Effects of logging-induced sediment loading on Chinook salmon rearing habitat in Tranquil Estuary, BC and Implications for Estuary Restoration

by

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Abstract

Research on estuaries has increased in recent years, however, the effects of logging on estuaries and the effects of estuary habitat loss on Chinook salmon (*Oncorhynchus tshawytscha*) in the Pacific northwest is limited. To address habitat loss associated with logging, I used an extensive aerial photo record for Tranquil Creek estuary and an unlogged control to analyze changes in salt marsh area, elevation and volume, supplemented with a grain size distribution analysis.

While I failed to find evidence of a difference between a logged and an unlogged estuary, some negative trends in salt marsh area and elevation observed over the observational period were indicative of changes that are unfavorable for juvenile Chinook salmon. Analytical methods presented here to assess changes in two remote coastal estuaries has contributed to the current knowledge on the effects of logging on estuarine ecosystems in coastal BC and provide tools for innovative estuary habitat restoration.

Keywords: *Chinook salmon (Oncorhynchus tshawytscha); salt marsh; estuary restoration; logging; sediment; aerial photograph analysis.*

Dedication

I dedicate this thesis to my father Ian Spice who unconditionally supports my academic and personal aspirations. I also dedicate this thesis to my mother Kim Spice, whose adventurous nature and love for the ocean lives in me.

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List of Acronyms

ANOVA	Analysis of Variance
BC	British Columbia
BCIT	British Columbia Institute of Technology
CAD	Computer Aided Design
CGVD28	Canadian Geodetic Vertical Datum of 1928
CSRT	Clayoquot Salmon Round Table
CU	Conservation Unit
CWFS	Central Westcoast Forest Society
DEM	Digital Elevation Model
DEM	Digital Elevation Model
DFO	Department of Fisheries and Oceans Canada
DoD	DEM of Difference
DTM	Digital Terrain Model
ESRI	Environmental Systems Research Institute
GIS	Geographic Information System
LF34	Limiting Factor 34
LWD	Large Woody Debris
MARS	Marsh Resilience to Sea-Level Rise
MC1	Moyeha Core 1
MHWL	Mean High Water Level
MLSL	Mean Local Sea Level
MLWL	Mean Low Water Level
NAD	North American Datum
NADV88	North American Vertical Datum of 1988
NERRS	National Estuarine Research Reserve System
NuSED	New Salmon Escapement Database System
PIT	Passive Integrated Transponder
PNW	Pacific Northwest
PSU	practical salinity unit
RCP	Representative Concentration Pathway
RST	Rotary Screw Trap
RTK	Real-time Kinetic
SA1	Study Area 1
SA2	Study Area 2
sd	Standard deviation
SE	Standard error
SET	Surface Elevation Tables
SFU	Simon Fraser University
SLR	Sea Level Rise
TC3	Tranquil Core 3
TSA	Terrain Stability Assessment
UTM	Universal Transverse Mercator
VICLMP	Vancouver Island Conservation Land Management Program

Aerial Photograph of Tranquil Creek Watershed in 1970. Sourced from GeodataBC (2019).

Applied Research Project By: Christine Spice

1. INTRODUCTION

1.1. Background

Historical land use practices have significantly changed the landscape in British Columbia (B.C.). As a result, watersheds affected by anthropogenic activities have been the focus of restoration studies in recent years (Keeley & Walters 1994; Simenstad & Cordell 2000; Simenstad et al. 2002; Hood 2014; Ellings et al. 2016). Vancouver Island has lost three quarters of its original old-growth forests to clear-cut logging practices (Mychajlowycz 2010). Clayoquot Sound, on the West coast of Vancouver Island (Fig.1), was historically a priority area for commercial timber harvest because of the extensive intact old-growth tributaries (CSSP 1995). Tranquil Creek was one of many watersheds in Clayoquot Sound that was logged. The Moyeha River is one of only two unlogged watersheds in Clayoquot Sound and was used as a reference study area for this project.

From 1950 to 1993, 27,000 ha of old-growth forest was logged in Clayoquot Sound (Mychajlowycz 2010). Industrial logging in Clayoquot Sound peaked in the late 80s with harvest rates reaching 1200 ha per year (CSSP 1995). Clear-cutting of intact old-growth forests in Clayoquot Sound triggered significant controversy over unsustainable forestry practices and the subsequent loss of ecosystem services on Meares Island (Berman & Leiren-Young 2011). The loss of old-growth forest ecosystems sparked the iconic Clayoquot Sound protests known as “War in the Woods” in 1993. Since then there have been significant changes to the forestry

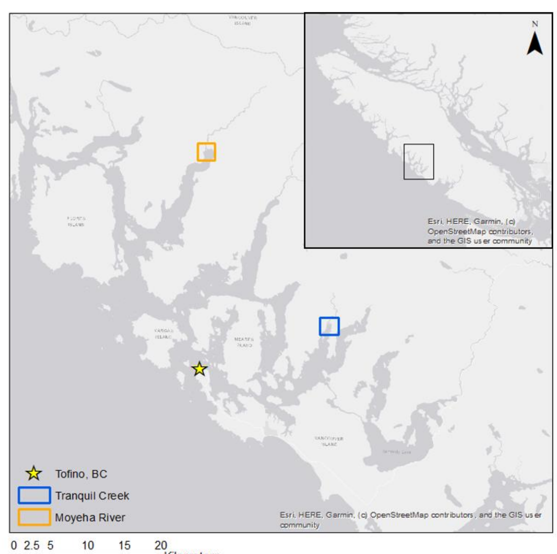

Figure 1. Study Sites: Tranquil Creek & Moyeha River. Both study sites are in Clayoquot Sound on the West Coast of Vancouver Island, BC, Canada. Tranquil Creek: Latitude: 49°13’54” N Longitude: 125°39’30” W Moyeha River: Latitude: 49°43’14” N Longitude: 125°90’95” W (Esri 2011).

industry both within B.C. and in Clayoquot Sound. The environmental legacy of these industrial harvests, however, are still apparent and will be for decades to come. It is for this reason that there is significant focus on restoring degraded watersheds and protecting intact old-growth forests (Berman & Leiren-Young 2011).

There is an extremely good historical aerial photograph record for Tranquil Creek and Moyeha River watersheds. This data can be analyzed to track environmental changes over time by comparing the impacts of land use at a large spatial scale (Morgan et al. 2010). This rare and invaluable resource provides crucial information on conditions prior to industrial land use and can be used to help set restoration objectives (Hales 2000; Ellings et al. 2016).

The effects of forestry practices on small coastal watersheds have been reported in Clayoquot Sound and have been rigorously studied using unlogged controls in Carnation Creek on Eastern Vancouver Island (Hartman et al. 1996) and in the Cascade Mountains of Western Oregon (Lyons & Beschta 1983; Montgomery et al. 2000). Logging activities affected stream temperature, stream hydrology, and trophic processes in Carnation Creek (Hartman et al. 1996). The cumulative effects on the hillslopes and riparian areas after logging to the stream margin, also decreased the quantity and quality of fish spawning and rearing habitat for stream-resident salmonids (Hartman et al. 1996).

In steep watersheds that are managed for timber harvest, accelerated rates of landslides and debris flows are common (Lee et al. 2005). In Carnation Creek, complete loss of forest cover altered hydrology, reduced slope stability and increased the amount of sediment delivered to the river system via mass wasting events (Hartman et al. 1996). The magnitude of sediment delivered from landslides associated with forestry practices in Carnation Creek increased 12-fold. In another study on the Queen Charlotte Islands 17 years post logging, they found a 34-fold increase in the volume of landslide material delivered to the river (Rood 1984). Guthrie (2005) noted that logging on Vancouver Island increased landslide frequency from 0.014 km^2 per year to 0.189 km^2 per year. Excess sediment is associated with road cuts, road surfaces, and ditches

(Hartman et al. 1996). Studies from other areas of coastal B.C. also support that clear-cutting and construction of logging roads increases the frequency of debris flows post-logging (O'Loughlin 1968; Rood 1984).

Hydrological changes as a result of logging practices can also disrupt a watershed (McMurray & Bailey 1998). Logging is known to cause increases in seasonal flows, and stream temperature and affect the timing of peak flows by shifting them earlier and increasing both the magnitude and duration of the peak flow (Hartman et al. 1996; McMurray & Bailey 1998). The magnitude of these changes is dependent on many site-specific factors such as the size of the watershed, the percent of the watershed that has been logged, and the extent of road networks intersecting the slopes (McMurray & Bailey 1998). Clear-cutting in the Pacific Northwest (PNW) temporarily exposes large areas of bare hillslopes to heavy precipitation. Loss of tree cover in these areas causes soils to become saturated and increases surface flow that discharges rapidly to the main stem of the river (McMurray & Bailey 1998).

Disturbances in headwaters and riparian areas of a watershed are inherently linked to downstream reaches. Disturbances in headwaters and riparian areas thus have the potential to seriously impact the morphology of downstream reaches and the estuary (Gomi et al. 2002; Lee et al. 2005). Currently the scientific community does not have a good understanding of the effects of logging on estuary and marine systems in coastal B.C. Although research on estuaries has increased dramatically (Levings 2016), the potential effects of increased sediment supply on estuary ecosystems is not well understood (Thrush et al. 2003). Furthermore, the effects of estuary habitat loss on Pacific salmon (*Oncorhynchus spp.*) in temperate regions is limited (Chalifour et al. 2019).

Aquatic-habitat loss associated with logging is one of the primary factors contributing to the decline of Pacific salmon species on the West coast of Vancouver Island, and more broadly throughout B.C. (Grant et al. 2019). This project examined how salt marsh habitat has changed over time, in a logged and an unlogged estuary, to determine if critical rearing habitat for juvenile Chinook salmon (*Oncorhynchus* *tshawytscha*) had been lost. Under Canada's Conservation of Wild Salmon Policy (WSP), Fisheries and Oceans Canada (DFO) has conducted standardized monitoring to determine the status of wild salmon stocks (DFO 2016). A Conservation Unit (CU) designation is assigned to stocks that hold biologically distinct characteristics that if lost, could not be recovered. Many Chinook salmon populations in PNW are endangered or threatened; 76 populations of Chinook salmon have been designated as Conservation Units in B.C. (Grant et al. 2019). This includes Chinook populations in my two study sites: Tranquil Creek and Moyeha River (Fig.1); Conservation Unit West Vancouver Island South (CK-31) (Government of Canada 2018). By increasing habitat complexity and diversity of habitat necessary to support juvenile Chinook salmon, it should be possible to accelerate the natural recovery process and restore populations that otherwise would take hundreds of years to recover on their own (Palmer et al. 2016). Knowledge on specific stock life histories, detailed habitat inventory data, appreciation for ecosystem complexity, and an understanding of watershed processes is needed to address and restore degraded salmonid habitat (Hartman et al. 1996).

At the heart of the partnership between Simon Fraser University (SFU), British Columbia Institute of Technology (BCIT), and Central Westcoast Forest Society (CWFS), is the desire to address the important issue of declining Chinook salmon populations in Clayoquot Sound through innovative research. The purpose of this research project was to gather baseline information to assess changes in the estuary as it relates to rearing habitat for juvenile Chinook salmon (Simenstad et al. 2002). Here I examine the dynamic interactions between complex watershed processes and the impacts on estuarine juvenile Chinook rearing habitat. This project aims to provide a comprehensive restoration plan to guide restoration of critical estuary rearing habitat for juvenile Chinook salmon in the Tranquil Creek Watershed in Clayoquot Sound.

1.2. Objectives

The goal of this research is to provide CWFS, The Salmon Round Table–Area 24 Clayoquot Sound, The Tla-o-qui-aht First Nation, and The Nuu-chah-nulth Tribal Council with a comprehensive restoration plan to restore critical rearing habitat for juvenile Chinook salmon in the Tranquil Creek estuary. This restoration plan will guide future restoration efforts to increase estuary habitat for juvenile Chinook, in hopes to address the problem of declining populations of Chinook salmon in the PNW.

The objectives of this study are as follows:

1. Determine if salt marsh habitat has changed over time as a result of historical industrial logging practices in the watershed.

2. Determine how the available salt marsh habitat has changed over time i.e. salt marsh area, elevation, and volumetric change.

3. Determine if ecological restoration treatments are feasible and develop a restoration plan to improve or increase the usable saltmarsh habitat for juvenile Chinook salmon.

I hypothesize that sediment input to the stream in the upper watershed, has been deposited in the estuary and has decreased the available salt marsh habitat for juvenile Chinook salmon. I predict that the area of vegetated salt marsh habitat has decreased over time. I also predict that excess sediment deposition in the salt marsh has resulted in an increased salt marsh elevation in Tranquil Creek estuary.

1.3. Estuary Dynamics

Estuaries are dynamic landforms that are continually formed and eroded by river, wind, and tidal processes (Government of BC 2006). To appropriately predict how an estuary will change over time, the suite of physical and environmental factors that influence sediment input and output need to be assessed. This may be limited by data available to produce a comprehensive evaluation of such a complex system (Simenstad et al. 2002). Currently, there are no models in the PNW that can be used to predict estuary habitat trajectories (Simenstad & Cordell 2000). It is difficult to develop reliable predictive models to predict changes, because coastal sediment budgets are not fixed, and are influenced by time-dependent factors such as land use changes (Warrick et al. 2013).

In forested mountain environments, sediment supplied to streams is directly related to the surrounding landscape and materials moved by colluvial and fluvial processes (Church 1992). Sediment yield and sediment load supplied in the river are positively correlated with estuary size (Levings 2016). The input and deposition of sediment in the estuary from hillslope processes, stream bank erosion, and tidal processes is dynamic and is important for habitat formation (CSSP 1995; Levings 2016).

Estuaries in Clayoquot Sound are categorized as coastal plain landforms containing glaciofluvial materials of mostly sand and gravel transported to the valley bottom as fluvial sediments which are then deposited as floodplains and alluvial fans (CSSP 1995). My study sites, Tranquil Creek and Moyeha River watersheds, are dominated by steep, rocky slopes of colluvium and glacial till and are subject to gully erosion, rock falls, debris slides and debris flows (CSTPC 2006). Changes in the frequency and magnitude of sediment input is relevant to restoration and protection of juvenile Chinook salmon rearing habitat.

To evaluate sediment loading to the estuary, one must also have a good understanding of the local climate (annual precipitation) and history of land use in the watershed (Ziemer et al. 1998; Warrick et al. 2013). Average surface run-off and sediment supply to the watershed are both affected by logging (Warrick et al. 2013). In a natural estuary, the river moves horizontally across the plane of the estuary and sediments are deposited in the floodplain where river and wave energy mix (Levings 2016). The extent of estuary habitat thus depends on this interaction. The establishment of salt marsh vegetation and habitat required for juvenile Chinook, is therefore also dependent on sediment input and transport, and wave action at the estuary (Wainger et al. 2017).

1.4. Salt Marsh Habitat Use by Juvenile Chinook

Estuaries are highly productive and ecologically diverse ecosystems that combine freshwater aquatic, terrestrial, and marine environments (Government of BC

2006). Juvenile Chinook salmon rely heavily on estuary habitat for rearing and spend three months to one year in the estuary before migrating to the Pacific Ocean (Levings 2016). Life-history terminology relating to migration timing and estuary residence time varies across the literature (Carl & Healey 1984; Bottom et al. 2005; Moran et al. 2013; Levings 2016). Two main life-history types have been defined for juvenile Chinook based on the amount of time spent in freshwater before migrating to marine environments (Teel et al. 2000). Sub-yearling or ocean-type Chinook migrate to the estuary within the first year in spring or summer as fry, while yearling or river-type, reside in freshwater for at least one year before they migrate into the estuary as smolts (Teel et al. 2000). Brannon et al. (2004) argue that population-specific life-history types vary along a temporal continuum related to incubation and rearing temperatures in addition to genetically derived life history traits. Interestingly, three distinct life history types of juvenile Chinook have been identified in the Nanaimo Estuary, Vancouver Island, B.C.: an estuary-type that migrate to the estuary almost immediately after hatching, a two-month river-type, and a stream-type that remain in the river overwinter before migrating to the estuary in the spring (Carl & Healey 1984; Levings 2016). The Nanaimo estuary provides a good example of the known life-history types found on Vancouver Island B.C. It is believed that only sub-yearling (ocean-type) juvenile Chinook inhabit my study sites, however, data on juvenile life history characteristics is lacking in B.C. (Levings 2016).

When Chinook salmon fry migrate from a freshwater river environment to a saline estuarine environment, they face many physiological and ecological challenges (Levings 2016). As they undergo smoltification, changes to osmoregulatory processes in the gills, liver, kidneys and bladder are necessary to adapt to changing physical environments (Levings 2016). Upon entering the estuary, juveniles also become exposed to new stressors such as prey that is more evasive and new predators (Simenstad et al. 2002). The surrounding landscape and structures within salt marshes directly influence the available refuge and prey abundance for juvenile fishes (Simenstad et al. 2002). Estuary restoration that maximizes the available refuge habitat and supports high prey abundance will improve survival of juvenile Chinook in this developmental stage.

Studies on juvenile salmonid habitat use conducted by Simenstad et al. (2002) captured Chinook salmon in the small first-order channels of the salt marsh and along the edge and above salt marsh vegetation but not within the marsh vegetation. Another study demonstrated with gill net surveys, that Chinook salmon fry demonstrate the highest density deeper in the marsh, likely due to food availability at higher marsh elevations (Levy & Northcote 1981). It was suggested by Simenstad et al. (2002) that most fish emigrate from channels with ebbing tides, however one study using PIT-tag-detection proved that juvenile Chinook salmon movement was volitional and was not passive with tidal flows (Hering et al. 2010). This highlights possible limitations to fyke net and beach seine sampling on falling tides in channels that remain inundated at low tides. Another study found that short-term storm driven increases in freshwater inflow to estuaries has the biggest potential to disrupt fish use of tidal marshes (Wissmar 1998). Hering et al. (2010) used PIT-tags to study juvenile Chinook salmon estuary movements and found that 94% of total detections in salt marsh habitat occurred in >0.4 m of water. This could provide implications for habitat use by Chinook if there have been changes that affect water depth in Tranquil Creek estuary. Juvenile Chinook salmon use of estuaries is greatly understudied in B.C., however, the data presented in these studies provide a baseline for how I can evaluate changes in salt marsh habitat as they relate to available and usable habitat for juvenile Chinook in Tranquil estuary.

