SKEENA RIVER ESTUARY SYNTHESIS REPORT

MARCH 2021

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Research supported by the Natural Sciences and Engineering Research Council, Liber Ero Foundation, Coast Opportunities Fund, Fisheries and Oceans Canada, and Lax Kw'alaams Fisheries

Citation: Sharpe, C., Moore, J.W., Carr-Harris, C., & Butts, K. 2021. Skeena River Estuary Synthesis Report. Simon Fraser University, Vancouver, BC.

ACKNOWLEDGEMENTS

We sincerely thank all the contributing authors and Lax Kw'alaams First Nation for inspiration and on-going support for this research. This research was generously supported by Liber Ero Foundation, Natural Sciences and Engineering Research Council, Coast Opportunities Fund, and Fisheries and Oceans Canada through the Oceans Protection Plan, the Fish Habitat Restoration Initiative, and Skeena Area Marine Research Collaboration. We also thank the Gitxsan Watershed Authority and Lake Babine Nation for collecting salmon smolts for genetics (Section 3.2). Additional thanks to Vancouver Aquarium, Active Pass charters, Dr. Doug Hay (Nearshore Research) for support with larval fish research projects (Section 5.5).

Many thanks to the Salmon Watershed Lab at Simon Fraser University and the Skeena Fisheries Technical Committee for providing technical advice on many sections. We also thank Kate Broadley and Matthew Pyper from Fuse Consulting for creating report illustrations, along with Brian Huntington from Skeena Watershed Conservation Coalition and Dave Herasimtschuk from Freshwater Illustrated for contributing professional photos for the report. Many thanks to Laura Grant for the detailed editing and formatting of this report, your thoughtful comments and feedback were invaluable. The authors would also like to thank the following researchers and organizations for their contributions to research summaries for Appendix A: Paul Covert (DFO Oceans Protection Plan), Ryan Flagg (Ocean Networks Canada), Penny White (Metlakatla Fisheries), Jason Scherr (Prince Rupert Port Authority), Angela Addison, and Maya Paul (North Coast-Skeena First Nations Stewardship Society).

This research is part of a collaboration between Lax Kw'alaams First Nation, Skeena Fisheries Commission, and Simon Fraser University. Any opinions, findings, conclusions, or recommendations expressed in this paper are those of the authors and do not necessarily reflect the views of the partner organizations.

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EXECUTIVE SUMMARY

Although estuary habitats support many economic and culturally important fisheries, baseline knowledge of the Skeena River estuary remains limited. Much of the limited research that does exist is not publicly available and/or is difficult to find, making it challenging to establish a common scientific understanding of the ecology of the Skeena River estuary. The goal of this report is to address the aforementioned issue by distributing research within the broader community and providing reference material to support future scientific studies (Appendix A). Additionally, this report synthesizes findings from five years of collaborative scientific research and monitoring efforts conducted by the Lax Kw'alaams Fisheries Program, the Skeena Fisheries Commission, and Simon Fraser University. We answer research questions relating to the estuarine environment (Section 2) and ecology of the Skeena River estuary, with a focus on juvenile salmon (Section 3 and 4). Additional topics include the estuarine fish community (Section 5), crab ecology (Section 5), and the estuarine food web (Section 4). This report summarizes these recent findings in the context of historical studies and identifies major knowledge gaps to be addressed in future research.

Section 2 – Estuarine Environment

The estuarine environment is influenced by both freshwater and marine conditions. Freshwater discharge into the ocean drives environmental gradients of salinity, temperature, and turbidity, which vary according to time of year and location (2.1.4). We identified several trends in the estuary environment over 80 to 90 year time periods from fixed sampling stations proximal to the Skeena River estuary. The magnitude of freshwater discharge during spring months (March – May) has increased overtime (2.1.1), while ocean conditions have become warmer (+0.09°C per decade) and fresher (less saline by 0.10 ppt per decade) (2.1.2) (Figure 0-1). Furthermore, the ocean is warming 2 – 4 times as fast from January – April and August – December (2.1.2). Changing spring discharge patterns and ocean temperatures may affect zooplankton prey for juvenile salmon and larval eulachon that use the estuary during the spring (2.1.3). Short-term patterns and long-term trends in the estuary environment provide information about conditions experienced by estuarine organisms. We also investigated the origins, transport processes (2.2.1), and eelgrass habitat (2.2.2) on Flora Bank, a unique sand bank composed of relic sediment that is likely approximately 8000 years old (2.2.1).

Section 3 and 4 – Juvenile Salmon and Associated Food Web

Estuaries represent a transition zone for young salmon migrating from freshwater to the ocean that provides optimal environmental conditions, high feeding opportunity, and protection from predators. The North Coast Juvenile Salmon Monitoring Program (NCJSMP) is run by Lax Kw'alaams Fisheries Program in collaboration with the Skeena Fisheries Commission and Simon Fraser University. The NCJSMP sampled juvenile salmon from 2013 - 2018 to collect baseline data on the factors contributing to salmon productivity during their young estuary life-stage in the Skeena River estuary. We found that all species of salmon from diverse populations use the Skeena River estuary (3.1), reside and feed for varying durations of time from weeks to months (3.2), and grow (3.3) in the estuary during a vulnerable life history stage (3.4) (Figure 0-1).

Juvenile salmon are a part of an estuary food web that is sustained by upriver and marine nutrients. We documented the food availability of the estuarine environment and determined the food web linkages of several juvenile salmon and forage fish species (4.1). We found that fish feed on estuary resources originating from terrestrial, benthic, plantonic, and larval fish sources. Fish larvae and insects appear to be an important food source for coho salmon, while harpacticoid copepods were the most important prey for juvenile sockeye (4.1). Harpacticoid copepods are associated with eelgrass habitats and are most abundant on Flora Bank, the largest eelgrass bed in the Skeena River (4.2). Understanding the connections between available food, fish diet, and habitat associations of the food web within the Skeena River estuary helps assess estuary use by juvenile salmon.

By answering the following questions, Sections 3 and 4 lay the foundation of understanding how Skeena River salmon use the estuary:

1. When and where are juvenile salmon found in the estuary (3.1)?
2. What populations of juvenile salmon use the estuary (3.1)?
3. What is the diversity of juvenile salmon migration timing and implications for the future (3.1)?
4. What habitats and biophysical metrics explain juvenile salmon use of estuary habitat (3.1)?
5. How long do juvenile salmon grow and reside in the estuary (3.2 and 3.3)?
6. What is the condition of juvenile salmon upon arrival in the estuary (3.4)?
7. What do juvenile salmon eat during this estuary phase (4.1)?

Section 5 – Fish and Shellfish

Estuaries provide critical nursery habitat that supports fisheries of economic and cultural importance including Pacific salmon, herring, eulachon, Dungeness crab, clams, and cockles. We found 34 species of fish, including seven species of salmonids, using the estuary during spring and summer (5.1.1). Pacific herring and surf smelt are the most abundant forage fish species throughout the season, with multiple age classes for both species using the estuary overtime including young of the year and spawning adults (5.1.1 – 5.1.3). Abundance pulses are characteristic of the Skeena River estuary, as pink and sockeye salmon represent the most abundant fish species caught during their annual outmigration (5.1.1), while high densities of larval eulachon are caught in spring, 6 weeks after peak spawning activity is observed in freshwater habitat (5.2) (Figure 0-1). We also documented the Skeena River estuary as year-round habitat for Dungeness crab for foraging, mating, wintering, and egg-brooding (5.3) (Figure 0-1). This section advances our understanding of the use of the estuary by several commercial and culturally important species.

Many knowledge gaps remain surrounding estuarine habitat use and are discussed in Section 6. Given that the Skeena River estuary is one habitat where all Skeena salmon populations transition, continued research assessing the importance of the Skeena River estuary habitat to salmon populations is needed to inform conservation and management. To support future research efforts, this report contains a comprehensive summary of all past and on-going scientific research conducted in the Skeena River estuary up to March 2021 (Appendix A).

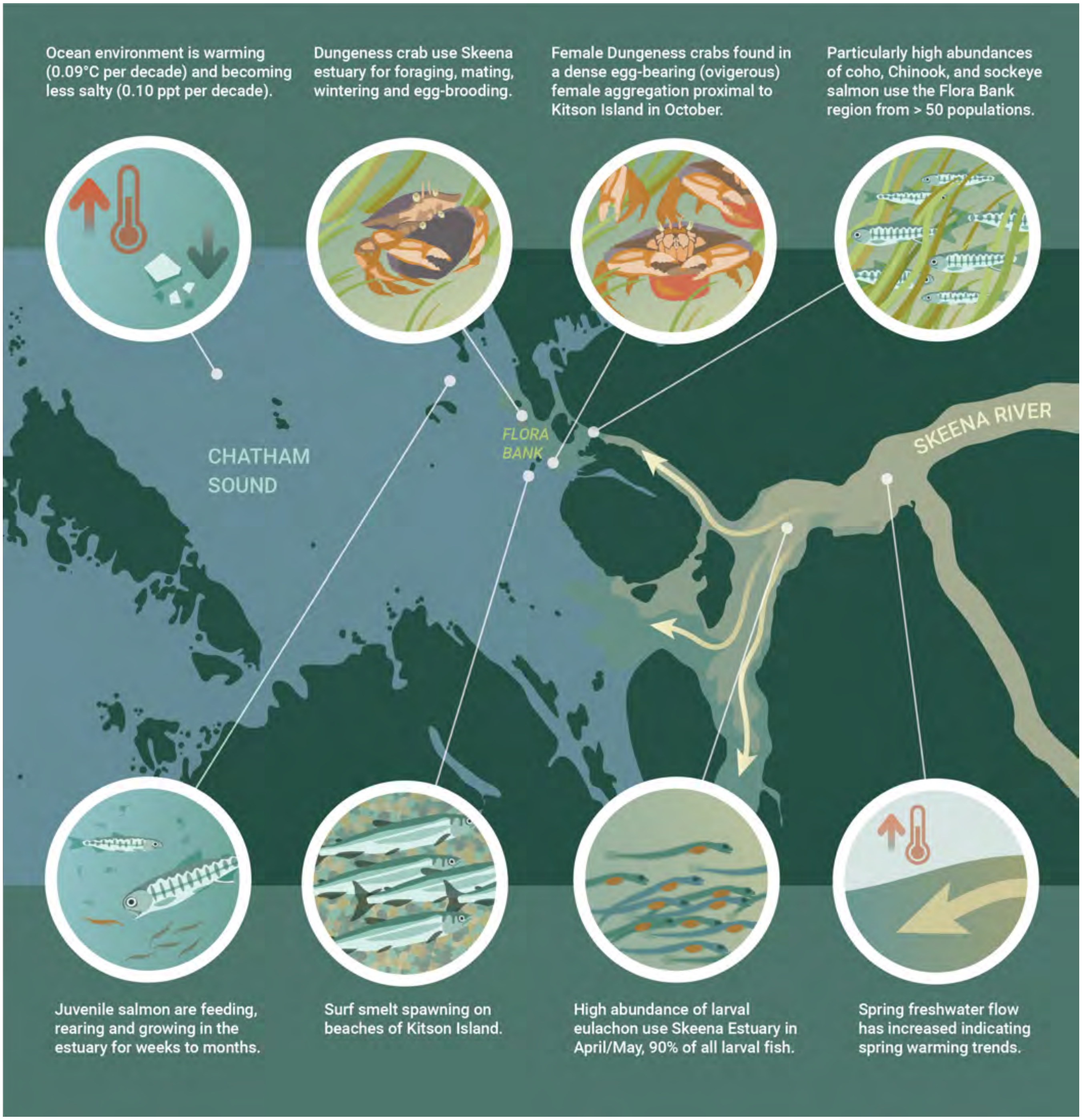

Figure 0-1. Map of the Skeena River estuary illustrating key results from sections of Skeena River Estuary Synthesis Report (illustration by Kate Broadly of Fuse Consulting).

1. INTRODUCTION

1.1. Rationale

Estuaries are productive transition zones between freshwater and marine environments (Simenstad 1983; Orth et al. 2006; Waycott et al. 2009). These ecosystems support many species of fish, birds, and marine mammals (Simenstad et al. 1979) and are commonly referred to as nursery habitats, as they provide a safe environment for the growth and development of many species of young fish (Beck et al. 2001; Dahlgren et al. 2006). In the northeast Pacific, estuaries support many economically and culturally important species including six species of Pacific Salmon (sockeye (*Oncorhynchus nerka*), coho (*O. kisutch*), Chinook (*O. tshawytscha*), pink (*O. gorbuscha*), chum (*O. keta*), steelhead (*O. mykiss*)) and (Pacific herring (*Clupea pallasii*), surf smelt (*Hypomesus pretiosus*), Dungeness crab (*Metacarcinus magister*), and eulachon (*Thaleicthys pacificus*). Pacific salmon and forage fish, such as Pacific herring, sustain indigenous fisheries and cultures (Thornton et al. 2010; DFO 2016; Nesbitt & Moore 2016), along with commercial and recreational fisheries, which significantly contribute to North American economies (Kristianson & Strongitharm 2006; BCMOE 2008; Schindler et al. 2010; DFO 2016).

The Skeena River has a drainage area of 55,000 km^2, supports all species of eastern Pacific Salmon, and is the second-largest salmon-producing watershed in British Columbia, after the Fraser River (Figure 1-1). The wild salmon of the Skeena River form the basis of internationally recognized sport, aboriginal food, and commercial fisheries, contributing $110 million per year to the regional economy (SWCC report 2003). As salmon depend on freshwater, estuarine, and marine habitat to survive at various life history stages, understanding how these ecosystems support salmon can help guide effective decision-making.

The Skeena River meets the ocean near Prince Rupert, British Columbia (54.13°N, -130.10°W) to create the Skeena River estuary (Figure 1-2). Here the freshwater of the Skeena River flows into Chatham Sound, where it meets the freshwater of the Nass River (Figure 1-2). Compared to other major salmon estuaries that have been well studied (e.g. Columbia River and Fraser River estuaries), the estuary of the Skeena River remains understudied despite its large size and economic and culturally importance. In addition, much of the historic research that exists has been performed by proponents in relation to industrial projects and/or may not be publicly available. Table A1 listed in Appendix provides a summary of known research that has been performed in the Skeena River estuary to date. In recent years, there is an emerging body of scientific research on the Skeena River estuary and its ecology. A synthesis of findings from both recent and historic research efforts can contribute to establishing a common scientific understanding of the ecology of the Skeena River estuary.

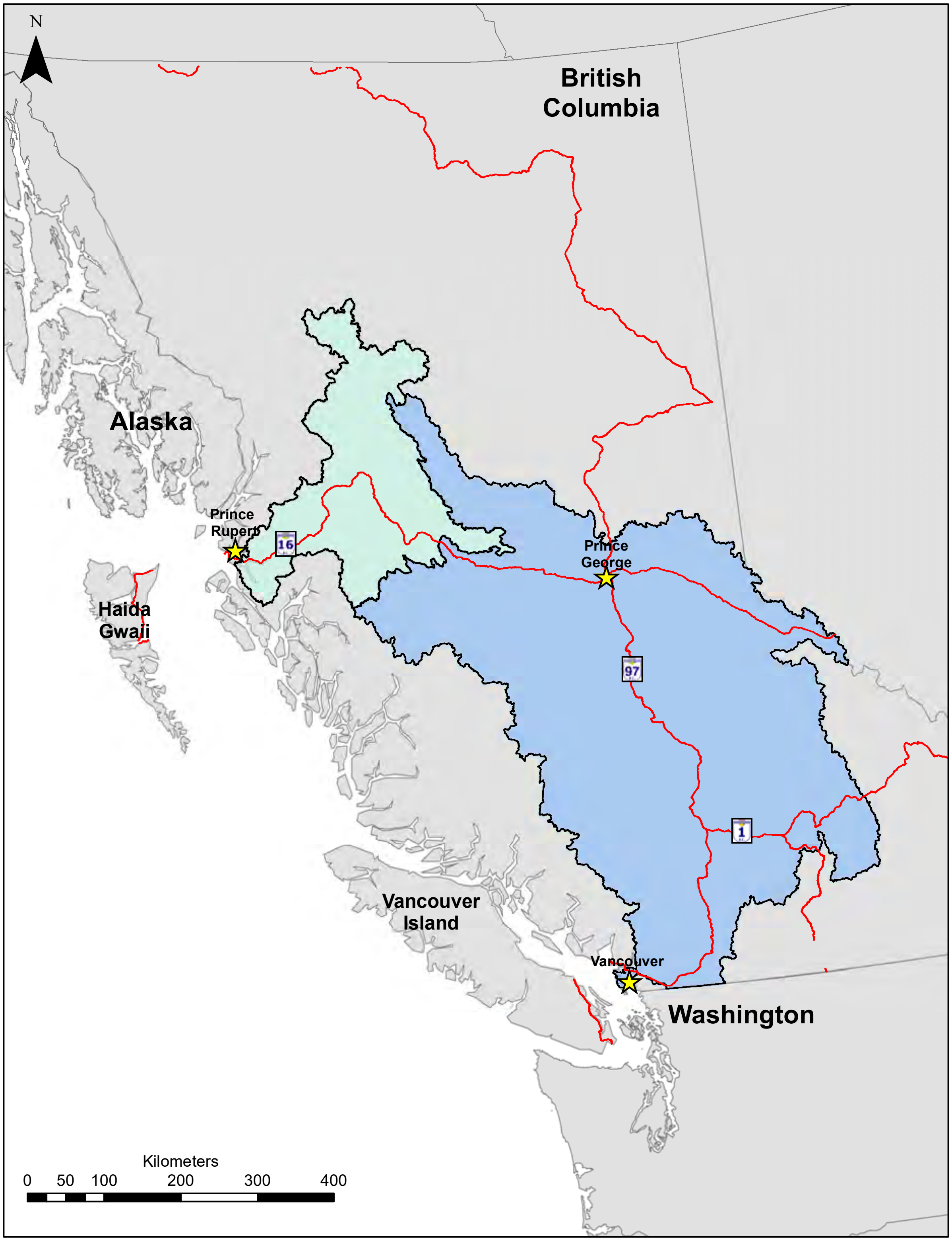

Figure 1-1. Areal extent of the Skeena River (green) and Fraser River (blue) watersheds in British Columbia in relation to major cities and highways in red (map created by John Latimer, Lax Kw'alaams Fisheries).

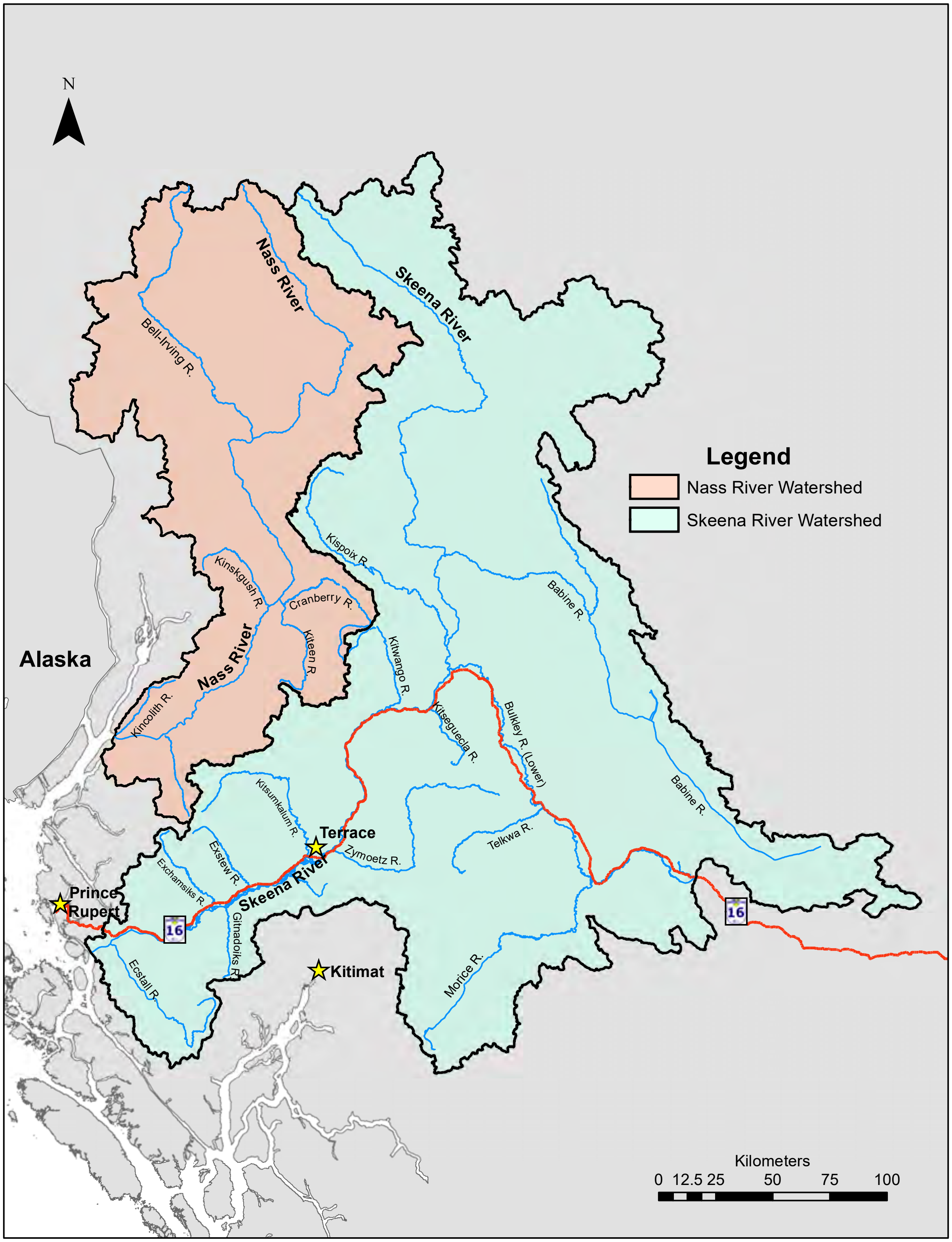

Figure 1-2. Skeena River and Nass River Watersheds with major rivers identified (map created by John Latimer, Lax Kw'alaams Fisheries).

1.2. Goals and Objectives

This report synthesizes the findings of five years of collaborative scientific research and monitoring efforts conducted by the Lax Kw'alaams Fisheries Stewardship Program, the Skeena Fisheries Commission, and Simon Fraser University. The goal of this report is to clearly synthesize research findings and outline historic studies in order to distribute knowledge within the broader community and provide reference material to support future scientific studies. The report is intended for both a scientific and public audience.

The research outlined in this report answers questions relating to the estuarine environment, fish community, crab ecology, and the estuarine food web, with a particular focus on juvenile salmon (see Table 1-1 for specific research objectives addressed). This report summarizes these recent findings in the context of historical studies and identifies major knowledge gaps to be addressed with future research. To support this, this report includes a comprehensive summary table of all past and on-going scientific research conducted in the Skeena River estuary up to March 2021 (Appendix A).

1.3. Partners

The research in this report was conducted in collaboration with the Lax Kw'alaams Fisheries Program, the Skeena Fisheries Commission (SFC), and Simon Fraser University (SFU).

The Lax Kw'alaams Fisheries Program was initiated in 2010 and conducts scientific research to monitor the condition and use of marine resources within the Lax Kw'alaams traditional territory. This program performs year-round research and monitoring on a wide range of marine projects and traditional foods including abalone, clams, cockles, crabs, salmon, and eulachon.

The Skeena Fisheries Commission is an aboriginal organization that focuses on fisheries management, science, and conservation throughout the Skeena River Watershed. The SFC has worked in the Skeena River estuary with the Lax Kw'alaams Fisheries Program from 2012 – 2016, providing technical support for research projects and industrial proposals around Prince Rupert.

Led by Jonathan Moore, the Salmon Watersheds Lab at Simon Fraser University has been working on juvenile salmon use of the Skeena River estuary with partner organizations since 2013. The Salmon Watersheds Lab conducts research on applied aquatic ecology in salmon-bearing watersheds throughout British Columbia including the Fraser, Keogh, Koeye, and Skeena watersheds.

Table 1-1. Description of research objectives addressed in this report.

Report Section	Objective	Project	Project lead(s)
Estuary Environment *Section 2*	Mapping and summary of salinity, temperature, and turbidity landscapes in the estuary across space and time (2.1)	1) Long-term hydrology data from the Skeena River at Usk (08EF001), 1928 – 2018, and Pacific Ocean at Langara Lighthouse, 1936 – 2018 2) North Coast Juvenile Salmon Monitoring Project (NCJSMP), 2013-2016 3) Larval Eulachon Project, 2016 – 2017	Ciara Sharpe, Jonathan Moore
	Flora Bank sediment process and eelgrass habitat mapping (2.2)	1) Sediment trend analysis of Prince Rupert Harbours, 2015 2) Eelgrass Survey of Flora Bank, 2016	Patrick McLaren, Kyla Warren
Juvenile Salmon *Section 3*	Abundance through space and time of different salmon species and populations (3.1)	NCJSMP, 2013 – 2018	Charmaine Carr-Harris, Ciara Sharpe, Jonathan Moore, Michael Arbeider, Sam Wilson
	Stable isotope study of residence patterns (3.2)	NCJSMP, 2013 – 2018	Jonathan Moore, Ciara Sharpe, Charmaine Carr-Harris
	Inferred growth of salmon overtime in the estuary (3.3)	NCJSMP, 2013 – 2018	Jonathan Moore, Ciara Sharpe, Charmaine Carr-Harris
	Energetic status of Skeena sockeye smolts in the estuary (3.4)	NCJSMP, 2013 – 2018	Sam Wilson
Salmon Food Web *Section 4*	The estuary food web study (4.1 – 4.3)	NCJSMP, 2013 – 2018 Fish diet and selectivity study, 2016	Charmaine Carr-Harris, Michael Arbeider, Jonathan Moore, Ciara Sharpe
Fish and Shellfish *Section 5*	Abundance of estuary fish species across time and space (5.1)	NCJSMP, 2013 – 2018	Ciara Sharpe, Charmaine Carr-Harris, Jonathan Moore
	Eulachon and larval fish distribution and speciation (5.2)	Larval Eulachon Project, 2016 – 2017	Charmaine Carr-Harris, Katherine Butts, Ciara Sharpe, Jonathan Moore
	Tracking crab use of the Flora Bank region (5.3)	Dungeness crab telemetry Project, 2016 – 2017	Janvier Doire

2. ESTUARINE ENVIRONMENT

As the place where the river meets the sea, estuaries are dynamic environments that support diverse communities of plants and animals. Estuaries extend from the tidally-influenced portions of rivers that have saltwater influence to the constituent marine bays that have freshwater influence (Perillo 1995). These dynamic environments see daily and seasonal changes in water depth, turbidity, salinity, and temperature, all of which influence the suitability of estuary habitat for fish and invertebrates (Marshall & Elliott 1998; Wagner & Austin 1999; Harrison & Whitfield 2006). Environmental variables, such as salinity and turbidity (Section 2.1 – 2.5), are the primary drivers of spatio-temporal dynamics of estuary dependent species (Jones et al. 1990; Bacheler et al. 2009a). However, biotic factors like the presence of marine vegetation (e.g. eelgrass (Section 2.6 – 2.7) or tidal marshes), can also influence patterns of estuary use for many species, including juvenile salmon (Semmens 2008; Waycott et al. 2009). Thus, understanding the environmental and biotic dynamics of the landscape are key to understanding the ecology of estuaries.

2.1. Abiotic landscape in the estuary across space and time

The tidally influenced portion of the Skeena River extends approximately 55 km upstream to the Kasiks River (~8 km downstream of the Exchamsiks River, labelled on Figure 1-2). At the mouth of the Skeena River (Figure 2-1), the freshwater flows through three channels, extending south to Telegraph Passage and Grenville Channel, and northwest through Inverness Passage into Chatham Sound (Figure 2-2). One quarter of the freshwater flows north through Inverness Passage, with the remainder flowing south and west into the sound (Lin & Fissel 2018). This freshwater extends into Chatham Sound, where it exerts influence on the saline environment for approximately 25 km west and 75 km north (Hoos 1975; Pickard et al. 2015) (Figure 2-2). The environment in Chatham Sound is also influenced by freshwater from the Nass River, which flows into Portland Inlet to the north of the Skeena River estuary (Trites 1956).

The Skeena River estuary is macrotidal (Coleman & Wright 1975; Wild 2020), with large semidiurnal tides of up to 7.5m influencing currents throughout passages and channels (Trites 1952). Currents in the estuary are strong and during ebb tides have been measured up to 2 m/s when combined with river currents (Hoos 1975; Ages 1995). The coastline in Chatham Sound is a fjord type coastline that was covered in ice during the last glaciation (Clague 1984). Outer islands shelter Chatham Sound from the open Pacific Ocean, however, strong winds and storms can generate wave heights of 1 to 3 m (Hoos 1975). The total amount of sediment discharged by the Skeena River into the estuary is estimated to be 21.2 – 25.5 million tons/year (Binda et al. 1986; Wild 2020)**.** Significant sediment plumes emanate through Marcus and Inverness Passages into Chatham sound. The division of the Skeena River into three channels prevents the formation of a large identifiable delta within the bedrock confined estuary (Wild 2020).

The estuarine environment is influenced by both freshwater and marine conditions. This environment can vary across short-term daily or seasonal time frames. The degree of freshwater influence varies throughout the year, with the highest influxes occurring during the spring run-off from

April–May. By mixing with saline water, this freshwater influx creates a gradient of salinity, temperature, and turbidity across the estuary. This is further influenced by semi-diurnal tides in Chatham Sound, with two high and low tides daily that vary in height and timing throughout the year (Trites 1956). In addition, there are global trends related to climate change that may be influencing the estuarine environment long-term. For example, increasing ocean temperatures, changes to salinity, and nutrient cycling have been documented globally (Richardson 2008; Statham 2012).

To fully understand estuary conditions experienced by estuarine organisms it is important to account for both long-term and short-term changes in the local environment. Historical studies on the biophysical landscape of the Skeena River estuary are limited and are summarized in recent research by McLaren (2016), Lin and Fissel (2018), and Wild (2020). Much of the historical data has not been synthesized (Ocean Networks Canada 2017) and much of the ongoing data collection is not publicly available (Port of Prince Rupert 2017). Thus, this report section uses recent fine-scale data to investigate both long-term and short-term trends in the estuary environment. The first sections investigate long-term trends in freshwater discharge (2.1.1) and ocean conditions (2.1.2) from fixed sampling stations with long time series (over 80 years). Results of long-term trends are discussed in relation to the potential impacts on the Skeena River estuary community (2.1.3). The final section explores environmental factors in the Skeena River estuary over a seasonal time scale (2.1.4).

Figure 2-1. View northeast from Smith Island (Marcus Passage) to the mouth of the Skeena River (orange arrow) where it meets the Pacific Ocean on June 27, 2013 (photo by Brian Huntington).

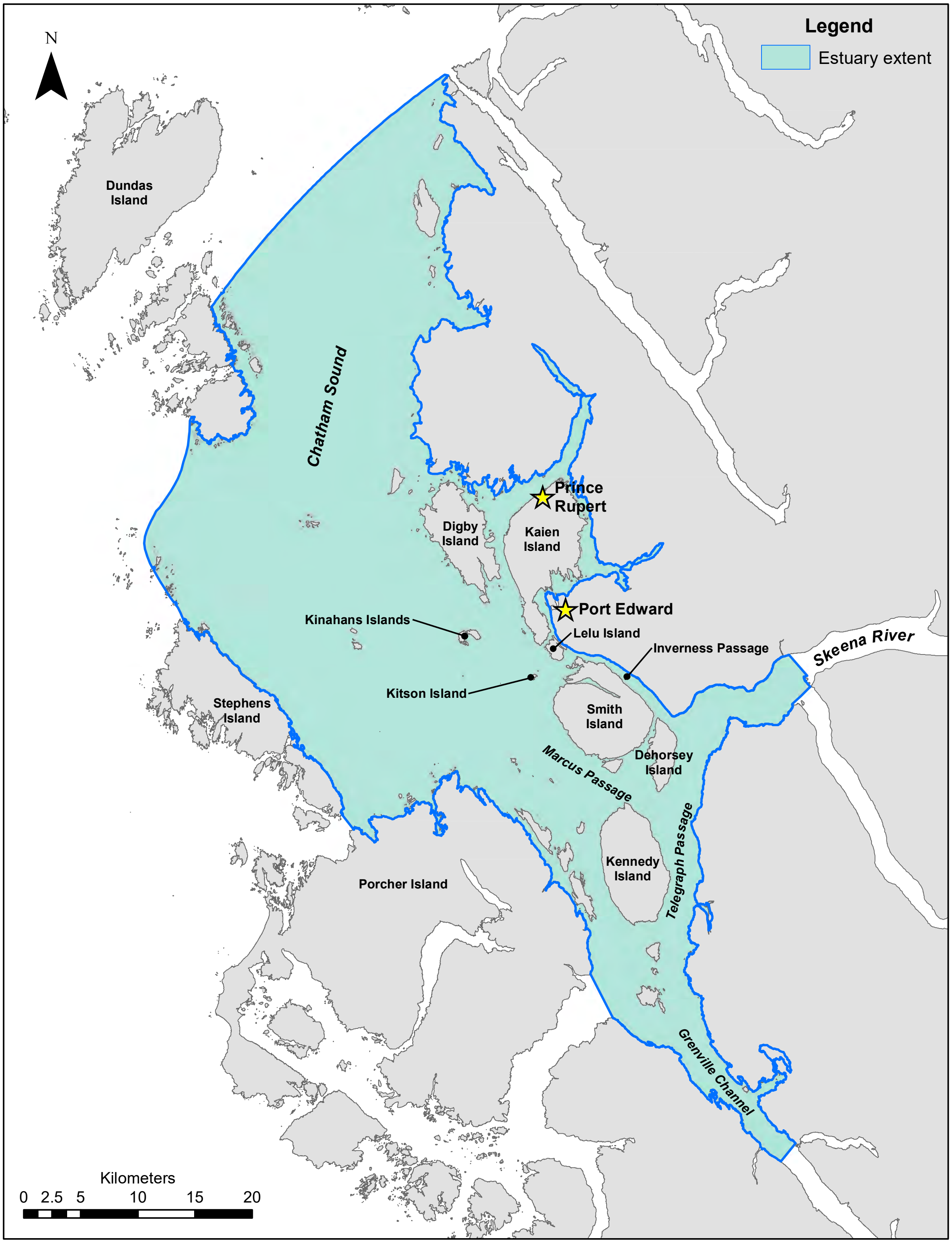

Figure 2-2. Maximum extent of the Skeena River estuary (freshet flow conditions) as defined by the influence of freshwater on salinity measurements, taken from Pickard et al. 2015 (map created by John Latimer, Lax Kw'alaams Fisheries).

2.1.1. Long-term patterns in the estuarine environment: freshwater discharge

INTRODUCTION

Freshwater input in the estuary is influenced by upstream water sources, namely snow and rain patterns across the Skeena watershed (Gottesfeld & Rabnett 2008). The flow of the Skeena River creates a current seaward as it meets the strong tidal current of the estuary (Lin & Fissel 2018). This freshwater regime is responsible for creating the river plume, which extends salinity, turbidity, and temperature gradients across the estuary (Figure 2-3).

Figure 2-3. Aerial photo facing southeast to Lelu Island, Flora Bank, Kitson Island, and Port Edward showing the river plume exiting the north arm (Inverness Passage) of the Skeena River estuary (photo by Brian Huntington).

The flow of the Skeena River is influenced by precipitation (rain and snow, Figure 2-4) regimes which are predicted to be impacted by climate change. This region has become warmer and wetter over the last century, with mean annual precipitation increasing by 5.2% and mean annual temperatures increasing by 0.8°C since 1886 (Foord 2016). Specifically, there have been observed temperature increases of 1.8°C during winter, 0.8°C during summer, and 0.6°C during spring over the last decade. The Skeena region is projected to warm by a further 3.1˚C by 2055, with minimum temperatures increasing more than maximum temperatures (Foord 2016). Although annual precipitation is expected to increase, decreasing snow-type precipitation and increasing temperatures are expected to lead to a decline in snowpack. In the Skeena region, snow-type precipitation is projected to decrease by 35% by 2055 (Foord 2016). Evidence of change caused by increasing temperatures and less snow-type precipitation has been identified in many river watersheds across western North America, including British Columbia (Barnett et al. 2005; Mote et al. 2005; Chezik et al. 2017). These changes are predicted to influence stream flow in the following ways. First, increased temperature is expected to produce earlier springs, and thus earlier periods of spring snow-melt derived streamflow (Regonda et al. 2005; Stewart et al. 2005; Rauscher et al. 2008). Second, as precipitation is less likely to occur as snowfall, watersheds are shifting from snow-dominated systems to rain-dominated systems (Kapnick & Hall 2012; Berghuijs et al. 2014). This may influence the availability and timing of freshwater patterns towards decreased freshwater discharge during spring freshet with more variable discharge patterns from increased rain precipitation (Hirabayashi et al. 2013). Changing freshwater regimes would ultimately influence the timing and distribution of the river plume.

We investigated the hydrology patterns of the Skeena River over time to determine if changes to riverine inputs to the estuary could be detected. We asked the following questions:

Q1 - What are the seasonal patterns of freshwater input into the Skeena River estuary?

Q2 - Has the magnitude and pattern of freshwater input changed over time?

Q3 - Is there evidence of earlier spring peak snowmelts?

Figure 2-4. Viewing west during a helicopter flight towards the Sicintine icefield, a snow-dominated basin in the Skeena River watershed. Snow-dominated basins may be changing to rain-dominated basins overtime (photo by Brian Huntington).

METHODS

We used publicly available discharge measurements from the hydrologic station at Usk (O8EF001) to investigate stream flow patterns from 1928 – 2018. This station is located 18 km north of Terrace (54.513795°N, -128.600619°W) approximately 125 km north of the mouth of the estuary. It is important to note that this hydrologic station is upstream and does not include account for several major coastal tributaries of the Skeena River that contribute substantial discharge, such as the Ecstall River. Discharge was recorded once a day from 1928 – 1971 and hourly from 1971 – 2018.

First, we visually assessed patterns of discharge across time by fitting data with trend lines from linear regression. We investigated changes to the mean and median monthly discharges along with maximum spring discharge values and timing of spring peak discharge. We also visually compared the magnitude of spring and fall peak discharges to look for trends in snow and rain produced discharge peaks. Next, we conducted a generalized least square (GLS) regression analysis to determine if changes over time were statistically significant. This type of regression is commonly used in time-series analysis to account for temporal autocorrelation (Gomi et al. 2006; Fox & Weisberg 2018). GLS regression was performed using the nlme package (Pinheiro et al. 2018) in R (R Core Team 2016).

RESULTS

Q1 - What are the seasonal patterns *of freshwater input into the Skeena River?*

The overall shape of the hydrograph in the Skeena watershed is driven by the combination of rain and snow-dominated basins (Figure 2-5) (Gottesfeld & Rabnett 2008). In the Skeena watershed, mean annual precipitation is 1393 mm, with most precipitation occurring in the fall due to low-pressure systems from the coast (Foord 2016). Snow-dominated basins in the interior have a dominant flood during spring snowmelt in May or June. Coastal drainages receive 2,500 mm of precipitation and have at least one fall or winter flood event. In addition, basins can have a combination of spring and flooding events. Together, the subbasins of the Skeena watershed produce the 'twin peaked' hydrograph shape shown in Figure 2-5, where there are pronounced discharge peaks in both spring and fall. Generally, the highest peak discharge occurs in the spring between May and June, except in 1974, 1978, and 1991 when maximum flood occurred from fall rains in October and November. During high flows in the spring, the freshwater concentration in Chatham Sound is three to four times the average (Trites 1956). Peak flows caused by spring snowmelt (greater than 3000 m^3/s) occur during May–June and the discharge remains high (above 1000 m^3/s) into July before it drops to levels of 800 m^3/s or less in August. Fall rains bring floods leading to another peak discharge period with discharges greater than 1000 m^3/s until mid-November when discharges drop to 300 m^3/s for the winter.

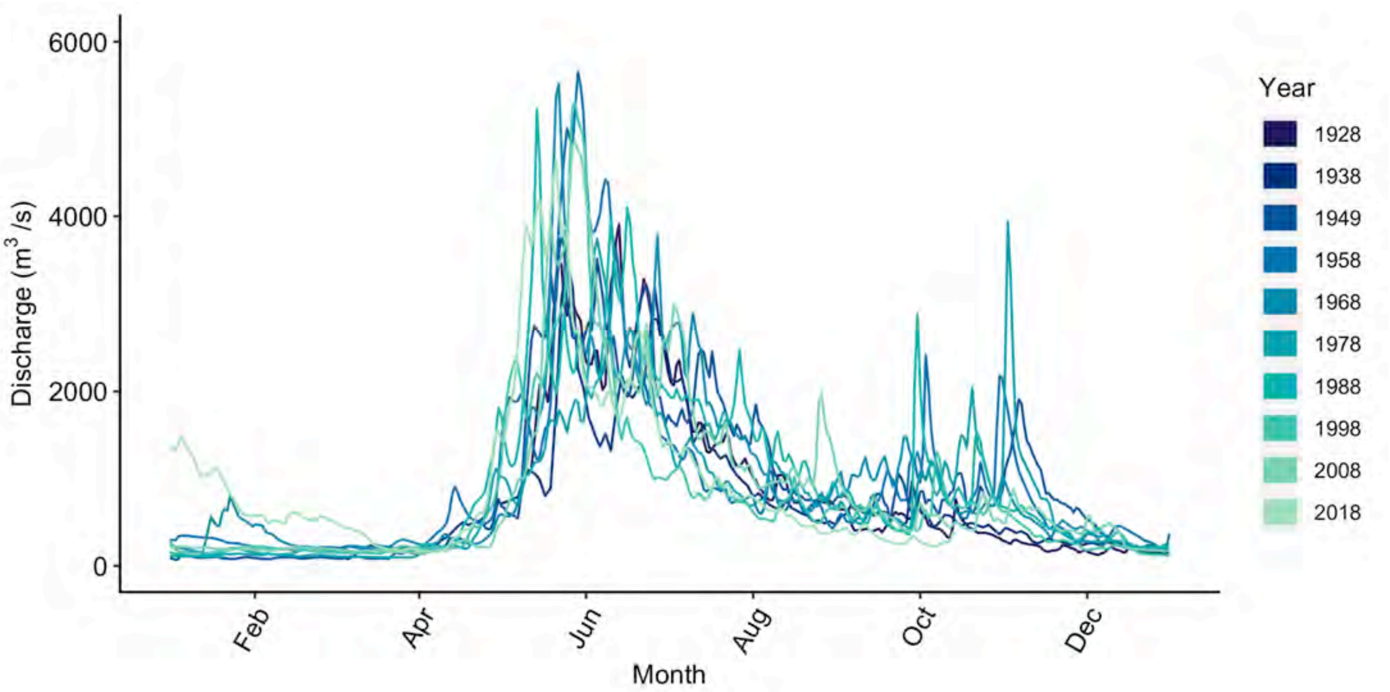

Figure 2-5. Monthly hydrograph discharge readings from Usk station for years: 1928, 1938, 1948, 1958, 1968, 1978, 1988, 1998, 2008, and 2018.

Q2 - Has the magnitude and pattern of freshwater input changed over time?

Evidence of a changing climate on freshwater discharge has been previously documented in the Skeena watershed. Declining flows, increasing water temperature, and diminishing snowpack have been documented for interior parts of the watershed, such as the Babine and Bulkley Rivers (Danard & Murty 1994; Gottesfeld & Rabnett 2008; WWF- Canada 2016). These interior systems are snow-dominated (Gottesfeld & Rabnett 2008) and are expected to be more affected by increasing winter temperatures (1.8°C in the last decade, Foord 2016) and decreasing winter precipitation (Barnett et al. 2005; Berghuijs et al. 2014; Foord 2016). Flow has been documented decreasing a rate of 1.5% per year (Skeena River at Glen Vowell, 1966 to 1989), 0.55% per year (Bulkley River at Quick, 1930 – 2013) and 0.95% per year (Bulkley River at Houston, 1930 – 2013)(Danard & Murty 1994, WWF- Canada 2016). Decreasing trends in January precipitation, and snow water equivalent found in April (0.9% per year, 1966 to 1989) in the Skeena River area identified by Danard and Murty (1994) provide some evidence to suggest drying trends in winter and reduced snowmelt in the spring. These climatic changes may ultimately influence the magnitude and pattern of freshwater input in the Skeena River estuary over time.

There is no evidence that annual discharge at the Usk station has changed overtime, as there have been no statistically significant changes to measures of annual mean, median, and maximum discharge over time (Figure 2-6). However, when looking at monthly trends we found that discharge in the spring months of March, April, and May has significantly increased over time (Figure 2-7, Table 2-1). Besides this increase in mean spring discharge, we found no other indication that overall discharge patterns have changed over time. Figure 2-8 illustrates the hydrograph discharge patterns by decade but no visual pattern for annual mean discharge is discernible. In addition, there are no obvious differences in the magnitudes of the spring or fall peaks in present years compared to historic years (Figure 2-9). This discharge station is the most representative available recording of freshwater input into the estuary and captures water from approximately 42,200 km^2 of the entire 55,000 km^2 Skeena River watershed (Gottesfeld & Rabnett 2008). However, this station does not account for freshwater inputs from many coastal tributaries downstream including the Exchamsiks, Exstew, Ecstall, and Gitnadoix rivers which have large catchment areas and are particularly wet. Although we do not have climate data from these catchments, trends in precipitation in the coastal regions of Terrace and Kitimat indicate a strong trend towards increased spring precipitation and a weak trend towards decreasing fall precipitation (Foord 2019). These freshwater inputs would likely influence the discharge patterns, further adding to trends supporting increased spring discharge.

Shifting away from snow-dominated flows would impact the magnitude of spring snow-melt streamflow, while moving towards a more rain-dominated system would potentially increase the magnitude and variability of fall peak streamflow due to flooding. We did not see evidence of decreased spring discharge or changes to the magnitude of spring/fall peak discharge at the Usk discharge station. However, declining flows supporting this hypotheses have been documented at upriver stations (Gottesfeld & Rabnett 2008; WWF- Canada 2016). This variation in trends observed suggests that any changes to precipitation regimes are not currently detected at this large geographical scale. Increasing spring rain-type precipitation observed within the Skeena region may be masking the effect of decreasing spring snow melt. Foord (2016) demonstrated that precipitation has increased during the spring, but decreased in the winter, summer, and fall seasons within the Skeena region. Specifically, precipitation, likely rain-type, has increased by 28% in the Smithers region and 13.5% in the Terrace region during spring, but decreased by 10.1% (Smithers) and 35.2% (Terrace) during the winter (Foord 2016, 2019). In addition, large river networks such as the Skeena River and the Fraser River (Chezik et al. 2017) can dampen the local long-term hydrological signals of climate change. The freshwater discharge of these larger watershed integrates diverse climate regimes from many subbasins, which can create downstream flow regimes that smooth local changes and dampen flow trends that may be detected at the local upriver scale. Chezik et al. 2017 identified a tenfold decrease in the variability of flow trends as watershed size increased, buffering many trends related to climate change. Our results indicate that trends previously found in upriver subbasins of the Skeena watershed (Gottesfeld & Rabnett 2008; WWF- Canada 2016) are likely dampened in the Skeena river closer to the river mouth. Although it is important to monitor these trends into the future, it appears that the amount and timing of freshwater inputs into the estuary has not changed significantly since 1928.

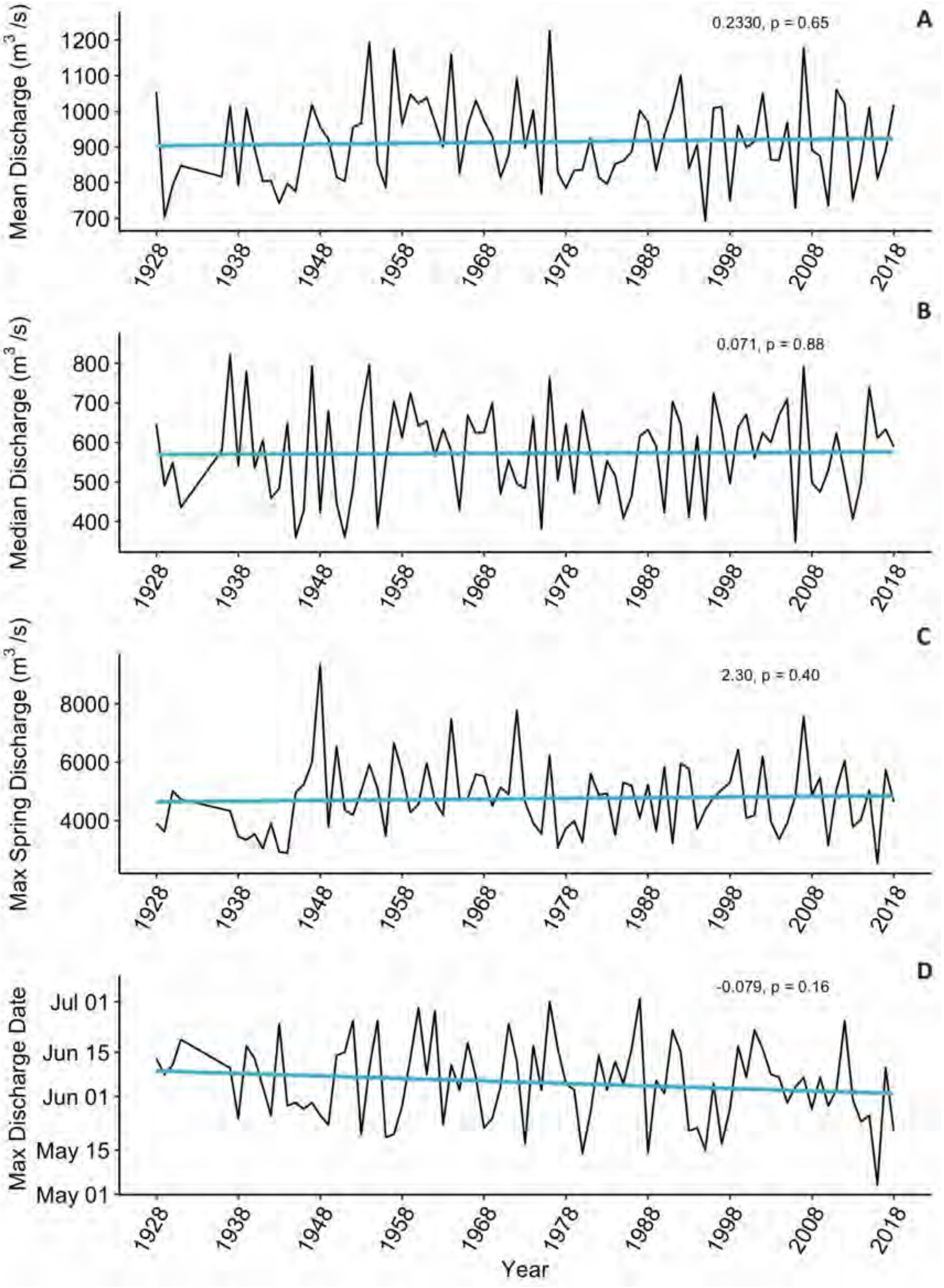

Figure 2-6. Time series of yearly mean (A) and median (B) discharges over time along with maximum spring discharge (C) and date of maximum discharge (D) during spring (prior to July 15). Effect of year on the variable determined from GLS regression and p-values are indicated in each plot (no statistically significant trends ($p > 0.05$)).

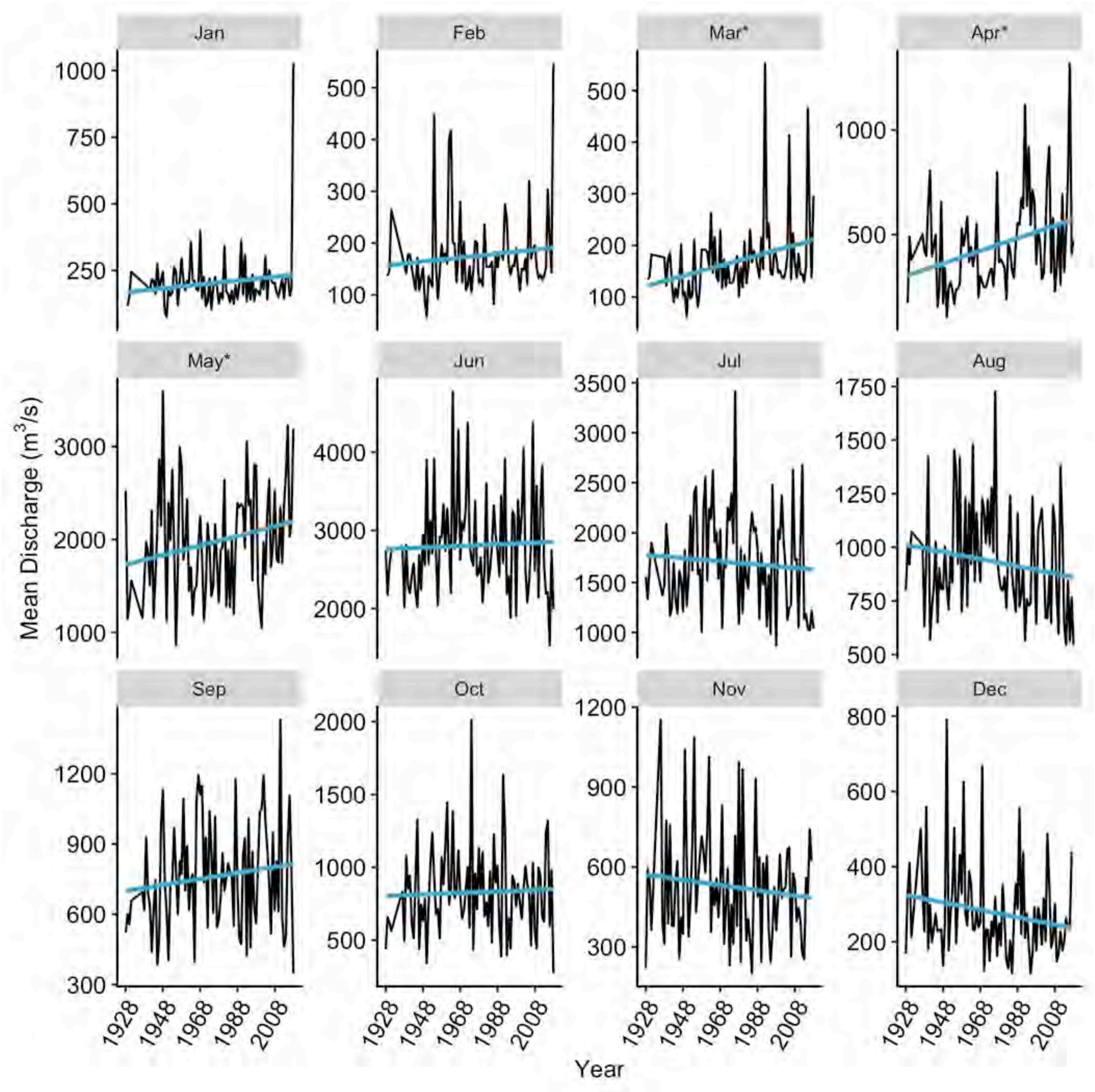

Figure 2-7. Mean discharge by month between 1928–2018 in the Skeena River estuary. * indicates a statistically significant trend ($p < 0.05$) from GLS regression.

Table 2-1. Results of GLS regression trend analyses of mean discharge (m3/s) by month with 95% confidence intervals and p-values. Trends are statistically significant at a p <0.05.

Month	Coefficient Estimate		95% CI	p-value	Trend
January	0.733	±	0.984	0.148	none
February	0.380	±	0.715	0.301	none
March	0.995	±	0.613	0.002	increasing
April	3.010	±	1.766	0.001	increasing
May	5.154	±	4.720	0.035	increasing
June	1.031	±	5.359	0.707	none
July	-1.629	±	4.126	0.441	none
August	-1.667	±	2.073	0.119	none
September	1.282	±	1.906	0.191	none
October	0.496	±	2.507	0.699	none
November	-0.987	±	1.776	0.279	none
December	-0.985	±	1.071	0.075	none

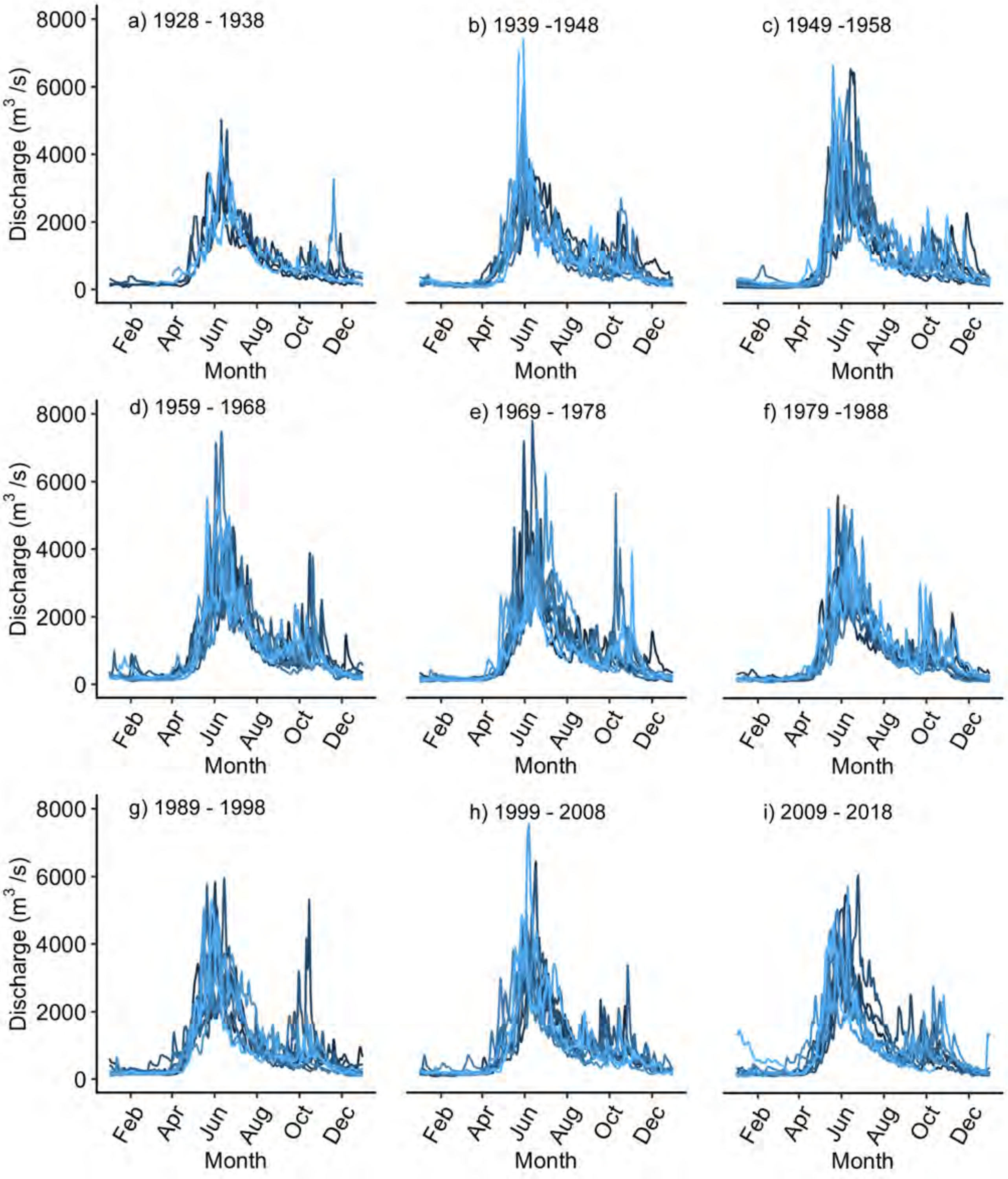

Figure 2-8. Freshwater discharge (Usk Sation 08EF001) for each year grouped into panels by decade. Darker blue colors represent earlier years within each decade, while lighter blue represent more recent years within the time period.

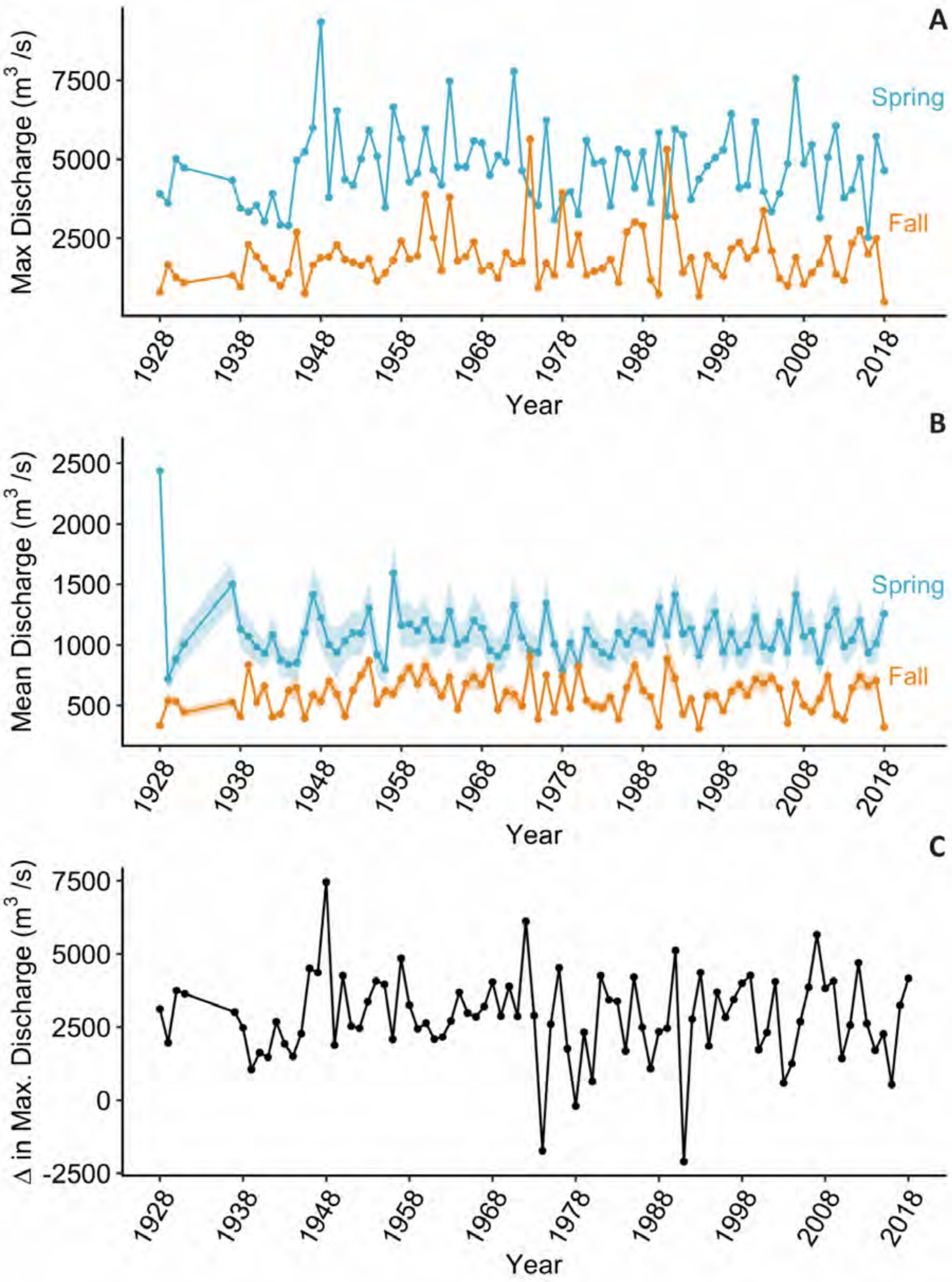

Figure 2-9. Maximum (A) and mean (B) yearly discharge values during spring and fall peak discharge between 1928–2018. 95% confidence intervals are illustrated as transparent ribbons for mean discharge (B). Panel C illustrates the difference between yearly maximum discharge values for fall and spring.

Q3 - Is there evidence of earlier spring peak snowmelts?

Changing climates are advancing the dates of spring to occur around 1-4 weeks earlier across western North America (Stewart et al. 2005), including British Columbia (Rodenhuis et al. 2007). Earlier springs influence the freshwater flow regime by causing earlier peak snow-melt periods (Burn 1994; Mote et al. 2005). Within the Skeena region, warmer spring and winter temperatures over the last decade (1.8°C and 0.8°C, respectively (Foord 2016, 2019)) may be linked to trends in earlier onset of spring snow-melt and decreasing water availability (less snow melt) in summer and fall (Burn 1994; Stewart et al. 2005; Kapnick & Hall 2012).

To investigate if spring snowmelts are happening earlier in the Skeena watershed, we used the date of maximum discharge during spring snowmelt for each year from 1928–2018 (Burn 1994). When looking at yearly trends, we did not find any significant trends to suggest that spring peak discharge was occurring earlier or changing in magnitude over time (Figure 2-6d). However, when looking at monthly trends, (Figure 2-7, Table 2-1) we found that discharge in March, April, and May was significantly increasing over time. This increasing trend for March–May months has been found in other watersheds across Canada and across North America (Burn 1994; Stewart et al. 2005; Kapnick & Hall 2012) and may be indicative that springs may be occurring earlier. Changes to snow-melt patterns are coupled with increasing spring rain-type precipitation observed within the Skeena region during spring months (Foord 2016, 2019). Thus, it is unclear if earlier snow-melt periods or increased spring precipitation is driving trends in increasing freshwater discharge during spring months.

Section 2.1.1. - Key Findings

1) The magnitude and pattern of annual freshwater inputs into the Skeena River estuary has not systematically changed over the last 90 years.

2) Freshwater discharge during spring months (March–May) has increased over the last 90 years.

2.1.2. Long-term patterns in the estuarine environment: ocean temperatures

INTRODUCTION

Consistent with increasing air temperatures, oceans temperatures are rising with climate change, with sea surface temperature rising an average rate of 0.11°C per decade (range of 0.09 to 0.13°C) (Ren & Riser 2010). In addition, sea surface salinities have changed over time, with the Atlantic becoming saltier and the Pacific and Southern Oceans becoming less saline (Ren & Riser 2010; Rhein et al. 2013). As estuaries are open to the ocean, these changes may ultimately influence temperature and salinity regimes in estuaries (Seekell & Pace 2011; Oczkowski et al. 2015). Warming temperatures in estuaries have been shown to impact abundance and composition of aquatic vegetation (Oviatt 2004), molluscs (Oviatt 2004), and fish (Seekell & Pace 2011; Oczkowski et al. 2015), and shift the timing of phytoplankton blooms and predation by zooplankton in estuaries (Oviatt 2004; Worden & Wilken 2016). Although there is undisputed evidence that oceans are warming, these trends are complex and vary according to location as well as seasonal, interannual, decadal, and centennial time scales (Rhein et al. 2013). In this section, we investigate if ocean temperature and salinity proximal to the Skeena River estuary are changing over time, in the absence of larger and more complex climate trends. Specifically, we ask: Is there evidence that the Pacific Ocean proximal to the Skeena River estuary is warming and becoming less saline?

METHODS

We used publicly available sea surface temperature (SST) and salinity (SSS) measurements from the Langara Point Lighthouse located on the north-western tip of Haida Gwaii (54.25528°N, -133.0594°W) to examine trends over time. This station is located 170 km northeast of the Skeena River mouth and represents the most proximal station with measurements from the longest time period (1928–2018). Data from Bonilla Island Lighthouse (77 km southwest of the river mouth) has SST and SSS data from a narrower time period (1960–2018) and has near identical recorded measurements.

As with the freshwater section above, we visually assessed patterns of SST and SSS across time fit with trend lines from linear regression. We investigated mean values along with the difference from the mean value for each month during 1936–2018 to determine if SST and SSS were changing over time. Next, we conducted a GLS regression time-series analysis to determine if changes over time were statistically significant (Fox & Weisberg 2018).

RESULTS

Q1 - Are the ocean conditions proximal to the Skeena River estuary becoming warmer and less saline?

We found evidence that the ocean environment proximal to the Skeena River estuary is warming over time. Using yearly mean, we found that the Pacific Ocean at Langara Point warmed 0.09°C per decade (Figure 2-10, 0.0089°C ± 0.005°C per year, 95% CI). This is on the lower end of global sea surface warming trends, with Rein et al. 2013 illustrating an increase of 0.09°C to 0.13°C per decade in the upper 75m of the water column globally. Figure 2-10 illustrates that temperatures have been warming since 1900. Overall, the Pacific Ocean near the Skeena estuary has warmed approximately 1°C over the last 80 years.

This warming trend was particularly pronounced in the winter, early spring, and late summer months (January–April, August, September) (Figure 2-11, Table 2-2). The temperature patterns observed at Langara Point on Haida Gwaii are consistent with those found at other stations across British Columbia (Chandler et al. 2017). Information on fine-scale trends in SST are related to Climate Indices (PDO, El Nino, etc) and are summarized in Chandler et al. (2017).

Studies have documented that the North Pacific Ocean has been freshening starting between 1984 and the mid 1990s (Riser et al. 2008; Ren & Riser 2010). Similarly, we found that SSS at Langara Point is becoming significantly less saline over time at a rate of 0.10 ppt per decade (Figure 2-12, -0.096 ppt ± 0.002 ppt per year, 95% CI), generating salinities that are consistently less saline than the long-term salinity average starting in 1984–1990. These trends were significant for all months during the time period from 1936–2018 (Figure 2-13, Table 2-2). Decreasing surface salinities in the North Pacific may be occurring due to an increase in riverine discharge or net precipitation over time, although other factors such as changing wind and ocean circulation patterns may also be effecting salinities in the Pacific (Nakano et al. 2007; Ren & Riser 2010). The impacts of decreasing salinities are complex, but have been shown to influence ocean stratification, carbon dioxide absorption, and ocean ventilation.

Section 2.1.2. - Key Findings

1) Ocean conditions proximal (Haida Gwaii) to the Skeena River estuary have become warmer and fresher since 1936.

2) Ocean warming is 2 – 4 times as fast during January–April and August–December.

3) Overall, the ocean is warming at a rate of 0.09°C per decade and becoming less saline by 0.10 ppt per decade.

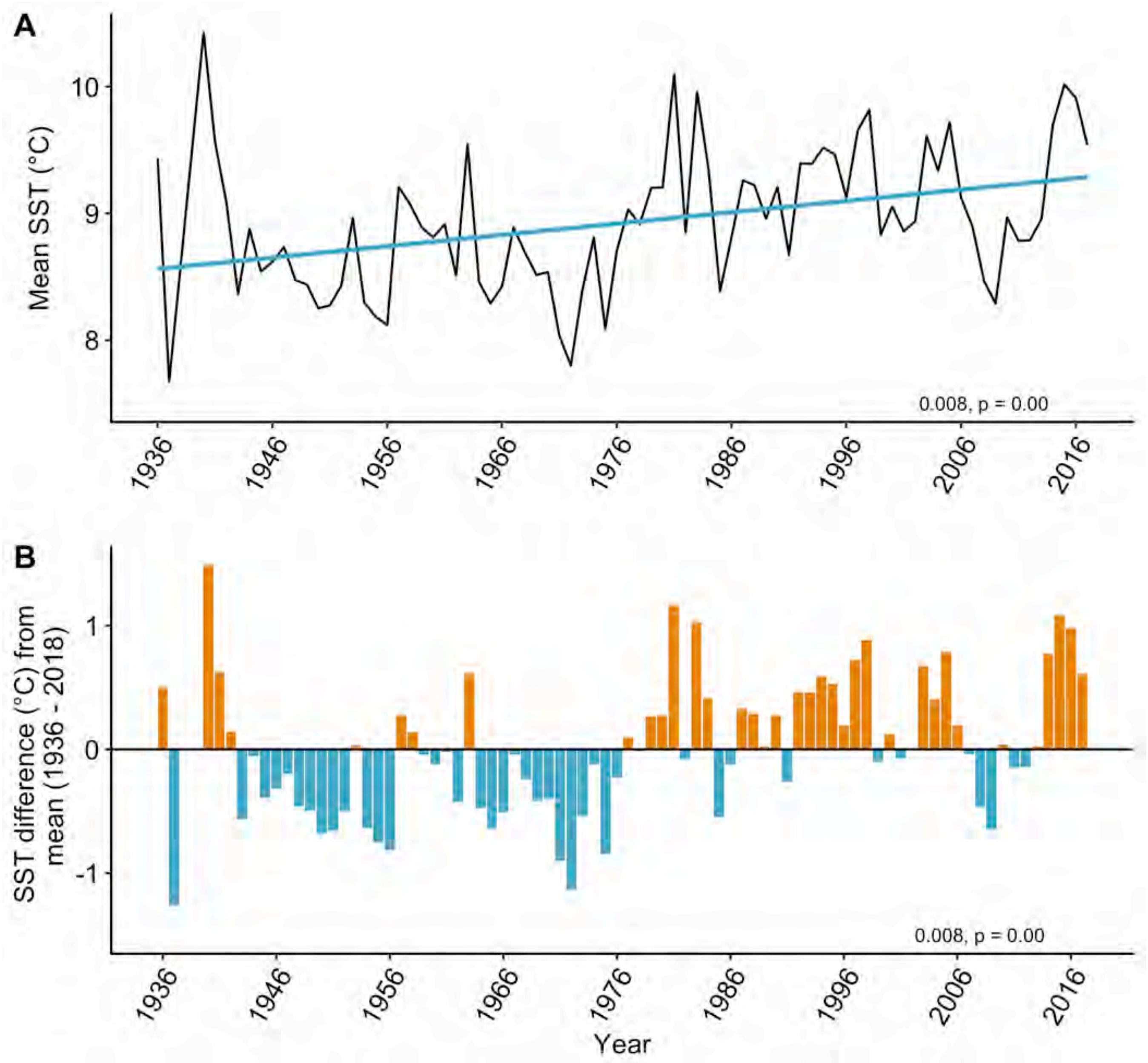

Figure 2-10. Mean SST (Sea Surface Temperature) with trend line (A) and the difference from the mean SST for each month during 1936–2018 (B) at Langara Point Lighthouse on Haida Gwaii. Blue bars in plot B indicate when the SST was colder than the mean SST from 1936–2018 and yellow bars indicate when SST was warmer than the mean SST. Effect of year on temperature and p-value determined from GLS regression are listed and indicate that trends are statistically significant ($p < 0.05$).

