The impacts of exotic *Typha* on benthic invertebrate communities in the South Arm of the Fraser River Estuary

by
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Abstract

In recent decades, the exotic cattail *Typha angustifolia* and its hybrid *Typha x glauca* have invaded the Fraser River estuary. The impacts from this invasion on benthic macroinvertebrate communities, however, are yet to be studied. Macroinvertebrates play important roles in food chains, trophic dynamics, and nutrient cycling and are potentially at risk from this invasion. In this study, I compared the benthic invertebrate communities between exotic cattail stands and native vegetation stands at 25 paired sites. Sediment cores were analyzed for invertebrate abundance, biomass, and Shannon Wiener diversity index, and it was found that biomass and abundance were lower in exotic cattail when compared to native vegetation, however, there was no difference in diversity. Given the proximity to side channels, tidal inundation time would be a logical explanation for the differences in the benthic communities; however, it was not found to be a significant predictor. Given the invasive nature of exotic cattail and the correlations that were found, cattail should be removed in restoration projects where possible.

Keywords: Fraser River; *Typha angustifolia*; *Typha x glauca*; Estuary; Invasive species

Dedication

I would like to dedicate this research project to my family. I will be forever grateful for the love, support, care, and patience you have given me. Mum, my source of reason. Dad, my teacher of practical skills. My dear sister Nelle, my life-long partner in crime. You have always been there for me, shaping me into who I am today and pushing me to be better and I cannot thank you enough. You are my daily inspiration, and I would not be here without you. If it is worth doing it is worth overdoing. I love you all.

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Introduction

Estuaries represent a unique combination of several ecosystems that include both freshwater and marine environments, while also including terrestrial areas as well. These environments act as foraging habitats, migration corridors, resting locations, and transition zones, but also provide ecosystem functions such as water filtration, water attenuation, carbon sequestration, and other ecosystem services (Barbier et al. 2011, Short et al. 2016). These estuary characteristics and processes are valuable to both humans and the hundreds of species that use them (Emmett et al. 2000, Barbier et al. 2011). In British Columbia, estuaries are a common feature along the 27,000 km of coastline in the province and are inhabited by a wide variety of animals and life history stages (Emmett et al. 2000). These estuaries are also some of the most impacted and threatened ecosystems around the world as the result of anthropogenic stressors, such as sea level rise, urbanization, and invasive species (Crooks and Turner 1999, Barbier et al. 2011, Thorne et al. 2015).

The Intergovernmental Panel on Climate Change has predicted up to a 0.98 m rise in mean sea level by 2100 and this increases the risk of losing estuarine habitat through increases in inundation, particularly in urban estuaries (Pontee 2013, Wong et al. 2014). Additionally, in some river systems sediment inputs have been declining, which is suggested to be a result of sediment trapping in reservoirs, limiting accretion in estuaries and further increasing the risk of estuarine habitat loss (Weston 2013). The combination of decreased supplies of sediment and increasing sea levels is predicted to result in an expansion of low marsh and tidal mudflats and a loss of middle and high marsh habitat on the west coast of North America (Thorne et al. 2015). This is further exacerbated by the armoring of urbanized areas and the construction of dikes and levees, which are resulting in ‘coastal squeeze’ where coastal habitats, such as marshes, are “squeezed” between armoring and rising water levels and are lost over time (Pontee 2013). In areas with no armoring, marsh habitat has the capacity to migrate inland and retain water, but with these barriers in place, marshes cannot maintain the same elevational relationship with increasing water levels (Pontee 2013).

This increasing water depth is likely to change vegetation communities, favoring species that are able to tolerate deeper water (Weisner 1993). This is potential problematic if native species are unable to colonize quickly, leaving open niches for faster growing invasive species such as *Spartina* spp. and *Typha* spp. (Weisner 1993, Selbo and Snow 2004).

As trade and globalization increase, invasive species are becoming more prevalent worldwide. Whether they are shipped incidentally as stowaways in goods or in ballast water, or purposefully traded as pets or food, these species threaten ecosystems, particularly those along trade routes (Sardain et al. 2019). Biological invasions by introduced, invasive species have been found to have many impacts, including increased competition with, and displacement of, native species which can alter species composition in an area, leading to changes in food web dynamics when prey availability and abundance change (Di Castri 1990, Pejchar and Mooney 2009, Simberloff et al. 2013). It is forecasted that by 2050 there will be a 209% increase in global shipping traffic, which could translate into a 3 to 20-fold increase in invasion risk (Sardain et al. 2019). While we are aware of some of the possible impacts, our comprehension of the extent and severity of this possible increase in invasion is limited and requires further investigation.

As human populations increase, urbanization and expanding land use have been found to negatively impact estuaries in many ways, including habitat loss and pollution (Yunker et al. 2002, Freeman et al. 2019). Habitat alterations that result from the expansion of cities and agricultural spaces physically remove or modify ecosystems, both of which cause the loss of ecological function (Pomeroy 1995, Hall and Schreier 1996). Pollution also has extensive impacts on inhabitants and processes in estuaries, particularly in areas with high proportions of impermeable surfaces. The presence of impervious surfaces has been shown to contribute copper, zinc, heavy metals, and other toxicants into watercourses, which can accumulate in estuaries and lead to adverse effects on the inhabiting species (Hall and Schreier 1996, Teuchies et al. 2013). As a result of urbanization and industrialization, the Fraser River estuary has lost

80% of its wetlands and it is at risk of losing two-thirds of associated species in the next 25 years without any intervention (Boundary Bay Conservation Committee 2016, Kehoe et al. 2020).

The Fraser River Estuary is located in Southern British Columbia and has three arms that flow into the Strait of Georgia. Approximately 16 km upstream of the Strait of Georgia, the North Arm meets the Middle Arm, and after an additional 5 km, the combined Middle and North Arm meet with the larger South Arm. The Fraser River estuary is important habitat for pacific salmon, an ecological and cultural keystone species on the Pacific Northwest (Nobel et al. 2016). The estuary provides an important environment for salmon during the transitional period of their lifecycle, where they adapt from fresh to saline water. It also represents important feeding habitat required for growth (Davis et al. 2020). This growth is particularly important as higher body weights are associated with higher survival rates in the first year of ocean life (Björnsson et al. 2011, Davis et al. 2020). Historically, the Fraser River has been home to the largest wild salmon runs in the world, but in recent decades pacific salmon populations have declined significantly. (Fraser et al. 1982, Noakes et al. 2000). Although these declines are likely a result of a combination of overfishing, climate change, and habitat alteration, additional factors such as degraded foraging conditions for benthic invertebrates in estuaries may also play a role (Noakes et al. 2000, Duffy et al. 2010).

Benthic macroinvertebrate communities are composed of many different taxa, including mollusks, segmented worms, crustaceans, insects, and other arthropods. These communities can be found in a variety of bottom substrates such as sand, mud, or gravel, where they have roles in maintaining water and sediment quality, and are important forage for many estuary inhabitants, including juvenile salmon (Roegner et al. 2004). Stressors such as dredging, pollution, and invasive species impact these communities and can directly and indirectly impact food webs in the ecosystem. Dredging can result in the mobilization of pollutants and toxicants, smothering and suffocation through mobilized sediment, physical removal of invertebrates, and suspension of invertebrates into the water column exposing them to predators

(Szymelfenig et al. 2006, Pledger et al. 2021). While the impacts of toxicants and dredging are more widely known and better understood, the impacts that invasive macrophytes have on benthic communities remain unclear.

