Restoring the Nanaimo River Estuary: vegetation and soil nutrient response to disturbance and treatment

by
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Approval

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Abstract

The Nanaimo River Estuary has been modified throughout the past century for agriculture, including the construction of embankments to limit tidal influence. In the summer of 2019, the embankments were removed. I carried out a series of vegetation surveys to compare impacted and natural regions of the estuary. I also collected soil cores from these regions and assessed them for soil organic matter content to determine the long-term effect of human impact from agriculture and embankment construction. Finally, I used Western Ag *Plant Root Simulators©* deployed before and after the embankment removal treatment to see if short-term changes in nutrient availability could be detected. Significant differences in both vegetation community and soil organic matter were found between impacted and non-impacted regions of the estuary. Soil nutrient results were unclear, although the technique shows promise as a way to measure near-term soil property changes after a restoration treatment.

Keywords: estuary; restoration; soil organic matter; soil nutrient availability; agriculture; de-embankment

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Background

The Nanaimo River estuary is the largest estuary on Vancouver Island, forming an important interface between freshwater, the ocean and nearshore environments by facilitating the exchange of sediments and nutrients (Hood, 2007). It is used by 58 species of fish and 200 species of birds (Catherine Berris Associates Inc., 2006). Unfortunately, the estuary has also been severely impacted by decades of human development, with a loss of over 50% of its historical area to conversion for agricultural land and settlement (Catherine Berris Associates Inc., 2006). It is also traditional land for the Snuneymuxw First Nation who still reside in the estuary today.

Throughout the past century, large sections of the Nanaimo estuary were diked for agriculture resulting in the loss of tidal flow and consequently, reduced soil moisture regime in the salt marsh (Armstrong, 1985). Disturbance to an ecosystem has long been known to influence the successional trajectory of communities (Watt, 1947), and plant communities in the salt marsh can be affected by changes in tidal inundation (De Jong & van der Pluijm, 1994). Un-impacted regions of the Nanaimo estuary marsh are defined by saline, saturated soils and flood twice daily. King tides and storm events provide the greatest inundation events.

Soil productivity refers to a soil's ability to support production of biomass and is dependent on both soil nutrient availability and soil fertility (Hillel and Hatfield, 2005). Soil fertility refers to quantitative measurement of *total* nutrients present in a soil while nutrient availability refers to the amount of soil nutrients in chemical forms accessible to plant roots (Fageria and Baligar, 2005).

Redox processes occur in any soil with plant cover and aerated soils use free oxygen as an electron acceptor in oxidation reactions (Van Breeman, 1988). Soil reduction processes change the availability of nutrients essential for plant function (Pezeshki, 2001). Waterlogged soils have a lower soil standard reduction potential (Eh) than dry soils (Reddy & Patrick Jr., 1975; Husson, 2013). Small scale elevation changes can affect tidal influence and the degree of waterlogging, enabling a more aerobic environment to persist (Armstrong, 1985; Pezeshki & DeLaune, 2012). Flooded soils in general have been found to have an increased cation exchange capacity (CEC) under anaerobic conditions particularly in the presence of clays (Favre et al., 2002).

The state of the redox environment in an estuary can alter the presence of porewater sulfides, ammonium, iron, phosphate mobilization, and toxic metal uptake (Portnoy and Giblin, 1997a; 1997b; Portnoy, 1999; Gambrell & Patrick, 1989). Drained salt marshes show a lower soil pH and increased dissolved iron, while re-flooding a salt marsh will display a lower redox potential, higher CEC, and affect the mobilization of NH_4, Fe(II) and PO_4 (Portnoy and Giblin, 1997b).

Saturation also influences both organic matter decomposition and nitrogen loss in soils (Reddy & Patrick Jr., 1975). Tidal marsh plants contribute to both soil organic matter and nutrients through decomposition of roots, rhizomes and stems (Gallagher and Plumley, 1979; Hackney and De la Cruz, 1980). *Juncus* species in particular can account for up to 12.4kg dry weight/m^2 of belowground biomass in a salt marsh (De la Cruz and Hackney, 1977) and are present in large monocultures in the Nanaimo River estuary.

It is important to study the role of plants in Pacific Northwest estuaries because of their influence on the physical features of this rare and unique landform. When marsh platforms flood, sheet flow over the marsh surface further sculpts the landform (Temmerman, 2005). Differences in type, height and density of vegetation that resides on the platform influence shear stress and erosional processes in the estuary (Leonard and Luther, 1995). Dense root systems also bind together the marsh sediments and help form deep-cut tidal channels featuring overhanging banks and low width to depth ratios (Garafalo, 1980, Chen et al., 2012). Some salt marsh species have been shown to be more effective at bank stabilization than others (Chen et al., 2019).

Historically, tidal channels provide marsh access for all five species of Pacific salmon in the Nanaimo Estuary (Catherine Berris Associates Inc., 2006). Visintainer et. al (2006) found that the presence of both high- and low-order tidal marsh channel systems improved survival of various fish species in addition to enhancing species richness, underscoring the importance of promoting channel formation in estuary restoration projects.

It is clear that there is a direct link between human modification, estuary soil properties and plant community composition (Portnoy et al., 1987, Warren and Niering, 1997; Pennington and Walters, 2006). It is also clear that land managers and restoration practitioners can manipulate these soil properties (intentionally or otherwise) through the construction or removal of berms in an estuary. Removing or re-introducing the tide to an estuary incites a cascade of biological and physical processes which affect marsh functioning **(Figure 1)**. If diking affects soil properties to a degree where the plant community changes, then the trajectory of future estuary ecology and morphology will also change. Monitoring estuarine vegetation change can demand a commitment of 18 months to 20+ years (Cordell et al., 1996; Morgan and Short, 2002; Williams and Orr, 2002). The need for a method of short-term evaluation of restoration success in estuaries is evident and may help restoration-focused organizations predict project outcomes more successfully.

Berm or dike removal has been proposed as a restoration treatment for estuaries in British Columbia and throughout the Pacific Northwest such as the Squamish River estuary (Squamish River Watershed Society, 2020). The treatment has been carried out in the Salmon River estuary and Skagit River estuary as part of on-going efforts to support local declining salmon populations (Gray et al., 2002; Furlong, 2017).

But is this enough to completely reverse the long-term impact of these embankments? Could there be unforeseen changes to soil properties which would affect vegetation and restoration management in the salt marsh? How can we monitor or predict these changes to support effective adaptive management?

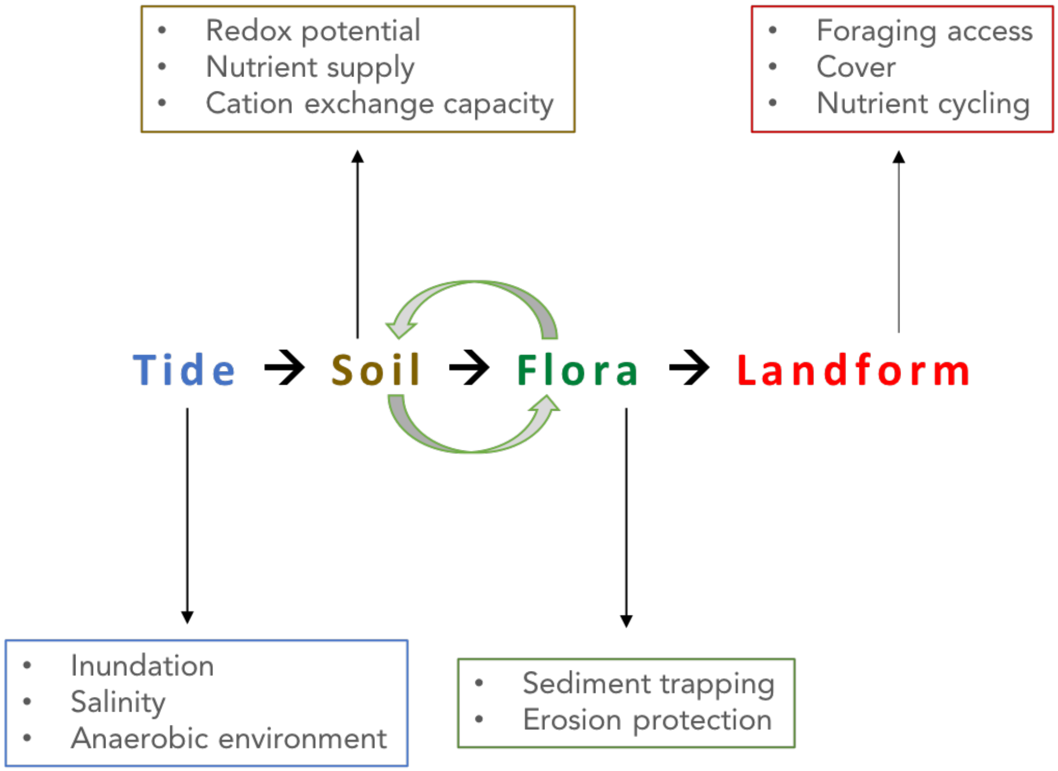

Figure 1 - Cascading effects on landform function when removing/re-introducing tidal influence in an estuary. Soil and flora form an additional feedback loop of processes.

Objectives

To answer the above questions, I assessed the plant community composition in both impacted and reference regions of the estuary. I also wanted to characterize the physical environment that determines the vegetation mosaic, and how the restoration treatment might affect that pattern.

Assessing the amount of organic material in both modified and un-impacted regions of the estuary will provide a long-term picture of the impact of the berms, and the role of vegetation in the process of soil formation.

I also wanted to see if short-term observation of soil nutrient change was possible, in order to provide restorationists with a rapid assessment of their results after a berm removal treatment. Developing an understanding of nutrient regime changes through soil testing in the marsh before and after restoration will provide a short-term look into the impact of de-embankment.

Studying the dynamics of estuaries can provide restoration practitioners with an additional piece of knowledge to predict the outcome of de-embankment treatments on Pacific Northwest estuary plants. This understanding will then present opportunities to manipulate soil nutrient properties in a way that governs the specific plants which grow on the restored estuary. Building on this knowledge in the future, targets can be set for precise species abundance, diversity and density which promote estuary health and resilience.

I hypothesized that un-impacted regions of the estuary would show greater plant species diversity than the regions which had been previously used for agriculture. I expected the impact regions to show poor soil organic matter accumulation due to the impacts of farming. I also predicted that the berm removal treatment would increase saturation in the impacted regions of the estuary and would be reflected in soil nutrient changes. Even though breaches had already been made in the berms, I expected to see a generally wetter environment as water would be able to move more freely through the estuary, particularly during king tides and storm events over the course of three months following the treatment. This would then manifest changes in nutrient availability in regions of the estuary formerly confined by embankments.

Methods and Study Location

Study Areas

The original inhabitants of the area were the Snuneymuxw First Nation who used the estuary for hunting and fishing and still reside on the banks of the river today. The Nanaimo River Estuary covers about 1000 ha and is currently managed by the Nature Trust of British Columbia.

The Nanaimo estuary is dominated by *Juncus gerardii*, in addition to numerous other salt marsh plants described below. Simple field tests indicated soils in the salt marsh are high in salinity, well-saturated and fine-textured with the presence of clays in all areas of the estuary. The estuary features dendritic networks of tidal channels, characteristic of tidally-dominated estuary formations on the west coast of North America.

During the early 1900’s berms and dikes were constructed to keep the tide out of the productive estuary soils, and further embankments were constructed in the 1940’s. These activities took place primarily in the region of the estuary on the east side of the east branch of the Nanaimo River. The presence of agricultural plots was confirmed from aerial photos from 1951 through to 1975 **(Figure 2)**. By 1982, farming appears to have ceased. Some parts of Oak Island (particularly south) were also embanked and farmed.

