Investigating the Effect of Salinity, Elevation, Redox Potential, and Geese Herbivory on Planting Success in a Pacific Northwest Salt Marsh

by
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Abstract

The MacKay Creek Estuary, is a severely altered estuarine ecosystem located within an active international port in Vancouver, British Columbia. Several elevated salt marsh terraces were constructed as part of a larger restoration project within the MacKay Creek Estuary. Site visits conducted in 2018 revealed 75% of the terrace surface area failed to establish salt marsh vegetation. Significant difference in soil pore-water salinity, oxidation reduction (redox) potential and tidal elevation were found between vegetated and unvegetated portions of the terraces. Additionally, exclusion from Canada Geese (*Branta canadensis*) herbivory increased total percent cover and colonization of the adjacent unvegetated area. The combination of soil parameters and herbivory, as well as potential interactions between factors may be responsible for the lack of vegetation progression within the constructed salt marsh terraces at MacKay Creek Estuary.

Keywords: Salt Marsh; Redox Potential; Pore-water Salinity; MacKay Creek; Fraser River; Estuary; Canada Geese; Restoration

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List of Acronyms

BEC	Biogeoclimatic Ecosystem Classification
BCIT	British Columbia Institute of Technology
CWHdm 06	Coastal Western Hemlock Dry Maritime Site Series 06
DEM	Digital Elevation Model
GIS	Geographic Information System
GNSS	Global Navigation Satellite System
GCM	Ground Control Monument
LiDAR	Light Detection And Ranging
NAPL	Non-Aqueous Phase Liquids
ORP	Oxidation-Reduction Potential
PAH	Polycyclic Aromatic Hydrocarbons
PVC	Polyvinyl Chloride
SFU	Simon Fraser University
TBT	Tributyltin
CGVD2013	Canadian Geodetic Vertical Datum of 2013
MAMSL	Metres Above Mean Sea Level

Introduction

Humans have caused major alternations in many of the world's coastal marine and inland ecosystems (Vitousek et al., 1997). Nearly one third of the world's populations are living in coastal areas, which account for only 4% of the total land surface area (UNEP, 2006). Multiple anthropogenic stressors, including ecosystem modification, contamination, and climate change, have placed a disproportionate strain on coastal ecosystems (Silliman et al., 2009; von Glaslow et al., 2013). Coastal colonization, development, and resource exploitation have resulted in a significant global decline in salt marshes (UNEP, 2006; Li et al., 2018). Research into the function of salt marshes has illuminated the wealth of ecosystem services provided by these habitats (Levin et al., 2001; Williams, 2002; Chmura et al., 2003; Carvalho et al., 2010). Recognition of the ecological importance of wetlands has galvanized numerous governmental and non-governmental organizations to conserve, restore, and remediate these valuable ecosystems (Joshi and Bhandari, 2016).

Tidal salt marshes are coastal wetlands that lie at the intersection of land and ocean. They are a transitional ecosystem that is regularly flooded by euhaline to mixohaline water brought in by the tides. Salt marshes occur worldwide but are particularly common throughout the middle to high latitudes. Despite their large distribution, salt marshes are incredibly rare, covering less than 1% of the British Columbia coast (MacKenzie and Moran, 2004). Salt marshes form where the abatement of hydrodynamic processes allows for sedimentation of suspended particulate matter from the water column. They are often found in estuaries, fjords, bays, and on the leeward side of large islands where sediments are protected from the destabilizing forces of wave, tidal, and fluvial processes (MacKenzie and Moran, 2004). In estuaries, fluvial deposition of sediments is facilitated by the reduction in water velocity as rivers flow into the relatively quiescent waters of the estuary. Sedimentation of suspended organic and mineral material gradually form expansive gently graded mudflats comprised of fine-sandy silty gleysols or humic gleysols (MacKenzie and Moran, 2004). Colonization of mudflats by salt-tolerant emergent, gramanoid, forb, and succulent plant species further aid in sediment accretion by attenuating hydrodynamic energy and stabilizing sediments (Seliskar, 1983; Bouma et al., 2005; Gutierrez et al., 2012).

Ecosystem Services of Salt Marshes

The importance of salt marshes is exemplified by the abundance of services and functions derived from them. These services are expressed in many different ways, including the fixation and transformation of solar energy, carbon sequestration, support of aquatic and terrestrial food webs, wildlife refugia, water filtration, nutrient cycling, and disturbance regulation (Chmura et al., 2003; Gedan et al., 2009; Barbier et al., 2011). The dual nature of intertidal ecosystems results in the provision of both aquatic and terrestrial ecosystem services (Barbier et al., 2011). Consequently, salt marshes are ranked as providing some of the most valuable ecosystem services of natural systems (Costanza et al., 1997; Levin et al., 2001).

Despite having low species diversity, the significance of salt marsh vegetation is pronounced (Levin et al., 2001). Net primary productivity of salt marshes ranges between 0.6 to 2.0 kg m^{-2} yr^{-1} above-ground dry matter rivalling tropical rain forests which produce an average of 0.31 to 2.2 kg m^{-2} yr^{-1} (Marinuccii, 1982; Clark et al., 2001). Sediments continually bury organic carbon, in the form of plant biomass, where anaerobic conditions drastically reduce decomposition (Batjes and Sombroek, 2003). Unlike freshwater wetlands, salt marshes are unique in that they emit significantly less methane (Poffenbarger et al., 2011). Tidal salt marshes therefore function as net carbon sinks due to their ability to produce, trap and sequester organic carbon. World wide, the carbon storage capacity of salt marshes is estimated to be 4.8-87.2 Tg C y^{-1} (Mcleod et al., 2011). This is the equivalent of storing CO_2 emissions from 1.0-18.6 million passenger vehicles per year, based on an emission rate of 4.42 t CO_2 yr^{-1} per vehicle (United States Environmental Protection Agency, 2018). Furthermore, restoration of these ecosystems has been shown to dramatically increase carbon sequestration rates, relative to nearby mature marshes (Wollenberg et al., 2018). Thus, coastal wetland restoration could potentially be deployed as part of a larger climate change mitigation strategy to reduce the concentrations of CO_2 in the atmosphere (Moomaw et al., 2018).

The high primary productivity of salt marsh ecosystems support complex bottom up and top down food web interactions (Valiela et al., 2004). As tidal salt marshes are a quintessential bridge between terrestrial and aquatic biomes, restoration of these ecosystems directly benefit food webs related to both. Coastal invertebrates, fish, birds, and mammals all rely on marsh vegetation to fix energy and redistribute it throughout the ecosystem. It is estimated that up to 80% of all coastal wildlife species use estuaries throughout their life, including the full suite of iconic species in British Columbia (Flynn et al., 2006). Many red-listed and endangered species, such as wolverine (*Gulo gulo*), horned Lark (*Eremophila alpestris strigata*), Pacific water shrew (*Sorex bendirii*), and red knot (*Calidris canutus*) utilize these areas for forage or shelter (B.C. Conservation Data Centre, 2018). Salt marshes also support many ecologically and economically valuable fisheries such as oyster, clam, shrimp, and finfish (Boesch and Turner, 1984). Particularly important to the Pacific Northwest are salmonid species that use salt marshes as juvenile nurseries and areas of acclimation for migrating adults (Levy and Northcote, 1980). Chinook salmon (*Oncorhynchus tshawytscha*) heavily rely on estuaries throughout their life cycle (Bottom et al., 2005). Alterations to Chinook salmon production directly affect southern resident killer whale (*Orcinus orca*) populations, who use Chinook salmon as a primary food source (Hilborn et al., 2012).

Coastal salt marshes also function as physical, chemical, and biological water purification systems (Nelson and Zavaleta, 2012; Nagabhatla and Metcalfe, 2018). Suspended particulate matter is filtered out of the water column by several processes including sedimentation, flocculation, and mechanical filtration (Stumpf, 1983; Moskalski and Sommerfield, 2012). Fine textured sediments can serve as adsorption sites for several classes of ionic contaminants such as heavy metals, phosphorous, and refractory compounds (Beeftink et al., 1982). Coastal wetlands also have a range of dynamic redox processes that influence the transformation and cycling of compounds (DeLaune et al., 1983). Anaerobic environments favour the reduction of sulphur and sulfate into hydrogen sulphide, giving salt marshes their characteristic rotten egg smell (Devai and DeLaune, 1995). Hydrogen sulphide complexes with dissolved heavy metal ions to form insoluble metal sulphides that remains in the substrate under reduced conditions (Smith, 1923; Jiménez-Rodríguez et al., 2009). Nutrients are intercepted by macrophytic vegetation and cycled through the ecosystem, where they are transformed by bacteria, adsorbed, or exported to the adjacent water column (Sullivan and Moncreiff, 1988). Vegetation and bacteria can also accumulate, store, and break down contaminants through a variety of enzymatic processes (Carvalho et al., 2010; Williams, 2010; Lian et al., 2017). The ability of wetlands to function as water purification systems is reflected in the surfeit of literature on this subject.

Salt marshes also function as coastal defence systems by attenuating wave energy from storm surge (Weinstein and Kreeger, 2000). Many regional management strategies now include salt marsh management as a vital defence against sea level rise and coastal erosion. Thus, salt marshes are important for the ecosystem services they provide humans and wildlife. They provide wildlife habitat, sequester carbon, support diverse food webs, filter water, cycle nutrients, and regulate against disturbance.

Common Stressors to Salt Marshes

Human use of salt marshes likely precedes ancient Egypt and Mesopotamia, where the Tigris and Euphrates Rivers shaped the productive lands of the Fertile Crescent, giving rise to the cradle of civilization (Silliman et al., 2009). Numerous indigenous groups have relied on estuaries for hunting and resource gathering (Ham, 1982; Bernick, 1998). Their accessibility, fertility, biodiversity, and proximity to freshwater have historically made salt marshes attractive places for human settlement (Gedan et al., 2009). Today, it is estimated that between 30-50% of the world’s population reside within 100 km of the coast (Culliton, 1998; Small and Nicholls, 2003; Gedan et al., 2009). Over 80% of the world’s megacities (cities with more than 10 million inhabitants) are coastal or have coastal influence (Glasow, 2012). Unsurprisingly, salt marshes now face a continuum of stressors arising from a long and varied legacy of colonization and anthropogenic modification.

Estuarine and salt marsh transformation have rapidly accelerated within the past three centuries (Lotze et al., 2006). Habitat loss, exploitation, and pollution have resulted in the depletion of historically important species and degradation of seagrass and wetland habitats. It is estimated that over 90% of economically valuable estuarine species have been overexploited and between 50-65% of coastal wetlands have been lost (Lotze et al., 2006; Li et al., 2018). Human mediated impacts are multifarious but stem from several broad categories of activities including development, alterations to tidal and flow regimes, harbor and navigation channel construction and maintenance, heavy metal and nutrient pollution, and biological invasions (Mccaffrey and Thomson, 1980; Anisfeld et al., 1999; Bart and Hartman, 2000; Pontee, 2004; Miller et al., 2008; Deegan et al., 2012). The effects of these stressor are compounded by climatic factors, such as sea level rise, subsidence, climate change, and ocean acidification (Crosby et al., 2016; Li et al., 2018). The interactions of human and climatic stressors have caused a 25% loss in global salt marsh area (Secretariat of the Convention on Biological Diversity, 2010).

Within the Fraser River estuary, this number is more dramatic, with over 70% of coastal wetlands lost due to agricultural and urban development (Flynn et al., 2006). Their proximity to navigable waterways make salt marshes economically valuable land for industrial and commercial enterprises alike. Port Metro Vancouver is one of North America's largest shipping ports, consisting of 28 major terminals for marine passengers and cargo, and four railroad connections (Port Metro Vancouver, 2014). Of the approximately 3,600 acres of coastal property owned by Port Metro Vancouver, only 200 acres remain undeveloped as of 2016. As of 2018, several major projects have been proposed or approved, including Roberts Bank Terminal 2, Kinder Morgan pipeline, Fortis LNG Facility on Tilbury Island, and the Vancouver Airport jet fuel marine terminal. Additionally, the Port of Metro Vancouver anticipates a 2,300-acre increase in demand for coastal land by 2025 (Port Metro Vancouver, 2014). Habitat loss through development is therefore expected to increase with rising demand for property.

As fish habitat loss has accelerated within the past century, Canada's freshwater fisheries resource has also been in decline (Kerckhove et al., 2015). The recognition that fish habitats are national assets worth protecting and restoring is exemplified by the development of the Government of Canada's No Net Loss principle outlined in the federal policy on wetland conservation (Department of Fisheries and Oceans, 1986). Although the policy aims to alleviate the ecological consequences of habitat loss, several challenges threaten the efficacy of the compensation and offsetting principle. Compensation and offsetting projects are often subject to a rubric that places greater value on habitats for economically valuable species (Lievesley et al., 2016). Intertidal mudflats, for example, are valued at 10-50% the value of intertidal marsh, resulting in a disincentive for project proponents to construct the variety of habitat types required to maintain coastal biodiversity (Lievesley et al., 2016). Lievesley et al. (2017), also found that only one third of marsh habitat compensation projects with the Fraser River estuary are deemed successful in actually compensating for habitat loss. This is similar to what has been observed nationally, where approximately 63-67% of projects resulted in net habitat losses (Quigley and Harper, 2006; Favaro and Olszynski, 2017).

The cumulative impact extends beyond the initial loss of habitat from each development. Increases in shipping traffic often result in the introduction of exotic and invasive algae species through the discharge of ballast water in coastal waters (Sutherland and Levings, 2013). Lievesley et al. (2017) cited high invasive and exotic plant cover coupled with a lack of native plant cover as the primary limiting factor for marsh compensation success. Of the 1,446 different vascular plant species that occupy the lower Fraser Basin, approximately 40% are introduced aliens (Gray and Tuominen, 1998). More development increases the potential for industrial runoff, wastewater discharge and accidental spills. Many smaller basins within the lower Fraser River have water quality issues from agricultural, industrial and urban run-off (Gray and Tuominen, 1998). Some locations within the Fraser River estuary have levels of polycyclic aromatic hydrocarbons (PAHs), dioxin and furan congeners that exceed limits set for the protection of aquatic wildlife (Gray and Tuominen, 1998). Wood pilings preserved with creosote have been associated with elevated concentrations of PAHs in nearby sediments. While not uncommon in densely populated coastal locations, contamination of marine sediments from point and non-point pollution sources is a pervasive threat not easily remedied by ecological restoration.

