Lower Columbia River Ecosystem Monitoring Program

Annual Report for Year 14

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Lower Columbia River Ecosystem Monitoring Program
Annual Report for Year 15 (October 1, 2018 to September 30, 2019)

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Executive Summary

The Ecosystem Monitoring Program (EMP) is managed by the Lower Columbia Estuary Partnership and is an integrated status and trends program for the lower Columbia River. Under the EMP, researchers collect key information on ecological conditions for a range of habitats throughout the lower river characteristic of those used by migrating juvenile salmon and provide information to aid the recovery of threatened and endangered salmonids. The program inventories the different types of habitats within the lower river, tracks trends in the overall condition of these habitats over time, provides a suite of reference sites for use as end points in regional habitat restoration actions, and places findings from management actions into context with the larger ecosystem. The EMP is implemented through a multi-agency collaboration, focusing sampling efforts on examining temporal trends within a study area that extends from the mouth of the river to Bonneville Dam. The goal of this executive summary is to provide a brief synopsis of the ecological conditions observed in the trend sites in 2019. The full report following this executive summary should be consulted for detailed scientific methods and findings.

In 2019, data were collected on fish and fish prey, habitat, hydrology, food web, abiotic site conditions, and mainstem river conditions at Ilwaco Slough (river kilometer; rkm 6), Welch Island (rkm 53), Whites Island (rkm 72), Campbell Slough (rkm 149), and Franz Lake (rkm 221). Habitat and hydrology data were also collected at Cunningham Lake (rkm 145) in 2019. The trends sampling sites are minimally disturbed, tidally influenced freshwater emergent wetlands with backwater sloughs that represent a subset of the eight hydrogeomorphic reaches across the lower river. In addition to tracking ecological changes in the Lower Columbia River, this year, a collaborative effort has been made to study the effect of varying flow regimes over the monitoring period, of the mainstem on site-specific biotic and abiotic conditions. Based on the size of the freshet between 2010 – 2019, an NMDS plot of differences in river discharge and river temp between years, hydrologic conditions or cumulative discharge of the Mainstem since 2010 were classified into four categories (Table 3). 2019 is classified as a dry year, due to low flows before and after the spring freshet. The spring freshet in 2019 was close to average. The research questions we have attempted to answer with this report is *"What is the influence, if any, of cumulative river discharge on juvenile salmon habitat and site-use pattern?"* We believe that exploring this question provides crucial information to restorative planning in the face of rising water levels and shifting climate patterns.

This report is a collaborative effort by many researchers. Habitat structure research leads from Lower Columbia Estuary Partnership are Dr. Sarah Kidd and Sneha Rao. Water Quality and Food Web dynamics research leads from Oregon Health and Science University are Dr. Joseph A. Needoba and Dr. Tawnya D. Peterson. Salmon Prey and Diet research leads from University of Washington are Dr. Jeff Cordell and Mary Ramirez. Fish community and genetic composition research leads from NOAA – Fisheries are Regan McNatt and Susan A Hinton. Sneha Rao and Dr. Sarah Kidd are the lead report Editors.

Mainstem Conditions of the Columbia River

Mainstem conditions are evaluated through measures of river discharge at Bonneville Dam and at Beaver Army Terminal (river mile 53). In addition, temperature data and other variables are provided through in situ sensor measurements at Camas (river mile 122) and at Beaver Army Terminal (BAT).

River Discharge

Compared to the previous nine years, discharge at Bonneville Dam during the freshet in 2019 can be characterized as dry, on the whole. Discharge was nearly as low as the long-term minimum until mid-March and again after the freshet subsided. The freshet itself was close to average. Thus, the water year could be described as having a lower-than-average baseline flow with an average-sized freshet. The freshet occurred in a series of peaks between April and early June.

The Columbia River accounts for the largest proportion of flow throughout the year; however, during the winter months, flows from the Willamette River increase in relative importance, as do flows from other tributaries. River discharge in the early spring of 2019 composed of a relatively small fraction of flow from the Willamette River and tributaries, which influences water quality parameters in the mainstem, including nutrients, turbidity, and colored dissolved organic matter. Compared to previous years, the proportional flow from the Willamette and other tributaries was very low and characterized by the absence of distinct peaks during the winter.

Water Quality

Water Quality parameters measured at RM-122 and RM-53 include Temperature, Conductivity, Turbidity, Nitrates, Colored Dissolved Organic Matter, Dissolved Oxygen and Chlorophyll a concentration. Temperature is an important variable that influences organismal physiology and particularly the performance and survival of salmonids. We compare the number of days in 2019 where temperatures exceeded thresholds associated with reduced performance or physiological stress with the years 2013, 2014, 2015, and 2016. The most conservative threshold (19 ^{o}C) was exceeded on 80 days in 2019, similar to many of the previous years (2013-2018, with the notable exception of 2015 which was an exceptionally hot and dry year). There were similar numbers of days exceeding 20^{o}C and 21^{o}C compared to previous year; however, the number of days with temperatures exceeding 22^{o}C or 23^{o}C was lower than in any previous year, with only 7 days exceeding 22^{o}C and no days exceeding 23^{o}C.

In 2019, peaks in tributary flow are associated with peaks in colored dissolved organic matter (CDOM), turbidity, and nitrate, underscoring the influence of water source on water quality parameters. Chlorophyll *a* peaked in April-May as well as in June-July, in association with changes in river discharge. Dissolved oxygen saturation exceeded 100% for nearly the entire year, with greater day-to-day variability observed during the summer months. Dissolved nitrate concentrations were lower in 2019 compared to previous years during peaks flows.

Tidal Wetland Habitat Conditions of the Columbia River

Native and non-native Plant Communities

Overall, 2019 total plant cover was relatively stable across Ilwaco Slough, Welch Island, Whites Island, and Franz Lake compared to levels observed in 2018 and 2017. Cunningham Lake total cover has continued to increase between 2017 and 2019, rebounding from the heavy cattle grazing observed in 2017, and Campbell Slough has exhibited variability in total cover levels from 2017 to 2019, which is also attributed to cattle grazing at that site.

Between 2012-2019 the six most common plant species identified throughout the tidal estuary (across the 6 trend sites) in order of overall abundance are *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Carex lyngbyei* (CALY, native), lyngby sedge, *Eleocharis palustris* (ELPA, native), common spikerush, *Sagittaria latifolia* (SALA, native), wapato, *Leersia oryzoides* (LEOR, native), rice cut grass, and *Ludwigia palustris* (LUPA, native), water purslane. While these species are the most common and abundant across all sites over the years, they are not necessarily present at all sites every year.

In 2019, *Phalaris arundinacea* cover levels stayed relatively consistent to those observed in 2018, however, at Cunningham, there was a significant increase in *P. arundinacea* levels from 17 to 38% and Franz Lake also experience a small increase from 8.7 to 13.1%. *P. arundinacea* frequency also increased at Cunningham, and overall *P. arundinacea* frequency dropped at Franz Lake. This shift in *P. arundinacea* levels observed at Cunningham and Franz Lake is likely a product of both low freshet flooding conditions in 2019 and, at Cunningham Lake, a break from grazing pressure. In 2019, data continued to support our findings that annual shifts in *P. arundinacea* cover are strongly correlated with

Columbia River discharge levels and site water levels during the growing season, with lower water levels (and lower discharge levels) favoring *P. arundinacea* growth and observed abundance. These findings indicate that annual flooding conditions within sites and across the river (freshet accumulated discharge) are important mechanisms driving much of the observed annual variability in *P. arundinacea* dominance across the estuary. The long-term trends in the abundance of native species *Carex lyngbyei, Sagittaria latifolia, Polygonum amphibium* have also been found to be strongly (and significantly) linked to annual river discharge conditions. Generally, *C. lyngbeyi* abundance has been found to increase in years of greater freshet and discharge levels, especially in Ilwaco Slough, where salinity levels are reduced during large discharge years, making growing conditions more favorable for *C. lyngbeyi*. *S. latifolia* has been found to have a delayed reaction to freshet and river discharge conditions, with lower discharge years resulting in an increase in *S. latifolia* abundance the following year. Additionally, *P. amphibium* levels at Franz lake have also be found to follow a similar trend to *S. latifolia* with a one-year delayed reaction (increase in abundance) to decreased river discharge conditions. For both species, this might be a result of increased rhizome stores from positive growing conditions (low water levels), providing for more robust growth in the following growing season.

Water Quality

Measurements of water quality parameters were made every 30 minutes at five trends sites (Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough) using sensor packages moored at fixed depths. From these data, daily averages were computed to look for seasonal trends.

Water quality parameters determined at the five fixed trends sites revealed similar patterns to previous years (Sagar et al. 2016, Hansen et al. 2017). Ilwaco stands out for the low dissolved oxygen saturation values, which were below 100% (relative to equilibrium with the atmosphere) throughout the spring and summer. Franz Lake Slough also had low dissolved oxygen saturation levels in the summer, while the other sites had minimal observations of low dissolved oxygen levels in 2019. Temperatures were high in 2019 during the late spring and summer, similar to observations from 2015 and 2016; the number of day-equivalents greater than thresholds of relevance to salmonid growth and survival was higher in 2019 than in 2018 or 2017 in the summer. The high-water temperatures following the spring freshet reflect the fact that aside from high flows occurring during the freshet, river flow was very low in 2019, making summer conditions more like those observed during dry years like 2015.

With the exception of Campbell Slough, pH values were in the target range for good water quality throughout the sampling season in 2019. The short-term fluctuations in pH occasionally fell outside the range for good water quality according to the Washington Department of Ecology's acceptable limits (7–8.5), but the daily averages were acceptable. In contrast, at Campbell Slough, pH values exceeded 8.5 in late June, and remained higher from June through September compared to the March-June period. Levels that exceed 9 lead to a shift in the speciation of ammonium, from its ionic form to the toxic gas, ammonia.

Phytoplankton and Zooplankton

In 2019, total algal biomass, as estimated by concentrations of chlorophyll a, was highest in March, prior to the spring freshet, at Welch Island and Whites Island; in contrast, the highest algal biomass at Campbell Slough and Franz Lake Slough was observed in August, with an exceptional peak in biomass at Franz Lake Slough in May. While there were a number of years between 2011 and 2019 where samples were not obtained in March, the values observed in March 2019 are relatively high for the early spring period. The low river flows in winter 2019 coincided with relatively high algal biomass, consistent with previous analyses that showed a negative correlation between river flow and algal biomass during the winter and spring months (Maier, 2014). The low levels of chlorophyll a observed after the freshet subsided and flows were reduced to some of the lowest rates in the 2011-2019 time series is also consistent with previous observations that in the summer months, river flow is positively associated with algal biomass (Maier, 2014).

In 2019, cyanobacteria accounted for a large proportion of the phytoplankton assemblage in the summer; however, the cell densities (of cyanobacteria and other phytoplankton) were not as high as in 2017 or 2018 at most of the trends sites, with the exception of Campbell Slough, which had high abundances of cyanobacteria in August. There were also relatively high densities of cyanobacteria in 2019 at Ilwaco. Total phytoplankton biomass was highest in early spring at Ilwaco, Welch Island, and Whites Island; in contrast, peak biomass occurred after June at Campbell Slough. Phytoplankton biomass was high both before and after the freshet at Franz Lake Slough. At Franz Lake Slough the first peak coincided with high nitrate concentrations while the second peak (after the freshet) coincided with high phosphate concentrations. The species composition of the first peak was dominated by diatoms and chlorophytes, whereas the second peak was dominated by cyanobacteria, where the assemblage was dominated by *Anabaena* spp. and *Microcystis* spp. Anabaena was also abundant at Campbell Slough, in addition to Merismopedia spp., in August. *Microcystis* and *Pseudo-anabaena* were the most abundant cyanobacteria taxa at Whites Island, Welch Island, and Ilwaco.

Analysis of relationships between environmental variables and phytoplankton assemblages revealed that high relative proportions of diatoms are associated with high concentrations of dissolved oxygen. Diatom growth is also associated with a reduction in nutrient concentrations and are indicative of good water quality. Diatoms tend to dominate in the spring months, where populations can get quite large; most of the annual growth of phytoplankton occurs in the spring and is accomplished by diatoms.

Zooplankton assemblages differ along the spatial gradient from Ilwaco Slough to Franz Lake Slough and over time from early spring to summer. Ilwaco Slough is consistently dominated by copepods, with inputs from rotifers, but very few cladoceran taxa. At the other sites, copepods generally dominated the zooplankton assemblages. At Welch Island and Whites Island, there was an increase in the proportional contribution by cladocerans from spring to summer in each of 2017, 2018, and 2019. At Campbell Slough and Franz Lake Slough, an increase in the proportional contribution of cladocerans was observed from March to June; however, by July, the relative proportions of cladocerans decreased at both sites in 2017 and 2018 and 2019

Stable isotopes ratios of Carbon and Nitrogen

Stable Isotope Ratios (SIR) is used to determine the relative importance of food sources including algae and wetland plants to the food web supporting juvenile salmonids at trends sites. Carbon and Nitrogen isotope ratios yield different information: $\delta^{13}C$ ($^{13}C/^{12}C$) ratio used to identify the primary source of organic matter (i.e., primary producers). In contrast, $\delta^{15}N$ ($^{15}N/^{14}N$) values are useful in determining trophic position. The SIR of C and N were measured in juvenile Chinook salmon muscle tissues and several potential food sources to provide information on the food web supporting juvenile salmonids. These were studied for influence of cumulative mainstem discharge.

In 2019, Isotopic values of carbon in particulate organic matter ($\delta^{13}C$-POM) collected onto filters revealed $\delta^{13}C$ signatures in the range of freshwater phytoplankton most of the time, with values closer to terrestrial vascular plants in May and June at Campbell Slough and Franz Lake Slough. $\delta^{13}C$-POM at Ilwaco was closer to marine values.

When stable isotope signatures for carbon and nitrogen associated with all primary producers is combined, two broad patterns emerge. The average $\delta^{13}C$ for all primary producers is slightly higher in very dry years (for example, 2015) as well as very wet years (for example, 2017), and lower for more moderate years. In the case of nitrogen, this effect is more pronounced. Heavier carbon isotope signatures in particulate organic matter (POM) were associated with dry years. In contrast, there were no significant differences in the stable isotope signatures of nitrogen in POM. The stable isotope signatures of periphyton collected across the trends sites between 2011 and 2019 varied widely across the data set. Average values of $\delta^{13}C$ for periphyton were higher during moderately wet and wet years. There was an increase in the average $\delta\,^{15}N$ for periphyton over a gradient of dry to wet years, with the largest spread in data observed for wet years. When the samples from EMP trend sites were grouped according to whether they came from years with low, moderate, or high cumulative discharge (very dry, dry, moderate, wet), there were significant differences in average $\delta^{13}C$, but not in $\delta^{15}N$.

According to a Bayesian Inference stable isotope mixing model, phytoplankton carbon contributes to the juvenile salmonid food web as part of the diet of chironomid prey, based on stable isotope signatures of carbon; this carbon is incorporated as particulate organic matter and as periphyton. Models looking at how different sources of primary production contribution to additional prey sources are being investigated as more data are gathered, but analysis thus far suggests that periphyton constitutes an important source of organic matter for the preferred prey of juvenile salmonids (i.e., amphipods and chironomids). Estimates of dietary contributions from different prey items inferred from stable isotope mixing models suggest that juvenile salmonid growth is supported by amphipods, chironomids, and other crustacean prey, which is consistent with observations derived from stomach analysis.

Macroinvertebrates

Salmon prey data from 2019 were still under analysis at the time of this report, hence this report presents the variability of juvenile salmon prey data in 2018. Juvenile salmon diets in the lower Columbia River are dominated by Amphipods and Dipterans (particularly chironomids). Amphipod abundance in benthic core samples in 2018 was greatest at Ilwaco Slough. Relatively few amphipods were collected from Welch and Whites Islands, and typically no amphipods were collected from Campbell Slough and Franz Lake. Benthic dipteran larval abundances also tended to be greater in the lower river sites. In contrast, counts of Diptera collected by Neuston tows were highest at the upriver sites. The distribution of benthic invertebrates in the environment is not uniform, and high variation occurred among benthic samples. Regardless, the pattern of declining abundance in amphipods upriver is consistent over time and is also reflected in the diets of juvenile Chinook salmon.

The EMP study has observed a consistent transition from the consumption of primarily dipterans and other wetland insects at Franz Lake and Campbell Slough, to the consumption of both dipterans and amphipods at Whites Island and Welch Island, to the consumption of primarily amphipods at Ilwaco Slough near the estuary mouth. Diets were collected from Welch Island, Whites Island, and Campbell Slough in 2018 and composition of IRI was generally consistent with the typical pattern described above. However, the smallest juvenile Chinook salmon (30-59mm length) did consume cladocerans in addition to dipterans, and the proportion of amphipods consumed at Whites Island was high in 2018 relative to past sample years.

Fish Communities

Examinations of fish communities for all years of sampling, show that all five trend sites are different from each other. The one exception is that Welch and Whites, when compared directly to each other, are similar. Thirteen major families of fishes have been consistently present at the trend sites. Within those families, the fish species range from native marine species at Ilwaco Slough, to freshwater native and non-native species at the remaining EMP trend sites sampled in 2019. Chinook salmon are captured at all five trend sites and are often the numerically dominant salmonid species. Although low in numbers, chum salmon (primarily at Ilwaco Slough) and Coho salmon (primarily at Franz Lake) have also been captured at the five sites.

Ilwaco Slough, sampled for fish since 2011, is the only trend site that is influenced by marine waters due to its proximity to the mouth of the Columbia River (rkm 6). Besides the numerically dominant Threespine stickleback and banded killifish (non-native), Pacific staghorn sculpin and shiner perch seasonally dominate the catches. Two salmon species, chum and Chinook, are regularly captured at this site. Chum salmon was the dominant species (>= 90% of the total salmon numbers) except during 2012 and 2015, when few salmon were captured. Through 2019, six or less individual Chinook salmon were captured during five of the eight years of sampling. Most were unmarked salmon (presumed wild), however, marked salmon (presumed hatchery reared) were captured in 2017 and 2018. The majority of Chinook salmon captured at Ilwaco Slough were subyearlings (fork length <60mm, weight < 2 grams). Genetic analysis of unmarked and marked Chinook salmon captured at Ilwaco Slough has identified two stocks, Spring Creek group-fall and West Cascade-fall.

Welch Island, sampled for fish since 2012, is a tidally influenced, freshwater marsh habitat in the lower Columbia River (rkm 53). The introduced banded killifish and native Chinook salmon were the other most consistently captured species and have been captured in every year of sampling. Chinook salmon comprised 98% or greater of the total numbers of salmon captured within a year and were capture each year. Chum were the second most frequently seen salmon, making up 2% or less of all salmon in a given year, and have been captured in six of eight years of sampling. Each year 70–100% of the Chinook salmon captured at Welch Island were unmarked (presumably wild) juveniles. Genetic composition of unmarked Chinook salmon capture at Welch Island, has been dominated by West Cascade-fall followed by Upper Columbia River-summer/fall. There have been minimal detections of Snake River-fall, Spring Creek-fall, and Rogue River. Genetic composition of marked Chinook salmon at Welch Island had been comprised primarily of two genetic stock groups, West Cascade-fall and Spring Creek Group-fall.

Whites Island, sampled for fish since 2009, is a freshwater, tidally influence marsh, located on the north side of Puget Island in the Columbia River (rkm 72). Five different species in the Salmonidae family have been identified at Whites Island since 2009. The site has been dominated by juvenile Chinook, followed by chum salmon. Coho, Sockeye and mountain whitefish are other species of the Salmonidae family caught at the site. The majority of Chinook salmon captured were unmarked, making up 70–100% of the yearly total. For six of the sampling years, unmarked juvenile chinook were mostly fry and fingerlings. Marked Chinook (presumed hatchery origin), were primarily fingerlings with the occasional yearling seen in 2009, 2010 and 2019. From the genetic stock analysis of unmarked Chinook salmon, seven different stocks have been identified since 2009. West Cascade-fall stock is the predominant group, comprising 80% or more of the fish analyzed. For marked Chinook, four genetic stocks have been identified at Whites Island since 2009. The two major groups are West Cascade-fall and Spring Creek Group-fall. Threespine sticklebacks, banded killifish, American shad and yellow perch have also been caught at the site.

Campbell Slough (rkm 149), sampled for fish since 2008, is a freshwater area that is highly influenced by Bonneville Dam discharge, and minimally by standard tidal fluctuations. Fish community composition at this site is highly variable, with combinations of native and non-native (warm water tolerant) fishes. Two native fish species that contribute a large portion to each yearly total are Chinook salmon and Threespine stickleback. Largescale suckers were also captured at the site every year. Banded killfish and yellow perch were the predominant non-native species at the site. Five predatory fish species identified in the Campbell Slough area are native northern pikeminnow and non-native largemouth and smallmouth bass, Warmouth and yellow perch. Six species of the family Salmonidae have been observed in Campbell Slough since 2008. The most common species is Chinook salmon followed by chum salmon. Coho, cutthroat trout, sockeye salmon and mountain whitefish are the remaining species in the Salmonidae family. Fry and fingerling unmarked juvenile Chinook make up most of the salmon catches at the site. No marked juvenile Fry chinook has been captured at the site. Marked Juvenile chinook are primarily fingerlings. Mean densities of both unmarked and marked Chinook were similar among all years at

Campbell Slough, never exceeding 20 individuals/1000m^2 of either category. Seven distinct genetic stocks of unmarked and unmarked Chinook salmon have been found in Campbell Slough. Most consistent stocks for both marked and unmarked chinook are Spring Creek Group-fall, followed by West Cascade-fall, although percentage contribution in catches vary extensively over the monitoring years.

Franz Lake is a freshwater site located at the confluence of the creek that drains Franz Lake and the Columbia River (rkm 221). The water levels at this site are almost exclusively controlled by discharge from the nearby Bonneville Dam, which regularly prevents sampling in the spring due to sub-optimal water levels and/or temperatures. Franz Lake sampling was initiated in 2008 but has been patchy since. Though this site was not sampled in 2019, we have summarized the community composition of the long term dataset. Nine species of Salmonidae have been captured at this site in the past years, contributing to less than 5% of total catches per year at this site. Salmon catches predominantly consisted of juvenile chinook and Coho. Majority of the catches from this site consisted of native and non-native warm water tolerant fishes like Threespine stickleback, banded killfish and largescale sucker. Juvenile chinook at Franz lake are primarily unmarked, however, we have seen numbers decline since sampling. This could also be an outcome of lack of optimal conditions for sampling at this site, among other environmental factors. The unmarked category of Chinook were predominantly fry (<60 mm fork length) making up more than 70% of those captured followed by fingerlings (60-100 mm fork length). These unmarked salmon were seen in densities between 1–10 individuals/1000m^2. Marked (presumed hatchery origin) Chinook have only been captured in 2008 and 2009. All marked Chinook salmon were fingerlings and had densities of 15/1000m^2 and 3/1000m^2 in 2008 and 2009, respectively. Genetics analysis of Chinook salmon over the course of the 12 years of sampling has been conducted on very few unmarked (23) and marked (41) individuals. No one group is dominant, and no discernable pattern can be seen among the stock groups identified. For unmarked Chinook salmon, stock groups include Spring Creek Group-fall, Upper Columbia River-fall, Snake River-fall, West Cascade-fall and Willamette River-spring. For marked Chinook salmon, Spring Creek Group-fall are the most common, and West Cascade-fall and Willamette River-spring have also been present.

Closing Summary

The Ecosystem Monitoring Program is the only study in the lower Columbia river that collects long-term habitat data from relatively undisturbed tidal freshwater marshes to upper freshwater reaches to allow researchers and restoration practitioners to differentiate between variability associated with natural conditions and variability resulting from human influence, and enhance our understanding of the degree to which these wetlands aid in supporting life-cycle and recovery of endangered and threatened salmonids. In 2019, we monitored water quality, habitat structure, food web dynamics and fish use at five trend sites from the mouth of the Columbia River to the Bonneville dam to assess habitat function at these sites. We also began a focused effort to evaluate the influence of river discharge on wetland habitat conditions. Results from our collective analyses indicate that that differences in annual Columbia River discharge and climate conditions are correlated with significant shifts in wetland food web and habitat conditions including plant community, plankton, and zooplankton abundance and composition and food web nitrogen and carbon dynamics. These findings are critical for evaluating how future environmental fluctuations predicted to be associated with climate change may impact salmonid habitat and food web dynamics. Future EMP research will focus on synthesizing these environmental observations and identifying how shifting climatic, and habitat conditions will impact the salmonid food web.

Management Implications

There are a number of questions that emerge based on several years of observations in the lower Columbia. Some of these include:

- ***How important are biogeochemical processes upstream of Bonneville Dam for the tidal freshwater estuary?*** It is unclear how conditions above Bonneville Dam influence water chemistry and plankton stocks observed downstream. Measurements of water quality and food web components from above the dam would help to determine the degree to which advection is important versus in situ processes such as growth and gas equilibration with the atmosphere.
- ***What is the importance of decomposition of organic matter by microbial organisms in determining its quality for salmon prey?*** Microbial decomposition often results in "trophic upgrading", whereby less labile compounds are transformed through microbial metabolism to compounds that are more easily assimilated. How are these processes influenced by water chemistry, temperature, and nature of the organic matter (e.g., non-native vs. native plant species)?
- ***What factors contribute to cyanobacteria blooms in Franz Lake Slough? Do these blooms pose a problem for wildlife, and if so, what is the extent of the problem?*** Over the last few years, elevated phosphorus concentrations have been observed at Franz Lake Slough in advance of cyanobacteria blooms, although the source is unknown.
- ***How do pulses in primary production from different sources vary in space and time, and how does this influence secondary production and salmon food webs?*** The timing of availability of different sources of organic matter produced through primary production varies between pelagic phytoplankton and marsh vegetation. It would be helpful to compare the magnitude of these stocks to identify patterns that could inform food web models. In addition, pulse events, such as the production and deposition of pollen, could produce reservoirs of organic matter originating from vascular plants in the water column that is independent of detritus transport.
- ***How does prey quality and quantity vary spatially and temporally across the estuary?*** While studies have shown that emergent wetlands are important for prey production and export, accurate assessments of information on prey source in the mainstem and floodplain habitats are yet to be made in the lower Columbia river. The spatial and temporal variation of energy densities of chironomids and amphipods in these undisturbed sites of the lower Columbia river would provide an important functional tool for restoration design.
- ***How does mainstem cumulative discharge effect prey availability and juvenile salmon health and habitat use?*** Additional information is needed to explore the effect of different mainstem hydrologic conditions on the food web and habitat structure for the EMP. Since many EMP sites serve as reference sites for restoration projects, additional information about changes in habitat use and structure under various freshet conditions would help determine crucial actions in restoration design, and mitigate effects of climate change.
- ***How much do specific environmental factors impact growth, fish condition, residence time, age at maturation and survival of anadromous salmonids in the estuary?*** Habitat use in the lower Columbia depends on a myriad of abiotic conditions, and a closer look into specific characteristics such as temperature, DO, discharge, etc. would provide critical information about juvenile salmonid behavior which can be used to inform landscape principles in restoration planning.
- ***How does sediment carbon interact with Greenhouse gases in EMP Trend Sites?*** In order to understand the effects of climate change on the EMP sites, another aspect that needs to be explored further are the exchanges between carbon and greenhouse gases in emergent wetlands. While some data is available from sediment analysis, further exploration is required in terms of accretion and nutrients and carbon sequestration.

The Estuary Partnership shares results from the monitoring program with other resource managers in the region and results from this multi-faceted program are applied to resource management decisions. Results from the EMP are presented and discussed at an annual Science Work Group meeting. The Science Work Group is composed of over 60 individuals from the lower Columbia River basin representing multiple regional entities (i.e., government agencies, tribal groups, academia, and private sector scientists) with scientific and technical expertise who provide support and guidance to the Estuary Partnership. In addition, EMP results will also be shared with regional partners at the Salmon Recovery Conference in April 2021. Data are often provided to restoration practitioners for use in restoration project design and project review templates (e.g., ERTG templates). Finally, data from the EMP are used to compare and contextualize results from the Action Effectiveness Monitoring Program (see Schwartz et al. 2019). A regional database to store and share these data would be a valuable contribution to this work and allow for wider data dissemination and use.

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This study could not have been completed without the help of our partners. We are grateful to the Northwest Power and Conservation Council and the Bonneville Power Administration for funding the Ecosystem Monitoring Program through the Columbia Basin Fish and Wildlife Program. We extend much gratitude to Lyndal Johnson who retired from NOAA Fisheries in 2017. Lyndal was a part of the Ecosystem Monitoring Program from the beginning and contributed to the sampling design and analysis of fish community and contaminants. We also thank Sean Sol who contributed over ten years of fish sampling effort. Amy Borde, Shon Zimmerman and others from PNNL were instrumental in setting up the overall sampling design both at site scale and across the lower river. They also collected vegetation composition, elevation, sediment accretion, and surface water elevation at EMP sites from 2005 through 2016. Jennifer Morace and Whitney Temple of USGS assisted with sampling design and prior years of data collection of abiotic conditions at four of the trends sites and portions of the food web study; we thank them immensely for their collaborative work on this program. This effort could not have been completed without the help of numerous field assistants: we would like to thank Cailene Gunn and Ethen Whattam from PNNL; Candice Trusty from Ocean Associates, Inc.; Stuart Dyer, Katherine Pippenger, and Lyle Cook from OHSU; Narayan Elasmar and April Silva from Columbia River Estuary Taskforce (CREST); Tiffany Thio and Matthew Schwartz from the Estuary Partnership. We also thank the landowners and managers who have allowed us to conduct research on lands they manage, including Alex Chmielewski (Ridgefield National Wildlife Refuge and Franz Lake National Wildlife Refuge), Paul Meyers (Lewis and Clark National Wildlife Refuge), Ian Sinks (Columbia Land Trust), and Stanley Thacker. USFWS Abernathy Fish Technology Center provided the fish feed samples for the stable isotope study. Finally, the Estuary Partnership's Science Work Group provided valuable input throughout the process and peer review on final drafts. The Science Work Group is composed of over 60 members and is integral in ensuring the Estuary Partnership represents the best available science.

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1 Introduction

1.1 Background

The Columbia River historically supported diverse and abundant populations of fish and wildlife and is thought to have been one of the largest producers of Pacific salmonids in the world (Netboy 1980). Anthropogenic changes since the 1860s including dike construction, land use conversion, and the construction of the hydropower system in the Columbia River basin have resulted in alterations to the hydrograph (i.e., timing, magnitude, duration, frequency, and rate of change in river flows); degraded water quality and increased presence of toxic contaminants; introduction of invasive species; and altered food web dynamics. Subsequently, these changes within the Columbia River basin have significantly reduced the quantity and quality of habitat available to fish and wildlife species. The quantity and quality of available habitats affect the diversity, productivity, and persistence of salmon populations (Fresh et al. 2005). Degradation and loss of estuarine habitats can threaten salmon population viability, thus highlighting the importance of identifying limiting factors to salmon survival and filling key knowledge gaps across the habitat gradient of the lower Columbia River to promote salmon recovery.

Threatened and endangered salmonids use shallow water wetland habitats of the lower Columbia River for rearing and refugia, with some stocks utilizing these habitats for long time periods before completing their migratory journey to the ocean (Bottom et al. 2005, Fresh et al. 2005, 2006, Roegner et al. 2008, McNatt et al. 2016). Traditionally, fish and fish habitat research and monitoring efforts were concentrated in the lower reaches of the estuary (nearest the mouth of the river), leaving knowledge gaps in the basic understanding of fish habitat use and benefits within the upper, freshwater-dominated reaches.

Tidal emergent wetland vegetation provides rearing and refuge habitat for juvenile fish and a source of organic matter to the mainstem and downstream habitats, while tidal channels provide access to wetlands and to foraging opportunities. The majority of emergent wetlands in the lower Columbia River cover a narrow elevation range (0.8 – 2.6 m, relative to the Columbia River Datum); thus annual fluctuations in hydrology drive the spatial and temporal variability of wetland vegetation (i.e., cover and species composition) and affect wetland inundation (Sagar et al. 2013). Vegetation species composition in the lower river is spatially variable, with the middle reaches generally showing the greatest species diversity; although some areas are dominated by non-native species such as reed canarygrass (*Phalaris arundinacea*), particularly in the river-dominated upper reaches (Sagar et al. 2013). Identification and quantification of habitat metrics allow for greater predictability biotic responses to changing environmental conditions and improves our understanding of ecological function in the lower river.

Salmonids occupy the upper trophic levels in the Columbia River system. They spend portions of their life cycle in fresh, estuarine, and oceanic waters. Thus, threats to their survival could arise from a variety of sources or stressors occurring at any one of several life stages or habitat types. Large-scale changes to the ecological characteristics of the lower Columbia River food web as a consequence of wetland habitat loss have resulted in a reduction of macrodetritus inputs to the system that historically formed the basis of the aquatic food web (Sherwood et al. 1990). Currently, it is believed that organic matter derived from fluvial phytoplankton (rather than macrodetritus) may be a seasonal driver of the salmon food web (Maier and Simenstad 2009). The consequences of this apparent shift in the type of organic matter fueling food web dynamics are uncertain, and the understanding of food web shifts requires a detailed examination of interactions between multiple trophic levels and environmental conditions. Studying the abundance and assemblage of phytoplankton and zooplankton over space and time provides important information on diets of preferred salmon prey (i.e., chironomids and benthic amphipods). In turn, characterizing the abiotic conditions within emergent wetlands, and in the river mainstem is essential for elucidating spatial and temporal patterns in primary and secondary productivity in the lower river.

The Lower Columbia Estuary Partnership (Estuary Partnership), as part of the Environmental Protection Agency (EPA) National Estuary Program, is required to develop and implement a Comprehensive Conservation and Management Plan. This Management Plan specifically calls for sustained long-term monitoring to understand ecological condition and function, evaluate the impact of management actions over time (e.g., habitat restoration), and protect the biological integrity in the lower Columbia River. The Estuary Partnership implements long-term monitoring through the Ecosystem Monitoring Program (EMP). Ultimately, the goal of the EMP is to track ecosystem condition over time, but also to allow researchers and managers the ability to distinguish between variability associated with natural conditions and variability resulting from human influence. The EMP partnership collects on-the-ground data from relatively undisturbed emergent wetlands to provide information about habitat structure, fish use, abiotic site conditions, salmon food web dynamics, and river mainstem conditions to assess the biological integrity of the lower river, enhance our understanding of estuary function, and support recovery of threatened and endangered salmonids. The creation and maintenance of long-term datasets are vital for documenting the history of change within important resource populations. Therefore, through this program, we aim to assess the status (i.e., spatial variation) and track the trends (i.e., temporal variation) in the overall condition of the lower Columbia River, provide a better basic understanding of ecosystem function, provide a suite of reference sites for use as end points in regional habitat restoration actions, and place findings from other research and monitoring efforts (e.g., action effectiveness monitoring) into context with the larger ecosystem.

Ecosystem-based monitoring of the fish habitat conditions in the lower river is a regional priority intended to aid in the recovery of the historical productivity and diversity of fish and wildlife. In addition to tracking ecological changes in the Lower Columbia River, this year, a collaborative effort has been made to study the effect of varying flow regimes over the monitoring period, of the mainstem on site-specific biotic and abiotic conditions. The hydrology of mainstem Columbia is strongly influenced by winter snow melt and precipitation between months of October and March (Arelia Werner et al., 2007) . The resulting cumulative discharge of the spring freshet depend on magnitude, frequency and duration of precipitation (Nilsson & Renöfält, 2008). Several studies indicate that river discharge exerts a significant influence on ecosystem processes like nutrient and sediment transport, organic matter transport as well as biotic structures. Moreover, studying these relationships will allow us to inform impacts associated with extreme high and low flow events, hence informing restorative actions (Bonada et al., 2006; Larned et al., 2007; Leigh et al., 2010; Rolls et al., 2012). The research questions we have attempted to answer with this report is ***"What is the influence, if any, of cumulative river discharge on juvenile salmon habitat and site-use pattern?"***

The EMP is funded by the Northwest Power and Conservation Council/Bonneville Power Administration (NPCC/BPA) and a primary goal for the action agencies (i.e., the BPA and US Army Corps of Engineers) is to collect key information on ecological conditions for a range of habitats and whether the habitats in the lower river are meeting the needs of outmigrating juvenile salmonids for growth and survival. Such data provide information toward implementation of the 2008 Federal Columbia River Power System (FCRPS) Biological Opinion (BiOp; NMFS 2008). Specifically, NPCC/BPA funding for this program focuses on addressing BPA's Columbia Estuary Ecosystem Restoration Program (CEERP) goal of improving habitat opportunity, capacity and realized function for aquatic organisms, specifically salmonids.

The EMP addresses Action 28 of the Estuary Partnership Comprehensive Conservation and Management Plan; Reasonable and Prudent Alternatives (RPAs) 161, 163, and 198 of the 2000 Biological Opinion for the Federal Columbia River Power System; and RPAs 58, 59, 60, and 61 of the 2008 Biological Opinion. The Estuary Partnership implements the EMP by engaging regional experts at Battelle-Pacific Northwest

National Laboratory (PNNL), National Oceanic and Atmospheric Administration National Marine Fisheries Service (NOAA-Fisheries), Estuary Technical Group (ETG), University of Washington (UW), and Oregon Health & Science University (OHSU).

1.2 Study Area

The lower Columbia River and estuary is designated as an "Estuary of National Significance" by the Environmental Protection Agency (EPA) and as such, it is part of the National Estuary Program (NEP) established in Section 320 of the Clean Water Act. The EMP study area encompasses that of the NEP (a.k.a., the Estuary Partnership), including all tidally influenced waters, extending from the mouth of the Columbia River at river kilometer (rkm) 0 to Bonneville Dam at rkm 235 (tidal influence is defined as historical tidal influence, relative to dam construction in the 1930s). The Estuary Partnership and monitoring partners collect data for the EMP from habitats supporting juvenile salmonids, in tidally influenced shallow water emergent wetlands connected to the Columbia River.

The Estuary Partnership and monitoring partners use a multi-scaled stratification sampling design for sampling the emergent wetland component of the EMP based on the Columbia River Estuary Ecosystem Classification (Classification). The Classification, a GIS-based data set, is a six-tier hierarchical framework that delineates the diverse ecosystems and component habitats across different scales in the lower river. The primary purpose of the Classification is to enable management planning and systematic monitoring of diverse ecosystem attributes. The Classification also provides a utilitarian framework for understanding the underlying ecosystem processes that create the dynamic structure of the lower river. As such, it aims to provide the broader community of scientists and managers with a larger scale perspective in order to better study, manage, and restore lower river ecosystems. The EMP sampling design has been organized according to Level 3 of the Classification, which divides the lower river into eight major hydrogeomorphic reaches (Figure 1).

