Change in sediment features and the macroinvertebrate community within an estuarine ecosystem two years post-restoration

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Abstract. Our objective was to assess the response of an estuarine ecosystem to restoration efforts, two years post-restoration. Sediment attributes of particle size distribution (PSD), %LOI, water content and amounts of fine wood debris (FWD), and the macroinvertebrate community were compared among three sites, two reference and the recently restored site. The restored region had been previously used as a log sorting facility. As indicated by PSD, the restored site showed signs of recovery. However, the macroinvertebrate community had still not responded to restoration efforts. Sediments of reference sites were comprised of fine sand, and the macroinvertebrate community was dominated by *Macoma* spp. By contrast, at the restored site, sediments were mainly comprised of silt followed by fine sand, *Macoma* spp. was absent, and the main macroinvertebrate was *Glycera americana*, a polychaeta characteristic of disturbed regions. The restored site still contained significance amounts of FWD as compared to the two reference sites attributed to its previous use. Although still early in its recovery stage, active restoration did have a positive effect and will have likely kick started the region toward recovery and further follow-up in five years is recommended.

Key words: estuary; macroinvertebrates; restoration ecosystem response; sediment particle size.

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INTRODUCTION

Estuaries are one of the most productive and diverse ecosystems in the world. As defined by Kennish (2016) and NOAA (1990), they are semi-enclosed bodies of water where freshwater from rivers and streams intermixes with saltwater of the ocean. Estuaries are naturally rare in British Columbia (BC) and provide critical habitat for a diverse range of species (e.g., shorebirds, fish, marine-dependent mammals), and help regulate the global carbon cycle (Campbell 2015, Levings 2016, Kennish 2016). However, decades of anthropogenic disturbances have resulted in the degradation of most estuaries within the Pacific North West (BCCDC 2006, Robb 2014).

In 2002, the then Canadian Federal Department of Fishers and Oceans identified nine estuaries within BC for habitat restoration. The objective was to improve salmonid habitat to help maintain healthy wild salmon populations (Williams and Langer 2002). Notable was the Squamish Estuary, within the Squamish Nation traditional territory and traditional fishing

grounds (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 many species including all six types of Pacific salmon (Golder Associates 2006, MoE 2007, IBA 2016).

The estuary has been significantly altered since 1921 including the construction of a dyke that impeded water flow, hindered fish passage, and increased sedimentation. In 1964, the estuary was used as a log handling facility which was operational for fifty years when in 2015–2016, it was decommissioned and underwent restoration (Hoos and Vold 1975, CORI 2017). Remedial treatment included re-establishing natural elevation gradients, tidal channel creation, and vegetation planting. The goal of these treatments was to re-establish marsh meadow, tidal channel, and mudflat ecosystems (SRWS 2016). Hence, the opportunity was presented to assess the efficacy of the restoration and the ecological response of the region to restoration efforts. Further, although estuarine restoration efforts have been completed within BC, follow-up studies that provide information on the success of the restoration efforts are relatively few but necessary as such information is needed to allow for the development of effective restoration strategies based on known outcomes. Indeed, the importance of follow-up has been recently demonstrated by Kodikara et al. (2017) who assessed the effectiveness of the restoration efforts of mangroves in Sri Lanka. In their extensive study, all lagoons and estuaries within Sri Lanka which had been restored were included. It was determined that of the 1000–2000 ha of mangroves that had been replanted <10% had been successfully restored. These authors concluded that their findings are a stark illustration of the mismatch between the aims of restoration and the realities on the ground.

Our objective was therefore, through a comparison of the restored site to two reference sites within the estuary, to assess the success of the restoration efforts two years post-restoration. To meet this objective, we compared sediment characteristics (organic matter as %LOI, water content, and particle size distribution), amounts of fine woody debris (FWD), and the macroinvertebrate community among the three sites. Such information should provide restoration practitioners' information on how to most successfully restore in this case degraded estuarine environments.

METHODS

Study site

The Squamish Estuary is a fjord head deltaic estuary and is the final drainage point for the Squamish River watershed—composed of ~3650 km^2 of coastal temperate rainforest (Golder Associates 2005; Fig. 1a, b). Historically, the Squamish Estuary was a deltaic fan with the main river channel fluctuating east to west; 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 ~6.4 m per year (Bell 1975).

