Boundary Bay, two locations are now being planned that will pilot the living dike concept over the next nine years. At Sturgeon Bank, north of the Steveston Jetty, a sediment augmentation pilot project has been proposed that can take advantage of mud motor concepts. The method would potentially utilize both small sand dunes placed on the tidal flats and fine sediments that would be pumped into the area during flood tides to mimic natural freshet sediment inputs. These locations are good options for sediment augmentation pilot projects in the Fraser River delta as Sturgeon Bank receives reduced supplies of fine sediments and Boundary Bay has been effectively separated from Fraser River sediment sources.

Pilot studies should be designed to take into consideration changes at control and augmentation sites before and after augmentation (e.g., BACI design) to factor in temporal changes. To avoid pseudo-replication, multiple control and treatment sites would improve the ability to detect effects, although may not be possible. Pilot projects would benefit from a variety of monitoring surveys (e.g., plant community surveys, sediment core analysis, marsh elevations, tidal creek accumulations, sediment characteristics such as nutrients, particle size and salinity, turbidity effects on eelgrass communities, mammal, fish, invertebrate, and avian use). Light Detection and Ranging (LiDAR) remote sensing, multispectral imagery and ground truthing can be used to map and monitor large scale changes in elevation and vegetation over time. Ultimately, successful implementation may be defined by whether sediments applied to the site increase rates of accretion, if elevation gains are maintained over the long term, and if marsh vegetation recovers. Adaptive management, especially in a complex system with many knowledge gaps, involves learning by doing and using the results of these initial smaller-scale projects to develop and improve subsequent management opportunities.

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Appendix A.

Thin Layer Sediment Augmentation Technique Observations & Results Summaries

Author(s)	Location	Dominant Species	Sediment Placement	General Methods Overview	Summary of Results
Reimold et al., 1978	Georgia; Glynn County	*Spartina alterniflora*	Manual; small enclosures Coarse sand, mixed sand and clay, and clay	Replicate plots (corrugated metal pipe) where three dredge material grain sizes (coarse sand, mixed sand and clay, and clay) were applied at six different thicknesses (8, 15, 23, 30, 61 and 91 cm). Materials placed on undisturbed marsh.	*Spartina alterniflora* was able to grow through up to 23 cm of material regardless of grainsize. Growth was comparable to reference marshes. Marshes did not recover when 60 cm or more material was applied.
Cahoon and Cowan, 1988	Louisiana; Dog Lake and Lake Coquille	*Spartina alterniflora*	Various	Qualitative evaluation at two disposal sites in a saline marsh 14 months after placement. Sediments deposited 10-15 cm deep at one location and 18-38 cm deep at the other location.	14 months after placement, vegetation was still smothered at both sites with recolonization beginning. Authors estimated three years to full revegetation. Lack of pre-placement elevation data limited interpretations.
DeLaune et al., 1990	Louisiana; Barataria Bay	*Spartina alterniflora*	Manual; small enclosures 40% fine sand, 28% coarse-fine silt, 32% clays and organics	Dredged sediment applied to 12 plywood enclosures in a Spartina alterniflora salt marsh. Enclosures were 1.44 square meters and 10 cm high. A randomized block design was used with two sediment treatments. Sediment was applied at thicknesses of 2 to 3 cm and 4 to 5 cm in 1986 and a second addition was applied to 4 to 6 cm and 8 to	Above ground biomass and density in both treatments was comparable to controls.