More recently, subsequent to the development of the sampling design, data collected as part of the EMP and other studies (Borde et al. 2011; Borde et al. 2012) have been used to define five emergent marsh (EM) zones based on spatial variation of the hydrologic regime and vegetation patterns observed in the lower river (Jay et al. 2016). Vegetation species assemblages vary temporally and spatially and are broadly grouped into categories, or (EM) zones, based on vegetation cover and species richness. EM zones are used here to evaluate vegetation patterns within the tidal wetlands of the lower river because they are more representative of vegetation patterns than hydrogeomorphic reach. The zone boundaries are meant to be broad, and variation of the zone boundaries is observed between years. The following river kilometers are currently used to delineate the zones:

EM Zone	River Kilometer (rkm)
1	0 – 39
2	39 - 88
3	89 - 136
4	137 - 181
5	182 - 235

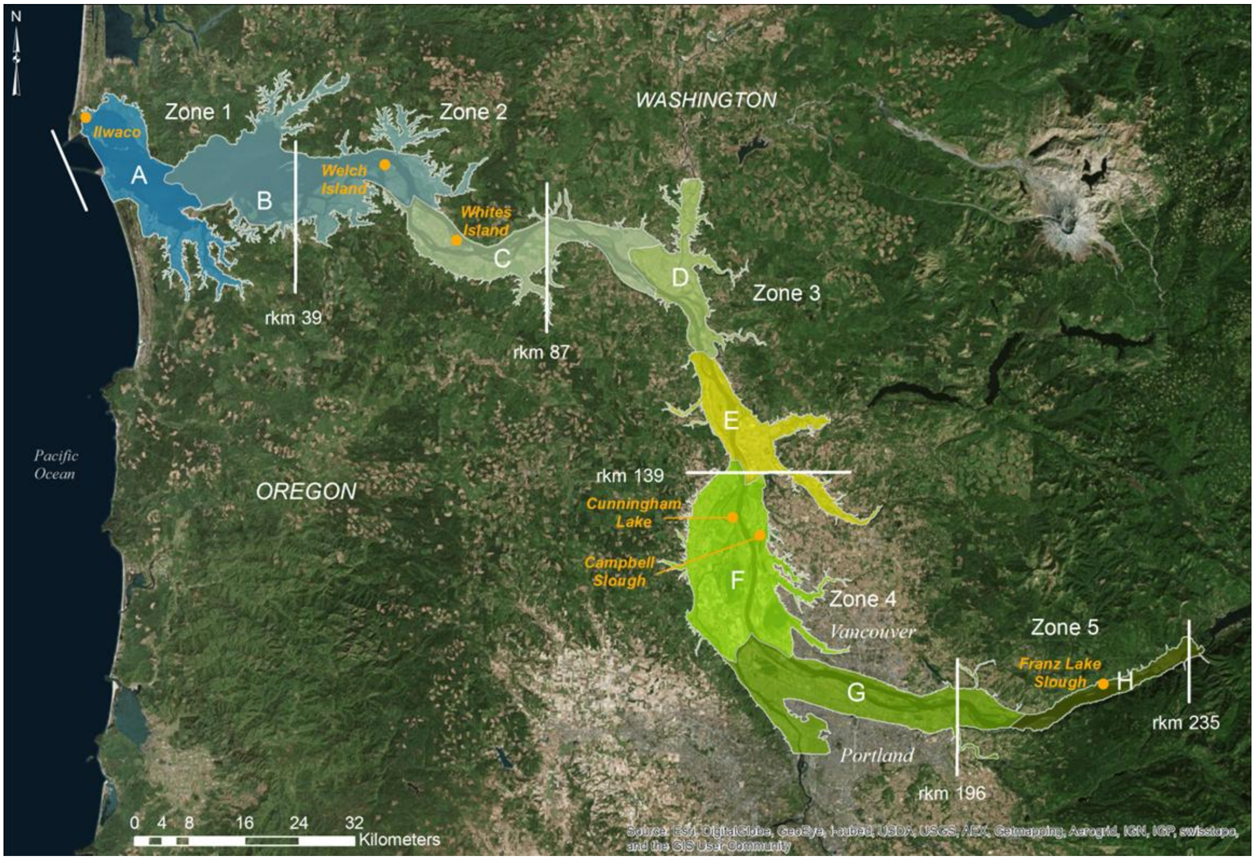


Figure 1. Lower Columbia River and estuary with hydrogeomorphic reaches (A-H) specified by color (Simenstad et al. 2011) and wetland zones (1-5) delineated by white lines (Jay et al. 2016). The 2019 EMP trends sites are shown in orange.

1.3 Characterization of Emergent Wetlands in the Lower Columbia River

1.3.1 Sampling Effort, 2005-2019

The objective of the EMP is to characterize habitat structure and function of estuarine and tidal freshwater habitats within the lower river in order to track ecosystem condition over time, determine ecological variability in these habitats, and provide a better understanding of ecosystem function. The EMP is largely focused on characterizing relatively undisturbed tidally-influenced emergent wetlands that provide important rearing habitat for juvenile salmonids, which also serve as reference sites for restoration actions. The Estuary Partnership and its monitoring partners have focused on providing an inventory of salmon habitats (or "status") across the lower river and including a growing number of fixed sites for assessing interannual variability (or "trends"). Between 2005 and 2012, three to four status sites in a previously unsampled river reach (as denoted in the Classification described above) were selected for sampling each year, along with ongoing sampling of a growing number of trends sites (Table 1). Since 2007, we have conducted co-located monitoring of habitat structure, fish, fish prey, and basic water quality metrics at multiple emergent wetland sites throughout the lower river. In 2011, the Estuary Partnership added food web and abiotic conditions (i.e., conditions influencing productivity such as temperature, turbidity, dissolved oxygen, nutrients) sampling and analysis in both the mainstem Columbia River and at the trend sites.

In 2013, the EMP sampling scheme was adjusted to no longer include data collection at status sites and monitoring efforts focused solely on the six trends sites. The six trends sites selected based on EM Zones were Ilwaco Slough (2010-2019), Secret River (2010-2016), Welch Island (2010-2019), Whites Island (2009-2019), Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2019), and Franz Lake (2008-2009, 2011-2019). Habitat and hydrology data were collected at Cunningham Lake (in addition to the trends sites) as a reference site for habitat and hydrology representative of Reach F sites because vegetation has been periodically trampled by livestock at Campbell Slough in past years. Sampling efforts was discontinued in Secret River from 2017. In 2018, Steamboat Slough, an Action Effectiveness Monitoring and Research site was included in the habitat biomass data collection efforts to aid in the applied interpretation of these data (Schwartz et al. 2019). Methods from the protocol Lower Columbia River Habitat Status and Trends (v1.0, ID 85) were used to monitor the status and trends of specified metrics.

Activities Performed, Year 15 Contract (October 1, 2018 – September 30, 2019):

- Salmonid occurrence, community composition, growth, condition, diet, prey availability, and residency
- Habitat structure, including physical, biological and chemical properties of habitats
- Food web characteristics, including the primary and secondary production of shallow water habitats and in the mainstem lower river and,
- Biogeochemistry of tidal freshwater region of the lower river for comparison to the biogeochemistry of the estuary, key for assessing hypoxia, ocean acidification, and climate change impacts.

Table 1. Summary of sampling effort by site and year(s) conducted at EMP sampling sites. Bold text indicates that data were collected in 2019.

Reach	Type of Site	Site Name	Site Code	Vegetation & Habitat[1]	Fish &Prey[5]	Abiotic Conditions	Food Web[4]
A	Trend	**Ilwaco Slough**	BBM	2011-2019	2011-2013, 2015-2019	2011-2013, 2015-2019	2011-2013, 2015-2019
B	Trend	Secret River	SRM	2008[2], 2012-2016	2012, 2013		2012, 2013
	Tributary	Grays River, lower	-		2015		2015
	Trend	**Welch Island**	WI2	2012-2019	2012-2019	2014, 2019	2012-2019
C	Status	Ryan Island	RIM	2009	2009		
	Status	Lord-Walker Island 1	LI1	2009	2009		
	Status	Lord-Walker Island 2[3]	LI2	2009			
	Trend	**Whites Island**	WHC	2009-2019	2009-2019	2009, 2011-2019	2011-2019
	Status	Jackson Island	JIC	2010	2010		
	Status	Wallace Island	WIC	2010	2010		
	Status	Bradwood Landing	BSM		2010		
D	Status	Cottonwood Island small slough	CI2	2005			
	Status	Cottonwood Island large slough	CI1	2005			
	Status	Dibble Slough	DSC	2005		2005	
E	Status	Sandy Island 1, 2	SI1, SI2	2007	2007		
	Status	Deer Island	DIC	2011	2011		
	Status	Martin Island	MIM	2007			
	Status	Goat Island	GIC	2011	2011		
	Status	Burke Island	BIM	2011	2011		
	Tributary	Lower Lewis River	-		2015		
	Status	Lewis River Mouth	NNI	2007			
F	Status	Sauvie Cove	SSC	2005			
	Status	Hogan Ranch	HR	2005			
	Trend	**Cunningham Lake**	CLM	2005-2019	2007-2009		
	Trend	**Campbell Slough**	CS1	2005-2019	2007-2019	2008-2019	2010-2019
G	Status	Water Resources Center	WRC	2006			
	Status	McGuire Island	MIC	2006			

Reach	Type of Site	Site Name	Site Code	Vegetation & Habitat[1]	Fish &Prey[5]	Abiotic Conditions	Food Web[4]
	Status	Old Channel Sandy River	OSR	2006			2006
	Status	Chattam Island	CIC	2006			
	Status	Government/Lemon Island	GOM	2012	2012	2012	
	Status	Reed Island	RI2	2012	2012	2012	
	Status	Washougal Wetland	OWR	2012	2012	2012	
	Trend	**RM122**	-			2012-2019	
H	Trend	**Franz Lake (slough)**	FLM	2008-2009, 2011-2019	2008-2009, 2011-2019	2011-2019	2011-2019
	Status	Sand Island	SIM	2008	2008	2008	
	Status	Beacon Rock		2008	2008		
	Status	Hardy Slough	HC	2008	2008		

[1] Vegetation biomass data were not collected at any EMP sites in 2014. Only the four upstream trends sites were sampled for biomass in 2015.

[2] Site sampled as part of the Reference Site Study; thus, only vegetation and habitat data were collected.

[3] Lord-Walker Island 2 was sampled by the EMP in conjunction with the Reference Site Study; thus, only vegetation and habitat data were collected.

[4] Phytoplankton and zooplankton only sampled from 2011 – 2019.

[5] Fish prey data were not collected for juvenile Chinook salmon diet and prey availability analyses in 2014.

1.3.2 Site Descriptions

In 2019, the EMP focused primarily on the five trends sites that were monitored over multiple years: Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough. Habitat and hydrology data were collected at all five trends sites plus Cunningham Lake, which is typically sampled for habitat and hydrology metrics as a control site since livestock grazing activities occasionally occur at Campbell Slough (Table 1). Coordinates for trends sites sampled in 2019 are listed in Table 2. The 2019 trends monitoring sites are described in order below, starting at the mouth of the Columbia River and moving upriver towards Bonneville Dam (Figure 1). Maps of the sites, including vegetation communities, are provided in Appendix A and photo points from all sampling years are provided in Appendix B.

Ilwaco Slough. This site is located in Reach A, EM Zone 1 at river kilometer (rkm) 6, southwest of the entrance of Ilwaco harbor, in Baker Bay, WA. The property is currently owned by Washington Department of Natural Resources. The site has developed in the past century as the bay filled in, likely due to changes in circulation from the construction of the jetties at the mouth of the Columbia River, the placement of dredge material islands at the mouth of the bay, and changes in river flows. Ilwaco Slough marsh is dominated by lush fields of Lyngby's sedge (*Carex lyngbyei*) with higher portions occupied by tufted hairgrass (*Deschampsia cespitosa*) and cattail (*Typha angustifolia*). Being so close to the mouth of the Columbia River, the tidal channel is regularly inundated with brackish water (average salinity < 10 Practical Salinity Units, PSU, however salinity up to 20 PSU occur in the late summer). Selected as a long-term monitoring site in 2011, Ilwaco Slough was sampled for all EMP metrics every year except 2014 when only habitat and hydrology were monitored.

Welch Island. The monitoring site on Welch Island is located in Reach B, EM Zone 2 on the northwest (downstream) corner of the island at rkm 53, which is part of the Lewis and Clark National Wildlife Refuge. The island was present on historical late-1800's maps; however, the island has expanded since then, and wetland vegetation has developed where there was previously open water near the location of the study site. The site is a high marsh dominated by *C. lyngbyei*, but with diverse species assemblage and a scattering of willow trees. Small tidal channels grade up to low marsh depressions within the higher marsh plain. The area was selected as a long-term monitoring site in 2012; two other areas of the island were monitored as part of the Reference Sites Study in 2008 and 2009 (Borde et al. 2011).

Whites Island. The Whites Island site is Reach C, EM Zone 2 located on Cut-Off Slough at the southern (upstream) end of Puget Island, near Cathlamet, Washington at rkm 72. A portion of the island is owned by Washington Department of Fish and Wildlife (WDFW) and is maintained as Columbia white-tailed deer habitat. Whites Island is not present on historical maps from the 1880s and was likely created from dredge material placement. The site is located at the confluence of a large tidal channel and an extensive slough system, approximately 0.2 km from an outlet to Cathlamet Channel; however, according to historic photos, this outlet was not present prior to 2006 and the connection to the river mainstem was approximately 0.7 km from the monitoring site. The site is characterized by high marsh, some willows, scattered large wood, and numerous small tidal channels. This long-term monitoring site has been surveyed annually since 2009.

Cunningham Lake. Cunningham Lake is a floodplain lake located in Reach F, EM Zone 4 at rkm 145 on Sauvie Island in the Oregon DFW Wildlife Area. The site is a fringing emergent marsh at the upper extent of the extremely shallow "lake" (Figure 2f) and at the end of Cunningham Slough, which meanders approximately 8.7 km from Multnomah Channel (a side channel of the Columbia River). The mouth of the Slough is located between rkm 142 and 143 near where Multnomah Channel meets the Columbia River. This long-term monitoring site has been sampled exclusively for habitat and hydrology data annually since 2005. In some years, the “lake” is covered with wapato (*Sagittaria latifolia*), however, in all years since 2005, this cover has been sparse or non-existent until 2016 when cover increased once again. In 2017 Cunningham Lake was heavily grazed by cattle.

Campbell Slough. The Campbell Slough site is located in Reach F, EM Zone 4 at rkm 149 on the Ridgefield National Wildlife Refuge in Washington. This long-term monitoring site has been surveyed annually since 2005. The monitoring site is an emergent marsh adjacent to the slough, approximately 1.5 km from the mainstem of the Columbia River. The site grades from Wapato up to reed canarygrass. The US Fish and Wildlife Service manages the impact of reed canarygrass within the extensive refuge by allowing cattle grazing in some areas. The site is usually fenced off from cattle except for times during and immediately after high freshets, which can cause holes in the fencing due to high flows and occasional woody debris. Extensive grazing occurred at the site in 2007, but vegetation appeared to recover in subsequent years. In 2010 and 2011, slight evidence of grazing was again observed. Since 2012 the site has been periodically grazed and trampled by cows, affecting primarily in the upper marsh portion of the site that is dominated by reed canarygrass. In 2017 this site was heavily impacted by cattle grazing due to the removal of the protective fence in the previous winter (2016). In 2018 an electric fence was installed however it failed to keep cattle out, and the wetland was grazed during the growing season prior to habitat monitoring. The electric fence was updated in 2019 in an attempt to prevent further grazing, and it failed. Due to COVID-19 no fence was installed in 2020, and grazing is expected to continue to impact the site.

Franz Lake. The long-term monitoring site located in Reach H, EM Zone 5, the furthest up river at rkm 221 is Franz Lake, which is part of the Pierce National Wildlife Refuge. The site has an expansive area of emergent marsh extending 2 km from the mouth of the slough to a large, shallow ponded area. Several beaver dams have created a series of ponds along the length of the channel resulting in large areas of shallow-water wetland with fringing banks gradually sloping to an upland ecosystem. The sample site is located approximately 350 m from the channel mouth, spanning an area impacted by a beaver dam. The site is primarily high marsh with scattered willow saplings, fringed by willows, ash, and cottonwood.

Table 2. Coordinates of the trend sites sampled in 2019.

Site Name	Latitude	Longitude
Ilwaco Slough	46°18.035'N	124° 2.784'W
Welch Island	45° 47.032'N	122° 45.291'W
Whites Island	45° 9.561'N	122° 20.408'W
Cunningham Lake	45° 48.448'N	122° 48.285'W
Campbell Slough	45° 47.032'N	122° 45.291'W
Franz Lake	45° 36.035'N	122° 6.184'W


a) Ilwaco Slough - 2016


b) Welch Island - 2016


c) Whites Island - 2016


d) Cunningham Lake - 2016


e) Campbell Slough - 2016


f) Franz Lake Slough – 2016

Figure 2. Ecosystem Monitoring sites sampled in 2019: (a) Ilwaco Slough; (b) Welch Island; (c) Whites Island; (d) Cunningham Lake; (e) Campbell Slough; (f) Franz Lake Slough. Updated site photos were unavailable at the time this report was compiled.

1.3.3 Water Year

River flows in the Columbia and its tributaries are influenced by a combination of winter snowpack and pluvial flows driven by rainfall. High snowpack arises from cold and wet winters, while low snowpack arises from dry conditions throughout the winter, which can be either warm or cold (Figure 4). The timing of precipitation and whether it falls as snow or rain influences the timing and magnitude of the spring freshet. Typically, the freshet begins in late April/early May and persists into June. After that, the summer period tends to be dry and river flows are low between June and October.

Compared to the previous nine years (Figure 5), discharge at Bonneville Dam during the freshet in 2019 can be characterized as dry, on the whole (Figure 3). Discharge was nearly as low as the long-term minimum until mid-March and again after the freshet subsided. The freshet itself was close to average. Thus, the water year could be described as having a lower-than-average baseline flow with an average-sized freshet. The freshet occurred in a series of peaks between April and early June.

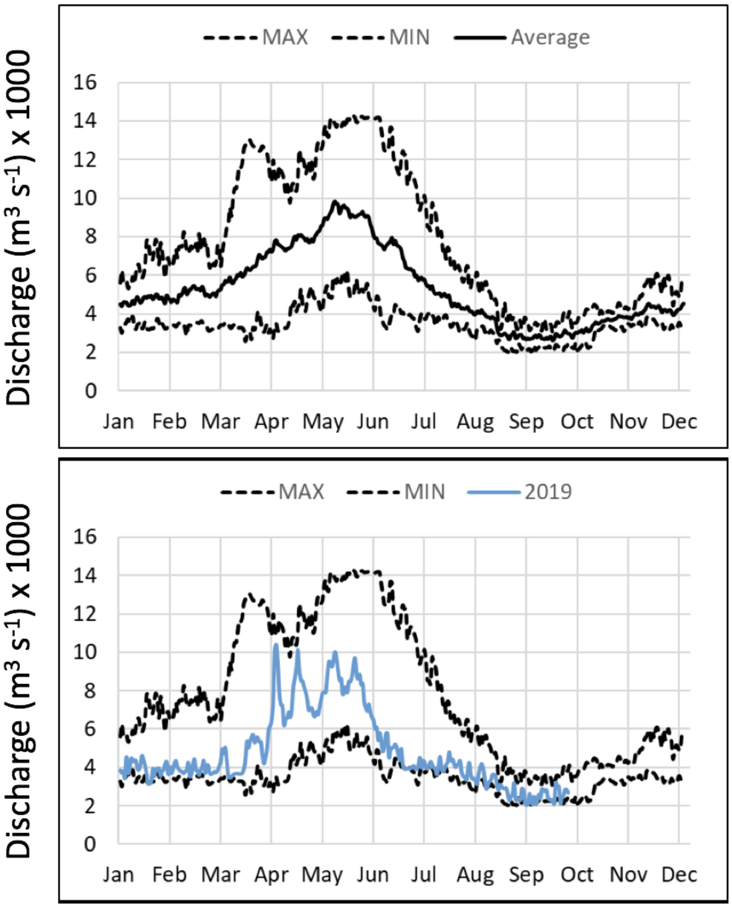


Figure 3. Top panel: Minimum, maximum, and average Columbia River discharge at Bonneville Dam between 2011 and 2019. Bottom panel: Minimum, maximum, and 2019 river discharge fluxes at Bonneville Dam.

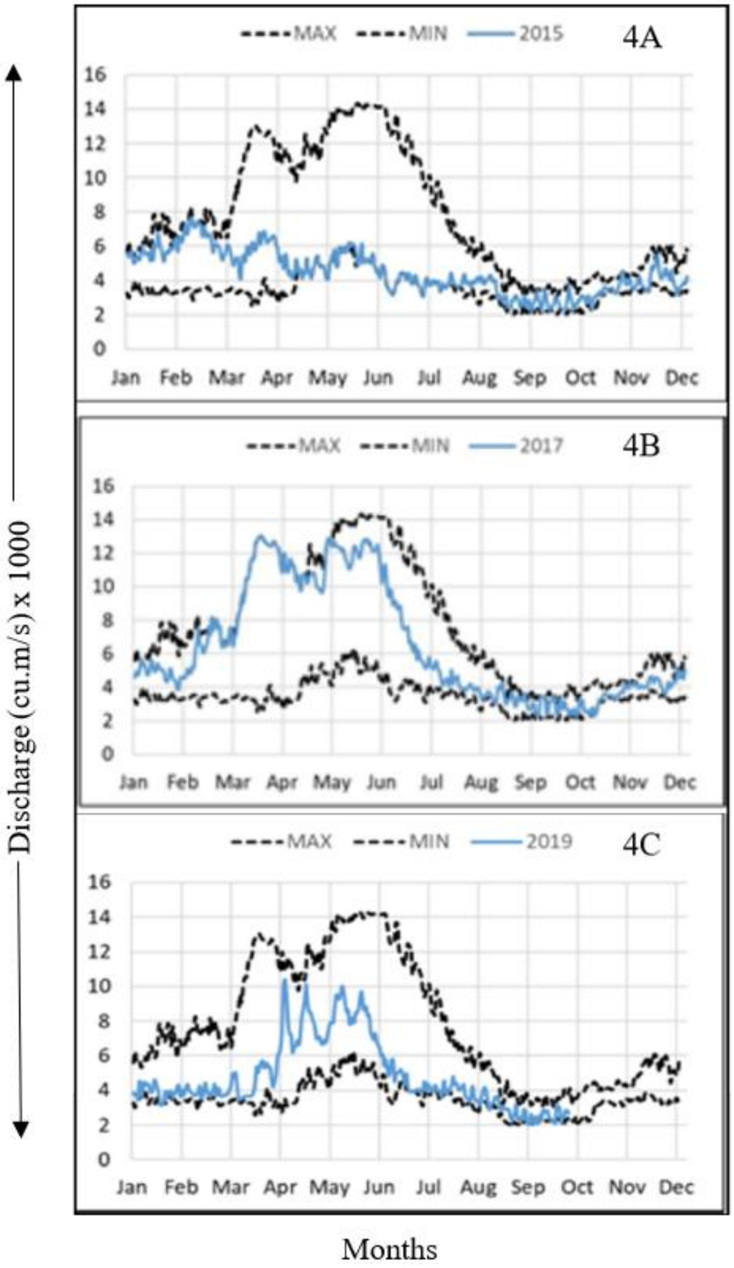


Figure 4. Comparative panels of minimum, maximum, and average river discharge at Bonneville Dam in 2015, 2017, and 2019. Panel 4A represents discharge for 2015 which consisted of warm rainy winter, low snowpack and summer drought. Panel 4B represents discharge for 2017 which consisted of high precipitation and large snowpack. Panel 4C represents discharge for 2019, described as an "dry" year.

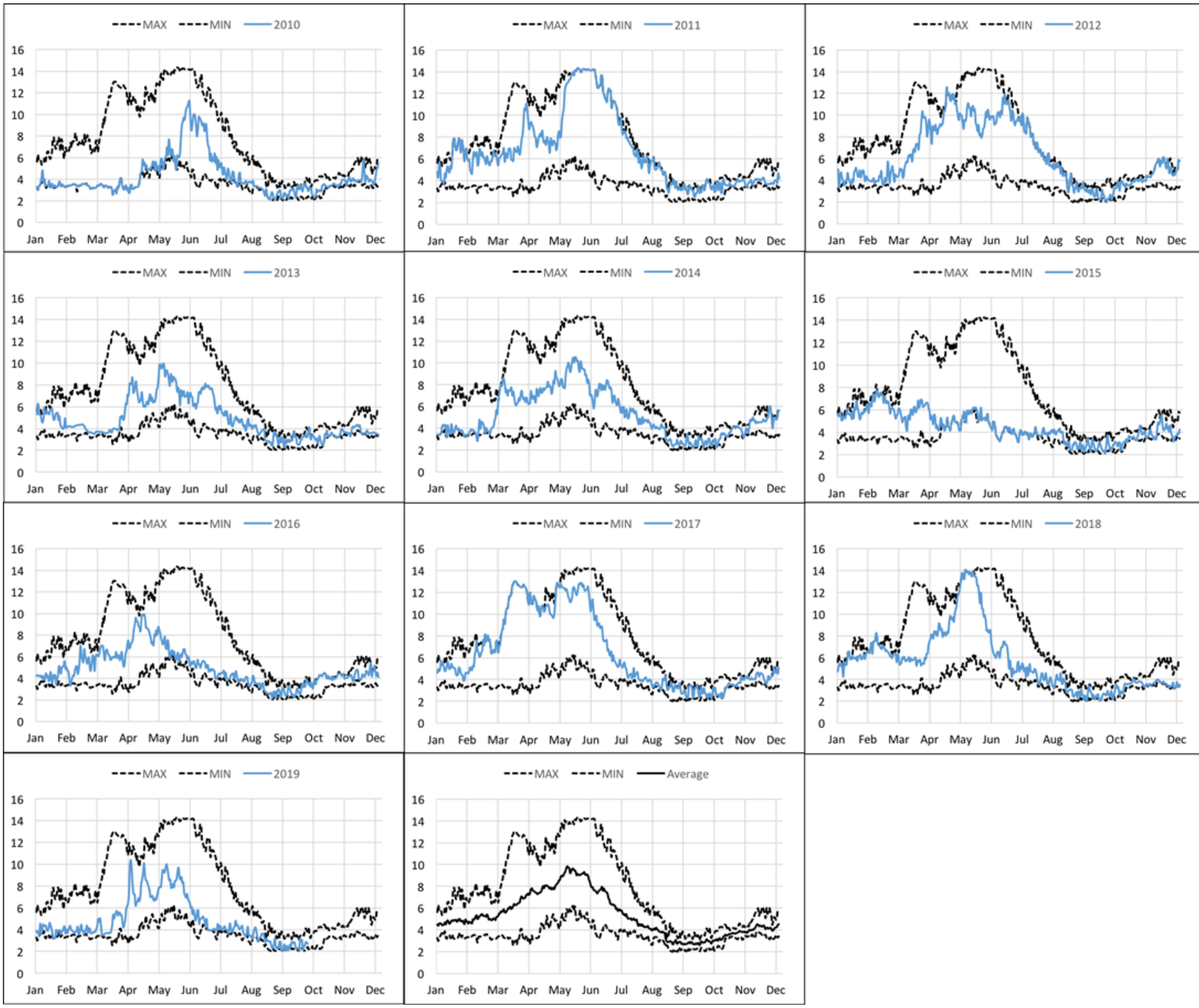


Figure 5. Daily water discharge (m^3/s) at Bonneville Dam. Panels show individual years between 2010-2019 (blue lines) and the daily max and min for all years combined. Vancouver gage web page shows recent flood stage years - https://water.weather.gov/ahps2/crests.php?wfo=pqr&gage=vapw1&crest_type=recent

Based on Figure 5 an NMDS plot of differences in river discharge and river temp between years, hydrologic conditions or cumulative discharge of the Mainstem since 2010 were classified into four categories (Table 3). The results presented in this report have compared the evolution of abiotic and biotic conditions over the monitoring years and differentiated the results between the tabulated categories. Any additional or modified freshet categories have been included in respective sub-sections.

Table 3: Classification of Monitoring years according to cumulative river discarge during the spring freshet between 2010-2019

Year	Cumulative River	River Temperature[1]	Classification

	Discharge ($m^3 \times 10^{10}$) for May – Aug[2]	(°C)	
2019	5.9	85	**dry**
2018	7.8	79	**mid/wet**
2017	8.7	78	**wet**
2016	5.5	85	**dry**
2015	4.7	102	**very dry**
2014	7.3	86	**mid**
2013	6.7	84	**mid**
2012	9.2	59	**Wet**
2011	10.4	59	**wet**
2010	6.3	47	**mid**

[1] River temperature: Number of days days that the river temp was >19 °C May –Sep

[2] Freshet: cumulative river discharge ($m^3 \times 10^{10}$) for May – Aug. Also referred to as "Freshet condition" in this report

1.4 Report Organization

We have divided this report into six sections, excluding References and Appendices. In section 2, we describe methods used to collect data from the mainstem and site-specific abiotic and biotic aspects. Methods of analysis are also described in this section. Section 3 presents results of 2019 monitoring effort. We begin by describing abiotic and nutrient characteristics of the mainstem, and then move onto site-specific abiotic conditions. We then report on site – specific hydrological patterns, sediment dynamics, habitat structure and channel morphology. We then move on to food web dynamics at the trend sites, reporting on primary and secondary productivities, plankton assemblages and report out on isotope analyses of carbon and nitrogen for vegetation and plankton. Stable isotope ratios for salmon prey and whole body salmon have also been presented in this report. Section 3.5 describes prey availability for 2019 and Section 3.6 reports out on Juvenile Chinook community and genetic stock composition for 2019 at the trend sites. GLM models have been used to study the influences of environmental variables and genetic stocks on growth rates in juvenile salmon. Salmon health were determined by lipid content in body samples. Due to a lack of significant differences between freshet conditions (Table 3) and salmon community composition or influence on growth rates or health, no results have been included for this aspect in this report. Based on the overall results, trends observed over the years have been discussed in Sections 4 and 5. In order to inform restorative actions in the study area, Adaptive management measures have been provided in Section 6.

2 Methods

2.1 Mainstem Conditions

2.1.1 Overview

There are two in-situ water quality monitoring platforms in the mainstem Columbia River that provide baseline water quality measurements in support of the Ecosystem Monitoring Program. The first platform, funded by the National Science Foundation, was installed in July 2009 at River Mile 53 (in Reach C) and is physically located on a USGS Dolphin piling (46 11.070 N, 123 11.246 W; Figure 6). A second platform, funded by the Ecosystem Monitoring Program, was installed in August 2012 at River Mile 122 (in Reach G) and is physically located on the outer-most floating dock at the Port of Camas-Washougal (45 34.618 N, 122 22.783 W; Figure 6). The monitoring protocol can be found on monitoringmethods.org (Protocol ID 459). Each instrument platform consists of a physical structure, sensors, sensor control, power supply and distribution, and wireless communication. Data transmitted from the sensors is available within 1–2 hours of collection. Raw data can be downloaded in near-real time from a dedicated webpage (http://columbia.loboviz.com/), and data that have been examined for quality assurance is available upon request). In addition to capturing spatial and temporal resolution of basic water quality and biogeochemical observations for the mainstem Columbia River, an outcome of this effort is to provide daily estimates of parameters necessary for the assessment of ecosystem conditions at sites upstream and downstream of the Willamette-Columbia confluence. Knowledge of daily conditions at these sites allows the identification of contributions from lower river tributaries. Availability of these data enables the calculation of fluxes of various inorganic and organic components, such as nitrate concentration or chlorophyll, an estimate of phytoplankton biomass. Knowledge of nutrients and organic matter flux for a large river is important for a variety of applications, including assessment of pollution, an indication of eutrophication, and quantification of material loading to the coastal zone, where many important ecological processes may be affected. Another product is the assessment of Net Ecosystem Metabolism (NEM), which provides a daily measure of the gross primary production and aerobic respiration occurring in the river as measured by hourly changes in dissolved oxygen. NEM is often used by managers to identify changes or impairments to water quality (Caffrey 2004).

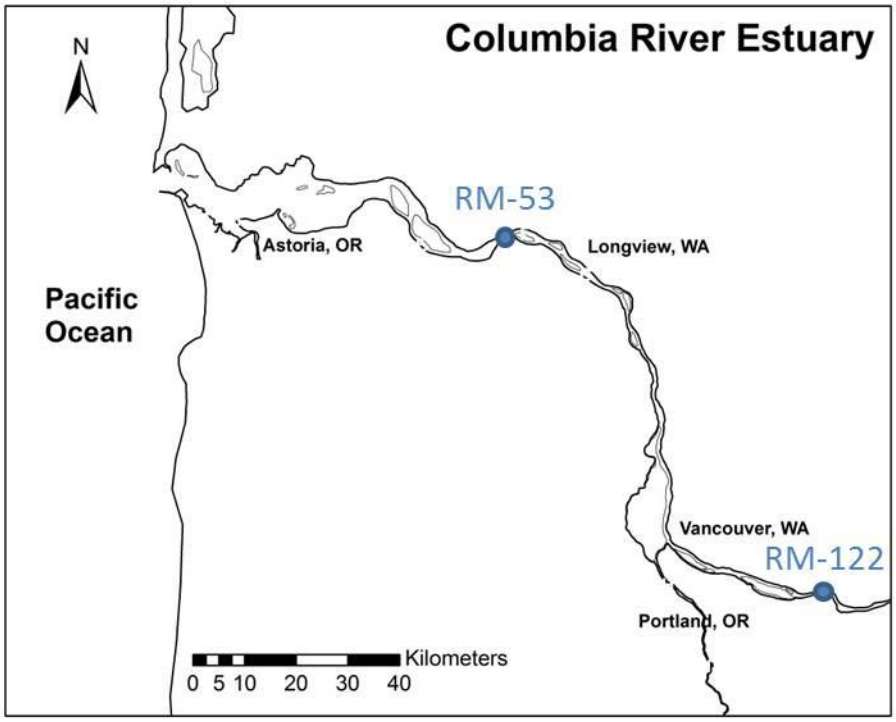


Figure 6. Station locations for the two in-situ water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program. RM-53 (river mile 53) is Beaver Army Terminal, while RM-122 (river mile 122) is located in Camas, WA.

2.1.2 Operation of RM-122 Platform at Port of Camas-Washougal

In 2019, the LOBO instrument platform was deployed from January through June 14; the LOBO sensor suite was replaced with a YSI water quality sonde due to a need for instrument repair. The YSI sonde was in the water from June 14 through October 31, 2019.

2.1.3 Sensor Configuration

Instruments and sensors common to both platforms are described in Table 4. Sensors are configured to collect a sample and telemeter the data every hour. In addition to the parameters listed in Table 4, the RM-122 station is designed to operate a WET Labs Cycle-PO4 to measure dissolved ortho-phosphate concentration. This measurement is a wet chemistry analysis, and therefore this instrument has reagent limitations, which restricts its operation to a reduced schedule (three consecutive measurements daily) and intermittent operation. The instrument was not operational in 2018.

Table 4. Description of the components on the LOBO sensor platforms located at RM-53 and RM-122. Note that the LOBO system was deployed from January through June; after this, the system consisted of a YSI sonde equipped with temperature, conductivity, and dissolved oxygen.

Company	Sensor	Parameters
SeaBird (formerly Satlantic)	LOBO	Power distribution Sensor control Wireless communication Data management
SeaBird (formerly Satlantic)	SUNA Nitrate	Nitrate Concentration
SeaBird (formerly WET Labs)	ECO-CDS	Colored Dissolved Organic Matter (CDOM)
SeaBird (formerly WET Labs)	WQM Water Quality Monitor	Conductivity, Temperature, Dissolved Oxygen, Turbidity, Chlorophyll *a* Concentration

2.1.4 Sensor Maintenance

The sensors are designed to operate autonomously, at high temporal resolution (hourly), and over long periods between maintenance (estimated at three months, although sensors are typically maintained at shorter intervals). This is achieved through a design that maximizes power usage and minimizes biofouling. Antifouling is achieved through the use of sunlight shielding (to prevent algae growth), window wipers, copper instrument surfaces, and bleach injection of the internal pumping chamber. Maintenance trips include cleaning of all sensors and surfaces and performing any other needed maintenance. Additionally, water samples are collected for laboratory analysis of nutrients and chlorophyll *a*. Maintenance activities took place approximately every three weeks in order to change the batteries, clean and calibrate the instruments, download data, and make any necessary adjustments.

2.1.5 Quality Control

Initial sensor calibration was performed by the manufacturer. Each instrument is supplied with a certificate of calibration, and where appropriate, instructions for recalibration. For example, the Seabird SUNA for nitrate measurements operates with a calibration file determined at the factory under strictly controlled environmental conditions but which can be periodically checked and modified for sensor drift by performing a "blank" measurement at our OHSU laboratory using deionized water. At longer intervals (every 1–2 years) the sensors are returned to the factory for maintenance and recalibration.

During periodic sensor maintenance, samples are collected for additional quality control criteria. At RM-53, nutrients, and chlorophyll *a* samples are returned to the laboratory at OHSU and analyzed using established laboratory techniques. Laboratory-based chlorophyll *a* measurements are used to correct the in situ fluorometer measurements. The discrete samples and the corresponding sensor data for nitrate and chlorophyll *a* are shown in Table 5.

Table 5. Comparison of in situ data with laboratory measurements of water samples.

Location/Parameter/# measurements	Regression equation
RM-122/Nitrate/46	$Y = 0.95x + 1$ $r^2 = 0.99$
RM-122/Chl/13	$Y = 0.8x + 1$ $r^2 = 0.93$

2.2 Abiotic Site Conditions

2.2.1 Continuous Water Quality Data (Temperature, DO, pH, Conductivity)

In 2019, water quality was continuously monitored at five trends sites, Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake (Table 6). The monitoring protocol can be found on monitoringmethods.org (Method ID 816). Figure 7 shows how the sensors were deployed to ensure ready access for servicing, and data downloads and Figure 8 shows the periods of deployment of in situ sensors between 2008-2019.

Table 6. Locations of water quality monitors (YSI sondes) at trends sites in 2019.

Site name*	USGS site number	Site name*	Reach	Latitude	Longitude	Monitor deployment date	Monitor retrieval date
Ilwaco Slough			A	46° 18’ 19”	-124° 02’ 06”	3/13/2019	9/11/2019
Welch Island	461518123285700	Unnamed Slough, Welch Island, Columbia River, OR	B	46° 15' 18.4”	-123° 28' 56.8"	3/21/2019	9/10/2019
Whites Island	460939123201600	Birnie Slough, White’s Island, Columbia River, WA	C	46° 09’ 39”	-123° 20’ 16”	3/19/2018	9/10/2019
Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47’ 05”	-122° 45’ 15”	3/23/2019	9/13/2019
Franz Lake	453604122060000	Franz Lake Slough Entrance, Columbia River, WA	H	45° 36' 04"	-122° 06’ 00”	3/20/2019	9/12/2019

*Site names used in this report differ from official USGS site names to be consistent with site names used by other EMP partners.

Figure 7. Images are showing deployment of water quality monitors (YSI sondes) at study sites.

The water quality monitors were Yellow Springs Instruments (YSI) models 6600EDS and 6920V2, equipped with water temperature, specific conductance, pH, and dissolved oxygen probes. In addition, YSI EXO2 units equipped with fluorometer were installed at Campbell Slough and Franz Lake Slough. Addition of a fluorometer provides a capability to detect and monitor chlorophyll and phycocyanin, pigments that approximates the biomass of total phytoplankton and cyanobacteria, respectively. Table 7 provides information on the accuracy and effective ranges for each of the probes. The deployment period for the monitors was set to characterize water quality at the trends sites during the juvenile salmonid migration period. In 2019, the monitors were deployed from mid-March through mid-September (Table 7). In this report, given that the majority of the trends sites are located within Washington State, site-specific water quality data are compared to standards for temperature, pH, and dissolved oxygen set by the Washington Department of Ecology to protect salmonid spawning, rearing, and migration, available at http://www.ecy.wa.gov/programs/wq/swqs/criteria.html. Note that water temperature standards set by the Washington Department of Ecology (threshold of 17.5°C) are more conservative than those outlined by the maximum proposed by Bottom et al. (2011) used for comparisons in the mainstem conditions section of this report (Section 2.1).

Table 7. Range, resolution, and accuracy of water quality monitors deployed at four trends sites. m, meters; °C, degrees Celsius; µS/cm, microsiemens per centimeter; mg/L, milligrams per liter.

Monitoring Metric	Range	Resolution	Accuracy
Temperature	-5–70°C	0.01°C	±0.15°C
Specific conductance	0–100,000 µS/cm	1 µS/cm	±1 µS/cm
ROX optical dissolved oxygen	0–50 mg/L	0.01 mg/L	±0–20 mg/L
pH	0–14 units	0.01 units	±0.2 units

Figure 8. Time periods are corresponding to sensor deployments at five trends sites (2008–2019).

2.2.2 Nutrients (N, P)

Nitrogen and phosphorus are dissolved nutrients that are often present at low enough concentrations to limit plant and phytoplankton growth in aquatic environments relative to other growth requirements. Conversely, in many water bodies, high levels of these nutrients arise from fertilizer and other inputs, which leads to the impairment of water quality following the stimulation of algal and bacterial growth. To analyze water column nutrient concentrations, two 1 L water grab samples were collected from representative open water areas within the sites and subsampled before processing. Three fractions were determined from the subsamples: (1) dissolved inorganic species of nitrogen and phosphorus (nitrate, nitrite, ortho-phosphate, ammonium), (2) total dissolved nitrogen and phosphorus (TDN, TDP), and (3) total nitrogen and phosphorus (TN, TP). Nitrate+nitrite and orthophosphate were determined according to EPA standard methods (EPA 1983a), ammonium was determined colorimetrically (APHA 1998), and total phosphorus were determined according to USGS (1989). Detection limits for each ion or species are given in Table 8. The dates corresponding to sample collection are discussed in Section 2.4.1.2. The monitoring protocol can be found on monitoringmethods.org (Method ID 1591).

Table 8. Detection limits for colorimetric analysis of nitrogen and phosphorus species. TDN = total dissolved nitrogen, TN = total nitrogen, TDP = total dissolved phosphorus, TP = total phosphorus.

Ion or element	Detection limit (mg/L)
Ammonium	0.00280134
Nitrate + Nitrite	0.00700335
Nitrite	0.00140067
TDN	0.01540737
TN	0.1960938
Phosphate	0.00619476
TDP	0.00619476
TP	0.9601878
Silicic acid	0.0280855

2.3 Habitat Structure

In 2019, LCEP and ETG collected field data on vegetation and habitat conditions at the six trends sites (Figure 1). Monitoring dates are provided in Table 9, and detailed maps of the 2019 monitoring sites are presented in Appendix A.

