Ecological restoration of the Little Qualicum River Estuary: Analysis of short-term sediment deposition

by

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Abstract

Restoration of the Little Qualicum River Estuary has focused on re-establishing the *Carex lyngbyei* channel edge vegetation lost to grubbing by the overabundant resident Canada goose population. Short-term sediment deposition rates were measured using weekly deployments of sediment traps between June and July 2019 to investigate how restoration is facilitating sediment retention to rebuild the marsh platform.

Deposition rates varied between 6.82-107.88 g/m^2/week with traps deployed on the denuded mud flat areas collecting more sediments than inside the older exclosures. It had been expected that the exclosures with a greater density of sedges would retain more sediment. Spatial variation may be attributed to differences in sampling elevations. Restoring *C. lyngbyei* may not increase localized sediment deposition directly but does protect the continued supply of organic input from the seasonal senescence of *C. lyngbyei*. The organic input from aboveground biomass may have a larger contribution to marsh accretion than allochthonous sediments.

Keywords: *Carex lyngbyei*; Estuary; Restoration; Canada Goose; Sediment deposition.

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LQRE	Little Qualicum River Estuary
LQRERCA	Little Qualicum River Estuary Regional Conservation Area
NWA	National Wildlife Area

Carex lyngbyei in the Little Qualicum River Estuary, June 2019

Chapter 1.

Introduction

1.1. Estuary ecosystems.

Estuaries are the meeting place between the rivers and the oceans of the world. Despite covering just 3% of British Columbia's 27,000 km long coastline (Williams & Langer 2003), these environments are universally recognized for their significant ecological, economical and aesthetic values. Estuaries provide stopover habitat for birds and support a rich network of birds, fish, amphibians and insects that make them amongst the most productive ecosystems on the planet (Robb 2014). These marshes provide the reproductive habitat, nursery grounds and protective cover to support commercially significant species including fish, shrimp, oysters and clams (Barbier et al. 2011). They are critical habitat for out-migrating juvenile salmonids, providing refugia from predation and foraging opportunities as the anadromous fish transition on to oceanic life stages and again when they return to freshwater to spawn (Simenstad & Cordell 2000).

Beyond the importance of supporting wildlife, tidal wetlands also provide valued ecosystem services. Marshes protect coastlines from wave and storm surges by stabilizing sediments, raising the intertidal height over time and supporting vegetation that dampens wave energy (Barbier et al. 2011). Salt marshes filter water that passes through the estuary, improving water quality, trapping pollutants, storing flood waters and excess nutrients and depositing suspended sediments (LQRERCA 2010; Barbier et al. 2011; Kirwan et al. 2016). Climate change forecasts predict an increase in the rate and intensity of storms and sea level rise, so estuaries will play a critical role in protecting coastal communities by facilitating sediment accretion, providing the structure to reduce erosion and providing greater resiliency during these peak flows (Barbier et al. 2011).

Estuaries have a high rate of primary productivity compared to other ecosystems (Jesperson & Osher 2007; Barbier et al. 2011). Autochthonous matter from photosynthesis by marsh vegetation and allochthonous matter transported into the estuary from the ocean or rivers are all transformed within the estuary (Jesperson & Osher 2007). Organic matter decay is accelerated by benthic organisms and moved around the estuary with tidal flushing (Jesperson & Osher 2007). Tall grasses provide structure to trap particles and accelerate sediment deposition. As more sediment layers accumulate, the anoxic conditions below the sediment surface inhibit microbial processing and photodegradation maximizing carbon storage (Jesperson & Osher 2007). The growing interest in global carbon trade has intensified interest in 'Blue Carbon' stored in coastal vegetated ecosystems including mangroves, seagrass beds and estuaries (Lavery et al. 2013; Siikamaki et al. 2013).

Despite their high ecological and ecosystem values, estuaries have not received adequate protection from human impacts. Land conversion for agriculture, forestry and residential development have all resulted in habitat loss and fragmentation (Robb 2014). Upstream land use changes can alter the flow of sediment, nutrient and pollutant delivery with harmful effects on the estuarine trophic structure (Robb 2014). BC estuaries cover a fraction of their previous extent; seventy percent of the Fraser River estuary wetlands have been converted for land development and on Vancouver Island, half of the Nanaimo and Cowichan estuary wetlands have been lost (Lievesley et al. 2017). Estuaries are often found at the epicenter of human development and face increasing stressors as coastal populations expand, leaving them vulnerable to invasive and introduced species.

1.2. Canada goose introductions.

Prior to the 1970s, nesting Canada Geese (*Branta canadensis*) were rarely seen on Vancouver Island. A small population of the native Vancouver Canada Goose (*B. c. fulva*) were present but thought to have only nested north of the Great Central Lake (Dawe & Stewart 2010). There is now an abundant population of year-round resident geese that were established through a series of early introductions in the 1930s from privately run game farms and later by government sponsored initiatives in the 1970s to enhance sport hunting opportunities on Vancouver Island.

In 1971, biologists with the BC Fish and Wildlife Branch began transplanting young of the year goslings of the subspecies *B.c. moffitii* to Vancouver Island.

Subsequent transplants of multi-race hybrid geese continued through the 1980s (Dawe 2010). With few predators and ample grazing opportunities the population quickly increased. For example, the Nanaimo River estuary had a maximum of nine Canada Geese between January and April 1973 (Dawe & Stewart 2010). More than 190 geese were counted during the same time period in 1999, and by July of that same year 420 geese were counted in the estuary (Dawe & Stewart 2010). Estimates of the current population of year- round resident Canada Geese along the east coast of Vancouver Island is as high as 12,000 individuals (T. Clermont 2020, Guardians of Mid Island Estuaries Society, Parksville, BC, personal communication).

These resident birds have had a marked negative effect in the region. Mitigation costs associated with crop damage in the Victoria Goose Management Area alone cost upwards of $300,000 per year (Regional Canada Goose Management Strategy 2012). Fecal contamination often creates water quality concerns at public beaches and pools, and geese are a nuisance in parks, fouling sports fields and benches. Of most ecological significance is that the goose population has also been linked to the loss of productive soils and channel edge structure in estuaries along the East coast of Vancouver Island though the over-grazing and grubbing of channel edge vegetation of (Fig 1.1) (Dawe & Stewart 2010; Clermont 2015).

Current management strategies require collaboration across a number of governing bodies and include intensive egg addling programs and nest surveys, scare programs to exclude geese from airports and sports fields and seasonal harvests with First Nations partnerships during the summer molt times to reduce local populations and provide a high-quality food source to the nearby Indigenous community.

Figure 1.1 Photos of the changes to the Little Qualicum River marsh structure between August 1980 and 2005.
Figure adapted from Dawe & Stewart (2010)

1.3. Little Qualicum River Estuary.

The Little Qualicum River is located approximately 40 km northwest of Nanaimo on Vancouver Island and flows Northeast from Cameron Lake to the Strait of Georgia. The river supports Chinook (*Oncorhynchus tshawytscha*), chum (*O. keta*), coho (*O. kisutch*) and pink (*O. gorbuscha*) salmon, as well as Dolly Varden (*Salvelinus malma*), rainbow trout (*O. mykiss*), cutthroat (*O. clarkii*) and brown trout (*Salmo trutta*) (Silvestri 2004; LQRERCA 2010). The Little Qualicum River Estuary (LQRE) is a complex site (Fig 1.2) surrounded by one of the few remaining undeveloped estuarine spits on the east coast of Vancouver Island. The spit forms the Little Qualicum River Estuary Regional Conservation Area (LQRERCA) managed in partnership between Ducks Unlimited and the Regional District of Nanaimo. The outer reaches are managed as part of the Parksville-Qualicum Beach Wildlife Management Area, a recognized Important Bird Area. East of the LQRERCA is the Marshall Stevenson Unit of the Qualicum National Wildlife Area (NWA), donated to Canadian Wildlife Services in 1974 to preserve the estuary and uplands areas. Permission to access the NWA was granted by the Canadian Wildlife Service and is the site of this applied research project.

Figure 1.2 Management areas of the Little Qualicum River Estuary

Source: 2010 Little Qualicum River Estuary Regional Conservation Area management plan (LQRERCA 2010).

The estuary was used from the early 1900s by settlers to raise cattle. From the 1930s to early 1950s the marsh was used as a log storage area and a sawmill operated near the mill pond along the spit (LQRERCA 2010). Following designation as a National Wildlife Area, the roads that used to serve the estuary business were removed to improve tidal marsh connectivity (LQRERCA 2010) and are now closed to the public, with management goals aimed to protect and conserve wildlife and their habitats.

The LQRE has experienced severe degradation in recent years as a result of increased goose pressure from non-native resident Canada geese. Through the 1970s, Canada geese were recorded as rare transients on the estuary. The first nesting pair were not recorded until 1984. The number of nests rapidly increased to the point of counting 45 nests containing 238 eggs in the spring of 2010 (Dawe & Stewart 2010).

Coinciding with the increase in goose presence, there was a marked change to the vegetation community. A 1978 vegetation survey found a lush marsh community dominated by *Carex lyngbyei, Potentilla pacifica, Juncus balticus,* and *Agrostis sp* (Dawe & White 1982). Revisiting the transects in 2005 revealed a significant decline in the frequency and cover of plants known to be a preferred diet of Canada Geese and changes to the channel edge structure (Dawe et al. 2011).

A main concern in this estuary was the significant loss of *Carex lyngbyei*. This sedge is considered a keystone pioneer species in eastern Pacific coast estuaries (Dawe et al. 2011). *C. lyngbyei* grows from short overwintering shoots at an impressive rate of up to 1.88 cm a day from April to June before approaching senescence after seeding in June (Kistritz et al. 1983). At its summer peak, the shoots can be over one metre tall. In a study comparing overwinter decay rates of the native sedge to the introduced exotic purple loosestrife (*Lythrum salicaria*), *C. lyngbyei* had an autumnal decay rate four times slower. These findings suggested it provides a significant detritus input through winter and early spring compared to introduced species (Grout et al. 1997). This organic input in turn supports the detrital invertebrate community that is important to the juvenile salmonid food web and for migratory birds.

Goose grubbing has removed *C. lyngbyei* rhizomes and exposed the organic soils along the lower marsh dendritic channels to erosion (Dawe et al. 2011). This action has caused the infilling of some channels that were formerly 30 cm deep, and a shift to ruderal species like *Spergularia canadensis* (Dawe et al. 2011). Channel infilling and the associated loss of the overhanging plant growth is also a major loss of important protective cover habitat for juvenile salmonids. Dawe and his co-investigators (2011) estimate that approximately 10,056 m^2 of Carex channel edge community (Fig 1.3) was lost between 1978 and 2005, corresponding with a minimum of 17 tonnes of above ground dry weight biomass lost annually to the detrital food web. An additional 5 tonnes of dry mass per year were estimated to be lost from the decline in the *Deschampsia*-flats community. The Deschampsia-flats community was representative of the mid marsh and was predominated by *Deschampsia cespitosa, Potentilla egedii, C. lyngbyei, Glaux maritima, Juncus balticus and Trigochlin maritimum* (Dawe et al. 2011).

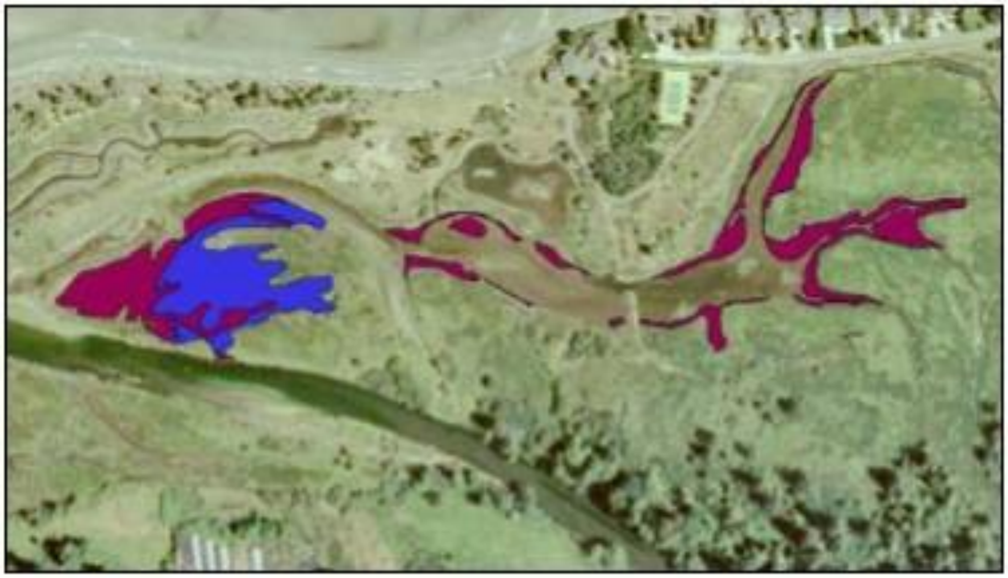

Figure 1.3 Locations of lost Carex channel edge in purple and Deschampsia flats communities in blue between the 1978 and 2005 vegetation surveys in the Little Qualicum River Estuary.