In the late-1980’s, some breaches were made in the berms, and tidal channels rapidly formed to adjust to the change in daily tidal forcing (P. Koenig, personal communication, July 2019). The complete berm removal took place from July 30th to August 2nd, 2019 when a total of 1.96 km of embankment was removed. The material was then spread out, flattened, and placed in drainage ditches which ran previously alongside the berms.

Figure 2 - Aerial photo from 1951 showing agricultural activity in the Nanaimo estuary (Nature Trust BC).

The Nanaimo River estuary study site was divided into four areas based on their historical use and intended future trajectory. Inside of each area, the longest continuous tidal channel branch was selected as the focal point. These channels are illustrated in **Figure 3**.

Site A was located on Oak Island, an isolated portion on the west side of the Nanaimo River east branch. This site is considered to be the reference site for this project, as no berms were ever constructed here, and the area was never used for agriculture.

Sites B, C and D are located on the east side of the Nanaimo River east branch. These sites are in the impact zone. Sites B and D in particular are located behind berms constructed prior to 1950. Site C is unique in that it is located on the sea-ward side of the berms. While not directly affected by tidal restriction, it is directly adjacent to the impact site.

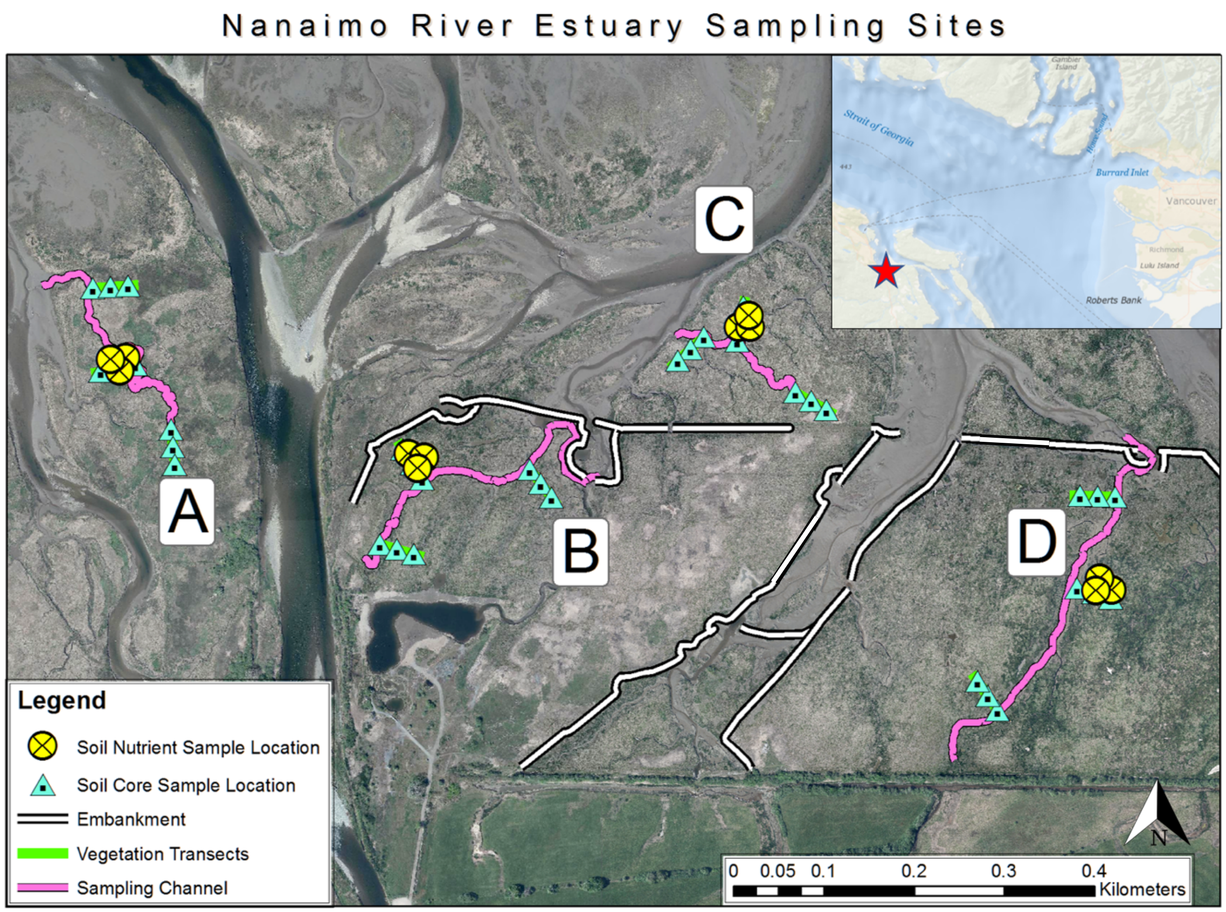

Figure 3 – Four study sites at the Nanaimo River Estuary Conservation Area. The Oak Island reference site is labelled A. Sites B and D were behind berms and C was in front. Vegetation transects align with soil core sample locations. The embankments on the map were removed between July 30th and August 2nd, 2019.

Vegetation Survey

For each selected channel, three 50m vegetation transect surveys were performed. The transects were located near the open end, mid-section and blind end of each channel, perpendicular to the banks. Where possible, the blind end transects continued in the direction of channel evolution. In the case of sites B and D, this was not possible due to obstruction, and a perpendicular survey was performed instead for the blind end.

At one-meter intervals along the transect, the plant species under a measuring tape was recorded along with its height. In cases where multiple species existed, the dominant species was selected. Using a LiDAR digital elevation model contributed by The Nature

Trust of BC, the elevation above sea level was recorded for each of the species observed. Elevation data was not available for Oak Island, as only the disturbed region of the estuary was scanned.

The Shannon-Weiner diversity index, Simpson diversity index, evenness and Berger-Parker Dominance for each transect was then calculated and total species diversity was recorded. These indices allow for an examination of all species present as well as their relative abundance in each area.

Soil Nutrient Sampling

While any changes in total soil nutrient content may not be observable for some time after the berm removal, changes in soil nutrient availability should be rapidly evident (Patrick Jr. & DeLaune, 1977). As tidal flow is restored, the rhizosphere will be significantly wetter due to inundation by high tides. This will create anaerobic conditions in the soil, changing the total availability of these nutrients for plant uptake (Favre et al., 2002).

Using Western Ag Plant Root Simulators©, we are able to measure the nutrient transfer rate to plant roots. These soil probes contain ion exchange resin membranes which adsorb soil cations and anions at a rate which depends on their activity and diffusion rate in the soil solution. Greer and Schoenau (1994) found a direct relationship between PRS Probe supply rate and actual plant uptake in saline soils.

In order to capture any changes in nutrient availability due to the berm removal, two rounds of nutrient sampling were carried out. The first deployment occurred before the berm removal, from July 25th, 2019 and July 31st, 2019. The second deployment occurred after the berm removal from October 27th, 2019 to November 2nd, 2019.

Near the middle transect of each selected channel, three soil nutrient samples were taken. Each sample consisted of three pairs of Plant Root Simulator© probes (one cation and one anion probe) and analyzed by the Western Ag lab as a composite sample. The probes were buried at 10cm depth and remained in the ground for exactly one week. Upon extraction, the probes were cleaned on-site with de-ionized water to ensure no further ions were taken up before analysis. The probes were then sent to the lab at Western Ag for a complete assessment of NO_3-N, NH_4+N, P, K, S, Ca, Mg, Al, Fe, Mn, Cu, Zn, B, Pb & Cd supply rate.

The data provided by the analysis is given in units of μg/10cm^2/7 days. This value represents the mass of each ion adsorbed by the 10cm^2 gel membrane surface over the seven-day deployment period. Because the Plant Root Simulators© initially adsorb ions at a higher rate during their initial days of deployment and slow with time (non-linear adsorption), a single per-day rate cannot not be calculated.

Typically, Plant Root Simulators © are used to test differences in treatments over the same time period. Less frequently they are used in two different time windows, such as in this study. Since factors such as temperature, precipitation and flooding will all affect nutrient adsorption rates, determining actual changes in nutrient availability is not as simple as comparing the exact values returned by the probes. **Table 7** in the results section summarizes the different environmental factors during the two deployments.

To properly assess effects of the treatment on soil nutrient availability, we will be looking for relative differences between the four study areas. An ideal comparison would have required both rounds of sampling to occur in identical environmental and weather conditions. Unfortunately, this was not possible due to project time constraints. Still, the change in weather would have been applied equally to all four sampling sites. To determine an effect from the treatment, I am looking for a signal that indicates some sort of change to have occurred at the two treatment sites (B and D), but not at the reference sites (A and C).

Soil Organics Assessment

Pilot soil cores revealed an obvious portion of the visible organic matter in the clay-rich soils of the estuary were plant roots and rhizomes (both living and dead).

Loss on ignition (LOI) is a method to calculate percent organic matter by comparing the weight of a soil sample before and after it has been exposed to high heat. Before ignition, the sample contains organic matter, and post-ignition all that remains is the mineral portion of the soil. LOI was found by Ball (1964) to be a simple and rapid method for assessing soil organic matter with an acceptable accuracy for ecological studies. Dean Jr. (1974) considered the method to be comparable in accuracy to dilute acid leaching and gas chromatography, both of which are more expensive, resource intensive and time consuming. Craft et al. (1991) found when the method was used in estuarine salt marsh soils, LOI was an 'excellent' estimator of organic carbon (R^2=0.990). It is important to note that results from this test will produce an estimate of soil organic matter (SOM) as opposed to total organic carbon, a commonly evaluated soil property. Some conversion factors have been proposed, but they can be highly dependent on other soil properties, such as texture (Craft et al., 1991; Pribyl, 2010).

In the field, three 5cm x 30cm soil cores were pulled from along each of the three transects at each channel. The cores were pulled at evenly spaced intervals, with some randomness added by throwing a flag into the air and sampling where it landed.

To analyze the cores, each sample was split into upper and lower halves with the upper portion representing a 0-15cm surface sample and the lower portion representing a 15-30cm depth sub-surface sample. From the top and bottom half of each core, a small 30-60g subsample was removed. The subsample was dried in a Quincy Lab Model 40 Lab Oven at 105°C for 24 hours, then ground to a fine powder to ensure homogeneity of the sample to be ignited. Before grinding, any large roots and rocks were removed. A 5-gram sample of ground material was then weighed using a milligram balance.

The samples were then placed in a Thermolyne™ FB1315M Benchtop Muffle Furnace at 550°C for four hours. This exposure time has been found to be optimal to ensure all the organic material has combusted (Heiri, 2001). Quartz crucibles were used to load the samples in the muffle furnace, eliminating the need to dry and store clay crucibles in a desiccator. A post-burn weight was then recorded, and the difference calculated to determine the percent organic matter.

The mass of organic matter in this method includes both decomposed organic material and 'living' organic material such as roots and rhizomes. It is important to note that results from this test will produce an estimate of soil organic matter (SOM) as opposed to total organic carbon, a commonly measured soil property.

Statistical Analysis

The data were analyzed using R Studio version 1.2.1335 for Mac OS X. The following packages were used in the analysis and graphing: car version 3.0-7, ggpubr version 0.2.5, ggplot version 3.3.0, ggstatsplot version 0.3.1, multcompview version 0.1-8, nortest version 1.0-4, plyr version 1.8.6, and Rmisc version 1.5.

Significance was set to $\alpha = 0.05$ and no outliers were removed owing to the small sample size (Van Selst & Jolicoeur, 1994). Shapiro-Wilk Normality tests were used to check the soil organics data for assumptions of normality, as well as to check the residuals from linear regression analysis. Bartlett's test was used to check for homogeneity of variance. A one-way ANOVA was used to check for significant differences between sites, and Tukey's range test was used to group the findings by significant difference. The Student's t-test was used to compare the impact and non-impact data sets.