Table 9. Site location and sampling dates for each site sampled in 2017. All habitat and hydrology metrics were sampled at these sites except as otherwise noted.

Site Name	Site Code	River kilometer (rkm)	Site Type	Sampling Date
Ilwaco Slough (Baker Bay)	BBM	6	Trend	7/30/19
Welch Island	WI2	53	Trend	8/1/19
Whites Island	WHC	72	Trend	7/31/19
Cunningham Lake	CLM	145	Trend	8/12/19
Campbell Slough	CS1	149	Trend	8/13/19
Franz Lake	FLM	221	Trend	8/6/19

2.3.1 Habitat Metrics Monitored

The habitat metrics in this study were monitored using standard monitoring protocols developed for the lower Columbia River (Roegner et al. 2009). In 2019, monitoring efforts were focused on vegetation cover, elevation, hydrology, sediment accretion, and the quantification of vegetative biomass production and breakdown. These metrics have been determined to represent important structural components, which can be used to assess habitat function. The rationale for choosing these metrics is discussed below.

Elevation, hydrology, and substrate are the primary factors that control wetland vegetation composition, abundance, and cover. Knowing the elevation, soil, and hydrology required by native tidal wetland vegetation is critical to designing and evaluating the effectiveness of restoration projects (Kentula et al. 1992). In the lowest part of the estuary, salinity is also an important factor determining vegetation composition and distribution. Sediment accretion is important for maintaining wetland elevation. Accretion rates can vary substantially between natural and restored systems (Diefenderfer et al. 2008, Borde et al., 2012); therefore, baseline information on rates is important for understanding the potential evolution of a site. Evaluating vegetative composition and species cover indicates the condition of the site. Vegetation composition is important for the production of organic matter (released to the river in the form of macrodetritus), food web support, habitat for many fish and wildlife species including salmon, and contributions to the biodiversity of the Columbia River estuarine ecosystem. Likewise, vegetative biomass is being collected at the trends sites to begin to quantify the contribution of organic matter from these wetlands to the ecosystem.

Assessment of channel cross sections and channel networks provides information on the potential for many important estuarine functions including fish access (i.e., habitat opportunity; Simenstad and Cordell 2000) and export of prey, organic matter, and nutrients. This information is also necessary to develop the relationship between channel cross-sectional dimensions and marsh size, which aids in understanding the channel dimensions necessary for a self-maintaining restored area (Diefenderfer and Montgomery 2009).

2.3.2 Annual Monitoring

The monitoring frequency for the habitat metrics depends on the variability of the metric between years. The composition, cover, and elevation of vegetation have been monitored annually since 2005. Plant species composition and cover can vary substantially from year to year, depending on climate and related water level differences. Beginning in 2009, we also measured channel cross sections, water surface elevation, and sediment accretion rates. Beginning in 2011 plant biomass was collected at the trends sites, excluding Cunningham Lake annually. In 2015, biomass was collected at the four upstream sites, including Cunningham Lake to maximize collection at sites with reed canarygrass. Sediment samples were collected once from each site to characterize sediment grain size and total organic content, but are not repeatedly collected.

Similarly, vegetation community mapping methods were used to characterize the landscape at the site. After repeated mapping at each site, we determined that large-scale changes were not occurring between years; therefore, this effort is no longer repeated during annual monitoring at trends sites unless vegetation changes are observed. Low inter-annual variability of channel morphology at the trends sites has been observed in prior sampling years. Thus only the cross-section at the channel mouth was measured in 2015. Photo points were also designated at each site from which photographs were taken to document the 360-degree view each year.

2.3.2.1 *Hydrology*

Continuous water level data is collected annually at all the trends sites.. Occasionally sensor failure or loss occurred; however, the sensors have been downloaded and redeployed every year since the initial deployment for the collection of a nearly continuous dataset (Appendix C). The sensors were surveyed for elevation so that depth data could be converted to water surface elevation and evaluated against wetland elevations. The water surface elevation data was used to calculate the following annual hydrologic metrics for each site:

- Mean water level (MWL) – the average water level over the entire year

- Mean lower low water (MLLW) – the average daily lowest water level *(this may shift slightly with different annual deployment elevations of the data logger)*
- Mean higher high water (MHHW) – the average daily highest water level
- Annual water level range – the average difference between the daily high and low water levels
- Annual maximum water level – the maximum water level reached during the year

The monitoring protocol can be found on monitoringmethods.org (Method ID: 3982).

2.3.2.2 *Sediment Accretion Rate*

At each site beginning in 2008, PVC stakes were placed one meter apart and driven into the sediment and leveled. The distance from the plane at the top of the stakes to the sediment surface is measured as accurately as possible every 10 cm along the one-meter distance. The stakes were measured at deployment then subsequently on an annual basis. Additional stakes were deployed in Whites island in 2012. New stakes were deployed at four of the five trend sites in 2015 to measure accretion at additional elevations within site. A new set of PVC stakes were installed at Campbell slough at a lower elevation in 2019. The stakes, termed sedimentation stakes or pins, are used to determine gross annual rates of sediment accretion or erosion (Roegner et al. 2009).

The accretion or erosion rate is calculated by averaging the 11 measurements along the one-meter distance from each year and comparing the difference with past year's average. The accretion or erosion rates were plotted against marsh elevation (m, CRD) to test the hypothesis that high accretion is observed at lower marsh elevations. The accretion or erosion rates were also regressed against annual cumulative discharge from the mainstem over the monitoring period. The monitoring protocol can be found on monitoringmethods.org (Method ID 818).

2.3.2.3 *Salinity*

In order to better assess the influence of salinity on habitat, a conductivity data logger (Onset Computer Corporation) was deployed at the Ilwaco Slough site in August of 2011. The data logger records conductivity and temperature within the slough and derives salinity from those two measurements based on the Practical Salinity Scale of 1978 (see Dauphinee 1980 for the conversion). The monitoring protocol can be found on monitoringmethods.org (Method ID 816).

2.3.2.4 *Vegetation Species Assemblage*

The vegetation sampling areas at each site were selected to be near a tidal channel and to be representative of the elevations and vegetation communities present at the site. This was easier in the upper portions of the study area, where the sites were generally narrower, and the entire elevation range could be easily covered in the sample area. In the lower estuary, the sites are broad and covered a larger area, so in some cases, multiple sample areas were surveyed if possible to cover different vegetation communities (e.g., low marsh and high marsh). The monitoring protocol can be found on monitoringmethods.org (Method ID 822).

Along each transect, vegetative percent cover was evaluated at 2 – 10 m intervals. This interval and the transect lengths were based on the marsh size and/or the homogeneity of vegetation. At each interval on the transect tape, a 1 m^2 quadrat was placed on the substrate and percent cover was estimated by observers in 5% increments. If two observers were collecting data, they worked together initially to ensure their observations were "calibrated." Species were recorded by four letter codes (1st two letters of the genus and 1st two letters of species, with a number added if the code had already been used, e.g., LYAM is *Lysichiton americanus*, and LYAM2 is *Lycopus americanus*). In addition to the vegetative cover, features such as bare ground, open water, wood, and drift wrack were also recorded. When plant identification could not be determined in the field, a specimen was collected for later identification using taxonomic keys or manuals at the laboratory. If an accurate identification was not resolved, the plant remained "unidentified" within the database.

2.3.2.5 *Elevation*

Elevation has been measured many times in previous monitoring years at all trends sites at the locations of vegetation quadrats, water level sensor, sediment accretion stakes, and in the channels. While elevations change over time, the change from one year to the next is minimal, so high-resolution elevation measurements are not collected each year. Elevations were surveyed in 2016 so were not re-surveyed in 2017. The elevation is surveyed using a Trimble or TOPCON real-time kinematic (RTK) GPS with survey-grade accuracy. All surveying was referenced to the NAVD88 vertical datum; the horizontal position was referenced to NAD83. Data collected from the base receiver were processed using the automated Online Positioning User Service (OPUS) provided by the National Geodetic Survey. OPUS provides a Root Mean Squared (RMS) value for each set of static data collected by the base receiver, which is an estimate of error. A local surveyed benchmark was located whenever possible and measured with the RTK to provide a comparison between the local benchmark and OPUS-derived elevations.

Trimble Geomatics Office (TGO) software was used to process the data. Each survey was imported and reviewed. Benchmark information was entered into TGO and rover antenna heights were corrected for disc sink (measured at each survey point to the nearest centimeter) at each point. The survey was then recomputed within TGO and exported in a GIS shapefile format. Surveys were visually checked within TGO and GIS software for validity. Historically elevations were then converted from NAVD88 to the Columbia River Datum (CRD) based on conversions developed by the USACE (unpublished). Using the CRD alleviates elevation differences associated with the increasing elevation of the river bed in the landward direction. Sites below rkm 37, the lower limit of the CRD, were converted to mean lower low water (MLLW). In 2019, NAVD88 elevations were not converted to CRD to aid in the translation of wetland plant community and elevation results to project sponsors implementing restoration projects throughout the river (CRD not being as accessible of a datum as NAVD88).

Quality assurance checks were performed on all data. Elevations from the RTK survey were entered into an Excel spreadsheet to correspond to the appropriate transect and quadrat location. All elevations in this report are referenced to NAVD88 unless noted otherwise. The monitoring protocol can be found on monitoringmethods.org (Method ID 818).

2.3.3 Analyses

2.3.3.1 *Inundation*

The data from the water level sensors were used to calculate inundation metrics from the marsh and channel elevations collected at the sites. The percent of time each marsh was inundated was calculated daily across each marsh's elevation gradient. The average inundation daily, as measured by the average numbers of hours a day (converted to a %) the water surface level is above the marsh elevation, is a means of comparing sites to each other and over time. This is similar to the historic sum exceedance value (SEV) analysis; however, it is summarized by day instead of over the entire growing season (Kidd 2017). The average inundation daily at each site is dependent on the elevation, the position along the tidal and riverine gradient, and the seasonal and annual hydrologic conditions. The average % of the day the mean marsh elevation is inundated for the month of August was calculated for all sites and years. The month of August was chosen because it is a critical time for plant development in the upper river sites, as the freshet draws down and exposes the marsh surface.

Additionally, we have the most consistent amount of data for the month of August all sites and all years monitored. Generally, the trends in % time inundated identified in August correlate well with average % daily inundation for the year. Freshet conditions were also used in the hydrologic analysis; Freshet conditions were defined as the accumulative river discharge at Bonneville Dam from May-August (m^3 x 1010), this metric was developed by J. Needoba at Oregon Health & Science University (OHSU).

The monitoring protocol can be found on monitoringmethods.org (Method ID 954).

2.3.3.2 *Vegetation Community Change Analysis*

Plant species composition and productivity in tidal wetlands respond to annual variability in key ecological processes such as hydrology, salinity, sediment dynamics, and biological interactions. These processes vary naturally but are also projected to change substantially with climate change. For this reason, understanding how key characteristics and functions of wetlands change in response to these processes is important to long-term salmon recovery.

Processes such as hydrology can vary due to normal inter-annual climate variation that affects the amount and form of precipitation. For example, the phases of ENSO (El Nino/Southern Oscillation) and PDO (Pacific Decadal Oscillation) differ regarding the volume of precipitation received in a year, and the relative ratio of snow to rain which affects the spring freshet. Similarly, sea level and the effects of storm waves can vary from year to year in response to ENSO and other climate patterns. Marsh inundation patterns also vary as a result of the actions of bioengineers such as beavers. Grazing by cattle or other herbivores can affect species composition and wetland biomass productivity. Finally, species interactions such as competition from invasive non-native species can alter vegetation composition and wetland function. The strength of biotic interactions is affected by environmental conditions such as inundation, so the effects of biotic elements like invasive species can also vary from year to year.

Data Classification

To begin to evaluate the spatial and temporal variations in vegetation composition, we calculated changes in species richness, percent cover, and relative % cover within and among trend sites over time. Species richness is simply the total number of plant species. Total richness was calculated for each site and each year, as well as average richness per plot. Percent cover is the % of the soil surface that is covered by a plant species. Total plant cover for a plot may exceed 100% when plants overlap. When recording percent cover, maintaining consistency among observers or between years can be difficult, and for this reason, we use relative percent cover to compare species with each other. The relative cover is the proportion of total vegetative cover represented by a species or guild of species. With relative cover, the sum of all species always adds up to 100. The relative cover is a more reliable method for comparing species with each other or evaluating the change in a species over time. We further segregated plant species by key characteristics including native/non-native provenance and wetland indicator status. Additionally, Shannon diversity (H) and evenness (J) indices were calculated from the relative plant cover data using the standard methods outlined by Magurran (1988).

Most plants were identified to the species level or finer, allowing for clear categorization as native or non-native. However occasionally at some growth stages, certain plants could not be identified to species level. A few of these taxa contained both native and non-native species or varieties and were classified as "Mixed." For example, at certain growth stages, several species of Agrostis (bentgrass) are difficult to tell apart and were lumped as "Agrostis species." Since this genus includes both native and non-native species, it was classified as "Mixed." In calculations involving native vs. non-native species, "Mixed" taxa were included with the non-native group.

Most species also have a clear wetland indicator status that has been identified in the literature. Wetland indicator values reflect how dependent on wetland hydrology a species may be (Reed 1988). Obligate wetland species (OBL) are those that appear in wetlands >99% of the time. Facultative-Wet wetland species (FACW) are those that occur in wetlands 67-99% of the time and occasionally are found in non-wetland habitats. Facultative wetland species (FAC) are those that appear in wetlands about half the time (34-66%), and in non-wetland habitats at other times. Facultative upland species (FACU) are those that occur mostly in upland habitats and less than 34% of the time in wetland habitats, and Upland species (UPL) are those that occur in wetlands less than 1% of the time. The relative proportion of species that fall into those categories, and their respective percent cover, change as the environmental conditions and biotic interactions vary. These changes can indicate changes in wetland functions and values with respect to salmon.

Long-term Trends and Drivers Analysis

Long-term plant community change analysis was conducted across all active EMP sites including annual plant community data starting in 2011 through 2019. When applicable plant community metrics were transformed and correlated with hydrologic conditions such as annual freshet conditions and daily inundation, only significant (p-value < 0.05) correlation and regressions were reported. Data analysis was conducted using Microsoft Office Excel (2016), Exploratory (2017), and R (2020) software.

2.4 Food Web

2.4.1 Primary Productivity

2.4.1.1 *Emergent Wetland Vegetation*

2.4.1.1.1 Aboveground Vegetation Biomass, Macrodetritus, and Soil

Starting in the summer of 2017 detritus sampling was included in the biomass sampling and analysis to evaluate detrital production and export. In the winter of 2018 (and all sampling events to follow) biomass sampling protocols changed slightly to accommodate detrital sampling and streamline data collection (Table 10, Table 11). This included shifting from “strata” mixed species designations to simple high and low marsh strata descriptions across all sites sampled. This change has also included species biomass weights to be recorded individually to assess species-specific contributions to each high and low marsh stratum (in the past mixes of species were assessed together). In general, these changes will allow for a more detailed understanding of species-specific biomass contributions and still allow for long-term comparisons to overall site, high and low marsh contributions.

Table 10: Seasonal data collection schedule Winter 2018-Summer 2019. 2019 results were unavailable at the time of this report. Sp= Species.

Season	Live Sp Cover	Live Sp Weights	Detritus Lignin	Detritus C:N	Live Sp Lignin	Live Sp C:N	Soil C:N	Soil Bulk Density	Soil Grain Size
Winter 2018	X	X	X	X			X		
Spring 2018	X	X	X	X					
Summer 2018	X	X	X	X	X	X	X	X	X
Winter 2019	X	X	X	X					
Summer 2019	X	X	X	X	X	X			
Winter 2020	X	X	X	X	X	X			

Field Methods

From Summer 2011 to Winter 2020, aboveground biomass was sampled to estimate the primary productivity at three trends sites. Samples were collected in the summer (July or August) during the peak biomass period and again during the winter (January or February) during the winter low biomass period. In 2018, Spring sampling also took place in March. For the emergent marsh biomass sampling, a $1m^2$ plot was randomly placed along the established vegetation transect, but off-set 2 m from the transect to ensure that the biomass plots did not intersect the vegetation percent cover plots. Biomass was randomly sampled within distinct vegetation strata as determined by plant species dominance, to 1) more clearly associate the samples with vegetation type, and 2) reduce the variability between samples within strata. Within the $1m^2$ biomass plot, a $0.1m^2$ quadrat was placed in a randomly selected corner and all rooted vegetation, live and dead, was removed using shears. Each sample was sorted in the field to separate the primary strata species from other species and to distinguish live from dead plant material. The biomass samples were placed in uniquely numbered bags and held in a cooler until samples were transported to the laboratory. Dominant vegetation species were recorded along with the corresponding biomass sample number. Submerged aquatic vegetation (SAV) plots were sampled in 2011-2013 using similar methods; however, due to the relatively low contribution of this strata to the overall macrodetritus production, the collection did not continue in subsequent years.

Beginning in summer 2018 at each site, we collected data and samples from at least 18 plots, nine high marshes, and nine low marshes. Plots were located in such a way to sample the dominant plant species present at each site in the high and low marsh and were distributed across the site while avoiding the permanent vegetation transects. During summer 2018, vegetation composition was assessed in a $1m^2$ plot by quantifying % cover for each species that had at least 5% cover and noting any species that was present with less than 5% cover (species denoted as “Other”). If a species had dead biomass present (dead stems or leaves that were still attached to the root system), the % cover of dead biomass was measured separately from the % cover of live biomass. For species with greater than 5% cover, we recorded the average maximum height for both live and dead biomass. The % of the plot that was covered by water was noted, and its depth in cm. The % cover of bare ground and detritus was also noted. Biomass and detritus collection occurred in a $0.1m^2$ subplot in one of the corners of the larger plot. For this subplot, we noted which of the “Other” species were present since these were collected for biomass analysis.

Beginning in winter 2019, the field data methods were changed, and vegetation height, species cover, water, bare ground, and detritus cover data were collected only for the small $0.1m^2$ plot.

Biomass and Detritus Collection

In a $0.1m^2$ plot in one of the corners of the larger $1m^2$ plot, we used clippers to cut all plant biomass at the soil surface. Plant matter was cut around the outer edge of the quadrat frame, and all material that was in or over the subplot was collected, whether or not it was rooted in the subplot. For plants rooted in the plot, only material that was in or over the plot was collected. The material was laid out on a plastic sheet in the field and separated by species and according to whether it was alive or dead. Species with cover >5% in the large plot were separated into separate plastic bags for analysis, while all species that were <5% in the large plot were combined into a single bag. All detritus within each subplot was also collected into a single plastic bag. Detritus was defined as any organic material that was not attached to roots. Samples were stored in coolers on ice until they returned to the lab where they were stored at $<5^{o}C$ until processing.

Soil Collection

Soils were collected during summer 2018 at five high marsh and five low marsh biomass plots at each site. PVC coring tubes were made with sharpened ends to facilitate soil penetration with minimal effects on soil compaction. Coring tubes had an internal diameter of 5.1cm and were marked around the outside at 10cm from the lip to indicate the depth of the sample to be collected. Spades and sharp knives were used to cut the soil around and beneath the cores. Samples were placed in plastic bags and stored on ice in a cooler until return to the lab.

Table 11. The number of samples collected in each year and season (S=summer, F=fall, W=winter, Sp=Spring) for all sample sites and vegetation strata. In 2017-2020 we also sampled at Steamboat Slough, a restoration site located near Whites Island.

		2011-12		2012-13			2013-14		2015-16		2016-17		2017-18			2018-19		2019		
Site[1]	**Strata**	**S**	**W**	**S**	**F**	**W**	**S**	**W**	**S**	**W**	**S**	**W**	**S**	**W**	**Sp**	**S**	**W**	**S**	**W**	**Total**
BBM	CALY	3	4	6		6	4	4			6	6								39
BBM	CALY/AGSP	4	3	4		4	6	6			6	6								39
BBM	SAV	4	4	6		6	6													26
SRM	HM			5		5	9	9			9	9								46
SRM	LM			5		5	9	9			9	9								46
SRM	SAV			6		6	6													18
WI2	HM			5		9	9	9			12	12	14	12	12	9	9	9	9	120
WI2	LM			4												9	9	9	9	40
WI2	SAV			4		4	6													14
WHC	CALY		1	3		3	3	3	3	3	3	3	3	3	3					31
WHC	HM												1		1	9	9	9	9	38
WHC	PHAR												1	1	1					2
WHC	PHAR/HM	6	4	5		5	6	6	9	9	9	9	9	8	8					85
WHC	SALA	2	3	3		3	6	6	6	6	6	6	6	6	6					59
WHC	SAV	8	8	6		6	6													34
WHC	LM															9	9	9	9	36
CLM	ELPA/SALA								6	6	6		5							
CLM	PHAR								7	7	7		6							27
CLM	SALA												1							1
CS1	ELPA/SALA	5	4				6		6	6	7	6								40
CS1	PHAR	3	4				6				6	6								25
CS1	SALA						5		6	6	6	6	6							35
CS1	SAV	8	8				6													22
FLM	HM															9	9	9	9	36
FLM	PHAR/HM	4	7	3	2	4	3	5	6	6	6									46
FLM	PHAR/POAM	2	5		2															9
FLM	POAM			3	2	1	6	4	6	6	6									34
FLM	SAV			5	8	6	6													
FLM	LM															9	9	9	9	36

[1] BBM – Ilwaco Slough, SRM – Secret River Marsh, WI2 – Welch Island, WHC – Whites Island, CLM – Cunningham Lake, CS1 – Campbell Slough, FLM – Franz Lake.

Laboratory Methods

Biomass and Detritus, dry weight

In the laboratory, live, dead, and detritus samples were stored in a refrigerator prior to processing. Samples were individually rinsed of all non-organic material over a 500μm sieve, and any obvious root material was removed. Pre-weighed paper bags or tinfoil were used to secure the individual biomass samples, a wet weight was measured, and the samples were placed in a drying oven set at 90°C for at least four days. When samples were deemed completely dry (checked by reweighing a subset of samples on consecutive days), a dry weight was measured for each sample and its corresponding bag or foil tray. If paper bags were used, they were re-weighed empty to account for any weight loss of the bag. The final sample dry weight was determined by subtracting the dry bag or foil weight from the dry weight of the container with the sample.

CN Analysis

All detritus samples and a subset of live and dead summer biomass samples were analyzed for carbon and nitrogen content. Live and dead summer biomass samples from each plot were selected for analysis if they covered at least 20% of that plot. Dried samples were pulverized with a small food processor and stored in a desiccator prior to analysis. Carbon and nitrogen content were analyzed with a FlashEA 1112 CN analyzer (Thermo Elecron Corp.). Approximately 18-22 mg of each subsample was packaged in a tin capsule. Chemical and soil standards were analyzed approximately every ten samples, and at least 10% of the samples were randomly selected and reanalyzed on a different day. Replicate measurements were averaged for reported results.

ADF Lignin

Dried and ground detritus samples were tested for ADF lignin following Soiltest 2016 Standard Operating Procedures for feed lignin (Section 50.400.600). Soiltest uses an acidified detergent solution with the Ankom digester to dissolve cell solubles, hemicellulose, and soluble minerals leaving a residue of cellulose, lignin, heat-damaged protein, a portion of cell wall protein, and minerals. This residue is then placed in an acid wash, and lignin is determined gravimetrically as the residue remaining after extraction, followed by an ash correction. Reference samples were run with each batch, and a duplicate sample was analyzed every ten samples.

Soil Bulk Density

Soil cores were frozen in the laboratory until processed. Each sample was oven-dried at 60°C for at least four days. The mass of each dried sample was recorded, and bulk density was calculated as the ratio of dry weight to wet volume. Wet volume was assumed to be 204.28 cm^3 based on a coring tube internal diameter of 5.1 cm and a coring depth of 10 cm.

Soil TOC/N Analysis

A subsample of each dried soil sample was pulverized and homogenized with a mortar and pestle, and large root fragments were removed. Soil subsamples were tested for the presence of inorganic carbon with a few drops of hydrochloric acid (HCl), which would cause the sample to effervesce with CO_2 bubbles if a significant quantity of carbonate were present. No effervescence was observed; therefore all soil samples were analyzed for total carbon under the assumption that total carbon measurements were representative of organic carbon content. Carbon and nitrogen content were analyzed with a FlashEA 1112 CN analyzer (Thermo Elecron Corp.). Approximately 100 mg of each subsample was packaged in a tin capsule. Chemical and soil standards were analyzed approximately every ten samples, and at least 10% of the samples were randomly selected and reanalyzed on a different day. Replicate measurements were averaged for reported results.

Soil Texture Analysis

Dried soil samples were sent to Materials Testing & Consulting, Inc. (MTC) in Olympia, Washington for particle size distribution following recommended protocols for measuring conventional sediment variables (PSEP Report TC-3991-04, 1986). Samples were shaken in appropriately sized sieves to separate gravel (>2000 microns), sand (between 62.5 - 2000 microns), and fines (<62.5 microns). The fines were further separated into silt (3.9 – 62.5 microns) and clay (<3.9 microns) using a pipetting technique to measure the differential settling rates of different sized particles. Samples were processed in batches of a maximum of 20 per batch. Each batch included one sample that was analyzed in triplicate.

Analysis

2019 results were not available at the time of this report, hence, 2018 analysis has been presented. Average dry weight was calculated for various strata and site values. For 2015 to 2018 data (Table 11), the proportion of the dominant species comprising each sample was calculated. Those data were used to identify samples that were primarily a single species. Those samples were then used to make estimates of the aboveground biomass for specific species within the study area. For long-term comparative analysis, all biomass data collected prior to 2018 was assigned wetland elevations based historic RTK survey data collected at plant community plots when elevation could not be determined it was left blank, and the biomass data point was not included in the high vs. low marsh long-term biomass assessment. Starting in 2018 all biomass plots were surveyed in directly with RTK equipment.

When applicable biomass, detritus, and soil metrics were transformed and correlated elevation and with hydrologic conditions such as annual freshet conditions and daily inundation, only significant (p-value < 0.05) correlation and regressions were reported. Data analysis was conducted using Microsoft Office Excel (2016), Exploratory (2017), and R (2018) software.

2.4.1.2 *Phytoplankton*

Abundance

Phytoplankton abundance was estimated in two ways: (1) from pigment concentrations, and (2) by direct counts using light microscopy. Phytoplankton abundance can be estimated by measuring the concentration of chlorophyll *a*, a photosynthetic pigment that is common to all types of phytoplankton. Surface water samples were collected into two 1 L brown HDPE bottles and sub-sampled prior to processing. A subsample of water (typically between 60–300 mL) was filtered onto a 25 mL glass-fiber filter (GF/F) for chlorophyll *a* and kept frozen (-80°C) pending analysis. Chlorophyll *a* was determined fluorometrically using a Turner Designs Trilogy fluorometer using to the non-acidification method, which is highly selective for chlorophyll *a* even in the presence of chlorophyll *b* (Welschmeyer 1994).

Phytoplankton abundance was also determined by enumeration of individual cells using inverted light microscopy. The dates corresponding to sample collection for determination of nutrient concentrations, zooplankton abundance, and phytoplankton abundance are shown in Table 12. Duplicate 100 mL whole water samples were collected from each of the trends sites. The samples were preserved in 1% Lugol's iodine and examined at 100, 200 and 400x magnification using a Leica DMIL and Zeiss Axiovert 200M inverted light microscopes following concentration achieved through settling 2.5–50 mL of sample in Utermohl chambers (Utermohl 1958) overnight (~24 h). Cell counts were performed at 200 and 400x magnification, with an additional scan done at 100x magnification to capture rare cells in a broader scan of the slide. The estimated error in abundance measurements was <5% at the class level and ~10% for genus-level counts. The monitoring protocol can be found on monitoringmethods.org (Method ID 1589 and 1590).

Table 12. List of samples analyzed (Xs) and data of collection from five trends sites in the Lower Columbia River in 2019.

Site	Zone	Reach	Date	Nutrients	Zooplankton	Phytoplankton
ILWACO SLOUGH	1	A	3/13/19	X	X	X
			4/8/19	X	X	X
			5/7/19	X	X	X
			6/11/19	X	X	X
			7/16/19	X	X	X
			8/15/19			
			9/11/19	X	X	X
WELCH ISLAND	2	B	3/21/19	X	X	X
			4/12/19	X	X	X
			5/16/19	X	X	X
			6/13/19	X	X	X
			7/18/19	X	X	X
			9/10/19	X	X	X
WHITES ISLAND	3	C	3/19/19	X	X	X
			4/11/19	X	X	X
			5/9/19	X	X	X
			6/13/19	X	X	X
			7/18/19	X	X	X
			8/21/19			
			9/26/19	X	X	X
CAMPBELL SLOUGH	4	F	3/23/19	X	X	X
			4/10/19	X	X	X
			5/8/19	X	X	X
			6/14/19	X	X	X
			7/19/19	X	X	X
			8/13/19			
			9/13/19	X	X	X
FRANZ LAKE SLOUGH	5	H	3/20/19	X	X	X
			4/9/19	X	X	X
			5/8/19			
			6/10/19	X	X	X
			7/17/19	X	X	X
			8/14/19			
			9/12/19	X	X	X

Multivariate Statistical Analyses

Nonmetric Multi-dimensional Scaling (NMDS) and Canonical Analysis of Principal Coordinates (CAP) routines were performed using PRIMER-E v.7 with PERMANOVA+. NMDS is a multivariate technique that identifies the degree of similarity among biological communities within a group of samples in a data set. In NMDS, samples are typically represented in 2-dimensional ordination space using the distance between sample points as a measure of similarity of biological communities; short distances represent the relatively high similarity between samples, while longer distances represent the relatively low similarity between samples.

Major phytoplankton taxa were selected for multivariate analyses if their abundance constituted at least 10% of total phytoplankton abundance in any sample. Taxa that did not meet these criteria were excluded from the analysis. Two NMDS analyses were run for this study that included (i) all major phytoplankton taxa ($\text{NMDS}_{\text{total}}$) and (ii) only major diatom taxa ($\text{NMDS}_{\text{diatom}}$). Abundances for 25 major phytoplankton taxa ($NMDS_{total}$) and ten major diatom taxa ($NMDS_{diatom}$) were standardized by sample, and the data were square-root transformed in order to achieve a normal distribution of the data prior to analysis. Canonical Analysis of Principal Coordinates (CAP) is an analytical technique that uses canonical correlation to determine the degree to which environmental factors explain variability among biological communities. A Bray-Curtis resemblance matrix was assembled using the standardized, square-root transformed phytoplankton abundance data and six environmental variables including NO_2^+ , NO_3^-, NH_4^+, PO_4^{3-}, mean daily water temperature, mean daily dissolved oxygen saturation, and mean daily discharge (at Bonneville Dam). Environmental data were normalized prior to analysis to compare variables on the same scale. Samples with missing environmental data were excluded from multivariate analyses. A total of 70 samples were analyzed in both NMDS analyses, and a total of 38 samples were included for CAP.

2.4.2 Secondary Productivity

2.4.2.1 *Zooplankton*

Secondary productivity (the rate of growth of consumers of primary production) was not measured directly but was estimated from the abundance of pelagic zooplankton. The samples were collected from near the surface of the water (< 1 m depth) using an 80 µm nylon mesh net with a mouth diameter of 0.5 m and a length of 2 m at five trend sites. A list of the collection dates and sampling sites are given above in Table 12.

Abundance

Zooplankton abundances collected via net tow were determined at each of the five trend sites. The net was fully submerged under the water and was dragged back and forth from a small boat through the water for approximately 3-5 min or over approximately 100 m. The samples were preserved in 1.5% formalin immediately after collection. A flow meter (General Oceanics Inc., Model 2030R) was mounted to the net's bridle to provide an estimate of the volume flowing through the net. The volume of water passing through the net was determined by knowledge of the distance of water passing through the net, the velocity of the water passing through the net, and the volume of water passing through the net, as calculated from both the distance traveled and the net diameter (as described in the flow meter manual). The distance covered (in meters) was determined from:

$$Distance = \frac{Difference\ in\ counts \times Rotor\ Constant}{999999} \qquad (1)$$

where the difference in counts refers to the difference between the initial and final counts on the six-digit counter, which registers each revolution of the instrument rotor. The speed is calculated from:

$$Speed = \frac{Distance\ in\ meters \times 100}{Time\ in\ seconds} \qquad (2)$$

The volume is determined as:

$$Volume\ in\ m^3 = \frac{3.14 \times net\ diameter^2 \times Distance}{4} \qquad (3)$$

For each net tow, the volume of material collected in the cod end of the net was recorded. From this, a concentration factor was calculated, and a final estimate of the volume examined was determined by multiplying the concentration factor by the final volume of the concentrated sample examined under the microscope.

Taxonomy

Zooplankton taxa were broadly categorized into one of the following groupings: rotifers, cladocerans, annelids, ciliates, and copepods, and ‘other.’ Within these groups, individuals were identified to genus or species where possible (rotifers, cladocerans, ciliates, annelids), or to order (copepods). Eggs of rotifers, cladocerans, and copepods were enumerated separately.

2.4.3 Stable Isotope Ratios

The ratios of carbon (C) and nitrogen (N) stable isotopes in tissues of consumers reflect the stable isotope ratios (SIR) of their food sources (Neill and Cornwell 1992, France 1995). Therefore, SIR are useful in the determination of major food sources, as long as the latter have distinct isotopic ratios that allow them to be distinguished. Within the scope of the EMP, SIR analysis is used to estimate the relative importance of food sources including algae and wetland plants to the food web supporting juvenile salmonids at trends sites including Ilwaco Slough, Whites Island, Campbell Slough, and Franz Lake Slough. SIR are suitable for identifying food sources assimilated over a longer time frame compared to point-in-time techniques such as gut content analysis; ideally, a combination of the two approaches provides the best indicator of diet.

C and N isotope ratios yield different information: since the $^{13}C/^{12}C$ ($\delta^{13}C$) ratio varies by only a small amount (<1‰) during the assimilation of organic matter, it is used to identify the primary source of organic matter (i.e., primary producers). In contrast, the ratio of $^{15}N/^{14}N$ ($\delta\,^{15}N$) changes markedly with trophic level, increasing by 2.2 to 3.4 parts per thousand (per mil, or ‰) with an increase of one trophic level (i.e., from a plant to an herbivore or an herbivore to a carnivore). Thus, $\delta\,^{15}N$ values are useful in determining trophic position.

The SIR of C and N were measured in juvenile Chinook salmon muscle tissues and several potential food sources to provide information on the food web supporting juvenile salmonids (Table **13**). Juvenile salmon were collected by NOAA Fisheries staff during monthly beach seine sampling and frozen (see Section 2.6). Skinned muscle samples were collected for analysis since SIR signatures are more homogeneous within muscle tissue and since muscle is a good long-term integrator of the food source.

Aquatic invertebrates were collected using a 250 μm mesh net with a rectangular opening in emergent vegetation at the water’s margin. The aquatic midge, Chironomidae, and amphipods were selected because they have been found to be preferred food sources for juvenile salmonids in the lower Columbia River (Maier and Simenstad 2009, Sagar et al. 2013, 2014, 2015). Most invertebrate specimens were found attached to submerged portions of vegetation. Invertebrates were collected by rinsing the exterior of the vegetation with deionized water and removing the invertebrates from the rinse water using clean forceps. Invertebrate samples were then rinsed with deionized water to remove algae or another external particulate matter. Salmon and aquatic invertebrate samples were frozen for later processing.

Table 13. Potential food sources for marked and unmarked juvenile Chinook salmon and invertebrate consumers.

Potential food sources for fish (marked and unmarked)	Potential food sources for invertebrates
Chironomidae	Particulate organic matter
Amphipoda	Periphyton
Oligochaetes	Live vegetation
Nematodes	Dead vegetation
Gastropods	
Zooplankton	

	Hatchery food*

*Only applicable to marked fish

A variety of autotrophs were sampled to characterize the range of potential food sources for invertebrates. Samples of terrestrial and emergent vegetation, aquatic macrophytes, and macroalgae (*Ulva* and miscellaneous seaweeds) were collected from representative areas within each site. Vegetation samples were rinsed at least five times in deionized water to remove external material, such as invertebrates and periphyton, and were kept frozen (-20°C) for later processing. Samples of particulate organic matter (POM) and periphyton were filtered onto combusted 25 mm glass-fiber GF/F filters and frozen (-20°C) for later processing.

Frozen filters, salmon tissue, invertebrate, and plant material were freeze-dried using a Labconco FreezeZone 2.5 L benchtop freeze dry system (Labconco Corp., USA). Plants were categorized as live or dead during field collections based on whether they were attached and by their physical appearance; mixtures of live plants from the same sampling date were composited and ground using a mortar and pestle, as were mixtures of dead vegetation (designated when plant material was detached rather than rooted). Freeze-dried invertebrates of the same taxa from the same collection site and collection date were composited, ground using a clean mortar and pestle, and subsampled when enough material was present. Otherwise, whole bodies of all individuals of the same taxa from the same site were composited into a single sample. Skinned muscle tissue samples from individual juvenile salmonids were analyzed separately by the individual; muscle tissue samples from different bodies were not composited.

SIR of carbon ($\delta\ ^{13}C$) and nitrogen ($\delta\ ^{15}N$) were determined at the UC Davis Stable Isotope Facility using a PDZ Europa ANCA-GSL elemental analyzer interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (Sercon Ltd., Cheshire, UK). The atomic ratios of the heavy isotope (^{13}C, ^{15}N) to the light isotope (^{12}C, ^{14}N) were compared to universal standards (Vienna PeeDee Belemnite and air for C and N, respectively) and reported in per mil (‰) units.

To estimate the proportional contributions of different food sources for juvenile salmon, the stable isotope mixing model, simmr was implemented in R.

2.5 Macroinvertebrates

2.5.1 Salmon Prey Availability Sampling

2.5.1.1 *Open Water and Emergent Vegetation*

To assess the availability of salmon prey at the trends sites, we conducted neuston tows in both open water (OW; in the center of the channel) and emergent vegetation (EV; along the edge of the wetland channel among vegetation). For OW samples, a Neuston net (250 µm mesh) was deployed from a boat for an average distance of 100 m and positioned to sample the top 20 cm of the water column. For EV samples, the Neuston net was pulled through a 10 m transect parallel to the water's edge in the water at least 25 cm deep to enable samples from the top 20 cm of the water column. From 2008 – 2016, neuston tows were taken concurrently with monthly beach seine collections when juvenile Chinook salmon were present at a site (i.e., captured during seine sets). Beginning in 2017, neuston tows were completed during every beach seine collection regardless of whether salmon were captured. Two OW and two EV samples were collected at each site per month; although, occasionally one or three tows were performed in each habitat type depending on field conditions (Table 14). Samples were preserved in 95% ethanol. The monitoring protocol can be found on monitoringmethods.org (Method ID 1622).

Table 14. The number of invertebrate tow samples (OW and EV) collected at each site per sampling event, 2008-2013, and 2015-2018.

		Ilwaco Slough	Secret River	Welch Island	Ryan Island	Bradwood Slough	Jackson Island	Whites Island	Wallace Island	Lord/Walker Island	Burke Island	Goat Island	Deer Island	Campbell Slough	Lemon Island	Washougal	Sand Island	Franz Lake	Hardy Slough	Total Tow Samples
2008	April													3			6	6		15
	May													6				6		12
	June																		4	4
2009	May				3			4		4				5				4		20
	June													4						4
2010	April					4	4	4	4					4						20
	May					4	4	4	4					4						20
	June					4	4	4	4					4						20
	July					4		4	4					2						14
2011	April	2																		2
	May	8						10			4	4	4	4				2		36
	June	4						4												8
2012	February		4																	4
	March			2				2							3					7
	April		4	5				6							4	2				21
	May		1	4				4						4	4	4				21
	June		6	4				4						4	2	4				24
2013	March			4																4
	May		4	4				4						4						16
	June		4	4				3						4						15
	July			4				6												10
2015	April	5												6				6		17
	May			2				4						2				5		13
	June			6				4												10
2016	February			2				6										2		10
	March							2												2
	April			2				4						6				4		16
	May			4				4						4						12
	June			6				4						6						16
	July			4				6												10
	August			4																4
	September			4																4
(Table 13 continued)																				

		Ilwaco Slough	Secret River	Welch Island	Ryan Island	Bradwood Slough	Jackson Island	Whites Island	Wallace Island	Lord/Walker Island	Burke Island	Goat Island	Deer Island	Campbell Slough	Lemon Island	Washougal	Sand Island	Franz Lake	Hardy Slough	Total Tow Samples
2017	February	4						4						4						12
	March	4		4				4						4						16
	April	4		4				4						4						16
	May	4		4				4						4						16
	June	4		4				4						4						16
2018	February			4				4						4						16
	March	4		4				4						4				4		20
	April	4		4				4						4						16
	May	4		4				4						4						16
	June/July	4		4				4						4						16
	October	4		4				4						4						16
Total Tow Samples		39	23	81	3	16	12	117	16	4	4	4	4	96	13	10	6	35	4	487

2.5.1.2 *Benthic Macroinvertebrates*

To characterize the benthic macroinvertebrate assemblage, benthic core sites were selected to correspond to locations directly adjacent those where the fish community, food web metrics, and vegetation were sampled. Benthic cores were collected monthly at the trends sites (n = 5 per site) between April and July. Cores were collected to a depth of 10 cm by driving a 2-inch diameter PVC pipe into the ground at each sampling location. Each core was then placed in a jar and fixed in 10% formalin. Core samples were collected at low tide from exposed sediments and among emergent vegetation. The monitoring protocol can be found on monitoringmethods.org (Method ID 1593).