Figure adapted from Dawe et al. (2011)

Despite an intensive annual egg addling program since 2010, Canada geese continue to have a physical presence in the LQRE as evidenced by grazed vegetation, nesting pairs on site, grubbing of the marsh platform and goose tracks covering high traffic channels at low tide (Fig 1.4). The 2019 nest survey documented 14 nests in the LQRE (G. Ashley, Guardians of Mid-Island Estuaries Society, unpublished data, 2019).

1.4. Guardians of Mid-Island Estuaries Society.

The Guardians of Mid-Island Estuaries Society was founded in 2002 with the goal of restoring degraded estuarine marshes while also addressing the grazing pressure exerted by this locally overabundant Canada goose population. The population growth is currently addressed through a seasonal First Nations led harvest, egg addling programs along the east coast of Vancouver Island and continuous population surveys.

Figure 1.4 Evidence of the continued presence of Canada geese in the Little Qualicum River Estuary.

Restoration efforts in the Little Qualicum River Estuary have centered on transplanting C. *Lyngbyei* from healthy donor sites on the estuary to degraded mud flats and protecting these zones from grazing with temporary fencing. The first ten exclosures were built in 2010 to protect existing channel edge communities and monitor the effectiveness of goose exclosures in preventing further degradation. Additional exclosures and the beginning of transplanting to denuded sites began in 2015 and has continued annually. Exclosures in Figure 1.5 are labelled with the year of construction and an identifying number (ex. second exclosure built in 2010 labelled 2010-02). This commitment over the past ten years to protecting the remaining vegetation structure allows the unique opportunity the study changes over time directly associated with estuary vegetation restoration activities.

While the work by the Guardians is a success visually, they have not had the resources to adequately quantify the physical effect they have had in this estuary. This applied research project will help to measure the success of previous and proposed restoration efforts led by the Guardians and their partners in rebuilding the structure of the marsh community.

Figure 1.5 Yellow outlines of the location of Canada goose exclosures installed to protect sensitive vegeation in the Little Qualicum River Estuary, Vancouver Island, BC, labelled with the year of installation, July 2019.

Source: Drone imagery by Joshua Prahl July 2019

1.5. Research objectives.

This project seeks to answer whether restoration activities in the Little Qualicum River Estuary are successfully contributing to rebuilding the structure of a *C. lyngbyei pre*dominated marsh estuary platform in two ways:

1. Do the transplanted exclosures facilitate the return *of C. lyngbyei* densities on previously denuded mud flats?
2. Are sediment deposition rates higher inside exclosures than denuded mud flats and does the rate of deposition increase with the time exclosures have been in place?

It was expected that when *C. lyngbyei* is transplanted into denuded mud flats and protected from grazing pressure, the overall shoot density will increase over time. Marsh vegetation reduces tidal currents locally and promotes sediment deposition (Temmerman et al. 2003). The increase in shoot density should result in an increased rate of sediment deposition compared to locations that have not received transplant which would suggest that restoration of the *C. lyngbyei* channel edge vegetation is facilitating the retention of sediments and rebuilding the marsh platform.

Chapter 2.

Methods

2.1. Exclosure construction.

Canada goose exclosures have been in place in the Little Qualicum River Estuary since 2010, however the associated changes to the marsh platform have not been well documented. The construction dates of each exclosure currently in place was determined by searching through the Guardians photo records and through personal communication with Garreth Ashley and Tim Clermont (Fig 1.5). The original exclosures were built in 2010 using heavy rebar poles and metal fencing. Three new exclosures were constructed in May and June of 2019 by driving 1.5 m long alder poles into the substrate and surrounding with green plastic snow fencing or by weaving alder poles through pole sections. The alder poles are being trialed as a more natural fencing material and have had a very positive response from the public in more accessible locations. A full 'eco-cultural' fencing design was inspired by traditional fish weir construction and involves weaving thin willow branches between the alder poles and secured in place with organic hemp string. This reduces the need to introduce plastic into the estuary and is well received in areas that are visible along popular walking paths.

Carex lyngbyei is a resilient sedge that rapidly colonizes the denuded mudflats when protected from grazing pressure in the constructed exclosures. Transplants are removed from the healthy donor zones on the upper reaches of the estuary that are considered the reference conditions in LQRE. Plugs are removed from healthy donor sites with a custom-built tool adapted from tulip bulb planter designs (Fig 2.1). The plugs are then moved to denuded mud flat zones and replanted. Survival has not needed to be monitored closely in the past due to the high survival rate and rapid colonization by rhizomal growth in the following growing season.

Observational note: in addition to rapidly increasing the shoot density of denuded sites, Carex is also an ideal estuary plant for transplantation due to its hearty characteristics. From the 143 plugs taken from the donor sites and transplanted in to *exclosures 2019-01 and 2019-02, only two plugs failed to establish and the remaining appeared healthy despite the stunted growth from the shock of transplanting.*

Figure 2.1 ***Carex lyngbyei*** **plugs are extracted with a customized tulip bulb tool from an intact donor site in the upper reaches of the Little Qualicum River Estuary. Plugs were then transplanted along the denuded mud flat in exclosure 2019-01, in June 2019.**

2.2. Vegetation surveys.

This project does not focus on the specific changes to biomass, but it was important to gather preliminary insight into the benefits of restoration activities on restoring a degraded estuary. The average shoot density and shoot length was sampled within each exclosure in early June of 2019. These surveys were completed to support the sediment deposition values but was not a comprehensive biomass survey. A 0.25-m^2 quadrat was randomly placed in 3 locations in each exclosure and the number of shoots in each was counted (n=3 for each exclosure). Five stem lengths were measured at random in centimeters inside each quadrat (n=15 for each exclosure). It was expected that protecting the vegetation from goose grazing and facilitating *C. lyngbyei* regeneration with transplanted plugs would show an increase in density over time. So exclosures that have been in place for longer would have more shoots per square meter. The length of *C. lyngbyei* was also expected to increase with time to near the length of the donor sites after the individual plants have recovered from the shock of transplant that may stunt growth for several growing seasons.

2.3. Short-term sediment deposition.

Short-term sediment deposition rates were measured by adapting the filter pad method described by Thomas & Ridd (2004). It is a low-cost method of collecting sediment at short time intervals for intertidal studies. Whatman glass microfiber filter pads (1.2 µM pore size, 7 cm diameter) were pre-weighed to 0.1 mg using a precision lab balance generously shared by the Chemistry Department at Vancouver Island University. They were then secured to thin plastic disks with plastic coated paper clips that were anchored to the sediment bottom and collected the following week (Fig 2.2). The plastic prevents adhesion to the sediment surface and acts as a place keeper between trials. Filter pads were stored in vented Petri dishes to remove excess moisture, then collectively dried at 35 °C for six hours prior to re-weighing to 0.1 mg. Filter pad trials were in place from June 14th - 21st, June 30th - July 7th, and July 12th - July 19th, 2019. The timing of Sediment trap deployment was aimed at capturing sediment deposition rates during the peak of above ground *C. lyngbyei* biomass that occurs mid-summer in the Pacific Northwest.

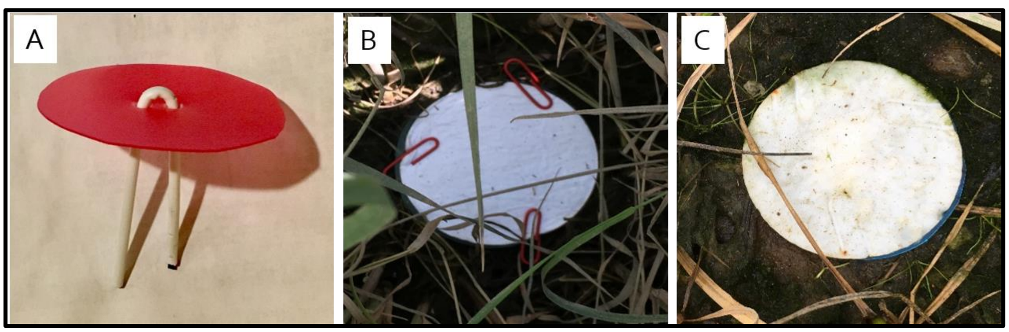

Figure 2.2 Sediment traps were anchored to sediment floor on plastic disks inside exclosures and on denuded mud flats in the Little Qualicum River Estuary, BC (A), filters papers were held in place by plastic coated paper clips (B) and collected after one week during the sampling periods in June and July 2019 (C).

In each round of sampling, 25 sediment traps were installed in exclosures of different ages and at three denuded mud flat zones. The denuded sites were a part of the carex channel edge community prior to goose impacts (Fig 2.3). Trials were randomly placed along the center of the *C. lyngbyei* zone inside the exclosures. Filter traps were deployed in the same location for each weekly trial. In total, six exclosures and three denuded mud flat areas were sampled per week. One-way ANOVA followed by Tukey’s multiple comparisons test was performed using GraphPad Prism version

8.4.1 for Windows (GraphPad Software, San Diego, California USA, www.graphpad.com) to examine differences in short-term sediment deposition by sampling location in each week of deposition collection and to generate summary figures. It was expected that older exclosures would house a denser stand of *C. lyngbyei* shoots and therefore trap more sediment than along the denuded mud flat sampling locations.

Sediment deposition rates were collected as grams of sediment per filter trap (surface area of 38.5 cm^2) per week of deposition and translated to the equivalent deposition rate per 1 m^2 per week. Each sampling location (exclosure ID label in the figures) had three filter traps deployed per sampling. If there were visible signs of damage to the filter papers such as by debris settlement from the last tidal inundation or algae mat settling on the trap, that specific filter trap was excluded from analysis for the weekly averages.

Individual flooding events were considered to calculate the cumulative inundation time at each sampling location. Tidal range heights were retrieved from the nearby Northwest Bay tidal station. On July 29th, 2019, the time that each sediment trap sampling location was submerged by the incoming tide was recorded. That time was correlated with a tidal height for that recorded time using the posted tidal charts and the 'Tides' iPhone app providing an estimate of the total submerged time at each sampling location per tidal cycle. The submerged time per tidal cycle was then added for a weekly average inundation time (Table 2.1). This was a rudimentary method for calculating tidal height and inundation time without specialized equipment.

Oceanside Geomatics donated their services to collect local elevation points within the estuary in August of 2019. Traditional RTK survey equipment was used to measure elevations accurate to < 1 cm in metres above sea level standardized to cgvd 1928 HT V2.0. Elevations were collected at each sediment trap and presented as the average elevation for each sampling location (Fig 2.4).

Figure 2.3 **Sediment traps (red dots) were deployed at each sampling location (outlined in red boxes) to compare rates of deposition inside exclosures installed in different years and along three denuded mud flat locations in the Little Qualicum River Estuary, Vancouver Island, BC in the summer of 2019.**

Source: Drone imagery by Joshua Prahl July 2019

Table 2.1 **Total inundation time for each sediment trap location (hours) as estimated by the tidal chart from Northwest Bay, BC.**

Exclosure ID	Week 1	Week 2	Week 3
2010-05	24.33	31.08	18.25
2010-08	24.33	31.08	18.25
2016-02	38.02	44.50	33.70
2017-02	31.17	37.33	25.83
2018-02	42.09	48.67	37.51
2019-01	38.02	44.50	33.70
Denuded 1	44.13	50.75	39.42
Denuded 2	37.65	44.04	32.63
Denuded 3	38.02	44.50	33.70
Average	35.31	41.83	30.33

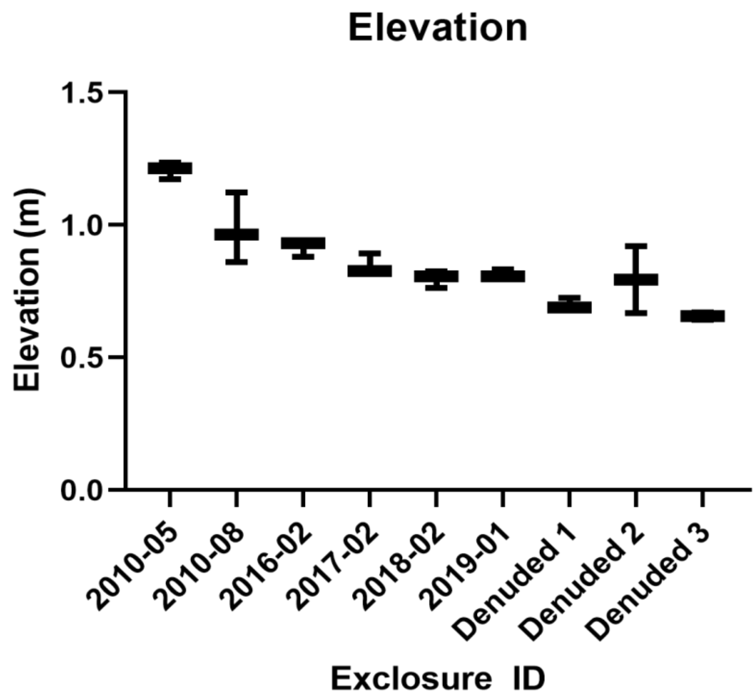

Figure 2.4 **Average elevation (metres above sea level) of sediment traps in each sampling location with whiskers representing the minimum and maximum values.**

Chapter 3.