Results

Vegetation Survey

In total, 14 different plant species were encountered. By far the most dominant species was *Juncus gerardii*, accounting for 52.96% of all observations. See **Table 1** and **Figure 3** below for complete results.

Table 1 - Observation frequency of all 14 species found during vegetation transect surveys conducted from mid-to-late June 2019.

Common Name	Scientific Name	Total Observations	Relative abundance
Mud rush	*Juncus gerardii*	367	52.96%
Seashore saltgrass	*Distichilis spicata*	124	17.89%
Patent saltbush	*Atriplex patens*	50	7.22%
Couch grass	*Elymus repens*	39	5.63%
Sea asparagus	*Salicornia pacifica*	34	4.91%
Blue wildrye	*Elymus glaucus*	21	3.03%
Yarrow	*Achellia millefolium*	16	2.31%
Tall fescue	*Festuca arundinacea*	10	1.44%
Puget Sound gumweed	*Grindelia integrifolia*	8	1.15%
Sea milkwort	*Lysimachia maritima*	6	0.87%
European beachgrass	*Ammophilia arenaria*	4	0.58%
Silverweed	*Potentilla anserina*	4	0.58%
Seaside Arrowgrass	*Triglochin maritima*	3	0.43%
Baltic rush	*Juncus balticus*	1	0.14%
	N/A - Bare ground	6	0.87%

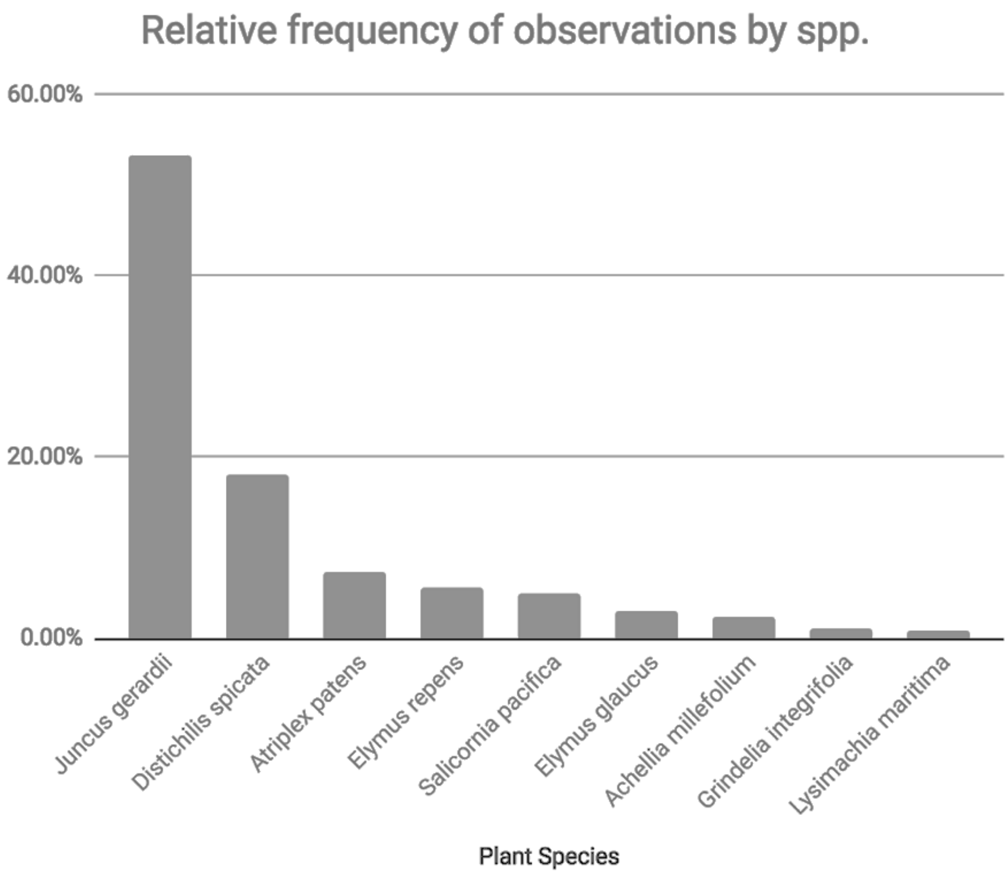

Figure 4 – Relative abundance of plant species in the Nanaimo River Estuary. Species with less than 5 observations are not included.

Species Elevation Profiles

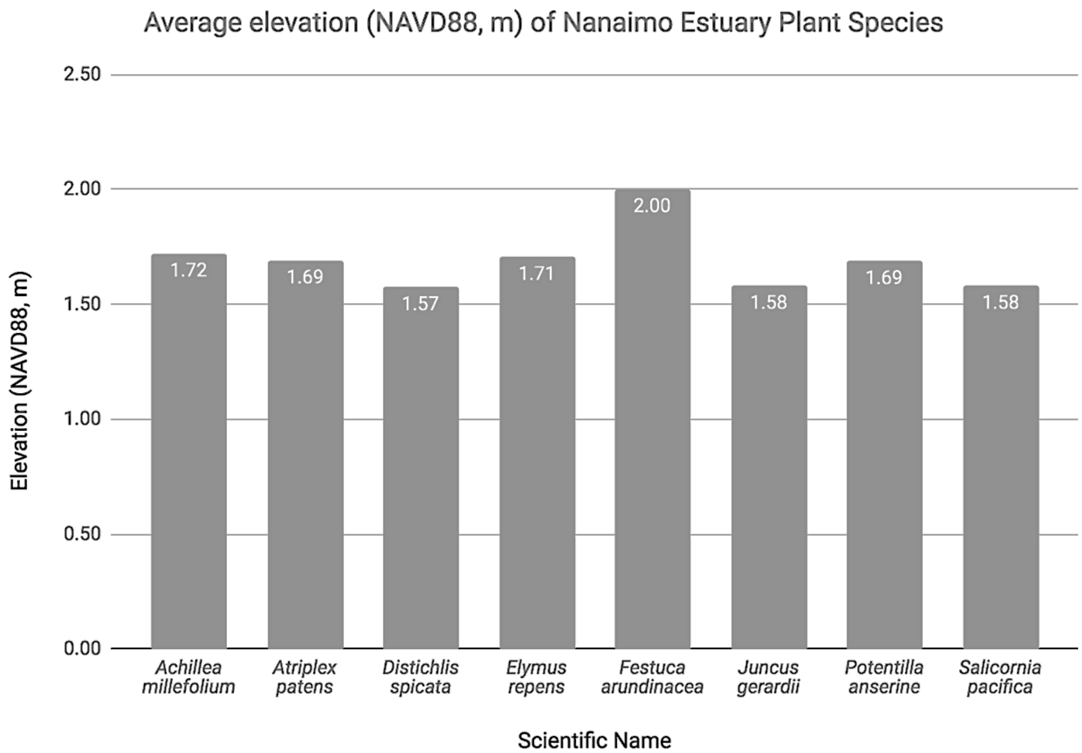

Figure 5 - Average elevation in meters for the eight species found in the eastern Nanaimo River Estuary. LiDAR data was not available for the Oak Island survey site.

The above eight species were present in regions of the estuary where LiDAR elevation data was available. The elevation is presented as North American Vertical Datum of 1988 (NAVD88) orthometric heights. The lowest and highest elevation for each species in the Nanaimo Estuary is presented below in **Table 2**.

Table 2 - Lowest and highest recorded elevation of eight plant species surveyed in the estuary.

Common Name	Scientific Name	Maximum Elevation (m)	Minimum Elevation (m)
Yarrow	*Achellia millefolium*	1.77	1.67
Patent saltbush	*Atriplex patens*	1.81	1.53
Seashore saltgrass	*Distichilis spicata*	1.95	1.13
Couch grass	*Elymus repens*	1.71	1.71
Tall fescue	*Festuca arundinacea*	2.09	1.96
Mud rush	*Juncus gerardii*	1.84	1.13
Silverweed	*Potentilla anserina*	1.70	1.67
Sea asparagus	*Salicornia pacifica*	1.78	1.31

Species Diversity by Area

At each transect, the species richness, Shannon Index, Simpson Index, Evenness and Berger-Parker Dominance were calculated (**Table 3**). The indices were also calculated for each of the four study areas by performing the same calculations with a combined dataset from each of the three transects.

Oak Island, the reference (Site A), was found to have the highest species diversity and most even distribution of species. It also was found to be the site that was least dominated by a single species.

The sea-ward high-marsh (Site C) and the East Berm (Site D) were found to be the least diverse sites and the most dominated by a single species (*Juncus gerardii*). For all surveys except for the Oak Island middle and Oak Island open end transects, *J. gerardii* was the most common species.

Elevation was not a significant predictor of Simpson Diversity Index ($p = 0.974$), although I did not have access to the elevation data from Oak Island.

Table 3 - Total plant species richness and vegetation diversity indices calculated for each study site and transect.

	Species Richness	Shannon Index	Simpson Index	Evenness	Berger-Parker Dominance
Oak Island (A)	**21 (Σ)**	**1.9920**	**0.1676**	**0.7766**	**0.2680**
Blind end	3	0.9971	0.3929	0.9076	0.5490
Middle	7	1.5750	0.2267	0.8094	0.3137
Open end	11	2.1000	0.1294	0.8758	0.2549
West Berm (B)	**15 (Σ)**	**1.3530**	**0.3297**	**0.6953**	**0.5163**
Blind end	4	1.1300	0.3906	0.8150	0.5882
Middle	7	1.5400	0.2698	0.7916	0.4706
Open end	4	1.2160	0.3263	0.8773	0.4902
Sea-ward Marsh (C)	**11 (Σ)**	**0.7792**	**0.6147**	**0.4841**	**0.7712**
Blind end	5	1.1420	0.4078	0.7099	0.6078
Middle	2	0.3999	0.7584	0.5770	0.8627
Open end	4	0.5498	0.7200	0.3966	0.8431
East Berm (D)	**11 (Σ)**	**0.7652**	**0.5624**	**0.4754**	**0.7105**
Blind end	3	0.6583	0.5835	0.5992	0.7255
Middle	5	0.7543	0.6229	0.4687	0.7800
Open end	3	0.7857	0.4965	0.7151	0.6275

Soil Organics Assessment

A Shapiro-Wilk normality test confirmed both the **upper** 0-15cm ($W = 0.9497$, $p = 0.1023$) and **lower** 15-30cm ($W = 0.96014$, $p = 0.2171$) sample sets to be approximately normally distributed. Homogeneity of variance across sample sites was confirmed using Bartlett's test for the **upper** ($K^2 = 6.2147$, df = 3, $p = 0.102$) and **lower** ($K^2 = 3.0634$, df = 3, $p = 0.382$). This allowed the use of a one-way ANOVA to ($F = 6.097$, df = 3, $p = 0.00211$) to confirm that there are significant differences between the four sites. A post-hoc Tukey HSD test showed significant difference between only Site B (West Berm) and Site C (Sea-ward Marsh) for both **upper** and **lower** samples.

Between sites for the **upper** samples, Site C (Sea-ward marsh) averaged the highest percent organic matter (36.49 ± 6.07) followed by Site D (East Berm) (29.56 ± 8.53), then Site A (Oak Island) (24.74 ± 3.54) while Site B (West Berm) (18.97 ± 8.37) was the lowest. The results are summarized in **Table 4** below and **Figure 5**. Means with no grouping letter in common are significantly different (HSD-test, $\alpha = 0.05$).

Table 4 - Ranked and grouped average percent soil organic matter for upper (0-15cm) samples by site.