Three sites were chosen for this study (Fig. 2), two reference sites, REF1 and REF2, and the restored site, REST (Fig. 2). The REST site was in the lower intertidal area located directly on the restored log handling site (0.39 ha). REF1 was located directly south of the restored log handling site (0.33 ha), and REF2 was approximately 50 m west of the restored 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 tidal regime is typical of the Pacific West coast, with two highs and two lows in a tidal day. Mean tidal range is 3.2 meters with a maximum of 5 m.

Collection and processing

Macroinvertebrates.—Field surveys were completed between 21 June and 4 August at the lowest annual mixed semi-diurnal tides. Survey design was based on Gillespie and Kronlund (1999) for intertidal invertebrate sampling. A transect located at the high tide mark was placed parallel to the low tide mark to delineate the intertidal region for study. Along this transect, four 50-m transects perpendicular to the shoreline were established using a random number generator. Fifteen quadrat plots (0.5 × 0.5 m) were placed along each transect (three quadrats/transect; Fig. 2). The location of each quadrat

Fig. 1. The Squamish Estuary and study site location (Source: Google Maps). Site location (a) and (b) estuary post-restoration (2017).

Fig. 2. Locations of study sites and sampling plots in the Squamish Estuary (Source: Google maps).

was determined using a random number generator. Sediment to 20 cm depth within the quadrat was excavated and stored in pails. We used a 6-mm sieve based on the methods of Whitely and Bendell-Young (2007) to quantify the macrobenthic invertebrate community, that is, the dominant biomass, within intertidal regions of coastal BC. All benthic specimens were identified to the lowest taxonomic level possible, counted, labeled, packaged, and stored in a cooler until transported to the laboratory freezer for later biomass determination. Biomass estimates on a wet weight basis for numbers of invertebrates recovered from each site were determined by first thawing collected samples, blotting dry then weighing, and recording total biomass in grams wet weight to 0.01 g accuracy. Hence, for each

site, both number of individuals and the weight of the total number individuals were recorded.

Sediment attributes.—A sediment core sample was collected using a polyvinyl chloride (PVC) tube (5 cm diameter × 25 cm depth) adjacent to each invertebrate quadrat plot. PVC tubing was inserted to a depth of 20 cm where possible, extruded from the tube and tightly wrapped to prevent mixing, in a re-sealable plastic bag, and transported to the laboratory. In the laboratory, the sample was cut into 1- to 2-cm increments (1-cm segments to 10 cm; 2–20 cm depth), homogenized, and stored in a freezer until detailed analysis was completed. Wet sieving was completed to determine particle size distributions. Samples weighing ~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 (Wright et al. 2007). 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 and clay fractions were 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 the reference sites due to time constraints. Loss on ignition (% LOI) was determined with a 2-g subsample 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. % LOI amount was determined from loss-on-ignition (LOI) calculation (i.e., difference in dry weight and after-burn weight) and recorded as a percentage of the sample (Wright et al. 2007). Sediment water content was determined by recording the percent difference in wet vs. dry sample weight after 48 h at 60°C.

Fine woody debris (FWD) 6–256 mm in length.— As the restored area had been used as a log sorting facility, a novel indicator of restoration success in this case would be the amounts of FWD that has remained. Therefore, in addition to macroinvertebrates, all FWD retained by the 6-mm sieve was collected, packaged into re-sealable plastic bags, labeled, and stored in a cooler until transported to the laboratory freezer. A total of 45 samples ($n = 15 \times 3$) were processed and weighed. In the laboratory, all obvious non-woody debris (i.e., stones) was removed and discarded. All woody debris >256 mm in length was also removed. The sample was washed through a 4-mm sieve in triplicate to remove sediment particles from wood (wash effluent was processed separately to capture fibrous FWD). Each sample was dried at 60°C for 72 h to remove water content (Fourqurean et al. 2014). After drying, any remaining pieces of stone were removed. The dried sample was weighed and dried again at 60°C for 24 h. The dry weight was recorded when the weight of the sample had stabilized (i.e., ±2%; Giese et al. 2003). If the weight

had not stabilized (from 72 to 96 h), dry weight was recorded every 24 h thereafter until stabilization was achieved. Wash effluent (i.e., fibrous FWD and residual inorganics) was placed in a separate quality assurance (QA) sample tray. The QA sample was homogenized, dried at 60°C for 72 h, and weighed. Then, a subsample of the QA sample (~5 g) was collected and burned at 400°C for 1 h. FWD amount in QA samples was determined from %LOI calculation. The final FWD weight was recorded as the sum of the sample and QA sample.