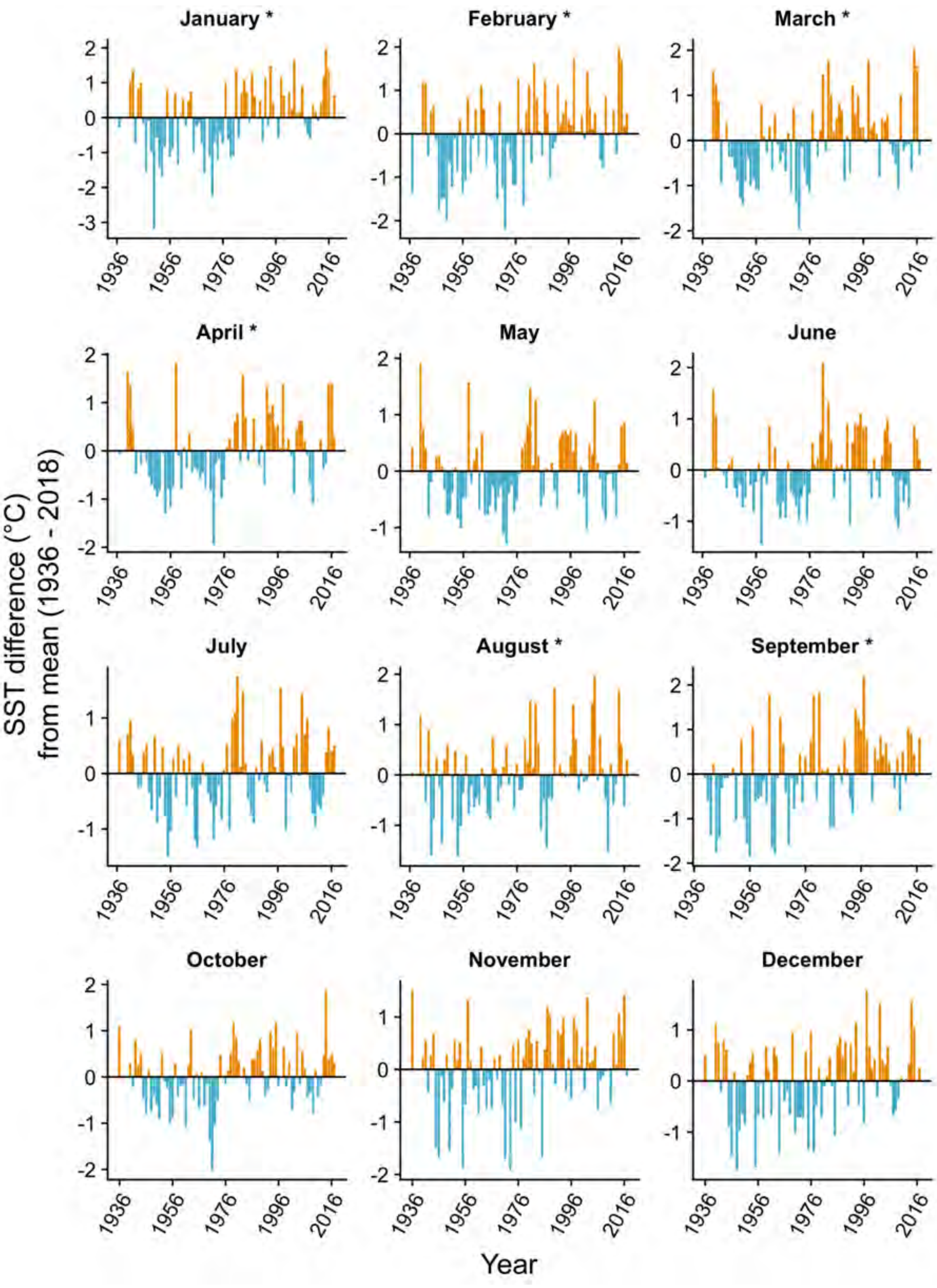

Figure 2-11. Difference in Sea Surface Temperature (SST) from the mean SST by month between 1936–2018 at Langara Point Lighthouse on Haida Gwaii. Blue bars indicate when the SST was colder than the mean SST from 1936–2018 and yellow bars indicate when SST was warmer than the mean SST. * indicates months with statistically significant trends ($p < 0.05$) from GLS regression (results listed in Table 2.2)

Table 2-2. Results of GLS regression of monthly mean sea surface temperature (SST) and sea surface salinity (SSS) with 95% confidence intervals (CI) from Langara Point Lighthouse on Haida Gwaii. Highlighted trends are statistically significant at a p-value <0.05.

	Temperature				Salinity			
Month	Est		95% CI	p-value	Est		95% CI	p-value
January	0.016	±	0.009	0.001	-0.008	±	-0.016	0.000
February	0.015	±	0.009	0.001	-0.01	±	-0.019	0.000
March	0.01	±	0.008	0.020	-0.011	±	-0.021	0.000
April	0.008	±	0.007	0.031	-0.011	±	-0.022	0.000
May	0.002	±	0.006	0.461	-0.012	±	-0.023	0.000
June	0.004	±	0.007	0.267	-0.012	±	-0.024	0.000
July	0.004	±	0.007	0.306	-0.011	±	-0.021	0.000
August	0.008	±	0.007	0.038	-0.011	±	-0.022	0.000
September	0.015	±	0.008	0.000	-0.009	±	-0.017	0.000
October	0.006	±	0.006	0.051	-0.008	±	-0.017	0.000
November	0.008	±	0.008	0.038	-0.008	±	-0.016	0.000
December	0.009	±	0.008	0.027	-0.007	±	-0.014	0.000

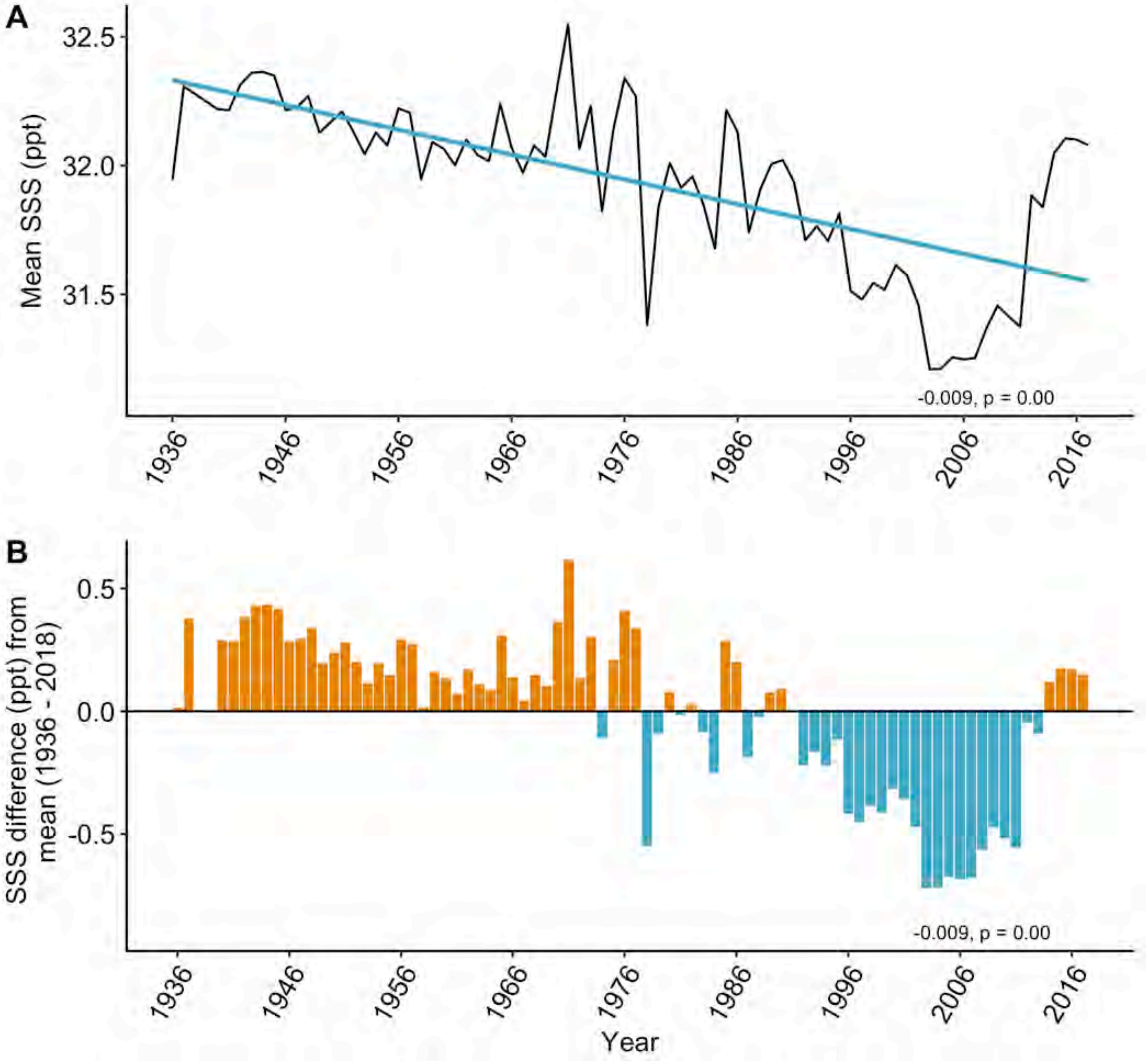

Figure 2-12. Mean SSS (Sea Surface Salinity) with trend line (A) and the difference from the mean SSS for each month during 1936–2018 (B) at Langara Point Lighthouse on Haida Gwaii. Blue bars in plot B indicate when the SSS was less saline than the mean SSS from 1936–2018 and yellow bars indicate when SSS was more saline than the mean SSS. Tau value and p-value from GLS regression analysis are listed and indicate that trends are statistically significant ($p < 0.05$).

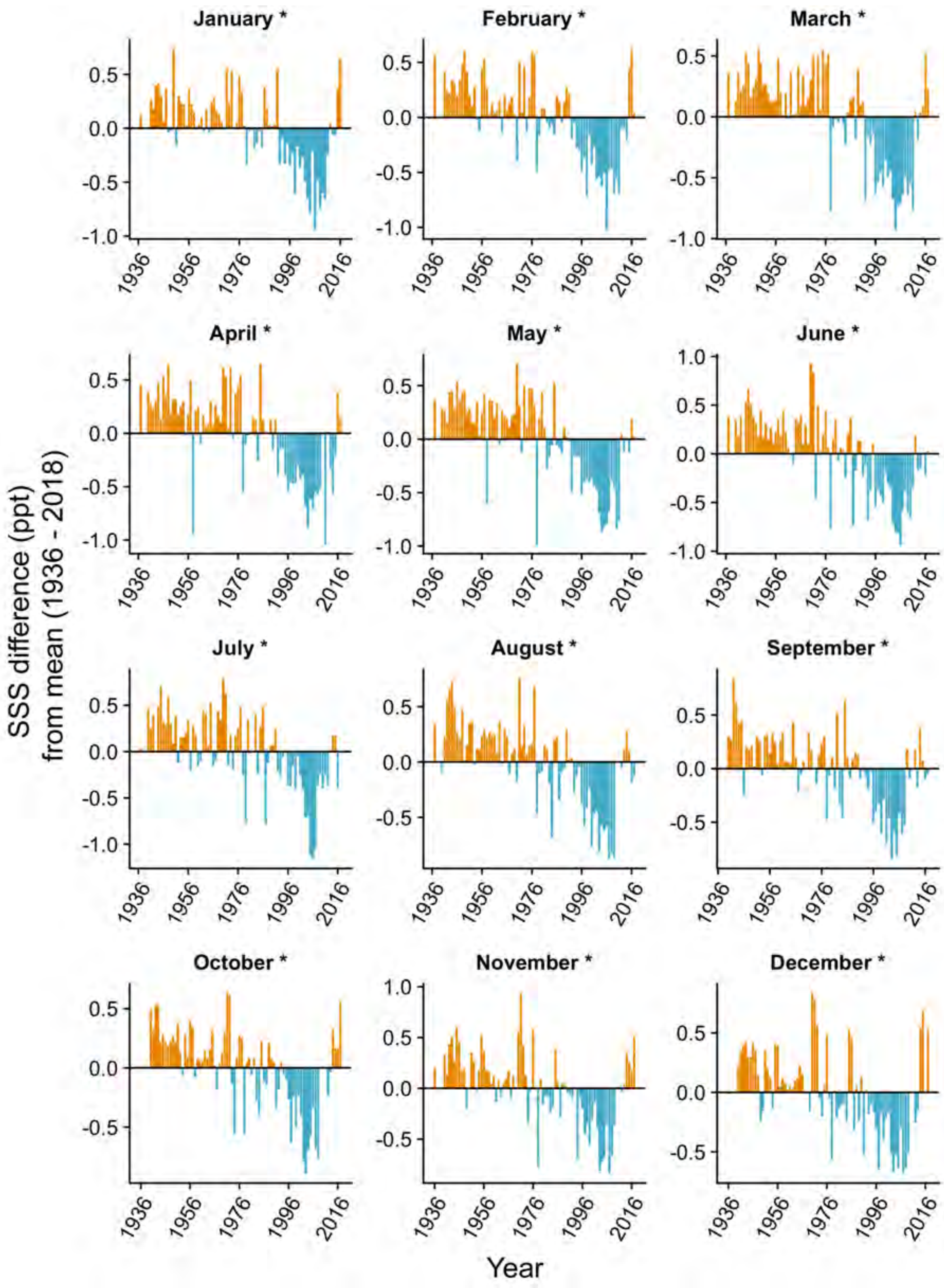

Figure 2-13. Difference in Sea Surface Salinity (SSS) from the mean SSS by month during 1936–2018 at Langara Point Lighthouse on Haida Gwaii. Blue bars indicate when the SSS was less saline than the mean SSS from 1936–2018 and yellow bars indicate when SSS was more saline the mean SSS. * indicates months with statistically significant trends ($p < 0.05$) from the GLS regression analysis (results listed in Table 2.2).

2.1.3. Potential implications of long-term changes on the Skeena River estuary

We found evidence that freshwater input patterns from the Skeena Watershed and local oceanic conditions have changed over time. Specifically, we found that springs are becoming warmer and earlier in the Skeena River watershed, advancing the timing of peak spring discharge periods. Local ocean conditions are becoming warmer and fresher over time. Evidence of warmer (ocean temperatures) and earlier springs (spring discharge timing) in the Skeena region are influencing the estuarine environment during spring (March–May). These changes will undoubtedly alter the estuarine environment and potentially influence estuarine community dynamics in a number of ways.

Changing spring conditions are particularly important for species such as juvenile salmon and eulachon, which use the estuary for a critical part of their life histories. Species from lower trophic levels, such as phytoplankton and zooplankton, that are preyed upon by salmon and eulachon are sensitive to changing environmental conditions in the estuary. For example, freshwater flow regimes are linked to the dynamics of spring phytoplankton blooms. The salinity, water column stratification, and euphotic depth of estuaries are influenced by freshwater discharge and contribute to the timing and productivity of spring phytoplankton blooms. Depending on the nutrient dynamics and circulation of estuaries, spring discharge can have positive (Malone et al. 1988; Cloern 1991; Yin et al. 1997), inverse (Cloern et al. 1982), and variable (Etherington et al. 2007) relationships to spring phytoplankton blooms. In addition, zooplankton blooms are advancing with increasing ocean temperatures (Richardson 2008; Poloczanska et al. 2013) and are becoming more variable overtime (Ji et al. 2010). Given the complex nature of these relationships, it is difficult to determine the effect of earlier spring discharge on phytoplankton blooms in the Skeena River estuary without further study. However, the timing of juvenile salmon migrations, which heavily feed on zooplankton populations in estuaries may not be changing at the same rate as their primary prey (Carr-Harris et al. 2018). As juvenile salmon survival is higher when migration timing aligns with peak zooplankton abundance (Chittenden et al. 2010; Malick et al. 2015), a de-coupling of the zooplankton blooms and salmon migration over time could influence the status of salmon populations.

Eulachon are also associated with spring freshets, with populations predominately accompanying snow-dominated or glacier-fed rivers (Hay et al. 2002; Schweigert et al. 2012). Skeena eulachon spawn in tidal regions of the Skeena River estuary (lower Skeena River) and are currently listed as a population of Special Concern (COSEWIC 2013). Generally, eulachon populations return to spawn the first week of March, but Skeena eulachon populations have been returning as early as mid-February (COSEWIC 2013). These earlier runs have been reported across British Columbia with populations in Bella Coola, the Columbia, Kemano, and Copper rivers and are related to milder springs with earlier spring freshets (Moody 2000). As the migration of larval eulachon has been linked to the timing of spring freshet (Hay et al. 2002; Sharma & Graves 2017), there is concern that earlier spring freshet timing may lead to mismatch with marine upwelling nutrients in the eastern Pacific (Gustafson et al. 2012; Schweigert et al. 2012; Sharma & Graves 2017).

We found local evidence that the Pacific Ocean conditions proximal to the Skeena River estuary have become warmer and less saline since 1936**.** This may influence estuarine communities in the Skeena River estuary in a number of ways. Temperature patterns control the distribution and phenology of zooplankton. For example, warmer ocean conditions linked to a warm water anomaly (“the blob”) from 2014–2016 caused changes in available zooplankton biomass and species composition (Gómez-Ocampo et al. 2018; McKinstry & Campbell 2018; Yang et al. 2018). Changes to available prey is thought to be a driving factor in significant declines for several species of forage fish in the Gulf of Alaska (Osmeridae and *Ammodytes hexapterus*) (Daly et al. 2017; Cornwall 2019). During this warm period, higher densities of less nutritious (low-lipid) zooplankton species were documented (Chandler et al. 2017; Brodeur et al. 2018), which is concerning given the projected warming of sea temperatures in the future.

Gradients of salinity also structure estuarine communities and thus, changes in salinity have the potential to influence estuarine communities. For example, freshwater inflow patterns have been correlated with species composition in low-salinity habitats, while large-scale climate indices, such as the Pacific Decadal Oscillation (PDO), have been correlated to the species occupying high-salinity habitats (Feyrer et al. 2015). As a result, changes in freshwater inputs and marine conditions influencing estuary salinities can lead to a shift in estuary community dynamics.

Changes to salinity and temperature are expected to influence feeding, respiration, growth, and reproduction of bivalve species (Matozzo et al. 2013; Velez et al. 2016), phytoplankton biomass (Cloern et al. 1982), zooplankton distribution, and timing (Richardson 2008) and spawning conditions for nearshore spawners, such as longfin smelt (*Spirinchus thaleichthys*) (Jassby et al. 1995; Kimmerer 2002). Additionally, warming marine conditions has resulted in a change in species’ distributions, with species expanding their distributions north in response to warmer water (Auth et al. 2018; Duguid et al. 2019).

Although it is unclear how the ecology of the Skeena River estuary will be impacted by warming ocean temperatures, decreasing ocean salinity, and earlier spring discharges, continued research and monitoring will allow for the detection of impacts to estuarine communities. Understanding these impacts will aid management and conservation efforts.

Section 2.1.3. - Key Findings

1) Earlier spring discharge and warmer ocean temperatures may impact zooplankton prey for juvenile salmon and larval eulachon that use estuary habitat in the spring.

2) Changes in temperature and salinity regimes in the estuary will influence the estuarine community structure and species distribution.

2.1.4. Seasonal patterns in the estuary environment

INTRODUCTION

Estuary environments change on a seasonal and daily basis in response to marine (tides, currents) and freshwater (freshwater discharge) conditions. These factors combine to create an environment that changes seasonally with a higher magnitude than marine environments. Figure 2-14 illustrates the seasonal fluctuations of temperature and salinity in the Skeena River estuary compared to the marine environment. Estuary environments are defined by gradients of salinity, turbidity, and temperature, which in turn structure estuary communities with preferences for different environmental regimes (Jones et al. 1990; Jassby et al. 1995; Baltz & Jones 2003; Bacheler et al. 2009a). Given the importance of these environmental variables, understanding their dynamics is fundamental to studying estuaries. In this section we investigated abiotic conditions in the Skeena River estuary across time and space. Here we specifically ask: What are the salinity, temperature, and turbidity conditions in the Skeena River estuary? How do they change over time, with water depth, and across regions in the estuary?

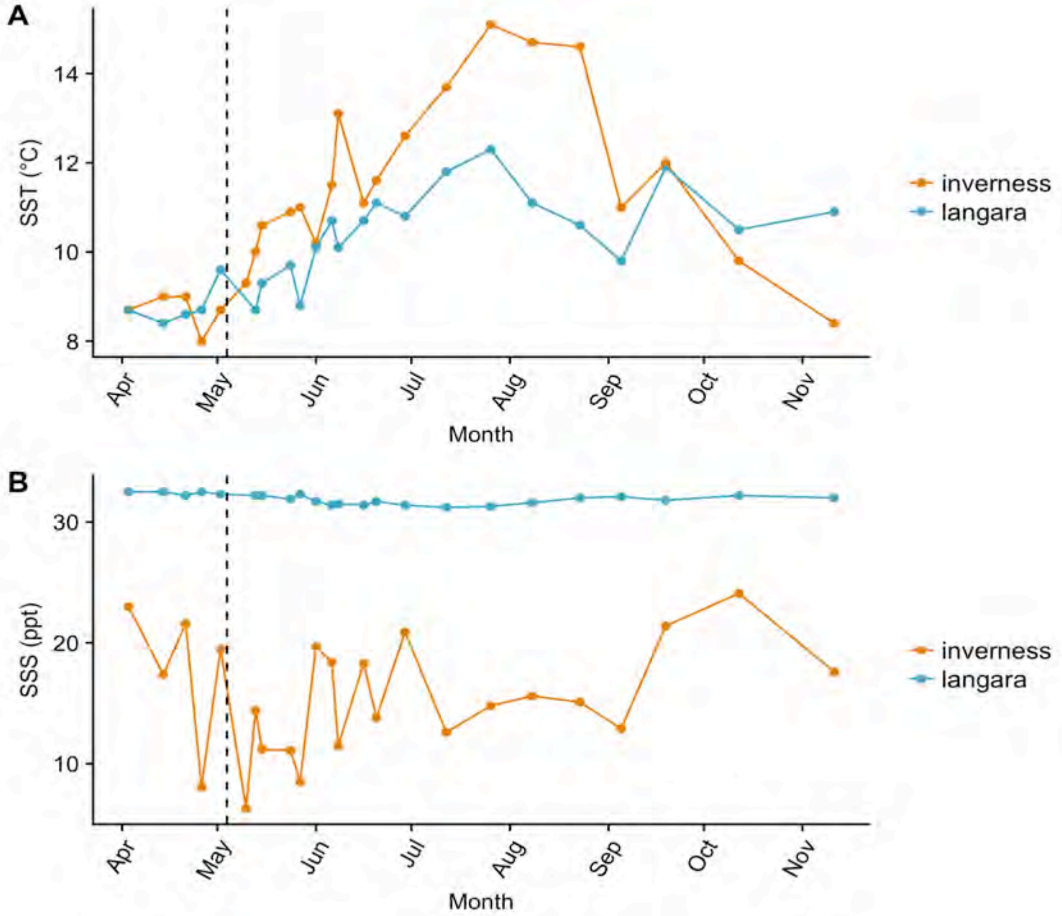

Figure 2-14. Sea Surface temperature (A) and salinity (B) during 2016 from Inverness Passage in the Skeena River estuary and from Langara Point Lighthouse station (Haida Gwaii). The dashed vertical line indicates the day of maximum discharge in 2016 (May 4).

METHODS

Measurements of water quality were collected as part of the NCJSMP and other targeted research projects. YSI profiles (YSI Pro2030) of temperature and salinity were collected at 21 sites across the estuary from 2013–2017, with measurements taken every 1 m from the surface to a maximum depth of 21 m (less at shallower sites). Turbidity was measured with a Secchi disk depth to the nearest 0.1 m.

As measurements were restricted to the operating schedules of the projects, the spatial and temporal extent of this analysis was limited. Months April–July have the highest density of measurements as they were collected during juvenile salmon monitoring projects. Measurements from a wider spatial coverage from March - December were recorded less frequently during the 2016 larval fish study. Using this information, we visually assessed the spatial and temporal landscape of environmental parameters.

First, we mapped surface salinity across the southern half of the estuary prior to and during peak spring discharge (Figure 2-15). Second, we plotted surface salinity, temperature, and Secchi disk depth over time at four locations in the estuary (Figure 2-16). Next, we mapped salinity depth profiles across time (Figure 2-17 and Figure 2-18). Finally, we plotted salinity measurements of four depths from four sites spanning the east to west estuarine gradient (Figure 2-19).

RESULTS

Q1 - What are the salinity, temperature, and turbidity conditions in the Skeena River estuary?

Environmental conditions in the Skeena River estuary change over time, geographic location, and depth. This change is driven by factors such as freshwater discharge, tides, currents, and circulation. Freshwater influx into the estuary creates salinity zones (Table 2-3) which can dictate the composition, abundance, and biomass of estuary communities utilizing certain geographical regions. Although many species are euryhaline and tolerate a range of salinity zones (Jones et al. 1990; Montagna et al. 2013), there can be more fine-scale structuring of these communities (Wagner & Austin 1999). For example, communities in low salinity regions of some estuaries show less invertebrate biodiversity (McInerney 1964; Rutger & Wing 2006; Montagna et al. 2013), but are important regions for juvenile salmon during seaward migration. Salinity gradients allow juvenile salmon to physiologically adapt to saline conditions (McInerney 1964; Toft et al. 2007). Surface salinities in the Skeena River estuary vary seasonally and spatially, but overall regions proximal to the mouth of the Skeena River (Inverness Passage) are generally mesohaline, while regions further west from Kinahan islands to the outer islands (Rachael Islands) are classified as polyhaline (Figure 2-15–Figure 2-19).

Freshwater entering estuaries is less dense then saltwater, often creating a salt-wedge, with the freshwater flowing above saltwater on the surface of the water column (Montagna et al. 2013). The formation of this salt wedge depends on the magnitude of freshwater discharge, tidal stage, and currents, which combine to cause fluctuations across the day, season, and between estuaries. While the salt wedge is apparent in some estuaries, others experience more mixing and estuaries can be classified as salt-wedges, partially mixed, well-mixed or fjord type estuaries (Montagna et al. 2013). Compared to other estuaries in British Columbia that have extensive salt-wedges, such as the Fraser and Squamish River estuaries, the Skeena River estuary experiences more mixing resulting in a less expansive salt-wedge (Hoos & Packman 1974; Hoos & Vold 1974; Hoos 1975; Kostaschuk et al. 1989). This is due to the geography of the Skeena River estuary, which is broken up by islands, and to the large tidal currents present (Hoos 1975; Lin & Fissel 2018). Based on depth profiles of salinity in the Skeena River, it can be classified as a partially-mixed estuary (Figure 2-15c), a conclusion supported by the concurrent research of Wild (2020). More horizontal stratification in Figure 2-15 would suggest more salt-wedge intrusion as seen in the Fraser River estuary (Kostaschuk et al. 1989). Tides often dominate mixing patterns in partially-mixed estuaries resulting in some stratification between top and bottom salinities, while winds dominate well-mixed estuaries (Montagna et al. 2013).

The influence of freshwater on the estuary environment varies throughout the year and across the geography of the estuary. As spring discharge increases, the influence of freshwater on the marine environment is higher, with lower salinities at the surface (Figure 2-15b) and in the water column (Figure 2-15c). Although the influence of freshwater during spring freshet is clear in Figure 2-15, the peak discharge values in spring were the lowest on record since 1928. Therefore, we expect that these patterns would be stronger during years of higher spring discharge (see maximum discharge values from Figure 2-9a). Higher stratification of salinity can be found at sites closer to the river mouth. This is illustrated in Figure 2-16, as depth profiles from Inverness Passage show vertical lines (less mixing) compared to sites by Kitson or Kinahans Islands, which are more inclined. This can also be seen in Figure 2-18, which compares salinity from April to December on a larger geographic scale. Salinity measurements from Rachaels and Stephens Island show less variation across depths (lines are closer together) (Figure 2-17). These sites are at the outer extent of the estuary, located 8 km and 22 km NW of Kinahan Islands, but demonstrate evidence of a peak fall discharge period despite being located far from the river mouth (Figure 2-17). When investigating finer-scale patterns closer to the river mouth, we found that southern channels (Marcus and Telegraph passages) have fresher environments (Figure 2-18). These southern passages receive approximately 75% of freshwater from the Skeena River, while the remaining 25% heads north through Inverness Channel (Trites 1956; Lin & Fissel 2018). After passing though these three channels, the majority of freshwater is then advected northward through the Chatham sound due to Coriolis force (Trites 1956; Lin & Fissel 2018).

Temperature in the estuary is also closely tied to freshwater discharge. Figure 2-14 compares temperature in the Skeena River estuary to marine SST from April to November. During May, freshwater from the Skeena River is colder than the marine environment (Figure 2-14) creating a temperature gradient across the estuary which is strongest during snow-melt freshet in May. During May, temperatures within the top 5m of the water column are colder at sites closer to the river mouth compared to sites further away (Figure 2-20a). By the end of June, sites closer to the river mouth are warmer (Figure 2-20b) and temperatures increase in the estuary throughout the summer (Figure 2-19) until they are warmer than the surrounding marine environment (Figure 2-14). This is likely due to shallower and more sheltered conditions within the Skeena River estuary and warmer freshwater temperature due to increasing air temperature in the interior.

Lastly, the Skeena River estuary is highly turbid, which is more pronounced during periods of high spring freshwater discharge (Figure 2-19) (Hoos 1975). For example, freshwater in Chatham sound is three to four times higher during peak spring discharge in May and June (Lin & Fissel 2018). During the spring of 2016 (April–June), Secchi disk depth, a coarse-scale measure of visibility, ranged from 0.2 m–7.0 m depending on date and location in the estuary (Figure 2-19).

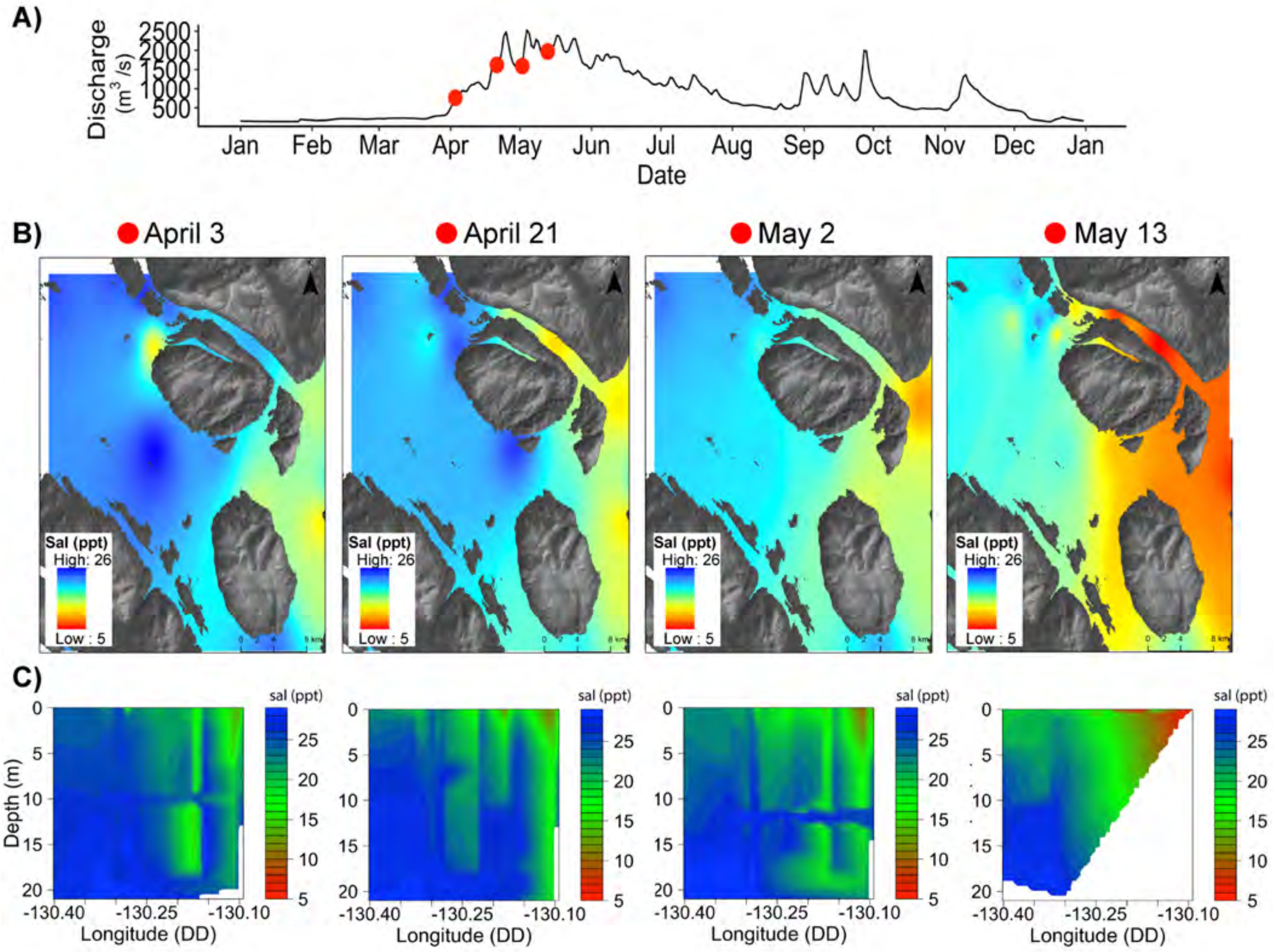

Figure 2-15. Gradient of surface salinity (B) in the Skeena River estuary over time at four time points (April 3, April 21, May 2, and May 13) during the period of spring discharge in 2016 (A). Vertical salinity profiles according to distance to the river mouth (longitude) are shown in panel C. Lower salinities (more freshwater) are indicated by red while higher salinities are indicated by blue (panels B and C). Maximum spring discharge in 2016 was on May 4 and was the lowest recorded since 1928.

In summary, sites closer to the river mouth have lower salinity, higher vertical stratification, and more turbid environments in the Skeena River estuary (Figure 2-21). These patterns are more pronounced in May during peak spring discharge which makes the estuary environment colder than the marine environment. Temperatures throughout summer increase and become warmer than the surrounding marine environment until September. Although we assessed these environmental variables across time of year, they are also influenced by the time of day due to tides, currents, winds, and daily discharge variations (morning compared to evening). We acknowledge that these gradients change according to several scales that were not assessed. Given that the measurements presented in the report were taken opportunistically during other projects, they do not have the resolution to assess the estuary environment at a finer time scale. These abiotic variables create a dynamic estuary environment which drive the spatio-temporal dynamics of estuary communities (Jones et al. 1990; Bacheler et al. 2009a; Cloern et al. 2017). We explore the links between the environmental variables discussed here, with the dynamics of fish and invertebrate species in subsequent sections. Forage fish species and juvenile salmon use of estuaries in relation to environmental variables is investigated in Section 5.4 and 3.1 respectively, while the dynamics of zooplankton populations to environmental changes are discussed in Section 4.2.

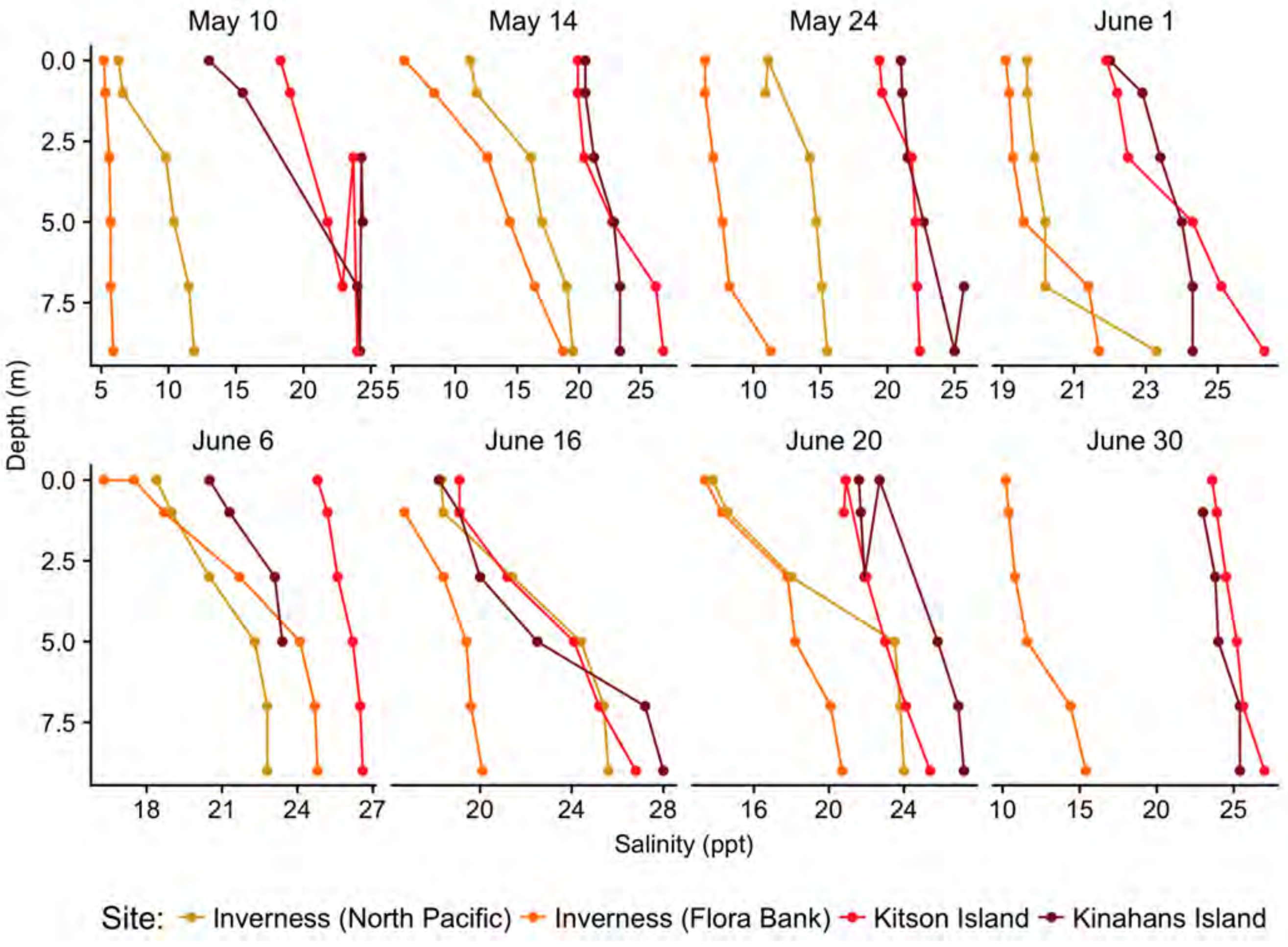

Figure 2-16. Vertical salinity profiles in the estuary from four sites at varying distance from the Skeena river mouth from May 10–June 30 in 2016 during peak salmon migration. Sites listed in order of decreasing proximity to the river mouth: Inverness Passage (yellow), Flora Bank (orange), Digby Island (red), and Kinahans Island (dark red).

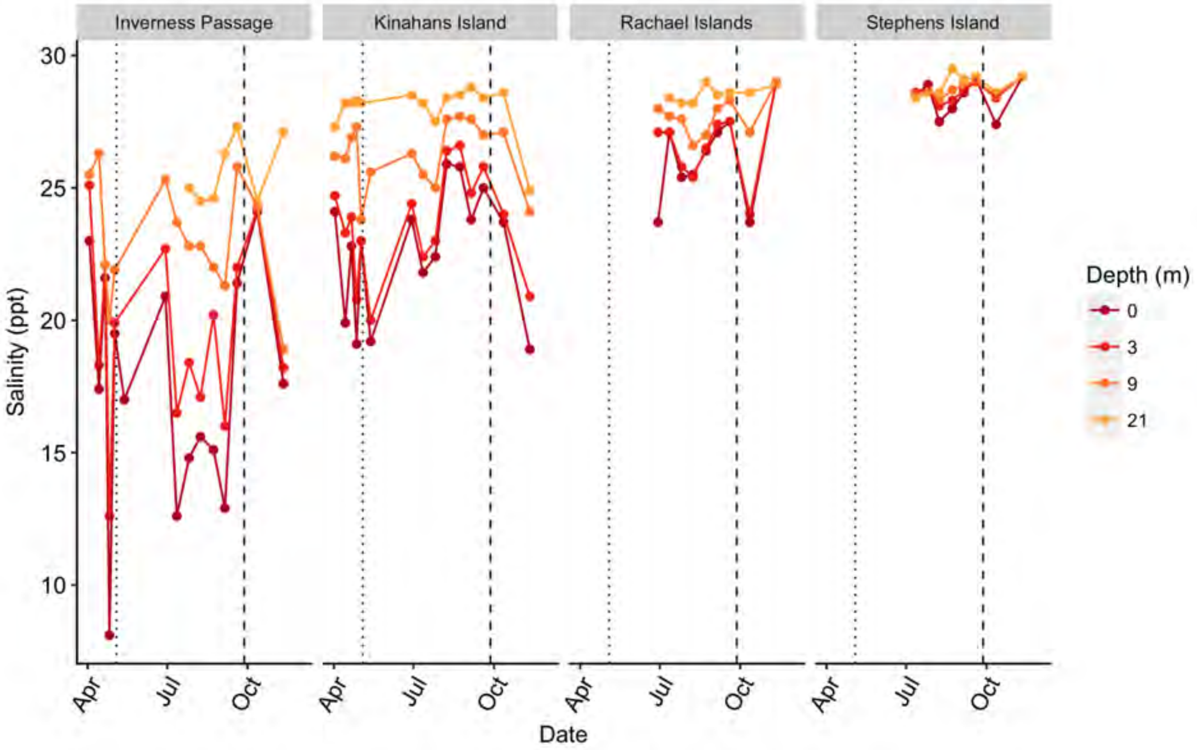

Figure 2-17. Salinity at four depths (0 m, 3 m, 9 m, 21 m) in 2016 from four estuary sites listed in order of increasing distance from the river mouth: Inverness Lelu, Kinahans Islands, Rachaels Island, and Stephens Island (NE point). Dotted lines (….) represent spring peak discharge while dashed lines (---) represent fall peak discharge. Sampling from Inverness-Lelu and Kinahans Island was performed from March–December, while sampling at Rachaels Island and Stephens Island was only performed from August to December.

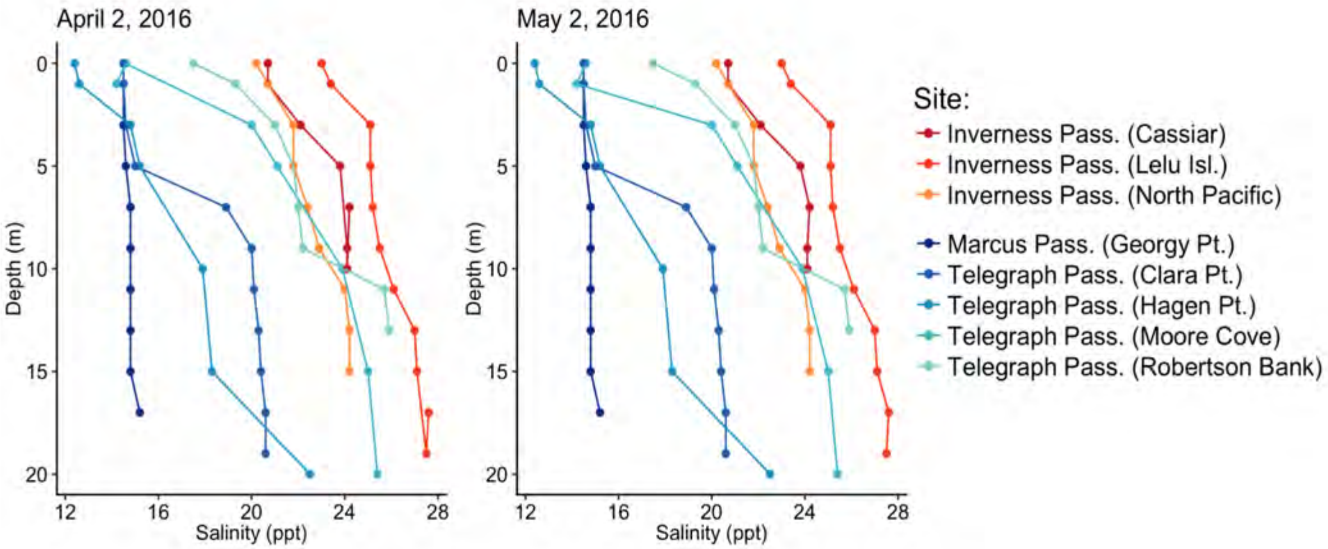

Figure 2-18. Vertical salinity profiles from sites in the Skeena River estuary from Inverness Passage (orange) and Telegraph and Marcus Passages (Blue) measured in April and May 2016.

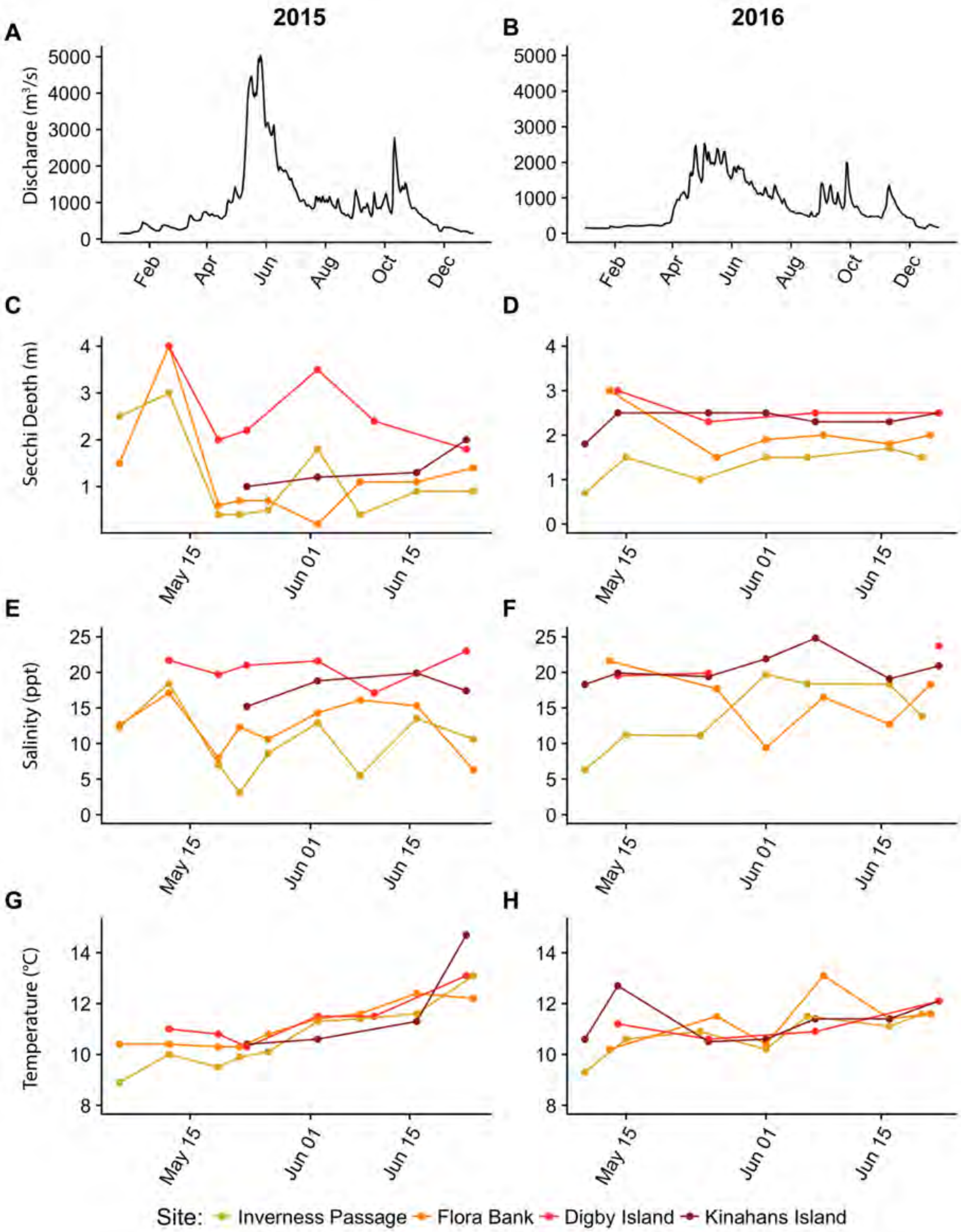

Figure 2-19. Environmental variables plotted at four sites across the Skeena River estuary at varying distance from the river mouth in 2015 and 2016. Freshwater discharge (A, B) was collected at Usk Discharge Station (08EF001) and Secchi depth (C, D), salinity at 0 m (E, F), and temperature at 0 m (G, H) were collected from the following sites listed in order of decreasing proximity to the river mouth: Inverness Passage (yellow), Flora Bank (orange), Digby Island (red), Kinahans Island (dark red).

Table 2-3. Classification of estuary zones according to salinity, adapted from Montagna et al. (2013).

Physiographic Area	Classification	Salinity (ppt)
River	Limnetic	< 0.5
Head of estuary	Oligohaline	0.5–5
Upper reaches	Mesohaline	5–18
Middle – Lower reaches	Polyhaline	18–30
Estuary Mouth	Euhaline	30–40

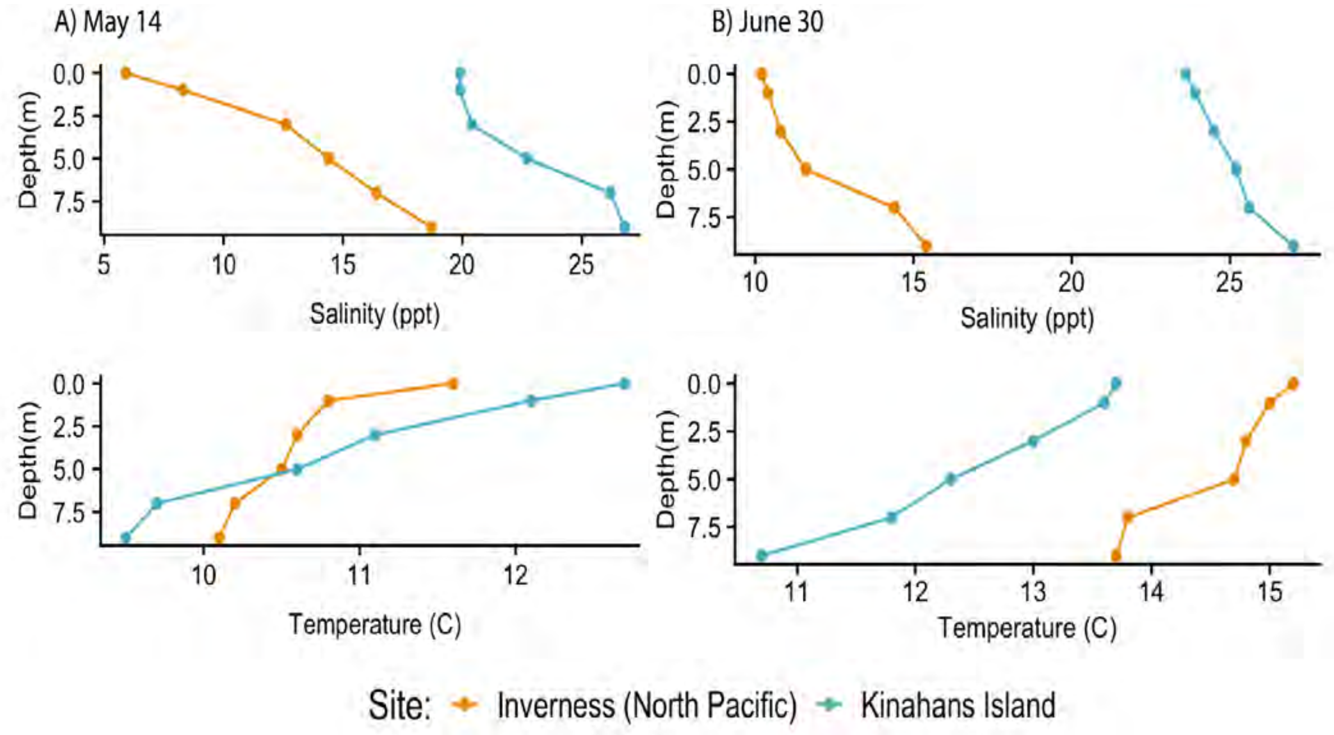

Figure 2-20. Vertical salinity and temperature profiles from two sites in the Skeena River estuary, Inverness (proximal to the river mouth) and Kinahans Island, from A) May 14 and B) June 30 in 2016.

Figure 2-21. Turbid conditions while purse seining across a plume front in the Skeena River estuary during May close to Inverness Passage (photo by Ciara Sharpe).

Section 2.1.4. - Key Findings

1) Freshwater discharge creates environmental gradients of salinity, temperature, and turbidity which vary according to time of year and location within the estuary.

2) The estuary environment is colder than the marine environment during periods of peak spring freshet.

3) Sites closer to the river mouth are fresher environments with higher turbidity and vertical stratification.

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2.2. Biotic environment

Estuary habitats in the NE Pacific can be broken into three main types: beaches (e.g., sandy, cobble, boulder), channels (e.g, tidal channels, riverine), and vegetative (e.g., eelgrass, saltmarsh, macroalgae) habitat (Levings 2016). These habitat types can be used as a practical framework for categorizing habitat availability for fish, such as juvenile salmon (Levings 2016). These estuary habitats are known to contribute different types and amounts of resources along with varying degrees of predator refuge. Vegetative habitats such as eelgrass, macroalgae, or salt-grass are known to support fish populations by providing shelter from predators and supporting increased secondary production of a diversity of prey (plankton and epiphytic fish) (Duggins et al. 1989, 1990; Semmens 2008; Kennedy et al. 2018a).

The availability of vegetative habitat is dependent on geography and environmental conditions which vary from estuary to estuary. Upstream of the river mouth, the Skeena River channel is confined by steep mountains. This results in the deposition of coarse fluvial deposits along the channel and fine sediments in protected embayments beyond the river mouth. These finer sediments are mainly deposited around a complex of islands and channels: mainly Kitson Island, Kennedy, and Smith Islands (Conway et al. 1996). This geology and pattern of sedimentation shapes the available habitat in the Skeena River estuary. Compared to the Fraser River, which has a large intertidal river delta and extensive saltmarsh habitat (Kostaschuk et al. 1989), sediment deposits in the Skeena River estuary are small in scale and located along channel margins (Conway et al. 1996). Thus, there are narrow bands of salt marsh habitat in the Skeena River estuary, particularly in the tidally-influenced lower river portion. The primary sub-habitat types are mainly mud and sand flats (“beach habitat”) (Levings 2016), rocky shores (“channel habitat”), and eelgrass beds and intertidal macroalgae, including kelp beds (“vegetation habitat”). Most of the eelgrass in the Skeena River estuary (50-60%) is located on Flora Bank (Figure 2-22) (Hoos 1975).

The following sections (2.7 and 2.8) present research on the Flora Bank eelgrass bed and summarize historical studies on Skeena River eelgrass.

Figure 2-22. Aerial view of Flora Bank eelgrass bed looking west. towards Kitson Island. (Photo by Brian Huntington)

2.2.1. Sediment transport regimes of the Flora Bank

This section summarizes the results of the following Lax Kw'alaams Fisheries funded research:

McLaren, P. (2015). A Sediment Trend Analysis (STA) of Prince Rupert Harbour and it's Surrounding Waters. Prepared for the Lax Kw'alaams Band.

McLaren, P. (2016). The Environmental Implications of Sediment Transport in the Waters of Prince Rupert, British Columbia, Canada: A Comparison Between Kinematic and Dynamic Approaches. Journal of Coastal Research, 32(3).

Vegetative habitat in an estuary is influenced by a complex serious of processes including environmental conditions (temperature and salinity), currents, tides, wave action (van Katwijk et al. 2009), light attenuation (Ralph et al. 2007; Thom et al. 2008,) and sediment movement (Cabaço et al. 2008). Processes surrounding sediment movement in an estuary can influence the suitability and maintenance of eelgrass habitat through erosion and deposition. Conversely, eelgrass habitat can act as an 'ecosystem engineer' and buffer wave action, stabilize sediment, and improve water clarity (van Katwijk et al. 2009; Hotaling-Hagan et al. 2017). The complexity of eelgrass habitat suitability and the interconnected nature of physical dynamics in the estuary makes eelgrass habitat restoration and bed creation challenging. This is evident in the varied success rates of such restoration projects (Tanner et al. 2008; van Katwijk et al. 2009).

Flora Bank is a large eelgrass bed composed of intertidal sand, located between Kitson and Lelu Island, and contains the majority of eelgrass in the Skeena River Estuary. Measuring around 4 km^2 (Figure 2-23), the geomorphological context of the bank is completely unique in the sense that there are no comparable features elsewhere in British Columbia (McLaren 2016). Proposed development (LNG terminal and shipping trestles) along Flora Bank during 2015 (Stantec 2015) ignited concerns over the impacts of development on the bank. Given that the sand on Flora Bank persists in a high-energy environment and that this region is known to support particularly high abundances of juvenile salmon during the smolt migration (Higgins & Schouwenburg 1973; Carr-Harris et al. 2015), Lax Kw'alaams First Nation in conjunction with SFC commissioned an in-depth sediment analysis of the Prince Rupert Region to understand how proposed development would impact natural sedimentation and erosion processes in the area. Dr. Patrick McLaren conducted a Sediment Trend Analysis (STA) in 2015, which identified the sediment transport patterns and source locations for all sediments.

Results of this study identified that the Flora Bank is a remnant feature from a pre-existing environment (Figure 2-24). Specifically, the sediments that compose Flora Bank are made up of glacial sediments that likely reflect a pause in the advance or retreat of the late Wisconsin ice cover, or a medial moraine from a glacial ice tongue movement. The constant ocean processes, such as tides and wave action, have held the sand in place over time. Thus, Flora Bank can be described as relic sediment, which is likely to be about 8000 years old. This understanding is further supported by a comparison of historical and current bathymetric charts and aerial photos, which illustrate the constant position and size of Flora Bank overtime (Figure 2-23). In-depth methods and results of this study can be found in McLaren (2015) and McLaren (2016a), while additional information on the ocean current regime over Flora Bank can be found in McLaren (2016b).

The novel discovery about the origins of Flora Bank has significant implications for development and management of this region of the estuary. Given that Flora Bank is not maintained by sediment that comes from elsewhere (i.e. The Skeena River), but is a lag deposit that was formed *in situ* and is held in place by surrounding processes, potential developments, such as docking facilities and pilings that have been previously proposed and federally approved (Pacific Northwest (PNW) LNG), could significantly alter the currents and processes holding Flora Bank in place. Given the strength of tidal currents and significant wave action that occur each flooding and ebbing tide (McLaren 2016b), it is likely that once lost, there would be no source or transport regime capable of replacing Flora Bank.

Section 2.2.1 - Key Findings

1) Flora Bank is a unique geological feature composed of glacial sediments, which likely formed about 8000 years ago.

2) Flora Bank is not maintained by sediments from other sources, but instead is held in place by surrounding processes such as tides and wave action.

3) Potential development on Flora Bank such as docking facilities and pilings could significantly alter the currents and processes holding Flora Bank in place and result in the loss of the bank overtime.

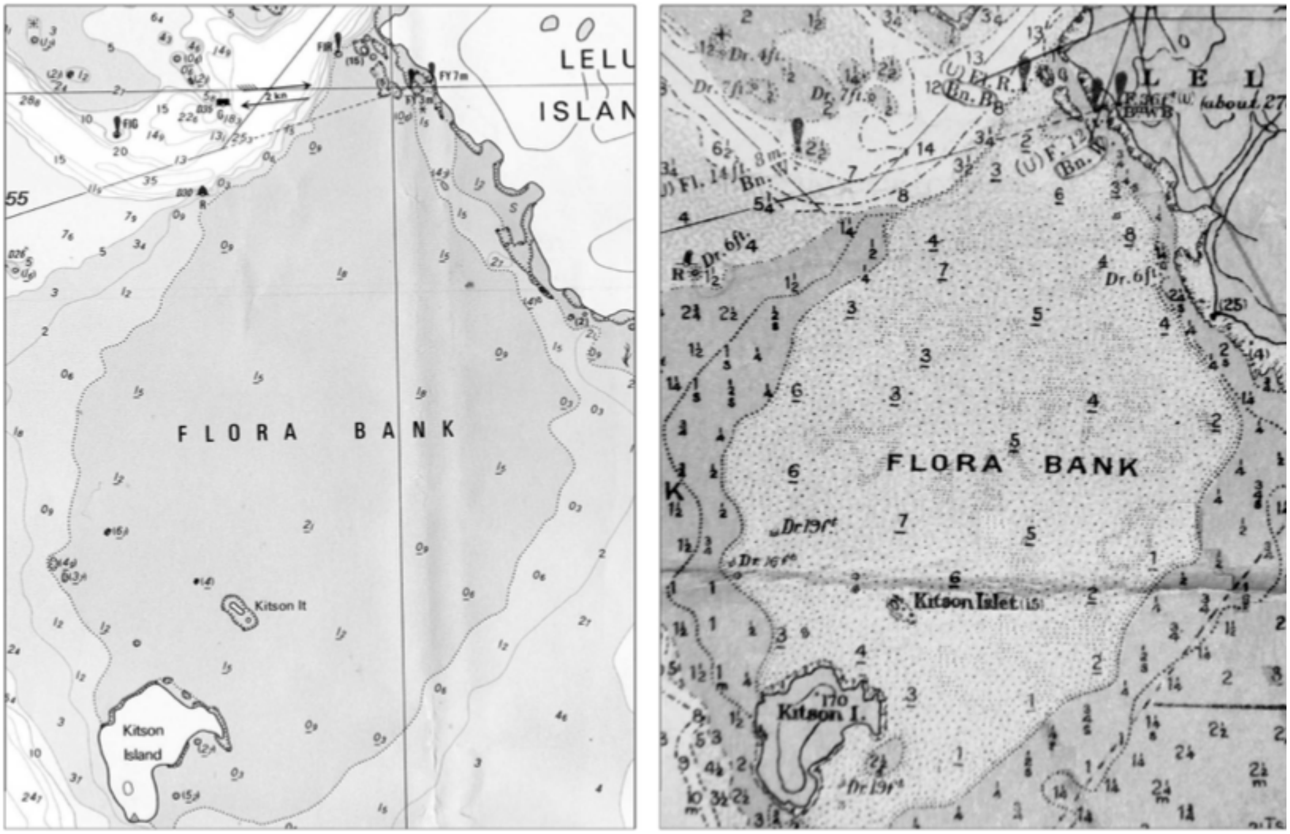

Figure 2-23. Comparison of nautical chart outlines of Flora Bank in 1991 (left) compared with the feature in 1907 (right). The outline of the low-water line has changed very little over 84 years of dynamic estuary processes (photos from McLaren 2015).

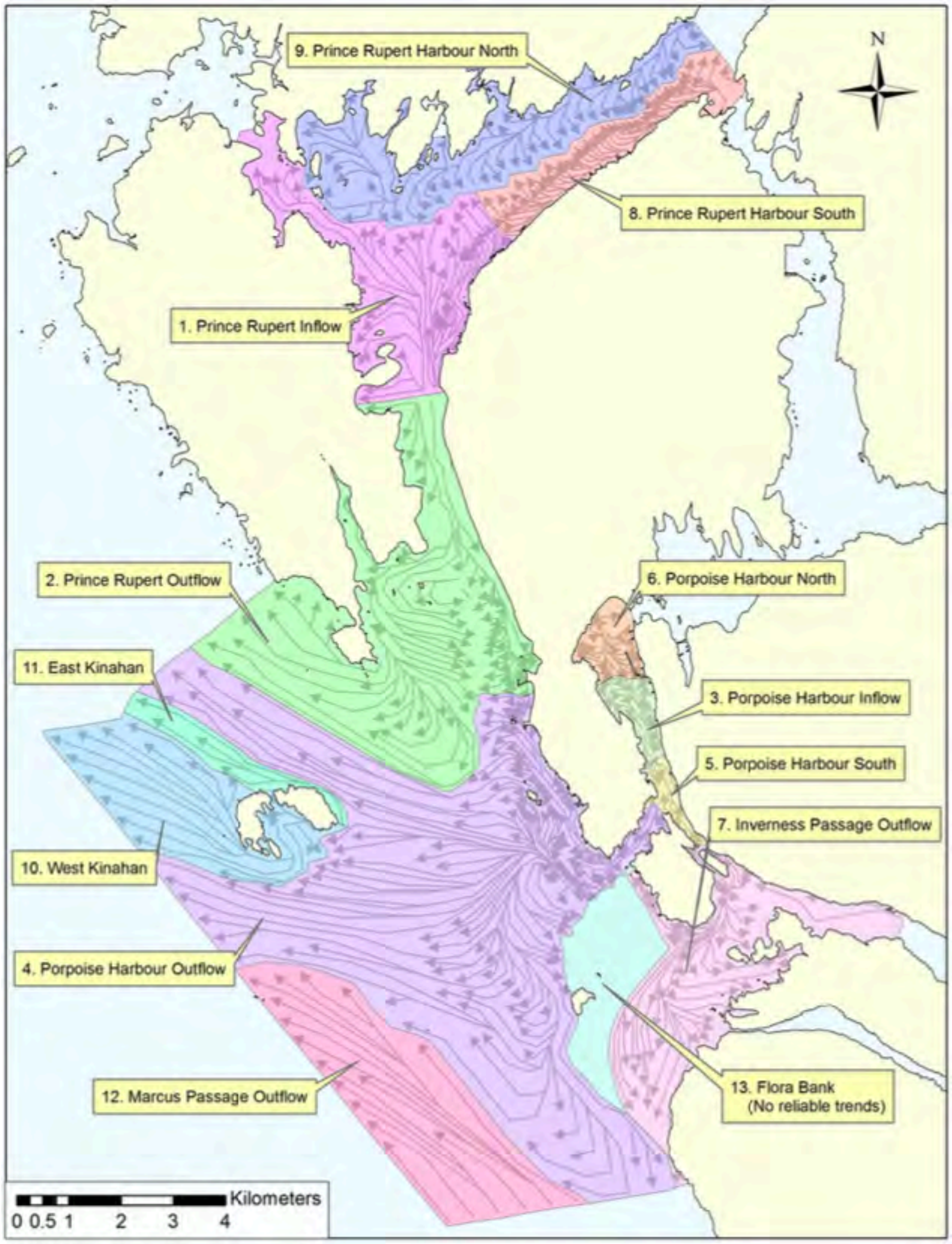

Figure 2-24. Sediment Transport Environments (TEs), as determined from Sediment Trend Analysis (STA), illustrating direction of sediment movement in the Prince Rupert and Flora Bank region of the Skeena River Estuary. This figure is from McLaren (2015) and illustrates the anomalous sediment transport regime around Flora Bank.

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2.2.2. Eelgrass survey of Flora Bank (Contribution by Kyla Warren)

This section contains an abridged version of the following SFC report:

Warren, K. (2017). 2016 Eelgrass Survey of Flora Bank, Prepared for the Lax Kw'alaams Band and Fish Habitat Restoration Initiative, Skeena Fisheries Commission, Kispiox BC.

INTRODUCTION

Within the Pacific north coast, few habitat types are as highly valued as eelgrass (*Zoster sp.*) beds and eelgrass itself is recognized as one of the most important marine macrophytes for maintenance of marine resources (Lucas et al. 2007). Eelgrass is a vascular, perennial plant that grows in the intertidal and shallow subtidal zones of coastal and estuarine waters. There are two main species of eelgrass on the Pacific north coast: the native *Z. marina* and the introduced *Z. japonica*. Both species spread through seeds and rhizomes to form dense mats that transform low-complexity soft sediment habitats into high complexity eelgrass beds. These beds are associated with higher survival and recruitment of juvenile Dungeness crabs (Fernandez et al. 1993; Plummer et al. 2013), provide spawning habitat for herring (Haegele et al. 1981), and are important nursery habitat for many economically important species, including rockfish (Murphy et al. 2000), cod, herring, and forage fish (Dean et al. 2000). Juvenile Chinook and coho salmon show a preference for eelgrass habitat even when compared to other structurally complex habitat types (Korman et al. 1997; Semmens 2008). However, eelgrass is sensitive to anthropogenic impacts from development and in many areas, eelgrass populations have experienced widespread declines and fragmentation of eelgrass beds (Lucas et al. 2007; Plummer et al. 2013).

In the Skeena River system, eelgrass has been hypothesized to be important habitat for juvenile salmon during the critical period where salmon smolts spend days to months rearing and feeding within the estuary (Carr-Harris et al. 2015; Moore et al. 2016). Surveys of juvenile salmon abundance within the Skeena estuary have shown an area, stretching from the southern end of Kaien Island to the northern portion of Smith Island, with some of the highest densities of juvenile sockeye, coho, and Chinook salmon found anywhere in the estuary (Carr-Harris et al. 2015). Within this larger area, the highest densities of juvenile Chinook and sockeye salmon were captured in the Flora Bank region (Moore et al. 2015; Sharpe et al. 2019). This area of importance also contains the community of Port Edward and much of the industrial infrastructure of the Port of Prince Rupert, raising concerns about anthropogenic impacts to critical habitat.

Given the sensitivity of eelgrass to human disturbance, there is concern regarding the ongoing health of eelgrass beds in the Skeena estuary. In particular, concerns were raised regarding the PNW LNG facility, a LNG shipping terminal that was proposed to be developed adjacent to Flora Bank is 2015. Although the project received federal approval, the development was ultimately cancelled in 2017. The PNW LNG facility shipping berths were proposed to be immediately adjacent to Flora Bank, with a bridge and trestle that ran along the bank's edge. Flora Bank is one of the largest eelgrass beds in British Columbia and the largest in the Skeena estuary and is located in the center of an area with high salmon density (Ocean Ecology 2009; Carr-Harris et al. 2015; Sharpe et al. 2019). Given the known importance of Flora Bank, one of the main objectives of this study was to investigate eelgrass habitat in this region and assess potential effects of the then-proposed project.

There have been several past attempts to assess eelgrass on Flora Bank using a variety of remote sensing and imaging methodologies. In 1997, an eelgrass survey was undertaken using Compact Airborne Spectrographic Imagery (CASI) that estimated eelgrass presence in the Prince Rupert Harbour area, including Flora Bank (Forsyth et al. 1998). In 2008, a towed video camera and side-scan sonar was used to assess eelgrass presence and estimate shoot length and percent cover on transects of Flora Bank at high tide, but due to the shallow intertidal nature of Flora Bank only the edges of the bank were surveyed (Ocean Ecology 2009). In 2015, PNW LNG purchased high-resolution multispectral satellite imagery of Flora Bank from 2007, 2009, and 2011 to support the project's Environmental Assessment (Stantec 2015). Spectral analysis of the photographs was used to identify eelgrass beds. In 2013, PNW LNG collected two additional satellite photographs and used visual analysis to delineate eelgrass beds (Stantec 2015), while an independent group used visual analysis to delineate eelgrass beds from a mosaic of aerial photographs (Faggetter 2014). PNW LNG also undertook a ground survey in 2013 to ground-truth the remote sensing data and to collect biological data (Stantec 2015).

The 1997 eelgrass survey and the 2008 towed video camera and sonar survey showed a close relationship between the aerial imagery and the ground observations, suggesting the beds were fairly stable in size and shape (Ocean Ecology 2009). The 1997 spectrographic analysis estimated that approximately 0.80 km^2 of Flora Bank was covered by eelgrass. In contrast, the multispectral imagery from 2007, 2009, and 2011 indicated that 0.64 km^2 of Flora Bank was covered by eelgrass beds (Stantec 2015) and the visual estimation from 2013 showed the lowest estimate of 0.33 km^2 (Stantec 2015). The independent aerial photography survey completed in 2013 estimated the eelgrass on Flora Bank covered 1.0 km^2 (Faggetter 2014). This represents an increase from the 0.80 km^2 estimated in 1997 and estimates 56% more eelgrass coverage than reported by Stantec (2015) from the satellite photography (Faggetter 2014).

These varied estimates highlight the challenges associated with using different methods across different seasons and sampling years to document eelgrass coverage. Eelgrass biomass is likely greatest from mid-July to August in the Prince Rupert region (Pedersen & Borum 1993; Ocean Ecology 2011), while studies conducted earlier in the growing season, such as those conducted by Stantec (2015), likely provide lower estimates. Faggetter (2014) also highlights other methodological inaccuracies with Stantec (2015) that would lead to lower estimates such as a sideways towed camera, low visibility, and incomplete coverage of the eelgrass bank. In addition, local intra-annual variation has been documented, suggesting that eelgrass extent can vary from year to year (Ocean Ecology 2011). Given the inconsistency regarding the areal extent of eelgrass coverage on Flora Bank, a significant near-shore habitat component in the Skeena River estuary, additional baseline data collection and monitoring is required. Repeatable and reliable methods measuring changes to eelgrass coverage and health should be employed to establish a successful long-term monitoring protocol.

Delineating and monitoring eelgrass beds through remote sensing is a well-established method. CASI (Gotceitas & Colgan 1989; Garono et al. 2007), towed camera transects with sonar (Lefebvre et al. 2009; Schubert et al. 2015), multispectral analysis of hyperspectral satellite photographs (Forster & Jesus 2006; Klemas 2011), and visual analysis of aerial photographs (Short & Burdick 1996; Chauvaud et al. 1998; Klemas 2011) are all regularly-used techniques with documented high accuracy. Nevertheless, the high variation in previous estimates on Flora Bank (Forsyth et al. 1998; Faggetter 2014; Stantec 2015) demonstrates that additional sampling is required to ground-truth previous estimates.

Many eelgrass surveys combine remote sensing data with field data to ensure accuracy (Mumby et al. 1999; Pasqualini et al. 2001). Monitoring based on field data removes the potential for habitat to be misclassified and allows for data on eelgrass health to be gathered. To date, field surveys have been limited in scale and have been designed to support industrial permit applications rather than a monitoring program (Faggetter 2014; Stantec 2014). The objective of the work conducted here was to collect high-quality georeferenced field data to establish potential monitoring sites and protocols for long-term monitoring. Measuring indicators of eelgrass health will provide additional information on the status of the eelgrass and support future monitoring activities.

METHODS

Study Area

Flora Bank is located approximately 2 km southwest of Port Edward, British Columbia in Chatham Sound. Flora Bank is a large intertidal sand bank with an area of approximately 3.31 km^2, as determined through orthophotography. The maximum elevation of Flora Bank is approximately 2 m and it is submerged at all but very low tides. Previous eelgrass surveys suggest that much of the surface of Flora Bank is covered by many small to medium eelgrass beds that form an eelgrass meadow complex. Given the size of the bank, the complex spatial structure of the eelgrass beds, and the limited time periods during which Flora Bank is exposed, it would be impossible to ground survey all eelgrass beds directly. Instead, representative areas of Flora Bank were sampled for this eelgrass survey program.

Nineteen sites were selected by generating random coordinates using ArcGIS software. Each randomly generated coordinate became the southwest corner of a 50 m by 50 m survey site. The sites were surveyed from July 4 to July 7, 2016 during the lowest tide of the day. During this period, the lowest daily tide ranged from 0.2–0.5 m.

Eelgrass Bed Delineation

Upon arrival at each site, a square study area was marked off by using a Trimble Nomad 1050 LC handheld GPS operated through Terrasync (version 5.86) data collection software. A GPS track was laid by walking the perimeter of every eelgrass bed within the site that was greater than 0.5 m in diameter. The GPS recorded its position once per second, with a pre-processing horizontal accuracy that ranged from 2.0 to 3.1 m. Eelgrass was considered to be part of a bed when more than 5% of the surface area within the perimeter was visually estimated to be covered in eelgrass. Where a bed extended beyond the study area, only the portion within the study area was mapped.

For each eelgrass bed, the species of eelgrass present was recorded (native species *Z. marina* or introduced *Z. japonica*) and a visual estimation was made of the amount of surface area the eelgrass covered within the bed. Surface coverage was recorded as sparse (<25%), moderate (25–74%), or high (≥75%).

Biological Data collection

The largest eelgrass bed within each site was selected for biological data collection. Three 0.25 m x 0.25 m quadrats were haphazardly placed within the bed (Figure 2-25). Inside each quadrat, the number of eelgrass shoots was counted, the percent cover was visually estimated, the lengths of at least five eelgrass shoots were measured, and the percent of eelgrass shoots that were flowering was recorded. In surveys conducted July 6 and July 7, the number of eelgrass shoots measured increased to 10 per quadrat and the widths of the ten shoots were recorded, in addition to length.