Salt marsh zones are described by Levings (2016) in terms of salinity, tidal activity and broad sediment characteristics. These definitions are broadly accepted here since estuaries on the West coast of Vancouver Island are unique in sediment structure and tend to have coarser sediments (Eggers & Ferguson 2018). Salt marshes display a vertical gradient of plant communities that reflect the frequency and duration of tidal flooding and vertical salinity variation in an estuary. A diverse plant community will likely increase habitat complexity and provide beneficial functions for Chinook and invertebrate prey species (Hood et al. 2018). Tidal sedge is the predominant plant species present at both my study sites, based on vegetation surveys, and appears to directly benefit juvenile salmon foraging potential (Hood et al. 2018). Small changes (cm) in salt marsh platform elevation can result in a large percent change in the frequency, duration, and depth of tidal inundation at the marsh surface (Raposa et al.

2016). Seasonal variability in river discharge and the resulting freshwater input to the estuary, can affect growing conditions for salt marsh vegetation (Frenkel et al.1981). There are three zones commonly used in estuary habitat classifications that are stratified by differences along a gradient of elevation and saltwater inundation and as a result differences in plant communities (Levings 2016). Zone definitions defined below were modified from field surveys conducted by CWFS from definitions provided by The National Oceanic and Atmospheric Administration (Robertson 2019).

- Low salt marsh: approximately Mean Low Water Level (MLWL) = -1.3 m CGVD28 to approximately Mean Local Sea Level (MLSL) = -0.3 m CGVD28

- Middle salt marsh: MLSL to approximately Mean High Water Level (MHWL) = 0.8 m CGVD28

- Upper salt marsh: MHWL to Riparian border, which is characterized by trees and riparian shrubs

Other studies evaluating salinity and elevation tolerances of common salt marsh species only refer to two salt marsh zones above (high) and below (low) MHWL. The average tolerance range estimated for all species surveyed at Tranquil was 0.49 m (Table 1).

Table 1. Vegetation Observed at Tranquil Creek Estuary
Vegetation surveys were conducted on 15 August 2019. Elevation data and salinity tolerance (PSU) were sourced from a study conducted in the Nisqually River Delta (Belleveau et al. 2015). Plants lacking elevation ranges were not observed in the Nisqually River Delta study. Elevation Ranges were converted from NADV88 to CGVD28 (2.7 m -0.8 m).

Common Name	Scientific Name	Salinity Tolerance (PSU)	Elevation Range (m, CGVD28)	Marsh Zone	Elevation Difference (m)
Lyngby's sedge	*Carex lyngbyei*	2 - 30	1.30 - 2.23	L/H	0.93
Silverweed	*Potentilla anserina*	10-30.	1.99 - 2.30	H	0.31
Sea-arrow grass	*Triglochin maritima*	10-45	1.39 - 2.25	L/H	0.86
Tuffed hairgrass	*Lysimachia maritima*	10-30.	1.98 - 2.19	H	0.21
Sea milkwort	*Lysimachia maritima*	10-45	1.77 - 2.26	L/H	0.49
Sea plantain	*Plantago maritima*	5-45	1.77 - 2.29	L/H	0.52
Pacific alkali grass	*Puccinellia nutkaensis*	5-33	2.01 - 2.28	H	0.27

Douglas aster	*Symphyotrichum subspicatum*	na	2.08 - 2.30	H	0.22
Meadow barley	*Hordeum brachyantherum*	5-21.	1.67 - 2.27	L/H	0.6
Sea Watch	*Angelica lucida*	-	-	L/H	-
Yarrow	*Achillea millefolium sp.*	-	-	L/H	-
Alaskan Plantain	*Plantago macrocarpa*	-	-	H	-
				Average Range Tolerance (m)	**0.49**

Absolute ranges in elevation tolerances are likely site specific since MHWL calculated for Tranquil Creek and Moyeha River estuaries differed from the study conducted by Belleveau et al. (2015) (0.8 m CGVD28 vs. 2.7 NAVD88 respectively). For my assessment of habitat changes I focused broadly on total vegetated salt marsh.

1.5. Study Sites in Clayoquot Sound, B.C.

The Tranquil Creek Estuary

Tranquil Creek watershed is located approximately 20 km northeast of Tofino, Vancouver Island, B.C. and is located within the traditional territory of the Tla-o-qui-aht First Nation. The drainage area of the watershed is 65 km^2 (Lerner 2011) and based on hydrometric data used to estimate Tranquil Creek, the mean annual discharge is 6.9 m^3/s (Kwasnecha & Hutchinson 2019; Government of Canada 2020a). Tranquil Creek historically supported Chinook (*Oncorhynchus tshawytscha*), Coho (*O. kisutch*), Chum (*O. keta*), Pink (*O. gorbusch*), and Sockeye salmon (*O. nerka*), also coastal cutthroat trout (*O. clarkii clarkia*) and steelhead (*O. mykiss*). However, Chinook salmon, among other species in this watershed have declined since the 1960’s (Smith et al. 2016). It is of note that logging in the watershed began in the 1960’s, however there are many other factors that are likely correlated to the observed decline (Lichatowich et al. 2017; Grant et al. 2019). Some of these factors include climate change and Northeast Pacific Ocean warming trends (Grant et al 2019), overfishing/poor fisheries management, pollution, and the presence of open-net fish farms (Lerner 2011).

DFO records indicate Tranquil Creek Chinook escapement counts were steady at 750 individuals from 1950-1960, after which numbers started to decline (Fig. 2a) (Brown et al. 1979; Government of Canada 2019). Escapement counts at Tranquil Creek were highly variable between 1963 and 1977 but remained relatively low compared to historic numbers. Recent data posted to the New Salmon Escapement DataBase System (NuSEDs) (Government of Canada 2019) indicate that Tranquil Chinook stocks have fluctuated dramatically between 1995-2018 (Fig. 2b) (Government of Canada 2019).

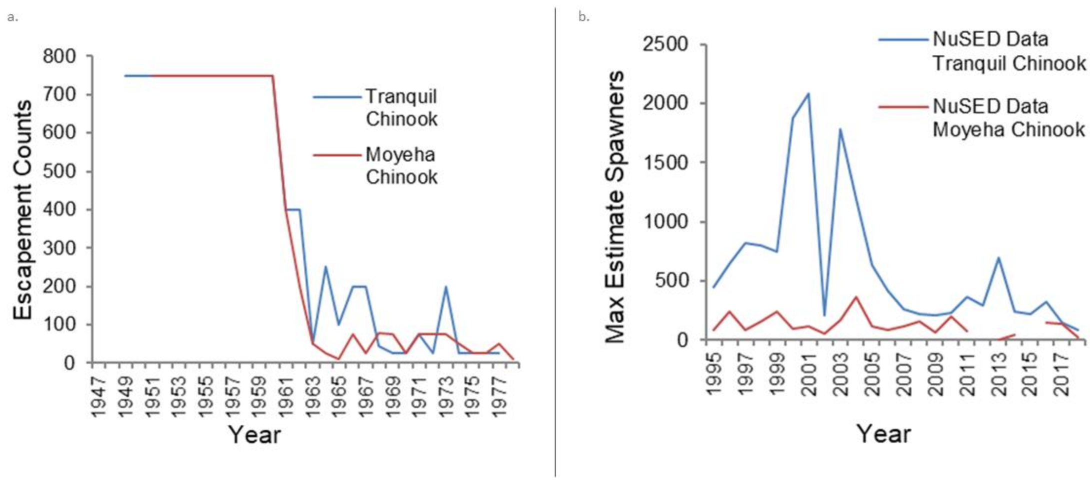

Figure 2. Chinook Escapement Counts and Maximum Spawner Estimates for Tranquil Creek and Moyeha River. Chinook Escapement Counts from 1947-1977 (Brown et al. 1979) (a) and Maximum Spawner Estimates from 1995-2018 (Government of Canada 2019) (b).

Tranquil Creek has experienced considerable resource extraction since the establishment of local fishing and logging towns Tofino and Ucluelet (located 40 km south of Tofino). Fandora Gold mining operations in the upper watershed of Tranquil Creek were completed in 1964 (Gary 2007). Shortly after, in 1965, logging began. During that time, it was standard practice to remove log jams from the river and so by 1968, all log jams had been removed from the stream (Brown et al. 1979). In 1977 bridges and culverts were constructed and by 1978 the second phase of logging operations began, which guided forestry workers to log all the way to the bank of the river (Brown et al. 1979). In 1995, the Forest Practices Code of B.C. and regulations came into effect (Province of B.C. 1997) the new Forestry Practices Code in B.C. was updated to incorporate the retention system, designed to maintain structural heterogeneity and protect ecological integrity (Mitchell & Beese 2002). Currently, the only forest management company operating in Tranquil Creek is Iisaak Forest

Resources Ltd., conducting forest harvesting within the scope of the Clayoquot Sound Land Use Decision under Tree Farm License 57 (Mychajlowycz 2010; Bryant 2016).

A terrain stability assessment (TSA) was conducted by Onsite Engineering Ltd. for Tranquil Creek watershed in 2018. The study was conducted to rate the likelihood of landslide occurrences, locate sources of sediment and to evaluate the risks associated with riparian restoration activities (Eggers & Ferguson 2018). Two study areas were identified in the assessment that are of interest to this project: Map 1 and Map 2 (Extra Fig. 1). These sites were determined to be historical and ongoing sources of landslide debris (coarse sediments and large woody debris (LWD)): Study area 1 (SA1)– east lower reach of Tranquil Creek, and study area 2 (SA2) – west lower reach of Tranquil Creek (Eggers & Ferguson 2018). Numerous large landslides (0.5 – 1 ha to >1 ha) occurred in SA1 in the early 1980s following road construction and harvesting operations. Although none of the landslides directly reached the main stem of Tranquil River, it is speculated that coarse sediments and LWD were transported to the main stem via tributary creeks during seasonal peak flows events (Eggers & Ferguson 2018). Five large landslides occurred in SA2 in the mid to late 1980s based on aerial photography and a Riparian Age Class Dating Map of the watershed (Eggers & Ferguson 2018). Active bank erosion on the east bank (~1 km upstream from tidewater) was also identified as a constant source of coarse and fine fluvial and glaciofluvial sediment (Eggers & Ferguson 2018). Clear-cut logging of old-growth-trees along the stream margin in the 1980s has contributed to bank instability after tree root systems stabilizing the soil were removed (Grant et al. 2019).

The Moyeha River Estuary

For my research project, I compared the Tranquil Creek estuary to an unlogged reference site, the Moyeha River estuary, which is located 40 km north of Tofino, in the traditional territory of the Ahousaht First Nation and within Strathcona Provincial Park. Having an unlogged control watershed as a comparison is rare in B.C. as is long-term environmental baselines for both watersheds that can be used to inform ecological restoration (Levings 2019). The drainage area of the Moyeha River watershed is 181

km^2 (Lerner 2011). To compare changes in hydrology between logged and unlogged watersheds, The CSSP (1995) recommended that a hydrometer be installed in Moyeha River, but no meter was ever installed in the river. Escapement records indicate that the number of returning Chinook spawners in the relatively pristine Moyeha River were relatively steady at 750 individuals from 1947-1960, after which, numbers started to decline and only 10 Chinook spawners were counted in 1978 (Brown et al. 1979; Government of Canada 2019). Furthermore, NuSEDs indicate that Moyeha Chinook spawners have remained low but have remained relatively steady between 1995-2018 (Fig. 2b) (Government of Canada 2019). This provides evidence that logging may not be the causal mechanism contributing to the decline of Chinook salmon populations in these watersheds.

Comparatively, the Moyeha River watershed is larger than the Tranquil Creek watershed with a drainage area of 181 km^2 vs 65 km^2, respectively. The Moyeha River watershed has a larger spring freshet peak due to greater snowpack in the surrounding mountains, while the Tranquil Creek watershed is medium-sized and has moderate peak flows (Lerner 2011). The Risk Assessment prepared for the Clayoquot Salmon Roundtable by Smith et al. (2016) determined that Moyeha river estuary had experienced significant deposition of sediment in the river and estuary between 1994 and 2012. This assessment compared changes in the extent of Chinook habitat based on aerial photographs and field interpretations (Smith 2020, M.C. Wright and Associates Ltd., Nanaimo, BC, personal communication).

2. METHODS

2.1. Aerial Photograph Analysis

Aerial photographs of Tranquil Creek estuary were available for the years 1937, 1956, 1968, 1970, 1981, 1996, 2014 and 2019. This time frame captures aerial images that illustrate industrial logging in 1965, and when logging to the stream margin began in 1977 (Brown et al. 1979). Aerial photographs of Moyeha River estuary were available for the years 1936, 1954, 1978, 1981, 1996, and 2014 (Table 2).

Table 2. Aerial photo years for Tranquil Creek and Moyeha River watersheds.

Tranquil Creek	Moyeha River
	1936
1937	
	1954
1956	
1968	
1970	
	1978
1981	1981
1996	1996
2014	2014
2019	

All publicly available aerial photographs from Provincial and Federal Government sources that reasonably captured both estuaries were ordered for this study (Natural Resources Canada 2019; Province of British Columbia 2019). Digital stereo pairs were ordered from the GeoData B.C. online database, however, not all flight lines and photo rolls were available for preview online. Therefore, an additional search from printed aerial photograph archives at the University of British Columbia Geography Library was conducted and digital copies were subsequently ordered (Table 4).

Once digital photos were acquired, they were uploaded to a file transfer protocol (FTP) site and orthorectified by IGI Consulting (Grady 2020). Orthorectification is required to accurately measure distances and interpret photographic imagery in a Geographic Information System (GIS). This is done by geometrically correcting each aerial photograph to an accurate digital elevation model (DEM), so that all pixels are in a precise horizontal orientation in space (x, y) (Schowengerdt 2006). The DEM used was available from the Province of BC TRIM DEM database for mapsheets 092F022 and 092F041 (Province of BC 2013; Grady 2020). Orthorectification adjusts for topographic relief, lens distortion, and camera tilt (Schowengerdt 2006). This allows for a comparable analysis across all photo years when photos are displayed using the same map projection. A map projection is a geographical transformation of the earth's surface onto a plane to create a map. The Universal Transverse Mercator Projection NAD 1983 UTM Zone 10N is standardly used for environmental mapping in B.C. and was used for this study. The result of this image processing was orthophotos generated for each set of imagery. All orthorectified photographs are listed in Table 5.

Light Detection and Ranging (LiDAR) imagery and aerial photos of Tranquil Creek were taken by Terra Remote Sensing in 2019. LiDAR surveying is a remote sensing method that uses a pulsed laser to measure light reflected off the earth's surface to create a 3-dimensional representation of the surveyed area. This data can be used to create highly accurate and detailed maps. LiDAR and aerial photographs taken by Terra Remote Sensing, were used by IGI consulting to create orthophotos, and were used to map habitat types at Tranquil Creek estuary in 2019.

2.1.1. Mapping Salt Marshes

Orthophotos were uploaded to a geodatabase and projected in ESRI ArcGIS Desktop 10.7 (ArcGIS) in the coordinate system NAD 1983 UTM Zone 10N. Habitat types were characterized as large woody debris (LWD), Islands, vegetated salt marsh and unvegetated areas and were manually digitized on orthophotos. Area of habitat types was calculated from resulting polygons features using ArcGIS calculate geometry function (Hood 2014). Differences in tide height at the time photos were taken, prevented accurate mapping of unvegetated areas in all photos and limited the comparable area across years. This resulted in the comparison of vegetated salt marsh habitat across years as it was the most relevant to salmon habitat (Levings 2016).

Experience delineating salt marsh vegetation on aerial photos and ground truth data from field surveys helped to determine salt marsh boundaries on orthophotos. Reference measurements or ground truth data is almost always required when conducting a spatial analysis. This data can assist with feature classification, can be used as calibration points and can help improve the accuracy of aerial photo analyses (Reif et al. 2012). Some photos differed in interpretability due to differences in resolution, image color balance, shadows from nearby forests, sun angle, vegetation growth, and tidal stage. These limitations are consistent with studies conducted by Hood (2014).

2.1.2. Analyzing Changes in Salt Marsh

Habitat types were color-coded, and maps were exported for all photo years to compare changes in habitat area. The total area of vegetated salt marsh was calculated for all years and the rate of change was calculated between photo years for both estuaries. Hydrometric data available from Tofino Creek Station: 08HB086 was available from 1995 to 2019 (Government of Canada 2020b). Data from this station has also been used by CWFS to estimate flows in Tranquil Creek for an annual rotary screw trap study conducted in partnership with DFO (Kwasnecha & Hutchinson 2019). Daily average river discharge (m^3 s^{-1}) for fall months (Sept-Dec) was calculated and plotted over time. This data was overlapped with the rate of change in salt marsh area (% $year^{-1}$) observed at the Tranquil Creek estuary from 1981 to 2019.

The average total salt marsh area was calculated before and after 1996, which represents pre and post logging activities. This is based on information provided in the TSA conducted for Tranquil in 2018 that stated, logging events in the anadromous reaches of Tranquil Creek ended in the late 1980's (Eggers & Ferguson 2018). Salt marsh mapping results for both estuaries were used to calculate respective average area pre and post logging activities. An analysis of variance (ANOVA) was conducted in R version 3.6.1 to determine whether there was an effect of site or logging on salt marsh area. A regression analysis could not be used to predict changes in salt marsh area due to the complexity involved in estuary dynamics.

2.1.3. Digital Elevation Model Differencing and Volumetric Change

Stereo models were created from stereo pairs by IGI Consulting to later produce DEMs of difference (DoD) in ArcGIS. DoDs can be created by subtracting one elevation model from another and can be used to map erosion, deposition, and volumetric change in a landscape (James et al. 2012). Stereo models were formed for 2014 imagery using referencing meta-data available from GeoBC (Province of BC 2013). Photo identifiable habitat features present in both the 1996 and 2014 imagery were selected, and horizontal and vertical coordinates were recorded into a control file. Using Summit Professional, the 1996 stereo models were formed using these control points to reference the images. The 1981 stereo models were formed using the 2014 control points as well as additional photo identifiable features selected in the 1996 imagery. This method of cascading control from photo identifiable features, continued down to the 1936 stereo models using control from all previous years of imagery: 2014, 1996, 1981, 1978 and 1954 stereo models (Grady 2020).

Digital terrain models (DTMs) were created in Summit Professional and MicroStation using summit models. Polygons from salt marsh mapping in ArcGIS were imported into MicroStation and a 10 m x 10 m grid was generated within the polygon boundary for DTM collection in Summit Professional (E. Saczuk 2020, British Columbia Institute of Technology, Burnaby, BC, personal communication). Elevation data was hand digitized and resultant CAD Point layer files were exported to ArcGIS.

