



## **Characterizing Biological Communities of Nearshore Habitats Associated with River Mouths**

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## Introduction

Estuarine habitats are an important piece of the nearshore ‘mosaic’ (Nagelkerken et al. 2015) and are regarded as critical nursery and feeding grounds for juvenile fish (Simenstad et al. 1982; Bennett 1989; Blaber et al. 1995) that enhance recruitment to adult populations, many of which are valued by Alaskans (Beck et al. 2001; Magnusson and Hilborn 2003; Dahlgren et al. 2006). Subsistence, recreational and commercial fisheries across the state have been valued at \$4 billion (Alaska Marine Conservation Council 2012). On the Kenai Peninsula, salmon alone underpin robust commercial and sport fisheries valued at \$80 million per year (ADFG 2014; Carson et al. 2009).

Nearshore habitats associated with river outflows are of particular interest because warming temperatures are anticipated to increase flow with accompanying sediment, nutrient, and organic matter inputs to those environments (Hood and Scott 2008; O’Neel et al. 2015), thereby changing the environmental drivers of those critical habitats. Furthermore, the nearshore habitats of lower Cook Inlet, including Kachemak Bay, are places of focused human activity (KBNERR and NOAA 2001). In light of these possible sources of disturbance and change, we proposed to the Kenai Peninsula Fish Habitat Partnership (KPFHP) the need for data-driven information on biological communities at nearshore habitats located at river mouths (herein, ‘nearshore estuarine’), as a useful tool for decision-makers focused on lower Cook Inlet and Kachemak Bay. This report documents results of a two-year study, undertaken with financial support from the KPFHP to respond to this need.

Kachemak Bay’s habitat diversity includes different estuary-nearshore settings, ranging from glacial rivers with river-flow dominated nearshore habitats to protected bar-built estuaries with wave dominated nearshore habitats. Existing databases include physical habitat variables in the intertidal nearshore, ongoing oceanographic and water quality monitoring, intertidal wetland and submerged aquatic vegetation mapping, and focused studies of biological communities, including nearshore fish (Wolf et al. 1983; Markis 2007) and benthic communities (e.g. Trasky et al. 1977; Ballachey et al. 2015). Abundance and distribution monitoring of intertidal communities occur annually for parts of Kachemak Bay (Konar and Iken 2013; Konar et al. 2015), but does not include nearshore estuarine communities. Nearshore fish assemblages have been surveyed along Kachemak Bay’s southern coast (Robards et al. 1999; Abookire et al. 2000; Harney et al. 2008); however, we recognized these surveys to be somewhat limited in terms of information for today’s decision-makers due to the decadal variability in the community (Robards et al. 1999).

Our project goal was to develop understanding of biological communities in nearshore estuarine habitats. To accomplish this, we proposed two broad objectives: (1) to characterize those communities (i.e., taxonomic abundance) and add data to existing nearshore habitat maps, and (2) to communicate conservation outcomes developed through this project with an engaged group of stakeholders and decision-makers.

Concerning the first objective, we report summary data on nearshore estuarine fish communities sampled during the summers of 2018 and 2019. We describe community structure for all sites visited and make available high-resolution aerial imagery for all sites (except Seldovia Harbor

due to flight restrictions). We highlight two sites as case studies for the usefulness of the aerial imagery data as supplementary material. Additionally, we report indices of species association to site groups, interannual comparisons for sites sampled in both 2018 and 2019, and the influence of different habitat structuring factors (i.e., flow and detached vegetation). Fish collection data from this study were contributed to NOAA's Fish Atlas database, which catalogues nearshore fish across Alaska in association to Shorezone's habitat classification (NOAA; Harney et al. 2008). Concerning the second objective (see section header *Conclusions*), we provide an overview of the communication of project findings and of ongoing efforts that were made possible with the support of this project.

## **Methods**

### *Study area and site selection*

The project was conducted in Kachemak Bay, Alaska (59°34'00" N, 151°33'00" W), which is a large estuarine embayment located on the eastern coast of lower Cook Inlet in the northern Gulf of Alaska (Figure 1). Data collection was conducted at six sites along the bay's southern coast: Glacier Spit, Halibut Cove, China Poot, Tutka Bay, Barabara and Seldovia Harbor, and one site on the bay's northern coast: Anchor River. In general, Kachemak Bay exhibits heterogeneous coastal formations with deep fjords in the outer bay, more bight-like formations in the inner bay, exposed sandy beaches on the northern coast, and heterogenous coastal formations on the southern coast. The coastline is interspersed with rivers that vary in freshwater outflow due to watersheds that differ in catchment size, elevation, gradient, and water source (e.g., glacial melt, snow melt, precipitation) (Hood and Scott 2008; O'Neel et al. 2015). Additionally, oceanographic patterns in temperature, salinity, dissolved oxygen, and turbidity generally differ from the outer and inner sections of the bay that are separated by a long spit jutting out from the middle of the northern coast (Holdereid et al. 2019).

The bay's circulation patterns are affected by a large diurnal tide range greater than 8 m (Adams et al. 2007), marine inputs from the Gulf of Alaska entering from the bay's mouth (Burbank 1977), and an accumulation of freshwater in the inner bay and northern coast as circulation generally moves inward then north to upper Cook Inlet (Bentz et al. 2018; Holdereid et al. 2019). The substrate in Kachemak Bay is also heterogeneous, containing steep sand-and-gravel mixed beaches in some areas and low-gradient mudflats in others (Harney et al. 2008). Due to the large tidal range, the water depth and substrate available to inshore fish could vary dramatically within a single tide exchange. Data collection was standardized as much as possible (e.g., samples taken only at low slack, fish were captured at beaches without vegetation).

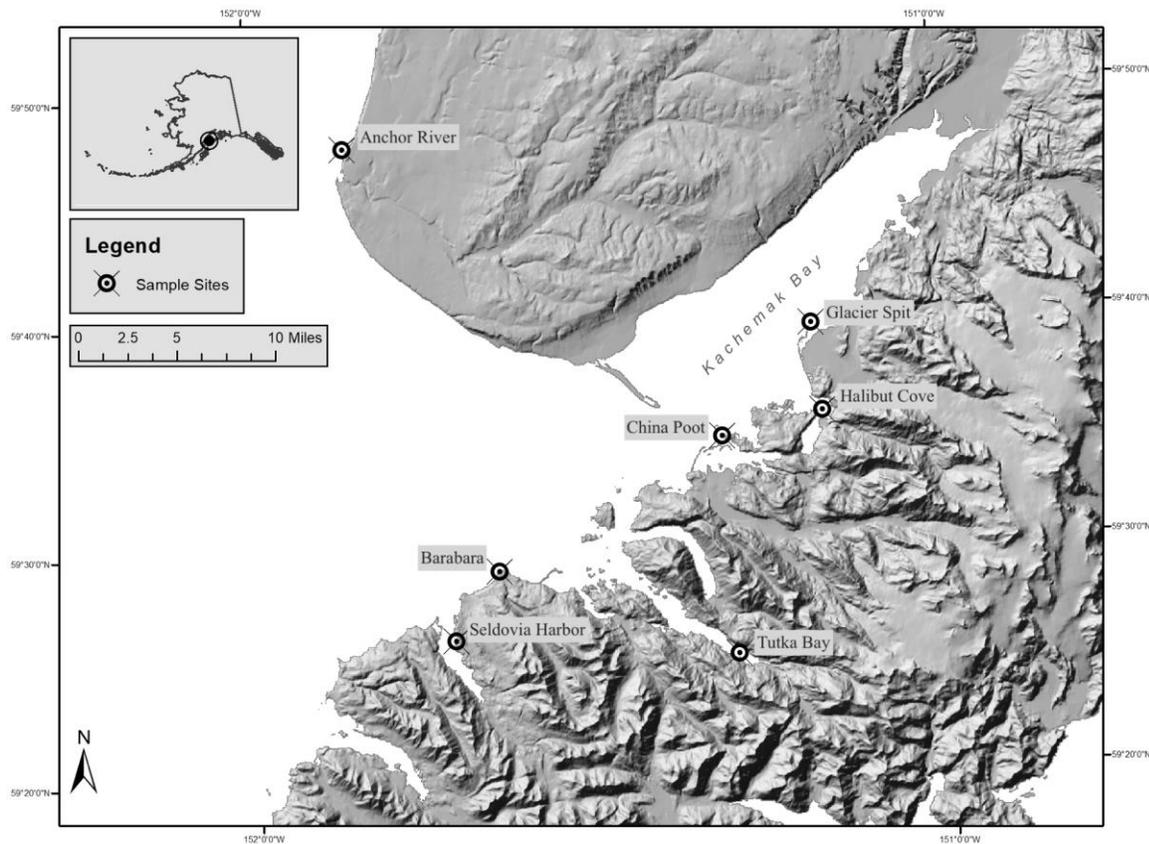


Figure 1. Geographic map of Kachemak Bay and project sites in relation to the state of Alaska, with surrounding area watershed contours shown in grayscale (J. Argueta, 2020).