Rank	**Sample Site**	**Site Name**	**Avg. Upper (0-15cm) % Organic Matter**	**Tukey HSD Grouping**
1	C	Sea-ward Marsh	36.49 ± 6.07	b
2	D	Berm East	29.56 ± 8.53	ab
3	A	Oak Island	24.74 ± 3.54	a
4	B	Berm West	18.97 ± 8.37	a

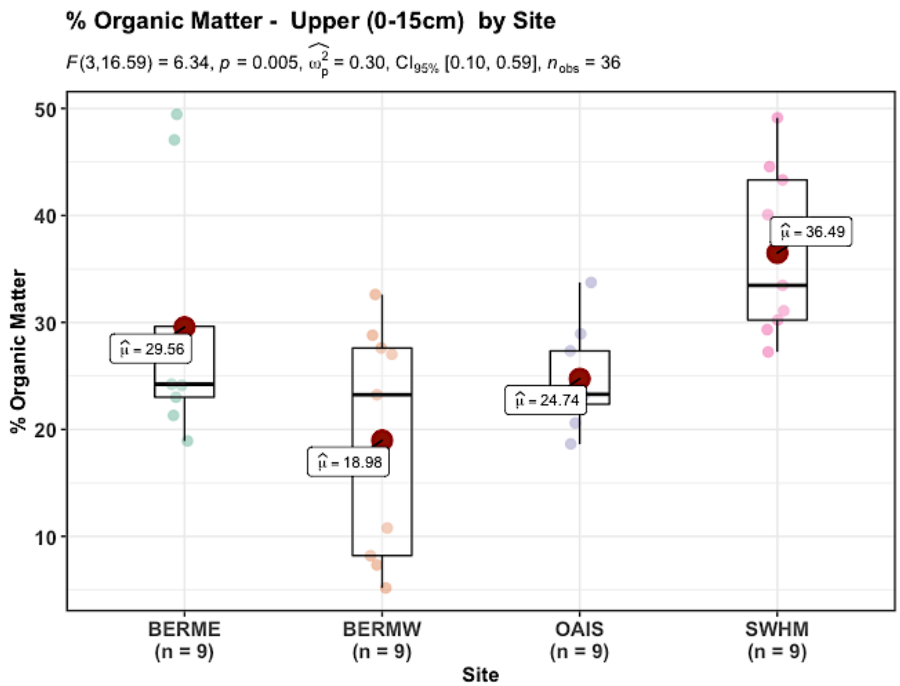

Figure 6 - Boxplot of upper (0-15cm) soil organic matter showing the highest average SOM at Site C (Sea-ward marsh) and the lowest at Site B (West Berm).

Between sites for the **lower** samples, Site C (Sea-ward marsh) again averaged the highest percent organic matter (23.74 ± 7.50) followed by Site A (Oak Island) (22.75 ± 3.54), then Site D (East Berm) (15.30 ± 4.23), and Site B (West Berm) (13.42 ± 6.30) was the lowest. The results are summarized in **Table 5** below and **Figure 6**.

Table 5 - Ranked and grouped average percent soil organic matter for lower (15-30cm) samples by site.

Rank	**Sample Site**	**Site Name**	**Avg. Lower (15-30cm) % Organic Matter**	**Tukey HSD Grouping**
1	C	Sea-ward Marsh	23.74 ± 7.50	b
2	A	Oak Island	22.75 ± 3.54	ab
3	D	Berm East	15.30 ± 4.23	ab
4	B	Berm West	13.42 ± 6.30	a

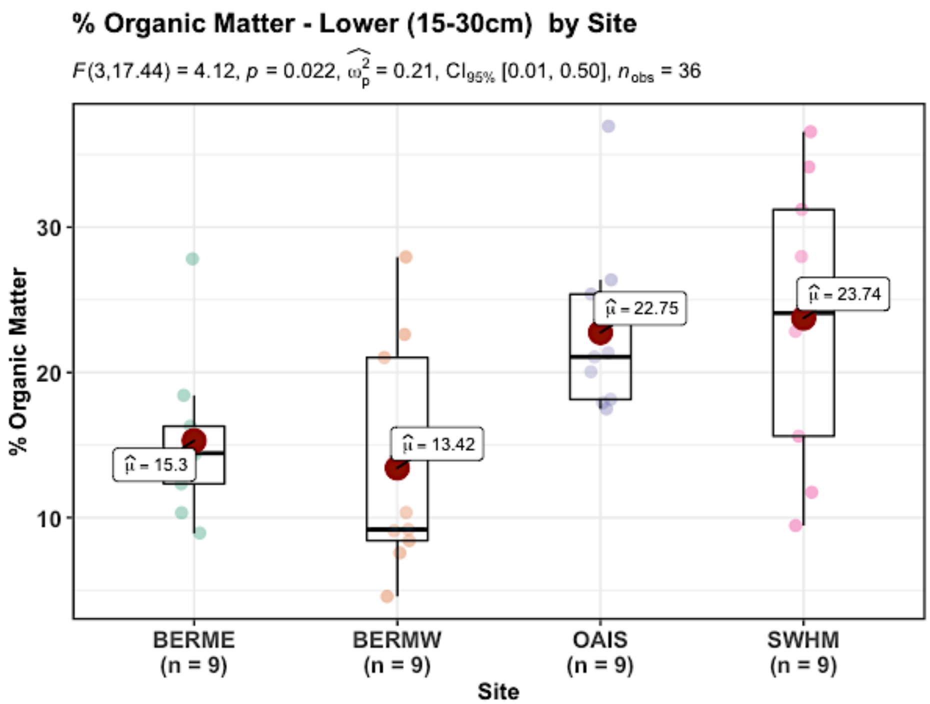

Figure 7 - Boxplot of lower (15-30cm) soil organic matter showing again the highest average SOM at Site C (Sea-ward marsh) and the lowest at Site B (West Berm).

Elevation was not found to be a significant predictor of organic matter in the upper ($p = 0.885$) or lower ($p = 0.965$) samples. Stem density was also not found to be a significant predictor of organic matter in both the upper ($p = 0.393$) and lower ($p = 0.243$) samples.

Soil organic matter was determined to be higher in the **upper** half (27.44% ± 3.66) of the soil cores than the **lower** half (18.80% ± 2.90) when compared across the estuary (paired t-test, $t = 4.6864$, 35 d.f., $p < 0.001$).

No significant difference was found for organic matter in the **upper** samples between non-impact sites (30.62 ± 4.34; n = 18) and sites which had been impacted by embankments (24.27% ± 5.96; n = 18) (two-sample t-test, $t = -1.819$, 34 d.f., $p = 0.0778$).

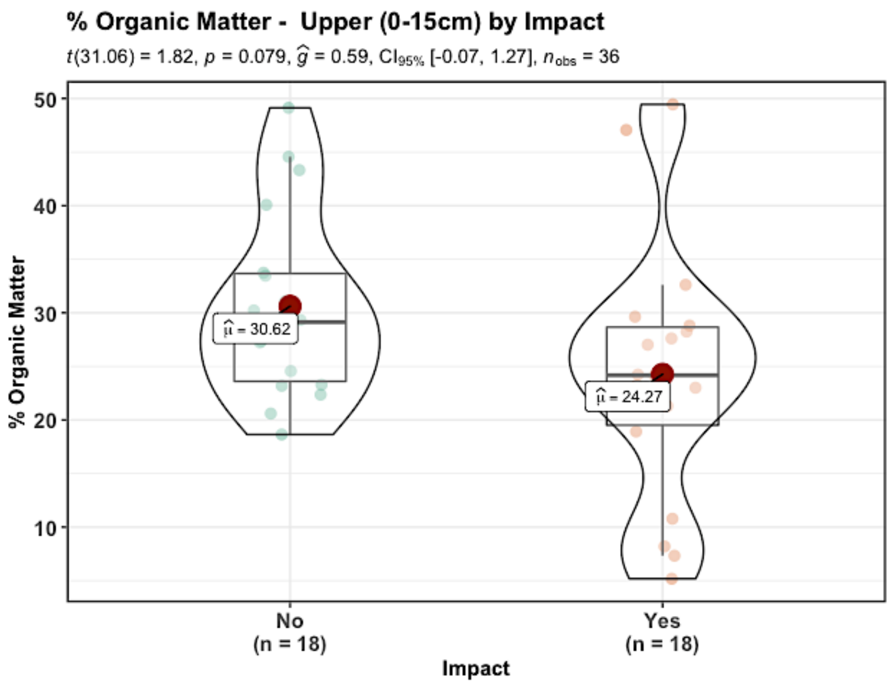

Figure 8 - Box/violin plot showing higher average soil organic matter in un-impacted sites for the upper (0-15cm) samples. Lines around the data show density of observations.

Organic matter in the lower samples was found to be greater in non-impact sites (23.24 ± 3.40) than at sites which had been impacted by berm construction (14.36% ± 3.95) (two-sample t-test, t = -3.594, 34 d.f., p = 0.001019).

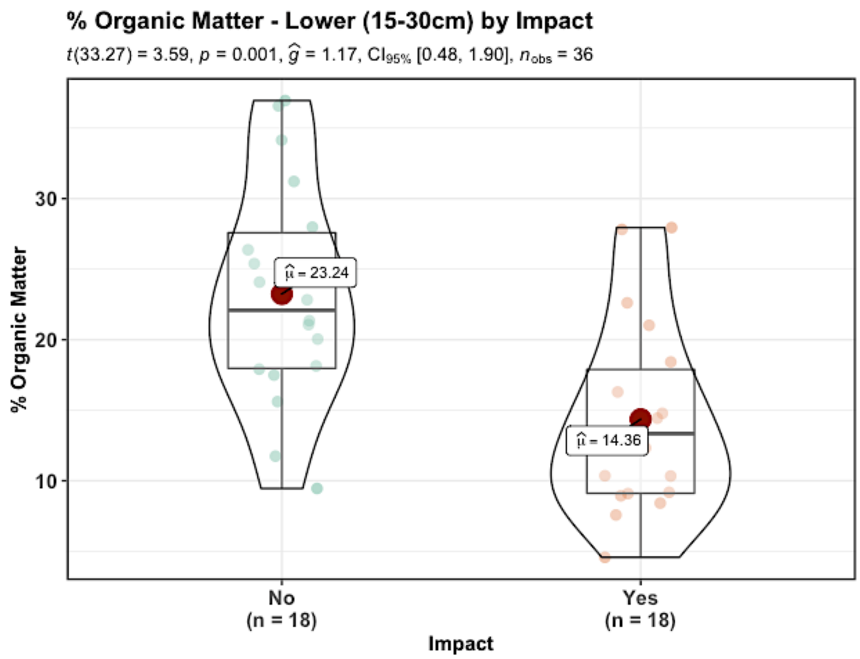

Figure 9 - Box/violin plot showing higher average soil organic matter in un-impacted sites for the lower (15-30cm) samples. Lines around the data show density of observations.

The Simpson index for each transect significantly predicted average organic matter content in the **upper** samples (b = 25.187, t = 2.962, p = 0.014) and also explained a significant proportion of variance in **upper** organic matter (R^2 = 0.47, F(1,10) = 8.772, p = 0.014). A visualization of the relationship is shown in **Figure 9**. The Simpson index was not a significant predictor for the **lower** organic matter. Other diversity indices showed weaker correlation, therefore the investigation continued using Simpson index.