Statistical analysis.—Data were analyzed using R version 3.3.2, R Studio version 1.0.136 for Mac OS × 10.9.5, and Microsoft Excel version 15.32. Data-frame manipulations, estimator predictions, and transformations were completed 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). Principal component analysis (PCA) and rendering completed using devtools (Wickham and Chan 2016), PerformanceAnalytics (Peterson and Carl 2014), and FactoMineR (Sebastien et al. 2008). A significance level of $P \leq 0.05$ was used for the statistical analysis and all outliers (>3 standard deviation from site mean) were removed from data set prior to analysis (Osborne and Overbay 2004).

Macroinvertebrate and FWD data were log-transformed $\log_{10}(x)+1$ prior to analysis to meet the assumptions of normality (McDonald 2014). Following transformation, data passed Levene's test for homogeneity of variance, but failed the Shapiro-Wilk normality test. Therefore, the Kruskal–Wallis test was used to determine significant differences. Coarse sand, silt, %LOI, and water content data also followed a non-normal distribution thus were arcsine square-root-transformed prior to conducting an ANOVA and PCA.

Results

Sediment characteristics

The PCA indicated that the first principal component (PC1) contributed to 66.77% of variation among sites, whereas the second principal component (PC2) accounted for 22.7% of variation
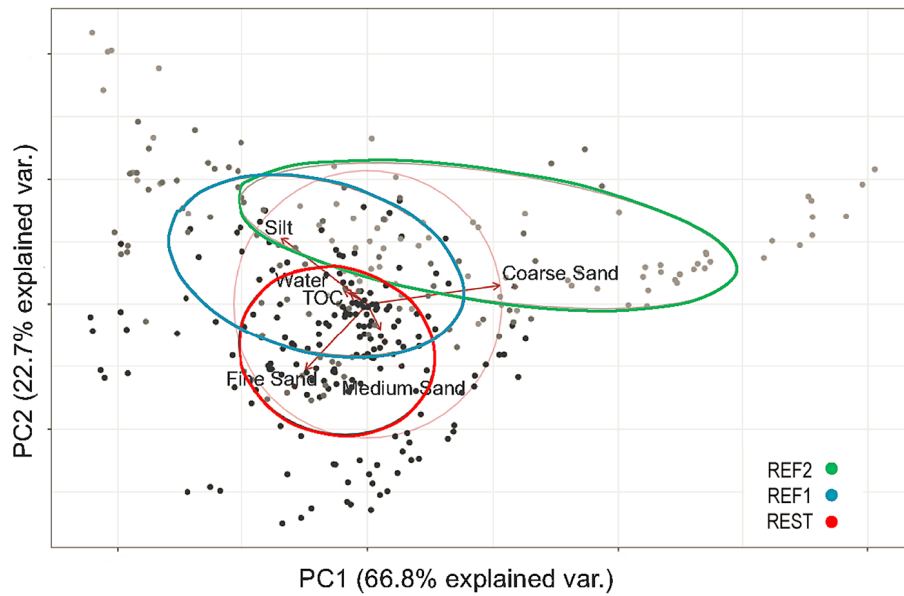
Fig. 3. Principal component analysis (PCA) with sediment variables: percent coarse sand, medium sand, fine sand, silt, TOC, and water content. The first principal component accounts for 66.8% variation among samples (based on eigenvalue calculations) with the REF2 site being most different from other sites. Ellipses delineate different sites and show variation of variables within sites.