Results

3.1. Vegetation surveys.

The vegetation surveys addressed the question of whether transplanted exclosures facilitate the return of *C. lyngbyei* densities to previously denuded mud flats. The density of *C. lyngbyei* differed within each exclosure in the LQRE (Fig 3.1). The exclosure ID refers to the label given to each exclosures (refer to figure 2.3) with a T denoting transplanted area in the exclosure, and an R denoting an area of natural regeneration in the exclosure. The difference in densities between exclosures was significant (ANOVA, $F(17,35) = 5.814$, $p < 0.0001$). Significant variations in densities are listed in Appendix A. Exclosures 2018-01, 2018-02, 2019-01 and 2019-02 had significantly lower densities of *C. lyngbyei* compared to the donor sites while all other exclosures were not significantly different from the donor sites.

Transplants are visually distinct for a few growing seasons before areas between plugs are filled in by rhizomatous growth. Exclosures constructed and transplanted in 2018 and 2019 showed the lowest density of carex as expected. Exclosures constructed in 2017 and earlier all begin to host densities that approached the reference donor site conditions. The original exclosures in LQRE were constructed in 2010 and now host a greater variety of plants compared to the most recently transplanted exclosures. Exclosure 2010-03 (16.7 shoots/m^2) had significantly lower mean density compared to 2010-06, 2016-02 and the healthy donor site. Exclosure 2010-06 had significantly higher mean density (149.7 shoots/m2) than exclosures 2018-01, 2018-02, 2019-01 and 2019-02.

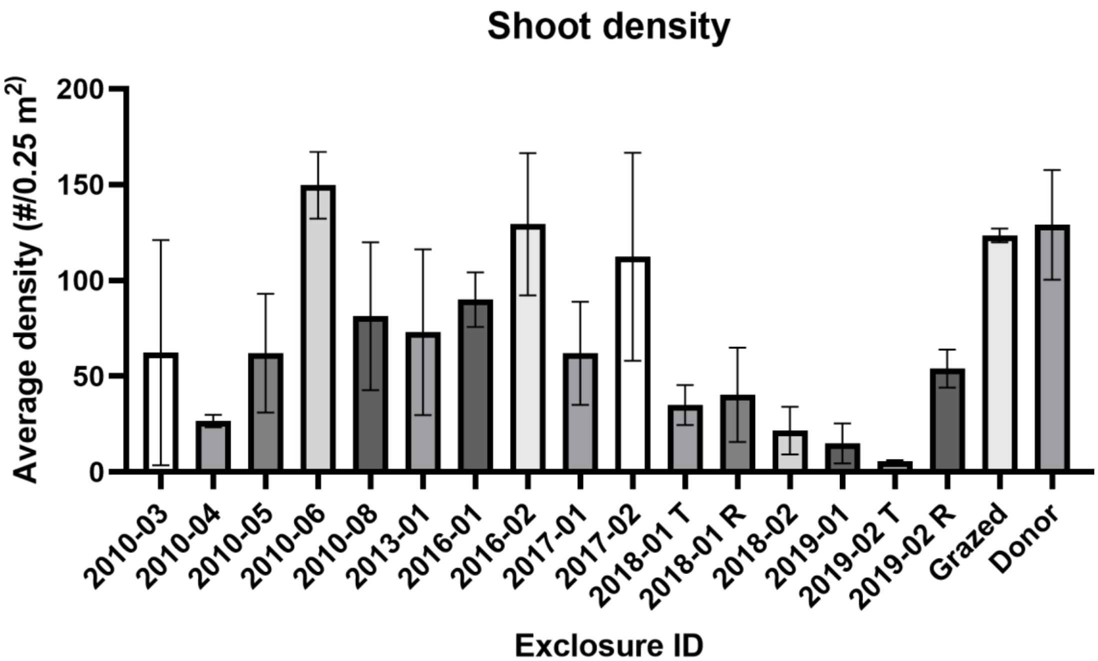

Figure 3.1 Average shoot density of *C. lyngbyei* (number of shoots per 0.25 m2 with standard deviation bars) inside exclosures constructed in different years, in a recently grazed zone and at the healthy donor zone in the Little Qualicum River Estuary, Vancouver Island, BC, June 2019. Error bars represent standard deviation of the mean.

The average shoot density by each exclosure age class illustrates how rapidly restoration of *C. lyngbyei* can return mud flat zones to conditions nearing unimpacted sites (Fig. 3.2). The average density between exclosures constructed in 2010, 20113, 2015, 2017, 2018 and 2019 was not significantly different (ANOVA, F (7,8) =1.909), p value=0.1923) but the trend is towards an increase in density over time.

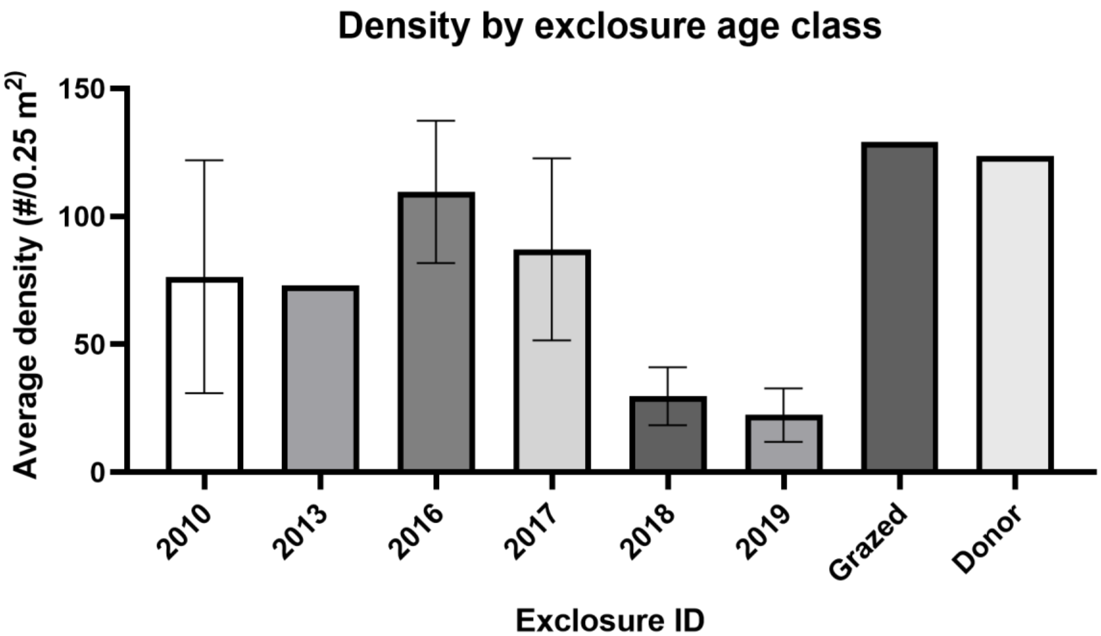

Figure 3.2 Average shoot density (number of shoots per 0.25 m^2 quadrat with standard error bars) for each exclosure age class compared to the grazed and healthy donor zones in the Little Qualicum River Estuary, Vancouver Island, BC, June 2019.. Error bars represent standard error of the means.

The average individual stem lengths also varied among sampling locations (Fig 3.3). Different exclosures had significant differences in average *C. lyngbyei* stem length (ANOVA, F (17,33) =11.19, p value = <0.0001). Exclosures built after 2016 have a shorter stem length on average than inside exclosures that have been protecting sedges for a longer time period. Combining the average stem lengths into exclosure age classes reveals a similar trend (Fig 3.4) though not significant (ANOVA, F (7,8) =3.352, p value=0.0558). Sedges in exclosures constructed in 2017, 2018, 2019 and the grazed locations all have stem lengths that are on average between 40 cm and 50 cm while the stem lengths in exclosures built in 2010, 2016 and the ungrazed reference sites average between 65 cm and 77 cm.

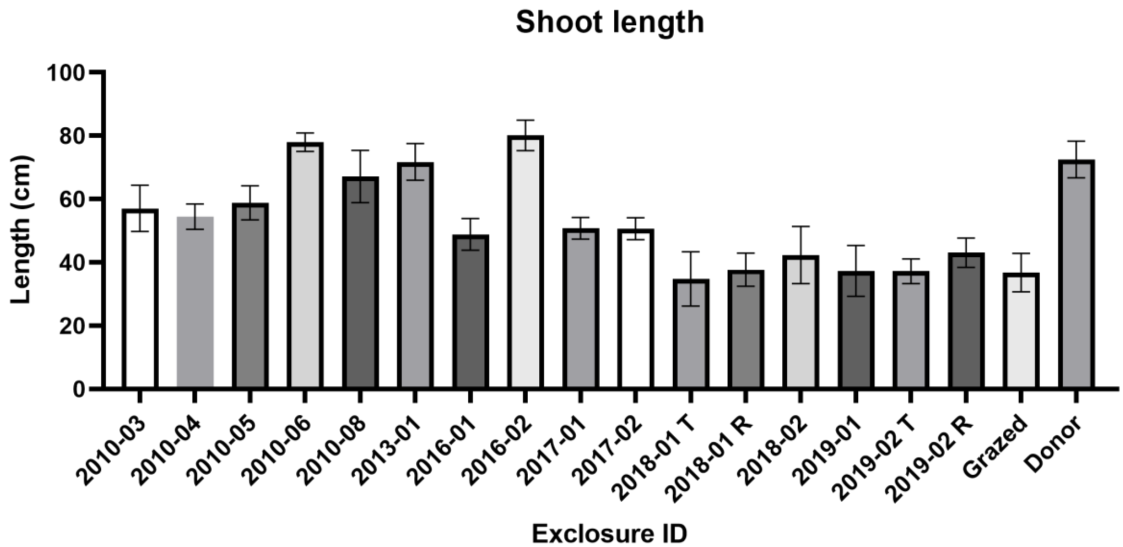

Figure 3.3 Average shoot length of *C. lyngbyei in centimetres* (with standard deviation bars) inside each exclosure constructed in different years, in a recently grazed zone and at the healthy donor zone in the Little Qualicum River Estuary, Vancouver Island, BC, June 2019. Error bars represent standard deviation of the mean.

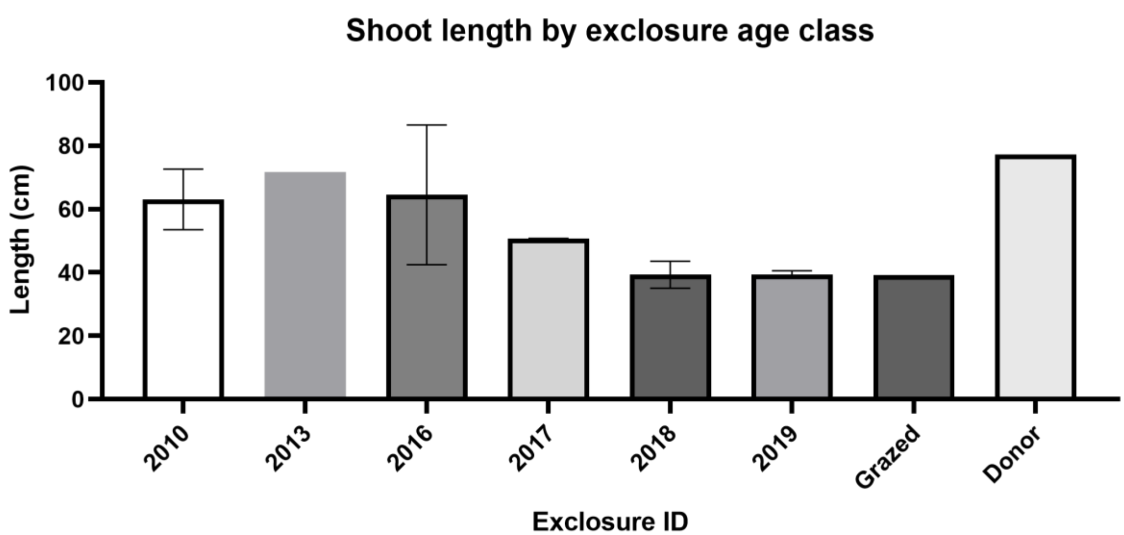

Figure 3.4 Average shoot length of *C. lyngbyei* in centimetres (with standard error bars) for each exclosure age class compared to the grazed and healthy donor zones in the Little Qualicum River Estuary, Vancouver Island, BC, June 2019. Error bars represent standard error of the means.