Figure 2-25. 25 cm by 25 cm quadrat placed in an eelgrass bed at Site 2.

Data Processing and Analysis

The geospatial data was post-processed using Trimble Pathfinder Office (version 5.85) software with differential correction, which increased the horizontal precision to less than 1 m. The geospatial data was imported into ArcMap for further analysis.

The total area within mapped eelgrass bed polygons was calculated for each site, generating a measurement of percent bed eelgrass coverage. The percent of the total bed area comprised of “sparse”, “moderate”, and “high” density beds was calculated.

The length, width, and number of eelgrass stems per quadrat were compared using a one-way analysis of variance (ANOVA) with a Tukey HSD post-hoc analysis. The photosynthetic area of the measured shoots was calculated by multiplying the shoot length by the width and was also analyzed by one-way ANOVA with a Tukey HSD post-hoc analysis. An analysis of residuals indicated that the number of eelgrass stems per quadrat and the percent cover were approximately normally distributed with constant variance and did not require transformation, while the lengths, widths, and photosynthetic area of eelgrass stems were log transformed to stabilize variance. Sites without eelgrass beds were omitted from the analyses. All statistical analyses were carried out in R, version 3.4.0 (R Core Team 2016).

RESULTS

Eelgrass Bed Delineation

Of the 19 sites surveyed, 13 contained eelgrass (68%), all of which was *Z. marina.* No *Z. japonica* was observed during the surveys. In general, sites near the center and north end of Flora Bank contained eelgrass beds, while sites located closer to the east and west edges of the bank were less likely to contain eelgrass beds (Figure 2-26). The percent of the study sites covered by eelgrass beds ranged from 1% (Site 20) to 72% (Site 17) with an average of 31% coverage in sites containing eelgrass (Table 2-4).

Table 2-4. Percent area covered by eelgrass beds at each site.

Site	Area within beds (%)
1	47.9
2	14.7
5	39.1
6	50.4
7	23.3
8	54.1
11	35.4
12	31.0
14	18.9
17	71.5
18	28.0
19	15.5
20	1.3
Mean	***30.8***
SD	***19.8***

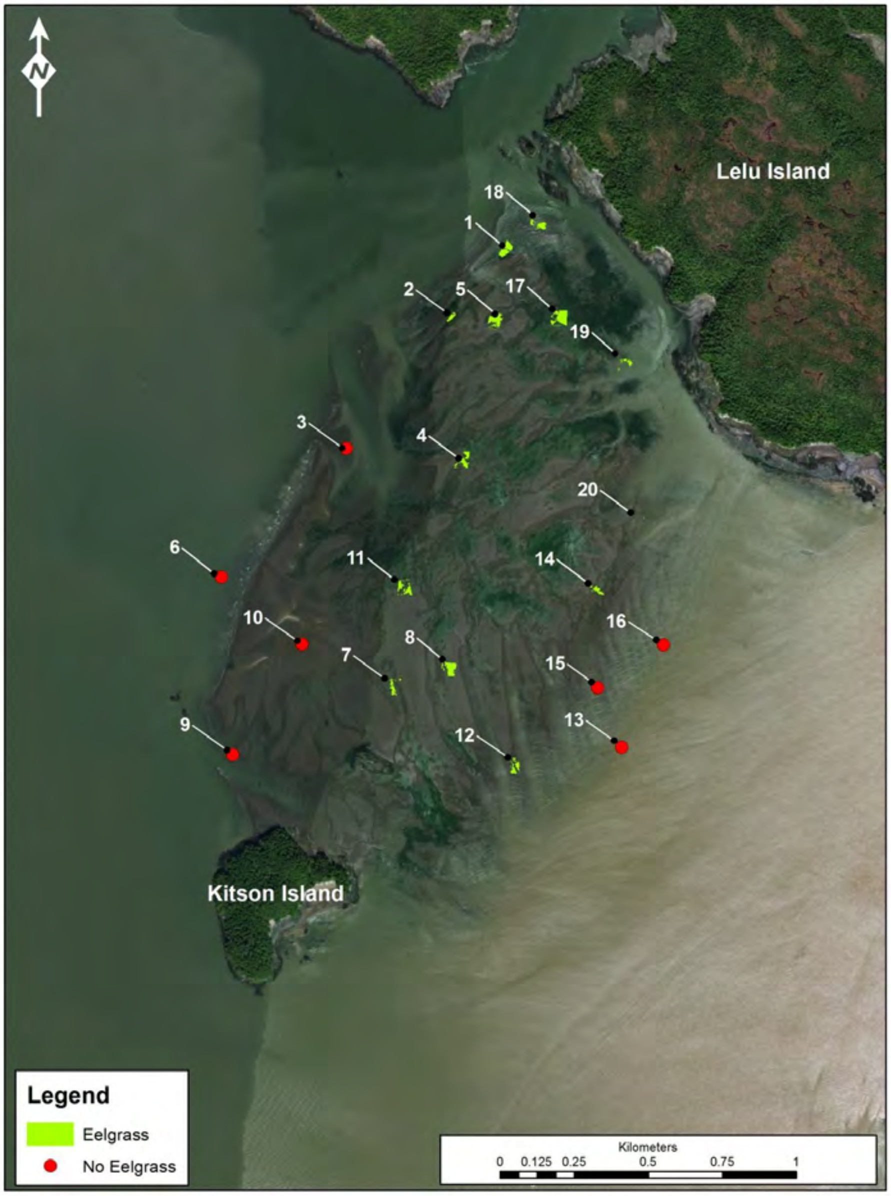

Figure 2-26. Locations and names of eelgrass survey sites on Flora Bank. Mapped eelgrass beds are shown in light green, while red circles indicate survey sites that contained no eelgrass.

When sites that did not contain any eelgrass were included in the summary, the average eelgrass coverage per site was 21.7%. If this average coverage is extrapolated to the entire estimated area of Flora Bank (3.31 km^2), there is an estimated 0.68 km^2 (+/- 0.72 km^2) of eelgrass beds on Flora bank. This study only surveyed a small portion of the overall area of Flora Bank and there was high variability between sites, resulting in a very large confidence interval for the total estimate.

The majority of eelgrass beds were of moderate density (35.8%), compared to sparse density (34.6%) and high density (29.4%), with several sites containing beds of different densities. Figure 2-27 shows the percentage of individual sites covered by eelgrass beds of each density category. No pattern relating location on the bank and density of eelgrass was immediately apparent.

Figure 2-27. Proportion of the area in each survey site with no eelgrass coverage (brown), sparse (light green), moderate (medium green), and high density (dark green) eelgrass coverage. Sites that did not contain eelgrass are not depicted.

Biological Data

Table 2-5 presents the mean of measured biological characteristics at each site. The mean number of eelgrass shoots per quadrat ranged from 12 to 69. The number of shoots per quadrat varied significantly among sites (ANOVA, $F_{12,29} = 5.7$, $p < 0.001$). This was primarily due to Site 19, which had significantly fewer shoots than nine of the other sites (Figure 2-27a; Tukey HSD, $p < 0.05$).

The mean length of eelgrass shoots ranged from 30 cm to 89 cm (Table 2-5) with the variation in length among sites was significant (ANOVA, F12,29= 8.37, $p < 0.001$). Eelgrass shoots at Site 1 and Site 2 were significantly longer than those at four and seven of the other sites, respectively, and shoots at Site 20 were significantly shorter than those at five of the other sites (Figure 2-28b; Tukey HSD, $p < 0.05$).

Table 2-5. The mean and standard deviation (SD) of the number of shoots counted per 0.0625 m^2 quadrat, the estimated percent cover within the quadrat, the length and width of measured eelgrass shoots, and the percentage of shoots on which flowers were observed. (-) denotes no eelgrass present.

Site	Shoots (No./0.0625 m^2)		Cover (%)		Length (cm)		Width (cm)		Flowering (%)
	Mean	*SD*	Mean	*SD*	Mean	*SD*	Mean	*SD*	
1	33	*3.1*	83	*7.6*	75.4	*18.3*	NA	*NA*	0
2	33	*9.0*	95	*0.0*	89.2	*22.9*	NA	*NA*	0
3	-	-	-	-	-	-	-	-	-
4	50	*7.0*	75	*21.8*	56.3	*3.2*	NA	*NA*	0
5	69	*31.0*	86	*22.2*	35.9	*6.8*	NA	*NA*	0
6	-	-	-	-	-	-	-	-	-
7	37	*5.3*	68	*10.4*	30.0	*5.0*	0.25	*0.07*	0
8	34	*10.8*	70	*21.8*	43.4	*16.6*	NA	*NA*	22
9	-	-	-	-	-	-	-	-	-
10	-	-	-	-	-	-	-	-	-
11	38	*8.5*	50	*10.0*	33.4	*8.7*	0.23	*0.04*	0
12	28	*8.6*	36	*12.9*	47.0	*9.5*	NA	*NA*	24
13	-	-	-	-	-	-	-	-	-
14	49	*21.1*	92	*7.6*	52.6	*15.5*	0.50	*0.12*	0
15	-	-	-	-	-	-	-	-	-
16	-	-	-	-	-	-	-	-	-
17	28	*14.7*	60	*22.9*	47.3	*13.9*	0.27	*0.06*	0
18	22	*4.4*	43	*7.6*	67.7	*24.2*	0.35	*0.05*	0
19	12	*5.7*	10	*5.0*	53.8	*10.8*	0.31	*0.05*	0
20	24	*4.0*	33	*5.8*	32.1	*11.0*	0.24	*0.09*	0

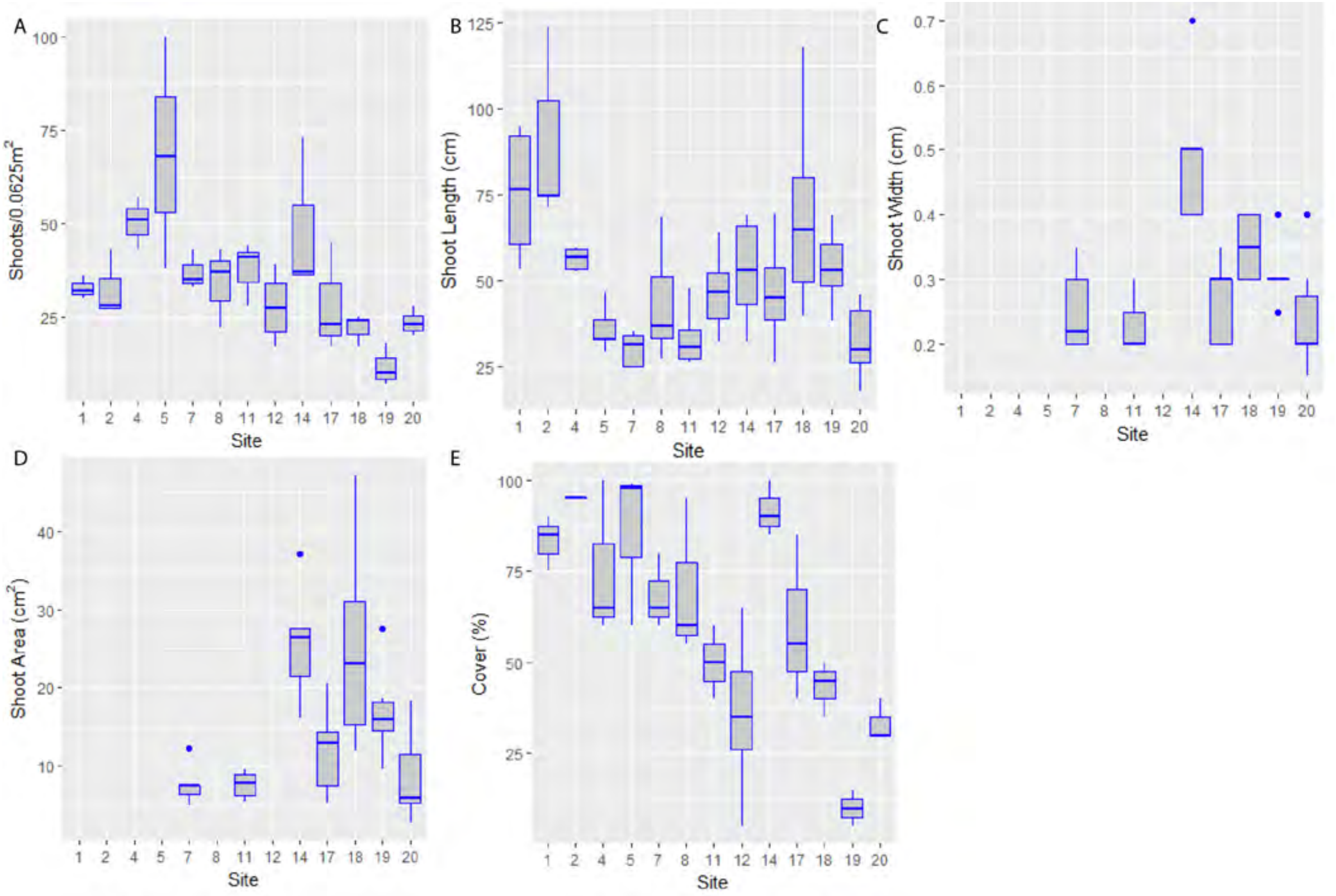

Figure 2-28. The A) number of eelgrass shoots, B) shoot length (cm), C) shoot width (cm), D) shoot photosynthetic area (cm^2) and, E) percent cover inside 0.0625 m^2 quadrats at each site on Flora Bank. The boxes indicate median, first, and third quartiles. Sites with no eelgrass beds have been omitted and sites where no width measurements were taken have no data in Panel C.

Eelgrass shoot width was only recorded at seven sites (Figure 2-28c), with mean width at each site ranging from 0.23 cm to 0.50 cm. There were significant differences between sites (ANOVA, F6,39= 10.1, $p < 0.001$). Eelgrass shoots at Site 14 were significantly wider than all other sites, and shoots at Site 11 were narrower than those at Site 18 (Tukey HSD, $p < 0.05$).

The comparison of calculated photosynthetic area of the eelgrass showed a similar pattern to shoot width (Figure 2-28d; ANOVA, F6,39= 7.53, $p<0.001$), with the eelgrass at Sites 14 and 18 having significantly greater area than four of the six other sites (Tukey HSD, $p < 0.05$). Site 20 had less photosynthetic area than three of the other six sites (Tukey HSD, $p < 0.05$).

Mean percent cover within quadrats ranged from 10% to 95%, with significant differences among sites (Figure 2-28e; ANOVA, F12,29= 8.90, $p < 0.001$). Eelgrass at Site 19 covered significantly less of the quadrat area than eelgrass at all other sites except Sites 17, 18, and 20 (Tukey HSD, $p < 0.05$). Site 18 and Site 20 also had significantly lower percent cover than Site 5.

Flowering was not widely observed during the biological surveys. While flowering eelgrass was observed in Site 8 and Site 12, it was noted that the flowering shoots within the surveyed quadrats was not representative of the eelgrass elsewhere in the survey site, most of which was not flowering.

Exposed rhizomes, a symptom of erosion, were observed in eelgrass beds outside of the study sites (Figure 2-29) but not in any of the study sites themselves. However, in Sites 5, 12, 14, and 19 it was observed that eelgrass was primarily growing on slightly raised hummocks of substrate, which may also indicate erosion of the areas surrounding the eelgrass beds.

Figure 2-29. Area of erosion between Site 2 and Site 4. Eelgrass is primarily on an elevated section of substrate (left) with exposed rhizomes indicating a zone of erosion (right).

DISCUSSION

Eelgrass on Flora Bank has been hypothesized to be limited due to erosion (Stantec 2014), turbidity (Faggetter 2014; Stantec 2015), and exposure to a high-energy environment (Stantec 2015). *Z. marina* has high phenotypic plasticity and has been documented responding to these stressors through distinct physiological changes. Under low light conditions, it is expected to produce larger (longer and wider) shoots at low density (Krause-Jensen et al. 2000). Where eelgrass is under stress from a high-energy environment it tends to produce thin, small shoots in high density clumps (Olesen and Sand-Jensen 1994). The eelgrass on Flora Bank is shorter and narrower than other eelgrass in adjacent but more sheltered areas of the Skeena estuary (Stantec 2015) suggesting that currents, wind, and wave energy are a more dominant influence than turbidity and low light levels. The eelgrass beds on Flora Bank have sometimes been described as "patchy", another characteristic of eelgrass growing in high-energy environments (Murphey & Fonseca 1995).

While patchy eelgrass beds with short and narrow shoots have been dismissed as low value habitat (Stantec 2015), research has shown that these eelgrass beds from high-energy environments have similar densities of rearing juvenile shrimp and larger on average shrimp when compared to low-energy beds (Murphey & Fonseca 1995). Patchy eelgrass may also result in higher survival of juvenile crabs than dense eelgrass (Hovel & Fonseca 2005). It is hypothesized that the physical characteristics of these high-energy eelgrass shoots allow for more efficient predation of benthic invertebrates by predatory fish (Irlandi 1997). This efficient feeding habitat may explain the high densities of juvenile salmon found in the Flora Bank region compared to other nearby eelgrass beds that have low-energy growth patterns (Moore et al. 2015).

The possible high-energy environment of Flora Bank has previously raised concerns regarding erosion and its impact on eelgrass mortality (Stantec 2014). Exposed rhizomes likely caused by erosion were observed in some eelgrass beds on the northwest side of Flora Bank. At several sites on the southeast edge of Flora Bank the eelgrass was primarily on hummocks of slightly elevated substrate, which is also common in high-energy environments where substrate not protected by eelgrass is eroded to a lower elevation (Murphey & Fonseca 1995). However, seagrass in general, including other *Zostera* species, appear to be generally robust to deep erosion (Cabaço et al. 2008) and the eelgrass rhizomes may prevent sediments from eroding (Plummer et al. 2013). *Zostera* species are much more sensitive to burial by fine sediments, which is less likely in a high-energy environment (Cabaço et al. 2008). No evidence of either deep, large-scale erosion or burial was noted on Flora Bank.

While the low percentage of flowering shoots would suggest low general reproductive effort, eelgrass often reproduces asexually through rhizomes so flowering does not capture all reproductive effort. Previous mapping of eelgrass on Flora Bank via remote sensing has shown eelgrass beds rapidly expanding into new areas, suggesting that the population is capable of effective reproduction if suitable habitat is available (Stantec 2015).

The biological characteristics of the eelgrass surveyed are consistent with a healthy eelgrass population in a high-energy environment. No observations were made that would suggest environmental stress or other factors that would support the 59% population reduction suggested by some remote sensing results. The estimated total area of eelgrass bed was 0.68 km^2. Although smaller than the initial 1997 estimate (0.80 km^2), this estimate is much greater than the 2013 0.33 km^2 estimate that had prompted concerns about severe population decline. While the current survey provides useful information, the small proportion of Flora Bank surveyed relative to its total area means it cannot be used to confidently determine which, if any, of the remote sensing estimates are accurate. As there is no current evidence to corroborate the extreme decline or to suggest a cause for a decline, no restoration efforts are recommended at this time. A systematic long-term monitoring program for the eelgrass on Flora Bank is recommended to confirm that the eelgrass population is stable and to identify potential impacts from industrial disturbances. Recommendations for future monitoring of eelgrass health on Flora Bank include:

- Conduct yearly remote sensing surveys of the overall eelgrass coverage of Flora Bank to determine long-term trends in eelgrass bed extent. These surveys should use consistent methodology to avoid introducing additional variation to the estimates.
- The field-based eelgrass monitoring should occur on an annual or biennial basis, visiting the

same sites and using the same methodology. This will provide information on inter-annual variation and long-term trends in bed form on a micro scale and identify changes in health through comparison of biological and reproductive data.

- Additional monitoring sites should be established in any areas in which specific impacts to eelgrass are of concern (e.g. near industrial infrastructure or at the margins of eelgrass-covered areas).

Section 2.2.2 - Key Findings

1) Eelgrass beds on Flora Bank are typical of eelgrass growing in high-energy environments and were generally patchy, containing less eelgrass than bare substrate.

2) Eelgrass density, percent cover, length, width, and photosynthetic area was not related to geography on the bank.

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3. JUVENILE SALMON

Estuaries are transition zones where young salmon graduate from freshwaters to the sea. All Skeena salmon must transit the estuary twice during their life cycle: as juveniles, when they migrate to the sea as smolts, and as adults, when they return to freshwaters to spawn. In the estuary, juvenile salmon undergo a physiological transformation in order to tolerate saltwater, and during this time they also must eat, grow, and avoid being eaten by predators. Although it is thought to be a key stage regulating population dynamics, this estuarine phase of the salmon life-history is more poorly understood than other phases (Weitkamp et al. 2014a). Thus, our research can help illuminate knowledge gaps on a critical phase of the salmon life-cycle.

Estuaries provide habitat for young salmon with good feeding opportunities and protection from predators. Near-shore habitat with benthic vegetation, such as eelgrass and kelp, are important feeding areas, presenting unique food sources not found in other estuary habitats. For example, previous research found that chum salmon in Padilla Bay, Washington primarily feed on plankton species that are only available in eelgrass habitat (Haas et al. 2002). The estuary also acts as a refugia from predators by providing enhanced cover for young salmon with vegetation, such as eelgrass and macroalgae, and murky water (high turbidity) caused by sediment carried into the estuary by the river (Macdonald et al. 1988; Semmens 2008).

The growing body of research on salmon in estuaries suggests that a number of factors may influence the suitability of estuaries for juvenile salmon. One study tagged juvenile Chinook salmon with acoustic tags and tracked how they used different types of habitats in a Washington estuary (Semmens 2008). Juvenile Chinook salmon preferred to use native eelgrass habitats compared with other habitat types. Furthermore, the individuals that used the eelgrass habitat were more likely to survive than individuals that used it less. Studies have compared the survival of salmon in estuaries that have been degraded to those that are pristine and found that a greater percentage of Chinook salmon survive in systems that are more pristine (less industrial development) (Magnusson & Hilborn 2003; Meador & MacLatchy 2014). However, much remains unknown in terms of the specific habitat attributes that define key estuary habitats for juvenile salmon as well as other estuary fishes.

In large river systems like the Skeena River, estuaries act as physical bottlenecks for all upstream salmon populations. The Skeena River estuary drains an area the size of Switzerland, and contains dozens of salmon populations, all of which must transit the estuary on the way to the ocean. Thus, every year, hundreds of millions, and in some years perhaps even more than a billion, young salmon transit the estuary (Carr-Harris et al. 2015).

During 2013–2016 and 2018, the North Coast Juvenile Salmon Monitoring Program (NCJSMP) sampled juvenile salmon in the Skeena River estuary to collect baseline data to better understand the factors that affect salmon productivity during their estuary and early marine stage of their life-cycle. This stage represents the transition point during their migration from freshwater to the ocean (Figure 3-1). Our program included collecting physical and biological data, in addition to directly sampling juvenile salmon and other fish from nearshore and offshore estuary habitats. In this report section, we use five years of data to answer questions relating to spatio-temporal abundance patterns (3.1), length of estuary residence (3.2), growth during estuary residence (3.3), and energetic status of smolts in the Skeena River estuary environment (3.4). All fish sampling methods are described below, while statistical analysis methods will be summarized in each sub-section. Much of this research has been published in scientific journals (Carr-Harris et al. 2015, 2018; Moore et al. 2015, 2016; Arbeider et al. 2019; Sharpe et al. 2019), but main findings are summarized and made publicly available in this section.

Figure 3-1. Juvenile salmon caught by purse seine net in the Skeena River estuary during the NCJSMP (photo by David Herasimtschuk of Freshwater Illustrated).

NCJSMP FISH SAMPLING METHODS

Fish communities in the Skeena River estuary were sampled from 2013–2016 and 2018 by beach seine, purse seine, and trawl gear types as part of the NCJSMP. Fish were sampled across the entire Skeena River estuary by trawl sampling in 2013 (Figure 3-2), and a smaller geographic extent (Flora Bank/Kitson Island region, see Figure 3-3) by purse and beach seine from 2014–2018. Both a historical and a recent study from our research group (Higgins & Schouwenburg 1973; Carr-Harris et al. 2015) identified that the highest abundances of sockeye, coho, and Chinook were found within the Flora Bank/Kitson Island region (IN region, Figure 3-2), thus it became the focus of several studies. Regular monitoring locations and several more in-depth studies were conducted within the IN region (Figure 3-2), which is located proximal to the Skeena River mouth. Regular monitoring sites were sampled with beach and purse seining on a weekly basis during spring and summer with the NCJSMP (Figure 3-3). Beach seining was used to sample nearshore fish communities at shoreline sites while purse seining was carried out to sample fish communities located further from shore. Data from fish sampling were used to investigate questions related to juvenile salmon (Section 3 and 4) and the estuary fish community (Section 5). Across all years and all gear types, fish were identified to species (when possible) and fork length was measured for 25 individuals from each species.

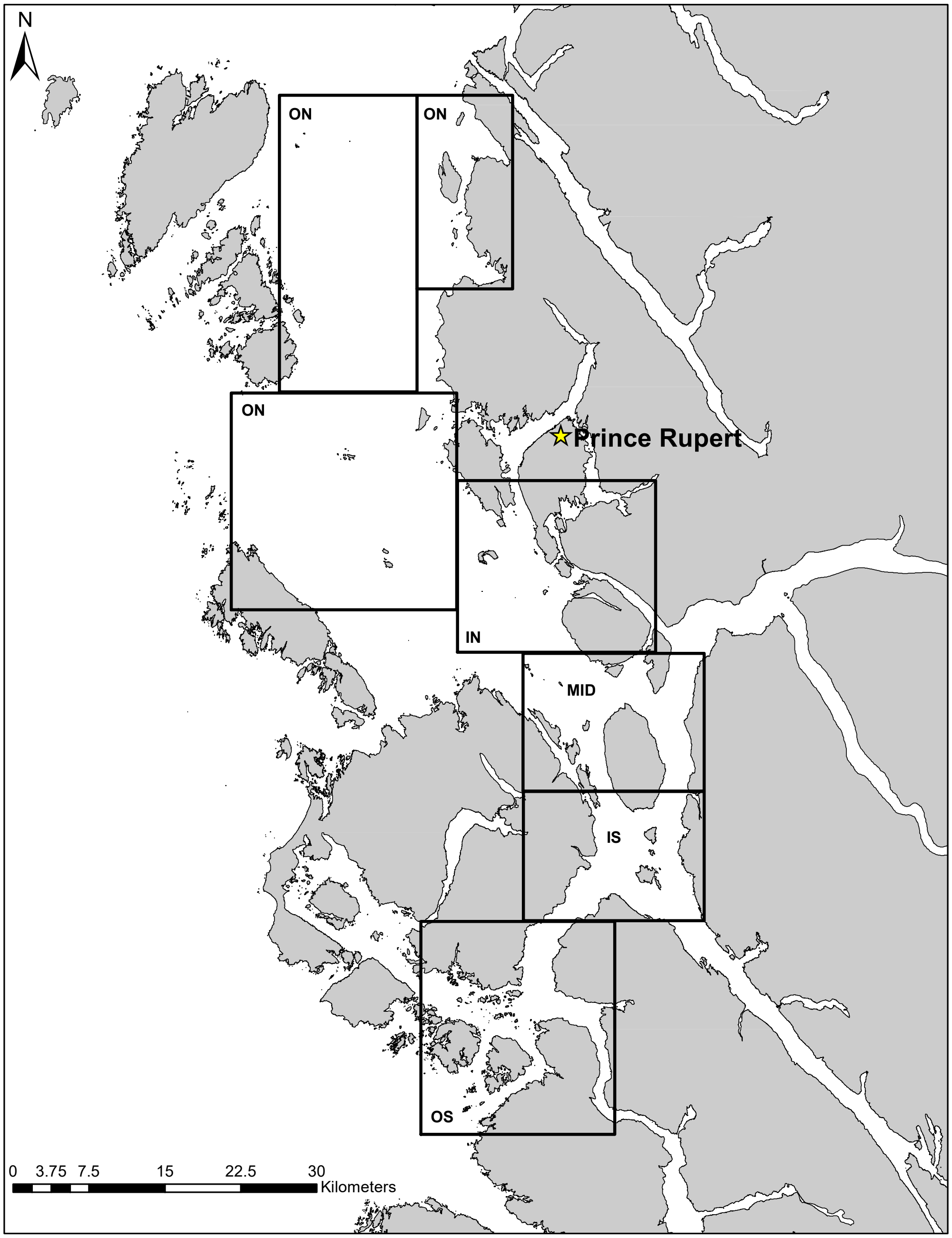

Figure 3-2. Regions sampled for fish within the Skeena River estuary. Trawl sampling occurred in all regions, while beach and purse seine sampling occurred within the IN region. The study area is shown divided into our analysis regions indicated by the letters IN for inside North, ON for outside north, MID for middle, IS for inside south, and OS for outside south. Note that the ON region includes two polygons. Figure is from Carr-Harris et al. 2015, map created by John Latimer, Lax Kw'alaams Fisheries.

The beach seine (35 m long, 3 m deep with 4 mm mesh webbing) was set using a 3 m skiff, deploying the net down current from an anchor point on the beach to enclose shoreline habitat. The beach seine was subsequently hauled into the shore from both ends, with the lead line on the bottom of the substrate to prevent fish from escaping. Beach seining occurred from April–June across all sampling years as part of the regular NCJSMP. Sampling occurred weekly at regular beach seine locations with the IN region (Figure 3-3).

Purse seine sampling occurred with two purse seine nets (Figure 3-4). A larger purse seine was used more consistently from 2014–2016 and 2018, with a smaller purse seine used to sample shallower pelagic waters in 2016. We deployed the larger purse seine (9.1 m deep by 73.2 m long, 5.1 cm webbing at the tow end, 1.3 cm webbing at the bunt) using a 3 m skiff to tow the bunt end away from a larger vessel, holding the net open into the tidal current for 5 mins per set. The smaller purse seine (5.5 m deep by 15.2 m long, 1.3 cm webbing at the tow end, 0.64 cm webbing at the bunt end) was deployed using two 3 m skiffs with the net held open for 2.5 mins per set. At the end of each set, the purse seine was closed and bagged by simultaneously pulling the purse line while hauling the web into the larger vessel. As one of the components of the regular NCJSMP, purse seine sampling occurred weekly at sites illustrated in Figure 3-3 during the salmon smolt migration season.

Trawl sampling was conducted with a chartered gillnet vessel (HMV Pacific Coast) from May–July in 2013 and 2014. The trawl net was 18 m long with an opening 5 m wide and 4.6 m deep and a rigid baffled holding box designed for live capture and sufficient floatation to maintain a position at the top of the water column while fishing. The trawl net was deployed for at least 15 mins and up to 20 mins for an approximate tow length of 1 km depending on current velocity. Trawl catches were normalized based on a trawl duration of 20 min to obtain a catch per unit effort (CPUE). Trawl sampling occurred across the extent of the estuary in all regions labeled in Figure 3-2.

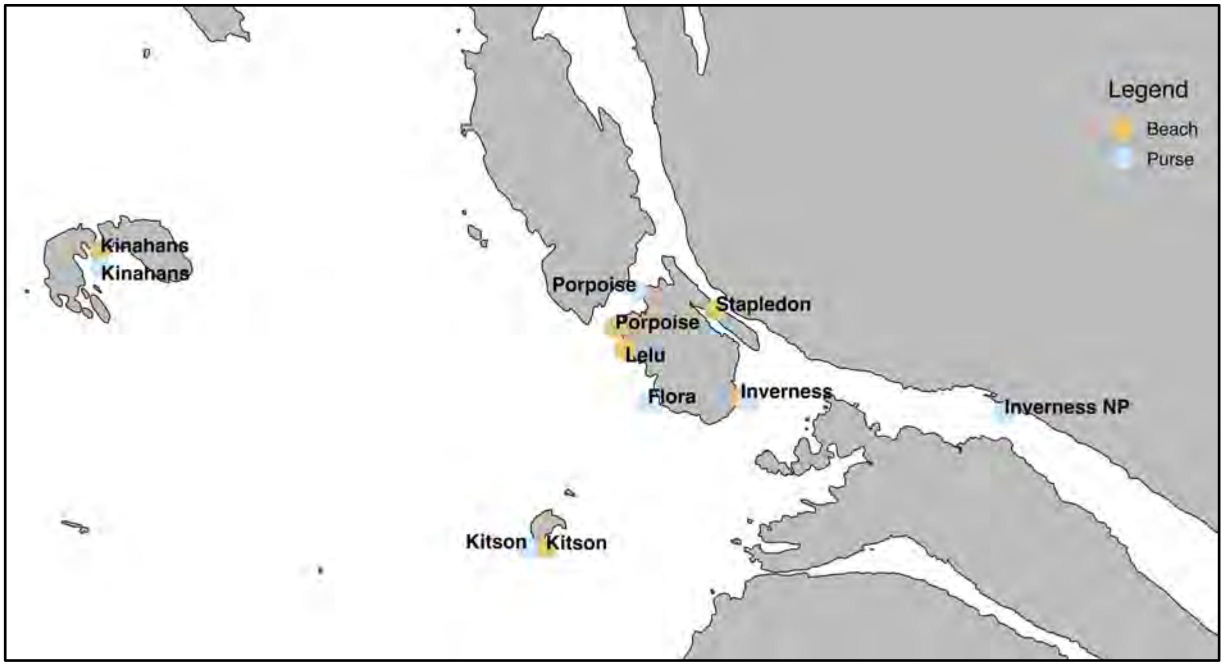

Figure 3-3. Map illustrating purse and beach seine sampling sites for the NCJSMP within the Skeena River estuary (map created by John Latimer, Lax Kw'alaams Fisheries).

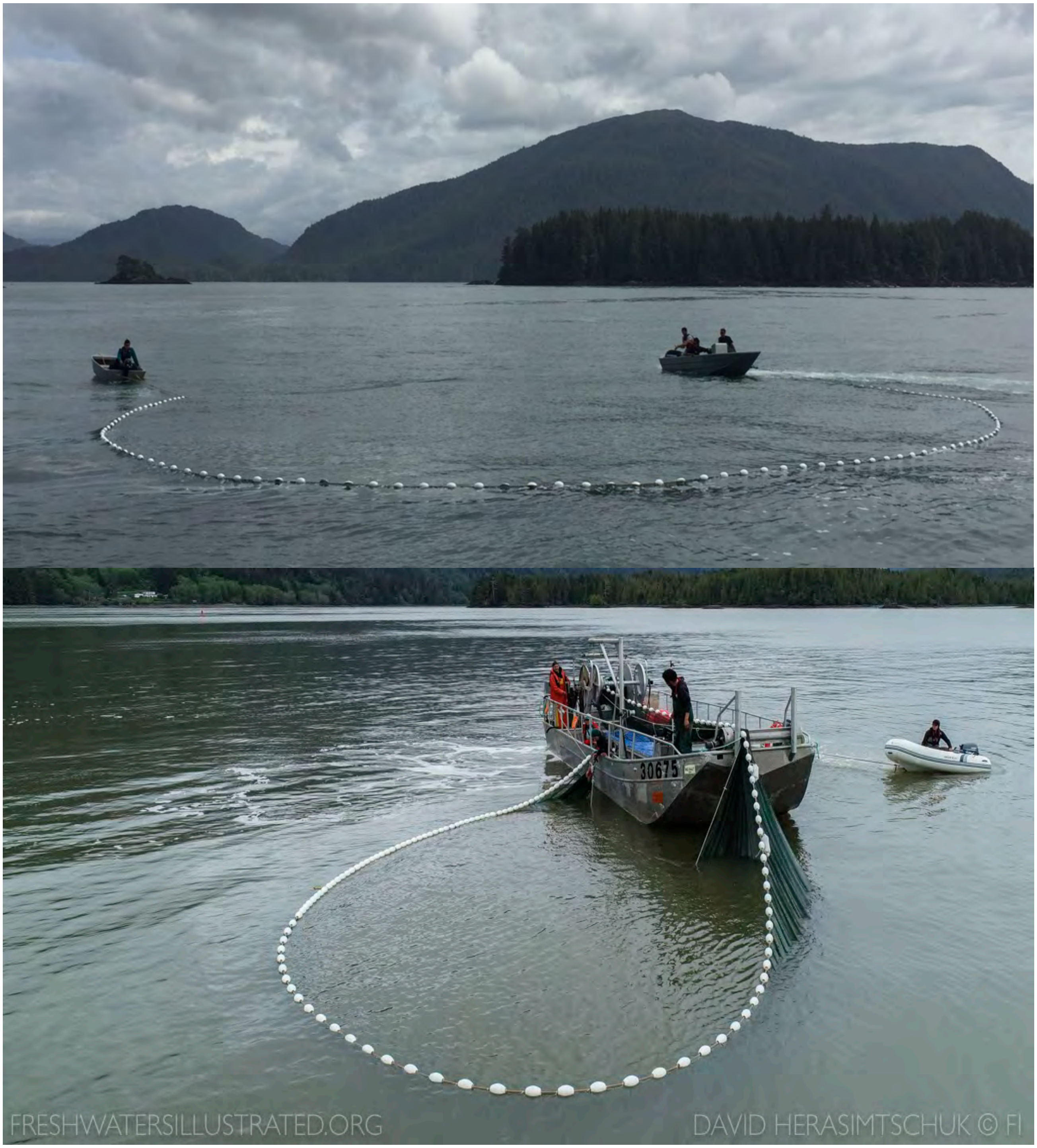

Figure 3-4. Purse seine sampling in the estuary with the small purse seine net (top panel) and large purse seine net (bottom panel) (photos by Samantha Wilson (top) and Dave Herasimtschuk from Freshwater Illustrated (bottom)).

3.1. Abundance patterns overtime of different salmon species and populations

In Section 3.1 we explore questions related to the abundance patterns of juvenile salmon during residence in the estuary. The research discussed here is the topic of Carr-Harris (2015), Moore et al. (2015), Carr-Harris (2018), and Sharpe et al. (2019). In this report section, we ask the following questions:

Q1 - How are juvenile salmon using the estuary through a) time and b) space?

Q2 - Can spatial patterns of use be explained with different habitat and biophysical factors?

Q3 - What populations of salmon are found transiting through the estuary?

Q4 - Does the timing of the estuary life-history phase differ between populations?

Q5 - How long does a juvenile salmon take to migrate from freshwater rearing habitat to the estuary?

Q1 - How are juvenile salmon using the estuary through a) time and b) space?

A) Temporal abundance patterns

The results from NCJSMP fish sampling indicate that the Skeena River estuary supports diverse and abundant populations of juvenile salmon. During our four years of purse seine and beach seine sampling, we found that the different species of juvenile salmon occupied the estuary from April until at least the end of our sampling period in the middle of July. However, abundance patterns of the different salmon species varied within this smolt outmigration period. While intra-annual patterns of abundance varied by species, overall patterns of abundance among species were consistent for our three years of sampling.

High abundances of juvenile pink salmon were observed during early-season beach seine sets, from the middle of April until the middle of May, subsequently diminishing in abundance for the remainder of the year (Figure 3-5). The highest abundances of juvenile chum salmon were captured by beach seine from the end of April until the middle of June in all years and smaller numbers of juvenile chum salmon were captured into July (Figure 3-5). Chum abundances were highest in 2016 compared to other sampling years (Figure 3-6). Juvenile coho salmon were captured in high abundances by beach seine, purse seine, and trawl from the middle of May onward (Figure 3-7). Juvenile Chinook salmon were captured primarily by purse seine and trawl from the middle of May onward in all years, with higher abundances observed in 2018 than previous sampling years (Figure 3-7 and Figure 3-8).

As with Chinook salmon, juvenile sockeye salmon were mostly captured by trawl and purse seine and were the most abundant salmon species captured by these offshore gear types in all years (Figure 3-8). Sockeye salmon were continually present in the study area from early May until the end of sampling in mid-July, with peak abundances observed between the last week of May and the first week of June in all years (Figure 3). The lowest abundances of juvenile sockeye salmon were observed in 2015, which was expected due to exceptionally low sockeye salmon returns to the Skeena River in 2013, the dominant brood year for the sockeye salmon smolts that we observed in the estuary in 2015 (Figure 3-8).

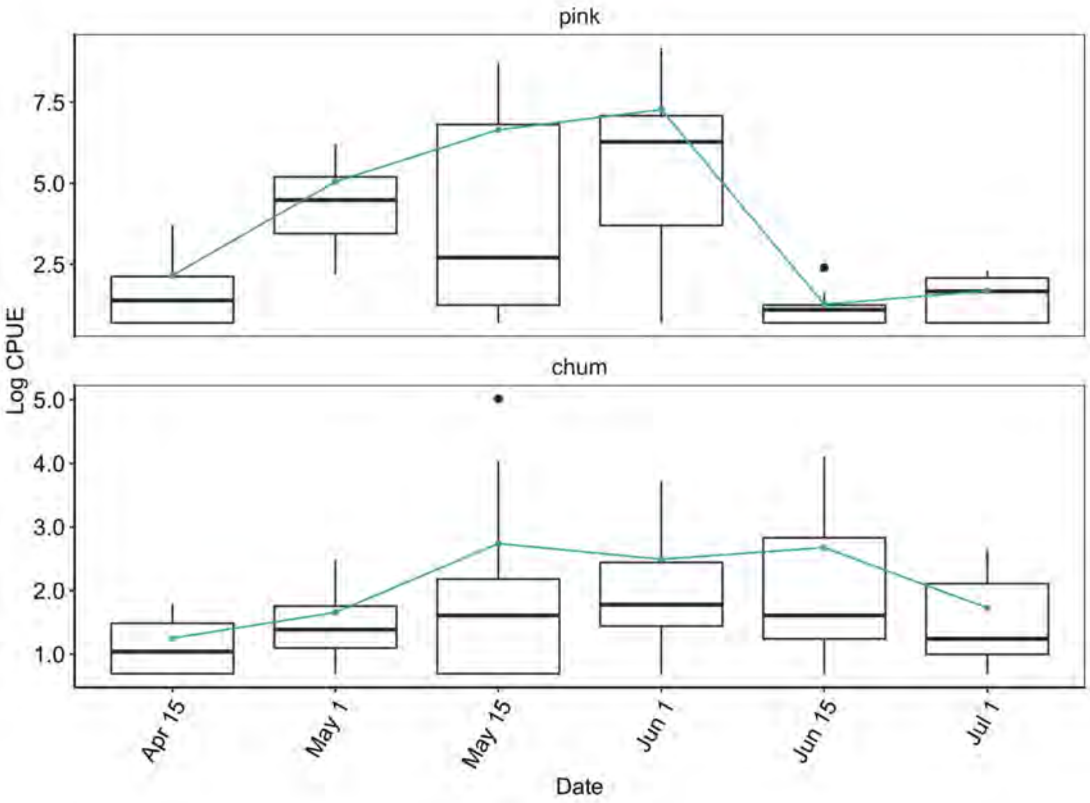

Figure 3-5. CPUE for pink and chum salmon species over time from regular beach seine sites (Flora Bank region, Figure 3-3) during 2014–2018 sampling. The solid black lines indicate median CPUE for each sampling period, while box boundaries indicate first and third quantiles and whiskers indicate the highest and lowest values of CPUE. Mean CPUE for each time period is overlaid with a blue dot and connected by blue lines to further visualize trends.

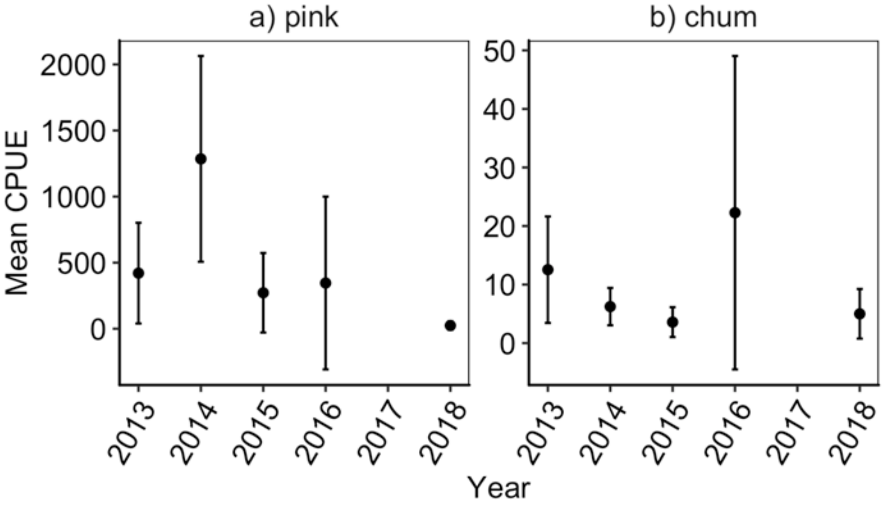

Figure 3-6. Mean CPUE with 95% confidence intervals for pink and chum salmon from sampling years 2014–2018 at regular beach seine sites.

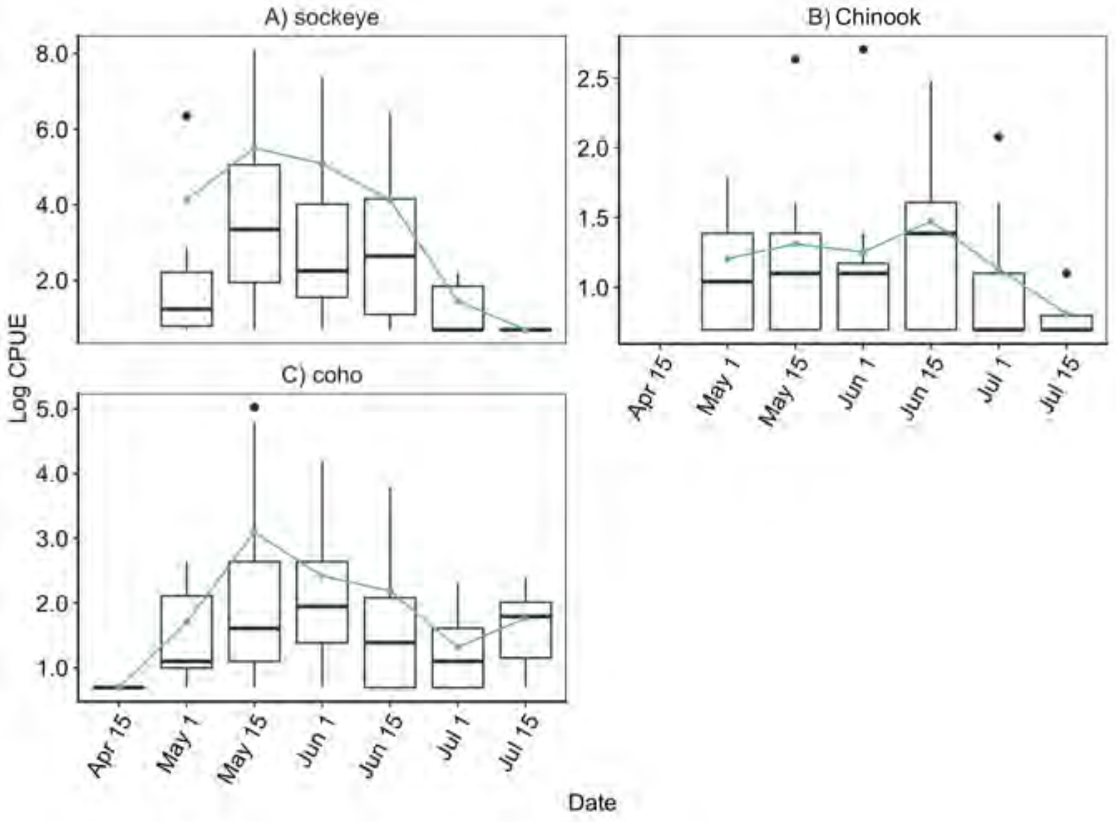

Figure 3-7. CPUE for dominant pelagic species A) sockeye, B) Chinook, and C) coho over time from 8 regular purse seine sites during 2014–2018. The solid black lines indicate median CPUE for each sampling period, while box boundaries indicate first and third quantiles and whiskers indicate the highest and lowest values of CPUE. Mean CPUE for each time period is overlaid with a blue dot and connected by blue lines to further visualize trends.

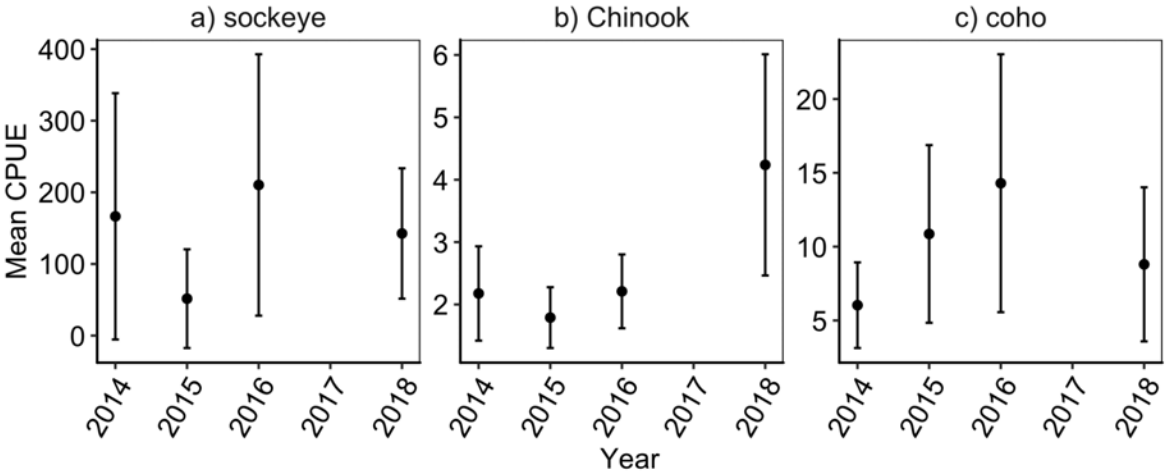

Figure 3-8. Mean CPUE with 95% confidence intervals for salmon species sockeye, Chinook and coho from sampling years 2014–2018 at regular purse seine sites.

B) Spatial Abundance Patterns

We looked at spatial abundance patterns for juvenile salmon species in the Skeena River estuary over two scales: a) the entire extent of the estuary (Figure 3-2) and, b) the IN region proximal to the Skeena River mouth (Figure 3-3). This section of the estuary was selected for regular monitoring and more in-depth analysis due to the high abundance of sockeye, coho, and Chinook salmon identified in historical (Higgins & Schouwenburg 1973) and recent findings by our research team (Carr-Harris et al. 2015).

Entire Estuary Extent

We analyzed trawl sample data collected during the NCJSMP in 2013 and by Skeena Fisheries Commission in 2007 to investigate the spatial distribution of different species of juvenile salmon throughout the Skeena River estuary. Trawl sites were aggregated into broad regions throughout the estuary according to their relative proximity to the northern or southern exit of the Skeena River (Figure 3-2). The 2007 trawl sampling program encompassed five regions (Inside North (IN), Outside North (ON), Middle (MID), Inside South (IS), and Outside South (OS)), and the 2013 program encompassed three of the five regions that were sampled in 2007 (IN, IS, and OS). Methods for fish sampling by trawl seines are described in Section 3.1.

We analyzed trawl catch-per- unit-effort (CPUE) for sockeye, coho and Chinook salmon with generalized additive models (GAM) to compare abundances across regions. These models examine the relative effect of each region on catch rate, after controlling for seasonal patterns in abundance for each species. Additional details about the statistical analysis can be found in the open access publication Carr-Harris et al. 2015.

We observed the highest densities of juvenile sockeye salmon in both years, juvenile coho salmon in 2013, and juvenile Chinook salmon in 2007 in the IN region (Figure 3-9). GAM statistically indicated that juvenile sockeye salmon were most abundant in the IN region in both years, and juvenile coho salmon were most abundant in the IN region in 2013 (Figure 3-10). The β coefficient for sockeye in the IN region was 1.74 + 0.36 ($p < 0.0001$, this and the following represent the best estimate of the coefficient + 1 SE and P of the coefficient) in 2007 and 1.56 + 0.34 ($p < 0.0001$) in 2013 (Figure 3-10). The predicted abundances for sockeye salmon in the IN region were 2–8x higher than in the other regions in both years. The β coefficients for coho salmon in the IN region were 0.63 + 0.28 ($p = 0.0262$) in 2007 and 0.45 + 0.19 ($p = 0.022$) in 2013 (Figure 3-10). Thus, predicted abundances for coho salmon were 2–7x higher in the IN than in other regions in 2013, and 2–7x higher in the IN and MID regions than in other regions in 2007. Chinook salmon appeared to be most abundant in the IN region in 2007 and in the IS region in 2013, however neither of these values were significant ($p < 0.05$).

Abundance patterns in the Flora Bank – Prince Rupert region (IN region)

Abundances of sockeye and coho salmon were consistently higher in the IN region compared with other regions in the two years sampled, suggesting that this region contains important rearing areas for out-migrating salmon smolts (Carr-Harris et al. 2015). These data provide evidence that while the Skeena River estuary in general contains high abundances and diversity of juvenile salmon, the Flora Bank region of the estuary contains some of the highest densities of the most ecologically and economically important species of Skeena River salmon.

In 2015 and 2016, we investigated the usage of different habitats by salmon at a more local scale. We used small and large purse seine nets to sample juvenile salmon during the spring–summer season (Figure 3-11). We sampled with a large purse seine at 25 sites every two weeks, of which a subset were sampled every week, while a smaller purse seine was used to survey habitat closer to shore every two weeks at an additional 18 sites. We used CPUE as a measurement of fish abundance to include catch from both nets in our analysis. While the CPUE values of the large seine net remained equal to the original catch data, relative abundance from the smaller seine net was standardized to the larger purse seine. We calculated CPUE for the small net by multiplying catches by the large net area (length by width) and net tow duration, and then dividing by the area and tow duration of the small net. Methods for fish sampling by small and large purse seines are described in Section 3.1.

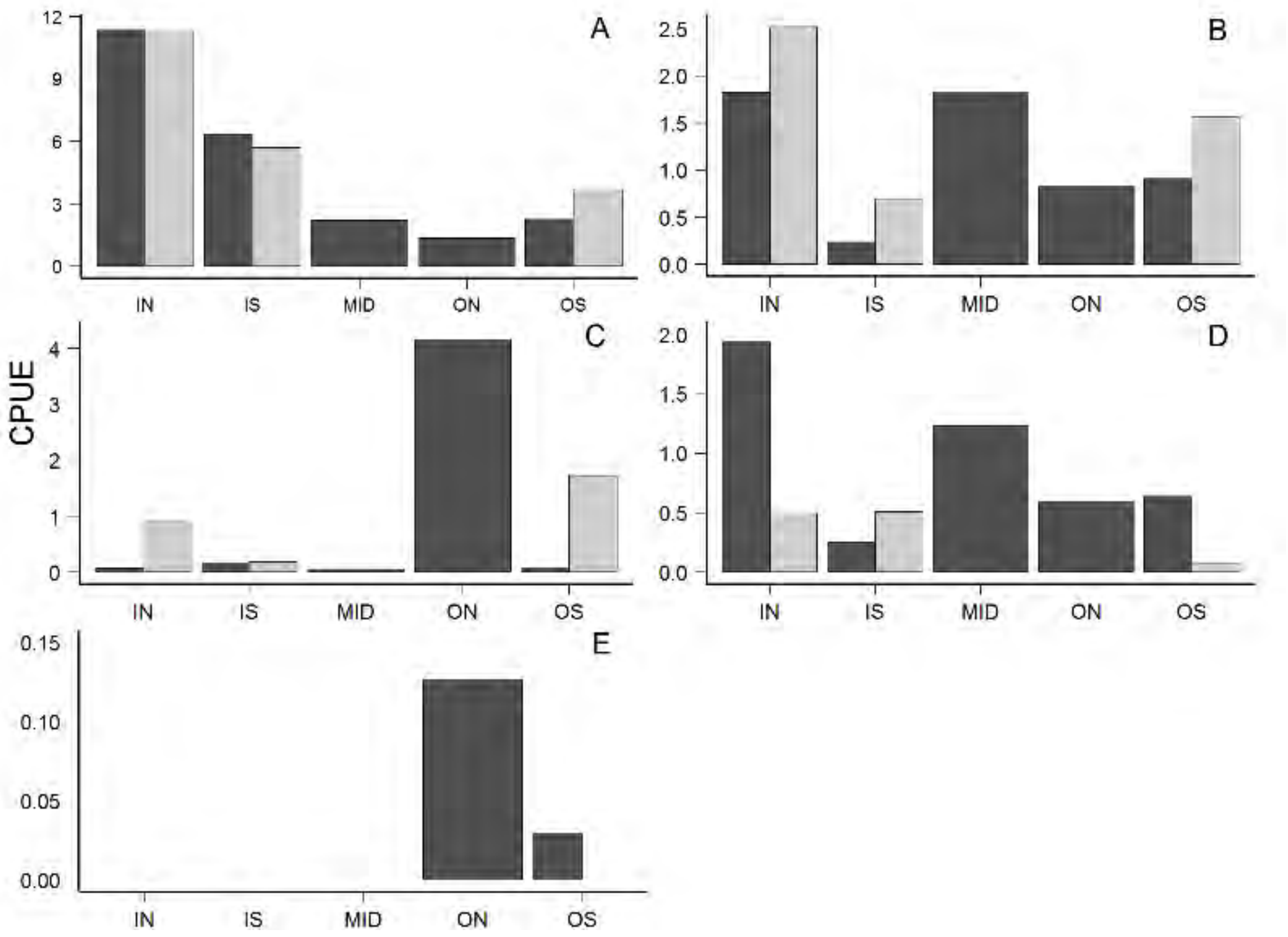

Figure 3-9. Average normalized trawl catch of juvenile sockeye (a), coho (b), pink (c), Chinook (d), and chum (e) salmon, pooled across all locations and sampling dates and normalized for 20 min sets. Dark grey bars indicate 2007 and light grey bars indicate 2013. Note different scales for y-axes for different species (Carr-Harris et al. 2015).

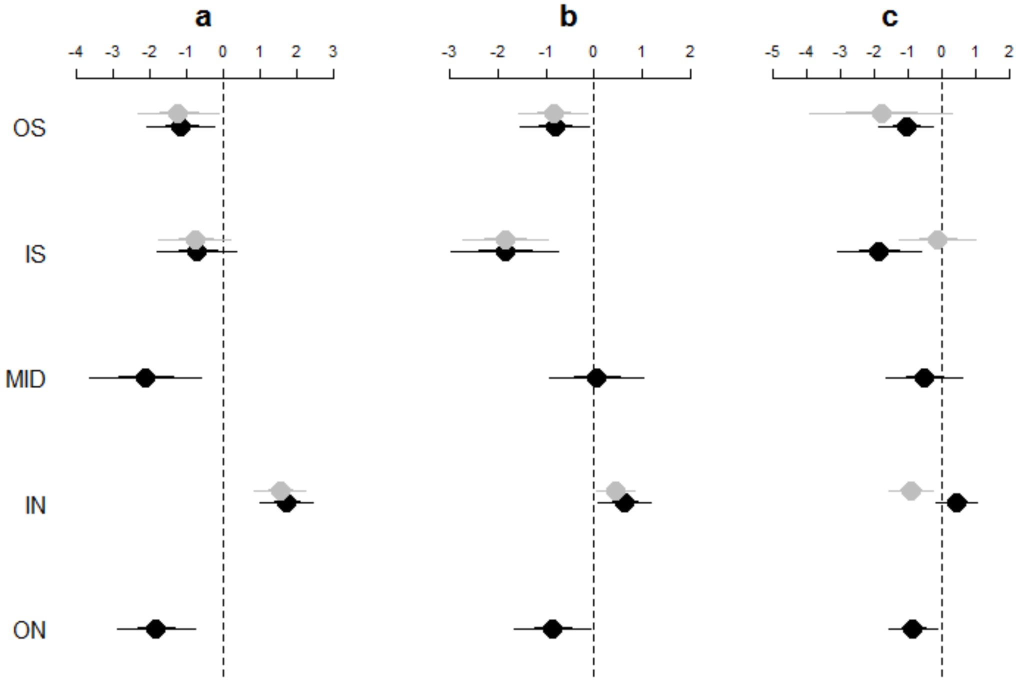

Figure 3-10. GAM coefficients for parametric region covariates for sockeye (a), coho (b), and Chinook (c) salmon. Coefficients are related to the (log) mean normalized catch per trawl set for each region in 2007 (black) and 2013 (grey). A value of 0 indicates a mean normalized trawl catch of 1 and error bars indicate +2 standard errors (Carr-Harris et al. 2015).

Sites were selected to represent different habitat types, such as eelgrass habitats and off-shore habitats. These data were collected to understand whether some locations in the greater Skeena River estuary support more juvenile salmon than others. We focused on juvenile coho, sockeye, and Chinook salmon because pink and chum salmon appear to reach peak abundances in the estuary earlier than we sampled and are often found in shallow nearshore habitats that were not sampled as effectively by our deep purse seine.

Within sampling years, fish were unevenly distributed across the estuary with higher catches (CPUE) of salmon consistently found at some sites compared to others (Figure 3-12 - Figure 3-14). Specifically, the relative abundance of all salmon species during the peak smolt out-migration (May 8–June 5) was highest at sites surrounding Lelu Island, including Flora Bank and Kitson Island. Comparing the grouped mean CPUE (using both the large and small purse seine) of sites around Flora Bank and Kitson Island (including Porpoise Harbour, Inverness Passage, and Agnew Bank) to elsewhere in the estuary illustrates these differences in salmon abundances across space. In 2015, CPUEs for sockeye, coho, and Chinook salmon were 8, 5, and 3 times higher, respectively, for the Flora Bank region compared to other sites on average, and 2, 3, and 19 times higher, respectively, in 2016. At the individual site level, we found the highest abundances of salmon at Flora Bank site in both years using the large purse seine. On average, the CPUE of sockeye, coho, and Chinook salmon in 2015 were 38, 8, and 5 times higher, respectively, at the Flora Bank large purse seine site alone compared to other sites during peak migration. Higher abundances were also found at the Flora Bank large purse seine site in 2016, where we collected 5, 9, and 2 times more sockeye, coho and Chinook salmon, respectively, on average compared to other large purse seine sites in the estuary.

These results suggest that all salmon species are using some sites more than others across the estuary. In particular, three salmon species are found to be using habitat around Flora Bank more consistently than others. This is an indication that this region is highly used by salmon and captures one metric of habitat “importance”, but does not capture other aspects such as residency, growth, and survival during the estuarine life-history stage. We discuss residency and growth of juvenile salmon species found in the Skeena River estuary in the Sections 3.2 and 3.3.

Section 3.1 - Q1 - Key Findings

1) The different species of juvenile salmon occupied the estuary from April until at least the end of our sampling period in the middle of July, with peak abundance of the different salmon species varying throughout this period.

2) The Flora Bank region on the north side of Inverness Passage has approximately twice as many juvenile coho, Chinook, and sockeye salmon compared to other regions across the Skeena River estuary extent.

3) Within this high density Flora Bank region, we caught more salmon at the site located on Flora Bank compared to other sites nearby (Inverness and Porpoise Channels, etc). At this Flora Bank site, we caught 38, 8, and 5 times more sockeye, coho, and Chinook compared to all other sites in 2015, and 5, 9, and 2 times for more sockeye, coho and Chinook in 2016.

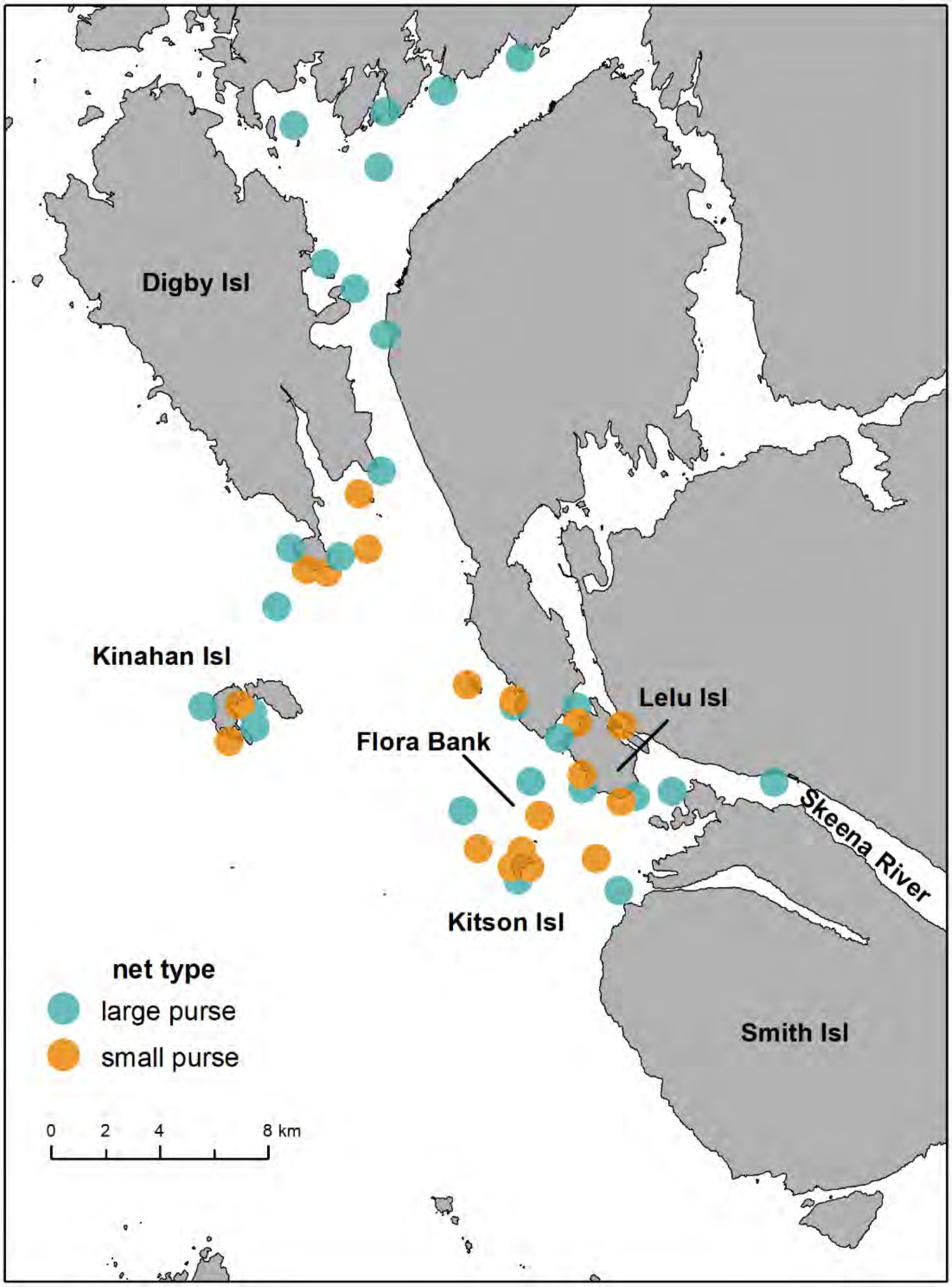

Figure 3-11. Purse seine locations for the large (blue) and small (orange) purse seine nets in 2015 and 2016 used to investigate herring and smelt associations with abiotic and biotic variables.

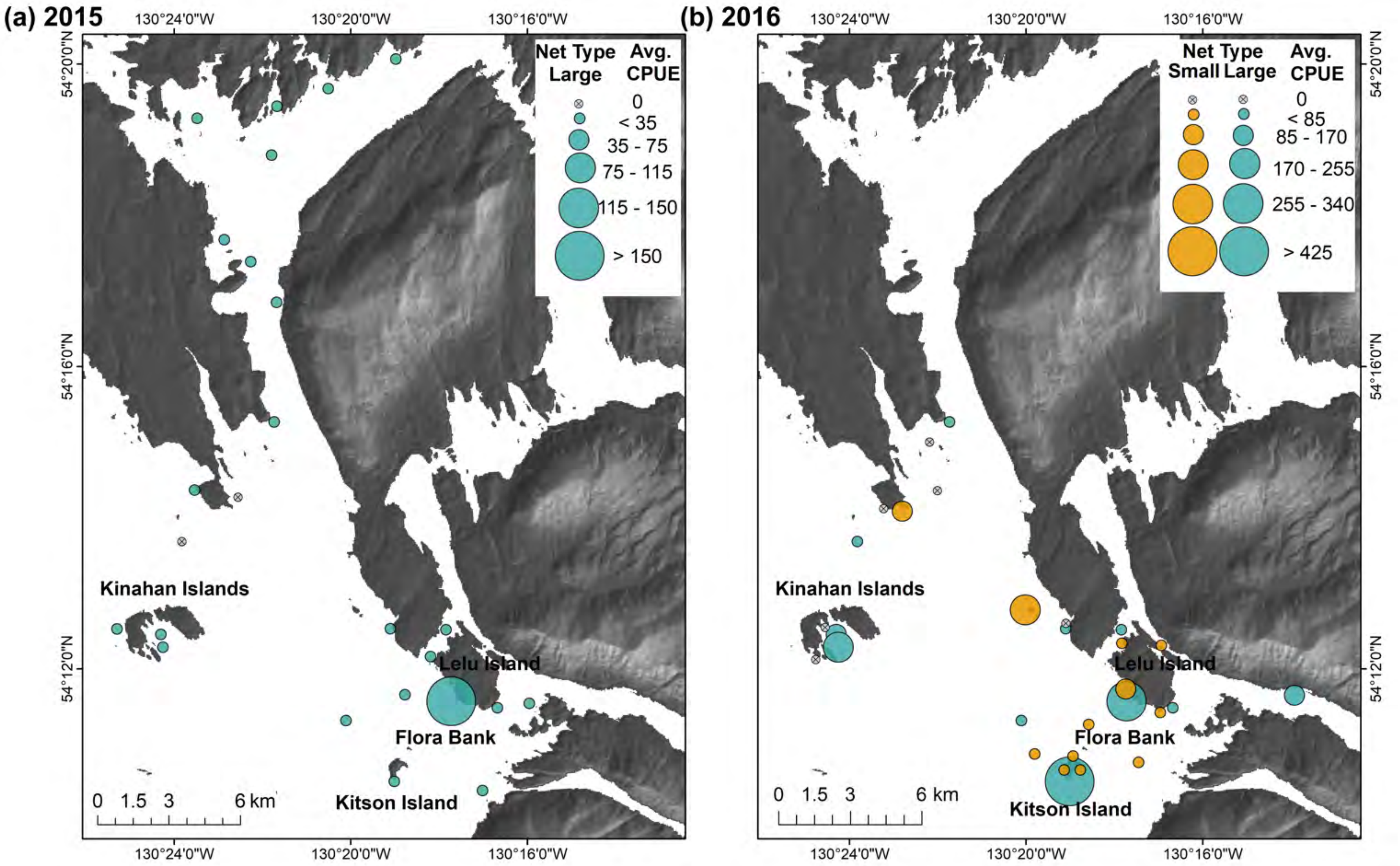

Figure 3-12. Mean sockeye salmon abundance during peak migration (May 8–June 5) at sampling locations in the Skeena River estuary in a) 2015 and b) 2016. Colours indicate the different net types (orange = small purse seine, blue = large purse seine). Point area represents average sockeye salmon CPUE abundance in 2015 and 2016.

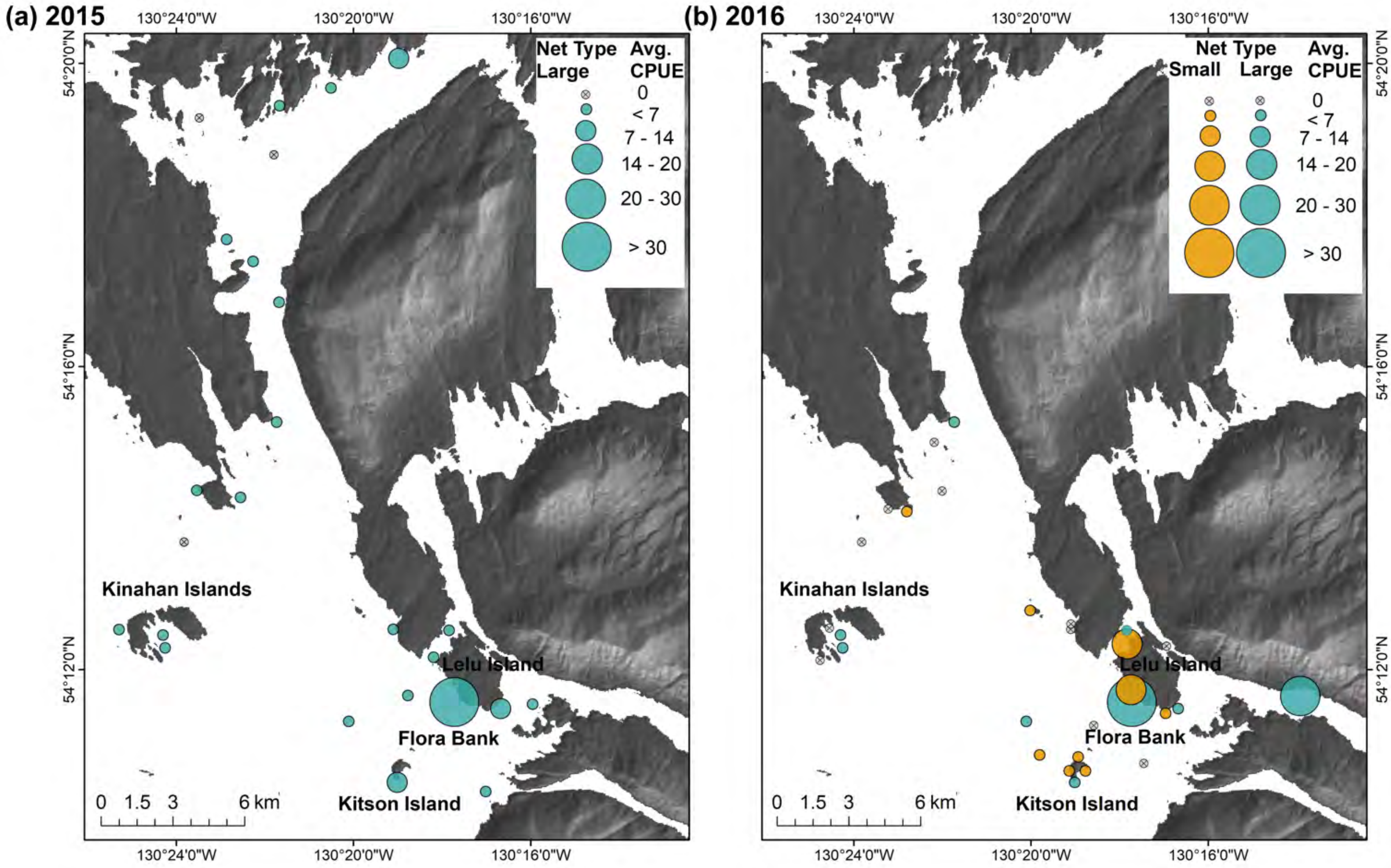

Figure 3-13. Mean coho salmon abundance during peak migration (May 8–June 5) at sampling locations in the Skeena River estuary in a) 2015 and b) 2016. Colours indicate the different net types (orange = small purse seine, blue = large purse seine). Point area represents average coho salmon CPUE abundance in 2015 and 2016.