### *Aerial imagery of physical habitat*

Aerial imagery was collected at each site (except Seldovia Harbor due to flight restrictions) using two platforms, a standard DJI Mavic Air and a DJI Phantom 4 Pro with a Sentra NDVI (Normalized Difference Vegetation Index) upgrade. The Mavic Air utilized a 12 megapixel 1/2.3" RGB sensor equipped with an electronic shutter. The Phantom 4 Pro utilized a 20 megapixel 1" RGB CMOS sensor with both a mechanical and electronic shutter. The Phantom 4 Pro included an additional 1.2 megapixel Near Infrared (NIR) sensor allowing for the creation of NDVI images.

Image acquisition missions were planned and flown using DroneDeploy software. Image processing included the creation of 'orthomosaics' for each site and was accomplished in Agisoft. Orthomosaics derived from the acquired imagery resulted in pixel resolution of approximately 4 cm. NDVI imagery was only captured at one sampling site. Each mission was flown during negative low tides at a height of 400 feet above ground level. Flight time averaged 15 minutes per 100 acres. Missions were generally flown midday on overcast days to avoid unnecessary noise from shadows.

High resolution orthomosaics of each study site allow visual exploration of various substrates present. Other factors such as glacial input, substrate grain size, and vegetation presence, among others, are easily seen in the high-resolution imagery. Further analysis of this recently acquired imagery could result in quantification of vegetation coverage and delineation of substrate types within each study area. Comparisons with previously available aerial imagery for Anchor River and Halibut Cove are provided as case studies in supplementary materials (Figures S1 & S2). Raw imagery data in .tif format are made available online at the KBNERR website or by request to the authors.

### *Fish collections*

Sites were sampled repeatedly for fish composition (i.e., species abundance and size) and water conditions (i.e., temperature, salinity, dissolved oxygen, and turbidity) during consecutive tide cycles during the summer and fall of 2018 and 2019. Fish were collected using a 34 m tapered nylon mesh beach seine. The net's bunt dimensions were eight m-width by five m-depth with three mm mesh, which is similar to gear used in prior studies conducted in the area (Blackburn et al. 1980; Robards et al. 1999; Abookire et al. 2000). Each set of the beach seine was considered a single replicate. A set involved deploying the net by motorized boat in a semi-circle open towards the shore. Thirty-meter ropes were attached to both ends of the net which samplers used to haul simultaneously and purse the seine such that fish ended up in the bunt of the net. During a site visit, up to four replicates would be conducted depending on the time required for processing the catch from previous sets. Combined sampling reach was used to measure the effort of a site visit, where reach was the linear distance from the middle of the net to the shoreline at deployment. Sampling was restricted to within two hours of the low slack tide to consistently sample nearshore (subtidal) habitat and reduce effects related to tidal stage. Collected fish were identified to the lowest taxonomic level (usually species), enumerated, and sized in length (mm) for up to 30 individuals of each species.



*Image: Aerial drone imagery of a seining replicate in action at Barabara in 2018. Samplers deploy the beach seine using a small motorized skiff and haul it into shore using 30 m ropes attached to either end of the net.*

Water conditions were measured immediately after each set at the same location used to determine sampling reach. Sea surface measurements for temperature (°C), salinity, dissolved oxygen (%) and turbidity (FNU) were made using a multi-parameter water quality sonde (YSI, EXO1 Water Quality Sonde) at one-meter depth or half the distance to the sea floor. The water

quality probes were calibrated monthly using manufacturer software (YSI, KOREXO v1.59). In 2018, we characterized local flow conditions for select sites (discussed in subsequent sub-header). In 2019, we characterized the organic physical structures associated with each replicate, where all non-living, organic matter (e.g., detached kelp, wood debris) were collected into permeable bags and wet-weighed.

### *Characterization of site flow*

Six sites were chosen based on exposure to sea surface circulation patterns (Bentz et al. 2018) to test the difference in fish community based on coastal exposure by proxy of flow. Their overall flow patterns were confirmed via field-based measurements. Three sites were located within small coastal embayments (Halibut Cove, Tutka Bay, and Seldovia Harbor), and were expected to have weaker flow conditions due to less coastal exposure. These were compared to three sites with relatively higher exposure (Glacier Spit, China Poot, and Barabara), which were expected to have stronger flow conditions. Sites were dispersed along the bay’s southern coast to account for known oceanographic gradients, some of which had been related to nearshore fish distributions in prior studies (Abookire et al. 2000; Speckman et al. 2005).

Water flow was measured using continuously recording Tilt Current Meter 1 data loggers (Lowell Instruments LLC) placed near the mouth of each site’s outflow channel and submerged to a standard two meters below the mean lower low water tide level. Each data logger contained an accelerometer that was programmed to make a 20 second ‘burst’ data record every 15 minutes from May 23 to June 25, 2018. Post data collection, a single vectorized water speed (cm/s) was calculated for each data record using manufacturer software (Lowell Instruments LLC) and compared among the sites (Figure 2). Only 2018 data were used for site flow comparisons.

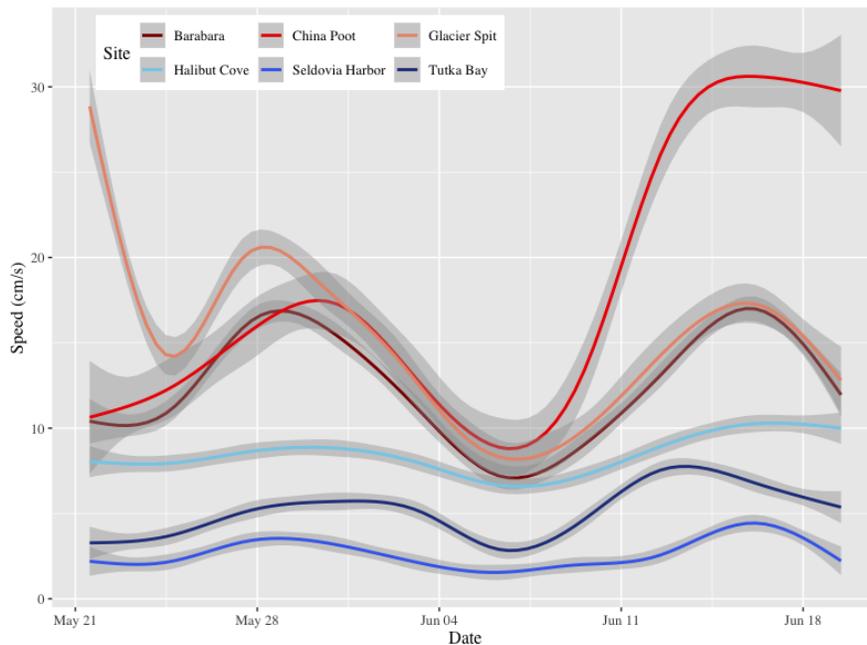


Figure 2. Site flow patterns were smoothed (GAM) using water speeds measured during May 21 to June 23, 2018; red hue lines represent high flow sites, blue hue lines represent low flow sites, and gray shading represent the 95% confidence intervals.

Water speed measurements were used to test for site differences in a one-way ANOVA. A significant difference between high flow sites and low flow sites ( $F = 4314.3$ ,  $p < 0.001$ ) provided evidence that community comparisons were valid based on this site grouping (herein, ‘flow group’).

### *Community analyses based on fish abundance*

Differences in fish community were determined using multivariate analyses in community ecology (Clarke 1993). All analyses were performed in the statistical software *R* (R Core Team 2019). Data manipulation and visualizations were conducted using *R*-packages in the *tidyverse* (Wickham et al. 2019). Multivariate tests were conducted with the *R*-package *vegan* (Oksanen et al. 2019). Species association to site groups was examined using the *R*-package *indicspecies* (De Cáceres and Legendre 2009).