3.2. Rate of sediment deposition.

It was expected that sediment deposition rates would be higher in the exclosures than have been in place for longer compared to younger exclosures and denuded mud flat sampling locations as they will house a greater density of *C. lyngbyei* and therefore trap more suspended sediments. Sampling location had a significant effect on deposition rates in week one of deployment (Fig 3.5) (ANOVA, $F (6,10) = 8.530$, $P=0.0018$). Sediment traps in exclosures 2010-05, 2010-08 and 2016-02 collected significantly less sediments than in exclosure 2019-01 and at denuded sampling site 1. Tukey multiple comparison test was used to determine which mean deposition rates were significantly different (Appendix A). The maximum average deposition occurred at denuded site 1 (101.82 g/m^2/week) and the lowest average deposition was collected from exclosure 2010-05 (27.77 g/m^2/week). Following the first week of sampling, sampling location 2018-02 and the third denuded mud flat were added.

Week two of deployment (Fig 3.6) showed a relaxation of the visual groupings of deposition rates seen in the first week of sampling. The sampling location had no significant effects on deposition during the second week of sampling (ANOVA, $F (8,15) =2.403$, $P=0.0682$). The maximum average deposition rate of 48.79 g/m^2/week was collected in exclosure 2017-02. The lowest average deposition was collected again from exclosure 2010-05 at just 6.82 g/m^2/week. The range of deposition values was lower at every sampling location in week two than in the first week of sampling.

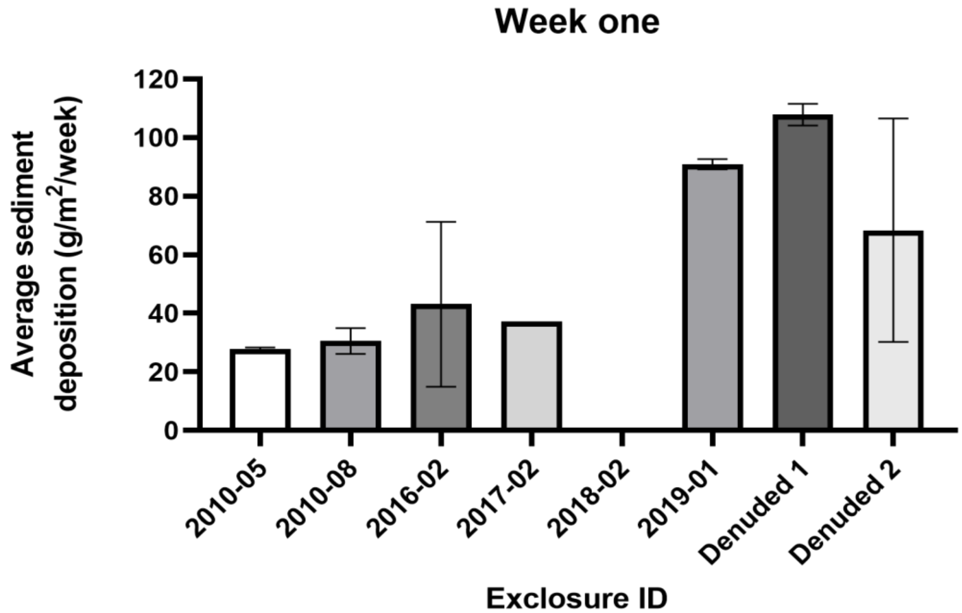

Figure 3.5 **Average sediment deposition in g/m²/week at each sampling location in the Little Qualicum River Estuary in the first week of deployment (June 14th-21st, 2019). Error bars represent standard deviation of the mean.**

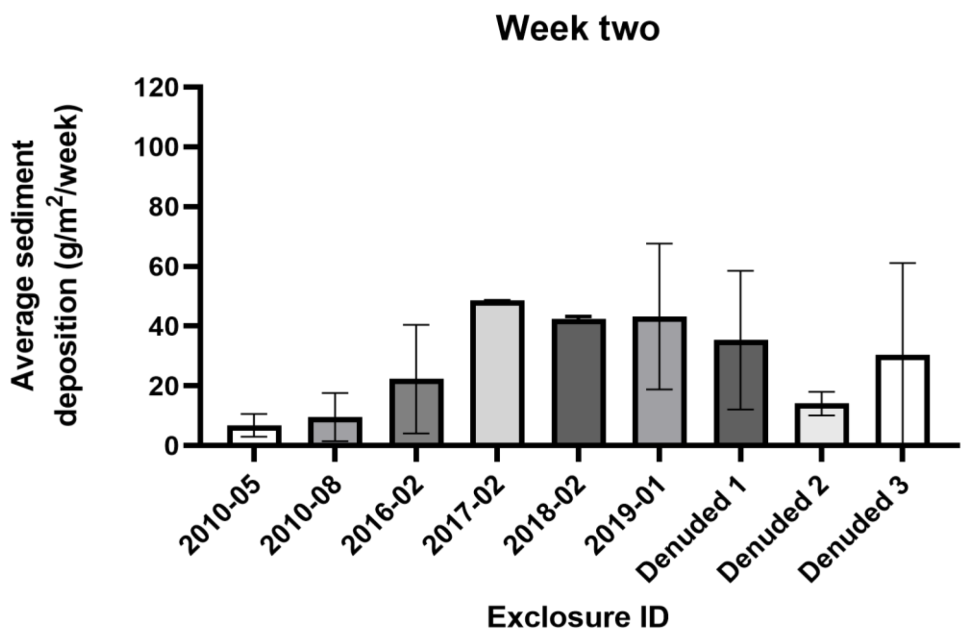

Figure 3.6 **Average sediment deposition in g/m²/week at each sampling location in the Little Qualicum River Estuary in the second week of deployment (June 30th-July 7th, 2019). Error bars represent standard deviation of the mean.**

The deposition rate trends by sampling location in week three (Fig 3.7) are similar to the first week. Location of the sediment traps had a significant effect on deposition rates (ANOVA, $F(8,16) = 23.83$, $P < 0.0001$). Deposition rates inside exclosures 2010-05, 2010-08, 2016-02 and 2017-02 were all significantly lower than inside exclosures 2018-02, 2019-01 and along the denuded mud flat locations 1,2 and 3. Exclosures 2010-05, 2010-08, 2016-02 and 2017-02 were also had a higher elevation of sampling than the remaining sampling locations. The maximum average deposition rate was collected in denuded mud zone 1 at 101.53 g/m^2/week and the lowest rate was again collected from 2010-05 at 37.36 g/m^2/week.

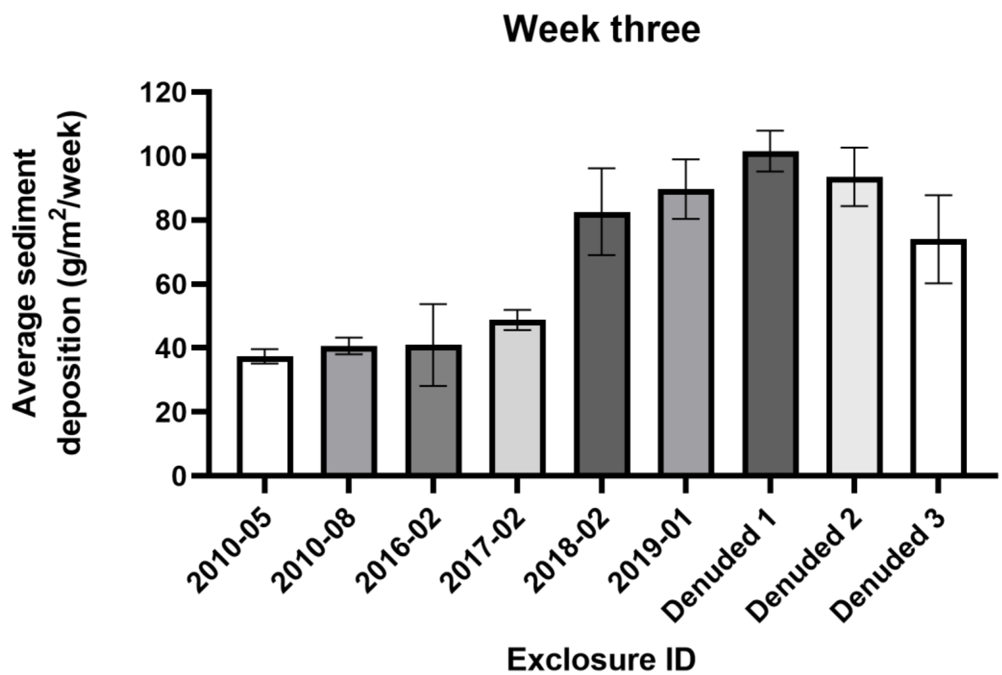

Figure 3.7 Average sediment deposition in g/m^2/week at each sampling location in the Little Qualicum River Estuary in the third week of deployment (July 12th-19th, 2019). Error bars represent standard deviation of the mean.

The overall highest average rate of sediment deposition occurred at denuded sampling site 1, placed between 2010-05 and 2017-01 (Table 3.1). This zone was completely lacking vegetation and collected 107.82 g/m^2/week. The smallest rate of deposition occurred in exclosure 2010-05 and was just 6.83 g/m^2/week. The rate of deposition varied from week to week of the trials with the second week (Fig 3.6) having lower amounts of sediment collected than both weeks one (Fig 3.5) and three (Fig 3.7).

Comparing the deposition rates among weeks of sampling did not have significant effect on the overall average sediment deposition (ANOVA, F (1,64) = 1.891, P=0.174).

Table 3.1 Average deposition rates (g/m^2/week) over three weeks of sediment trap deployment.

	Average deposition (g/m^2/week)		
Exclosure ID	**Week 1**	**Week 2**	**Week 3**
2010-05	27.77	**6.83**	37.37
2010-08	30.54	9.62	40.62
2016-02	43.11	22.34	40.93
2017-02	37.21	48.79	48.84
2018-02	-	42.32	82.60
2019-01	90.89	43.22	89.65
Mud flat 1	**107.82**	35.33	101.53
Mud flat 2	68.39	14.14	93.50
Mud flat 3	-	30.46	74.01

Chapter 4.

Discussion

4.1. Sediment deposition variation between sampling locations

Short-term sediment deposition varied spatially and temporally in the LQRE. This applied research project was developed to focus on two main questions concerning sediment deposition rates and how they relate to rebuilding the marsh platform that was lost due to Canada goose grubbing.

The first of which was: *Do the transplanted exclosures facilitate the return of C. lyngbyei densities on previously denuded mud flats?* The preliminary vegetation surveys completed in June of 2019 suggests that protecting vegetation from grazing and transplanting *C. lyngbyei* plugs on to denuded mud flats does allow plants to regrow and colonize the mud flats. The density did increase over time after initial restoration activities. It does take several years for the densities to fill in by rhizomatous growth of the sedges but over time we expect the channel edge vegetation to return to pre-impacted conditions. The variability of carex density in the 2010 exclosures was due to the additional plant species not included in the shoot density assessment, and that these exclosures were now at higher elevations on the upper marsh platforms instead of along the lower mud zones. The 2010 exclosures were constructed largely for monitoring purposes while the exclosures installed in later years were constructed primarily for restoration.

The second proposed question asked: *Are sediment deposition rates higher inside exclosures than denuded mud flats and does the rate of deposition increase with the time exclosures have been in place?* Contrary to the expected mechanism of a greater density of sedges trapping higher sediment deposition rates, the denuded mud flat zones in nearly all trials had higher rates of deposition than within the oldest exclosures that shelter denser carex stands. The density and stem length of *C. lyngbyei* was greater in older exclosures than more recently constructed exclosures suggesting vegetation was restored to the denuded mud flats but this did not translate to increases in deposition. This suggest that other factors have a more direct influence on sediment deposition rates than the age of the exclosures. In most cases the younger exclosures (i.e. 2018 and 2019) collected greater sediment than the older exclosures.

The lack of influence of the density of *C. lyngbyei* implies that there are other factors at play that influence local rates of sediment deposition. Sediment deposition is the settling of inorganic and organic particles by gravity during tidal inundation (Temmerman et al. 2003; Butzeck et al. 2015). Marsh vegetation can reduce the tidal currents locally and promote sediment deposition (Temmerman et al. 2003; Austin et al. 2017) but additional forces are also involved. Sites that are flooded more frequently and for a longer duration exhibit higher rates of sediment deposition (Neubauer et al. 2002). The mud flats are, by function of being a denuded site, at a lower elevation than the sediment traps in the marsh vegetation. The preliminary estimates for inundation rates, noted in table 2.1, suggest this observation is true. The mud flats were submerged by the tide for longer periods than the older exclosures on the upper marsh platform. The RTK elevation survey further confirmed that the mud flat sampling locations tended to be at a lower elevation than the vegetated exclosures (Fig 2.4).