In recent decades, the invasive macrophyte *Typha angustifolia* (european cattail or narrow leaf cattail) has invaded the Fraser River estuary. The exact year of arrival is unknown, but it is present in herbarium collections as early as 1990 (PNWHerberia, V202108). Previous studies have indicated that the species can have three main ecological impacts on the surrounding area: 1) changes to gene pool composition in native cattail 2) competition with native vegetation and 3) alteration of trophic level dynamics (Heimer and Parsons 2013).

Gene pools of native cattail are altered through hybridization with the native *Typha latifolia* (broad leaf cattail) to form the hybrid *Typha x glauca*. The hybrid can also back-cross with, and is more competitive than, either parent species, which can further change gene pool composition. Both the hybrid and *T. angustifolia* have better nitrogen uptake, are more tolerant of deeper water and higher salinity, and can utilize allelopathic root exudates to out compete *T. latifolia* and other native vegetation (Jarchow and Cook 2009). The resulting exotic stands are generally monotypic, highly dense, and have been shown to alter trophic level dynamics (Schultz and Dibble 2012).

Exotic cattail stands have been associated with a reduction in the performance of amphibians, lower use by waterfowl, and lower fish diversity and abundance (Maerz et al. 2010, Shrank and Lishawa 2019, Lishawa et al. 2020). There has been limited examination, however, of the impacts on benthic macroinvertebrate communities and the resulting bottom-up effects that may occur for important species like salmon that rely on these communities both directly and indirectly as food sources. Decreased use of exotic cattail stands by waterfowl may be the result of impacts to benthic communities, as invertebrates are a driver in waterfowl use of wetlands (Anderson and Smith 2000). Similarly, a decrease in abundance and diversity of non-salmonid species in wetland areas with exotic cattail may also be driven by changes in benthic communities (Shrank and Lishawa 2019). Impacts on salmonid species may be asymmetrical in nature as the result of differences in feeding behaviours. Species like pink (*Onchorhynchus gorbuscha*), chum (*Onchorynchus keta*), and chinook (*Onchorynchus tschawytscha*) would be directly impacted by decreases in benthic invertebrates, and chinook in particular rely heavily on benthic invertebrates while in estuaries (McPhail 2007, Daly et al. 2019). In the case of coho (*Onchorynchus kisutch*), which are piscivorous, one of their main forage fishes is juvenile pink salmon (Mcphail 2007). Declines in pink salmon associated with decreases in benthic invertebrates would then likely have a trickle-down effect on coho populations. These declines could lead to impacts on higher trophic levels that rely on salmon as a main food source.

Given the importance of estuaries as foraging habitat for juvenile salmonids and the fact that a large part of salmon success in the ocean is dependent on their size when they leave the estuary, further understanding of the factors affecting benthic communities in estuaries is crucial. If exotic cattail negatively impacts benthic invertebrate communities, this could add to the threats faced by Pacific salmon populations, which are already experiencing large population declines. If this is the case, restoration activities such as exotic cattail control or removal should be considered to improve the likelihood of survival of juvenile salmonids.

Study Area

This study occurred in a cluster of islands in the marshes of the South Arm of the Fraser River estuary. The group of islands are in an area of the estuary that is approximately 6 km long and 2.2 km wide and is located east of the Alaksen National Wildlife area. The islands include Duck, Gunn, Frenchies, Barber, and Kirkland Islands (Fig 1).

Figure 1 Map of the lower mainland of BC, showing the study area on a regional scale.

The Fraser River estuary is a drowned river, salt wedge estuary, which is subject to mixed tides, resulting in two unequal tidal periods over a 24-hour period. The surrounding area experiences moderate temperatures throughout the year, with a yearly mean maximum of 21.7 °C in July and a yearly mean minimum of 5.1 °C in December and January (Meteorological Service of Canada 2020). The area receives approximately 900 mm of rain each year, and the highest volumes fall between October and January. Very little of the annual precipitation falls in the form of snow, with a yearly amount of 275 mm (Meteorological Service of Canada 2020).

The estuary is subjected to many sources of pollution such as urban runoff, commercial and recreational watercraft, trash, and noise from surrounding industrial areas. The estuary is also a popular site for recreational boaters and is a transit corridor for locals. The largest port in the country is also found in the estuary, and it has four terminals directly upstream of the study area, two automotive, one breakbulk, and one oil and chemical (Port of Vancouver 2021). As the result of ship traffic, certain areas of the estuary are dredged to maintain channel depth for ships (Port of Vancouver 2016). Agriculture is also a common land use in the area, occurring from Richmond and Delta at the mouth of the river to Hope, which is located approximately 150 km upstream of the study area.

Objectives

It is unclear what impacts invasive cattail may be having on benthic invertebrate communities in the Fraser River estuary. Ducks Unlimited Canada is currently undertaking several restoration projects in the Fraser River estuary, many of which are focused on ameliorating conditions for sockeye, coho, chum, pink, and chinook salmon. The goal of this study was to investigate possible associations between invasive cattail stands and benthic invertebrate communities, which could lead to larger impacts on sockeye, coho, chum, chinook, and pink salmon that may result from shifts in invertebrate communities. With a focus on potential bottom-up trophic processes, the main objective of this study was to identify possible correlations between tidal inundation and vegetation origin and invertebrate biomass, abundance, and diversity.

Methods

Sampling Dates and Site Access

Sampling occurred in the last two weeks of May and first two weeks of June 2020 for a total of six sampling days (Appendix A). In 2019, Ducks Unlimited Canada completed a benthic pilot study and a four-week period was used in this study to complement the 2019 data set. This time frame also aligns with the occupancy period in the estuary by juvenile sockeye, coho, chum, chinook, and pink salmon (Chalifour et al. 2019).

Time of site access was determined based on the occurrence of an ebbing tide and a low tide occurring before 2:00 pm PST. This time was selected to allow for all sites to be sampled before nightfall, with additional time allotted in the event of an emergency.

Exotic and Native Site Selection

An exotic *Typha* distribution model was used locate stands of exotic *Typha* (Stewart et al. 2021). This model was then filtered for a minimum stand size of 10 m^2, which was selected to have a sufficiently large patch to minimize the likelihood of edge effects from all sides (Cooper et al. 2012). From this, 54 sites were identified and then ground-truthed to ensure stands were present and of appropriate size. Once confirmed, they were pinned in google maps on a smart phone. The pins used to mark the confirmed stands were also used as the locations for the exotic benthic core samples. These locations were exported as a .kml file and then imported into ArcMap (Appendix B). Using ArcMap's randomization tool, 25 exotic stands were randomly selected from the 54 ground-truthed stands. The 25 exotic stands were plotted into ArcMap and divided into six groups based on proximity to each other (Fig 2). Each group area was saved as a separate georeferenced map to be used in PDF Avenza for site location in the field. Grouped sampling areas were assigned randomly to a sampling date to reduce spatio-temporal effects.