Per replicate, species abundance was measured as the number of individuals per linear meter of seining reach (catch-per-unit-effort or CPUE). Community samples consisted of species CPUE averaged across the replicates per site visit. CPUE data were treated with a fourth-root transformation prior to analyses due to highly abundant catches for some species. Associated with each community sample was a set of environmental factors that included the day of the year and site visit averages for temperature, salinity, dissolved oxygen, and turbidity. For 2018, we included flow as a factor. For 2019, we included vegetation as a factor. Turbidity data were treated with a log transformation due to a strong, rightward skew. Environmental variables were standardized to a mean of zero and deviation of one prior to analyses. Community samples missing data for one or more factors were not included in analyses.

Ecological distances among community samples were calculated using Bray-Curtis distance at both the family and species level. Dissimilarities among fish community samples were visualized with a non-metric multidimensional scaling (NMDS) ordination at the family level. Species-level distances were used for all other analyses. The relationship between environmental variability and fish community was interpreted by indirect gradient analysis (BIOENV, Clarke and Ainsworth 1993). The relationship was tested for significance by Mantel randomization, providing a ‘best set’ of explanatory variables based on Spearman rank correlation. Variance in fish community was partitioned among environmental factors and tested for significance with permutation-based multivariate analyses of variance models (PERMANOVA, McArdle and Anderson 2001).

### *Site group comparisons and indicator species*

Site groupings based on flow, inner-vs-outer bay, and year were tested for homogeneity in multivariate dispersion (beta-diversity), determining whether variability differed among groups based on Bray-Curtis dissimilarities (BETADISPER, Anderson et al. 2006). An analysis of similarities was used to test the significance that between-group dissimilarities were greater than within-group dissimilarities (ANOSIM, Clarke 1993). The species most associated with each site group were identified using the indicator value (*IndVal*) approach (Dufrêne and Legendre 1997). This approach is similar to testing for similarity percentages in site group association (SIMPER,

Clarke 1993). However, indicator values (*IndVal*) were calculated directly from species abundances (CPUE) instead of Bray-Curtis dissimilarities. Additionally, the *IndVal* provides a level of interpretation that SIMPER does not in that the reported value is a product of the species positive prediction (i.e., site specificity) and its sensitivity to site group (i.e., site fidelity).

## Results and discussion

### Overview

In large part, the purpose of this project was to provide a benchmark dataset of nearshore estuarine fish communities found in the Cook Inlet region, using the diverse habitats of Kachemak Bay as study sites. Although we performed a number of analyses on the community data collected, we report only the significant trends here. Un-reported findings were either insignificant or can be found elsewhere in primary or gray scientific literature. We make all data publicly available online at the KBNERR website or by request (see authors' contact information).

Sites were unevenly sampled in 2018 and 2019 due to adjusted research requirements between the years. While seven total sites were sampled during the project, only four sites were sampled in both 2018 and 2019 (Table 1). In 2018, we focused on high-frequency sampling at sites that differed in overall flow condition (see also Figure 2), where site visits occurred about once every two weeks. In 2019, we continued site visits for select sites (Anchor River, China Poot, Halibut Cove, Tutka Bay) but reduced visits to once per month. Sampling efforts in 2019 focused on collecting data such that two years of community information were available to test the effects of site location within Kachemak Bay and interannual differences. The influence of year, flow group, and bay location are reported in detail in following sub-headers.

Table 1. Summary of fish data collection and grouping classification for all sites sampled in 2018 and 2019. Totals are reported in bold print.

<i>Site</i>	<i>Location</i>	<i>Flow group</i>	<i>Number of site visits (seine replicates) and total fish collected</i>			
			2018		2019	
Anchor River	Outer bay	<i>ns</i>	3 (9)	626	6 (17)	1,093
Barabara	Outer bay	High	8 (27)	2,959	<i>ns</i>	<i>ns</i>
China Poot	Inner bay	High	6 (17)	3,792	5 (16)	4,146
Glacier Spit	Inner bay	High	6 (18)	1,807	<i>ns</i>	<i>ns</i>
Halibut Cove	Inner bay	Low	5 (16)	19,636	5 (18)	20,595
Seldovia Harbor	Outer bay	Low	6 (13)	4,558	<i>ns</i>	<i>ns</i>
Tutka Bay	Outer bay	Low	7 (22)	1,314	5 (15)	295
<b>Totals</b>			<b>41 (122)</b>	<b>34,692</b>	<b>21 (66)</b>	<b>26,129</b>

*ns* = not sampled

Raw fish abundances varied quite drastically according to the site. Tutka Bay in 2019 had the least abundant fish abundances compared to the other sites. Whereas, Halibut Cove had much higher abundances compared to other sites in both 2018 and 2019. These abundances were primarily due to catches of large schools (usually >1,000 individuals) of juvenile Pacific herring (*Clupea pallasii*, herein ‘herring’). Prior to the collapse of herring populations in the 1990’s, Halibut Cove was a documented spawning area and the location of a local canning factory (Rounsefell 1930). The large numbers of juvenile herring found at Halibut Cove during this project suggest that this area still supports local populations. A summary of abundances (CPUE) for all species captured per site is provided (Table 2).



*Images: Juvenile herring were abundant at Halibut Cove (left); Oftentimes, predators would be documented consuming herring as in this great sculpin caught at China Poot (right).*

The majority of fish collected were likely juveniles based on lengths at maturity as reported elsewhere (e.g., FishBase.org); however, since we could not be certain that this information was applicable to fish collected in this study, life stages were not assigned to most species except for salmonids whom we had high confidence in juvenile/adult identifications. In general, species sizes increased at the seasonal scale; species average length increased over the course of 2018 and 2019 collection periods respectively. Since our sampling did not include recapture protocols, we cannot conclusively report the size frequency of species populations. However, bay-wide trends in size variability were likely captured. A summary of sizes for each species collected is provided as supplementary material (Table S1).

Table 2. Summary of species abundance as catch-per-unit-effort (CPUE) for all sites during 2018 and 2019 sampling efforts and combined for sites sampled in both years. Fish are listed alphabetically by family then species. Totals are reported in bold print.

Family	Scientific Name	Common Name	2018 only			2018 – 2019				Total
			Barabara	Glacier Spit	Seldovia Harbor	Anchor River	China Poot	Halibut Cove	Tutka	
<i>Agonidae</i>	<i>Pallasina barbata</i>	Tube-nose Poacher	1.4	--	--	5.4	--	--	--	<b>6.8</b>
	<i>Podothecus accipenserinus</i>	Sturgeon Poacher	1.3	0.5	--	--	--	0.4	0.5	<b>2.8</b>
<i>Ammodytidae</i>	<i>Ammodytes hexapterus</i>	Pacific Sand Lance	6.6	10.3	--	0.4	24.0	12.2	0.5	<b>54.1</b>
<i>Clupeidae</i>	<i>Clupea pallasii</i>	Pacific Herring	2.4	8.6	--	1.0	14.7	25.6	1.2	<b>53.4</b>
<i>Cottidae</i>	<i>Artedius fenestralis</i>	Padded Sculpin	--	--	0.7	0.7	1.9	--	--	<b>3.3</b>
	<i>Artedius harringtoni</i>	Scalyhead Sculpin	--	0.5	1.5	0.4	7.6	2.9	--	<b>12.9</b>
	<i>Clinocottus acuticeps</i>	Sharpnose Sculpin	--	--	--	1.1	2.0	--	1.4	<b>4.5</b>
	<i>Cottidae spp.</i>	Unidentified Sculpin (J)*	1.9	4.7	1.1	2.5	5.6	2.7	7.2	<b>25.7</b>
	<i>Enophrys bison</i>	Buffalo Sculpin	6.9	2.3	1.5	--	8.9	1.0	3.0	<b>23.5</b>
	<i>Gymnocanthus galeatus</i>	Armorhead Sculpin	--	--	2.3	--	2.7	0.5	0.5	<b>6.0</b>
	<i>Hemilepidotus hemilepidotus</i>	Red Irish Lord	--	0.6	--	--	--	--	0.8	<b>1.4</b>
	<i>Leptocottus armatus</i>	Staghorn Sculpin	5.4	2.3	0.9	13.6	--	9.9	7.6	<b>39.7</b>
	<i>Myoxocephalus polyacanthocephalus</i>	Great Sculpin	7.4	8.0	13.2	1.6	15.7	9.6	8.6	<b>64.0</b>
	<i>Myoxocephalus scorpius</i>	Shorthorn Sculpin	--	--	--	0.6	--	--	1.0	<b>1.6</b>
	<i>Oligocottus maculosus</i>	Tidepool Sculpin	0.8	3.8	1.1	0.5	4.2	1.0	2.0	<b>13.3</b>
	<i>Gadidae</i>	<i>Eleginus gracilis</i>	Saffron Cod	15.8	5.4	19.9	0.6	3.2	7.1	9.4
<i>Gadus chalcogrammus</i>		Walleye Pollock	1.1	1.0	2.1	--	--	--	--	<b>4.2</b>
<i>Gadus macrocephalus</i>		Pacific Cod	7.6	3.4	5.1	0.8	1.4	3.1	1.8	<b>23.1</b>
<i>Microgadus proximus</i>		Pacific Tom Cod	5.6	0.6	3.9	0.9	0.4	4.5	1.3	<b>17.3</b>
<i>Gasterosteidae</i>		<i>Gasterosteus aculeatus</i>	Three Spine Stickleback	--	0.5	1.7	--	0.7	--	0.7
<i>Hemitriptoridae</i>	<i>Blepsias cirrhosus</i>	Silver Spotted Sculpin	1.3	--	--	1.0	0.5	--	--	<b>2.8</b>
<i>Hexagrammodae</i>	<i>Hexagrammos decagrammus</i>	Kelp Greenling	1.3	--	0.6	0.8	--	--	--	<b>2.7</b>
	<i>Hexagrammos lagocephalus</i>	Rock Greenling	0.6	--	--	0.8	0.6	--	--	<b>2.1</b>