Neubauer et al. (2002) found that deposition rates in a Virginia tidal marsh were consistently higher on the creekbank than on the interior sampling blocks especially during the growing season. The authors attributed this observation to the rapid decrease in turbulent energy as the water floods onto the marsh platform which causes sediments to fall out of suspension. Though water turbulence was not measured during this study, that may contribute to why the denuded mud flats and sediment traps within exclosures constructed in 2018 and 2019 collected higher rates of deposition. As the flooding tides reached the marsh platform, sediment may have come out of suspension and settled before flooding the marsh platform. More sediment is deposited in proximity to the tidal creek as that is the primary source of mineral sediment (Christiansen et al. 2000; Neubauer et al. 2002). A study in the Scheldt estuary, Netherlands, also found sedimentation rates decreased with the distance from the marsh edge and further attributed surface elevation as a main driver (Temmerman et al. 2003).

The overall mean sediment deposition rate measured during this project was 50.72 g/m^2/week. The deposition rates measured varied between 6.82- 107.88 g/m^2/week. Local deposition rates vary based on many factors. Marsh elevation has a significant role on sediment deposition, but the effect can also be dampened by local hydrology, plant/ flow interactions and total suspended solid loads (Leonard 1997). Leonard (1997) suggested that tidal creek geometry, creek channel position and tidal stage all act synergistically to control sediment and particulate delivery on the surfaces of tidal wetlands. As such it is difficult to directly compare sediment deposition rates among studies. Deposition rates in a Virginia tidal marsh varied from 0.1 g/m^2/day during the winter to 284.2 g/m^2/day for July and August in 1999 (Neubauer et al. 2002). Hensel el al. (1998) measured average sedimentation rates of 0.8 to 5.4 g/m^2/day at the Rhone river delta. Sediment deposition rates in this study were as such within a range of expected values and could be comparable to the rates that would be measured in similarly sized watersheds on Vancouver Island.

The Scheldt estuary, Netherlands, study measured greater deposition during the winter than in the summer (Temmerman et al. 2003) while many studies link the greatest rates of sediment deposition with the seasonal freshet input of sediments. This highlights the importance conducting regional experiments rather than comparing to studies around the world. Deposition rates may be more valuable when being used to inform seasonal variation in deposition and accretion rates for long term studies in changes to marsh elevation.

4.2. Sediment deposition variation between weeks

Trends in deposition in LQRE also varied among the weeks of sediment collection. Though the overall average deposition by week was not significantly different, clear trends between figures 3.1, 3.2 and 3.3 show that weeks one and three of sampling had higher variability and higher rates of sediment deposition than week two. Timing, frequency and height of inundation, distance to the sediment source, and seasonal variations in water and wind levels all affect sediment deposition (Betzeck et al. 2015). The potential sources of temporal variability in sediment deposition are discussed in this section.

4.2.1. Tidal influence

Temporal variations in sediment deposition are often attributed to tidal inundation periods (Temmerman et al. 2003; Betzeck et al. 2015). Vancouver Island experiences semi-diurnal tides in that one tidal cycle contains two high and two low tides of different size every lunar day. Tidal forces are influenced by the gravitational pull of the moon and the sun. The moon exerts greater force on earth's surface and so the tidal cycle varies with the rotation of the moon around the earth (Fig 4.1). The lunar day is the time it takes for the moon to rotate around the Earth, 24 hours and 50 minutes (Dohler 2007). The highest tides occur with the new moon and full moon (spring tides) and the lowest high tide levels occur with the first or last quarter moon (neap tides) (Dohler 2007).

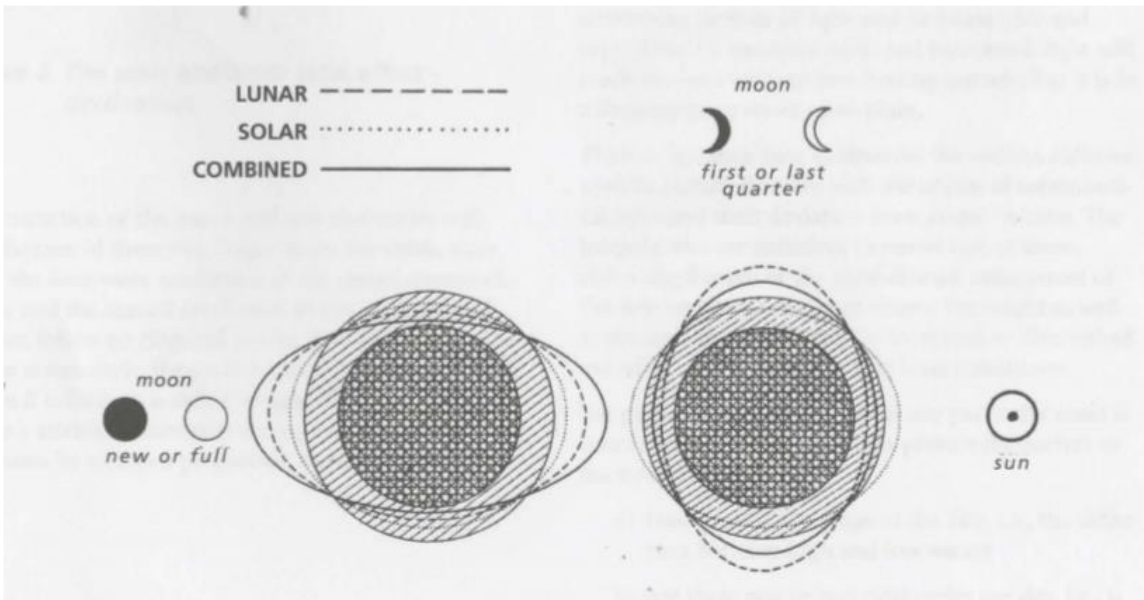

Figure 4.1 Solar and lunar tidal effect.

Source: Dohler (2007)

As this experiment was designed to measure short-term sediment deposition over a week, each trial experienced slightly different tidal cycles and therefore a different resulting inundation period. (The tidal cycles from the nearby Northwest Bay tidal station are presented in Appendix B). Week two of sampling experienced the greatest average daily tidal ranges, the highest high and lowest lows, out of the three weeks of sampling (Table 4.1). The average daily tidal height was comparable in each of the three weeks of sampling (Table 4.1). Christiansen et al. (2000) found that suspended sediment concentrations increased with tidal amplitude and subsequently promoted higher rates of disposition on higher tides. If the differences in tidal amplitude were the most significant influence on sediment deposition in LQRE we would have expected to see higher rates of deposition in week two of sampling based on their findings.

The tidal range may influence the rate of sediment deposition, but three weeks of limited sampling is not enough data to draw definite conclusions.

Table 4.1 Average minimum and maximum daily tidal heights, overall average tidal heights and average daily range of tidal heights recorded at the Northwest Bay tidal station during the three sampling periods.

	Average daily maximum height	Average daily minimum height	Daily average height	Average daily range
Week one	4.68	0.88	3.19	3.80
Week two	4.84	0.63	3.19	4.21
Week three	4.60	0.94	3.21	3.66

4.2.2. Wind influence

Waves and currents are the primary mechanisms of resuspension in estuaries and are generated locally by wind acting on the fetch of the estuary basin (Green and Coco 2013). Increased wave energy could increase the suspended sediment load in the LQRE but the developed spit to the north, the partial berm to the west that divides the national wildlife management unit from the regional conservation unit and the shallow nature of the estuary channels all limit the fetch length (see Figure 1.2). The nearest weather station with available wind speeds in LQRE was a part of the school-based weather station program developed by Andrew Weaver and Ed Wiebe at the University of Victoria. The Parksville Elementary School station (49°18'53.7"N 124°18'41.1"W) recorded minimum, maximum and average hourly wind speeds (Weaver and Wiebe 2006) presented in Table 4.1. Week two of the sediment trials did have a lower average wind speed compared to week one and three which could account for the reduced sediment deposition if resuspension is the key influence on temporal variation in deposition rates. Wind speeds and duration may play a role in sediment resuspension in the LQRE but the full impact to sediment deposition in this study is unclear.

Intense weather patterns with periods of heavy rainfall could also affect the amount of sediment retained by the filter traps and would need to be considered in future studies. The sediment trap start dates were chosen to avoid any forecasted rainfall and was not an issue during this study.

Table 4.2 Minimum, average and maximum wind speeds recorded by the Parksville Elementary weather station during the three sampling periods.

	Minimum	Average	Std Dev	Maximum
Week 1	0	4.34	4.20	25.91
Week 2	0	2.54	2.68	16.65
Week 3	0	2.99	3.17	28.16

4.2.3. Little Qualicum River discharge

Peak river discharges are correlated with an increase in sediment load (Woodruff et al. 2001; Kitheka et al. 2005; Snedden et al. 2007). The Little Qualicum River water station (49°21’16’’N, 124°29’00’’ W) near Qualicum Beach has been recording continuous water levels for 36 years (Government of Canada 2020). Figure 4.2 displays the discharge rate (orange line) and water level (green line) of the Little Qualicum River from June 1st, 2019 to July 31st, 2019 (Government of Canada 2020). The red boxes were added to the figure to denote the three sampling periods. The first and third week have a clear spike in discharge rates while the second week remains consistent. These discharge spikes may be delivering additional sediment into the estuary accounting for the observed increase in rates of deposition during these weeks. Major sediment inputs typically occur during the spring freshet (Geyer et al. 2001) but this pulsed water releases may be mobilizing remaining sediment that remain upstream into the LQRE.

The Little Qualicum River is the largest watershed in the Little Qualicum Water Region (WR2LQ) draining an approximate 251.7 km^2 (Waterline Resources Inc 2013). Cameron Lake is the major surface water feature. Two hydrometric stations and 42 water diversion points exist in the region. Flows in the Little Qualicum River are managed by DFO to protect sensitive fish habitat. Flows are controlled by a weir at the outlet of Cameron Lake.

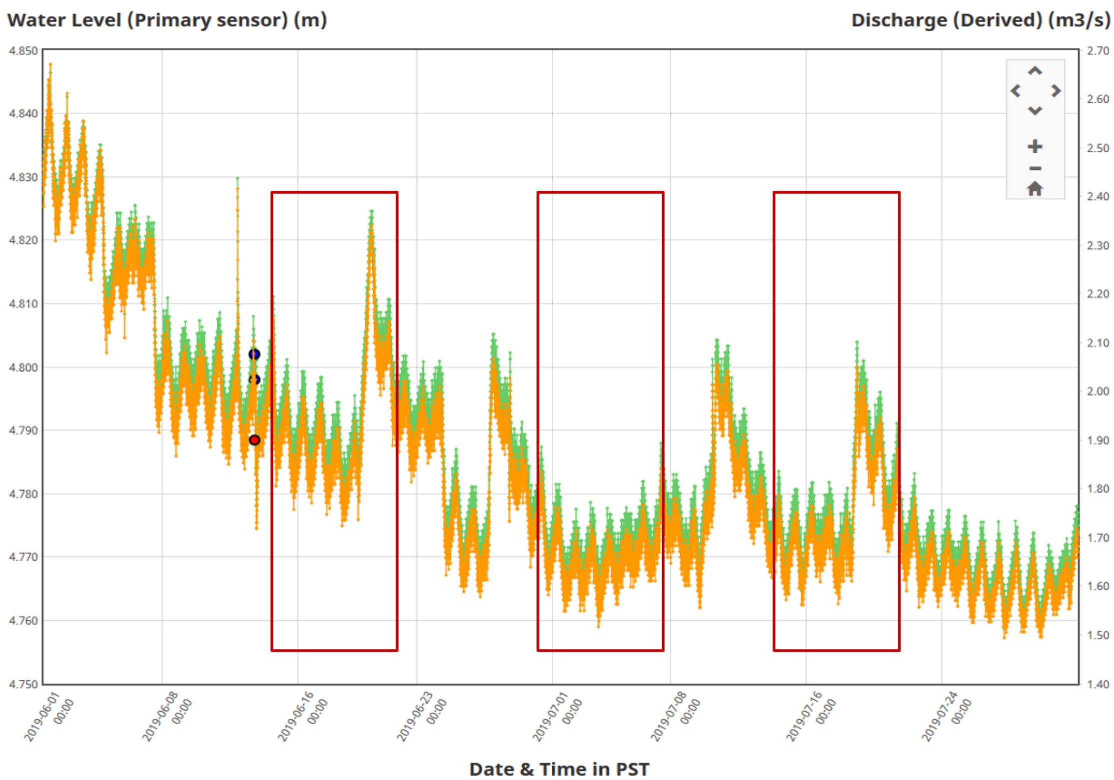

Figure 4.2 Little Qualicum River discharge with boxes outlining the three weeks of sediment deposition trials.