Each exotic stand was always bordered by native vegetation stands. The native benthic core locations were selected in the field based on the distance from the exotic benthic core location to the nearest edge of the exotic stand. This distance was then used to determine the location of the native benthic core. For example, if the exotic core was taken 3 m into the exotic stand, then the native core would need to be located at least 3 m out of the exotic stand, measured from the nearest edge. This distance was sometimes extended to account for edge effects such as other invasive plants or proximity to side channels. This yielded 25 native stands that were paired with the 25 randomized exotic stands.

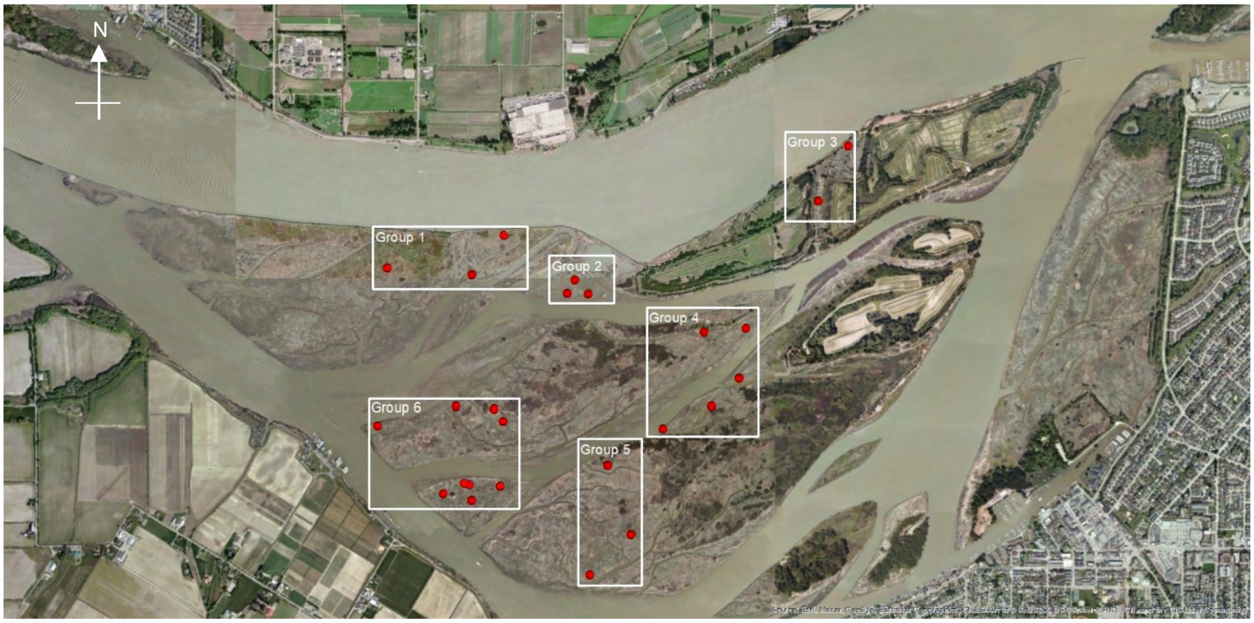

Figure 2 Local view of study area. Sampling sites were grouped to locate sites efficiently and are outlined with white borders. The red circles are the randomized exotic cattail sampling sites.

Sediment Cores and Sifting

Sediment cores were collected with a 14 cm barrel diameter stainless steel clam gun to a depth of 10 cm. To ensure a consistent 10 cm core depth was achieved, the clam gun was marked with permanent marker to a 10 cm depth. The core was then placed in a ZipLock Freezer bag and labelled with the site ID and date.

Cores were sieved on the same day they were collected using a 20.32 cm brass soil sieve with 500 μm stainless steel #35 mesh (Dual MFG Co.) to separate sediment from the invertebrate sample (Yozzo and Osgood 2013, Woo et al. 2018). Large pieces of rhizomes were washed and removed before the sample was transferred to 500 mL LDPE sample jars. Jars were filled to the top with 10% formalin for fixation (by request of Biologica Labs) and labelled with sample site ID and sample date. Samples were refrigerated at 1 °C until delivery to Biologica Labs for sorting and identification.

Identification and subsampling

Invertebrate identifications were completed by Biologica Labs, located in Victoria, BC. Samples were rinsed to remove remaining sediment and preserved in ethanol. Once sorted, the samples were stained with Rose Bengal and identified by taxonomic specialists. The invertebrates were identified to taxonomic family and wet weight biomass measurements were taken. Sub-sampling was used to account for budget limitations and increased sorting requirements due to the varying amounts and types of debris present in samples.

Sub-sampling to 1/6 was completed using the Environmental Effects Monitoring Protocol (Environment Canada 2002). Organisms larger than 1 mm were sorted and counted as a whole sample and the remaining sample was spread across a Caton tray and the fraction to be subsampled was randomly selected and removed for sorting. Sorting was done to an efficiency of >95% (Caton 1991).

Statistical Analysis

Statistical analysis was completed using R-Studio (Version 1.3.959) and spatial analysis was completed in ArcMap (Version 10.8). A Levene's test was used to determine if the variance between the two stands was the same, after which ANOVA was selected for the analysis as a result of unequal variances. The ANOVAs were completed in R-Studio, and were used to compare stand inundation time, biomass, abundance, and Shannon Wiener Diversity index as a function of stand origin. The Shannon Wiener Diversity index was calculated as:

$$H = -1\sum_{i=1}^{s} p_i \ln p_i$$

where H is the family diversity of a benthic core, s is the totally number of families in a core, and p_i is the proportion of the core abundance in a given family (Routledge 1979). A multiple linear regression was completed to model the impacts of duration of tidal inundation and origin of the vegetation community (native versus non-native vegetation) on community abundance, biomass, and diversity. ArcMap was used to determine the tidal inundation time as calculated by Carew and Hickey (2000) and Hickey (2019) using elevation data from the City of Richmond LiDAR and tide data from the Steveston Tide Station to determine tidal inundation time (Fisheries and Oceans Canada n.d.) (Appendix F). The tide data was then corrected for the study area using the CGVD 28 HTv2.0 datum.

Results

In the 50 samples, 31 different families were identified. The most numerous families in both exotic and native vegetation were Naididae (annelids), Chironomidae (dipterans), Enchytraeidae (annelids), and Hydrobiidae (molluscs), all of which had counts over 1700 (Table 1). The remaining 27 families had counts of 294 or less (Appendix C). The results of the ANOVAs (Appendix D) showed that biomass was higher in native stands than in exotic stands, with mean sample weights of 0.0571 g and 0.0064 g respectively ($P=0.017$, $F=6.0859$, $df=48$). Abundance was also higher in native stands compared to exotic stands, at 296.12 and 168.8 mean individuals per sample respectively ($P=0.0207$, $F=5.7262$, $DF=48$) (Fig 2, Fig 3.). Shannon Wiener Diversity ($P=0.911$, $F=0.1226$, $DF=48$) was not significantly different between the stands, with a native stand average of 1.2406 and an exotic stand average of 1.2263 (Fig 4).

