

Geochemical and Biological Response of an Intertidal Ecosystem to Localized Restoration Efforts: Squamish East Delta

**by
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ABSTRACT

Geochemical and biological attributes of three intertidal areas in the Squamish Estuary with different levels of disturbance (low, medium, and high) were assessed to determine short-term ecosystem responses to localized restoration efforts conducted one year previously on a former log handling site. Sediment and macroinvertebrate variables were analyzed among sites to characterize the ecosystems response and provide insight on the nature and process of an assisted successional trajectory. Invertebrate composition and biomass were lowest on the site with the highest level of disturbance. The high disturbance site also contained the highest percentage of fine sand (0.0067 mm to 0.25 mm). This confirms that in the short term there are distinct site responses to disturbance and ameliorative restoration efforts – even in a highly dynamic estuarine environment. The medium site contained more invertebrates than the low disturbance site indicating that something other than localized disturbance is affecting the invertebrate community on the low site. All sites exhibited a less-rich and less diverse invertebrate community than that of historical records (circa. 1970-1980). Invertebrate community in the east delta today is more typical of estuarine environments with higher salinity levels - which indicates more widespread levels of disturbance throughout the Estuary is affecting the study sites. This study highlights the importance of considering temporal and spatial scales when setting restoration goals, objectives and creating monitoring plans. Additional monitoring of sediment, invertebrate, and other variables on restored and reference sites is recommended to characterize typical recolonization and reassembly attributes of restoring intertidal estuaries in coastal British Columbia. This would provide evidence and rigor in determining effective restoration techniques and management strategies for a critical and increasingly threatened ecosystem.

Keywords: Benthic ecology; estuaries; intertidal flats; macroinvertebrates; restoration; sediment.

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TABLE OF CONTENTS

<i>Approval</i>	<i>ii</i>
<i>Abstract</i>	<i>iii</i>
<i>Acknowledgments</i>	<i>iv</i>
<i>List of Tables</i>	<i>vi</i>
<i>List of Figures</i>	<i>vi</i>
1.0 Introduction	1
2.0 Methods	2
2.1 Study Sites	2
2.2. Collection and Processing	4
2.2.1 Invertebrates	4
2.2.2 Sediment Variables	5
2.3. Statistical Analysis	6
3.0 Results	6
3.1 Sediment Characteristics	7
3.2 Sediment Core Data	7
3.3 Invertebrate Community.....	9
4.0 Discussion	10
4.1 Sediment Response.....	11
4.2 Invertebrate Response.....	12
4.3 Cumulative Effects	13
5.0 Recommendations	15
5.1 Monitoring Variables	15
5.2 Monitoring Locations.....	16
6.0 Summary	16
References	18
Appendix. Site Photographs	23

LIST OF TABLES

Table 1. Eigenvalues of principle components (PC) and identities 7
Table 2. Summary of ANOVA test results for sediment characteristics 8
Table 3. Coefficients and 95% confidence interval limits for sediment characteristics 8
Table 4. Endobenthic macroinvertebrate specie characteristics for study sites 10
Table 5. Results from Kruskal-Wallis rank sum test analyzing differences in invertebrate community among sites..... 10

LIST OF FIGURES

Figure 1. Map depicting of deltaic regions of the Squamish Estuary, current and historic hydraulic features, and study site location. 3
Figure 2. Locations of study sites and sampling plots in the Squamish Estuary 5
Figure 3. Principle component analysis (PCA) with sediment variables: percent coarse sand, medium sand, fine sand, silt, TOC, and water content. 8
Figure 4. Predictions of sediment variables disturbance sites from ANOVA model. 9
Figure 5. Estimated macroinvertebrate biomass by total and by specie among study sites 10

1.0 INTRODUCTION

Estuaries are one of the most productive and diverse ecosystems in the world; they are semi-enclosed bodies of water where freshwater from rivers and streams intermixes with saltwater of the ocean (Kennish, 2016; NOAA, 1990). Estuaries provide critical habitat to a diverse range of species (e.g. shorebirds, cetaceans, salmon, forage fish, marine-dependent mammals) and provide valuable ecological services to humans (e.g. dispersing nutrients, reducing effects from storm events, carbon sequestration) (Levings, 2016; Kennish, 2016; Campbell, 2015). At least 80% of coastal wildlife in British Columbia (B.C.) use estuaries for at least one life stage or behavior (BCCDC, 2006). Intertidal macroinvertebrates (e.g. bivalves, worms, barnacles, snails, crabs, limpets, amphipods) are a critical component of the estuarine ecosystem as they: i) provide a substantial food source to fish and wildlife species, ii) can be used as an indicator of contamination, and iii) perform ecosystem engineering functions (Heerartz *et al.* 2016, Borja *et al.* 2000).

Estuaries are naturally rare in B.C. and provide a critical ecological niche. However, they are among the most impacted ecosystems due to anthropogenic development along coasts and in upstream watersheds (BCCDC, 2006; Robb, 2014). Upstream land conversion for forestry, agricultural, urban, or industrial development can intensify water runoff, sediment and contaminant inputs to estuaries by increasing impervious surface areas, and erosion potential downstream (Kingsford *et al.* 2016; Kennish, 2002). Berms, jetties and river training dikes can cut off estuarine areas from sediment sources, disrupt salinity gradients, and thereby affecting invertebrate community response and recovery (Levings, 1980). In B.C., climate change is expected to increase precipitation and sea levels, as well as the frequency of storm events, which will affect sedimentation, temperature, and salinity gradients thus further altering trophic structure in estuaries (Austin *et al.* 2008). The Strait of Georgia has a higher than average amount of anthropogenic threats than other estuaries in the region and these threats are likely to increase in frequency and magnitude as populations continue to expand along coastlines (Kennish, 2002; Robb, 2014).

For the purposes of this study, restoration will be defined as: *“the act of partially or, more rarely, fully replacing structural or functional characteristics of an ecosystem that have been reduced or lost”* (Elliot *et al.*, 2007). Many studies have been completed both in B.C. and abroad to identify technical and management strategies for estuary restoration (e.g. identify drivers, remove stressors, reinstate structural features, public education) (Elliot *et al.* 2007; Ellings, *et al.* 2016; Kennish, 2012). However, only one-third of intertidal compensation sites along the Fraser River have been successful (Lievesley *et al.* 2016). Hence, additional research is needed to: i) determine factors contributing to restoration failure and ii) to identify opportunities to improve structure, function, and resilience of restoring- and restored- intertidal estuaries.

Borja *et al.* (2000) summarized that benthic invertebrate communities respond to habitat improvements (e.g. restoration) progressively over three stages: i) increasing abundance, ii) increasing diversity, iii) increasing pollution-sensitive species. Such community structure is typically improved by changing abiotic factors (e.g. sediment or water quality characteristics, or contamination removal) as many invertebrates have specific physical and chemical property thresholds (Gusamo *et al.* 2016). Therefore, it is important to monitor abiotic conditions as well as invertebrate communities in restored and reference intertidal estuary sites to determine trends.