2.5.1.3 *Laboratory Methods*

Invertebrates collected in neuston tows and benthic cores were identified in the lab using high-resolution optical microscopy and taxonomic references (Mason 1993, Kozloff 1996, Merritt and Cummins 1996, Thorp and Covich 2001, Triplehorn and Johnson 2005). Most individuals were identified to family, although some groups/individuals were identified to coarser (e.g., order) levels. For each sample, the number of individuals in each taxonomic group was counted, then each group was blotted dry and weighed to the nearest 0.0001 g.

Analysis of neuston tow data included all invertebrates except the single-celled protist Foraminifera, which were abundant in samples from Ilwaco Slough in April. In benthic core samples, taxa that were not aquatic and/or benthic in their ecology (e.g., adult flies) were considered contaminants and were excluded from analyses of benthic core data.

2.5.2 Salmon Diet

2.5.2.1 *Field Data Collection*

When juvenile Chinook were captured at a site, fish were typically euthanized within an hour of collection. Fish were kept on ice until arrival at the NOAA field station laboratory where they were stored in a -80°F freezer. Chinook salmon bodies were necropsied at the end of the sampling season. Whole stomach samples were preserved in 10% formalin until delivered to the laboratory for processing. The total number of diet samples collected at the EMP sites since 2008 is provided in Table 15.

2.5.2.2 *Laboratory Methods*

Organisms in the diets were identified in most cases to the family level, although some groups/individuals were identified to coarser (e.g., order) levels, and crustaceans were usually identified to genus or species. Some contents were unidentifiable due to digestion. Each prey taxon was counted, blotted on tissue, and weighed to the nearest 0.0001 g.

Table 15. The number of Chinook salmon diet samples collected at each site per sampling event, 2008-2013, 2015-2018.

		Ilwaco Slough	Secret River	Welch Island	Ryan Island	Bradwood Slough	Jackson Island	Whites Island	Wallace Island	Lord/Walker Island	Burke Island	Goat Island	Deer Island	Campbell Slough	Lemon Island	Washougal	Sand Island	Franz Lake	Pierce Island	Hardy Slough	Total Tow Samples
2008	April													6			13	15	9		43
	May													19				7			26
	June																			13	13
2009	May				9			10		6				10				8			43
	June				10									9							19
2010	April					10	19	16	6					12							63
	May					17	15	14	14					24							84
	June					9	8	18	11					18							64
	July					10		19	11					15							55
	August					8		13													21
2011	May							10			10	13	10	22							65
	June							25													25
	July							2			2										4
2012	February		15	16																	31
	March			14				13							13						40
	April		15	14				10							7	15					61
	May			30				11						18	15	18					92
	June		14	15				15						15	15	36					110
2013	March			9																	9
	May		12	30				15						34							91
	June		1	23				13						9							46
	July		2	25										1							28
2015	April	6																			6
	May			15				15						15				4			49
	June			7				13													20
2016	April			13				13					7					12			45
	May			15				19					13								47
	July			3				8													11

		Ilwaco Slough	Secret River	Welch Island	Ryan Island	Bradwood Slough	Jackson Island	Whites Island	Wallace Island	Lord/Walker Island	Burke Island	Goat Island	Deer Island	Campbell Slough	Lemon Island	Washougal	Sand Island	Franz Lake	Pierce Island	Hardy Slough	Total Tow Samples
2017	February	2						2													4
	March	1						1													2
	April			15				8						1							24
	May			30				30						34							94
	June			32				5						23							60
2018	February			30				4													34
	March			30				30													60
	April			31				30													61
	May			30				30						32							92
	June/July													2							2
	October																				N/A
Total Tow Samples		9	59	427	19	54	42	412	42	6	12	13	30	319	50	69	13	46	9	13	1644

2.5.3 Salmon Prey Data Analysis

Descriptive statistical analysis of the whole invertebrate community was calculated, in addition to specific analyses of the order Diptera (flies) and amphipod taxa that have been shown to be important prey of juvenile Chinook salmon in the lower Columbia River (Lott 2004, Spilseth and Simenstad 2011). For benthic cores, the density and biomass of taxa in each sample were calculated as the total count or weight for a given taxon divided by the core volume (# individuals m^{-3}, g m^{-3}). For neuston tows, the density and biomass of taxa in each sample were calculated as the total count or weight for a given taxon divided by the meters towed (# individuals m^{-1} towed, mg m^{-1} towed). To compare taxa densities and biomass between study sites, density and biomass data for each taxon were summed across replicate samples taken within a given site each month and then divided by the number of replicates to give an average total density and biomass at each sampling site per month.

Multivariate analyses were used to examine differences in the invertebrate assemblage between sites using the PRIMER (Plymouth Routines In Multivariate Ecological Research) software package developed at the Plymouth Marine Laboratory (Clarke and Warwick 1994, Clarke and Gorley 2006). Taxa were initially combined into taxonomic groups for analysis of community composition. Similarity indices were calculated for the average site abundance of each invertebrate taxon using the Bray-Curtis similarity coefficient as a measure of the distance between sites. The density data were log transformed prior to analysis. A non-metric, multi-dimensional scaling (MDS) ordination plot was used to show similarity. The MDS plots observations as points such that those close together represent samples similar in community composition and points far apart correspond to different composition values.

Due to unequal variances and sample sizes, a Welch's ANOVA test was used to compare average log-transformed neuston densities and juvenile Chinook feeding rates across years and sites, respectively. Where a significant difference was found, the Games-Howell post hoc test was used to make pairwise comparisons.

Diet composition was assessed as the percent of the total index of relative importance (% IRI) for each taxon, as calculated in Liao et al. 2001, where:

$$\mathrm{IRI}_i = (\mathrm{P}_{i,_{\mathrm{numeric}}} + \mathrm{P}_{i,_{\mathrm{gravimetric}}}) \times \mathrm{FO}_i$$

and % IRI is the percentage of the total IRI for prey taxa *i*. In the equation, *P* is the percent numeric and gravimetric composition of total prey and FO is the percent frequency of occurrence of prey *i*. This index is recommended because it accounts for prey weight and numbers, as well as the likelihood of taxa appearing in the diet of individuals (frequency of occurrence; Liao et al. 2001). Because the index incorporates taxa counts, items that were not countable (e.g., plant matter, unidentifiable, highly-digested material), were removed from descriptive analyses of diet composition.

Instantaneous ration (IR) was calculated as a measure of fish condition or fitness. IR is the ratio of the total diet weight to the total fish mass. Total diet weight was calculated as the sum of the weights of all individual taxa counted in the diet, except that only nutritious diet items were included in IR calculations; sediment and plant matter were excluded. Energy ration (ER), was calculated as a measure of energy consumption. For each juvenile Chinook salmon, the sum of individual prey taxon masses were multiplied by the energy density (kJ g^{-1} wet mass) of each prey taxon, divided by the total fish mass. Thus, energy ration equals kilojoule consumed per gram of fish. Energy densities of prey taxa were compiled and acquired from David et al. (2016). For descriptive analyses, IR and ER was calculated for each individual salmon diet and averaged across all fish within a given habitat and month.

Following methods in Fiechter et al. (2015), maintenance metabolism was calculated for all juvenile Chinook salmon used in diet analyses between 2008-2013, and 2015-2016. Maintenance metabolism (J_M) represents the cost of metabolic upkeep and varies with temperature and body mass, such that:

$$J_M = j_m * e^{dT} * W$$

where j_m is the mass-specific maintenance cost at 0' C (0.003), d is the temperature coefficient for biomass assimilation (0.068), T is the temperature at the time of capture, and W is fish body mass. There were instances in June 2015 where fish were not weighed upon capture at Welch Island and Whites Island. Fork length (mm) was used to estimate the weight (R^2=0.972) for use in the analysis.

Maintenance metabolism and energy ration were plotted on a quadrant chart, divided by the 50th percentile, to evaluate the two metrics of potential growth together. For juvenile Chinook salmon, low metabolic cost and high energy assimilation represent relatively positive growing conditions (lower right quadrant), while high metabolic cost and low energy assimilation represent relatively poor growing conditions (upper left quadrant).

2.6 Fish

2.6.1 Fish Community

In 2019, NOAA Fisheries monitored habitat use by juvenile Chinook salmon and other fishes at four trends sites, Campbell Slough in Reach F (sampled from 2007-2019), Whites Island site in Reach C (sampled from 2009-2019), Welch Island in Reach B (sampled from 2012-2019), and Ilwaco Slough in Reach A (sampled from 2011-2019), in order to examine year-to-year trends in fish habitat use in the lower river. In 2019, sampling was attempted at a fifth trend site, Franz Lake in Reach H (sampled in 2008 – 2016, no sampling in 2017, March only in 2018) during April, but river levels were already too high for effective sampling. The site was not sampled again due to high river levels and warm temperatures. Coordinates of the sampling sites are shown in Table 16**Error! Reference source not found.**.

Fish were collected from March (Welch only) through June, and October 2019 using a 38 x 3-m variable mesh bag seine (10.0 mm and 6.3 mm wings, 4.8 mm bag). Bag Seine sets were deployed using a 17 ft Boston Whaler or 9 ft inflatable raft. Up to three sets were performed per sampling month, as conditions allowed. At each sampling event, the coordinates of the sampling locations, the time of sampling, water temperature, weather, habitat conditions, and tide conditions were recorded. Fish sampling events conducted as part of our regular EMP sampling in 2019 are shown in Table 16**Error! Reference source not found.**. The monitoring protocol can be found on monitoringmethods.org (Method ID 826). All non-salmonid fish were identified to the species level and counted. For salmonid species other than Chinook, up to 30 individuals were measured (fork length, nearest mm), weighed (nearest gram), and released. Up to 30 juvenile Chinook salmon were euthanized in the field, measured, weighed, and retained for subsequent laboratory analyses (diet, genetic, lipid, and otolith). If present, an additional 70 Chinook were measured and released. Any additional Chinook were counted and released. All salmonids were checked for adipose fin clips, or other external marks, coded wire tags, and passive integrated transponder tags to distinguish between marked hatchery fish and unmarked (presumably wild) fish.

Fish bodies retained in the field were frozen and stored at -80°C. At the end of the sampling, season fish were necropsied, and samples were collected for laboratory analyses. Stomach samples for taxonomic analyses were preserved in 10% neutral buffered formalin. Fin clips for genetic analyses were collected and preserved in alcohol, following protocols described in Myers et al. (2006). Otoliths for age and growth determination were also stored in alcohol. Whole bodies (minus stomachs) for measurements of lipids remained frozen until processed.

Table 16. Location of EMP sampling sites in 2019 and the number of beach seine sets per month. ns = not sampled.

Site	Feb	Mar	Apr	May	Jun	Oct	Total
Ilwaco Slough (Reach A) 46.300530° N, 124.045893° W	ns[1]	ns[1]	3	3	3	3	12
Welch Island (Reach B) 46.255011° N, 123.480398° W	ns[1]	3	2	1	2	2	10
Whites Island (Reach C) 46.159350° N, 123.340133° W	ns[1]	ns[1]	3	1	3	3	10
Campbell Slough (Reach F) 45.783867° N, 22.754850° W	ns[1]	ns[1]	3	2	2	3	10
Franz Lake (Reach H)* 45.600583° N, 122.103067W	ns[1]	ns[1]	0[2,3]	ns[3]	ns[4]	ns[5]	0
Total	0	3	12	7	10	11	43

[1] permit not issued[2] pole seine was attempted
[3] site flooded
[4] water temperature exceeded sampling criteria
[5] water too low

Fish species richness (*S*; the number of species present) and fish species diversity for each site were calculated by month and year. Fish species diversity was calculated using the Shannon-Weiner diversity index (Shannon and Weaver 1949):

$$H' = -\sum(p_i \ln p_i)$$

Where

p_i = the relative abundance of each species, calculated as the proportion of individuals of a given species to the total number of individuals in the community.

Catch per unit effort (CPUE) and fish density were calculated as described in Roegner et al. (2009), with fish density reported in number per 1000 m^2.

Multivariate analyses were used to examine differences in the fish community between sites using the PRIMER (Plymouth Routines In Multivariate Ecological Research) software package (Clarke and Warwick 1994, Clarke and Gorley 2006). A Bray-Curtis index of similarity coefficients was calculated for the square-root transformed species abundance data at each site. A non-metric, multi-dimensional scaling (nMDS) plot was used to graphically examine variation in the fish community between sites. We used a multivariate analog to ANOVA called analysis of similarity (ANOSIM) to quantitatively assess the variation in fish community based on site. The global R-value generated from this analysis indicates the degree of separation, with 0 representing no separation and 1 representing complete separation. ANOSIM also produces pairwise tests which compute an R-value for comparisons of different site locations. Statistical probabilities of both R-values are generated through permutation.

2.6.2 Salmon Metrics

2.6.2.1 *Genetic Stock Identification*

Genetic stock identification (GSI) techniques were used to investigate the origins of juvenile Chinook salmon captured in habitats of the Lower Columbia River Estuary (Manel et al. 2005, Roegner et al. 2010, Teel et al. 2009). From 2008–2013 juvenile Chinook salmon stock composition was estimated by using a regional microsatellite DNA data set (Seeb et al. 2007). Beginning in 2014 stock composition was estimated by using a Single Nucleotide Polymorphism data set that includes baseline data for spawning populations from throughout the Columbia River basin (described in Hess et al. 2014). The overall proportional stock composition of Lower Columbia River samples was estimated with the GSI computer program ONCOR (Kalinowski et al. 2007), which implemented the likelihood model of Rannala and Mountain (1997). Probability of origin was estimated for the following regional genetic stock groups: Deschutes River fall; West Cascades fall; West Cascades spring; Middle and Upper Columbia River spring; Spring Creek Group fall; Snake River fall; Snake River spring; Upper Columbia River summer/fall; Upper Willamette River spring; Rogue River fall; and Coastal OR/WA fall (Seeb et al. 2007, Teel et al. 2009, Roegner et al. 2010). West Cascades and Spring Creek Group Chinook are Lower Columbia River stocks. The monitoring protocols can be found on monitoringmethods.org (Method ID 948)(Method ID 1356)(Method 1332) (Method 5446).

Multivariate analyses were used to examine differences in the genetic stock groups between sites using the PRIMER (Plymouth Routines In Multivariate Ecological Research) software package (Clarke and Warwick 1994, Clarke and Gorley 2006). A Bray-Curtis index of similarity coefficients was calculated for the square-root transformed stock abundance data at each site. A non-metric, multi-dimensional scaling (nMDS) plot was used to graphically examine variation in genetic stock abundance between sites. We used a multivariate analog to ANOVA called analysis of similarity (ANOSIM) to quantitatively assess the variation in salmon stock composition based on site. The global R-value generated from this analysis indicates the degree of separation, with 0 representing no separation and 1 representing complete separation. ANOSIM also produces pairwise tests which compute an R-value for comparisons of different site locations. Statistical probabilities of both R-values are generated through permutation.

2.6.2.2 *Lipid Determination and Condition Factor*

As part of our study, we determined total, nonvolatile, extractable lipid (reported as percent lipid) and lipid class content in Chinook salmon whole bodies. Lipid content can be a useful indicator of salmon health (Biro et al. 2004) and also affects contaminant uptake and toxicity (Elskus et al. 2005). Studies show that the tissue concentration of a lipophilic chemical that causes a toxic response is directly related to the amount of lipid in an organism (Lassiter and Hallam 1990; van Wezel et al. 1995); in animals with high lipid content, a higher proportion of the hydrophobic compound is associated with the lipid and unavailable to cause toxicity. While lipids may help sequester toxins and protect fish from contaminants, an overabundance of lipids can interfere with buoyancy regulation during early ocean entry and may increase vulnerability to surface predators (Weitkamp 2008).

Prior to analyses, whole body samples from salmon collected in the field were composited by genetic reporting group, date, and site of the collection into a set containing 3-5 fish each. The composited salmon whole body samples (~ 2 g) were homogenized, mixed with drying agents (sodium sulfate and magnesium sulfate), packed into extraction cells, and then extracted with dichloromethane using an accelerated solvent extractor. The sample extracts were collected into pre-cleaned, pre-weighed sample tubes. Approximately 1-2 mL of sample extract was transferred to a pre-weighed sample vial to determine the amount of total, nonvolatile, extractable lipid (reported as percent lipid) by gravimetric analysis as described in Sloan et al. (2014). Another sample extract aliquot (1- 2 mL) was transferred to a second pre-weighed sample vial to measured lipid classes (i.e., sterol esters/wax esters, triglycerides, free fatty acids, cholesterol, phospholipids/polar lipids) using thin-layer chromatography–flame ionization detection (TLC–FID) (Ylitalo et al. 2005; Sloan et al. 2014). In this method, each sample extract was spotted on a silica rod (Chromarod) and developed in a chromatography tank containing 60:10:0.02 hexane:diethyl ether:formic acid (v/v/v). The lipid classes were separated based on polarity and measured using flame ionization detection, using the mean of two measurements. The percent contribution of each lipid class to the total lipid were calculated by dividing the concentration of each lipid class by the total lipid measured.

For all salmonid species, Fulton's condition factor (*K*; Fulton 1902; Ricker 1975) was calculated as an indicator of fish health and fitness, using the formula:

$$K = [\text{weight (g)}/\text{fork length (cm)}^3] \times 100$$

The monitoring protocol can be found on monitoringmethods.org (Method ID 952).

2.6.2.3 *Otoliths (Growth Rates)*

Otoliths from fish ranging in fork length from 35-111 mm (mean = 66 mm, SD = 14.4 mm) were processed for microstructural analysis of recent growth (see Chittaro et al. 2018). Specifically, left sagittal otoliths were embedded in Crystal Bond and polished in a sagittal plane using slurries (Buehler©'s 600 grit silicon carbide, 5.0 alumina oxide, and 1.0 micropolish) and a grinding wheel with Buehler© 1500 micropolishing pads. Polishing ceased when the core of the otolith was exposed, and daily increments (Volk et al. 2010, Chittaro et al. 2015) were visible under a light microscope. We photographed polished otoliths using a digital camera (Leica DFC450) mounted on a compound microscope (Zeiss©). Using Image Pro Plus© (version 7, Mediacybernetics), we took two measurements from each otolith; distance from otolith core to edge (i.e., otolith radius at time of capture, *Oc*) and distance from otolith core to seven daily increments in from the otolith edge (i.e., otolith radius measured at seven days before capture, *Oa*). For each individual, fork length at seven days prior to capture (*La*) was estimated using the Fraser-Lee equation:

$$La = d + \frac{Lc - d}{Oc} Oa$$

where d is the intercept (3.98mm) of the regression between fish length and otolith radius ($R^2 = 0.81$, n = 855) where Lc represents fork length (mm) at capture. Next, the average daily growth rate (mm/day) was calculated for an individuals' last seven days of life (a),

$$Average\ daily\ growth = \frac{Lc - La}{a}$$

Seven days of growth was a reasonable amount of time to estimate growth while in estuarine habitats because, depending on migratory type (i.e., ocean-type versus stream-type) and timing of migration (i.e., sub-yearling versus yearling migrant), Chinook salmon may inhabit estuaries for weeks or months (Healey 1991, Thorpe 1994, Weitkamp et al. 2014).

We used a generalized linear modeling (GLM) approach to investigate the extent to which variability in somatic growth rate (dependent variable) was explained by a suite of independent variables; collection year and day, river discharge, off-channel distance, river kilometer, genetic stock, hatchery or unmarked classification, and fork length. River kilometer and off-channel distance are defined as the distance (km) a site is from the mouth of the Columbia River and the distance (m) between a site and the Columbia River channel respectively. If an individual had a clipped fin or coded wire tag, then it originated from a hatchery and was categorized as “hatchery.” If a fish did not have a mark or tag, then the individual was labeled as “unmarked.” The term “unmarked” is used instead of “naturally produced” or “wild” because some hatcheries do not clip fins nor inject coded wire tags or mark only a fraction of their releases (Sagar et al., 2013).

For all models, we used a gamma family distribution with a log link to account for the normally distributed, but positive, growth rate data. Preliminary analyses indicated a nonlinear relationship between growth rate and day of the year, and therefore, the day of the year was also included in our analyses. In addition, fork length was included in our analyses so as to account for the linear relationship we observed between growth rate and fish size. We ran all possible GLM model combinations of the independent. All model parameters were estimated by maximizing the likelihood function. To compare models, we calculated four values for each model; Akaike’s information criterion (AIC), delta AIC, relative likelihood, and AIC weight. Smaller AIC values indicate “better” models, and when comparing two models, we calculated the difference in AIC values (delta AIC; Akaike, 1973; Burnham & Anderson, 2002). A delta AIC of less than 2 indicates little difference between competing models; a delta AIC of 2–10 indicates moderate support for a difference between the models, and a delta AIC of greater than 10 indicates strong support (Burnham & Anderson, 2002). Relative likelihood represents the likelihood of a model given the data, whereas AIC weight is the discrete probability of each model (Burnham & Anderson, 2002). The best model was defined as having a delta AIC of 0.00, although preference was given to the simplest model if two or more models had a delta AIC of less than 2.

2.6.2.4 *PIT Tag Array*

A passive integrated transponder (PIT) tag detection system has been operating at Campbell Slough since June 2011, with a hiatus in 2012 and 2017. It is located approximately 150 m into the slough channel from the mainstem Columbia River. The system consists of a Destron-Fearing FS1001-MTS multiplexing transceiver, which simultaneously receives, records and stores tag signals from six antennas measuring 4’ by 10’. The system is powered by a 470W solar array with battery backup and is also connected to a wireless modem that allows for daily data downloads. The array is intended to monitor the presence and to estimate residency of PIT tagged fish in Campbell Slough.

The previous detection system at Campbell Slough, consisting of two antennas measuring 4’ by 20’ was in place from 2011-2017. It was not operational in 2017 due to power cables having been severed by rodents and failed structural integrity of one of the antennas. We revamped the PIT detection array at Campbell Slough in 2018 by installing six antennas measuring 4’ x 10’. The antennas were arranged in a vertical “pass-through” configuration (Figure 9) which allowed greater detection capability at a larger range of water levels. An elevated platform was installed to keep the electronic telemetry equipment above potential water levels. The system continued to run a Destron-Fearing FS1001-MTS multiplexing transceiver and was powered by a 470W solar array with battery backup. A new modem was installed to update the equipment from 3G technology, which is no longer supported by cellular providers, to 4G technology. The location of the interrogation site was moved approximately 90 m further upstream.

Figure 9. Image of the new PIT detection system at Campbell Slough, installed February 2018.

In 2013, a second PIT detection system was installed near the confluence of Horsetail and Oneonta Creeks in the Columbia River Gorge where substantial restoration actions were completed. The Horsetail PIT detection arrays aids in evaluating the effectiveness of the restoration actions by monitoring use of the habitat by fish in the mainstem Columbia River (*Horsetail Restoration Project*). Antennas are located on both sides of the culvert allowing determination of whether salmon pass through the culvert to access the restored floodplain.

The array consists of a Biomark FishTRACKER IS1001-MTS distributed Multiplexing Transceiver System (MTS), which powers ten antenna units mounted within the culvert system at Horsetail/Oneonta Creek site (Columbia River, OR) beneath Interstate-84. The MTS unit receives, records and stores tag signals from these ten antennas, which all measure approximately 6’ by 6’ and are mounted on both ends of the 5-barrel culvert system running under the freeway. The system is powered by an 840 watt (W) solar panel array and supported by a 24-volt, 800 amp-hour battery bank back up. The unit is also connected to a fiber optic wireless modem that allows for daily downloads of tag data and system voltage monitoring updates.

3 Results

3.1 Mainstem Conditions

3.1.1 Continuous Data From the Mainstem

Mainstem conditions are evaluated through measures of river discharge at Bonneville Dam and at Beaver Army Terminal (river mile 53). In addition, temperature data and other variables are provided through in situ sensor measurements at Camas (river mile 122) and at Beaver Army Terminal (BAT).

3.1.1.1 *Discharge at Beaver Army Terminal (RM-53)*

River discharge at BAT is shown in Figure 10**Error! Reference source not found.**. BAT discharge includes inputs from tributaries, including the Willamette River (Figure 12), in addition to flows from the Columbia River. In 2019, the pattern of flows was similar to that observed in 2018 and 2010, with small to moderate peaks in January and February and a large peak in April. The April peak in 2019 was larger than in 2018 or 2010 and was the highest in the 2010-2019 period.

Flows associated with the freshet at BAT were close to the average for the 2010-2019 period. The period of high flow, however, was short. From mid-June through the fall, flows were close to the 8-year minimum. Unlike in 2017, when late-autumn flows at BAT were high relative to other years—which was not observed in the data from Bonneville (Figure 11**Error! Reference source not found.**)—flows were low at both sites during the summer and autumn. Thus, winter river discharge at BAT was relatively high, as was the peak freshet, relative to other years; however, the decline in discharge following the freshet was rapid and flowed in the summer and autumn were very low.

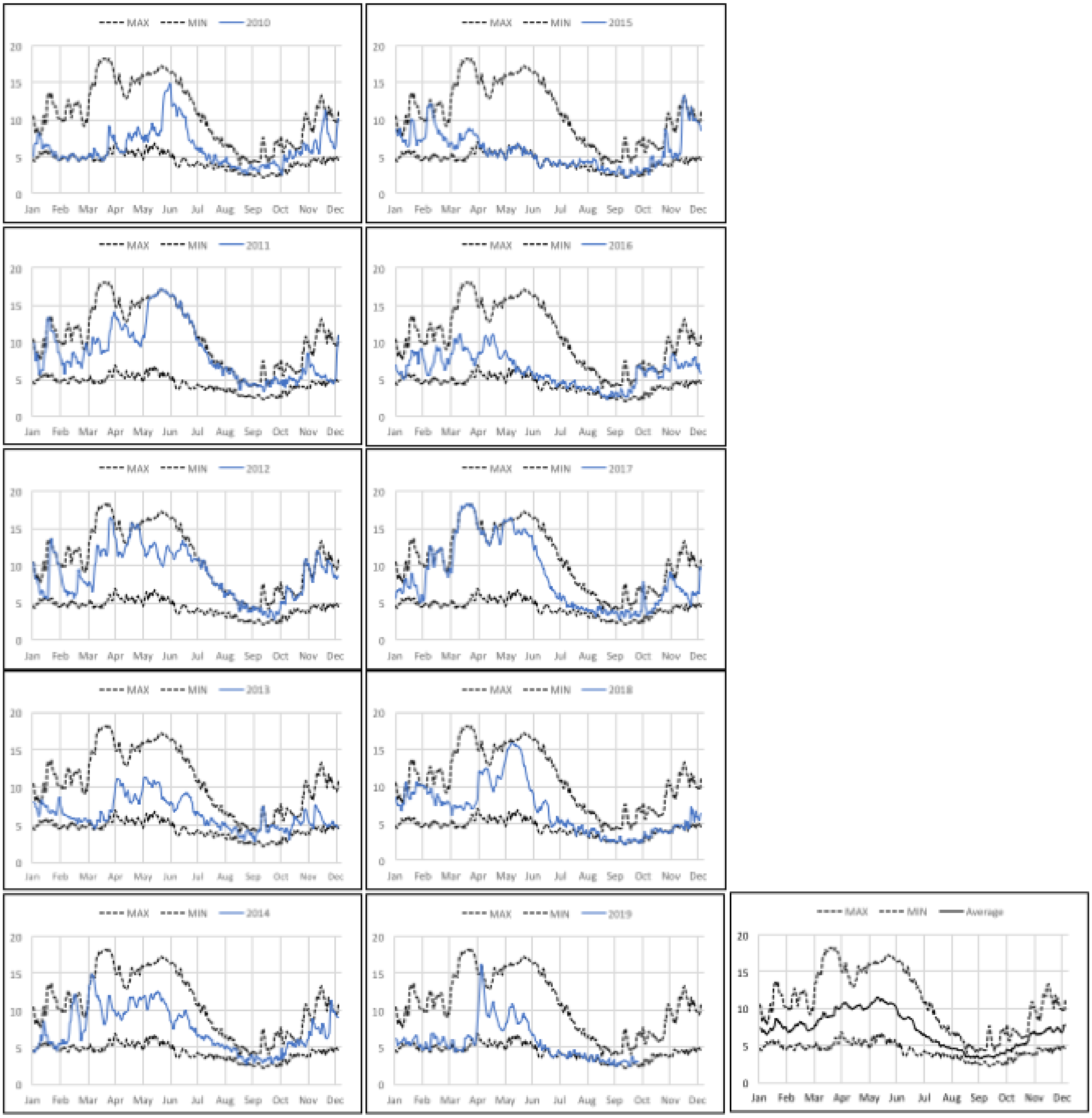


Figure 10. Daily water discharge (m^3/s) at Beaver Army Terminal (RM-53) from 2010-2019. Panels show individual years (blue lines) with the daily maximum and minimum indicated (upper and lower dashed lines) in each panel. The final panel shows the maximum (upper dashed line), minimum (lower dashed line)

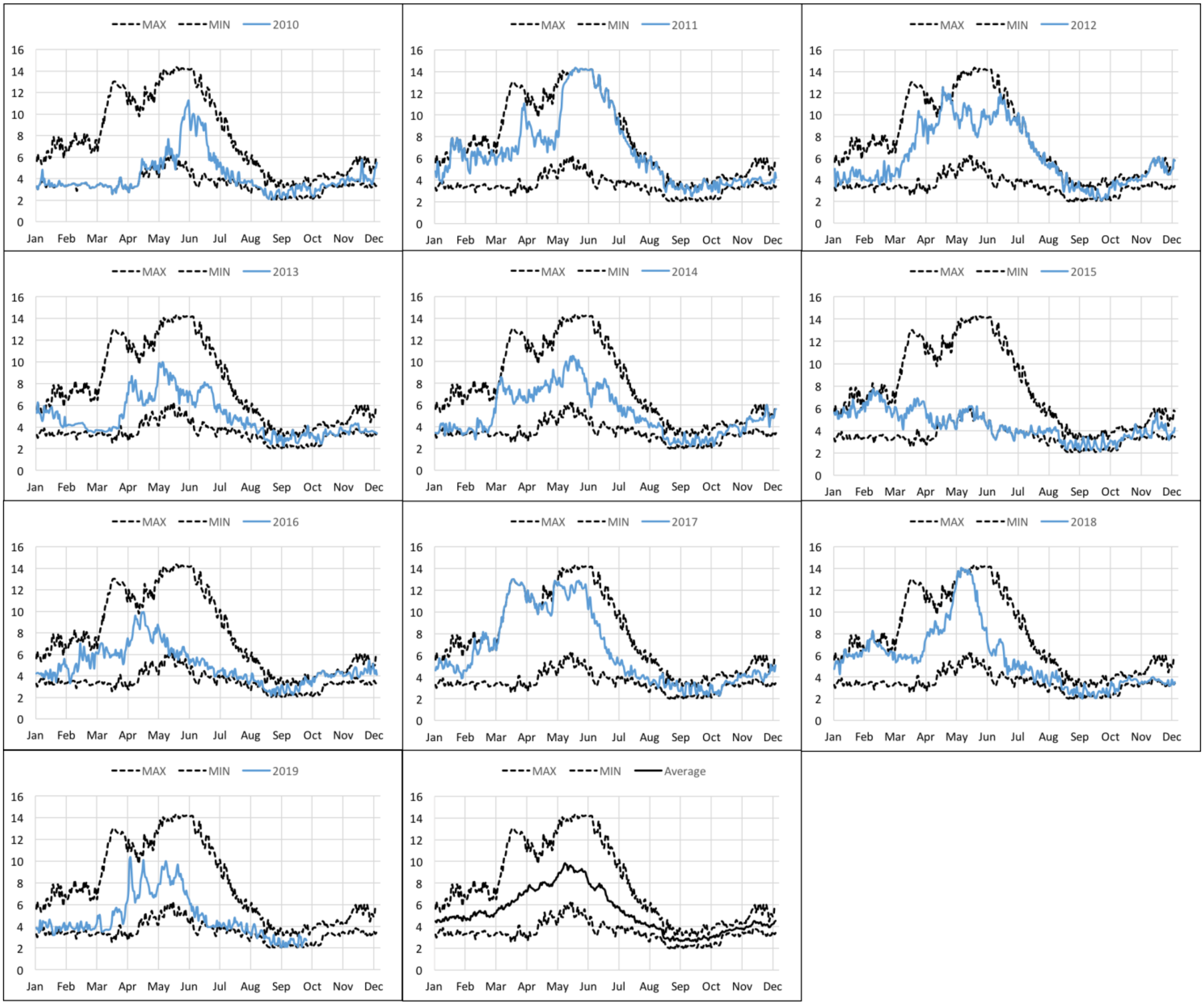


Figure 11. Daily water discharge (m^3/s) at Bonneville Dam from 2010-2019. Panels show individual years (blue lines) with the daily maximum and minimum indicated (upper and lower dashed lines) in each panel.

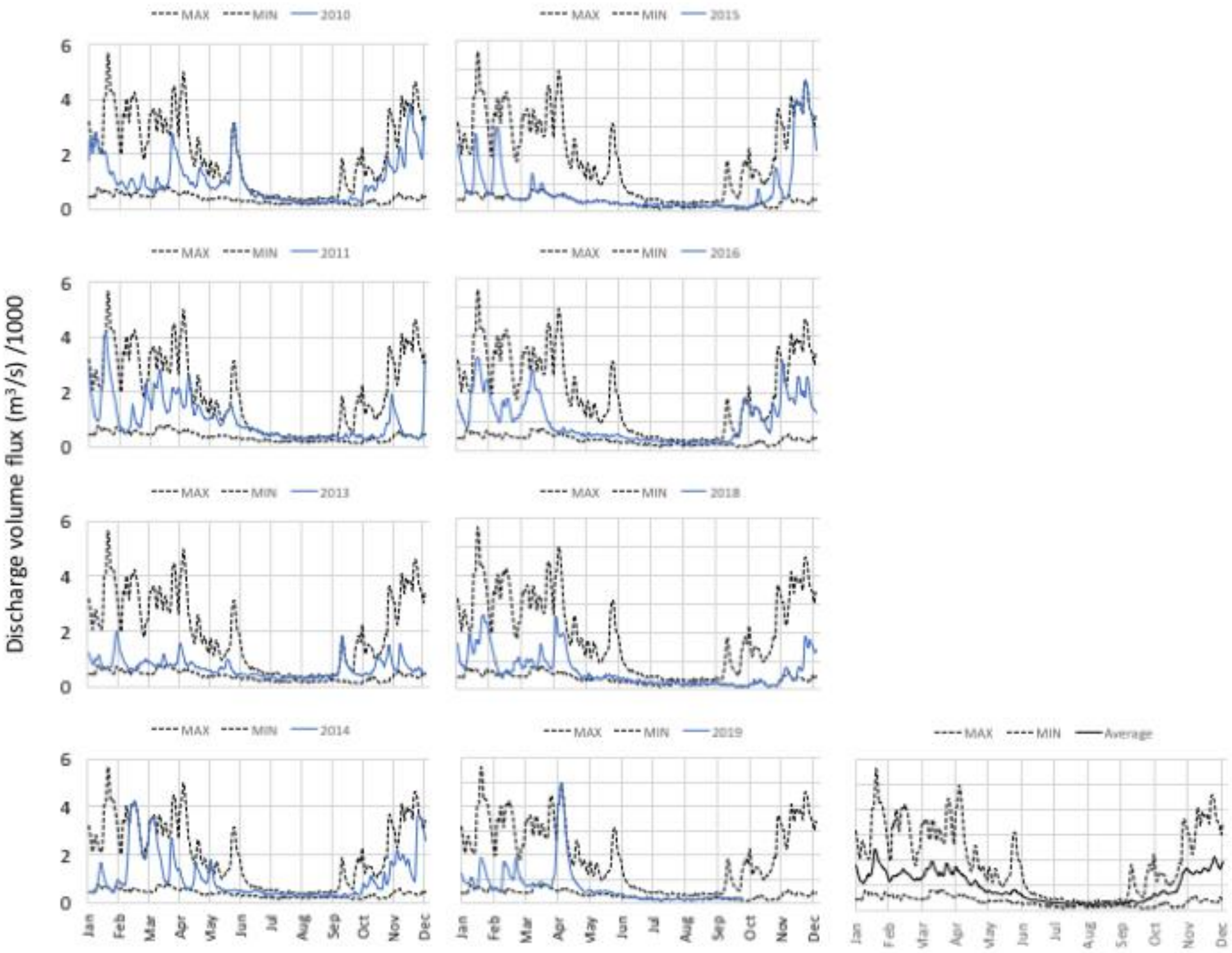


Figure 12. Daily river discharge of the Willamette River measured near the Morrison Bridge for years 2010–2019. Also shown are the daily maximum and minimum values for the years 2010–2019. Data from USGS 14211720.

Time series of river discharge associated with tributaries excluding the Willamette River are shown in Figure 13**Error! Reference source not found.**. Discharge fluxes were computed from the difference between observations at Beaver Army Terminal and observations at Bonneville Dam and the Willamette River near the Morrison Bridge (i.e., Tribs = BAT – Bonneville – Willamette). These data show that the discharge associated with small tributaries follows the patterns observed in the Columbia River during the spring and the Willamette River during the winter, with moderate-to-low flows associated with the winter and spring and a short-lived freshet. Tributary flows were among the lowest in the 8-year data set.