	<i>Hexagrammos octogrammus</i>	Masked Greenling	4.2	1.1	--	2.0	--	2.1	--	<b>9.3</b>
	<i>Hexagrammos stelleri</i>	White Spotted Greenling	5.3	1.7	3.6	5.4	3.5	2.6	1.4	<b>23.5</b>
	<i>Ophiodon elongatus</i>	Lingcod	1.3	1.2	0.8	--	0.5	2.2	2.7	<b>8.6</b>
<i>Larval**</i>	--	Larval Fish	0.6	--	0.8	2.8	4.2	2.0	2.8	<b>13.2</b>
<i>Liparidae</i>	<i>Liparis spp.</i>	Snailfish	1.7	0.7	0.8	3.2	4.9	--	--	<b>11.3</b>
<i>Osmeridae</i>	<i>Hypomesus pretiosus</i>	Surf Smelt	--	1.1	--	6.7	0.7	1.9	--	<b>10.4</b>
	<i>Mallotus villosus</i>	Capelin	1.9	4.5	--	1.0	1.0	4.1	0.8	<b>13.3</b>
	<i>Spirinchus thaleichthys</i>	Longfin Smelt	--	1.2	--	1.0	--	0.4	--	<b>2.6</b>
	<i>Thaleichthys pacificus</i>	Eulachon	--	--	--	0.6	--	--	--	<b>0.6</b>
<i>Pholidae</i>	<i>Apodichthys flavidus</i>	Penpoint Gunnel	--	--	--	--	--	--	1.2	<b>1.2</b>
	<i>Pholis laeta</i>	Crescent Gunnel	3.0	3.3	11.7	1.9	--	7.1	8.9	<b>35.9</b>
<i>Pleuronectidae</i>	<i>Lepidopsetta spp.</i>	Rock Sole	15.9	3.0	5.9	5.9	3.0	4.2	21.0	<b>58.9</b>
	<i>Parophrys vetulus</i>	English Sole	5.9	0.5	3.1	1.0	--	1.5	1.5	<b>13.5</b>
	<i>Platichthys stellatus</i>	Starry Flounder	2.2	3.6	--	16.7	--	11.6	6.9	<b>41.0</b>
	<i>Pleuronectidae</i>	Unidentified Flatfish (J)*	3.1	--	0.6	1.3	0.4	1.6	3.2	<b>10.2</b>
	<i>Psettichthys melanostictus</i>	Sand Sole	4.8	--	--	6.8	--	--	--	<b>11.7</b>
<i>Salmonidae</i>	<i>Oncorhynchus gorbuscha</i>	Pink Salmon (J)	1.7	--	2.1	--	2.2	2.7	3.1	<b>11.8</b>
		Pink Salmon (A)	--	0.6	--	--	0.5	--	5.8	<b>6.9</b>
	<i>Oncorhynchus keta</i>	Chum Salmon (J)	--	--	2.5	0.6	0.8	3.5	4.2	<b>11.6</b>
		Chum Salmon (A)	--	--	--	--	--	--	1.5	<b>1.5</b>
	<i>Oncorhynchus kisutch</i>	Coho Salmon (J)	--	--	2.8	3.3	1.4	3.3	3.5	<b>14.3</b>
	<i>Oncorhynchus nerka</i>	Sockeye Salmon (J)	--	0.6	2.6	0.5	1.9	0.9	2.0	<b>8.4</b>
	<i>Oncorhynchus tshawytscha</i>	Chinook Salmon (J)	--	--	0.7	--	--	--	--	<b>0.7</b>
	<i>Salvelinus malma</i>	Dolly Varden (J)	7.6	8.0	1.2	9.9	9.3	11.3	4.3	<b>51.5</b>
		Dolly Varden (A)	--	2.0	--	--	--	--	--	<b>2.0</b>
<i>Stichaeidae</i>	<i>Anoplarchus insignis</i>	Slender Cockscomb	--	--	--	--	--	--	0.4	<b>0.4</b>
	<i>Lumpenus fabricii</i>	Slender Eelblenny	--	--	--	--	--	1.2	0.6	<b>1.8</b>
	<i>Lumpenus sagitta</i>	Snake Prickleback	1.7	0.5	4.2	--	--	11.7	0.9	<b>19.0</b>
<i>Trichodontidae</i>	<i>Trichodon trichodon</i>	Sandfish	--	0.5	--	--	0.5	--	--	<b>1.0</b>

Juvenile (J) and adult (A) life stages were based on size at maturity pulled from literature sources.

\* Sculpin and flatfish < 20 mm in total length were unidentifiable and classified as juvenile.

\*\*Individuals (generally < 20 mm in total length) that were unidentifiable to family were classified as 'Larval'.

A detailed analysis of seasonality in fish community and environmental measurements are not reported in this report as they are discussed elsewhere (Guo 2019; Guo, *submitted for review*). Here, we present an overview of our findings. In general, seasonal patterns were found similar to that reported in prior studies conducted in the area (Robards et al. 1999; Abookire et al. 2000). Generally, species diversity and abundances were highest during mid-summer and decreased during fall months. Another notable trend was seen in the punctuated occurrences of seasonally migrating species, such as spawning runs of Pacific salmon and fall-spawning Pacific sand lance (*Ammodytes hexapterus*, herein ‘sand lance’).

Water quality measurements also exhibited seasonal trends similar to that reported in monitoring efforts that are ongoing in Kachemak Bay (Holdereid et al. 2019). Overall, these water quality measurements accounted for approximately half of the proportion of variance in community that seasonality, flow group, or bay location did in a PERMANOVA test. Although, the combination of water characteristics (temperature, salinity, dissolved oxygen, turbidity) were better ‘explainers’ of fish community differences than any combination including detached vegetation. As a result of data collection in 2018, we had hypothesized that detached vegetation (e.g., floating kelp, marine debris) was potentially associated with larger abundances of fish. However, when we tested this in 2019, detached vegetation alone was not a significant environmental factor when fitted against the overall fish community variability (Mantel’s  $r = -0.041$ ,  $p = 0.679$ ). It is possible that individual species alone could be related to detached vegetation, but those results were not tested in analyses under this project.

#### *Community comparison by year*

Only sites sampled in both 2018 and 2019 were included in analyses of interannual variability in fish community. These sites were Anchor River, Tutka Bay, China Poot, and Halibut Cove (Table 1). The variability in community dispersion (beta-diversity) was not different between years (ANOVA,  $F = 0.216$ ,  $p = 0.645$ ). In other words, the ‘spread’ of community variance in 2018 was not different than in 2019, which is apparent in the NMDS ordination (Figure 3). It should be noted that the stress level of the ordination in two dimensions is higher than preferred (typically,  $<0.2$  stress value is ‘good’); however, we did not find that interpretability increased by comparing more dimensions (e.g., NMDS1 x NMDS3).