(PC1 + PC2 = 89.46%; Fig. 3). Including PC3 and PC4 accounts for a total of 98% of the variation in the sediment data (Table 1). Percent coarse sand was negatively correlated with fine sand, silt, %LOI, and water along the PC1 axis. It also shows that percent water, silt, and %LOI are correlated to each other (Fig. 3). REF1 and REF2 sample plots are grouped by similar values of % LOI, water, and medium sand. REST and REF1 do not overlap, demonstrating distinct characteristics, particularly in coarse and fine sand values. Larger ellipses indicate more variation among sediment variables.

Sediment core: depth

Sample profile analyses supported the PCA findings and detail how response variables vary across site and depth (Tables 2 and 3). Coarse sand on REF2 was significantly different than on REF1 and REST throughout core depth. Fine sand was significantly different among all three sites. REF1 and REST exhibited some trait convergence at lower depths. REST and REF1 contained similar, constant proportions of medium sand throughout core depth, whereas REF2 exhibited a decrease in medium sand proportions as the depth increased.

Table 1. Eigenvalues of principal 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

Silt proportions were consistently greater on REF1. Over total depth, proportions of %LOI were lower and less variable on REF2 than REF1 and REST. 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.

Table 2. Summary of ANOVA (F) test results for sediment characteristics.

Source	Coarse sand	Medium sand	Fine sand	Silt	TOC	Water content
Depth	6474.000***	5447.366***	5687.102***	5376.034***	7645.68***	4576.890***
Site	848.060***	878.295***	985.303***	783.087***	973.03***	548.022***
Depth × site	12.250***	25.227***	1.656 NS	14.536***	4.705***	0.772 NS

Note: df depth: 1, df site: 3, df depth × site: 2.
*** $P < 0.01$, **$P < 0.05$, *$P < 0.1$, NS is not significant.

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

Characteristic	REF1 Coefficient	REF1 95% CI	REF2 Coefficient	REF2 95% CI	REST Coefficient	REST 95% CI
Coarse sand	0.6	0.56–0.64	0.44	0.4–0.48	0.43	0.4–0.6
Medium sand	0.17	0.16–0.18	0.16	0.15–0.17	0.16	0.15–0.17
Fine sand	0.24	0.22–0.27	0.34	0.32–0.37	0.47	0.45–0.5
Silt	0.52	0.48–0.56	0.57	0.53–0.61	0.41	0.37–0.45
TOC	0.11	0.1–0.11	0.16	0.15–0.17	0.12	0.11–0.13
Water content	0.45	0.41–0.5	0.5	0.45–0.53	0.48	0.43–0.52

Macroinvertebrate community

A total of 4646 individuals were collected from the sample sites ($n = 15 \times 3$), of which the dominant biomass wet weight was represented by two taxonomic groups, *Macoma* spp. and *Glycera americana*. Total and *Macoma* spp. wet weight biomass were determined to be significantly different among all sites (Kruskal-Wallis rank-sum test, $P < 0.05$; Fig. 4a, b). *G. americana* wet weight biomass was more prevalent at the restored site; however, due to the high variability, differences were not statistically significant ($P > 0.05$; Fig. 4c).

Fine woody debris

The mean FWD mass (gm^{-2}) was greatest for the REST, followed by REF1, then REF2 (Fig. 5; Kruskal-Wallis rank-sum test, $P < 0.05$).

Discussion

Just as important as the initial restoration of an anthropogenically disturbed site is the follow-up to ensure that restoration efforts were successful. Wortley et al. (2013) reviewed the literature from 1984 to 2012 to evaluate the number of studies where there had been follow up as to the success of the restoration activity. For this time period, follow-up studies were non-existent until 1994; however, there was a steady linear increase with the majority of studies occurring post-2008. However, during this time period, only 9% were conducted in riparian zones with no representation of estuarine ecosystems. Hence, while Wortley et al. (2013) noted an increase in the follow-up of restoration efforts, there was a strong bias toward terrestrial ecosystems with no reference to marine ecosystems.

Since 2012, there have been a number of estuaries within the Pacific Northwest that have undergone restoration (e.g., Lievesley et al. 2017), providing the opportunity to evaluate the success of restoration efforts as well as adding to the literature information on the success of the methods applied. Here, we report on the short-term response of an estuary as measured by changes in sediment attributes and the macroinvertebrate community to restoration efforts and provide information on whether restoration efforts were indeed successful.