Source: Government of Canada (2020)

4.3. Research limitations

Precipitation and weather patterns were unusual throughout 2019. In April of 2019, the River Forecast Centre was measuring the Snow Basin Index at 66% of the normal range (Fig 4.3). The snowmelt component of seasonal runoff was therefore below normal. The typical freshet occurs in late May but with warmer weather through April and less snowpack, the freshet occurred earlier in the season than usual evident by the spike in discharge in mid April (River Forecast Centre 2019) (Fig 4.4). The timing of sediment trap deployment beginning in mid June had been designed to trap the freshet sediment delivery but may have missed the initial influx by up to a month. Extending sediment deposition studies for an entire year would provide greater insight into temporal variability in estuary deposition.

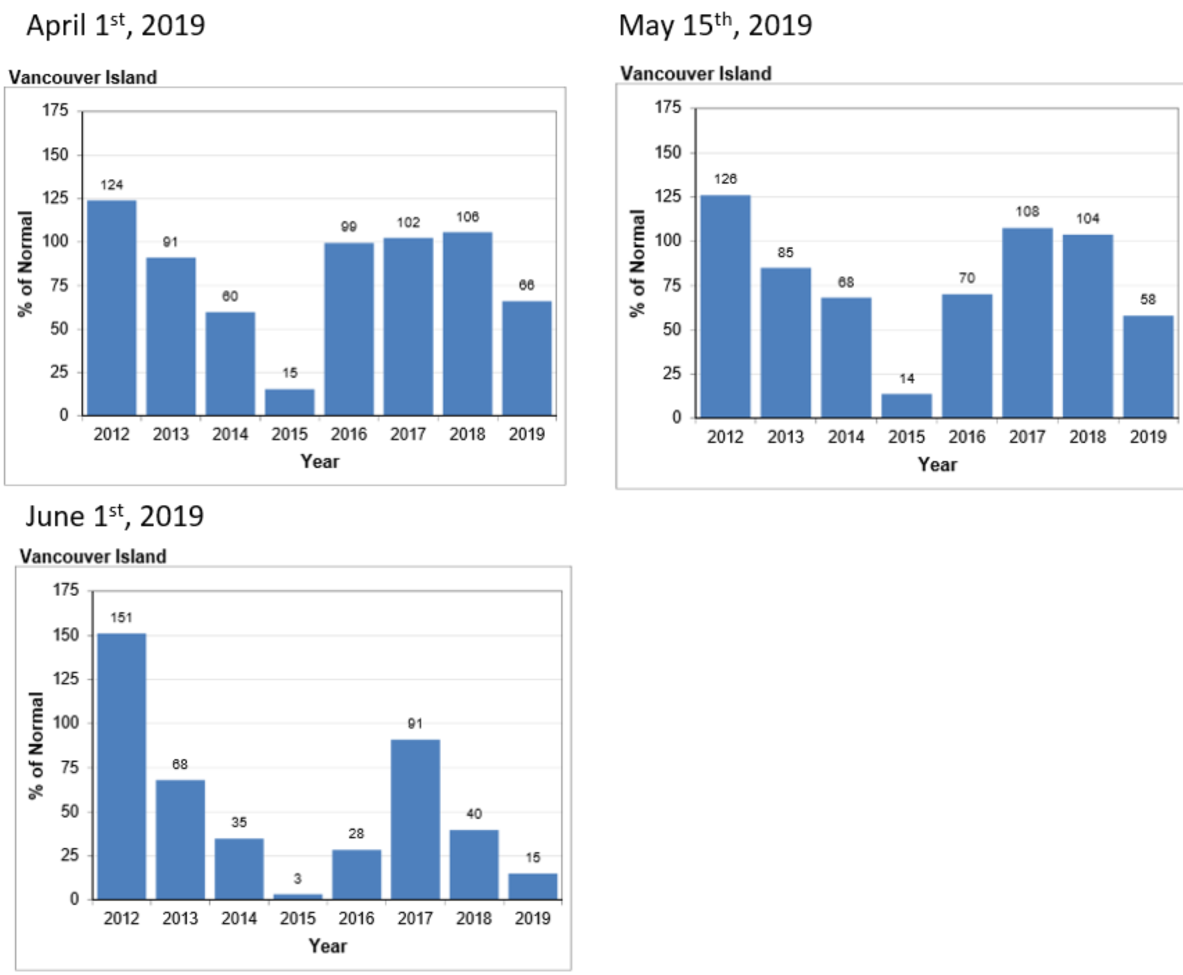

Figure 4.3 Snow basin index for Vancouver Island in April, May and June of 2019.

Figure adapted from the River Forecast Centre (2019)

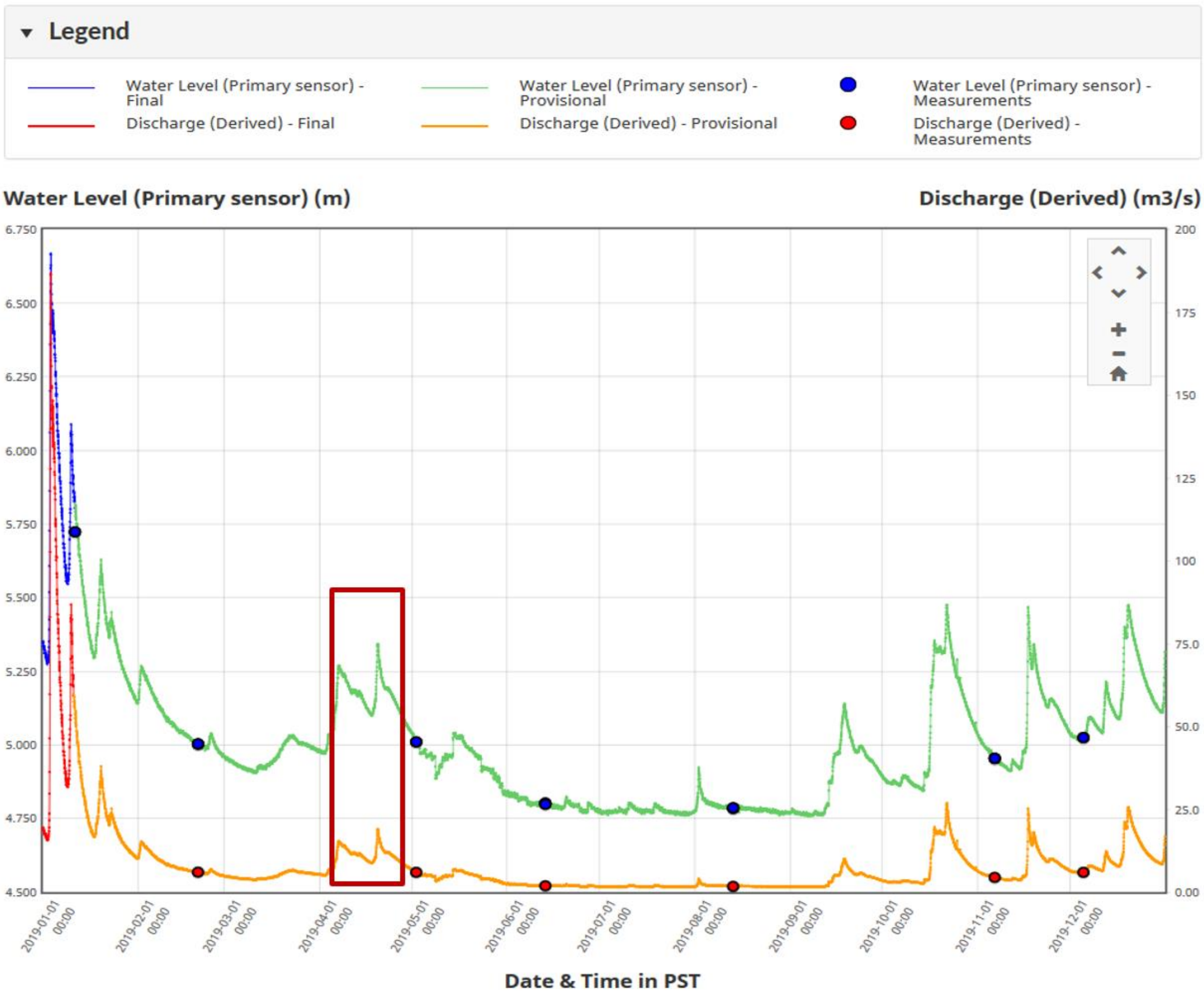

Figure 4.4 Water level and discharge rate recorded at the Little Qualicum River Station 08HB029 from January 2019 to December 2019.

Source: Government of Canada (2020)

This applied research project focused on measuring the rate of sediment deposition (the settling of material on the marsh surface) and though we can start to draw conclusions towards the source of this settlement I cannot say for certain that the sediments measured were transported in to the estuary rather than being mobilized by the resuspension of existing marsh sediments. Long-term studies of estuary restoration should consider the use of erosion pins to monitor for erosion/accretion of the marsh platform (Couper et al. 2002; Lawler and Leeks 1992) or rod surface elevation tables (RSET) to monitor changes in relative elevation (Cahoon et al. 2002; Callaway et al. 2013). These methodologies would not have contributed valuable information towards this study as it occurred over one short field season. In depth studies on the effects of flow velocity and turbulence on a tidal marsh surface in Virginia concluded that fine sediment was not remobilized by tidal flows after initial deposition, with no indication of resuspension of sediment at any time including falling tide when the velocities and stresses on the marsh surface were at their greatest (Christiansen et al. 2000). Aboveground plant biomass reduces energy in the water column of incoming tides and typically increases sediment deposition and decreases erosion and remobilization of sediments (Christiansen et al. 2000; Butzeck et al. 2015). If resuspension of sediments is a concern it is more likely to be sediments from the denuded mud flats further exemplifying the importance of restoring *C. lyngbyei* channel edge vegetation to prevent further marsh loss.

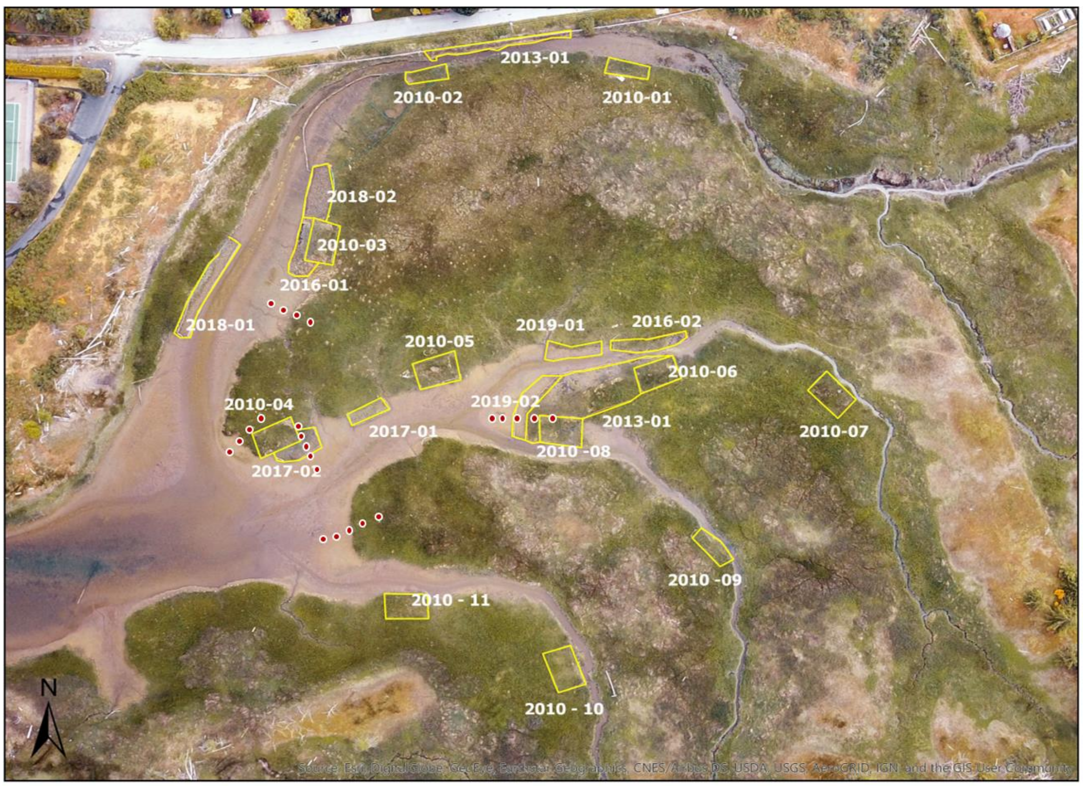

Figure 4.5 Suggested sediment deposition sampling locations for future studies.

Future studies of sediment deposition in the LQRE should consider a stronger experimental design to remove the confounding factor that elevation plays on sediment deposition. Figure 4.5 contains the suggested sampling locations for additional studies on sediment deposition. Each row of sediment traps should begin and end at the same relative elevation allowing for more direct observation of the influence of the edge of the marsh platform on deposition rates. The lack of replicate sediment collection at the same elevation makes it impossible to completely draw conclusions when comparing the denuded mud flats to restored sampling locations. Having replicates of the same elevation within vegetated stands and along denuded mud flat sites would allow the investigation of the role of elevation and inundation rates on deposition rates. Sediment trials should continue through an entire year to compare seasonal variability.