				10 cm in 1987. Plant aboveground biomass was assessed.	
Pezeshki et al., 1992	Louisiana; Barataria Bay	*Spartina alterniflora*	Manual; small enclosures	Study enclosures were the same as those described in DeLaune et al. (1990). 22 months after the initiation of the study, leaf conductance, transpiration and photosynthesis measurements were conducted in the different treatments	Leaf conductance and transpiration rates were higher in sediment augmentation sites when compared to controls.
Wilber, 1992	North Carolina; Gull Rock	*Spartina alterniflora*	Various	Thin-layer disposal of dredge material was applied to a marsh at a depth of 5 cm along the canal and 10 cm at an island site. This assessment was completed approximately 10 years after placement of material to characterize long-term effects. In addition to the disposal sites, two reference sites were examined. Marsh characteristics were assessed quantitatively for plant biomass, density, relative elevation, soil bulk density, soil organic content and macroinfauna density. Qualitative assessments included fiddler crab and fish abundances and soil layering.	Lack of predisposal data limit interpretations. Site characteristics such as soil bulk densities, marsh vegetation, organic contents and faunal distribution indicate productive marshes. Disposal sites consisted of higher elevations, higher soil bulk densities, lower soil organic content and lower vegetation cover compared to the reference sites. Comments from the author indicate that placing layers of dredged material 5 cm thick did not lead to major changes in vegetation (25% less shoot density noted). A 10 cm thick layer showed a different dominant species than the reference site (D. spicata and S. alterniflora vs. J. memerianus and D. spicata) and may have altered soil drainage resulting in altered salt content.
Ford et al., 1999	Louisiana; Venice	*Spartina alterniflora*	Spray dredging	Impact of spray dredging assessed on a 0.5-hectare salt marsh. Thickness measured from artificial soil marker horizons. Elevation	Measurements immediately following application (July 1996) showed that the marsh was covered with 2.3 cm of dredged material and that stems of *S.*

				change measured from sedimentation-erosion tables (SET) installed in treatment and reference sites prior to spraying.	*alterniflora* were knocked down. Stems were observed to recover and by July 1997 the percent cover of *S. alterniflora* had tripled. The layer of dredge was thin enough to allow plant survival with no higher marsh zone species colonization. Within a year of spraying, soil bulk density, percent organic matter, root/rhizome biomass and newly laid sediment volumes had returned or exceeded levels measured prior to spraying.
Mendelssohn and Kuhn, 2003	Louisiana; Venice	*Spartina alterniflora*	Spray-dredging Sediment slurry (85% liquid, 15% solids)	Hydraulic dredging for a gas pipeline was completed in 1992 resulting in material being placed in an adjacent marsh at varying depth gradients. A laser level was used to measure depth of added sediment. The marsh was divided into five regions (reference, trace, <15cm, 15-30cm and >30cm sediment additions). 25 sampling plots were used to collect vegetation data (biomass and percent cover) in 1993 and 1994. Soil physico-chemical characteristics were also measured.	Observed increase in total vegetative cover, plant height and biomass (S. alterniflora) with increasing levels of added sediment. No change in species composition observed. *S. alterniflora* was observed to re-establish from seed in the 15-30cm and >30cm site. In the <15cm sediment subsidy site revegetation was observed to occur mostly from vegetative regrowth. Soil characteristics indicated an increase in redox potential and soil aeration, bulk density iron and manganese concentrations corresponding to increasing levels of added sediment. Decrease in interstitial sulfide concentrations was observed with increasing levels of added sediment.
Slocum et al., 2005	Louisiana; Venice	*Spartina alterniflora*	Spray-dredging	Continuation of the Mendelssohn and Kuhn (2003) study. Sample plots and survey parameters were	Trend seven years after treatment showed better growth conditions under moderate elevation

			Sediment slurry (85% liquid, 15% solids)	replicated from Mendelssohn and Kuhn, 2003.	enhancements. Areas receiving 5-15cm of sediment had 10% more cover than areas receiving greater amounts of sediment. Potential reasons for the initial linear response were discussed relating to an initial pulse of growth due to sediment enrichment. Long lasting effects appeared to be due to increased elevation more so than nutrient additions.
Leonard et al., 2002	North Carolina; Masonboro Island	*Spartina alterniflora*	8 cubic meters of dredge material taken from dredge disposal area and manually placed in deteriorating marsh plots. Placement at high tide to reduce impact on vegetation, simulate slurry disposal, and create uniform distribution Medium to coarse grained material; comparable to surrounding conditions (50% fine sand and 50% muds)	Dredged sediment manually placed in four study plots: 2 deteriorated marshes and 2 non-deteriorated at varying thicknesses (0cm to 10cm). Parameters included response of plants, microalgae (BMA), benthic infauna, and sediment redox potential. Additionally, short-term deposition rates, flow, and changes in sediment composition were also studied. Plots were monitored from May 2000 to October 2001.	Placement of dredge material on deteriorating marshes led to increases in stem densities and microalgal biomass with minimal impacts to non-deteriorating marshes. Thickness of sediment added did not significantly affect stem densities or benthic microalgae; however, stem density was more comparable to reference conditions where sediment additions were thickest. Marsh sediments were sandier where sediments treatments occurred (and where thickest). Benthic invertebrate populations were initially lower than reference; however quickly recovered in all plots.