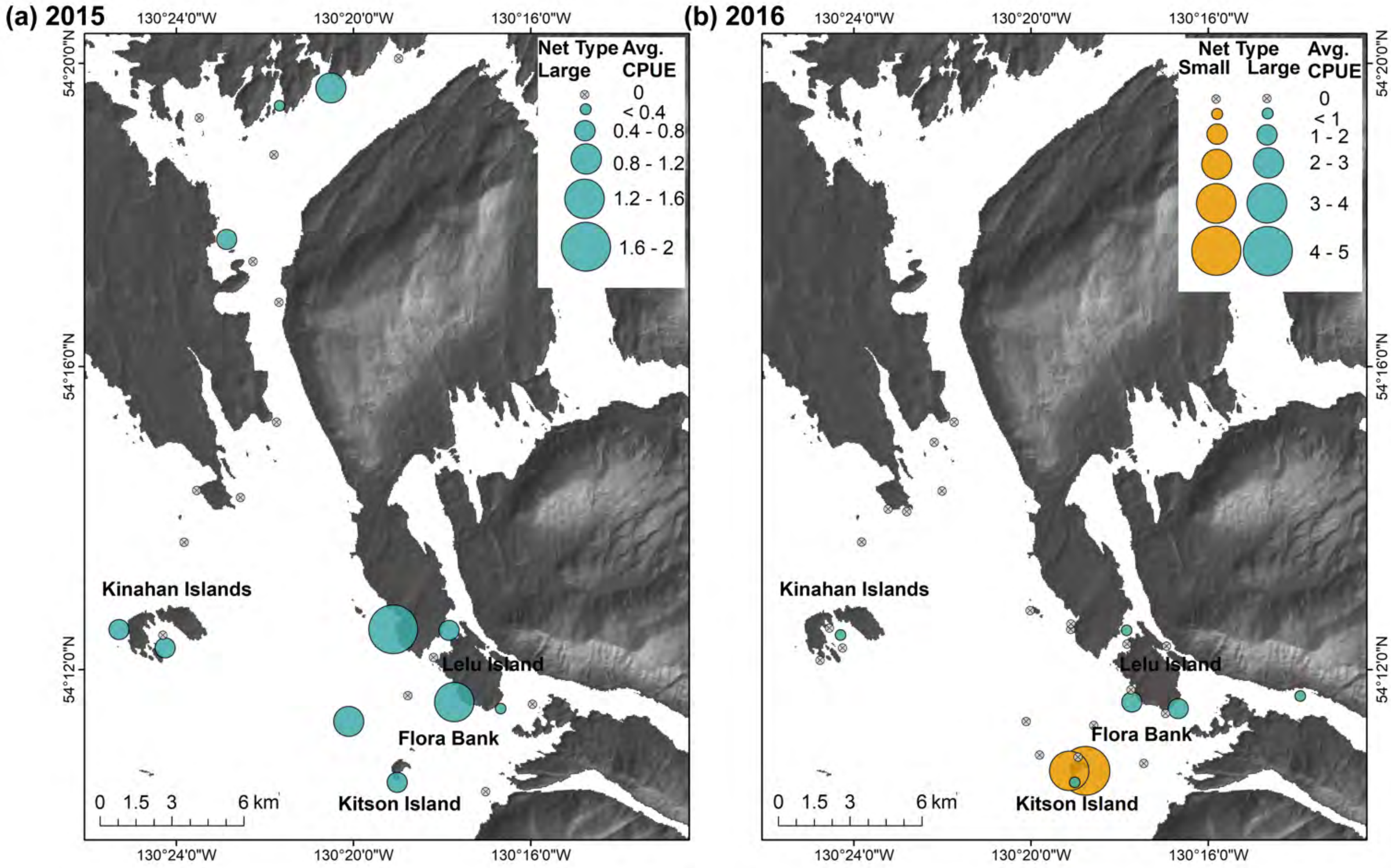

Figure 3-14. Mean Chinook salmon abundance during peak migration (May 8–June 5) at sampling locations in the Skeena River estuary in a) 2015 and b) 2016. Colours indicate the different net types (orange = small purse seine, blue = large purse seine). Point area represents average Chinook salmon CPUE abundance in 2015 and 2016.

Q2 - Can spatial patterns be explained with different habitat and biophysical factors?

We investigated abundance patterns of juvenile salmon species (sockeye, coho, and Chinook) within the Skeena River estuary across various habitat types, such as eelgrass and rocky shores, and biophysical factors, such as salinity, turbidity, and spatial habitat distribution. A suite of different abiotic and biotic factors have been strongly associated with structuring abundance and distribution of estuary fish communities (Marshall & Elliott 1998; Wagner & Austin 1999; Harrison & Whitfield 2006). Investigating the relationships between fish abundance and various abiotic and biotic factors can advance our understanding of fish distribution patterns in the estuary. This question was the topic of Sharpe et al. (2019), which investigated these fish-habitat associations for herring and surf smelt along with two salmon species (sockeye and coho salmon). We summarize some of the methods and results for juvenile salmon below. Additional details of sampling and statistical analysis can be found in the open access publication Sharpe et al. 2019.

METHODS

We sampled fish in the Skeena River estuary during the peak salmon smolt migration from April to mid-July in 2015 and 2016 with the large and small purse seine nets (Figure 3-11). Fish sampling occurred at four of the most abundant habitat types in the Skeena River estuary (Levings 2016): sandy banks, rocky shores, open water, and eelgrass beds.

Subsequent to each sampling event, we collected data for various habitat and environmental variables known to influence estuarine fish habitat utilization including water quality and tidal variables, along with vegetative and spatial attributes (Bacheler et al. 2009). Salinity (ppt), temperature (°C), and turbidity (Secchi disk depth) were measured after each set at a depth of one meter with a YSI. For each site, we determined the distance from the river, proximity to the shore, and tidal height from the Prince Rupert tide station (Station Number 9354, 54.317°N, 130.324°W) located within our study area. This data was used to generate tide height and tide phase. We surveyed benthic substrate along three parallel 20 m transects at each site with an underwater camera (HD Sea-Drop 1080p Sea-Viewer Inc.) and video footage was analysed to determine percent cover of substrate (sediment, macroalgae, and eelgrass). Percent cover was estimated by overlaying 30 points per frame on 20 still frames per transect using the program Coral Point Count with Excel extensions (CPCe) (Leonard & Clark 1993; Ninio et al. 2003; Kohler & Gill 2006; Guinan et al. 2009).

Statistical analysis investigated which combination of abiotic and biotic variables was most effective at explaining fish abundance across the estuary using multi-modal inference. Sockeye, coho, and Chinook salmon abundance patterns (CPUE) were modelled across the estuary as a function of various habitat features and biophysical factors using negative binomial generalized linear mixed effect models (R package glmmTMB, Magnusson et al. 2016). The importance of each predictor variable was quantified based on cumulative Akaike weight of the set of candidate models (ΔAIC_c less than two), creating a measure of Relative Variable Importance (RVI) (Burnham & Anderson 2002). Results are presented in a coefficient plots below (Figure 3-15 and Figure 3-16) with dots that represent parameter estimates with 95% confidence intervals from averaged models. The larger the confidence intervals, the higher the standard error associated with the coefficient. In addition, if the confidence intervals cross 0 (dotted line), the estimated value has large uncertainty which likely indicates that there is no strong effect. RVI values are listed beside each estimate and are an indication of how important each variable is to explaining fish abundance in relation to the other variables. For example, if an RVI value is 1, it indicated that this variable was in all models (100%) with higher statistical support.

RESULTS

We found that the biophysical factors that best predicted salmon abundance differed among salmon species. Juvenile sockeye salmon abundance was related to environmental conditions with higher temperatures and lower water visibility (higher turbidity) (Figure 3-15). Environmental variables temperature and turbidity had RVI values of 1.00 (Figure 3-15) which suggests that they were relatively important in explaining abundance of sockeye salmon compared to the other variables. Juvenile coho salmon and Chinook salmon were caught more frequently at locations closer to shore with increased macroalgae cover. In addition, coho salmon were associated with lower water visibility, while Chinook salmon was associated with warmer water temperature in the estuary. We also found juvenile sockeye and Chinook salmon more abundantly at eelgrass habitats compared to other habitat types sampled (sandy banks, rocky shores and open-water habitats, Figure 3-16).This aligns with previous research that has shown these environmental attributes influence juvenile salmon distribution across estuaries (Straty & Jaenicke 1980). Coho and sockeye had an affinity for locations with lower water visibility (Figure 3-15). This result supports the notion that turbidity in estuaries is key to providing increased refuge from visual predators by decreasing visibility, thus reducing the chance of being seen and eaten (Blaber & Blaber 1980; Phillips et al. 2017). We also found temperature to be an important variable in explaining sockeye and Chinook abundances – higher juvenile sockeye and Chinook salmon abundance was associated with higher temperatures (Figure 3-15). Given that fish can detect temperature gradients as small as 0.03°C (Murray 1971), selecting for warmer temperatures (within an optimum temperature range) may allow fish to capitalize on the productive food supply in estuaries by increasing scope for growth (Javaid & Anderson 1967; Straty & Jaenicke 1980). Increased growth rates during estuary rearing produce larger fish with an increased chance of survival in marine environments (Beamish et al. 2004; Duffy & Beauchamp 2011). During juvenile salmon migration, freshwater from the Skeena River is colder than the marine environment, creating a temperature gradient across our study region (strongest during snow-melt freshet in May, ranging from 9.1°C–12.6°C). As juvenile salmon did not show distribution patterns in relation to distance to the river, sockeye and coho salmon may have an affinity for temperature at a site level as opposed to a larger geographical gradient across the estuary.

Vegetation attributes were also key in predicting salmon abundance across the estuary. Juvenile coho and Chinook salmon were caught more frequently at locations closer to shore with increased macroalgae cover, while juvenile sockeye and Chinook salmon were found more frequently on eelgrass banks compared to other habitat types (open-water, rocky shores and sandy banks, Figure 3-16). Eelgrass is commonly cited as being an important habitat for estuary fish, including juvenile salmon, as it contributes to high prey diversity and has been shown to reduce predation rate (Gregory & Levings 1996; Heck et al. 2003; Semmens 2008). Juvenile Chinook salmon have demonstrated both neutral and strong association with eelgrass habitats in other estuaries (Hosack et al. 2006; Semmens 2008; Dumbauld et al. 2015; Rubin et al. 2018). Furthermore, an analysis of diets of juvenile salmon from the Skeena River estuary found that harpaticoid copoepods, a benthic prey item found in association with eelgrass habitats (see Section 4 and Arbeider et al. 2019), was the single most important prey item for juvenile sockeye salmon. Although an association with eelgrass has not been previously documented in other estuaries for sockeye salmon, this food web connection supports the link found in this section between juvenile sockeye salmon and eelgrass habitats.

Although we found that different biophysical factors were statistically associated with salmon abundance patterns, a large degree of uncertainty remained when predicting salmon abundance across time and space. For example, we found that juvenile sockeye were found more frequently at some eelgrass beds and not others. The highest abundances of sockeye, coho, and Chinook were found at Flora Bank, although we were not able to predict why this was. It is possible that this preference is a result of the combination of preferred turbidity, temperature, and spatial arrangement of the Flora Bank eelgrass bed in relation to the river mouth. These results suggest that identifying important habitat for juvenile salmon remains a key challenge which should be considered when assessing environmental risks in management and conservation planning. Local conditions, including the spatial arrangement of habitats, environmental gradients, prey distribution, and tides and currents, likely contribute to the differential use of habitats across estuaries and species. Managing key habitat for mobile and aquatic species, such as juvenile salmon, is particularly challenging because they require a continuous corridor of habitat arranged in particular distributions over their journey. This research identifies many contributing factors of estuary habitat for sockeye, coho, and Chinook salmon and supports our understanding of estuaries as dynamic habitat mosaics for migrating juvenile salmon.

Section 3.1 - Q2 - Key Findings

1) Juvenile sockeye and Chinook salmon preferred eelgrass habitats in the estuary over other habitat-types (open-water, rocky shores, and sandy bays).

2) Juvenile sockeye and coho were more abundant in locations with higher turbidity, while sockeye and Chinook salmon preferred locations with warmer water temperatures.

3) A diversity of biophysical variables can be used to explain spatial abundance patterns for juvenile salmon in the estuary, however, they underrepresent the complexity of fish-habitat associations. This supports our understanding of estuaries as dynamic habitat mosaics for migrating juvenile salmon.

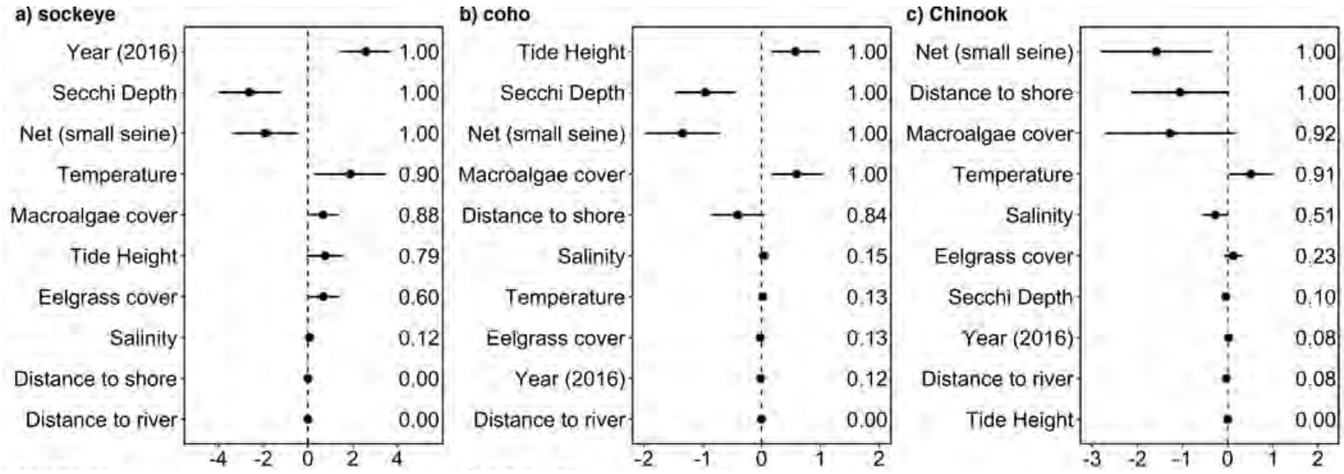

Figure 3-15. Standardized model-averaged coefficients (points) presented in decreased order of relative variable importance (RVI) and 95 % confidence intervals (bars) used to describe abundance of a) sockeye, b) coho and, c) Chinook salmon. RVI values are shown on rightmost of each panel and coefficients are related to the (log) mean of the normalized CPUE. Parameter year (2016) compares catches to the sampling year 2015 as a baseline, the small purse seine net is being compared to the large purse seine net, and low exposure is used as a reference to compare moderate and high exposed sites.

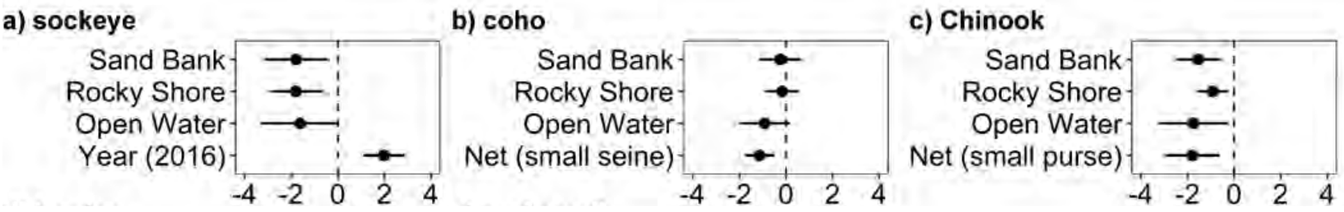

Figure 3-16. Standardized parameter estimates (dots) and 95% confidence intervals (bars) from top models (most parsimonious within ΔAIC_c less than 2) for modeling habitat types for a) sockeye, b) coho and, c) Chinook salmon. Open-water, sand banks, and rocky shores habitat types are compared to the eelgrass bed habitat type as a baseline (dashed line). Coefficients are related to the (log) mean of the normalized CPUE.

Q3 - What populations of salmon are found transitioning through the estuary?

Because salmon return to the same place where they were born, they have evolved remarkable adaptations to their ecosystems. For example, there are hundreds of uniquely adapted salmon populations in the Skeena River watershed (Gottesfeld & Rabnett 2008). Working with the Fisheries and Oceans molecular genetics laboratory, we performed genetic analysis of salmon caught in the Flora Bank region of the estuary to examine which salmon populations use this part of the estuary. Genetic analyses were performed on sockeye and Chinook salmon but were not performed on pink, chum, and coho salmon juveniles as their genetic population structure is poorly refined. There are approximately 60 total populations (Chinook and sockeye) that can be reliably separated with microsatellites in the Skeena River watershed (Beacham et al. 2005). Small pieces of the caudal fins were collected for genetic analyses from a subsample of Chinook and sockeye salmon collected in the Flora Bank region of the estuary (Figure 3-17). DNA was extracted and amplified by polymerase chain reaction at 13 and 14 microsatellite loci for Chinook and sockeye salmon. The genetic assignment outputs the probability that a given fish is from the population of interest. As a conservative approach, we only used the most likely probability in our calculations.

Figure 3-17. Lax Kw'alaams Fisheries personnel collecting genetic DNA samples and recording data on fish abundance during purse seine sampling for NCJSMP (photo by David Herasimtschuk from Freshwater Illustrated).

We collected four years of genetic stock identification data. Genetic data from the 2013 field season were previously published (Carr-Harris et al. 2015) and updated with the 2014 genetics in a more recent publication (Moore et al. 2015). Here we provide the most up-to-date data based on the 2013–2016 field seasons, during which we collected genetics from juvenile salmon from dozens of populations. We found that Chinook salmon collected in the Flora Bank region came from 21 different populations, ranging from as far inland as the Morice and Bear River populations, to more coastal populations, such as the Ecstall and Kalum Rivers. Sockeye salmon collected on Flora Bank had DNA that assigned to 31 different populations (Figure 3-18). Sockeye salmon included individuals genetically assigned to coastal populations (e.g., Diana Creek of the Kloiya), major Skeena producers (e.g., Fulton and Pinkut populations of the Babine system), lower Skeena populations (e.g., Williams of Lakelse Lake), and inland populations (e.g., Bear and Sustut populations). Some of the sockeye salmon collected were identified as originating from populations of conservation concern, such as the Nanika population from Morice Lake and sockeye salmon from Kitwanga Lake. Genetics data also revealed that juvenile sockeye salmon from other watersheds, both to the north and the south, also use the Flora Bank region—some sockeye salmon juveniles were identified as from the Stikine, Bowser Lake of the Nass, and Marble Creek from the Rivers Inlet area. Thus, more than 50 populations of salmon use the Flora Bank region. It should be noted that these are minimum estimates—the more fish we sample, the more populations we discover that use the Skeena River estuary.

These data provide evidence that the estuary habitats support fish that are harvested in commercial, recreational, and Aboriginal fisheries throughout the Skeena watershed and beyond. The Flora Bank region of the Skeena River estuary supports fish that originated from the traditional territories of at least 11 different First Nations.

Section 3.1 - Q3 - Key Finding

The Skeena River estuary supports more than 50 genetically-unique salmon populations that come from the traditional territories of at least 11 First Nations.

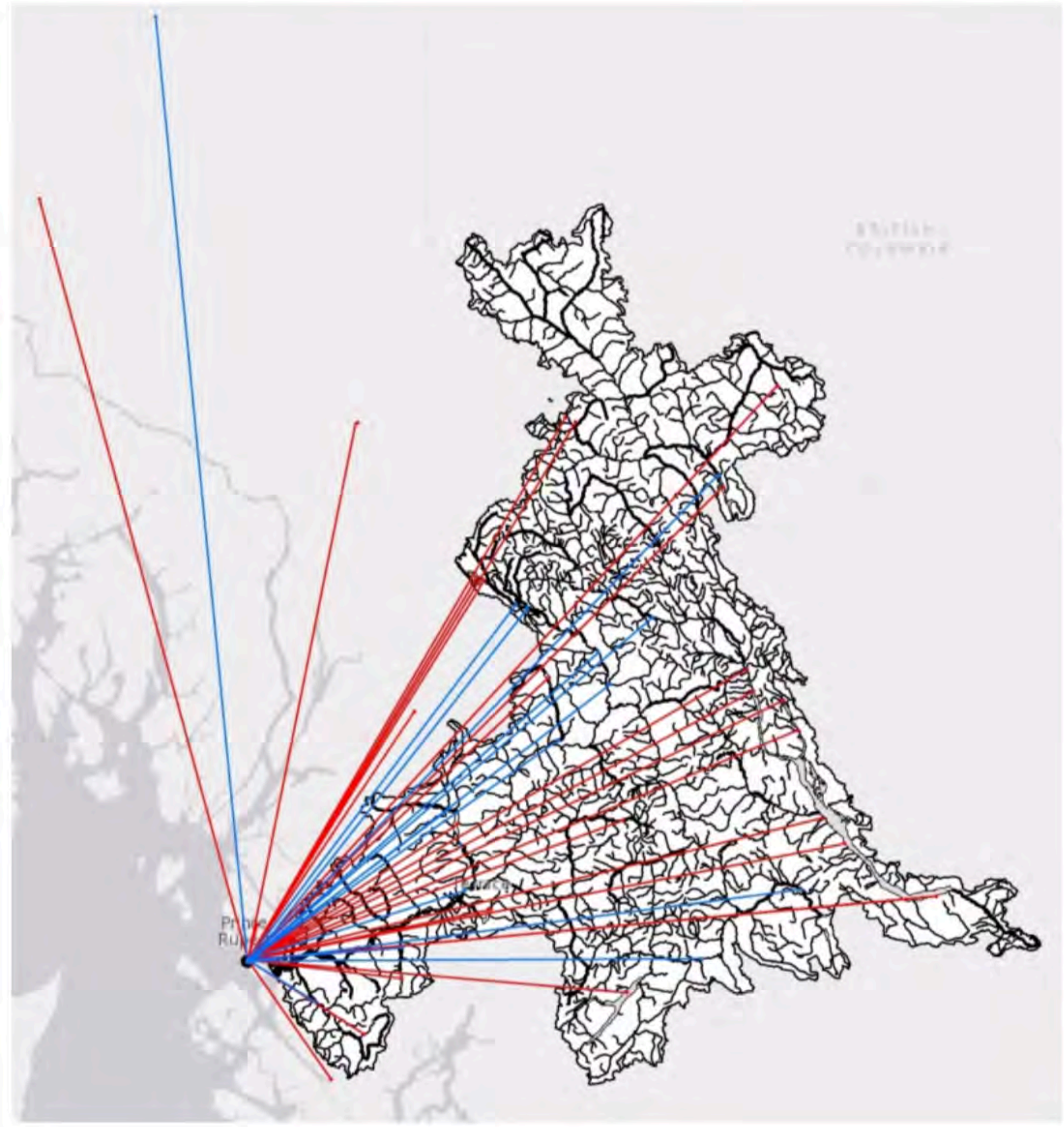

Figure 3-18. The Flora Bank region supports salmon from throughout the Skeena River watershed and beyond. Lines connect fish collected in the estuary with where they are from. Red lines represent sockeye salmon and blue lines represent Chinook salmon. Map created by John Latimer of Lax Kw'alaams fisheries, based on results from 2013 and 2014 (Moore et al. 2015).

Q4 - Does the timing of the estuary life-history phase differ between sockeye populations?

Using genetic data from juvenile sockeye salmon caught in 2013 and 2014 (methods described above), we explored the different in timing of estuary use among populations of sockeye salmon across the Skeena watershed. We found that different populations of sockeye from across the Skeena watershed were captured in the estuary at different times in the spring (Figure 3-19). For example, sockeye salmon smolts from the Kalum River had the earliest peak of abundance in the estuary, while Bear and Sustut River sockeye smolts were the latest to arrive (Figure 3-19). Peak abundances from the earliest to the latest populations were up to 5 weeks apart, highlighting the diversity in timing of different populations. Populations that arrived later in the estuary typically originated from further upstream and from higher latitudes and elevations. The timing of migration departure in populations originating in higher latitudes and elevations may be constrained by conditions in rearing lakes, such as timing of ice breakup. Further details about this study can be found in the following recent publication (Carr-Harris et al. 2018).

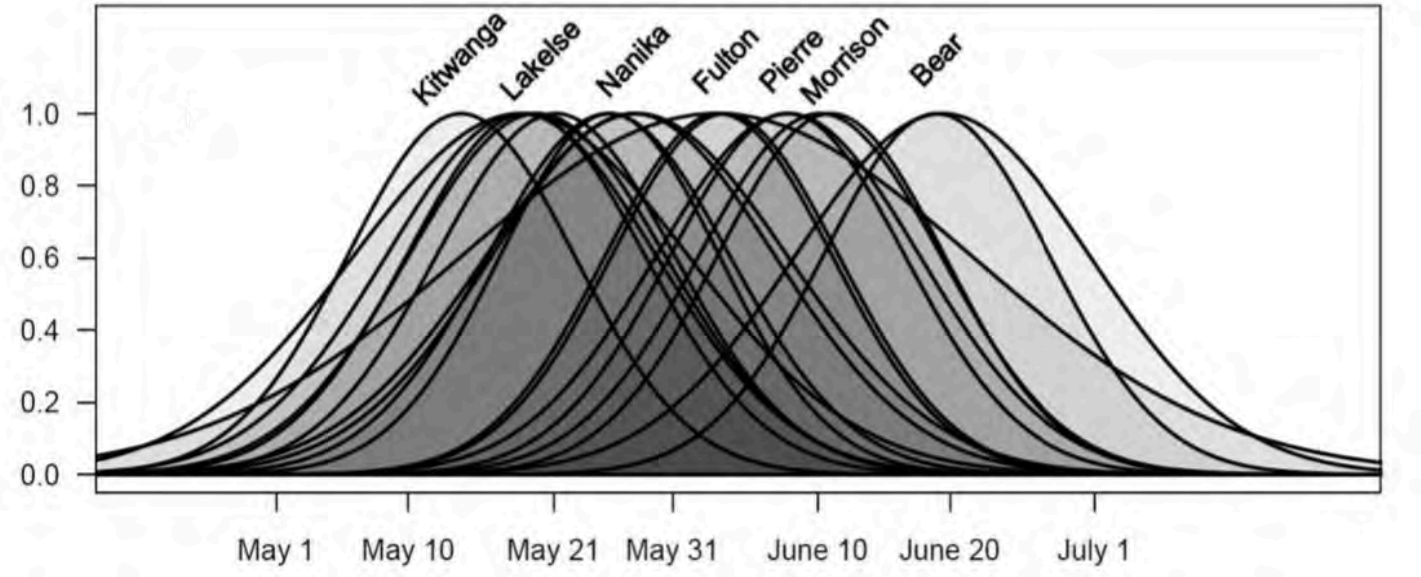

Figure 3-19. Probability distribution of timing of estuarine residence for different Skeena populations estimated from linear modeling of peak dates of estuarine capture of Skeena sockeye populations caught in 2013 and 2014 sampling (figure is amended from Carr-Harris et al. 2018).

Section 3.1 – Q4 - Key Finding

1) Througout the spring, different populations of juvenile sockeye salmon from across the Skeena River watershed were captured in the estuary at different times.

2) Sockeye salmon smolts from the Kalum River were the earliest peak of abundance in the estuary, while Bear and Sustut River populations were the latest to arrive. Late arrival of populations in the season was typically associated with higher latitudes and elevations.

Q5 - How long does a salmon take to migrate from freshwater rearing habitat to the estuary?

During estuary sampling, we captured tagged sockeye smolts from the enumeration project below Lake Babine that uses tagging as a mark and recapture method to determine population size (Figure 3-20). Re-capturing smolts that were tagged at Lake Babine, allowed an estimation of travel time between the lake and the estuary. For example, we captured nine smolts bearing tags from the Lake Babine enumeration fence during 2015 purse seine sampling on May 5, 9 days after tagging upstream and on May 19, 16 days after original tagging. Assuming that an extra day is required to swim from the tagging release location to the outlet, the mean duration for these smolts was 10.8 days, or an average velocity of 40 km per day. This velocity is the approximate speed of the river current. If we assume the same downstream velocity, the expected duration for different populations would range from two days for Alastair lake smolts that migrate from 67 km upstream to 14 days for Sustut Lake smolts migrating from 575 km upstream.

Figure 3-20. Photo of smolt originating from Lake Babine enumeration fence and being re-captured in the estuary. Smolt migrated for an average of 10.5 days (small sample size approximation) to reach the estuary (photo by Ciara Sharpe).

Section 3.1 – Q5 - Key Finding

On average, it takes juvenile sockeye 10.8 days, or an average velocity of 40 km per day, to migrate from Lake Babine to the Skeena River estuary.

3.2. *Stable isotope study of residence patterns*

Q1 - How long are different juvenile salmon species residing in the estuarine environment?

Stable isotopes can be used to understand how animals move across habitats, such as young salmon moving into the marine ecosystem (Kline Jr & Willette 2002; Kline et al. 2008). The delay in isotope incorporation into tissues can be used as a clock to time movements across ecosystems, like the movement of salmon from freshwater to estuaries. We used stable isotopes to learn about the degree to which juvenile salmon are feeding and residing in the Skeena River estuary. Stable isotopes are naturally occurring varieties of elements that have an extra neutron, so are slightly heavier. The ratio of the heavier isotope to the lighter isotope can be measured accurately, and thereby provides a natural chemical tracer that is passed up the food chain. This project was predominantly performed during the outmigration season of 2014. This research has been published but methods and results of the stable isotope study are summarized here. Additional details can be found in the open access paper Moore et al. 2016.

METHODS

In order to use stable isotopes to determine how juvenile salmon use the estuary, we collected three main types of samples. First, we collected stable isotopes from juvenile salmon in the estuary between April 9th and July 7th, 2014. Second, we collected stable isotopes from juvenile salmon from their freshwater rearing grounds to describe the chemical signature of where they were coming from (the "freshwater baseline"). Third, we collected stable isotopes from a fish that resides in the estuary (adult surf smelt) to characterize the chemical signature of this food web (the "estuary baseline"). By comparing the chemical signatures of juvenile salmon captured in the estuary to the chemical signatures of freshwater (where they came from) and the estuary (where they were caught), we can characterize the degree to which salmon are residing and feeding in the estuary.

Whenever possible, we collected both liver and muscle tissue from each fish. Liver tissue turns over extremely rapidly, on the order of days, while muscle tissue turns over more slowly, on the order of weeks. If fish were too small, we aggregated tissue samples from multiple individuals to generate pooled estimates. Slow- and fast-turnover tissues provided different timeframes of inference for their use as clocks (see below). Fish were collected from Flora Bank, Kinahans Island, Agnew Bank, Kitson Island, Porpoise Channel, and Lelu Island (Figure 3-11).

We used adult surf smelt, a zooplanktivorous estuary fish, as our estuary baseline. Freshwater baselines for each salmon species were collected by collecting salmon parr/smolts (Chinook, coho, and sockeye salmon) and fry (chum and pink) from rearing habitats in the Skeena River watershed. At least five samples were collected for each baseline. We collected pink, chum, sockeye, coho, and Chinook salmon juveniles from Flora Bank. Samples were frozen and then freeze-dried prior to shipping for analyses.

Stable isotope analyses were analyzed by mass-spectrometry at the Stable Isotope Laboratory at University of California Davis. Samples were run for Carbon ($\delta^{13}C$), Nitrogen ($\delta^{15}N$), and Sulfur ($\delta^{34}S$).

Carbon (C) and sulfur (S) are often used as tracers for habitat type, while nitrogen (N) is used as an indicator of trophic position, although many factors can influence isotope values. All samples are run against standards and values are shown as differences from these standards. Sulfur isotope analyses were not completed by the UC Davis laboratory at the time of writing this report; these results thus focus on C and N.

Freshwater baseline isotope signatures were different from estuary baseline isotope signatures among all the different salmon species. Estuary fish were more enriched in δ^{13}C. In contrast, stable isotopes of juvenile salmon in their freshwater phase were more depleted in δ^{13}C. This difference between freshwater and estuary isotope baselines enables us to distinguish between freshwater- and estuarine-derived tissues (Figure 3-21). There were less systematic differences between freshwater and estuary baselines for δ^{15}N.

RESULTS

Isotopes of juvenile Chinook salmon collected in the estuary were spread out between their freshwater baseline and the estuary baseline (Figure 3-21). Individuals that had isotopic signatures that were close to the freshwater baseline likely entered the estuary recently prior to capture. Other estuarine juvenile Chinook salmon had isotope signatures that appeared to be fully estuarine-derived. These individuals had likely been eating and rearing in the estuary for longer, allowing their tissues to become almost fully derived from estuary resources. Specifically, we found that 50% of the Chinook salmon had been in the estuary for at least 26 days, 25% of Chinook had been in the estuary for 33 days, and 5% had been in the estuary for 54 days (Figure 3-22).

Coho salmon smolts appeared to reside in the estuary for shorter times than Chinook salmon (Figure 3-21 and Figure 3-22), with 50% of the individuals having been in the estuary for at least 15 days, 25% of individuals residing for 22 days, and 5% of individuals residing for at least 43 days (Figure 3-21).

The isotope signatures of sockeye salmon smolts from the estuary were more freshwater in origin than those of Chinook salmon (Figure 3-21 and Figure 3-22). These data illustrate that most of the sockeye salmon caught had likely entered the estuary recently with 50% of individual sockeye smolts having been in the estuary for 2 days or less. Some individuals had isotope signatures that were shifted towards the estuarine baseline – these individuals had been rearing and feeding in the estuary, likely for days to weeks. We found 25% of sockeye tested had been rearing for at least 5 days, while 5% had been rearing for 18 days or more. These results support previous findings from the Skeena estuary (Higgins & Schouwenburg 1973), showing that juvenile sockeye salmon migrate through estuaries fairly rapidly, but also illustrates that some sockeye salmon smolts are rearing and feeding in the Skeena River estuary.

Pink salmon showed high individual variation in their isotope signatures. Some estuary-collected individuals had isotope signatures that matched the freshwater baseline, evidence that they had recently entered the estuary (50% of pink individuals had resided for at least 6 days) (Figure 3-21 and Figure 3-22). Other individuals had isotope signatures that were close to the estuary baseline, evidence that they had been feeding and growing in the estuary for some time (25% of pink individuals had been in the estuary for at least 30 days). Perhaps not surprisingly, the freshwater baseline of pink salmon was characterized by isotope signatures that are typical of marine ecosystems. Because pink fry migrate immediately down to the estuary prior to extensive feeding, freshwater pink fry have an oceanic isotopic signature that is derived from their mother. This creates less differentiation between freshwater and estuary baselines for this species.

Chum salmon had somewhat similar isotope patterns as pink salmon, with high variation in isotope signatures. These data are evidence that some juvenile chum salmon are residing and feeding in the estuary for an extended period of time. Similar to pink salmon, chum salmon baselines were not strongly differentiated between estuary and freshwater habitats. As chum fry leave the freshwater habitat immediately upon emergence, similar to pink salmon, the chum freshwater baseline had a “marine” isotope signature.

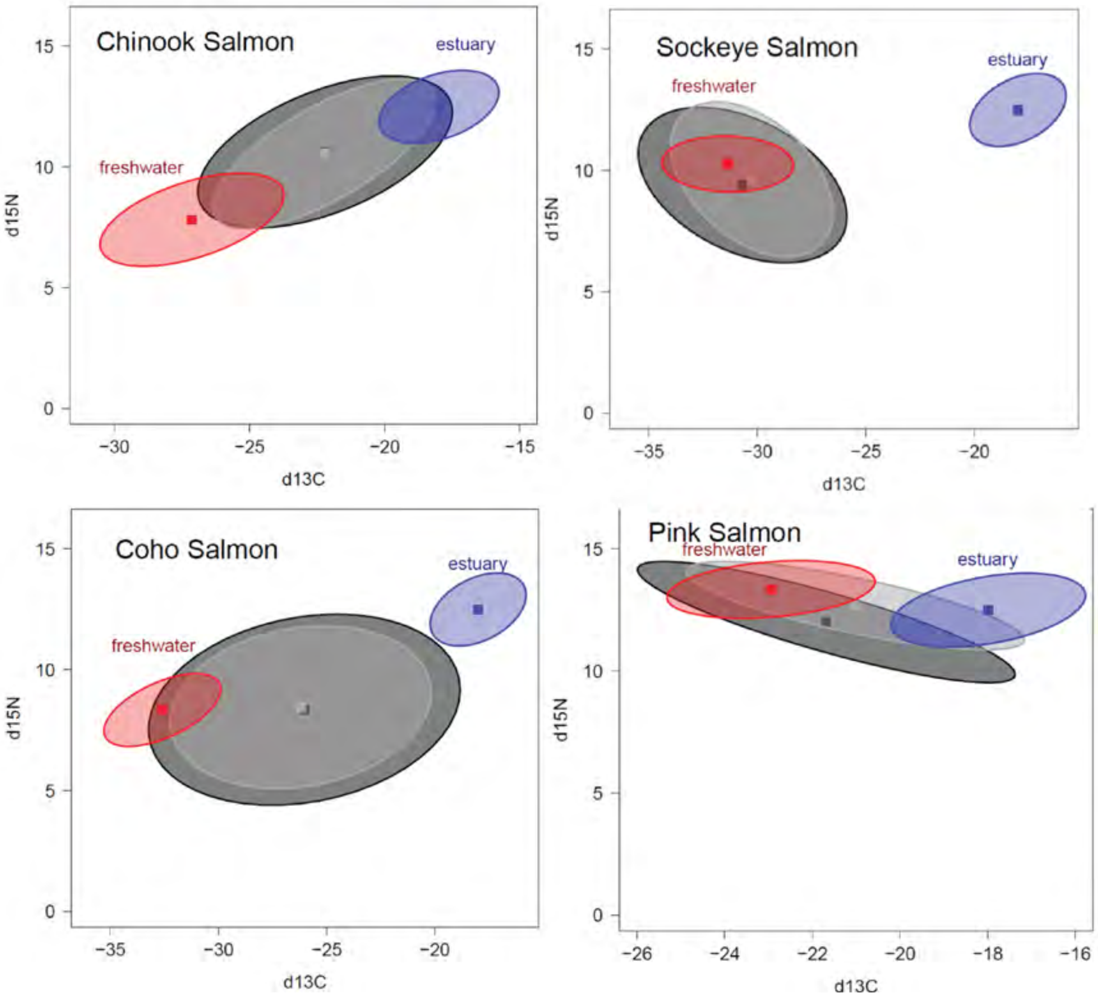

Figure 3-21. Stable isotopes of juvenile salmon in the Flora Bank region of the Skeena River estuary and their freshwater and estuary baselines. 90% confidence ellipses are shown, meaning 9 out of 10 data points should occur within this region. The black-lined ellipse that is darker gray represents isotope values of liver tissues, while the lighter gray ellipse represents muscle tissues. Different panels show different species of salmon that were sampled as juveniles in the estuary.

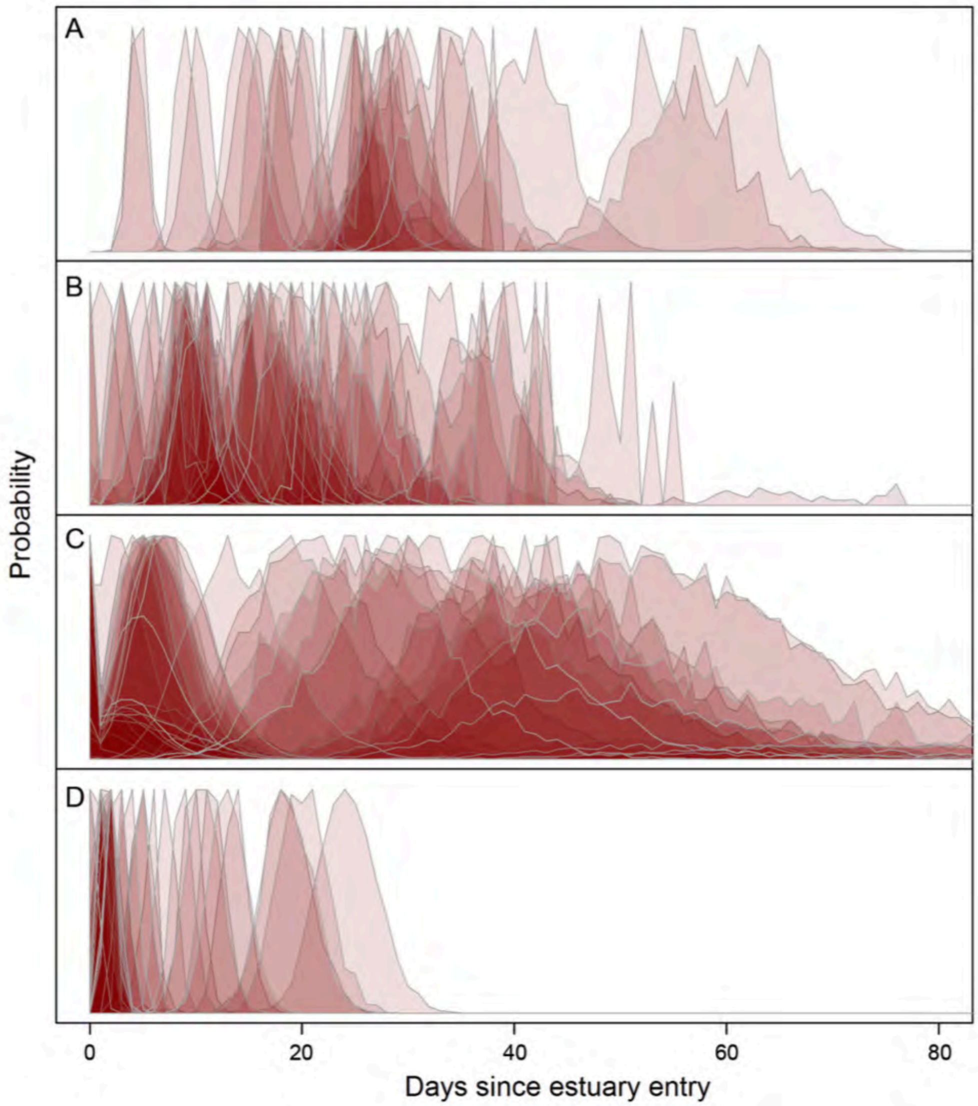

Figure 3-22. Estimates of days since entry into the estuary for A) Chinook, B) pink, C) coho, and D) sockeye salmon, shown as probabilities of individual fish based on estimates from liver tissue and muscle tissue and their sulphur and carbon isotopes (Moore et al. 2016).

Section 3.2 - Q1 - Key Findings

1) Juvenile salmon are feeding and rearing in the estuary for upwards of weeks to months.

2) Chinook salmon resided in the estuary for the longest time period: 50% of the Chinook salmon sampled had been in the estuary for at least 26 days.

3) Sockeye salmon resided in the estuary for the shortest amount of time (around 2 days), but some individuals resided for 18 days or more (5% of individuals sampled).

Q2 - Are salmon growing during estuarine residence?

Using results from the stable isotope analysis, we related estimated estuary residence time to fish size to determine if larger individuals of each species were residing in the estuary for longer. We found that larger Chinook, coho, and pink salmon individuals had resided for longer in the Skeena River estuary.

Juvenile Chinook used the Skeena River estuary extensively for rearing for up to one month and grew around 0.48 ± 0.09 (SE) mm per day (Moore et al. 2016). The length of residence time for Chinook salmon varies between watersheds (Miller & Simenstad 1997; Volk et al. 2010), however, the growth rates observed for Chinook salmon in the Skeena are similar to what has been observed in other estuaries (Miller & Simenstad 1997).

We found pink salmon used estuaries longer than previously documented. Past studies have shown that pink salmon move quickly through estuarine environments as fry upon emergence (Levy & Levings 1978; Weitkamp et al. 2014a). On the contrary, we found that many pink salmon remained in the estuary for over a month and larger individuals remained in the estuary for longer periods. We estimated that the growth rate of pink salmon was 0.22 ± 0.035 (SE) mm per day (Moore et al. 2016). This growth rate is lower than observed for juvenile Chinook and coho salmon.

The majority of sockeye salmon migrated through the estuary more quickly compared to other salmon species. However, we found that some individuals resided in the estuary for 18 days or more. Variation in residence time may be related to genetic population, however, this conclusion was based on a small sample size. This initial finding indicated that individuals from the Sustut resided the longest, while populations from the Babine, Four Mile, and Alastair populations resided for a shorter period of time (Moore et al. 2016). There are documented cases of sockeye populations rearing for longer periods of time, such as the Alaska Peninsula (Simmons et al. 2013). The average sockeye size captured did not seem to increase over time in the estuary, which is likely related to the variable age and size of sockeye populations migrating through the estuary (Gottesfeld & Rabnett 2008).

We found juvenile coho rearing in the Skeena River estuary for up to several weeks, and potentially months. Interestingly, smaller coho individuals were found to have resided longer in the estuary, which is a different relationship than observed for Chinook and pink (Moore et al. 2016). It is unclear why this relationship exists, however, it may be indicative of individuals migrating to the estuary as fry, and rearing in the lower Skeena River and estuary for extended periods of time. Other possibilities include juvenile coho migrating during the fall and winter. Although these alternative life-history strategies have not been documented for coho populations in the Skeena River previously, they have been documented in other regions (Koski 2009; Bennett et al. 2015).

Collectively, these data provide several key insights into juvenile salmon and their use of the estuary. First, for all species of juvenile salmon, individuals showed evidence of feeding and rearing in the estuary. These fish did not simply swim through the estuarine environment, but actively fed in the region. Second, these data provide estimates on the amount of time that individual salmon reared in the estuary. For example, stable isotope analysis of Chinook salmon revealed that many individuals entered the estuary weeks to months prior to their collection (Figure 3-22). Furthermore, larger individuals entered the estuary earlier, providing strong evidence that these individuals substantially grew in size in the estuarine environment.

Section 3.2 - Q2 - Key Findings

1) Larger juvenile Chinook, pink, and sockeye spent more time in the estuary compared to smaller fish. This provides evidence that Skeena River salmon are rearing (feeding and growing) in the estuarine environment (Figure 3-23).

2) The smallest juvenile coho salmon resided in the estuary for the longest time period, which may be evidence of different life-history stages that have not been previously documented.

Figure 3-23. Juvenile sockeye salmon found in the Skeena River estuary during the NCJSMP sampling program (photo by David Herasimtschuk from Freshwater Illustrated).

3.3. Inferred growth of juvenile salmon

In Section 3.2, we found that sockeye, pink, and Chinook were growing in the Skeena River estuary throughout the spring and summer. Growth rates were created by comparing days since estuary entry across length of individual fish. However, we can also investigate growth by looking at size of fish captured in the estuary over time to infer patterns of growth. The size distribution of juvenile salmon in the estuary over time will be controlled by both the growth of individuals within the estuary, as well as the arrival of new smolts. A positive relationship between time and size supports the hypothesis that fish are rearing and growing in the estuary, however, juvenile salmon might enter the estuary at different sizes or at different times, which would obscure the relationship between size and time.

Q1 - What are the size distributions of juvenile salmon found in the Skeena River estuary?

Juvenile pink and chum are the smallest juvenile salmon captured in the estuary, with an average fork length of 74 mm and 75 mm (Figure 3-24). Sockeye salmon were slightly larger, averaging 86 mm in size, while average sizes of juvenile coho and Chinook salmon were 108 mm and 109 mm, respectively (Table 3-1).

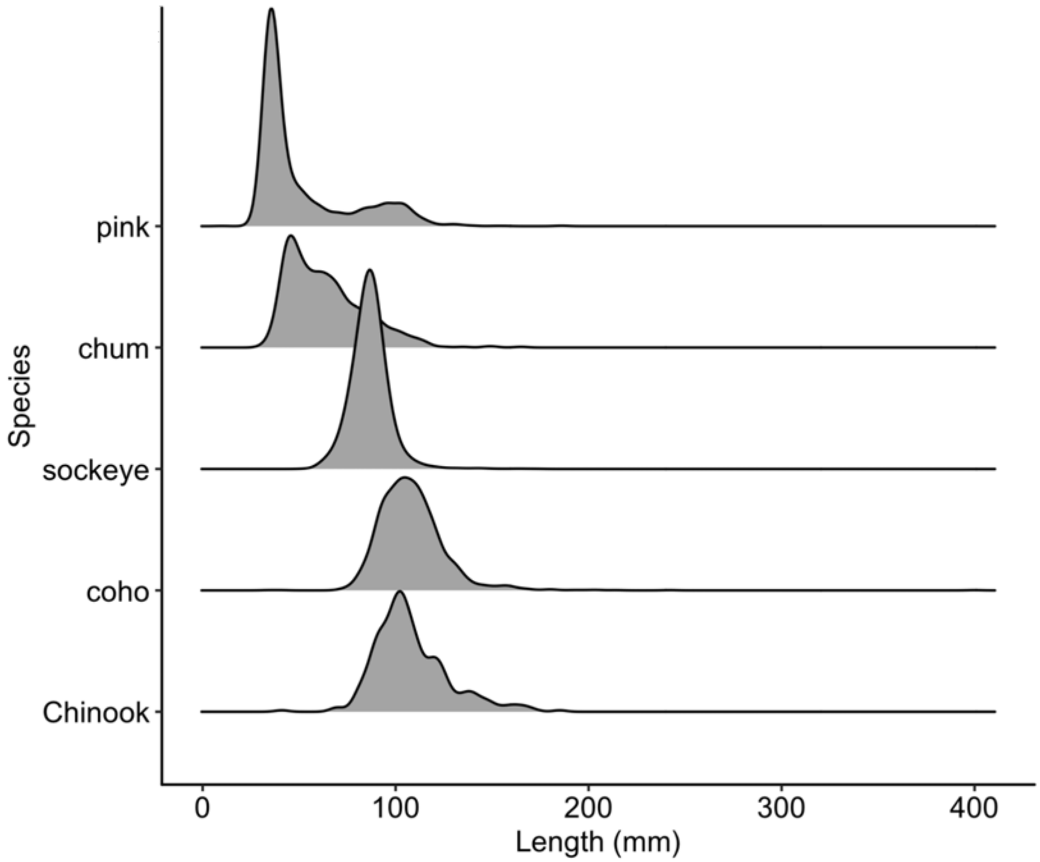

Figure 3-24. Density of juvenile salmon lengths for each species (pink, chum, sockeye, coho, and Chinook) from all fish sampled (purse seine and beach seine) in the Skeena River estuary from 2013–2018.

Table 3-1. Summary of juvenile salmon fork lengths (FL) caught by purse and beach seine in the Skeena River estuary from 2013–2018.

	Purse Seine		Beach Seine	
Salmon Species	FL (mm)	Mean FL (mm)	FL (mm)	Mean FL (mm)
pink	31–187	74	26–75	36
chum	40–165	75	32–101	51
sockeye	55–174	86	62–120	83
coho	34–400	109	41–160	103
Chinook	69–195	108	41–112	91

Q2 - Are juvenile salmon growing in the estuary over time?

The average length of juvenile pink, which enter the estuary immediately after emergence, increased more rapidly than the other species of salmon (Figure 3-25). Pink salmon were 0.76 ± 0.01 (SE) mm larger per day (p = 0.00) during spring and summer, suggesting that the average size of pink salmon increased almost 50 mm in two months. During April and early May, pink salmon captured in the estuary were small (<50 mm), but pink salmon seen in the estuary in June were large (Figure 3-26). This is a higher growth rate estimate than that derived from stable isotope data (0.22 ± 0.035 (SE) mm per day) (Moore et al. 2016). It is possible that the juvenile pink salmon captured in June are a completely different cohort, originating from the Skeena River or a different estuary. Although pink genetic baselines are not available to determine the origin of pink salmon captured in the Skeena River estuary, advancing technology will likely allow this question to be investigated in the near future.

The size of captured juvenile chum (0.32 ± 0.03 (SE) mm per day, p = 0.00), coho (0.32 ± 0.03(SE) mm per day, p = 0.00), and Chinook (0.31 ± 0.07 (SE) mm per day, p = 0.00)) increased over time at a similar rate to each other, but were lower than that of juvenile pink salmon (Figure 3-25). Average juvenile sockeye salmon size increased at the slowest rate of 0.17 ± 0.02 (SE) mm per day (p = 0.00) (Figure 3-25). The rate of size increase for these salmon species supports previous research from the Skeena River and other estuaries which suggest that chum, Chinook, and some populations of coho spend more time rearing and growing in the estuary. However, the majority of sockeye salmon in the Skeena River estuary were found to be spending less time in the estuary (Moore et al. 2016). Chinook and chum are known to be the most estuarine dependent species and may be more susceptible to impacts from degradation to estuary habitat or changes to the estuarine environment.

We found some similarities and differences when comparing the rate of increase in size over time to growth rates identified in Section 3.2. Firstly, the overall size distributions for pink, Chinook, coho, and sockeye salmon increased overtime, which is supported by growth rates identified in Moore et al. (2016). In the stable isotope study, Chinook salmon had the highest growth rate, while sockeye salmon had the lowest. Using inferred growth, we found that the size distributions for pink salmon increased at the fastest rate overtime, juvenile Chinook, chum, and coho salmon increased at a similar rate overtime, and sockeye salmon increased at the lowest rate. Estimating chum salmon growth rates were not possible during the stable isotope study, but the average size of chum salmon throughout the spring and summer was clearly increasing (Figure 3-25). This suggests that chum salmon are also using the Skeena estuary as rearing habitat. Interestingly, coho salmon decreased in size over estuarine residence in Section 3.2 (negative growth rate) but the average size of coho was found to increase throughout the summer. As mentioned previously, this may be evidence of certain coho having different life-history strategies. We also found that the average size of pink salmon increased at a faster rate than the estimated growth rate, suggesting that there may be other populations using the estuary.

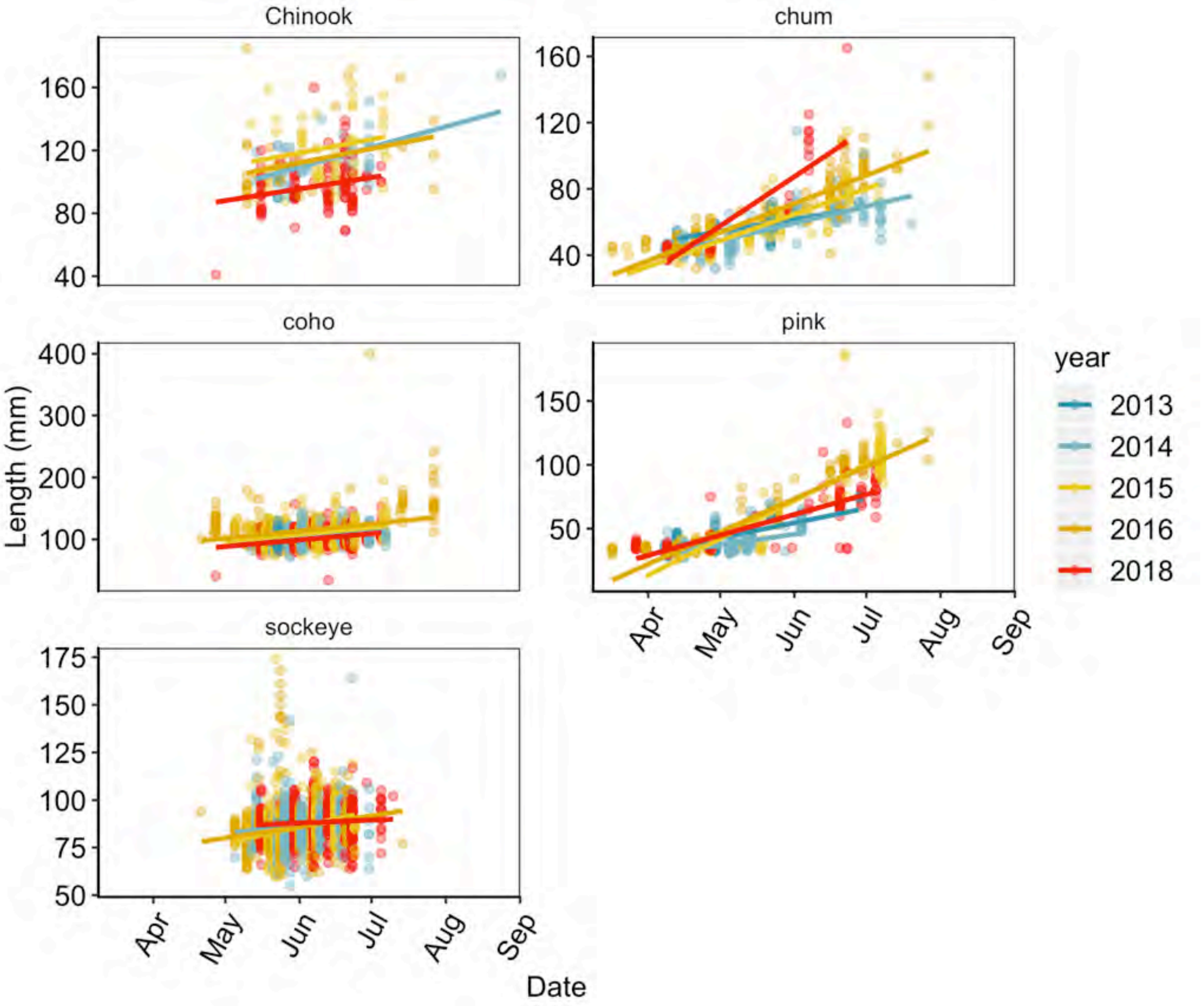

Figure 3-25. Lengths of juvenile Chinook, coho, sockeye, chum, and pink salmon captured from 2013–2018. Corresponding regression lines showing positive linear relationships are shown for each year.

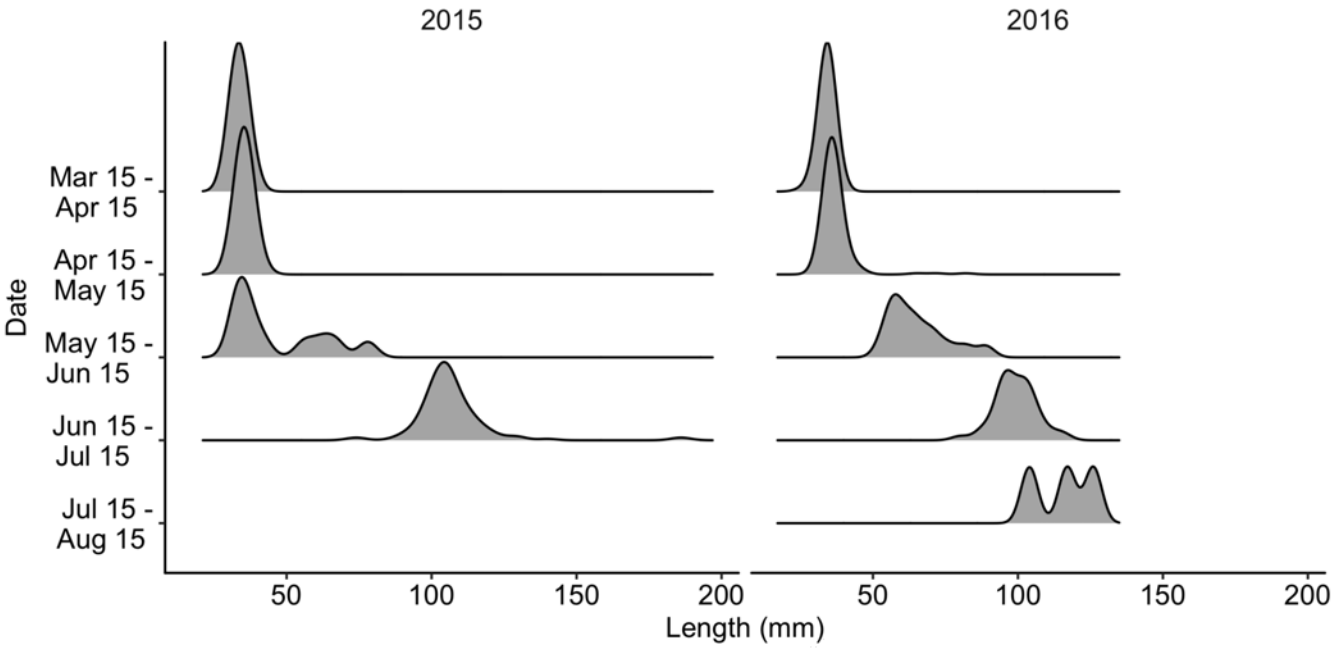

Figure 3-26. Size distributions of juvenile pink salmon captured in the estuary by purse seine and beach seine over two week time periods in 2015 and 2016.

It is important to note that none of these estimates of growth rate were derived from individual measurements of growth, but rather from changes in populations through time. It is likely that individuals enter the estuary at different times in the season and leave the estuary at different times, and that these seasonal patterns of migration are likely size-dependent. In addition, individuals from across species and populations are arriving in the estuary with varying fork lengths. For example, sockeye salmon leaving Babine Lake, the largest sockeye salmon population in the Skeena River watershed, are known to increase in size throughout the smolt migration period. Sockeye salmon individuals that reside in the lake longer continue to grow over the migration period and are larger when they arrive in the estuary compared to earlier migrating smolts from the same population. Thus, our estimates of changes in size should be interpreted with caution and with consideration of these assumptions.

Section 3.3. - Q1 & Q2 - Key Findings

1) Average size of all juvenile salmon species increased overtime in the estuary.

2) The size of pink salmon increased at the fastest rate, while sockeye salmon increased at the lowest rate.

3) Although different populations and life-history stages may be contributing to variation, the strong trends from multiple years of data indicate that all salmon species are likely growing in the estuary over time.

Q3 - *Do different sockeye salmon populations have different size distributions?*

The length of sockeye salmon smolts found in the Skeena River estuary varies based on the productivity of their respective rearing lakes (Figure 3-27 and Figure 3-28). Generally, clear lakes have higher productivity, greater prey availability, and produce larger smolts. Glacial lakes, which have lower prey availability, tend to produce smaller smolts. Smolt size is also affected by density dependence related to brood year abundance of the parent generation and fry abundance for each cohort. By comparing length data to genetic stock identification (Section 3.1 Q3) of individual sockeye salmon collected across four years of NCJSMP sampling (2013–2016), we investigated if the size of sockeye salmon smolts found in the estuary varies based on the origin population of the smolts.

For juvenile sockeye sampled in the Skeena estuary and Prince Rupert Harbour from 2013–2016, the largest smolts came from Slamgeesh and Bear Lakes in the upper Skeena, Kitwanga Lake in the middle Skeena, and Lakelse Lake in the lower Skeena, which are all clear, productive lakes (Figure 3-27). In recent years, spawner abundance for each of these systems has been much lower than the optimal spawner abundance estimated to meet the rearing capacity for each lake (Shortreed et al. 1998). Thus lower abundances may also be contributing to larger smolt sizes originating from these lakes due to density dependent processes, such as less competition for food resources.

The smallest smolts came from Kitsumkalum Lake and Johnston Lakes in the lower Skeena watershed. Kitsumkalum Lake is a glacial lake with low productivity (Figure 3-27). Johnston Lake is more productive than Kitsumkalum Lake, however, recent hydroacoustic surveys have shown that Johnston Lake has the highest densities of sockeye fry among Skeena sockeye rearing lakes, which may negatively influence smolt size (Doire 2019).

Section 3.3 - Q3 - Key Findings

1) The largest juvenile sockeye salmon captured in the estuary came from clear, productive lakes from across the Skeena River watershed (Slamgeesh Lake, Bear Lake, Kitwanga Lake, and Lakelse Lake).

2) The smallest juvenile sockeye salmon caught in the estuary came from lower productivity lakes, such as Kitsumkalum Lake and Johnston Lake in the lower Skeena Watershed.

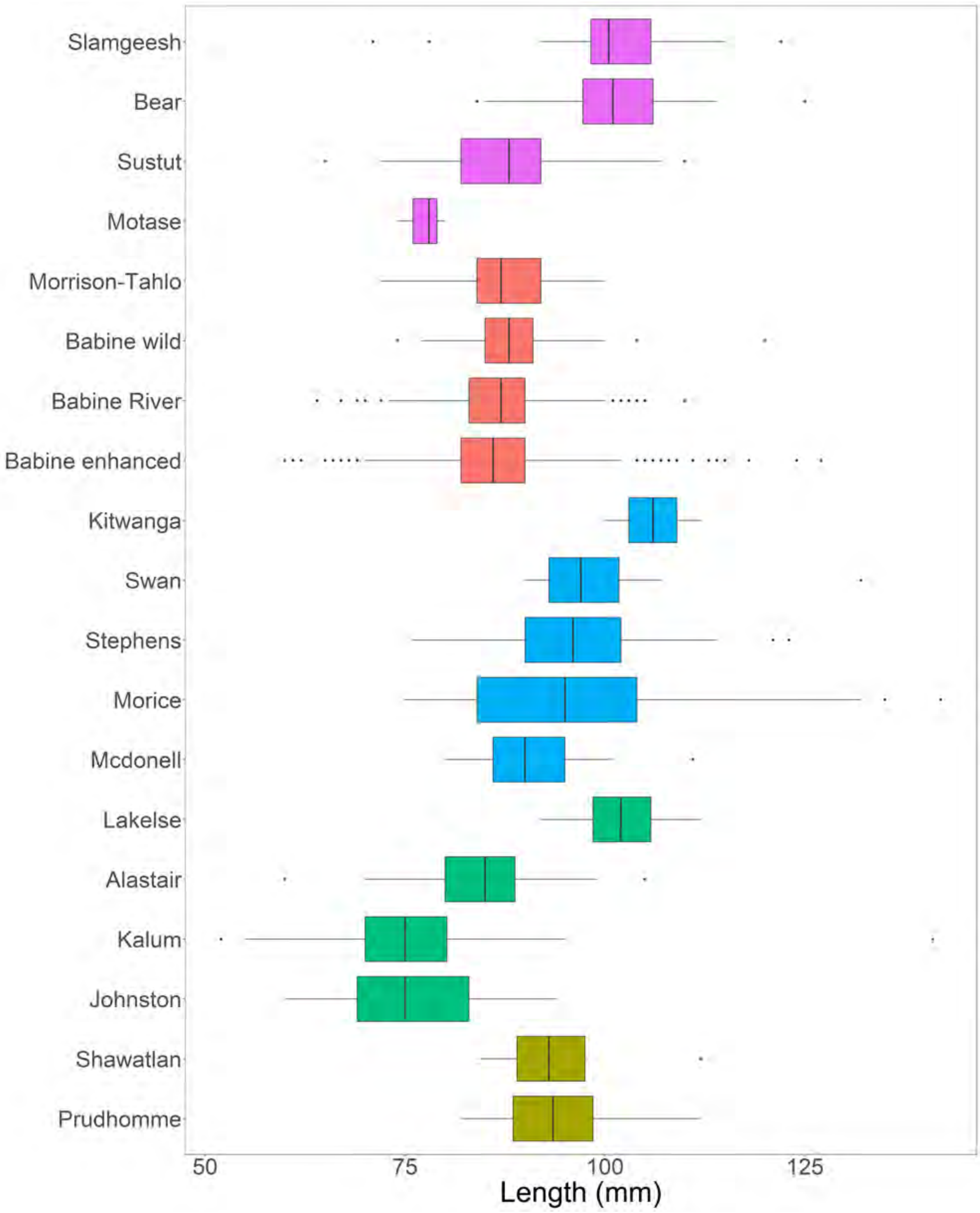

Figure 3-27. Distribution of lengths of genetically identified sockeye salmon captured in the Skeena River estuary and Prince Rupert harbour from 2013–2016, by lake of origin (y-axis). Different colours indicate different regions within the Skeena watershed and adjacent systems (pink = upper Skeena systems, red = Babine sockeye, blue = middle Skeena, green = lower Skeena, olive = coastal (non-Skeena) sockeye rearing lakes).

Figure 3-28. Various sizes of juvenile salmon captured in the same purse seine net during outmigration in the Skeena River estuary (photo by Samantha Wilson).

3.4. Energetic status of Skeena sockeye smolts in the estuary

We initiated a preliminary study into the energetics of juvenile sockeye salmon that were entering the Skeena River estuary. In large river systems like the Skeena River, juvenile salmon have to migrate enormous distances from where they rear downstream to the estuary. In some cases, the energetic costs of migration may push smolts to the point where they are ‘running on empty’ by the time they get to the estuary. For example, research on Chinook salmon in the Columbia River has shown that some juvenile salmon are dangerously close to starvation by the time they reach the estuary. Estuary resources are required to replenish energy stores and for growth during the critical early marine life-history stage. Understanding the energy stores of juvenile salmon as they enter the estuary can provide insight into how sensitive salmon may be to changes in estuary prey resources.

Q1 - What are the energy levels of smolts once they reach the estuary?

In the 2015 field season, we collected 78 sockeye smolts in Inverness Channel and Flora Bank and analyzed them in the lab for lipid content. Lipid constitutes the best energy source for fish and measuring the percent lipid (gram of lipid per gram body weight) reflects the amount of energy fish have stored.

We found that on average sockeye smolts were 2.6% lipid and ten percent of fish sampled were less than 2% lipid. The threshold where salmon performance begins to deteriorate (death) is 1.5%. These data indicate that a substantial proportion of juvenile sockeye salmon have low energy stores and are close to starvation when they reach the estuary (Figure 3-29).

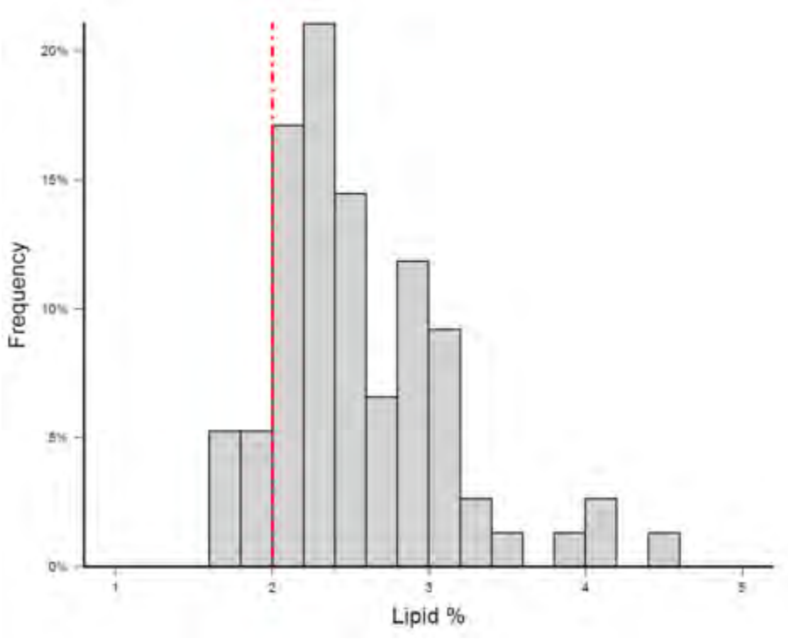

Figure 3-29. Frequency of energy storage levels of smolts captured in the Skeena River estuary. Red line at 2% indicates energy ‘danger zone’, where 1.5% is thought to be a threshold where salmon performance begins to deteriorate (death).

Q2 - How do the energy levels of smolts vary between different populations of salmon?

By collecting and processing lipid levels of sockeye smolts from freshwater populations, we compared the ‘pre-migration’ energy levels of various populations. We collected individuals from smolt weirs on the Kitwanga (36 smolts), Babine (91 smolts), and Slamgeesh (35 smolts) rivers, as sockeye salmon were leaving lake outlets on their migration to the estuary. Sockeye salmon collected from these populations were examined to determine the variability in energy storage between different populations.

Thirty-six smolts were captured at the Kitwanga smolt fence during the spring out-migration in 2016 and the amount of lipid (energy stored as fat) in each individual was assessed. Kitwanga smolts had the highest lipid content at 7%, which is comparable to lipid levels observed in hatchery smolts. Babine Lake smolts had an average of nearly half the lipid observed in Kitwanga smolts at 3.77%, while Slamgeesh sockeye smolts had on average 4.22% lipid content (Figure 3-30). None of the smolts sampled from Kitwanga Lake had lipid values of less than 2%.

Similar to what has been observed in other systems, the earliest migrating smolts that leave Kitwanga lake have higher energy storage than smolts that leave later (Figure 3-30). Interestingly, smolts beginning their migration from Kitwanga Lake appeared to have stopped feeding prior to their migration with more than 80% of smolts captured having empty stomachs.

There was huge variability in starting lipid levels across populations within the Skeena River, suggesting that different lake-rearing conditions have a strong influence on the energetic content of smolts. This variability does not appear to correlate with migration distance as we originally hypothesized, rather it may be related to the availability of resources in the rearing lake. For example, Kitwanga smolts appear to have access to high levels of food resources.

Kitwanga Lake sockeye smolts are not likely limited by the amount of lipid stores on an individual basis, as they have nearly double the amount of energy as Babine smolts and have a freshwater migration that is half the distance of Babine smolts. Potential trade- offs associated with such high lipid levels include a potential for decreased swim performance, but our understanding of these potential trade-offs is low. The results presented here in Section 3.4 (Q1 and Q2) assessing smolt lipid content are preliminary results from research performed by Samantha Wilson from Simon Fraser University. Further analyses, including comparing these lipid levels to lab experiments on how lipid levels influence fish performance and modeling population specific sensitivity to starvation are now completed and will be published in the future.

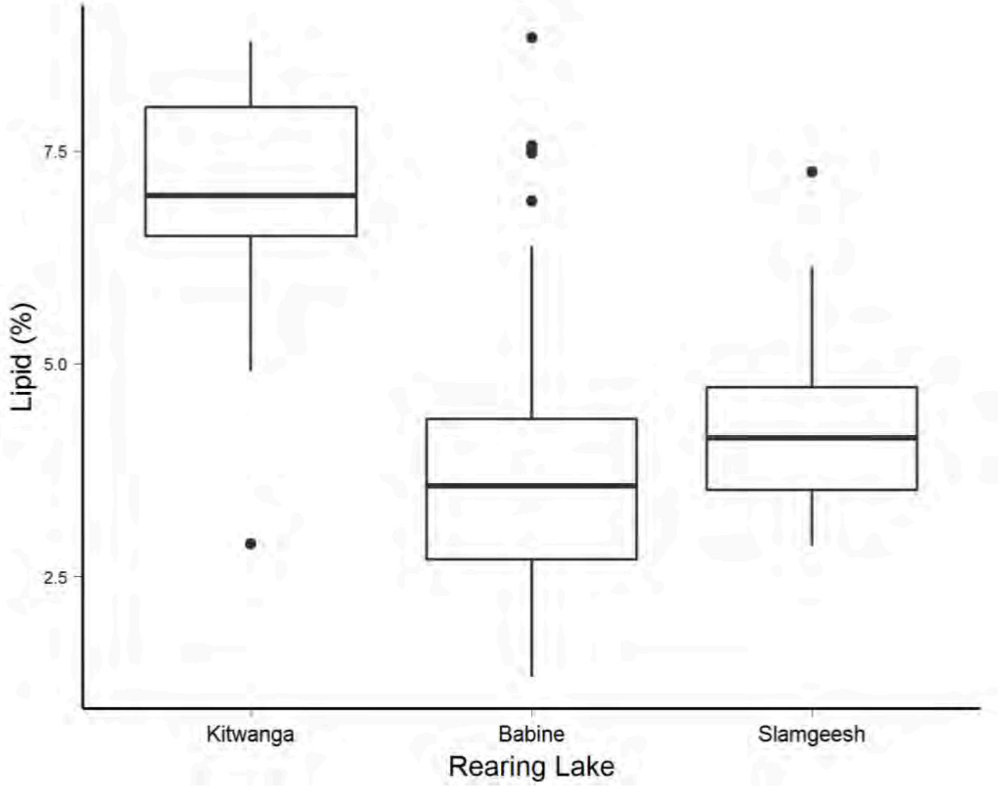

Figure 3-30. Comparison of the average percent lipid for sockeye salmon smolts collected from Kitwanga, Babine, and Slamgeesh populations. Dark center line is median and upper and lower lines of the box represent quartiles (25% and 75%).