4.4. Importance of estuary restoration

Patterns of sediment deposition observed in this study did not confirm the original hypothesis proposed in this applied research project that older exclosures will shelter denser stands of *C. lyngbyei* and therefore have greater sediment deposition rates. Increasing the density *of C. lyngbyei* channel edge vegetation was not correlated with an increased rate of sediment retention, but a stronger experimental design is needed to further confirm this finding. The results of this project however do not negate the importance of restoring estuaries degraded by the non-migratory population of Canada geese.

Tidal wetlands are sensitive to processes that affect their elevation relative to sea level because they occupy a narrow band of elevation (Callaway et al. 2013) with plants specialised to occupy a specific inundation regime. Estuarine plants are constrained as they cannot tolerate frequent inundation at lower elevations and are outcompeted by terrestrial vegetation at higher elevations (Morris 2006). Allochthonous mineral sediments and organic marsh plants contribute to raising elevations while sea level rise, regional subsidence and compaction of sediments all act to decrease relative elevations (Callaway et al. 2013) but so do the organic inputs from vegetation. The surface elevation determines the tidal inundation frequency and duration which is a major component in temporal patterns of accretion in intertidal systems where the sediment supply is largely allochthonous (Marion, Anthony & Trentesaux 2009). High volumes of allochthonous mineral sediments promote an increase in elevation over time which then in turn reduces the sediment input (Krone 1987).

The Little Qualicum River Estuary marsh platform may be composed of largely organic soils rather than mineral soils and so depends on the detrital input from plant matter to build up the marsh elevation to meet sea level rise and subsidence rates.

Further studies on the organic composition of the soils is required. A large watershed has the capacity to transport a large volume of sediment over the course of the year. The Fraser River for example has an estimated annual suspended sediment load of 17 million tonnes per year (McLean et al. 1999) and the watershed covers approximately 217,000 km^2 (Morrison et al. 2002). Estuary soils form from incremental additions of glaciofluvial and marine sediments and organic matter from estuarine plants accumulating on the soils surface (Jesperson & Osher 2008). A small watershed such as LQRE would not receive the same volume of sediments and rely more on the organic input for sediment accretion over time. The potential loss of organic input associated with Canada goose overgrazing could greatly impact the ability to build up marsh soils over time and reduce the capacity to keep up with sea level rise.

The continued loss of channel edge vegetation and decline in frequency and cover of marsh plants could have serious impacts on the detrital food web and its ability to support the species that depend on it (Dawe et al. 2011). There are growing concerns for the viability of many wild salmon populations in British Columbia (Darimont et al. 2010). Only 4% of monitored streams in BC are meeting management targets for escapement set by Fisheries and Oceans Canada (Price et al. 2008). Various salmonid life stages depend on marine, estuarine and riverine environments. Anadromous salmon that survive to reproduce pass through estuaries at least twice: first as out-migrating juveniles and again as adults retuning to freshwater to spawn (Bottom et al. 2005a). Estuaries provide three main advantages to salmon during the transition to a marine environment. The first is it acts as a productive feeding area capable of sustaining increased growth rates (Healey 1981; Bottom et al. 2005a). The major prey of juvenile salmonids tends to be detritus feeders, indicating the importance of detritus and the estuaries capacity to trap allochthonous organic carbon (Healey 1981). Estuaries provide temporary refuge from predators. And finally, estuaries act as a physiological transition zone where fish can gradually acclimate to the altered salinity regime (Simenstad et al. 1982; Bottom et al. 2005a). The duration of estuary residence differs between Pacific salmon species. Pink salmon tend to pass right through, chum salmon stay in estuaries for days to weeks, and sub-yearling chinook salmon may remain for several months (Thorpe 1994; Bottom et al. 2005a). Studies in Northwest estuaries indicate that estuary restoration can be a cost-effective measure to improve salmonid rearing conditions (Bottom et al. 2005b; Darimont et al. 2010). Restoring the extent of marsh vegetation provides refugia zones with overhanging vegetation. Wetland recovery can expand the life history variation in a salmonid population by allowing greater expression of estuarine behaviours (Bottom et al. 2005b) allowing greater resilience to persist in salmon stocks.

Restoring the channel edge vegetation in LQRE has benefits that extend beyond fish and wildlife. Estuaries are among the most productive ecosystems in the world (Couto et al. 2013) and recent interest has been growing in the capacity for carbon storage in estuary soils (Jespersen & Osher 2007). Estuary soils form by incremental additions from mineral sediments and organic matter to the soil surface. Estuary vegetation withdraw carbon dioxide from the atmosphere and store it in plant tissues. Organics are then protected from decomposition by anaerobic conditions as they are buried (Jespersen & Osher 2007). Soil carbon has a high capacity for carbon sequestration from this long-term storage capacity (Wang & Hseih 2002) Tidal marsh and coastal ecosystem storage of carbon has been termed “Blue Carbon”. Estuary restoration could provide a valuable tool for increase carbon sequestration to combat increase CO_2 atmospheric levels. Rates of CO_2 sequestration could be an order of magnitude larger in tidal marshes than terrestrial forests (Chmura 2011). To measure the capacity for carbon sequestration, researchers must consider the permanence of Blue Carbon storage (Chmura 2011). Sea level rise is a threat to permanence.

Marsh vegetation plays a large role in responding to changes in local sea level by building soils vertically through the accumulation of inorganic and organic material that maintain elevation levels. Small estuaries like the LQRE are threatened by both the abiotic threat of sea level rise and the biotic threat of invasive species such as the overabundant resident population of Canada geese. Restoration of the marsh vegetated platform by organizations like the Guardians of Mid Island Estuaries Society therefore plays a crucial role in acting against climate change. Without significant efforts to reduce the population size of resident geese, the grazing pressure will continue to increase. Restoring marsh vegetation will be significantly more costly and challenging if there are no remaining nearby stands dense enough to act as a donor site. These small estuaries area t a permanent risk of being degraded and require immediate funding and effort to prevent the complete loss of function.

Chapter 5. Conclusions

Restoration of the *Carex lyngbyei* channel edge vegetation community is successfully achieved by transplanting donor plugs of the sedge to denuded mud flat zones and protecting these plants from grazing. As expected, the density of *C. lyngbyei* increased over time. Contrary to the original research hypothesis though, this increase in density did not translate to an increase in localized sediment deposition.

Sources of temporal and spatial variation in sediment deposition may include a combination of local elevations, tidal range, river discharge rates and weather patterns. The most recently transplanted exclosures and mud flats had higher rates of sediment deposition than the sampling locations within older exclosures. The existing vegetation on the marsh platform acts as a barrier to slows turbulent energy and allows suspended sediments to settle. This results in the apparent increase in sediment deposition in the mud flats and young transplant zones compared to the upper marsh zones sampled. Localized deposition rates also increase with the duration of inundation and the sampling locations that had the highest rates of deposition occurred at the lowest elevations. Temporal variation in deposition rates occurred from the interaction of tidal influence, wind influence and differing discharge rates from the Little Qualicum River. Deposition rates measured in this study were within the range of expected values compared to other deposition studies but are dependent on local estuary morphology and sediment sources. The methodology described in this applied research project offers a low cost and easily replicated design to explore deposition rates.

These findings do not negate the importance of restoring estuaries degraded by the non-migratory population of Canada geese. Estuaries such as the Little Qualicum River Estuary may be more disconnected from the glaciofluvial deposits larger watersheds contribute and rely instead on the autochthonous organic input from vegetation. This distinction strengthens the need to restore degraded estuary ecosystems and prevent further damage, so the organic input is not further reduced over time. Promoting the growth *C. lyngbyei* in LQRE and in all degraded estuaries on Vancouver Island will continue to facilitate the accretion of organic sediments and promote long term carbon storage and resilience against sea level rise.

Restoration of estuaries impacted by the non-migratory population of Canada geese is achieved through the reduction of the local population, continued monitoring and egg addling programs and restoring estuary vegetation. Supporting the initiatives of organizations actively working to do so not only benefits fish, wildlife and native migratory bird species but provides ecosystem services that combat climate change.

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Appendix A.

Significance tables

Table A1 Holm-Sidak's multiple comparisons test results of significant variations in mean *C.lyngbyei* density between exclosures.

Holm-Sidak's multiple comparisons test	Mean Diff.	Significant?	Summary	Adjusted P Value
2010-04 vs. 2010-06	-123	Yes	**	0.0032
2010-04 vs. 2016-02	-102.7	Yes	*	0.033
2010-04 vs. Donor	-102.3	Yes	*	0.0155
2010-06 vs. 2018-01 T	114.7	Yes	**	0.0085
2010-06 vs. 2018-01 R	109.3	Yes	*	0.0155
2010-06 vs. 2018-02	128	Yes	**	0.0018
2010-06 vs. 2019-01	134.7	Yes	***	0.0008
2010-06 vs. 2019-02 T	144.2	Yes	**	0.0016
2016-02 vs. 2018-02	107.7	Yes	*	0.0187
2016-02 vs. 2019-01	114.3	Yes	**	0.0088
2016-02 vs. 2019-02 T	123.8	Yes	*	0.0133
2018-01 T vs. Donor	-94	Yes	*	0.0419
2018-02 vs. Donor	-107.3	Yes	**	0.0085
2019-01 vs. Donor	-114	Yes	**	0.0037
2019-02 T vs. Donor	-123.5	Yes	**	0.007

Table A2 Tukey's multiple comparison test results of significant variations in mean stem length by exclosure.

Tukey's multiple comparisons test	Mean Diff.	95.00% CI of diff.	Significant?	Summary	Adjusted P Value
2010-06 vs. 2010-04	23.47	0.2626 to 46.67	Yes	*	0.0452
2016-02 vs. 2010-04	25.67	2.463 to 48.87	Yes	*	0.0186
2016-01 vs. 2010-06	-29.07	-52.27 to -5.863	Yes	**	0.0043
2017-01 vs. 2010-06	-27.13	-50.34 to -3.929	Yes	**	0.01
2017-02 vs. 2010-06	-27.27	-50.47 to -4.063	Yes	**	0.0094
2018-01 T vs. 2010-06	-43.13	-66.34 to -19.93	Yes	****	<0.0001
2018-01 R vs. 2010-06	-40.2	-63.40 to -17.00	Yes	****	<0.0001
2018-02 vs. 2010-06	-35.6	-58.80 to -12.40	Yes	***	0.0002
2019-01 vs. 2010-06	-39.91	-63.45 to -16.37	Yes	****	<0.0001
2019-02 T vs. 2010-06	-40.73	-66.68 to -14.79	Yes	***	0.0001
2019-02 R vs. 2010-06	-34.83	-60.78 to -8.890	Yes	**	0.0017
Grazed vs. 2010-06	-39.23	-65.45 to -13.00	Yes	***	0.0003
2018-01 T vs. 2010-08	-32.33	-55.54 to -9.129	Yes	***	0.001
2018-01 R vs. 2010-08	-29.4	-52.60 to -6.196	Yes	**	0.0037
2018-02 vs. 2010-08	-24.8	-48.00 to -1.596	Yes	*	0.0265
2019-01 vs. 2010-08	-29.11	-52.65 to -5.567	Yes	**	0.0051
2019-02 T vs. 2010-08	-29.93	-55.88 to -3.990	Yes	*	0.0116
Grazed vs. 2010-08	-28.43	-54.65 to -2.205	Yes	*	0.0229
2018-01 T vs. 2013-01	-36.93	-60.14 to -13.73	Yes	***	0.0001
2018-01 R vs. 2013-01	-34	-57.20 to -10.80	Yes	***	0.0005
2018-02 vs. 2013-01	-29.4	-52.60 to -6.196	Yes	**	0.0037
2019-01 vs. 2013-01	-33.71	-57.25 to -10.17	Yes	***	0.0006
2019-02 T vs. 2013-01	-34.53	-60.48 to -8.590	Yes	**	0.0019
2019-02 R vs. 2013-01	-28.63	-54.58 to -2.690	Yes	*	0.019
Grazed vs. 2013-01	-33.03	-59.25 to -6.805	Yes	**	0.004
2016-02 vs. 2016-01	31.27	8.063 to 54.47	Yes	**	0.0016
Donor vs. 2016-01	23.67	0.4626 to 46.87	Yes	*	0.0418
2017-01 vs. 2016-02	-29.33	-52.54 to -6.129	Yes	**	0.0038
2017-02 vs. 2016-02	-29.47	-52.67 to -6.263	Yes	**	0.0036
2018-01 T vs. 2016-02	-45.33	-68.54 to -22.13	Yes	****	<0.0001
2018-01 R vs. 2016-02	-42.4	-65.60 to -19.20	Yes	****	<0.0001
2018-02 vs. 2016-02	-37.8	-61.00 to -14.60	Yes	****	<0.0001
2019-01 vs. 2016-02	-42.11	-65.65 to -18.57	Yes	****	<0.0001
2019-02 T vs. 2016-02	-42.93	-68.88 to -16.99	Yes	****	<0.0001
2019-02 R vs. 2016-02	-37.03	-62.98 to -11.09	Yes	***	0.0007
Grazed vs. 2016-02	-41.43	-67.65 to -15.20	Yes	***	0.0001
Donor vs. 2018-01 T	37.73	14.53 to 60.94	Yes	****	<0.0001
Donor vs. 2018-01 R	34.8	11.60 to 58.00	Yes	***	0.0003
Donor vs. 2018-02	30.2	6.996 to 53.40	Yes	**	0.0026
Donor vs. 2019-01	34.51	10.97 to 58.05	Yes	***	0.0005
Donor vs. 2019-02 T	35.33	9.390 to 61.28	Yes	**	0.0014

Donor vs. 2019-02 R	29.43	3.490 to 55.38	Yes	*	0.0141
Donor vs. Grazed	33.83	7.605 to 60.05	Yes	**	0.0029

Table A3 Tukey's multiple comparison test for significant difference in deposition rates by location in week one.