The objective of this study was to quantify macroinvertebrate community and sedimentation responses to restoration efforts at three sites with varying levels of anthropogenic impact in the Squamish Estuary's east delta. A recently restored site is expected to have different characteristics than a site less impacted by anthropogenic development, due to altered successional trajectories (Walker *et al.* 2007). By performing multivariate analysis, the components potentially accounting for similarities and differences among the sites can be identified. For example, differences in invertebrate community abundance/diversity could indicate recolonization responses; additionally, dissimilarities in sediment characteristics may explain invertebrate zonation and heterogeneity on sites (Borja *et al.* 2010). Therefore, examining localized differences among sites could provide insight to the process and nature of an assisted successional trajectory – and thereby could help inform priorities for restoration treatments.

2.0 METHODS

2.1 STUDY SITES

The Squamish Estuary is a fjord head deltaic estuary located at the northern extent of the Howe Sound – an inlet draining to the Strait of Georgia in southern coastal B.C. It is the final drainage point for the Squamish River watershed which is composed of approximately 3650 km² of coastal temperate rainforest (Golder, 2005). The Estuary is within Squamish Nation traditional territory and has significant cultural and historical value including traditional fishing for pacific salmon, char and steelhead (MOE, 2007). The Estuary is a federally recognized Important Bird Area, forms a portion of the Skwelwil'em Squamish Estuary Wildlife Management Area, and supports a wide range of species including all six types of migratory pacific salmon (IBA, 2016; MOE, 2007; Golder, 2006). The steep cliff faces of the fjord geform help to maintain higher concentrations of salt water at lower depths which keeps the Estuary largely freshwater-to-brackish (Levings, 1980). Historically, the Estuary was a deltaic fan with the main river channel fluctuating east to west over time; the main channel was last observed flowing in the east delta in 1960 (Levings, 1976). River-derived sediment is the main source of deposits for deltaic estuaries and accretion has been known to occur rapidly in steep-sided fjord estuaries (Bianchi

et al. 2009). Prior to 1972, the Squamish Estuary was estimated to be building seaward at approximately 6.4 m per year (Bell, 1975). For descriptive purposes, the Squamish Estuary has been divided into four sections: west-, central-, east-, and Mamquam deltas (Figure 1).

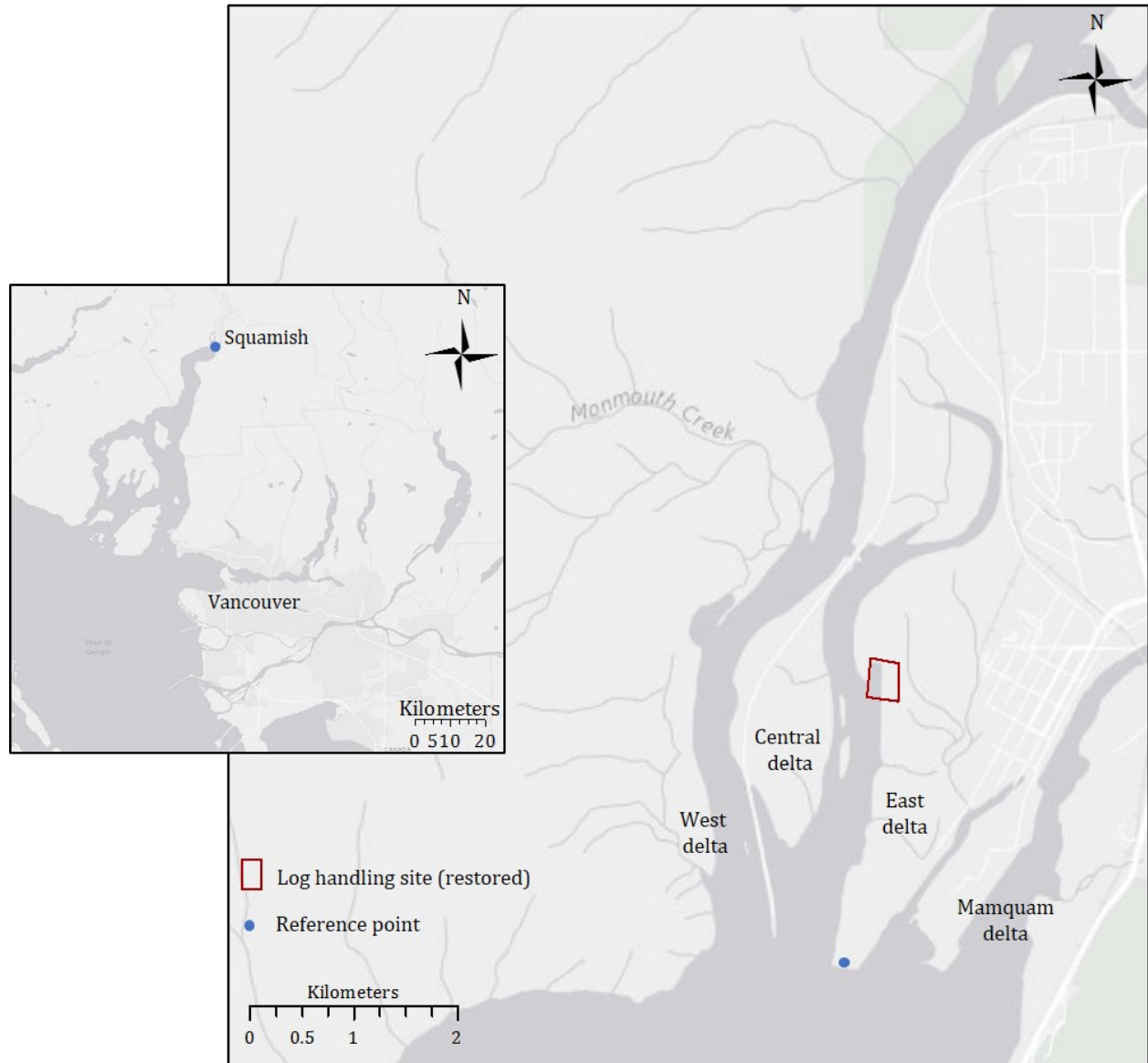


Figure 1. Map depicting of the deltaic regions of the Squamish Estuary and study site location. (Source: ESRI, HERE, DeLorme, MapmyIndia, © OpenStreetMap contributors and the GIS User Community)

There is an extensive history of development and restoration in the Squamish Estuary. The Mamquam River was redirected in 1921, which removed a significant amount of freshwater flow from the Mamquam delta to the west, central and east deltas (Levings, 1976; Stathers, 1958). In 1971, a large dike was constructed to train Squamish River through the western channel (Hoos and Vold, 1975). The dike is still in place today and has disconnected the east and central deltas from the main river

channel (MOE, 1982; MOE, 2007). From 1999 – 2013, ten (10) culverts were installed along the training dike to improve connectivity, but there is still evidence of elevated salinity levels, decreased fluvial sediment deposition, hindered fish passage, and increased tidal action in the east/central deltas (CORI, 2017; Golder, 2005; Levings, 1980). After the dike construction, the western channel was channelized and dredged spoils were placed in the central delta creating a disturbed area approximately 15 ha in size (Levings, 1973). Efforts to restore the dredge spoils area to a *Carex lyngbei* sedge meadow have largely been considered successful by returning structural function to the central delta (CORI, 2017; MOE, 2007). Furthermore, a log handling facility was constructed in the intertidal zone of the east delta in 1964. It was operational for fifty (50) years, and subsequently decommissioned, undergoing remedial restoration treatment in 2015-2016 (CORI, 2017; Hoos and Vold, 1975). Remedial treatment included re-establishing natural elevation gradients, tidal channel creation and vegetation planting. The goal of these treatments was to establish marsh meadow, tidal channel, and mudflat habitat types in the former log handling facility footprint (SRWS, 2016).