Development in the Lower Mainland has also led to a loss in habitat connectivity. The once expansive floodplain of the Fraser River has been gradually narrowed to several channels through the installation of an extensive network of flood control structures. Dikes and floodgates serve to mitigate flood risk for the estimated one-third of British Columbia's population that lives within the lower Fraser River area (Gray and Tuominen, 1998). However, research on the impact of dikes on ecosystem function illustrates the importance of maintaining tidal connection to salt marshes (Ibarra-Obando and Poumian-Tapia, 1991; Portnoy and Giblin, 1997). Flood control structures impact flow regimes, which is a major determinant of physical habitat, and the subsequent species that inhabit it (Bunn and Arthington, 2002). Diking has been associated with salt marsh recession and loss, while tidal floodgates have resulted in upstream water quality issues, invasive species abundance, and shifts in animal and plant communities (Hood, 2004; Gordon et al., 2015; Scott et al., 2015). The result is a reduction in habitat quantity, diversity, and ecosystem services.

MacKay Creek Estuary

Site Introduction

The MacKay Creek Estuary is a small drowned-river valley estuary located on unceded Squamish and Tsleil-Watuth traditional territory on the southwest coast of British Columbia, Canada. It is situated between the municipal borders of the City and District of North Vancouver and flows into Burrard Inlet (Fig. 1).

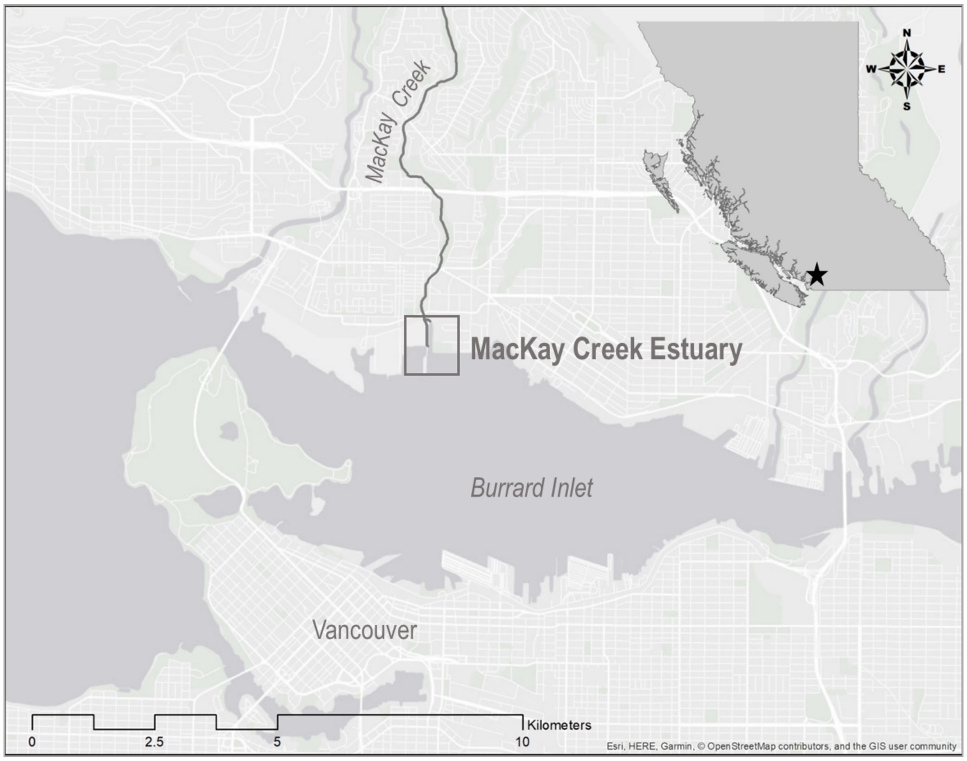

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Figure 1. Map of MacKay Creek Estuary location. The estuary is located within a highly ubanized matrix and borders the City and District of North Vancouver, BC. The mouth of the estuary flows into Burrard Inlet and the Port of Metro Vancouver.

The estuary falls within the Coastal Western Hemlock dry maritime site series 06 (CWHdm 06) biogeoclimatic (BEC) subzone (Pojar et al., 1991). Mean annual precipitation of Burrard Inlet is 1,827 mm, with the majority falling as rain (Pojar et al., 1991). The climate is mesothermal, with warm dry summers that average 17.6 °C and a mean precipitation rate of 53 mm during the driest month. During fall and winter, temperatures seldom permit snow cover to remain and mid-latitude cyclonic storms frequent the area resulting in a peak monthly precipitation rate of 292 mm (Pojar et al., 1991).

The watershed is 7.8 km^2, feeding into an 8.1 km long creek that flows through second-growth coniferous forests and a highly urbanized residential and commercial matrix. Approximately 66% of the watershed is currently developed for residential and commercial land use that account for 50% and 16% of the watershed, respectively (CH2M Hill, 2002). Despite having an impervious surface area of 29%, precipitation intensity and stream discharge are poorly correlated at MacKay Creek (Kerr Wood Leidal Associates Ltd, 2002). Hydrometric data in upper MacKay Creek at Montroyal Boulevard (Government of Canada Station 08GA061) show an average flow of 0.232 m^3 s^{-1} with a calculated 56 m^3 s^{-1} 100 year peak flow (Northwest Hydraulic Consultants, 2013).

Several aquatic wildlife inventories have been conducted in lower MacKay creek revealing that at least ten species of fish, and one species of crayfish use the estuary (Table 1).

Table 1. Summary results of aquatic species inventories conducted at MacKay Creek Estuary, BC between 2005 to 2016.

Scientific Name	Common Name
Cottus aleuticus	Coastrange Sculpin[e]
Cottus asper	Prickly Sculpin[bcf]
Cottus cognatus	Slimy Sculpin[ab]
Gasterosteus aculeatus	Threespine Stickleback[ac]
Lampetra richardsoni	Western Brook Lamprey[f]
Oncorhynchus clarkii	Cutthroat Trout[bcf]
Oncorhynchus keta	Chum Salmon[d]
Oncorhynchus kisutch	Coho Salmon[bcf]
Oncorhynchus mykiss	Rainbow Trout[a]
Oncorhynchus gorbuscha	Pink Salmon[b]
Pacifastacus leniusculus	Signal Crayfish[a]

Note. Data acquired from species inventories and scientific sampling conducted by Coast River Environmental Services Ltd ([a]2005, [b]2006) Golder Associates ([c]2008, [d]2009) North Shore Streamkeepers ([e]2009) and Dillon Consulting Ltd ([f]2016).

A variety of other mammals and birds also use the estuary. Black-tailed deer (*Odocoileus hemionug columbianus*), North American beaver (*Castor canadensis*), North American river otter (*Lontra canadensisten*), osprey (*Pandion haliaetus*), and great blue heron (*Ardea herodias fannini)* have been spotted using the estuary at MacKay Creek, as well as many species of water birds. Burrard Inlet is part of the Pacific Flyway and is internationally recognized as an Important Bird Area where water birds from three different continents congregate for nesting, roosting, and staging (Gray and Tuominen, 1998; IBA Canada, 2018). MacKay Creek Estuary is heavily used by Canada geese (*Branta canadensis*), mallard (*Anas platyrhynchos*), American wigeon (*Anas americana*), and a variety of other waterfowl.

Historical Land Use and Disturbances

The North Shore of Burrard Inlet has been home to the Skwxwú7mesh (Squamish), Tsleil-Waututh and xʷməθkʷəy̓əm (Musqueam) First Nations from time immemorial. Prior to the arrival of European explorers and settlers, as many as 23 permanent and seasonal villages belonging to the Coast Salish peoples punctuated the coastal rainforest from Whistler to Port Moody (Squamish Nation, 2013). Their oral histories indicate that over 10,000 First Nations people cohabitated the Lower Mainland area, which included Burrard Inlet and the watersheds draining into it (Musqueam Indian Band, 2019; Tsleil-Waututh Nation, 2019).

Following European colonization in the mid-nineteenth century, North Vancouver founded its first sawmill in what was once the small community of Moodyville (Kahrer, 1988). The opening of the Pioneer Sawmill in 1862 marked the beginning of industrialized forestry practices in North Vancouver. Between the 1860’s and the 1930’s advances to West Coast logging technology accelerated the decline in forest canopy cover throughout the North Shore Mountains (Kahrer, 1988). By the 1920’s, most of the easily accessible timber was gone, leaving behind visible scars in the landscape and dramatically increasing erosion (Kahrer, 1988). Environment Canada reports from the 1950’s indicate that forestry and residential development of the MacKay Creek watershed were causing unstable stream flows, flash floods, scouring, and heavy erosion (Marshall et al., 1978). In 1995, a 7,000-m3 landslide occurred in the upper portions of MacKay Creek, prompting the District of North Vancouver to install a debris flow barrier (Seyd, 2012).

The lower portions of MacKay Creek have similarly suffered a long history of alteration and industrial land use. Two foreshore parcels of land flank the estuary to the east and west. The earliest land use record for the eastern parcel of land is by Wallace Shipyards Ltd (Later becoming the Burrard Dry Dock Company) in 1905. Having moved their operations from False Creek to North Vancouver, their shipbuilding yard was located at the foot of Lonsdale Avenue where they constructed both steel and wood vessels (Webb, 1996). During World War I they also manufactured 18-pound high explosive shells to support the war efforts (Woodward-Reynolds, 1943). In 1912, Lonsdale Estates filled in the tidal flat east of MacKay Creek as part of a local improvement project colloquially known as the Fell Avenue fill (City of North Vancouver, 1912). Thus started the process of infilling the natural MacKay Creek Estuary. The Fell Avenue fill was later leased to the Lyall Shipbuilding Company Ltd who constructed wooden ships on the site until 1919 (Woodward-Reynolds, 1943). The parcel of land was then used for a lumber mill from the 1930’s to the 1970’s. Land use had been primarily zoned for industrial purposes until the 1970’s when the development of the Harbourside Business Park and Harbourside Waterfront made the area more accessible to the public (City of North Vancouver, 2014).

The west adjacent parcel to MacKay Creek is the former Capilano Timber Company which operated a saw and shingle mill on the foreshore of Burrard Inlet between 1917 and 1932 (Woodward-Reynolds, 1943). Aerial photographs from the 1960’s show MacKay Creek Estuary was used as a log sort. Further west and adjacent to the Capilano Timber Company, Domtar Inc. operated a creosote wood preserving facility between 1933 and 1965 (Woodward-Reynolds, 1943). Both sites were taken over by the shipbuilding company Seaspan ULC (formerly Seaspan International Ltd.) in 1965 where they currently repair, maintain, and produce large steel vessels. In 2010, a remediation order was issued to Seaspan and Domtar due to elevated levels of highly mobile nonaqueous phase liquids (NAPL), polycyclic aromatic hydrocarbons (PAH), metals, and tributyltin (TBT) in the soil, groundwater, and sediments. (Environmental Appeal Board, 2013). The decision went through the appeals process until 2014 when the Environmental Appeal Board deemed Seaspan as the “responsible party” and they were ordered to uphold the 2010 remediation order (Environmental Appeal Board, 2014).

Water quality and sediment contaminant levels appear to vary at MacKay Creek. Benthic index of biological integrity (B-IBI) surveys done for Greater Vancouver Regional

District in 2003 rated the stream as “very poor” (Page and Sylvestre, 2016). This is in contrast to invertebrate surveys done by local Streamkeepers in 2007 and 2008 who found the presence of pollution sensitive mayflies (North Shore Streamkeepers, 2009). In 2014, a study conducted at MacKay Creek Estuary found elevated levels of copper within the sediments (Barlow et al., 2014). Several plots vastly exceeded the 187 mg/kg copper threshold set by the Canadian Sediment Quality Guidelines for the Protection of Aquatic Life. Additionally, a distinct “plume” of metals containing copper, zinc, cadmium, and lead was found to originate from a nearby sewer outfall that empties directly into the estuary (Barlow et al., 2014). The presence of zinc and lead appeared to be legacy based contaminants originating from historical land and resource uses, while copper was theorized to originate from the current use of copper-based antifouling paints and vehicle brake pads (Barlow et al., 2014).

Restoration of MacKay Creek

A portion of the estuary was restored in 2014 as part of a habitat compensation project funded by the Habitat Conservation Trust Foundation and Seaspan. The habitat creation project included removal of invasive plants, riparian revegetation with native plants, removal of a dam that blocked upstream fish passage, placement of large woody debris, and construction of a series of elevated salt marsh terraces on the western bank of MacKay Creek. The terraces were backfilled with sediments and revegetated using a mixture of transplants from a nearby salt marsh and nursery plant stock. Orange plastic snow fencing was installed along the eastern perimeter of the terraces to prevent Canada geese from grazing the newly planted vegetation. Despite these measures, post-restoration monitoring in 2014 and 2015 revealed that the terraces had variable success.

In May of 2018, vegetation mapping of the terraces found approximately 75% of the terraces substrate failed to establish salt marsh vegetation. Majority of successful transplants were located in the northern terraces, which had high rates of percent cover. Percent cover followed a general decreasing trend from north to south, with the southern terraces nearly devoid of plant cover.

Hypothesized Stressors

The stressful environment found in salt marshes makes them difficult places for many plants to inhabit. Species zonation refers to the distribution of species into characteristic zones based on physical parameters, such as elevation or soil water content. Plant zonation patterns in intertidal areas are often the result of tolerance thresholds to environmental stressors. Plants that colonize these dynamic areas must be able to withstand tidal inundation, summer desiccation, soil waterlogging, elevated salinity, hydraulic forces, and grazing, amongst others. The ability to withstand the unique set of environmental stressors found in coastal wetlands are what give salt marsh species their distinct advantage over upland species (Adams, 1963; Pennings et al., 2004; Silvestri et al., 2005). In mature marshes, inter and intraspecific competition plays a role in determining species compositions. In the early stages of salt marsh succession, however, physical limitations are likely pre-eminent in determining the colonization and establishment of salt marsh plant cover (Davy et al., 2011).

Elevation is one of the primary factors influencing a wide suite of environmental parameters that affect species zonation in salt marshes (Adams, 1963; Davy et al., 2011). Changes to elevation through sediment accretion is a fundamental feature of salt marsh succession (Figueroa et al., 2003; Davy et al., 2011). As mudflat sediments accumulate, they are eventually colonized by salt marsh species, which transition to upland species as elevation increases (Figueroa et al., 2003; Davy et al., 2011). Elevation controls plant distributions by mediating the degree to which plants and soils are exposed to the hydraulic forces of waves and currents. More importantly, elevation dictates the duration and frequency of tidal inundation, which in turn affect a variety of soil physicochemical processes (Anastasiou, 2003; Ferronato et al., 2018). Tidal inundation affects ion exchanges, replenishes nutrients, buffers pH, and removes toxic metabolites in salt marshes (Craft, 2016).