*Table 1 Families of benthic macroinvertebrates and their abundances in native and exotic vegetation stands in the South Arm Marshes of the Fraser River Estuary**

Family	Native Vegetation Abundance	Exotic *Typha* Abundance
Naididae	2334	1990
Chironomidae	1593	389
Enchytraeidae	1328	620
Hydrobiidae	1114	666

*Complete table is in Appendix C

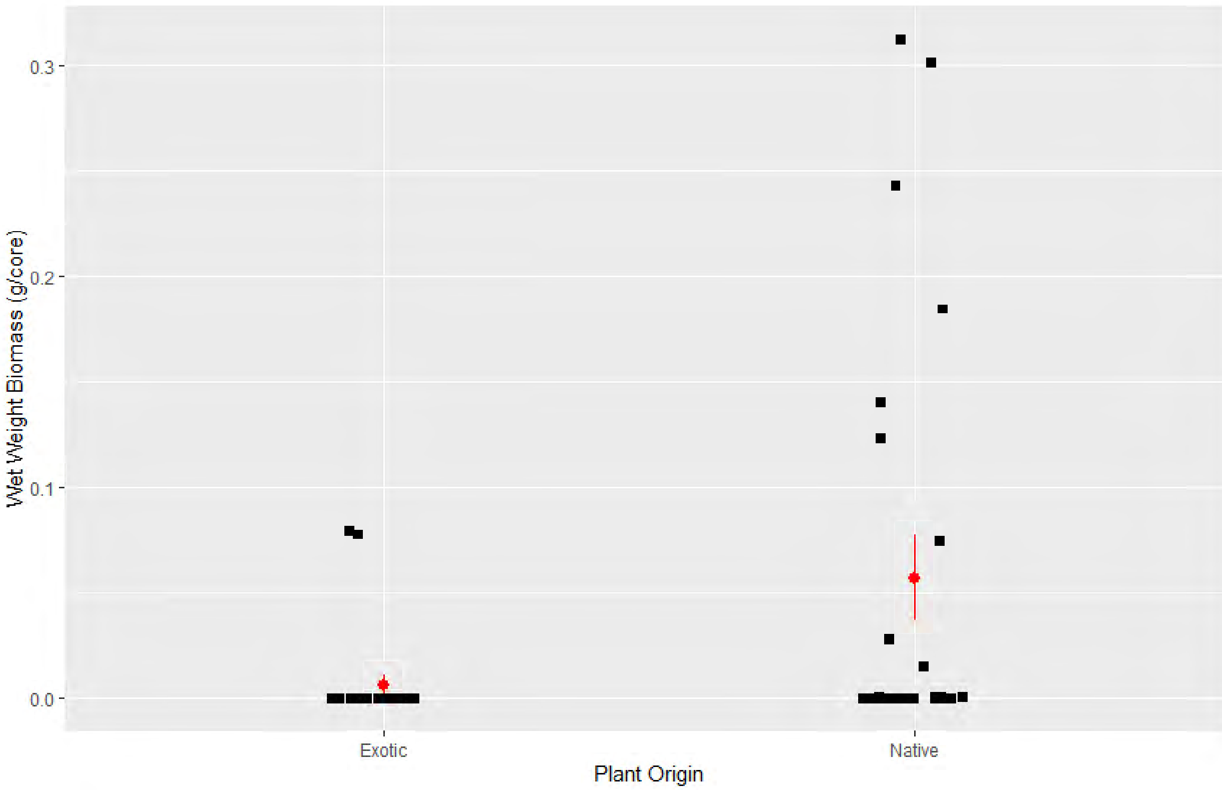

Figure 3 Dotplot of invertebrate wet weight biomass in grams/core, obtained from sediment cores, comparing stands of exotic cattail and stands of native vegetation in the South Arm marshes of the Fraser River Estuary. Each black dot represents a specific core, the red dot represents the mean, and the red lines on either side represent the standard error.

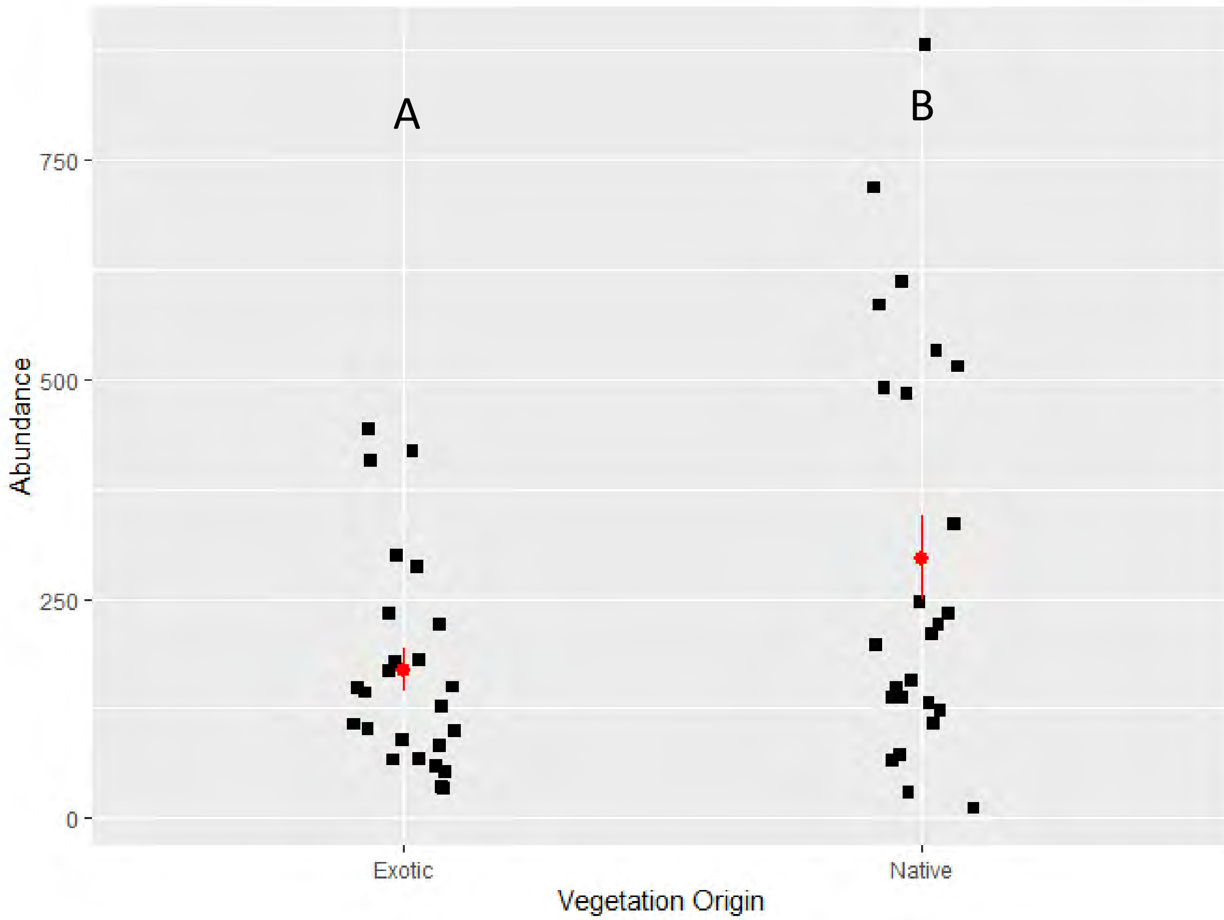

Figure 4 Dotplot of sediment cores benthic invertebrate abundance comparing stands of exotic cattail and stands of native vegetation in the South Arm marshes of the Fraser River Estuary. Each black dot represents a specific core, the red dot represents the mean, and the red lines on either side represent the standard error.