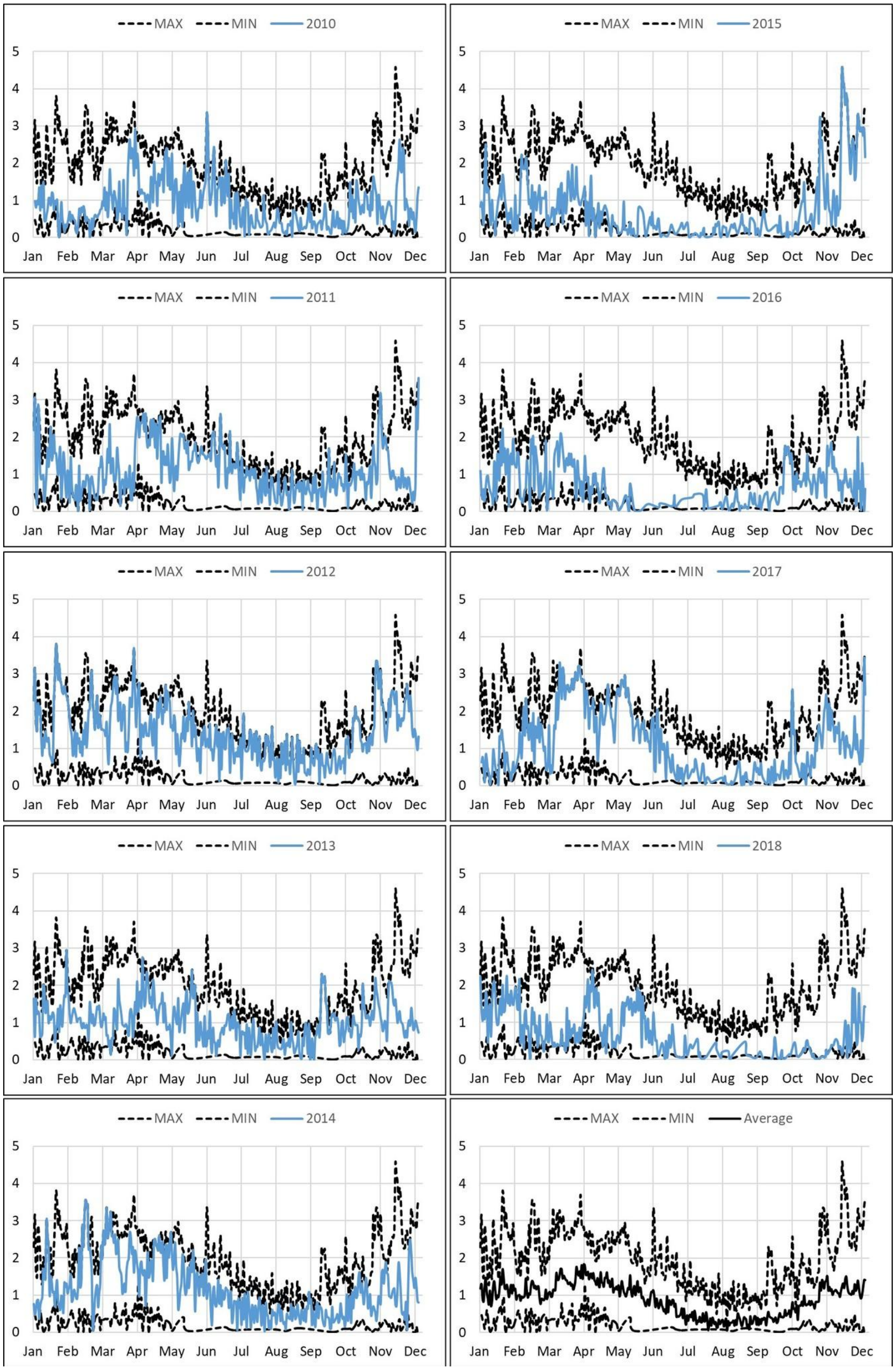
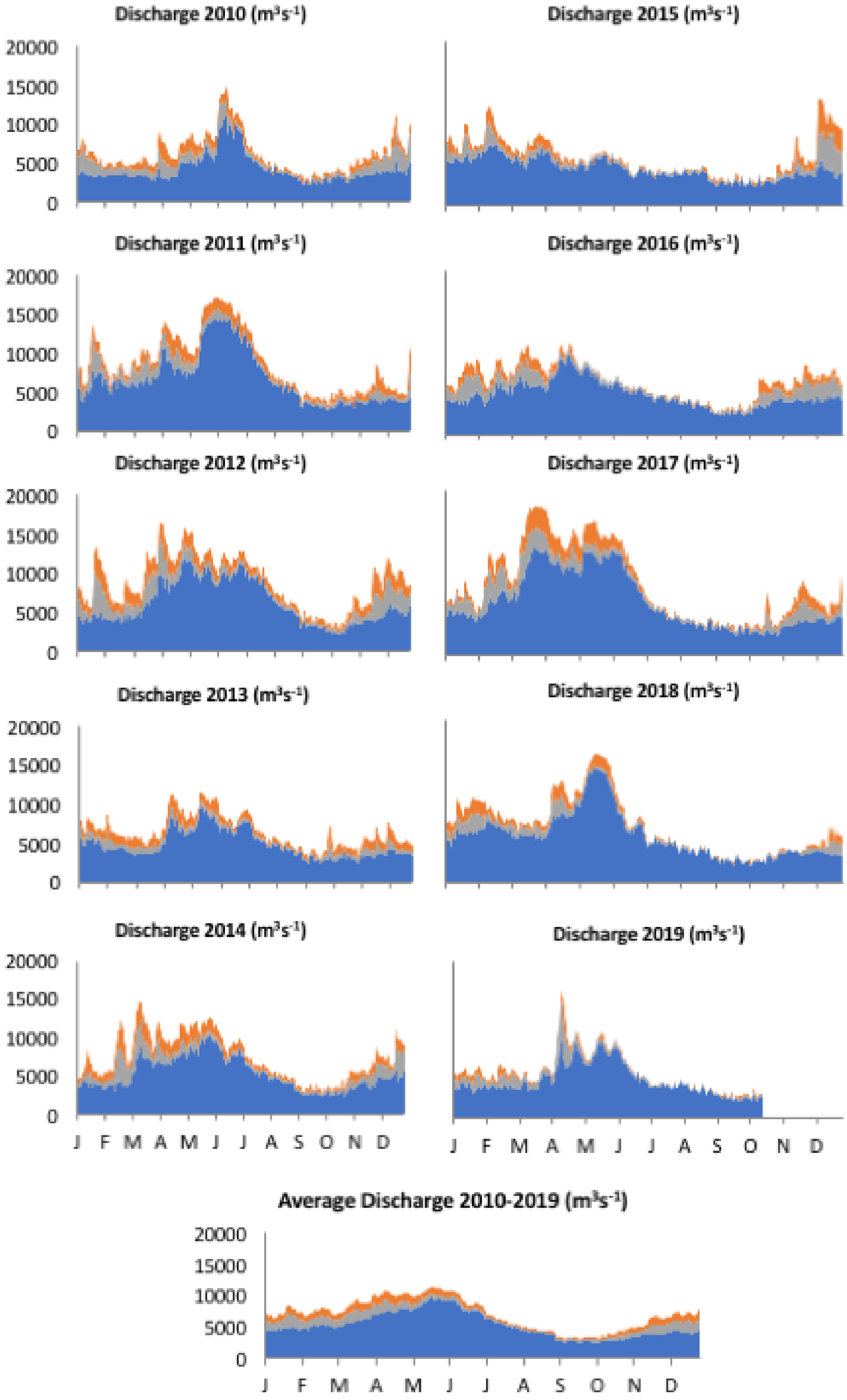


Figure 13. Estimated daily water discharge (m^3/s) associated with tributaries (excluding the Willamette River). Panels show individual years from 2010–2018 (blue lines) and the daily maximum and minimum values.

The proportion of flow associated with the Columbia River, the Willamette River, and other tributaries at Beaver Army Terminal (RM-53) for years 2010–2019 are shown in Figure 14**Error! Reference source not found.** and Figure 15**Error! Reference source not found.**. The Columbia River accounts for the largest proportion of flow throughout the year; however, during the winter months, flows from the Willamette River increase in relative importance, as do flows from other tributaries (Figure 15**Error! Reference source not found.**). River discharge in the early spring of 2019 was lower than average and composed of a relatively small fraction of flow from the Willamette River and tributaries, which influences water quality parameters in the mainstem, including nutrients, turbidity, and colored dissolved organic matter (see later sections). Compared to previous years, the proportional flow from the Willamette and other tributaries was very low and characterized by the absence of distinct peaks during the winter (January through March). The initial phase of the spring freshet had a large contribution from the Willamette and other tributaries; however, by May flows were again strongly dominated by the Columbia. The plots in Figure 15**Error! Reference source not found.** show more closely how the fractional composition of river discharge varies over the year among the years investigated, highlighting the low contribution from tributaries in 2019. The temporal extent of the freshet was similar in 2019 to 2018, with more gradual increases and decreases in flow compared to some years, such as 2010, 2011, and 2018.



Discharge 2010 (m^3s^{-1})

Discharge 2015 (m^3s^{-1})

Discharge 2011 (m^3s^{-1})

Discharge 2016 (m^3s^{-1})

Discharge 2012 (m^3s^{-1})

Discharge 2017 (m^3s^{-1})

Discharge 2013 (m^3s^{-1})

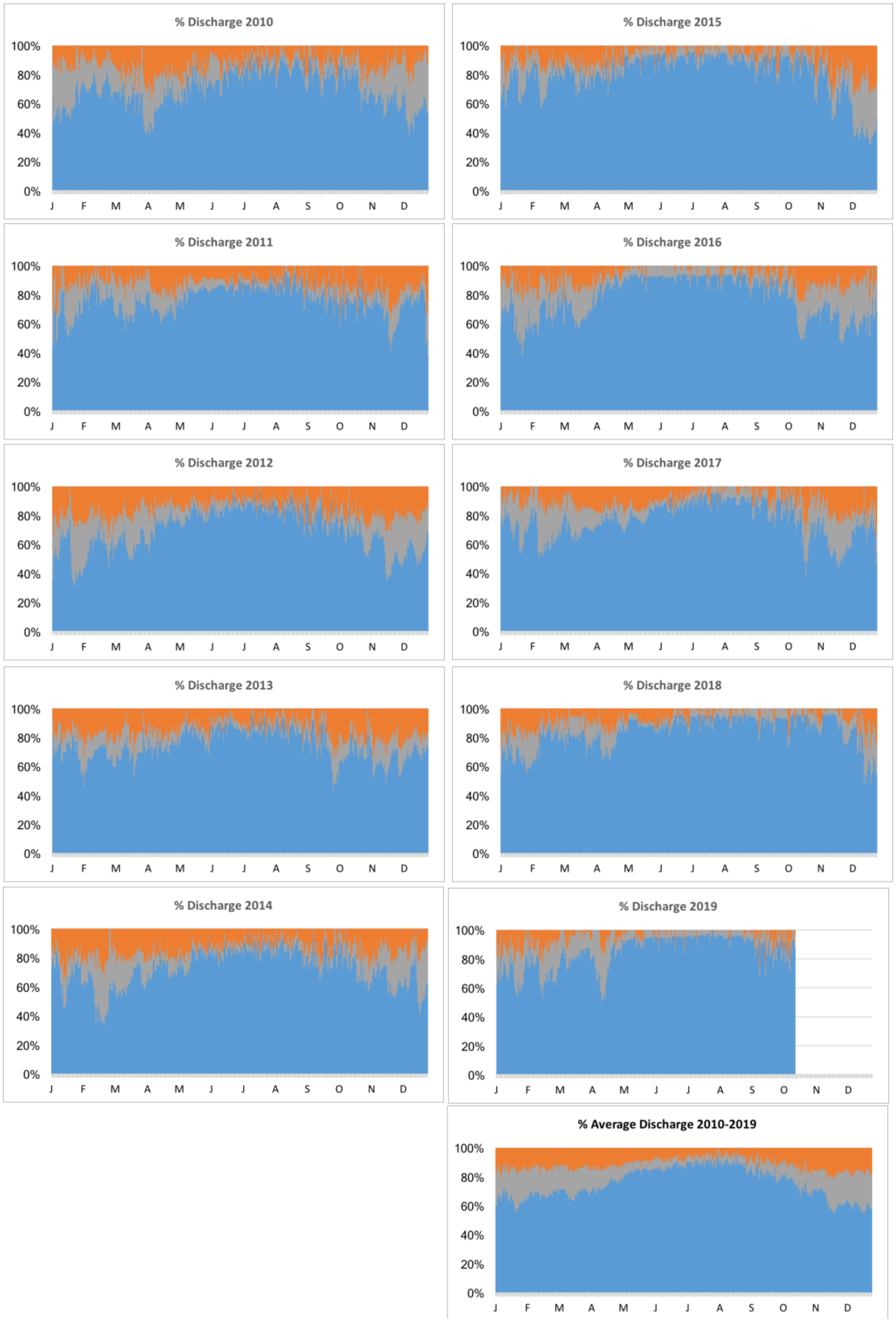
Discharge 2018 (m^3s^{-1})

Discharge 2014 (m^3s^{-1})

Discharge 2019 (m^3s^{-1})

Average Discharge 2010-2019 (m^3s^{-1})

Figure 14. Daily discharge fluxes (m^3/s) associated with Columbia River flow (blue), Willamette River flow (orange), and other tributaries (grey). Discharge from the Willamette was determined at the USGS stream gage at the Morrison Bridge; the contribution from other tributaries was computed by subtracting flows observed in the Willamette from those in the Columbia.


% Discharge 2010

% Discharge 2015

% Discharge 2011

% Discharge 2016

% Discharge 2012

% Discharge 2017

% Discharge 2013

% Discharge 2018

% Discharge 2014

% Discharge 2019

% Average Discharge 2010-2019

Figure 15. Daily percentage contributions of the Columbia (blue), Willamette (orange), and other tributaries (grey) to total river discharge observed at Beaver Army Terminal (RM 53) for the years 2011-2019. Discharge from the Willamette was determined at the USGS stream gage at the Morrison Bridge; the contribution from other tributaries was computed by subtracting flows observed in the Willamette from those in the Columbia.

3.1.1.2 *Water Temperature in the Mainstem at Camas (RM-122)*

We showed in previous reports that mainstem hourly temperatures did not vary substantially between Camas (RM-122) and Beaver Army Terminal (RM-53); data from Camas are shown here. Since temperature is an important variable that influences organismal physiology and particularly the performance and survival of salmonids, we compare the number of days in 2019 where temperatures exceeded thresholds associated with reduced performance or physiological stress with the years 2013, 2014, 2015, and 2016 (Figure 16**Error! Reference source not found.**). The number of days was computed by summing the number of hours for which a threshold was exceeded and then dividing by 24 to produce a day-equivalent. According to Oregon State Water Quality Standards (code 340-041) and Washington State Water Quality Standards (code 173-201A), water temperature should be less than 16^{o}C for optimal performance; rearing and migratory habitats should be less than 18.0°C (Oregon standards) or less than 17.5°C (Washington standards). The Columbia is considered spawning/rearing habitat (Washington State Water Quality Standards) between the mouth and rkm 497 (Oregon–Washington border). Within the migration corridor, temperatures should be less than 20°C (Oregon standards), with a recommendation that water bodies have cold-water refugia having temperatures at least 2^{o}C colder than the daily maximum temperatures of the adjacent water body that are sufficiently distributed to allow salmon/steelhead migration without significant adverse effect. It is recommended that the Columbia River may not exceed a one-day maximum of 20.0 °C (Washington standards). According to recommendations from DEQ/OWEB, year-round temperatures should not exceed 18 ^{o}C (Kidd, 2011), with an ideal range of 7.2-15.6 ^{o}C for healthy adults and 12.2-13.9 ^{o}C for healthy juveniles (Kidd, 2011).

The most conservative threshold (19 ^{o}C) was exceeded on 80 days in 2019, similar to many of the previous years (2013-2018), with the notable exception of 2015. There were similar numbers of days exceeding 20^{o}C and 21^{o}C compared to previous year; however, the number of days with temperatures exceeding 22^{o}C or 23^{o}C was lower than in any previous year, with only 7 days exceeding 22^{o}C and no days exceeding 23^{o}C.

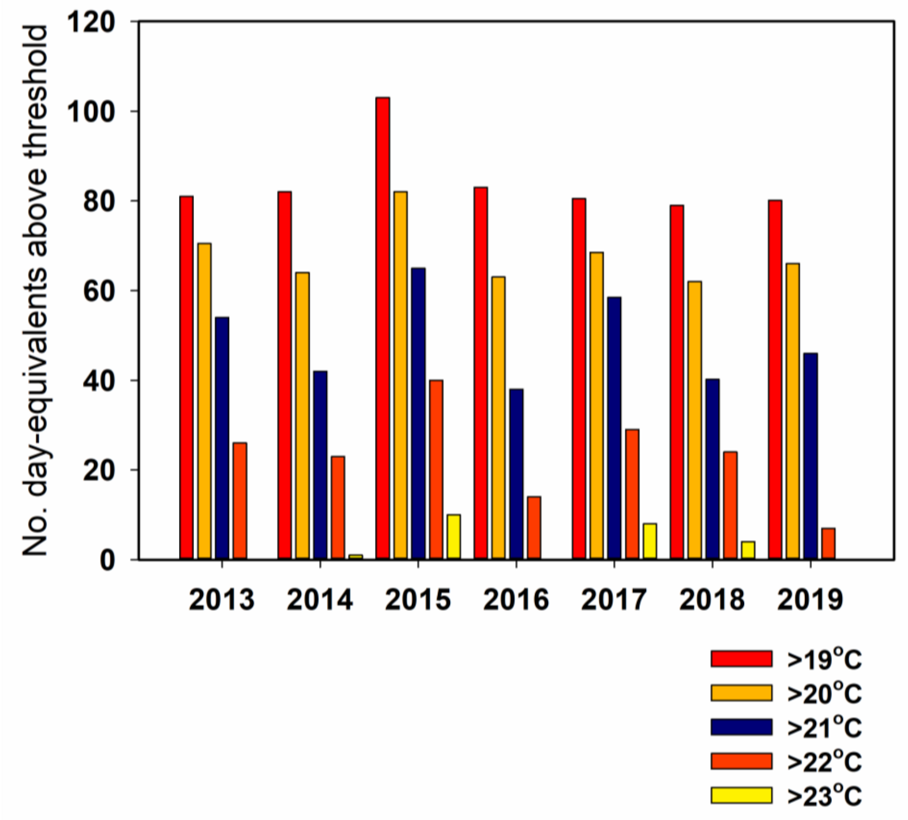


Figure 16. Number of days (computed from hourly data and converted to days, i.e., day-equivalents) where water temperatures exceeded the indicated temperature thresholds in the Columbia mainstem at Camas, WA (RM-122) (19 °C, 20 °C, 21 °C, 22 °C, 23 °C between 2013 and 2019).

3.1.1.3 *Water Quality Parameters in the Mainstem*

Time series of water quality parameters measured hourly at RM-122 (Camas) are shown in Figure 17**Error! Reference source not found.**. The difference in discharge between RM-122 and Beaver Army Terminal (RM-53) is highlighted to show the contribution of tributaries, which is focused on the late-autumn, winter, and spring periods. Similar to 2018, the tributaries had a relatively large contribution to total discharge during the peak in April 2019, as well as during January-February. Peaks in tributary flow are associated with peaks in colored dissolved organic matter (CDOM), turbidity, and nitrate, underscoring the influence of water source on water quality parameters. Chlorophyll *a*, a proxy for the contribution of fluvial phytoplankton to primary production in the river, peaked in April-May as well as in June-July, in association with changes in river discharge.

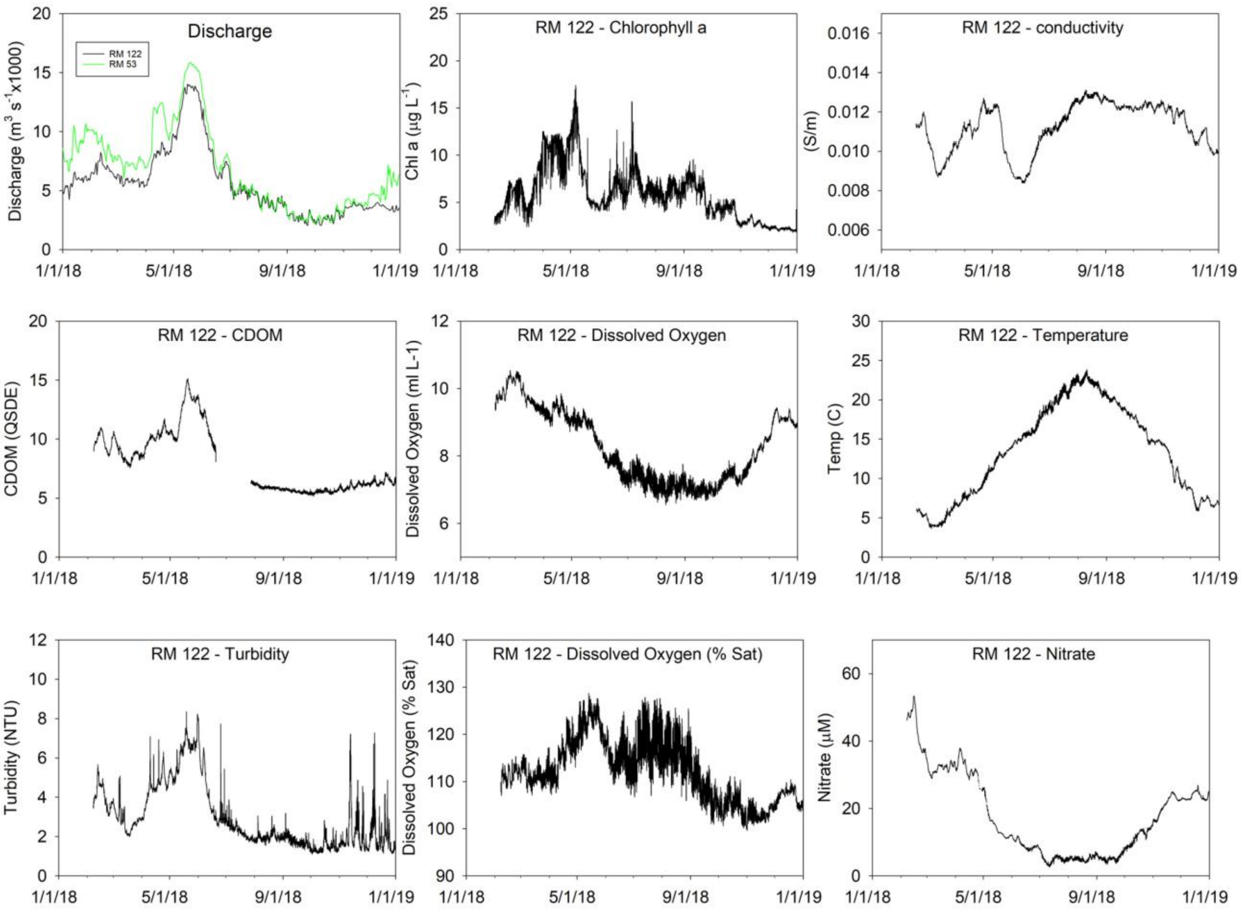


Figure 17. Time series of hourly water quality parameters measured at River Mile 122 (Camas, WA) in 2018 - 2019.

The percent saturation of dissolved oxygen (i.e., saturation relative to atmospheric equilibrium) reflects the balance between oxygen produced through photosynthesis and oxygen consumed through respiration. Dissolved oxygen saturation exceeded 100% for nearly the entire year, with greater day-to-day variability observed during the summer months (Figure 17**Error! Reference source not found.**).

Dissolved nitrate concentrations were lower in 2019 compared to previous years during peaks flows; in 2017, nitrate exceeded 50–60 µM in March–April, declining rapidly during the month of April when phytoplankton growth was strong (as evidenced by the increase in chlorophyll *a* during the same period). In 2018, although nitrate was high in February (~50 µM), concentrations were ~20-40 µM during the period of strong growth of phytoplankton in the spring. The time series of nitrate concentrations closely matches that of dissolved oxygen, partly due to drawdown by primary production and partly due to the contribution of winter sources. Even though nitrate concentrations were lower than in 2017, they were still frequently higher than the recommended benchmark for good water quality (<0.399 mg L^{-1}, or 28.5 µM; Oregon's National Rivers and Streams Assessment 2008-2009).

3.2 Abiotic Site Conditions

3.2.1 Continuous Water Quality

Measurements of water quality parameters were made every 30 minutes at five trends sites (Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough) using sensor packages moored at fixed depths. From these data, daily averages were computed to look for seasonal trends.

3.2.1.1 *Ilwaco Slough*

Water temperature at Ilwaco peaked at just over 20°C in July and August (**Figure 18**). Ilwaco is strongly influenced by tidal exchange with marine waters from the coastal ocean, particularly in the summer months. Salinity, which is computed from conductivity, is the clearest indicator of this influence: conductivity was highest between late June and mid-September. The maximum salinity (17 PSU) was observed in late July-early August, while minimum values were observed between April and June, in association with the spring freshet. The daily range in dissolved oxygen saturation was higher prior to and following the spring freshet, likely reflecting the higher levels of primary production associated with the lower river flows. The daily percent saturation of oxygen was between 60 and 80% in June, similar to 2018. pH values were high relative to previous years in April, when values greater than 8.5 were observed in a few cases. In general, values fluctuated between 6.6 and 8.0, with daily values falling between 7.0 and 8.0. These values fall within the recommended range for good water quality (6.5–8.5; Washington State Water Quality Standards).

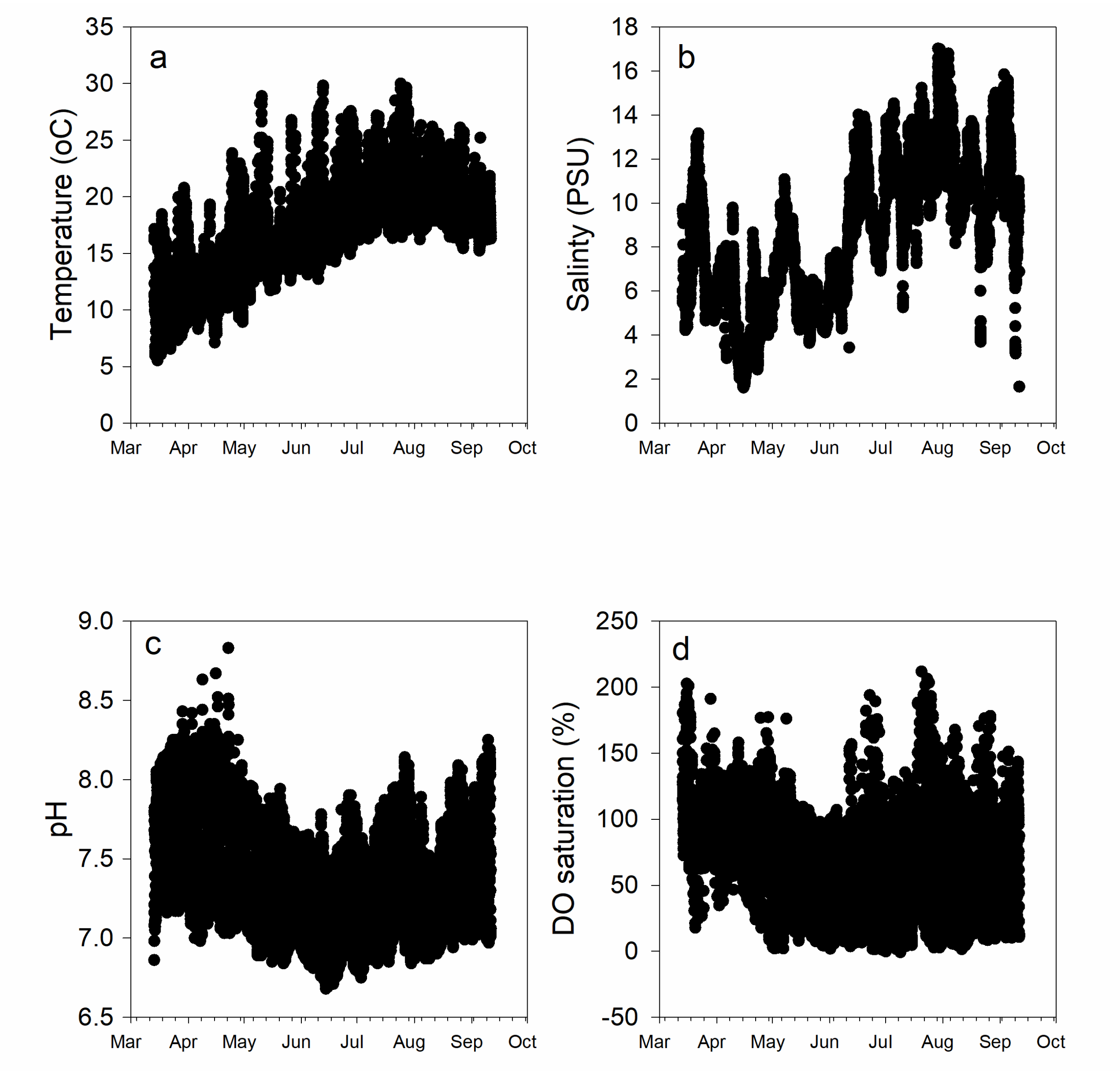


Figure 18. Time series of daily averaged measurements of water quality parameters made at Ilwaco Slough, 2019.

3.2.1.2 *Welch Island*

Maximum summer temperatures at Welch Island reached nearly 23°C in August, which was slightly higher than observed at Ilwaco (Figure 19**Error! Reference source not found.**). Dissolved oxygen saturation was >90% from March through the middle of June; after that, there was greater variation in DO saturation (between mid-June and September), with values dipping to ~70% on a few occasions. In general, patterns in dissolve oxygen saturation in 2019 were similar to 2018. Conductivity was highest when river flow was dominated by pluvial processes rather than snowmelt-driven flow; following the decline in freshet flows, levels increased slowly over the summer months (Figure 19). pH ranged from 7.0 to 7.9 in 2019, which falls within the recommended range (6.5–8.5; Washington State Water Quality Standards). Chlorophyll *a* was highest in early April and early July, with relatively high values between late May and mid-July; minima in chlorophyll fluorescence were observed in associated with the freshet and with warm, late-summer conditions. Phycocyanin fluorescence was low (<1 RFU) throughout 2019.

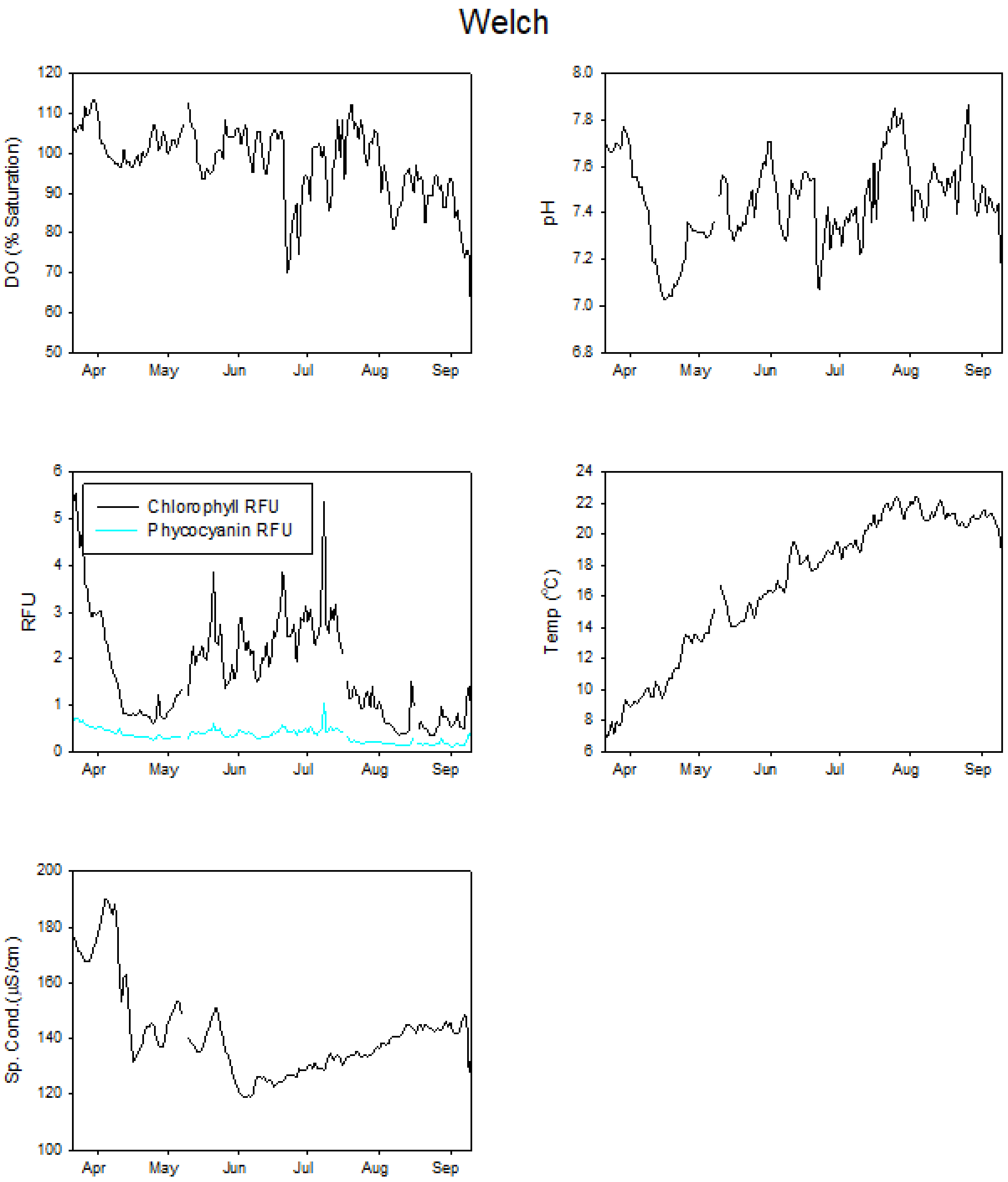


Figure 19. Time series of daily averaged measurements of water quality parameters made at Welch Island, 2019.

3.2.1.3 *Whites Island*

Dissolved oxygen saturation at Whites Island was >90% until late-August; after that, there were wide variations in oxygen saturation that persisted into September, peaking at ~130%. Similarly, daily variations in pH were relatively small over the same time period, ranging from just over 7.0 to ~7.9; by late August, there were a number of peaks in pH with values between 8.0 and 8.8. A change in the frequency of peaks was also observed in the temperature data from late-August into September. Specific conductivity was highest before the freshet, when river flows were dominated by pluvial processes and runoff; after mid-June, there was a slow increase in conductivity values as river water temperatures increased.

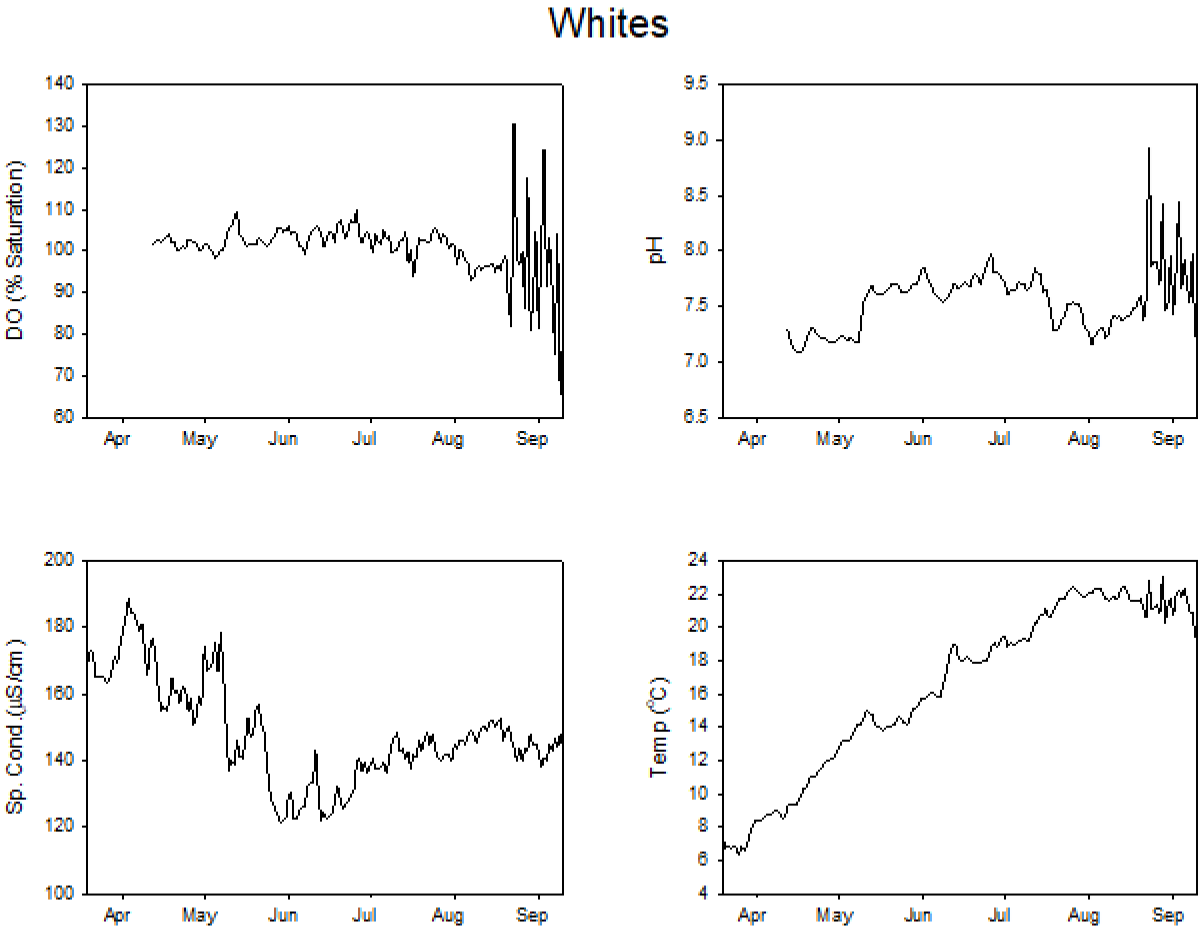


Figure 20. Time series of daily averaged measurements of water quality parameters made at Whites Island, 2019.

3.2.1.4 *Campbell Slough*

Of the five off-channel trends sites, Campbell Slough had the highest summer water temperatures, with values exceeding 25°C from mid-July to mid-August (Figure 21). The influence of river flow is evident in the increased depth that coincides with the spring freshet. All biogeochemical properties reflect the influence of the freshet, including a reduction in chlorophyll, dissolved oxygen saturation, conductivity, temperature, and pH. pH increased to values close to 10 by mid-July, which exceeds water quality benchmarks (Washington State Water Quality Standards). Chlorophyll concentrations observed at Campbell Slough were below the recommended benchmark of 15 µg L^{-1} (based on three samples collected over three consecutive months; Washington State Water Quality Standards). However, chlorophyll concentrations approached this level in July.

There were dramatic fluctuations in the percent saturation of dissolved oxygen throughout the year at Campbell Slough, indicating high biological activity at this site. Values dropped to as low as 60% relative to atmospheric values during the spring freshet and reached ~145% consistently between late June and early August (Figure 21**Error! Reference source not found.**).

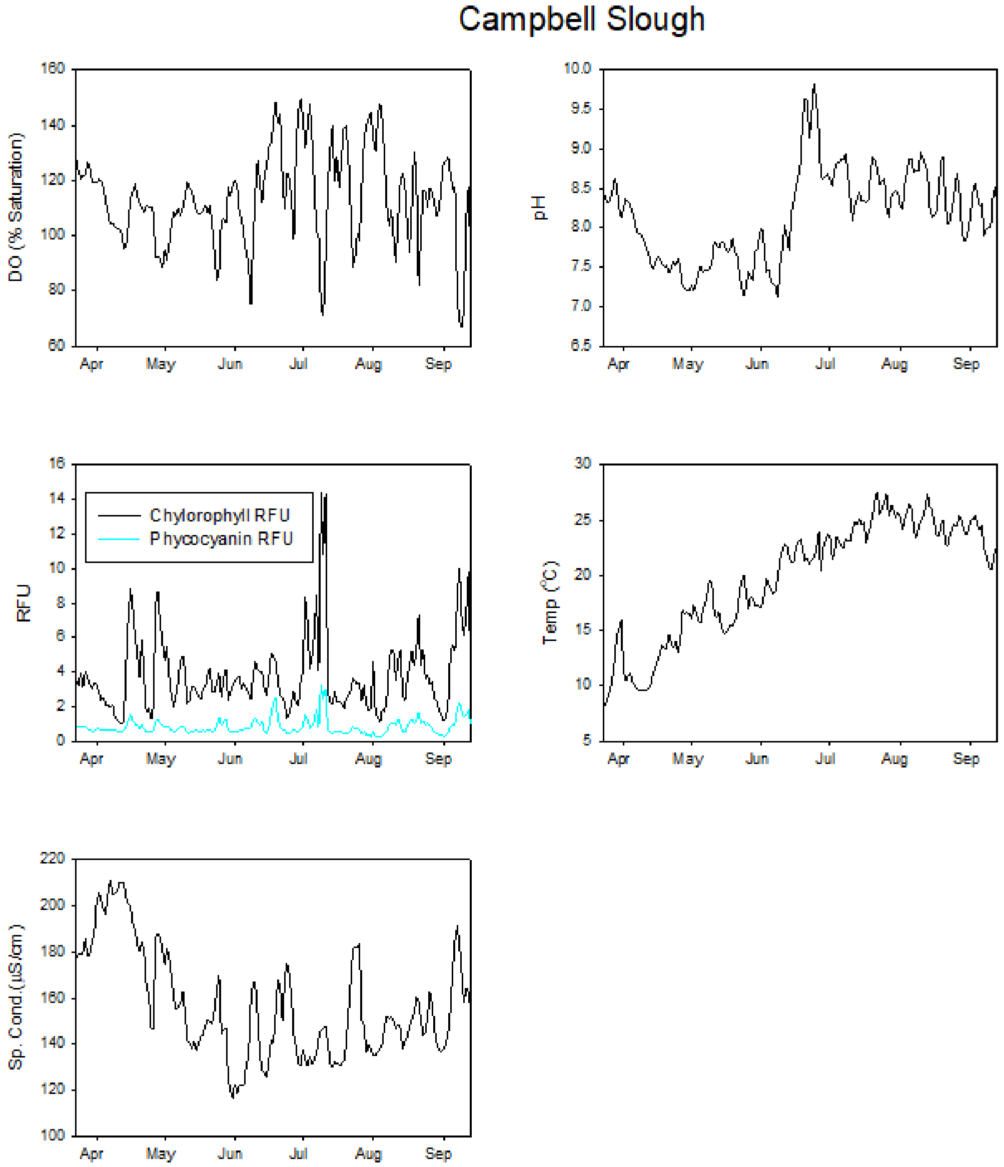


Figure 21. Time series of daily averaged measurements of water quality parameters made at Campbell Slough, 2019.