Sites that are lower in elevation are subject to longer periods of inundation, which also limits aerobic respiration. Oxygen is required by plants for various metabolic processes, therefore different salt marsh plants have evolved adaptations, such as aerenchyma that allow them to tolerate intermittent submergence from tides. Several studies have examined the effect of soil waterlogging on vegetation growth (Bandyopadhyay et al., 1993; Li et al., 2005; Pezeshki and DeLaune, 2012; Ferronato et al., 2018). Soils that remains saturated for extended periods often become strongly reduced and anoxic or anaerobic. Besides directly impacting plant aerobic respiration, a lack of oxygen in the soil can lead to the reduction of several compounds that are used as electron acceptors for microbial metabolic processes (Kralova et al., 1992). Nitrates (NO_3^-) are reduced at redox potentials of just ~340 mV, limiting their uptake by plants (Bandyopadhyay et al., 1992). With decreasing redox potential, sulfate is reduced to sulfide (H_2S) while manganese (Mn^{4+}) and iron (Fe^{3+}) are reduced to Mn^{2+} and Fe^{2+}. Ammonium nitrogen uptake by some plants is limited by sulfide, while the reduced forms of manganese and iron are toxic to some vascular plants (El-Jaoual and Cox, 2008; Fageria et al., 2008).

Salinity is another factor that plays a significant role in determining species compositions in salt marshes (Silvestri et al., 2005). Salt marsh plants employ different strategies to withstand the desiccating effect of salinity that arises from ionic, osmotic, and nutritional interactions (Shannon, 1997). Individual marsh plants have varying preferences to salt concentrations, with some species colonizing areas of hypersaline conditions and others tolerating brackish to freshwater soils (Hutchinson, 1988). Although many salt marsh species have unique adaptations that allow them to withstand elevated salinities, they often grow better in freshwater environments (Barbour and Davis, 1970). This has been demonstrated in several laboratory experiments where increases in soil salinity resulted in decreases in plant growth parameters (Barbour and Davis, 1970; Phleger, 1971; Reynolds et al., 2001; Redondo-Gómez, 2007).

Herbivores significantly influence vegetation community dynamics. Consumption of vegetation by migratory and non-migratory geese has been linked with loss of vegetation cover, exposure of bare sediments, and the development of hypersaline soil which preclude the reestablishment of marsh vegetation (Hik et al., 1992; Jefferies and Rockwell, 2002; Dawe et al., 2011; Dawe et al., 2015). In British Columbia, non-migratory Canada Geese are of particular concern. Non-migratory Canada geese refers to the exotic Canada geese that were introduced to Vancouver Island in the 1960’s and appear to be non-migratory as opposed to the native subspecies *Branta canadensis fulva* (Dawe et al., 2011). Studies done in the Little Qualicum and Campbell River estuaries found that grazing from non-migratory Canada geese (henceforth referred to as Canada geese) contributed to significant changes in frequency and/or mean cover values for up to twenty-four plant species (Dawe et al., 2011; Dawe et al., 2015).

Goals and Objectives

The factors that influence salt marsh plant zonation are often related and difficult to disentangle. The goal of this study is to determine the primary contributing factors to the lack of vegetation progression within the constructed salt marsh terraces at MacKay Creek Estuary. By comparing vegetation growth characteristics with elevation, soil waterlogging, soil salinity, as well as examining the effects of excluding non-migratory geese from portions of the salt marsh, I hope to identify the probable causes for the lack of planting success at MacKay Creek. This information can then be used to inform MacKay Creek stakeholders and provide land managers with information relating to decision-making.

Goal 1: Identify potential stressors that are preventing the establishment of planted salt marsh vegetation at MacKay Creek Estuary in North Vancouver.

> **Objective 1:** Establish thirty-eight random 1-m^2 plots to compare soil pore-water salinity, redox potential, and elevation in vegetated and unvegetated areas.
>
> **Objective 2:** Establish nine fenced and nine unfenced 1-m^2 plots to determine the effects of excluding geese herbivory on salt marsh plant live shoot frequency, percent cover, and height.

Goal 2: Determine which salt marsh species at MacKay Creek Estuary are most successful in terms of abundance and distribution.

> **Objective 1:** Using established plots, analyse species composition and canopy percent cover to determine relative percent cover and frequency for each plant species found at MacKay Creek Estuary.

Methods

Study Site

MacKay Creek Estuary is located on the North Shore of Burrard Inlet, between the municipal boundaries of the City and District of North Vancouver, B.C. (Fig. 2). The estuary currently falls within jurisdictions of three different land managers. City of North Vancouver manages the upland areas east and west of MacKay Creek, while Vancouver Port Authority has jurisdiction over the channel. Seaspan manages the land that directly borders their shipbuilding yard, including a small off channel habitat to the west of MacKay Creek (Fig. 2).

The site has a mixed tidal regime with two sets of unequal high and low tides occurring within a cycle. Between 2011 and 2018 tides of MacKay Creek Estuary ranged from +1.1 m Mean Higher High Water to -1.6 m Mean Lower Low relative to the Canadian Geodetic Vertical Datum of 2013 (CGVD2013).

In 2014, six terraces were constructed within the within the intertidal zone of the estuary on the City of North Vancouver's property. The terraces were constructed to elevate the substrate and make them suitable for planting salt marsh species. The terraces run parallel to MacKay Creek and range in size from 95 m^2 to 360 m^2, occupying a total area of 1,612 m^2. Each terrace was constructed using riprap retaining walls backfilled with sandy loam sediments. According to the technical specifications, the substrate was designed to be 55-75% sand, 7-25% silt, 4-16% clay, and 10-30% organic material (Northwest Hydraulic Consultants, 2013).

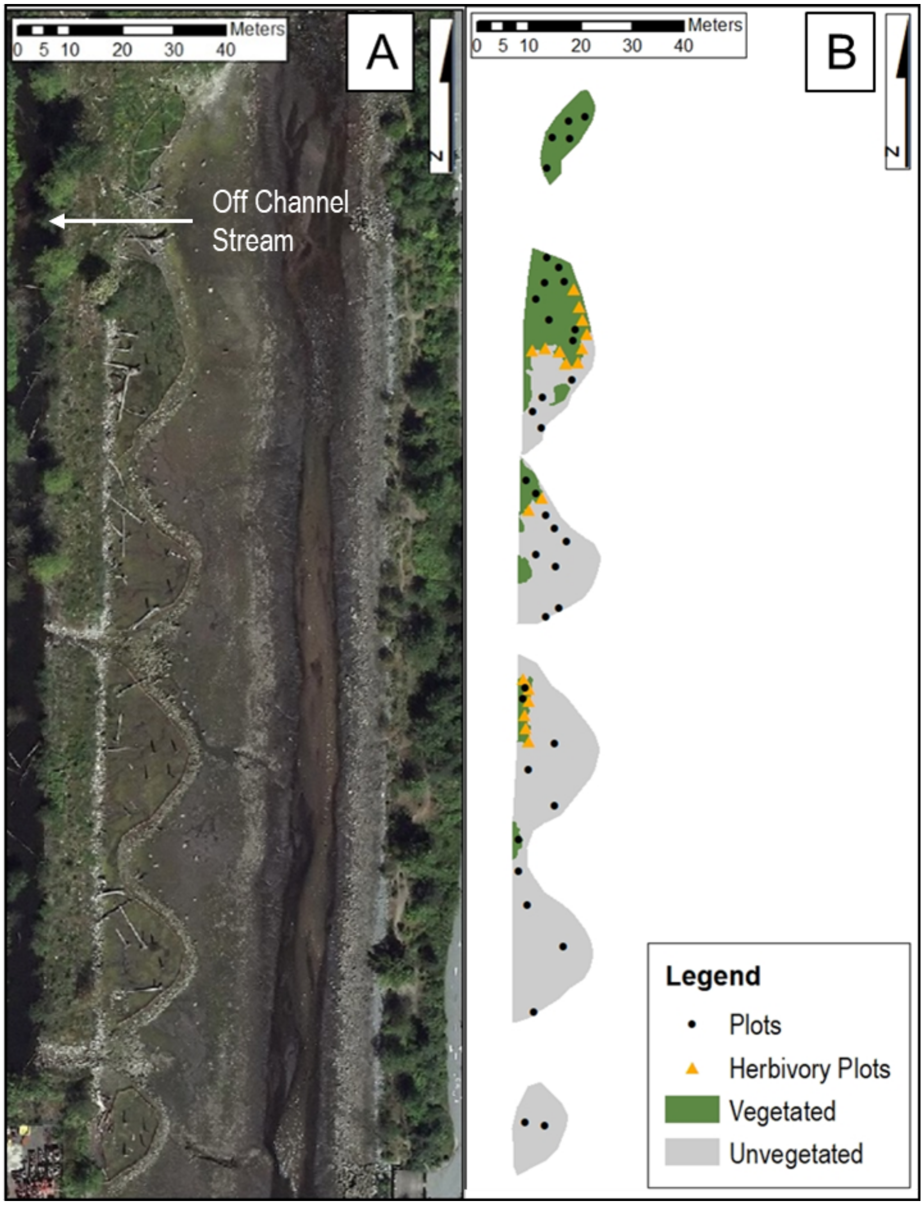

Figure 2. **(A) Map of the 2018 MacKay Creek Estuary study site, located in North Vancouver, BC (49°18’54.49” N 123°06’ 08.21” W). Constructed elevated terraces span the west bank of the estuary. West of the terraces, an off channel stream runs parallel to the salt marsh terraces. (B) Map of existing surface vegetation and plot locations.**

The terraces were planted in 2014 with seven salt marsh species: Lyngbye’s sedge (*Carex lyngbei*), pacific silverweed (*Potentilla anserina*), pickleweed (*Sarcocornia pacifica*), saltgrass (*Distichlis spicata*), seacoast arrowgrass (*Triglochin maritima*), seacoast bulrush (*Bolboschoenus martimus*), and threesquare bulrush (*Schoenoplectus pungens*). The soil preferences of each species are varied but generally fall within four main categories: salinity, elevation, soil drainage, and substrate type (Hutchinson, 1982).

Vegetation Survey

Thirty-eight 1-m^2 plots were used to monitor changes in vegetation dynamics from 2014 to 2018 and to compare soil parameters in vegetated and unvegetated plots. Vegetation data from 2014 came from a planting plan with area specifications for each of the seven planted species. The perimeter of existing vegetation patches were mapped during an early May, 2018 site visit using a handheld global navigation satellite system (GNSS) receiver (Trimble GeoExplorer 6000). Using GIS (ArcMAP 10.6), GNSS vegetation data were used to classify the marsh surface into the binary categories; vegetated or unvegetated.

Using a random point generator tool in ArcMAP 10.6, nineteen plots were created in both the vegetated and unvegetated areas. To avoid overlapping, the random point generator tool used a minimum spacing of three meters between plot centres. The coordinates of each random point were recorded and in early June 2018, the plots were located using a handheld GNSS receiver and marked with 60-cm wooden stakes.

Vegetation surveys were conducted between June-September, 2018. Using a 1-m^2 PVC quadrat, species presence/absence and canopy percent cover were recorded for each plot using a modified Braun-Blanquet (1932) cover score (Table 1).

Table 2. Modified Braun-Blanquet scale and midpoint values used for determining percent cover of salt marsh vegetation. Vegetation surveys were conducted between June to September 2018 at MacKay Creek Estuary, BC.

BB Value	Percent Cover	Midpoint
0	0 %	0%
1	<1 %	0.5 %
2	1-5 %	3 %
3	6-25 %	15.5 %
4	26-50 %	38 %
5	51-75 %	63 %
6	76-100%	88 %

Frequency, mean percent cover, relative cover and species cover were calculated using Rstudio. Frequency is defined as the proportion of all plots (n = 38) that contain a specific plant species. The midpoints of each Braun-Blanquet midpoint cover class were used to calculate the mean percent cover of each species. Mean percent cover was defined as the average percent cover of an individual species and included zero values.

Mean percent cover of each species was then used to calculate relative cover and species cover. Relative cover is the proportion of the vegetation cover comprised by a particular species, whereas species cover is the total estimated area occupied by a particular species. The following equations were used:

$$Spp\ A\ Frequency = \frac{Number\ of\ Plots\ Containing\ Spp\ A}{Total\ Number\ of\ Plots}$$

$$Mean\ \%\ Cover\ Spp\ A = \frac{\sum_{i=1}^{\#\ of\ Plots} Midpoint\ \%\ Cover_{(Spp\ A)}}{Total\ Number\ of\ Plots}$$

$$Spp\ A\ Relative\ Cover\ = \frac{Mean\ \%\ Cover_{(Spp\ A)} * Frequency_{(Spp\ A)}}{\sum_{i=1}^{\#of\ Species} Mean\ \%\ Cover_{(Spp\ i)}\ * Frequency_{(Spp\ i)}}$$

$$Spp\ A\ Cover = Relative\ Cover_{(Spp\ A)}\ * Total\ Vegetated\ Area$$

Soil Parameters

Pore-Water Salinity

Pore-water salinity was measured monthly from June through September, 2018 in thirty-eight plots. For the purposes on this study, the total concentration of pore-water salts was considered. Pore-water salinity, in parts per thousand (PPT) was measured using a handheld refractometer (HANSTRONIC). The refractometer was calibrated before each sampling day using distilled water as a ‘zero’ reference point.

Soil was sampled to a depth of 10 cm using a 2.2-cm diameter metal soil probe. The substrate sample was then placed within a coffee filter and squeezed until several drops of pore-water were extracted through the filter and onto the surface of the refractometer. In between each measurement, the refractometer was rinsed with distilled water and dried using paper towel. Each plot was sampled three times outside the quadrat using ~25-cm spacing to capture within plot variability. Pore-water salinity measurements were then averaged for use in statistical analysis. Salinity tolerances for each of the 2017 salt marsh species were assessed using mean literature values and compared to what was found at MacKay Creek estuary.

Soil Redox Potential

Soil oxidation-reduction (redox) potential was measured *in-situ* using a portable handheld pH/mV meter (HANNA INSTRUMENTS HI8424) with a single-junction platinum-tipped probe (HANNA INSTUMENTS HI3230B). The probe uses a built-in Ag/AgCl reference-probe from which oxidation-reduction potential (ORP) is measured against. Prior to each sampling day, the ORP probe was tested using a 470-mV calibration fluid to ensure that readings were within +/- 10 mV. Measurements were taken at the average rooted depth of 10 cm for salt marsh vegetation. A small soil core was removed using a 2.2-cm diameter metal soil probe. The ORP probe was inserted into the hole and lightly pushed into the substrate at bottom to ensure sufficient soil contact with the platinum pin. Readings were allowed to stabilize and subsequently recorded. A total of thirty-eight plots were measured monthly between June and September 2018. Redox potential was measured outside each plot a total of three times using 25 cm spacing. The ORP probe was rinsed with tap water followed by distilled water between each measurement. The mean of each plot was then used for statistical analysis.