McAtee, 2018	California; Seal Beach	*Spartina foliosa; Batis maritima*	Slurry spray: Hay bales, straw wattles, sandbags and geotextile fabric were placed along the perimeter to minimize sediment loss during augmentation 85-95% sand with little silt and clay content	Study examining short-term impact of sediment augmentation on vegetation and invertebrate communities. Two areas used including an augmentation site and a control site; 25 cm of sediment was sprayed over 7.9 acres; monitoring occurred one month after augmentation in the spring (2016) and 12 months after augmentation (spring 2017). Monitored effects of sediment augmentation (before and after) on vegetation (percent cover, community composition, photosynthetic rates), abiotic parameters (temperature, pore water salinity), and benthic invertebrate community (abundance, species richness, diversity and community composition).	Post augmentation, there was a decrease in plant cover and benthic invertebrate communities due to smothering. Dominance of benthic invertebrate species shifted from oligochaetes and polychaetes to insects. At six months, Spartina foliosa started to return through spread from the edges of the site. There was no evidence that any vegetation regrowth was from previously existing plants at the treatment site. Use of coarse sediment dredged material potentially resulted in slower growth and colonization of plants. Coarse sediment drains quicker and reduced the amount of organic matter. Increased drainage leads to higher evaporation rates which may explain the increased sediment salinity at the treatment sites. Increased salinity may inhibit germination and growth of vegetation as well as invertebrate species.
Graham and Mendelssohn, 2013	Louisiana; Vermillion Parish	*Spartina patens*	Sediment pumped from adjacent canal. 70-80% water and 20-30% sediment using a hand-operated dredge 82% Silt and Clay	Application of varying thicknesses of sediment slurries to deteriorating marsh. Objective to determine elevation change and sediment effects on ecosystem processes 20 3mx4m vegetated plots were used in a deteriorating spartina patens marsh. Elevated wooden board walks were constructed	2.3 to 20.3 cm of sediment nourishment increased soil elevation initially; however, after 2.5 years, sediment nourished areas subsided to pre-sediment surface elevations and were no different than reference sites. Soil compression and consolidation were related to sediment application thickness. Plots where sediment nourishment was <10 cm thick resulted in elevations lower than pre-

				around each plot and wire-backed silt fencing was attached to the boardwalk to contain the sediment.	sediment levels (approx. 2cm). Plots that received >15cm had small elevation gains (approx. 3 cm). Plots that received >15cm stimulated plant growth.
Streever, 2000	Various	*Spartina alterniflora*	Various	Quantitative review of three decades on marsh creation and restoration in *Spartina alterniflora* marshes using dredge material	Dredged material sites don't necessarily become increasingly similar to natural marshes. They provide some of the functions but likely do not replace all the functions of a natural marsh. Long-term trajectories may be different.
Tong et al., 2013	Louisiana; Leeville	*Spartina alterniflora*	Hydraulically dredged sediment slurry pumped into 1.5ha cells	Sediment was pumped into cells controlled by levees. After sediment addition, the levees were broken to allow tidal exchange.	Moderate sediment additions restored macroinvertebrate species richness, diversity, density and total biomass. Species and taxa variably recovered depending on level of treatment. High sediment additions resulted in impaired recovery across all metrics.