3.2.1.5 *Franz Lake Slough*

Summer temperatures at Franz exceeded 20°C from late-July through mid-August, with several days reaching 24^{o}C (Figure 22). Peaks in chlorophyll and phycocyanin fluorescence were observed in late May and in late June; there was a notable peak in phycocyanin in late August, which was ~5 times higher than any other observation from the other sites. Unlike in 2018, when spring primary production—as inferred from chlorophyll fluorescence—occurred in a large, long-lived peak lasting from mid-April to mid-May, in 2019 the peaks were sharp and short-lived. The longest-lived peak occurred in early to mid-May, prior to the spring freshet. Chlorophyll concentrations observed after late June were lower than those observed from May through June. Conductivity values were higher in early spring than in late spring or summer, with values declining slowly from June through September. Dissolved oxygen saturation values generally declined from spring to summer, with the exception of a period associated with higher values in late August, coincident with the peak in phycocyanin (Figure 22). Finally, pH values were within the limits set for good water quality throughout the spring and summer, where they ranged from 6.5 to 7.5.

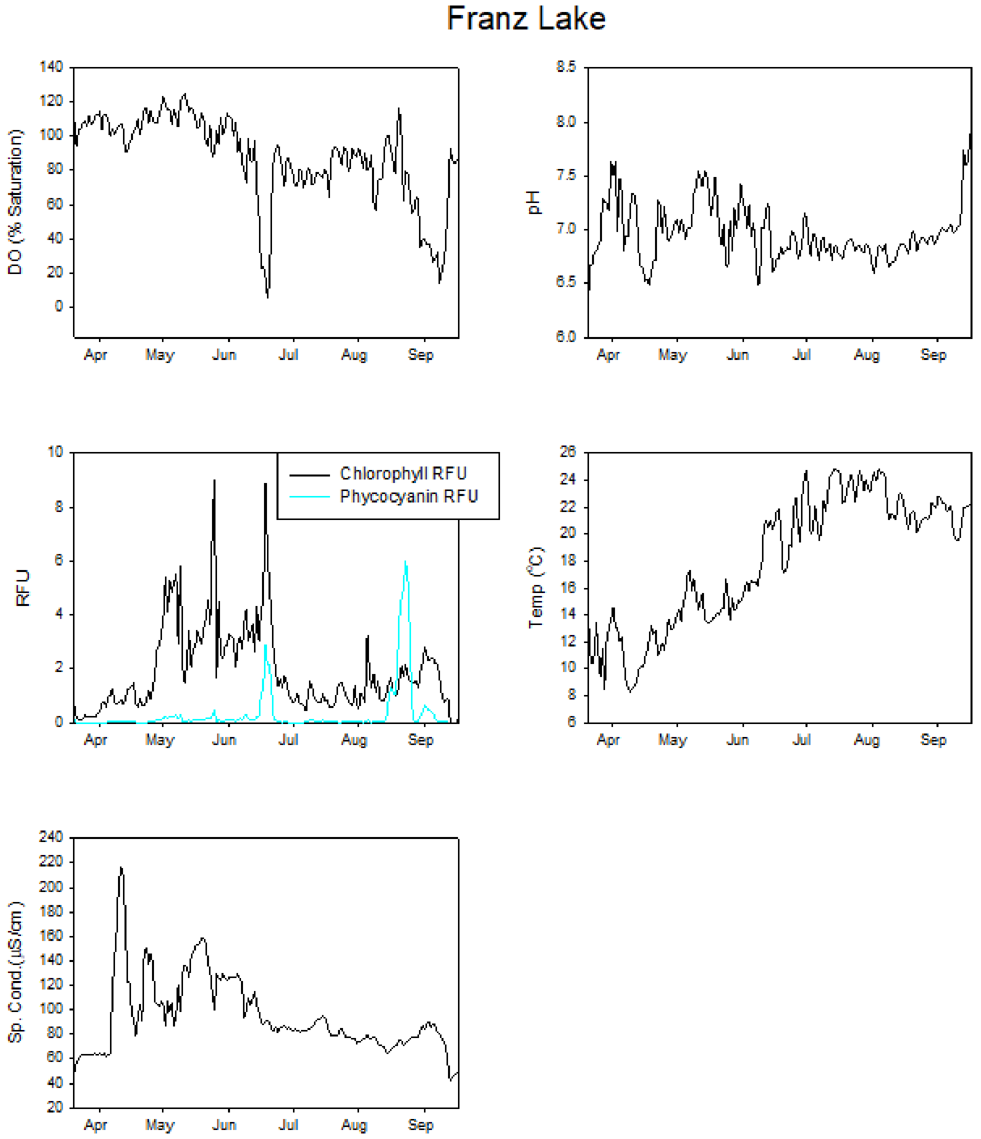


Figure 22. Time series of daily averaged measurements of water quality parameters made at Franz Lake Slough, 2019.

3.2.2 Dissolved Oxygen at Trends Sites

There was a wide range of values corresponding to dissolved oxygen saturation relative to the atmosphere at the off-channel trends sites (Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). It is recommended that dissolved oxygen should not fall below 6.0 mg L^{-1} for cold-water species, including native salmon (Oregon State Water Quality Standards); lower concentrations (4 and 2 mg L^{-1}) are considered to be increasingly detrimental to aquatic life. Using these thresholds to estimate stress associated with suboptimal levels of dissolved oxygen, we computed the number of hours below 6, 4, and 2 mg L^{-1} for each of the five trend sites (Figure 23**Error! Reference source not found.**). Ilwaco had the greatest number of hours where dissolved oxygen concentrations were below each of the three threshold levels (>1400 in 2019), similar to observations in 2017 and 2018 (Figure 24). Dissolved oxygen concentrations below the 2 mg L^{-1} threshold were observed at Ilwaco and Franz Lake slough in 2019. Dissolved oxygen levels below 6 mg L^{-1} were observed at all of the sites.

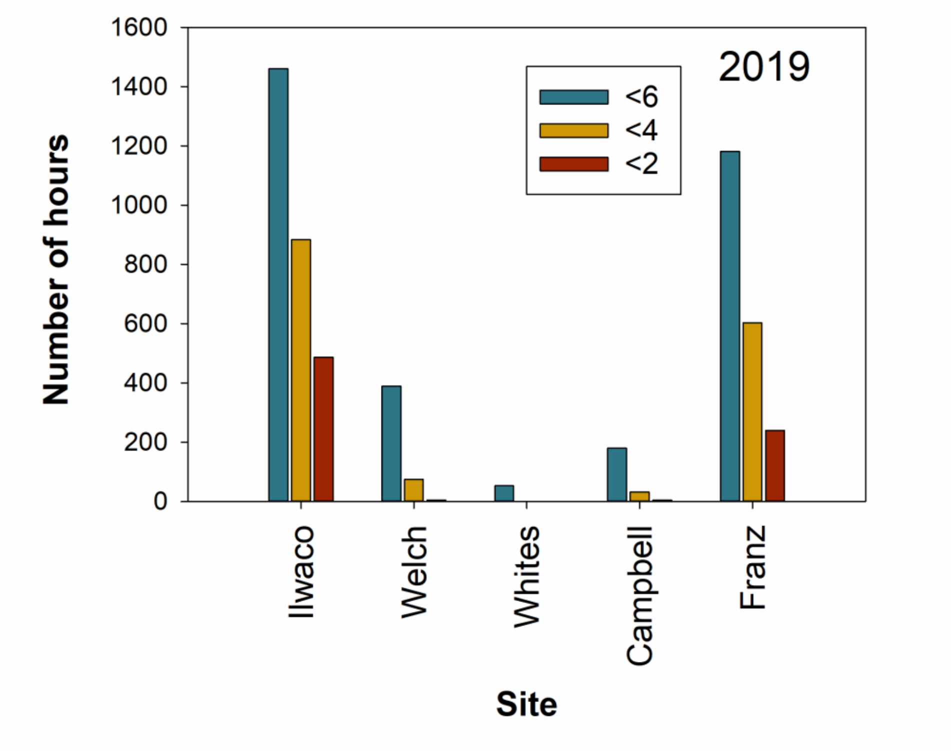


Figure 23. Number of hours in 2019 when dissolved oxygen concentrations (in mg/L) were either below 6 mg/L, 4 mg/L, or 2 mg/L at the five trends sites.

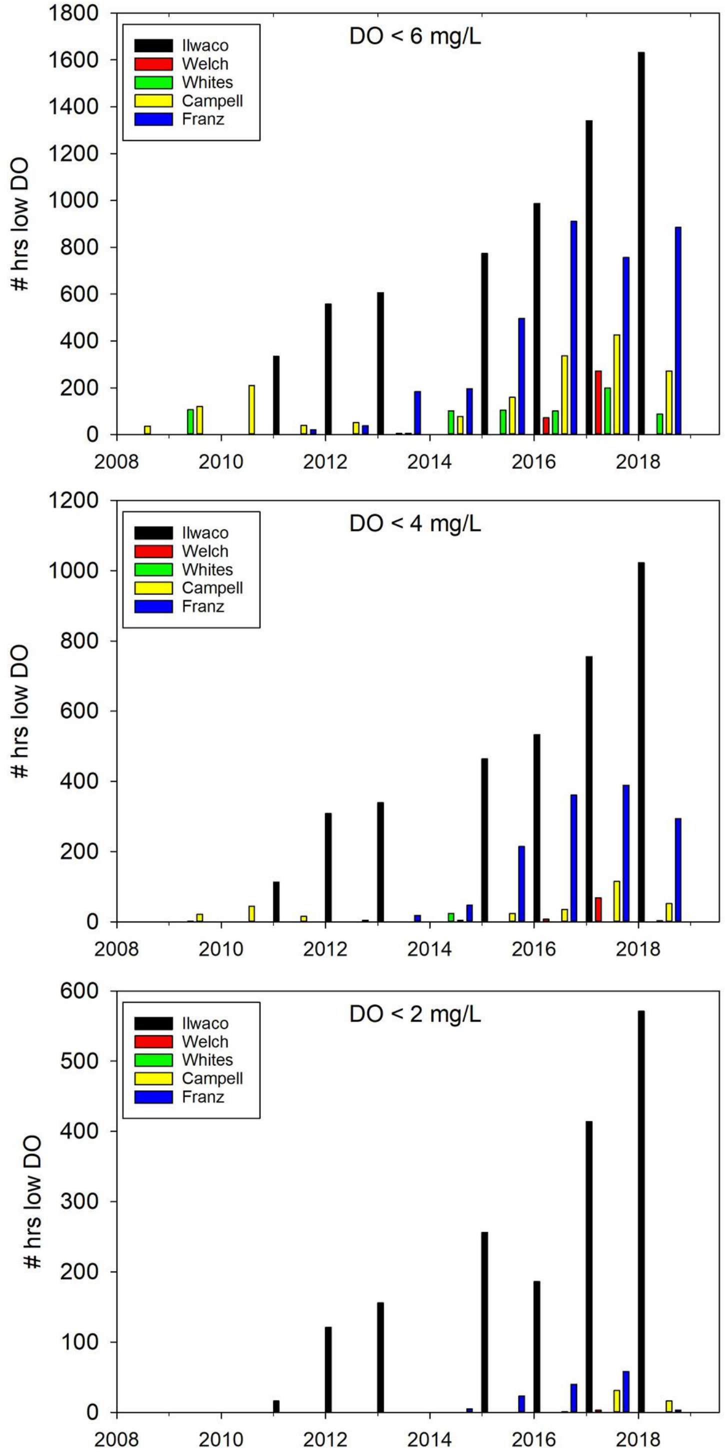
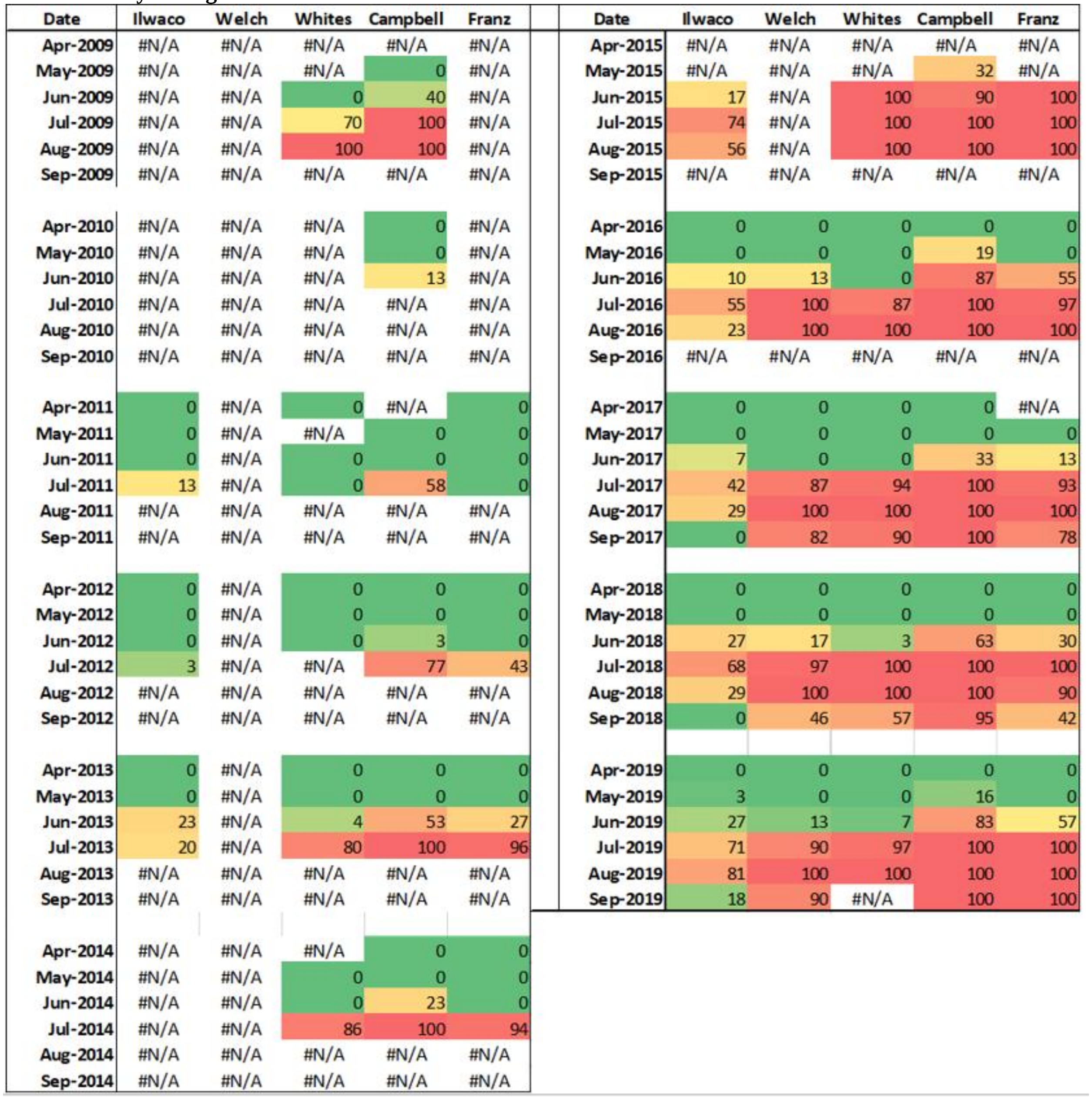


Figure 24. Time series of past years' (2008-2018) observations corresponding to three thresholds of dissolved oxygen concentration (mg/L): top panel, dissolved oxygen concentrations <6 mg/L; middle panel, dissolved oxygen concentrations <4 mg/L; bottom panel, dissolved oxygen concentrations <2 mg/L. The data show the number of hours where dissolved oxygen concentrations were below each threshold, as determined by in situ sensors.

3.2.2.1 *Temperature Thresholds at Trends Sites*

Water temperature in the river mainstem was discussed earlier as an important variable and potential stressor to salmonid populations. Here, we present a time series showing the percentage of days within each month where temperatures corresponded to threshold exceedance of 19^{o}C for five off-channel trends sites (Table 17**Error! Reference source not found.**). According to these criteria, high temperatures posed a potential problem for salmonids during July and August of all years where there are data, with a few exceptions (for example, July 2009 at Whites Island). It is perhaps more useful to look at June temperature exceedance, since salmonids are typically present in June. With the exception of 2015, the only site where temperature exceedance in June is a persistent problem is Campbell Slough. Between 2009 and 2019, the percentage of June days where the 19^{o}C threshold was exceeded at Campbell Slough ranged from 0 in 2011 to 83% in 2019. This is interesting, given that the number of days in 2019 where temperature thresholds were exceeded in the mainstem was similar to previous years, highlighting the important role water level and river discharge plays in moderating temperatures in off-channel sites.

Table 17: Temperatures exceeding 19°C in the EMP sites from 2009-2019. Data show the percentage of each month that the daily average temperature exceeded 19°C. No calculation was performed if data collection was less than 7 days of a given month.

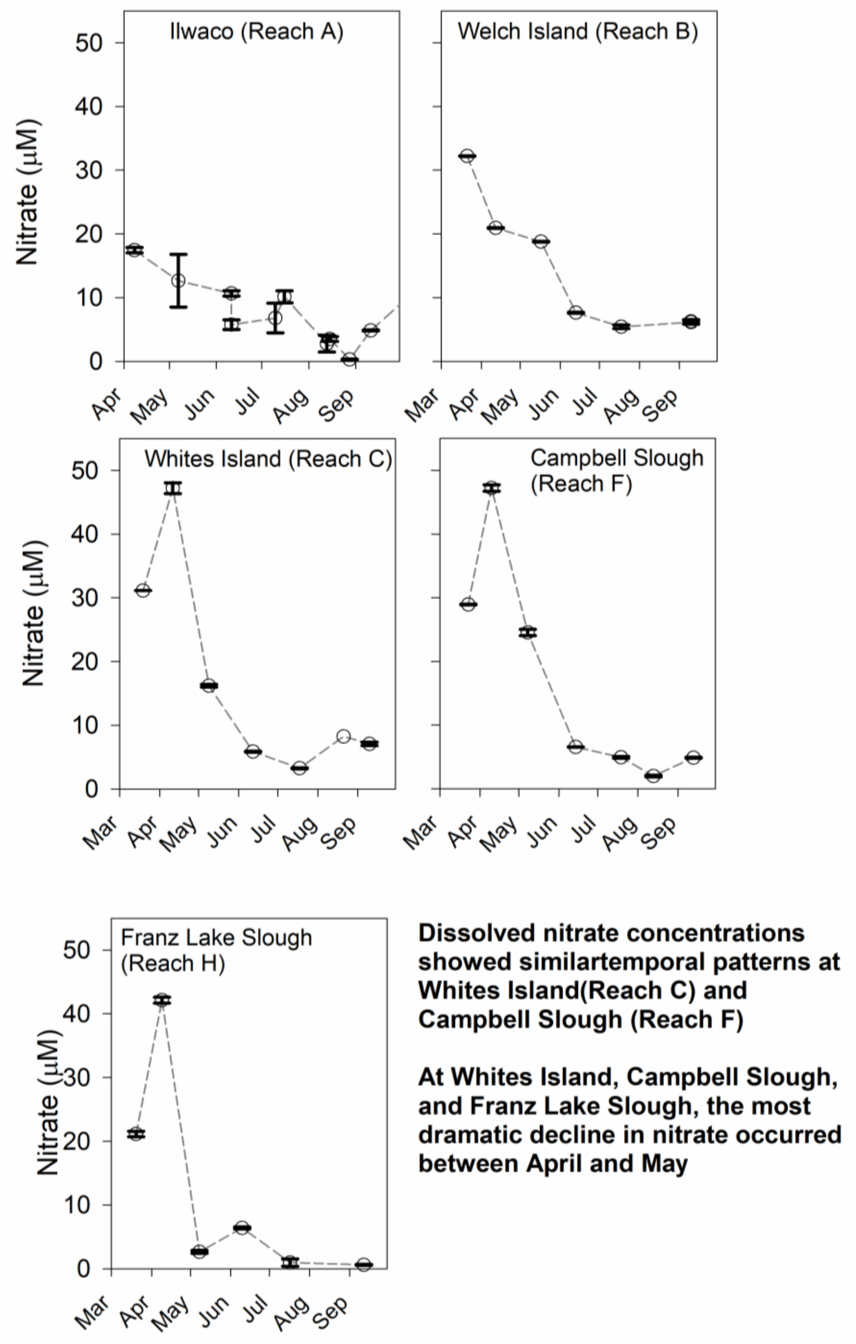


Date	Ilwaco	Welch	Whites	Campbell	Franz
Apr-2009	#N/A	#N/A	#N/A	#N/A	#N/A
May-2009	#N/A	#N/A	#N/A	0	#N/A
Jun-2009	#N/A	#N/A	0	40	#N/A
Jul-2009	#N/A	#N/A	70	100	#N/A
Aug-2009	#N/A	#N/A	100	100	#N/A
Sep-2009	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2010	#N/A	#N/A	#N/A	0	#N/A
May-2010	#N/A	#N/A	#N/A	0	#N/A
Jun-2010	#N/A	#N/A	#N/A	13	#N/A
Jul-2010	#N/A	#N/A	#N/A	#N/A	#N/A
Aug-2010	#N/A	#N/A	#N/A	#N/A	#N/A
Sep-2010	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2011	0	#N/A	0	#N/A	0
May-2011	0	#N/A	#N/A	0	0
Jun-2011	0	#N/A	0	0	0
Jul-2011	13	#N/A	0	58	0
Aug-2011	#N/A	#N/A	#N/A	#N/A	#N/A
Sep-2011	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2012	0	#N/A	0	0	0
May-2012	0	#N/A	0	0	0
Jun-2012	0	#N/A	0	3	0
Jul-2012	3	#N/A	#N/A	77	43
Aug-2012	#N/A	#N/A	#N/A	#N/A	#N/A
Sep-2012	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2013	0	#N/A	0	0	0
May-2013	0	#N/A	0	0	0
Jun-2013	23	#N/A	4	53	27
Jul-2013	20	#N/A	80	100	96
Aug-2013	#N/A	#N/A	#N/A	#N/A	#N/A
Sep-2013	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2014	#N/A	#N/A	#N/A	0	0
May-2014	#N/A	#N/A	0	0	0
Jun-2014	#N/A	#N/A	0	23	0
Jul-2014	#N/A	#N/A	86	100	94
Aug-2014	#N/A	#N/A	#N/A	#N/A	#N/A
Sep-2014	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2015	#N/A	#N/A	#N/A	#N/A	#N/A
May-2015	#N/A	#N/A	#N/A	32	#N/A
Jun-2015	17	#N/A	100	90	100
Jul-2015	74	#N/A	100	100	100
Aug-2015	56	#N/A	100	100	100
Sep-2015	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2016	0	0	0	0	0
May-2016	0	0	0	19	0
Jun-2016	10	13	0	87	55
Jul-2016	55	100	87	100	97
Aug-2016	23	100	100	100	100
Sep-2016	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2017	0	0	0	0	#N/A
May-2017	0	0	0	0	0
Jun-2017	7	0	0	33	13
Jul-2017	42	87	94	100	93
Aug-2017	29	100	100	100	100
Sep-2017	0	82	90	100	78
Apr-2018	0	0	0	0	0
May-2018	0	0	0	0	0
Jun-2018	27	17	3	63	30
Jul-2018	68	97	100	100	100
Aug-2018	29	100	100	100	90
Sep-2018	0	46	57	95	42
Apr-2019	0	0	0	0	0
May-2019	3	0	0	16	0
Jun-2019	27	13	7	83	57
Jul-2019	71	90	97	100	100
Aug-2019	81	100	100	100	100
Sep-2019	18	90	#N/A	100	100

3.2.3 Nutrients

3.2.3.1 *Dissolved Inorganic Nutrients (nitrate, phosphate)*

Dissolved nitrate concentrations reach high levels in the Columbia in the winter. Observations from trends sites in 2019 began in March at the end of winter/early spring when nitrate was nearly 30 μM at three sites most influenced by the Willamette River, the Columbia's largest tributary (Whites Island, Campbell Slough, and Welch Island; Figure 25**Error! Reference source not found.**). Nitrate concentrations at Ilwaco Slough and at Franz Lake Slough were ~20 μM. Similar to 2018, nitrate concentrations were higher at all sites in March and April compared to May, June, and July. Interestingly, in 2019 we did not observe the decline in nitrate concentration between March and April that was observed in 2018 at both Campbell Slough and Franz Lake Slough, but instead captured a peak in concentrations in April prior to the period of drawdown that accompanies spring primary production. This was not observed at Ilwaco Slough, Welch Island, or Whites (Figure 25**Error! Reference source not found.**). Similar to 2018, there were smaller differences among the monthly observations of nitrate concentration at Ilwaco compared to the other sites due to tidal exchange with the coastal ocean. At the other sites, nitrate concentrations were lowest during the summer months, reaching minimum values in August, with the lowest values observed at Franz Lake Slough. The recommended benchmark for maximum total nitrogen concentration in waters of the Columbia is <0.255 mg L^{-1}, or 18.2 μM according to the Department of Environmental Quality (DEQ; Oregon's National Rivers and Streams Assessment, 2008-2009), with levels exceeding 0.399 mg L^{-1} (28.5 μM) considered to be of poor quality. This level was exceeded during the spring at all sites except Ilwaco. The difference in nitrate concentration between the spring and the summer arises in part from dilution by the freshet and drawdown by primary producers. Thus, changes in the annual freshet volume influence the margin of safety for good water quality in waters of the lower Columbia.



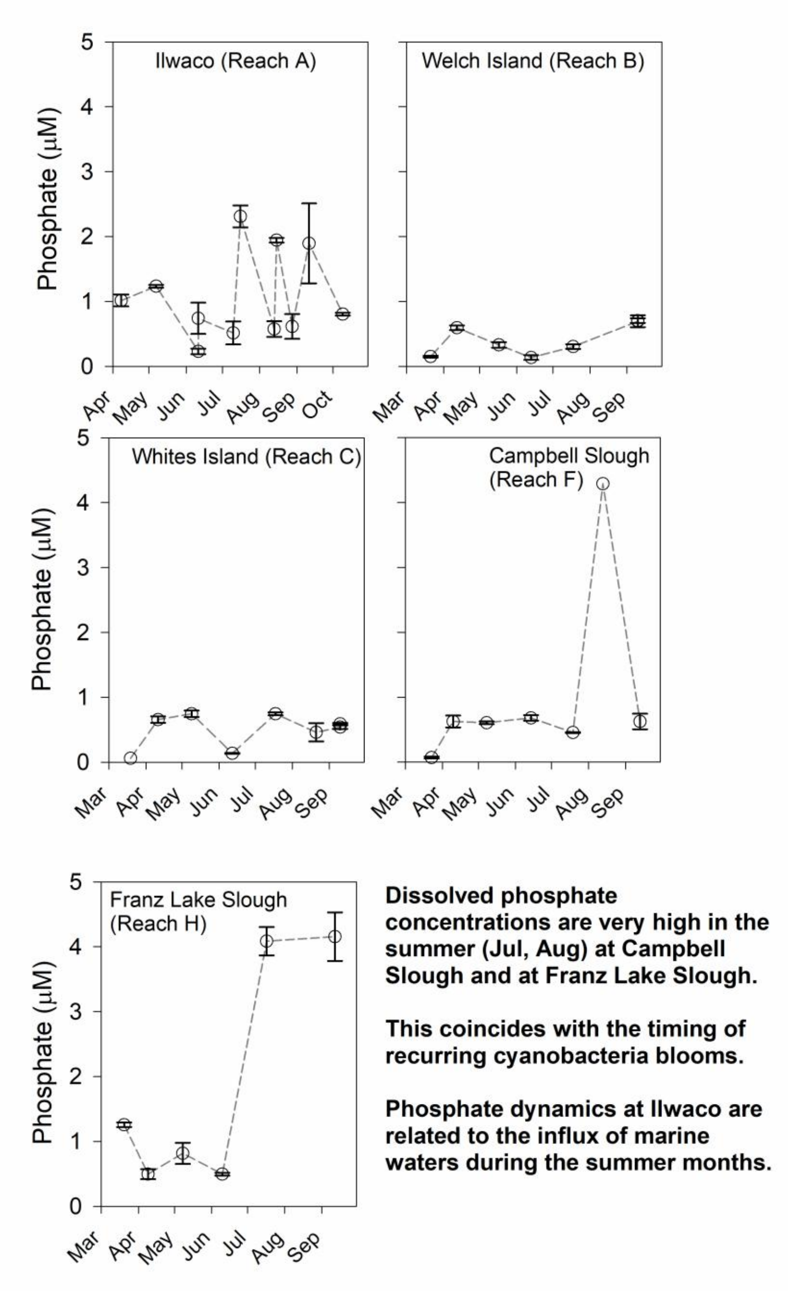
Dissolved nitrate concentrations showed similartemporal patterns at Whites Island(Reach C) and Campbell Slough (Reach F)

At Whites Island, Campbell Slough, and Franz Lake Slough, the most dramatic decline in nitrate occurred between April and May

Figure 25. Time series are showing concentrations of dissolved nitrate at the five trends sites in 2019.

Dissolved phosphorus (determined as soluble reactive phosphorus, or ortho-phosphate) were lowest in late June after the freshet subsided (Figure 26**Error! Reference source not found.**), but were otherwise similar before and after than at Welch Island and Whites Island. Phosphate concentrations were more variable at Ilwaco Slough, with higher levels observed during the summer compared to the spring. In contrast, very high phosphate concentrations were observed during the summer in both Campbell Slough and Franz Lake Slough (Figure 26**Error! Reference source not found.**). The DEQ benchmark for total phosphorus is <0.044 mg L^{-1} (1.42 µM) for good water quality and >0.069 mg L^{-1} (2.23 µM) indicating poor water quality. Summer concentrations of phosphate at Campbell Slough and Franz Lake Slough exceeded this level by two-fold.

Ammonium concentrations tend to increase as a result of microbial activity. The highest concentrations were observed at Ilwaco, particularly during the summer (Figure 27**Error! Reference source not found.**). At the other sites, ammonium concentrations tended to decrease in concert with the spring freshet, likely due to dilution and slower microbial growth.

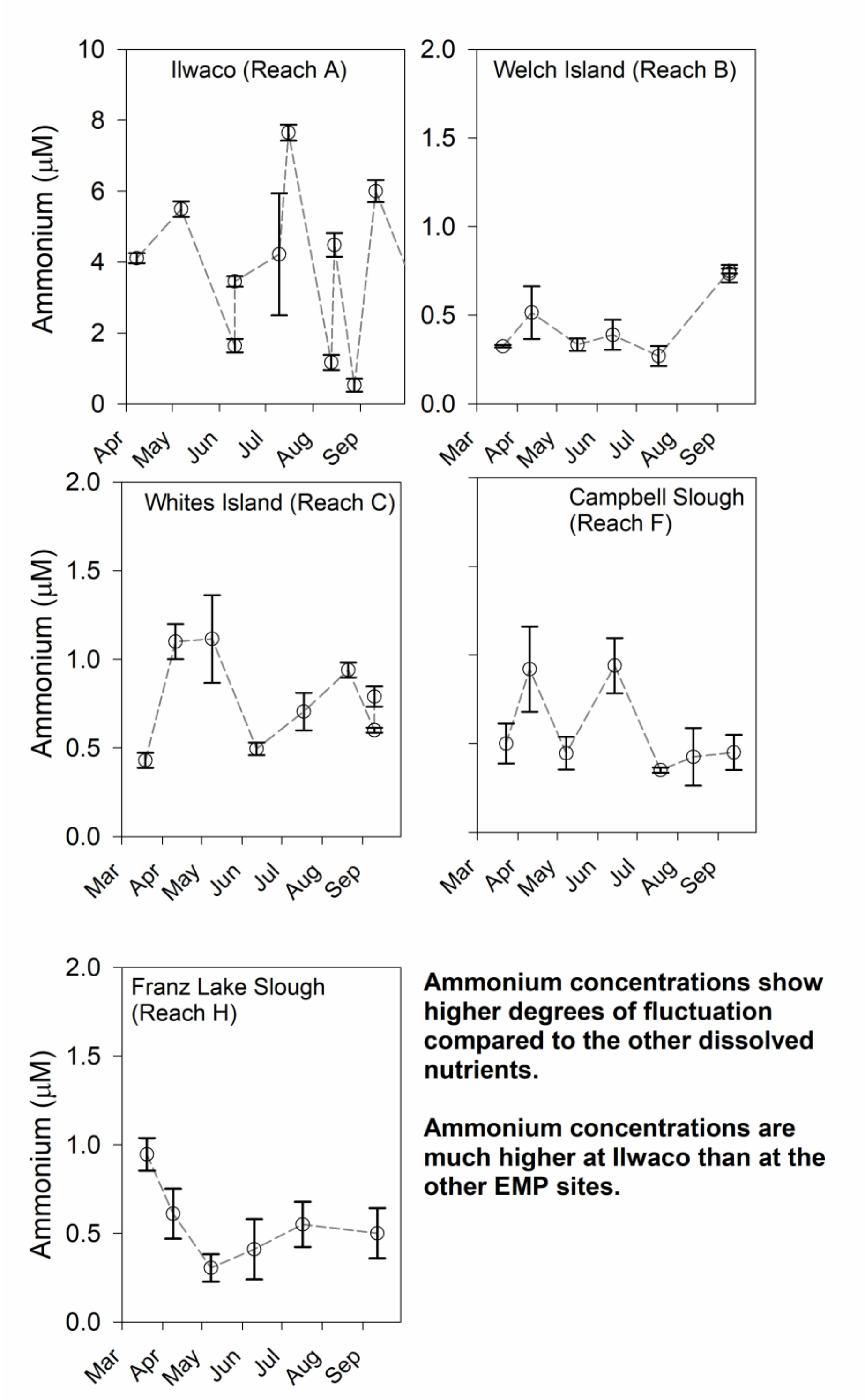


Dissolved phosphate concentrations are very high in the summer (Jul, Aug) at Campbell Slough and at Franz Lake Slough.

This coincides with the timing of recurring cyanobacteria blooms.

Phosphate dynamics at Ilwaco are related to the influx of marine waters during the summer months.

Figure 26. Time series showing concentrations of dissolved phosphate (ortho-phosphate) at the five trends sites in 2019.



Ammonium concentrations show higher degrees of fluctuation compared to the other dissolved nutrients.

Ammonium concentrations are much higher at Ilwaco than at the other EMP sites.

Figure 27. Time series showing concentrations of dissolved ammonium at the five trends sites in 2019.

3.3 Habitat Structure

3.3.1 Hydrology

Hydrologic patterns vary from year to year at all but the most tidal sites. Ilwaco is tidally dominated, while this tidal influence is reduced and traded for greater fluvial influences as you move up river towards Franz Lake. Hydrographs from all the years in which water surface elevation (WSE) was sampled at the trends sites, including the 2019 water year, are provided in Appendix C. Due to sensor failure at Ilwaco Slough, 2019 data has not been updated for this site. Some data loss occurred during the first half of 2019 at Welch Island. The following observations were made for these sites:

- The WSE at the Ilwaco Slough (rkm 6) is very minimally affected by the spring freshet but is elevated by winter storm events and extreme high tides. Low-water elevation measurements are truncated at the site because the elevation of the tidal channel is above that of extreme low water. The average tidal range across all monitoring years for this site varies between 1.4m and 1.5m annually (Table 18). Surveyed the wetland plant community range was 1.58 m in 2019 (Table 19).

- Welch Island site (rkm 53) hydrology is predominantly tidally driven. Annual maximum WSE at this site were observed in December 2019, which coincides with king tide elevations for winter 2019. The average tidal range at this site across all monitoring years is between 2.1m and 2.2m (Table 18). Slightly elevated WSE were detectable during the prolonged spring freshet in 2012, 2014, 2017, 2018. Although the spring freshet for 2019 peaked several times between April and June, due to sensor malfunction, this effect could not be captured for 2019. Winter storms also drive higher water levels at this site, particularly elevating the low tide levels. The surveyed wetland plant community range was 0.92 m in 2019 (Table 19).

- The hydrology of Whites Island (rkm 72) is influenced by tidal action as well as periods of prolonged freshets. Annual maximum WSE measurements for 2019 were observed in January and May. These elevations coincide with king tide duration for that month, as well as peak flow durations of the spring freshet in 2019. Elevated water levels were also observed during winter storm events and large spring freshets in May 2018. The average tidal range varies between 1.7-1.6 m (Table 18) across monitoring years and the surveyed wetland plant community range was 1.29 m in 2019 (Table 19).

- The Cunningham Lake site (rkm 145) and Campbell Slough site (rkm149), have similar hydrologic patterns. Annual tidal ranges between 2009 and 2019 vary between 0.4m to 0.6m at Cuningham Lake, and between 0.3m and 0.4m at Campbell slough. This indicates minimal influence of tides at these sites. Historic data indicate that Cunningham Lake has a slightly greater tidal range and slightly lower WSE during flood events compared to Campbell Slough. In most years, the primary hydrologic driver at both sites is the spring freshet, although from 2013 to 2018 winter storms also increased the WSE at these sites. The freshet caused greater flooding at Cunningham Lake in 2018 (Table 18). Similar influences were not observed in 2019. Annual maximum WSE at both sites coincided with the first peak flow durations during the 2019 spring freshet. The sensor at Cunningham Lake is in the very upper reach of the channel and is therefore elevated above the lowest water levels. The Campbell Slough sensor is in a deeper channel, however, a weir located at the mouth of the slough limits drainage. The topographic range of the wetland monitoring at these sites is significantly different, with Cunningham Lake wetland being more of a shallow gradient the elevation ranging only 0.68 meters on average, while the Campbell Slough wetland has a steeper gradient the elevation ranging 1.52 meters on average (Table 19). Shallow vs. steep wetland topography can significantly alter the hydrology and plant communities observed between the sites.

- The Franz Lake site (rkm 221) has the smallest tidal signal (in between 0.2-0.3 m, Table 18 and Appendix C) which is difficult to distinguish from diurnal variation from dam operations (Jay et al. 2015). The beaver dam that has been present in most years just below our sample area was gone in 2016, resulting in lower water levels in the channel. The beaver dam was re-established during 2017, 2018 and 2019, elevating the water level in the sampling area above that of the tidal exchange signal in the dry months of August and September (Appendix C). In most years, the winter and spring high WSE are both discernable. However, the spring levels are usually considerably higher than those in winter. The elevation range of the wetland at Franz Lake is 1.11 meters on average, and not well predicted by tidal signal (Table 19).

Table 18. Water surface elevation (WSE) metrics calculated at each site for the sensor deployment period from 2016-2018. All metrics are in meters, relative to the North American Vertical Datum of 1988 (NAVD 88). MWL = mean water level; MLLW = mean lower low water; MHHW = mean higher high water. Full hydrographs and annual summaries for each year are in Appendix C.