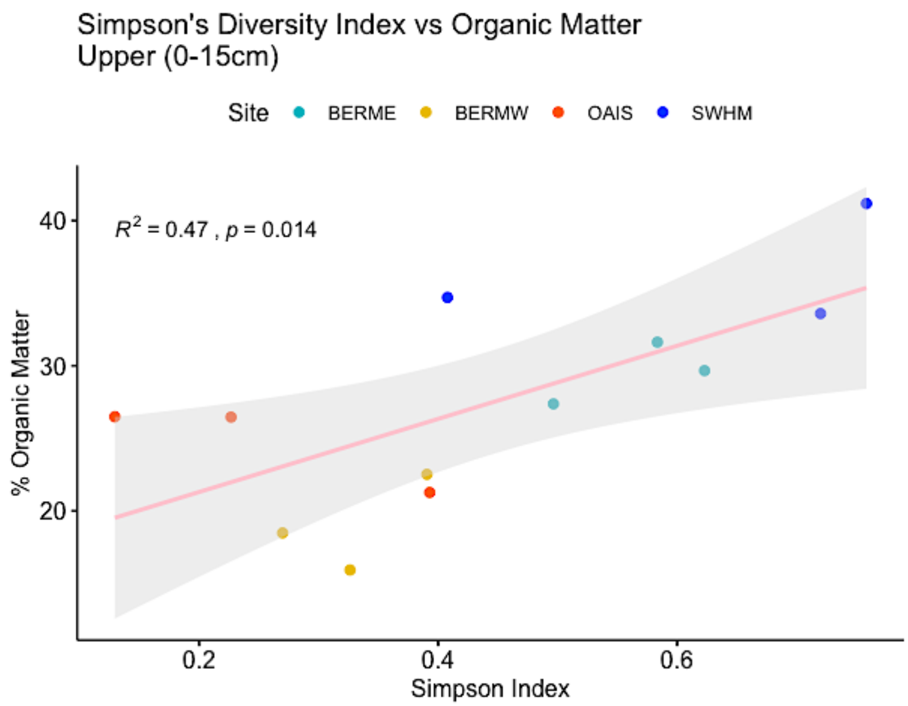

Figure 10 - Linear model of Simpson's Diversity Index as a predictor for percent soil organic matter in the upper (0-15cm) samples, with measured data plotted by sampling transect. It is important to note that a lower value for Simpson Index corresponds to increased diversity and a higher value means less diversity. Therefore, we are seeing an inverse relationship between diversity and organic matter.

To answer the question of whether vegetation diversity and impact affected organic matter, I decided to perform a multiple regression analysis using the Simpson index and presence of berms, without interaction. A multiple linear regression was calculated to predict organic matter throughout the entire 0-30cm profile and a significant regression equation was found ($R^2 = 0.7018$, $F(2,9) = 13.94$, $p = 0.001751$). When the individual predictors were inspected, Simpson Index was significant ($t = 3.392$, $p = 0.00797$) as well as impact ($t = -4.127$, $p = 0.00257$). A Shapiro-Wilk normality test showed the residuals to be normal ($W = 0.99019$, $p = 0.9998$). A visualization of the regression is shown below in **Figure 10**.

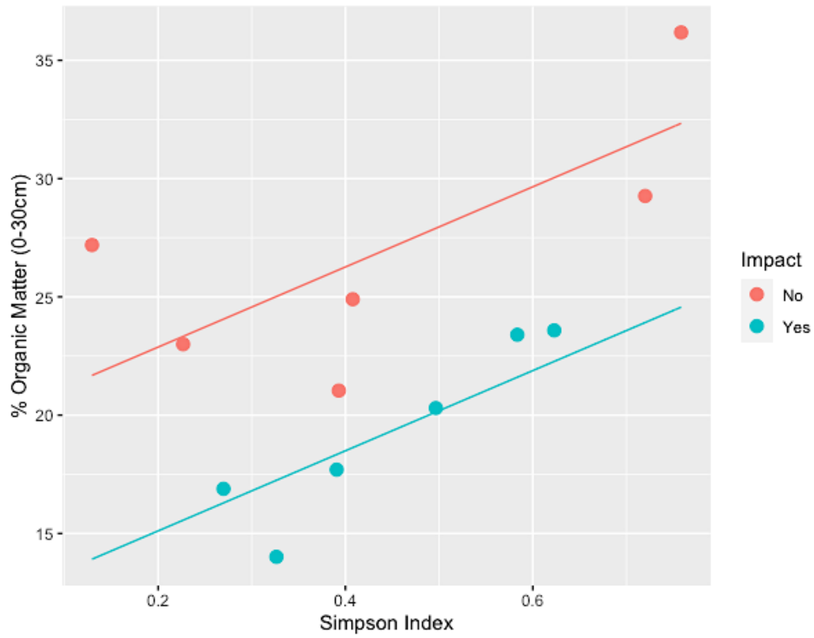

Figure 11 - Linear model predicting organic matter throughout the 0-30cm soil core using Simpson's Diversity Index and Impact as predictors (R^2 = 0.7018, F(2,9) = 13.94, p = 0.001751).

Since *Juncus gerardii* was the dominant plant in the estuary (and possibly exhibiting the most influence over the soil environment) we decided to test the whether the frequency of *J. gerardii* in a transect could predict organic matter. The frequency of *J. gerardii* alone did not significantly predict **upper** matter (p = 0.096) or **lower** organic matter (p = 0.9485) at α = 0.05 using a linear model. Interestingly, the data fit a polynomial regression equation quite well for both **upper (**R^2 = 0.6493 F(2,9) = 8.33, p = 0.00896) and **lower (**R^2 = 0.7775, F(2,9) = 15.73, p = 0.00116) organic matter, but it is unclear if this simply due to over-fitting caused by too small a sample size.

To investigate whether vegetation diversity and *Juncus gerardii* frequency had an effect on organic matter throughout the 0-30cm profile, a multiple linear regression without interaction was also calculated. A significant regression equation was computed (R^2 = 0.6544, F(2,9) = 11.41, p = 0.00339). The individual predictors Simpson index (t = 4.481, p = 0.00153) and *J. gerardii* (t = -3.670, p = 0.00516) were both significant. A Shapiro-Wilk normality test showed the residuals to be normal (W = 0.95248, p = 0.6735).

Soil Nutrient Availability

Environmental Variability

Below is a summary of the environmental variables collected using observations from Environment Canada's Nanaimo Airport weather station and Fisheries and Oceans Canada's Nanaimo (#7917) tidal observation station.

Table 6 - Differences in Environmental variables during the two deployments of Plant Root Simulators ©.

	'Before' Deployment Jul 25-Aug 1	**'After' Deployment Oct 27-Nov 3**
Precipitation	1.3 mm	Trace
Highest Tide	4.627 m	4.526 m
Lowest Tide	0.306 m	0.350 m
Mean Tide	3.067 m	2.884 m
Maximum Air Temperature	29.5°C	13.5°C
Minimum Air Temperature	11.5°C	-3.0°C
Mean Air Temperature	19.6°C	5.5°C

Macronutrients – Nitrogen, Phosphorus, Potassium

Ammonium results showed significant variation in the two treatment sites, with more precise results in the two reference sites (**Figure 11**). No noticeable pattern was detectable.

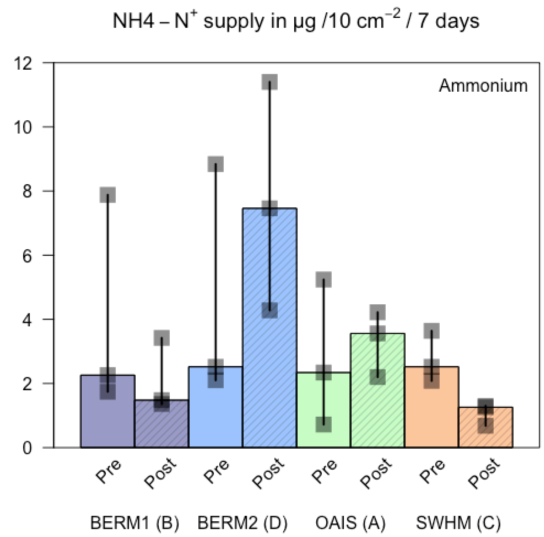

Figure 12 - Change in ammonium availability before and after treatment.

While there was a decrease in phosphide availability at all sites, there was a substantial decrease at the two treatment sites after the berm removal compared to the two reference sites (**Figure 12**).

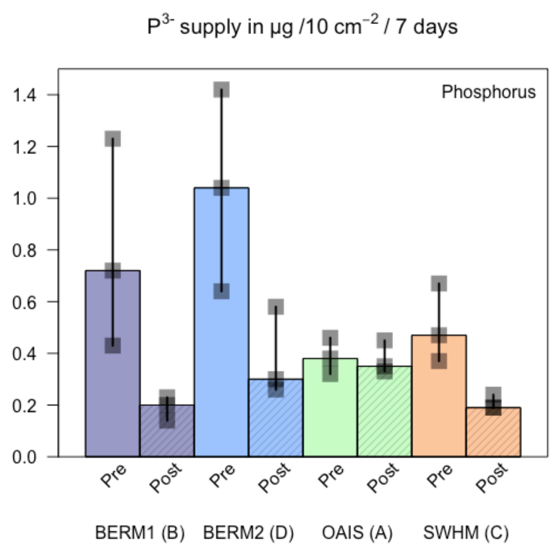

Figure 13 - Change in phosphide availability before and after treatment.

Potassium probe results indicated stable supply rates at all four sites both before and after the treatment. This set of results also showed the least variation among the soil nutrient data (**Figure 13**).

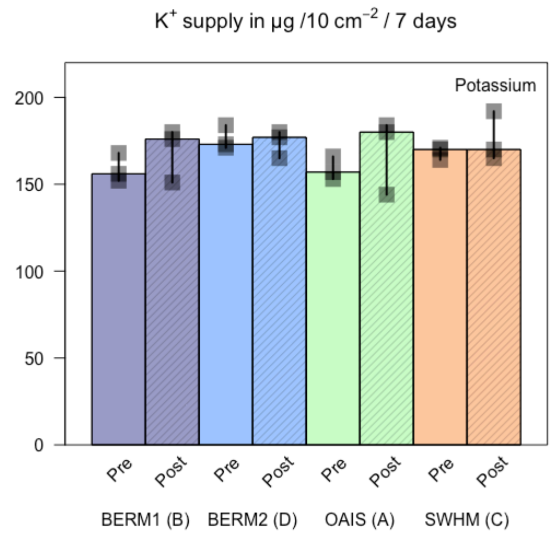

Figure 14 - Change in potassium cation availability before and after treatment.

Micronutrients – Copper, Iron, Sulphur

Copper probe analysis showed a decrease in availability in the treatment sites and an increase in the two reference sites (**Figure 14**). Most probes returned copper ion values below the advisable minimum detection level, and these results should be interpreted which caution. This set of results best displayed the signal I had hoped to see.

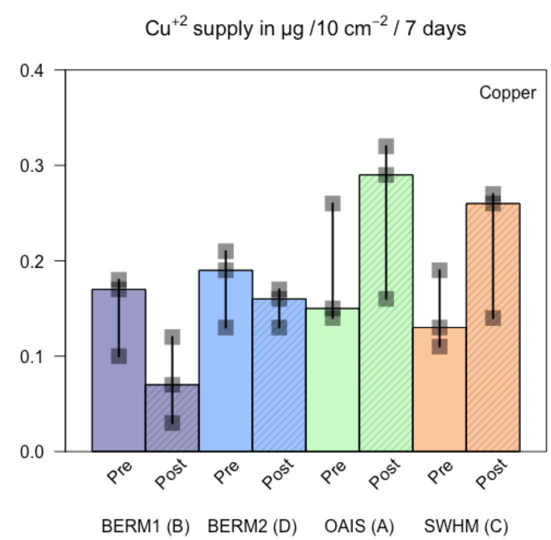

Figure 15 - Change in cupric ion availability before and after treatment.

Iron probes analysis showed minimal change at all sites, with some decrease at the impact sites (**Figure 15**). The two treatment sites showed much larger variability both before and after restoration compared to the reference sites.