Figure 3. NMDS ordination of dissimilarities (Bray-Curtis) in fish community based on species abundance (CPUE), comparing samples from sites visited in both 2018 and 2019. Red points represent 2018 site visits, blue points are 2019 site visits, and the underlying species scores are superimposed in dark gray.

The insignificant difference in beta-diversity between years was not entirely expected because the frequency of site visits was higher in 2018 (roughly bi-weekly) than in 2019 (roughly monthly), and because sampling began later in 2018 (late June) than in 2019 (mid-May). In an analysis of similarity test, we found that there was actually a significant difference between samples from the two years but that the strength of the year-factor was weak (ANOSIM,  $R = 0.124$ ,  $p = 0.004$ ). While these findings suggest that the year-factor is not a strong influence on dissimilarities in fish community, it should not be misinterpreted to say year-to-year shifts in abundances do not structure community. Prior studies have documented interannual variability occurs at the decadal scale and can result in large differences in community (Robards et al. 1999). Our findings were based on a relatively limited sampling effort regarding interannual variability (i.e., only two years of data collection), and we expect that stronger variances in community would be reflected by more years of data collection.

Table 3. Species most associated to community samples grouped by year. The species indicator value (*IndVal*) is a product of the positive predictive value (*Specificity*) and the sensitivity of the species to flow group (*Fidelity*). Only species with significant results are reported (*p-value* < 0.10); number of permutations = 4999.

<i>Common name</i>	<i>Specificity</i>	<i>Fidelity</i>	<i>IndVal</i>	<i>p-value</i>
<b>2018</b>				
Saffron cod	0.933	0.524	0.699	<0.001
Unidentified sculpin* (J)	0.948	0.476	0.672	0.001
Tidepool sculpin	0.804	0.524	0.649	0.005
Pacific cod	0.842	0.429	0.601	0.009
Unidentified flatfish* (J)	0.916	0.333	0.553	0.023
Lingcod	1.000	0.286	0.535	0.020
Pacific tom cod	0.807	0.286	0.480	0.053
<b>2019</b>				
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Juvenile (J) life stages were based on size at maturity pulled from literature sources.

\*Sculpin and flatfish < 20 mm in total length were unidentifiable and thus classified as juvenile.



*Images: Various juvenile gadid species (left) and juvenile lingcod (above) were more abundant in 2018 sampling than 2019.*

To examine which species were driving the difference in community between 2018 and 2019, we used the Indicator Value approach (Dufrêne and Legendre 1997) that measures the species most associated to samples grouped by year. Sampling in 2018 found higher abundances of juvenile sculpin (*Cottidae*) and flatfish (*Pleuronectidae*), cod (*Gadidae*, herein ‘gadid’), and lingcod (*Ophiodon elongatus*); whereas in 2019, there were no significantly associated species (Table 3). Specificity for the species most associated to 2018 was quite high. For example, lingcod was caught in 2018 but not 2019 (*Specificity* = 1.000) at these sites. In 2019, there were generally high specificity values for juvenile salmon species (i.e., sockeye *Oncorhynchus nerka*, pinks *O. gorbuscha*, and chums *O. keta*) but insignificant results suggest that there were too few instances for there to be confidence. Furthermore, fidelity values were low in both years but more so in 2019 (<0.3), meaning that species abundances were more-or-less spread out amongst sites within each year. It should be noted that species identification likely increased from 2018 to 2019, which may help explain the increased abundance of unidentified juvenile sculpin and flatfish found in 2018. An abundance of juvenile cod in 2018 was reported by researchers in other nearshore regions of Alaska, such as Kodiak and Prince William Sound (C. Guo, *pers. comm.*);

although, while other regions reported large abundances of Pacific cod and pollock, our sampling found that saffron cod was by far the most abundant gadid in Kachemak Bay.

### Community comparison by flow group

Only sites sampled in 2018 where flow was measured were included in analyses to compare fish community by flow group. These sites were Barabara, China Poot, Glacier Spit, Halibut Cove, Seldovia Harbor, and Tutka Bay (Table 1). The variability in community dispersion (beta-diversity) was not different when comparing by flow group (ANOVA,  $F = 0.995$ ,  $p = 0.326$ ), so differences in community could be attributed to group centroids rather than dispersion. Separation between flow groups was evident in the two-dimensional NMDS ordination of community samples (Figure 4). All sites shared a general association with sculpin, flatfish, adult and juvenile salmonids (*Salmonidae*), and larval fish driven by common nearshore and estuarine species (e.g., great sculpin *Myoxocephalus polyacanthocephalus*, rock sole *Lepidopsetta spp.*, juvenile dolly varden *Salvelinus malma*). Community samples of low flow sites were associated with large-schooling gadids and with non-gregarious but relatively common fish in gunnels (*Pholidae*), sticklebacks (*Gasterosteidae*), and pricklebacks (*Stichaeidae*). High flow sites were associated with forage fish, which include sand lance, herring, and smelts (*Osmeridae*). High flow sites were also associated relatively rare fish in silver spotted sculpin (*Hemitripteridae*), sandfish (*Trichodontidae*), and tubenoses (*Agonidae*).



Figure 4. NMDS ordination of dissimilarities (Bray-Curtis) in fish community based on family abundance (CPUE) comparing flow group using samples from sites visited in 2018 where flow was measured. Red and blue points represent high and low flow sites, respectively (site colors match Figure 2), and the underlying family scores are superimposed in dark gray.

In an analysis of similarity test, we found a significant difference between flow groups (ANOSIM,  $R = 0.205$ ,  $p < 0.001$ ). The strength of the flow-factor was somewhat weak as seen in the low  $R$ -statistic, yet stronger than that in the year-factor test. This suggests that the flow can significantly influence the dissimilarities in fish community based on site groups. Although we did not test flow as a continuous variable, we suggest that it is actually more appropriate as a site-level factor considering our fish capture method was beach seining. By standardizing fish collection to slack low tide, we essentially removed the immediate effect of flow but maintained a consistent collection protocol. For instance, if we had collected fish exactly when site conditions were high or low flow, then the comparability of our capture efficiencies would become questionable (Rozas and Minello 1997). It stands to reason that lower flow conditions would make fish easier to capture. Passive observational techniques could have been used (e.g., underwater cameras) but the results would likely have suffered from coarse abundance data and unreliable taxonomic identifications. Furthermore, flow as a ‘meta-influencer’ is likely acting on fish on multiple levels, such as through bottom-up forces driving planktonic food availability (Kimmerer et al. 2002). Thus, characterizing a site’s overall flow pattern seems sufficient to determine its effects at the community level.

Table 4. Species most associated to flow group. The species indicator value (*IndVal*) is a product of the positive predictive value (*Specificity*) and the sensitivity of the species to flow group (*Fidelity*). Only species with significant results are reported ( $p$ -value  $< 0.10$ ); number of permutations = 4999.

<i>Common name</i>	<i>Specificity</i>	<i>Fidelity</i>	<i>IndVal</i>	<i>p-value</i>
<b>High flow sites</b>				
Pacific sand lance	0.791	0.842	0.816	<0.001
Buffalo sculpin	0.748	0.842	0.793	<0.001
Masked greenling	0.810	0.421	0.584	0.032
Snailfish	0.819	0.316	0.509	0.079
<b>Low flow sites</b>				
Snake prickleback	0.877	0.588	0.718	0.001
Chum salmon (J)	1.000	0.294	0.542	0.018
Unidentified flatfish (J)*	0.789	0.353	0.528	0.045
Coho salmon (J)	0.906	0.294	0.516	0.027

Juvenile (J) life stages were based on size at maturity pulled from literature sources.

\* Flatfish  $< 20$  mm in total length were unidentifiable and classified as juvenile.

Species indicative of high flow sites included sand lance, buffalo sculpin (*Enophrys bison*), masked greenling (*Hexagrammos octogrammus*), and snailfish (*Liparis spp.*). Species indicative of low flow sites included snake pricklebacks (*Lumpenus sagitta*), juvenile chum salmon, juvenile coho salmon (*O. kisutch*), and juvenile flatfish. All indicator species had relatively high specificity values, meaning that abundances were largely contained within the respective flow groups. High specificity was counteracted with low site fidelity in most species except in the case of sand lance and buffalo sculpin. For instance, juvenile chum salmon were only caught within the low flow group (*Specificity* = 1.000) but mostly occurred at Seldovia Harbor (*Fidelity* = 0.294). By comparison, sand lance were predominantly caught within the high flow group (*Specificity* = 0.791) and occurred in samples collected from most of those sites (*Fidelity* =

0.816). Low fidelity values suggest that other site-level factors were influencing differences in fish community. Scientific literature reported elsewhere by the project team further examines the role of flow in structuring fish community using this data (Guo 2019; Guo et al. *submitted for review*).