2019	Rkm	MWL	MLLW	MHHW	Avg Tidal Range (m)	Maximum WSE	Date of Maximum WSE	Period of Record	Days
Ilwaco	6	NA	NA	NA	NA	NA	NA	NA	NA
Welch	53	1.7	0.6	2.8	2.1	3.3	Dec 12	Sept-Dec	113
Whites	72	1.8	1.1	2.8	1.7	3.7	Jan 20	Jan-Dec	365
Cunningham	145	2.8	2.6	3.1	0.5	5.4	Apr 12	Jan-Aug	223
Campbell	149	2.9	2.7	3.2	0.4	5.6	Apr 12	Jan-Aug	225
Franz	221	5.5	5.4	5.6	0.3	10.3	Apr 11	Jan-Dec	341
2018	**Rkm**	**MWL**	**MLLW**	**MHHW**	**Avg Tidal Range (m)**	**Maximum WSE**	**Date of Maximum WSE**	**Period of Record**	**Days**
Ilwaco	6	1.5	1.0	2.4	1.4	3.1	Mar 02	Jan-Nov	311
Welch	53	1.9	0.9	3.0	2.1	4.0	Jan 30	Jan-Feb, May-July	149
Whites	72	2.2	1.4	3.4	1.7	4.1	May 17	Jan-Dec	365
Cunningham	145	3.3	3.0	3.5	0.5	7.0	May 17	Jan-Dec	365
Campbell	149	3.1	3.0	3.4	0.4	5.9	May 17	Jan-Dec	365
Franz	221	5.6	5.5	5.8	0.3	8.9	May 19	Jan-Jul	218
2017	**Rkm**	**MWL**	**MLLW**	**MHHW**	**Avg Tidal Range (m)**	**Maximum WSE**	**Date of Maximum WSE**	**Period of Record**	**Days**
Ilwaco	6	1.5	1.0	2.4	1.5	3.3	Feb 09	Jan-Feb, Aug-Dec	216
Welch	53	2.0	0.9	3.0	2.1	3.9	Feb 09	Jan-Dec	365

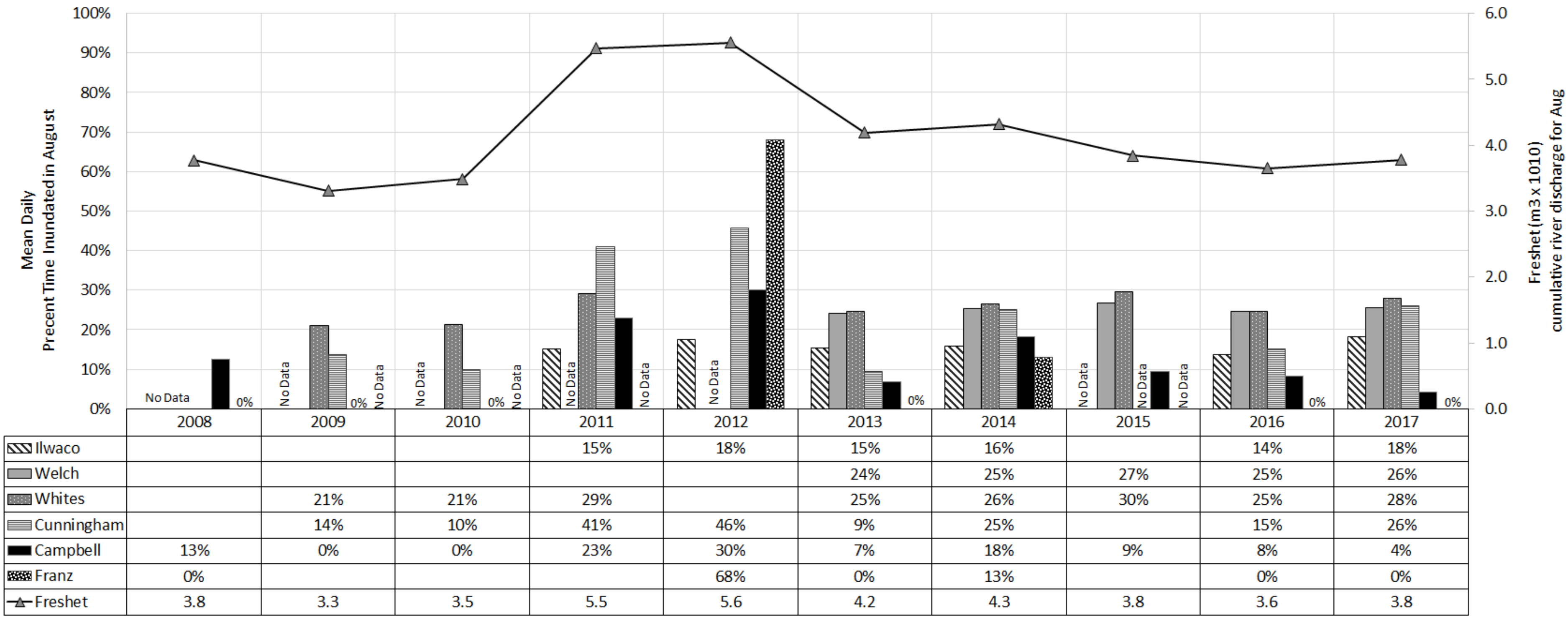
Whites	72	2.2	1.5	3.1	1.6	4.0	Dec 03	Jan-Dec	365
Cunningham	145	2.7	2.4	3.0	0.6	4.2	Dec 30	Jan, Aug-Dec	193
Campbell	149	3.6	3.4	3.8	0.4	6.3	Mar 30	Jan-Dec	365
Franz	221	5.2	5.1	5.3	0.2	8.2	Mar 25	Jan-Dec	365
2016	**Rkm**	**MWL**	**MLLW**	**MHHW**	**Avg Tidal Range (m)**	**Maximum WSE**	**Date of Maximum WSE**	**Period of Record**	**Days**
Ilwaco	6	1.4	0.9	2.4	1.5	3.2	Oct 15	Aug-Dec	147
Welch	53	1.7	0.6	2.8	2.2	3.9	Jan 02	Jan-Dec	366
Whites	72	1.9	1.1	2.8	1.7	4.7	May 10	Aug-Dec	153
Cunningham	145	2.7	2.5	2.9	0.4	3.6	Nov 26	Aug-Dec	152
Campbell	149	3.0	2.8	3.2	0.4	4.5	Mar 12	Jan-Dec	362
Franz	221	3.8	3.7	3.9	0.2	4.6	Dec 23	Aug-Dec	152

Table 19. Site marsh elevation range in meters based on the vegetation plot elevation (with ≥5% absolute living plant cover), relative to the North American Vertical Datum of 1988 (NAVD88). Mean number of plots, mean elevation, standard deviation (SD), minimum elevation (Min), and maximum elevation (Max). Marsh elevation ranges for all years can be found in Appendix D.

Long-term Elevation, m, NAVD88		Mean (SD)	2019	2018	2017
Ilwaco Slough	Plots (n)	40 (1)	40	40	40
	Mean	1.95 (0.05)	1.86	1.94	1.91
	SD		0.30	0.23	0.23
	Min	1.04 (0.29)	0.70	0.95	0.94
	Max	2.35 (0.08)	2.28	2.31	2.31
	Range	1.31 (0.29)	1.58	1.36	1.38
Welch Island	Plots (n)	41 (1)	43	40	41
	Mean	2.04 (0.03)	1.98	2.00	2.06
	SD		0.21	0.17	0.16
	Min	1.31 (0.06)	1.22	1.22	1.34
	Max	2.19 (0.03)	2.14	2.14	2.20
	Range	0.88 (0.03)	0.92	0.92	0.86
Whites Island	Plots (n)	41 (6)	45	44	42
	Mean	2.09 (0.09)	2.33	2.09	2.08
	SD		0.45	0.36	0.38
	Min	1.21 (0.03)	1.29	1.21	1.20
	Max	2.57 (0.12)	2.92	2.63	2.53
	Range	1.36 (0.10)	1.63	1.42	1.33
Cunningham Lake	Plots (n)	57 (15)	69	67	68

	Mean	2.7 (0.07)	2.67	2.69	2.78
	SD		0.20	0.19	0.19
	Min	2.31 (0.11)	2.30	2.29	2.41
	Max	2.99 (0.07)	3.01	3.07	3.02
	Range	0.67 (0.07)	0.71	0.77	0.60
Campbell Slough	Plots (n)	57 (14)	62	60	60
	Mean	2.98 (0.05)	2.96	3.01	2.95
	SD		0.37	0.37	0.32
	Min	2.46 (0.09)	2.50	2.40	2.51
	Max	3.99 (0.08)	3.97	4.01	3.76
	Range	1.53 (0.11)	1.47	1.61	1.25
Franz Lake	Plots (n)	56 (11)	66	64	60
	Mean	4.59 (0.04)	4.63	4.62	4.60
	SD		0.30	0.29	0.23
	Min	3.92 (0.13)	3.97	3.87	3.95
	Max	5.04 (0.04)	5.05	5.05	5.01
	Range	1.11 (0.11)	1.08	1.19	1.06

Annual variability in indundation patterns were not assessed for 2019, hence, the historic dataset has been further expolored. Inter-annual variation in inundation patterns is much greater at the upper estuary sites (Table 18, Figure 28), where seasonal flooding can result in months of continuous inundation during high-water years. In contrast, at the lower estuary sites dominated by tidal patterns, inundation lasts just a few hours during high tide, but occurs frequently, usually one to two times daily. The average inundation daily, as measured by the average numbers of hours a day (converted to a %) the water surface level is above the marsh elevation, is a means of comparing sites to each other and over time. The average inundation daily at each site is dependent on the elevation, the position along the tidal and riverine gradient, and the seasonal and annual hydrologic conditions. The average % of the day the mean marsh elevation is inundated for the month of August shown in Figure 28-Figure 34. The month of August was chosen because it is a critical time for plant development in the upper river sites, as the freshet draws down and exposes the marsh surface. Additionally, we have the most consistent amount of data for the month of August all sites and all years monitored. Generally, the trends in % time inundated identified in August correlate well with average % daily inundation for the year (unpublished data).



	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
▧ Ilwaco				15%	18%	15%	16%		14%	18%
▭ Welch						24%	25%	27%	25%	26%
▦ Whites		21%	21%	29%		25%	26%	30%	25%	28%
▤ Cunningham		14%	10%	41%	46%	9%	25%		15%	26%
■ Campbell	13%	0%	0%	23%	30%	7%	18%	9%	8%	4%
▩ Franz	0%				68%	0%	13%		0%	0%
▲ Freshet	3.8	3.3	3.5	5.5	5.6	4.2	4.3	3.8	3.6	3.8

Figure 28: Mean daily % time inundation for the month of August at all five trends sites between 2008-2017; Sites are ordered from left to right starting at the river mouth. Average site elevations are given in Table 19.

The lower river sites, Ilwaco Slough, Welch Island, and Whites Island, showed similar and consistent trends of daily flooding during the month of August (Figure 29-Figure 31). The hydrology of these sites are more tidally driven than the mid and upper river sites, especially in the summer after the high winter and spring flows have dissipated, generally showing less annual variability in their hydrology and plant community compositions than the mid and upper river sites (Figure 28-Figure 34).

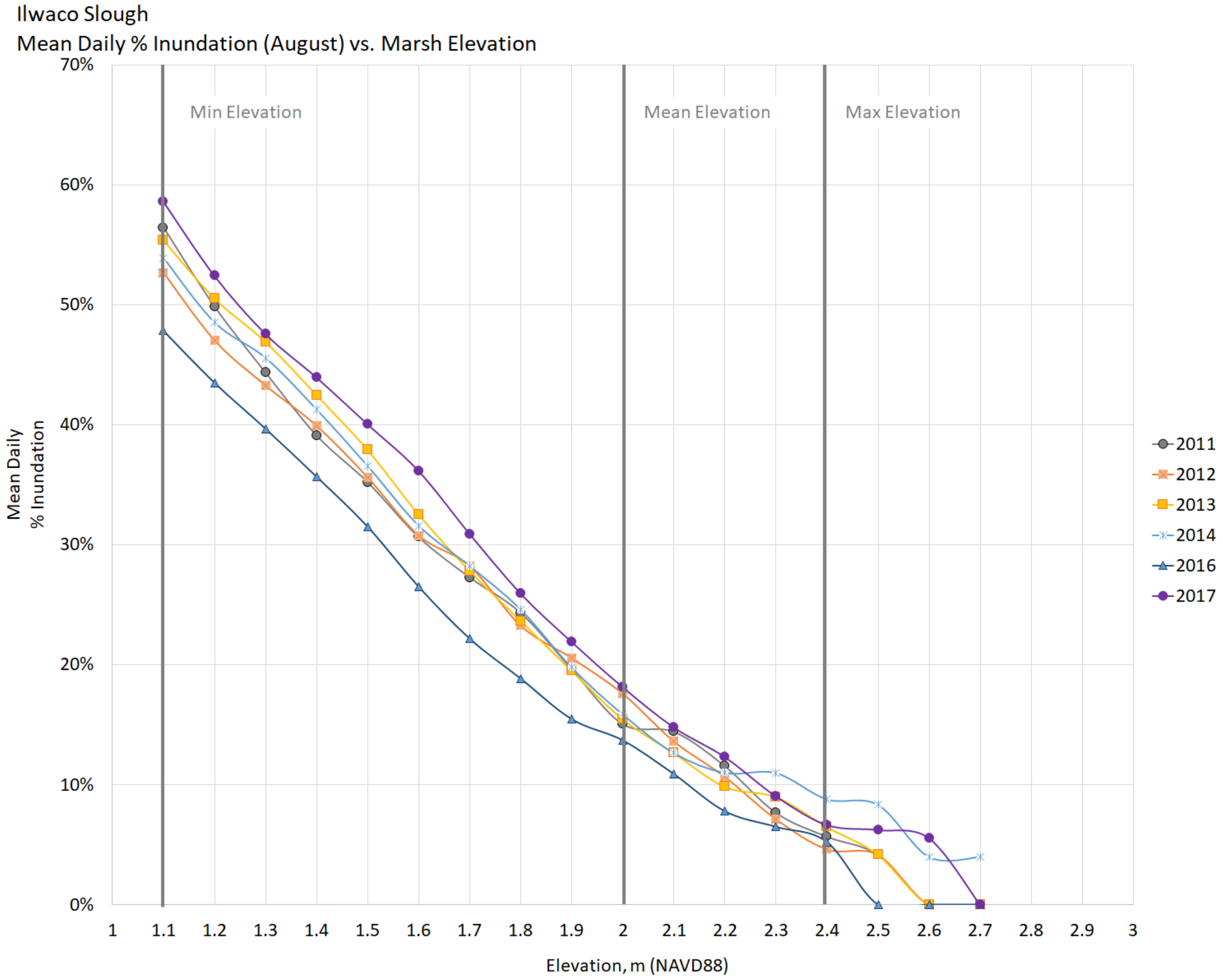


Figure 29: Ilwaco Slough: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2011-2017.

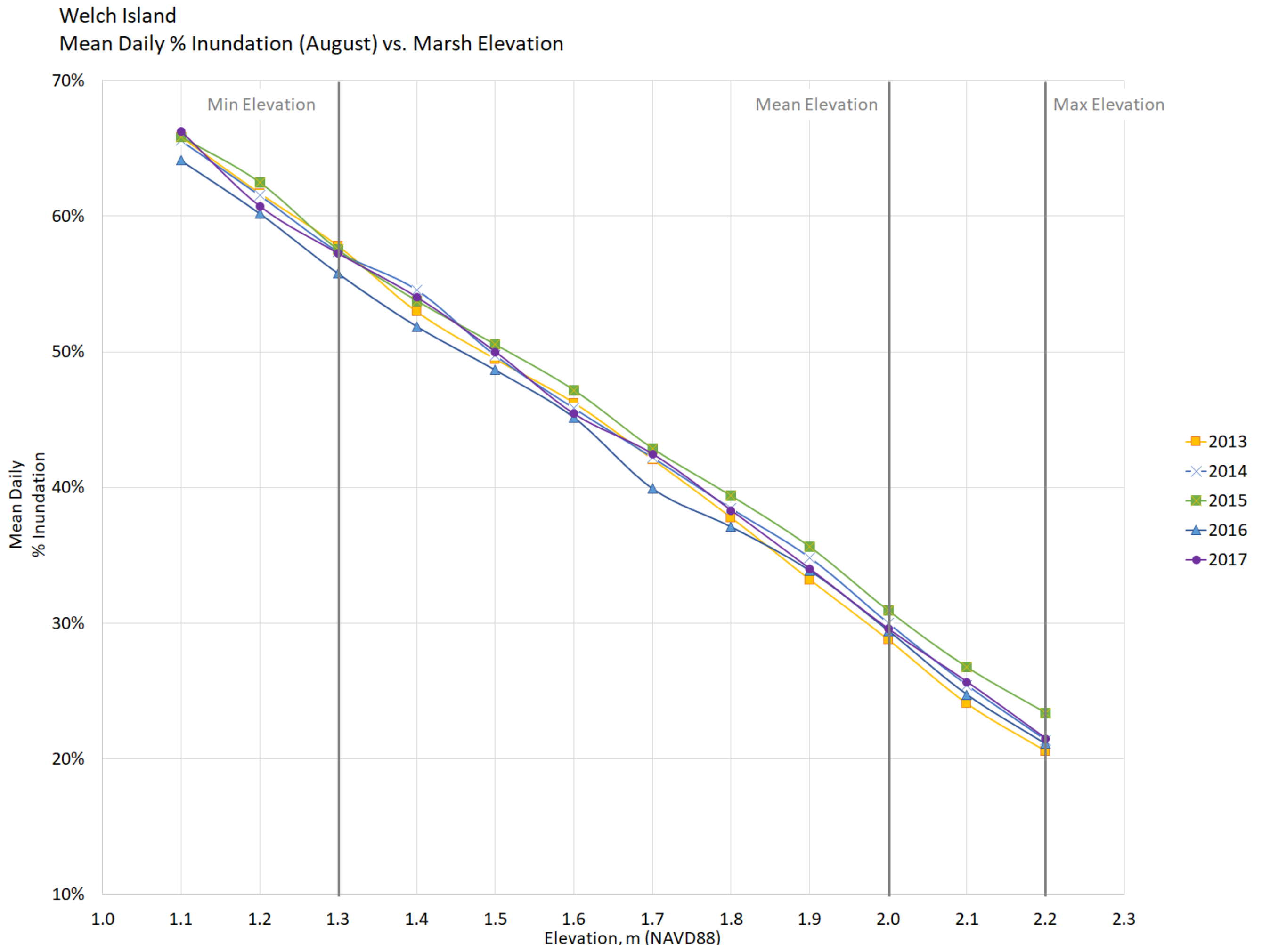


Figure 30: Welch Island: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2013-2017.

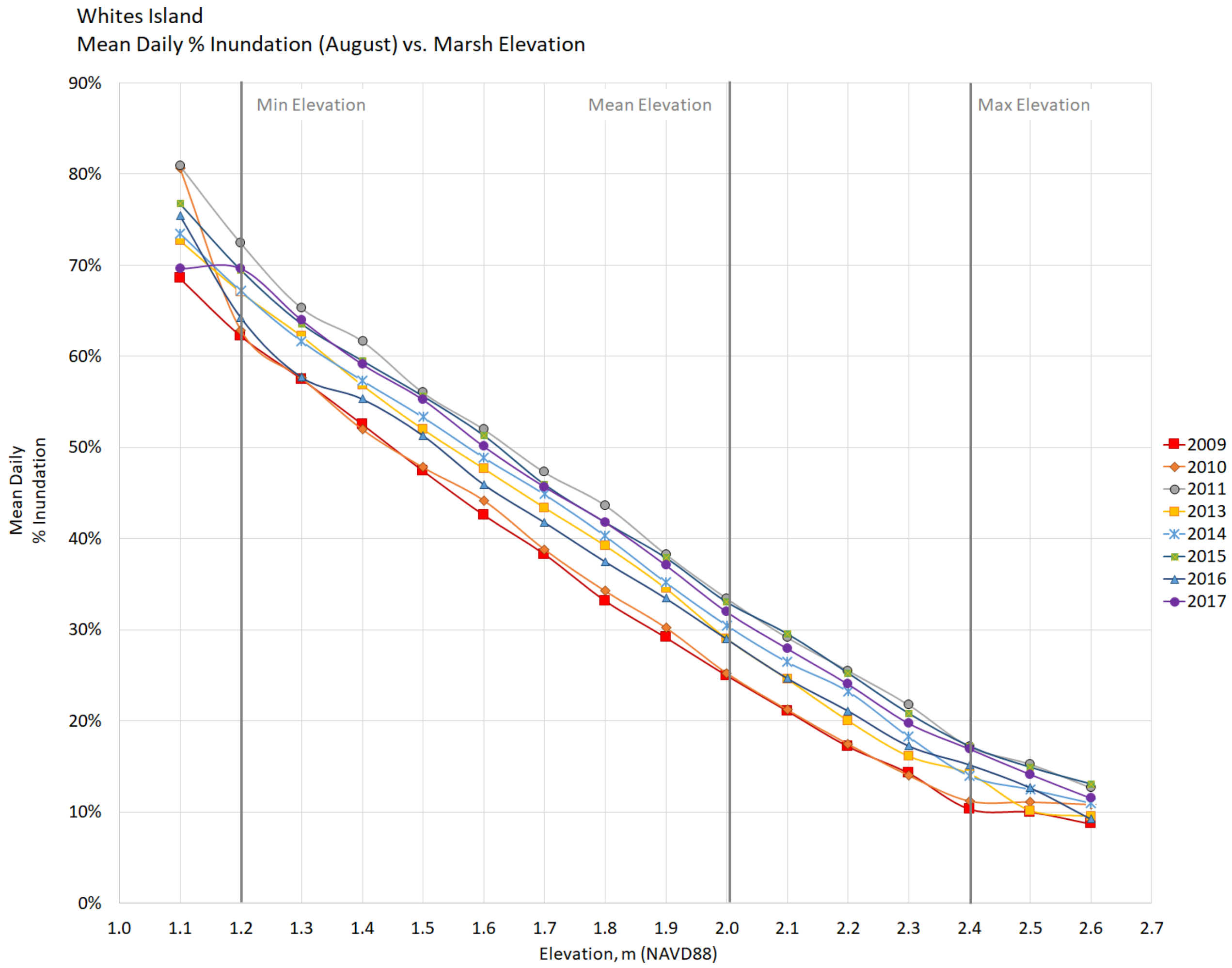


Figure 31: Whites Island: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2009-2017.

Inundation levels (mean % time flooded daily) at Cunningham Lake, mean marsh elevation of 2.7 meters (NAVD88) in 2017 were about 10% greater than 2016 (up from 15 to 26%), while Campbell Slough, at mean marsh elevation of 3 meters (NAVD88), inundation levels were down from 8 to 4% in 2017 (Figure 28, Figure 32, Figure 33). Consistently, for all years monitored, Cunningham Slough experiences greater levels of flooding than Campbell Slough (Figure 28, Table 18). Cunningham Lake hydrology appears to be slightly more sensitive to shifts in the Columbia River discharge than Campbell Slough, likely due to differences in their connectivity and proximity to the mainstem (Figure 1).

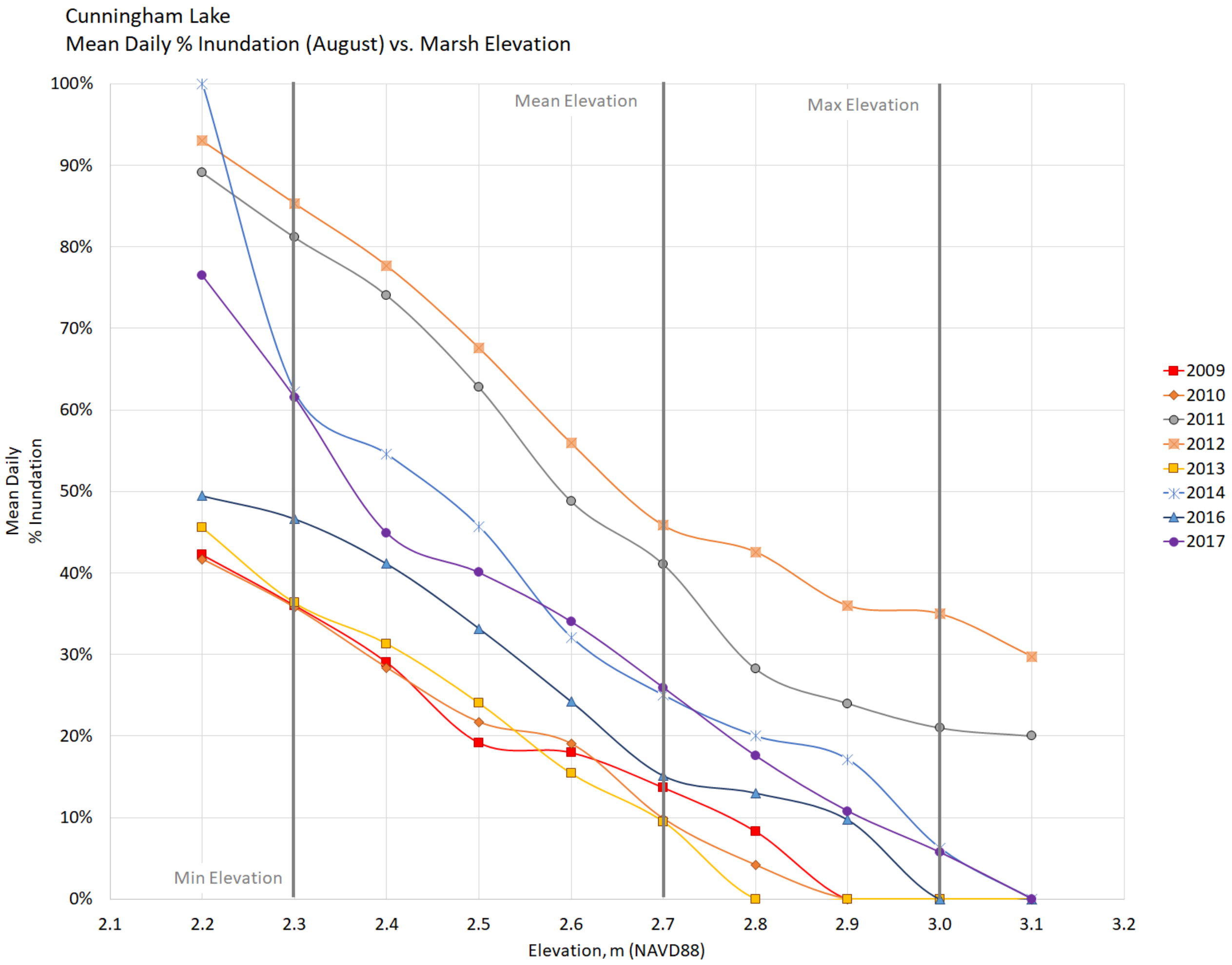


Figure 32: Cunningham Lake: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2009-2017.

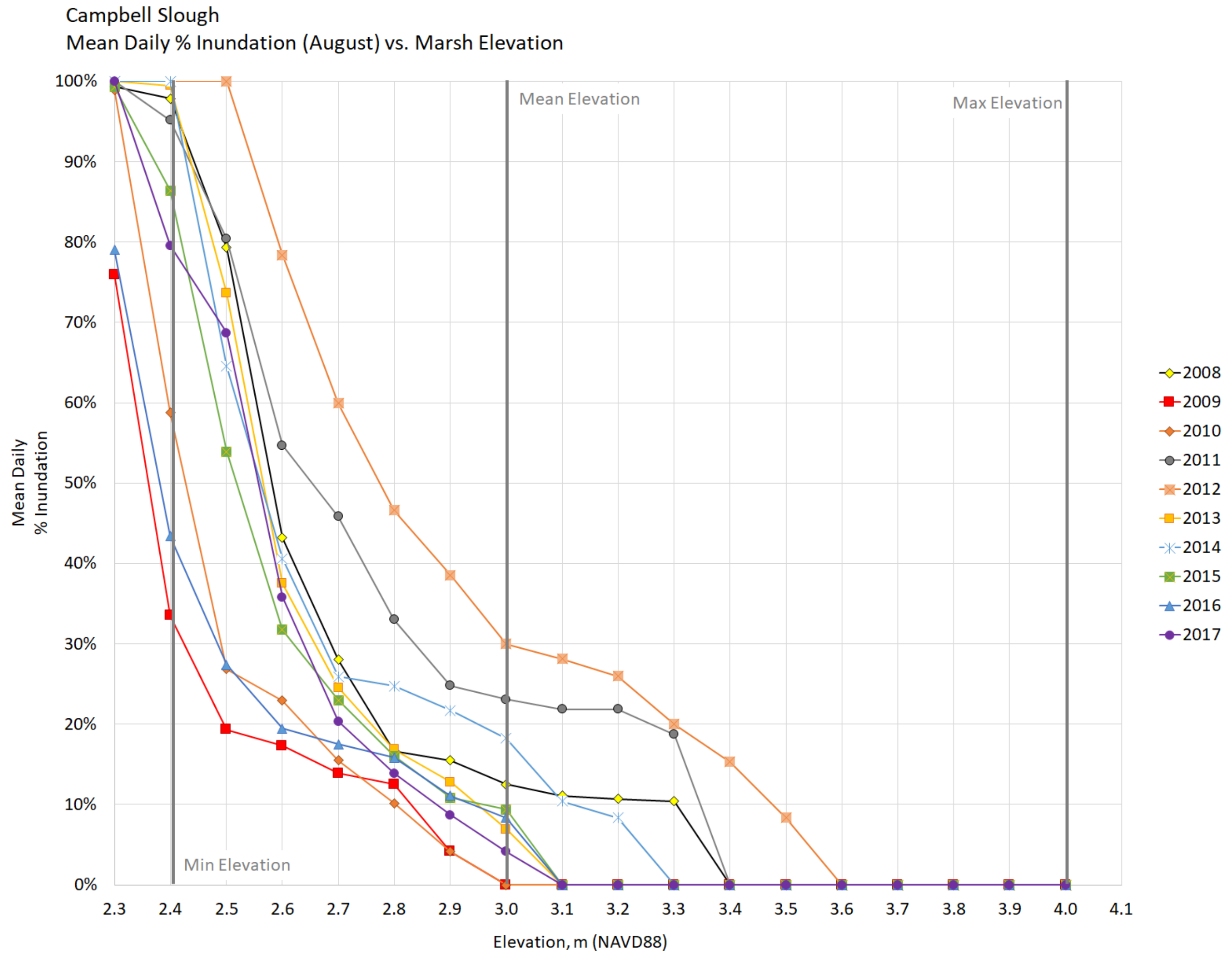


Figure 33: Campbell Slough: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2008-2017.

Franz Lake (mean marsh elevation of 4.6 m, NAVD88) was not flooded during the month of August in 2008, 2013, 2016 and 2017 in contrast to 2012, when the mean marsh elevation was flooded 68% of the time (Figure 34). Significant flooding during August for the upper river and mid-river sites (Cunningham and Campbell) indicates flooding for a majority of the growing season, freshet levels being elevated for a longer duration than normal. Extended periods of flooding during the peak of the growing season can significantly alter wetland plant community compositions. For example, Franz Lake transitioned from being a *Phalaris arundinacea* dominated to a *Polygonum amphibium* dominant wetland following the 2011 and 2012 growing seasons, these years exhibiting extremely high and extended Columbia River discharge levels compared to previous and following years (Figure 34, Table 25).

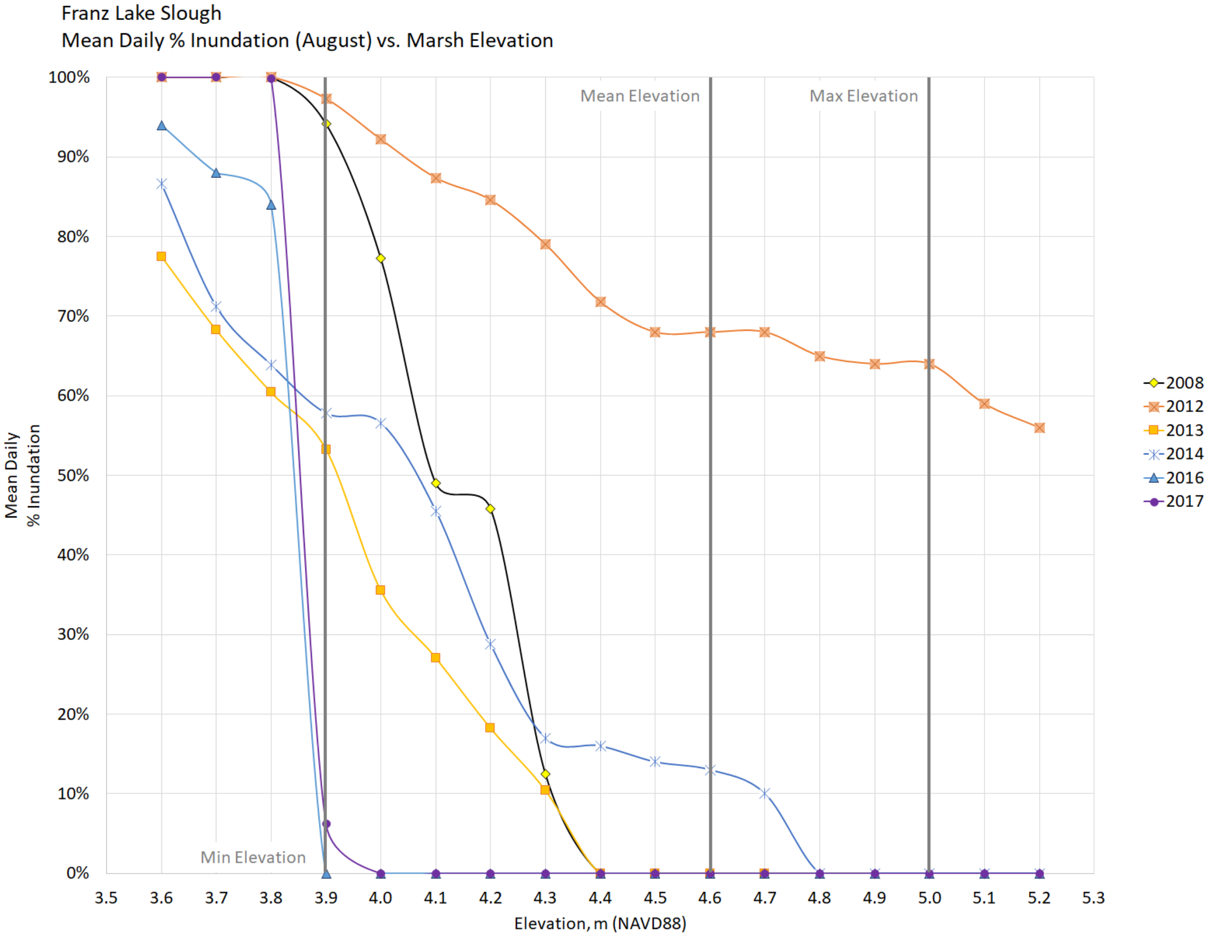


Figure 34: Franz Lake: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2008, 2012-2017.

3.3.2 Sediment Accretion Rates

In 2019, average sediment accretion or erosion rates at the five trend sites ranged from -0.4cm to 2.3cm per year. Franz Lake high marsh stakes had the highest rate of average erosion, with greatest variability (FLM-2: -0.7±1.8cm), which has been a consistent trend since installation of these stakes in 2015. Campbell slough also showed minor rates of erosion, which have been observed at Campbell Slough in 2015 and 2017 (Table 20). This is attributed to cattle grazing and repeated trampling of the site. Sedimentation stakes installed in Whites Island low marsh displayed the maximum accretion rate (2.3±0.9cm) for 2019. Cunningham Lake low marsh sedimentation stakes were measured just once after installation in 2016, hence, only high marsh accretion rates are available for 2019. It should be noted that accretion and erosion rates in the long term dataset are accompanied by high degree of variability, as shown in Figure 35.

Table 20. Sediment accretion rates at the trends sites between 2008 and 2019. WHC-M and WHC-H represent mid-elevation and high-elevation marsh locations, respectively.

Site Code:	BBM-1	BBM-2	WI2-1	WI2-2	WHC-1	WHC-2	CLM-1	CLM-2	CS1-1	CS1-2	FLM-1	FLM-2
Elevation (m, NAVD88)	**2.61**	**2.49**	**2.83**	**2.71**	**3.09**	**2.46**	**3.53**	**3.25**	**3.71**	**4.08**	**5.28**	**5.71**
Dominant Species	**CALY3**	**LIOC**	**CAOB3**	**CALY3**	**TYLA**	**CALY3**	**PHAR3/ SALA**	**Mud**	**ELPA3**	**PHAR3**	**POAM**	**PHAR3**
Year												
2008-2009	ND	ND	ND	ND	-1.2	ND	ND	ND	ND	ND		ND
2009-2010	ND	ND	ND	ND	1	ND	1.9	ND	0.4	ND	ND	ND
2010-2011	1.7	ND	ND	ND	0.1	ND	1.6	ND	1.7	ND	3	ND
2011-2012	0.1	ND	ND	ND	0.9	ND	1.4	ND	0.9	ND	-0.2	ND
2012-2013	0.6	ND	0.8	ND	0.2	1.2	1.3	ND	0.2	ND	3	ND
2013-2014	0.4	ND	0.6	ND	0.8	2.3	0.5	ND	1.5	ND	0.7	ND
2014-2015	1	ND	0.7	ND	0	2.7	-0.5	ND	-2.4	ND	1.2	ND
2015-2016	0	0.3	ND	1.0	ND	2.6	0.9	2.9	1.4	0.8	-0.6	-2.3
2016-2017	0.4	-2.5	0.6	1.8	0.4	3.7	0.1	ND	-4.2	-0.6	0.6	-2.1
2017-2018	0.9	1.1	-2.5	4.0	2.1	2.4	1.5	ND	2.2	0.6	3.3	1.4
2018-2019	-0.3	-0.5	1.7	-3.0	-0.3	1.3	-1.1	ND	-3.2	-1.2	0.4	0.2
Average	**0.5**	**-0.4**	**0.9**	**0.7**	**0.4**	**2.3**	**0.8**	**ND**	**-0.2**	**-0.1**	**1.2**	**-0.7**
Std Dev	0.7	1.6	0.5	3.4	0.9	0.9	1.0	ND	2.3	1.0	1.4	1.8

ND No data.

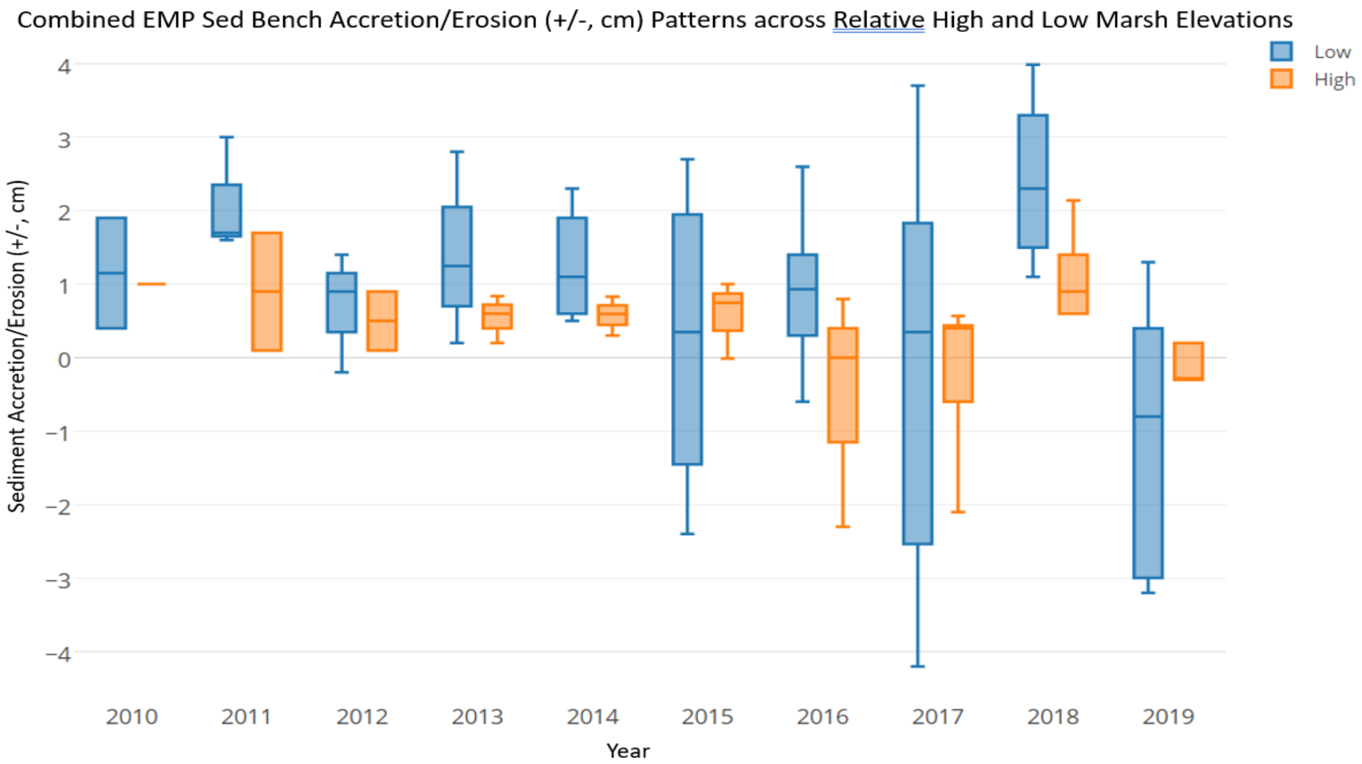


Figure 35: Long-term accretion rate variability across all 6 trend sites for 2010-2019

When long-term accretion and erosion rates were plotted against marsh elevation (m,CRD, Figure 36), we obtained a significant ($R^2 = 0.57$, p-value = 0.005) correlation between marsh elevation and sediment accretion rates. Low marsh elevations at the trend sites having higher accretion rates than high marsh elevations, thereby supporting our previously stated hypothesis in Section 2.3.2.2.

Regression analysis of mainstem cumulative discharge over average accretion and erosion rates at trend sites did not show any significant relationship ($R^2 = 0.034$, $p>0.05$). This can be attributed to high variability of the long-term dataset (Figure 35).