Contour maps for each year were generated by the SFU Geography Department (Song & Ng 2020). DoDs were created to compare the average elevation pre 1996 and post 1996 for both sites. To create contour shape files, CAD files were converted to ESRI Geodatabase files using the ArcGIS tool: CAD to Geodatabase, with spatial reference NAD 1983 UTM Zone 10N. To interpolate discrete point elevation values into a surface raster file, ArcGIS tool: IDW, with cell size 1 m^2, was used to create output inverse distance weighted (IDW) shape files. To clip the raster to the study area the ArcGIS tool: Extract by mask, was used to mask the surface raster file with the boundary of the study area polygon file. To calculate the average elevation for each time period (19xx ~ 19xx and 19xx~ 20xx) ArcGIS tool: Cell statistics, was used. To calculate the elevation difference between the two average values ArcGIS tool: Minus was used. The resultant IDW shape files were exported as maps (Song & Ng 2020).

Volumetric change was calculated pre and post 1996 for both sites using frequency of elevation data extracted from ArcGIS shape files. The elevation difference value (m) was exported into a .csv file (for 1 m^2 grid squares). Volume (m^3) was calculated by adding all the values (Song & Ng 2020).

2.2. Stratigraphic Methods

2.2.1. Sediment Core Sampling

Stratigraphy of sediment cores can be used to better understand the dynamic episodic regimes in coastal estuaries (Watson et al. 2013). To supplement salt marsh digitization, sediment cores were taken at both the Tranquil Creek and Moyeha River estuary to support inferences on hydrogeomorphic mechanisms behind the observed changes and to gain insight on sediment transport and deposition in the estuary system (Poppe et al. 2000). Grain size distribution is a powerful tool used to interpret the geomorphic significance of fluid dynamics in the natural environment and can be used to distinguish local sediment transport mechanisms (Hunt & Jones 2019).

Sediment core locations were determined along exposed vertical channel banks using a stratified random sampling design (Extra Fig. 2). Cores were evenly distributed in each salt marsh zone (lower, middle, and upper), with a total of 6 cores taken at Tranquil Creek estuary and 6 cores taken at Moyeha River estuary (Fig. 3). UTM coordinates were recorded prior to digging at each sediment core location (Table 6).

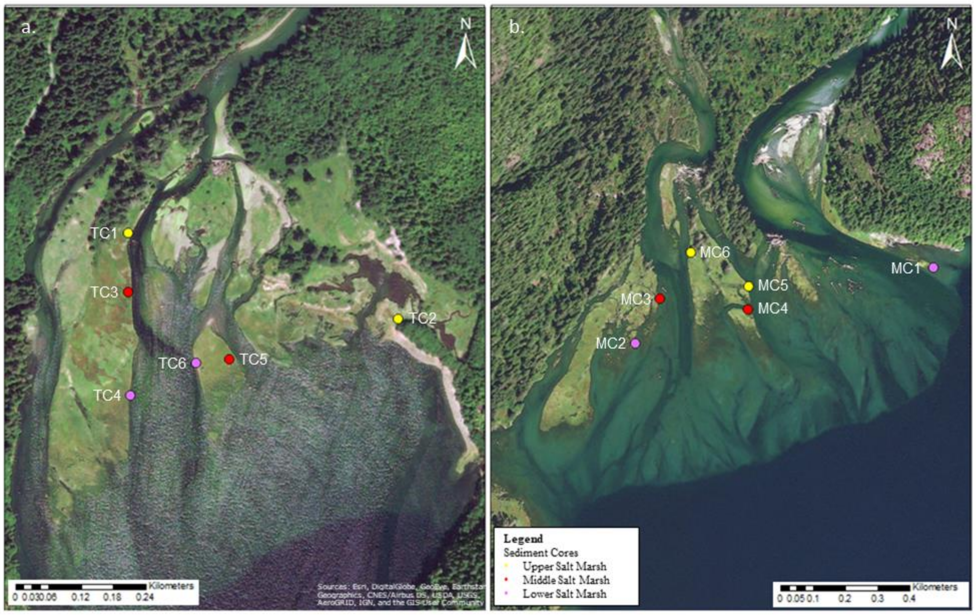

Figure 3. Sediment Core Locations at Tranquil Creek estuary and Moyeha River estuary.
Sediment cores at Tranquil Creek estuary were taken 26-August 2019 and 30-August 2019 (a) and sediment cores at Moyeha River estuary were taken 29-August 2019, 15-Sept. 2019 and 16-Sept. 2019 (b), in Clayoquot Sound, Vancouver Island, BC. Labels correspond to Core ID (ex. Tranquil Core 1: TC1; and Moyeha Core 1: MC1) (Esri 2009)

Estuaries on the West coast of Vancouver Island are unique to most estuaries studied in B.C. such as the Fraser River Estuary and the Nanaimo, Cowichan, Chemainus, and Comox estuaries on the East coast of Vancouver Island (Flynn et al. 2006). In the steep headwaters of coastal tributaries, fluvial sediment load transported can be significantly coarser than in larger rivers (Lee et al. 2005). Due to the coarse grain size characteristic of my study locations, I designed a simple sampling method described below, that allowed me to sample deeper into the coarse estuary sediments typically found at my study sites.

I considered ^{14}C and ^{210}Pb isotope dating in this study because it had been used in other studies to date sediment deposition in estuaries (Hales 2000; Watson et al. 2013).This would have been an informative way to correlate major logging events to sediment loading in the estuary however funding was not available for these analyses.

Sampling at Tranquil Creek estuary occurred 26-August 2019 and 30-August 2019. Sampling at Moyeha River estuary occurred: 29-August 2019, 15-Sept. 2019 and 16-Sept. 2019. The boundary of each salt marsh zone in the Tranquil Creek estuary was established using a map of the estuary with DTM polylines (Robertson 2019). The boundary of each salt marsh zone in the Moyeha River estuary was demarcated on a 2019 drone photo. Zone boundaries were verified in the field based on predominant vegetation types known to exist in each zone (Pojar 1994; Belleveau et al. 2015). Potential sampling points were evenly marked along the perimeter of each zone and numbered 1 through k (k=total number of possible sampling locations). Two random points were selected in each salt marsh zone (upper, middle and lower) using a random number generator. Selected sampling points were estimated in the field using a Garmin Global Positioning System (GPS)–this likely contributed to some bias. A crude core sampling method was used to extract core samples that would later be used to determine grain size distribution in the lab.

Cutback banks were most easily sampled at low tide. Sampling windows were roughly 4-6 hours, from low tide to high tide. Tides and physical effort greatly restricted this method and is one reason for a small sample size used in this study. Cores were dug by hand at each randomly selected sampling location using a pickaxe and shovel. Cores were roughly 0.1 m in from the bank edge and 1.0 m deep. A transect was set from the top edge of the bank to the base of the core. Samples were taken at 10-cm intervals along the transect using a 555 ml sampling device designed by Douglas Crocker (Extra Fig. 3). Photos were taken to help with qualitative analysis (Fig. A2. a-e).

I attempted to extract 10 samples for each 1.0 m core (roughly 1 sample every 0.1 m). I was only able to extract 7 samples from TC5 at Tranquil 30-Aug 2019 because of rising tides. Samples were used if they filled >50% of the can. If samples were <50% a second sample was taken at the same depth. Rocks that prevented successful extraction were taken out and included in the sample, followed by a second attempt to extract sediment from that depth to fill the sampling device. Each sample was placed in a labeled plastic Ziplock bag. Date, estuary name, UTM, core number, sample number, and sample depth were included on every sample. Samples from each core were stored in separate large plastic Ziplock bags and all samples collected in a day were stored in a 14 L plastic tote until they could be processed in Dr. Jeremy Venditti’s sediment lab at

SFU. A total of 12 cores were taken and a total of 117 samples were collected and processed (6 cores x10 TRANQ – 3 (TC5 had 7 samples) + 6 core x10 MOYEHA).

2.2.2. Sediment Processing

Sediment processing occurred in two parts: drying and sorting by grain size. First samples were emptied from Ziplock bags into aluminum drying trays labeled with core number, sample number and depth of sample. Samples were then placed in a drying oven set at 150°C until dry. Drying time was ~8 hours per core but varied depending on the moisture content in the samples and number of samples in the oven. Samples were deemed dry by touch. After drying, dry weight of each sample was noted on a spreadsheet. Next, sieve trays were assembled from smallest to largest screen size. Sieve tray sizes were <0.063 mm, 0.063-0.2 mm, 0.2-4.0 mm, 4-8 mm, 8-16 mm, 16-31.5 mm and >31.5 mm respectively. Sieve sizes chosen were based on grain sizes outlined by Shreve & Downs (2005) and what was available in the sediment lab.

Once sieves were assembled, the deepest sample taken from Tranquil Creek core 1 was poured into the stack of sieves and run in a mechanical sieve machine for 5 minutes or until all sediment was sufficiently sorted. Sieve trays were separated carefully and the weight of sediment in each sieve was recorded on a spreadsheet. Relative weight fractions of: clays and silt (<63 µm), sand (0.063-0.2 mm), very fine gravel (0.2-4.0 mm), fine gravel (4-8 mm), medium gravel (8-16 mm), coarse gravel (16-31.5 mm), and very coarse gravel (>31.5 mm), were recorded and compared to the total dry weight of that sample to calculate percent grain size distribution (Shreve & Downs 2005). Sorting was repeated for each sample in ascending order until all samples were sorted from the respective core. Sorting procedures were repeated for all 12 cores.

2.2.3. Analysis of Sediment Core Data

Depth of core samples were standardized for the analysis, and percent grain size distribution was determined for each core (Hales 2000). The median proportion for each grain size category was determined at each depth for the six cores from each site (n=6).

Grain size distribution was presented in a stacked bar graph to compare the relative proportional differences in grain size categories between the Tranquil Creek and Moyeha River estuaries.

A community analysis of sediment core results was conducted by Joy (2020). The dimension reduction took the proportion data of all the different substrate types and tried to explain the variability in just two axes (NMDS1 and NMDS2). This analysis was used to evaluate community level differences between study sites.

3. RESULTS

3.1. Changes in Salt Marsh Habitat Area

Salt marsh habitat in the Tranquil Creek estuary and Moyeha River estuary varied over the observational period. Salt marsh habitat surface area in Tranquil estuary (Fig. 4) and Moyeha estuary (Fig. 5) were mapped on orthophotos in ArcGIS 10.7. Based on mapping results, the total salt marsh area in 1937 in the Tranquil Creek estuary was 88,614 m^2, reached a maximum area of 122,415 m^2 in 1956 and a minimum area of 58,600 m^2 in 1970. The total salt marsh area in 2019 was calculated to be 112,741 m^2. Indicating a net gain of 24,127 m^2 (27%), between 1937 and 2019. The most notable changes seem to have occurred on the east side of the estuary. A logging camp, log storage sites and roads are visible for the first time in the 1968 aerial photograph. It is likely that the east bank of the salt marsh was dredged to facilitate ocean-transportation of logs. This area of the salt marsh experienced a loss of 34,342,17 m^2, between 1956 and 1968. The unvegetated area in the estuary also appears to have increased over the observational period and is largest in 2019.

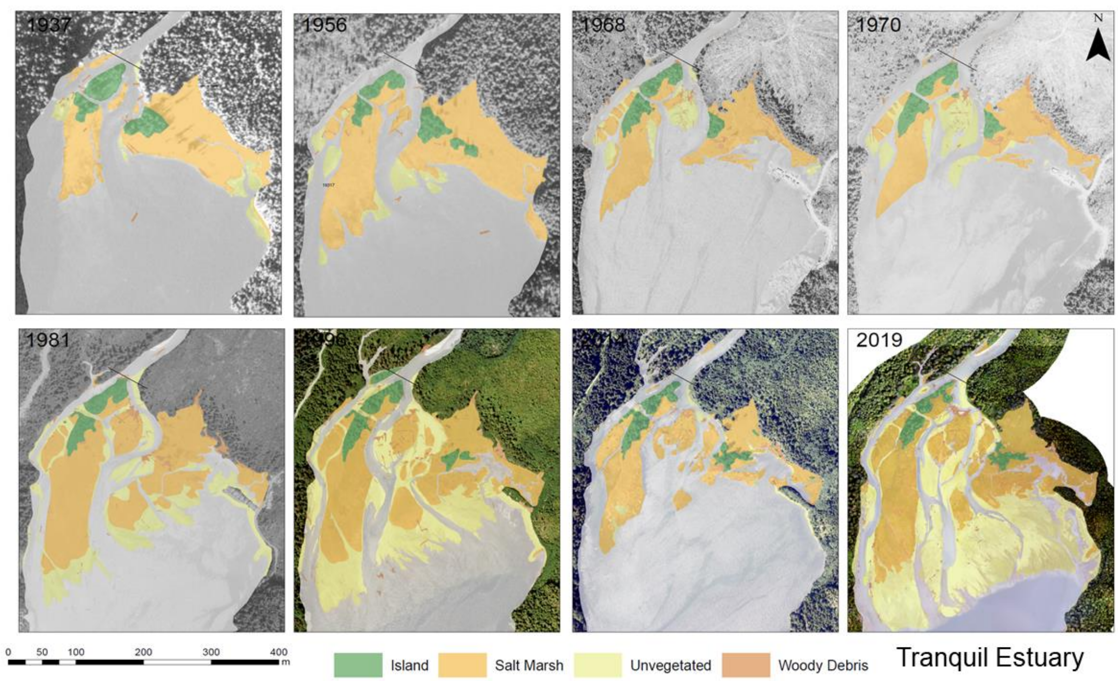

Figure 4. Estuary Habitat Mapping Tranquil Creek
Mapping for these photos was conducted by Tuzlak (2019).

The total salt marsh area in 1936 at the Moyeha River estuary, the reference estuary, was 124,983 m^2. The minimum area mapped was 98,490 m^2 in 1954 and the total salt marsh area in 2014 was 158,487 m^2 which was also the maximum area mapped over the observational period indicating a net gain of 33,503 m^2 (27%), between 1936 and 2014.

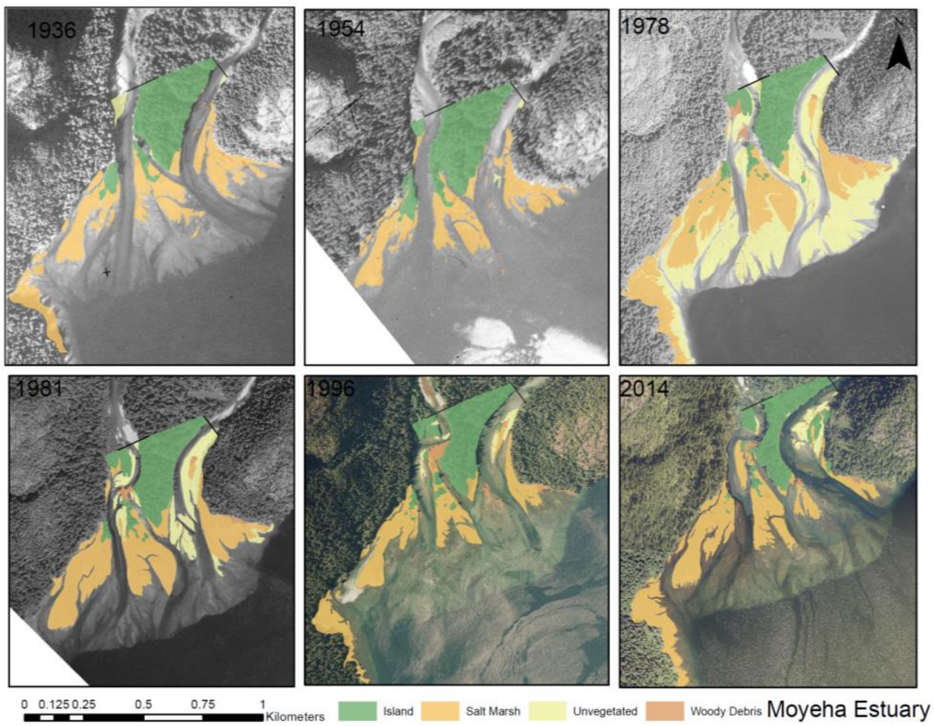

Figure 5. Estuary Habitat Mapping Moyeha River.

The average total salt marsh area across the observation period in the Tranquil Creek estuary was 93,210.52 m^2 with a standard deviation of 24,724.21 m^2 and the average total salt marsh area across the observation period in the Moyeha River estuary was 122,441.7 m^2 with a standard deviation of 22,970.49 m^2. The total surface area (m^2) mapped on orthophotos across the observational period is depicted in Figure 6. The beginning of logging in Tranquil Creek (1965) and the start of logging to the stream margin in Tranquil Creek (1977) is also depicted in Figure 6. No significant trend in total salt marsh area was evident in the data for Tranquil Creek or Moyeha River over the observational period, however both estuaries experienced a positive net gain in surface area.

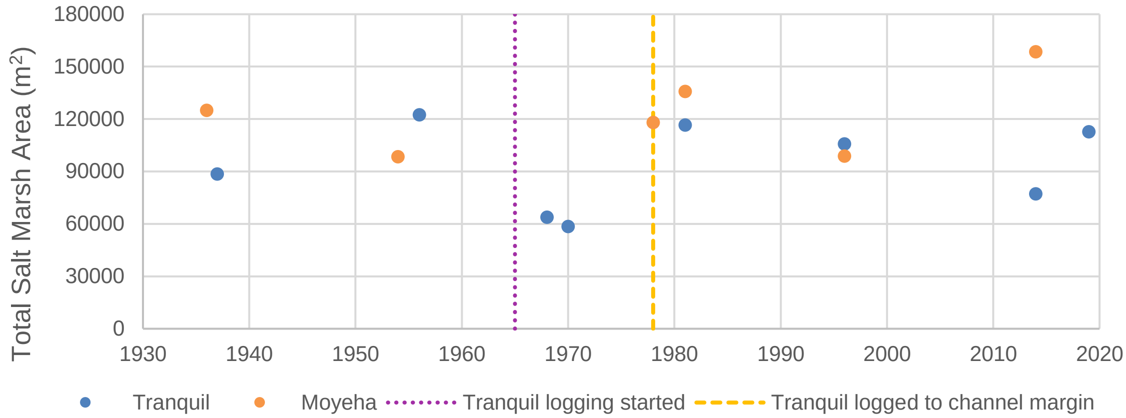

Figure 6. Total Salt Marsh Area.
Total salt marsh area (m^2) in Tranquil Creek estuary calculated from polygons mapped on orthorectified aerial photographs from 1937, 1956, 1968, 1970,1981, 1996, 2014 and 2019, and in the Moyeha River estuary from 1936, 1954, 1978 1982, 1996 and 2014. Logging began in Tranquil watershed in 1965 and was logged to the stream margin in 1977.