Three (3) lower intertidal areas in the east delta on and adjacent to the log handling facility, were chosen to determine localized sediment and invertebrate responses to a restoration treatment (Figure 2). Sites were classified with three levels of disturbance based on temporal and spatial distances to anthropogenic disturbance and a review of historical GIS imagery. The entire Squamish Estuary is considered disturbed from a landscape perspective due to dredging, dike construction, channel redirections, as well as urban and industrial development (CORI, 2017; Hoos and Vold, 1975). Disturbance levels for this study (high, medium, low) refer to differences in localized disturbance (i.e. log sort removal and restoration treatments). The high site (HS) was in the lower intertidal area of the historical log handling site footprint (0.39 ha). The medium site (MS) was located directly south of the log handling site footprint (0.33 ha). By contrast, the low site (LS) was located approximately 50 m west of the log handling site separated by a reconstructed tidal pool (0.35 ha). Study sites were established in discrete regions of lower intertidal flats between 0.4 and 2.0 m above chart datum. The HS was observed to have a large wood fiber mat which may be a relic of the log handling activities or decomposing sedge material from pre-handling facility era.

2.2. COLLECTION AND PROCESSING

2.2.1 INVERTEBRATES

Field surveys were completed between 21 June 2016 and 4 August 2016 at the lowest annual mixed semi-diurnal tides. Survey design was based on Gillespie and Kronland (1999) for intertidal invertebrate sampling and modified to include both epi- and endo- benthic communities. A y-axis transect was established perpendicular to the shoreline to isolate the tidal flat ecosystem. A 50-m x-axis transect was established along the low tide line; four additional transects were placed randomly

along the y-axis to ensure adequate site coverage. Fifteen quadrat plots (0.5 m x 0.5 m) were placed along each transect (three quadrats/transect). The location of each quadrat was determined using a random number generator. Each quadrat was sampled once, epi-flora and epi-fauna were identified, percent cover was estimated, and photographic evidence was taken. Sediment to 20 cm depth within the quadrat was excavated and stored in pails for separating. Sediment was passed through a 6-mm sieve to expose all macroinvertebrates (Whiteley, 2005). All benthic specimens were identified to the lowest taxonomic level possible, counted, labelled, packaged, and stored in a cooler until transported to the laboratory freezer.



Figure 2. Locations of study sites and sampling plots in the Squamish Estuary. (Source: ESRI DigitalGlobe, GlobeEye, Earthstar Geographics, CNE S/Airbus DS, USDA, USGS, AeroGRID, IGN, and the GIS User Community)

2.2.2 SEDIMENT VARIABLES

A sediment core sample was collected using a polyvinyl chloride (PVC) tube (5 cm diameter x 25 cm depth) adjacent to each invertebrate quadrat plot. The sediment specimen was stored in a re-sealable

plastic bag and transported to the laboratory. In the laboratory, the sample was cut into 1-2 cm increments (1 cm segments to 10 cm; 2 cm to 20 cm depth), homogenized, and stored in a freezer until detailed analysis was completed. Wet sieving was completed to determine grain size proportions. Samples weighing ca. 10 g were dried for at least 72 h at 60°C and burnt for 1 h at 400°C to remove particulate organic content. Three sieves were stacked together (coarse sand >0.5 mm, medium sand >0.25 mm, and fine sand >0.063 mm) and the sample was washed through three times with distilled water. Each fraction was then dried for 48 h prior to final weighing. The silt fraction was determined from the difference in weights from the dried sample and sum of sand fractions. All fractions were recorded and analyzed as percentages of the total dried sample weight. A portion of sediment cores (n = 8, randomly selected) were omitted from grain size analysis for low and medium sites due to time constraints. Total organic carbon (TOC) was determined with a >2 g sediment sample in accordance with Schumacher (2002). Wet weight was recorded and each sample was dried at 60°C for at least 48 h. Then, dry weight was recorded and the sample was placed in an oven for 1 h at 400°C to remove all organic content. TOC amount was determined from loss-on-ignition (LOI) calculation (i.e. difference in weights between the dry sample and after-burn sample) and recorded as a percentage of the sample (Wright *et al.* 2007). Sediment water content was determined by the recording the percent difference in wet vs. dry sample weight after 48 h @ 60°C.

2.3. STATISTICAL ANALYSIS

Data were analyzed using R version 3.3.2 and R Studio version 1.0.136 for Mac OS X 10.9.5. Data-frame manipulations, estimator predictions and transformations were completed in Microsoft Excel version 15.32 and with R packages: dplyr (Wickham and Chan 2016) and nlme (Pinheiro *et al.* 2016). Standard errors were calculated using plotrix package (Lemon, 2006). Graphing was completed using ggplot2 (Wickham, 2009), ggbiplot (Vu, 2011), ggthemes (Arnold, 2017), and plotly packages (Sievert *et al.* 2016). Principle Component Analysis (PCA) and rendering completed using devtools (Wickham and Chan, 2016), PerformanceAnalytics (Peterson and Carl, 2014), and FactoMineR (Sebastien *et al.* 2008). Significance was set to 0.05 and all outliers (>3 standard deviation from site mean) were removed from data set prior to analysis (Osborne and Overbay, 2004). Invertebrate data were log transformed $\log_{10}(x)+1$ prior to analysis to attempt to meet the assumptions of normality (McDonald, 2014). After transformation, data still followed a non-normal distribution but variances were equal thus, the Kruskal-Wallis Test was used to test for significant differences of invertebrate biomass among sites. Coarse sand, silt, TOC and water content data also followed a non-normal distribution, thus were arcsine square root transformed prior to conducting an ANOVA and PCA.

3.0 RESULTS

3.1 SEDIMENT CHARACTERISTICS

The PCA analyzes sediment characteristics and determine which variables account for the most variation among all sample plots and study sites. It indicates that the first principle component (PC1) contributes to 66.77% of variation among sites; whereas, the second principle component (PC2) accounts for 22.7% of variation (PC1 + PC2 = 89.46%). Including PC3 and PC4 accounts for a total of 98% of the variation in the sediment data. Table 1 lists eigenvalues and the identities of the variables within the first four PC.