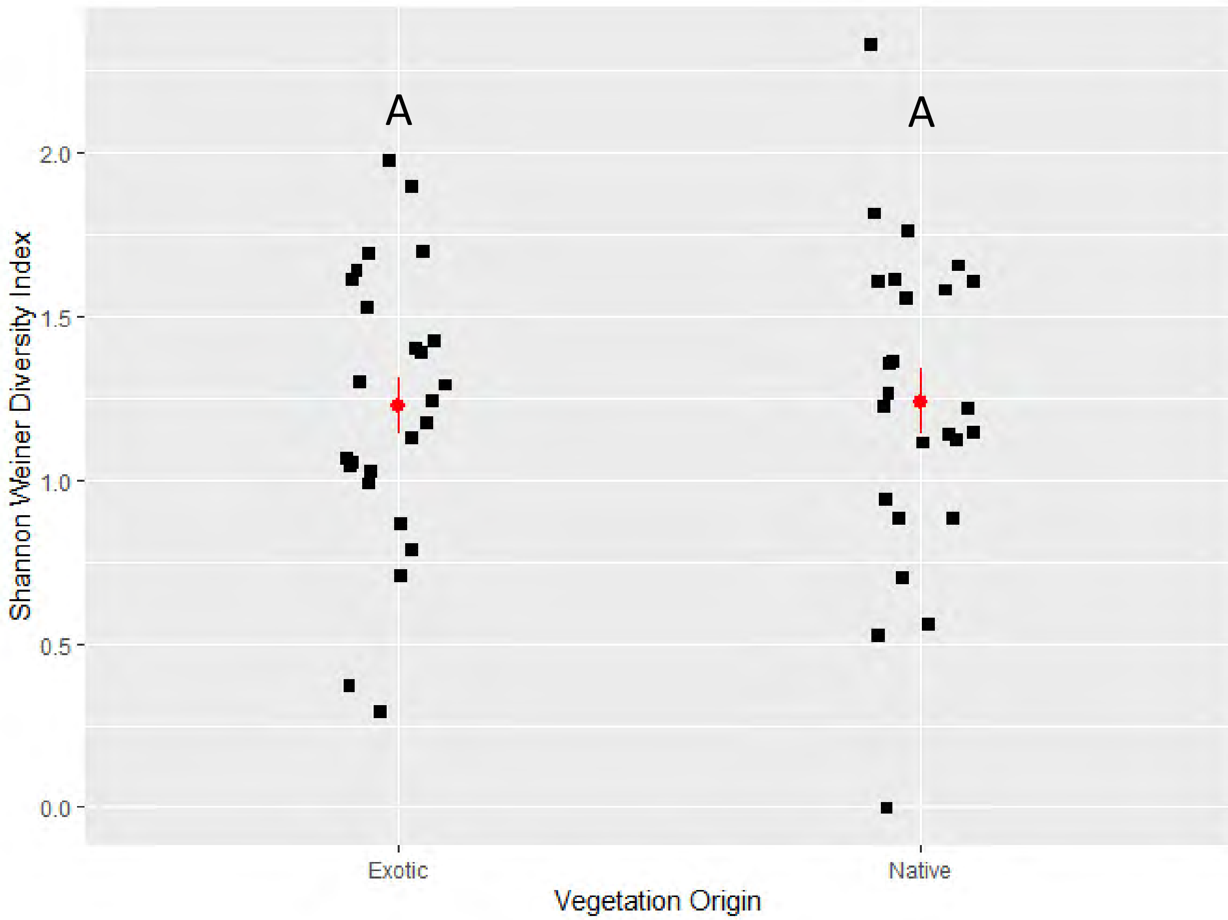

Figure 5 Dotplot of Shannon Wiener Diversity indices of benthic invertebrate communities compared between stands of exotic cattail and stands of native vegetation in the South Arm marshes of the Fraser River Estuary. Each black dot represents a specific core, the red dot represents the mean, and the red lines on either side represent the standard error.

Each multiple linear regression model consisted of a combination of duration of tidal inundation and vegetation origin as predictors for biomass, abundance, or diversity. For biomass, there was a weak correlation for native vegetation and a very weak correlation for exotic vegetation (Fig 6). Abundance and diversity had a very weak correlation for both native and exotic vegetation (Fig 7, Fig 8). For time of inundation, no difference was detected between exotic cattail stands and stands of native vegetation (P=0.3338, F=0.9531, DF=48) (Fig 9).