Sediment response

Findings of our study, specifically the similarity of REST and REF1 sediment attributes, as shown in the PCA, indicate restoration treatments were successful in matching certain aspects of the sediment regime. Re-establishment of the sediment regime, that is, geomorphological conditions would then meet one of the prerequisites for estuary restoration as noted by
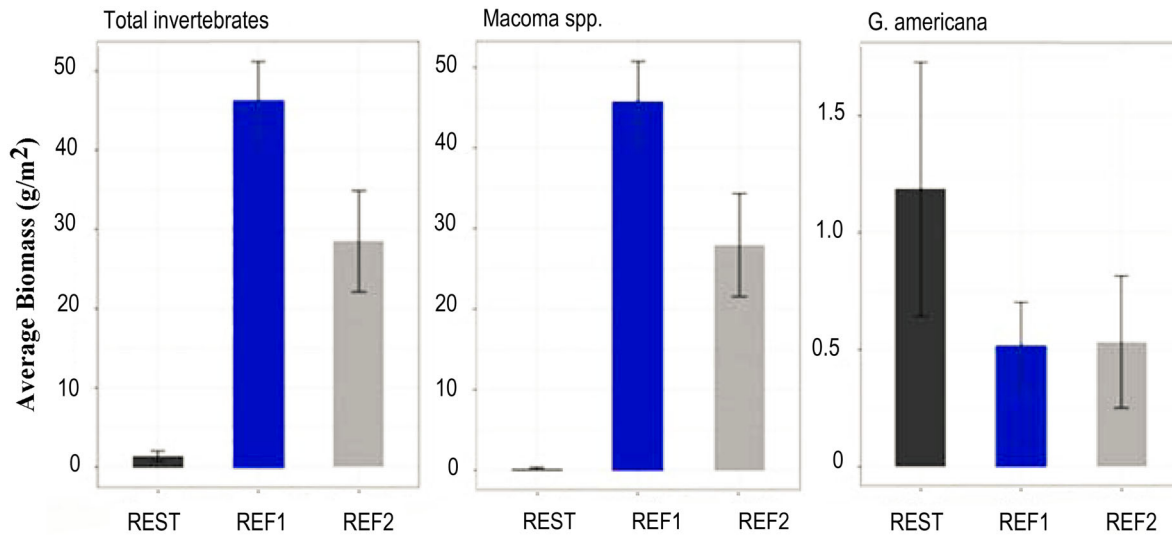
Fig. 4. Estimated wet weight macroinvertebrate biomass g/m^2 (mean ± standard error) by total and by genera among the three sites.

Nienhuis et al. (2002). However, the REST site was characterized by much more fine sand and silt, than both REF1 and REF2. Seventy percent of the variation 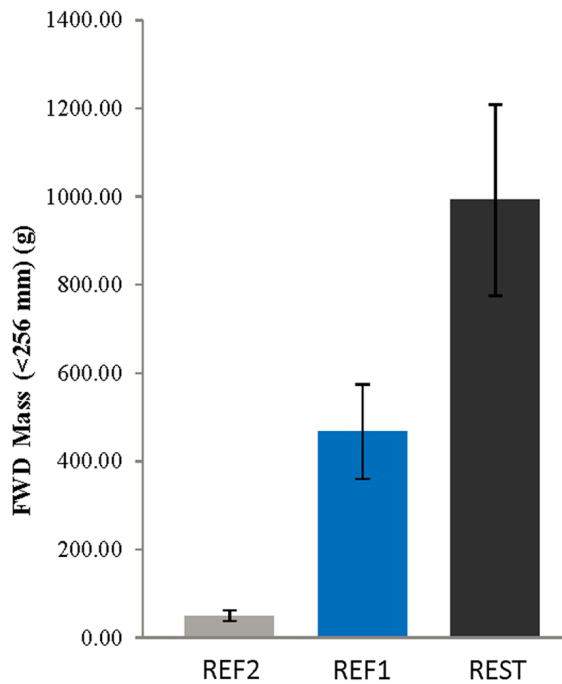in sediment grain size among the three sites was explained predominantly by differences in coarse and fine sand. A Nanaimo BC case study also noted former log handling sites contained slightly more fine grain size on average than reference sites (McGreer et al. 1984). Log handling facilities have been 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 that have altered sediment properties by using different grain size fractions can accelerate or lower invertebrate population recovery to a site (Peterson et al. 2000, Bilodeau and Bourgeois 2004). Higher proportions of silt on the restored site could be indicative of increased compaction and decreased dissolved oxygen levels. Therefore, comparing sediment grain size attributes among sites was an important diagnostic tool of this study.