Elevation

Elevation was determined using 2018 light detection and ranging (LiDAR) data from the District of North Vancouver. LiDAR data was converted into a digital elevation model (DEM) using GIS software (ArcMAP 10.6) and georeferenced to the Canadian Geodetic Vertical Datum of 2013 (CGVD2013). Plot locations were marked using a handheld GNSS receiver and uploaded into GIS software where elevation was then derived using the Add Surface Information function in the 3D analyst toolbox.

To determine if there is an approximate elevation where the seaward edge of the salt marsh vegetation transitions into mudflat, the mean elevation of vegetated and unvegetated plots was calculated then averaged. The average elevation was then used to create a contour line in ArcMAP which was superimposed over preexisting maps of vegetation cover. Goodness of fit was then assessed visually.

Herbivory Experiment

Eighteen 1-m^2 plots (nine fenced and nine unfenced) were established on the low edge of the vegetated zone of MacKay Creek in June 2018. The fenced plots were enclosed using 1.2-m high orange plastic fencing with a 2.5-cm mesh size. The unfenced plots remained unprotected, however, their locations were marked using 60-cm wooden stakes. The plots were placed on the lower edge of the vegetated marsh so that approximately 50% of the ground cover was vegetated and 50% was unvegetated. This was done to provide sufficient space for marsh colonization or recession to occur as a result of the treatments. Each plot had three meter spacing between centers and alternated between fenced and unfenced treatments.

Between June and September 2018, monthly measurements of species maximum height, species percent cover, total plot percent cover, and live shoot frequency were monitored in fenced and unfenced plots. Maximum height (henceforth referred to as just 'height') was determined by recording the tallest individual of each species from the substrate level to extended leaf tip. Live shoot frequency and species percent cover were measured using a 1-m^2 polyvinyl chloride (PVC) quadrat with one hundred 10-cm^2-subplot divisions. Live shoot frequency was defined as the proportion subplots that contained at least one living shoot of salt marsh species originating from the subplot. Shoots were considered living if at least part of the shoot contained green colour. Monthly measurements were then plotted to assess changes in fenced and unfenced plots over time.

Soil pore-water salinity, redox potential, and elevation were measured in fenced and unfenced plots were also measured to control for potential differences in soil parameters between treatments. Photos were also taken during each sampling period to visually assess changes in vegetation structure over time.

Statistical Analysis

All statistical analyses were done using Rstudio Version 1.1.463 for Windows (R Core Team, 2018). Correlation between variables was done using spearman's rank correlation. Tests for normality were performed on all data using the Shapiro-Wilk test and the appropriate significance tests applied to the distribution type. T-tests and the non-parametric Whitney–Wilcoxon U test were used to test differences in the means of soil parameters for vegetated and unvegetated plots. All significance was tested at the alpha 0.05 level.

For the herbivory experiment, T-tests and Whitney–Wilcoxon U tests were also used to test for differences in elevation, soil pore-water salinity and soil redox potential between fenced and unfenced treatments. Simple summary statistics were then used to compare species height, percent cover, and total plot percent cover in fenced and unfenced grazing plots. Data were plotted using line graphs to show changes in plant parameters throughout the sampling period.

Results

Sample Distributions and Covariate Interactions

Results of the Shapiro-Wilk tests for normality found that soil pore-water salinity and elevation of the sample plots were normally distributed, whereas soil redox potential had a skewed distribution. Results from a spearman ranks correlation matrix indicated there was a high degree of multicollinearity between sample data (Fig. 3). The strong interactions of elevation with soil pore-water salinity and redox potential precluded the use of multiple logistic regression for statistical analysis. Instead, simple summary statistics and two-sample significance tests were used to determine trends and make comparisons.

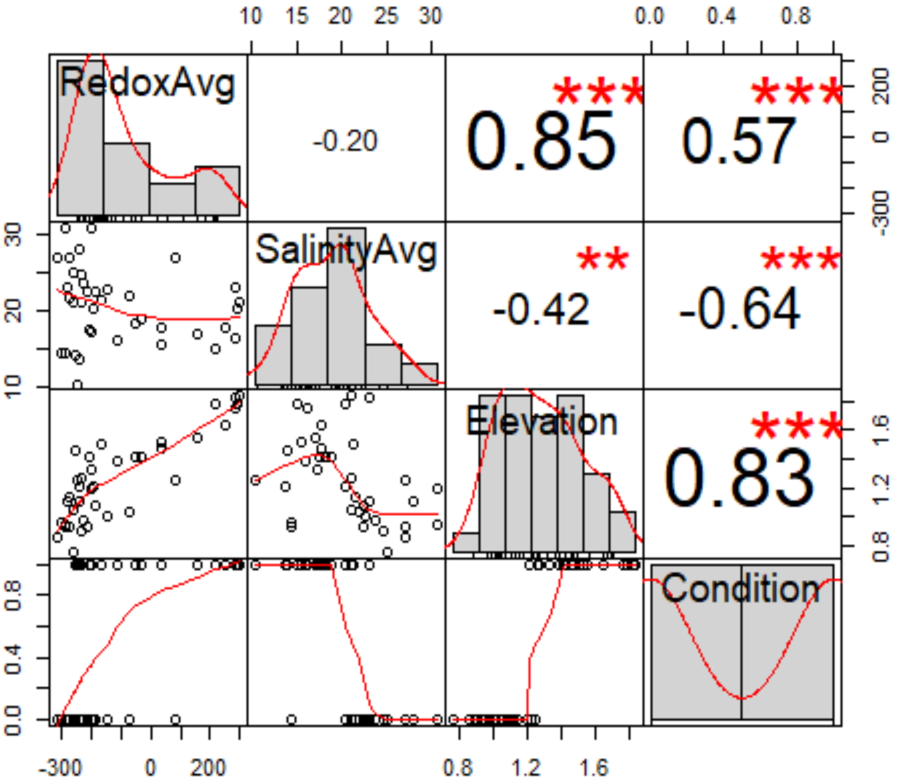

Figure 3. **Spearman ranks correlation matrix depicting relationships between redox potential, soil pore-water salinity, elevation, and condition in study plots at MacKay Creek Estuary, BC. Parameters were measured between July to September, 2018. Condition is a binary categorical variable (0 or 1) that represents the two classifications of land cover (unvegetated and vegetated).**

Soil Parameters

Two-sample T-tests were used to test differences in mean elevation and mean soil pore-water salinity. Due to the skewed distribution of redox potential, the non-parametric Whitney–Wilcoxon U test was used for differences in median soil redox potential. Unvegetated plots were generally higher in salinity, more reduced, and lower in elevation compared with vegetated plots (Table 3; Fig. 4).

Pore-Water Salinity

Mean soil pore-water salinity was significantly higher in unvegetated plots compared with vegetated plots (unpaired *T-test*, $p = 0.0000184$; Table 3; Fig. 4). Unvegetated plots had a mean pore-water salinity range of 14.6 ppt to 31.1 ppt during the July to September sampling period, while vegetated plots ranged from a mean of 10.6 ppt to 22.4 ppt. Both vegetated and unvegetated plots showed high variability within plots, among plots, and throughout the sampling period. Unvegetated plots were consistently more variable than vegetated plots (Table 4). Repeated measurements within unvegetated plots sometimes differed by as much as 26 ppt in some locations, indicating a high spatial heterogeneity even within a 1-m^2 sampling area. This variability was represented by the significant differences in variance for three of the sampling dates (July 17, August 15, and September 26), when unvegetated plots had approximately twice the standard deviation as vegetated plots (Table 4).

Mean soil pore-water salinity experienced fluctuations throughout the sampling period (Fig. 5). Average pore-water salinity was highest in July and decreased steadily until September. Soil pore-water salinity was significantly different between vegetated and unvegetated plots for three of the four sampling dates (2018-07-17, 2018-08-15, 2018-09-26; unpaired *T-test*, $p < 0.05$; Table 5; Fig. 5). Mean soil pore-water salinity in July was also significantly different than September (paired *T-test*, $p < 0.05$) for both vegetated and unvegetated plots.

Redox Potential

Median soil redox potential was significantly different in unvegetated plots compared with vegetated plots (Whitney–Wilcoxon U test, $p = 0.0002158$; Fig. 4; Table 3). In unvegetated plots, median redox potential was -254.9 ± 122.7 (s.d.) mV, though two plots had soils that fell within the positive range. Vegetated plots had a median of -6.0 ± 219.5 (s.d.) mV but had a larger range of redox potential values than unvegetated plots (Fig. 4). Throughout the July to September sampling period, redox potential was consistently more variable in vegetated plots compared with unvegetated plots (Table 5). This pattern was particularly pronounced during the August 15th sampling date, when vegetated plots had a standard deviation nearly quadruple what was found in unvegetated plots. This high variability was reflected by field measurements where within-plots redox potentials differed by as much as 300 mV in vegetated plots.

Redox potential was also significantly different between vegetated and unvegetated plots for all of the sampling dates (paired Whitney–Wilcoxon U test; $p < 0.05$; Table 5), However, it did not significantly change between the July to September sampling dates for either condition (paired Whitney–Wilcoxon U test, $p > 0.05$).

Elevation

Vegetated plots were higher in elevation than unvegetated plots (unpaired *T-test*, $p = <0.0001$; Fig. 4; Table 3). Vegetated plots had a mean of 1.52 MAMSL compared with unvegetated plots which had a mean of 1.02 MAMSL. Both vegetated and unvegetated plots had a similar variance, however vegetated plots were slightly more variable (s.d. = 0.19) than unvegetated plots (s.d. = 0.13).

The midpoint elevation value between vegetated and unvegetated plots was found to be 1.27 MAMSL. The creation of a contour line at this elevation yielded a map that visually lined up with the lower seaward salt marsh edge (Fig. 6).

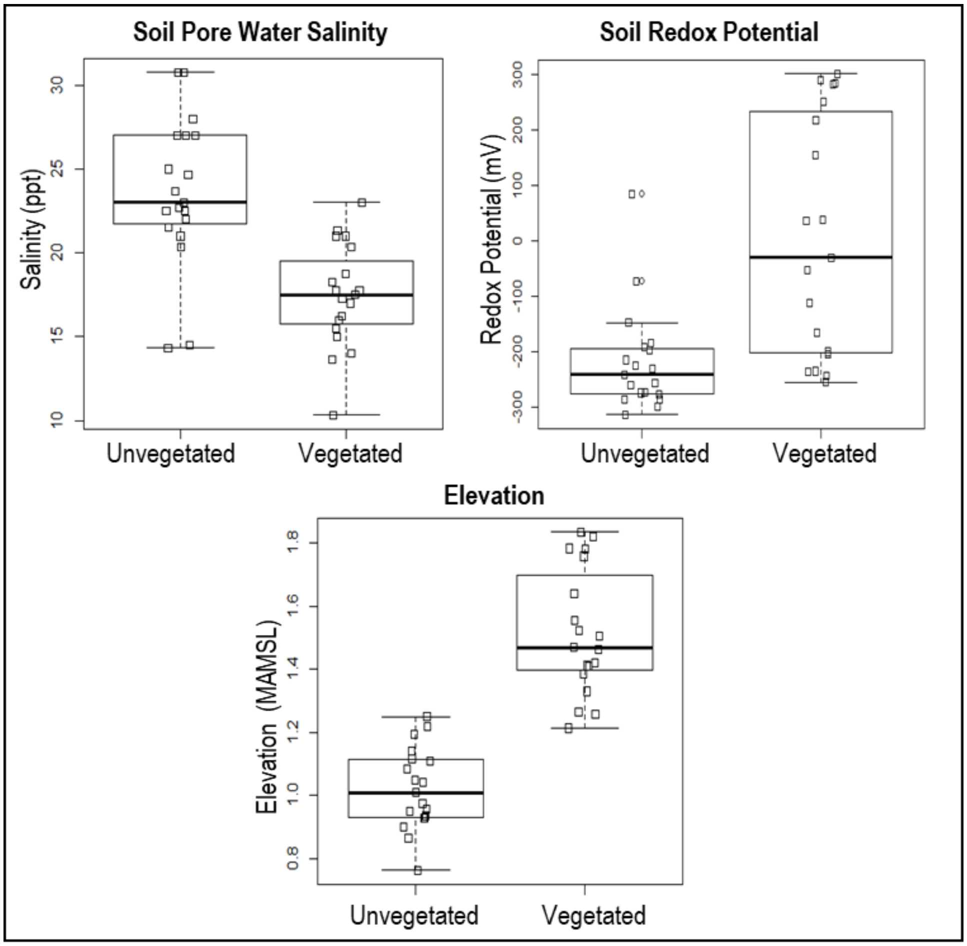

Figure 4. Boxplots comparing soil pore-water salinity, redox potential, and elevation in unvegetated (n = 19) and vegetated (n = 19) plots at MacKay Creek Estuary, BC. Data points represent the 2018 July to September mean of each plot.

Table 3. Summary values for pore-water salinity, redox potential, and elevation in vegetated (n = 19) and unvegetated plots (n = 19) at MacKay Creek Estuary, BC. Values represent the 2018 July to September mean or median.

Soil Parameter	Vegetated Plots	Unvegetated Plots	p-value
Mean Pore-Water Salinity ± s.d (ppt)	17.1 ± 5.8	23.7 ± 8.2	0.0000184
Median Redox Potential ± s.d. (mV)	-6.0 ± 219.5	-254.9 ± 122.7	0.0002158
Mean Elevation ± s.d. (MAMSL)	1.52 ± 0.19	1.02 ± 0.13	0.00000000008027

Note. Values represent the average of each parameter over the July to September sampling period and expressed as mean or median ± standard deviation (n = 19 per treatment). Significance tested using a 95% confidence interval.