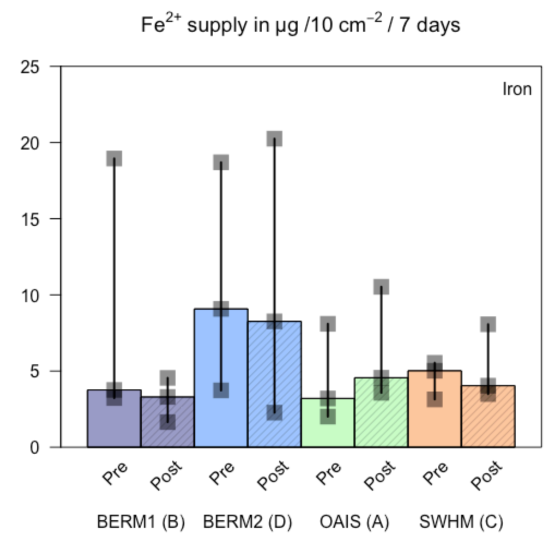

Figure 16 - Change in ferrous ion availability before and after treatment.

Sulphur nutrient supply showed a decrease at one of the treatment sites, and remained relatively unchanged at the other. The two reference sites both displayed an increase in sulphur availability post-treatment, indicating the berm removal may have had an effect on this nutrient (**Figure 16**).

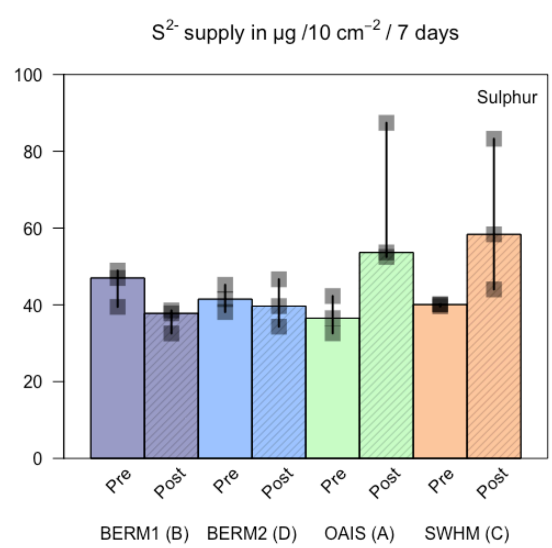

Figure 17 - Change in sulfide cation availability before and after treatment.

Aluminum, calcium, and magnesium exhibited minimal change between pre- and post-restoration at all sites, and will not be further evaluated. Boron, manganese, nitrate, and total nitrogen suffered from significant variability in the data and will not be interpreted further.

The probes returned no detectable Zinc in the estuary soils, but trace amounts of both lead and cadmium. The second round of sampling in particular returned the highest levels of lead, found on Oak Island.

Discussion

Vegetation Survey

Unsurprisingly, the only site that was not impacted by agriculture or embankments (Site A - Oak Island) was found to have the greatest diversity of plant species and the most even distribution of species. This suggests that physical habitat disturbance in Pacific Northwest estuaries can contribute to declining vegetation diversity and dominance by opportune species. This is further evidenced by the dominance of *Juncus gerardii* in particular, which is an exotic species (BC Conservation Data Centre, 2020). Bouzille et al., (1997) found *J. gerardii* to exhibit a logistic growth curve when colonizing disturbed saltmarshes, which explains the dominance of this plant in the former agricultural sites of the Nanaimo Estuary. After the disturbance to the salt marsh, it is likely that other species could not compete with the rapid growth rate of *J. gerardii*. Increased salinity was shown by Rozema and Blom (1977) to have no effect *J. gerardii* growth.

While Site C (Sea-ward High Marsh) was also not farmed or diked it showed the least diversity among species, and its various diversity indices were similar to Site D (East Berm). Even though not directly impacted, Site C is adjacent to an embanked area of the estuary and lies on the sea-ward side of the berms. The low diversity and dominance of *J. gerardii* implies an *indirect* disturbance from farming and diking can affect plant communities in regions of an estuary beyond the direct impact zone. Hood (2004) explored the indirect effects of dikes on estuarine tidal channel morphology and concluded that loss of tidal channels sea-ward of a dike can exceed the loss inside, with possible effects on plant species succession. My results also suggest this effect may extend to vegetation community composition.

Warren and Niering (1993) found that increased tidal flooding in a New England salt marsh caused *Juncus gerardii* vegetation to be replaced by forbs such as *Triglochin maritima* and *Spartina alterniflora*, the former being found in the Nanaimo River estuary. If this held true in the Namaimo estuary, we would expect to see *J. gerardii* in the formerly diked areas of the study site to change over time as well. The difference is that *J. gerardii* is considered to be native to the east coast of North America, while its presence in Nanaimo may be influenced more by disturbance and inundation time both have effects on the growth of *J. gerardii* (Rozema and Blom, 1977; Watson et al., 2015).

Eilers (1974) showed that salt marsh vegetation species diversity increased with elevation in an Oregon estuary. I was not able to find a similar pattern in the Nanaimo River Estuary, further supporting the idea that human impact played a significant role in the dominance of *J. gerardii*.

This information will also be useful for long-term tracking of any vegetation change, either due to the berm removal treatment, sea-level rise or any other climate change impacts.

Soil Organics Assessment

Converse to our results, Portnoy (1999) found the organic matter volume of a diked-waterlogged marsh to be higher than the natural marsh. This was attributed to the dikes physically blocking inorganic sediment normally supplied by flooding tides, therefore increasing the ratio of organic to inorganic material. The Nanaimo River estuary berms were breached in the 1980’s and this may have allowed additional infiltration of inorganic sediment, causing our results from impact sites to show lower percent organic matter. Weston et al., (2011) showed that increased salt-water intrusion significantly stimulated soil microbes to mineralize soil organic matter and accelerated the loss of organic carbon from soils. Similar to the above, this would also suggest that the reference sites would have been expected to show lower soil organic matter. I propose that the specific plant species composition present in the Nanaimo estuary was able to counter this effect through its substantial contributions to soil organic matter.

The two study sites in the estuary which displayed the greatest organic matter content in the upper half of soil cores were also the two sites with the highest Berger-Parker dominance index scores (Site C and D), and also the greatest abundance of the dominant species *(Juncus gerardii)*. This plant is capable of achieving a stem density higher than all other plants surveyed at the Nanaimo River Estuary and both of these two sites featured large monocultures of the plant. Stems also contribute organic matter to soil (Gallagher and Plumley, 1979) and my informal survey of *Juncus gerardii* density in the Nanaimo Estuary counted upward of 5000 stems/m^2, similar to numbers surveyed by Bouzille et al. (1997). As previously stated, *Juncus* species can contribute upwards of 12 kg/m^2 of belowground biomass to estuary soils (De la Cruz and Hackney, 1977).

Our results also suggest that *J. gerardii* contributes a significant amount of organic matter to this specific estuary's soils in the form of live and dead roots, particularly in the upper soil profile. De la Cruz and Hackney (1977) found that 94% of the halophytic *Juncus* species *J. roemerianus* biomass occurred in top 20cm of soil. Hackney and De la Cruz (1980) studied decomposition of roots and rhizomes of *J. roemerianus* and found the greatest decomposition rate to occur in the top 10cm of tidal marsh soil, with no measurable decomposition below 20cm depth. Chen et al., (2019) found root depth for *Juncus maritima* to reduce rapidly below 10cm depth, from 0.12g g^{-1} (gram of dry root mass per gram of dry sediment mass) to 0.04g g^{-1}. My results from the upper (0-15cm) soil samples would support this, particularly at the sea-ward marsh site which featured *J. gerardii* monocultures.

Still, our modelling showed a negative relationship between *J. gerardii* frequency for both the upper and lower soil layers when combined with Simpson index, implying that while *J. gerardii* contributes to SOM through decomposition, too much of the species alone may actually prevent SOM from accumulating to its full potential throughout the full 0-30cm soil column. Increasing diversity also limited SOM, suggesting there is a 'golden mean' of diversity in estuaries to maximize organic matter accumulation. Therefore, to achieve maximum SOM in a salt marsh, an optimal vegetation mosaic may consist of as few plant species as possible which produce plentiful roots and rhizomes, and well as a few additional species which can penetrate deeper into the soil horizon, regardless of their root density. This was demonstrated at the Oak Island site, which showed the most consistent organic matter throughout the 0-30cm soil cores (24.74% in the upper and 22.75% in the lower).

Groenendijk, A. M., & Vink-Lievaart (1987) found that the halophytic *Elymus* species *E. pycanthus* contributed 7.79kg/m^2 of belowground biomass in a salt marsh and that the majority of the production occurred in the 20-60cm root environment. This is substantially deeper than the depth of *Juncus* decomposition reported by Hackney and De la Cruz (1980). This could explain why the Oak Island site, which featured the highest occurrence of *Elymus* species, displayed a higher percentage of organic matter in the lower samples as well as the least difference between upper and lower samples. It also helps to explain why a diversity of plant species can increase organic matter to a degree, as some species will contribute more biomass to the upper soil profile, and other species contribute more in the lower profile.

Eisenhauer et al. (2013) showed that soil organic matter increased in soil with higher plant diversity due to improved diversity and abundance of soil microorganisms. This improvement in the soil's food web assisted in the breakdown of organic residues.

Hood (2004) suggests my classification of Site C as a non-impact site may have been a mis-representation of the degree of impact to the area. It is possible that being directly adjacent to the impact region may have affected the soil organic matter results. A re-classification of the site as an impact site, and the addition of more 'true' non-impact sites could have yielded better results.

Soil Nutrient Availability

Portnoy and Giblin (1997) suggested that reflooding a diked estuary would mobilize a large pulse of NH_4. While my results did show an increase in availability of ammonium at the East Berm site, a decrease was seen at the West Berm site. The two reference sites also showed differing ammonium mobilization, making these results difficult to interpret. A longer deployment or larger sample size may have improved the noise in the data.

Potassium mobility was shown by Kuchenbuch et al., (1986) to increase with increasing soil moisture. They also measured and showed an increased uptake in plant roots. My results showed very little (if any) change in potassium at all four sampling sites. The potassium results were some of the most reliable, with the smallest variation between all four sites, possibly suggesting that soil saturation did not markedly change between probe deployments.

Giblin and Howarth (1984) found that aeration increases the solubility of iron minerals and iron levels in soil porewater increases. From this, we would have expected to see a decrease in iron availability. Conversely, Shaheen et al. (2017) described higher iron concentrations under reducing conditions in both the Nile and Mississippi River estuaries. Our results showed a very slight decrease in available Fe(II) supporting the latter publication, but the sample size was too limited to make a confident conclusion. Rozema and Blom (1977) found an increase in iron in the root material of *Juncus gerardii* after increased inundation in a lab experiment while our plant root simulators showed a slight decrease in iron after the berm removal treatment. Whether this is due to less available iron in the soil solution or increased competition by other plants (leaving less iron for the probes) is unclear. It is worth noting that the increase in iron in the roots of *J. gerardii* was also associated with growth inhibition in the study.

Portnoy (1997) reported that restoring seawater to a diked marsh would cause porewater sulfides to increase. Gilblin and Howarth (1984) also found that with the influence of tidal flushing removed in a diked estuary, pyrite (FeS_2) formation increases in the following years and sulfide concentrations in porewater would then decrease in the long-term. My results showed the opposite effect in the short-term. A decrease in sulphide availability was detected at the restored sites with reintroduced flushing after restoration. If this effect could be confirmed in future research, it would suggest reduced stress on marsh macrophytes immediately after de-embankment, as soil sulfide has been shown to inhibit growth of these plants (Havill et al., 1985; Koch and Mendelssohn, 1990). Another possible explanation for my results is the previous breaches in the berms made in the late 1980's limited the influx of sulfides after the berm removal, as most of the re-inundation had already occurred.