The average salt marsh area in the Tranquil Creek estuary prior to the end of logging activities in the lower watershed (pre 1996) was 89,994.75 km^2 and after (post 1996) was 98,570.14 km^2 (Fig. 7). The average salt marsh area in the Moyeha River estuary pre and post 1996 was 119,337.10 km^2 and 128, 651.0 km^2 respectively (Fig. 7). No significant difference was detected between the average salt marsh area in the Tranquil Creek estuary and the Moyeha River estuary pre 1996 and post 1996. Thus I was unable to detect an effect of site or logging on salt marsh area (ANOVA, $F(3)= 1.5923$, $P=0.2523$).

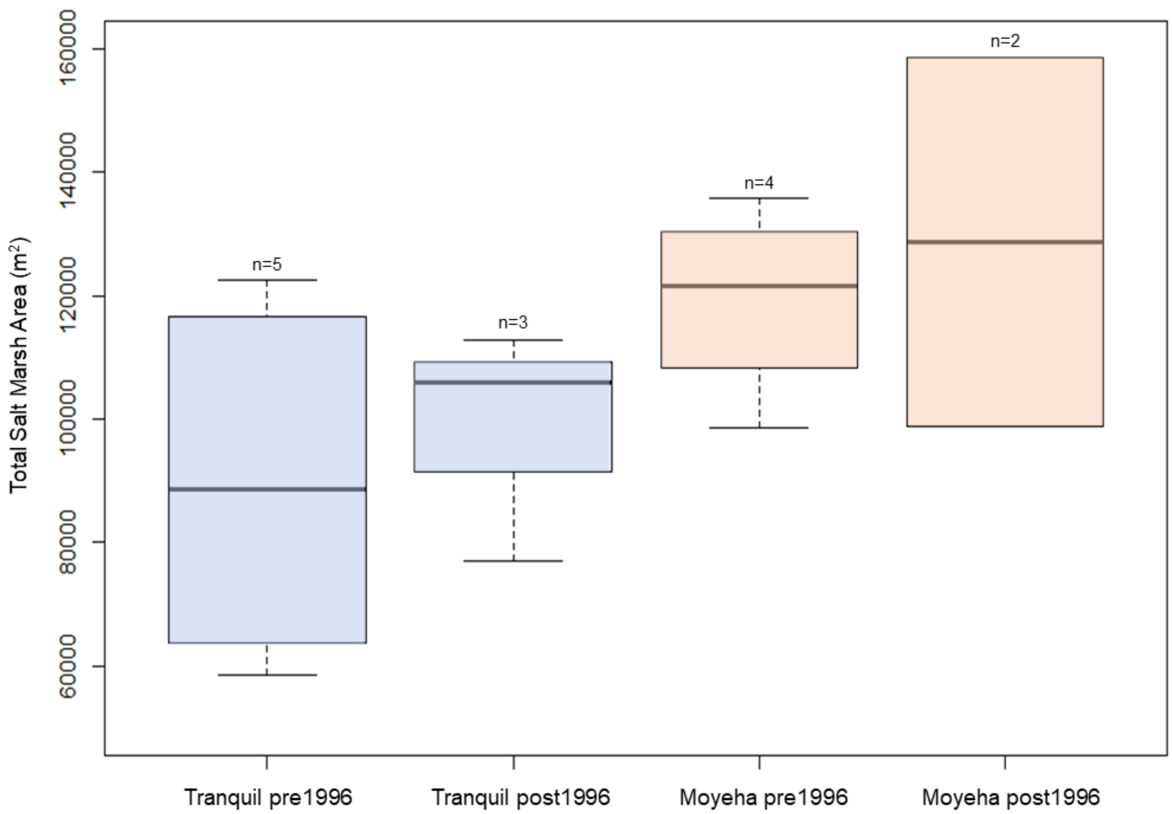

Figure 7. Average Salt Marsh Area pre 1996 and post 1996.
Boxplot depicting average total salt marsh area at Tranquil pre 1996 and post 1996 (blue boxes n=5 and n=3, respectively) and at Moyeha pre 1996 and post 1996 (orange boxes n=4 and n=2, respectively).

To visualize times within the observational period when the greatest changes occurred, I calculated rate of change (Table 2) and plotted the rate of change in total salt marsh area for both watersheds (Fig. 8). It is important to note that the aerial photos available for both watersheds were not consistent across years. Aerial photos were not available for Moyeha River between 1955-1977 and two photos were available for Tranquil Creek in that time. Temporal changes that may have occurred in the Moyeha River estuary during that time will not be considered in this analysis. The most notable changes in the Tranquil Creek salt marsh area occurred between 1956 and 1968, with a loss of 7.65 % in total salt marsh area per year, and between 2014 and 2019 with an increase of 6.32 % in total salt marsh area per year (Fig. 8). The greatest surface area changes in the Moyeha River salt marsh occurred between 1978 and 1981 with an increase of 4.39 % in total salt marsh area per year. Tranquil Creek salt marsh experienced a similar increase of 4.52 % in total salt marsh area from 1970 to 1981 (Fig. 8).

Table 2. Rate of Change for Tranquil and Moyeha Salt Marsh Area

Study site	Study period (time 1–2)	Year	Habitat type	Area at time 1 (m2)	Area at time 2 (m2)	Change (%)	Rate of change (% yr−1)
Tranquil	1937	1937	Salt Marsh	-	88614.92	0	0
Tranquil	1937-1956	1956	Salt Marsh	88614.92	122415.93	27.61	1.45
Tranquil	1956-1968	1968	Salt Marsh	122415.93	63816.12	-91.83	-7.65
Tranquil	1968-1970	1970	Salt Marsh	63816.12	58600.56	-8.90	-4.45
Tranquil	1970-1981	1981	Salt Marsh	58600.56	116526.20	49.71	4.52
Tranquil	1981-1996	1996	Salt Marsh	116526.20	105836.97	-10.10	-0.67
Tranquil	1996-2014	2014	Salt Marsh	105836.97	77131.63	-37.22	-2.07
Tranquil	2014-2019	2019	Salt Marsh	77131.63	112741.81	31.59	6.32
Moyeha	1936	1936	Salt Marsh	-	124983.41	0	0
Moyeha	1936-1954	1954	Salt Marsh	124983.412	98490.79535	-26.90	-1.49
Moyeha	1954-1978	1978	Salt Marsh	98490.79535	117987.7226	16.52	0.69
Moyeha	1978-1981	1981	Salt Marsh	117987.7226	135886.286	13.17	4.39
Moyeha	1981-1996	1996	Salt Marsh	135886.286	98814.685	-37.52	-2.50
Moyeha	1996-2014	2014	Salt Marsh	98814.685	158487.2292	37.65	2.09

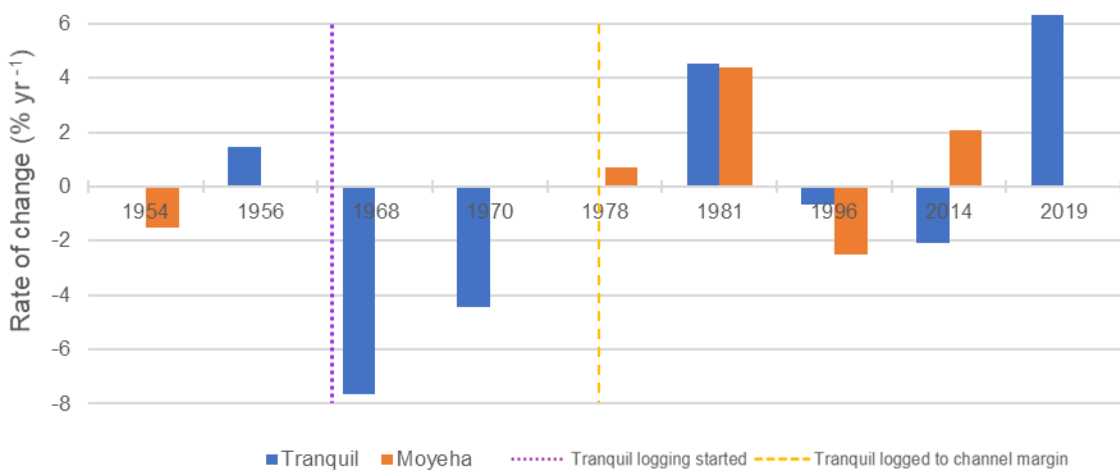

Figure 8. Rate of Change for Salt Marsh Area
Rate of change, percent area (m^2) per year, of salt marsh in Tranquil Creek estuary and Moyeha River estuary. Aerial photos used to calculate area were from 1936-2014 for Moyeha River and 1937-2019 for Tranquil Creek.

Daily average river discharge for fall months (Sept-Dec) in Tranquil Creek were estimated using hydrometric data from Tofino Creek Hydrometric Station 08HB086 (Government of Canada 2020b) and overlaid with rate of change for Tranquil Creek salt marsh area from 1995- 2019 (Fig. 9). High peaks in river discharge coincided with negative rates of change observed from 1981 to 1996 and from 1996 to 2014 and lower peaks in river discharge coincided with a positive rate of change observed between 2014 and 2019.

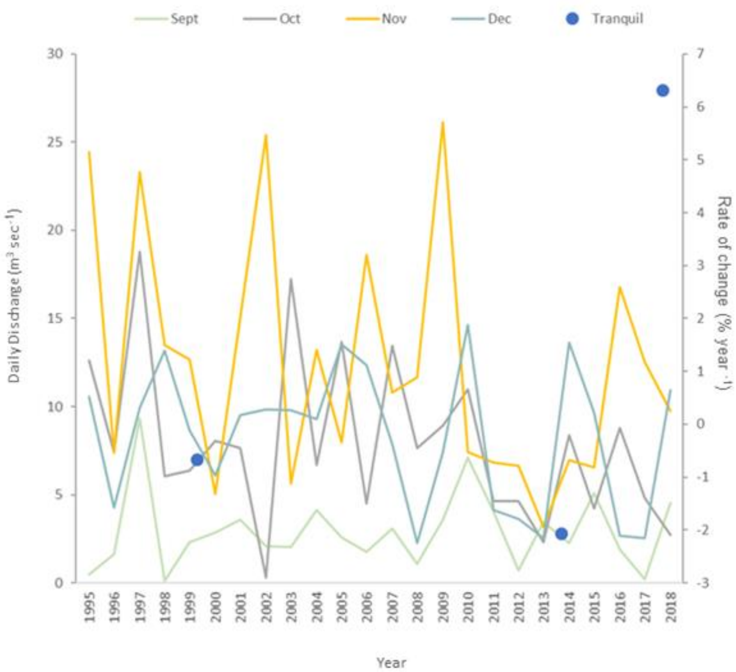

Figure 9. Hydrometric Data and Rate of Change in Salt Marsh Area
Daily discharge from hydrometric Station:08HB086 was averaged for September, October, November and December and was used to estimate fall peak flows in Tranquil Creek.

3.2. Changes in Salt Marsh Elevation and Volume

Elevation was compared spatially across years (Extra Fig. 4 and 5), and mean elevation was determined before and after 1996 for both watersheds to compare how elevation has changed before and after logging activities at Tranquil Creek (Fig. 10). The study area commonly mapped across all photo years in the Moyeha River estuary was 39,107.00 m^2 (31% of the average total area), which when superimposed presented an average elevation difference of -0.190 m (SE +/- 0.031 m) pre and post 1996. The total volume change calculated from the DEM-differencing of mean elevation pre and post 1996 in the Moyeha estuary was -5,816.04 m^3. The study area commonly mapped across all photo years at Tranquil Creek estuary was 21,810.00 m^2 (23% of the average total area), which when superimposed presented an average elevation difference of -0.88 m (SE +/- 0.049 m) pre and post 1996. The volume change calculated from the DEM-differencing of mean elevation pre and post 1996 in the Tranquil Creek estuary was -11,799.874 m^3.

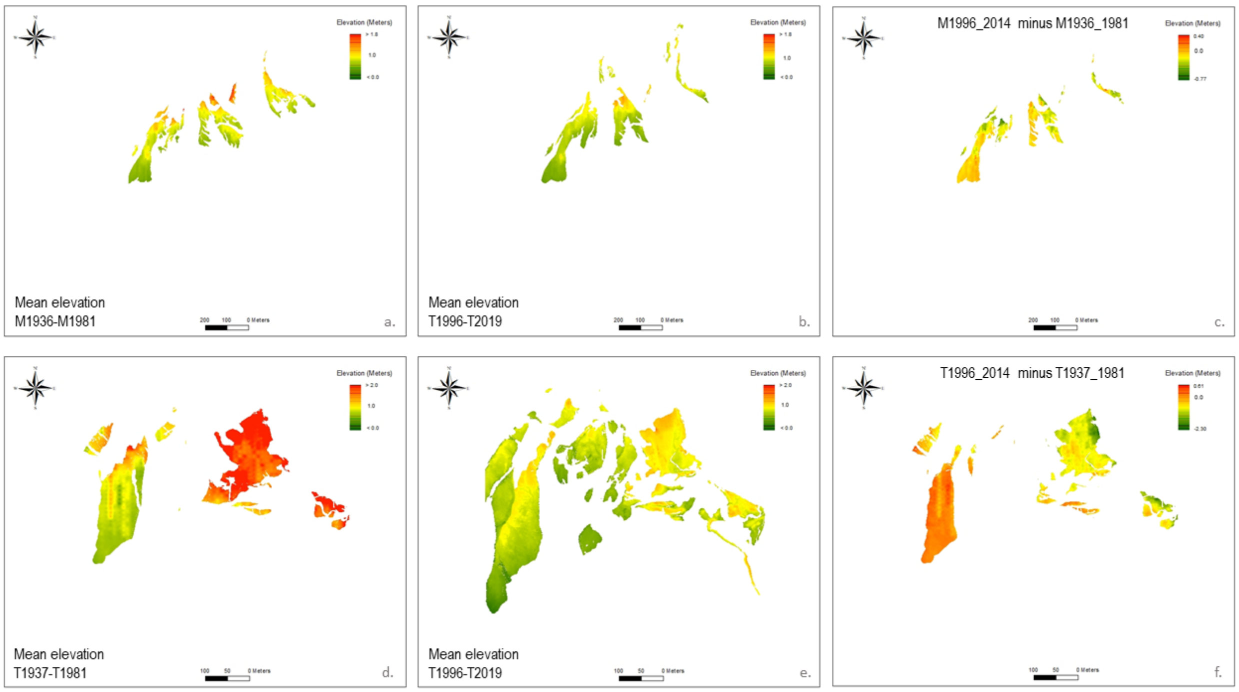

Figure 10. Digital Elevation Models and DEM of Difference.
Elevation at Moyeha averaged pre 1996 from 1936-1981, n=4 (a) and post 1996 from 1996-2019 n=2 (b) Elevation at Tranquil averaged pre 1996 from 1937-1981, n=5 (d) and post 1996 from 1996-2019, n=3 (e) Changes in elevation at Moyeha with a range of >1.8m and <0.0m (c) and at Tranquil with a range of >2.0m and <0.0m (f) (Song & Ng 2020).

3.3. Grain Size Distribution and Stratigraphy

Grain size proportions were calculated for all cores (Tranquil n=6; Moyeha n=6) at both sites to determine differences in relative band width of different grain size proportions at different depths (Fig. 11). Sand (0.063-0.2 mm) was the predominant type of sediment found across both sites; 42.8% of all sediment in the sediment cores at the Tranquil Creek estuary and 50.8% in the sediment cores at the Moyeha River estuary. The proportion of gravels at different depths varied between both sites. There was a noticeable difference in the combined proportion of gravels, in the first 5 depth categories between both sites. Depths 1, 2, 3, 4, and 5 at Tranquil Creek estuary show relatively smaller band widths (proportions) of silt and clays compared to Moyeha River estuary, with 2.6% and 12.4% fines respectively. Tranquil Creek estuary cores also displayed greater combined bandwidth of the three coarsest grain size categories compared to Moyeha River estuary cores, with 28.9% and 6.3% respectively.

Sediment core stratigraphy at each core location differed (Fig. 12, Fig. A2. a-e). Moyeha Core 1 – the closest sample to the river mouth (Fig. A2.d), and Tranquil Core 3 – adjacent to a historic logging camp (Fig. A2.c), displayed the least distinct stratification. Vegetation density also appeared to be greater in the lower salt marsh in the Moyeha River estuary compared to the Tranquil Creek estuary (Fig. 12).

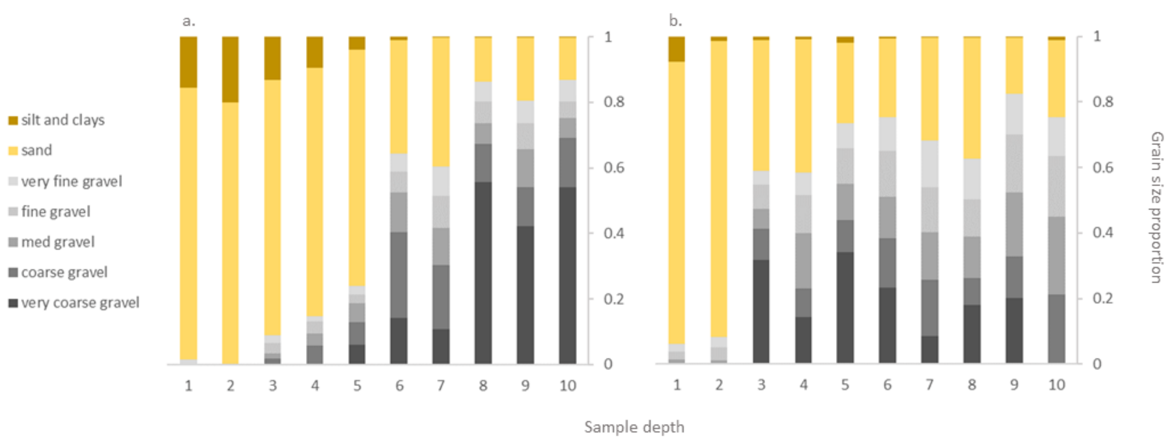

Figure 11. Grain Size Distribution of Sediment Cores
Median Grain size proportion of seven grain size categories: silt and clays (<63 μm), sand (0.063-0.2 mm), very fine gravel (0.2-4.0 mm), fine gravel (4-8 mm), medium gravel (8-16 mm), coarse gravel (16-31.5 mm), and very coarse gravel (>31.5 mm) at standardized 10 cm intervals to a depth of 1 m, for 6 cores (n=6) (a) taken in Moyeha estuary taken on 29 August, 15 and 16 September 2019 and (b) taken in Tranquil Creek estuary taken on 26 and 30 August 2019.