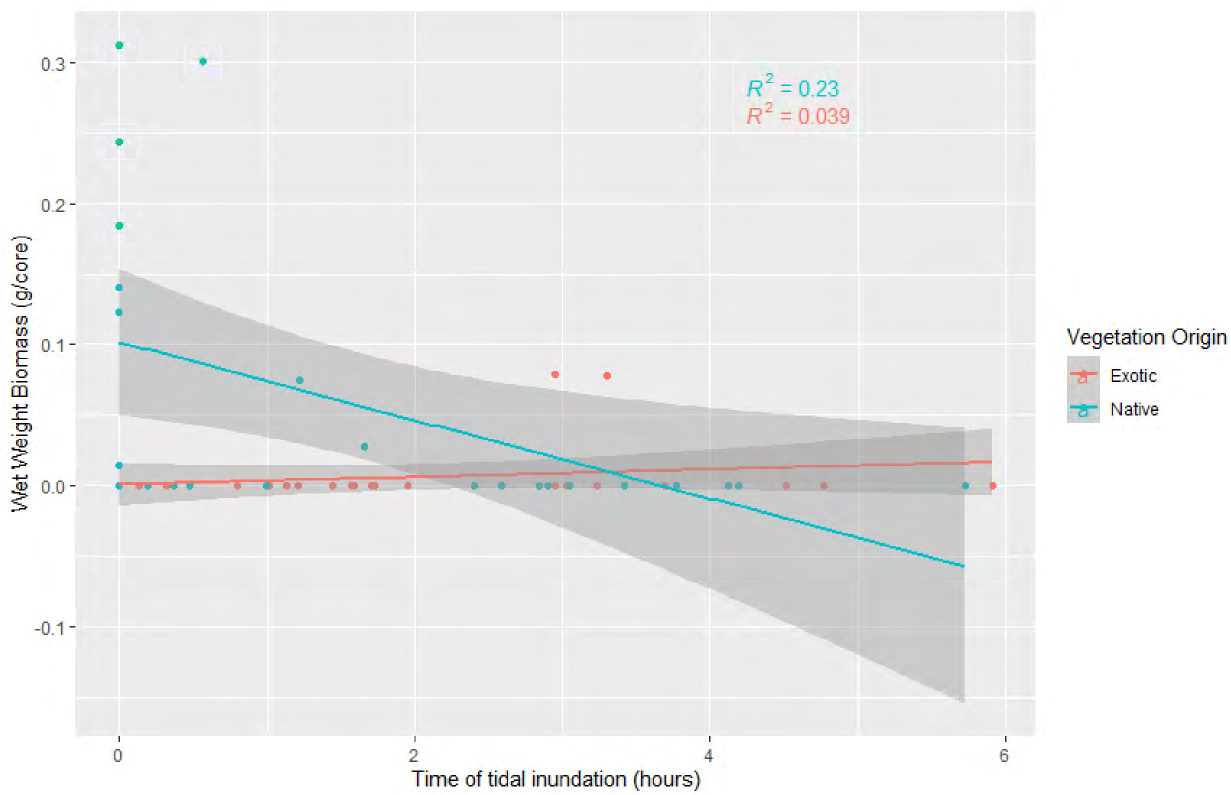

Figure 6 Scatter plot of biomass in grams/core, as a function of time of tidal inundation in hours and vegetation origin. Exotic vegetation is in coral and native vegetation is in turquoise. The points are individual samples, the solid line is the regression fit, the shaded region is the standard error of the fit, and the R^2 is the proportion of variation explained by the model.

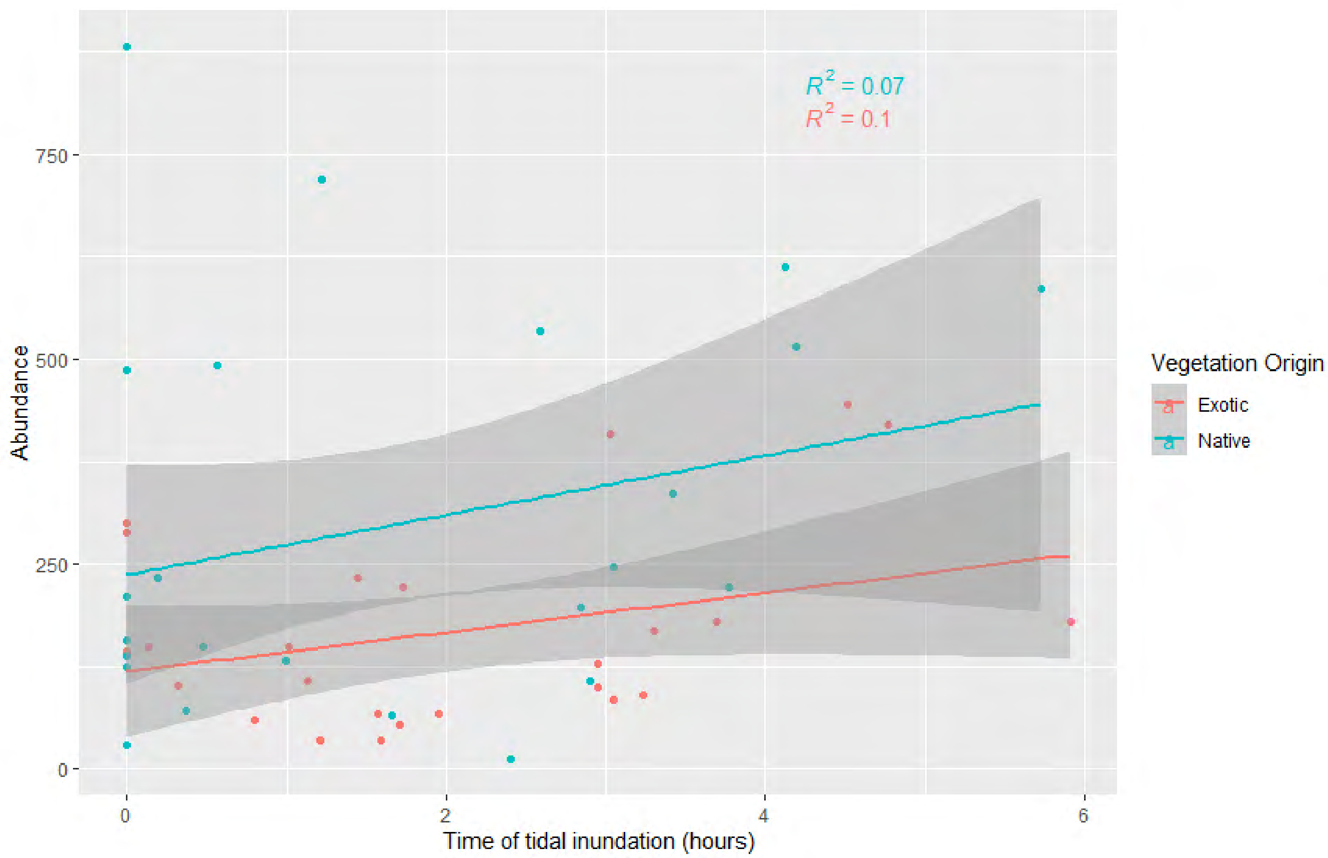

Figure 7 Scatter plot of abundance as a function of time of tidal inundation in hours and vegetation origin. Exotic vegetation is in coral and native vegetation is in turquoise. The points are individual samples, the solid line is the regression fit, the shaded region is the standard error of the fit and the R^2 is the proportion of variation explained by the model.