Section 3.4 - Key Findings

1) Juvenile sockeye salmon have variable energetic stores in the estuary; some individuals had low energy levels indicative of starvation.

2) Different populations of sockeye smolts in the Skeena watershed had different 'pre-migration' lipid content.

3) Sockeye smolts from Kitwanga lake had the highest lipid levels (7.0%) compared to Slamgeesh (4.22%) and Babine Lake (3.77%) sockeye smolts.

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4. JUVENILE SALMON FOOD WEB

This section contains results summarized from the following two references:

A) Carr-Harris C. (2017). Fish diet and selectivity study of Flora Bank and adjacent areas. Prepared for the Lax Kw'alaams Band and Habitat Restoration Initiative, Skeena Fisheries Commission, Kispiox, BC.

B) Arbeider, M., Sharpe, C., Carr-Harris, C., & Moore, J.W. (2019). Integrating prey dynamics, diet, and biophysical factors across an estuary seascape for four fish species. Marine Ecology Progress Series, 613

Juvenile salmon are a part of an estuary food web that is sustained by a variety of energy sources. Terrestrial nutrients and energy are transported downstream by rivers into the estuary (Naiman & Sibert 1979). Estuaries are also sustained by oceanic sources of nutrients and energy. Both of these sources, as well as internal cycling of nutrients from decomposition of detritus, fuel primary production that is both benthic (e.g., eelgrass and macroalgae like kelp) and pelagic (e.g., phytoplankton up in the water column) (Sigmon & Cahoon 1997). Additionally, there are often high rates of direct uptake of dissolved carbon by microbes in estuaries. Collectively, these microbes and phytoplankton are consumed by a variety of zooplankton, such as copepods (Naiman & Sibert 1979). These zooplankton, as well as some benthic invertebrates, terrestrial invertebrates, and small fishes, are the predominant food sources for juvenile salmon. Juvenile salmon, as well as other small fish such as smelt and herring, are important food sources for other fish, birds, like ospreys and murrelets, and mammals, like river otters and seals (Macdonald et al. 1988; Dolloff 1993).

Previous research has found that salmon species tend to feed and utilize habitat in the estuary in different ways. Generally, it is thought that coho and sockeye salmon move through the estuary in a short amount of time, while chum and some Chinook salmon will reside in the estuary for weeks or months (Weitkamp et al. 2014b). There is also variation within species in the degree to which certain populations use estuaries; for example, some populations of sockeye salmon reside in estuaries for extended periods of time (Simmons et al. 2013). Despite these differences in estuary residence time, active feeding and growth in the estuary has been observed in all salmon species, including those that migrate rapidly to the ocean. When Chinook, pink, and chum salmon migrate into the estuary, they are thought to spend this phase in the shallow near-shore environments feeding on epibenthic zooplankton, such as harpacticoid copepods and epiphytic crustaceans (Naiman & Sibert 1979; Healey 1982). Juvenile coho, sockeye, and steelhead are thought to reside in deeper water environments feeding on neritic zooplankton and small fish (Healey 1982; Simenstad et al. 1982). Productive feeding on zooplankton allows for salmon to achieve high growth rates in the nursery habitat of the estuary before migrating to the open ocean (Simenstad et al. 1982).

Understanding the connections of the juvenile salmon food web within the Skeena River estuary contributes to our understanding of how juvenile salmon are using the estuarine environment and is important for assessing the potential effects of development in the area. In this section, we investigate the available food (zooplankton community) and diet of juvenile salmon species captured in the Skeena River estuary to determine important food resources for each species. We conducted zooplankton and benthic invertebrate sampling in the study area during the spring and summer to determine prey availability for key estuarine resources. Here we report the results of the fish diet, zooplankton, and benthic invertebrate surveys. The research discussed here is the topic of Carr-Harris (2017) and Arbeider et al. (2019). Specifically, we ask the following questions:

Q1 - What do juvenile salmon species eat in the estuary? How does this compare to common forage fish species like herring and surf smelt?

Q2 - What are the spatio-temporal dynamics of zooplankton communities in the estuary?

Q3 - What are the potential food web linkages in the Skeena River estuary?

METHODS

This sub-section contains details of fish and zooplankton sampling (benthic, oblique tows, and vertical tows) and laboratory processing applicable to results presented in Sections 4.1 and 4.2

Fish Sampling

Purse seining was conducted from early March until the end of July in 2016 to obtain fish stomach samples for diet analysis. Samples of juvenile sockeye, coho, herring, and smelt were collected with large and small purse seine nets. See methods in Section 3.1 and sampling map (Figure 3-11) for details on purse seine sampling. The number of diet samples retained after each purse seine set varied according to diet availability, resulting in 111 sockeye salmon, 57 coho salmon, 57 Pacific herring, and 35 surf smelt samples from 17 locations. Fish samples were retained when there were at least five individuals of a species from a given set. We also retained small numbers of chum and Chinook salmon that were captured with focal species during two sets in 2016, and 25 sockeye salmon samples that were collected during the 2015 sampling season, in order to improve our understanding of interspecific competition between salmon species and year to year variability. Fish that were selected for diet analyses were anaesthetized with a lethal dose of MS-222, preserved in 10% formaldehyde, and transferred to a laboratory for further processing. In the lab, fish fork length was measured to the nearest mm and weighed to the nearest 0.01 gram (wet weight) after blotting with paper towel. Stomachs were excised and stored in 10% formaldehyde in individual vials. Fish stomachs were shipped to Biologica Environmental Services Ltd., where they were assigned qualitative indices of fullness and digested material, and their content was identified to the highest possible taxonomic resolution.

Zooplankton Sampling

Zooplankton were sampled with double-oblique tows (Carr-Harris 2017) and vertical tows (Arbeider et al. 2019). Oblique zooplankton sampling was performed at four locations near fish collection sites in Porpoise Channel, Inverness Channel on the southeast side of Lelu Island, off the northwest side of Kitson Island, and from Kinahans Bay. The samples were collected biweekly from April–September 2016, and monthly in October, November, and February 2016. Additionally, we collected zooplankton samples from a sampling station further upstream in Inverness Channel from April–June 2016. Zooplankton samples were collected using double-oblique tows with a 350-micron mesh bongo net

(Figure 4-1). During each sampling event, a bongo net weighted with 9–27 kgs of lead was lowered to the bottom of the water column and immediately retrieved using a hydraulic winch at a velocity ranging from approximately 0.5–1.5 m/s. The water column depth was estimated using the vessel sounder and target depth was calculated using the line length and angle during each tow. In order to maintain an ideal tow angle of approximately 45°, the vessel speed varied depending on the prevailing current at a given sampling location. The actual tow angle was estimated during each sampling event using a handheld angle meter. The actual depth of each tow was recorded using a Sensus Reefnet depth logger, which was attached to the bongo net frame and downloaded at the end of each cruise. The water flow through the bongo nets was metered using a TSK flowmeter. At the end of each tow, the zooplankton nets were rinsed thoroughly, and their contents were preserved in 5% formaldehyde buffered with seawater. Zooplankton samples were shipped to Biologica Environmental Services Ltd. for taxonomic analysis. In the lab, large organisms (>1mm) were identified to species and enumerated, while smaller organisms were subsampled using a Folsom plankton splitter and identified to the lowest practical taxonomic level up to a maximum count of 300 organisms per sample. The total density of each taxonomic group was determined by dividing the total abundance in each sample by the volume of water filtered through the plankton net, which was estimated by multiplying the length of line deployed by the area of the net mouth.

Figure 4-1. Photograph of bongo nets used for zooplankton sampling.

Vertical zooplankton sampling concurrently occurred after every purse seine set at all 17 of the small purse seine sites (Figure 3-11). We used a 250 µm WP2 plankton net towed by hand vertically from a boat at 5m below the surface to standardize the volume of water that was sampled. Samples were stored in a 5% seawater buffered formalin solution. We stained zooplankton with Rose Bengal to make them more visible (Figure 4-2), partitioned them with a Folsom plankton splitter, and sorted them until at least 400 individuals or the entire sample had been identified. We used a taxonomic level that was comparable to zooplankton identified within the diet samples and enumerated each group. We used abundance, corrected for the size of partition, as the final variable because all samples were from the same depth (5m) and, therefore, volume of water (3.9 L).

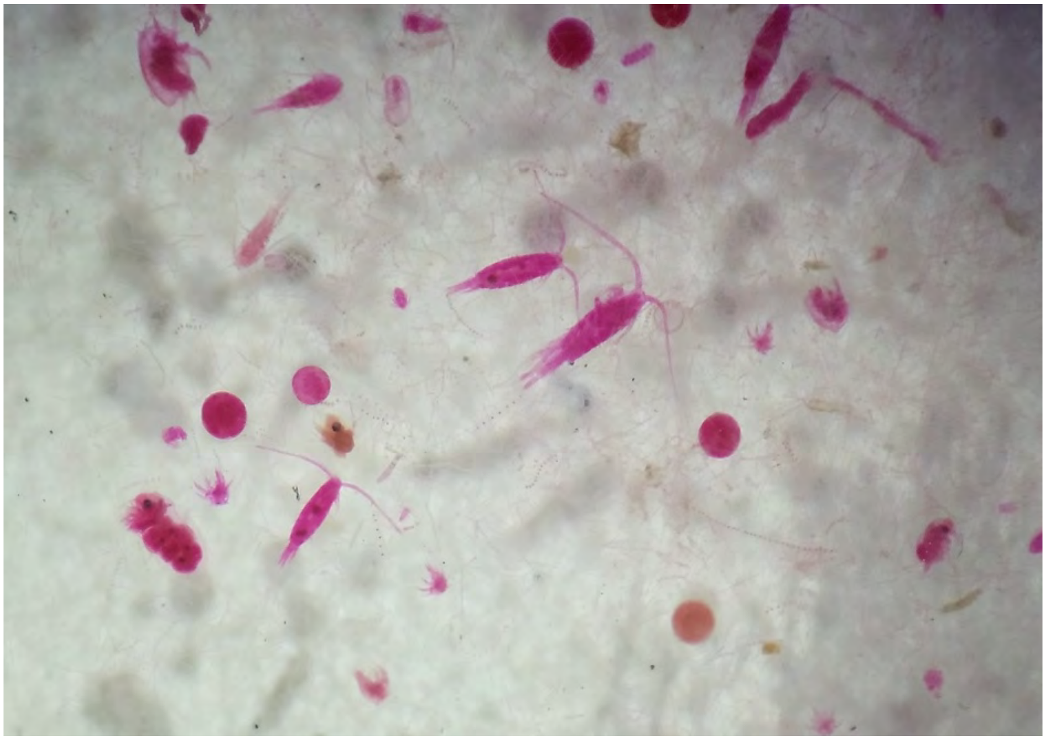

Figure 4-2. Zooplankton collected with vertical tow and stained pink with Rose Bengal to increase visibility.

Benthic invertebrate sampling

Benthic invertebrate samples were collected between July 17 and 27, 2016, primarily from Agnew, Flora, and Horsey Banks. Additionally, we collected one sample each from Porpoise Channel, Stapledon Passage, and Kinahans Bay (Figure 4-3). Samples were collected from small and large purse seine sampling stations sampled in 2016. Additional benthic invertebrate sampling stations on Agnew, Flora, and Horsey Banks were selected by generating random coordinates using ArcGIS software. Altogether, benthic invertebrates were collected from 25 sampling stations using a Van Veen-type grab sampler deployed from a small vessel. For each sampling event, the sampler was lowered by hand from the vessel to the substrate, and quickly retrieved after it reached the substrate and closed. Contents were rinsed through a 500-micron sieve, and the remaining fraction was collected into jars and preserved in 10% formaldehyde. Samples were shipped to Biologica Environmental Services Ltd. for taxonomic analyses. Benthic invertebrates were identified to the lowest possible taxon, usually to species.

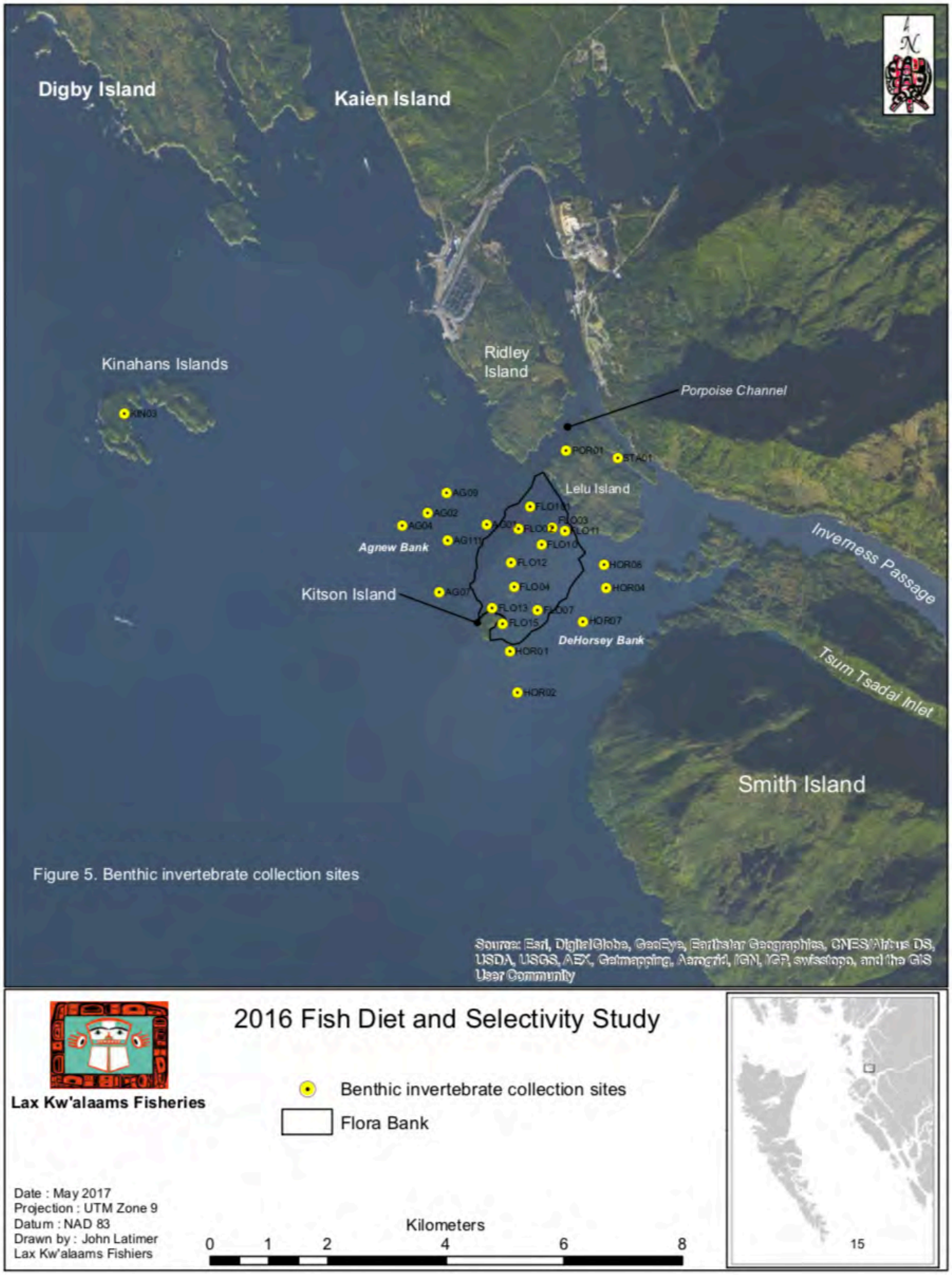

Figure 4-3. Map of benthic invertebrate collection sites sampled in 2016 (map created by John Latimer, Lax Kw'alaams Fisheries).

4.1. Feeding patterns of juvenile salmon

Q1 - What do juvenile salmon species eat in the estuary? How does this compare to common forage fish species like herring and surf smelt?

METHODS

In order to answer these questions, we collected and analyzed stomach samples from salmon and other abundance forage fish species (herring and surf smelt). These methods are described in detail above. We looked at general patterns of food consumption by sockeye and coho salmon, along with consumption by herring and smelt for comparison. Next, we calculated two indices to describe the consumption and selectivity of zooplankton prey to determine what prey are most important and selected for by juvenile salmon, herring, and smelt.

First, an Index of Relative Importance (IRI) was calculated to determine how much of each zooplankton species contributed to a fish species' total diet, essentially providing a measure of importance for each zooplankton species (Bottom & Jones 1990). The formula for an IRI is as follows: IRI = (% prey abundance + % prey biomass) x (% frequency of occurrence). By considering abundance and weight relationships, this index scores rare prey lower than common prey and helps standardize IRI scores across varying individuals. For example, one larval fish may account for a high percentage of prey biomass but a low percent of abundance, while a smaller copepod may do the opposite.

Second, Chesson's α-electivity index (Chesson 1978; Robert et al. 2008) was used to relate the abundance of food available in the stomach to the abundance of food available in the environment. This prey selectivity analysis provides a measure of how fish species may be selecting for certain zooplankton species. Chesson's (1978) alpha: $\alpha_j = \frac{d_j}{p_j} / \sum(\frac{d_i}{p_i})$, for $i = 1, \ldots N$ where N is the number of prey taxa considered, (d_j/p_j) the relative frequency ratio of prey j in the diet (d) and in the plankton (p) and $\sum(d_i/p_i)$ the sum of this ratio for all prey taxa. We removed some species from analysis including prey that could readily avoid capture in the plankton net (e.g. larval fish, crab megalopa, cumaceans, and isopods) or occurred in less than 5% of tow samples (e.g. terrestrial insects) because they artificially inflated the electivity denominator (p) due to systemic sampling error or general rarity (Brodeur et al. 2011).

RESULTS

Patterns of prey consumption

All of the fish species examined during our sampling period actively fed on estuary resources at all sites that were surveyed. Very few of the fish stomachs submitted for diet analyses were empty, indicating active feeding close to the time of capture. All focal species (coho, sockeye, herring, and surf smelt) consumed a combination of benthic, epibenthic, planktonic, terrestrial, and fish prey, and the relative importance of each prey group varied by species. Juvenile coho salmon consumed the highest proportion of fish prey, sockeye salmon consumed the highest proportion of benthic prey, and herring and surf smelt consumed the highest proportions of planktonic prey throughout the sampling season. For juvenile salmon, the relative proportions of benthic, planktonic, terrestrial, and fish prey varied by sampling location and date of sampling. Juvenile coho salmon stomachs contained different combinations of the different prey groups for almost every sampling event. Juvenile sockeye stomachs collected in April and May contained more planktonic prey than those collected in June, which contained more benthic prey.

There was variation in diet for sockeye, coho, herring, and smelt in the Skeena River estuary across sites, time periods, and from year to year. Here we highlight results from the overarching trends and present figures with more details about the variation in diet for each species.

Sockeye

Compared to coho salmon, herring, and smelt, sockeye salmon consumed the most benthic prey (Figure 4-4), and their stomach contents revealed heavy feeding activity on resources unique to the study area. Harpacticoid copepods were the most important contributor to sockeye salmon diet and were the most highly selected for in the environment. This is demonstrated by the highest mean IRI score for harpacticoid copepods (2X higher than the next highest ranked prey), and the highest electivity index (Figure 4-5b). This was followed by decapod zoea, amphipods, and barnacle cyprids (Figure 4-5b). Harpacticoid copepods, which were present in 57% of sockeye stomachs examined, appeared in sockeye salmon stomachs from June 1 onwards at sites on Flora Bank, off Kitson Island, Porpoise Channel, and Inverness Passage following the peak of the sockeye salmon migration. It is interesting to note that five individuals also consumed adult stages of the salmonid parasite from the family Caligidae (Figure 4-5b), which has also been observed in southern British Columbia (Price et al. 2013b).

Harpacticoid copepods were a key diet item for juvenile sockeye and chum salmon at sampling stations on Flora Bank and off Kitson Island. Although chum salmon were not a focus of the current study, the small sample of juvenile chum salmon stomachs submitted for diet analyses suggest that they also consume large numbers of harpacticoid copepods. While previous studies have identified harpacticoids as important prey for chum salmon in estuaries (Healey 1982), their importance to juvenile sockeye salmon is less documented. We observed relatively low abundances of harpacticoid copepods during benthic invertebrate sampling despite their high prevalence in sockeye and chum salmon stomach contents, which may be the result of feeding pressure. In the Nanaimo River estuary, chum salmon grazed most of the standing biomass of harpacticoids in a single migration season and left the estuary when this prey type was no longer available (Healey 1982). To assess the degree to which salmon predation may deplete benthic invertebrate populations, particularly harpacticoid copepods, annual monitoring should be conducted and should be carried out before and after the salmon migration period.

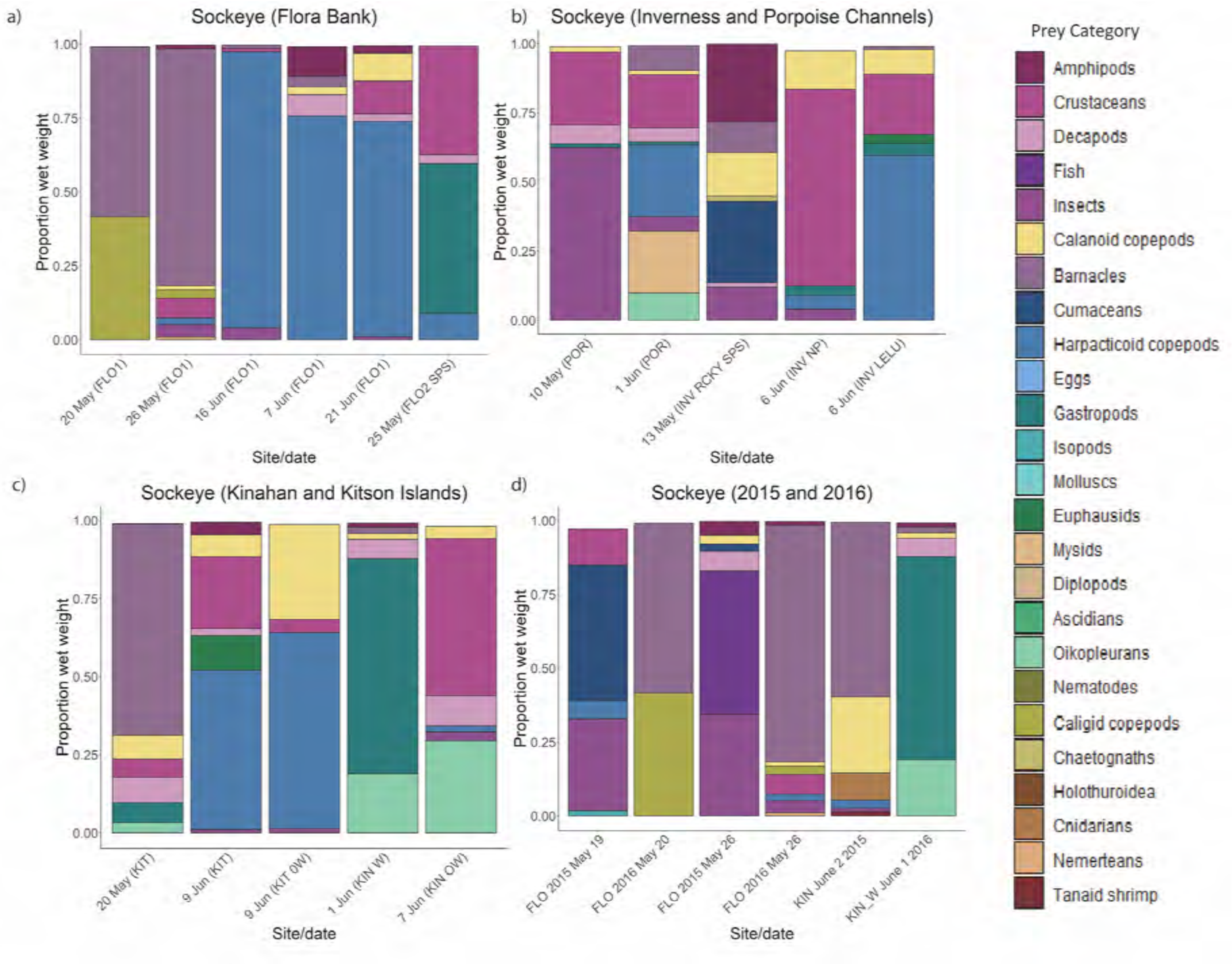

Figure 4-4. Mean proportion of wet weight of prey categories for sockeye salmon sampled during May and June at Flora Bank, Kinahan, and Kitson Islands, and Inverness and Porpoise Channels in 2016 (a, b, c), and Flora Bank and Kinahan Islands on similar dates in 2015 and 2016 (d) (amended from Carr-Harris 2016).

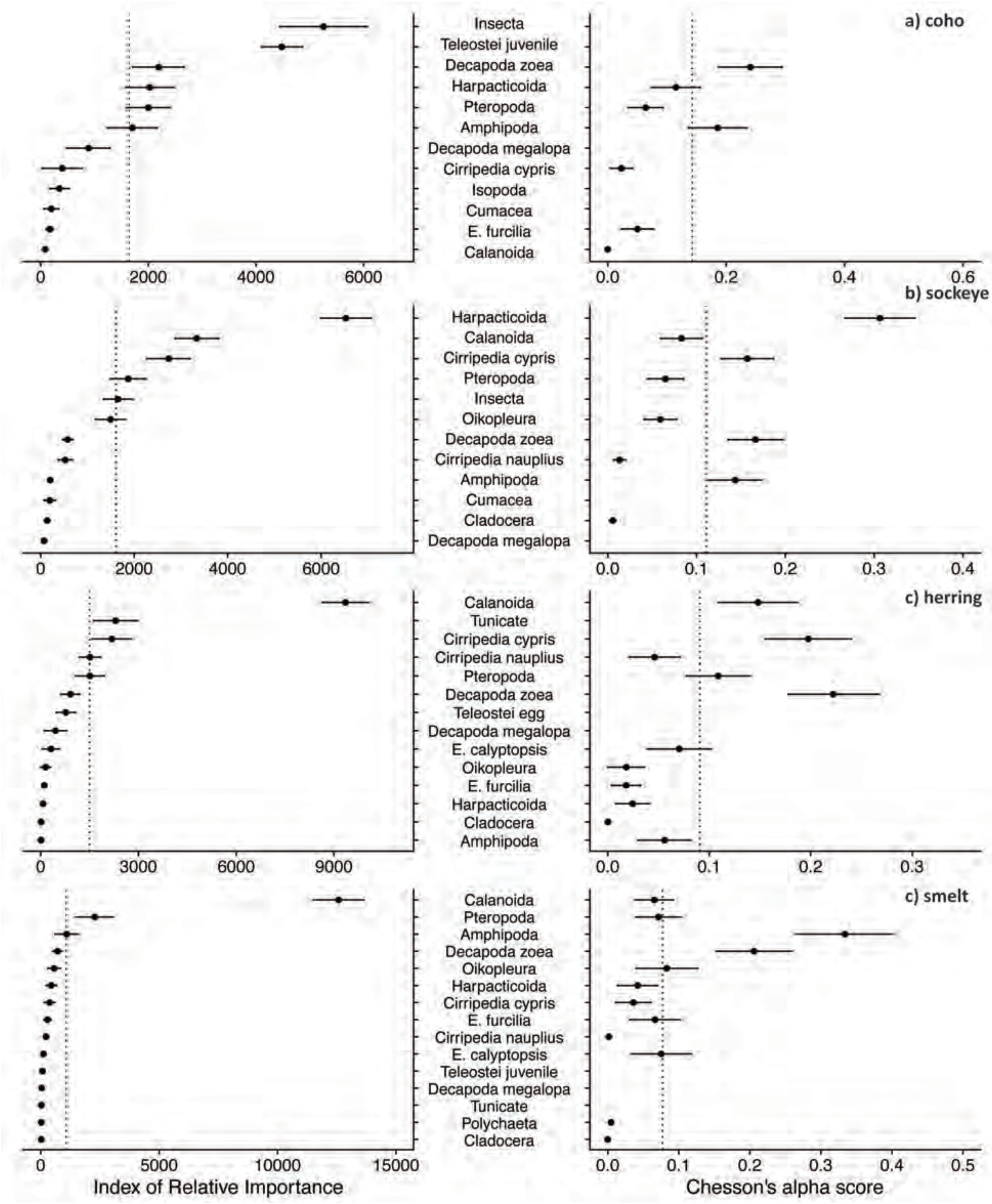

Figure 4-5. Index of relative importance (IRI) and Chesson's electivity scores of each prey species with standard error for a) juvenile coho salmon, b) juvenile sockeye salmon, c) Pacific herring, and, d) surf smelt in the estuary. Dotted lines represent average IRI scores for sockeye salmon and the neutral alpha selectivity threshold for each fish species. An alpha score above the dotted line suggests that a species was selected for consumption more than is present in the environment (amended from Arbeider et al. 2019).

There was considerable variation in sockeye salmon stomach contents between sampling stations and sampling years (Figure 4-4). For example, although the small purse seine sampling station FLO2 was <1 km away from the large purse seine sampling station FLO1, samples captured one day apart on May 25 and 26, respectively, revealed considerable differences in prey composition. Sockeye salmon stomachs from the FLO1 site contained mostly barnacle cyprids, and small numbers of planktonic crustaceans and terrestrial insects, while molluscs and planktonic crustaceans were the most important components in stomachs taken from the FLO2 site. Similarly, there was variation in the relative proportions of prey taxa observed in the stomachs of sockeye salmon captured in two sets collected on the same day at two sites in Inverness Channel, with notably lower abundances of planktonic crustaceans and higher abundances of harpacticoid copepods at INV-Lelu compared with the more riverine INV-NP site. Previous surveys of juvenile sockeye salmon diet have reported high variability in sockeye salmon stomach contents and selectivity of food items with respect to the available zooplankton (Healey 1991; Price et al. 2013b). Juvenile sockeye salmon consume a variety of prey in the months following marine entry. Previous diet studies conducted in northern British Columbia found copepods, euphausids, decapods, amphipods, larvaceans, fish, and terrestrial insects in juvenile salmon stomach contents (Healey 1991; Brodeur et al. 2007). Calanoid copepods were a major component of sockeye salmon stomachs in previous studies, but were under-represented compared to the high proportion of calanoid copepods available in the water column (Price et al. 2013b).

Coho and Chinook

Larval fish and terrestrial insects were the largest components of juvenile coho salmon diet (Figure 4-6). Larval and juvenile stages of prey fish were present in 33% of juvenile coho stomachs examined, were more than 4x more abundant than any other prey category found in the stomachs, and accounted for 53% of the total wet weight of coho stomach contents (Figure 4-6). IRI scores for fish and insects were 2.4 and 2.1 higher than the third highest prey Diptera (64% by abundance), followed by Hemiptera (26%), Colleoptera (6%), and others (Collembola, Hymenoptera, Trichoptera, and Ephemeroptera) (Figure 4-5a). Fish were not present in the stomachs of coho salmon sampled in Porpoise Channel or in one set of juvenile coho captured on Flora Bank on May 20 (Figure 4-6). Only 23% of juvenile and larval fish were identified to family or lower, which were either Pleuronectidae (86%) or Pacific herring (14%). Other important prey items for juvenile coho salmon included insects, decapods, and other crustaceans and gastropods, including pteropods.

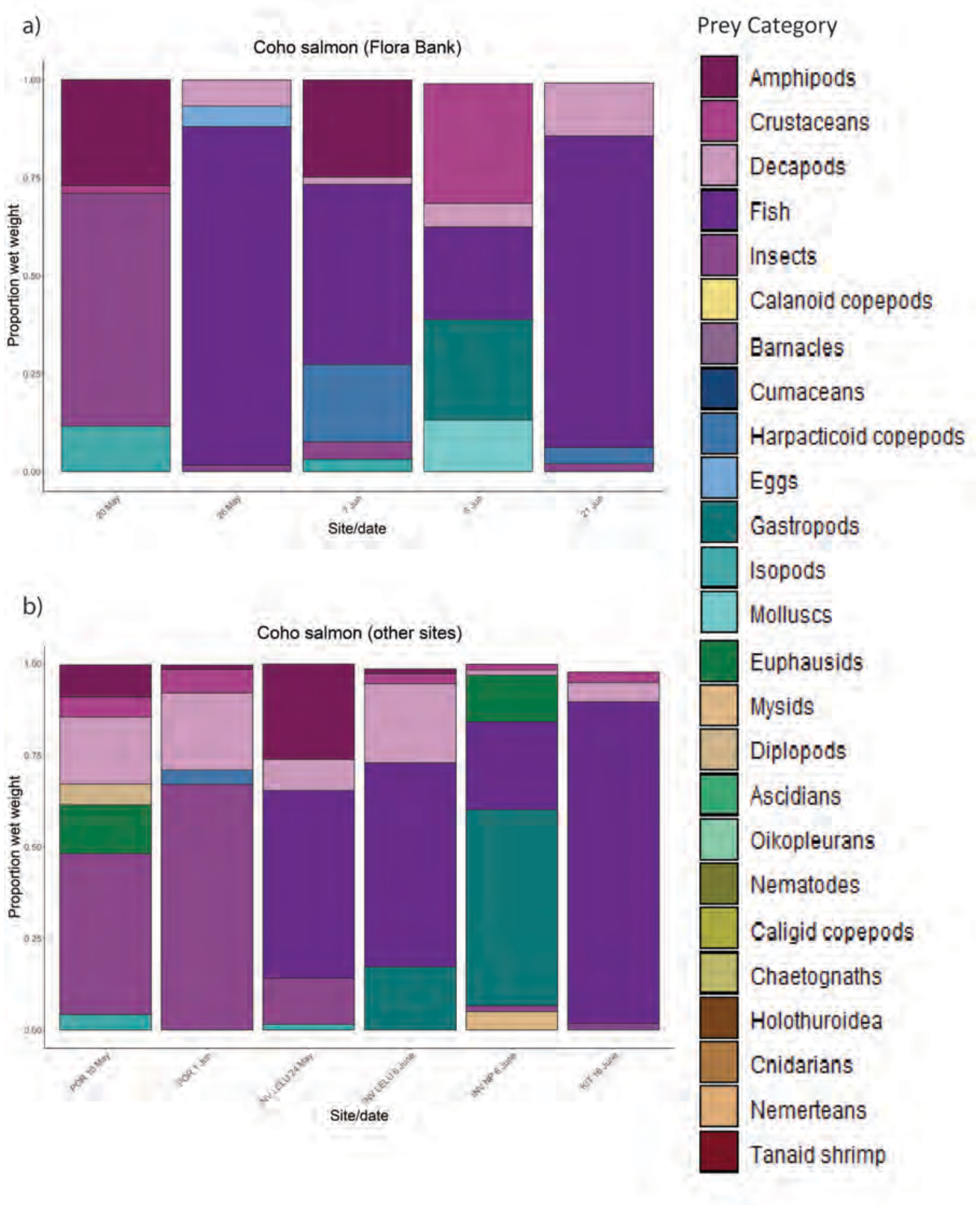

Figure 4-6. Relative abundance (wet weight) of prey items in coho salmon stomachs during May and June sampling events from a) Flora Bank and b) other sampling locations (Porpoise Channel, Inverness Channel and Kitson Island) in 2016.

We submitted a small number of stomachs collected from juvenile Chinook (n=8) and chum salmon (n=10) that were captured with other salmon species during three purse seine sets in Porpoise Channel and on Flora Bank. Although the sample sizes were small, the stomach contents for both species demonstrated some overlap with other salmon species. Fish prey were present in 5 of 8 Chinook stomachs examined and comprised 86% of the total wet weight of Chinook stomach contents (Figure 4-7). As with coho and sockeye salmon, insects comprised the highest proportion of Chinook salmon stomach contents sampled in Porpoise Channel on May 13 (Figure 4-7). The juvenile Chinook salmon captured on Flora Bank on May 20 and June 16 were the most piscivorous of all salmon species (Figure 4-7). The only recognizable fish species in the Chinook salmon stomach contents was a Pacific sand lance (*A. hexapterus*). The stomach contents from the juvenile chum salmon most closely resembled those of juvenile sockeye salmon captured in the same sets, with barnacle cyprids and harpacticoids comprising the highest proportion of chum salmon stomach contents from Flora Bank on May 20 and June 16, respectively (Figure 4-7). Like sockeye, chum salmon fed heavily on harpacticoid copepods with 900 individuals enumerated in a single chum salmon stomach collected from Flora Bank on June 16 (Figure 4-7).

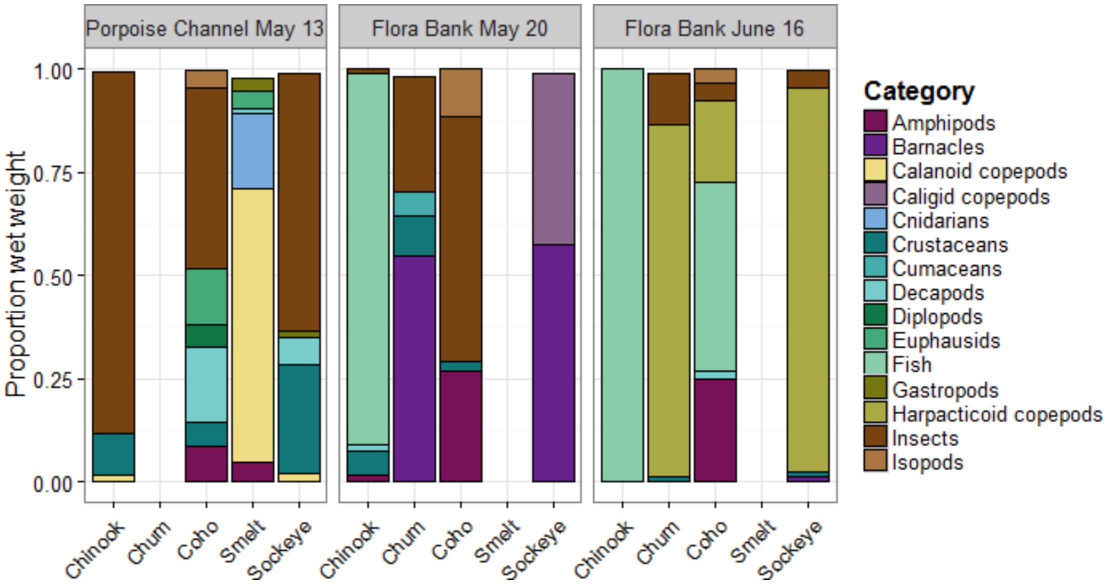

Figure 4-7. Relative abundance (wet weight) of all species of juvenile salmon and surf smelt prey items captured at Porpoise Channel on May 13 and Flora Bank on May 20 and June 16, 2016.

Coho and Chinook salmon consumed the most fish, and in the absence of larval fish prey, appeared to consume terrestrial insects, which is consistent with previous studies (Manzer 1969; Magnusson & Hilborn 2003; Osgood et al. 2016). The scope of the current study did not include detailed surveys of potential terrestrial insect or larval fish prey. While some larval fish were captured in zooplankton sampling, the zooplankton survey did not encompass the full size range of larval fishes consumed by juvenile Chinook and coho salmon. We observed high abundances of osmerid larvae, most likely larval eulachon, at all zooplankton sampling stations in April, prior to the start of the outmigration period for coho and Chinook salmon. It is not known how long larval eulachon are retained in the estuary following their downstream migration from the Skeena River, however one larval eulachon was identified in zooplankton sampling off Kitson Island as late as September 2016. A detailed study of the timing and availability of larval fish prey and terrestrial insects would contribute to understanding the estuary food web in the Skeena River estuary.

Herring and Surf Smelt

The stomach contents of herring and surf smelt revealed more generalist prey selection than for the juvenile salmon species examined, with higher reliance on pelagic zooplankton. Pacific herring and surf smelt primarily consumed calanoid copepods, a prey which had IRI scores 4 and 5.5 higher than the next ranked prey for herring and surf smelt, respectively (Figure 4-5c and Figure 4-5d). calanoid copepods, which were found in 74% of herring and 91% of surf smelt (Figure 4-8) stomachs examined, comprised the largest component of herring stomach contents overall (31% of total wet weight). Herring selected for calanoid copepods, barnacle cyprids, and decapod zoea (Figure 4-5c), while surf smelt only selected for amphipods and decapod zoea, with neutral affinity for calanoid copepods (Figure 4-5d).

Summary

Stomach contents analyses revealed estuary resource partitioning between juvenile salmon and forage fish species, and between different species of juvenile salmon, with considerable diet overlap observed between coho and Chinook salmon, and between chum and sockeye salmon. Forage fish species (herring and surf smelt) consumed the highest proportions of planktonic prey, including calanoid copepods and other crustaceans. Amphipods, euphausids, and different groups of decapods were also important and preferred prey items for coho, sockeye, and Chinook salmon. Peak abundances of calanoid copepods off Kitson Island in the middle of June coincided with the highest observed abundances of Pacific herring and surf smelt at the same location.

The high degree of selectivity exhibited by sockeye salmon may relate to the quality of available prey (Trudel et al. 2007; Tanasichuk & Routledge 2011). For example, growth and survival of juvenile coho salmon has been correlated with the lipid content of consumed prey (Orsi et al. 2004; Trudel et al. 2007). Sockeye salmon returns to Alberni Inlet, British Columbia were strongly correlated with the abundance of one species of euphausid (*Thysanoessa spinifera*) during their first summer at sea (Tanasichuk & Routledge 2011). Therefore, decapods and euphausids, which were minor components of the total zooplankton biomass but have higher lipid content than calanoid copepods (Tanasichuk & Routledge 2011), may be disproportionately important for juvenile salmon and forage fish growth in the estuary even though calanoid copepods were the most numerous species captured in our zooplankton survey.

The relative importance of different prey sources to different fish species varied across the different locations surveyed and throughout the sampling season, likely in response to prey availability. Diet composition varied for juvenile sockeye salmon captured in adjacent habitats with the two different nets; the large purse seine which sampled deeper habitats and the smaller purse seine which sampled shallower habitats. Juvenile coho and sockeye salmon captured in Porpoise Channel and Flora Bank earlier in May consumed a high proportion of terrestrial insects and switched to fish prey (for coho) by May 26. Insects were also an important component of sockeye salmon stomach contents collected from Porpoise Channel in early May, but not for any other sampling event in 2016. Harpacticoid copepods, which were the most important prey item identified for sockeye salmon in this study, only appeared in sockeye salmon stomachs in June, replacing barnacle cyprids as the most common diet item in sockeye salmon stomachs collected from Flora Bank. Although juvenile sockeye salmon are thought to spend the least amount of time in the estuary of all salmon species, Moore et al. (2016) found that some individuals remained in the estuary for longer, with the longest estuary residence times estimated for juvenile sockeye salmon that were captured later in the season. Thus, juvenile sockeye captured in late June may feed on estuary resources, including harpacticoid copepods, for a period of time exceeding 10 days and may switch between planktonic and benthic resources depending on prey availability.

Section 4.1 - Q1 - Key Findings

1) Salmon and forage fish species actively feed in the Flora Bank region on a wide variety of estuary resources originating from terrestrial, benthic, planktonic, and larval fish sources.

2) Coho salmon consumed high proportions of fish larvae and insects when fish larvae were not available.

3) Harpacticoid copepods, which are benthic, proved to be the single most important prey item for juvenile sockeye salmon.

4) Forage fish species, herring and smelt, fed on calanoid copepods, the most ubiquitous prey in the estuary.

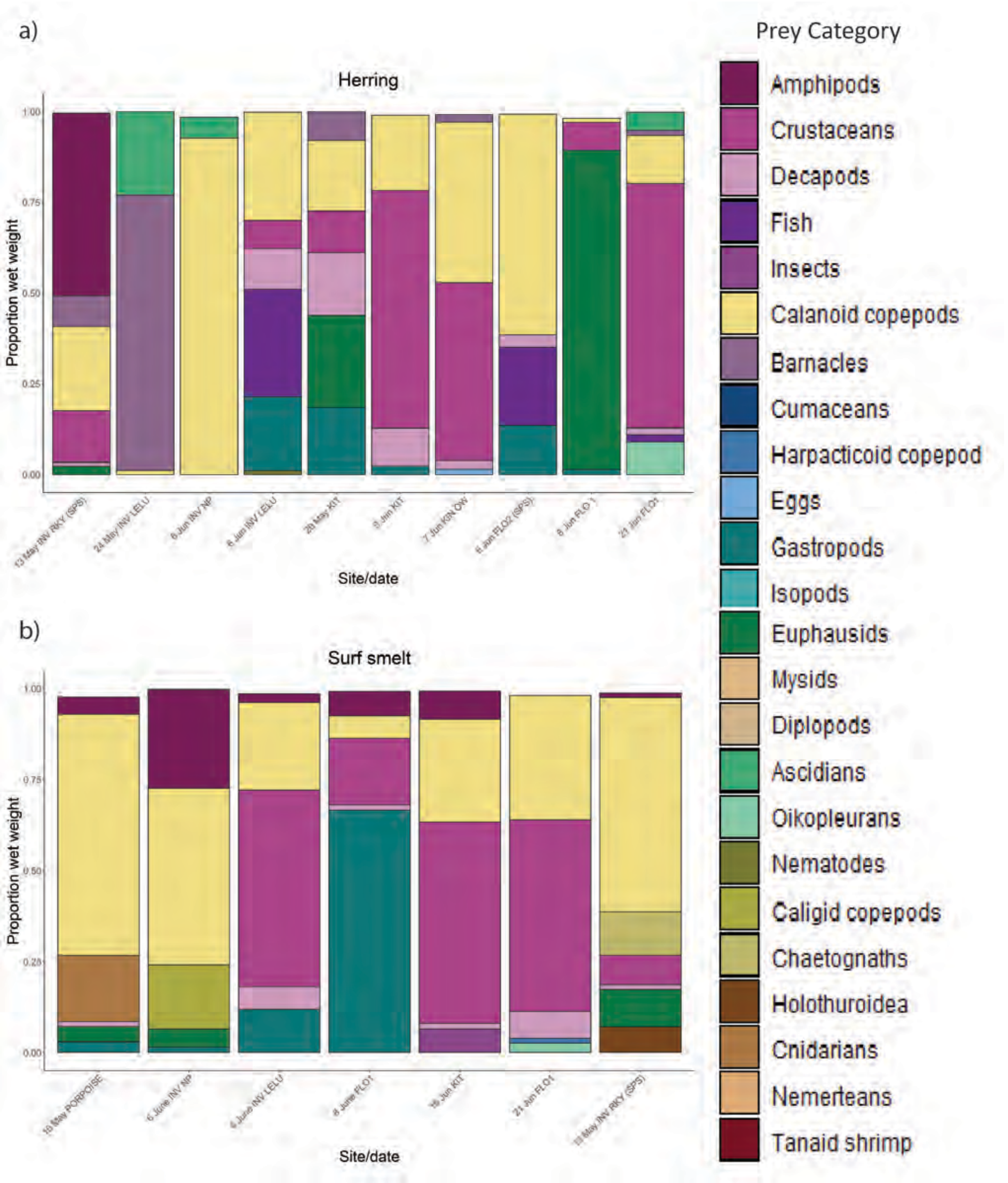

Figure 4-8. Relative abundance (wet weight) of prey items from a) Pacific herring and b) surf smelt stomach samples captured at all sampling locations from May 13–June 21, 2016.

4.2. Dynamics of benthic and zooplankton communities

Q2 - *What are the spatio-temporal dynamics of available benthic and zooplankton communities in the estuary?*

We examined the food available to fish in the estuarine food web by sampling for benthic invertebrates and zooplankton in the water column. Benthic invertebrate sampling occurred across a smaller spatial and temporal scale compared to zooplankton. We first describe the benthic community sampled around the Flora Bank and Kinahans Island locations from July 17 – 21, 2016 (Figure 4-3), followed by results describing the zooplankton community sampled by oblique tows at fixed locations between April 2016 and February 2018 (longer temporal scale). Information from vertical tows was used to explore the spatio-temporal dynamics of zooplankton communities during spring and summer of 2016. Methods for field collection and laboratory processing for benthic sampling, oblique tows, and vertical tows are provided at the start of Section 4.

RESULTS

Spatial description of benthic invertebrates

A total of 24 benthos samples were submitted for taxonomic analysis, of which 7 were drawn from Agnew Bank, 10 from Flora Bank, 5 from Horsey Bank, and one each from Stapledon Passage, Kinahans Bay, and Porpoise Channel (Figure 4-3). Benthic invertebrates were collected once from all sampling stations between July 17–21, 2016. The benthic invertebrate collection contained considerable diversity. Altogether 6,137 organisms were enumerated, representing 181 unique species in 98 families. The most abundant taxa enumerated in the benthic invertebrate samples were polychaete and oligochaete worms and bivalves. The benthic community structure varied across the different sites surveyed. The highest diversity of benthic invertebrates was found on Agnew Bank where an average of 42 unique taxa were identified among 7 samples submitted. Benthos samples from Agnew Bank exhibited the highest diversity and abundances of bivalves, with at least 29 species represented in the samples (Figure 4-9c). Benthic invertebrate samples collected on Horsey Bank had lower diversity and abundances of benthic invertebrates, with the exception of one station, which had a relatively high count of bivalves, most of which were *Nutricola* spp. (Figure 4-9d).

Flora Bank eelgrass sites had intermediate species diversity, with higher counts of amphipods, gastropods, and egg masses (which were attached to eelgrass blades) (Figure 4-9a). The non-eelgrass sites on Flora Bank had the lowest species diversity of all sites surveyed and were dissimilar with respect to benthic invertebrate community composition (Figure 4-9b). Surprisingly, the highest counts of harpacticoid copepods were from a sandy sample drawn from the northeast quadrant of Flora Bank (Figure 4-9b). The lowest diversity and abundances of benthic invertebrates of all sites surveyed were found in Flora Bank samples collected from the southeast and northeast corners of Kitson Island, which had unique taxon counts of only 1 and 4 species, respectively (Figure 4-9b and Figure 4-9e).

Single samples submitted from Porpoise Channel and Stapledon Passage had relatively high abundances of benthic invertebrates compared to samples from Agnew, Horsey, and Flora Banks (Figure 4-9). The sample from Stapledon Passage had the highest overall abundance of benthic invertebrates, mostly polychaetes, and relatively low species diversity. The Stapledon Island sample contained the second highest count of harpacticoid copepods observed in this study. The sample from Porpoise Channel had higher diversity with 24 unique taxa. The benthic invertebrate sample submitted from Kinahans Bay had relatively low species diversity, with 13 unique taxa (Figure 4-9d).

Benthic invertebrates comprised 8% of identified prey items for coho salmon, 46% for sockeye salmon, and 73% for chum salmon that were examined as part of this study. However, most of the taxa identified in the benthic invertebrate collection were not or were rarely consumed by juvenile salmon. The most important benthic invertebrate species present in fish stomachs included harpacticoid copepods, amphipods, cumaceans, and some marine insects. The highest abundances of these groups were found in different habitats throughout the study area. For example, the highest abundances of amphipods were sampled at Flora Bank eelgrass sites, the highest abundances of harpacticoids were sampled from a single sandy site on Flora Bank and in Stapledon Passage, while the highest abundances of cumaceans were collected in Stapledon Passage and Agnew Bank. It should be noted that benthic invertebrate samples were collected in July after most salmon had migrated through the area.

Spatio-temporal dynamics of the zooplankton community

Calanoid copepods were by far the most abundant category of zooplankton identified in both oblique and vertical tow samples. In 2016, we observed two peaks of calanoid copepod abundance on June 17 and July 26 (Figure 4-10). Calanoid copepod densities ranged from 214/m^3 at Kinahans Islands on February 2, 2017, to 11,473/m^3 at Kitson Island on July 26. High densities of calanoid copepods were observed around Kitson Island and Inverness Passage in both oblique (Figure 4-10) and vertical tow samples (Figure 4-11) suggesting that this region might be a prey hotspot for calanoid copepods.

The species composition of copepods identified in the zooplankton collection was similar across the different sites surveyed, and dominated by small calanoid copepods including *Acartia longerimis*, *Centropages abdominalis*, and *Pseudocalanus* spp for most of the year. Peak abundances of large calanoid copepods, including *Calanus marshallae* and *Calanus pacificus,* were recorded at the beginning of May at all stations, and periodically between the beginning of June and middle of September off Kitson Island (Figure 4-10).

Barnacle nauplii were very abundant in April and May, and were the dominant taxa in zooplankton samples collected at Inverness-Lelu and Kinahan Island sites from the beginning of April until the end of May. Barnacles were the most abundant taxa in Porpoise Channel throughout the salmon migration season from April–July, with densities exceeding 17,000/m^3 on May 13 (Figure 4-10). Although the most abundant stage of barnacles identified in juvenile salmon stomachs were cyprids, they comprised just a small proportion of the total barnacles identified in the zooplankton samples.

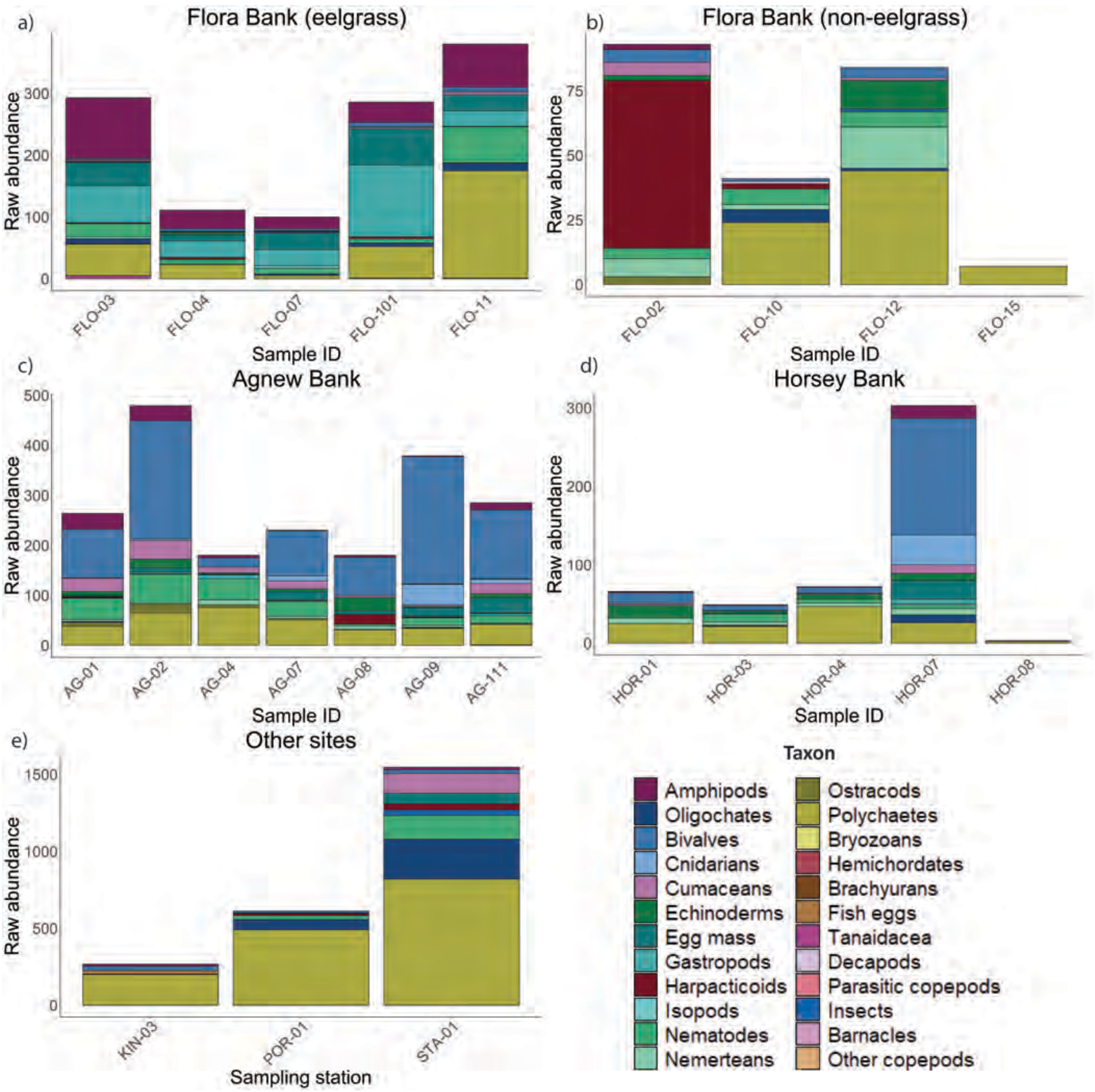

Figure 4-9. Total (raw) abundance of major benthic invertebrate groups sampled on Flora Bank (a and b – eelgrass and non-eelgrass sites), Agnew Bank (c), Horsey Banks (d) and other locations (e) – Kinahans Island, Porpoise Channel, and Stapleton Passage in July 2016. Note different scales on y-axis of each panel.

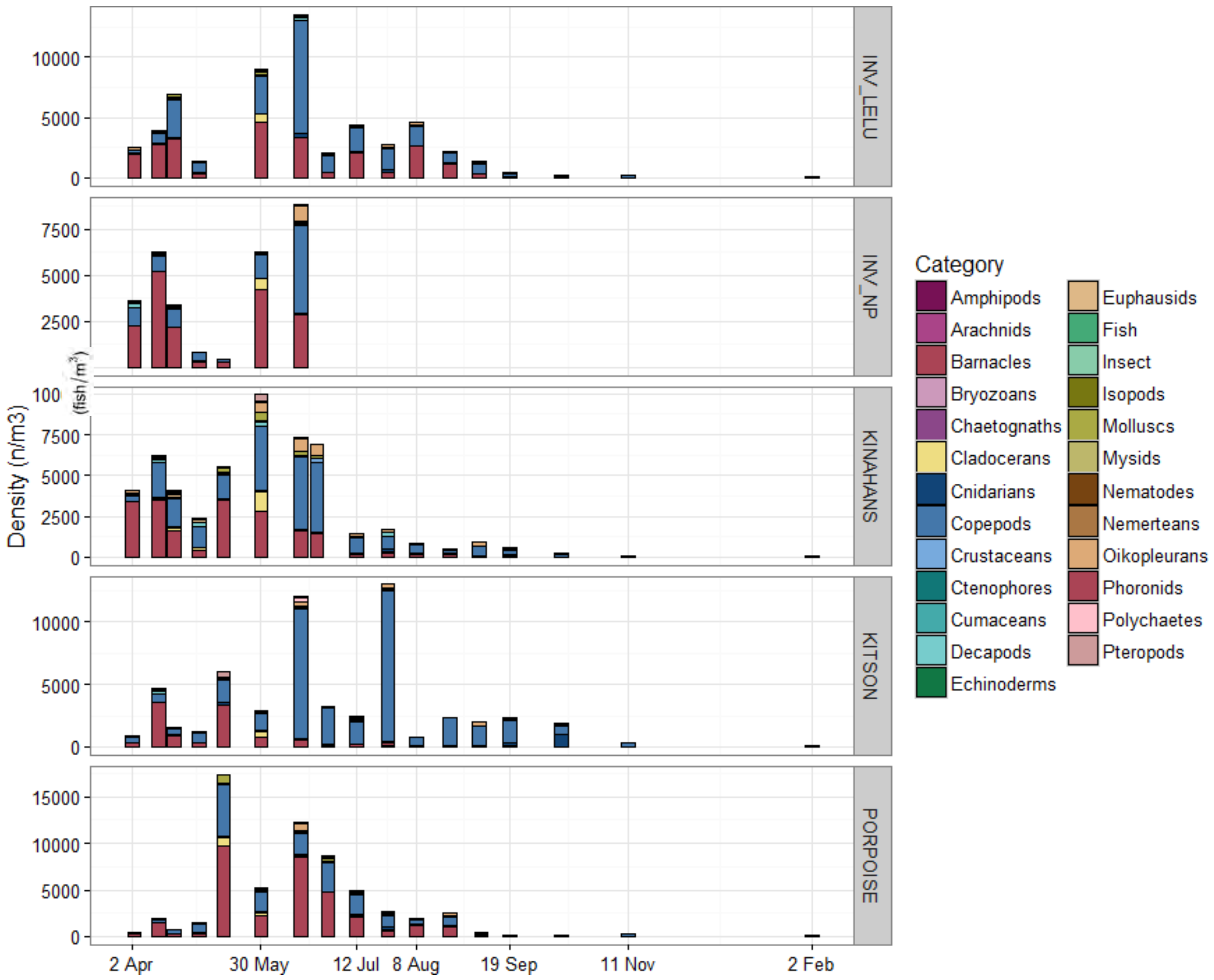

Figure 4-10. Overall densities of major categories of zooplankton sampled at 5 sites from April 2–November 11, 2016. Note different scales on y-axes for all sites.

The timing and abundances of rarer zooplankton taxa, including those that were also identified as prey items in the contents of fish stomachs (Section 4.1), varied across sampling stations throughout the study period. Euphausid densities ranged from 0.1–210 individuals/m^3 and were most abundant at Kinahans throughout April and May, and off Kitson Island in July and September (Figure 4-10). Pteropod blooms were observed at Kinahans on May 24 and off Kitson Island on May 30 with densities exceeding 400 individuals/m^3 on both occasions. Hyperiid amphipods were most abundant in the middle of May in Porpoise Channel and the middle of July at Inverness-Lelu and Kitson Island, and in the middle of July and in September at Kinahans. Oikopleurans densities ranged from 16–791 individuals/m^3, with the highest abundances observed at all sites on June 16.

The zooplankton collection included at least 10 families of decapods, most of which were captured as zoea. The most abundant decapod taxa in the zooplankton collection included caridean shrimp; spider crabs (Majidae), which were most abundant in April and May at all sites; and ghost shrimp (Callianasidae), which were more abundant from June–August at Inverness-Lelu and Porpoise Channels. Decapods were important prey for sockeye and coho salmon, herring, and surf smelt. Brachyuran (crab) zoea were the most common decapods identified in fish stomach contents.

We captured larval fish at all zooplankton sites surveyed throughout the sampling period. In total, 48 species of larval fish were captured representing 18 distinct larval fish families (Table 4-1). Abundant families included Cottidae (sculpins), Gadidae (cods), Pleuronectidae (flounders), Sebastidae (rockfish), and Clupeidae (herring). Diverse families of fish captured included sculpin (10 species) and flatfish (8 species). In addition, large abundances of larval eulachon were present at all sampling stations in the middle of April, concident with the outmigration of larval eulachon.

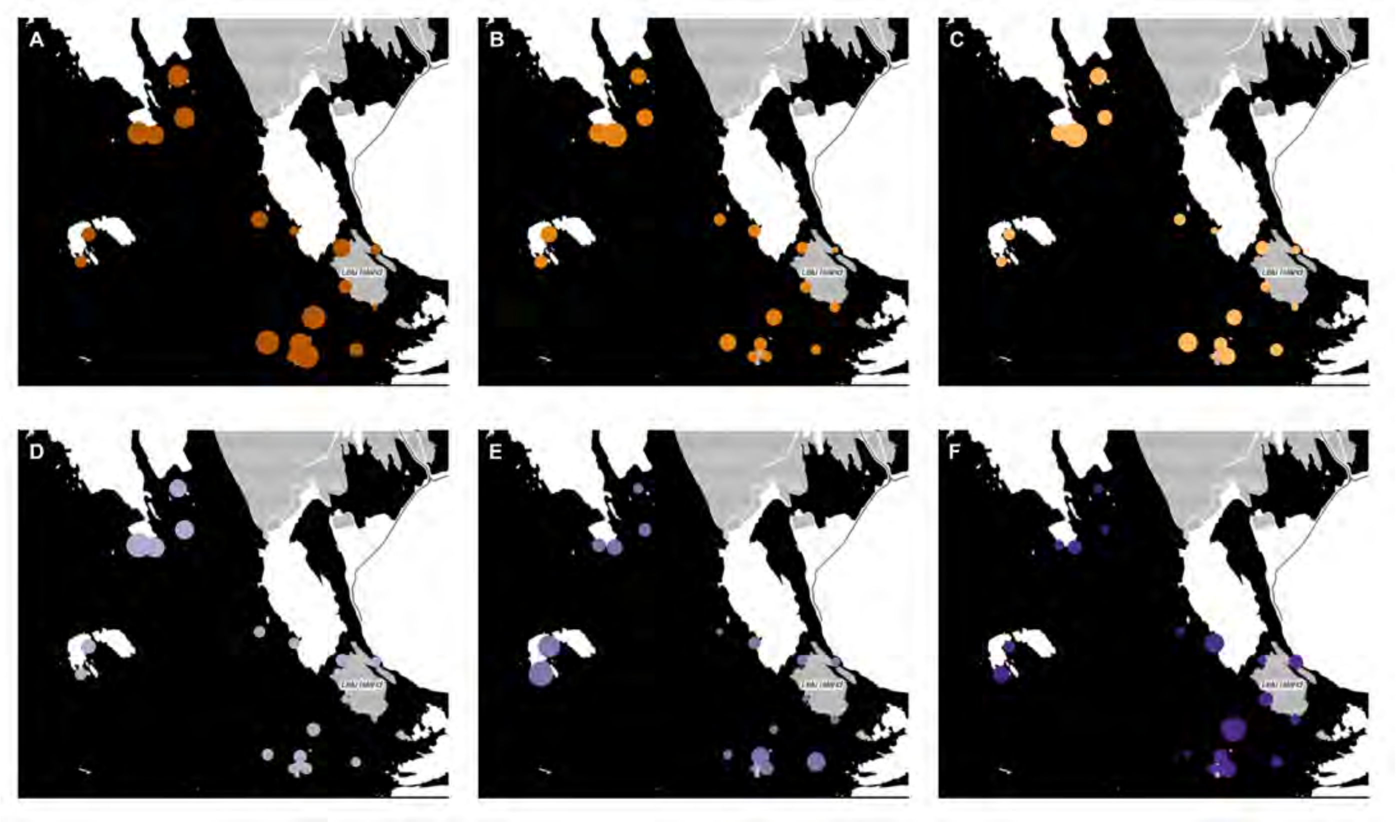

Figure 4-11. The spatial distribution and relative mean abundance of a) Calanoida, b) Pteropoda, c) Decapoda zoea, d) Oikopleura, e) Cirripedia cypris, and f) Harpacticoida across sampling sites in the Skeena River estuary (water is black). Bubble size is scaled by the relative abundance within a species, i.e. the size of bubbles is not comparable between species, only within. Large bubbles of species that had site level differences (top row) could represent hotspots for those prey.

Linking zooplankton abundance with estuarine conditions

We used generalized linear mixed models (GLMMs) to determine if zooplankton abundance was linked to estuarine conditions. Single fixed-effects GLMMs were fit for each zooplankton species, to determine if zooplankton abundance was related to temperature, salinity, time of day, main habitat type (eelgrass, sandy bay, rocky shore, or open water), and site distance from shore. A random effect for site was included in each model to account for the covariation within sites that may be present across time. We created models for zooplankton groups with higher than average IRI and alpha electivity scores because they were deemed to be important contributors in the juvenile salmon diet analysis.

Salinity was correlated with the abundance of several groups of zooplankton. Calanoid copepod and oikopleuran abundance were positively correlated with salinity. This may be a reflection of these species' natural salinity tolerances, as salinity gradients can drive zooplankton distributions (Telesh & Khlebovich 2010). Barnacle cyprid and pteropoda abundance were negatively correlated with temperature, however, these relationships were weak and likely driven by outliers. No biophysical factors predicting decapoda zoea abundance had statistical support.

Harpacticoid copepods were found in higher abundances on eelgrass habitat compared to rocky shores, sandy bays, and open-water habitats. Eelgrass is known to support higher densities of harpacticoid copepods and is likely a population source for them (Hosack et al. 2006; Kennedy et al. 2018). Multiple studies have shown that juvenile salmon are capable of consuming large proportions of total harpacticoid production (Healey 1979; Godin 1981; Fujiwara & Highsmith 1997). Therefore, degradation of eelgrass, and thus prey productivity, could affect salmon foraging behaviour and survival. Exploring these food web habitat connections is an important step in understanding the potential habitat value of estuaries for migratory juvenile salmon and forage fish.

Table 4-1. Larval fish species captured during zooplankton tows conducted between April–November 2016 and in February 2017, with fish identified to species when possible (continues on page 131).

Family	Species	Common Name
Agonidae	*Odontopyxis trispinosa*	Pygmy poacher
Ammodytidae	*Ammodytes hexapterus*	Pacific sand lance
Anoplopomatidae	*Anoplopoma fimbria*	Sablefish
Bathymasteridae	*Ronquilus jordani*	Northern ronquil
Bythitidae	*Brosmophycis marginata*	Red brotula
Clupeidae	*Clupea pallasi*	Pacific herring
Cottidae	*Artedius fenestralis*	Padded sculpin
	Artedius harringtoni	Scalyhead sculpin
	Artedius lateralis	Smoothhead sculpin
	Clinocottus acuticeps	Sharpnose Sculpin
	Clinocottus embryum	Calico sculpin
	Cottus asper	Prickly sculpin
	Enophrys bison	Buffalo sculpin
	Leptocottus armatus	Pacific staghorn sculpin

Table 4-1 cont'd. Larval fish species captured during zooplankton tows conducted between April–November 2016 and in February 2017, with fish identified to species when possible.

Family	Species	Common Name
	Radulinus asprellus	Slim sculpin
	Ruscarius meanyi	Puget sound sculpin
Gadidae	*Gadus chalcogrammus*	Walleye pollock
	Gadus macrocephalus	Pacific cod
	Gadus sp.	Cod
Gobiidae	*Lepidogobius lepidus*	Bay goby
	Rhinogobiops nicholsii	Blackeye goby
Hexagrammidae	*Oxylebius pictus*	Painted greenling
Liparidae	*Liparis callyodon*	Spotted snailfish
	Liparis fucensis	Slipskin snailfish
	Liparis sp.	Snailfish
Osmeridae	*Hypomesus pretiosus*	Surf smelt
	Mallotus villosus	Capelin
	Thaleichthys pacificus	Eulachon
Paralichthyidae	*Citharichthys sordidus*	Pacific sanddab
Pholidae	*Pholis laeta*	Crescent gunnel
Pleuronectidae	*Hippoglossoides elassodon*	Flathead sole
	Lepidopsetta bilineata	Southern rock sole
	Lepidopsetta polyxystra	Northern rock sole
	Lyopsetta exilis	Slender sole
	Parophrys vetulus	English sole
	Platichthys stellatus	Starry flounder
	Pleuronichthys coenosus	C-O sole
Psychrolutidae	*Psychrolutes sigalutes*	Soft sculpin
Sebastidae	*Sebastes sp.*	Rockfishes
Stichaeidae	*Anoplarchus purpurescens*	High cockscomb
	Lumpenus sagitta	Snake prickleback

Connecting juvenile salmon with the zooplankton community

Previous research has demonstrated that the study area supports particularly high abundances of ecologically and economically important fish species (Carr-Harris et al. 2015; Moore et al. 2015), and that some species rear in the estuary for extended periods of time. Some of the salmon species captured during sampling, such as Skeena chum salmon, are of conservation concern (Price et al. 2013a). Conserving the integrity of rearing habitats is especially important for maintaining productivity of these species. We identified important prey resources consumed by juvenile salmon and forage fish during the spring of 2016. The different fish species examined as part of this study consume a wide variety of estuary resources originating from benthic, planktonic, and terrestrial sources throughout the project area. Diet composition varied across the different habitats and throughout the salmon migration, and the juvenile salmon and forage fish species that we examined exhibited a high degree of prey selectivity with respect to available zooplankton resources. The forage fish species consumed the highest proportion of zooplankton prey throughout the season. Coho salmon consumed high proportions of fish larvae and insects when fish larvae were not available. Harpacticoid copepods, which are benthic, proved to be the single most important prey item for juvenile sockeye salmon.

The availability of key prey resources shifted across habitat types and throughout the sampling season for all species of juvenile salmon and forage fish examined. For example, the results of our benthic invertebrate survey suggested that the highest abundances of amphipods and benthic gastropods were found in Flora Bank eelgrass habitats, while the highest abundances of harpacticoid copepods were sampled at a non-eelgrass station on Flora Bank.

The timing of the downstream migrations of juvenile salmon and their key zooplankton prey resources varies across years. Here we report the results of zooplankton sampling conducted between April 2016 and February 2017. In 2016, the timing of the juvenile sockeye salmon migration occurred approximately a week earlier than observed in previous sampling years (Lax Kw'alaams, unpublished). This coincided with peak abundances of zooplankton at most of the sampling stations that we examined and with peak abundances of some sockeye salmon prey items that were identified during this study, such as pteropods and some groups of decapods. We found that the prey assemblages in the stomach contents of juvenile sockeye salmon sampled from Flora Bank on similar dates in 2016 were vastly different from samples collected in 2015. Furthermore, the most important prey items identified in the 2015 samples from Flora Bank, including cumaceans, larval fish, and insects, were not present in 2016 samples. The identification and continued monitoring of key prey resources for focal fish species is important for maintaining estuarine food webs.

Lax Kw'alaams Fisheries has conducted annual juvenile fish sampling in the nearshore Skeena River estuary since 2013, and has created a time series of juvenile salmon and forage fish abundance data at regularly sampled sites that will become richer over time. Based on the results of this study, we recommend the following:

1) continued fish sampling to monitor timing, abundance, and distribution of juvenile salmon following their downstream migration,
2) multiple years of sampling from different habitat types to assess salmon diet preferences across the full spectrum of potential prey items,

3) annual zooplankton and benthic invertebrate sampling to capture inter-annual variability in the timing and abundance of these prey resources, and
4) further investigations into the abundance and distribution of larval fish and terrestrial insect prey to provide a better understanding of prey availability for juvenile coho and Chinook salmon in particular, and to assess the effects of clearing forested areas on the availability of terrestrial insect prey in the study area.

Section 4.2 - Q2 - Key Findings

1) The highest abundances of amphipods and benthic gastropods were found in Flora Bank eelgrass habitat while the highest abundances of harpacticoid copepods were sampled at a non-eelgrass station on Flora Bank.

2) Harpacticoid copepods were found in higher abundances on eelgrass habitat compared to rocky shores, sandy bays, and open-water habitats.

3) Calanoid copepods were by far the most abundant category of zooplankton and abundance was positively correlated with salinity.

4.3. Skeena River estuary food web

Q3 - What are the potential food web linkages in the Skeena River estuary?

The following figure is a conceptual diagram of the potential juvenile salmon food web linkages during spring and summer in the Skeena River estuary based on results presented throughout Section 4 (Figure 4-12).