Figure 12. Photos of Sediment Cores in the Lower Salt Marsh
Sediment cores were taken in the lower salt marsh at (a) Tranquil (TC4) and (b) Moyeha (MC4) taken on 30 August 2019 and 15 September 2019 respectively.

Lastly, the multivariate analysis displayed a significant overlap in the sediment core data for both sites (Joy 2020) (Fig. 13).

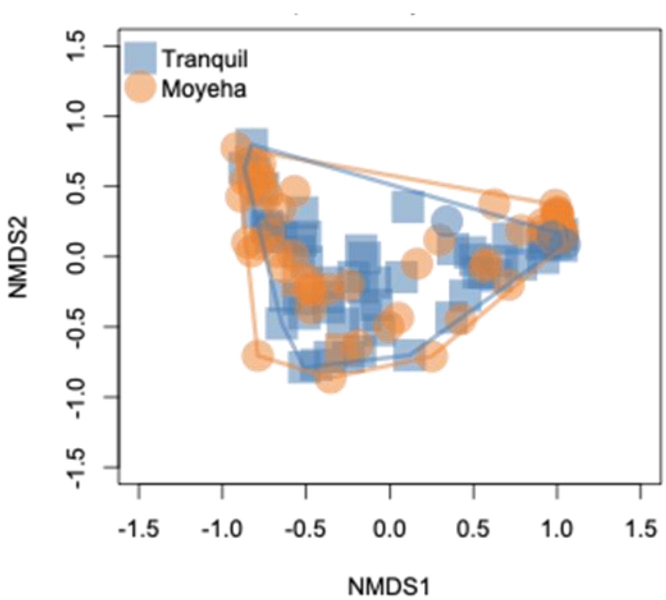

Figure 13. Multivariate Analysis of Sediment Composition in Tranquil and Moyeha estuaries.
Variability in the proportion data was reduced into two axes (NMDS1 and NMDS2). Figure was created by (Joy 2020).

4. Discussion and Project Limitations

Change in Salt Marsh Area

The extent of submerged aquatic vegetation, in this case represented as the surface area of salt marsh visible in aerial photos, has been described as a measure of resilience by Wainger et al. (2017). Salt marsh vegetation is a good indicator of estuary ecosystem resilience due to this ability to stabilize sediments and various habitat types that support numerous aquatic and marine species (Wainger et al. 2017).

Changes in the extent of vegetated salt marsh area reflects changes in salt marsh platform elevation or changes in inundation and thus growing conditions for salt marsh vegetation (Gurbisz et al. 2016). The difference between the average salt marsh area in the Tranquil Creek estuary and the Moyeha River estuary measured over the observation period, mirrors the notable difference in watershed drainage area, 65 km^2 and 181 km^2, respectively. The size of a watershed influences sediment loading and thus influences estuary size (Levings 2016). Regrettably, there were no photos available for Moyeha River that captured the estuary from 1968 or 1970 in the Provincial Government aerial photograph database (Province of British Columbia 2019); thus, preventing an equal comparison between Tranquil Creek and Moyeha River. The ANOVA comparing Salt marsh area in the Tranquil Creek estuary and the Moyeha River estuary pre and post 1996, reflects a classic statistical type II error (R. Joy 2020, Simon Fraser University, Burnaby, BC, personal communication). The small sample size resulted in insufficient power to detect a significant difference between sites. Collecting additional photos from other sources to fill data gaps across the observational period could result in a larger sample size for a more robust analysis.

Contrary to my predictions, my results showed a net increase in total salt marsh area at both watersheds. These results suggest that sediment from hillslopes has not been deposited in the salt marsh, burying vegetation, as I expected. I have come up with two very different hypotheses, involving competing processes to explain the results presented here. It is not clear which process is dominating; however, I suggest further studies that may be useful to gain a deeper understanding of how estuary habitat has changed in both watersheds. One hypothesis for these results is that excess sediments have filled estuary channels. This might explain the net increase in salt marsh area observed over the observational period. If low order channels deep in the estuary have infilled, vegetation may have opportunistically expanded towards the front of the estuary (Levings 2016). There could also be infilling of tidal channels in the Moyeha River estuary because of climate change factors such as increases in annual precipitation, as well as an increase in the frequency and magnitude of precipitation events (Bush & Flato 2018). This hypothesis supports comments about channel infilling, made in the Clayoquot Sound Watershed Risk Assessment conducted by Smith et al. (2016). However, it could be due to natural variation in estuary geomorphology (Levings 2016). The relative difference in the changes observed between the two estuaries may suggest a cumulative effect of logging and climate change on salt marsh habitat in Tranquil Creek estuary. Although salt marsh area is increasing, the importance of maintaining deep tidal channels for Chinook is key (Hering et al. 2010). Infilling of channel habitat could result in added stress to juvenile salmon populations as this decreases access deep into the salt marsh, decreasing foraging potential (Simenstad et al. 2002). Quantification of available channel habitat for juvenile Chinook salmon has been identified as a data gap that warrants future study.

A second hypothesis that could be considered, is the possibility that excess sediment is being transported directly to the subtidal area, and so is not directly affecting the estuary. Relative to large estuaries like the Fraser River, these steep tributaries end at short floodplains extending into deep inlets (Flynn et al. 2006; Levings 2016). Given the influence of high flows on sediment bedload movement and high flows attributed to high rain events in this region, it is possible that excess sediment has been pushed into the subtidal zone, beyond the boundary of my analyses (Guthrie 2005). Historical subtidal areas may now exhibit favorable growing conditions for salt marsh vegetation as a result of sediment deposition and increased elevation. This may have resulted in the succession of salt marsh vegetation in those areas and may explain the increase in area measured at both sites.

Surprisingly, both sites displayed an increase in mean total salt marsh area post logging activities. The net increase over the observational period in the Tranquil estuary could indicate that vegetation is re-establishing in areas that were degraded during logging activities in the estuary. This supports my second alternative hypothesis and could mean that the estuary is on a trajectory towards recovery since 1970 when the salt marsh reached its smallest. These results warrant further monitoring to determine trends and to determine if restoration is feasible.

Another important factor to consider is the stability of excess gravel in the river (Warrick et al. 2013). Sediment aggradation in the channel has occurred since logging ended along the channel margin in Tranquil Creek watershed. Large gravel bars were visible in the 1981, 1996 and 2014 orthophotos, and were mapped in ArcGIS by Tuzlak (2019) (Extra Fig. 6). It is likely that the sediment in these gravel bars will remain in place until high flow events mobilize sediment downstream into the estuary (Gurbisz et al. 2016). It is unknown exactly how long the effects of logging on landslide frequency in a watershed will last, and may reach upwards of 50 years until larger trees become established to secure sediments (Hartman et al. 1996). Sediment from hillslopes may not have entered the estuary yet, highlighting the importance of monitoring the success of upriver restoration and sediment stabilization prior to estuary restoration.

Interestingly, river discharge may help explain the rate of change in salt marsh habitat at Tranquil Creek estuary between 1995 and 2019. Warrick et al. (2013) compiled hydrologic and suspended sediment data from six watersheds in Northern California from 1955-2010. Sediment concentrations in a regularly logged watershed were roughly 10-times greater compared to an unlogged watershed surveyed in the study (Warrick et al. 2013). A positive relationship between suspended sediment concentration and river discharge was observed for all watersheds and an increase in suspended sediment was observed after a major precipitation event in 1964 (Warrick et al. 2013). In a study conducted by Gurbisz et al. (2016), higher flows led to greater overall plant loss in salt marsh habitat. This could indicate that high flows before 1996 contributed to the negative rate of change between 1978 and 1996 and between 1996 and 2014 at Tranquil Creek estuary. Furthermore, lower flows between 2014 and 2018 may have allowed vegetation to recover and contributed to the positive rate of change observed between 2014 and 2019. River discharge can also affect water levels in the estuary and can increase depth during max discharge or freshets (Levings 2016). This could have made conditions unfavorable for salt marsh vegetation that prefer a higher salinity (Table 1). Estuarine plants are known to occupy different zones depending on their ability to tolerate salinity, wave action, river flow, tidal changes, and sedimentation levels (Government of BC 2006). A Study conducted by Janousek & Folger (2014) found that salinity and tidal elevation explained most of the variation observed in common salt marsh vegetation species while hydrology, soil nitrogen and soil clay content were secondary influences. As limited data was available to estimate river discharge in Tranquil Creek, this relationship requires further research to confirm a causal effect of river discharge on sediment transport, deposition and erosion in the estuary.

Inherent limitations are involved when working with historical photographs (Hales 2000; Hood 2014). Resolution limited accurate delineation of habitat types in the estuary, particularly in photos from 1930s and 1950s. Furthermore, the Ocean tidal height was not identical across photo years. Tide estimates were back calculated to confirm this. The tide height at the time photos were taken was calculated by the Canadian Hydrographic Service (Ballantyne 2020) (Table 5). Unequal tide levels prevented an accurate assessment of total intertidal unvegetated areas which are likely an important area of gravel deposition in the estuary. Light reflection off the water at the time photos were taken also prevented delineation of the outer subtidal estuary sediment boundary in some of the photos (Tranquil 1968, 1970 and 1981). For these reasons, the area presented here focused only on salt marsh habitat so a reasonable comparison could be made. Lastly, inevitable bias exists in delineating polygon boundaries in ArcGIS. This bias was consistent across all photo years and thus allows for a reasonable comparison for the purpose of this study but should be considered if mapping is used in the future to delineate habitat in these watersheds.

Changes in Elevation and Volume

The Geological Survey of Canada has been measuring land uplift and sea level rise across the country to predict future levels based on Representative Concentration Pathway (RCP) scenarios (James et al 2014). The elevation change recorded for Ucluelet British Columbia is +1.46 mm +/- 0.55 mm per year, therefore, over the observational period the land has risen ~119.72 mm (James et al. 2014). This broad trend for the Clayoquot Sound area is not consistent with local changes calculated in this study for Tranquil Creek or Moyeha River estuary. Sea level rise projections for Ucluelet, based on the highest RCP8.5 scenario, predict an increase of 29.7 cm by 2100 compared to the global mean sea-level projection of 74 cm by 2100 (James et al. 2014). Large-scale land uplift in this region can counteract global sea-level rise, leading to reduced rise or even fall of relative sea level (James et al. 2014). On the other hand, loss in elevation adds to absolute sea-level rise and would theoretically increase the effects of relative sea-level rise on salt marsh habitat. Furthermore, high river discharge can add to the effects of sea level rise on salt marsh vegetation (Levings 2016). Higher tides combined with high river discharge, increases water levels in the estuary and may result in shifts of plant communities to those that can tolerate prolonged inundation, or may result in the loss of vegetation in those areas all together (Belleveau et al. 2015).

The Nature Trust, located in Nanaimo, B.C., is currently conducting research on estuary restoration and resilience to sea-level rise. This work will help prioritize restoration efforts based on distribution of elevation and change in elevation in 15 estuaries in B.C. This work is based on the U.S. Marsh Resilience to Sea-Level Rise (MARS) tool developed by The National Estuarine Research Reserve System (NERRS) (Raposa et al. 2016). Results from this study will contribute to meta-data on the susceptibility of estuaries on Vancouver Island to the effects of sea level rise.

A study conducted by Dean et al. (2000) identified and prioritized areas for habitat restoration in the Skagit River Estuary in Puget Sound, USA. The authors used DEMs to determine elevation ranges in the estuary. Estuary platform elevation less than 2.87 m was considered mudflat and at elevations ranging from 2.86 **-** 3.87 m, it was typical to see salt marsh vegetation. Although the absolute elevation measured in this report differs slightly from those measured in this study, this information offers another explanation for the changes observed at both sites, since elevation and salinity largely influence salt marsh vegetation (Janousek & Folger 2014). Morris et al. (2002) determined that productivity in salt marsh vegetation was most stable in the range of 10-60 cm below MHWL, and no productivity was observed 60 cm below MHWL. MHWL level estimated for Tranquil Creek estuary was 0.8 m (Robertson 2019). Similar MHWL should be used to estimate local tide levels at Moyeha River estuary since the same tidal station (Tofino Station 8615), would be used in the calculation. This implies that any part of the estuary platform measured at an elevation below 0.2 m would theoretically exhibit no plant biomass and can be considered mudflat. Elevation loss in the salt marsh area analyzed in this study could have been beneficial for vegetation growth after a historic influx of sediment. However, excess sediment could also have been deposited in the channels and in subtidal areas and remained unvegetated outside the boundary analyzed in this study due to unfavorable growing conditions.

Volume changes represent a change in sediment storage in the estuary and reflect aggradation and erosional processes. Loss of elevation in both estuaries could have occurred due to higher river discharge (Gurbisz et al. 2016). Dredging and compaction that occurred prior to 1968 on the east side of the salt marsh, to facilitate construction of a logging camp in the Tranquil Creek estuary could also have contributed to elevation changes seen in the DoD. Furthermore, loss of vegetation in the Tranquil Creek estuary during logging activities could have decreased wave attenuation and sediment trapping in those areas and contributed to local elevation loss (Barbier 2013). It is also possible that the average loss in elevation observed in both estuaries is a result of sediment compaction caused by the weight of the sediment deposit (Cahoon 2003).

Lack of experience and minimal training working with MicroStation and Summit Professional likely contributed to vertical error in elevation data collected for this study. A more experienced person selecting elevation points in MicroStation would increase elevation point accuracy and improve the evaluation of minor changes in the estuary platform. For this reason, elevation changes stated here should be considered estimates. Factors that could have been included to increase the accuracy of elevation measurements include: estuary transition zone polylines created by Robertson (2019) from ground truth elevation data from an RTK and total station survey, and estimated: maximum high water, MHWL and MLSL to delineate salt marsh zone boundaries and as control points across years.

DoDs created for Tranquil Creek estuary and Moyeha River estuary provide local changes in elevation and are specific to the common mapped area across all photograph years. The common area measured in the DEM differencing analysis was only a fraction of the total average area at Tranquil estuary and Moyeha estuary (23% and 33%, respectively). This speaks to the level of spatial variation in salt marsh habitat across years and resulted in the assessment of localized changes and regrettably is not representative of the entire estuary. The changes found in this study, however, warrant further investigation to address watershed processes influencing salt marsh habitat as it relates to Chinook salmon habitat.

Sediment Grain Size and Stratigraphy

Visible differences in the proportions of grain size categories could be due to changes in the amount of sediment delivered to the respective systems associated with landslides and debris flows (Cui et al. 2003). To better understand the differences observed between estuary sites and to include the effects of historic land use on sediment delivery, I use depth as a proxy for time. It was not surprising that cores in areas of the estuary that had been disturbed most recently, by river flow or logging activities showed less stratification i.e. Moyeha Core 1 (Fig. A2.d) and Tranquil Core 3 (Fig. A2.c). Coarse sediment in estuary cores can also be indicative of abandoned channels (Watson et al. 2013). Higher proportions of coarse grain size in the Tranquil Creek estuary reflects the spatial variability in salt marsh channels compared to the reference site, the Moyeha River estuary.

Sediment transport in mountainous streams has been thoroughly studied by Cui et al. (2003). Sediment pulses with large amounts of sand elevate the transport rate of ambient gravel creating a morphodynamic response in the river of moving gravel sheets downstream (Cui et al. 2003). High proportions of thick glaciofluvial sediments mostly gravels, and sand with minor silts are characteristic of hillslopes in both watersheds and is evident in the sediment core results for both sites (Eggers & Ferguson 2018). The connectivity between the upper and lower watershed and the forces acting on gravel deposited in the river to re-establish morphologic integrity of the river after a pulse, will eventually disperse or translocate the excess sediment downstream (Cui et al. 2003). When sediment is translocated, degradation can occur as a result of deposited sediments that then propagate downstream, causing channel erosion and deposition of sediment in the floodplain (Cui et al. 2003).

Coarser sediment is typically derived from hillslopes (Attal et al. 2015). Coarser intervals in the sediment cores can thus act as indicators that higher energy flow events are responsible for gravel deposition in the estuary derived from landslide events (Watson et al. 2013). Sediment particles larger than 10 mm represented roughly 70 % mass of landslide deposits investigated by Attal et al. (2015). This includes the three largest grain size categories included in this analysis (medium gravel (8-16 mm), coarse gravel (16-31.5 mm), and very coarse gravel (>31.5 mm). Coarse gravel found at shallower depths in Tranquil Creek estuary may be attributed to sediment pulses as a result of historic landslide events (Eggers & Ferguson 2018).

Higher levels of cohesive sediment along the channel bank reduce the ability of the system to entrain deposited sediment, making the salt marsh edge more stable to hydraulic forces (Edmonds & Slingerland 2010). This may explain the difference in variation observed between Moyeha River estuary and Tranquil Creek estuary, as Moyeha River estuary sediment cores displayed a higher proportion of fine sediments in the top five depth categories. Higher vegetation density could also describe the relative stability observed in the Moyeha River estuary since dense vegetation reduces the likelihood of resuspension of sediments (De Boer 2007). Comparatively, the Tranquil Creek estuary is at a greater risk for resuspension of sediment in the event of high flows during major flood events or wind-driven wave action because of lower levels of cohesive sediments in the top layers of the salt marsh platform (Gurbisz et al. 2016).

Higher sedge density in the reference estuary compared to the Tranquil Creek estuary (Fig. 12), could explain the higher proportions of fine sediment observed in the Moyeha River estuary since vegetation acts as a cohesive sediment agent (Edmonds & Slingerland 2010). When salt marsh becomes fragmented, it also reduces salt marsh capacity to maintain itself through deposition of detritus to build and maintain the estuary platform. Less dense vegetation exposes banks to resuspension and erosion with high flows (river and wave action).

Sediment mineralogy and stable isotope dating of sediment core samples was not conducted in this study but could have provided useful support for the origin and timing of sediment deposition (Hales 2000). Organic carbon dating has also been used to date estuary sediment core samples in the Cowichan Estuary, Cowichan Bay, BC, however coarse floodplain sediments tend to result in mineralization rather than sequestration of organic matter and may not have provided useable samples for organic carbon dating analysis (L. Kimpe 2019, Chronos Scientific Inc., personal communication).