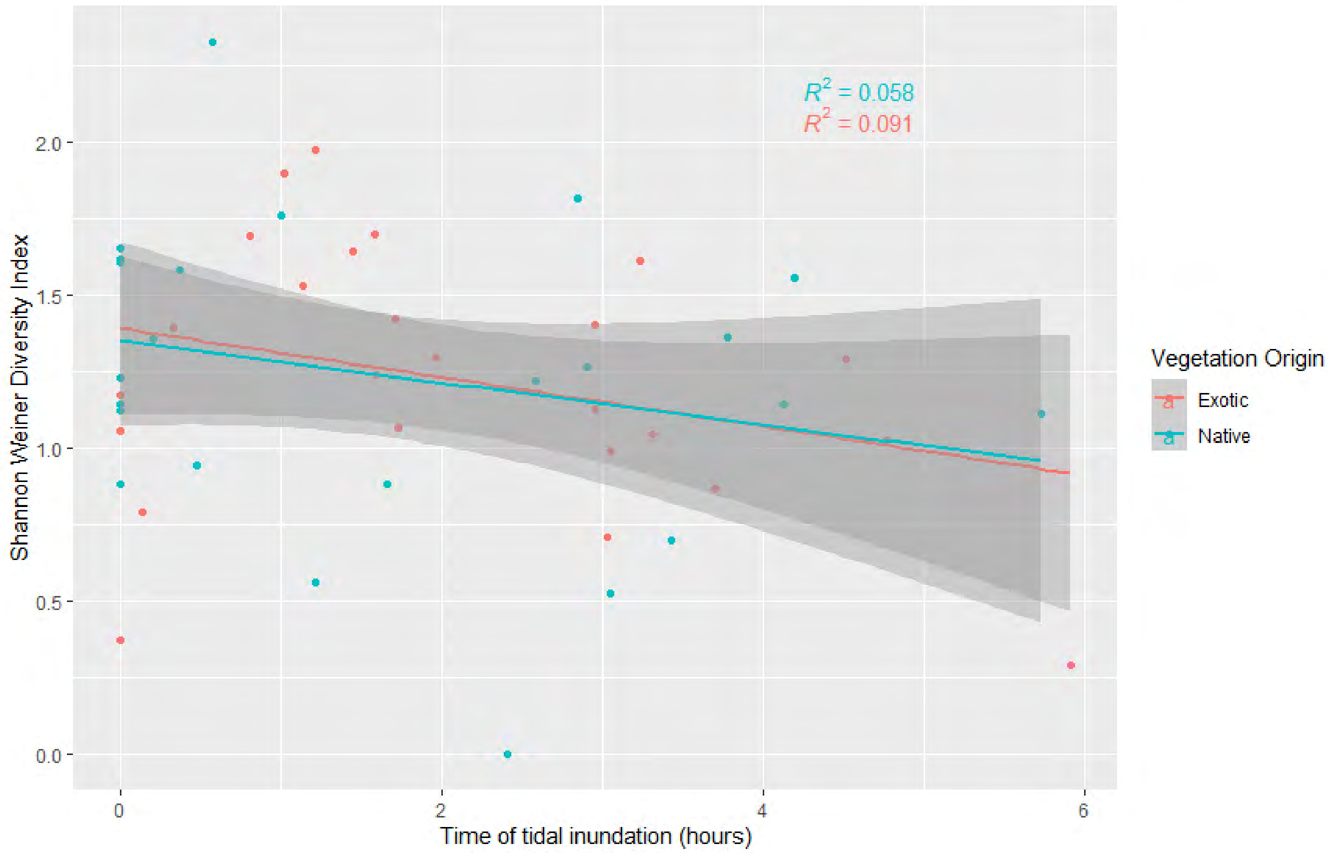

Figure 8 Scatter plot of Shannon Wiener Diversity indices as a function of time of tidal inundation in hours and vegetation origin. Exotic vegetation is in coral and native vegetation is in turquoise. The points are individual samples, the solid line is the regression fit, the shaded region is the standard error of the fit and the R^2 is the proportion of variation explained by the model.

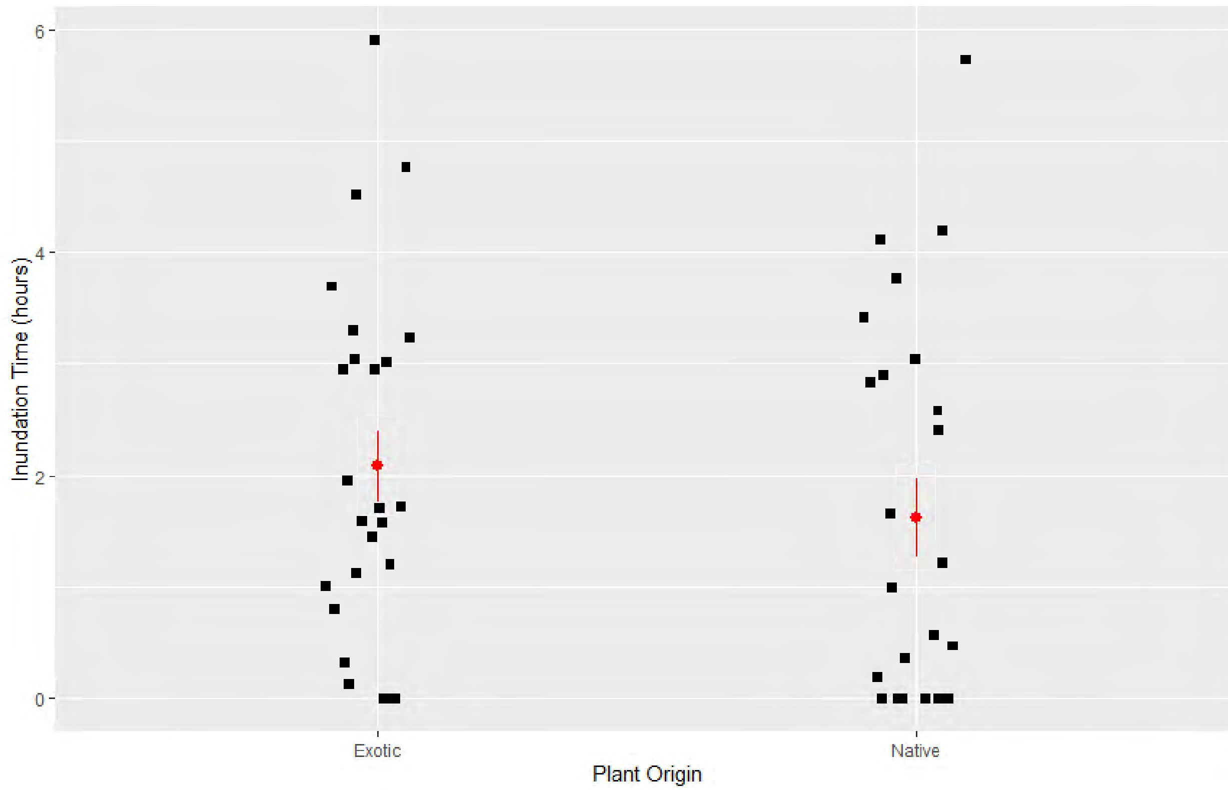

Figure 9 Dotplot of inundation time in hours of exotic cattail and stands of native vegetation in the South Arm marshes of the Fraser River Estuary. Each black dot represents a specific core, the red dot represents the mean, and the red lines on either side represent the standard error.

Discussion

Benthic macroinvertebrates were found to have lower abundances and biomasses in exotic cattail stands when compared to stands of native vegetation in the Fraser River estuary, but there was no detectable difference in diversity. This could be the result of a decrease in light penetration through the cattail, resulting in less light reaching the sediment, and a subsequent decrease in soil temperature (Shultz and Dibble 2012, Lawrence et al. 2016). A decrease in temperature resulting from the lack of solar energy could have a negative impact on invertebrates. Invertebrates are exothermic and decreases in temperature would result in delayed or impaired metabolism, growth, and activity (Mellanby 1939). The decrease in growth and metabolism could lead to a lower biomass and fewer reproducing individuals, resulting in lower abundances in subsequent generations. It has also been shown that impaired growth can have negative impacts on invertebrate fecundity (Tikkannen et al. 2000). The decrease in light penetration may be a result of the dense monoculture nature of the exotic cattail. It can grow well to water depths of 0.5 m, with a density of up to 108 stems/m^2, and grows up to 3 m tall (BCCDC 2020, Yang et al. 2020). This would shade the sediment more than native stands which are open in comparison.

Native stands also had a larger variation in the data for abundance and biomass within stand origin compared to the variation within exotic stands. This variation could be a result of cattail density. The density and height of the sampled exotic stands could create a more uniform environment within the stand, which could result in lower and more consistent benthic biomass and abundance in exotic stands. Native stands, however, were composed mostly of soft stem bullrush and lyngbei sedge. The native stands are less dense, have higher species diversity, shorter vegetation, and could have more dynamic abiotic conditions throughout the tidal cycle. The subsequent stand environment could be more variable, resulting in differing benthic abundance and biomass throughout the sampled native stands.