Table 1. Eigenvalues of principle components (PC) and identities of variables contributing to the first four PC

PC	Variables Represented	Eigenvalue	Variance Percent	Cumulative Variance Percent
1	Silt, Water, Coarse Sand, TOC	3.032	66.77	66.77
2	Fine Sand, Coarse Sand, Medium Sand	1.334	22.68	89.46
3	Medium Sand, Water	0.772	6.03	95.49
4	Silt, TOC	0.485	2.57	98.06
5	NA	0.340	1.27	99.33
6	NA	0.027	0.67	100.00

Percent coarse sand was negatively correlated with fine sand, silt, TOC, and water along the PC1 axis; it also shows that percent water, silt, and TOC are correlated to each other (Figure 3). MS and HS sample plots are grouped by similar values of TOC, water, and medium sand. HS and LS do not overlap, demonstrating distinct characteristics, particularly in coarse and fine sand values. MS is also grouped with HS, indicating some similarities in coarse sand and silt proportions. Larger ellipses indicate more variation among sediment variables.

3.2 SEDIMENT CORE DATA

Sample profile analyses support PCA findings and detail how response variables vary across site and depth (Table 2, 3). Coarse sand on LS was significantly different than on MS and HS throughout core depth. Fine sand was significantly different among all three sites; MS and HS exhibited some trait convergence at lower depths. HS and MS contained similar, constant proportions of medium sand throughout core depth; whereas, LS exhibited a decrease in medium sand proportions as the depth increased (Figure 4). Silt proportions were consistently greater on MS; again, LS revealed a gradual decrease in proportion as depth increased. Over total depth, proportions of TOC were lower and less variable on LS than MS and HS. There was no apparent difference among sites for proportions of water content. ANOVA results (Table 2) show that depth is the factor accounting for the most differences among sediment variables. Site is also a significant factor however, the interaction of depth: site together is not.

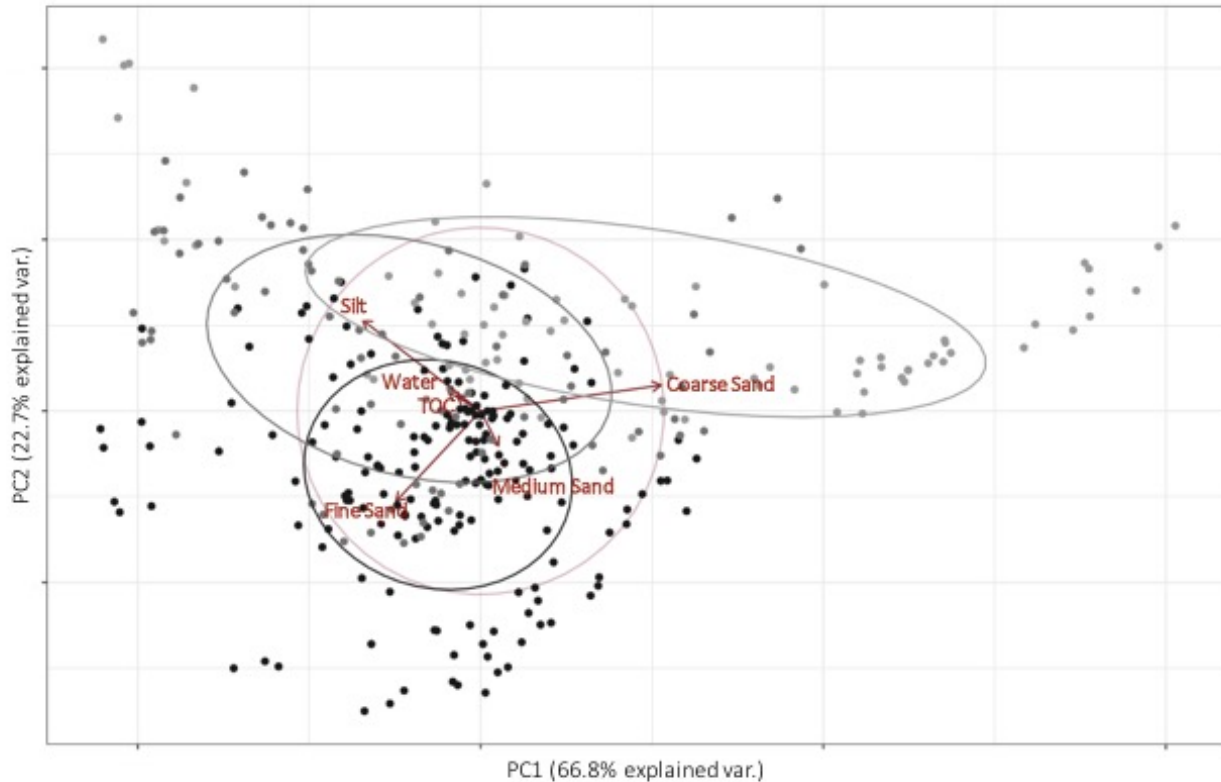


Figure 3. Principle component analysis (PCA) with sediment variables: percent coarse sand, medium sand, fine sand, silt, TOC, and water content. The first principle component accounts for 66.8% variation among samples (based on eigenvalue calculations) with the low site (LS) being most different from other sites. Ellipses delineate different sites and show variation of variables within sites. Black is high site (HS); grey is medium site (MS); light grey is low site (LS).

Table 2. Summary of ANOVA test results for sediment characteristics; df depth: 1, df site: 3, df depth: site: 2

	COARSE SAND	MEDIUM SAND	FINE SAND	SILT	TOC	WATER CONTENT
Depth	F: 6474.000 p: <2.2e-16***	F: 5447.366 p: <2.2e-16***	F: 5687.102 p: <2.2e-16***	F: 5376.034 p: <2.2e-16***	F: 7645.68 p: <2.2e-16***	F: 4576.890 p: <2.2e-16***
Site	F: 848.060 p: <2.2e-16***	F: 878.295 p: <2.2e-16***	F: 985.303 p: <2.2e-16***	F: 783.087 p: <2.2e-16***	F: 973.03 p: <2.2e-16***	F: 548.022 p: <2.2e-16***
Depth: Site	F: 12.250 p: 0.0001***	F: 25.227 p: 2.247e-07	F: 1.656 p: 0.206	F: 14.536 p: 2.92e-05***	F: 4.705 p: 0.0155*	F: 0.772 p: 0.479

Significance codes: 0 '***' 0.01 '**' 0.05 '*' 0.1 '.' 1

Table 3. Summary of coefficients and 95% confidence interval limits for sediment characteristics

	COARSE SAND		MEDIUM SAND		FINE SAND	
	Coefficient	95% CI	Coefficient	95% CI	Coefficient	95% CI
site: low	0.60	0.563 – 0.640	0.171	0.160 – 0.183	0.24	0.216 – 0.268
site: medium	0.44	0.394 – 0.477	0.161	0.149 – 0.173	0.34	0.316 – 0.372
site: high	0.43	0.393 – 0.465	0.157	0.147 – 0.168	0.47	0.448 – 0.497
	SILT		TOC		WATER CONTENT	
	Coefficient	95% CI	Coefficient	95% CI	Coefficient	95% CI
site: low	0.52	0.483 - 0.559	0.107	0.098 – 0.115	0.451	0.412 – 0.491
site: medium	0.57	0.533 – 0.616	0.156	0.146 – 0.167	0.491	0.446 – 0.536
site: high	0.41	0.375 – 0.447	0.123	0.114 – 0.133	0.475	0.434 – 0.518