Based on the multivariate analysis results, it does not appear that there is an obvious difference in sediment from a principle component perspective and thus I was unable to detect a direct correlation of site or logging on grain size distribution (Fig. 13). The reduction of site variability into two axes, suggests that there is a lot of overlap in the data at a community level. Sediment grain size proportions may be different however, at finer scales, as was observed in the stacked bar comparison, or at individual core locations (Ruth 2020). It is likely that the failure to detect an effect was due to a lack of replicate sites and a small sample size. It is also possible that disturbances responsible for local differences observed in grain size were not large enough to change the entire estuarine system.

Sediment core sampling methods used here were unconventional, however are easily replicable and low cost. Sampling equipment needed to take cores in regions with coarse sediments such as this would most likely not be feasible for such remote sites.

5. Implications for Estuary Restoration

5.1. Restoration Goals and Objectives

If it had been conclusively demonstrated that Chinook estuary habitat had been lost or degraded, the primary goal of a restoration plan for the Tranquil Creek estuary would be to increase the amount of available rearing habitat for juvenile Chinook salmon. The second would be to develop a monitoring plan to better understand changing estuary conditions and limiting factors affecting the quality and quantity of salt marsh habitat. Regardless of the analytical strength, the results here provide tools for design, prediction, and evaluation of habitat restoration. Given the results presented here, it may be feasible to construct new areas of salt marsh and to stabilize tidal estuary banks to increase refuge habitat and support high food production (Levings & Macdonald 1991). The next step to acquire information needed for a comprehensive restoration plan is to accurately quantify juvenile Chinook salmon distribution in the estuary to gauge restoration success in the future.

Preliminary watershed restoration should focus on hillslope and riparian stabilization to reduce sediment input from the upper watershed to the stream (Hartman & Scrivener 1990). To reduce sediment supply to the system side-casts should be stabilized, roads deactivated, and old culverts should be removed. One major constraint to restoration of fish habitat is the lack of hydrologic, hydraulic and geomorphic data (Hartman & Scrivener 1990). This leads to ad-hoc restoration treatments that rely on industry standards that may not be adequate/suitable for every watershed. Restoration projects need to be multidisciplinary and require a systematic analysis before restoration treatments can occur. Detailed planning and data collection will ultimately minimize costs and improve the success of the proposed project. Based on the hierarchical analysis proposed in the Watershed Restoration Management Program No. 7, estuary restoration as it relates to fish habitat should be done following: restoration of hillslopes, riparian areas and channels (Keeley 1994). Restoration activities conducted by CWFS to address degraded hillslope and riparian sections of the watershed are described in The Tranquil Creek Riparian Restoration Plan (Hutchinson & Warttig

2018). A report on in-stream restoration works conducted by CWFS has yet to be published, however, are described below. Recommendations for preliminary estuary restoration treatments and pilot studies follow.

5.2. Restoration Completed by CWFS

In 2016, the Clayoquot Sound Roundtable hosted a Workshop a Chinook Risk Assessment conducted on six watersheds in Clayoquot Sound to identify limiting factors affecting Chinook salmon populations and prioritize restoration efforts (Smith et al. 2016). Limiting factor number 34 (LF34) was identified as “loss of good quality foreshore, estuary and nearshore habitat i.e. loss of natural abundance and composition of benthic communities, eelgrass habitat, kelp forests and associated ecological communities” (CSRT 2016). Tranquil Creek and Moyeha River watersheds were assigned a moderate risk level rating at the time of the assessment and it was predicted that both watersheds would reach a high-risk level rating in the future (Smith et al. 2016). Smith et al. 2016 speculated that estuary habitat loss was due to channel infilling and increases in in-stream winter flows (CSRT 2016).

Since the workshop, restoration work has been completed by CWFS in the main stem of the Tranquil River and in the riparian zone to address limiting factors addressed in the CSRT (2016) summary report for Tranquil Creek Chinook. The goal of the Tranquil River Restoration Initiative implemented by CWFS is to restore wild Chinook populations by restoring critical habitat that has been degraded. Activities completed to date include 8 in-stream structures, 8 bar-top structures, and riparian restoration activities. Riparian activities in a previously logged 100 m riparian buffer zone were completed in March 2019 and included 19 ha of brushing activities and planting of 9,800 seedlings (Sitka Spruce and Western Red Cedar) (Hutchinson & Warttig 2018). Channel excavation and engineered logjams were created in summer 2019 to restore riffle-pool habitat and to mitigate erosion, bedload movement and to increase habitat complexity (Roni et al. 2014). Bar-top willow-staking was also used to catch fine sediments, stabilize active gravel bars and promote vegetation growth (Polster 2015).

To stabilize channel banks and gravel bars, 30,000 alder, 10,000 native conifers, and 500 native shrub rooted seedlings were planted (Polster 2015).

5.3. Preliminary Restoration Plan for Tranquil Creek Estuary

Methods to improve habitat quality and quantity of rearing habitat in the estuary have yet to be implemented. The timing of estuary restoration implementation is dependent on the success of restoration efforts conducted by CWFS. If vegetation plantings in the riparian zone and on gravel bars establish, this will help prevent further movement of coarse and fine sediments into the estuary (Polster 2015).

5.3.1. Site Assessment

Historic Conditions

Historic salt marsh conditions in the Tranquil Creek estuary are described above, based on historical aerial photograph analysis over the observational period of 1937-2019. Salt marsh area and elevation were quantified to determine how current estuary habitat compares to historical conditions prior to industrial clear-cut logging in the Tranquil Creek watershed.

Current Conditions

Vegetation surveys were conducted at Tranquil Creek estuary on 15-August 2019. Comparatively, Lyngby’s sedge (*Carex lyngbyei*) density was lower in the lower salt marsh at Tranquil Creek estuary compared to reference conditions at Moyeha River estuary. Vegetation species observed at Tranquil are listed in Table1. All species listed are native in the PNW and should be considered when selecting plants for restoration purposes (Pojar 1994).

Juvenile Chinook salmon monitoring with a beach seine was conducted by Central Westcoast Forest Society in the Tranquil estuary on 31-May 2019. Distribution of juvenile Chinook in Tranquil Creek estuary is displayed in Extra Figure 7. Data from previous years was not available and thus could not be compared across the observational period of this study.

Localized elevation changes measured at Tranquil creek estuary, calculated using DEM differencing, indicate areas that are lower in elevation compared to historical conditions. Although the salt marsh area has varied over time, it is its largest at this time.

5.3.2. Stressors and Impacts

Current stressors that should be the focus of preliminary restoration activities in the estuary include aggradation from sediment that resides in the main river channel and flashy high flows that have the potential to deliver coarse sediment to the floodplain. Hydraulic forces and coarse bedload movement are likely to cause erosion of the channel edge and elevate water levels in the estuary. Both stressors may cause loss of estuary salt marsh vegetation needed to maintain salt marsh channel form to provide cover and invertebrate food production (Simenstad et al.2002; Gurbisz et al. (2016).

Legacy effects in areas where logging camps were built, logs were stored, and roads were constructed (visible in Tranquil photo 1968) likely persist on the east side of the Tranquil estuary. These activities likely resulted in compaction and degradation of foreshore marsh habitat and contributed to a transition from a diverse tidal marsh community to a monoculture in those areas. Vegetation loss and land scarring from moving logs in the estuary is also visible in the 1968 aerial photographs (Extra Fig. 8). This area in the estuary has experienced the largest change over the observational period and should be the focus of pilot restoration efforts.

5.3.3. Restoration Treatments and Pilot Study

1. Lyngby's sedge (*Carex lyngbyei*) transplants

Using DEM differencing results, it is possible to identify areas across the identified salt marsh that have elevation to support optimum sedge productivity.

Combined with ground truthing vegetation surveys or current large-scale, high-resolution aerial photos, we can prioritize areas across the estuary that could benefit from sedge transplants. Lyngby's sedge produces deep root systems and acts as an ecosystem engineer to stabilize the deepest channel edges where Chinook are often found (Simenstad et al. 2002; Hood et al. 2018). Vegetation should be planted using an experimental design to rigorously quantify establishment success in such a highly variable and under-studied system.

Increasing salt marsh area with transplants can also increase estuary ecosystem resilience (Wainger et al. 2017). Increasing sedge density will result in wave attenuation and sediment trapping and may reduce the effects of high flows on salt marsh channel bank erosion (Barbier 2013). Sedges produce large above ground biomass and thus transplants will also contribute to a positive feedback of organic sediment supply in the form of detritus to counteract sea level rise. More vegetation and detritus will also benefit invertebrate prey for Chinook (Simenstad & Cordell 2000). Simenstad & Cordell (2000) also suggest that capacity to produce invertebrates will increase as the marsh becomes more like reference ecosystems.

Foreshore areas adjacent to the deactivated logging road may also benefit from sedge transplants however the entirety of this area was not captured in my analysis. Subsequent elevation and vegetation surveys in these areas can be conducted to determine if terrestrial vegetation is encroaching on historic estuary habitat. It may be feasible to lower elevation in these areas to recreate salt marsh elevation and environment as was successful in a marsh re-leveling project in the Effingham River estuary conducted by M.C. Wright and Associates Ltd. (Wright 2016). It is also likely that these areas have been impacted by compaction, which decreases the ability of water and roots to penetrate the sediments and limits nutrient uptake (Polster 2015). Seeds that are dispersed in the estuary on compact surface sediments may also be washed away in high flows on compact sediment (Polster 2015). These areas may benefit from structural components such as large woody debris that can trap fine sediments until sedges or other early successional marsh vegetation can establish (Polster 2015).

2. Excavation of tidal channels

A second restoration treatment that has been used to maximize habitat for juvenile Chinook is the excavation of tidal channels that have aggraded (Wallace 2005). It may, however, take 4-13 years for constructed channels to stabilize in an estuary setting (Williams et al. 2002). This should be a small-scale project involving only 2-3 channels and should target 1st order channels (Simenstad et al. 2002; Hood 2014). This work should be supplemented with juvenile salmon sampling the following rearing season on falling tides with pole seines or fyke nets, to determine if juvenile Chinook are using the habitat following restoration (Levings 2019). The target depth of excavated channels should be >0.4 m based on surveys conducted by Hering et al. (2010). It is important that a comparative study be conducted that focuses on channel bathymetry at Tranquil Creek estuary compared to the Moyeha River estuary, or another reference system. The decision to excavate channels, or not, will be based on these findings, and should evaluate the total available channel habitat, based on depth conditions outlined by Hering et al. (2010). Ground-based topographic surveys using a rod and level for cross sections of major channels would be a low-cost method to determine the elevation in estuary channels and to monitor channel changes over time (Williams et al. 2002).

To maximize capacity for rearing juvenile Chinook salmon, restoring a large, complex tidal channel system may be most important (Simenstad et al. 2002). This will facilitate access deep into the marsh, maximize marsh edge for foraging and generate the highest possible water residence time (Levings 2016; Simenstad et al. 2002). The importance of connectivity in tidal channel networks is paramount (Ellings et al. 2016). Restored habitat in the Nisqually Delta, WA, was used by juvenile salmon within one year of restoration activities despite habitat quality differences between restored and reference sites (Ellings et al. 2016).

3. Raise Salt Marsh Platform

Lastly, I recommend efforts to raise the salt marsh platform in areas that have experienced a loss in elevation based on DEM differencing (Fig. 14) to counteract the effect of SLR. These areas are at a higher risk of SLR, despite regional uplift, and innovative methods should be used to prevent loss to these important habitats. This presents an opportunity to test the transplant potential of other native salt marsh species. Estuary studies in Oregon found that salt marshes that were left to recover on their own after dike removal became predominantly Lyngby's sedge and that these estuaries were unlikely to regain natural structural complexity due to a loss in elevation (Hoobyar 2007). Establishing high salt marsh vegetation diversity will maximize access to area above marsh vegetation as opposed to tall monotypic sedge meadows that restrict access with increasing vegetation height (Hood et al. 2018).

Excavated sediments from the above mentioned first-order channels or gravel excavated from in-stream restoration efforts could be used as a potential restoration strategy to raise the estuary platform followed by salt marsh vegetation transplants. Elevating the marsh platform helps to restore marsh structure and function (Burdick et al. 2019). A study in the Eastern United States showed that adding 5 - 20 cm of sediment in sparsely vegetated areas of degraded salt marsh supported the reestablishment of marsh vegetation (Berkowitz et al. 2017).

Infilling with sediment and excavation activities should occur in the appropriate fish work window and all fish species should be removed via seine from the targeted area in accordance with the *Fisheries Act* (2012) and the *Fish Protection Act* (1997). This proposed pilot project will require adequate funding to account for site remoteness and heavy machinery required to conduct this work.

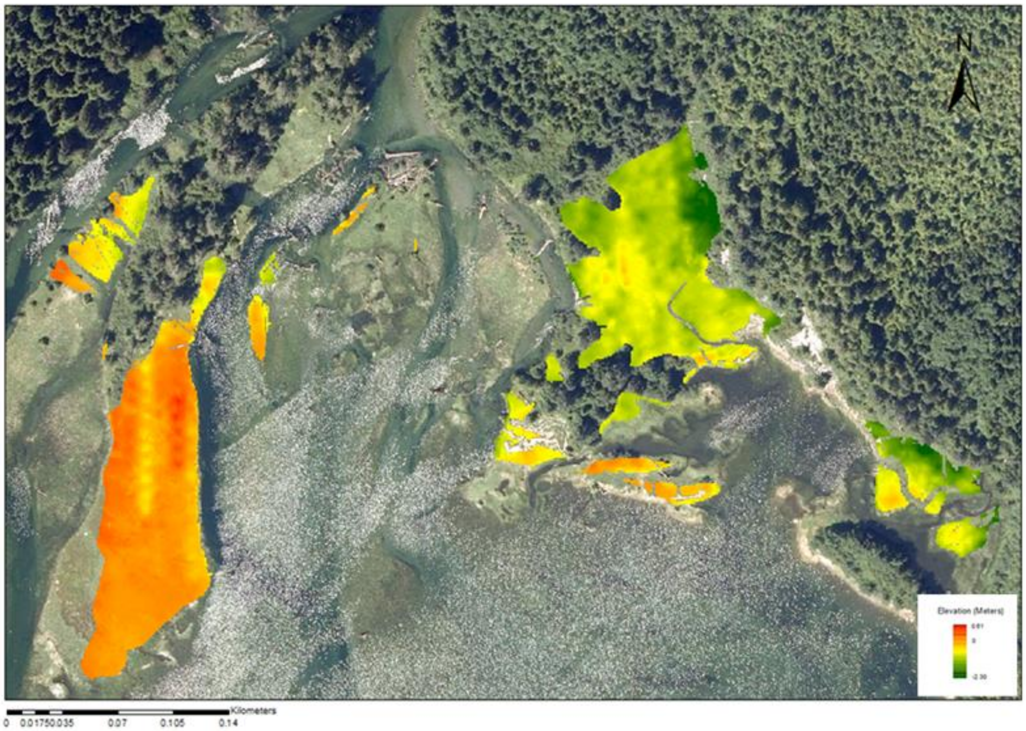

Figure 14. Priority Restoration Areas in Tranquil Creek Estuary.
Areas that display a negative change in elevation since 1996 (Yellow and Green) – DEM differencing results overlaid on the 2014 orthophotograph of Tranquil Creek estuary. Elevation change range: -2.3 m to +0.6m.

5.3.4. Monitoring and Future Research

Building on the data collected and presented here, is necessary to develop a full-scale restoration plan for the Tranquil Creek estuary. Regardless, if CWFS wishes to proceed with the proposed restoration treatments, an essential component is a dedicated monitoring plan to evaluate restoration success (Dionne & Peter 2012). Elevation, sediment deposition, tide levels (inundation), sediment nutrient levels, dissolved oxygen levels, and juvenile salmonid distribution should be monitored prior to and following restoration treatments. The Restoration Performance Index (RPI) used by Dionne & Peter (2012) can also be applied to gauge restoration success.

Transport and deposition of gravel and sand in the main stem of Tranquil River needs to be monitored to determine how these processes will affect in-stream restoration projects and to predict the amount of sediment that will be delivered to the estuary. Sediment deposition in the river can be monitored using weirs after major flood events to evaluate local sedimentation trends and model bedload movement in response to high flows (Chartrand et al. 2015). Cross section transects can also be used to monitor relative elevation change in river channels and can be applied to monitor changes in estuary channels over time (Chartrand et al. 2015). Sediment traps or weirs can also be used as an integral part of monitoring the effectiveness of restoration projects in logged watersheds involving erosion control (Larkin et al. 1998).

Surface Elevation Tables (SET) have been installed in several estuaries on Vancouver Island including: the Quatse River, Cluxewe River, Salmon River, Englishman River, Nanaimo River and Cowichan River estuaries, as part of The Vancouver Island Conservation Land Management Program (VICLMP) (HCTF 2016). These devices measure elevation of substrate on the marsh platform and help to quantify trends in deposition and erosion over time. SETs can be used to determine if transplants are establishing and stabilizing estuary sediments (HCTF 2016). This method can also be supplemented with current aerial photographs to monitor changes in salt marsh area, channel connectivity and elevation (Ellings et al. 2016). Ideally photos should be taken at regular intervals to accurately capture changes outside the natural variation of the system. Photos should certainly be taken pre and post restoration treatments to accurately monitor changes at the landscape scale (Ellings et al. 2016). It is also important that the scale of the photos be consistent with historic photos to accurately compare changes over time (A. Pearson 2020, Simon Fraser University, Burnaby, BC, personal communication).

These data will also contribute to a SLR susceptibility analysis to determine if restoration is feasible given the projected sea levels (Reposa et al. 2016). Numeric thresholds and color codes for individual metrics and all categories and indices are presented by Raposa et al. (2016). The B.C. Salmon Restoration and Innovation Fund may offer an opportunity to acquire funding to determine marsh resilience to sea-level rise (MARS) scoring, developed by the U.S. National Estuarine Research Reserve System, for Tranquil Creek estuary. This would involve collecting data for five indicators of estuary resilience: 1. distribution of elevation, 2. change in elevation, 3. sediment supply, 4. tidal range, and 5. rate of sea level rise. This would determine if restoration is feasible given the trajectory of the system in the face of sea-level rise (Raposa et al. 2016). The highest rating that warrants pursuit of restoration activities is yellow, and a red rating indicates that recovery is slim given SLR predictions.