Soils with high levels of organic carbon can display increased phosphorus availability due to the effects of organic matter on iron reduction (Willet, 1986). Reddy and Rao (1983) showed when flooding organic soils, a phosphorus release occurs due to mineralization of organic phosphorus. Turner and Haygarth (2001) showed re-wetting of soils increases soluble phosphorus after being released from the soil microbial mass. My results showed a decrease in available phosphate after the treatment, suggesting there was limited increased saturation occurring after the treatment. Again, this may have been due to previous breaches in the berms.

Copper ion results showed reduced availability after the restoration treatment, although some probes returned values below the manufacture's recommended minimum. Copper in agricultural soils can appear due to the use of pesticides, fungicides, industrial effluent and wastewater irrigation (Adrees et al., 2015) and is also generally toxic to plants due to interference with photosynthetic processes (Fernandes and Henriques, 1991). Shaheen et al. (2017) found lower copper mobilization in reducing condition initiated by sulfide precipitation, suggesting this may be the primary process for copper availability changes in the Nanaimo River Estuary.

Overall, the method appeared to capture changes in nutrient availability for *some* specific nutrients. To restate my objective, I was looking for a signal where some effect would be seen in the treatment areas, and either an opposite effect or no effect would be observed at the non-impact sites. This signal was only present in the iron, phosphorus, sulphur and copper, while the remaining 11 nutrients tested by the Western Ag labs did not show a similar signal. Whether this was due to poor sampling technique, a limited increase in marsh saturation or because of the complexity of redox processes in estuaries remains unknown.

Restoration and Management Implications

Hood (2004) published an important study with implications for restoration in the Nanaimo River Estuary. The study looked at the effects of dike construction and salt marsh conversion *beyond* the areas confined by dikes. The paper proposed that indirect impacts of diking can occur in sea-ward marshes outside of converted marsh land. Additionally, it was proposed that restoration treatments can benefit these areas as well. In my Nanaimo estuary study, the sea-ward marsh site (Site C) may very well have been an example of this phenomenon, adding to the body of evidence that drives the theory of 'thinking outside the dike'. At Site C, we saw a monoculture of an exotic species, even though the area was never farmed or diked. Unfortunately, there is no way to know how or when *J. gerardii* ended up in the Nanaimo estuary. The earliest documentation that could be found for this species in the estuary was published 1976 (Bell and Kallman, 1976).

De Jong and Pluijm (1994) noted that vegetation change four years after the construction of embankments in the Oosterschelde estuary was still in flux and had not stabilized. This reinforces the need to monitor vegetation change for multiple years after a modifying the tidal regime in an estuary.

One of the goals of my study was to evaluate the potential to use nutrient availability and Plant Root Simulators as short-term monitoring tools. While there may be limited potential for predicting vegetation change, the probes may be still useful to assess whether substantial inundation change has occurred after a berm removal. Permanently installed electrodes to measure redox potential in combination with soil nutrient probes could provide a more complete picture of changing estuary soil chemistry (Armstrong, 1985).

Kindscher and Tieszen (1998) showed that replanting of native grass species in a prairie restoration treatment did little to improve soil organic matter content. Our results suggest that invasive species, such as *J. gerardii* can actually improve SOM in estuaries, when the species is a large belowground biomass producer. Still, I would not necessarily recommend planting exotic species as a restoration treatment. But my results do show that if restoration goals are to improve soil organic matter, the elimination of invasive species may not always be a priority. This is particularly true when the species in question provides additional ecosystem services.

Portnoy (1997 & 1999) cautioned that seawater restoration to a previously diked marsh should be gradual and carefully monitored due to the possible release of sulfide toxins. This should be taken into consideration for all berm removal projects in estuaries. Since we did not detect a large sulfide influx in our sampling, this suggests the previous berm breaches helped to slow the reintroduction of seawater in the Nanaimo River estuary. Had the inital breaches not taken place, the embankment removal instead would caused a large initial sulfide release, and we may have seen vegetation struggle to survive. This may have had the further effect of allowing more invasive *Juncus gerardii* to take over the estuary if the natural vegetation could not survive the sulfide influx.

The higher stem density of *J. gerardii* compared to other species in the estuary may actually make it beneficial to sediment accretion in the face of sea-level rise, regardless of its exotic status (Gleason et al., 1979; Warren and Niering, 1993). Estuary marsh plants play an essential role in estuary morphology by influencing the rate of sedimentation and in some cases, building a marsh platform that can keep pace with sea-level rise (Craft et al., 1993; Boorman et al., 1998). Salt marsh resilience in the face of climate change is frequently cited as rationale for estuary de-embankment, as it was for the work carried out in the Nanaimo Estuary. As sea levels rise with climate change, salt marshes may be exposed increased inundation, with potential impacts on vegetation succession (Warren and Niering, 1993; Olff et al., 1997).

We know that the vegetation in an estuary has significant control over the landforms, and this accentuates the importance of including vegetation in any estuary restoration plan. This is particularly true when considering tidal channels in Pacific Northwest estuaries. Tidal channels are troughs within the salt marsh, which periodically or continuously contain moving water (Simenstad, 1983) and are flooded and drained twice daily in the Nanaimo estuary. Chinook salmon (*Onchorynchus tshawytscha*) which have access to estuary tidal channels display superiority in growth versus those without access (Levy, 1982). Juveniles may rear for 25 days or more in this estuary; here they feed on invertebrates, insects and mysids, making it a crucial early-life nursery (Healy, 1980). Chinook salmon juveniles can also feed on Dungeness crab megalopae (Hunt, 1999) which reside in large numbers in the tidal channels of the Nanaimo Estuary. Intertidal foraging has been demonstrated to be necessary for young Cancer magister (Holsman, 2003), emphasizing the importance of these channels in the food web.

A berm removal also increases inundation, albeit on a shorter timescale. We may be able to use the results from long-term vegetation monitoring after a berm removal to predict the change we will see in estuaries from rising oceans, since, a berm removal treatment, in effect, acts as a rapidly rising sea in the estuary. The outcome of the berm removal also extends beyond soil nutrient changes. For example, tidal wrack is organic material deposited by a high tide or storm tide. *J. gerardii* has been shown to have poor resistance to burial by tidal wrack (Brewer, 1998) and the removal of berms will may result in increased wrack accumulation in the future, another factor to consider in the grand scheme of restoring this estuary.

All but one of British Columbia's biogeoclimatic zone site series for estuaries are red-Listed (see **Appendix C**) under the British Columbia *Wildlife Act*. The remaining association (Em06) is blue-listed. Three of the red-listed site series were surveyed in the Nanaimo River Estuary (Em02, Em03, Em04). The total area of these plant communities throughout the province is extremely limited, and their conservation status enforces the importance of conserving these communities. It also emphasizes the importance of vegetation monitoring in the estuary and the value of predicting vegetation change. The estuary's most dominant species (*Juncus gerardii)* is not listed as threatened and, as previously stated, is considered an exotic species (BC Conservation Data Centre, 2020).

The use of PRS or similar technology/sampling techniques shows great potential for future use in before/after studies where short-term changes to soil conditions need to be measured. Pezeshki & DeLaune (2012) emphasized the need for further research and techniques which can assess plant function in wetland ecosystem in response to soil redox condition, and our study demonstrates a relatively low cost, replicable method to meet this need. This technique may be more effective in different restoration scenarios where soil moisture regime remains more consistent, without several daily fluctuations.

My copper soil nutrient results showed one of the strongest signals of an effect in the treatment regions of the marsh versus the reference regions. It would be prudent to further investigate this potential release of copper after a berm removal, particularly since many berm treatments take place on former agricultural land, a common source of copper pollutants (Adrees et al., 2015).

Limitations

A limited sample size using the nutrient probes limits the interpretation of these results. Some results were tightly clustered (indicating higher precision) and conclusions can be drawn from these data, whereas others exhibited far too much variability and no noticeable patterns from which to make interpretations.

Additionally, the deployment period may have been too short to pick up truly representative masses of ions for some elements. Some ions returned results below what the supplier suggests as the ‘minimum detectable level’, and these could only be interpreted with caution and where variability was low. Nutrient supply from soil to plant occurs in the presence of moisture, and moister soils will increase adsorption to the probe’s membrane. The one-week deployment time was chosen based on general recommendations from Western Ag for use in PRS © Probes in wetlands. Since the salt marsh fluctuates in soil moisture, the soil at 10cm depth may have been drier than anticipated and required a longer burial to push all results to detectable levels. Practitioners should be mindful of the environment in which they are sampling, paying particular attention to the wetness of soils. Several variations of deployment length should be trialed prior to the main nutrient assessment. When assessing the results of these trials, practitioners should make sure that the probes have adsorbed masses of ions considered to be above the minimum detection level for each nutrient of interest. If results fall below these thresholds, the deployment period should be extended.

Differing environmental variables (weather, tides, plant growth, etc.) between the two PRS© deployments may have also obscured the interpretation of results. While all four samples sites would have received an equal ‘treatment’ from these environmental conditions, ideally the two sampling periods should have occurred one-year apart. This would have ensured estuary plants (which compete with the PRS© probes) to would have been in similar growth and productivity stages during both sampling periods.

Conclusion

Estuaries are highly dynamic and complex environments due to multiple daily inundations of seawater. This has implications for soil properties, flora in the estuary and even the physical landforms present. The construction of embankments for the purpose of agriculture in these environments affects all of the above and their related processes. I was able to demonstrate that plant species (as influenced by the long-term effect of diking) had more control over soil organic matter in this disturbed estuary than elevation and natural inundation-influenced processes such as microbial activity. I was also able to show that the availability of some soil nutrients appears to be affected by berm removal in the short-term, but this requires further investigation. Previous dike breaches in this estuary appears to have affected the inundation and soil saturation effects of the berm removal treatment, emphasizing the importance of considering previous restoration work in restoration planning. Using our data as a baseline, managers of the Nanaimo River Estuary property should monitor for long-term vegetation change in the salt marsh. This would give an indication of changing inundation in the marsh due to the restoration treatment, as well as changes in soil properties. Managers should also monitor physical change in the estuary. Given the necessity for estuary restoration throughout British Columbia and the Pacific Northwest, the need to understand our restoration actions and the cascading series of events that follow is becoming increasingly important for pre-project planning and adaptive management.