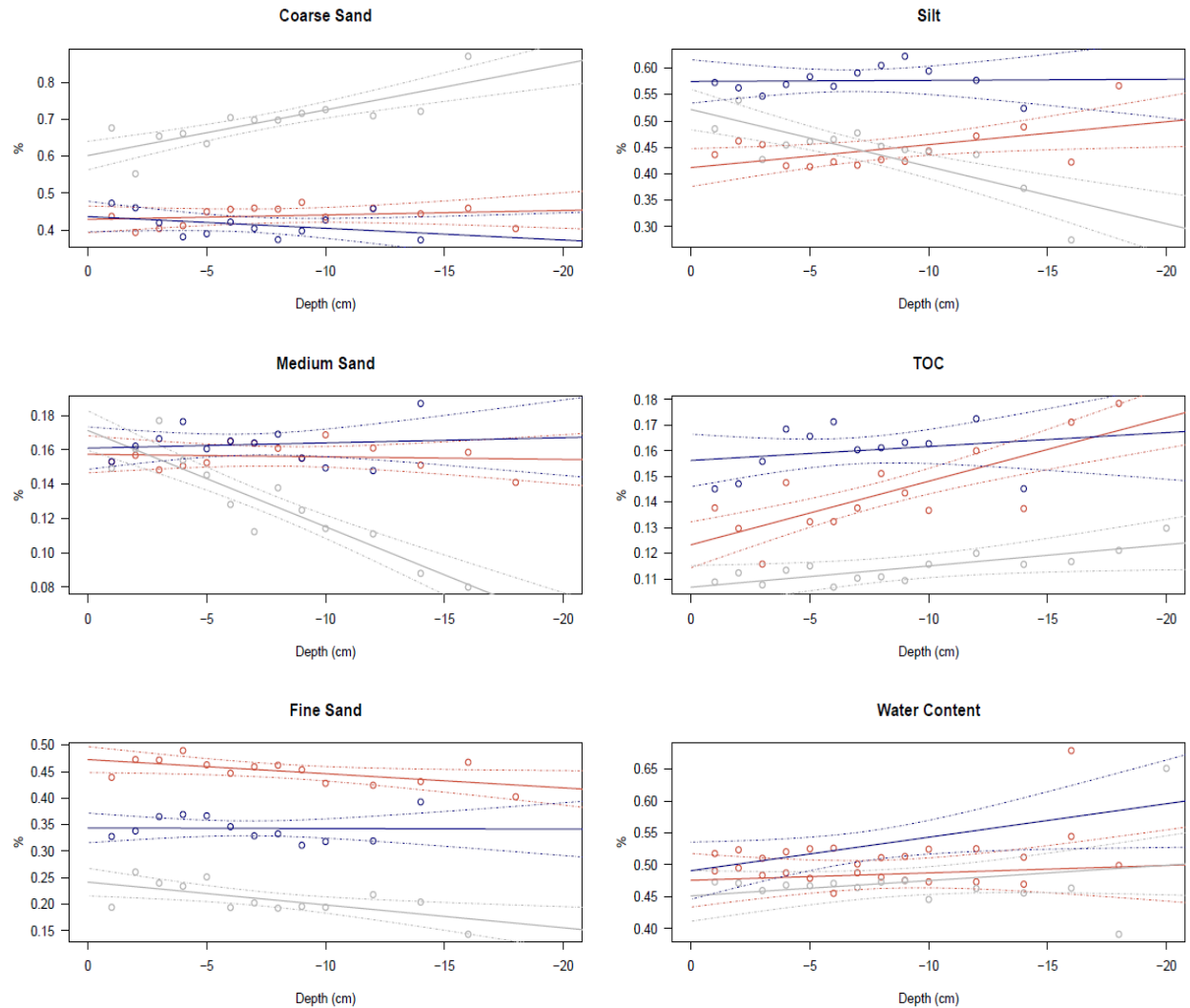


Figure 4. Predictions of sediment variables disturbance sites from ANOVA model. Solid lines are best fit; dotted lines are 95% confidence intervals. Red is high site (HS); blue is medium site (MS); grey is low site (LS). Proportions of coarse sand, silt, TOC, and water content were non-normally distributed and arcsine square-root transformed prior to modeling and graphing. Coarse sand is > 0.5 mm, medium sand is 0.50 mm to 0.25 mm, fine sand is 0.25 mm to 0.067 mm and silt is < 0.067 mm.

3.3 INVERTEBRATE COMMUNITY

A total of 4646 individuals were collected from the sample sites ($n = 15 \times 3$), which cover two taxonomic groups (*Macoma spp.* and *Glycera americana*). All sites contained both taxonomic groups in the endobenthic community; in the epibenthic community, two *Macoma spp.* were observed on two separate sample plots (one on HS; one on MS). The species assemblage was similar on MS and LS as *Macoma spp.* as dominant; whereas on HS, *G. americana* was dominant. MS has the highest biomass and species count, followed by LS. HS contained the least number of species and biomass among sites (Table 4 and Figure 5). The distribution of bivalves was patchy as these were only found on 33 of 45 quadrats sampled (2-398 individuals/m²). Total and *Macoma spp.* biomass were

determined to be significantly different among all sites; however, *G. americana* biomass is statistically equal among sites (Table 5).

Table 4. Summary of endobenthic macroinvertebrate specie characteristics for study sites

	Average species (count/m ²)			Average wet weight biomass (g/m ²)		
	<i>Macoma spp.</i>	<i>G. americana</i>	Total	<i>Macoma spp.</i>	<i>G. americana</i>	Total
High	3.1	8.1	11.2	0.24	1.24	1.48
Medium	211.7	4.3	216.0	45.74	0.51	46.25
Low	79.1	3.3	82.4	29.36	0.52	29.87

Table 5. Results from Kruskal-Wallis rank sum test analyzing differences in invertebrate community among sites

	chi-squared	df	p-value
<i>Macoma spp.</i>	30.011	2	3.043e-07
<i>G. americana</i>	0.1489	2	0.9282
Total	28.888	2	5.34e-07