In exotic stands, impaired growth and reproduction could also be the result of allelopathic compounds and higher amounts of detritus. Exotic *Typha* have been shown to produce allelopathic compounds and some allelopathic compounds can impair growth and development of invertebrates which, again, can have negative impacts on future abundance (Erhard 2005).

Exotic cattail has been shown to have higher accumulated detritus, which is a result of mature cattail stems that scenesce at the end of the growing season, some of which can remain erect for two years post senescence (Vaccaro et al. 2009). Over those two years the stems can constantly add to the fallen detritus on top of the sediment, and frequent detrital enrichment is shown to decrease invertebrate abundance (Bishop and Kelaher 2007).

No differences in benthic diversity were detected between exotic cattail stands and native vegetation. This may indicate that stressors resulting in decreased abundance and biomass are impacting all taxonomic groups symmetrically. Alternatively, it could be the result of sample timing. The short sampling window may not incorporate the period where exotic cattail has asymmetrical impacts on invertebrate taxonomic groups. For example, some insects such as beetles use horizontally polarized light that is reflected off the surface of the water to find potential colonization sites, and cattail has been shown to impair water reflected polarized light (Molnár et al. 2011). Sampling at a time of year where the cattail is its tallest and has the least amount of reflected polarized light could result in a difference in diversity.

The differences in abundance (1.75 times higher in native stands) and biomass (8.85 times higher in native stands) are concerning with regards to salmon food webs and trophic implications. Future investigations should consider the direct impact this change in benthic invertebrates could have on salmonids given their cultural and ecological significance on the Pacific Northwest Coast (Willson and Halupka 1995, Noble et al. 2016).Future projects should also include variables that may impact invertebrate health such as temperature, rhizome density, and amount of detritus.

Given the invasive nature of exotic cattail and the lower abundances and biomasses that are associated with exotic cattail stands, these species should be removed in restoration projects where possible. Its removal can only benefit a system given these results and its known impacts on fish, amphibians, waterfowl, and vegetation (Heimer and Parsons 2013, Lishawa et al. 2015, Shrank and Lishawa 2019). Of the limitations to cattail removal in the South Arm Marshes of the Fraser River Estuary, the most difficult to address is likely site access. The estuary is in the middle of a fast-flowing river that sees high amounts of shipping traffic and recreation use, which can make traveling to the area a challenge. The estuary also experiences tides that can make parts of the marsh inaccessible for several hours during each tidal cycle. Complete eradication or removal is unlikely given the high cost and effort required, so long term management will be required to minimize spread (Stewart et al. 2021).

Given these results and the potential impact to higher trophic level consumers, restoration planners may need to rethink how invasive vegetation is managed. Considerations of how or when to prioritize invasive vegetation when trying to restore populations of species at higher trophic levels, such as salmon, may need to change. Further understanding of the impacts to benthic invertebrates and impacts to consumers will be important in long term management and restoration benefits for declining salmonid populations and associated habitats. There could also be similar impacts in other systems that may be subject to bottom-up impacts from invasive vegetation that could benefit from a better understanding of these bottom-up controls.

Conclusion

As expected, there were lower biomasses and abundances of benthic macro invertebrates in exotic cattail stands compared to native stands of vegetation. Contrary to expectations, however, no differences were detected in benthic invertebrate diversity. While further investigation is needed to explore impacts on juvenile salmonids, these results demonstrate a possibility for impacts to higher trophic levels. Given the negative impacts that have been documented in other ecosystems, exotic cattail should be removed where possible until we have a more comprehensive understanding of its negative impacts.

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Appendices

Appendix A

Sampling dates with corresponding sites. Site codes are EX for exotic cores (green on the map) and NAT for native cores (black on the map). The first number is the group number (1-6), and the second number is the core number.

Sampling Date	Sample Site
May 23, 2020	EX5-6, EX5-7, EX5-8, EX6-1, EX6-2, EX6-3, NAT5-6, NAT5-7, NAT5-8, NAT6-1, NAT6-2, NAT6-3
May 28, 2020	EX4-1, EX4-2, EX4-3, NAT4-1, NAT4-2, NAT4-3
May 30, 2020	EX3-1, EX4-4, EX4-5, NAT3-1, NAT4-4, NAT4-5
June 5, 2020	NAT5-1, NAT5-2, NAT5-3, NAT5-4A, NAT5-4B, NAT5-5, EX5-1, EX5-2, EX5-3, EX5-4A, EX5-4B, EX5-5
June 10, 2020	EX3-2, NAT3-2
June 12, 2020	NAT1-1, NAT1-2, NAT1-3, NAT2-1, NAT2-2, NAT2-3, EX1-1, EX1-2, EX1-3, EX2-1, EX2-2, EX2-3

Appendix B

Aerial view of field sites with site codes. Exotic sites are in green, native sites are in black

Appendix C

Map of the 54 exotic Typha stands, 10 m x 10 m or greater, in the South Arm Marshes of the Fraser River Estuary

Appendix D

Families of benthic macroinvertebrates and their abundances in the South Arm Marshes of the Fraser River Estuary

Family	Native Vegetation Abundance	Exotic *Typha* Abundance
Ampharetidae	16	16
Anisogammaridae	72	30
Ceratopogonidae	222	72
Chironomidae	1593	389
Chrysomelidae	32	20
Corophiidae	0	6
Dolichopodidae	31	6
Empididae	6	0
Enchytraeidae	1328	620
Entomobryidae	1	0
Ephydridae	0	12
Erpobdellidae	3	0
Halacaridae	36	12
Hydrobiidae	1114	666
Hydrophilidae	12	6
Hydrozetidae	0	6
Hygrobatidae	12	0
Hypogastruridae	0	6
Isotomidae	16	48
Limoniidae	92	37
Lumbricidae	10	9
Lumbriculidae	0	30
Naididae	2334	1990
Nereididae	1	0
Onychiuridae	78	21
Pediciidae	30	12
Psychodidae	60	55
Sphaeromatidae	21	14
Stygothrombidiidae	0	4
Tetrastemmatidae	0	1
Tipulidae	16	20

Appendix E

i) ANOVA table for benthic macroinvertebrate abundance

	DF	Sum Sq	Mean Sq	F Value	P-value
Time	1	83122	83122	2.4312	0.12580
Elevation	1	28269	28269	0.8268	0.36793
Origin	1	217082	217082	6.3494	0.01528
Residuals	46	1572716	34189		

ii) ANOVA table for benthic macroinvertebrate biomass

	DF	Sum Sq	Mean Sq	F Value	P-value
Time	1	0.032771	0.032771	6.6684	0.01306
Elevation	1	0.008274	0.008274	1.6837	0.20090
Origin	1	0.017745	0.017745	3.6109	0.06368
Residuals	46	0.226060	0.004914		

iii) ANOVA table for benthic macroinvertebrate Shannon Wiener Biodiversity Index

	DF	Sum Sq	Mean Sq	F Value	P-value
Time	1	0.7278	0.72784	3.5709	0.06511
Elevation	1	0.0899	0.08992	0.4412	0.50987
Origin	1	0.0009	0.00087	0.0042	0.94832
Residuals	46	9.3759	0.20382		

Appendix F

Inundation map for the study period of May 23rd to June 12th 2020. Pink dots are exotic cattail benthic cores, green dots are native vegetation benthic cores.