Table 4. **July to September 2018 mean soil pore-water salinity ± s.d. in vegetated (n = 19) and unvegetated plots (n = 19) at MacKay Creek Estuary, BC.**

Condition	July 17***	August 2	August 15***	September 26***
Unvegetated	30.1 ± 8.0	20.8 ± 6.2	24.3 ± 6.7	15.0 ± 4.4
Vegetated	20.8 ± 4.0	19.3 ± 5.5	17.1 ± 3.1	9.0 ± 2.2

Note. Unpaired T-test was used to test for significant differences between vegetated and unvegetated plots for each sampling date * = $p<0.05$; ** = $p<0.01$; *** = $p<0.001$

Table 5. **July to September 2018 median soil redox potential ± s.d in vegetated (n = 19) and unvegetated plots (n = 19) at MacKay Creek Estuary, BC.**

Condition	July 17**	August 2***	August 15***	September 26**
Unvegetated	-254.9 ± 161.9	-235.7 ± 57.6	-272.6 ± 117.3	-221.2 ± 151.6
Vegetated	-99.5 ± 214.6	-106.3 ± 233.1	-6.0 ± 209.1	164.8 ± 217.2

Note. Whitney–Wilcoxon U test was used to test for significant differences between vegetated and unvegetated plots for each sampling date. * = $p<0.05$; ** = $p<0.01$; *** = $p<0.001$

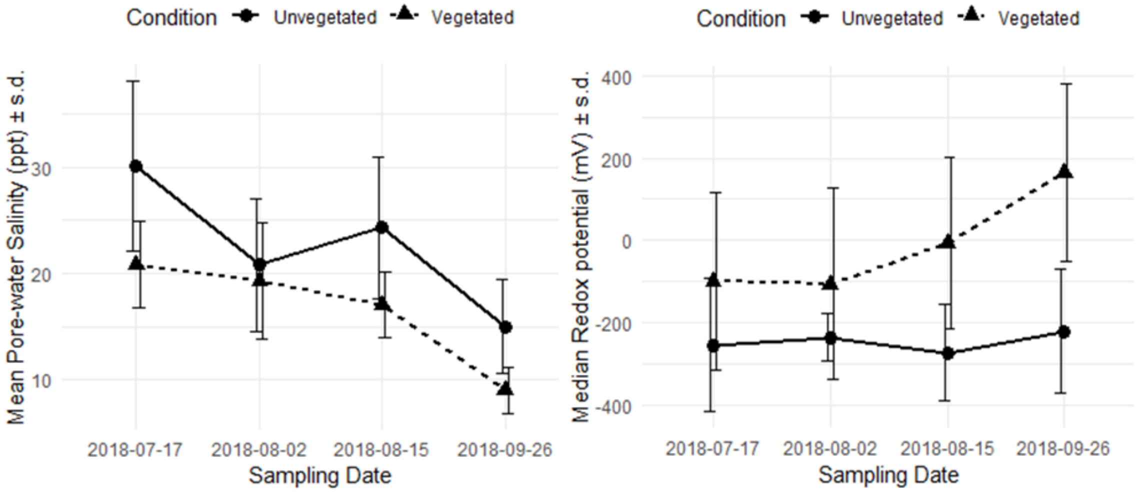

Figure 5. **Mean pore-water salinity and median redox potential in unvegetated (solid line; n = 19) and vegetated plots (dashed line; n = 19) from July to September 2018 at MacKay Creek Estuary, BC.**

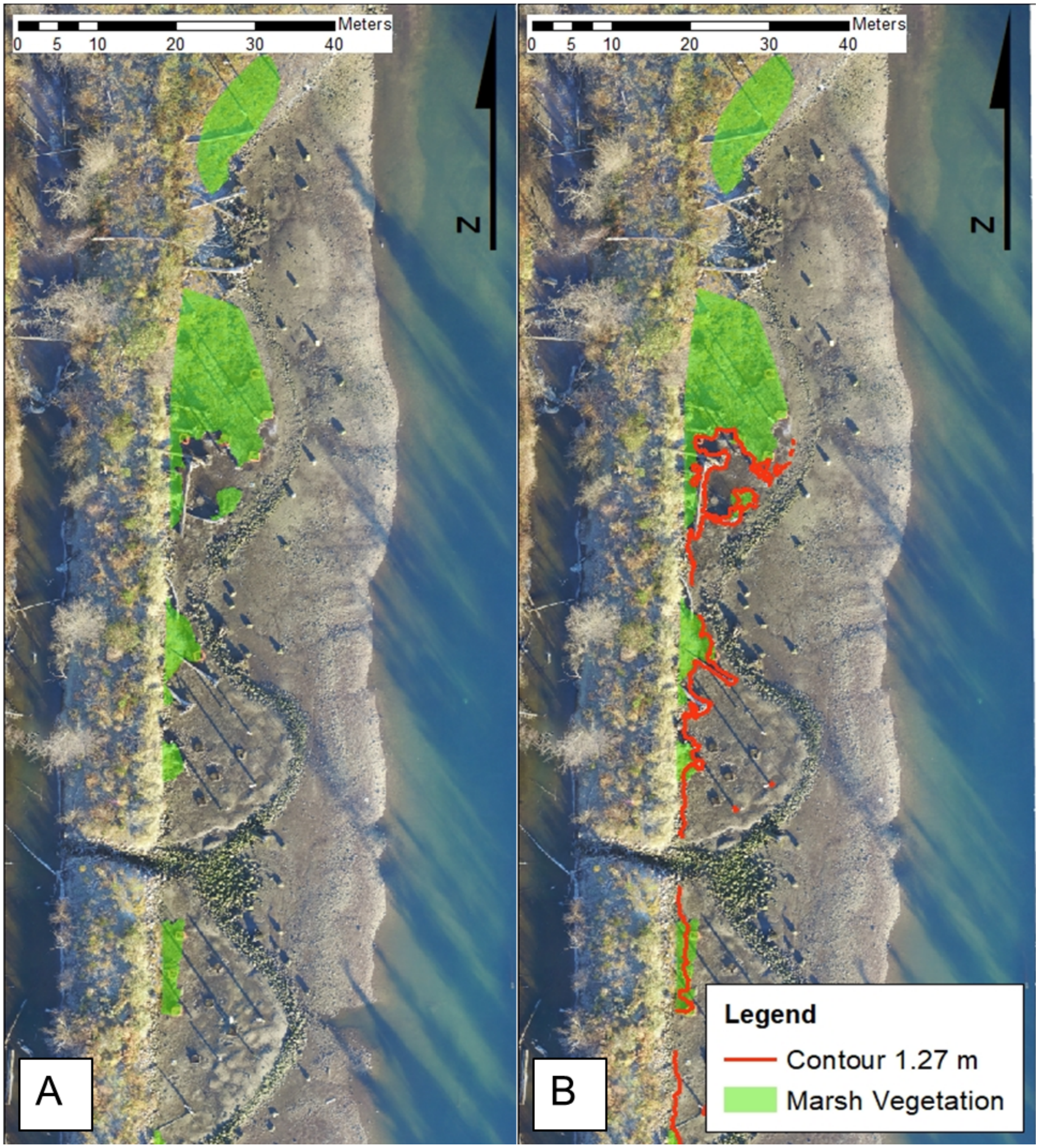

Figure 6. **Maps of MacKay Creek Estuary, BC (2018) depicting (A) Vegetated extent within the terraces (B) Vegetated extent with 1.27 MAMSL contour line (shown in red). Contour lines created using DEMs from 2018 District of North Vancouver LiDAR data georeferenced to CGVD2013.**

Planted Species Distributions by Soil Parameters

Throughout the thirty-eight plots that were surveyed, no planted species were found in plots that had pore-water salinities above 23 ppt, elevations below 1.13 MAMSL, or soil redox potentials below -268.2 mV. Although soil texture analysis was not conducted, the unvegetated substrate was generally fine-textured and poorly drained (D. Fiddler 2018, personal observation).

Bolboschoenus maritimus was only found in one plot at MacKay Creek Estuary (Fig. 7). Mean soil pore-water salinity was 16 ppt, which was slightly more saline than the mean salinity threshold of 14 ppt (Table 6). Mean elevation and median soil redox potential were 1.38 MAMSL and -113.3 mV respectively, which were consistent with this species low marsh designation (Table 6).

Carex lyngbyei was found in seven plots with pore-water salinities ranging from 16 ppt to 23 ppt. All of the plots were above the mean salinity threshold of 15 ppt (Table 6), with three of the plots exceeding the 20 ppt threshold described by Hutchinson (1982). *Carex lyngbyei* survived in plots with that had high elevations, a trait it shared with *Potentilla anserina.* Elevation ranged between 1.49 MAMSL to 1.81 MAMSL, and soil redox potentials were between +38.6 to +301.2 mV. Both of these observations were consistent with the literature which describe this species occupies the mid to high marsh, on well-drained, fine soils (Table 6).

Distichlis spicata was found in nineteen of the plots, making it the most frequently found of the species planted in 2014. It occupied plots with the largest soil pore-water salinity range (10 ppt to 23 ppt), elevation range (1.22 MAMSL to 1.81 MAMSL), and redox potential range (-256.1 mV to +301.7 mV), compared with any other planted species. *Distichlis spicata* was also found in several plots that had redox potentials near the median value for unvegetated sites (Fig. 7). Results were also consistent with literature observations, which indicated *Distichlis spicata*, had a high salinity threshold (27 ppt) and colonized sites with variable elevations, soil drainages and textures (Table 6).

Potentilla anserina was the second most widely distributed species, occupying 10 plots with a wide range of pore-water salinities, redox potentials and elevations. It was found in plots with mean pore-water salinities between 14 ppt to 23 ppt, 90% of which were higher than the mean salinity threshold of 15 ppt (Table 6). This species occupied higher elevation sites (1.37 MAMSL to 1.81 MAMSL) with *Carex lyngbyei* and had the same redox potential range as *Distichlis spicata* (-256.1 mV to +301.7 mV), though only one plot had soils that were near the median redox potential for unvegetated sites. Congruent with observations by Hutchinson (1982) and Belleveu (2012), *Potentilla anserina* mostly occupied high elevation sites with well-drained soils but was more varied in elevation than *Carex lyngbyei* or *Triglochin martima* (Table 6; Fig. 7).

Triglochin martima was only found in two plots with soil pore-water salinities of 17 ppt and 21 ppt. One plot had salinities greater than the mean salinity threshold of 18 ppt (Table 6). *Triglochin martima* was found in the lowest elevation (1.12 MAMSL) plot with the lowest mean redox potential (-268.2 mV) compared with other species planted in 2014. Similar to literature observations, *Triglochin martima* occupied the low to mid marsh but only one plot appeared to be on well-drained soils (approximated by the measured redox potential; Table 6; Fig. 6).

Sarcocornia pacifica and *Schoenoplectus pungens* were both absent from the vegetation survey. Mean pore-water salinity in vegetated sites was 17.1 ppt, which was slightly above the *Schoenoplectus pungens* salinity tolerance found in literature (16 ppt) but fell well below the salinity tolerance of *Sarcocornia pacifica* (29 ppt; Table 6). Similar to *Triglochin maritima* and *Bolboschoenus maritimus*, both *Sarcocornia pacifica* and *Schoenoplectus pungens* are low marsh species that prefer well-drained soils. However, *Sarcocornia pacifica* typically colonizes sandier substrates than *Schoenoplectus pungens* (Table 6).

Table 6. Summary table of soil preferences for the seven salt marsh species planted in 2014 during the restoration of MacKay Creek Estuary, BC

Species	Salinity Tolerance	Elevation Preference	Soil Drainage and Substrate Type
Bolboschoenus martimus	<14 ppt[c]	Low to mid[b]	Poorly-drained, fine[b]
Carex lyngbei	<15 ppt[cf]	Mid to High[ab]	Well-drained, fine[bd]
Distichlis spicata	<27 ppt[cf]	Low to High[a]	Poorly-drained to well drained, fine[b]
Potentilla anserina	<15 ppt[cf]	High[bd]	Well-drained, sand[e]
Sarcocornia pacifica	<29 ppt[cf]	Low[a]	Well-drained, sand[e]
Schoenoplectus pungens	<16 ppt[c]	Low[b]	Well-drained, silty sand[b]
Triglochin maritima	<18 ppt[cf]	Low to Mid[ab]	Well-drained, fine[b]

Note. Species preference data compiled from Weinnmann et al. ([a]1984), Hutchinson ([b]1982, [c]1988), Pojar et al. ([d]2004), MacKenzie and Moran ([e]2004), and Belleveau ([f]2012). Salinity tolerance represents the mean thresholds described by associated authors.

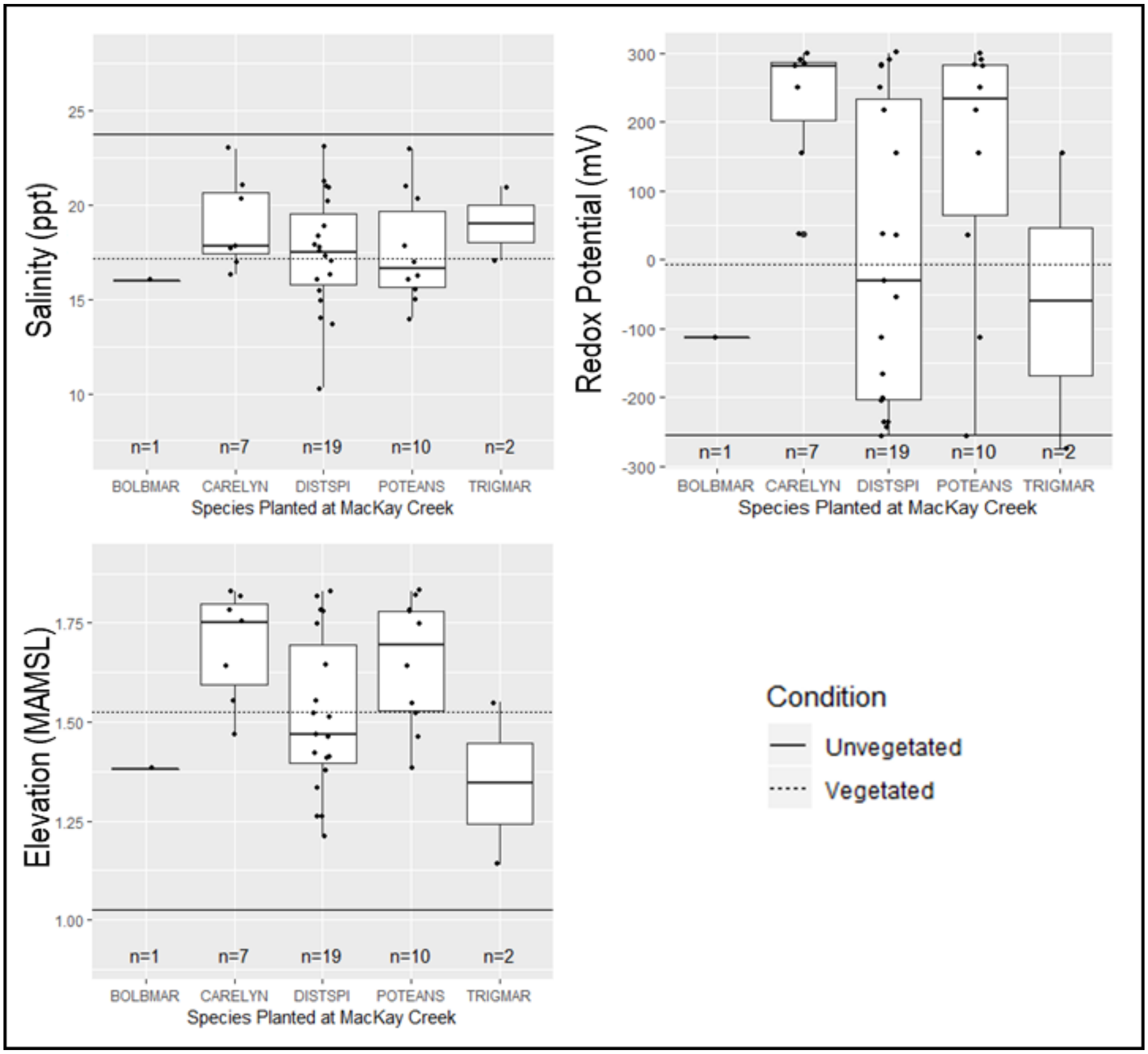

Figure 7. Planted species distributions and abundance relative to plot (n = 38) soil pore-water salinity, redox potential, and elevation at MacKay Creek Estuary, BC. Horizontal reference lines illustrate the 2018 July to September mean pore-water salinity, mean elevation, and median redox potential of vegetated (dashed line; n = 19) and unvegetated (solid line; n = 19) plots.