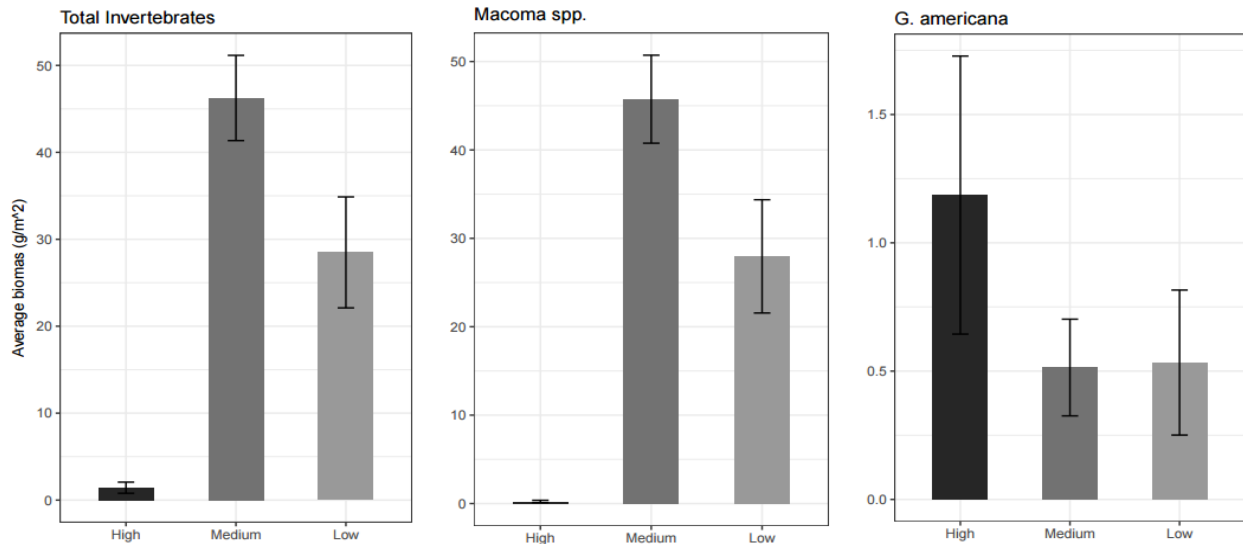


Figure 5. Estimated macroinvertebrate biomass per m² (mean +/- SE), by total and by specie among study sites

4.0 DISCUSSION

The objective of this study was to determine short-term sediment and invertebrate responses to a localized disturbance and restoration effort. Generally, disturbance and restoration are shown to affect proportions of sand (coarse, medium, fine), as well as invertebrate composition and biomass. However, as the study sites are all located within an estuary with a long history of disturbance, contrasting results from this study to: i) historical ecological records, and ii) known cumulative effects from regional disturbances, is valuable for determining restoration objectives and indicator variables for monitoring similar ecosystems in the region.

4.1 SEDIMENT RESPONSE

Results show that restoration on HS has generally matched localized sediment conditions (MS and LS) for water content; HS also has similar medium and coarse sand proportions to MS. This similarity of HS and MS sediment attributes, as shown in the PCA, indicates restoration treatments were successful in matching certain aspects of the sediment regime. However, HS shows different trend throughout the sediment core for TOC, and silt; additionally, HS contains much more fine sand than both MS and LS. Seventy percent of the variation in sediment grain size among the three sites was explained predominantly by differences in coarse and fine sand. A Nanaimo B.C. case study also noted former log handling sites to contain slightly more fine grain size on average than reference sites (McGreer *et al.* 1984). Log handling facilities have been shown to increase sediment compaction, reduce pore water space, decrease interstitial water circulation, and affect grain size proportions (McGreer *et al.* 1984). Furthermore, restoration projects using different grain size fractions can accelerate and or lower invertebrate population recovery to a site (Bilodeau and Borgeois, 2004; Peterson *et al.* 2000). Higher proportions of fine sediment and silt on the high-disturbance site could be indicative of increased compaction and decreased dissolved oxygen levels. Therefore, comparing sediment and grain size attributes among sites was an important feature of this study.

Though some site attributes of HS are distinctly different than MS and LS, it is important to consider the historical sediment regime to determine broader implications of the study on landscape ecology. Delta areas are heterogeneous environments where sediment deposition is primarily driven from gradual channel migration, abandonment, and fill (Gibson, 1994). Therefore, only general trends can be noted due to high local variability. There is an absence of sediment surveys in the intertidal flats in Squamish; earliest field-recorded information was recorded in 1973, one year post dike construction (Bell, 1975). Fortunately, there have been many analyses of historical photos as well as inferences made on the nature of the pre-disturbance Squamish estuarine environment (Gibson, 1994; Bell, 1975, Levings, 1976). These studies indicate that pre-1972 sediments were primarily river-derived and were deposited during the spring freshet (Bell, 1975). Since 1972 there has been increased erosion in the west delta, due to channelization and increased velocities; the central and east deltas are receiving fewer sediment inputs due to the hydrological restriction of the dike (Gibson, 1994).

The 1972 dike construction has been extensively studied confirming it has restricting freshwater surface flows to the central/east deltas (Golder, 2006; Gibson, 1994). Culverts have been installed since its construction with the primary goal of increasing fish passage for migrating salmonid species (CORI, 2017; Golder, 2006). The change in sediment and hydraulic distribution pattern is likely to have also affected vegetation and benthic populations. For example, increased salinity and finer-sand

proportions in the central/east delta will provide habitat suitable for different species than the coarse sand, fresh-water environment of the west delta (MOE, 2007).

In summary, the restoration efforts indicate that some sediment traits on HS are similar to those on MS (and to a lesser degree LS). Additional years of monitoring are needed to determine if there is a converging trend across ecosystem traits on study sites. However, if the objective is to restore the former log handling site and east delta to a free-flowing and sediment depositing environment, more representative of historical conditions, larger landscape level treatments will need to be considered.

4.2 INVERTEBRATE RESPONSE

The common clam *Macoma* spp. is an important species in the intertidal food web as it links primary producers to fish and shorebirds. (Harrison *et al.* 1999). As larvae, it is also a common food source for juvenile salmonids and flat fish (Cranford *et al.* 1985). Hence, it is promising to see high recordings of *Macoma* spp. biomass among LS and MS (29.36 g/m² and 45.75 g/m² respectively). These quantities are significantly higher than those previously reported in a contaminated and reference intertidal sites in the Fraser River Estuary (up to 10.9 g/m²) (Levings and Coustalin, 1975). This could be indicative of a 'Stage 1' response (i.e. increasing abundance of stress tolerant species; Borja *et al.* 2000) in the medium- and low disturbance sites. MS contained a larger quantity of *Macoma* spp. and total invertebrates, which indicates preferable environmental conditions in comparison to HS and LS. The elevated quantity of *Macoma* spp. on MS could be evidence of the Intermediate Disturbance Hypothesis – which proposes that plant and animal community diversity will increase with intermediate levels of disturbance as opposed to pristine or highly impacted conditions (Connell, 1978). Additional research is required to determine what factor (e.g. elevation, gradient, salinity, tidal influence, etc.) is contributing to the increased abundance of *Macoma* spp. on the MS.

Macoma spp. are stress tolerant and resilient to discharge levels, contamination, nutrient loading, grain size, and carbon loading (Harrison *et al.* 1999; McGreer *et al.* 1984). It is often one of the first species to recolonize intertidal sites – known to appear within 2-5 months' post-disturbance (Rossi and Middelburg, 2011; McGreer *et al.* 1984). Subspecies *M. balthica* can vary their feeding strategy from suspension to deposit feeding depending on the sediment grain size, confirming trait plasticity (Kamermans, 1994). *Macoma* spp. have been used as a bio-monitor of contamination for heavy metals as it feeds directly on deposited sediments and is often present (Harrison *et al.* 1999); thus, contamination is likely not an exclusive force explaining its absence from HS. It is unclear if differences in grain size, or another factor is contributing to the absence of this specie on HS; hence, additional monitoring of invertebrate recolonization to HS is recommended to better understand this observation.