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Appendix A – Soil Organics and Nutrients - Complete Results

BERM1							
Berm East							
			ID	**Sample #**	**Pre-Burn**	**Post-burn**	**% OM**
Transect 1	**Core 1**	Upper	BERME-B1-U	049	5.007	2.531	49.45%
Blind	Close	Lower	BERME-B1-L	050	5.008	4.192	16.29%
	Core 2	Upper	BERME-B2-U	051	5.007	3.799	24.13%
	Middle	Lower	BERME-B2-L	052	5.000	4.261	14.78%
	Core 3	Upper	BERME-B3-U	053	5.003	3.937	21.31%
		Lower	BERME-B3-L	054	5.014	4.290	14.44%
Transect 2	**Core 1**	Upper	BERME-M1-U	057	5.005	4.058	18.92%
Mid	Close	Lower	BERME-M1-L	058	5.012	4.494	10.34%
	Core 2	Upper	BERME-M2-U	059	5.008	2.651	47.06%
	Middle	Lower	BERME-M2-L	060	5.002	4.284	14.35%
	Core 3	Upper	BERME-M3-U	061	5.015	3.861	23.01%
		Lower	BERME-M3-L	062	4.997	3.607	27.82%
Transect 3	**Core 1**	Upper	BERME-O1-U	065	4.998	3.586	28.25%
Open	Close	Lower	BERME-O1-L	066	5.012	4.564	8.94%
	Core 2	Upper	BERME-O2-U	067	5.008	3.524	29.63%
	Middle	Lower	BERME-O2-L	068	5.005	4.388	12.33%
	Core 3	Upper	BERME-O3-U	069	5.010	3.796	24.23%
		Lower	BERME-O3-L	070	5.000	4.079	18.42%
	Core 4	Upper					
BERM2							
Berm West							
			ID	**Sample #**	**Pre-Burn**	**Post-burn**	**% OM**
Transect 1	**Core 1**	Upper	BERMW-B1-U	001	5.007	3.374	32.61%
Blind	Close	Lower	BERMW-B1-L	002	5.014	3.960	21.02%
	Core 2	Upper	BERMW-B2-U	003	5.005	4.638	7.33%
	Middle	Lower	BERMW-B2-L	004	5.002	4.581	8.42%

	Core 3	Upper	BERMW-B3-U	005	5.002	3.621	27.61%
		Lower	BERMW-B3-L	006	4.999	4.540	9.18%
Transect 2	**Core 1**	Upper	BERMW-M1-U	009	4.999	4.740	5.18%
Mid	Close	Lower	BERMW-M1-L	010	5.006	3.607	27.95%
	Core 2	Upper	BERMW-M2-U	011	5.077	3.705	27.02%
	Middle	Lower	BERMW-M2-L	012	5.003	4.624	7.58%
	Core 3	Upper	BERMW-M3-U	013	5.000	3.838	23.24%
		Lower	BERMW-M3-L	014	5.003	4.485	10.35%
Transect 3	**Core 1**	Upper	BERMW-O1-U	017	5.001	4.591	8.20%
Open	Close	Lower	BERMW-O1-L	018	5.009	4.780	4.57%
	Core 2	Upper	BERMW-O2-U	019	5.003	3.562	28.80%
	Middle	Lower	BERMW-O2-L	020	5.006	3.874	22.61%
	Core 3	Upper	BERMW-O3-U	021	5.008	4.468	10.78%
		Lower	BERMW-O3-L	022	5.004	4.549	9.09%
Oak Island							
			ID	**Sample #**	**Pre-Burn**	**Post-burn**	**% OM**
Transect 1	**Core 1**	Upper	OAIS-B1-U	073	5.004	4.071	18.65%
Blind	Close	Lower	OAIS-B1-L	074	5.011	4.102	18.14%
	Core 2	Upper	OAIS-B2-U	075	5.005	3.974	20.60%
	Middle	Lower	OAIS-B2-L	076	5.013	3.691	26.37%
	Core 3	Upper	OAIS-B3-U	077	4.999	3.771	24.56%
		Lower	OAIS-B3-L	078	4.997	4.102	17.91%
Transect 2	**Core 1**	Upper	OAIS-M1-U	081	5.000	3.313	33.74%
Mid	Close	Lower	OAIS-M1-L	082	5.008	3.953	21.07%
	Core 2	Upper	OAIS-M2-U	083	5.000	3.882	22.36%
	Middle	Lower	OAIS-M2-L	084	5.004	4.001	20.04%
	Core 3	Upper	OAIS-M3-U	085	5.003	3.838	23.29%
		Lower	OAIS-M3-L	086	5.002	4.127	17.49%
Transect 3	**Core 1**	Upper	OAIS-O1-U	089	5.011	3.849	23.19%
Open	Close	Lower	OAIS-O1-L	090	5.007	3.157	36.95%
	Core 2	Upper	OAIS-O2-U	091	5.001	3.554	28.93%
	Middle	Lower	OAIS-O2-L	092	5.014	3.741	25.39%
	Core 3	Upper	OAIS-O3-U	093	5.004	3.636	27.34%

		Lower	OAIS-O3-L	094	5.003	3.935	21.35%
SWHM							
			ID	**Sample #**	**Pre-Burn**	**Post-burn**	**% OM**
Transect 1	**Core 1**	Upper	SWHM-B1-U	025	5.024	2.785	44.57%
Blind	Close	Lower	SWHM-B1-L	026	5.004	3.799	24.08%
	Core 2	Upper	SWHM-B2-U	027	5.008	3.539	29.33%
	Middle	Lower	SWHM-B2-L	028	4.999	4.412	11.74%
	Core 3	Upper	SWHM-B3-U	029	5.009	3.495	30.23%
	Far	Lower	SWHM-B3-L	030	5.013	4.539	9.46%
Transect 2	**Core 1**	Upper	SWHM-M1-U	033	5.007	2.838	43.32%
Mid	Close	Lower	SWHM-M1-L	034	5.014	3.302	34.14%
	Core 2	Upper	SWHM-M2-U	035	5.007	2.547	49.13%
	Middle	Lower	SWHM-M2-L	036	5.004	3.174	36.57%
	Core 3	Upper	SWHM-M3-U	037	5.012	3.454	31.09%
		Lower	SWHM-M3-L	038	5.021	3.875	22.82%
Transect 3	**Core 1**	Upper	SWHM-O1-U	041	5.005	2.999	40.08%
Open	Close	Lower	SWHM-O1-L	042	5.011	3.609	27.98%
	Core 2	Upper	SWHM-O2-U	043	5.002	3.328	33.47%
	Middle	Lower	SWHM-O2-L	044	5.010	3.446	31.22%
	Core 3	Upper	SWHM-O3-U	045	5.011	3.646	27.24%
		Lower	SWHM-O3-L	046	5.016	4.233	15.61%

Soil Nutrient Analysis

Blue highlight = Below detectable levels

Soil Nutrient Availability (before)

		Sample ID	Sample #	Total N (µg/10cm2/7 days)	NO3-N	NH4-N	Ca	Mg	K	P	Fe	Mn	Cu	Zn	B	S	Pb	Al	Cd
BERM2	Site 1	**BERMW-1-B**	001	10.28	2.40	7.88	447	785	168	0.72	3.76	0.27	0.18	0.00	1.12	47.00	0.01	12.91	0.02
West	Site 2	**BERMW-2-B**	002	3.28	1.02	2.26	491	826	156	0.43	3.23	0.24	0.17	0.00	0.86	39.46	0.18	12.60	0
	Site 3	**BERMW-3-B**	003	2.02	0.28	1.74	428	757	152	1.23	18.95	1.70	0.10	0.00	0.84	48.92	0.00	11.07	0
SWHM	Site 1	**SWHM-1-B**	004	4.08	0.44	3.64	363	655	164	0.67	5.02	0.34	0.13	0.00	0.66	40.17	0.10	11.19	0
	Site 2	SWHM-2-B	005	2.68	0.60	2.08	437	748	170	0.47	5.54	1.33	0.19	0.00	0.64	39.63	0.03	11.51	0
	Site 3	**SWHM-3-B**	006	2.52	0.00	2.52	377	696	171	0.37	3.14	0.38	0.11	0.00	0.40	40.07	0.39	9.15	0.02
BERM1	Site 1	**BERME-1-B**	007	2.10	0.00	2.10	501	844	171	1.04	9.08	0.73	0.13	0.00	2.41	45.21	0.27	19.60	0.02
East	Site 2	**BERME-2-B**	008	2.78	0.26	2.52	389	742	173	1.42	18.70	1.12	0.19	0.00	0.32	41.49	0.06	7.10	0.02
	Site 3	**BERME-3-B**	009	9.58	0.74	8.84	461	823	184	0.64	3.73	0.40	0.21	0.00	0.35	38.13	0.32	9.72	0.02
OAIS	Site 1	**OAIS-1-B**	010	2.10	1.38	0.72	383	714	157	0.38	8.10	0.78	0.14	0.00	0.30	36.53	0.08	7.12	0
	Site 2	**OAIS-2-B**	011	6.54	1.30	5.24	428	743	166	0.32	2.01	0.11	0.15	0.00	0.44	32.58	0.37	10.82	0
	Site 3	**OAIS-3-B**	012	3.40	1.06	2.34	414	734	153	0.46	3.20	0.15	0.26	0.00	0.43	42.29	0.17	8.01	0.02

Soil Nutrient Availability (after)

				Total N (µg/10cm2/7 days)	NO3-N	NH4-N	Ca	Mg	K	P	Fe	Mn	Cu	Zn	B	S	Pb	Al	Cd
BERM2	Site 1	**BERMW-1-A**	013	5.16	1.74	3.42	567	949	180	0.23	3.30	0.27	0.12	0.00	1.19	37.78	0.00	15.11	0.01
West	Site 2	**BERMW-2-A**	014	2.04	0.56	1.48	425	708	151	0.14	1.65	0.14	0.03	0.00	0.33	32.54	0.00	8.34	0.03
	Site 3	**BERMW-3-A**	015	1.94	0.58	1.36	492	827	176	0.20	4.55	0.99	0.07	0.00	0.77	38.54	0.00	13.92	0.01
SWHM	Site 1	**SWHM-1-A**	016	1.20	0.52	0.68	382	812	192	0.24	8.07	0.40	0.26	0.00	0.59	83.26	2.70	13.42	0.01
	Site 2	**SWHM-2-A**	017	2.54	1.28	1.26	471	874	165	0.19	4.04	0.84	0.27	0.00	0.77	58.36	1.80	10.37	0.02
	Site 3	**SWHM-3-A**	018	1.30	0.00	1.30	391	711	170	0.19	3.49	0.38	0.14	0.00	0.94	44.01	1.33	11.92	0.02
BERM1	Site 1	**BERME-1-A**	019	11.40	0.00	11.40	414	700	165	0.26	2.27	0.08	0.17	0.00	0.36	34.31	0.02	7.94	0.01
East	Site 2	BERME-2-A	020	7.96	0.50	7.46	465	796	180	0.58	20.25	0.28	0.13	0.00	1.78	46.65	0.13	21.46	0.02
	Site 3	**BERME-3-A**	021	4.92	0.64	4.28	461	789	177	0.30	8.26	0.76	0.16	0.00	0.41	39.67	0.10	9.88	0.01
OAIS	Site 1	OAIS-1-A	022	5.62	2.06	3.56	458	899	180	0.33	10.53	0.86	0.29	0.00	2.67	87.41	0.52	24.69	0.01
	Site 2	**OAIS-2-A**	023	6.22	2.00	4.22	549	1002	184	0.35	3.57	0.27	0.16	0.00	1.29	52.48	0.92	16.76	0.02
	Site 3	**OAIS-3-A**	024	4.00	1.80	2.20	413	736	144	0.45	4.56	0.22	0.32	0.00	0.90	53.64	0.47	14.69	0.01

Appendix B – Red and Blue-listed Plant Communities

BEC Classification	Community Name	Latin Name	BC List Status	Found in Nanaimo Study Area
Em01	Beaked ditch-grass herbaceous vegetation	*Ruppia maritima*	Red	No
Em02	**American glasswort and sea-milkwort**	***Salicornia virginiana – Glaux maritima***	**Red**	**Yes**
Em03	**Seashore saltgrass herbaceous vegetation**	***Distichilis spicata***	**Red**	**Yes**
Em04	**Sea plantain - dwarf alkaligrass**	***Plantago maritima - Puccinellia pumila***	**Red**	**Yes**
Em05	Lyngbye's sedge herbaceous vegetation	*Carex lyngbyei*	Red	No
Em06	Lyngbye's sedge – Douglas' water-hemlock	*Carex lyngbyei – Cicuta douglasii*	Blue	No
Ed01	Tufted hairgrass - meadow barley	*Deschampsia cespitosa ssp. beringensis - Hordeum brachyantherum*	Red	No
Ed02	Tufted hairgrass - Douglas' aster	*Deschampsia cespitosa ssp. beringensis - Symphyotrichum subspicatum*	Red	No

Ed03	Arctic rush - Alaska plantain	*Juncus arcticus* - *Plantago macrocarpa*	Red	No