Vegetation Survey

Sixteen plant species were found within the terraces (Table 7). Five of the seven species planted in 2014 (*Distichlis spicata, Potentilla anserina*, *Carex lyngbyei, Triglochin maritima,* and *Bolboschoenus maritimus*) were present in the 2018 vegetation survey.

Schoenoplectus pungens was the only planted species not found in any of the 2018 plots. *Sarcocornia pacifica,* was found in one herbivory plot but was absent from vegetation survey plots. Nine non-planted species colonized the site between 2014 and 2018. Three of the non-planted species found at MacKay Creek were classified as exotic (*Atriplex* patula and *Juncus gerardii*) or invasive (*Lythrum salicaria*) and three plants belonging to the *Cirsium, Juncus,* and *Puccinellia* genera were not identified down to the species level (Table 7). Of the non-planted species, *Eleocharis parvula, Juncus gerardii,* and *Juncus Balticus* occurred the most frequently and had the highest relative cover. For the planted species, *Distichlis spicata* and *Potentilla anserina* were the most successful in terms of frequency and relative cover.

Differences in species cover and relative cover were observed for all planted species between 2014 and 2018. Species cover was dramatically reduced for each of the seven plant species used in the 2014 restoration planting plan. *Distichlis spicata* and *Potentilla anserina* had the least dramatic reductions in species cover, occupying 34% and 38% of their former 2014 extent. *Carex lyngbyei* species cover was estimated to be 3% of its 2014 cover, while *Bolboschoenus maritimus,* and *Triglochin maritima*, were found to be less than 1% each (Table 7).

Shifts in planted species predominance (as assessed through estimates of relative abundance) were also observed between 2014 and 2018. *Potentilla anserina* nearly doubled in relative abundance (12.7% to 22.8%) and *Distichlis spicata* increased by over one-third (22.6% to 36%). *Carex lyngbyei, Sarcocornia pacifica, Bolboschoenus maritimus,* and *Triglochin maritima, Schoenoplectus pungens* all dramatically decreased in predominance with relative abundance values of less than 2.0% each in 2018.

Table 7. Changes in estuarine marsh plant species cover and relative cover between 2014 and 2018 (n = 38) at MacKay Creek Estuary, BC.

	Frequency	Species Cover (m^2)		Relative Cover (%)	
Species	**2018**	**2014**	**2018**	**2014**	**2018**
Distichlis spicata+	50	428.0	147.0	22.6	36.5
Eleocharis parvula	39		92.7		23.0
Potentilla anserina+	26	240.0	91.8	12.7	22.8
*Juncus gerardii**	29		38.3		9.5
Juncus balticus	24		24.0		6.0
Carex lyngbyei+	18	259.0	7.8	13.7	1.9
Puccinellia sp.	8		0.3		0.1
Spergularia canadensis	13		0.3		0.1
Triglochin maritima+	5	202.0	0.2	10.6	0.1
*Atriplex patula**	8		0.1		<0.05
Bolboschoenus maritimus+	3	144.0	0.1	7.6	<0.05
Cirsium sp.	3		0.1		<0.05
Juncus sp.	3		0.1		<0.05
*Lythrum salicaria**	3		0.1		<0.05
Plantago Major	3		0.1		<0.05
Sarcocornia pacifica+	0	440.0	0	23.2	0
Schoenoplectus pungens+	0	180.0	0	9.5	0

Note. + Indicate species planted at MacKay Creek in 2014; * indicates an invasive or exotic species. Species cover = the total estimated land cover occupied by an individual species. Frequency equals the number of times the plant species was found within a plot divided by the total number of vegetated and unvegetated plots (n = 38) and expressed as a percentage.

Herbivory Experiment

Morning goose counts conducted from January 16-23, 2019, found that approximately 350 Canada geese regularly use the site for grazing, roosting and staging. Geese regularly occupied the estuary at night and swam out into Burrard Inlet following sunrise.

Analysis of soil treatment parameters between the two treatments (fenced and unfenced) found there were no significant differences between elevation, salinity and redox potential in fenced and unfenced plots (unpaired *T-test*, $p > 0.05$; Whitney–Wilcoxon U test, $p > 0.05$; Table 8).

Table 8. Summary of 2018 July to September mean elevation, mean pore-water salinity, and median redox potential for fenced (n = 9) and unfenced (n = 9) plots at MacKay Creek Estuary, BC.

Factor	Condition	Mean ± s.d.	p-value
Elevation (MAMSL)	Fenced	1.28 ± 0.11	0.8591
	Unfenced	1.29 ± 0.12	
Pore-Water Salinity (ppt)	Fenced	18.9 ± 3.9	0.2232
	Unfenced	22.5 ± 7.6	
Redox Potential (mV)	Fenced	-234.1 ± 771.4	0.6665
	Unfenced	-200.6 ± 143.7	

Note. Unpaired T-tests were used to test differences in elevation and soil pore-water salinity. The non-parametric equivalent (Whitney–Wilcoxon U test) was used to test differences in redox potential. Mean and median values calculated over the July to September sampling period.

At the beginning of the sampling period, unfenced plots had a higher mean live shoot frequency than fenced plots. Mean live shoot frequency increased from 54% in June to 81% in September (paired *T-test*, $p < 0.001$; Fig. 8). Unfenced plots decreased in mean live shoot frequency from 73% to 37% during the same period (paired *T-test*, $p < 0.05$; Fig. 8). By end of the sampling period in September, fenced plots had a significantly higher mean live shoot frequency than unfenced plots (unpaired *T-test*, $p < 0.001$; Fig. 8).

Mean plot percent cover also changed as a result of the fencing treatment. Fenced plots increased in cover from 15% to 68% between the June to September sampling period (paired *T-test*, $p < 0.001$; Fig. 8). Percent cover in unfenced treatment fluctuated between 8% and 25% throughout the entire sampling period but by September percent cover was not significantly different from June (paired *T-test*, *p* > 0.05; Fig. 8). At the end of the experiment, percent cover in the fenced treatment was significantly higher than the unfenced treatment (paired *T-test*, *p* > 0.001; Fig. 8).

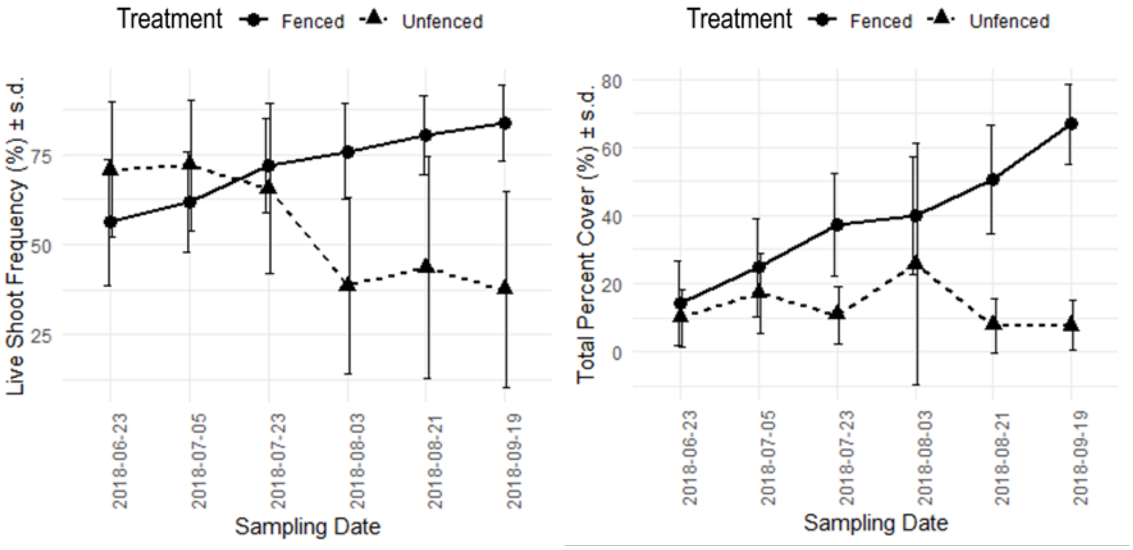

Figure 8. **June to September 2018 live shoot frequency (left) and percent cover (right) for fenced (solid line; n = 9) and unfenced (dashed line; n = 9) plots at MacKay Creek Estuary, BC.**

Comparisons of individual species cover found that *Distichlis spicata* was the only species to significantly change percent cover over the July to September sampling period (paired *T-test, p* < 0.001; Table 9; Fig. 9). *Distichlis spicata* percent cover increased in fenced plots by 23.8% whereas unfenced plots experienced a 4.5% decline (paired *T-test, p* < 0.001). Other species such as *Juncus gerardii*, and *Spergularia canadensis* had marginal increases in mean species cover in fenced plots but was not considered significant.

Table 9. **Changes in estuarine marsh plant species mean cover values for fenced (n = 9) and unfenced (n = 9) plots between July and September, 2018 at MacKay Creek Estuary, BC.**

		Mean Percent Cover (%)		Mean Difference
Species	**Condition**	**July**	**September**	**July-September (%)**
Atriplex patula	Fenced	0.22	0.33	0.11
	Unfenced	0.11	0.11	0
Bolboschoenus maritimus	Fenced	0.44	0.56	0.11
	Unfenced	0	0	0
Carex lyngbyei	Fenced	0.56	1.22	0.67
	Unfenced	0.67	0.22	-0.44
Distichlis spicata	**Fenced**	**30.67**	**54.44**	**23.78*****
	Unfenced	**7.78**	**3.22**	**-4.50***
Juncus balticus	Fenced	0.00	0.00	0
	Unfenced	0.11	0.00	-0.11
Juncus gerardii	Fenced	3.56	5.22	1.67
	Unfenced	0.78	0.56	-0.22
Sarcocornia pacifica	Fenced	0.11	0.11	0
	Unfenced	0.11	0.0	-0.11
Spergularia canadensis	Fenced	2.56	3.56	1
	Unfenced	0.56	0.33	-0.22
Triglochin maritima	Fenced	0.00	0.33	0.33
	Unfenced	0.00	0.11	0.11

Note. Species in bold showed significant change in mean cover values from July to September. Paired T-test * = $p<0.05$; ** = $p<0.01$; *** = $p<0.001$

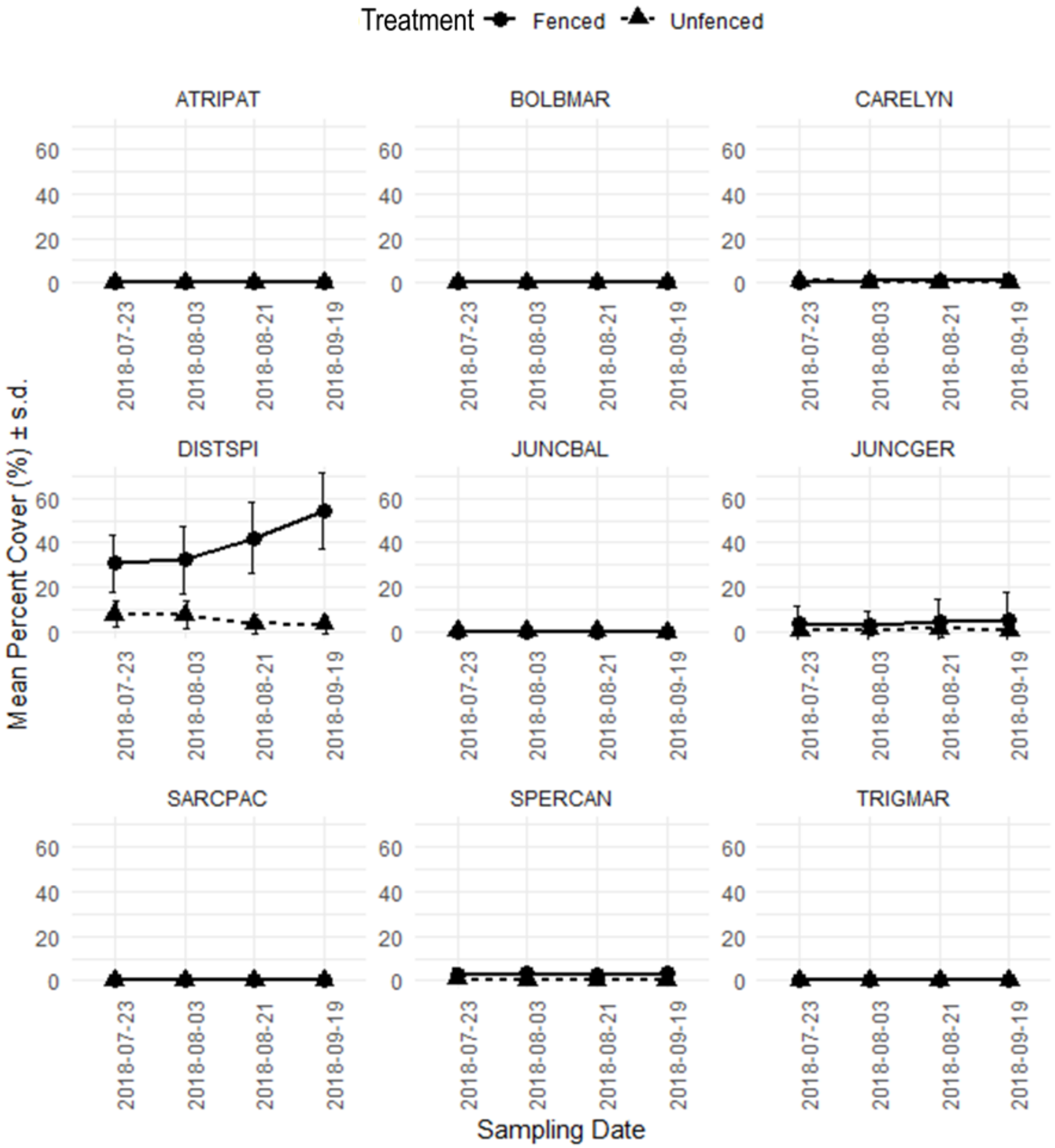

Figure 9. **June to September, 2018 mean percent cover for *Atriplex patula, Bolboschoenus maritimus, Carex lyngbyei, Distichlis spicata, Juncus balticus, Juncus gerardii, Sarcocornia pacifica, Spergularia canadensis,* and *Triglochin maritima* in fenced (n = 9) and unfenced plots (n = 9) at MacKay Creek Estuary, BC.**

Analysis of species height was conducted, however, due to the small sample size, comparisons of height were not possible. Many species were present in small numbers, unequally distributed between and within treatments, or present in one treatment but not the other. The results of this analysis were excluded from this report due to limited data availability.