To understand the effectiveness of restoration with regards to juvenile Chinook, a rigorous juvenile salmon monitoring protocol must be implemented. Sample sites need to be chosen carefully, maintained at standard locations and sampled regularly e.g. every two weeks (Levings 2019). A study on resilience factors in estuaries conducted by Wainger et al. (2017) recommends using spatial distribution of fish populations as an indicator of ecosystem resilience. This measure can help us understand the response of Chinook populations to changes in habitat through restoration. RST data from the annual juvenile salmon monitoring project (Kwasnecha & Hutchinson 2019), combined with PIT-tagging to conduct a mark recapture study would be the ideal method to determine Chinook distribution and habitat use (Hering et al. 2010). Fyke nets and pole seining on falling tides should be used over beach seining when possible, since beach seining can be limited to areas that avoid LWD snags that may provide suitable habitat complexity for salmonids (Levings 2019). Snorkel surveys can also be used to monitor juvenile Chinook habitat use but this method lacks a well-defined protocol in B.C. (Levings 2016). One factor that needs to be addressed to accurately determine natural Chinook distribution in the estuary is juvenile Chinook from The Tofino Salmon Enhancement Society that are released downstream of the RST. Currently there is no way to undoubtedly determine that a juvenile chinook caught in the estuary is of natural or hatchery origin since hatchery-origin Chinook have never been clipped or tagged (Kwasnecha & Hutchinson 2019).

6. Conclusions and Future Research

While I failed to find any evidence of a difference between a logged area and an unlogged area, some negative trends in salt marsh area and elevation observed were indicative of changes that are unfavorable for juvenile Chinook salmon. Changes in hydrology likely contributed to trends in the rate of change experienced at Tranquil Creek estuary over a portion of the observational period (1995-2019). High flows and vegetation loss at that time likely contributed to erosion of salt marsh habitat and resulted in a decrease in elevation where elevation data was compared. Relatively lower flows at the end of the study period, may have promoted vegetation recovery and contributed to the increase in salt marsh area observed in both estuaries. However, further studies are needed to confirm my theory of channel infilling and natural succession of vegetation into these areas. Furthermore, fine-scale sediment grain size differences were indicative of sediments derived from landslides that are more prone to erode in high flows and it is spatially plausible that areas with less defined stratigraphy have been disturbed more recently in time.

Logistical constraints prevented the study of multiple replicates of logged and unlogged sites. Had this been possible, I could have made inferences towards causal effects of logging. The analytic methods presented here to assess changes in two remote coastal estuaries have contributed to the current knowledge on the effects of logging on estuarine and marine ecosystems in coastal B.C. This study can be replicated and refined elsewhere or expanded upon to further the current understanding of landform evolution and how historic land use affects complex estuary dynamics.

The areas identified to have decreased in elevation are targets for restoration as their risk to the effects of SLR are significantly higher. Given current SLR predictions, we can expect to see changes in distribution, community composition and species richness of salt marsh vegetation (Janousek & Folger 2014). Preliminary restoration techniques have been suggested including, sedge transplants, channel excavation, and elevation correction, as well as rigorous juvenile salmon distribution surveys to guide future restoration actions conducted by CWFS.

Improving our understanding of ecosystem processes, and the underlying causes of ecological changes observed over time can help develop reasonable ecosystem-based management plans to improve the resilience of ecosystems (Weinstein 2008). It is paramount to evoke that restoration is a poor substitute for habitat protection in the first place (Hartman et al 1996). To protect and restore weak Chinook stocks identified as CUs in Clayoquot Sound, conservation areas that encompass entire watersheds (including a generous riparian buffer and estuary) – involving crown land designation and purchase of private land, should be considered (Stalberg et al. 2009).

7. References

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8. Tables

Table 3. List of Aerial Photographs
Ordered from GeoData BC (Province of British Columbia 2019) and The National Air Photo Library (Natural Resources Canada 2019).

Aerial Flight Line/Photo Number –Type of photo	Watershed	Year
A5778/43 - Digital Air Photo	M	1936-50
A5778/44 - Digital Air Photo	M	1936-50
A5778/45 - Digital Air Photo	M	1936-50
A5778/46 - Digital Air Photo	M	1936-50
BC2310/31 - Digital Air Photo	M	1954
BC2310/32 - Digital Air Photo	M	1954
BC2310/33 - Digital Air Photo	M	1954
BC2310/34 - Digital Air Photo	M	1954
BC7085/227 - Digital Air Photo	T	1968
BC7085/228 - Digital Air Photo	T	1968
BC7085/229 - Digital Air Photo	T	1968
BC7238/218 - Digital Air Photo	T	1970
BC7238/219 - Digital Air Photo	T	1970
BC7238/220 - Digital Air Photo	T	1970
BC78074/119 - Digital Air Photo	M	1978
BC78074/120 - Digital Air Photo	M	1978
BC78074/121 - Digital Air Photo	M	1978
BC78074/122 - Digital Air Photo	M	1978
BC78074/123 - Digital Air Photo	M	1978
BC81072/169 - Digital Air Photo	M	1981
BC81072/170 - Digital Air Photo	M	1981
BC81072/171 - Digital Air Photo	M	1981
BC81072/172 - Digital Air Photo	M	1981
BC81072/173 - Digital Air Photo	M	1981
BC81072/184 - Digital Air Photo	M	1981
BC81072/185 - Digital Air Photo	M	1981
BC81072/186 - Digital Air Photo	M	1981
BC81072/187 - Digital Air Photo	M	1981
BC81072/296 - Digital Air Photo	T	1981
BC81072/297 - Digital Air Photo	T	1981
BC81072/298 - Digital Air Photo	T	1981
BC81072/299 - Digital Air Photo	T	1981
BC81073/42 - Digital Air Photo	T	1981
BC81073/43 - Digital Air Photo	T	1981
BC81073/44 - Digital Air Photo	T	1981
BC81073/45 - Digital Air Photo	T	1981
BCC96001/148 - Digital Air Photo	M	1996
BCC96001/149 - Digital Air Photo	M	1996
BCC96001/150 - Digital Air Photo	M	1996
BCC96005/21 - Digital Air Photo	M	1996
BCC96005/22 - Digital Air Photo	M	1996
BCC96005/23 - Digital Air Photo	M	1996
BCC96005/24 - Digital Air Photo	M	1996
BCC96006/100 - Digital Air Photo	T	1996
BCC96006/101 - Digital Air Photo	T	1996
BCC96006/102 - Digital Air Photo	T	1996
BCC96006/127 - Digital Air Photo	T	1996
BCC96006/128 - Digital Air Photo	T	1996
BCC96006/129 - Digital Air Photo	T	1996
BCC96009/24 - Digital Air Photo	T	1996
BCC96009/25 - Digital Air Photo	T	1996
BCC96009/26 - Digital Air Photo	T	1996
BCC96009/75 - Digital Air Photo	T	1996
BCC96009/76 - Digital Air Photo	T	1996
BCC96009/77 - Digital Air Photo	T	1996
BCD14101/783 - Digital Air Photo	M	2014
BCD14101/784 - Digital Air Photo	M	2014
BCD14101/785 - Digital Air Photo	M	2014
BCD14102/73 - Digital Air Photo	M	2014
BCD14102/74 - Digital Air Photo	M	2014
BCD14102/75 - Digital Air Photo	M	2014
BCD14102/76 - Digital Air Photo	M	2014
BCD14103/699 - Digital Air Photo	T	2014
BCD14103/700 - Digital Air Photo	T	2014
BCD14103/701 - Digital Air Photo	T	2014
10962 - TRIM Orthophoto bc_092f041_xc500mm_utm10_2014.tif	M	2014
10986 - TRIM Orthophoto bc_092f022_xc500mm_utm10_2014.tif	T	2014
6172 - TRIM Orthophoto bc_092f022_xc1m_utm10_1996.tif	T	1996
6190 - TRIM Orthophoto bc_092f041_xc1m_utm10_1996.tif	M	1996

Table 4. Aerial Photograph Calibration Data and Tide Estimates
Tide estimates were conducted by the Canadian Hydrographic Service (Ballantyne 2020).

Watershed	Year	Date	Time	Photo Roll	Frame Numbers	Scale	Focal length (mm)	Tide Height Estimate
T	1937	1937-10-09	1442-1526	A5776	65, 66	1:20,000		2.874-2.829m
T	1956	1956-10-07	1255-1337	BC2309	67, 68	1:30,000		2.790-3.226m
T	1968	1968-07-26	1320	BC7085	227, 228, 229	1:16,000	305	2.95m
T	1970	1970-06-01	900-1230	BC7238	218, 219, 220, 246, 247	1:15,000		2.086-2.329m
T	1981			BC81072	298, 299			
T	1981	1981-07-25	no time specified	BC81073	43, 44			1.90-1.410m
T	1996	1996-07-07	840-1421	BCC96006	100, 101, 102; 127, 128, 129	1:30,000	153	0.902m
T	1996	1996-07-07	1302	BCC96009	24, 25, 26; 74, 75, 76, 77	1:15,000	305	1.127m
T	2014	2014-06-27	1410	bcd14103	700, 701	1:20,000	80	2.969-2.961m
T	2019			LiDAR data				
M	1936			A5778	43, 44,45, 46	1-20 chains		
M	1954			BC2310	31,32,33,34	1-40 chains		
M	1978	1978-06-23	925-10	BC78074	119, 120, 121, 122, 123	1:20,000		0.194-0.444m
M	1981	1981-06-25	954-1142	BC81072	169, 170, 171, 172, 173; 184, 185, 186, 187, 187	1:20,000		1.427-0.923m
M	1996	1996-07-06	1536	BCC96001	148, 149, 150	1:30,000	153	2.849m
M	1996	1996-07-06	1536	BCC96005	21, 22, 23, 24	1:15,000	305	2.849m
M	2014	2014-07-10	1034-1035	bcd14101	783, 784, 785	1:20,000	80	2.847-2.853m
M	2014	2014-07-08	1243-1244	bcd14102	73, 74, 75, 76	1:20,000	80	1.819-1.817m

Table 5. UTM Coordinates for Sediment Cores

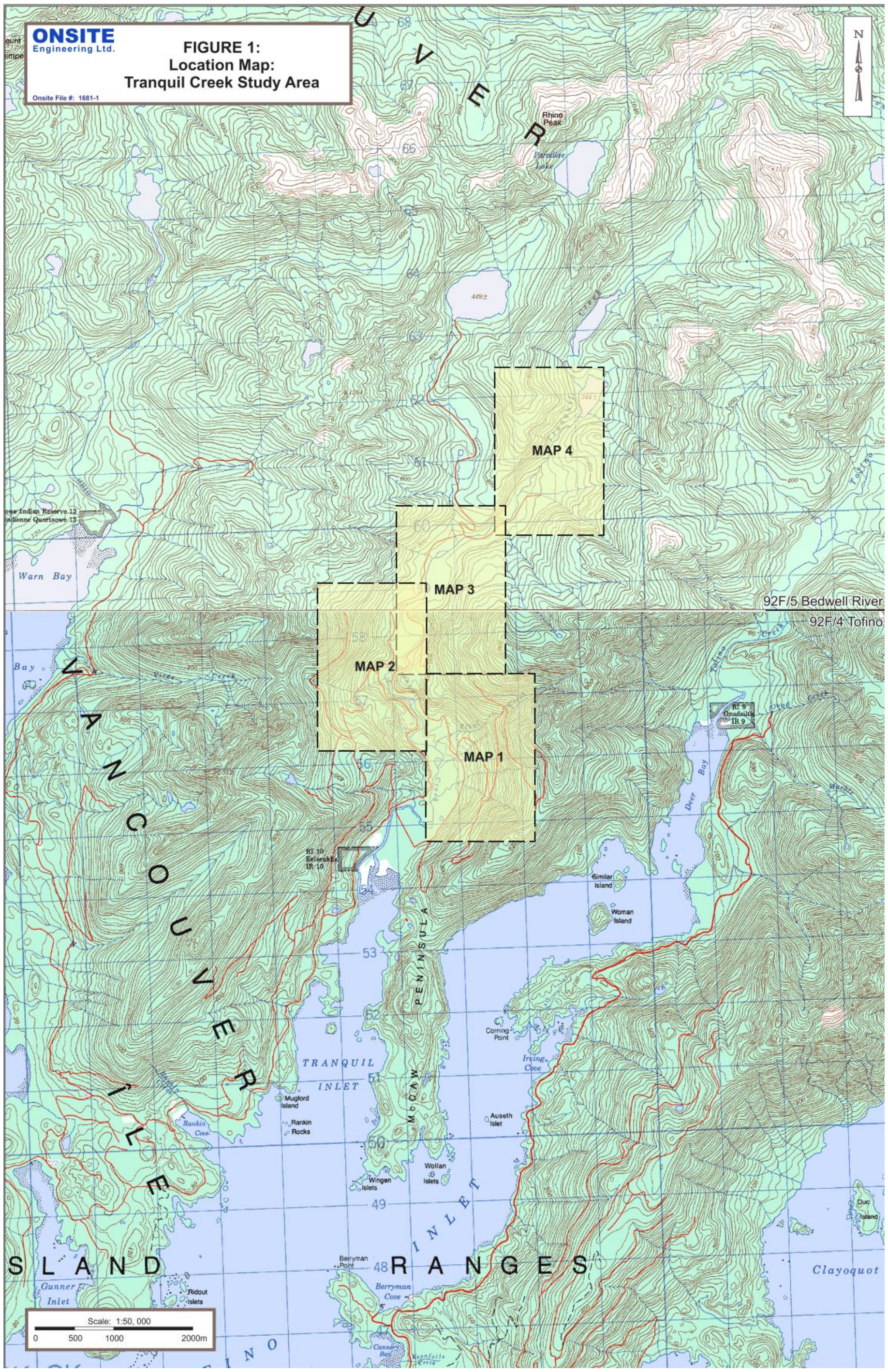
Tranquil Estuary Sediment Core Sampling locations		
Core number	UTM	
1	10U0305385	5454456
2	10U0305779	5454441
3	10U0305385	5454480
4	10U0305387	5454329
5	10U0305532	5454382
6	10U0305483	5454376

Moyeha Estuary Sediment Core Sampling Locations		
Core number	UTM	
1	10U0289351	5478172
2	10U0288696	5478005
3	10U0288750	5478104
4	10U0288943	5478080
5	10U0288946	5478131
6	10U0288818	5478205

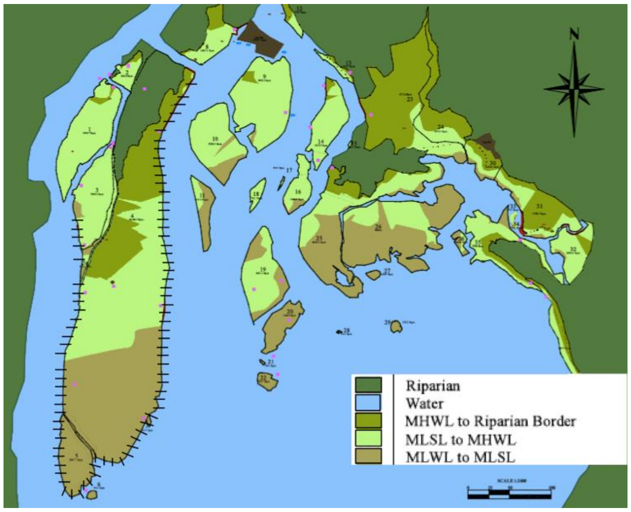
Table 6. Salt Marsh Vegetation in the Nisqually River Delta (Belleveau et al. 2015).
"Soil pore-water salinity ranges, and salt marsh zone of plant species encountered in the Nisqually Delta. Species with a significant relationship of <0.05 for salinity or elevation are indicated for percent cover (P), height (H), and Density (D) for the nine most common species. Marsh zones of species occurrence are specified as low marsh (L:2.0-2.7 NAVD88 m) and high marsh (H:>2.7 NAVD88 m)." Table and table caption are sourced directly from (Belleveau et al. 2015).