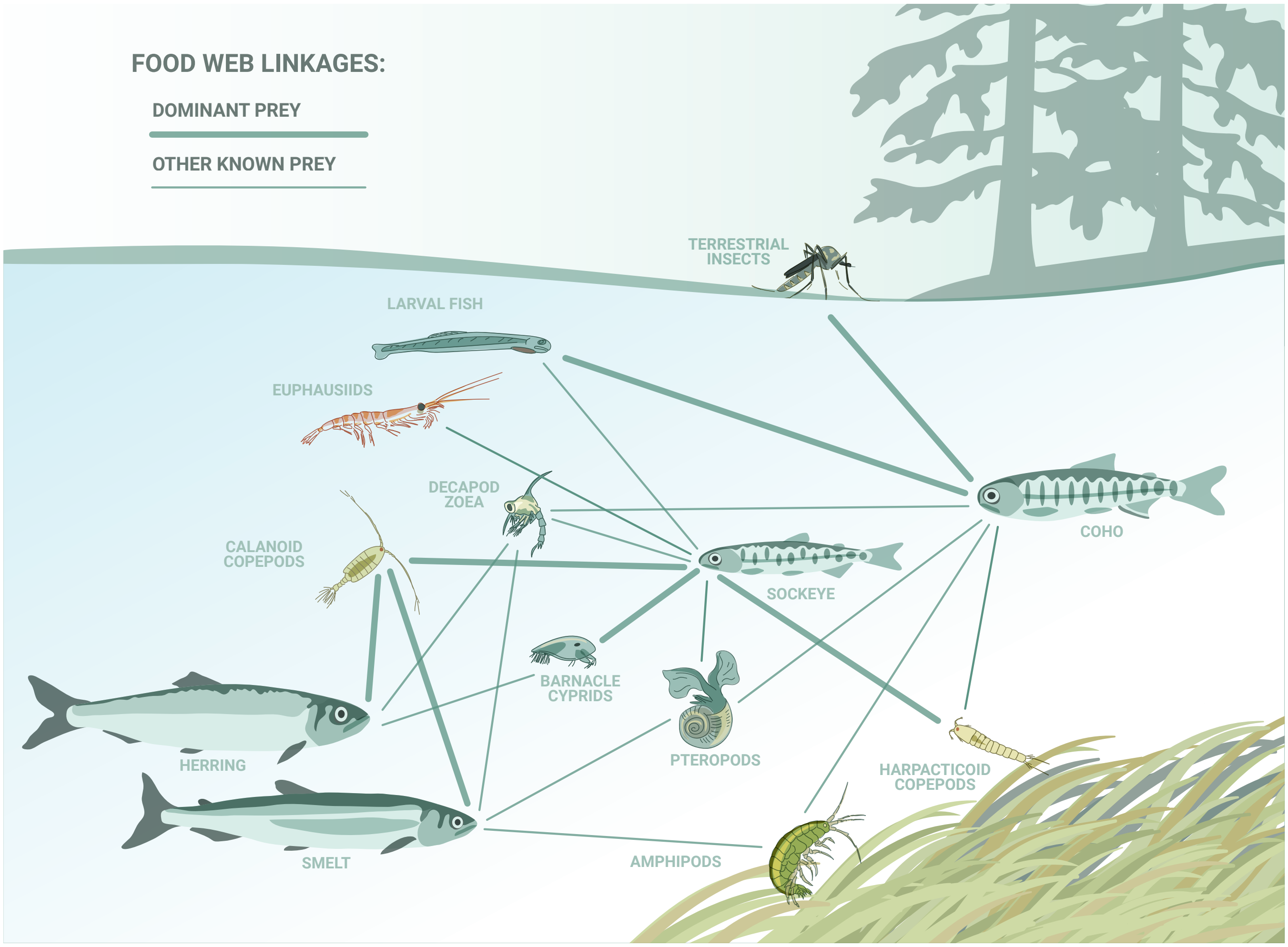

Figure 4-12. Juvenile salmon food web linkages in the Skeena River estuary during spring and summer created using results from Section 4 (created by Fuse Consulting).

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5. FISH AND SHELLFISH

Estuaries provide important breeding and feeding habitat for invertebrates and fish. In addition to Pacific salmon, this crucial habitat supports many First Nations traditional food resources in British Columbia including herring, eulachon, Dungeness crab, clams, and cockles (Williams 1989). For example, larval eulachon may rear in shallow estuarine habitats and juvenile forage fish, such as Pacific herring, feed in estuaries during spring and summer. Estuaries also act as nurseries for juvenile crabs, where they are associated with eelgrass and other aquatic vegetation that provide feeding opportunities and protection from predators (Gunderson et al. 1990; Rooper et al. 2002).

In Section 4, we describe juvenile salmon use of the Skeena River estuary in detail. This section advances our understanding of the use of the Skeena River estuary by other commercially and culturally important fish and invertebrate species within the estuary community. There are only a handful of publicly available resources describing other fish or invertebrate species using the estuary. The following references are listed in Appendix A and represent the available scientific resources on the Skeena River estuary, beyond that of recent research: Higgins and Schewenberg 1973 (fish, benthic and plankton communities), Kelson 2011 (fish), Stantec Consulting Ltd. 2015 (fish), and Stoffels 2001 (eulachon). This section answers questions surrounding the use of the Skeena River estuary by estuary fish species, larval fish, and Dungeness crab. First, the spatial and temporal abundance of estuary fish species in spring and summer is described (5.1), with an emphasis on forage fish species, such as herring and surf smelt. Second, larval fish distribution and speciation is presented with a focus on larval eulachon (5.2). Third, the utilization of Flora Bank region by Dungeness crab highlights the importance of estuary habitat for this species (5.3).

5.1. Spatial and temporal abundance of estuary fish species

INTRODUCTION

Compared to adjacent marine or freshwater environments, estuaries can act as nurseries, supporting fish communities through two main mechanisms. First, turbid waters and nearshore structural complexity (St. John et al. 1992; Fukuwaka & Suzuki 1998; Bottom et al. 2005; Brodeur & Morgan 2016; Levings 2016) in estuarine environments can provide refuge and decrease predation risk to young fish (Beck et al. 2001; Heck et al. 2003; Alofs & Polivka 2004; Sheaves et al. 2015). Second, higher food availability in the estuary than surrounding marine or freshwater ecosystems provides elevated prey resources (Grimes & Finucane 1991; St. John et al. 1992, 1992; Grimes & Kingsford 1996; Hill & Wheeler 2002; De Robertis et al. 2005; Ware 2005; Selleslagh et al. 2012; Brodeur & Morgan 2016). Driven by a combination of detrital production and the mixing of nutrients from marine upwelling and freshwater inputs (Hill & Wheeler 2002; Maier & Simenstad 2009), high basal production provides increased prey resources that contributes to increased growth of juvenile fish (Naiman & Sibert 1979; Sobczak et al. 2002). As larger fish can have increased survival (Sogard 1997), conditions in estuaries favouring increased growth are important to the health of fish populations. Finally, environmental gradients in estuaries facilitate the osmoregulatory transition of migrating fish species, such as juvenile salmon (Thorpe 1994; Zydlewski & Wilkie 2012).

Given that estuaries provide refuge and support growth for fish, it is important to understand how fish communities are using estuaries. All previous studies performed in the Skeena River estuary have small spatial and temporal coverage and many uncertainties remain about estuary fish communities in the Skeena. The data collected during the North Coast Juvenile Salmon Monitoring Program (NCJSMP) that occurred over five sampling years is an ideal dataset to identify fish communities and further explore how fish are using the estuary. While the main focus of the program is Pacific salmon, substantial information has been collected on other estuary fish species. In this section, we investigate the abundance of estuary fish species across time and space. We ask the following questions:

Q1 - What fish species are using the Skeena River estuary from March to August?

Q2 - How does the estuary fish community change over time?

Q3 - What are the size distributions of abundant forage fish like herring and smelt? Do these distributions change over time?

Q4 - How are dominant forage fish species like herring and surf smelt using the estuary spatially?

a) Are there spatial distribution patterns in herring and surf smelt abundance?

b) How are forage fish oriented along environmental gradients in the estuary?

METHODS

As part of the NCJSMP, fish communities in the Skeena River estuary were sampled from 2013–2016 and 2018 by beach seine, purse seine, and trawl gear. Due to a variety of sampling locations and net types used during the NCJSMP, different combinations of gear types and timeframes are used to answer questions in Section 5. Within the Flora Bank region, regular monitoring sites were sampled with beach and purse seining on a weekly basis during spring and summer (Figure 3-3). During the program timeframe, we set 878 nets, a combination of trawl, beach, and purse seine, with the largest sampling effort occurring from 2014–2016. For example, 170 larger purse seine and 54 beach seine sets were completed in 2015. While the entire suite of data across gear types and sampling years is used in Q1 and Q4a, only the data from regular monitoring sites (beach and purse seine) within the Flora Bank region were used to investigate Q2, Q3, and Q4b (2014 – 2016 and 2018). Additional details pertaining to fish sampling methods, timing, and frequency are presented in Section 3.

5.2. *Estuary fish community of the Skeena River*

Q1 -What fish species are using the Skeena estuary from March to August?

We caught 34 species of fish in the estuary using all gear types (trawl, beach, and purse seine) from 2013–2016 and 2018 (Table 5-1). There is considerable overlap of fish species captured during the NCJSMP sampling program and in historical studies (Higgins & Schouwenburg 1973; Kelson 2011; Stantec 2015). The most abundant species of fish caught in the estuary were pink salmon, which were caught early in the spring (April–May) in large schools by beach seining. Similar to historic sampling by purse seining (Higgins & Schouwenburg 1973), catches of sockeye salmon were consistently high from late May through early June. Herring and surf smelt were abundant and consistently caught from

March–August in both NCJSMP and historic sampling (Higgins & Schouwenburg 1973). Capelin (*Mallotus villosus*) was captured during historic (Higgins & Schouwenburg 1973) and more recent studies (Kelson 2011), but no adult capelin were captured during our field program. It is possible that they were misidentified as longfin smelt, but they would have been rare in capture as they are not known to be common in the region (Hart & McHugh 1944). Larval capelin were captured during NCJSMP zooplankton sampling near Kitson and Kinahan Islands and in Porpoise and Inverness Channels on July 12 and 26, 2016 (Table 4-1, Lax Kw'alaams unpublished data). The size of larval capelin identified in the Skeena River estuary ranged from 6.0 - 22mm.

Other abundant species such as surf perch (Embiotocidae), tubesnout (*Aulorhynchus flavidus*), sculpin spp. (Cottoidea), starry flounder (*Platichthys stellatus*), and pricklebacks (*Lumpenus sagitta*) are commonly associated with estuary communities across British Columbia (Hoos & Packman 1974; Hoos & Vold 1974; Bell & Kallman 1976a, 1976b; Morris et al. 1978; Gordon & Levings 1984). Pacific sand fish (*Trichodon trichodon*), found in moderate abundances in the Skeena River estuary (Figure 5-1), have not been reported in BC estuaries further south, but are commonly found in Alaskan estuaries (Thedinga et al. 2006). A list of species caught in the Skeena River estuary and Chatham Sound are listed in Appendix 7.2 of Hoos (1975). All species caught in the NCJSMP are also listed in Table 5-1.

Notably, we caught seven species of salmonids in the estuary: pink, chum, sockeye, coho and Chinook salmon, along with steelhead trout and Dolly Varden. Four species of forage fish were caught consistently across all sampling years: Pacific herring, surf smelt, longfin smelt, and Pacific sand lance. These fish species are known to be important food fish for many species of marine birds, marine mammals and larger fish across British Columbia (Therriault et al. 2009). Thus, we focus on these four forage fish species in the following sections.

Section 5.2 - Q1 - Key Findings

1) We caught 34 fish species over 5 years of sampling (2013–2018) in 878 sets of beach, purse, and trawl seine.

2) The fish catch included seven species of salmonids, including steelhead and Dolly Varden.

3) Sockeye and pink salmon were the most abundant salmon species caught during the NCJSMP spring sampling program in the estuary, while Pacific herring, surf smelt, sandfish, and surf perch were the most abundant forage fish species.

Table 5-1. Summary of species caught in the Skeena River estuary during trawl, beach, and purse seine sampling from 2013–2018 conducted as part of the NCJSMP (continues on page 142). Fish species were identified visually in the field to the lowest known taxonomic level by field crews. Fish species with difficult ID features, such as surf perch, sculpin, and flat fish were not identified to species due to the large catches of fish that need to be processed quickly.

		Number of Sets					
	Gear Type	2013	2014	2015	2016	2018	Total
	beach seine	36	91	54	40	40	261
	purse seine (large)	0	63	170	103	53	389
	purse seine (small)	0	0	0	74	0	74
	trawl	84	65	0	5	0	154

Species Name		Abundance					
Common	Scientific	2013	2014	2015	2016	2018	Total
Salmonid							
juvenile pink salmon	*Oncorhynchus gorbuscha*	12823	90603	5045	7435	259	116166
juvenile chum salmon	*Oncorhynchus keta*	167	307	158	457	39	1128
juvenile sockeye salmon	*Oncorhynchus nerka*	562	3950	1749	8739	4280	19280
juvenile coho salmon	*Oncorhynchus kisutch*	20	443	721	755	281	2403
juvenile Chinook salmon	*Oncorhynchus tshawytscha*	34	79	58	81	93	345
Dolly Varden	*Salvelinus malma*	6	26	48	7	29	116
juvenile steelhead salmon	*Oncorhynchus mykiss*	3	7	4	7	1	22
adult pink salmon	*Oncorhynchus gorbuscha*	0	42	1	0	0	43
adult sockeye salmon	*Oncorhynchus nerka*	0	34	6	0	7	40
adult coho salmon	*Oncorhynchus kisutch*	0	1	0	0	0	1
adult Chinook salmon	*Oncorhynchus tshawytscha*	0	2	0	0	0	2
adult steelhead salmon	*Oncorhynchus mykiss*	0	0	1	0	0	1
Non-salmonid							
Pacific herring (juv. & adult)	*Clupea pallasii*	2224	10378	19512	14237	6390	52741
surf smelt (juv. & adult)	*Hypomesus pretiosus*	432	13989	18290	4867	4486	42064
Pacific sandfish	*Trichodon trichodon*	355	14087	105	621	1	15169
surfperch spp.	Embiotocidae	362	6433.5	143	25	128	7092
three spine stickleback	*Gasterosteus aculeatus*	1	4	5069	4	0	5078

Table 5-1 cont'd. Summary of species caught in the Skeena River estuary during trawl, beach, and purse seine sampling from 2013–2018 during the NCJSMP.

Species Name		Abundance					
Common	Scientific	2013	2014	2015	2016	2018	Total
Sand lance	*Ammodytes hexapterus*	245	830.5	498	203	419	2196
tubesnout	*Aulorhynchus flavidus*	16	177	570	2	2	767
snake prickleback	*Lumpenus sagitta*	78	338	13	1	3	433
sculpin spp.	Cottoidea	106	129	49	18	7	309
gunnel spp.	Pholidae	140	92	28	20	11	291
longfin smelt	*Spirinchus thaleichthys*	2	0	60	74	15	151
flat fish spp.	Pleuronectiformes	28	96	17	3	2	146
starry flounder	*Platichthys stellatus*	8	34	71	11	20	144
Dolly Varden	*Salvelinus malma*	6	26	48	7	29	116
dungeness crab	*Metacarcinus magister*	0	37	67	2	7	113
kelp greenling (juv.)	*Hexagrammos decagrammus*	3	6	1	3	6	19
rockfish spp.	*Sebastidae*	0	11	0	0	6	17
soft sculpin	*Psychrolutes sigalutes*	0	11	2	0	0	13
Pacific spiny lumpsucker	*Eumicrotremus orbis*	9	0	0	0	0	9
Pacific lamprey	*Lampetra tridentata*	1	0	2	2	0	5
Pacific tomcod	*Microgadus proximus*	1	1	2	0	0	4
bay pipefish	*Syngnathus leptorhynchus*	0	0	0	1	1	2
sturgeon poacher	*Podothecus accipenserinus*	0	0	2	0	0	2
Northern anchovy	*Engraulis mordax*	0	0	0	1	0	1
spiny dogfish	*Squalus acanthias*	0	0	1	0	0	1
Northern Clingfish	*Gobiesox maeandricus*	0	1	0	0	0	1
copper rockfish	*Sebastes caurinus*	1	0	0	0	0	1
Pacific hagfish	*Eptatretus stoutii*	0	0	0	0	1	1

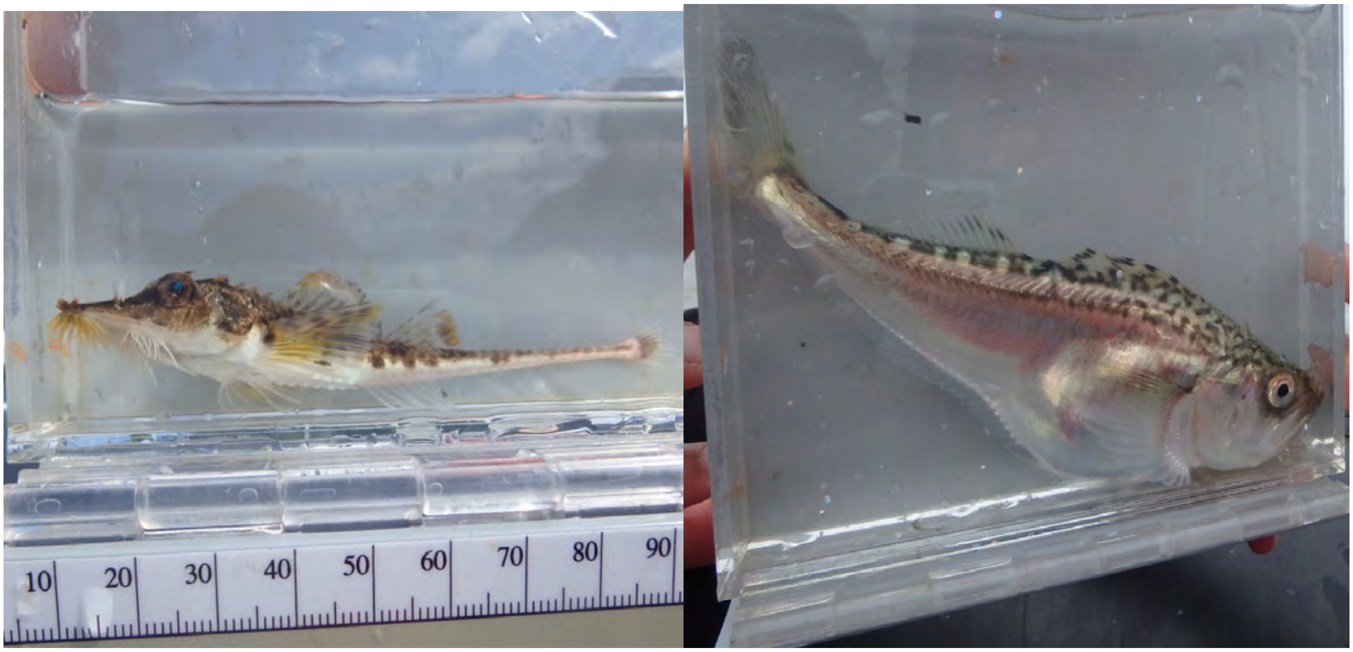

Figure 5-1. Sturgeon poacher (*Podothecus accipenserinus*) and Pacific sandfish (*Trichodon trichodon*) caught in the Skeena River estuary (photos by Ciara Sharpe).

Q2 -How does the estuary fish community change over time?

Estuary fish communities in the Northeast Pacific are comprised of freshwater, marine, and anadromous species (Hoos & Packman 1974; Bottom & Jones 1990). These communities can vary across seasons, with abiotic conditions and life history stages. For example, seasonal changes to freshwater discharge impact the physical environment and can contribute to the temporal distribution and abundance of fish species (Armor & Herrgesell 1985; Bottom & Jones 1990). In addition, anadromous fish species, such as juvenile salmon and eulachon, arrive in the estuary as a seasonal 'pulse' on their migration from freshwater to the ocean. Abundance and diversity of estuary fish communities are influenced by a variety of factors, such as seasonal spawning migrations, reproductive cycles, and the recruitment of juvenile fish that use the estuary as a nursery (Bottom & Jones 1990). We compare the fish community across time in the Skeena River estuary to further define the estuary fish community during the spring and early summer.

METHODS

We used fish catch (CPUE) from regular NCJSMP beach and purse seine sites (Figure 3-3) in the Skeena River estuary from March–August of 2014–2018 to assess trends in fish community composition, diversity, and abundance. Since beach seine and purse seine gear types cannot be compared, we assessed trends based on each type separately. We visually assessed estuary fish community composition and used two metrics of biodiversity to investigate trends over time in early spring and summer months. Rare species, caught only once, were removed from visualization and analysis. Species richness and the Shannon diversity index are commonly used metrics to explain the diversity of a community (Gallardo et al. 2011; Santini et al. 2017). Species richness is the number of species present in a community, while the Shannon diversity index is a measure of species diversity. This index accounts for the relative abundance of different species (evenness). For example, if only one species is abundant in a community of ten species, then this community is considered uneven and thus not diverse.

RESULTS

The estuary community changed in composition, abundance, and diversity during spring and summer in the Skeena River estuary. We discuss the results in the context of the estuary pelagic community (purse seine) and the nearshore community (beach seine).

Composition

Surf smelt and herring were the dominant species sampled by purse seine in the estuary's forage fish community (Figure 5-2a and Figure 5-4a). During mid-March, herring were the most abundant fish species, comprising 80% of the total catch, while surf smelt were the most abundant in early-April, mid-April, and early-May, comprising 85%, 60%, and 80% of the catch, respectively (Figure 5-2a). Later in the season (mid-June to mid-July), herring were the most dominant species, comprising 57% percent of the fish community. An increase in all age classes of herring can be seen during this period, with a pulse of young-of-the-year in mid-July (Figure 5-4a) and an increase of 1+, 2+, and 3+ year cohorts starting in mid-June (Figure 5-9). It is common for fish communities in other northeast Pacific estuaries to be dominated (over 75%) by a few low trophic level species (Horn & Allen 1976; Allen 1982; Gordon & Levings 1984; Bottom & Jones 1990). On average, herring and surf smelt comprised 38% and 39% of the total catch, respectively, representing a mean composition of 77% of the pelagic fish community throughout the spring and summer.

Juvenile salmon species, including sockeye, pink, and coho, increased in abundance starting in April until they represented the majority of individuals sampled in pelagic communities (purse seine sampling) in mid-May and early-June (Figure 5-2a and Figure 5-4a). As the earliest salmon migrants, juvenile pink made up 25% (Figure 5-2a) of the total catch in mid-April, although this likely under-represents the abundance of pink salmon, as during early spring pink salmon are smaller than the purse seine mesh and can escape capture. Pink salmon abundance is more apparent in the nearshore estuary communities captured in the beach seine (smaller mesh size). On average, sockeye made up 50% of the estuary fish community in mid-May, dropping to 10% by July (Figure 5-2a). On average, sockeye and coho both made up 12% of abundance for a total of 24% of the pelagic fish community in spring and summer.

Other marine species such as sand fish, surfperches, and Pacific sand lance were mostly captured later in the season from mid-May to July. An increase in marine species in late-summer and fall is likely associated with favourable environmental and feeding conditions, spawning habitat, and recruitment of young (Armor & Herrgesell 1985; Bottom & Jones 1990). For example, sand lance abundance is seasonal and influenced by spawning and feeding requirements across life stages (adults and juveniles) (Robards et al. 2000). Sand lance captured in the Skeena River estuary in the late-summer may be seasonally present to spawn on intertidal as they are known to spawn after peak water temperatures (Robards et al. 1999). In the San Francisco Bay, higher abundance and diversity is associated with marine species that are known to spawn offshore and move into the estuary to rear (e.g. English Sole), or species, such as Pacific herring, which rear in the Bay and migrate to the Pacific Ocean after summer (Armor & Herrgesell 1985). Additionally, freshwater discharge begins to decrease after peak freshet at the end of May in the Skeena River estuary, resulting in increased salinity and temperature which can be favorable to many marine species. We hypothesize that a combination of environmental factors and recruitment of younger age classes during the summer are related to an increase in the abundance of marine species in the late-summer.

Pink salmon and surf smelt dominate nearshore communities of estuaries sampled by beach seine (Figure 5-2b and Figure 5-4b). Surf smelt was the most dominant species in mid-March at 75% of the total catch, while pink salmon became more dominant in mid-April and early-May when they comprised 85% and 75% of the total catch, respectively (Figure 5-2b). Nearshore communities in the Skeena River estuary were typically comprised of fewer salmonid species (excluding pink salmon) and a higher proportion of other species such as surf perch, prickleback, sculpin, and longfin smelt. For example, surfperches and snake prickleback made up 25% and 30% of the community in early June and July, respectively.

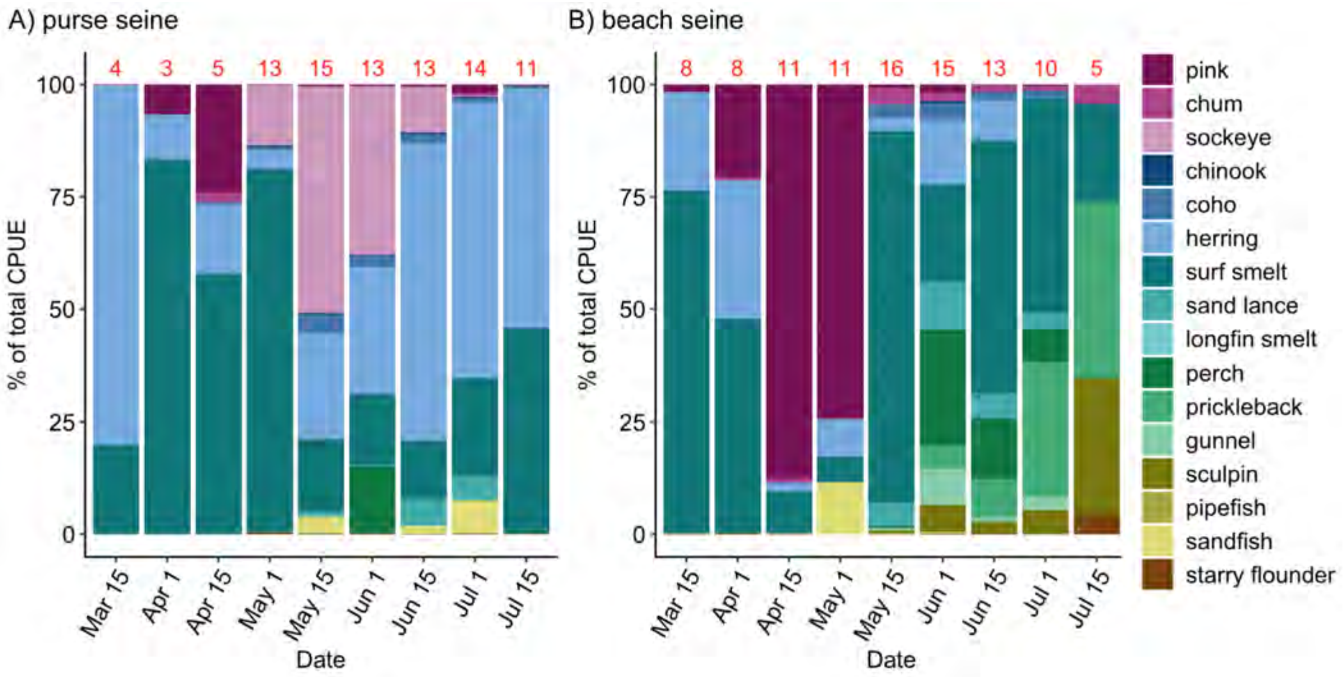

Figure 5-2. Proportion of CPUE for each species from A) purse seine and B) beach seine sampling at monitoring sites from 2014–2018. For each time period, mean CPUE for each species as a proportion of the total CPUE is presented. Total number (cumulative species richness) of each species captured is listed in red for each time period.

Diversity

Overall, the estuary community was most diverse from mid-May to June in both purse and beach seine communities (Figure 5-2 and Figure 5-3). Total cumulative species richness (number of species) per time period is included in Figure 5-2, while Figure 5-3 uses mean species richness to account for the variation between sets. The Shannon diversity index increases with species richness in the estuary. This suggests that when the number of species increases, these communities are increasing in diversity as well. The Shannon diversity index ranged from 0.41–1.49 for purse seine sets and 0.58–2.07 for beach seine sites throughout the season (Figure 5-3). These values are similar to Shannon diversity values for other estuaries in the northeast Pacific (Archipelago Marine Research 1999; Abookire et al. 2000).

Species richness and diversity increases with the size of bays and estuaries (Horn & Allen 1976; Bottom & Jones 1990; Nicolas et al. 2010). Studies of the Fraser River estuary have reported a higher species richness, with 52 species caught on Roberts Bank (Gordon & Levings 1984) compared to the 34 species found in the Skeena River estuary during spring and summer sampling. Although this can be partially explained by a higher level of species identification for groups such as sculpin, more species were captured regardless. Smaller estuaries such as the Kitimat (Bell & Kallman 1976a), Somass (Birtwell et al. 1983), or Courtenay River (Hamilton et al. 2008) estuaries have been reported to have lower species richness.

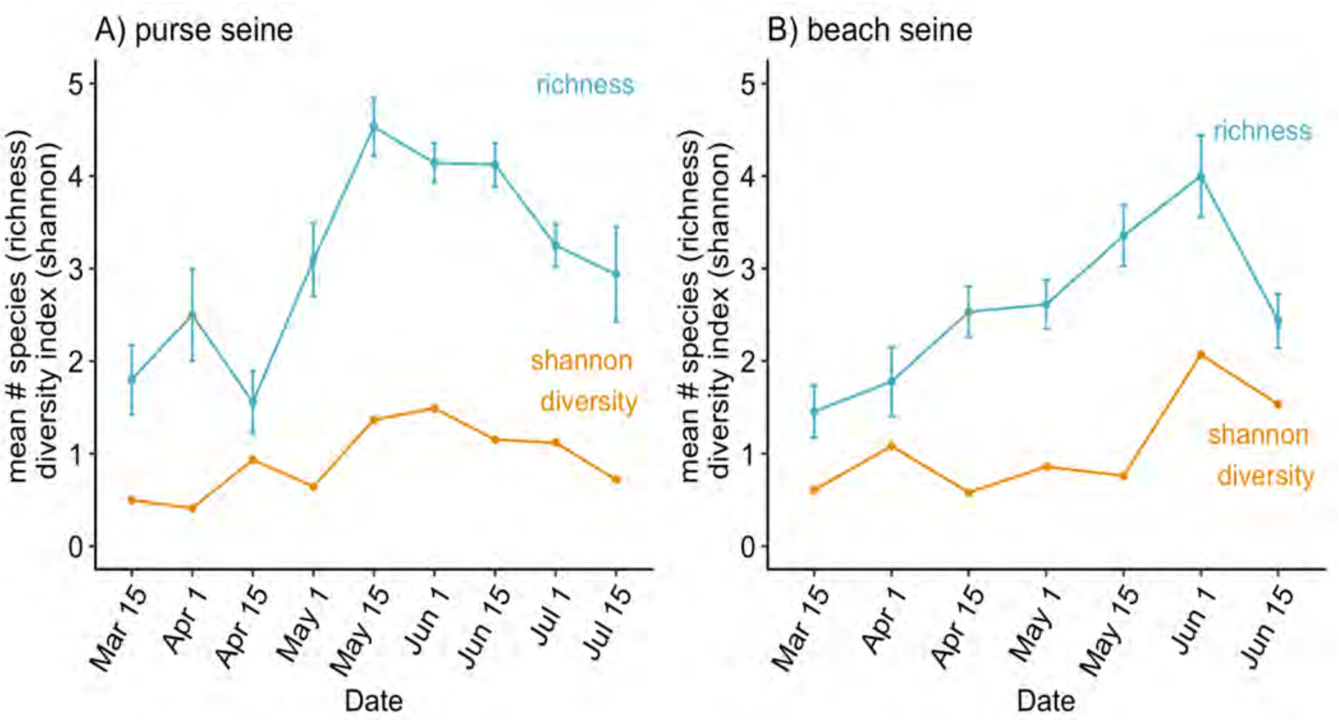

Figure 5-3. Mean species richness and Shannon diversity index of A) purse seine and B) beach seine sets over time in the estuary during 2014–2018 sampling.

Abundance

Estuaries are productive environments and abundance of fish in temperate estuaries is generally highest in summer (Gordon & Levings 1984; Raymond et al. 1985). In the Skeena River estuary, herring increased in abundance throughout the sampling season, with the highest catches in mid-June to July when sampling ended for the season (Figure 5-4 and Figure 5-5). In contrast, surf smelt were caught in high abundance across all time periods with no obvious peak (Figure 5-4 and Figure 5-5). Similar results were found for both herring and surf smelt in the Fraser River estuary, with herring demonstrating temporal variation (high catches in late summer) but no temporal patterns in surf smelt abundance (Gordon & Levings 1984). Several sets in early-May had the highest abundance of surf smelt but overall there was no evident trend across seasons. Other forage fish species, including sand lance and longfin smelt, were not caught until early spring or caught as consistently as herring and smelt (March or April) (Figure 5-4 and Figure 5-5).

Abundance of forage fish species varied across sampling years (Figure 5-6). Results from the 8 index sites in the Skeena River estuary illustrate that surf smelt catches declined from 2014–2018. Herring abundance was lowest in 2014 but had similar mean catch in 2015, 2016, and 2018. It is unclear what causes inter-annual variation but from 2013–2016 the warm water anomaly off the northeast Pacific Ocean caused widespread impacts to pelagic communities (Bond et al. 2015; Cornwall 2019). Documented impacts include lowered phytoplankton and zooplankton biomass (Gómez-Ocampo et al. 2018; Yang et al. 2018), changes to available zooplankton prey (McKinstry & Campbell 2018), and significant declines in the abundance and condition of fish and marine mammals higher up the food web, including several forage fish species in the Gulf of Alaska (Osmeridae and *Ammodytes hexapterus*)

(Daly et al. 2017; Cornwall 2019). Given the widespread effects of the warm water anomaly on marine food webs in the northeast Pacific, it is likely that pelagic species residing in this region were impacted to some degree during the time of this study. A declining trend in surf smelt since 2014 may be evidence of lower survival during the warm water anomaly.

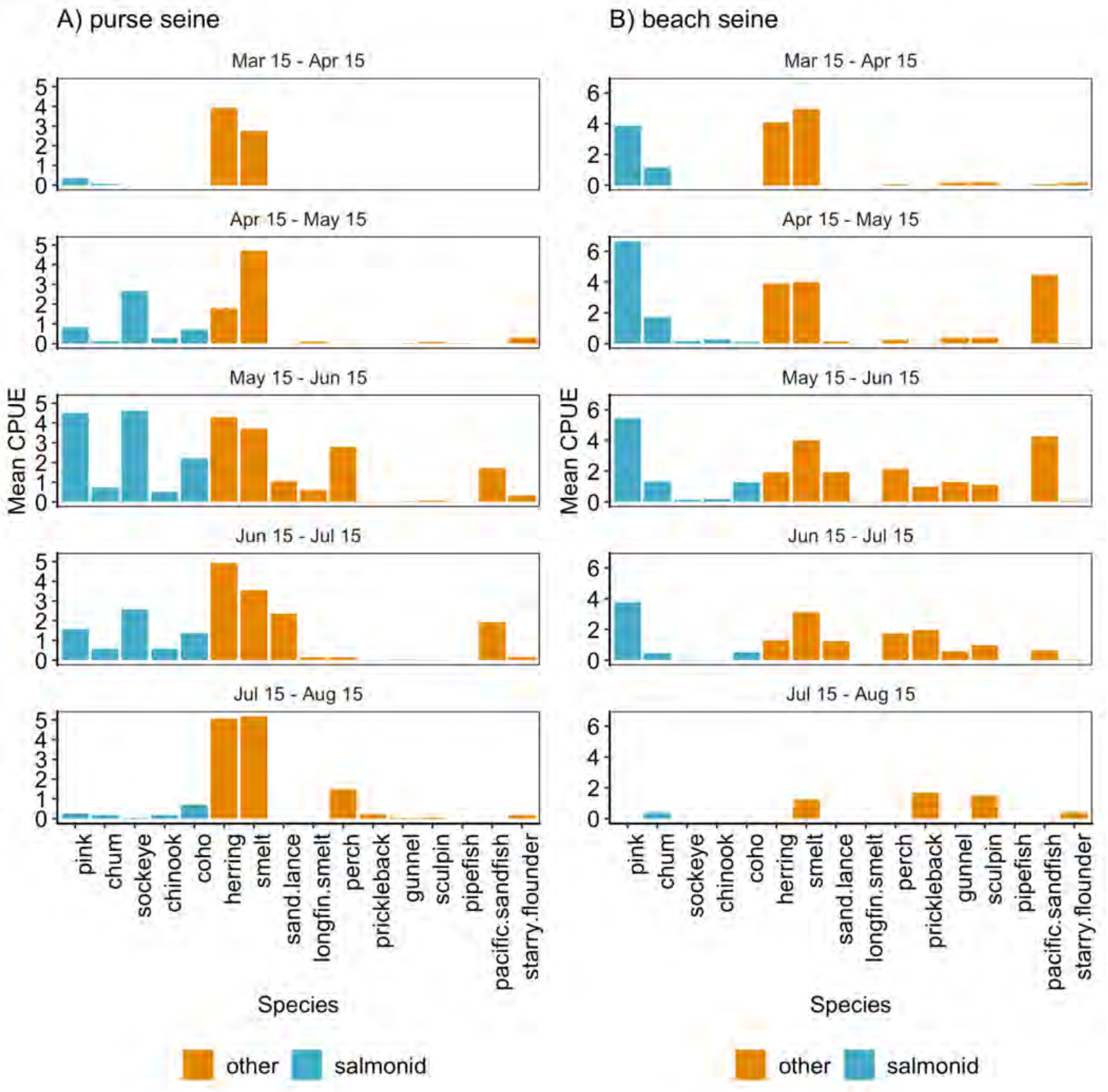

Figure 5-4. Mean CPUE for A) purse seine and B) beach seine sets grouped by seasonal time periods between March 15–August 15 in the Skeena River estuary from 2014–2018 sampling.

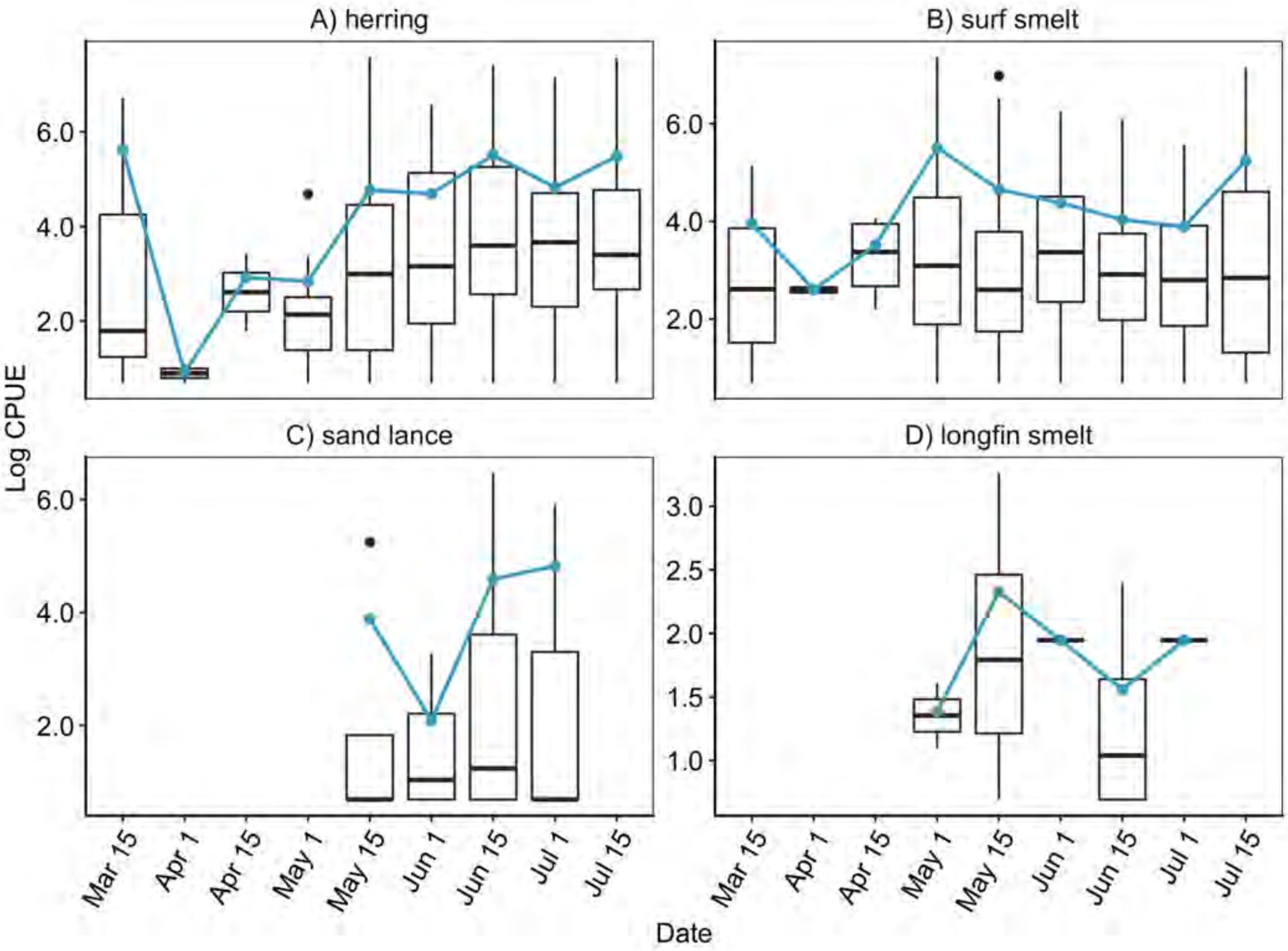

Figure 5-5. CPUE (log-scale) for dominant forage fish species A) herring, B) surf smelt, C) sand lance, and, D) longfin smelt between March 15–July 15 from regular purse seine sites (8 sites) sampled from 2014–2018. Mean CPUE (log-scale) for each time period is overlaid in blue. The solid black lines indicate median CPUE for each sampling period, while box boundaries indicate first and third quantiles and whiskers indicate the highest and lowest values of fish CPUE.

CONCLUSION

Species richness, diversity, and abundance in the Skeena River estuary was highest in summer months. This is typical of estuaries across the northeast Pacific which experience high richness, diversity, and abundance during warm summer months and declines during winter (Gordon & Levings 1984; Moyle & Baltz 1986; Bottom & Jones 1990). Fish are typically least abundant when river outflows are highest and most abundant when salinities and temperature are highest (Moyle & Baltz 1986). A combination of spawning migrations, reproductive cycles, and summer recruitment of juvenile fish cause temporal variation in composition, abundance, and distribution of species (Moyle & Baltz 1986; Bottom & Jones 1990). Over the last five years, beach seine and purse seine sampling ended in June and July, respectively. Therefore, trends in the estuary fish community in the fall and winter cannot be assessed using this data. Considering trends in other estuaries of the northeast Pacific, it is likely that fish diversity and abundance decreases in the winter. Previous sampling by Stantec (2016) identified an overall trend of decreased fish abundance in the Skeena River estuary in fall and winter, however certain survey areas observed similar fish abundance across seasons. This was largely driven by adult surf smelt and juvenile Pacific herring, which were caught in the estuary year-round. For example, surf smelt were caught in relatively high abundances from December to January in a less saline estuary region around Inverness Passage and De Horsey Island (Robertson Bank) (Figure 2-2). In general, fish capture incidence and relative abundance throughout the Stantec (2016) study were significantly lower than found with NCJSMP fish sampling, thus any trends should be interpreted with caution and followed up with additional research. Additional sampling and extending the sampling season into the late summer and fall will increase our understanding about fish communities year-round.

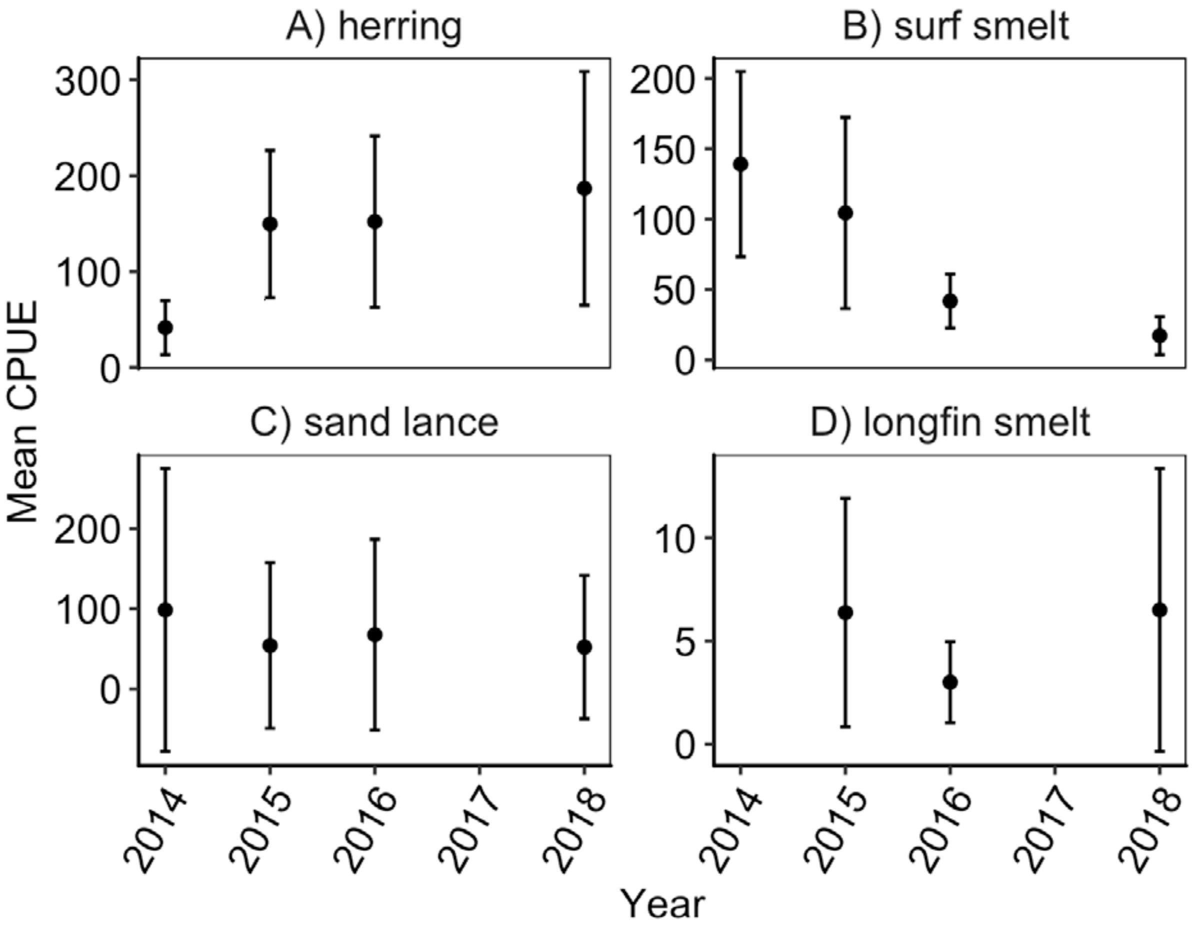

Figure 5-6. Mean CPUE with 95% confidence intervals for forage fish species A) herring, B) surf smelt, C) sand lance, and D) longfin smelt from regular purse seine sites from 2014–2018.

Section 5.2 - Q2 -Key Findings

1) Overall, the estuary community was most diverse from mid-May to June in both purse and beach seine communities.

2) Surf smelt and herring are the most dominant forage fish species in estuary communities from March–July.

3) Juvenile salmon increase in abundance starting in April and represent a majority of individuals in pelagic communities in mid-May and early June.

4) There were consistent catches of herring in the estuary across sampling years (2014–2018) while surf smelt abundances decreased in relative abundance since 2014.

5.3. Size distributions of abundant fish species: Pacific herring and surf smelt

Q3 -What are the size distributions of abundant forage fish species like herring and surf smelt over time?

Size distributions for fish in the estuary were determined by assessing fork length (Figure 5-7). After each beach and purse seine set during 2014–2018 NCJSMP sampling, fork lengths (mm) of 25 individuals of each species were measured. Table 5-2 provides a summary of fork lengths in the estuary. To assess how size distributions changed over time, we visually plotted fork length by time period (two week increments) and year (Figure 5-8). We selected 2015 and 2016 to include in Figure 5-8 because they had the highest replication of samples across the same site locations.

Individual herring and surf smelt, which are present in the study area year-round (e.g. Stantec 2015), ranged from young-of-the year fish to sexually mature adults (Figure 5-9). Sexually mature gonads were noted in larger individuals of both species in 2015 and 2016. We also observed surf smelt spawning activity in June 2016 off Kitson Island and noted herring spawn on eelgrass on Flora Bank in May and June of 2015 and 2016. As large spawning events for herring typically occur in outer regions of the Skeena River estuary in March and April (DFO 2016), it is unclear what proportion of these fish are migratory and what proportion reside in estuary habitat year-round. The presence of mature adults and separate small spawning events may be indicative of a resident population (DFO 2012).

Herring spawn in inter-tidal nearshore environments, including estuaries, in February to April (Haegele & Schweigert 1985; Lassuy & Moran 1989, DFO 2016). After spawning, herring larvae are distributed through water circulation patterns for two to three months, with survival rates being highest in nearshore environments (Stevenson 1962). Juvenile herring begin aggregating in estuarine environments at lengths of 25–40 mm (Lassuy & Moran 1989) and are found in higher abundances in these environments when compared to surrounding freshwater or nearshore marine environments (Bottom & Jones 1990; St. John et al. 1992). The warmer temperatures, refuge from adverse weather, and high food availability associated with estuarine environments provide essential nursery services for juveniles (Hourston 1959; Abookire et al. 2000). Juvenile herring may migrate offshore after their first summer or remain in the nearshore environment until maturity (2–5 years). Generally, adults migrate from offshore environments to spawn in the spring, however, many resident populations remain in coastal inlets and bays.

In the Skeena River estuary, the individual lengths of herring captured by beach and purse seine sampling in 2016 ranged from 36–270 mm (Table 5-2, Figure 5-8). During certain time periods in spring and summer, herring size distribution was bimodal, suggesting that two size classes were present (Figure 5-8a). Figure 5-10a illustrates the size distribution according to age class using fork length cut-off values from Lassuy & Moran (1989) and Hay & McCarter (1999). Most herring caught were younger than 2+ years of age (below the age of first maturity, <150 mm) (Figure 5-10a). In 2016, an increase in the abundance of smaller herring captured at the end of the sampling period on July 15 and August 15 indicates that a new brood of juvenile herring entered the area during the 2016 sampling period (Figure 5-10a and Figure 5-11a).

Herring from the 0+ age class increased in length 0.27 mm ± 0.03 mm (95% CI, p = 0.00) per day from mid-March to August (Figure 5-11a). Thus, herring from 0-2 years of age were on average 32 mm ± 3.6 mm larger than individuals of the same age class that were caught in April. Mature herring (5+ years) had a decreasing trend in size over time (0.23 mm ± 0.21 mm, 95% CI, p = 0.04, Figure 5-11a). Although this may be evidence of larger adult herring (past the age of first maturity) moving out of nearshore environments to reside in deeper, offshore environments, this trend has a larger degree of uncertainty due to variability in the data. Lastly, there was no trend in length captured overtime for the 2+ (2-5 years old) age class (Figure 5-11a).

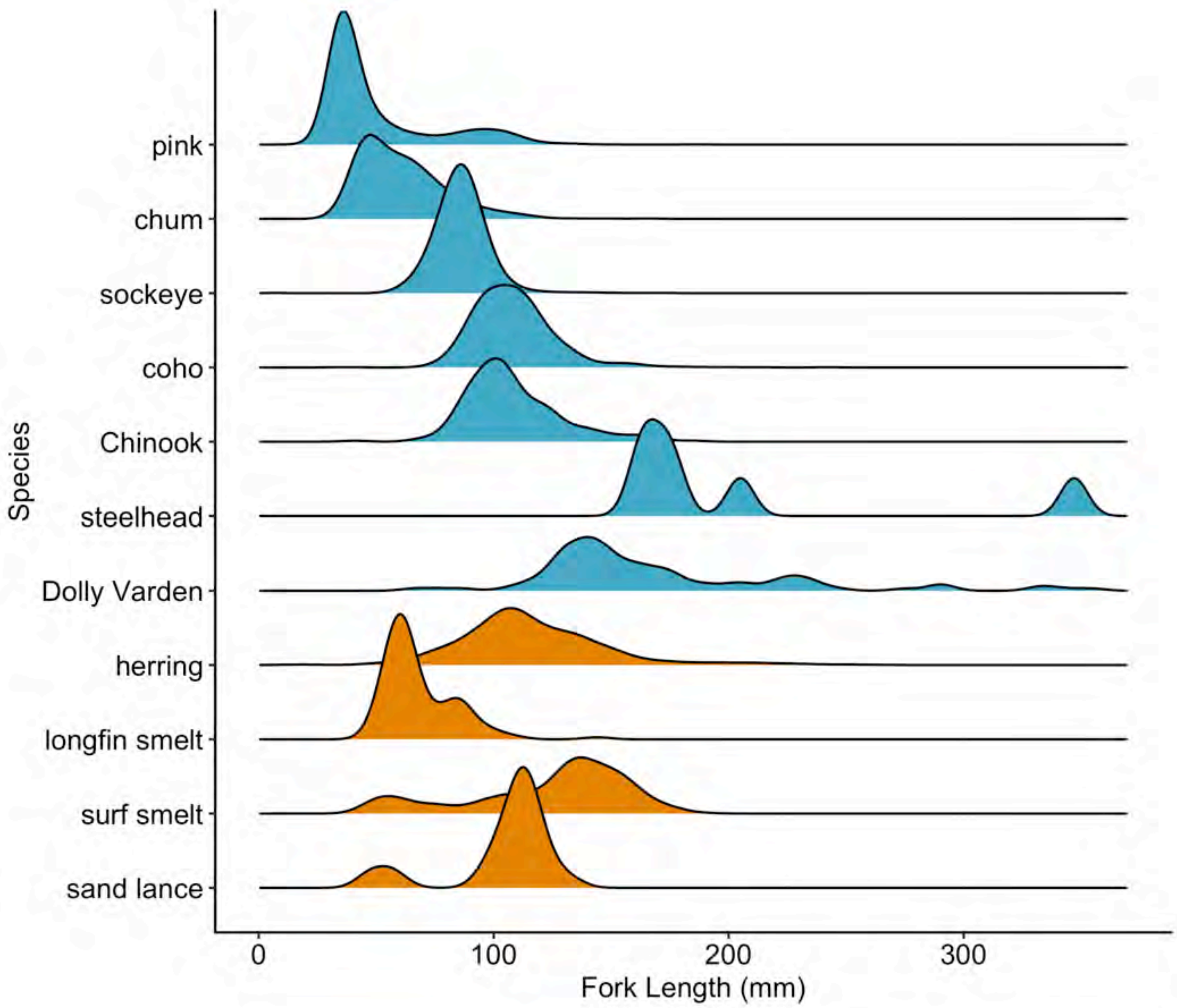

Figure 5-7. Density of fish fork lengths for the most abundant salmonid (blue) and forage (orange) fish species of all fish caught by purse seine and beach seine in the Skeena River estuary from 2014–2018.

Table 5-2. Summary of fish fork lengths (FL) caught by purse and beach seine in the Skeena River estuary from 2013–2018.

	Purse Seine		Beach Seine	
Species	FL (mm)	Mean FL (mm)	FL (mm)	Mean FL (mm)
pink salmon	31–187	74	26–75	36
chum salmon	40–165	75	32–101	51
sockeye salmon	55–174	86	62–120	83
coho salmon	34–400	109	41–160	103
Chinook salmon	69–195	108	41–112	91
steelhead salmon	162–347	204	N/A	N/A
Dolly Varden	76–520	182	66–292	151
herring	36–270	120	35–225	96
surf smelt	36–199	133	40–180	89
longfin smelt	49–144	69	65	N/A
sand lance	95–135	112	46–102	57

We have also included length data from longfin smelt and sand lance in Figure 5-7 and Figure 5-10. Longfin smelt ranged in size from 49–144 mm and were only caught in beach seines once during our five-year study period (Table 5-2, Figure 5-7). Longfin smelt size distributions appear to be bimodal with the majority of fish younger than two years of age (Figure 5-10c) (Robinson & Greenfield 2011). Longfin smelt generally spawn within their second year (Robinson & Greenfield 2011), suggesting that most of the fish caught in the estuary were juvenile fish. Sand lance in the Skeena River estuary ranged in size from 46 mm–135 mm (Table 5-2, Figure 5-7) and similar to surf smelt, herring, and longfin smelt, had a bimodal distribution. Although the sample size was small, sand lance younger than a year were caught in the beach seine in the spring, while larger fish were caught in the purse seine in May and June (Figure 5-10d). Given that smaller age classes of sand lance are smaller than the purse seine mesh, it is unclear whether they would be found later in the season as beach seine sets become less frequent. Sand lance generally spawn once a year in intertidal and possiblly subtidal habitat (burrowing into sand or gravel substrate), with populations returning to the same locations year after year at various times depending on location (Robards et al. 2000). Sand lance are associated with freshwater influence (estuaries) and are found in depths less than 50 m deep in nearshore and intertidal environments (Robards et al. 1999). Robards et al. 1999 found that age 1 sand lance begin maturing in the fall although most matured at the age of 21 months, with age 1 (50%) and age 2 (31%) fish dominating spawning schools in Kachemak Bay, Alaska. Given that the smallest maturing male and female sand lance were 88 mm and 113 mm, respectively (Robards et al. 1999), the majority of sand lance found in the Skeena Estuary are likely over a year old and at an age where spawning is possible (Figure 5-10d).

Forage fish of the NE Pacific rely on estuarine and nearshore environments during several life-history stages. Pacific herring and surf smelt support commercial, traditional, and recreational fisheries in British Columbia (Therriault & Hay 2003; DFO 2014, 2016). These species, along with longfin smelt and Pacific sand lance, generally make up a large portion of the forage fish community (Bottom & Jones 1990) and are crucial components of the broader marine food web (Robards et al. 2000; Therriault et al. 2009; Robinson & Greenfield 2011).

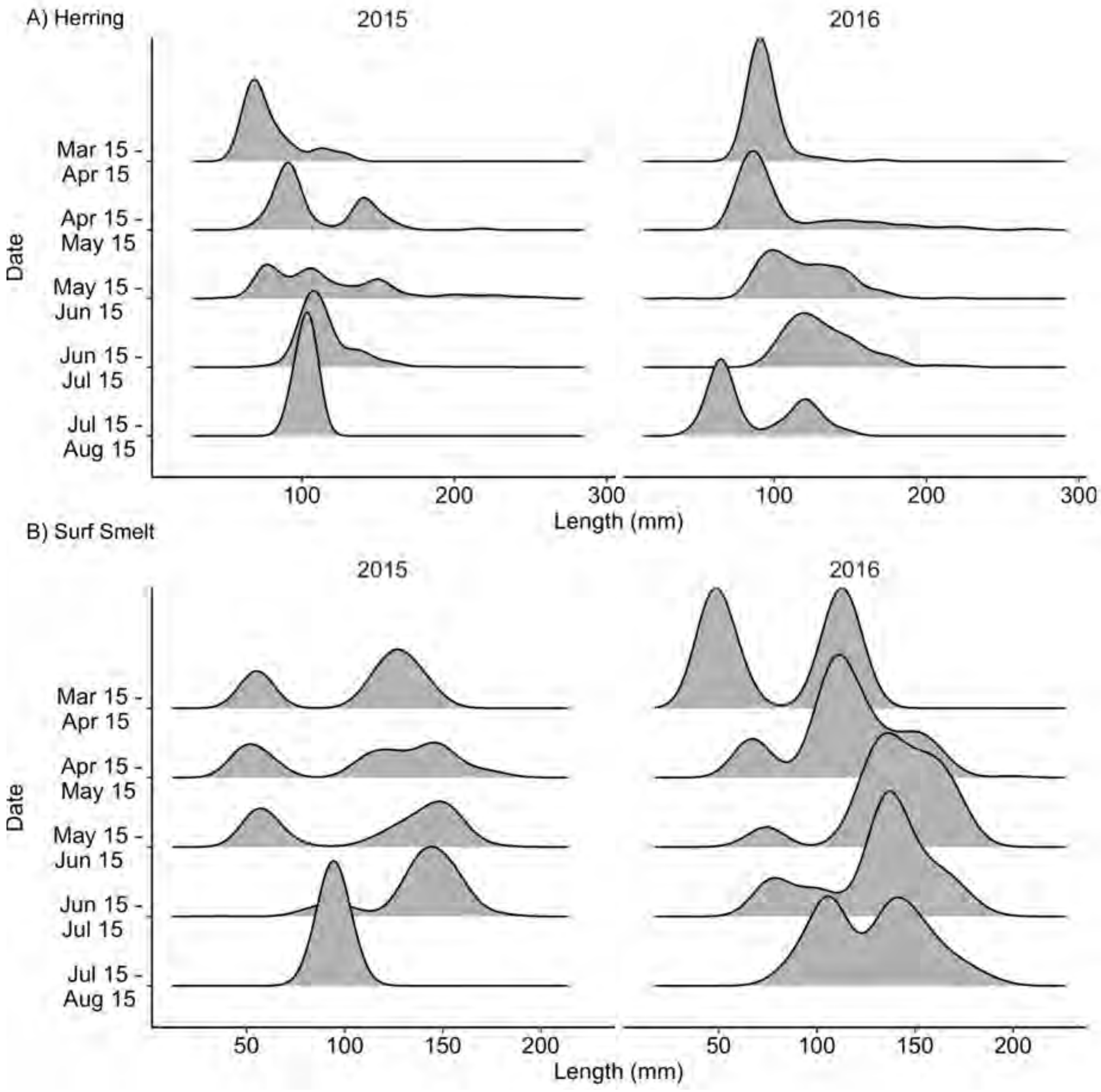

Figure 5-8. Fork length distributions of herring and surf smelt measured over two week long periods from beach and purse seine sets between March 15–August 15 of 2015 and 2016. Purse seine sampling occurred across the same 8 sites in both years and beach seine sets represent compiled data from the Flora Bank and Lelu Island region.

Figure 5-9. Photo of mature herring caught in the Flora Bank region of the Skeena River estuary (photo by Ciara Sharpe).

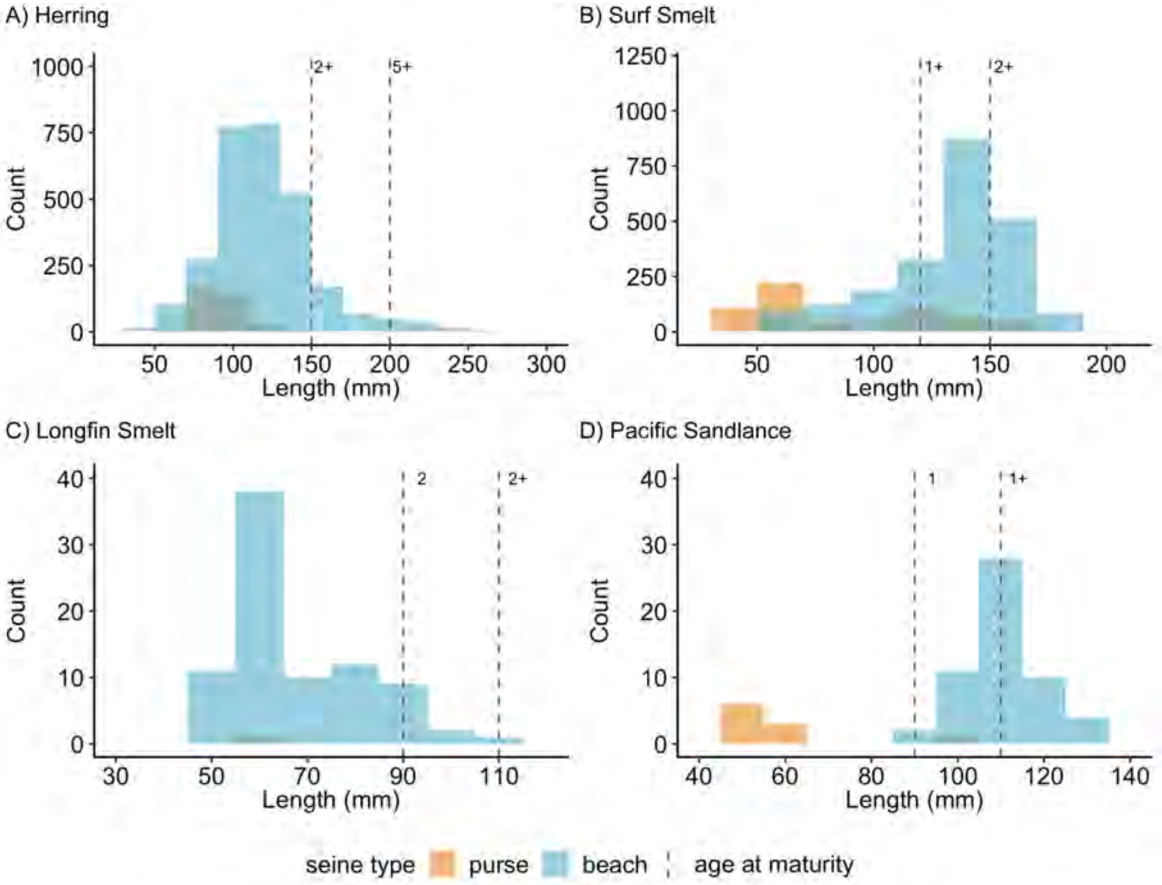

Figure 5-10. Fork length distributions of forage fish A) herring, B) surf smelt, C) longfin smelt, and D) Pacific sand lance captured in beach and purse seine sets in the Skeena River estuary between 2014–2018.

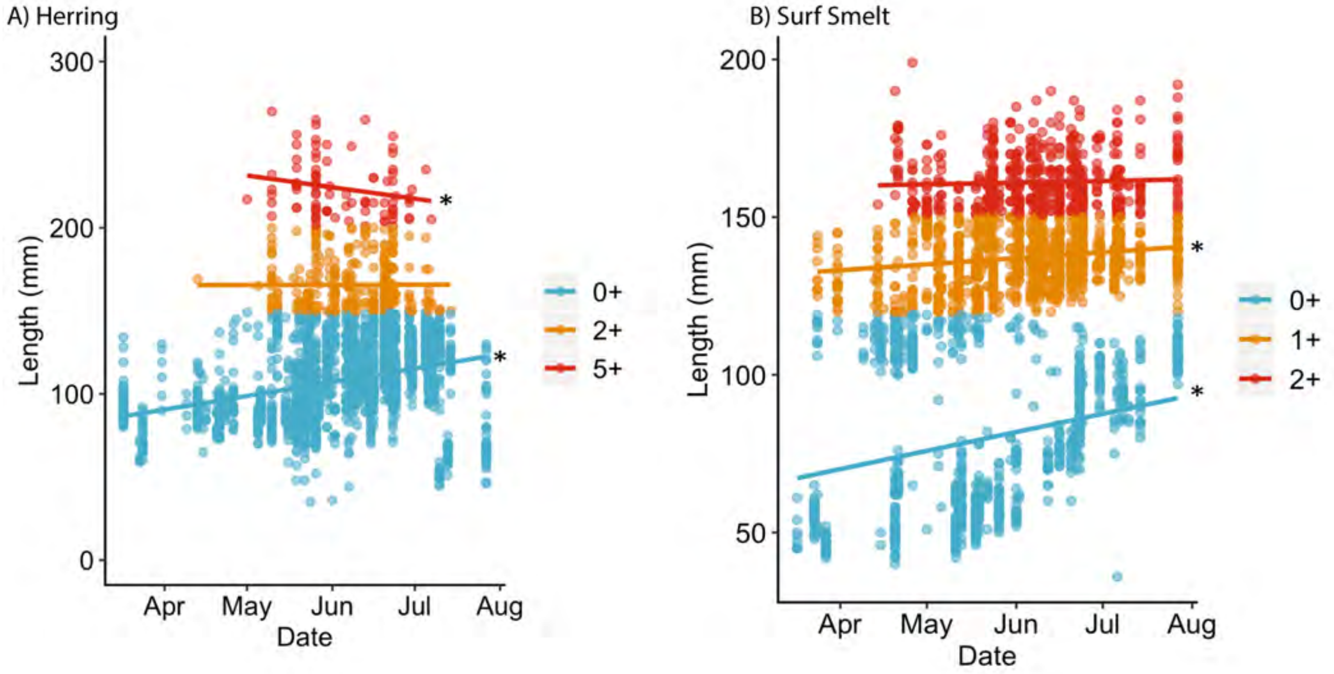

Figure 5-11. Fork length distributions of forage fish A) herring and B) surf smelt captured by beach and purse seine sets in the Skeena River estuary between 2014–2018. Fish were assigned age cohorts based on previous studies. Linear trend line is fitted for each age class separately, with * representing a statistically significant trend at a ($p < 0.05$).

Section 5.3 - Q3 - Key Findings

1) Herring and smelt increased in size throughout the sampling season.

2) All size distributions of forage fish species (herring, surf smelt, longfin smelt, and sand lance) were bimodal, suggesting that multiple age classes ranging from young of the year to adult fish use the estuary.

3) Length distributions indicate that most of the herring captured in spring and early summer were immature, while most surf smelt and sand lance were large enough to be mature fish.

5.4. Spatial patterns of abundance for Pacific herring and surf smelt

Q4 - How are dominant forage fish species like herring and surf smelt using the estuary spatially?

a) Are there spatial distribution patterns in herring and surf smelt abundance?

In this section, we look at spatial patterns of abundance for herring and surf smelt over two spatial scales: a) the entire extent of the estuary (Figure 5-12) and b) the IN region proximal to the Skeena River mouth (Figure 5-13). More detailed methods can be found in Section 3.1, where a similar analysis was performed for juvenile sockeye and coho salmon. The IN section of the estuary was selected for regular monitoring and more in-depth analysis due to the high abundance juvenile salmon identified in historical (Higgins & Schouwenburg 1973) and recent research findings by our research team (Carr-Harris et al. 2015).

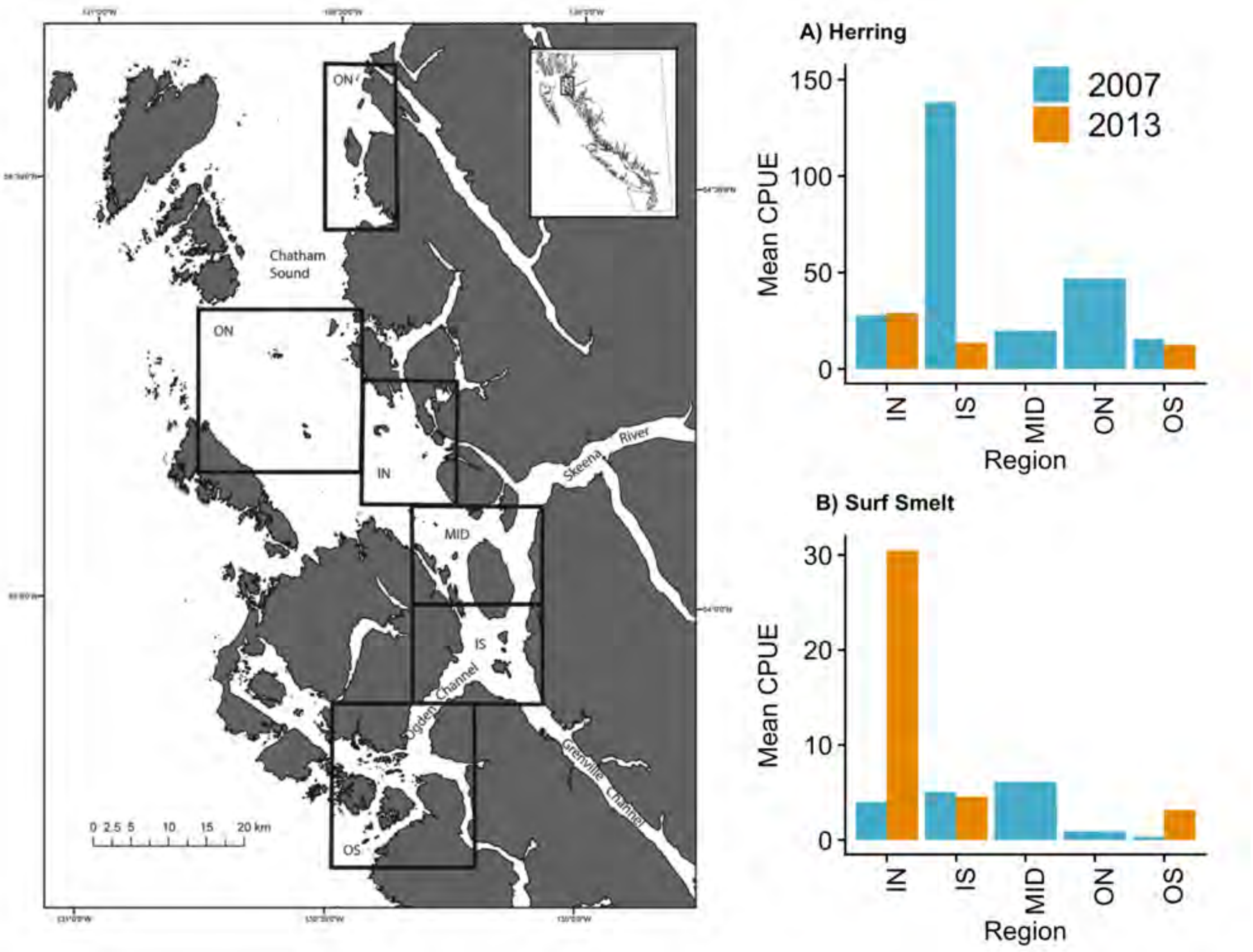

Figure 5-12. Mean trawl catch (CPUE, normalized for 20 minute trawl sets) of A) herring and B) surf smelt by region pooled across all locations and sampling dates during 2007 and 2013. Regional abbreviations in graphs correspond to regions in the map panel, amended from Carr-Harris et al. (2015) (map by John Latimer, Lax Kw'alaams Fisheries).

Herring were relatively evenly distributed across all regions in the estuary, with the largest catches occurring in the IS region in 2007, but not in 2013, demonstrating yearly variation in catch data (Figure 5-12a). High catches were also found in the ON region, which is farthest from the Skeena River mouth. Surf smelt were also caught in high abundances in many of the regions, with lower abundances found within the ON and OS regions (Figure 5-12b). These are the regions farthest from the river mouth and as a result, are more saline.

Next, we mapped abundance patterns on a smaller spatial scale using purse seine catch data from 2015 and 2016 in the IN region proximal to the mouth of the Skeena River, from Inverness Channel north into Prince Rupert Harbour (Figure 5-13). High densities of herring and surf smelt were caught during purse seine sampling in 2015 and 2016, and were caught in all sampling locations during all months of regular sampling. Figure 5-13 illustrates the spatial abundance patterns of herring and smelt from May until mid-July in this section of the estuary. Although we found herring and smelt ubiquitously throughout this region, we found that certain sites consistently supported high abundances. Sites further north into Prince Rupert Harbour had high abundances of both forage fish species during 2015 sampling, while Kitson Island sites had high abundances of herring and smelt in 2015 and 2016 (Figure 5-13). For example, we caught four times more herring on average around Kitson Island compared to all other sites combined in 2016. This aligns with results from a historical study by Higgins and Schouwenberg (1973) that found high abundances of herring in proximity to Kitson Island and Ridley Island. Ridley Island is currently the site of large shipping terminals for coal, grain, and propane (Ridley Terminals Inc., Prince Rupert Grain Terminal Ltd. and Ridley Island Propane Export Terminal).