### Community comparison by location within Kachemak Bay

Inner and outer bay fish communities were tested for differences using two subsets of the total data: samples collected only in 2018 and those from sites sampled in both 2018 and 2019 (Anchor River, Tutka Bay, China Poot, Halibut Cove). The full dataset was not tested because of the ‘imbalance’ of sampling effort between 2018 and 2019 (see also Table 1). The variability in community dispersion (beta-diversity) was not different between locations in the 2018-only data (ANOVA,  $F = 3.331$ ,  $p = 0.76$ ) nor in the shared-year data (ANOVA,  $F = 3.157$ ,  $p = 0.083$ ). In other words, the ‘spread’ of variability in either dataset is not causing community differences. Inner bay sites appear to separate from outer bay sites when community samples were visualized in an NMDS ordination (Figure 5). It should be noted that the stress level of the ordinations in two dimensions is higher than preferred (typically,  $<0.2$  stress value is ‘good’); however, we did not find that interpretability increased by comparing more dimensions (e.g., NMDS1 x NMDS3).



Figure 5. NMDS ordination of dissimilarities (Bray-Curtis) in fish community based on species abundance (CPUE) comparing locations in the bay using samples from sites visited in 2018 only (A) and those from sites visited in both years (B). Red points represent inner bay site visits, blue points are outer bay site visits, and the underlying species scores are superimposed in dark gray.

Analyses on both subsets of data generally exhibit the same pattern in community differences and indicator species. Although in the shared-year dataset, bay location was a much stronger factor (ANOSIM,  $R = 0.415$ ,  $p < 0.001$ ) compared to the 2018-only dataset (ANOSIM,  $R = 0.193$ ,  $p < 0.001$ ), which is apparent in the relatively large difference in R-statistic. Since similar results were found when testing for indicator species (i.e., the same species were significantly

associated to bay locations using either dataset), we report test results only from the shared-year dataset but discuss findings based on all results.

Table 5. Species most associated to bay location (inner or outer) based on samples from sites visited in both 2018 and 2019 (see also Table 1). The species indicator value (*IndVal*) is a product of the positive predictive value (*Specificity*) and the sensitivity of the species to flow group (*Fidelity*). Only species with significant results are reported (*p-value* < 0.10); number of permutations = 4999.

<i>Common name</i>	<i>Specificity</i>	<i>Fidelity</i>	<i>IndVal</i>	<i>p-value</i>
<b>Inner Bay</b>				
Pacific sand lance	0.952	0.905	0.928	<0.001
Pacific herring	0.912	0.714	0.807	<0.001
Scalyhead sculpin	0.804	0.524	0.702	<0.001
Buffalo sculpin	0.710	0.476	0.581	0.069
Snake prickleback	0.865	0.333	0.537	0.032
Armorhead sculpin	0.856	0.286	0.495	0.083
<b>Outer Bay</b>				
Sand sole	1.000	0.286	0.535	0.020
Tubenose poacher	1.000	0.286	0.535	0.017

Similar to tests based on flow group, we found evidence of additional site-level factors at play. This is most evident in the weak fidelity of species in general, especially those associated with the outer bay: sand sole (*Psettichthys melanostictus*) and tubenose poachers (*Pallasina barbata*). Although, we found sand lance and herring were strong indicators of the inner bay with relatively stats for specificity and fidelity. Prior studies from the area also found that herring among other species were more abundant in the inner bay and attributed this pattern to gradients in water temperature and salinity (Abookire et al. 2000).



Images: Armorhead sculpin (left) and buffalo sculpin (right) both associated with sites located in the outer Kachemak Bay.

Site-level influences were further evident when comparing results of the species associations tests with patterns in the NMDS ordinations. For example, certain sculpin were strongly associated to inner bay sites but had low fidelity scores (see ‘scalyhead sculpin’ and ‘armorhead sculpin’ in Table 5); this was driven by abundances in China Poot samples but not at Halibut Cove resulting in a clear segregation between the two sites (Figure 5B). As another example,

Seldovia Harbor appears tightly grouped compared to other sites from 2018 (Figure 5A), which was driven by high abundances of juvenile gadids and juvenile salmonids. Similarly, Anchor River samples appear tightly grouped in the shared-year dataset (Figure 5B), driven by abundances of starry flounder (*Platichthys stellatus*), staghorn sculpin (*Leptocottus armatus*), and tubenose poacher. The same pattern was not seen in the 2018-only dataset perhaps because of limited samples collected at Anchor River that year. Although, starry flounder and staghorn sculpin significantly contributed to group differences in percent similarity tests (SIMPER), high abundances of those species in general likely confounded those results and were not reflected in the *IndVal* analyses. These results suggest that further studies on these communities might also consider site comparisons between northern and southern bay locations. This concurs with findings by oceanographic monitoring efforts that report environmental drivers (e.g., water quality, plankton communities) exhibit a north-south gradient in Kachemak Bay (Holderied et al. 2019).

## Conclusions

The goal of this project was to develop understanding of biological communities at nearshore estuarine habitats within the KPFHP. Originally, we had proposed to characterize benthic infauna, phytoplankton, and zooplankton communities in addition to fish communities; however, it became clear upon initial data collection that our goal would be best served by focusing on fish and by applying more in-depth data collection (e.g., intensive temporal sampling, additional environmental measurements, site grouping factors). Furthermore, relating fish communities to other aspects of the food web could be better served in partnership with other ongoing research and monitoring efforts in the study area (i.e., Gulf Watch Alaska collaborative oceanographic monitoring program, EPSCoR Fire & Ice initiative by UAF). As a result of this project, KBNERR is better positioned to explore broader food web dynamics with other agencies (discussed in subsequent paragraphs)

Our first objective was to characterize nearshore estuarine fish communities and add data to existing nearshore habitat maps. Summary information for total and per-site species abundance (CPUE) are provided (Table 2), and sizes of all species collected are provided as supplementary material (Tables S1). The raw fish collection data were contributed to NOAA's Fish Atlas database, which catalogs nearshore fish across Alaska in association to Shorezone's habitat classification (NOAA; Harney et al. 2008). Upon further investigation into the gathered data, we found interannual and spatial patterns in overall community among the sampled sites.

The effect of year on fish community was significant but weak, suggesting that interannual shifts in community may occur over longer time periods (e.g., 5-year or decadal cycles) or perhaps as response to infrequent, large-scale disturbances (e.g., the *Exxon Valdez* oil spill). Whereas, spatial factors (i.e., site flow conditions and site location within the bay) significantly influenced fish community. However, additional site-level factors are likely at play considering that species were often significantly associated to specific sites. The influence of environmental measurements, including water quality and detached vegetation biomass, were weakly significant factors in shaping fish community compared to spatial effects. Although these environmental measurements can influence community, these are likely species-specific relationships since they do not appear to have a strong effect at the community level.

The KBNERR has conducted a number of research projects on estuarine fish communities (Hoem-neher et al. 2013a; Hoem-neher et al. 2013b; Walker et al. 2013; Walker and Pierce 2016; Walker and Pierce 2017), particularly concerning intertidal estuarine habitat use by juvenile salmonids, which became the impetus for this project proposed to the KPFHP. Those prior efforts demonstrated a varied use of habitat types (e.g., salt marsh channels, distributary channels, tidal guts) by relatively few estuarine resident species in addition to juvenile salmonids, such as staghorn sculpin, starry flounder, and three spine sticklebacks. In examining the nearshore habitats beyond the river mouth estuary, we have a better understanding of the estuarine habitat use by those species as well as a benchmark for relative abundances in downstream habitats (nearshore) along with other community members.