Fig. 5. Estimated FWD biomass (g/m^2) among sites: mean ± SE.

Macroinvertebrate response

The macroinvertebrate community differed among the three sites indicating different environmental conditions. *Macoma* spp. dominated in the two reference sites and was absent at the restored site. By contrast, although not statistically significant due to high variability, *G. americana* occurred in greater amounts on the restored

site as compared to the two reference sites. 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). *Macoma* spp. are stress tolerant and resilient to discharge levels, contamination, nutrient loading, grain size, and carbon loading (McGreer et al. 1984, Harrison et al. 1999). It is often one of the first species to recolonize intertidal sites—known to appear within 2–5 months post-disturbance (McGreer et al. 1984, Rossi and Middelburg 2011). *Macoma balthica* can vary its feeding strategy from suspension to deposit feeding depending on the sediment grain size, confirming trait plasticity (Kamermans 1994). Despite this plasticity, *M. balthica* had yet to establish itself within the restored site, 2 yr post-restoration. The presence of *G. americana* at the restored site and the absence of *M. balthica* suggest that intertidal sediments tended toward anoxic conditions and had yet to recover to a healthy state as defined by the two reference sites. *G. americana* in contrast to bivalves have highly adapted branched gills that increase gaseous exchange allowing it to tolerate low oxygen conditions in organically rich sediments (Mangum 1976) such as those found at the restored site.

Fine woody debris

FWD biomass was significantly different among the three sites with the restored site containing significantly greater amounts as compared to the two reference sites. As the restored is on the footprint of the historic log handling site and it is likely that there is remaining excess wood waste that was not entirely removed through the remediation efforts (MOE 2007, Hodgson and Spooner 2016).

Recently, Campbell et al. (2019) measured sediment attributes and infaunal community of a soft-sediment estuary undergoing a passive reclamation from historical anthropogenic activities including salmon canneries and pulp mills. Impacts to the estuary occurred from 1889 to 2001 allowing the estuary close to 15 yr to recover. These authors found that indeed a passive approach was sufficient to turn the intertidal mudflat to a relatively productive, functioning,

and diverse ecosystem. Also noted was that a passive approach which produces similar outcomes as active restoration may be more cost-effective and hence preferable to active restoration.

Lv et al. (2019) measured response of macrobenthoic functional groups as indicators of ecological restoration in reclaimed intertidal wetlands within China's Yangtze Estuary. As with Campbell et al. (2019), time was the main factor contributing to the recovery of the wetlands however also noted was that active restoration of the wetlands yielded short-term positive outcomes. Woo et al. (2018) were also able to demonstrate that three years after active estuarine restoration, the ecosystem returned to its natural tidal influence, which resulted in the increase of numbers of key forage items for juvenile salmonid, such as amphipods and copepods in the newly accessible marsh channels. These authors concluded that restoration of estuaries may take decades, however, with active restoration, there was an initial rapid response resulting in enhancement of prey items for estuary-dependent wildlife.

Within the Squamish estuary, active restoration appears to have facilitated the recovery of the impacted site as indicated by sediment particle size returning to that found at one of the reference sites. Although the 2-yr time period did not provide the amount of time for the restored site to obtain a similar macroinvertebrate community, active restoration may serve to enhance and speed up the process. Time lines required for a passive restoration may not work well for species that rely on a healthy functioning ecosystem for specific stages in their life cycle such as the Pacific salmon that require estuaries for breeding on a yearly basis. Therefore, an advantage of active restoration could be to kick start the process of ecological succession. However, unknown is the trajectory that the restored ecosystem will take hence continued follow-up will be necessary to ensure that the restoration efforts are indeed successful.

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