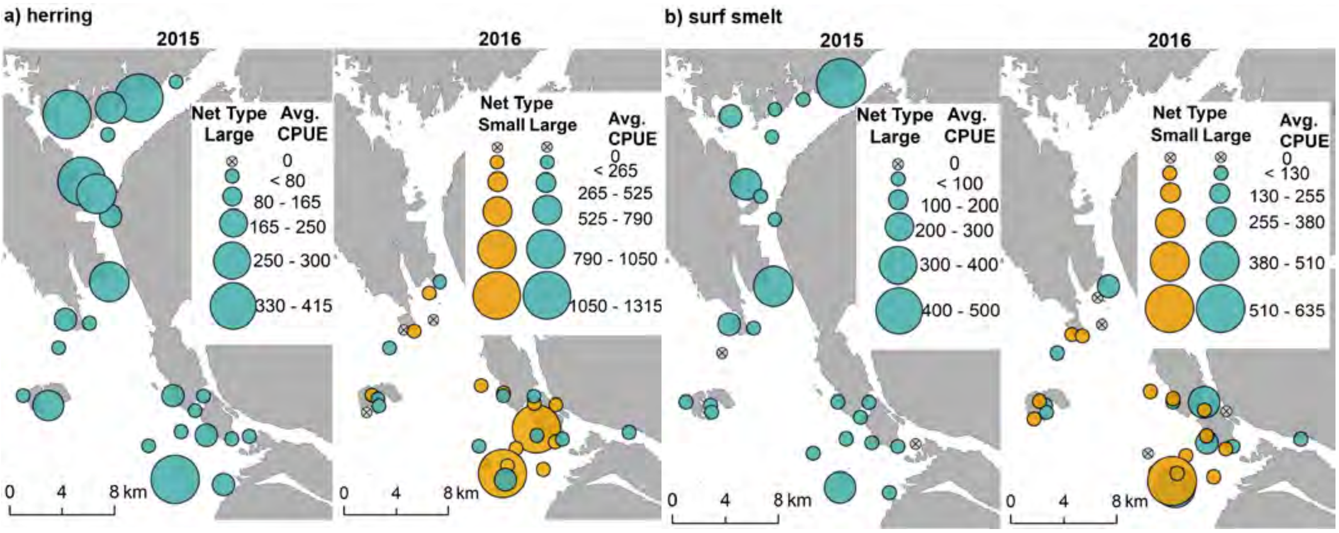

Figure 5-13. Mean fish abundance (CPUE) of A) herring and B) surf smelt at locations sampled during peak migration (May–mid July) in the Skeena River estuary in 2015 and 2016. Colours indicate the different net types used for sampling (orange = small purse seine, blue = large purse seine).

b) How are forage fish oriented along environmental gradients in the estuary?

Different abiotic and biotic factors have been strongly associated with structuring abundance and distribution of estuary fish communities (Marshall & Elliott 1998; Wagner & Austin 1999; Harrison & Whitfield 2006). These variables, include water quality (salinity, turbidity, and temperature), spatial habitat distribution, temporal factors, vegetative habits, tidal influences, and food web dynamics (Bacheler et al. 2009b). Investigating the relationships between fish abundance and various abiotic and biotic factors can advance our understanding of fish distribution patterns in the estuary. Similar to Section 3.1 on juvenile salmon species, we investigated abundance patterns of forage fish species (herring and surf smelt) within the Skeena River estuary across various habitat types, such as eelgrass and rocky shores, and biophysical factors such as salinity, turbidity, and spatial habitat distribution. This question was the topic of Sharpe et al. (2019), which investigated these fish-habitat associations for herring and surf smelt along with three salmon species (sockeye, coho, and Chinook salmon). We summarize some of the methods and the results in this section. For more details about sampling or statistical methods see Section 3.1 and Sharpe et al. (2019).

METHODS

We sampled fish in the Skeena River estuary during the peak salmon smolt migration from April to mid-July in 2015 and 2016 with large and small purse seine nets (Figure 3-11). Fish sampling occurred at four of the most abundant habitat types available in the Skeena River estuary: sandy banks, rocky shores, open water, and eelgrass beds. Subsequent to each sampling event, we collected data on various habitat and environmental variables known to influence estuarine fish habitat utilization including water quality, tidal variables, vegetative, and spatial attributes (Bacheler et al. 2009).

We used multi-modal inference to determine which combination of abiotic and biotic variables was most important in explaining fish abundance across the estuary, and used negative binomial generalized linear mixed effect models (GLMM) to investigate these fish-habitat relationships (R package glmmTMB, Magnusson et al. 2016). The importance of each predictor variable was quantified based on cumulative Akaike weight of the candidate model set (ΔAIC_c less than two), creating a measure of Relative Variable Importance (RVI) (Burnham & Anderson 2002). Additional details on statistical analysis can be found in Section 3.1 and Sharpe et al. (2019).

Results are presented in a coefficient plot below (Figure 5-14) with dots that represent parameter estimates with 95% confidence intervals from averaged models. The larger the confidence intervals, the higher the standard error associated with the coefficient. In addition, if the confidence intervals cross 0 (dotted line), the estimated value has large uncertainty which likely indicates that there is no strong effect. RVI values are listed beside each estimate and are an indication of how important each variable is to explaining fish abundance in relation to the other variables. For example, an RVI value of 1 indicates that this variable was in all models (100%) with higher statistical support. The variables in Figure 5-14 are ranked according to RVI values.

RESULTS

Herring abundance was positively correlated to environmental conditions with higher temperatures and lower water visibility (higher turbidity), while smelt were associated with increased salinity (Figure 5-14). These variables had RVI values of 1.00, which suggests that they were relatively important in explaining abundance of these species compared to the other variables. In addition, both forage fish species were associated with sites closer to shore.

Warmer temperature and low visibility are hypothesized to contribute to the nursery function of estuaries. In general, turbidity from river plumes are thought to provide increased cover from predation (Blaber & Blaber 1980; Simenstad et al. 1982; Gregory & Levings 1998), but may also enhance visual contrast of prey resulting in higher feeding success for juvenile herring (Boehlert & Morgan 1985). Increased biomass of forage fish species, including herring, has been positively associated with warmer temperatures in other estuaries (Marshall & Elliott 1998; Abookire et al. 2000) and nearshore environments (Cross et al. 1980; Reum et al. 2013). Fish have been observed distributing towards preferred temperatures in fresh and marine water environments to increase metabolism and support increased growth rates (Garside & Tait 1958; Javaid & Anderson 1967; Straty & Jaenicke 1980; Armstrong et al. 2013). The optimal temperature for juvenile herring was reported as 12.2°C in one study (Haist & Stocker 1985), which is on the warm end of the range of temperatures we observed in the Skeena River estuary (9.2–12.6°C). Interestingly, temperature and turbidity did not generally co-vary with each other; except during spring freshet when colder river waters create a temperature gradient across the estuary. Collectively these findings suggest that temperature and turbidity are two major environmental gradients that structure herring distribution patterns across the estuary.

We found higher abundances of both herring and smelt closer to shore in the Skeena River estuary. Previous work has documented that juvenile herring concentrate in shallow, sheltered regions and hypothesized that nearshore environments were likely driving this trend (Hourston 1959). Nearshore environments provide shelter from adverse wind and wave activity, and shoreline complexity creates refuges from strong currents, such as back eddies (Hourston 1959). Additionally, nearshore environments can have higher habitat complexity, including intertidal vegetation, which provides more shelter from predation and may provide higher concentrations of zooplankton trapped along the shoreline (Heck et al. 2003; Alofs & Polivka 2004; David et al. 2016). Our results support the importance of nearshore habitat for both herring and smelt.

Eelgrass is commonly cited as being an important habitat for estuary fish as it contributes high prey diversity and has been shown to reduce predation rate (Gregory & Levings 1996; Heck et al. 2003; Semmens 2008). This was not the case for herring and smelt populations in the Skeena River estuary, the abundance of which were not associated with vegetative habitats, such as eelgrass or macroalgae. In addition, analysis of fish diets in the Skeena River estuary did not find that herring and smelt were eating high abundances of harpacticoid copepods, a species of zooplankton associated with eelgrass habitat (Arbeider et al. 2019). Herring and surf smelt consumed primarily calanoid copepods, which were the most abundant prey in the estuary. Calanoid copepod abundance was positively correlated with salinity, which may explain the higher abundances of smelt in locations with higher salinities (Figure 5-14). Higher abundances of surf smelt have been found at lower salinities on a similar geographic scale in the Skagit River estuary (Reum et al. 2011) and in the lower saline conditions of the estuarine mixing zone of the Columbia River estuary (Bottom & Jones 1990). Given that smelt have been shown to prefer regions of lower salinity, it is possible that they are orientating towards higher salinity in the Skeena River estuary due to the increased abundance of calanoid copepods, their main prey.

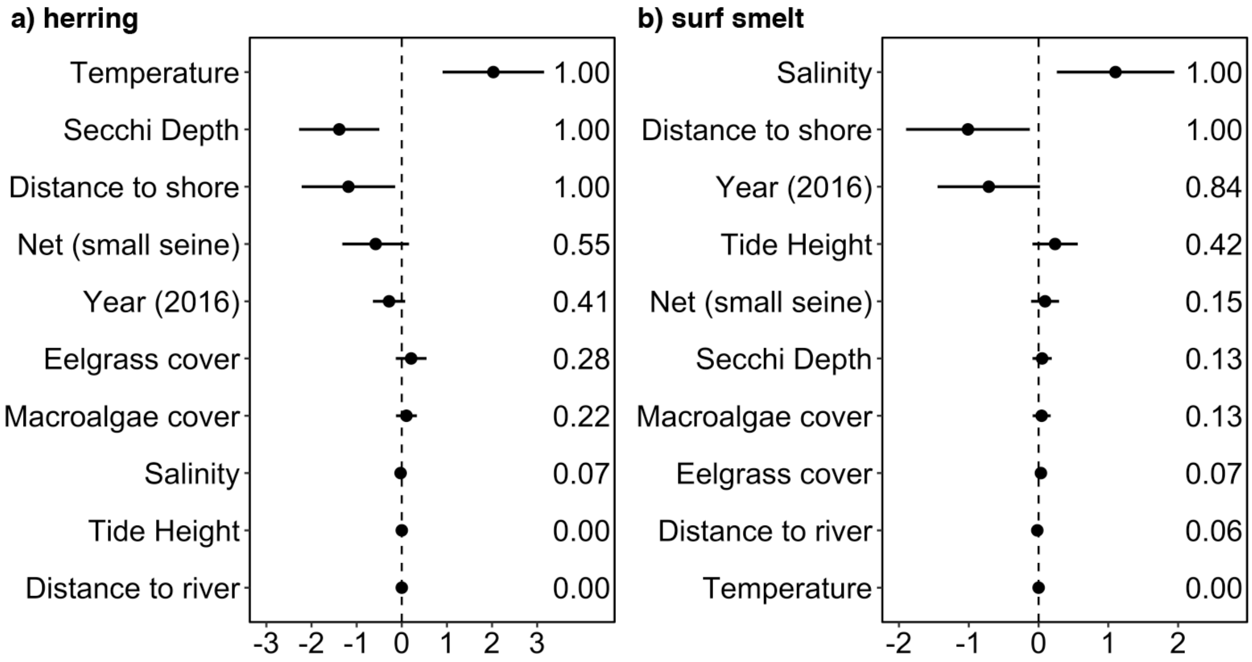

Figure 5-14. Standardized model-averaged coefficients (points) and 95% confidence intervals (bars) used to describe abundance of (a) herring and b) surf smelt with variables presented in decreased order of relative variable importance (RVI). RVI values shown on rightmost of each panel. Coefficients are related to the (log) mean of normalized CPUE. Parameter year (2016) compares catches to the sampling year 2015 as a baseline and the small purse seine net is being compared to the large purse seine net.

Section 5.4 - Q4 - Key Findings

1) Herring and smelt were found in high abundances in most regions and sites in the estuary.

2) Herring and smelt were more abundant closer to shore.

3) Herring abundance was associated with increased temperature and turbidity, while surf smelt were found at sites with higher salinities.

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5.5. *Larval fish survey for eulachon*

This section contains combined and abridged information from the following reports:

Carr-Harris, C. (2017). Early life history of eulachon (Thaleicthys pacificus) in the Skeena River Estuary. Prepared for the Lax Kw'alaams Band and Skeena Area Marine Research Collaboration, Skeena Fisheries Commission, Kispiox BC.

Butts, K., Sharpe, C., & Carr-Harris, C. (2019). Early life history stages of eulachon (Thaleicthys pacificus) in the Skeena River Estuary, part II. Amended to include 2017 sampling results. Prepared for the Vancouver Aquarium Marine Research, Lax Kw'alaams Fisheries, Prince Rupert, BC.

INTRODUCTION

Eulachon are an ecologically and economically important anadromous fish species that spawns in large river systems throughout the Northeast Pacific. Eulachon, which support traditional Indigenous fisheries and are highly prized for their high oil content, were historically harvested and traded by First Nations throughout their range. Although eulachon fisheries are still carried out by some First Nations groups, eulachon returns to systems in British Columbia have declined drastically over the past three decades (Hay & McCarter 2000), and most British Columbia eulachon populations are currently designated as threatened or endangered (COSEWIC 2011). Skeena River eulachon, which are listed as threatened, support modest First Nations fisheries in years of higher abundance (COSEWIC 2011, 2013).

Although variable marine survival has been suggested as a limiting factor for eulachon productivity, the movements and distribution of eulachon during the marine life history stages are not well understood. Eulachon spend more than 95% of their life history in saltwater and typically return to freshwater to spawn at 3 or 4 years of age (Willson et al. 2006). Eggs are deposited on the substrate, and following incubation, newly hatched larvae drift downstream into tidewaters, becoming benthic during the first few months at sea (Willson et al. 2006). There are many unknowns surrounding use of estuary habitat during the larval and juvenile stages, including how long they remain in the estuary. Hay and Schweigert (2015) suggest that larvae and small juveniles remain in the estuary to benefit from increased access to food and decreased risk of predation in the productive and turbid estuary environment.

Skeena eulachon typically spawn during February or March in the lower reaches of the Skeena River. There are currently no direct estimates of Skeena eulachon run sizes (COSEWIC 2011), however, the relative abundance of a given brood year is inferred using indices such as catch data and predator densities. Although little is known about their marine distribution, juvenile eulachon are captured regularly during autumn shrimp trawl sampling in Chatham Sound, which overlaps with the Skeena estuary (MacConnachie et al. 2016). Eulachon length frequencies collected during the shrimp trawl sampling program in November 2015 indicated that at least three age classes were present, and genetic sampling determined that these juvenile eulachon originated from populations in the Skeena and Nass Rivers (which are not currently genetically distinguishable), and the Central Coast (MacConnachie et al. 2016). While it is probable that a portion of Skeena eulachon rear in Chatham Sound, the timing of their transition from estuarine pelagic to marine benthic stages is unknown.

In-river egg and larval drift surveys, which have been developed to estimate the spawning stock biomass for eulachon in other river systems such as the Fraser (McCarter & Hay 2003), are currently being developed for the lower Skeena River (Kitsumkalum Fisheries Manager, pers. comm.). Larval fish surveys may also be used to establish the presence and quantify densities of larval eulachon in coastal and estuarine waters (Hay & McCarter 2000). Larval fish data collected over multiple years has the potential to provide more information about the timing of spawning and abundance for many different fish species, including eulachon. In 2016 and 2017, Lax Kw'alaams Fisheries and Skeena Fisheries Commission conducted larval eulachon surveys using zooplankton sampling methodology to characterize the timing and distribution of larval eulachon in the Skeena River estuary during their downstream migration. Here we summarize results of the 2016 and 2017 larval eulachon estuary sampling program.

METHODS

Larval Fish Sampling

Larval fish samples were collected from 16 sampling locations in 2016 and 10 locations in 2017, all of which were distributed through the northern, middle, and southern exits of the Skeena River (Figure 5-15). Specifically, stations were located between Clara Point, east of De Horsey Island at the mouth of the river, and Genn and Kinahan Islands, the most distal sites surveyed, approximately 10 and 15 km northeast and east of Clara Point, respectively.

Larval fish samples were collected during five sampling periods (approximately 10 days apart) between April 2 and May 13 in 2016, and six sampling periods between April 14 and June 9 in 2017. Samples were collected using double-oblique tows with paired 350-micron mesh zooplankton nets mounted on a bongo frame and deployed from a research vessel. During each sampling event, the bongo net, weighted with 20–60 lbs of lead, was lowered to the bottom of the water column and retrieved immediately using a hydraulic winch with the vessel underway. The water column depth was estimated using the vessel sounder, and target depth calculated using the line length and angle during each tow. The vessel speed varied between 1–3 knots, depending on the prevailing current at a given sampling location to maintain an ideal tow angle of approximately 45°. The actual tow angle was estimated during each sampling event using a handheld angle meter. The maximum depth of each tow was recorded using a Sensus Reefnet depth logger, which was attached to the bongo net frame and downloaded at the end of each cruise. Water flow through the bongo net was metered using a TSK flowmeter. Equipment malfunction resulted in a change of sampling methods for the last sampling events on May 28 and June 9, 2017. We used a 250 µm WP2 plankton net towed by hand vertically at sample locations and recorded maximum depth for vertical tows. At the end of each tow (both oblique and vertical types), the zooplankton nets were rinsed and the contents preserved in 5% formaldehyde buffered with seawater (Figure 5-16). Temperature and salinity profiles were collected at 2m intervals to a maximum depth of 21m using a YSI meter before each sampling event.

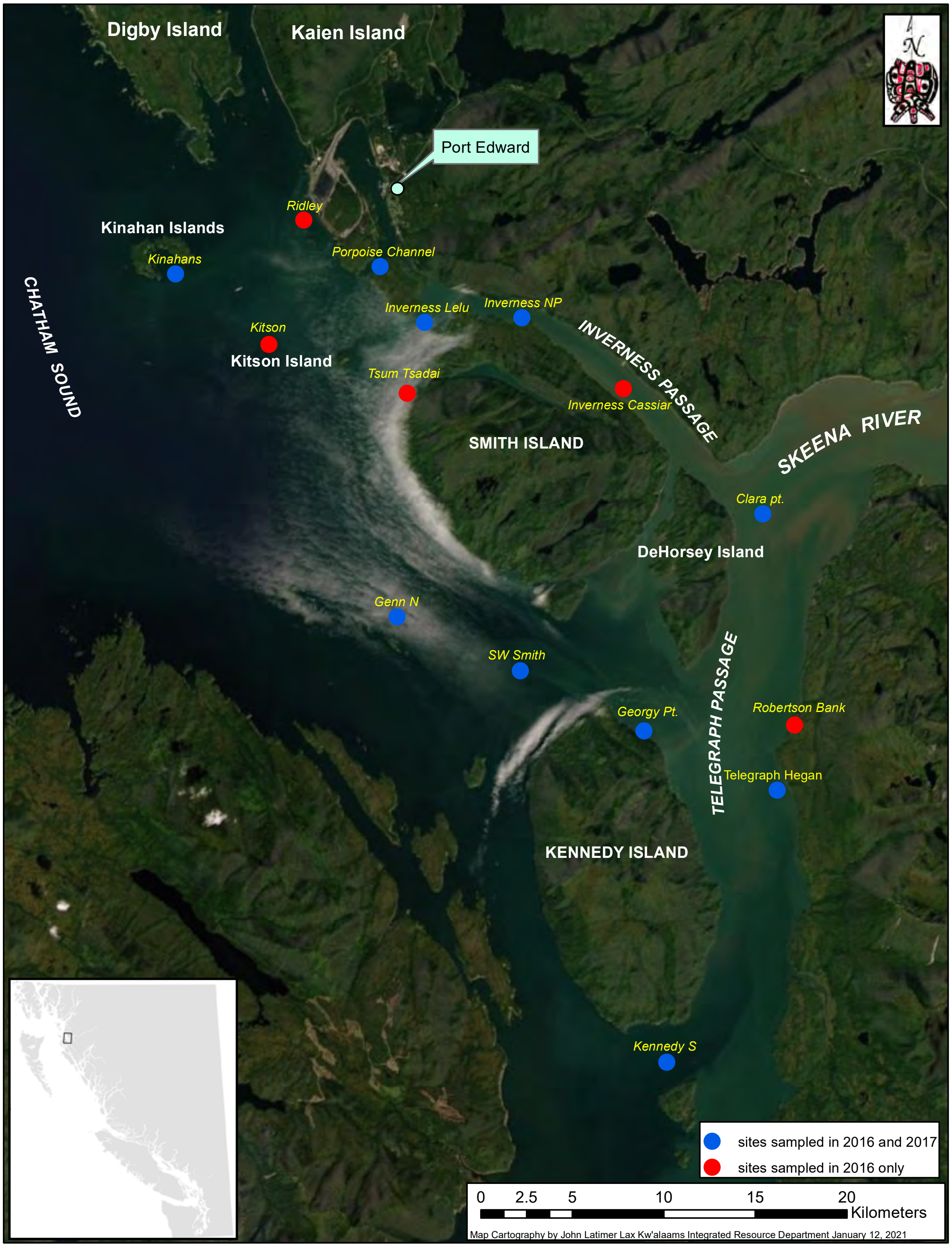

Figure 5-15. Map of larval fish stations in the Skeena River estuary sampled in 2016 and 2017. Blue sites were sampled in both 2016 and 2017, while red sites were sampled in 2016 only (map by John Latimer, Lax Kw'alaams Fisheries).

Laboratory analyses

Larval fish were enumerated at Lax Kw'alaams Fisheries field laboratory in Prince Rupert in 2016. Samples were rinsed through a 150-micron sieve and divided into fractions using a Folsom plankton splitter. Up to 200–400 larval fish per sample were enumerated using a Leica M80 dissecting microscope, and stored in vials for species identification. The total abundance of larval fish in each sample was estimated by multiplying the inverse of the total fraction sampled by the number of larval fish enumerated. The total density of larval fish was calculated by dividing the total abundance in each sample by the volume of water filtered through the plankton net, which was estimated by multiplying the length of line deployed by the area of the net mouth. Subsamples of larval fish samples were submitted to Biologica Environmental Services Ltd. for species identification.

Figure 5-16. Lax Kw'alaams field technician rinsing bongo net used to sample larval eulachon between sets.

RESULTS

Fish larvae were captured at all surveyed locations and during all sampling events throughout the duration of the study from April 2–May 13, 2016 and April 14–June 9, 2017 (Figure 5-17). Larval fish densities for all collections during 2016 and 2017 are provided in Table 5-3. Overall, higher densities of larval fish were caught in 2016, with a mean density of 49.4 fish/m^3 (range 4.0 fish/m^3–1963.8 fish/m^3), compared to 2017 which had a mean density of 14.8 fish/m^3 (range 0.32 fish/m3–330.4 fish/m^3) (Figure 5-17). Peak abundance of larval eulachon was observed at all sampling stations on April 14 in 2016 and May 17 in 2017 (Figure 5-17 and Figure 5-18).

Larval fish captured in 2016 and 2017 were identified to species and represented 14 families (Figure 5-18) and 31 species (Table 5-4). Species present in higher abundances included smelts (Osmeridae), sculpins (Cottidae), cods (gadoids), rockfish (Sebastidae) and flatfish (Pleuronectidae). Three families (Bathylagidae, Gobiidae and Myctophidae) were caught in low abundances in 2016 but not in 2017. Most of the fish identified were eulachon, accounting for 91% and 88% of identified larval fish in 2016 and 2017, respectively (Figure 5-18). Additionally, eulachon accounted for 99.9% (2016) and 98.7% (2017) of larval fish from the family Osermeridae (Figure 5-18). Eulachon larvae were present in most samples collected during both years and ranged in size from 4.0–12.0 mm, with the majority of eulachon measuring between 5.0–7.0 mm. Eulachon densities ranged from 0.12 fish/m^3–1962 fish/m^3 throughout the 2016 sampling season, while densities in 2017 ranged from 0.14 fish/m^3–329 fish/m^3 (Figure 5-18).

Spatial distribution patterns of eulachon abundance were similar across years, with the highest densities of eulachon recorded at sampling stations closest to the mouth of the Skeena River estuary (Figure 5-19 and Figure 5-20). During periods of high abundance in 2016, eulachon densities were highest at Georgy Point, off the northern tip of Kennedy Island, and Clara Point off the eastern side of De Horsey Island (Figure 5-19). In 2017, the highest densities were recorded again at Clara Point and within Inverness Passage proximal to the North Pacific Cannery (Inverness NP) (Figure 5-20).

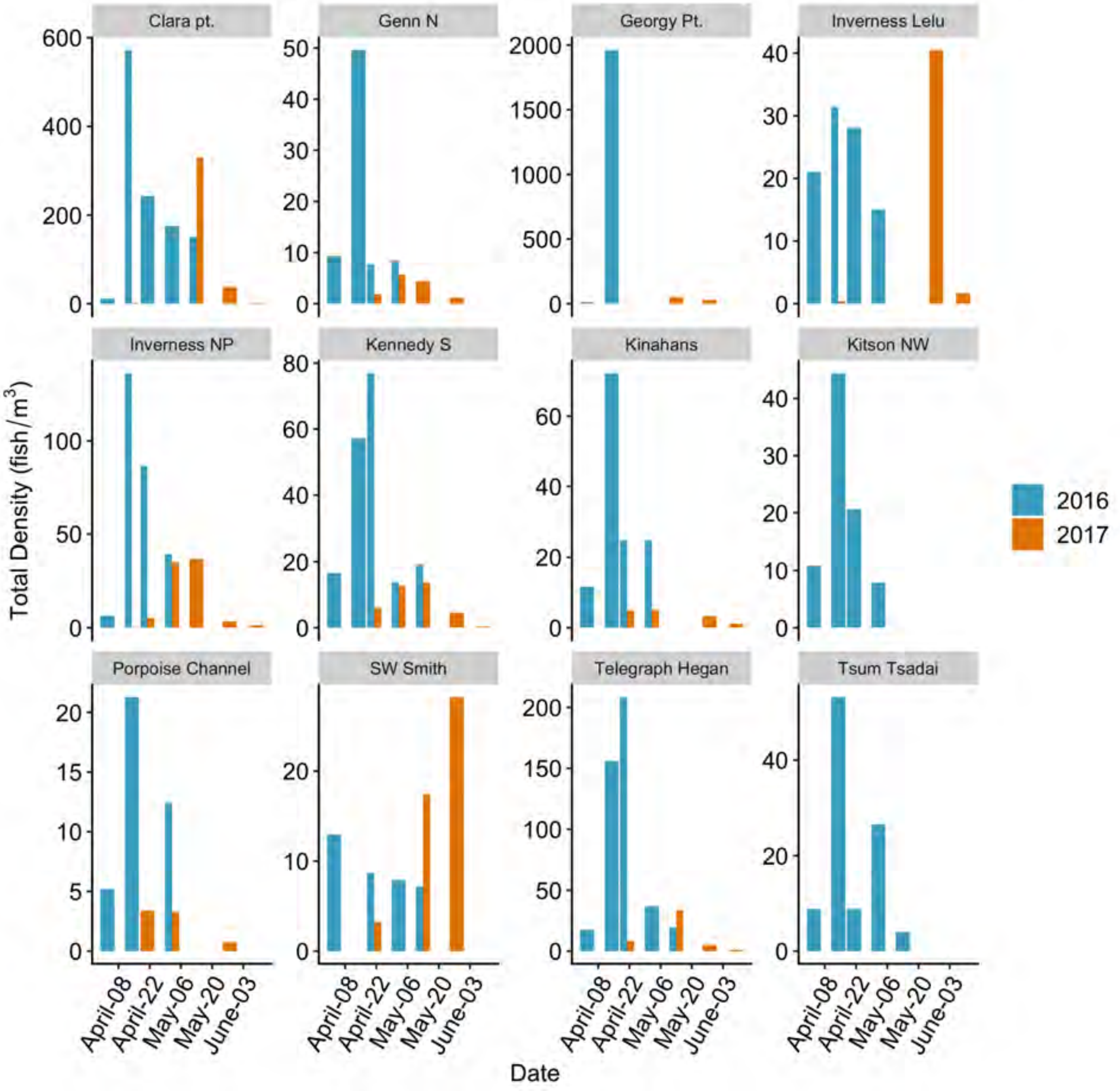

Figure 5-17. Densities of larval fish at all sampling stations in the Skeena River estuary by time period (April 8–June 3) from 2016 and 2017. Note different scales on y axes of different locations (figure from Butts et al. 2019)

Table 5-3. Larval fish densities (fish/m^3) present in zooplankton samples captured at various locations in the Skeena River estuary in 2016 and 2017 (table from Butts et al. 2019).

Date	Clara Pt.	Genn N	Georgy Pt.	Inverness Lelu	Inverness NP	Kennedy S	Kinahans	Kitson NW	Porpoise Channel	SW Smith	Telegraph Hegan	Tsum Tsadai
2016-04-02	11.9		10.2				11.6	10.9		12.9	17.3	8.8
2016-04-03				21.1	6.6	16.5			5.2			
2016-04-04		9.3										
2016-04-14	572.5	49.6	1962.8	31.4	136.3	57.2	72.2	44.4	21.3		155.9	53.2
2016-04-21	243.0	7.7		28.1	86.6	76.9	24.7	20.6		8.7	208.3	8.8
2016-05-02	175.3	8.6		15.1		13.9	24.7	7.9	12.4	7.9	36.8	26.6
2016-05-13	150.5				39.6	19.2				7.2	19.5	4.1
2017-04-14	2.8			0.4	0.4							
2017-04-27		1.9	4.2		5.2	6.1	4.9		3.4	3.3	8.4	
2017-05-03	330.4	5.6			35.2	12.7	5.0		3.3	17.4	33.5	
2017-05-17		4.3	52.8		37.0	13.7				5.5		
2017-05-28	37.9	1.2	28.3	1.4	3.5	4.5	3.3		0.7	28.2	5.1	
2017-05-31				40.5								
2017-06-09	2.5			1.7	1.3	0.3	1.1				1.1	

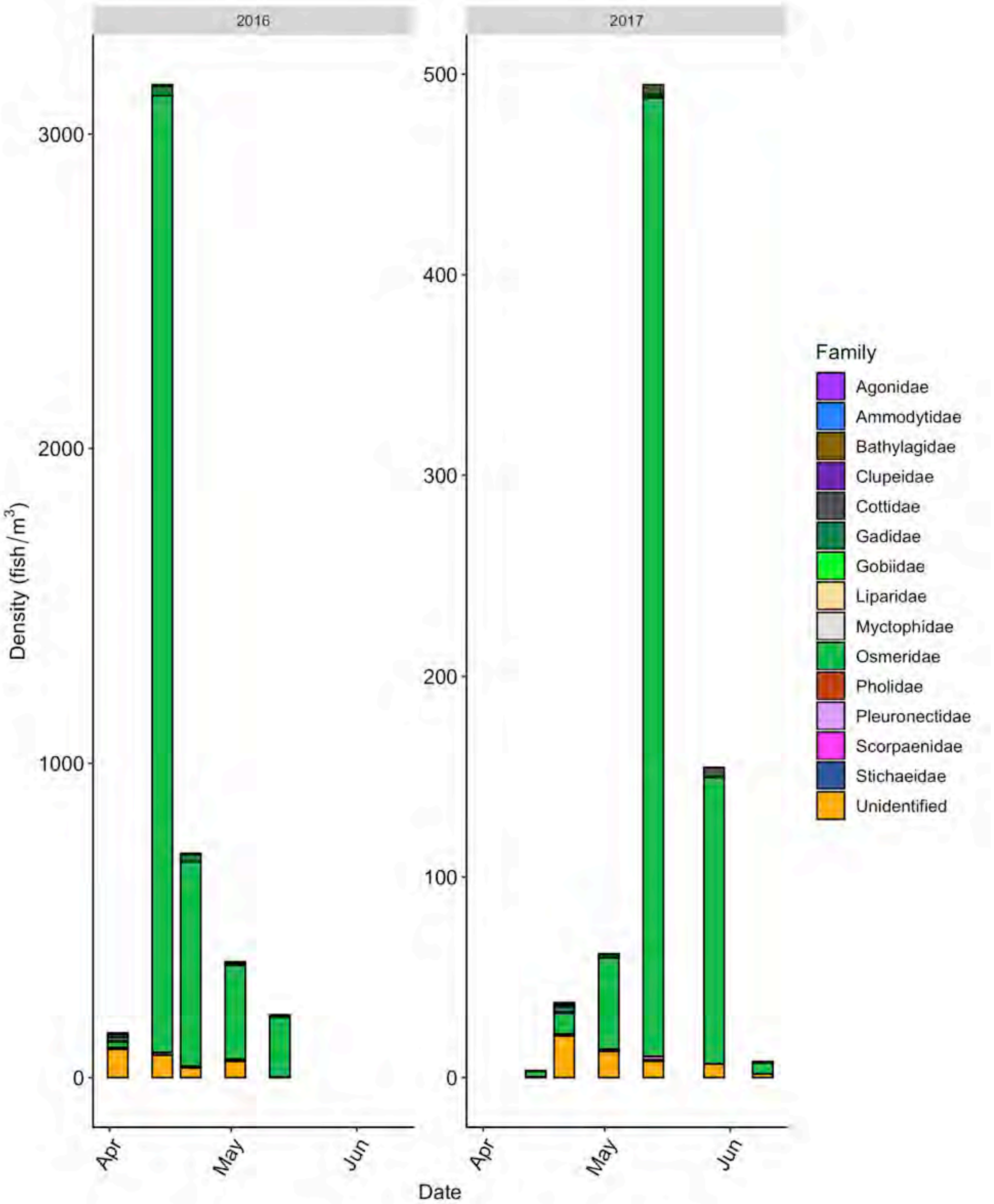

Figure 5-18. Larval fish densities by family across sampling dates from 2016 and 2017 samples collected in the Skeena River estuary. Eulachon accounted for 99.9% (2016) and 98.7% (2017) of larval fish from the family Osermeridae, the most abundant family shown in the figure (green).

Table 5-4. Total annual density (fish/m^3) by taxon of larval fish captured in the Skeena River estuary during April–June sampling in 2016 and 2017 (continues on page 179). When possible, larval fish were identified to species. Those lacking identifying features were classified to family and are represented as indenterminate (indet.), while fish identified to genus are represented as sp.

Family	**Species**	**Common Name**	**Density (fish/m^3)**	
			2016	**2017**
Agonidae	*Agonidae* indet.	Poacher sp.	0.16	
	Bathyagonus sp.	Poacher sp.		0.09
	Odontopyxis trispinosa	Pygmy Poacher		0.14
Ammodytidae	*Ammodytes personatus*	Pacific sand lance	4.70	0.76
Bathylagidae	*Leuroglossus schmidti*	Northern Smoothtongue	0.16	
	Leuroglossus stilbius	California Smoothtongue	0.14	
Clupeidae	*Clupea pallasii*	Pacific herring	7.00	0.16
Cottidae	*Artedius fenestralis*	Padded Sculpin	0.12	
	Artedius harringtoni	Scalyhead Sculpin	3.54	0.30
	Artedius lateralis	Smoothhead Sculpin	8.48	1.14
	Chitonotus pugetensis	Roughback Sculpin	0.27	
	Clinocottus acuticeps	Sharpnose Sculpin		0.12
	Cottidae indet.	Sculpin sp.	0.16	0.08
	Cottus asper	Prickly Sculpin	6.14	8.66
	Leptocottus armatus	Pacific staghorn sculpin	7.63	1.12
	Radulinus asprellus	Slim Sculpin	0.06	0.31
	Ruscarius meanyi	Puget Sound Sculpin	0.85	0.23
	Gadus chalcogrammus	Walleye Pollock	65.53	5.32
Gadidae	*Gadus* sp.	Cod		0.08
	Rhinogobiops nicholsii	Blackeye Goby	0.06	
Liparidae	*Liparis fucensis*	Slipskin Snailfish	0.76	0.94
	Liparis sp.	Snailfish sp.	2.56	0.16
Myctophidae	*Stenobrachius leucopsarus*	Northern Lampfish	0.26	
Osmeridae	Osmeridae indet.	Smelt sp.	22.15	9.20
	Thaleichthys pacificus	Eulachon	4175.44	675.21
Pholidae	*Pholis laeta*	Crescent Gunnel	4.12	0.14

Table 5-4 cont'd. Total annual density (fish/m3) by taxon of larval fish captured in the Skeena River estuary during April–June sampling in 2016 and 2017.

Family	Species	Common Name	Density (fish/m^3)	
			2016	2017
Pleuronectidae	*Hippoglossoides elassodon*	Flathead sole	1.85	
	Lepidopsetta bilineata	Southern Rocksole	4.10	0.08
	Lyopsetta exilis	Slender Sole	2.25	0.17
	Parophrys vetulus	English Sole	5.69	0.66
	Platichthys stellatus	Starry Flounder	4.98	
	Pleuronectidae indet.	Flatfish sp.	1.01	1.24
Scorpaenidae	*Sebastes* sp.	Rockfish sp.	4.44	1.92
Stichaeidae	*Lumpenus sagitta*	Snake Prickleback	0.12	
	Poroclinus rothrocki	Whitebarred Prickleback	0.12	
	Stichaeidae indet.	Prickleback sp.	0.26	0.57
Unidentified	Pisces indet.	Unknown sp.	246.61	51.30

DISCUSSION

Large numbers of eulachon larvae were present in the water column at all sampling stations in the middle of April during 2016 and May in 2017, representing 90% of the total larval fish caught during the sampling period. In addition, significantly higher densities of larval eulachon were recorded in 2016 compared to 2017; approximately six times more eulachon were captured in 2016 during peak abundances. These results suggest that the in-river eulachon spawning migration was likely earlier and larger in 2016 when compared to 2017.

Although no publicly available data exists on annual spawner biomass or timing windows, Lax Kw'alaams eulachon harvesters found the eulachon migration in 2016 to be larger and last longer than the eulachon run in 2017 (Scott Campbell and James Harvey Russell, Lax Kw'alaams First Nation, pers. Comm. July 2017). Eulachon abundance in both years seemed to be significantly higher than the poor eulachon returns documented in the early 2000s but lower than the large run documented in 2019. In 2016, eulachon spawned in the lower Skeena River from the fourth week of February until the second week of March 2016, with peak spawning activity observed between February 27 and March 6, 2016. This is approximately 6 weeks prior to peak abundances of larval eulachon detected during 2016 estuary sampling (Mark Biagi, Kitsumkalum Fisheries, pers. comm). In 2017, peak spawning activity was reported to have occurred between March 14–19, 2019, approximately two weeks later than in 2016. We found the highest densities of eulachon in the Skeena River estuary approximately eight weeks after peak spawning was reported to have occurred within the river in 2016. These results coincide with the reported eulachon egg incubation period, which is temperature dependent and typically lasts for 2–8 weeks before the larvae hatch and are rapidly carried downstream to the estuary (Howell 2001). Thus, earlier timing of larval eulachon observed in the Skeena River estuary during 2016 is likely related to earlier timing of peak in-river spawning of adult eulachon.

Larval eulachon densities in the Skeena River estuary may also be related to environmental conditions. The timing of egg hatching is thought to coincide with peak spring river discharge (Hay and McCarter 2000), however, it is unclear how differences in timing and magnitude of annual discharge will influence factors such as length of egg incubation, egg and larval survival, and subsequent arrival of eulachon larvae in the estuary environment. Sharma et al. (2017) found that higher eulachon survival and abundance in the Columbia River was dependent on higher spring flows. In contrast, Langness et al. (2020) found no change in density of larval eulachon across varying daily discharge levels. Given that eulachon spawn in freshwater prior to spring discharge periods and are subsequently flushed downstream during peak spring freshwater flow (Howell 2001), the arrival of eulachon larvae in the river mouth and estuary may be linked with discharge conditions. In the Skeena River estuary. the volume of the spring freshet in 2017 was greater and later in the spring when compared to 2016. Maximum spring discharge recorded at the Usk discharge station was 5730 m^3/s and occurred on June 10 in 2017, which was twice as large when compared to the maximum discharge on May 4, 2016 (2530 m^3/s). A later spring discharge period in 2017 potentially influenced the timing of peak larval eulachon abundances found in the estuary. Multiple years of study are required to investigate how the magnitude of the Skeena River water levels and discharge during the downstream migration period for larval eulachon may influence their timing, distribution, and density throughout the Skeena estuary.

Section 5.5. - Key Findings

1) High densities of larval eulachon (90% of all larval fish caught) were caught in the Skeena River estuary between April 13 and May 13, approximately 6 weeks after peak spawning activity was observed in the Skeena River.

2) Larval eulachon were caught in highest abundance at locations close to the mouth of the Skeena river.

3) The Skeena River estuary supports at least 14 families of larval fish including sculpin (Cottidae), cods (gadoids), rockfishes (Sebastidae), flatfishes (Pleuronectidae) (most abundant at sites further from the river mouth), and Pacific herring (most abundant in early April).

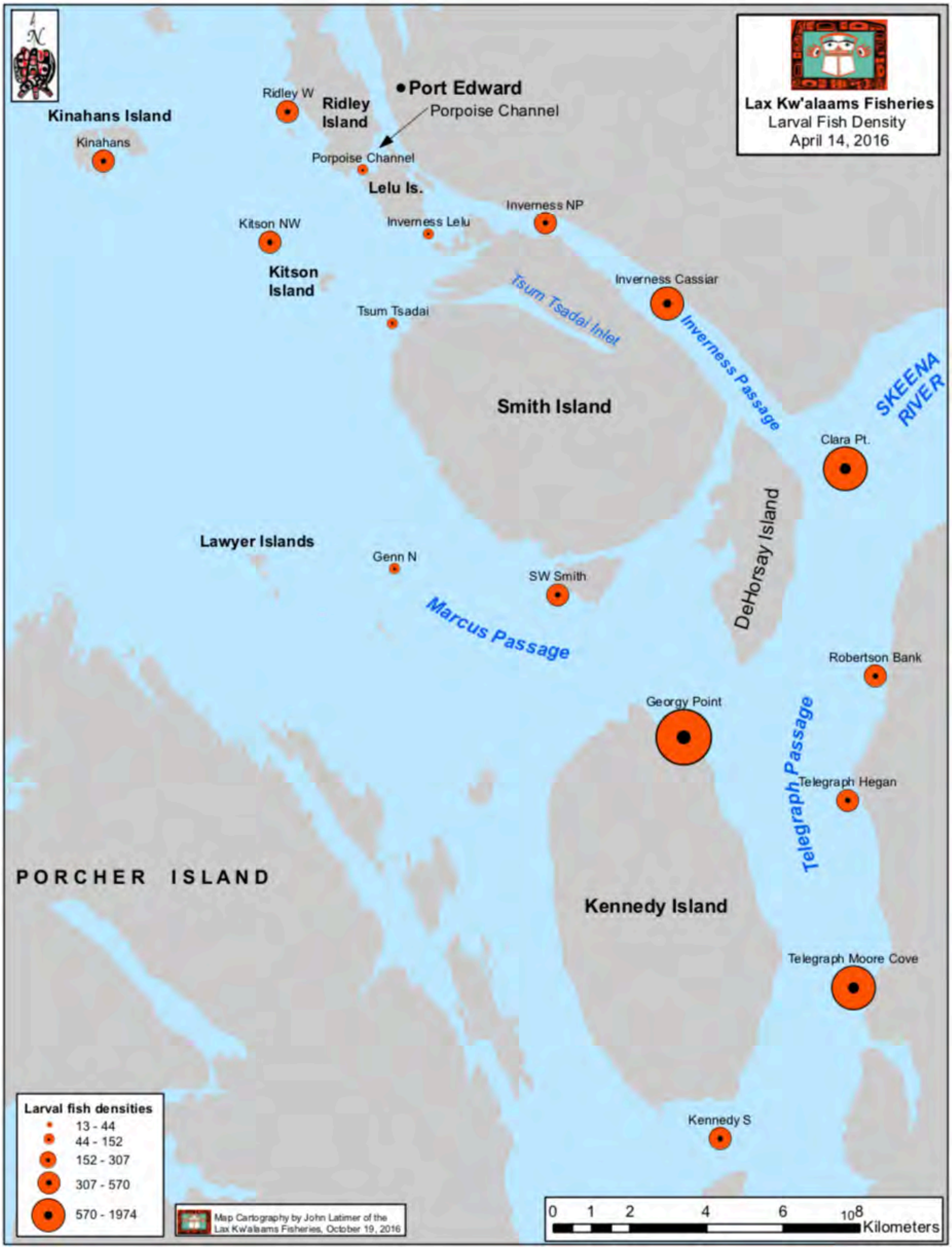

Figure 5-19. Spatial distribution of larval fish in the Skeena River estuary study area on April 14, 2016. Point area represents larval fish density (fish/m^3) at each site (map by John Latimer, Lax Kw'alaams Fisheries).

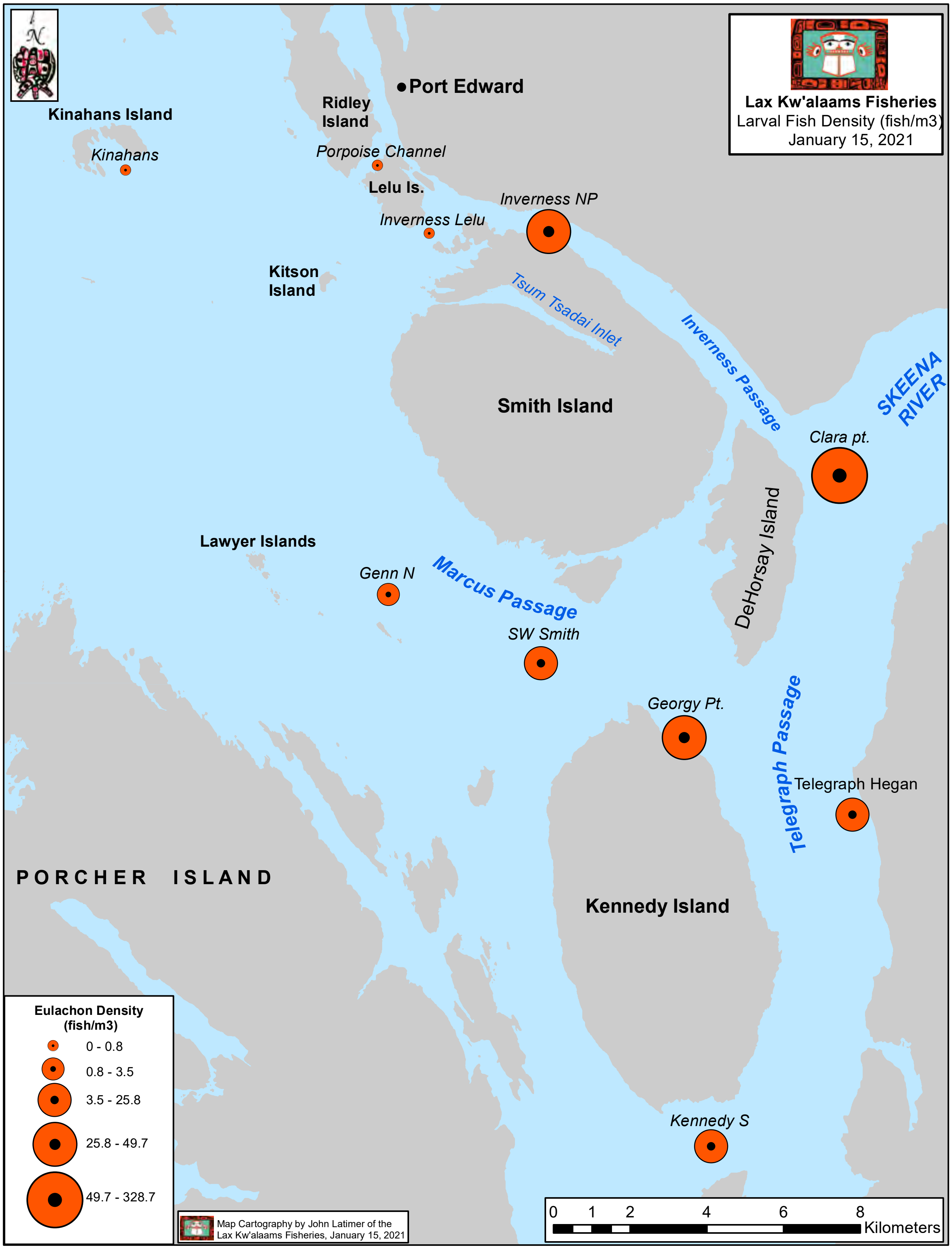

Figure 5-20. Spatial distribution of larval fish in the Skeena River estuary study area on May 17, 2017. Point area represents larval fish density (fish/m^3) at each site (map by John Latimer, Lax Kw'alaams Fisheries).

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5.6. Crab utilization of the Flora Bank Region (Contribution by Janvier Doire)

This section contains an abridged version of the following report:

Doire, J. (2017). 2016 Skeena estuary dungeness crab telemetry survey. Prepared for the Lax Kw'alaams Band and Fish Habitat Restoration Initiative (DFO), Skeena Fisheries Commission, Kispiox BC.

INTRODUCTION

The Dungeness crab is among the most conspicuous benthic predators in coastal waters of the Northeast Pacific. It ranges from the Aleutian Islands to central California (MacKay 1942), and is exploited in commercial and recreational fisheries throughout most of its range. Along the north coast of British Columbia, the Dungeness crab is a significant marine resource, with a commercial catch of 5,081 tonnes valued at $29.4 million in 2009 (Dunham et al. 2011). The Dungeness crab is also a focal species in the culture and diet of coastal First Nations in BC, with many communities harvesting crabs year-round for sustenance. This species is also ecologically important as both predator and prey at all life stages, in all the ecological niches it occupies. Planktonic larval stages of the species are preyed upon by whales and many fish species including coho and Chinook salmon, and herring. Juveniles crabs are consumed by small benthic flatfish, and adults fall prey to bigger bottom feeders. Sub-adult and adult Dungeness crab occur year-round in the Skeena River estuary, on and around Flora, Agnew, and Horsey banks (Stantec 2016). Pelagic larvae occur in the estuary during the spring. Flora Bank specifically supports important commercial, recreational, and First Nation Dungeness crab fisheries (Hinton 1975).

The Dungeness crab is the only commercially-important invertebrate species to utilize estuaries as nursery habitat in the northeast Pacific (Gunderson et al. 1990; Rooper et al. 2002). Juvenile crabs (young-of-the-year and sub-adults) are found buried in intertidal sand (MacKay 1942) and/or associated with eelgrass (*Zostera* sp.) and other aquatic vegetation in estuaries that provide prey and protection from predators (Hoopes 1973; Stevens & Armstrong 1984, 1985; Rooper et al. 2002). While predation rates in estuaries are high, juvenile crab survival may be higher due to faster growth rates than in the open ocean (Gunderson et al. 1990). Faster growth in the estuary would offset high predation rates by reducing the period of vulnerability to predators. After reaching a size refuge from predation (>20 mm), age 0+ crabs move into deeper subtidal areas as they grow (MacKay 1942; Armstrong et al. 1995), although sub-adults require littoral habitats as important foraging areas (Armstrong et al. 2003; Holsman et al. 2006). Adult Dungeness crabs generally occupy subtidal sandy, muddy, or silty habitats of estuaries, near eelgrass beds (Hoopes 1973; Alaska Department of Fish and Game 1978; Karpov 1983), and venture into nearby littoral areas during night time, high tides to forage on the abundant prey (Stevens & Armstrong 1984; Holsman et al. 2006; Curtis & Armstrong 2007).

Dungeness crabs exhibit annual migration patterns. Adult crabs inhabit deep, offshore habitats during the winter months, and return to nearshore waters in the early spring and summer for mating and foraging (MacKay 1942; Hoopes 1973; Alaska Department of Fish and Game 1978; Stone & O'Clair 2002). Mating takes place largely on tide flats, and can occur only immediately after a female has molted in late spring to early summer (MacKay 1942). Males will carry females in a mating embrace when they are about to moult (Hoopes 1973; Dunham et al. 2011). Females store sperm so they can fertilize the eggs at a later date. In BC, adult female Dungeness crabs extrude fertilized eggs in the fall and are relatively inactive while egg-bearing during the winter (Stone & O'Clair 2001), when they remain buried in the soft bottom sediment much of the time and seldom feed (MacKay 1942; Stone & O'Clair 2002; Dunham et al. 2011). The eggs hatch in December to June, with hatching reaching a peak in March (MacKay 1942). The crab larvae or zoeae stay in the water column and are distributed offshore and alongshore by ocean currents for up to 4 months (MacKay 1942; Hoopes 1973). The megalopae, the final larval stage, then settle on soft bottom substrate in shallow coastal zones or estuaries (Lough 1976; Gunderson et al. 1990; McConnaughey et al. 1992) to become the first true non-larval stage (young-of-the-year). Megalopae are strong swimmers, and seek out appropriate habitats for settlements, and large numbers enter the estuaries of the northeast Pacific as late megalopae. As with many marine invertebrate species, Dungeness crabs form a metapopulation whereby local populations of relatively sedentary juveniles and adults are interconnected by dispersing larvae (Dunham et al. 2011). This means individual crab populations are sustained either by larvae originating from the parent stock and/or from other populations over a broader geographical area.

Because estuaries provide excellent habitat to Dungeness crabs, dredging and development in estuaries can have serious consequences for this species. Studies have shown that crab populations are likely to be reduced by habitat alteration from dredging and development unless proper precautions are taken to reduce losses (Armstrong et al. 1982; Stevens & Armstrong 1984). Additionally, the loss of vital estuarine habitat could significantly reduce recruitment to the offshore habitat (Armstrong & Gunderson 1985). Within the time frame of this project (2015–2017) there were significant concerns that the construction and infrastructures related to the approved Pacific Northwest LNG project may have impacted the Dungeness crabs' migratory pathways and access to important habitat present on and around Flora, Agnew, and Horsey Banks. Given the unique nature of Flora Bank (Section 2.2), one of the main objectives of this study was to assess the utilization of this region by Dungeness crab and the potential effects of the proposed project.

METHODS

The study area for this project falls within the Skeena River estuary and is comprised of Flora, Agnew, and Horsey Banks (Figure 5-21). The study area has a surface area of approximately 13.5 km^2 (Stantec 2016). Dungeness crab habitat use and migration patterns were studied using an acoustic telemetry methodology to track crab movement in the study area from June 2016 to early January 2017. Acoustic telemetry has been used successfully to observe the movement patterns of Dungeness crab and other crab species (Wolcott 1995; Stone & O'Clair 2002; Holsman et al. 2006), and provides a method for observing in situ activity of sub-adult and adult Dungeness crab without dramatically altering their behavior. Fifty sub-adult and adult Dungeness crabs were captured, tagged, and released during two distinct tagging periods: a) early summer (June) and b) late summer (August). Tagging was focused on Agnew and Horsey Banks, immediately around Flora Bank, with supplemental captures and releases on Flora Bank. Six stationary acoustic telemetry receivers were deployed and maintained on Flora Bank (Table 5-5 and Figure 5-21). A grid of active telemetry survey locations was established over most of the survey area (Figure 5-21) and surveys to determine the position of tagged crabs occurred during early summer to fall.

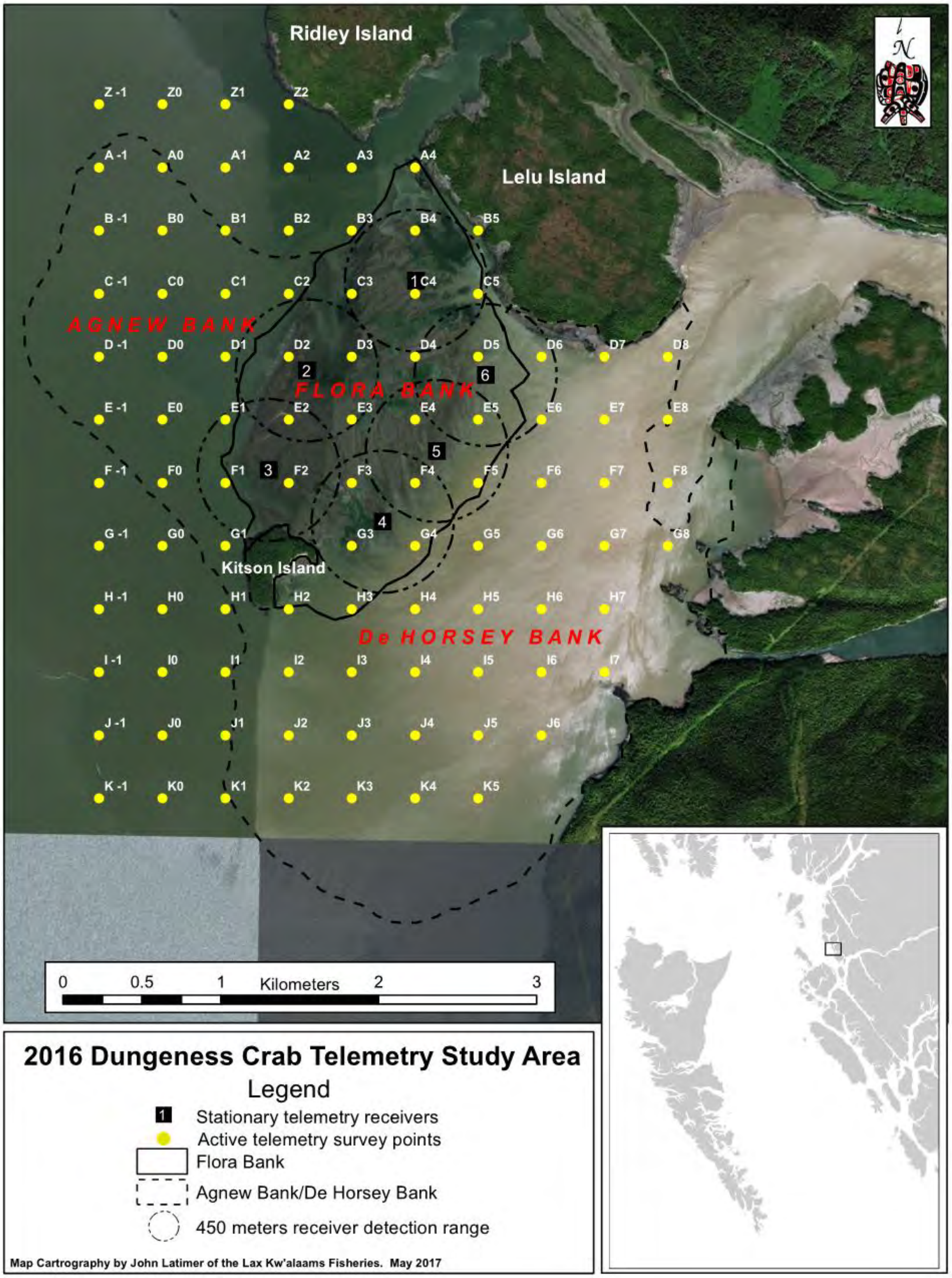

Figure 5-21. Map of the 2016 Dungeness crab study area in the Skeena River estuary (map by John Latimer, Lax Kw'alaams Fisheries).

Table 5-5. Dates and locations of stationary acoustic telemetry receiver installation conducted in 2016 to monitor Dungeness crab in the Skeena River estuary.

Stationary receiver #	Date installed	Date retrieved	Easting	Northing
1	June 13	January 4	0415008	6005932
2	June 13	January 4	0414330	6005369
3	June 13	January 4	0414082	6004724
4	July 14	January 4	0414795	6004403
5	June 14	January 4	0415142	6004854
6	June 14	January 4	0415460	6005335

Crab Capture and Tagging

Dungeness crabs were captured using commercial grade traps modified to prevent smaller crabs from escaping. Traps were baited with salmon or halibut carcasses (heads and/or spines) and set to soak overnight for 18 to 24 hours. Upon retrieval of traps, all crabs were identified to species, sexed, measured for carapace width, tested for shell hardness, and assessed for injuries and abnormalities (e.g. missing claw/legs or cracked shell). Shell condition was assessed based on DFO’s shell condition scale (Table 5-6), which is a subjective estimate of the intermolt stage of the crab based on characteristics of shell plasticity, overall body wear, and epiphytic growth (Dunham et al. 2011).

Table 5-6. Dungeness crab shell condition, approximate time since the last molt (from Dunham et al. 2011).

Shell Condition	Time since moult
Soft Plastic	2–6 days
Soft Crackly	6 days–1 month
Soft Springy	1–3 months
New Hard	3–6 months
Between New and Old	6–12 months
Old Shell	12–24 months
Very Old Shell	> 24 months

We tagged a total of 50 Dungeness crabs, including 26 females (12 tagged in early summer and 14 in late summer) and 24 males (13 in early summer and 11 in late summer). The notch widths of females ranged from 119 to 154 mm and males from 105 to 152 mm. The acoustic tags were attached in the cervical groove of the crabs’ carapace using Gorilla Super Glue GelTM brand cyanoacrylate adhesive as done by Holsman et al. (2006) (Figure 5-22). To minimize the loss of tagged crabs to fisheries in the project area, only males measuring less than the legal size of 154 mm notch width were tagged and released. Furthermore, as adult Dungeness crabs typically molt once per year (McKay 1943), only crabs with newer shells and little epibionts were tagged, ensuring the longest possible intermolt time and tag retention. Tagged crabs were released at their capture location once the adhesive had hardened. The tagging process was brief, requiring only a few minutes to complete.

The Dungeness crabs were tagged with coded 69 kHz acoustic transmitters (model V9 manufactured by Vemco Inc.) (Figure 5-22). The acoustic transmitters were 9 mm in diameter, 29 mm in length, 4.7 grams in weight (in air), and had a power output of 151 dB. The transmitters used during the early summer tagging period had an average delay of 60 seconds between pings and an estimated battery life of 189 days, and those used during the late summer tagging period had an average delay of 30 seconds between pings and an estimated battery life of 102 days.

Figure 5-22. A Dungeness crab with a Vemco V9 acoustic tag attached to its carapace with cyanoacrylate adhesive.

Tracking Equipment and Procedures

The proper function of acoustic tags was confirmed before deployment. Once released, the tagged Dungeness crabs' locations were monitored by tracking the location of the tags using stationary acoustic receivers and loggers, and mobile acoustic tracking equipment.

Stationary acoustic receivers/data loggers

The stationary acoustic receivers/loggers were 69 kHz, model VR2W manufactured by Vemco Inc., with 16 Mb in storage capacity and Bluetooth® wireless technology to offload data (Figure 5-23). The receivers were secured to 60 x 60 x 8 cm concrete pads using PVC tubing and stainless steel eye-bolts (Figure 5-24). The concrete pad/receiver assemblies were installed on Flora Bank with buoys to mark the receiver locations (Figure 5-24).

A detection range test was conducted prior to installation of the receivers on Flora Bank to determine the approximate range at which the VR2W acoustic receiver could detect a V9 acoustic transmitter on Flora Bank. The results showed that the VR2W receiver's detection range was highly variable and dependent on water levels, which vary by more than 7 m on Flora Bank over a tide cycle and bottom undulations. It was determined that a tag would be detected if it was within a range of 450m or less from a receiver. Considering these results, it was established that six VR2W receivers separated by 600 to 800 m would suffice to detect the presence of a transmitter on Flora Bank (Figure 5-21). Five receivers were installed in mid-June 2016, and the last receiver was installed in mid-July 2016 (Table 5-5). Data recorded by the receivers was regularly offloaded onto a computer over the study period so the loss of a receiver would not result in the loss of a significant amount of data. There were no losses of receivers, or instances of receiver failure during the whole duration of the study. All receivers were retrieved on January 4, 2017.

Figure 5-23. Acoustic telemetry tracking equipment used in the 2016 Dungeness crab study: Vemco VR2W stationary receiver/logger (left), Vemco VR100 acoustic receiver (center), pole mounted Vemco VH110 directional hydrophone (right), and Vemco VH165 omni-directional hydrophone (bottom).

Figure 5-24. A stationary acoustic receiver/logger with its concrete pad and PVC tubing assembly (left) and installation on Flora Bank during low tide (right).

Active telemetry surveys

Active telemetry surveys were conducted from a boat in June (7 days), July (11 days), August (9 days), September (5 days), and October (6 days), for a total of 38 days. A typical survey would begin with listening for tagged crabs at predetermined mobile survey locations (Figure 5-21) using a model VH165 69 kHz omni-directional acoustic hydrophone connected to a VR100 acoustic receiver, both manufactured by Vemco Inc. (Figure 5-23). Once a tagged crab was detected, a pole mounted Vemco Inc. VH110 (69 kHz) directional acoustic hydrophone (Figure 5-23) was connected to the VR100 receiver instead of the omni-directional hydrophone, and the tagged crab was tracked. The geographical location of the tagged crabs could be identified and recorded with relatively good accuracy. An active telemetry tracking test was conducted using a transmitter at a known location. This test established that the location of the transmitters could be easily identified to within 50m with the VH110 and VR100 combination, and to less than 10m if more time was spent detecting the transmitter location. Low speed was required for accurate telemetry tracking of tagged crabs and at times, wind and/or waves made this impossible or the tracking crew did not have enough time to track. In those instances, the location where the tagged crab was detected while listening with the omni-directional hydrophone was recorded.

Baited crab traps were set opportunistically near some of the tagged crab locations to attempt to re-capture them. In addition to the survey area, Porpoise channel from its entrance to Port Edward, and Inverness Passage to the North Pacific cannery, were each surveyed twice using active telemetry tracking.

Individual Tagged Crab Telemetry Data Mapping

All the detections of tagged crabs made using the directional hydrophone were documented including the date of detection. The detections made while listening with the omni-directional hydrophone were mapped with associated tracking date only if these detections were not immediately followed by a more accurate tracking detection. The stationary receivers' detections were compiled in tables showing the dates an individual crab was detected by each individual receiver. These tables were incorporated in the individual crab maps.

RESULTS

Tagged Dungeness crabs were detected throughout the study area, on Flora, Agnew, and Horsey Banks from late June 2016 to late October 2017 (Figure 5-25). A total of 103 and 144 detections of tagged crab were made during the study while listening with the omni-directional hydrophone and tracking with the directional hydrophone, respectively (Table 5-7). The stationary receivers recorded a total of 177,526 tagged crab detections between mid-June 2016 and early January 2017. Receivers 1, 2, 5, and 6 recorded between 23,000 and 25,000 tagged crab detections each, while receivers 5 and 6 recorded over 40,000 tagged crab detections each. All tagged Dungeness crabs were detected at least once after their release, except for crab #384. No crabs were detected in Inverness Passage, nor in Porpoise channel. For more detailed information and figures on individual crab movement see the original report (Doire 2017).

Tagged female and male Dungeness crabs all remained in the study area for a considerable amount of time following their release, either in June 2016 or August 2016. The majority of the crabs released in June 2016 (60%) and August 2016 (64%) were last detected in October 2016 or later (Figure 5-26). On average, 86.3 and 67.1 days elapsed between the release and the last detections of female crabs released in June 2016 and August 2016, respectively (Table 5-8). For male crabs, an average of 94.4 days elapsed between their June 2016 release and last detection, whereas an average of 36.0 days elapsed between their August 2016 release and last detections (Table 5-8). The average amount of time elapsed between the releases and last detections were not statistically different between males and female released in June 2016 nor in August 2016 ($p = 0.1794$ for August releases).

Female crabs tagged and released in June 2016 were detected an average of 4.7 times during the active surveys, whereas the males tagged and released at the same time were detected 7 times on average. In contrast, female crabs tagged and released in August 2016 were detected more often than the males (4.6 versus 3.2 on average). However, these differences in the number of detections for males versus females were not statistically significant ($p > 0.05$) (Table 5-8).

On average, the stationary receivers installed on Flora Bank detected the presence of male crabs more often than females for the crabs released in both June 2016 (4,770 versus 578 detections) and August 2016 (6,877 versus 4,770 detections), however, these differences were not statistically significant ($p > 0.05$).

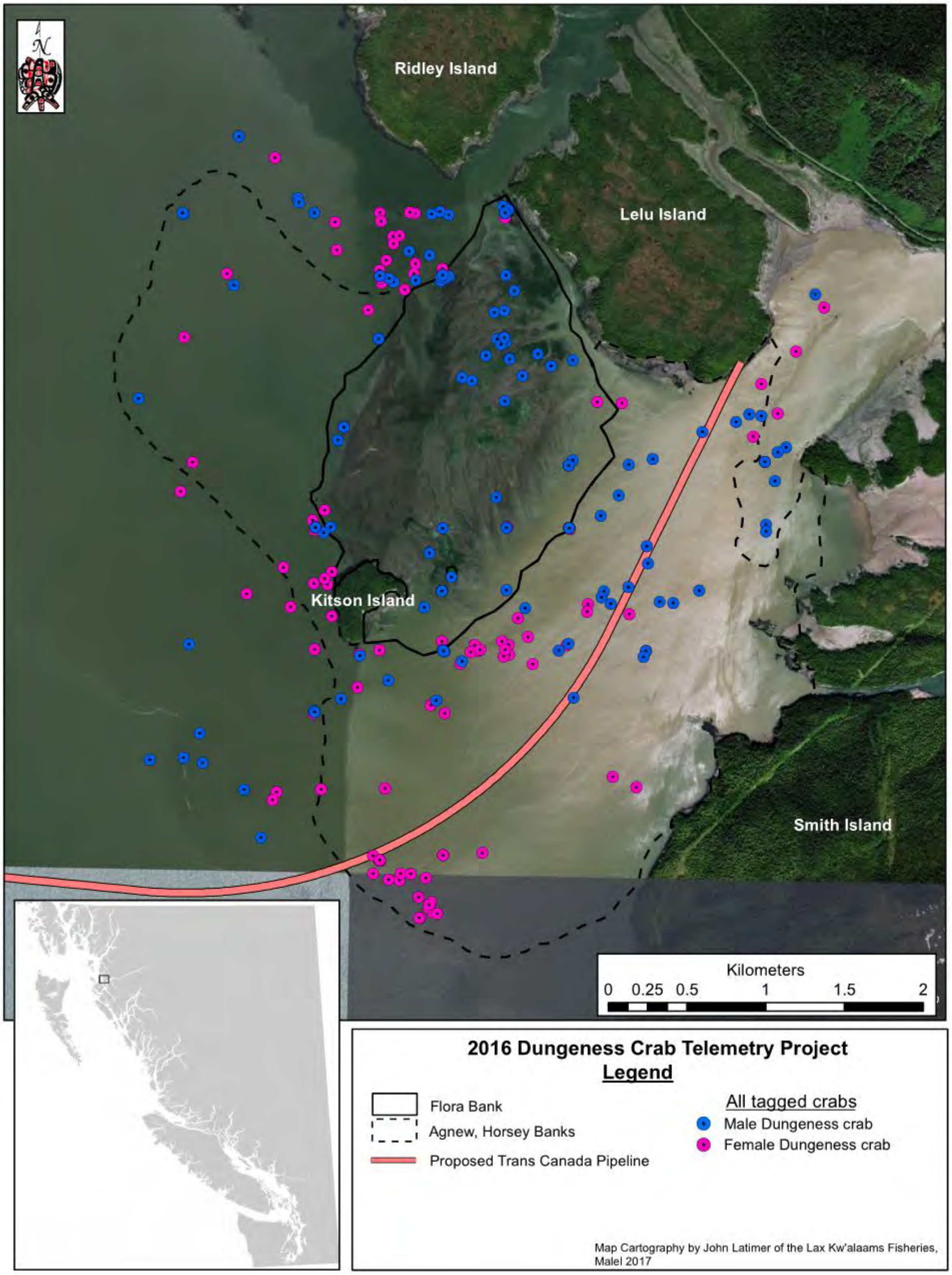

Figure 5-25. Map showing tagged Dungeness crab detections made during the active tracking surveys conducted between June and October 2016 (map by John Latimer, Lax Kw'alaams Fisheries).

Table 5-7. Number of tagged crab detections made while listening and tracking, and recorded by the stationary receivers.

	Listening	Tracking	Total receivers	Receiver 1	Receiver 2	Receiver 3	Receiver 4	Receiver 5	Receiver 6
# of detections	103	144	177,526	23,139	25,067	41,901	41,322	22,814	23,282

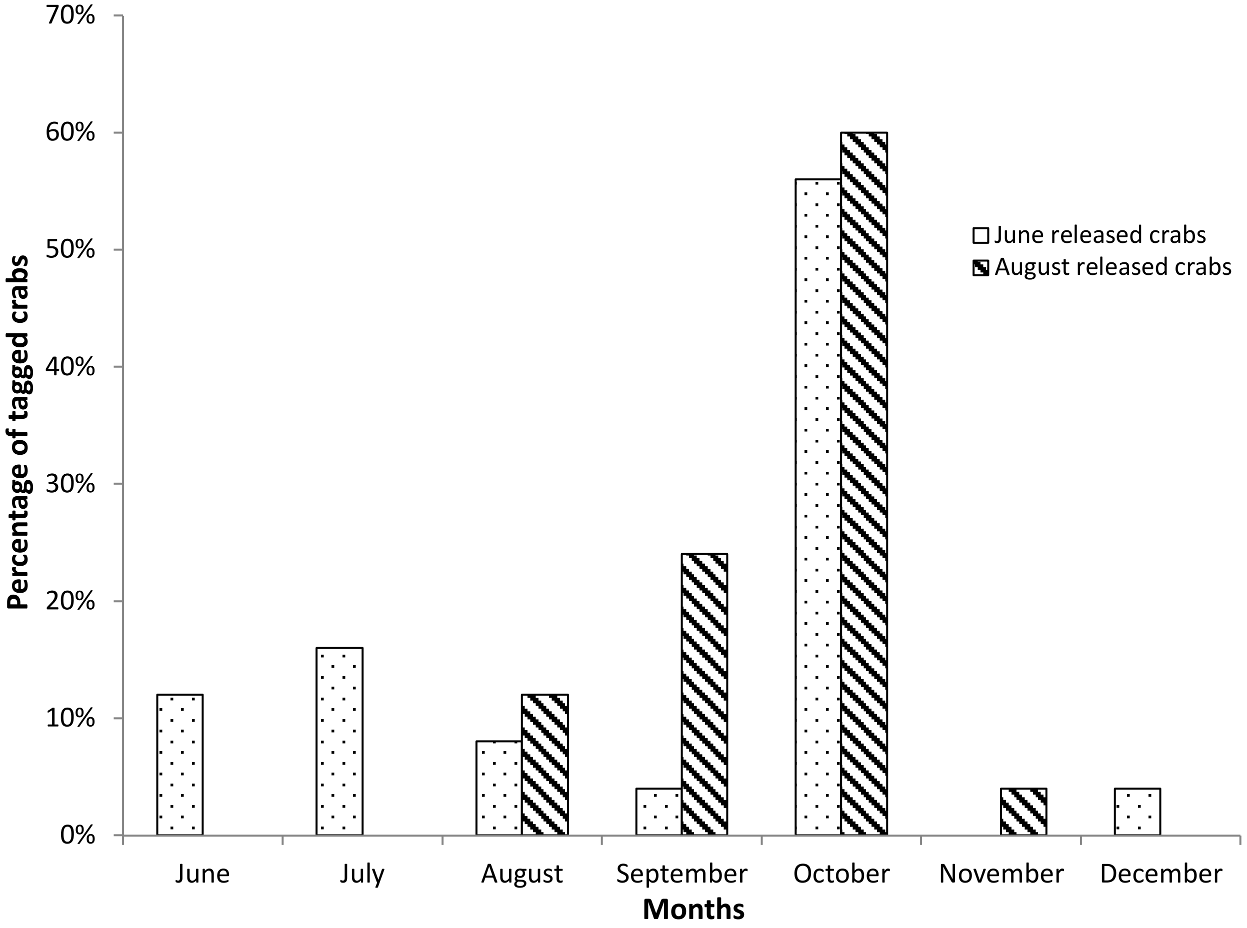

Figure 5-26. Monthly percentage of last detections either by active telemetry or stationary receiver.

Tagged male Dungeness crabs released in June traveled a greater distance on average (6,959 m) than the females released in June (4,055 m). In contrast, the females released in August travelled a greater distance on average than the males released at the same time (3,277 m). None of these differences were found to be statistically significant ($p > 0.05$). Figure 5-27 shows that the hourly distribution of the detections recorded by the stationary receivers was heavily weighted towards hours of darkness for both female and male tagged Dungeness crabs. Three-quarters of the detections of female crabs were recorded within 8 hours, between 22:00 and 6:00, and two-thirds of the male crabs detections were recorded during the same period.

Table 5-8. Mean time between release and last detection, number of detections during active surveys and by stationary receivers, and distance between consecutive active survey detections for female and male Dungeness crabs tagged in June and August 2016. One-way ANOVA statistical test results comparing values between female and male are included.