Though *G. americana* was present on all sites, on HS it was dominant with a virtual absence of any macroinvertebrate competition. *G. americana* assemblages are typically related to well sorted, saline intertidal areas with fine-sand to silt and a low density of bivalves (Pastor de Ward, 2000). *G. americana* was not noted in historical invertebrate studies in the east delta; as it is a saline-associated species, this could be an indicator of elevated salinity levels in this region. Elevated salinity levels have been noted in post-dike studies showing surface salinity in the east delta at 30 ‰; compared to 6 - 14 ‰ in the west delta by the mouth of the Squamish River (Levings, 1976). The branched gills of *G. americana* increase gaseous exchange ability and allow it to tolerate low oxygen conditions in organically rich sediments (Mangum, 1976). This indicates probable elevated salinity levels and organic enrichment in the study sites compared to historical conditions.

Historical studies show that the east delta was once much richer and diverse for macroinvertebrate species. Surveys were completed throughout 1972-1977 in the east delta to determine invertebrate response from the training dike installation (Levings, 1980). Barnacles (*Balanus glandula*) and blue mussels (*Mytilus edulis*) were observed on solid substrates in the central delta in a summer field survey in 1972. However, neither of these species were observed within ten (10) months post dike completion indicating a delayed community response to disturbance (Levings, 1980). In soft-bottom habitats, *M. edulis* are correlated most strongly with positive terrestrial gravel, and negative organic enrichment; they also exhibit weak negative correlations to proportions of silt-clay (Commito *et al.* 2008). Other macroinvertebrates noted during central delta surveys include the cockle (*Clinocardium nuttallii*), *Macoma* spp., and Polychaetes (*Amphicteis* spp., *Eteone longa*, and *Pygospio elegans*) (Levings and McDaniel, 1976). Though few macroinvertebrates were recorded on the west delta, identified mesio- and micro- invertebrates were fresh water tolerant crustaceans and amphipods (Levings and McDaniel, 1976). By contrast, the Mamquam delta included more saline tolerant species including: Polychaetes (*Pygospio elegans*, *Manayunkia aestuarina*, *Glycera* spp., *Lepidonotus* spp. and *Eteone longa*), *Macoma* spp., *Mytilus edulis*, as well as crustaceans and amphipods (Levings and McDaniel, 1976). This historical date (circa 1972-1977) indicated a lateral salinity gradient across the Squamish Estuary. The historical presence of *Glycera* in Mamquam and current abundance found in the central delta indicate that the central delta has become more saline over time. In 1975, a marine-obligate specie, shipworms (*Bankia setacea*), were observed colonizing and destroying large wood debris structures in the central delta (Levings, 1980). Additional water quality and invertebrate sampling across the entire Squamish Estuary would be needed to confirm this notion.

4.3 CUMULATIVE EFFECTS

Differences in invertebrate community composition, and sediment characteristics among sites in this study show a possible correlation with level of disturbance (i.e. high- medium- low-). However, it is

difficult to determine causal relationships between anthropogenic activities and a specific environmental receptor in an ecosystem with many types of disturbances (e.g. effect from pulp/paper mills, hydroelectric, mining contamination, or aquaculture, etc.). In addition to the localized disturbances discussed (i.e. dike construction, long handling facility), there are many regional stressors. The Squamish Estuary and watershed contained a pulp and paper mill, which was known to discharge effluent and organic carbon into sediments of the Howe Sound (Burd *et al.* 2008). Log storage areas can cause sediment compaction, accelerated erosion and disruption of habitat on site and in adjacent areas, particularly in the lower-intertidal and sub-shallow subtidal areas (Burd *et al.* 2008). Wood fiber mats, deposited from pulp/paper mills or log handling facilities (such as observed on the high-study site) can hinder the recovery of benthos due to reduced biotic activity and low penetrability (Burd *et al.* 2008; Conlan and Ellis, 1979). Heavy metal contamination (e.g. copper, aluminum, iron, zinc and manganese) from mining historic and current mining and industrial activities (e.g. Britannia Mine) have been documented throughout the Estuary and Howe Sound (MacDonald, 1991; CORI, 2017). There are two (2) run-of-river and one (1) hydroelectric dam on the Squamish River and its' tributaries upstream of the Estuary (Energy B.C., 2017). Hydroelectric development inhibits water flow creating water holding areas and sediment impoundments upstream which can negatively impact estuarine bivalve population structure (Boominathan *et al.* 2014). The Squamish Wastewater Treatment Plant (SWWTP) discharges treated effluent to the Squamish River and removes dewatered sludge from its system for shipment to a nearby treatment plant for disinfection (Urban Systems, 2015). By 2031, the District of Squamish estimates the SWWTP will service 27,000 residents with a high estimate effluent rate of 0.1857 m³/s (based on high 7-day flow rate estimates) (Urban Systems, 2015). Wastewater treatment effluent release nutrients (i.e. nitrogen, phosphorous) which must be monitored to ensure the downstream ecosystem is not affected from oligo- or eutrophication (Efroymsen *et al.* 2006). Historically, the FMC Chlor-Alkali Plant was operational in the South Squamish Estuary (west delta). Several studies have been completed showing contamination and eventual remediation of methylmercury from the tidal flats and estuarine ecosystem (MOE, 2009). Analyzing regional and local disturbance histories is important in understanding the recovery of invertebrate communities and sediment characteristics in restoring a specific ecosystem (Lee and Khim, 2016). Failing to identify cumulative effects from regional disturbances could result in applying the wrong solution to the wrong place (Hobbs *et al.* 2007). To achieve successful restoration, investigation into the relative contributions of major regional stressors is needed (Davis, 2012; Baird 2005). This analysis will help scope restoration objectives and recommendations for returning structure and function to the Squamish Estuary and central delta.

5.0 RECOMMENDATIONS

5.1 MONITORING VARIABLES

This study represents the first investigation into the invertebrate community of the Squamish Estuary since a series of publications in the 1970s and 1980s. There are also many records and studies on micro- and meso-invertebrates (i.e. <6 mm) in the Squamish Estuary, which form an important component of the diet of young salmon (Levings, 1973; Levings, 1976; Levings, 1978; Levy and Levings, 1978; Levings *et al.* 1983). Though macroinvertebrates are a primary food source for many species (e.g. marine shore birds) including additional invertebrate size classes in future monitoring studies would provide additional rigor to trophic energy analysis and modeling. Robust historical records would offer a comprehensive baseline to render comparisons.

Many environmental variables affect invertebrate habitat suitability and could provide insight as to why the invertebrate community varies across disturbance sites and in contrast to historical records. A review of water quality gradients would confirm and/or clarify this study's suspicion that a hydraulic factor is contributing to differences in the invertebrate community among sites and across the entire Estuary. Water quality parameters to consider including: salinity, dissolved oxygen, turbidity, and temperature. Additionally, measurements accretion and erosion (e.g. sediment plates and erosion pins) would offer confirmation/clarification this study's analysis of local sediment deposition patterns.

Furthermore, studies have shown that woody debris are an important structural component of the estuarine environment affecting species behavior and utilization (McMahon and Holtby, 1992; Hood, 2007). Woody debris in estuaries are described over four size classes: organic detritus (<4 mm), fine woody debris (4-64 mm), coarse woody debris (64-256 mm), and very coarse woody debris (256-4,096 mm); and can be floating or embedded into substrates (FGDC, 2012). There are two identified knowledge gaps/opportunities relating to woody debris: i) analyzing variations of woody debris inputs (by size class) to sites in the Squamish Estuary, and ii) conduct an experimental treatment manipulating quantities of woody debris to determine affects to estuarine species.