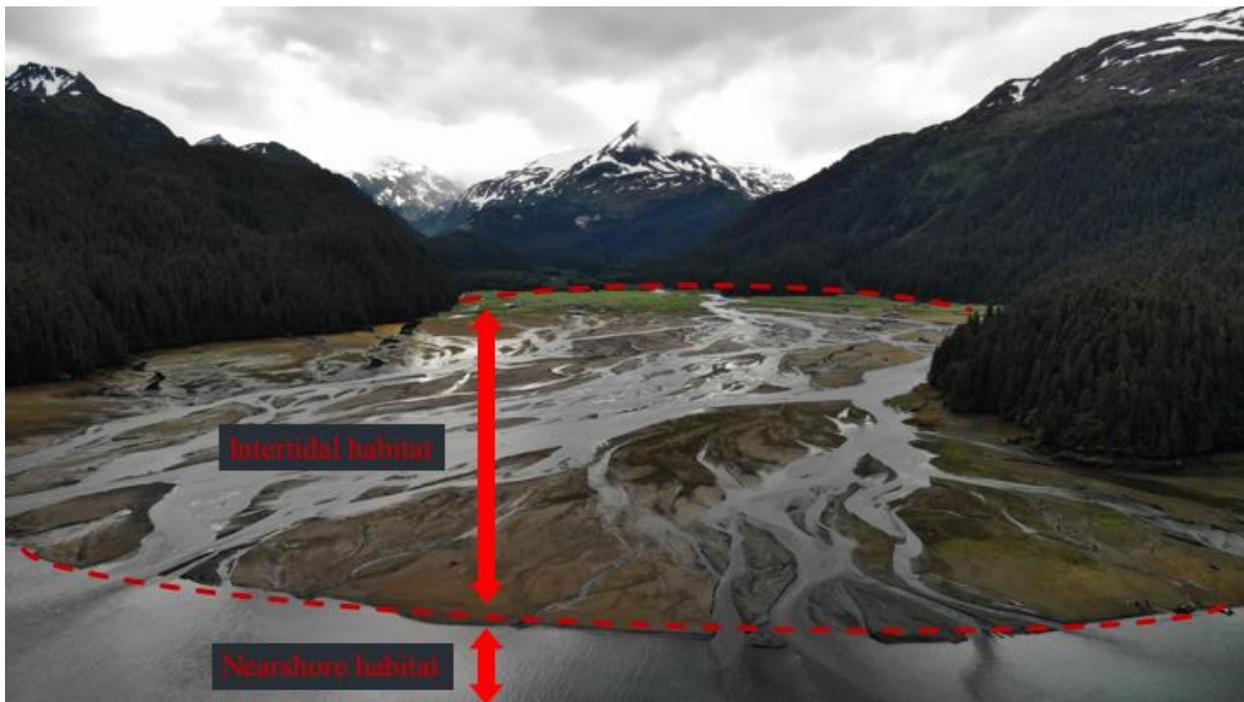


Figure 6. Conceptual figure of the relative location and type of habitat found in the intertidal and nearshore areas of an estuary, where the perspective faces upstream towards the estuary's freshwater source. Background image was taken using an aerial drone at Tutka Bay in 2018.

On a system-wide scale (i.e., throughout Kachemak Bay), we found that nearshore estuarine sites exhibited stronger spatial patterns in fish community compared to no strong patterns in community among intertidal estuarine sites (Walker and Pierce 2017). This suggests that those intertidal habitats contain more features in common than nearshore habitats do, and that only certain estuarine species benefit from those features. The relatively increased species diversity in nearshore habitat compared to the intertidal highlights the value of those upstream areas for juvenile salmon in that individuals likely experience less interspecies competition and potentially less predation due to their tolerance of euryoecious conditions. Still, the importance of the nearshore for juvenile salmonids cannot be discounted because of their inevitable migration route. As such, protected nearshore areas with lower flow conditions may benefit from increased

study and management focus due to the significant associations with juvenile salmon and other juvenile species.



*Images: Juvenile coho salmon utilize habitats in the intertidal (left) and nearshore (right) of the Anchor River estuary. Juvenile salmon sampled in the nearshore often exhibit smolting characteristics, such as a torpedo shape and a ‘silver-ing’ coloration (right).*

Our second objective was to communicate conservation outcomes developed through this project with an engaged group of stakeholders and decision-makers. Project updates and findings were reported to stakeholders during the Lower Cook Inlet and Kachemak Bay Marine Ecosystem Workgroup held bi-annually in the spring and fall each year of this project. These meetings serve as a communication hub for professionals working in the area to develop collaboration and enhance their respective efforts. Additional opportunistic stakeholder meetings were held throughout the project, including sharing with NOAA fisheries scientists located in Juneau, AK, and academic researchers with the University of Alaska located in Fairbanks and Juneau, AK.

Scientific presentations based on our findings were held at multiple venues throughout the project’s duration. These include oral presentations at the College of Fisheries and Ocean Sciences (CFOS) Seminar (2019), the KPFHP Science Conference (2019), and the Coastal Marine Institute’s (CMI’s) Annual Meeting (2019, 2020). Scientific posters were presented at the Kachemak Bay Science Conference (2018) and the Alaska Marine Science Symposium (2018, 2019, 2020).

A KBNERR staff member (C. Guo, co-author on this report) was able to apply the project’s data towards a master’s degree in marine biology with CFOS at the University of Alaska Fairbanks. A graduate student award obtained by the graduate student (\$25K award, CMI BOEM) was leveraged for collection of data in support of this project. A total of eight KBNERR interns were able to gain experience while working on activities associated with this project. Dozens of other secondary education students (undergraduate and graduates) received learning opportunities derived from this project through KBNERR’s education and outreach efforts and partnerships with other local education organizations (e.g., Alaska Center for Conservation Science, Kenai Peninsula College’s Kachemak Bay Campus). Additionally, KBNERR continues to engage with local education groups (e.g., Kenai Peninsula School District; Kenai Peninsula College’s Kachemak Bay Campus) focused on nearshore fish topics related to our findings.

Furthermore, this project has spawned multiple additional research efforts. These opportunities include stewardship-building strategies, such as citizen science and student or intern-led research

projects, as well as pursuit of collaborative funding opportunities for continuing lines of research. Examples of research being pursued include stomach content analysis of predator fish, further habitat associations to fish communities (e.g., beach and vegetated areas adjacent to river mouths), investigations into forage fish as trophic transferrers of marine toxins, and predictive spatial analyses of nearshore estuarine fish communities.

### **Acknowledgements**

This project has filled information gaps identified by the KPFHP and has had far-reaching consequences that provide opportunities for further knowledge-building and stewardship concerning nearshore fish. This would not have been possible without the funding provided by the KPFHP. We wish to acknowledge the Cook Inlet Regional Citizens Advisory Council who provided matching funds in the form of environmental data collection equipment and use of their marine research vessel. We also wish to thank NOAA's Kasitsna Bay Laboratory for helping to facilitate research efforts with materials and personnel. The CFOS at UAF provided invaluable academic guidance and support for graduate student work associated with this project, as well as co-authorship by Dr. Brenda Konar on expected publications (*submitted for review*). Data collection was completed with the help of multiple non-project staff, student-interns, and volunteers with the KBNERR.