Discussion

The primary goal of this research was to determine the potential factors inhibiting the establishment of planted salt marsh vegetation at a recently restored site in MacKay Creek Estuary. Three soil parameters were sampled to test for differences between the successfully established salt marsh areas of the site and the areas that failed to establish salt marsh vegetation. I found differences in soil pore-water salinity, redox potential, and elevation between vegetated and unvegetated sites. Furthermore, the exclusion of Canada geese from grazing was associated with increases in species cover and colonization. Difficulties arose when attempting to disentangle parameters that determine the limiting factors to salt marsh plant colonization. High correlation between soil parameters precluded the use of multiple regression models to determine which factors had stronger relationships to the presence of salt marsh vegetation. Despite the inability to use regression to correlate soil parameters with the presence or absence of salt marsh vegetation, it is likely a combination of several factors with potential compounding effects that are preventing the progression of salt marsh vegetation.

Elevation

Soil characteristics in coastal salt marshes often have biogeochemical gradients that are linked to elevation (Hutchinson, 1982; Pennings and Callaway, 1992). Elevation directly impacts the frequency and degree to which a site is exposed to tidal submergence. Tidal inundation is known to influence edaphic characteristics such as salinity, pH, substrate flushing, and nutrient concentration (Craft, 2016). As elevation mediates tidal inundation, it is often used as a proximate parameters in salt marsh ecology. This is exemplified by the frequent categorization of salt marshes into low, medium, and high elevation zones which are observed to form distinct and predictable plant communities perpendicular to the elevation gradient (Weinnman et al., 1984; Pennings and Callaway, 1992; MacKenzie and Moran, 2004; Brooks et al, 2014).

The factors that influence spatial bounds of salt marshes are driven by intraspecific competition in the less severe high marsh, and the ability to withstand harsh environmental stressors in the low marsh (Pennings and Callaway, 1992). If the tidal inundation period controls the lower limit of salt marsh species, one would expect to find a line of equal elevation, past which no salt marsh species would be able to establish. This is what was observed at MacKay Creek Estuary when a contour line of 1.27 MAMSL was superimposed on a map of existing vegetation cover. The contour line approximately followed the lower edge of the vegetated areas and only deviated where estimations of elevation were altered by the presence of large woody debris. The map indicates that elevation is highly correlated with the presence/absence of vegetation, however, this does not necessarily indicate causation.

The assumption that low substrate elevation is the primary limiting factor influencing planting success is likely an oversimplification. If tidal submergence is the sole parameter limiting species colonization, it stands to reason that plant community development at MacKay Creek estuary should replicate natural sites that occupy similar elevations. Westham Island and Mud Bay both have salt marshes that terminate below the 1.27 MAMSL elevation found at MacKay Creek Estuary, although Sturgeon Bank appears to be similar (Table 10). Research by Hutchinson (1982) found that salt marsh communities within the Fraser River Delta were primarily associated with elevation but also had interactions with salinity, sediment texture, and substrate water content. Given the range of elevations found at nearby marshes, tidal inundation appears to be highly correlated with the presence/absence of salt marsh species but is likely acting in concert with one or more other parameters.

Table 10. Salt marshes in close proximity to MacKay Creek Estuary, BC with approximate low marsh edge elevations measured between 2017 and 2018.

Salt Marsh	Location	Elevation of Low Marsh Edge (MAMSL)
Westam Island Foreshore Marsh[a]	Delta	-0.6 m
Sturgeon Bank Marsh[a]	Richmond	1.1 m
Mud Bay[b]	Surrey	0 - 0.5 m
MacKay Creek Estuary	North Vancouver	1.27 m

Note. Elevation data gathered from Balke ([a]2018) and City of Surrey ([b]2018).

Substrate Drainage

Although no texture analysis was done on the substrate of MacKay Creek terraces, the sediment in unvegetated areas appeared to be primarily composed of fine textured, poorly-drained soils with limited organic content (D. Fiddler 2018, personal observation). Groundwater upwelling seemed to be occurring in several locations throughout the marsh, leading to the formation of rivulets that flowed across the substrate throughout the tidal cycle. Additionally, the presence of Iron-oxidizing bacteria (an indicator of de-oxygenated water) was noted in many of the locations where groundwater upwelling appeared to be occurring. Many of the rivulets originated from high elevation sites where a riprap-enforced peninsula separates the salt marsh from a small stream to the west that was designed to be an off-channel habitat. The small stream is fed by MacKay Creek at the north and empties into the estuary through two spillways that bisect the salt marsh terraces. The off-channel habitat is basin-like in design, storing a significant amount of water despite its low flow. Water from the off-channel habitat may be causing ground water upwelling within the salt marsh terraces and contributing to permanently saturated soils with low redox potentials.

Most species require at least partial oxygenation of the roots to support respiration and power metabolic processes (Mendelssohn et al., 1981; Pezeshki, 2001; Li et al., 2005). Highly reduced soils affect species differently, the degree to which is dependent on individual adaptations to cope with anaerobic conditions. *Spartina patens* for example has well-developed aerenchyma (air space) tissues to convey oxygen to the rhizosphere when soils are moderately reduced (Mendelssohn et al., 1981; Howes et al., 1981). The ability of certain species to locally oxygenate soils may be responsible for the higher redox potentials observed in vegetated sites at MacKay Creek Estuary. Interspecific differences in plant soil-oxygenation may also be responsible for the high variance observed within vegetated plots compared with unvegetated plots.

Even species with morphological adaptations that facilitate oxygen transfer are susceptible to highly reduced conditions. Research on *Spartina patens* found that redox potentials of -200 mV were enough to deplete oxygen in the roots and trigger anaerobic respiration (Mendelssohn et al., 1981). Redox potentials between -100 mV to -300 mV can cause iron toxicity in plants where increases in the concentration of reduced iron oxide ($2e^- + 6H^+ + Fe_2O_3 \rightarrow 2Fe^{2+} + 3H_2O$) trigger cell death or, more commonly, reduce nutrient uptake (Fageria et al., 2008). The median redox potential of unvegetated plots at MacKay Creek Estuary was -254.9 mV, indicating these soils were highly reduced. Furthermore, median redox potential was likely underestimated as measurements were made during low tide when the salt marsh substrate was temporarily exposed to oxygen.

Five of the species planted at MacKay Creek (*Carex lyngbei*, *Potentilla anserina, Sarcocornia pacifica, Schoenoplectus pungens,* and *Triglochin maritima*) prefer well-drained soils. *Triglochin maritima* has been found to tolerate highly reduced conditions in laboratory experiments (Skelly, 2015), and was the only planted species found in a plot with redox potential of -268 mV. Several plots containing *Distichlis spicata, Potentilla anserina* and *Bolboschoenus maritima* were also found in highly reduced soils, indicating that redox potential alone is likely not the limiting factor. Research by Ungar (1991) and Colmer and Flowers (2008) found the ability of perennial halophytes to tolerate elevated soil salinities depends on the maintenance of ion transport processes that can be hampered in anaerobic conditions. This hypothesis is exemplified by Mahall and Park (1976) and Pennings (1992) who explain that *Sarcocornia pacifica* is able to oxygenate its roots, yet remains sensitive to flooding.

Soil Pore-Water Salinity

In terrestrial environments, the relative salt responses of various species are strongly dependent on the soil type and composition of minerals (Shannon, 1997). In salt marshes, the composition of ions in seawater is relatively uniform within a site, consisting of chloride (Cl^-), sodium (Na^+), sulfate (SO_4^{2-}), magnesium (Mg^{2+}), calcium (Ca^{2+}), potassium (K^+), and bicarbonate (HCO^{3-}) (Morcos, 1970). As the ionic composition of seawater is relatively uniform, plant zonation is more so a product of salinity gradients rather than specific ionic composition of minerals.

Within salt marshes, small-scale salinity gradients can result in distinct shifts in species compositions. An example of this phenomenon is observed in salt pans, where sea water evapoconcentrates in marsh depressions and results in salinities that far exceed the surrounding soil substrates (Craft, 2016). These areas often have sparse vegetation cover consisting of halophytic *Salicornia* spp., *Batis* spp., or *Suaeda* spp (Craft, 2016). Salinity gradients are thus an important consideration for estuarine planting plans but planting success can be confounded by landscape-mediated shifts in gradients.

Research into the literature values of salinity tolerances found that only three species (*Sarcocornia pacifica, Distichlis spicata and Triglochin maritima*) had the necessary adaptations to withstand the mean pore-water salinities observed in unvegetated portions at MacKay Creek Estuary (Fig. 10). *Distichlis spicata* was by far the most abundant species found at MacKay Creek Estuary and occupied the widest range of pore-water salinities and redox potential values. *Triglochin maritima* was present but in very low numbers. *Sarcocornia pacifica* was not found at all during the 2018 vegetation survey but interactions with poorly drained substrates may have precluded the successful colonization of this species (Mahall and Park, 1976; Pennings, 1992).

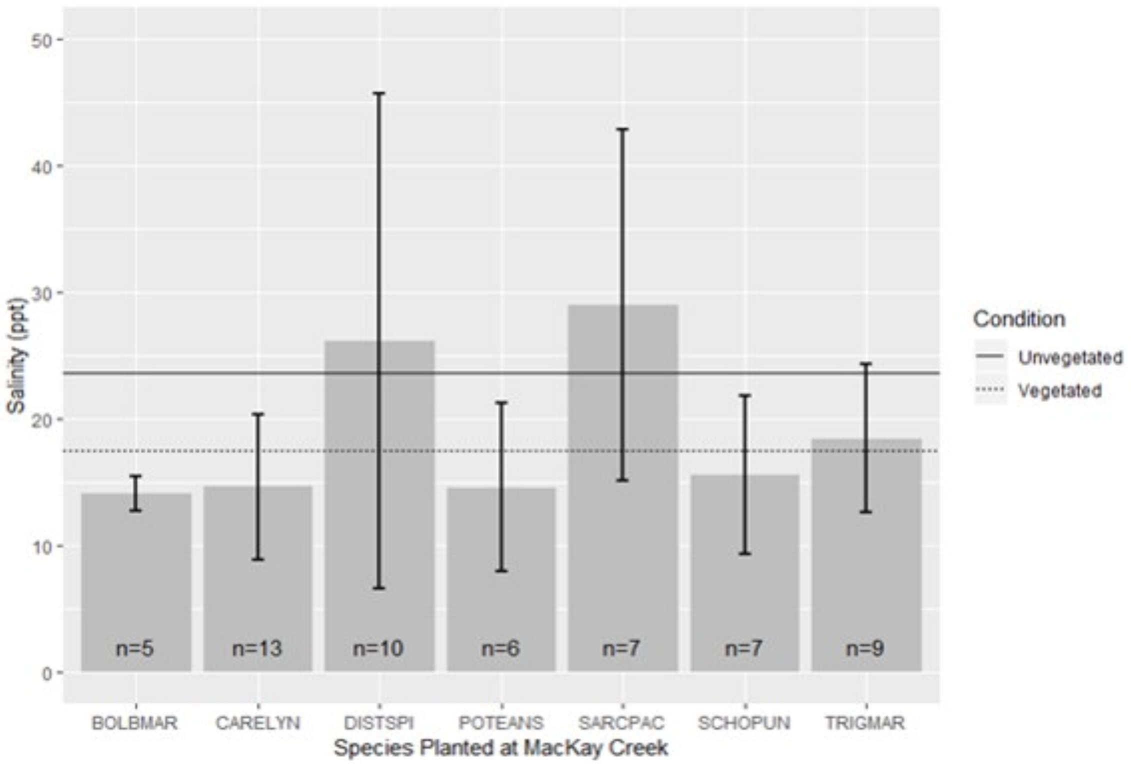

Figure 10. **Salinity tolerances of seven salt mash species planted at MacKay Creek, BC in 2014. Mean salinity tolerances with corresponding sample sizes are based on literature by Hutchinson (1988) and Belleveau (2012). Horizontal reference lines illustrate the mean salinities of vegetated (dashed) and unvegetated (solid) plots.**

One of the more perplexing observations was the deficiency of low marsh species within the salt marsh terraces. All of the four low marsh species (*Bolboschoenus maritimus, Schoenoplectus pungens, Sarcocornia pacifica* and *Triglochin maritima*) planted in 2014 were either absent or occupied less than 0.5% of the total vegetated area. Similar observations have been made at the salt marshes at Sturgeon Bank, Brunswick Point, and Westham Island in Vancouver, BC. (Boyd et al., 2012; Balke, 2018). Boyd et al. (2012) found that 100% (17.4 ha) of the *Schoenoplectus pungens* and *Bolboschoenus maritimus* low marsh community had died and transitioned into mud flat at Sturgeon Bank. The lack of low marsh species at MacKay Creek provides further evidence that elevation is not the sole factor limiting marsh colonization. If the terraces were constructed too low in the tidal prism, one would expect the vegetated areas to be primarily dominated by low marsh species, rather than a mixture of mid to high marsh species. One possible explanation may be herbivory by Canada geese.