Species Code	Common Name	Scientific Name	Salinity Range (PSU)	Elevation Range (m, NAVD88)	Marsh Zone
CALY	Lyngby's sedge	*Carex lyngbyei*	2 – 30[PHD]	2.10 - 3.03	L/H
DISP	Salt grass	*Distichlis spicata*	4 – 45[PD]	2.08 - 3.11	L/H
GRIN	Entire-leaved gumweed	*Grindelia integrifolia*	20 – 45[PHD]	2.80 - 3.08[PHD]	H
JACA	Salt-marsh daisy	*Jaumea carnosa*	4 – 45[PD]	2.57 - 3.08[PHD]	L/H
JUBA	Baltic rush	*Juncus balticus* ssp. *ater*	5 – 28[H]	2.51 - 3.09[PHD]	L/H
POAN	Silverweed	*Potentilla anserina*	10 - 30	2.79 - 3.10[PHD]	H
SAPE	Pickleweed	*Sarcocornia perennis*	5 – 45[PHD]	2.33 - 3.08	L/H
SPSP	Sand spurry	*Spergularia* spp.	11 - 32	2.08 - 2.95	L/H
TRMA	Sea arrow-grass	*Triglochin maritima*	10 - 45	2.19 - 3.05[PH]	L/H
AGGI	Redtop	*Agrostis gigantea*	12 - 23	2.75 - 3.07	H
ATPA	Spear saltbush	*Atriplex patula*	5 - 45	2.54 - 3.08	L/H
BOMA	Seacoast bulrush	*Bolboschoenus maritimus*	2 - 16	2.60 - 2.93	L/H
COCO	Brass buttons	*Cotula coronopifolia*	10 - 11	2.77 - 2.91	H
CUPA	Salt-marsh dodder	*Cuscuta pacifica*	28 - 45	2.57 - 3.01	L/H
DECE	Tufted hairgrass	*Deschampsia cespitosa*	10 - 30	2.78 - 2.99	H
ELPA	Dwarf spikerush	*Eleocharis parvula*	11 - 28	2.36 - 2.91	L/H
ELRE	Quackgrass	*Elymus repens*	13 - 18	2.78 - 3.29	H
LYMA	Sea milkwort	*Lysimachia maritima*	10 - 45	2.57 - 3.06	L/H
HOBR	Meadow barley	*Hordeum brachyantherum*	5 - 21	2.47 - 3.07	L/H
HOJU	Foxtail barley	*Hordeum jubatum*	16 - 30	2.81 - 2.91	H
LACA	Canadian lettuce	*Lactuca canadensis*	NA	2.90 - 3.07	H
LIOC	Western lilaeopsis	*Lilaeopsis occidentalis*	12 - 17	NA	NA
PLMA	Sea plantain	*Plantago maritima*	5 - 45	2.57 - 3.09	L/H
PUNU	Pacific alkali grass	*Puccinellia nutkaensis*	5 - 33	2.81 - 3.08	H
SPCA	Canadian sand spurry	*Spergularia canadensis*	15 - 45	2.47 - 3.08	L/H
STHU	Salt-marsh chickweed	*Stellaria humifusa*	22 - 45	2.90 - 2.98	H
SYSU	Douglas' aster	*Symphyotrichum subspicatum*	NA	2.88 - 3.10	H

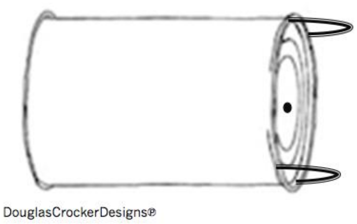
9. Extra Figures



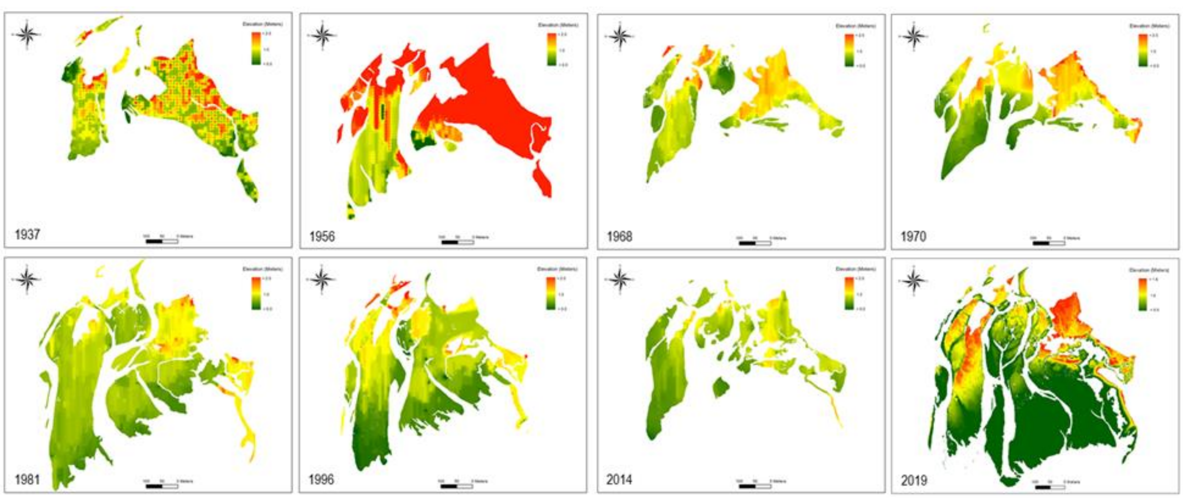
Extra Figure 1. Study sites for Terrain Stability Assessment.
Map 1 and 2 represent study area 1 and study area 2 respectively. Study areas were identified for a Terrain Stability Assessment conducted by Onsite Engineering Ltd. (Eggers & Ferguson 2018)



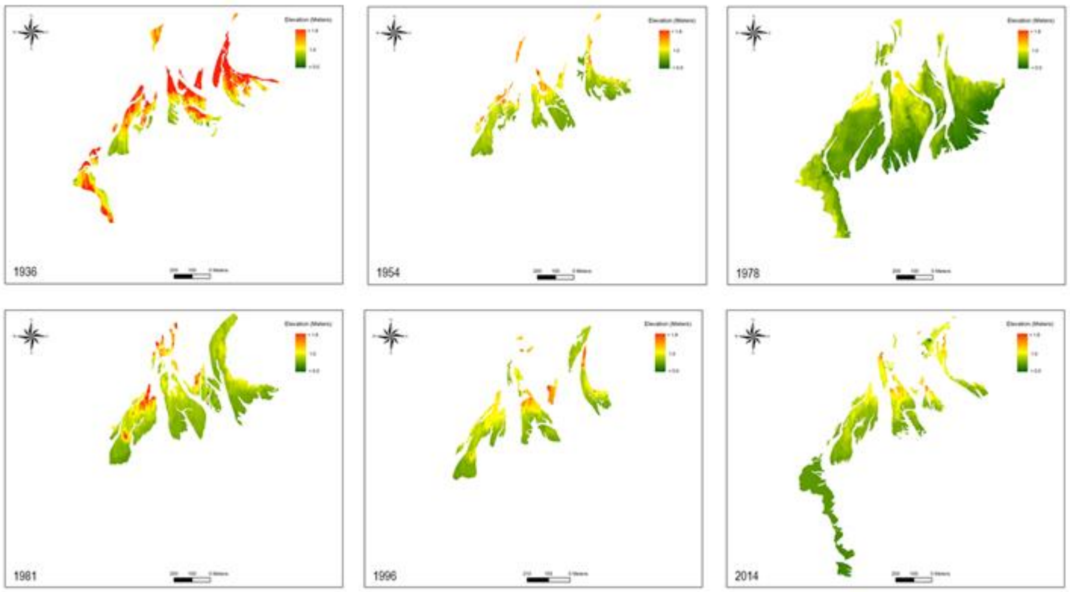
Extra Figure 2. Stratified Systematic-Random Sampling Design.



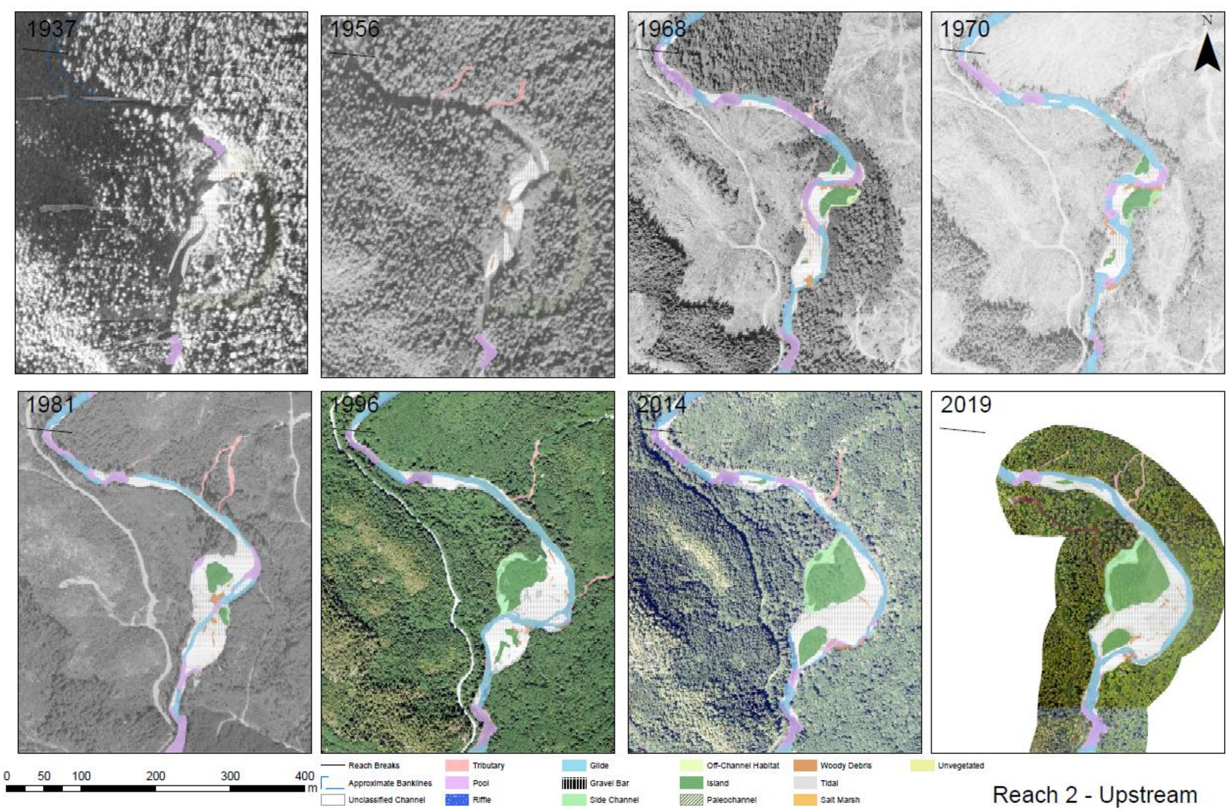
Extra Figure 3. Sampling Device.
555ml can with two handles and a hole in the top to add suction, used to extract sediment samples. Douglas Crocker Designs ℗



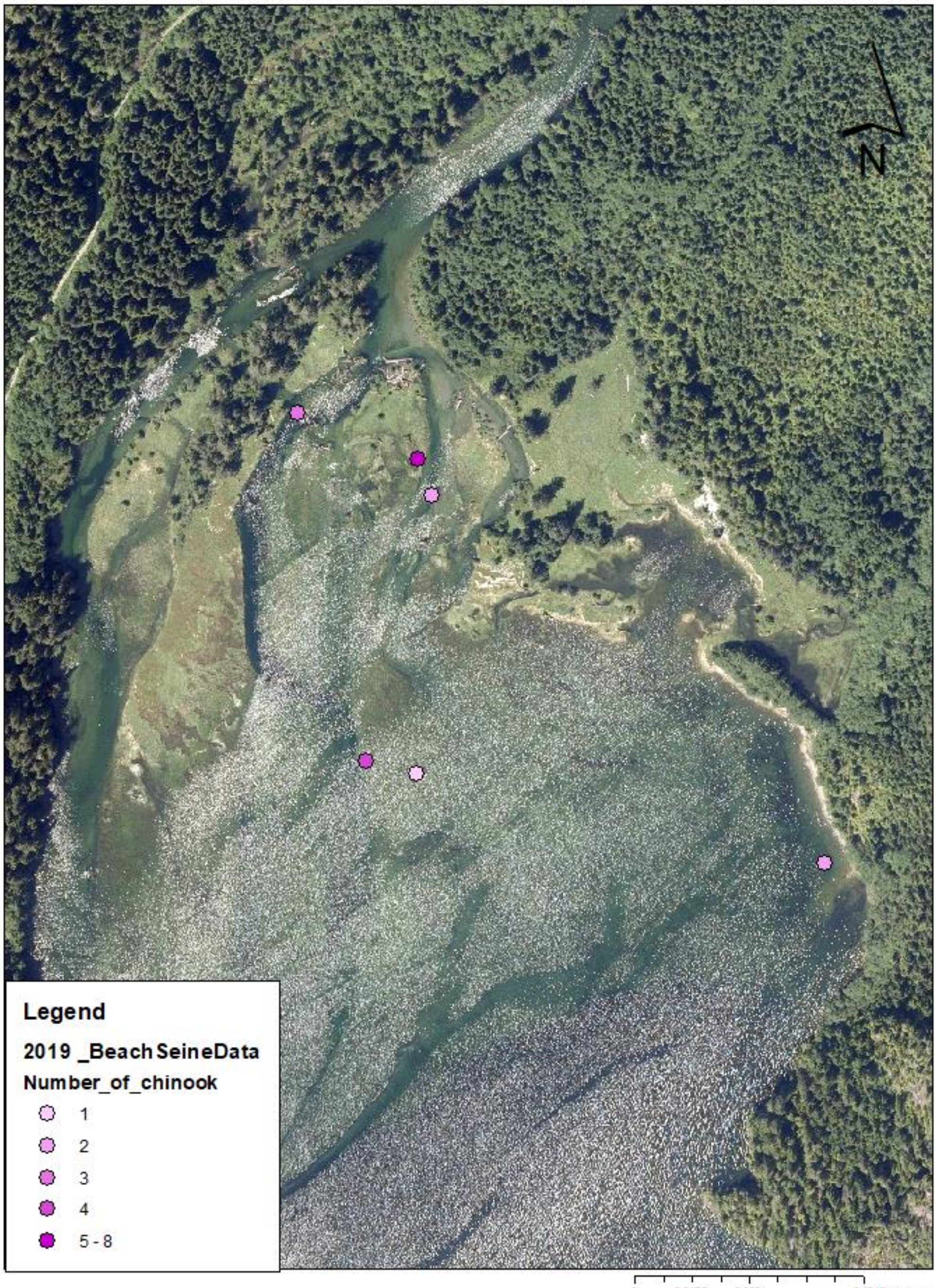
Extra Figure 4. Digital elevation models of Tranquil Creek Estuary.
DEMs were created from point elevation data collected in Summit Professional and MicroStation. Years correspond to orthophoto sets used to create summit models. Contour maps were created in ArcGIS by SFU Geography Department (Song & Ng 2020).



Extra Figure 5. Digital elevation models of Moyeha River Estuary
DEMs were created from point elevation data collected in Summit Professional and MicroStation. Years correspond to orthophoto sets used to create summit models. Contour maps were created in ArcGIS by SFU Geography Department (Song & Ng 2020).



Extra Figure 6. Mapping results of reach 2 – Upstream of Tranquil Creek estuary
Habitat types were hand digitized in ArcGIS 10.7 on orthophotos of Tranquil Creek watershed from 1937, 1956, 1968, 1970, 1981, 1996, 2014, and 2019. Mapping was conducted by Tuzlak (2019).



Extra Figure 7. Juvenile Chinook Distribution Map in Tranquil Estuary
Juvenile Chinook captured in beach seining conducted by CWFS on 31-May 2019.

Extra Figure 8. Close up of the east side of Tranquil Creek estuary visible in the 1968 aerial photograph. The photo shows land scarring from logs being dragged through the estuary, log storage areas, a logging camp and a logging road.

10. Appendix A. Sediment Cores

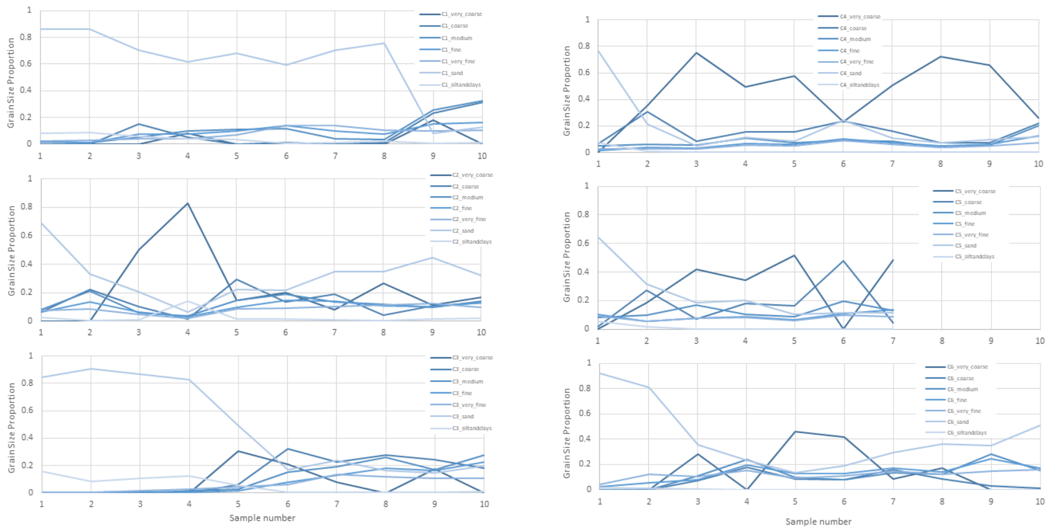


Figure A1 a. Grain Size distribution of Cores - Tranquil Creek Estuary
Grain size proportion of 7 grain size categories (gravel: very coarse, coarse, medium, fine and very fine; sand; and silt and clays) for 6 cores taken in Tranquil Creek estuary taken on 26 and 30 August 2019. Sample numbers correspond to the distance that each sample was taken from the estuary surface.

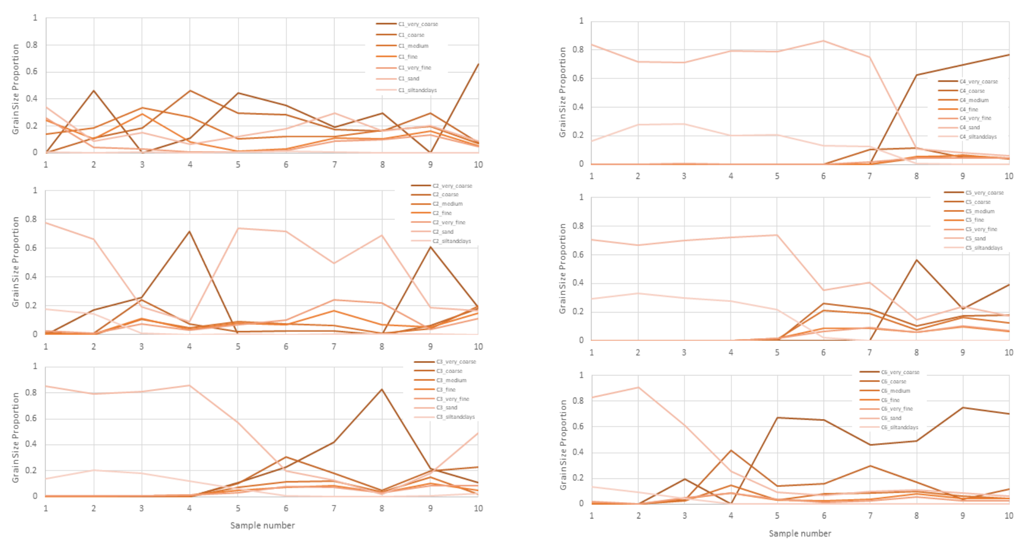


Figure A1 b. Grain Size distribution of Cores - Moyeha River Estuary
Grain size proportion of 7 grain size categories (gravel: very coarse, coarse, medium, fine and very fine; sand; and silt and clays) for 6 cores taken in Moyeha estuary taken on 29 August 2019 and 15, 16-September 2019. Sample numbers correspond to the distance that each sample was taken from the estuary surface.

Photos of Sediment Cores

Figure A2 a. Photos 1-3. Tranquil Core 1
Photos were taken 26 August-2019.

Figure A2 b. Photos 4-6. Tranquil Core 2
Photos were taken 26 August 2019.


Figure A2 c. Photos 7-9. Tranquil Core 3
Photos were taken 26 August 2019.


Figure A2 d. Photos 10-12. Moyeha Core 1
Photos were taken 31 August 2019.

Figure A2 e. Photos 13-14. Moyeha Core 2
Photos were taken 31 August 2019.