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## Supplementary material

16 Centimeter Resolution NIR Imagery



4 Centimeter Resolution RGB Imagery

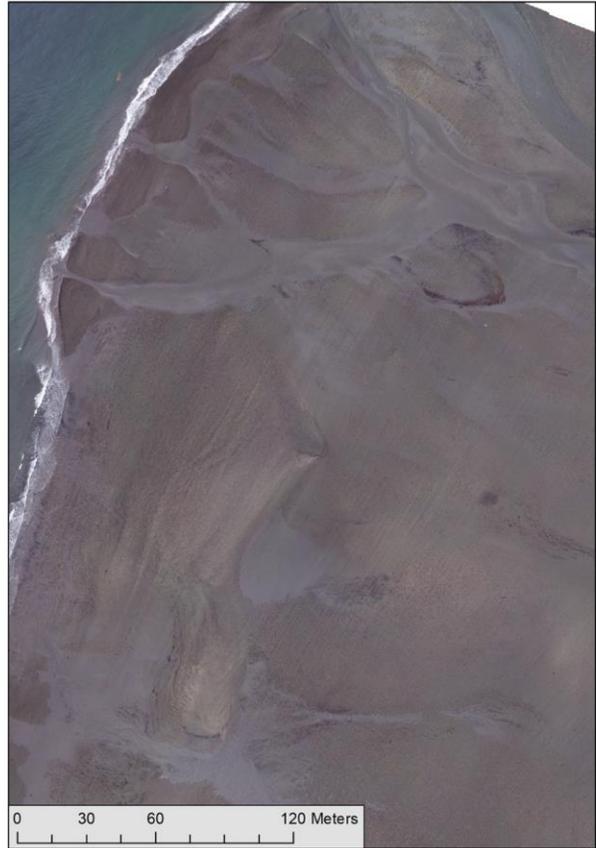
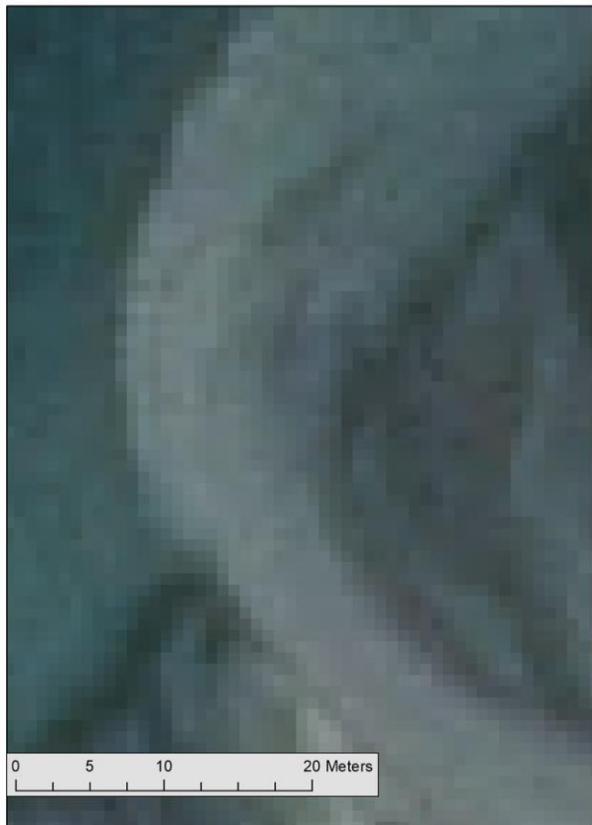


Figure S1. Case study of imagery data collected at the Anchor River nearshore site in 2019 showing side-by-side comparisons for NIR (left) and RGB imagery (right).

1 meter resolution circa 2013



4 centimeter resolution circa 2019



Figure S2. Side-by-side comparison of previously available imagery data circa 2013 (left) and this project's data collected in 2019 (right) at the Halibut Cove nearshore site.

Table S2. Summary of fish sizes as lengths to nearest mm for all fish caught. Metrics include average (Avg), standard deviation (sd), minimum-to-maximum range (Range), measurement type (FL/TL), and total number of measured individuals (N). Fish are listed alphabetically by family then species.

Family	Scientific Name	Common Name	Avg (sd)	Range	Type	N	
<i>Agonidae</i>	<i>Pallasina barbata</i>	Tube-nose Poacher	81 (31)	40 - 138	TL	30	
	<i>Podotheucus accipenserinus</i>	Sturgeon Poacher	40 (11)	28 - 62	TL	8	
<i>Ammodytidae</i>	<i>Ammodytes hexapterus</i>	Pacific Sand Lance	91 (28)	34 - 164	TL	726	
<i>Clupeidae</i>	<i>Clupea pallasii</i>	Pacific Herring	49 (17)	24 - 115	FL	511	
<i>Cottidae</i>	<i>Artedius fenestralis</i>	Padded Sculpin	85 (26)	33 - 115	TL	8	
	<i>Artedius harringtoni</i>	Scalyhead Sculpin	56 (26)	18 - 119	TL	82	
	<i>Clinocottus acuticeps</i>	Sharpnose Sculpin	34 (5)	23 - 45	TL	38	
	<i>Enophrys bison</i>	Buffalo Sculpin	72 (45)	19 - 280	TL	102	
	<i>Gymnocanthus galeatus</i>	Armorhead Sculpin	85 (44)	30 - 145	TL	17	
	<i>Hemilepidotus hemilepidotus</i>	Red Irish Lord	218 (252)	40 - 397	TL	2	
	<i>Leptocottus armatus</i>	Staghorn Sculpin	208 (60)	22 - 352	FL	330	
	<i>Myoxocephalus polyacanthocephalus</i>	Great Sculpin	77 (56)	16 - 420	TL	430	
	<i>Myoxocephalus scorpius</i>	Shorthorn Sculpin	286 (219)	46 - 475	TL	3	
	<i>Oligocottus maculosus</i>	Tidepool Sculpin	29 (11)	12 - 79	TL	89	
	<i>Gadidae</i>	<i>Eleginus gracilis</i>	Saffron Cod	113 (45)	40 - 291	FL	685
		<i>Gadus chalcogrammus</i>	Walleye Pollock	70 (30)	31 - 107	FL	9
		<i>Gadus macrocephalus</i>	Pacific Cod	87 (17)	32 - 150	FL	175
		<i>Microgadus proximus</i>	Pacific Tom Cod	89 (38)	48 - 276	FL	129
<i>Gasterosteidae</i>	<i>Gasterosteus aculeatus</i>	Three Spine Stickleback	29 (18)	15 - 87	TL	18	
<i>Hemitripterae</i>	<i>Blepsias cirrhosus</i>	Silver Spotted Sculpin	88 (37)	45 - 150	TL	7	
<i>Hexagrammodae</i>	<i>Hexagrammos decagrammus</i>	Kelp Greenling	66 (13)	50 - 85	FL	6	
	<i>Hexagrammos lagocephalus</i>	Rock Greenling	62 (7)	55 - 72	FL	9	
	<i>Hexagrammos octogrammus</i>	Masked Greenling	84 (24)	54 - 154	FL	27	
	<i>Hexagrammos stelleri</i>	White Spotted Greenling	82 (23)	40 - 205	FL	104	
	<i>Ophiodon elongatus</i>	Lingcod	96 (15)	79 - 154	FL	33	
<i>Liparidae</i>	<i>Liparis spp.</i>	Snailfish	39 (25)	8 - 138	TL	40	
<i>Osmeridae</i>	<i>Hypomesus pretiosus</i>	Surf Smelt	126 (43)	34 - 200	FL	169	
	<i>Mallotus villosus</i>	Capelin	50 (27)	22 - 182	FL	127	
	<i>Spirinchus thaleichthys</i>	Longfin Smelt	66 (30)	38 - 115	FL	8	
	<i>Thaleichthys pacificus</i>	Eulachon	136 (6)	132 - 140	FL	2	
<i>Pholidae</i>	<i>Apodichthys flavidus</i>	Penpoint Gunnel	79 (3)	75 - 81	TL	3	
	<i>Pholis laeta</i>	Crescent Gunnel	114 (37)	40 - 224	TL	153	
<i>Pleuronectidae</i>	<i>Lepidopsetta spp.</i>	Rock Sole	141 (54)	30 - 375	TL	607	
	<i>Parophrys vetulus</i>	English Sole	80 (34)	32 - 177	TL	71	
	<i>Platichthys stellatus</i>	Starry Flounder	208 (90)	44 - 587	TL	401	
	<i>Psettichthys melanostictus</i>	Sand Sole	100 (38)	30 - 240	TL	104	
<i>Salmonidae</i>	<i>Oncorhynchus gorbuscha</i>	Pink Salmon (J)	76 (73)	31 - 420	FL	106	
		Pink Salmon (A)	501 (48)	425 - 640	FL	102	
	<i>Oncorhynchus keta</i>	Chum Salmon (J)	76 (18)	33 - 134	FL	110	

		Chum Salmon (A)	671 (40)	610 - 720	FL	6
	<i>Oncorhynchus kisutch</i>	Coho Salmon (J)	107 (19)	54 - 182	FL	221
	<i>Oncorhynchus nerka</i>	Sockeye Salmon (J)	68 (24)	22 - 181	FL	99
	<i>Oncorhynchus tshawytscha</i>	Chinook Salmon (J)	76 (28)	47 - 103	FL	3
	<i>Salvelinus malma</i>	Dolly Varden (J)	205 (73)	102 - 420	FL	427
		Dolly Varden (A)	460 (27)	430 - 502	FL	12
<i>Stichaeidae</i>	<i>Anoplarchus insignis</i>	Slender Cockscomb	72 (--)	--	TL	1
	<i>Lumpenus fabricii</i>	Slender Eelblenny	81 (30)	60 - 125	TL	4
	<i>Lumpenus sagitta</i>	Snake Prickleback	161 (68)	54 - 324	TL	215
<i>Trichodontidae</i>	<i>Trichodon trichodon</i>	Sandfish	92 (15)	82 - 103	FL	2

Juvenile (J) and adult (A) life stages were based on size at maturity pulled from literature sources. Lengths were measured using fork length (FL) or total length (TL) depending on the species.