	Tagged female crabs - mean		**Tagged male crabs - mean**		**Statistical difference between female/male**	
	June	August	June	August	June	August
Time between release and last detection (days)	86.3	67.1	94.4	36.0	No (p=0.9428)	No (p=0.1794)
# detections – active tracking	4.7	4.6	7.0	3.2	No (p=0.6182)	No (p=0.5816)
# detections – stationary receivers	578	2,352	4,770	6,877	No (p=0.4011)	No (p=0.6288)
Total distance between active tracking detections (m)	4,055	4,285	6,969	3,277	No (p=0.2896)	No (p=0.6444)

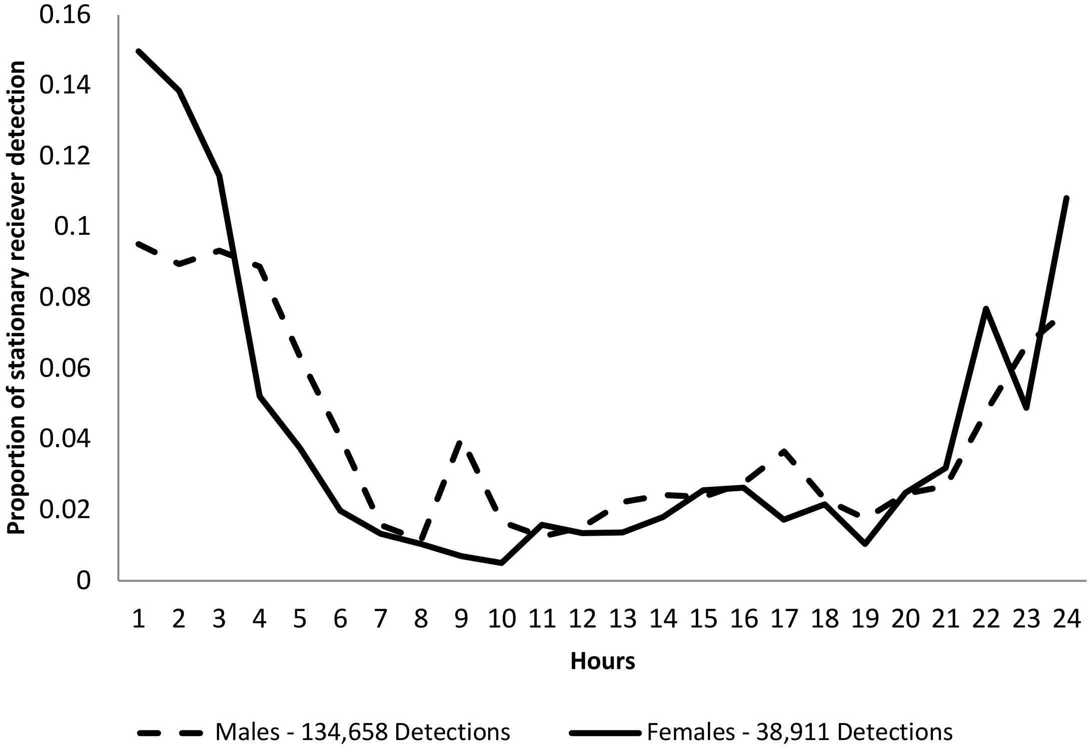

Figure 5-27. Hourly distribution of stationary receiver detections for male and female tagged crabs during the duration of the 2016 study.

DISCUSSION

The results of this telemetry study tracking the movements of Dungeness crabs in the Skeena River estuary confirm that this estuary is not unlike others in the northeast Pacific with respect to Dungeness crab. The Skeena River estuary provides very important habitat for Dungeness crab, at least from the early summer to fall each year, but also provides valuable habitat year-round for ovigerous female Dungeness crabs. Most of the crabs tagged and released either in June 2016 or August 2016 were detected well into October 2016, the last month that active telemetry surveys were conducted. Many crabs travelled long distances throughout the study area (e.g. crabs #387, 392, 833, 380, and 388), while others remained in the same area for long periods of time (e.g. crabs #830, 378, 377, 375, 829, and 390).

Dungeness crab summer/fall forage habitat

Tagged crabs were detected by stationary receivers more often during darkness hours than during daylight hours (Figure 5-27), demonstrating that the crabs were closer to the stationary receivers located on Flora Bank at night versus during the day. These diel movement patterns, where Dungeness crabs move onto littoral and tidal habitat at night to forage, were observed for Dungeness crab from Grays Harbor, Washington (Stevens et al. 1984), the Sarita River estuary, Barkley Sound, British Columbia (Curtis & Armstrong 2007), and Willapa Bay, Washington (Holsman et al. 2006). Many other crab species are also known to have daily cycles, with nocturnal peaks in activity commonly found in the Cancer genus (Novak 2004). The nocturnal timing of crab migration onto Flora Bank, the shallowest portion of the study area, is most likely to benefit from the abundance of prey provided by the extensive eelgrass patches on Flora Bank (Warren 2017) while minimizing visual predation (Stevens et al. 1984; Gunderson et al. 1990). Furthermore, many tagged crabs were located close to the edges of Flora Bank, in deeper waters, during daytime active tracking surveys (e.g. crab #'s 829, 375, 377, 379, 388, 821, 393, and 816). These crabs were potentially waiting for darkness or high tide before moving onto Flora Bank to forage.

Despite observing that crabs preferentially move onto Flora Bank during hours of darkness, some tagged male crabs (crabs #378, 387, 392, 832, and 839) were tracked and even re-captured (crab #839) on Flora Bank during daytime surveys. Some male crabs stayed on Flora Bank for numerous days to a couple of months (e.g. crab #378). On September 20, 2016, crab #832 was tracked on Flora Bank immediately preceding and following a low tide during which Flora Bank was completely above water. This crab must have spent the duration of this large low tide buried in the sand as the crew looked for it, but could not find it. We observed one crab actively burying itself in the sand on Flora Bank as it was getting stranded due to another large low tide.

Dungeness crab mating habitat

The fact that only males were observed on Flora Bank for extended periods of time and appeared to be barely moving (e.g. crab #378), and only females that had already molted (hence were not available to mate) were tagged may indicate that Flora Bank is not only used as foraging habitat, but also as mating habitat. Further evidence of this was observed on June 6, 2016, when two crab couples (male and female) were observed in an embrace partially buried in the sand and eelgrass of Flora Bank during low tide (Figure 5-28). The male was holding the female in an embrace, most likely waiting for it to molt, as described by Dunham et al. (2011) and Hoopes (1973). This demonstrates that Flora Bank is used as mating habitat by Dungeness crabs.

Wintering and egg-brooding habitat

After the month of October 2016, only crab #822 in November 2016 and crab #396 in December 2016 were recorded by the stationary receivers. Most tagged crabs were not detected by stationary receivers after October, regardless of time of tag activation, suggesting that both female and male Dungeness crabs migrated away from Flora Bank and did not forage there in late fall and early winter.

No active tracking surveys were conducted after October 29, 2016, however, in mid-October 2016 five tagged males (crabs #379, 380, 387, 820, and 840) were tracked in the deepest portion of the study area, southwest of Kitson Island. These observations support findings from Tofino (Smith & Jamieson 1991) and Fritz Cove in southeast Alaska (Stone & O'Clair 2001) that found males retreated to deeper water in fall and winter and returned to shallower habitats the following spring.

The active telemetry surveys conducted in October 2016 found over 40% of tagged females (total of eleven females including crabs #383, 388, 394, 817, 818, 823, 824, 825, 827, 830, and 833) occupying an area of less than 1km in diameter at the southern end of Horsey Bank, 1.5 km south of Kitson Island (Figure 5-25). These tagged females were most likely part of a dense ovigerous female aggregation, like that observed by Stone and O'Clair (2001) in Fritz Bay. The aggregation found south of Kitson Island was so dense that when four baited crab traps were set to recapture one of the aggregating tagged females, we captured 139 females and 19 males after a total soak time of only six hours (Figure 5-29).

Before being detected among this egg-brooding female aggregation on October 28, 2016, female crabs #830 and 833 were tracked northwest of Flora Bank on October 12, 2016, where #830 had remained since its release on August 15, 2016. In just over two weeks, crabs #830 and 833 travelled approximately 3.5 km and 5.5 km (geodesic distances), respectively, to join the aggregation. Female crab #383 is another example of such efforts to reach the egg-brooding habitat south of Kitson Island. This crab was detected west of Kitson Island on October 9, 2016 at 16:05 PST and then again two days later on October 11, 2016 at 13:35 PST 2 km south (geodesic distances) within the ovigerous female aggregation. This is equivalent to a moving rate of at least 1 km/day. Although it was over only two days, this moving rate is much greater than rates observed by Smith and Jamieson (1991) and Stone and O'Clair (2001).

These observations strongly validate evidence for habitat selection by ovigerous female crabs put forth by Stone and O'Clair (2002). Furthermore, these examples of females travelling great distances relatively quickly to reach the dense group of females 1.5 km south of Kitson Island proves that the habitat in this area must have specific characteristics that are optimal for brooding eggs. Stone and O'Clair's (2002) in situ observations of female crab aggregations indicated that egg-brooding females were buried in sand, and the density of crabs measured within the Fritz Cove aggregation was at least 20 crabs/m^2 in early March 1999. The sediments in the area where ovigerous female crabs aggregated in Fritz Cove had low consolidation, permitting upwelling of water at the sediment interface, and consisted of moderately well-sorted fine sand with a low proportion of silt and clay

(Stone & O'Clair 2002). Such sediment properties provide a high rate of exchange of oxygen and metabolites, whereas finer sediments such as muds have poor water circulation and often low oxygen tension. Dungeness crabs can also bury more easily in unconsolidated sands (Stone & O'Clair 2002). The ovigerous female crab aggregations on Horsey Bank likely contained sediments similar to those observed by Stone and O'Clair (2002) in Fritz Cove.

It is likely that the Horsey Bank brooding site is used annually by ovigerous female Dungeness crab. For example, O'Clair et al. (1996) found repeated use of the same brooding sites between years in Glacier Bay, Alaska. A study in Fritz Cove, Alaska showed further evidence of site fidelity with purposefully displaced ovigerous females homing back to the brooding sites they were displaced from within 13 to 20 days (Stone & O'Clair 2002). The proposed location of the Trans-Canada pipeline to the historically proposed PNW LNG plant on Lelu Island ran very close to the egg-brooding female aggregation found on Horsey Bank.

Tag retention and post-tagging mortality

A number of results from the study would suggest that tag retention on Dungeness crab was good, and post-tagging mortality was low:

1) all tagged Dungeness crabs were detected at least once after their release, except for crab #384
2) the re-capture of crab #839 alive,
3) the observation of 40% of the tagged females within an area of only 1.5 km in diameter, and,
4) the detection of the majority of tagged crabs during the whole survey period.

CONCLUSIONS

Findings from this Dungeness crab telemetry project confirm what was expected: Flora, Agnew, and Horsey Banks (significant parts of the Skeena River estuary) provide important, year-round habitat to Dungeness crab and egg-brooding habitat for females. As a result, the PNW LNG terminal proposed to be constructed on Lelu Island which includes a bridge-trestle structure over Flora Bank, a docking facility 1 km northwest of Kitson Island, and a pipeline delivering LNG to the plant, is likely to have a negative impact on the Dungeness crab population in this area. The installation of the pipeline during the egg-brooding period would have the potential to remove at least 40% of the breeding female population of the area. Although the PNW LNG terminal was ultimately cancelled, projects that support shipping activities within the Prince Rupert region continue to be proposed. The results of this study provide further evidence regarding the importance of the Flora Bank region (see Section 2.1, Section 3.1 and Section 5.3).

Figure 5-28. Photos of a male and female Dungeness crab partially buried in sand and eelgrass (within yellow circle – top photo) on Flora Bank, with the male holding the female in an embrace (bottom photo) observed on June 6, 2016.

Figure 5-29. View of the catch (38 females and 6 males) from one baited crab trap soaked for 95 minutes 1.5 km south of Kitson Island on October 11, 2016.

Section 5.6 - Key Findings

1) A large region proximal to the mouth of the Skeena River estuary (Flora, Agnew, and Horsey Banks) provides year-round habitat for Dungeness crab for foraging, mating, wintering, and egg-brooding.

2) Over 40% of tagged females were found in a dense ovigerous female aggregation proximal to Kitson Island.

3) Females travelled great distances (over 5 km at rates of 1 km per day) to reach egg brooding habitat, illustrating the importance of this region for Dungeness crab.

4) Crabs had diel habitat migrations, moving onto Flora Bank more often at night compared to daytime hours.

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6. GAPS IN KNOWLEDGE

This report was initiated to clearly synthesize research findings and outline historic studies related to the Skeena River estuary in order to distribute knowledge within the broader community and provide reference material to support future scientific studies. A variety of partnership driven research programs have been conducted between 2013–2018 with a major focus on how juvenile salmon use the Skeena River estuary during the transition from freshwater to saltwater. The results of the research conducted by the NCJSMP over the last five years is summarized in this report and addresses a diverse array of questions, however, many knowledge gaps surrounding juvenile salmon use of the Skeena River estuary remain.

In 2015, Pickard et al. performed a knowledge gap analysis on the Skeena River estuary and classified gaps on a variety of factors influencing key juvenile salmon habitat and condition in the estuary (water quality, habitat and lower food-web, salmon population and wild salmon impact). Next, research activities were organized according to priority based on size and importance of the knowledge gaps to the understanding of juvenile salmon use of the Skeena River estuary. Pickard et. al (2015) presented five high priority indicators for prioritization of research and monitoring activities of juvenile salmon in the Skeena River estuary: smolt density and residence time (distribution and abundance), smolt growth (growth and condition), eelgrass extent, and zooplankton density and diversity. In this section, we discuss how we addressed many knowledge gaps presented by Pickard et al. (2015) within these five indicator categories and suggest additional topics for further research.

1) Smolt density and residence time (distribution and abundance of juvenile salmon).

Investigating smolt density and residence time was a substantial focus of our 2013–2018 research program, which included extensive sampling juvenile salmon by beach, trawl, and purse seine from April to August (Section 3.0). We believe our results have created a suitable baseline understanding of temporal patterns of all species of salmon and spatial distribution patterns around the Flora Bank and Kitson Island region (Figure 6-1, Section 3.1). We also created residence timelines for Chinook, coho, sockeye, and pink salmon in the Skeena River estuary to provide an understanding of how long each species resides within the estuary environment (Figure 6-1, Section 3.2). As there were difficulties associated with using stable isotope clock analysis for the chum salmon collected, residence time for this species remains unknown. Given that chum salmon populations are poorly understood and of conservation concern (Gottesfeld & Rabnett 2008), further research on the residence time for chum salmon should be conducted.

The majority of the information presented in the report occurred around the Flora Bank and Kitson Island region of the estuary. Although this region only represents a portion of the Skeena River estuary, local traditional knowledge, historical studies, and research from our collaboration indicated that it contains high-value habitat for juvenile salmon and was used extensively by all species. During the timeline of much of the research presented in this report, this region was the proposed location of 8 liquid natural gas terminals and several other industrial projects, such as the Fairview Terminal Port Expansion (currently being constructed). Thus, conducting research in this region was determined to be ecologically interesting and important for scientific study, an effective place to capture high volumes of juvenile salmon, and of pressing conservation concern.

Gaps remain in our understanding of juvenile salmon distribution patterns across a greater spatial coverage in the estuary. This is especially true for chum salmon which were caught in low abundances in our trawl sampling of the entire estuary extent. Conducting further research in northern and southern reaches of the estuary extent (using purse and beach seine) will enhance our understanding of the communities and processes of the Skeena River estuary. In addition, the tidal extent of the lower Skeena River is understudied (upper Skeena River estuary), and it is unclear how juvenile salmon may be using this habitat to rear during estuary migration in the winter, spring, summer, and fall.

2) Growth and condition of juvenile salmon

We investigated growth of salmon using stable isotope analysis (Section 3.2) and visualized size distributions over time in the estuary (Section 3.3). Using stable isotopes, we identified residence time and growth rate estimates for juvenile Chinook, pink, sockeye, and coho salmon for the regions around Flora Bank and Kinahans Island. However, complications with chum sampling resulted in uncertainty with estimates and remains an important topic for future research. We have found species of juvenile salmon utilizing the estuary environment at the end of our sampling period in July, specifically juvenile Chinook and chum salmon, which are known to reside in estuaries for the longest. Extending sampling past July could provide additional information on salmon that may be using the estuary for longer periods. In addition, sampling fish across a greater extent of the estuary and comparing growth rates and genetic information would improve our understanding of estuarine residence.

Growth rates used in this report should be applied with caution because of assumptions associated with these methods which do not account for several important factors. Rates were derived from changes in populations over time without accounting for individual differences, such as estuary arrival date and initial size. Other methods mentioned in Pickard et al. (2015), including otolith structure and chemistry, scales and bio-chemical investigations of growth factors (blood), and/or RNA:DNA ratios (muscle tissue), may be useful to understanding growth of individual fish within the estuary.

In this report we provide preliminary results from our study aimed at determining the energetic status of sockeye smolts found in the estuary. This provides one measure of condition of juvenile salmon rearing in the estuary. Other methods such as condition factors (fork length and weight relationships), stomach fullness, and new technologies like Fit Chips can be employed to further explore the baseline of smolt condition during estuarine residence (MacLean et al. 2008; Ferriss et al. 2014). Fit Chips (biomarker panels) can detect variability in short-term and chronic stressors on salmon such as viral disease, thermal stress, inflammation, and immanent mortality (Miller-Saunders & Pavlidis 2017).

3) Extent of eelgrass

We performed one eelgrass mapping study, aimed at understanding the distribution of eelgrass on Flora Bank in relation to proposed development in the area (Section 2.2). This research summarized findings and methods from eelgrass research in the Skeena River estuary and advanced our

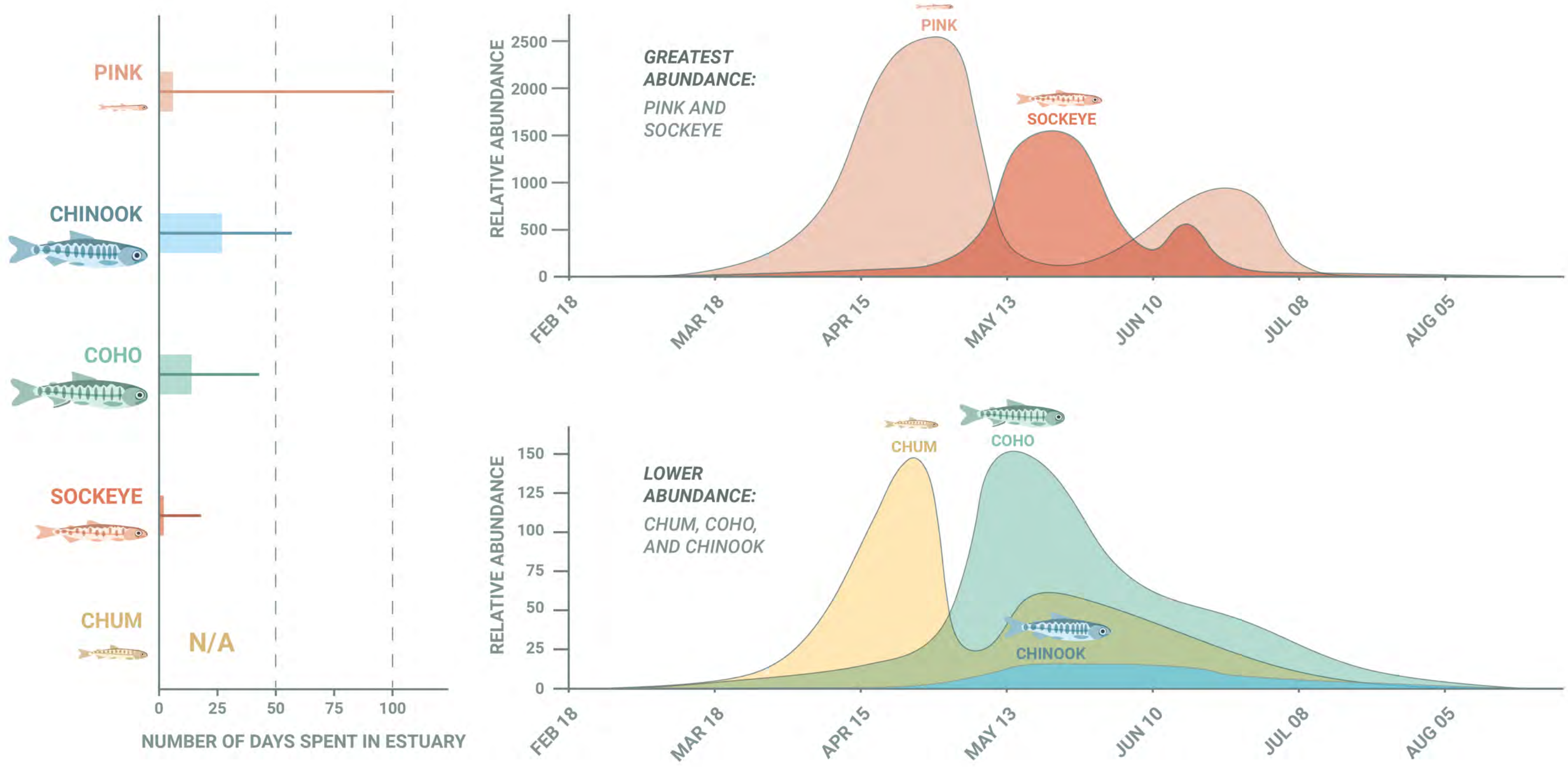

Figure 6-1. Summary of temporal abundance patterns (Section 3.1), estimated residence time (Section 3.2) and size (Section 3.3) of juvenile salmon species utilizing the Skeena River estuary from March to August.

understanding of Flora Bank, the largest eelgrass bed in the estuary. Additional research since our study has been conducted in the estuary by the MaPP/ESI North Coast Integrated Cumulative Effects (NC CE) Program (Appendix A), and mapping of intertidal and subtidal research on the west side of Ridley Island (Hemmera Envirochem Inc. 2018).

While the research conducted on Flora Bank is valuable, the high priority knowledge gaps identified by Pickard et al. (2015) to create a robust and complete baseline on the extent of both intertidal and subtidal eelgrass in the Skeena River estuary remain. Our current understanding of eelgrass habitat in the estuary is based on a combination of projects using several methodologies over many years, with the larger survey efforts having occurred ten years ago. Pickard et al. (2015) suggest that the distribution of eelgrass beds be quantified in a one-time census to fill in the data gaps of regions not recently surveyed. This has not been completed and remains a research priority.

Other habitat types that are important to juvenile salmon species include salt-marsh habitats. Although there is less salt-marsh habitat in the Skeena River estuary compared to estuaries with larger flood deltas like the Fraser River, salt-marsh habitat along the upper Skeena estuary (lower river section) could provide important food and shelter for juvenile salmon. In addition, ocean-type Chinook from certain populations are reported to be found in low abundances in this area of the estuary (Gottesfeld & Rabnett 2008). The north side of the lower Skeena River (upper estuary) was degraded historically as it parallels both a railway and highway. Given the limited knowledge of this region, available habitat, and restoration potential, an in-depth study of this region is recommended. Research investigating juvenile salmon use of salt-marsh habitat in the Skeena estuary should involve mapping and field components that assess food web linkages and use by salmon species and populations from March to November.

4) Density and diversity of key salmon food

We conducted research linking the temporal and spatial distribution of juvenile salmon to available food sources, and zooplankton communities to estuarine environmental factors (Section 4). First, we mapped zooplankton densities spatially and temporally during spring and summer, and conducted modeling to assess if certain environmental factors were correlated with their distributions (Section 4.2). We performed most zooplankton sampling around the mouth of the river, including all channels, and conducted an in-depth study of zooplankton communities within the Flora Bank region. Next, we determined what food juvenile coho and sockeye salmon were consuming in the estuary, and investigated patterns of prey selectivity (Section 4.1). We also performed a study looking at the benthic invertebrate community from sediments collected around the Flora Bank and Kitson Island region of the estuary (Section 4.2).

This research informs zooplankton density and diversity indices in the estuary, one of the high priority knowledge gaps previously identified by Pickard et al. (2015). Future research should focus on understanding food web linkages for juvenile Chinook, pink, and chum salmon, which remain undetermined. It is also unknown whether prey resources are limited for juvenile salmon, especially during years of high smolt numbers. Furthermore, food-web linkages connecting upper trophic levels (marine mammals, birds, and larger piscivorous predators) to juvenile salmon use of estuaries remain unknown. Given that juvenile salmon represent a significant prey pulse into the estuarine and marine environment in the spring, understanding the links to the upper food web will provide a more complete picture of the importance of juvenile salmon migrations to the ecosystems.

In addition, further investigation into the zooplankton communities and diet trends across a wider extent of the estuary (southern and northern regions), including in tidal extents of the lower Skeena River, is a priority. The NCJSMP is currently (2019–2021) addressing this priority by conducting oblique zooplankton tows across two transects, once a month from April until November. On the first transect, sites start at the river mouth and move towards Stephens Island, and for the second sites start close to Port Simpson and move towards Chatham Sound. This study will contribute findings across a coarser spatial and temporal scale than what is described in this report and will provide baseline information on the spatio-temporal dynamics of zooplankton communities at the outer extents of the estuary into Chatham Sound.

5) *Other topics of research priority*

The above section focuses on knowledge gaps surrounding juvenile salmon use of the Skeena River estuary as outlined by Pickard et al. (2015). However, there are many additional species of social, economic, and/or ecological importance that utilize the Skeena River estuary including Pacific herring, surf smelt, Dungeness crab, and eulachon. In Section 5 of this report, we presented research conducted on these focal species in the Skeena River estuary. Given that this research was the first of its kind for this estuary, the knowledge gaps and research potential on the importance of the estuary for these species are vast. Priority questions include:

- Are the populations of Pacific herring utilizing the Skeena River estuary migratory or resident?
- Do larval, juvenile, and adult forage fish species (Pacific herring, surf smelt, and Pacific sand lance) use the Skeena River estuary more than surrounding marine nearshore habitat?
- When and where do Pacific sand lance and surf smelt spawn in the Skeena River estuary?
- How do estuarine fish communities differ in the inner and outer regions of the Skeena River estuary?
- Can larval eulachon densities be used to reflect patterns in adult spawning biomass as a measure of Skeena River eulachon populations?

We note that there are many aspects of this ecosystem that remain poorly understood, ranging from the physical sciences (e.g., movements of sediments and water), applied sciences (e.g., efficacy of restoration opportunities), as well as the integrative systems science of the estuary. One pressing topic is how sea-level rise, and other symptoms of climate change, will impact the physical dynamics of the estuary, its habitats, and the functions it provides for species of importance. Thus, a research priority would be to forecast sea-level rise and predict changes in estuary habitats.

This report summarizes information that has been revealed over years of study and also identifies key uncertainties and unknowns. We hope that it will be a useful resource for potential researchers, be of interest to readers learning about this important and dynamic region, and inform future science-based decision-making.

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7. APPENDIX - Complete list of research conducted in the Skeena River estuary

Table A1. Comprehensive list of all known historic and ongoing research conducted in the Skeena River estuary up to March 2021. Projects are sorted by research categories coinciding with those used in this report: All Categories, Estuary Environment, Fish and Shellfish, Foodweb and Juvenile Salmon and Miscellaneous (Marine Mammal, Vessel Noise, Emissions Data).

Category	Year	Citation	Objective
All Categories	1975	Hoos, L.M. (1975). The Skeena River estuary status of environmental knowledge to 1975 (Special Estuary Series No. 3). Environment Canada, Vancouver, British Columbia.	Summary of studies and knowledge up to 1975 in the estuary. Sections on geology, climate, water quality, estuary hydrology, benthic invertebrates, plankton, fish, eelgrass and other vegetation, and pollution.
All Categories	2015	Pickard, D., Porter, M., Olson, B., Connors, B., Kellock, K., Jones, E. & Connors, K. (2015a). The Skeena River Estuary: Technical Report. Vancouver, BC. Pacific Salmon Foundation.	PSF undertook the following assessment of the health and current condition of the Skeena River estuary: 1. Compiled datasets from multiple sources 2. Identified indicators and pathways of impact 3. Set benchmark values for indicators 4. Described and assessed datasets for key indicators for scientific quality and relevance 5. Identified data gaps and limitations. 6. Summarized existing monitoring efforts and main limitations
All Categories	2015	Pickard, D., Porter, M., Olson, B., Connors, B., Kellock, K., Jones, E. & Connors, K. (2015b). Skeena River Estuary Assessment Supplemental: Data Quality Assessment. Vancouver, BC. Pacific Salmon Foundation.	Authors assessed scientific rigor of existing datasets in the estuary. Detailed information for how, when, and what data were recorded along with possible data limitations.
All Categories	2016	Mazumder, A. (2016). A review and assessment of the project plans and their potential impacts on marine fish and fish habitat in the Skeena estuary.	Independent review of aquatic findings related to the PNW LNG terminal around Lelu Island and Flora Bank.

Category	Year	Citation	Objective
		Prepared for Lax Kw'alaams Band.	
All Categories	2018	Hemmera Envirochem Inc. (2018). Aquatic Effects Assessment: Fairview/Ridley Island Connector Road Project (Prepared for Port of Prince Rupert). Prince Rupert, BC.	Biophysical documentation of habitat on Kaien Island in proximity to Fairview Terminal for 5.5 km of road connecting Fairview Container Terminal and Ridley Island industrial area. Surveys of intertidal, subtidal, and riparian habitat to document fish species (beach seining) and bivalve species. Water quality surveys were also conducted.
All Categories	2018	WWF- Canada. (2018). Skeena Cumulative Effects Assessment: Advancing policy options for the conservation of the Skeena Watershed and Estuary.	Using an expert-based approach including 30 interviews with traditional knowledge holders, academic researchers, and experts from federal agencies, this report provides an analysis of the cumulative effects of individual and multiple physical, biological, and chemical stressors associated with human activities on three key species: eulachon, Chinook salmon, and eelgrass. This study also assessed how these species could be impacted by climate change and future human activities such as mining, liquefied natural gas, and port development.
All Categories	2017-2021	MaPP/ESI North Coast Integrated Cumulative Effects Program (NC CE) Program - Estuary	Development of an interim current condition assessment protocol for estuaries in British Columbia to inform cumulative effects assessment with an initial application to the Skeena Estuary. Work includes the collection of field data in the Skeena estuary on a range of parameters including oceanography, eelgrass, kelp, and contamination. This is paired with desktop assessment on pressure and state indicators to generate spatial maps to inform strategic, tactical and operational level monitoring, assessment and

Category	Year	Citation	Objective
			management discussions and implementation efforts.
All Categories	2017-2021	MaPP/ESI North Coast Integrated Cumulative Effects (NC CE) Program - Food Security & Access to Resources	Development and implementation of an interim cumulative effects assessment protocol for Food Security and Access to Resource to be applied across the NC Plan area, which includes the land and marine territories of the Gitga'at, Gitxaala, Kitselas, Kitsumkalum, Metlakatla, and Haisla Nations. This includes engaging in community based data collection and desktop analysis, generating tools to inform decisions such as mitigating barriers to access and development of tools to support integration of cumulative effects considerations in activities such as the evaluation of project referrals.
All Categories	2017-2021	MaPP/ESI North Coast Integrated Cumulative Effects (NC CE) Program - Salmon	Work is currently focussed on refinement of a Fish and Fish Habitat assessment protocol applicable to the Skeena River Drainage at an individual watershed level. Establishing an implementation plan for this desktop protocol will help to develop a preliminary understanding of the potential of existing freshwater fish habitat data to support further, focussed assessment with the intention to broaden the scope to include estuarine and marine habitats. Recognising the many jurisdictions and active parties currently working on Salmon research and monitoring along the BC Coast, the work aims to generate value-added cumulative effects assessments of the salmon value, enhanced by filling in gaps in data through field-based activities, as needed.
All Categories	2018-2021	Lax Kw'alaams Fisheries (https://www.dfo-mpo.gc.ca/science/environmental-	This project will involve collecting data on key components of the coastal ecosystem within the

Category	Year	Citation	Objective
		environnement/cebp-pdecr/projects/prince-rupert-eng.html)	traditional territory of the Lax Kw'alaams First Nation, including juvenile salmon, clams, zooplankton and eelgrass. The data will provide information on the status of the coastal ecosystem, wildlife, and environmental conditions in the Port of Prince Rupert and surrounding areas.
Estuary Environment	1948	Cameron, W.M. (1948). Fresh Water in Chatham Sound. Fisheries Resources Bd. Pac. Prog. Report, 76, 72–75.	Unknown content.
Estuary Environment	1952	Trites, R.W. (1952). The oceanography of Chatham Sound, British Columbia. (Master of Science).	Detailed analysis of oceanographic survey of Chatham Sound in the spring and summer of 1948. Includes information on discharge, tides, freshwater transport patterns, and coriolis force.
Estuary Environment	1956	Trites, R.W. (1956). The oceanography of Chatham Sound, British Columbia. J. Fish. Board Can., 13, 385–434.	Detailed observations of salinity and temperature in the estuary through time. Includes data on circulation and tides. Sampling was conducted May-September of 1956.
Estuary Environment	1975	Lee Doran Associates Ltd (1975). Prince Rupert Bulk-Loading Facility. Phase II. Vol. 4. Appendix C: Existing aquatic environment. For: Federal-Provincial Joint Committee of Tsimpsean Peninsula Port Development.	Assessment of various locations on the Tsimpsean Peninsula for suitability and impact of the Prince Rupert Bulk Loading Facility including descriptions of the area and known fish use of estuary in 1975.
Estuary Environment	1980	McGreer, E., Delaney, P.W. & Vigers, G.A. (1980). Review of Oceanographic Data Relating to Ocean Dumping in the Prince Rupert Area with Comments on Present and Alternate Dump Sites (No. 645). E.V.S. Consultants Ltd.	Summary of oceanographic data including salinity, temperature, and ocean currents. Discusses estuary according to the following regions: Prince Rupert Harbour, Tuk inlet, Porpoise Harbour, Morse Basin, Chatham Sound. Includes a list of all oceanographic studies in the Skeena River estuary (pg. 22) and a summary of all contaminated sediment sampling according to regions (pg. 47,

Category	Year	Citation	Objective
			PCBs and heavy metals).
Estuary Environment	1983	Pomeroy, W.M. (1983). Timber pulp mill, Porpoise Harbour- an assessment of mill impact on the receiving environment. Environmental Protection Service (Regional Program Report No. 83–9).	Unknown content.
Estuary Environment	1986	Binda, G.G., Day, T.J. & Syvitski, J.P.M. (1986). Terrestrial sediment transport into the marine environment of Canada. Annotative bibliography and data. (Environmental Canada Sediment Survey Section Report No. IWD-HQ-WRB_SS-86-1). Environment Canada.	Analysis of sediment transport load of the Skeena River into the Skeena River estuary.
Estuary Environment	1992	Dwernchuk, L.W. & Bruce, G.S. (1992). Dioxin and furan levels in marine biota near eight coastal mills relative to Tofino/Queen Charlottes/Port Hardy/Victoria and Vancouver Harbours. Hatfield Consultants Ltd. Report prepared for Howe Sounds Pulp and Paper Limited, Western Pulp Limited Partnership, MacMillan Bloedel Limited, Skeena Culluslose and Fletcher Challenge Canada Limited, West Vancouver, B.C.	Unknown content.
Estuary Environment	1993	Stucchi, D.J. (1993). Circulation and Water Property Study of Prince Rupert Harbour, Summer 1992. Can. Tech. Rep. Hydrogr. Ocean Sci., 154, 46p.	Assessment of circulation and water property data in Prince Rupert Harbor to understand the degree of flushing and exchange of harbor waters.
Estuary Environment	1993	Tera Planning Ltd. (1993). Bulk Liquids Terminal South Kaien Island Prince Rupert, BC: Volume III-Environmental Report. Consultant report for Prince Rupert Port Corporation.	Environmental inventory of oceanography, vegetation, and wildlife including assessment of eelgrass transplant project.
Estuary Environment	1994	Hatfield Consultants Limited. (1994). Skeena Cellulose Environmental Effects Monitoring (EEM): Pre-design Reference Document. Skeena Cellulose	Unknown content.

Category	Year	Citation	Objective
		Inc.	
Estuary Environment	1995	Ages, A.B. (1995). The Salinity Intrusion in the Skeena River: Observations of Salinities, Temperatures and Currents 1979, 1984. Can. Tech. Rep. Hydrog. Ocean Sci., 138.	Measured salinity, temperature, and currents from vertical profile stations in the lower Skeena River and approaches to the estuary. Sampled during spring freshet of 1979 and 1984.
Estuary Environment	1995	Remington, D. (1995). Review and Assessment of Water Quality in the Skeena River Watershed British Columbia, 1995 (Canadian Data Report of Fisheries and Aquatic Sciences No. 1003).	Summary of water quality in the Skeena River Watershed including hydrology, water quality, and water use discharge. Information concerning the Lower Skeena River (part of the tidal extent) includes potential water contamination from upriver sources.
Estuary Environment	1995	Yunker, M. & Cretney, W.J. (1995). Chlorinated Dioxin Trend between 1987 and 1993 for Samples of Crab Hepatopancreas from Pulp and Paper Mill and Harbour Sites in British Columbia (Canadian Technical Report of Fisheries and Aquatic Sciences No. 2082). Department of Fisheries and Oceans, Sidney, BC.	Analysis of chlorinated dibenco-p-dioxins and dibenzofurans in samples of crab hepatopancreas from locations in the vicinity of B.C. coastal mills and principal harbours. Includes information from Prince Rupert.
Estuary Environment	1996	Borstad Associates Ltd. (1996). Mapping Intertidal Habitat in Prince Rupert Harbour.	Aerial mapping of Prince Rupert Harbour using Compact Airborne Spectographic Imager (CASI). Ground-truthed 40 transects at Ridley, Kaein, and Digby Islands to create spectral signatures for various habitat classes (eelgrass, algae, etc). Used along with Archipelago (1999) and Forsyth et al. (1998) to create spatial layer of eelgrass extent in the Skeena.
Estuary Environment	1996	Conway, K.W., Bornhold, B.D. & Barrie, J.V. (1996). Surficial geology and sedimentary processes, Skeena River delta, British Columbia. In: Curr. Res. 1996-E Geol. Surv. Can. Natural Resources Canada = Ressources naturelles Canada, Ottawa, pp. 23–	Mapped the superficial geology and seafloor morphology in the Skeena River delta. High resolution seismic data, sonar, and sediment cores taken including radiocarbon dating of wood in cores.

Category	Year	Citation	Objective
		32.	
Estuary Environment	1998	Forsyth, F., Borstad, G., Horniak, W. & Brown, L. (1998). Prince Rupert intertidal habitat inventory project. Unpublished report to the Prince Rupert Port Corporation, the Canadian Department of Fisheries and Oceans, and the City of Prince Rupert.	Surveys of Flora Bank eelgrass using a towed video camera. 97% eelgrass was intertidal and 96% of eelgrass was within or in close proximity to location indicated by Borstad 1996 survey (ground-truthing for CASI technology). Used along with Archipelago (1999) and Borstad (1996) to create spatial layer of eelgrass extent in the Skeena mentioned in PSF report (Pickard et al. 2015b).
Estuary Environment	1999	Archipelago Marine Research. (1999). Prince Rupert Harbour Foreshore Habitat Classification and Proposed Development Study.	Identified valued habitats, performed habitat inventory and foreshore classification, and prepared development criteria and considerations.
Estuary Environment	2000	MacKenzie, W., Remington, D. & Shaw, J. (2000). Estuaries on the North Coast of British Columbia: a Reconnaissance Survey of Selected Sites. Ministry of Environment, Lands and Parks, and Ministry of Forests, Research Branch, Smithers, BC.	Inventory and mapping of estuary classifications on the North Coast including plant community descriptions, additional biological (birds, mammals, and fish) data, and social and development information.
Estuary Environment	2003	Bennett, K. (2003). Haegele Eelgrass Metadata Report: Source Metadata and Digital Data Specification. Geostreams Consulting.	Mapping of vegetation related to herring spawning grounds along BC's coast by aerial photography from 1980. No field verification.
Estuary Environment	2007	Johannessen, D.I., Harris, K.A., Macdonald, J.S. & Ross, P.S. (2007). Marine Environmental Quality in the North Coast and Queen Charlotte Islands, British Columbia Canada: A Review of Contaminant Sources, Types, and Risks (Canadian Technical Report of Fisheries and Aquatic Sciences No. 2717).	Summary of sources of disturbance and contamination on the North Coast.
Estuary Environment	2009	Ocean Ecology. (2009). Flora Bank Eelgrass Survey.	Surveyed Flora Bank eelgrass with a towed video camera. Identified intertidal and subtidal eelgrass

Category	Year	Citation	Objective
		Prepared for WWF-Canada.	composition and assessed changes in the Flora Bank eelgrass bed. Report includes experimental testing of sonar (sidescan technology) to identify eelgrass.
Estuary Environment	2011	Ambach, M. & Casey, J. (2011). Final report: Identification and mapping of fish habitat within and around Prince Rupert Harbour. WWF-Canada.	Ground-truthed shoreline habitat data from aerial photos (Borstad Associates Ltd. 1996; Archipelago Marine Research 1999) at 126 sites. A GIS layer with eelgrass extent was created using this data.
Estuary Environment	2011	Faggetter, B. (2011). Macroinvertebrate Baseline Study for the Canpotex Potash Terminal Project Disposal at Sea Application. Ocean Ecology.	Benthic macroinvertebrates were sampled at two disposal sites using Ponar grab (10 samples). Observations about sediment size and other physical features were recorded.
Estuary Environment	2011	Ocean Ecology. (2011). Lucy Island Eelgrass Study. Prince Rupert, BC.	Mapped the eelgrass around Lucy Island and assessed the productivity and ecology of eelgrass beds on the North Coast. Methods included towed video camera, drop camera, side sonar imagery, and multi-beam and single-beam bathymetry. Assessed the damage caused to the Lucy Islands eelgrass bed by the use of anchors.
Estuary Environment	2011	Stantec. (2011). Canpotex Potash Export Terminal and Ridley Island Road, Rail and Utility Corridor. Sediment Technical Data Report.	Main objective was to characterize the areas that may be affected by the proposed Canpotex Potash Export Terminal on the west and northeast coasts of Ridley Island. Looked at the characteristics of sediment to be dredged in the marine area, including: particle size, moisture content, total organic carbon, total metals, PAHs, and PCBs. Sampled sediment at eight stations in December 2008 and June 2009, which included depth samples from cores taken in the geotechnical drilling programs. Sampling occurred around Ridley island where proposed construction of wharf, trestle, and causeway are proposed.

Category	Year	Citation	Objective
			Concentrations of metals and organic compounds including nickel, cadmium, mercury, polychlorinated biphenyl (PCB), and PAH, were assessed.
Estuary Environment	2013	Baird, J.B. (2013). Baseline Marine Water Quality Sampling Program Summary- Q1 & Q2. Memorandum prepared by SNC Lavalin for the Prince Rupert Port Authority.	Water quality sampling across a spatial extent of 22% of the estuary.
Estuary Environment	2013	Ocean Ecology. (2013). Chatham Sound Eelgrass Study Final Report. Prepared for WWF-Canada.	Surveyed subtidal eelgrass with benthic video footage at 29 sites in the Skeena River Estuary.
Estuary Environment	2014	Fissel, D.B. (2014). Sediment Transport into the Project Development Area from the Skeena River. ASL Environmental Sciences consulting report to Stantec. May 27, 2014.	Unknown content.
Estuary Environment	2015	McLaren, P., M. (2015). A sediment trend analysis (STA) of Prince Rupert Harbour and its surrounding waters (Prepared for Lax Kw'alaams Band).	Established baseline for sediment (grain-size and composition) and sediment movement patterns (STA: Sediment Trend Analysis) in the Prince Rupert Harbour. Assessed the potential impacts of proposed developments surrounding the Port of Prince Rupert by understanding the movement and behaviour of sediments.
Estuary Environment	2015-2020	Porpoise Harbour Intertidal Reef Survey, Prince Rupert Port Authority and DFO.	Reefs constructed as compensation for RRUC construction. Two transect surveys are conducted on 10 of the 40 reefs in Porpoise Harbour at low, mid and high mark of each reef. Species percent cover or count are collected from each quadrat as well as substrate information. All reefs are also surveyed for % sedimentation at each intertidal level. Beach seines were also included at each site. An annual surveys of a small eelgrass patch that was found near the reefs, from 2017-2019. This research is ongoing with additional data

Category	Year	Citation	Objective
			collection planned for future years.
Estuary Environment	2015-2020	Subtidal reef surveys, Prince Rupert Port Authority	Prince Rupert Port Authority conducts annual dive surveys at seven subtidal reefs located in front of south Kaien Island and within the Port of Prince Rupert. Reefs built as habitat compensation, with goal to increase biodiversity in that area and improve bull kelp canopies. Surveys look at biodiversity at each reef, primarily settlement of seaweeds and invertebrates and use by fishes. Compared against reference locations in proximity to the reefs.
Estuary Environment	2016	Jossul, S. & Robinson, C. (2016). Marine Environmental Water Quality Program Summary. SNC-Lavalin Inc. Prepared for the Port of Prince Rupert, Burnaby, BC.	Collected baseline water quality data at 32 Stations in Prince Rupert Harbour in August. Vertical profiles of temperature, conductivity, DO, pH, salinity, and TDS were collected. Lab analysis assessed total suspended solids (TSS), turbidity, metals (42 metals including mercury), bacteriological (Enterococcus and fecal coliform), PAHs, nutrients (total phosphorus, total dissolved phosphorus, nitrates, ammonia, total nitrogen and silicate), and chl a.
Estuary Environment	2016	McLaren, P. (2016a). The Environmental Implications of Sediment Transport in the Waters of Prince Rupert, British Columbia, Canada: A Comparison Between Kinematic and Dynamic Approaches. J. Coast. Res., 319, 465–482.	Published version of Mclaren (2015) report with emphasis on the Flora Bank region, including a discussion of the relationship between percent mud content in the sediment and eelgrass (<20% mud for eelgrass habitat).
Estuary Environment	2016	McLaren, P. (2016b). Flora Bank Current Survey, June 4-6, 2016. (Report prepared for Gitanyow Hereditary Chiefs Office).	Measured of currents across Flora Bank to calculate tidal velocities. Assessed potential impacts of PNW LNG development on Flora Bank sediment stability.
Estuary Environment	2016	Rinaldi, J. & Robinson, C. (2016). Marine Environmental Water Quality Program Q3. SNC-	Report on fourth year of collecting 19 water quality parameters at 29 stations throughout the

Category	Year	Citation	Objective
		Lavalin Inc. Prepared for the Port of Prince Rupert, Burnaby, BC.	Prince Rupert Port Region.
Estuary Environment	2017	Warren, K. (2017), 2016 Eelgrass Survey of Flora Bank, Prepared for the Lax Kw'alaams Band and Fish Habitat Restoration Initiative, Skeena Fisheries Commission, Kispiox BC.	Conducted baseline sampling of the Flora Bank eelgrass bed for long-term monitoring of eelgrass. Data included eelgrass bed delineation, shoot density, and percent cover of eelgrass.
Estuary Environment	2017-2020	Intertidal biodiversity surveys on Kaien and Ridley Island, Prince Rupert Port Authority	Nine intertidal areas on Kaien Island (7) and Ridley Island (2) are surveyed based on an alternating sampling plan (i.e. once every two years). Surveys are done using the transect-quadrat method, with one transect per location set from high-low intertidal. Species percent cover or count are collected from each quadrat as well as substrate information. The program started in 2017 and surveys take place every August low tide.
Estuary Environment	2018	Gerwing, T.G., Cox, K., Gerwing, A.M.A., Carr-Harris, C.N., Dudas, S.E. & Juanes, F. (2018). Depth to the apparent redox potential discontinuity (aRPD) as a parameter of interest in marine benthic habitat quality models. International Journal of Sediment Research, 33, 149–156.	Assessment of marine benthic habitat quality at the Cassiar Cannery intertidal mudflats in the Skeena River estuary to evaluate apparent redox potential discontinuity (aRPD).
Estuary Environment	2018	Lin, Y. & Fissel, D.B. (2018). The Ocean Circulation of Chatham Sound, British Columbia, Canada: Results from Numerical Modelling Studies Using Historical Datasets. Atmosphere-Ocean, 56, 129–151.	Advanced the understanding of water circulation within Chatham Sound using extensive forcing datasets and high-resolution three-dimensional numerical modelling to investigate tidal and wind-driven currents.
Estuary Environment	2018	Shaw, J., Conway, K.W., Wu, Y. & Kung, R. (2018). Distribution of hexactinellid sponge reefs in the Chatham Sound region, British Columbia (No. Current Research 2018-1). Geological Survey of Canada.	Systematic mapping of Chatham Sound, including 3km of the Skeena River delta, identified that hexactinellid sponge reefs are a significant component of the seafloor mosaic.

Category	Year	Citation	Objective
Estuary Environment	2020	Wild, A.L. (2020). Morphodynamics of a Bedrock Confined Estuary and Delta: The Skeena River Estuary (Master of Science). University of Victoria.	Descriptions of sub-environments, stratification, and sediment accumulation within the Skeena River estuary. Information on sediment cores, grain size, current profiles, and conductivity-temperature-depth measurements.
Estuary Environment	2016 - current	Oceans Network Canada (2016-onwards), Environmental Data from Chatham Sound, https://www.oceannetworks.ca/, https://data.oceannetworks.ca/home, https://catalogue.cioospacific.ca/	Since 2016, Ocean Network Canada has collected publically accessible environmental data from fixed stations in Chatham Sound. CODAR (monitoring surface currents) and meteorological stations are located on Digby and Ridley Island to collect the following: absolute air pressure, absolute humidity, air density, air temperature, dew point, global radar, magnetic compass heading, mixing ratio, relative air pressure, relative humidity, specific enthalpy, wet bulb temperature, wind chill temperature, wind direction, and wind speed. Underwater stations on Digby and Kaein Islands (Atlin Terminal) collect information with: CTD (conductivity, density, practical salinity, pressure, sound speed, temperature, turbidity), Fluorometer (chl-a), and an Oxygen Sensor (oxygen uncorrected, oxygen corrected).
Estuary Environment	2020	Canadian Coast Guard and Ocean Network Canada, Vessel traffic data, timeframe unknown. https://www.oceannetworks.ca/	Ocean Networks Canada stores real-time data streamed from the Canadian Coast Guard, which contains all of their AIS (Automatic Information System) vessel traffic data from across the country. Data are still owned by CCG but can be accessed on a case by case basis upon request from Ocean Networks Canada.
Estuary Environment	2018-2021	Gitxaala Nation (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/projects/prince-	This project will involve collecting data to characterize the state of the coastal ecosystem, with a focus on estuarine health, in the Skeena

Category	Year	Citation	Objective
		rupert-eng.html)	River Estuary (within the traditional territory of the Gitxaala Nation), which includes the Port of Prince Rupert and surrounding areas. The Gitxaala Nation will undertake systematic data collection, focusing on sediment, water, and tissue sampling, to better understand the health of the coastal environment.
Estuary Environment	2013-2021	Prince Rupert Port Authority Marine Environmental Water Quality (MEWQ) (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/projects/prince-rupert-eng.html)	The PRPA MEWQ program is a collaborative initiative between the Prince Rupert Port Authority, port partners, First Nations and government agencies. The MEWQ program was developed to assess the impacts of port-related activities on marine water quality (e.g. heavy metal and organic contaminants, bacterial levels, oxygen concentrations. Quarterly water samples have been collected since 2013. In 2018, the project scope was expanded through the DFO OPP Coastal Environmental Baseline. This expanded scope includes more frequent field campaigns using new oceanographic equipment that provides more accurate and precise measurements and is collected at higher vertical resolution than in previous PRPA studies.
Estuary Environment	2019-2021	Skeena Fisheries Commission (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/projects/prince-rupert-eng.html)	An inventory of dissolved organic matter in the nearshore coastal waters of the Skeena River estuary and Prince Rupert region, including measures of dissolved organic carbon (DOC) and dissolved organic matter (DOM) (i.e., concentration and composition).
Estuary Environment	2019-2021	Coastal and Ocean Resources; funded through Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	ShoreZone imaging and mapping of the Prince Rupert Port Area done through a contract with Coastal and Ocean Resources. Includes complete coastal imaging, aerial surveys, ground truth

Category	Year	Citation	Objective
			stations, and habitat mapping and classification of shoreline using the standardized coastal imaging and habitat classification ShoreZone protocol.
Estuary Environment	2019-2021	Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	Describing biogeochemical cycles in the Port of Prince Rupert and Chatham Sound through sampling aboard the PRPA vessel for DIC, O2, Nuts, del18O, major ions, salinity, and sample analysis for DOC.
Estuary Environment	2019-2021	Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	Mapping of the historical distribution of kelp forests along the British Columbia coast, including Port of Prince Rupert, based on historical British Admiralty charts which date back to the late 1800s.
Estuary Environment	2019-2021	Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	Quantify nutrient fluxes and budget in the Prince Rupert and Chatham Sound areas, including contributions by river inputs (Skeena, etc.), and removal through sedimentation.
Estuary Environment	2019-2021	Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	Benthic habitat mapping and multi-species dive surveys conducted in Prince Rupert in September of 2019.
Estuary Environment, Fish and Shellfish	2018-2021	Metlakatla Stewardship Society (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/projects/prince-rupert-eng.html)	This project consists of a baseline study of shellfish, nearshore habitat, and marine contaminants in Metlakatla traditional territory, including the Port of Prince Rupert and surrounding areas. The study focuses on data collection and characterization of various habitats and species (including shellfish), as well as the identification and measurement of contaminants in water, sediment, and biota through the PollutionTracker program. These data may be used to support a framework on the cumulative effects of marine shipping, other environmental

Category	Year	Citation	Objective
			assessments, and evidence-based decision-making.
Estuary Environment, Fish and Shellfish	2019-2021	Pollution Tracker; funded through Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	Conducting contaminant and microplastic analysis through the collection of sediment, and mussel, crab and clam samples at five sites around Prince Rupert Harbour and the surrounding area (Lax Kw'alaams village, Smith Island, Tuck Inlet, Porpoise Harbour and Lucy Islands) in collaboration with Metlakatla First Nation, Lax Kw'alaams Band, and the Prince Rupert Port Authority.
Fish and Shellfish	1970	Haegele, C.W. & Miller, D.C. 1979. Assessment of 1979 Herring Spawnings in Chatman Sound, British Columbia. Fisheries and Oceans Canada. Nanaimo, BC.	Investigated the relationship between eelgrass and herring spawn in Chatham Sound.
Fish and Shellfish	1975	Levings, C.D., Pomeroy, W.M. & Prange, R. (1975). Sampling Locations for Intertidal Biota and Preliminary Observation of Habitat at some British Columbia Estuaries (Manuscript Report Series No. No. 1345). Pacific Environment Institute, West Vancouver, B.C.	Intertidal organisms sampled at six intertidal quadrats to examine the effect of pulp mill effluent on intertidal amphipods (page 13).
Fish and Shellfish	1999	Boutillier, J.A., Bond, J.A., Nguyen, H. & Gueret, D. (1999). Chatham Sound Shrimp Survey and Resulting Management Actions, August to September, 1998. Canadian Manuscript Reports of Fisheries and Aquatic Sciences, Issue 2496, 38 pp.	Investigated fish and invertebrate distribution using trawl sampling in Chatham Sound.
Fish and Shellfish	2001	Stoffels, D. (2001). Eulachon in the North Coast-Background Report. BC Ministry of Sustainable Resource Management- Skeena Region.	Status and life history of eulachon in the Skeena River region.
Fish and Shellfish	2004	Kingzett, B.C., Norgard, T., & White, P. (2004). Tsimshian Shellfish Aquaculture Development	Unknown content.

Category	Year	Citation	Objective
		Project: 2004 Interim Site Monitoring Report. For Ecotrust Canada on behalf of: Tsimshian Stewardship Committee. 59pp.	
Fish and Shellfish	2004	Norgard, T., Kingzett, B.C., & White, P. (2004). Turning Point Initiative Shellfish Aquaculture Development Project: 2004 Interim Site Monitoring Report. For Coastal First Nations Turning Point Initiative. 117pp.	Unknown content.
Fish and Shellfish	2005	White, P. 2005. Metlakatla First Nation Territory Crab (*Cancer magister* et al.) Survey 2005: Duncan Bay, Metlakatla Bay, and Venn Passage. For the Metlakatla First Nation and the Department of Fisheries and Oceans. 85pp.	Crab surveys, unknown content.
Fish and Shellfish	2007	Clarke, A.D., Lewis, A., Telmer, K.H. & Shrimpton, J.M. (2007). Life history and age at maturity of an anadromous smelt, the eulachon *Thaleichthys pacificus* (Richardson). Journal of Fish Biology, 71, 1479–1493.	Determined age at maturity and repeat spawning potential of eulachon populations from the Columbia River to the Copper River, Alaska, including Canadian populations from the Fraser, Kemano, and Skeena Rivers.
Fish and Shellfish	2007	White, P. (2007). Metlakatla First Nation: Duncan Bay Habitat Survey 2006. For the Metlakatla First Nation and the Department of Fisheries and Oceans. 26pp.	Crab surveys, unknown content.
Fish and Shellfish	2008	White, P. (2008). Metlakatla First Nation Territory Crab (*Cancer magister* et al.) Survey 2006-2007 Interim Report. For the Metlakatla First Nation and the Department of Fisheries and Oceans. 29pp.	Crab surveys, unknown content.
Fish and Shellfish	2011	Kelson, J. (2011). Skeena Estuary Study. Funded by Driftwood Foundation.	Sampled abundance of salmon and forage fish and investigated their diets (emphasis on larval fish) in August 2011. Genetic ID for longfin and eulachon larval fish. Objective was to examine what marine

Category	Year	Citation	Objective
			resources are found in the mixing zones of the estuary and to look at the presence and distribution of fish species in the area.
Fish and Shellfish	2014	Fisheries and Oceans Canada (DFO). (2014). Integrated Fisheries Management Plan, Crab by Trap: January 1 to December 31, 2015. Prince Rupert, BC. 149 pp.	Crab surveys in the Prince Rupert area.
Fish and Shellfish	2015-ongoing	Aquatic Invasive Species Settlement Plate Program conducted by DFO, Lax Kw'alaams Fisheries, Metlakatla Fisheries and Prince Rupert Port Authority.	Monitoring the distribution and presence of sessile aquatic invasive species across Chatham Sound using settlement plates during the summer and early fall of 2007 and 2015 - 2020. Settlement plates were placed annually at various locations throughout the area, including several core sites that are always set, and have detected a number of invasive species, primarily tunicates (e.g., violet and golden star tunicates)and bryozoans. Through the Oceans Protection Plan, Coastal Environmental Baseline Program (DFO) core sites and new sites will continue to be surveyed annually
Fish and Shellfish	2014-2020	Surveys monitoring for Eurpean Green Crab, Prince Rupert Port Authority, Coast Mountain College, and DFO	Trapping for adult European Green Crab at 3-4 locations in Prince Rupert harbour during summer/early fall. Traps were set in the intertidal zone for 24 hour periods. Vertical plankton tows also occur in the Prince Rupert region annually at 7-10sites, intended to detect invasive European green crab in their larval stage. The program is currently being developed in conjunction with DFO to set up a more robust monitoring program and involving Metlakatla and Lax Kw'alaams Fisheries training and monitoring of respective

Category	Year	Citation	Objective
			territories. This increased sampling effort is in response to the recent sighting in Haida Gwaii in July 2020.
Fish and Shellfish	2015	Stantec Consulting Ltd. (2015b). Unpublished data for Prince Rupert Gas Transmission Project – 2014, 2015 Marine Fish and Fish Habitat Surveys. Prepared for Pacific Northwest LNG Limited Partnership. Vancouver, BC.	Examined fish distribution and abundance, eelgrass beds, oceanography, ocean currents, marine birds, and marine mammals.
Fish and Shellfish	2017	Carr-Harris C. (2017). Fish diet and selectivity study of Flora Bank and adjacent areas. Prepared for the Lax Kw'alaams Band and Habitat Restoration Initiative, Skeena Fisheries Commission, Kispiox, BC.	Examined the diet of juvenile salmon and forage fish species in the Skeena River estuary. Analysis involved gut content analysis and benthic invertebrate and pelagic zooplankton sampling.
Fish and Shellfish	2017	Carr-Harris, C. (2017). Early life history of eulachon (*Thaleicthys pacificus*) in the Skeena River Estuary. Prepared for the Lax Kw'alaams Band and Skeena Area Marine Research Collaboration, Skeena Fisheries Commission, Kispiox, BC.	Characterized the timing and distribution of larval eulachon in the Skeena River estuary using oblique tows in 2016.
Fish and Shellfish	2017	Doire, J. (2017). 2016 Skeena Estuary Dungeness Crab Telemetry Survey. Prepared for the Lax Kw'alaams Band and Fish Habitat Restoration Initiative (DFO), Skeena Fisheries Commission, Kispiox, BC.	Examined Dungeness crab habitat use and migratory patterns on and around Flora, Agnew, and Horsey Banks using acoustic telemetry. Assessed the potential impacts of the proposed PNW LNG terminal on Dungeness crab in the region.
Fish and Shellfish	2018	Butts, K. (2018). Microplastic and microfibers in Butter clams (*Saxidomus gigantea*) on the Pacific North Coast of Canada.	Assessed the distribution and magnitude of microplastics and microfibers in Butter Clam tissue within 10km of the Port of Prince Rupert.
Fish and Shellfish	2019	Butts, K., Sharpe, C. & Carr-Harris, C. (2019). Early	Characterized the timing and distribution of larval

Category	Year	Citation	Objective
		life history stages of eulachon (*Thaleicthys pacificus*) in the Skeena River Estuary, part II. Amended to include 2017 sampling results. Prepared for the Vancouver Aquarium Marine Research, Lax Kw'alaams Fisheries, Prince Rupert, BC.	eulachon in the Skeena River estuary using oblique tows in 2017.
Fish and Shellfish	2021	Dealy, L.V. and Hodes, V.R. 2021. Monthly distribution and catch trends of Eulachon (*Thaleichthys pacificus*) in Chatham Sound, British Columbia, July 2018 to March 2019. Can. Manuscr. Rep. Fish. Aquat. Sci. 3187: ix + 37 p.	Assessing the non-spawning distribution and migratory patterns of eulachon using monthly bottom trawl surveys in Chatham Sound from July 2018 to March 2019. Data collected on abundance, size, sexual maturity and stomach content.
Fish and Shellfish, Food Web	2018-2021	Kitsumkalum First Nation (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/projects/prince-rupert-eng.html)	Characterizing phytoplankton, zooplankton, kelp beds, and eulachon in the Port of Prince Rupert and the surrounding area. Gathering data on key features within the traditional territory of the Kitsumkalum First Nation to characterize the status of the coastal ecosystem and environmental conditions. It will focus specifically on detecting inter-annual variability in the abundance, composition, and spatial-temporal distribution of Eulachon, plankton, and kelp.
Foodweb, Estuary Environment	2004	Raedemaecker, F.D. (2004). A GIS-approach to compare intertidal diversity and contaminant loading in the marine receiving environment of two pulp mills in British Columbia, Canada.	Using a GIS-approach, compared intertidal diversity and contaminant loading in the marine receiving environment of two pulp mills. Surveyed gradient of effluence exposure and measured species richness, trophic composition, and algal composition and assessed spatial trends related to dioxin/furan contamination.

Category	**Year**	**Citation**	**Objective**
Foodweb, Estuary Environment	2017	Gerwing, T.G., Allen Gerwing, A.M., Macdonald, T., Cox, K., Juanes, F. & Dudas, S.E. (2017). Intertidal soft-sediment community does not respond to disturbance as postulated by the intermediate disturbance hypothesis. Journal of Sea Research, 129, 22–28.	Evaluation of mudflat community biodiversity and species richness at Cassiar Cannery in the Skeena River estuary after experimental disturbance. Testing of the Intermediate Disturbance Hypothesis.
Foodweb, Estuary Environment	2019	Sizmur, T., Campbell, L., Dracott, K., Jones, M., O'Driscoll, N.J. & Gerwing, T. (2019). Relationships between Potentially Toxic Elements in intertidal sediments and their bioaccumulation by benthic invertebrates. PLoS ONE, 14, e0216767.	Collected sediment cores and benthic invertebrates five locations in the Skeena River estuary to understand the relationships between concentrations of Potentially Toxic Elements (PTE) and benthic invertebrate communities.
Foodweb, Estuary Environment	2019	Campbell, L., Sizmur, T., Juanes, F. & Gerwing, T.G. (2019). Passive reclamation of soft-sediment ecosystems on the North Coast of British Columbia, Canada. Journal of Sea Research, 155, 101796.	Surveyed intertidal mudflat surrounding Cassiar Cannery to document nutrient availability (chlorophyll a concentration/organic matter content), sediment variables (particle size, water content, penetrability, woody debris/macroalgae cover, apparent redox potential discontinuity depth), and infaunal community composition. Data was used to determine the success of reclamation from historic degradation.
Foodweb, Estuary Environment	2020	Campbell, L., Dudas, S.E., Juanes, F., Allen Gerwing, A.M. & Gerwing, T.G. (2020). Invertebrate communities, sediment parameters and food availability of intertidal soft-sediment ecosystems on the north coast of British Columbia, Canada. Journal of Natural History, 54, 919–945.	Assessment of benthic invertebrate community, sediment conditions, and food availability of intertidal mudflats in the Skeena River estuary.
Juvenile Salmon	1956	Manzer, J.I. (1956). Distribution and Movement of Young Pacific Salmon During Early Ocean Residence. Fisheries Research Board of Canada Progress Reports. No. 106: 24-28	Investigated juvenile Pacific salmon distribution in Chatham Sound, Dixon Entrance, and Hecate Strait.

Category	Year	Citation	Objective
Juvenile Salmon	1956	Manzer, J.I. (1969). Stomach contents of juvenile Pacific salmon in Chatham Sound and adjacent waters. Journal of the Fisheries Board of Canada, 26, 2219–2223.	Evaluated stomach contents of juvenile Pacific salmon in Chatham Sound from June to August, including pink, chum, coho, and sockeye salmon.
Juvenile Salmon	1969	Manzer, J.I. (1969). Food and Feeding of Juvenile Pacific salmon in Chatham Sound and Adjacent Waters (Manuscript Report Series No. 1020). Biological Station, Nanaimo, BC.	Described the food consumed by juvenile pinks, chums, sockeye, and coho during marine residence in Chatham Sound from June to late August, 1955.
Juvenile Salmon, Fish and Shellfish	1973	Higgins, R.J. & Schouwenburg, W.J. (1973). A biological assessment of fish utilization of the Skeena River estuary, with special reference to port development in Prince Rupert (Fish. & Mar. Ser. Tech. Rep.). Department of the Environment, Vancouver, BC.	Demonstrated fish distribution and utilization in the Skeena River estuary and related it to fish diet and food availability. Study investigated the relationships between the physical and chemical water characteristics associated with fish distribution and utilization.
Juvenile Salmon, Fish and Shellfish	1986	Anderson, E. (1986). Beach Seine Samples from the Skeena River Estuary, 2 May to 20 August 1986. Employment and Immigration Canada.	Examined fish distribution, habitat use, and the diet of salmon in the Skeena River Estuary using beach seine sampling from May to August 1986.
Juvenile Salmon	2007	Krkosek, M., Gottesfeld, A., Proctor, B., Rolston, D., Carr-Harris, C., & Lewis, M. (2007). Effects of host migration, diversity and aquaculture on sea lice threats to Pacific salmon populations. Proc. R. Soc. B (2007) 274, 3141–3149.	A large scale, multi-year (2004-2006) study of sea lice abundance on juvenile salmonids sampled throughout the Skeena estuary and Chatham Sound.
Juvenile Salmon	2008	Gottesfeld, A.S., C. Carr-Harris, Proctor, B. & Rolston, D. (2008). Sockeye Salmon Juveniles in Chatham Sound 2007. Report to Pacific Salmon Forum, July 2008. Skeena Fisheries Commission. Hazleton, BC. 33 pp.	Observed potential interactions between juvenile wild and farmed salmon in the Skeena and Nass estuaries as well as defined the distribution of sockeye salmon smolts in those areas.
Juvenile Salmon	2009	Gottesfeld, A., Proctor, B., Rolston, L., & Carr-Harris, C. (2009). Sea lice, *Lepeophtheirus salmonis*, transfer between wild sympatric adult and juvenile salmon on the north coast of British	Described mechanism of transfer of sea lice between outmigrating juvenile and sympatric returning adult salmonids in Chatham Sound.

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		Columbia, Canada. Journal of Fish Diseases 2009, 32, 45-57	
Juvenile Salmon	2014	Ocean Ecology. (2014). Skeena River Estuary Juvenile Salmon Habitat. Report prepared for Skeena Wild Conservation Trust and Skeena Watershed Conservation Coalition, BC.	Analyzed the habitat in and around the Skeena River estuary in terms of suitability as valuable or critical habitat to juvenile salmonids using Ecosystem Diagnosis and Treatment (EDT) methodology.
Juvenile Salmon	2015	Carr-Harris, C., Gottesfeld, A.S. & Moore, J.W. (2015). Juvenile Salmon Usage of the Skeena River Estuary. PLOS ONE, 10, e0118988.	Examined the habitat utilization of five species of Pacific salmon in the greater Skeena River estuary.
Juvenile Salmon	2015	Moore, J.W., Carr-Harris, C. & Gordon, J. (2015). Salmon science as related to proposed development in the Skeena River estuary. Prepared for the Lax Kw'alaams Band Council.	Research update from North Coast Juvenile Salmon Monitoring Program investigating the ecology of juvenile salmon in the Skeena River estuary. Includes discussion of environmental risks of proposed development to juvenile salmon in the Skeena River estuary.
Juvenile Salmon	2015	Moore, J.W., Carr-Harris, C., Gottesfeld, A.S., MacIntyre, D., Radies, D., Cleveland, M., Barnes, C., Joseph, W., Williams, G., Gordon, J. & Shepert, B. (2015). Selling First Nations down the river. Science, 349, 596–596.	Highlighting the genetic diversity of Skeena River salmon populations using the estuary environment and discussion of management implications, including the appropriate spatial scale for environmental assessment of industrial developments.
Juvenile Salmon	2016	Moore, J., Gordon, J., Carr-Harris, C., Gottesfeld, A., Wilson, S. & Russell, J. (2016). Assessing estuaries as stopover habitats for juvenile Pacific salmon. Mar. Ecol. Prog. Ser., 559, 201–215.	Investigation of residence timing and growth of juvenile salmon in the Skeena River estuary using stable isotopes.
Juvenile Salmon	2018	Carr-Harris, C.N., Moore, J.W., Gottesfeld, A.S., Gordon, J.A., Shepert, W.M., Henry, J.D.J., Russell, H.J., Helin, W.N.B., Doolan, D.J. & Beacham, T.D. (2018). Phenological Diversity of Salmon Smolt Migration Timing within a Large Watershed.	Quantified phenological diversity of out-migration timing among salmon populations within the Skeena watershed and it's implications for climate mis-matches with marine zooplankton. Genetic stock assessment of juvenile Sockeye salmon from

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		Transactions of the American Fisheries Society, 147, 775–790.	the Skeena River estuary, in addition to zooplankton sampling to compare out-migrating timing and zooplankton dynamics during spring blooms.
Juvenile Salmon, Fish and Shellfish	2019	Sharpe, C.E., Carr-Harris, C., Arbeider, M., Wilson, S.M. & Moore, J.W. (2019). Estuary habitat associations for juvenile Pacific salmon and pelagic fish: implications for coastal planning processes. Aquatic Conservation: Marine and Freshwater Ecosystems, 29, 1636–1656.	Analysis of how juvenile salmon species (sockeye, coho, Chinook salmon) and forage fish species (Pacific herring and smelt) utilize the Skeena River estuary in relation to biotic and abiotic factors. Discussion of management implications for environmental impact assessment including loss of eelgrass habitat.
Juvenile Salmon, Fish and Shellfish	2019	Arbeider, M., Sharpe, C., Carr-Harris, C. & Moore, J. (2019). Integrating prey dynamics, diet, and biophysical factors across an estuary seascape for four fish species. Marine Ecology Progress Series, 613, 151–169.	Examined the interaction of fish with zooplankton prey in the estuary and biophysical factors influencing prey dynamics in the Skeena River estuary. Included analysis of diet (sockeye salmon, coho salmon, Pacific herring, and surf smelt), zooplankton communities, and the estuary environment.
Miscellaneous: Vessel Noise	2018	Warner, G. and Hannay, D. 2018. Modelling Cumulative Vessel Noise near the Port of Prince Rupert. Prepared for Port of Prince Rupert by JASCO Applied Sciences (Canada). Victoria, BC.	Vessel traffic noise from ships operating in the Port of Prince Rupert were modelled to help understand the potential effects of increased underwater noise on marine fauna. Noise was modelled for three vessel traffic scenarios: 1) current (2016) traffic, 2) projected (future-case) vessel traffic from terminals currently under construction, and 3) additional projected traffic for terminals that might be built under high-growth conditions.
Miscellaneous: Emissions Data	2019	SNC-Lavalin Inc (2019). Port of Prince Rupert Energy and Emmissions Inventory 2018 Summary Report. Prepared for the Port of Prince Rupert.	Annual energy and emissions inventory is conducted by the Port of Prince Rupert using the Port Emission Inventory Tool. All port terminal activity is included, as well as activity associated

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			with commercial shipping, rail and trucking providers.
Miscellaneous: Marine Mammal	2018-2021	OceanWise Conservation Association (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/projects/prince-rupert-eng.html)	This project involves collecting data for baseline characterization of marine mammals in the coastal waters of the area surrounding the Port of Prince Rupert, through standardized surveys of nearshore cetaceans, seasonal and spatial habitat use by Humpback Whales, and seasonal and daily patterns of habitat use by Harbour Porpoises, as well as public communication.
Miscellaneous: Marine Mammal	2019-2021	Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	PRISMM coastwide marine mammal survey by the Cetacean Research Program including inshore and offshore transects in parts of Hecate Strait, Howe Sound, and Strait of Georgia
Miscellaneous: Marine Mammal	2019-2021	Oceans Protection Plan, Coastal Environmental Baseline Program (DFO)	Sea Lion aerial survey photo analysis to update pinniped population abundance and distribution in the Port of Prince Rupert.
Miscellaneous: Marine Mammal	2021	Dracott, K. and Merchel, J. (2021). North Coast Cetacean Research Initiative, Annual Report 2020 - 2021. Ocean Wise Research Institute.	The North Coast Cetacean Research Initiative (NCCRI) monitors cetacean populations in Chatham Sound through field research and citizen science. This research includeds seven years of documenting Humpack Whales in Chatham Sound and three years of nearshore transects to document cetacean distribution and abundance. In addition, it encorporates cetacean sightings from citizen science programs.
Miscellaneous: Marine Mammal, Fish and Shellfish	2018-2021	Kitselas Band Council (https://www.dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/projects/prince-rupert-eng.html)	Kitselas eulachon-related marine mammal and seabird baseline surveys in the Skeena River estuary. This study will gather data on marine mammals, including Humpback Whales and Harbour Porpoises, and seabirds to further characterize current ecosystem conditions in

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			relation to eulachon life stages, including seasonal migration and spawning. Land-based point-count sighting surveys and vessel-based transect surveys of marine mammals and seabirds will be conducted along the Skeena River Estuary (Port of Prince Rupert surrounding area), within the traditional territory of the Kitselas Nation.

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