Lastly, contamination from nearly a century of industrial activity around the Howe Sound is a present stressor to the Squamish Estuary. Historical sources of pollutants include: heavy metal leachates from the Britannia Mine site, dioxin and furan contamination from decommissioned pulp and paper mills. Whereas, ongoing pollution sources include: wastewater treatment effluent, spills from residential and commercial boat traffic as well as sunken vessels (CORI, 2017). Ecotoxicology analysis could determine if contamination factors are affecting biotic communities in the Squamish Estuary.

5.2 MONITORING LOCATIONS

Including more study sites in the Squamish Estuary, specifically in the west and central delta would establish a complete assessment of sediment and invertebrate characteristics in the entire Squamish Estuary. This would help to confirm this study's analysis of regional changes to the sedimentation regime as well as water quality gradients (i.e. suspected lateral salinity gradient). Documentation of invertebrate community assemblage (all size classes) across the estuary has not been completed since 1973 (Levings, 1976). There is proven value in updating regional baseline studies of ecosystems, particularly in environments subject to multiple stressors and disturbances (Lee and Khim, 2017; Kennish 2012). This baseline data would establish 'before' conditions which are vital for determining the magnitude and severity of impacts to critical ecosystems.

Additionally, other estuary habitat types (e.g. salt marsh, tidal channels subtidal) were omitted from this study and may have different responses to disturbance in the Estuary. Analyzing additional ecotypes could provide indication of common stressors or limiting factors. For example, subtidal eelgrass beds in the Squamish Estuary have been identified as an at-risk ecosystem (CORI, 2017). Log handling facilities, and the associated wood waste, have been identified as a potential threat to eelgrass survival and transplant success; investigative research is being conducted to determine if wood waste is a factor causing transplanted beds to fail (Adamson, unpublished data).

Finally, establishing a reference ecosystem in an estuary without large scale disturbance would be valuable for conducting monitoring and completing BACI experimentation. The Homathko Estuary (located at the head of the Bute Inlet,) is glacier fed with similar surface water temperature regime (4-12°C) and surface current speed (up to 700 cm/s⁻¹) to that of the Squamish Estuary (Levings *et al.* 1983). There are historical accounts of the Homathko Estuary geomorphology, vegetation and hydrological regimes which would be useful to compare to present conditions in Homathko and Squamish Estuaries (Levings, 1980).

6.0 SUMMARY

Estuaries are dynamic and productive ecosystems providing many valuable ecological services to wildlife and people. The variables analyzed here indicate that some sediment characteristics of a highly-disturbed site similar to, and some are different than less disturbed sites in a local environment. Localized disturbances from log handling and industrial activities in estuaries affect invertebrate species assemblages and a delayed recolonization response was observed on the highly-disturbed site. In comparison to earliest available data (circa. 1970-1980), there are significant differences in the present sedimentation and invertebrate characteristics of this Estuary. This difference is likely due to cumulative effects from urban and industrial developments throughout the Howe Sound and Squamish

Watershed. The persistence of a riverine training dike appears to continue to alter salinity gradients, sediment inputs and riverine influence rendering the central and east delta a lower-energy estuarine system. Monitoring additional parameters and locations in the restored sites in the Squamish Estuary as well as in reference ecosystems will help determine ecosystem responses to disturbance and effective techniques for restoring ecosystems in coastal B.C.

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APPENDIX. SITE PHOTOGRAPHS



Photo 1. Aerial view of high disturbance site post restoration. Photograph taken at high tide facing south west (C. Bates, 2015).



Photo 2. Aerial view of high, low, and a portion of medium disturbances site post restoration. Photograph taken low tide facing northwest (J. Smith, 2016).



Photo 3. High disturbance site, facing west. Photograph taken from perimeter road (E. Roberts, 2016).



Photo 4. Medium disturbance site. Photograph taken from high disturbance site, facing south (E. Roberts, 2016).



Photo 5. View of tidal channel separating low disturbance site from medium and high disturbance sites. Photograph taken from low disturbance site, facing northeast (E. Roberts, 2016).



Photo 6. View of 6-mm sampling sieve, invertebrates and fine woody debris on the medium disturbance site. (E. Roberts, 2016).

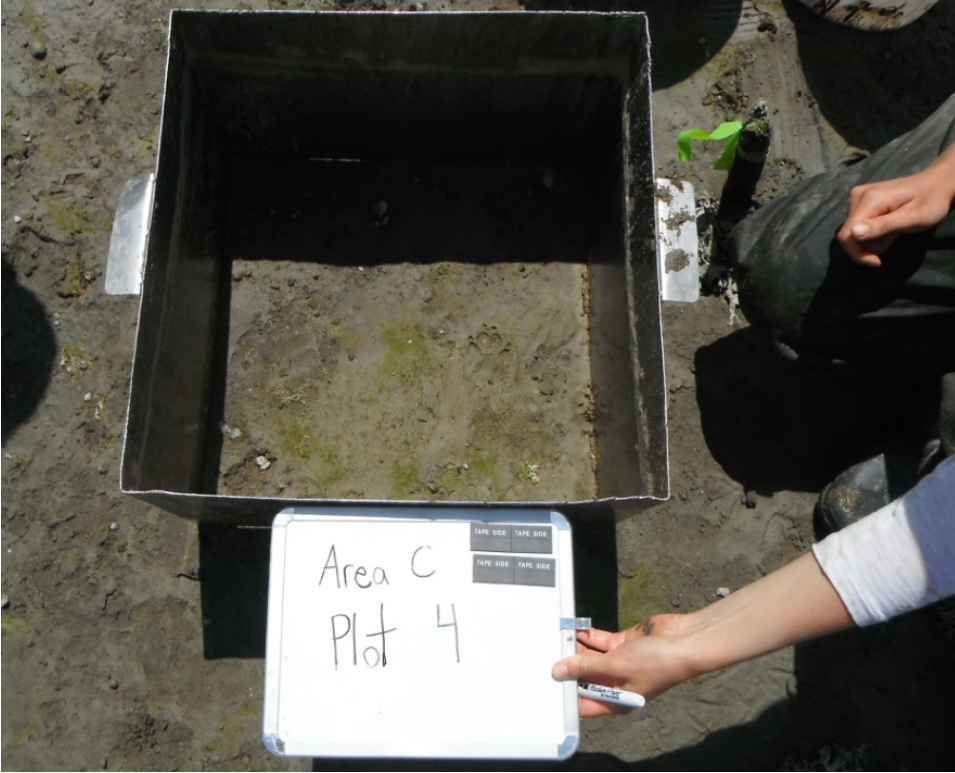


Photo 7. Typical quadrat sampling plot. Photograph taken at low disturbance site, facing down (E. Roberts, 2016).



Photo 8. Report author and supervisor identify invertebrates over 6-mm sieve. Photograph taken at medium disturbance site (E. Tobe, 2016).