Tukey's multiple comparisons test	Mean Diff.	95.00% CI of diff.	Significant?	Summary	Adjusted P Value
2010-05 vs. 2010-08	-2.765	-54.95 to 49.42	No	ns	>0.9999
2010-05 vs. 2016-02	-15.34	-67.53 to 36.85	No	ns	0.9251
2010-05 vs. 2017-02	-9.441	-83.24 to 64.36	No	ns	0.9989
2010-05 vs. 2019-01	-63.11	-121.5 to -4.769	Yes	*	0.0321
2010-05 vs. Denuded 1	-80.05	-132.2 to -27.86	Yes	**	0.0031
2010-05 vs. Denuded 2	-40.62	-98.97 to 17.72	No	ns	0.2461
2010-08 vs. 2016-02	-12.57	-64.76 to 39.61	No	ns	0.969
2010-08 vs. 2017-02	-6.675	-80.48 to 67.13	No	ns	0.9998
2010-08 vs. 2019-01	-60.35	-118.7 to -2.003	Yes	*	0.0415
2010-08 vs. Denuded 1	-77.28	-129.5 to -25.10	Yes	**	0.004
2010-08 vs. Denuded 2	-37.86	-96.20 to 20.49	No	ns	0.3081
2016-02 vs. 2017-02	5.899	-67.90 to 79.70	No	ns	>0.9999
2016-02 vs. 2019-01	-47.78	-106.1 to 10.57	No	ns	0.1321
2016-02 vs. Denuded 1	-64.71	-116.9 to -12.52	Yes	*	0.0138
2016-02 vs. Denuded 2	-25.28	-83.63 to 33.06	No	ns	0.7024
2017-02 vs. 2019-01	-53.67	-132.0 to 24.61	No	ns	0.2589
2017-02 vs. Denuded 1	-70.61	-144.4 to 3.196	No	ns	0.0632
2017-02 vs. Denuded 2	-31.18	-109.5 to 47.10	No	ns	0.7693
2019-01 vs. Denuded 1	-16.93	-75.28 to 41.41	No	ns	0.929
2019-01 vs. Denuded 2	22.49	-41.42 to 86.41	No	ns	0.8487
Denuded 1 vs. Denuded 2	39.43	-18.92 to 97.77	No	ns	0.2716

Table A4 Tukey’s multiple comparison test for significant difference in deposition rates by location in week three.

Tukey's multiple comparisons test	Mean Diff.	95.00% CI of diff.	Significant?	Summary	Adjusted P Value
2010-05 vs. 2010-08	-3.251	-29.09 to 22.59	No	ns	>0.9999
2010-05 vs. 2016-02	-3.565	-29.41 to 22.28	No	ns	0.9998
2010-05 vs. 2017-02	-11.47	-37.31 to 14.38	No	ns	0.8028
2010-05 vs. 2018-02	-45.23	-71.07 to -19.39	Yes	***	0.0003
2010-05 vs. 2019-01	-52.28	-78.13 to -26.44	Yes	****	<0.0001
2010-05 vs. Denuded 1	-64.16	-90.00 to -38.32	Yes	****	<0.0001
2010-05 vs. Denuded 2	-56.14	-85.03 to -27.24	Yes	****	<0.0001
2010-05 vs. Denuded 3	-36.64	-65.53 to -7.750	Yes	**	0.0081
2010-08 vs. 2016-02	-0.3143	-26.16 to 25.53	No	ns	>0.9999
2010-08 vs. 2017-02	-8.216	-34.06 to 17.63	No	ns	0.9602
2010-08 vs. 2018-02	-41.98	-67.82 to -16.14	Yes	***	0.0007
2010-08 vs. 2019-01	-49.03	-74.88 to -23.19	Yes	***	0.0001
2010-08 vs. Denuded 1	-60.91	-86.75 to -35.07	Yes	****	<0.0001
2010-08 vs. Denuded 2	-52.88	-81.78 to -23.99	Yes	***	0.0002
2010-08 vs. Denuded 3	-33.39	-62.28 to -4.498	Yes	*	0.0175
2016-02 vs. 2017-02	-7.901	-33.74 to 17.94	No	ns	0.968
2016-02 vs. 2018-02	-41.67	-67.51 to -15.82	Yes	***	0.0008
2016-02 vs. 2019-01	-48.72	-74.56 to -22.88	Yes	***	0.0001
2016-02 vs. Denuded 1	-60.6	-86.44 to -34.75	Yes	****	<0.0001
2016-02 vs. Denuded 2	-52.57	-81.46 to -23.68	Yes	***	0.0002
2016-02 vs. Denuded 3	-33.08	-61.97 to -4.184	Yes	*	0.0188
2017-02 vs. 2018-02	-33.76	-59.61 to -7.923	Yes	**	0.0062
2017-02 vs. 2019-01	-40.82	-66.66 to -14.98	Yes	***	0.001
2017-02 vs. Denuded 1	-52.69	-78.54 to -26.85	Yes	****	<0.0001
2017-02 vs. Denuded 2	-44.67	-73.56 to -15.78	Yes	**	0.0012
2017-02 vs. Denuded 3	-25.17	-54.07 to 3.717	No	ns	0.1144
2018-02 vs. 2019-01	-7.054	-32.90 to 18.79	No	ns	0.9836
2018-02 vs. Denuded 1	-18.93	-44.77 to 6.912	No	ns	0.2566
2018-02 vs. Denuded 2	-10.9	-39.80 to 17.99	No	ns	0.9033
2018-02 vs. Denuded 3	8.59	-20.30 to 37.48	No	ns	0.9727
2019-01 vs. Denuded 1	-11.88	-37.72 to 13.97	No	ns	0.7739
2019-01 vs. Denuded 2	-3.85	-32.74 to 25.04	No	ns	0.9999
2019-01 vs. Denuded 3	15.64	-13.25 to 44.54	No	ns	0.6075
Denuded 1 vs. Denuded 2	8.026	-20.87 to 36.92	No	ns	0.9817
Denuded 1 vs. Denuded 3	27.52	-1.373 to 56.41	No	ns	0.0683
Denuded 2 vs. Denuded 3	19.49	-12.16 to 51.14	No	ns	0.4538

Appendix B.

Northwest Bay tidal charts

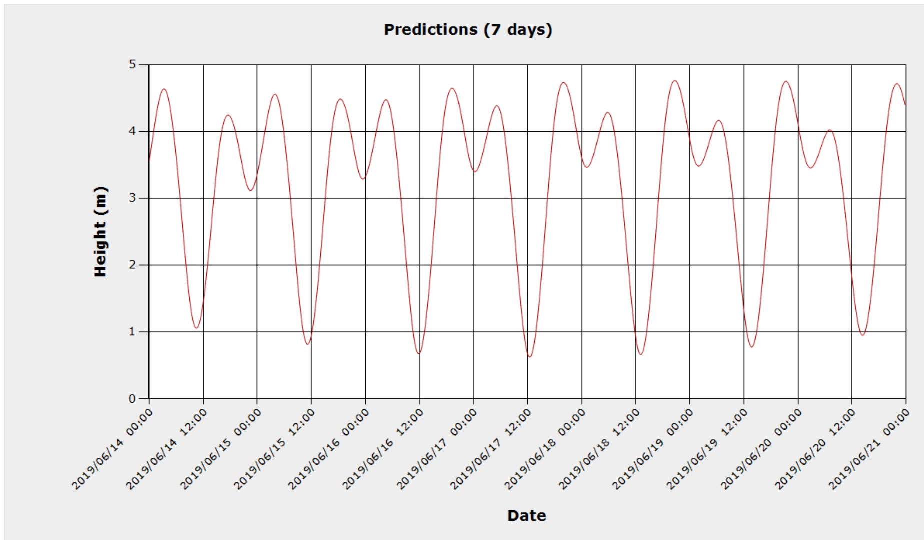

Figure B1 Northwest Bay tidal chart for the week of June 6th, 2019.

Source: Fisheries and Oceans Canada (2020)

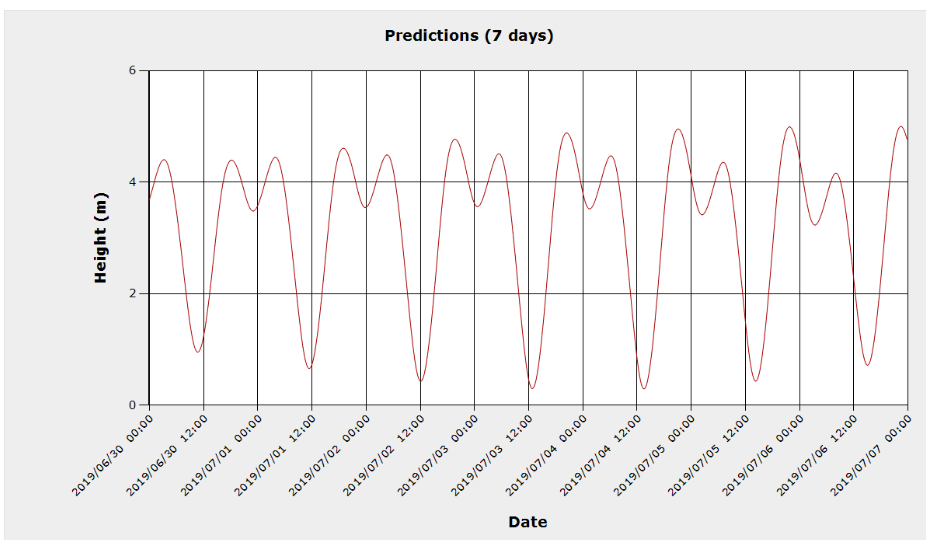

Figure B2 Northwest Bay Tidal chart for the week of June 30th, 2019.

Source: Fisheries and Oceans Canada (2020)

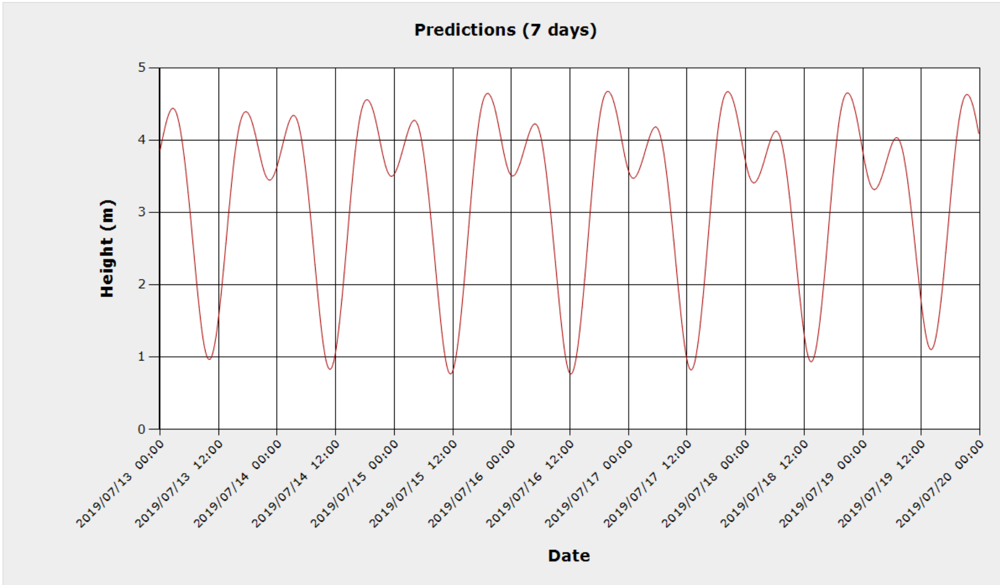

Figure B3 Northwest Bay Tidal chart for the week of July 13th, 2019.

Source: Fisheries and Oceans Canada (2020)