Herbivory

There is some evidence to suggest that edaphic characteristics in salt marshes may influence palatability and thus grazing behaviour in herbivores. Tall form plants, which are typically located at lower elevations, may be more palatable to herbivores than short form plants (Pennings, 2004). Studies examining intraspecific differences in biochemical composition found that low marsh species differ from high marsh species in two important ways. Low marsh species typically have higher concentrations of leaf nitrogen and lower concentrations of phenolic compounds (used in chemical defences) compared with high marsh species (Pennings, 2004). Increased concentrations of phenolic compounds are negatively correlated with grazing selection by Canada geese (Buchsbaum et al., 1984), whereas high leaf nitrogen content has been associated with positive preferential selection (Cadieux et al., 2005). Pennings (2004) therefore makes the case that variation in salt marsh plant palatability cause herbivores to preferentially select low marsh species over high marsh species.

According to Dawe et al. (2011), many Canada geese that occupy the estuaries of British Columbia during winter months are a hybrid subspecies deliberately released in the 1920’s, 1930’s, 1970’s and 1990’s. Prior to the late 1970’s Canada geese were a transient species and rarely seen throughout Vancouver Island except during spring and fall migration. This new hybrid subspecies does not migrate, allowing it to dedicate a significant amount of time and resources to feeding, breeding, and rearing. According to the U.S. Fish and Wildlife Service, non-migratory geese have a younger breeding age, have higher incubation constancy, and produce larger clutch sizes, compared to migratory Canada geese (U.S. Fish and Wildlife Service, 2005). Non-migratory Canada geese are thus prolific breeders with an altered life strategy that results in a concentrated impact to their reduced habitat range. An example of the detrimental effect non-migratory geese can have on estuarine vegetation is documented by Dawe et al. (2011) in the Little Qualicum Estuary on Vancouver Island.

The non-migratory population at Little Qualicum Estuary has been growing at a rate of 8.5% each year, increasing by over 1 500 individuals between 1986 and 2006. Dawe et al. (2011) noted significant shifts in frequency and/or mean cover values for up to twenty-four plant species (Dawe et al., 2015). Particularly relevant were a recorded decrease in *Carex lyngbyei* and *Triglochin maritima* abundance and increase in *Distichlis spicata* and *Juncus balticus* abundance (Dawe et al., 2011; Dawe et al., 2015; Table 11). Additionally, frequency and abundance dramatically increased for two ruderal species, *Atriplex patula* and *Spergularia canadensis*, which were posited to be colonizing the areas of salt marsh disturbed by grazing. Vegetation communities also appeared to change from salt-intolerant species like broadleaf cattail (*Typha latifolia*) to more salt-tolerant species like *Distichlis spicata.* Grazing by lesser snow geese (*Chen caerulescens* L.) has been associated with similar community shifts. Jefferies and Rockwell (2002), noted a considerable loss of intertidal salt marsh graminoid vegetation as a result of grazing and grubbing by lesser snow geese. Plant cover was reduced to <2% over six years and resulted in hypersaline bare sediments that further prevented the establishment of marsh species (Jefferies and Rockwell, 2002).

Table 11. Changes in estuarine marsh plant species frequency of occurrence and mean cover values between 1978 and 2005 (n = 270) on the Little Qualicum River estuary. Table adapted from Dawe et al., (2011)

	Frequency		**% Change in p^2**	**Mean Cover**		**% Change in p^2**
Species	**1978**	**2005**	**frequency**	**1978**	**2005**	**mean cover**
Atriplex patula	2	34	**1464 *****	<0.05	3	**3142*****
Carex lyngbyei+	72	59	**-17***	25	30	**20**
Distichlis spicata+	0	3	-*	0	13	-*
Juncus balticus	67	64	-3	22	18	**-18****
Plantago major	3	8	136	<0.05	7	**7186****
Spergularia canadensis	0	26	-***	0	3	-***
Triglochin maritima+	49	54	10	9	9	-1

Note. Table modified to contain only those species found at MacKay Creek Estuary. Species in bold showed significant change in frequency or mean cover between years. + Indicates species planted at MacKay Creek in 2014; * = p<0.05; ** = p<0.01; *** = p<0.001

Table 12. **Changes in estuarine marsh plant species frequency of occurrence and mean cover values between 1994 and 2012 (n = 151) on the Campbell River Estuary. Table adapted from Dawe et al., (2015)**

	Frequency		% Change in p^2	Mean Cover		% Change in p^2
Species	1994	2012	frequency	1994	2012	mean cover
Carex lyngbyei+	91	52	**-43*****	38	1	**-99*****
Juncus balticus	70	83	**19****	36	67	**85*****
Triglochin maritima+	20	5	**-75*****	1	<0.05	**-100*****

Note. Table modified to contain only those species found at MacKay Creek Estuary. Species in bold showed significant change in frequency or mean cover between years. + Indicates species planted at MacKay Creek in 2014; * = p<0.05; ** = p<0.01; *** = p<0.001

Similar circumstance may be causing a community shift at MacKay Creek Estuary. Morning geese counts revealed that approximately 350 individuals routinely used the site for roosting. This estimate is likely conservative as surveys were done in the early morning when geese had already begun leaving the estuary into Burrard Inlet. Evidence of site use by Canada geese were found in the form of grubbing, feathers, feces, footprints, and grazed leaves on plants.

Distichlis spicata and *Juncus balticus* were among the top five predominant species and found in 100% and 48% of the vegetated plots. The two ruderal species (*Atriplex patula* and *Spergularia canadensis*) were similarly widely distributed occurring in 16% and 26% of the vegetated plots respectively. Additionally plots that were excluded from grazing showed significant increases in percent cover and live shoot frequency. Increases to shoot frequency in fenced treatments indicate that that the absence of grazing promotes recolonization of the adjacent mudflat. Although species identification is difficult for small propagules, *Distichlis spicata* and *Spegularia canadensis* appeared to be the primary colonizers of the adjacent mudflat when grazing was excluded (D. Fiddler, 2018, personal observation). This effect was almost certainly an effect of a rapid increase in above-ground and rhizomatous growth, represented by the significant increase in percent cover for *Distichlis spicata* in fenced treatments (Fig. 11). Unfenced treatments showed poor colonization by marsh species (as measured by live shoot frequency) and in some plots, resulted in the exposure of bare sediment with no vegetation cover.

Grazing and grubbing also may be partially responsible for the salinity differences observed between vegetated and unvegetated sites at MacKay Creek. Salt marsh substrate is dark in colour owing to the reduction of sulfur and metals into metal sulfides

(Otero, 2002). Unvegetated marsh substrates may have higher measured salinities as a resulting from increased exposure to sunlight and wind. Evapoconcentration could locally increase salinities in bare sediments, particularly where Canada geese grubbing has created shallow depressions that mimic natural salt pans.

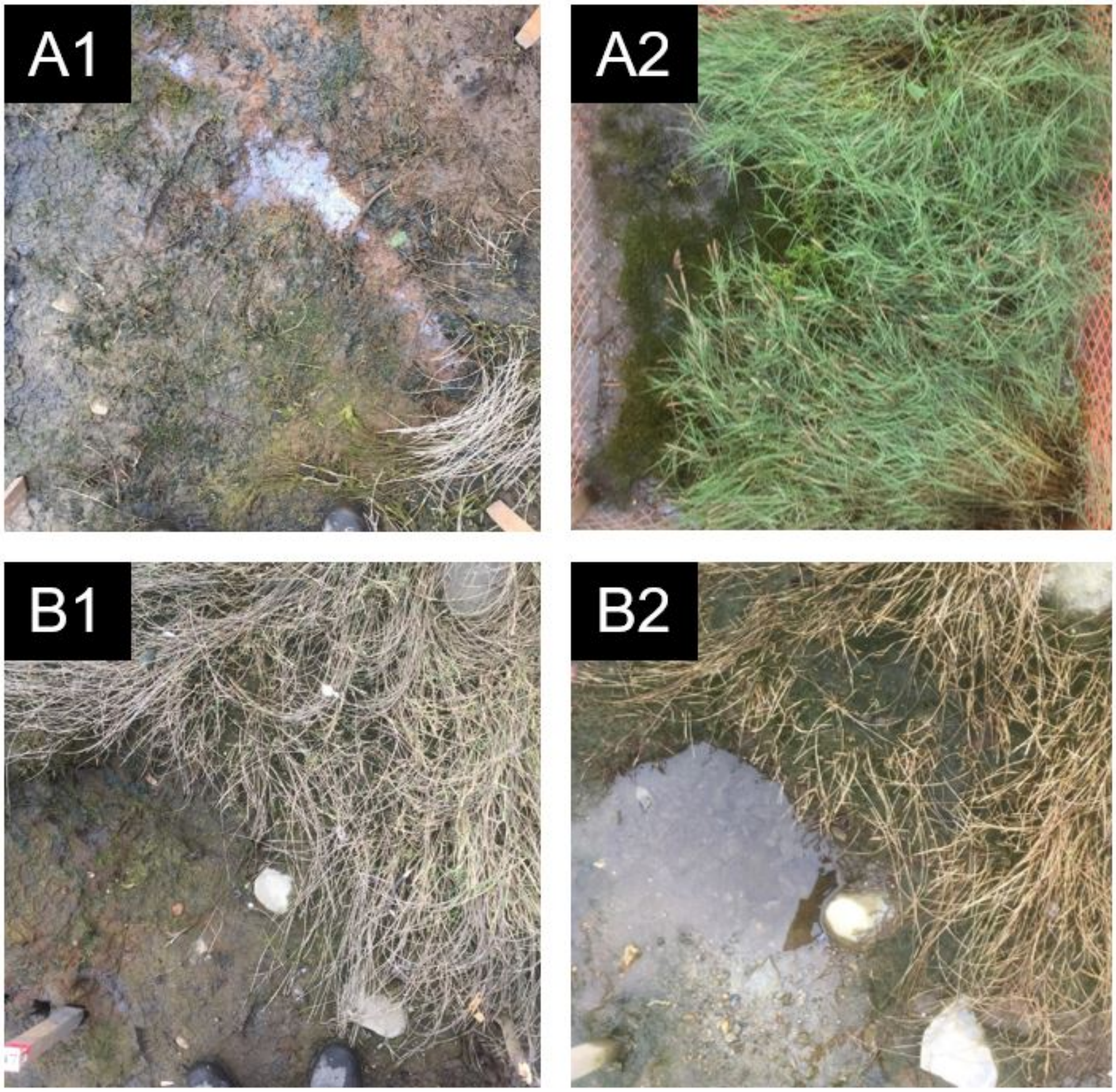

Figure 11. **June to September (2018) comparison of two treatments on vegetation cover and live shoot frequency at MacKay Creek Estuary, BC. (A1) Fenced plot during the June sampling period (A2) The same fenced plot during the September sampling period (B1) Unfenced plot during the June sampling period (B2) The same unfenced plot during the September sampling period.**

Future Research Recommendations

Understanding how environmental parameters impact the spatial distribution of vegetation communities is key to informing the field of Restoration Ecology and guiding future restoration efforts. Although there were significant differences in soil pore-water salinity, redox potential and elevation between vegetated and unvegetated areas within the salt marsh terraces, the exact mechanisms driving the low planting success are not fully understood. An increase in sample size would strengthen the statistical power of this analysis and allow for a more precise data interpretation. Resource limitations also prevented the laboratory analysis of substrate characteristics to determine if other parameters not examined within this study may also be inhibiting colonization. Future research should consider measuring other soil parameters associated with salt marsh growth and colonization such as pH, nutrient concentration, and soil texture.

In salt marshes, pH can exert an influence on vegetation zonation patterns (Rogel, 2000). In 1993, a salt marsh constructed in Port Moody Arm was designed with an elevation range of 1.0 m to 1.3 m (similar to MacKay Creek) but ultimately failed to establish vegetation. Monitoring revealed certain areas had soil with a pH as low as 3.9. Remedial activities included the rototilling of lime and manure to increase soil alkalinity, which ultimately resulted in suitable conditions for replanting (Keskinen, 2014). Future research may want to test differences in pH, between vegetated and unvegetated areas.

Substrate nutrient concentrations were not considered for this study due to research limitation and time. Additionally, there was assumption that the initial marsh substrate used in the 2014 restoration of MacKay Creek was homogenous in composition and therefore would likely not be significantly different in nutrient composition. However, as tidal inundation period is a keystone variable, it may have imparted differences in nutrient concentrations during the four years since construction. Fertilization experiments in intertidal salt marshes have induced variable responses in plant parameters. Some have shown a negative correlation between marsh species root growth and nitrogen additions (Vakueka et al., 1976; Darby 2008), while others had significant increases in above-ground biomass (Broome et al., 1975). One study found that nutrient additions enhanced plant growth, particularly in the low marsh zone where environmental stress was greatest (Langley et al., 2013). Therefore nutrient analysis of substrate is recommended to determine if soil amendments may be necessary to promote plant colonization and growth.

Summary and Conclusions

The individual parameters measured in the salt marsh terraces at MacKay Creek Estuary may each be contributing to the lack of vegetation development. Significant differences in elevation between vegetated areas and bare sediments were observed and a visual analysis of established vegetation cover in relation to an elevation contour line of 1.27 MAMSL illustrated a strong relationship. Other nearby estuarine marshes were found to have a seaward edge lower than or comparable to what was observed at MacKay Creek Estuary, indicating that elevation may not be the sole factor responsible for the lack of planting success. The distinct lack of low marsh vegetation also bolstered this supposition.

Soils within the unvegetated areas were highly reduced and continued to shed water throughout the tidal cycle. Ground water upwelling from the adjacent off-channel habitat may be responsible for this intrusion and the low redox potentials observed within the terraces. Although many marsh species are able to tolerate reduced soils, antagonistic interactions between salinity and saturated soils have been noted for several species. Pore-water salinity was significantly higher in unvegetated sites and comparisons of literature salinity tolerances found that four of the seven species had salinity threshold that fell below what was observed in the unvegetated substrates.

The exclusion of grazing through fencing treatments had a strong positive relationship to mudflat colonization and percent cover, however this mostly benefited one species (*Distichlis spicata*). Grazing has been shown to alter vegetation communities and shifts to more salt tolerant species have been associated with a reduction in plant cover. Grazing may be locally increasing pore-water salinities by increasing the evapoconcentration of exposed marsh substrates, especially when grubbing results in marsh depressions, which may form artificial, salt pans.

The interactions of soil pore-water salinity, redox potential, elevation, and grazing may be greater than the sum of their individual effects, however, an analysis of cumulative dynamics and impacts was outside the scope of this research.

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