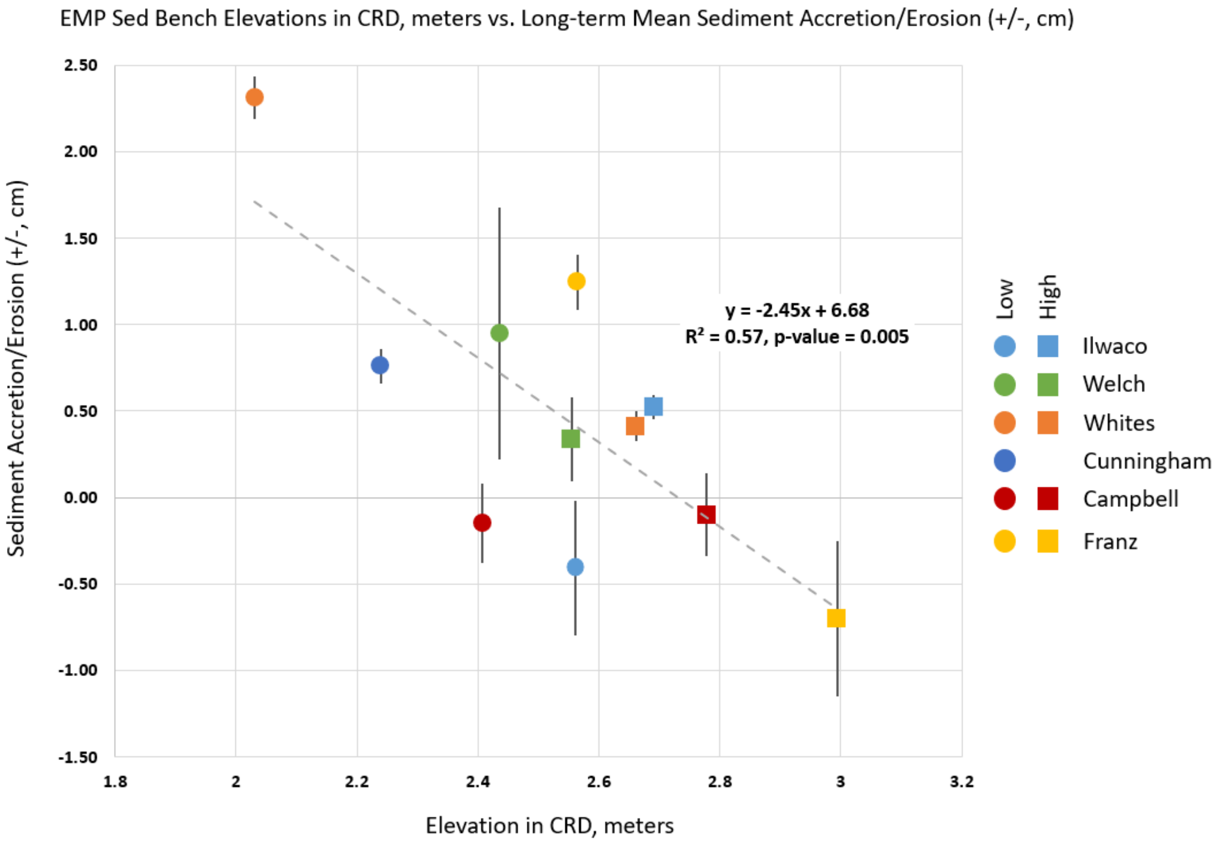


Figure 36: EMP site sediment bench elevations in CRD, meters vs. the long-term mean sediment accretion/erosion (+/-, cm) observations with Standard Error provided. Low (circles) and high (squares) relative within marsh elevations highlighted for each site. No long-term data was available for Cunningham Lake high marsh. Linear regression (y = -2.45x + 6.68), R² = 0.57, p-value = 0.005. For full summary of these data see Table 20.

3.3.3 Vegetation Species Assemblage

3.3.3.1 *2019 Vegetation Patterns Across Estuary Zones*

Trends in species richness and diversity

In 2019, the highest overall total species richness was found at Welch Island (50 species, Table 21), and the highest average richness per plot was also at Welch Island (12 species, Table 21). Welch Island occurs in wetland zone 2, in the lower river. As in previous years, the lowest species richness was at the brackish Ilwaco Slough (22 species) located in zone 1, near the mouth of the Columbia River. In wetland zone 4, mid-river sites, Campbell Slough, and Cunningham Lake both had similar richness (35, 27 respectively), and average richness per plot (5.8, 5.2 respectively). Franz Lake, wetland zone 5, the upper river site, was intermediate in total richness (25) and similar in average plot richness (5.9). The mid-river sites located in zone 4, Cunningham and Campbell, had substantially lower species richness than lower river sites despite having about 50% more sample plots than Welch and Whites (zone 2). A summary of all plant community species richness data for 2019 can be found in Table 21 and the long-term total species richness data can be found in Table 22 and Table 23.

Generally, trends in site level mean plant species richness and diversity are consistent across all years of data collection, mean species richness being greatest at Welch Island (zone 2), followed by Whites Island (zone 2), Franz Lake (zone 5), Ilwaco (zone 1), and Cunningham and Campbell Slough (zone 4) (Figure 38). Additionally, across time, a slight increase in mean total species richness has been observed, with both native and non-native species richness increasing annually, non-native species richness has increased at a slightly greater rate than native species (Figure 37- Figure 39). This overall long-term increase in species richness over time could be caused by several factors including an increased survey effort and familiarity with site flora, increases in non-native introduced species which have been aided in recent years (2016-2019) at Campbell Slough and Cunningham Lake by exposure to cattle grazing. The trends in mean species richness increasing across time are also seen in the Shannon Diversity values; these values are slightly increasing annually in response to increases in species richness (Table 23). Species evenness, a diversity measure of plant community cover and species richness distributions, show less change over time, species evenness remaining consistent across all sites over the long-term data set (Table 23). Indicating, while species compositions may be shifting slightly year to year the general distribution of species dominance have not changed substantially across the sites (Table 23). In 2019, Ilwaco and Campbell slough had the greatest evenness of 0.8, followed by Welch and Whites Islands 0.7, Cunningham Lake 0.6, and Franz Lake having the lowest at 0.4 (Table 23).

Table 21. 2019 and 2018 tidal marsh species richness, diversity, and areal cover at the trend sites, all values are overall site averages. OBL species are obligate wetland indicator species or species that are found in wetlands more than 99% of the time (Reed 1988). To see values in context with previous years and with standard deviations listed see Table 22-Table 24.

2019 Plant Community Summary		Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
	Site Rkm	6	53	72	145	149	221
	# Survey Plots	40	44	45	70	62	70
Species Richness	**Total**	4.6	12	9.2	5.2	5.8	5.9
	Native	3.6	9.3	5.6	3.7	3.8	4.9
	Non-native	1.0	2.3	3.4	1.4	2.0	0.8
Diversity	**Shannon**	1.2	1.8	1.6	1.0	1.3	0.8
	Evenness	0.8	0.7	0.7	0.6	0.8	0.4
Cover (%)	**Total (Living)**	102	151	136	129	84	105
Relative Cover (%)	**Bare ground**	5.0	0.2	2.2	6.7	22	0.5
	Native	84	77	45	58	66	84
	Non-native	16	22	55	42	34	14
	OBL	82	64	50	52	63	64
	Phalaris arundinacea	0.0	5.9	30	39	26	13
2018 Plant Community Summary		**Ilwaco Slough**	**Welch Island**	**Whites Island**	**Cunningham Lake**	**Campbell Slough**	**Franz Lake**
	Site Rkm	6	53	72	145	149	221
	# Survey Plots	40	41	44	69	65	61
Species Richness	**Total**	4.8	10.3	7.2	5.7	3.2	6.9
	Native	5.0	10.5	5.2	5.1	2.9	5.2
	Non-native	1.1	2.6	3.4	1.2	1.4	1.2
Diversity	**Shannon**	1.1	1.8	1.5	1.2	0.7	0.9
	Evenness	0.6	0.7	0.7	0.6	0.4	0.4
Cover (%)	**Total (Living)**	122	153	106	113	66	95
Relative Cover (%)	**Bare ground**	5.6	0.2	3.4	18	33	8.9
	Native	64	76	41	66	39	64
	Non-native	29	20	52	20	37	8
	OBL	70	71	52	70	61	55
	Phalaris arundinacea	0.0	7.3	30	17	32	8.2

Table 22. Overall total species richness, total native and total non-native species richness over time at the six trend sites, standard deviation included in parentheses for the long-term average for each site.

		Long-term	Year													
Site		**Average**	**19**	**18**	**17**	**16**	**15**	**14**	**13**	**12**	**11**	**10**	**09**	**08**	**07**	**06**	**05**
Ilwaco Slough	# of plots	40 (0)	40	40	40	39	39	40	40	40	40						
	Total	19 (2)	22	19	15	20	22	19	19	17	19						
	Native	16 (2)	18	16	13	17	20	14	16	14	16						
	Non-Native	2 (1)	4	3	1	1	1	3	2	2	2						
Welch Island	# of plots	40 (1)	44	40	42	40	40	40	40	40							
	Total	45 (5)	50	46	38	42	48	49	45	39							
	Native	33 (2)	34	32	29	32	36	34	34	30							
	Non-Native	10 (3)	15	12	7	8	12	10	9	8							
Whites Island	# of plots	41 (7)	45	44	44	44	51	43	45	42	42	35	25				
	Total	43 (6)	48	51	49	46	47	48	45	32	39	38	35				
	Native	30 (5)	32	33	38	31	32	30	31	21	25	27	25				
	Non-Native	11 (3)	15	14	9	10	11	13	11	9	13	9	6				
Cunningham Lake	# of plots	58 (14)	70	68	69	69	69	36	37	61	59	62	64	63	65	62	28
	Total	18 (6)	27	32	19	23	23	16	12	11	13	21	20	14	14	12	15
	Native	13 (4)	18	22	15	17	16	11	8	8	9	14	13	12	11	9	13
	Non-Native	4 (2)	8	5	3	5	6	3	2	2	3	4	5	2	2	2	1
Campbell Slough	# of plots	59 (10)	62	61	65	61	62	61	61	61	61	62	61	64	62	61	24
	Total	25 (10)	35	35	21	38	37	29	39	23	16	27	24	22	17	13	9
	Native	15 (6)	17	19	13	24	26	17	22	15	12	17	14	12	8	9	7
	Non-Native	8 (4)	17	12	6	12	8	8	12	5	3	8	8	5	9	3	2
Franz Lake	# of plots	57 (10)	70	67	61	61	68	61	60	64	58		36	40			
	Total	23 (7)	25	27	28	32	33	27	21	17	19		18	12			
	Native	18 (5)	18	19	21	24	24	21	15	14	16		15	9			
	Non-Native	4 (2)	6	4	5	6	6	4	4	2	1		2	2			

Table 23. Average (±SD) species richness, native and non-native species richness over time at the six trend sites. TSR=total species richness, NSR=native species richness, NNSR=Non-native species richness, H = Shannon diversity, and J = Species evenness index. Total number of plots surveyed for each year at each site can be found in the previous table (Table 22).

Site		Long-term	Year													
		Average	19	18	17	16	15	14	13	12	11	10	09	08	07	06	05
Ilwaco Slough	TSR	4.7 (0.7)	4.5 (2.5)	6.1 (3.2)	4.8 (2.5)	5.2 (2.7)	4.7 (2.9)	4.2 (2.9)	4.0 (3)	4.0 (2.6)	4.6 (3.4)						
	NSR	3.8 (0.6)	3.5 (2.4)	5 (3.1)	3.9 (2.3)	4.2 (2.3)	4.1 (2.5)	3.5 (2.5)	3.3 (2.8)	3.1 (2.3)	3.9 (3.2)						
	NNSR	0.4 (0.4)	1 (0.6)	1.1 (0.5)	0.1 (0.3)	0.1 (0.3)	0.1 (0.2)	0.8 (0.6)	0.1 (0.3)	0.2 (0.4)	0.1 (0.3)						
	H	0.9 (0.1)	1.2 (0.6)	1.1 (0.6)	0.9 (0.6)	1.0 (0.5)	0.9 (0.6)	0.9 (0.7)	0.7 (0.7)	0.8 (0.7)	0.9 (0.8)						
	J	0.6 (0.1)	0.8 (0.4)	0.6 (0.2)	0.6 (0.3)	0.6 (0.2)	0.5 (0.3)	0.5 (0.4)	0.4 (0.3)	0.5 (0.4)	0.5 (0.4)						
Welch Island	TSR	11.4 (1)	11.8 (4.6)	13.5 (3.7)	10.3 (3.3)	11.4 (3.3)	11.8 (3.6)	11.2 (4.2)	11.2 (3.4)	10.4 (3.5)							
	NSR	9.1 (0.8)	9.3 (3.8)	10.5 (3.1)	8.0 (2.8)	9.2 (3.1)	9.7 (3.3)	9.1 (3.3)	8.9 (2.9)	8.2 (2.9)							
	NNSR	1.9 (0.5)	2.3 (1.3)	2.6 (1.3)0	1.6 (1)	1.1 (0.9)	2.1 (1.2)	1.8 (1.2)	1.8 (1.2)	1.8 (1.1)							
	H	1.5 (0.2)	1.6 (0.5)	1.8 (0.4)	1.3 (0.4)	1.5 (0.5)	1.5 (0.5)	1.6 (0.6)	1.5 (0.5)	1.5 (0.5)							
	J	0.6 (0.1)	0.7 (0.1)	0.7 (0.1)	0.6 (0.1)	0.6 (0.2)	0.6 (0.2)	0.7 (0.2)	0.6 (0.2)	0.6 (0.2)							
Whites Island	TSR	7.2 (1.2)	9.2 (2.8)	9.0 (3)	7.3 (3)	8.0 (3.1)	6.3 (2.1)	7.3 (3.4)	6.7 (3.1)	5.4 (2.4)	6.0 (2.8)	7.5 (3.5)	6.6 (2.7)				
	NSR	4.6 (0.7)	5.6 (2.4)	5.2 (1.9)	5.0 (2.6)	5.3 (2.5)	4.0 (1.4)	4.5 (2.4)	4.3 (2.3)	3.4 (1.7)	3.5 (1.9)	5.1 (3.1)	4.4 (1.9)				
	NNSR	2.3 (0.6)	3.4 (1.7)	3.4 (1.4)	2.2 (1.4)	2.2 (1.2)	2.0 (1.4)	2.4 (1.6)	2.1 (1.5)	1.9 (1.3)	2.3 (1.3)	2.2 (1.2)	1.4 (1)				
	H	1.2 (0.2)	1.6 (0.4)	1.5 (0.5)	1.2 (0.5)	1.2 (0.4)	1 (0.5)	1.2 (0.6)	1.1 (0.5)	1.1 (0.6)	1 (0.5)	1.3 (0.6)	1.2 (0.6)				
	J	0.6 (0.1)	0.7 (0.1)	0.7 (0.1)	0.6 (0.3)	0.7 (0.3)	0.6 (0.2)	0.6 (0.2)	0.6 (0.2)	0.6 (0.3)	0.6 (0.2)	0.6 (0.2)	0.7 (0.2)				
Cunningham Lake	TSR	3.9 (1.3)	5.2 (2.5)	7.1 (2.4)	5.7 (2.3)	4.6 (2.4)	4 (2.2)	3.8 (1.8)	2.8 (1.6)	2.5 (1.1)	2.1 (1.1)	3.7 (1.6)	3.8 (1.5)	3.3 (1.7)	3.8 (1.5)	3 (1.2)	3.5 (2.1)
	NSR	2.8 (1)	3.7 (1.9)	5.1 (1.9)	4.0 (1.7)	3.5 (2.0)	3.0 (2.0)	2.8 (1.5)	1.8 (1.3)	1.8 (1.1)	1.3 (1.2)	2.5 (1.5)	2.9 (1.4)	2.5 (1.4)	2.8 (1.4)	2.2 (1.2)	2.7 (2.2)
	NNSR	0.9 (0.2)	1.4 (0.8)	1.2 (0.7)	1.1 (0.7)	1.1 (0.7)	1 (0.7)	0.9 (0.6)	0.8 (0.7)	0.7 (0.5)	0.8 (0.5)	0.9 (0.6)	0.9 (0.8)	0.8 (0.7)	0.9 (0.5)	0.7 (0.5)	0.7 (0.5)
	H	0.7 (0.2)	1 (0.5)	1.2 (0.4)	0.9 (0.5)	0.8 (0.5)	0.8 (0.4)	0.7 (0.4)	0.5 (0.4)	0.6 (0.4)	0.4 (0.4)	0.7 (0.4)	0.8 (0.4)	0.8 (0.5)	0.9 (0.4)	0.6 (0.4)	0.5 (0.4)
	J	0.5 (0.1)	0.6 (0.3)	0.6 (0.2)	0.5 (0.2)	0.5 (0.3)	0.6 (0.2)	0.5 (0.3)	0.4 (0.4)	0.5 (0.3)	0.4 (0.4)	0.5 (0.3)	0.6 (0.3)	0.6 (0.3)	0.7 (0.3)	0.6 (0.3)	0.4 (0.3)
Campbell Slough	TSR	3.5 (1.1)	5.8 (3)	4.8 (2.4)	3.2 (2)	4.5 (2.8)	4.9 (2.9)	3.6 (1.8)	3.9 (2.5)	2.6 (1.2)	2.2 (1.2)	3.2 (1.8)	3.2 (1.7)	3.0 (1.3)	2.4 (1.4)	2.3 (1.2)	2.3 (1.6)
	NSR	2.3 (0.7)	3.8 (2.3)	2.9 (2.2)	2.1 (1.2)	3.0 (2.3)	3.7 (2.9)	2.3 (1)	2.5 (1.4)	1.7 (1.1)	1.5 (1.3)	2.4 (1.6)	2.3 (1.5)	2.0 (1.1)	1.7 (1.3)	1.7 (1.1)	1.5 (1.4)
	NNSR	1 (0.4)	2 (2.1)	1.4 (1.6)	0.9 (1.1)	1.3 (1.4)	0.9 (1.1)	1.2 (1.3)	1.1 (1.4)	0.7 (0.7)	0.7 (0.5)	0.8 (0.9)	0.8 (0.9)	0.7 (0.7)	0.8 (0.9)	0.5 (0.6)	0.8 (0.7)
	H	0.6 (0.2)	1.3 (0.5)	0.7 (0.5)	0.6 (0.4)	0.7 (0.4)	0.7 (0.5)	0.7 (0.4)	0.7 (0.4)	0.6 (0.4)	0.4 (0.4)	0.5 (0.4)	0.6 (0.4)	0.7 (0.5)	0.5 (0.5)	0.4 (0.4)	0.4 (0.4)
	J	0.5 (0.1)	0.8 (0.3)	0.4 (0.3)	0.4 (0.4)	0.4 (0.2)	0.5 (0.3)	0.6 (0.3)	0.6 (0.3)	0.6 (0.3)	0.4 (0.4)	0.4 (0.3)	0.5 (0.3)	0.6 (0.4)	0.5 (0.4)	0.5 (0.4)	0.4 (0.4)
Franz Lake	TSR	5.2 (1.1)	5.3 (3.8)	7.1 (3.4)	7.1 (3.3)	6.3 (3.2)	5.3 (2.5)	4.7 (2.2)	5.6 (2.4)	4 (1.8)	4.2 (1.5)		4.6 (2.3)	3.7 (1.4)			
	NSR	3.8 (0.8)	4.4 (3.1)	5.2 (2.5)	5 (2.6)	4.4 (2.1)	3.9 (1.6)	3.4 (1.6)	4.1 (1.8)	2.6 (1.3)	3.3 (1.4)		3.8 (2.1)	2.7 (1.2)			
	NNSR	0.9 (0.2)	0.7 (0.8)	1.2 (0.9)	1.4 (0.9)	1.1 (1.0)	1.0 (1.1)	0.5 (0.6)	0.8 (0.6)	0.7 (0.5)	0.8 (0.4)		0.8 (0.6)	1.0 (0.4)			
	H	0.9 (0.1)	0.7 (0.6)	0.9 (0.5)	1 (0.5)	1 (0.5)	0.9 (0.5)	0.8 (0.5)	0.8 (0.5)	0.8 (0.5)	0.9 (0.4)		0.9 (0.6)	0.9 (0.5)			
	J	0.5 (0.1)	0.4 (0.3)	0.4 (0.3)	0.5 (0.2)	0.5 (0.2)	0.5 (0.3)	0.5 (0.3)	0.5 (0.2)	0.5 (0.3)	0.6 (0.2)		0.6 (0.3)	0.7 (0.3)			

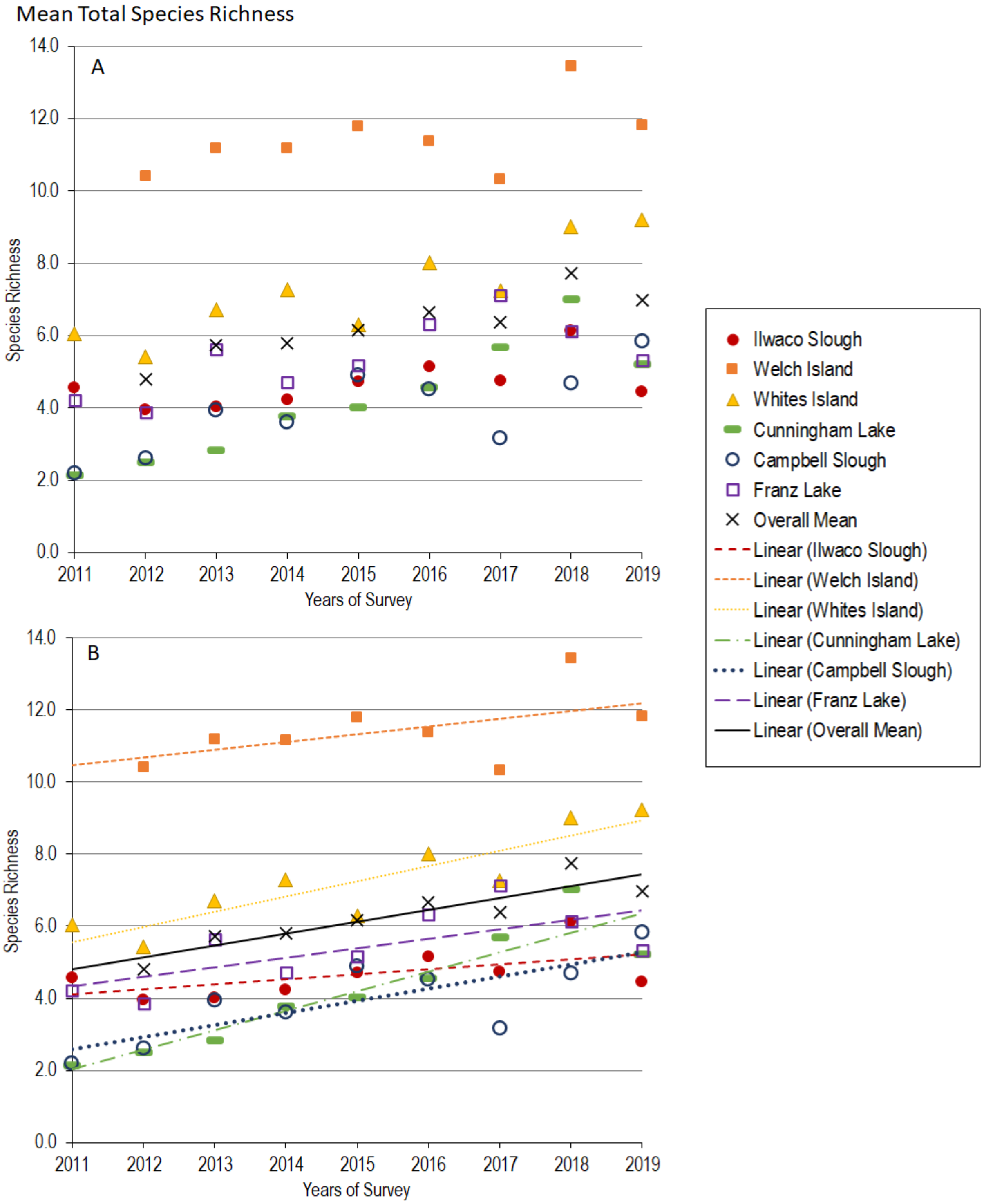


Figure 37. Changes in mean total species richness over time at each trend site. Only the years for which most current trend sites were sampled are shown (2011-2019).

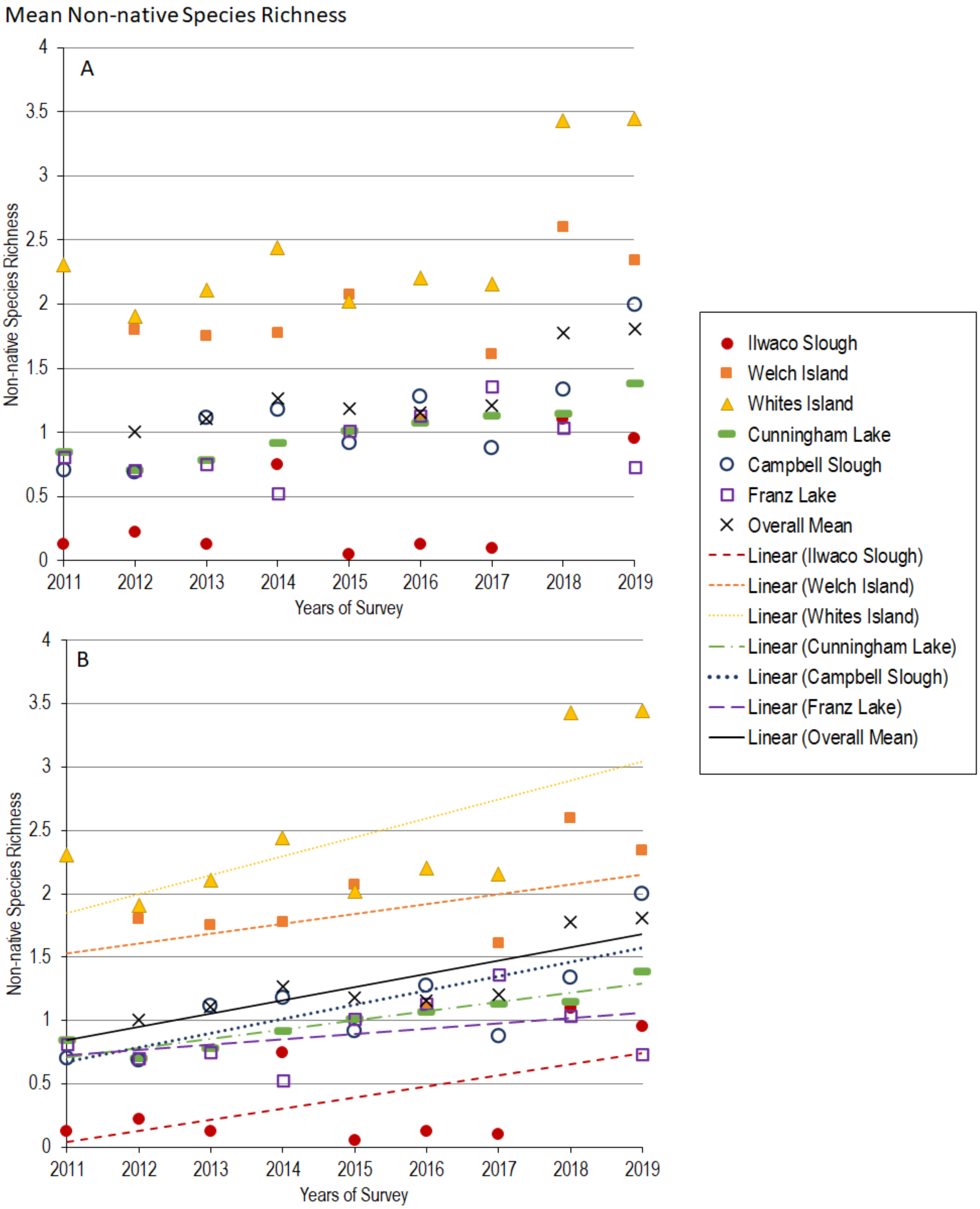


Figure 38. Changes in mean non-native species richness over time at each trend site. Only the years for which most current trend sites were sampled are shown (2011-2019).

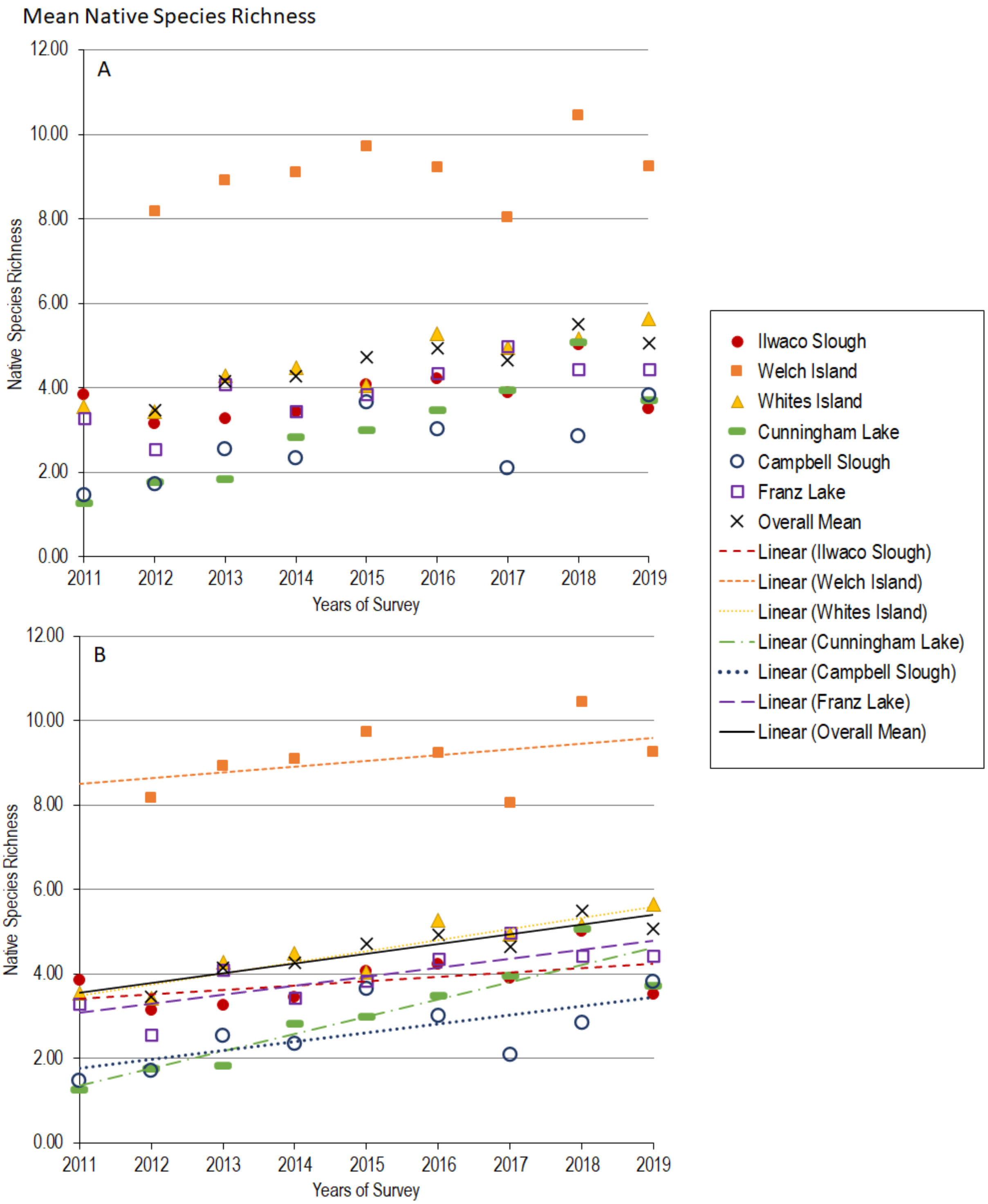


Figure 39. Changes in mean native species richness over time at each trend site. Only the years for which most current trend sites were sampled are shown (2011-2019).

Trends in plant cover

When comparing overall % total cover in 2019, Campbell Slough had substantially lower % vegetative cover (84%) than all other sites (105-151%) (Table 24, Figure 40-Figure 41). This is likely due to extensive cattle grazing removing vegetation from the site. Comparatively, there was a significant increase in cover at Cunningham Lake between 2017-2019 from 67% to 129% which was also grazed in 2017 but not in 2018 or 2019. Welch Island had the highest cover (151%), while the sites at the two ends of the tidal estuary, Ilwaco Slough (102%) and Franz Lake (105%) were similar to each other, with relatively high total cover. Total cover estimates follow trends with mean total species richness, as total cover is accumulative as the number of species observed increases so does the total % cover (Table 22, Table 24).

Relative cover of native and non-native plant communities across the sites have followed a less linear trend than total cover overtime (Figure 42). Generally, native and non-native cover are more similar from year to year in zone 1 and 2 sites (Ilwaco, Welch, Whites) compared to the zone 4 and 5 sites (Cunningham, Campbell, and Franz) (Figure 42 & Figure 43), this is likely due to the general hydrology of these sites, inundation patterns being much more stable from year to year in the tidally driven lower river, zone 1 and 2, sites compared to the fluvially dominated mid and upper river, zone 4 and 5, sites (see section 3.3.1). In 2019, mean relative native cover was greatest at Franz Lake (86±20%, long-term average 69±14%) and Ilwaco Slough (82±20%, long-term average 81±8%), both sites seeing an increase in about 20% relative native cover between 2018-2019 (Table 24, Figure 42 & Figure 43). At Ilwaco this was account for through a general reduction in non-native *Agrostis stolonifera* cover and a corresponding increase in native *Carex lyngbyei* cover, in addition to increases in other natives such as *Argentina egedii ssp. Egedii*, *Deschampsia cespitosa*, *Lilaeopsis occidentalis*, and *Symphyotrichum subspicatum*. At Franz Lake, this shift was a result of a general mixed increase in native species cover including *Argentina egedii ssp. Egedii*, *Fraxinus latifolia*, *Salix lucida*, and *Helenium autumnale*. Species specific cover for each site can be found in Appendix D.

Campbell Slough also showed a general increase in native relative cover shifting from 39±32% to 66±14% (Table 24, Figure 42 & Figure 43). This shift can be accounted for by an increase in native herbs such as *Eleocharis ovata*, *Helenium autumnale*, *Lindernia dubia*, and *Ludwigia palustris* which were found growing in the plots heavily disturbed by grazing. This shift caused by grazing, indicates that these native species are found in the seed bank but are normally (under no grazing) suppressed by more dominate non-native species such as *Phalaris arundinacea* (Kidd 2017). Comparatively, Cunningham lake, which has not experienced heavy grazing since 2017, had a decrease in native cover from 66 in 2018 to 58% in 2019 (Table 24, Figure 42 & Figure 43).This decrease in native cover was accompanied by a general increase in non-native cover including at 22% increase in *Phalaris arundinacea* cover between 2018 and 2019 (Table 21).

Overall, relative native cover at Whites and Welch Islands has remained relatively stable only fluctuating around 10% between 2012-2019. Welch Island (77±23%, long-term average 81±3%) and Whites Island (41±21%, long-term averages 40±6%) experienced almost no change in their average relative native plant cover between 2018-2019 (Table 24, Figure 42 & Figure 43).

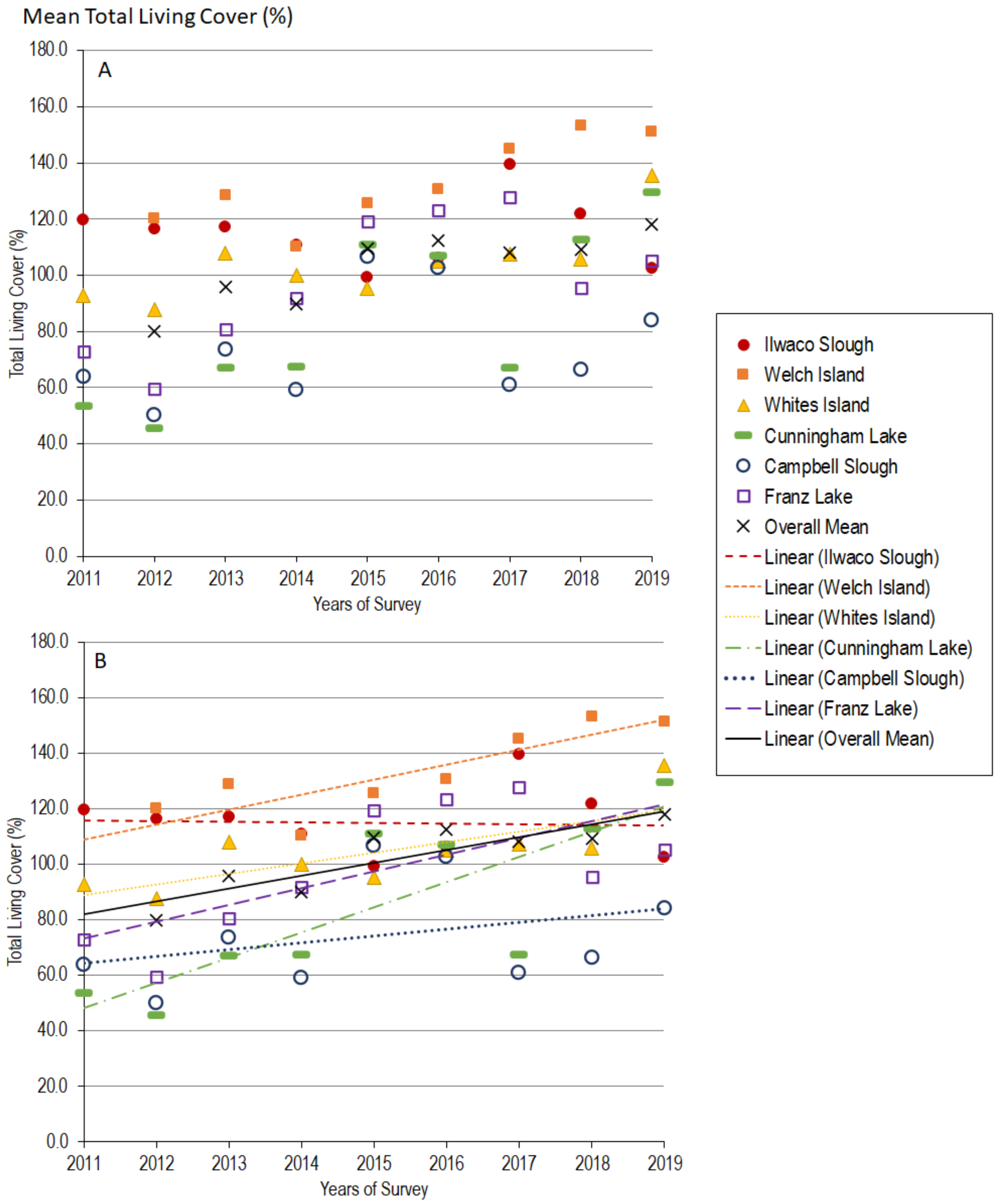


Figure 40. Trends in Total Living % Cover at each site since 2011, when most sites except Welch Island began being monitored. Only the years for which most current trend sites were sampled are shown (2011-2019).

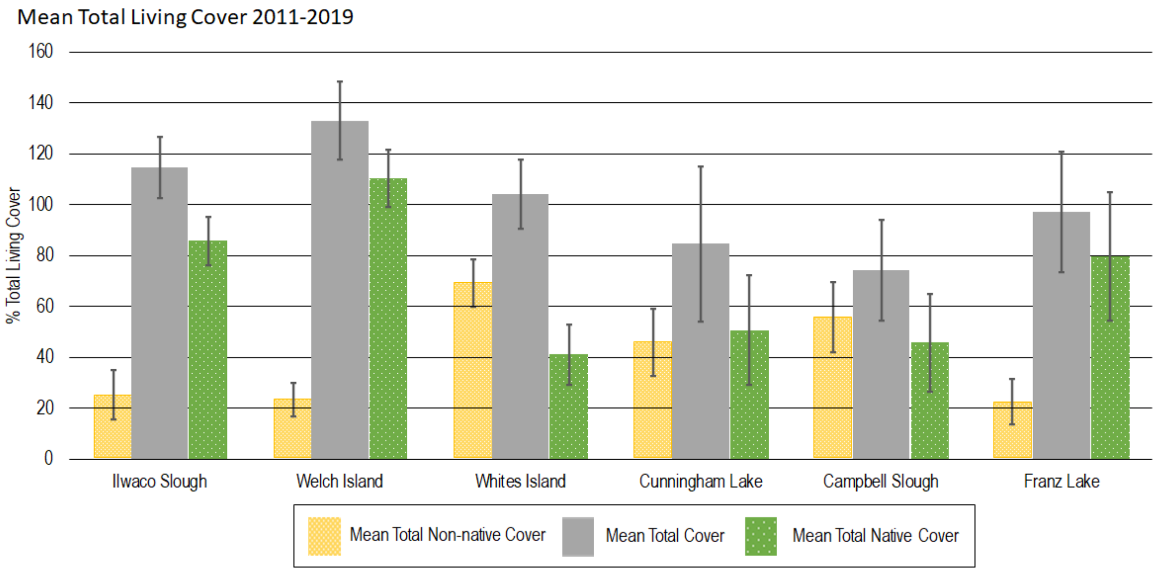


Figure 41. Average % total cover (not relative cover) of vegetation at each trend site, 2011-2019, which is the period during which all six current trend sites were being monitored. Error bars are standard deviations.

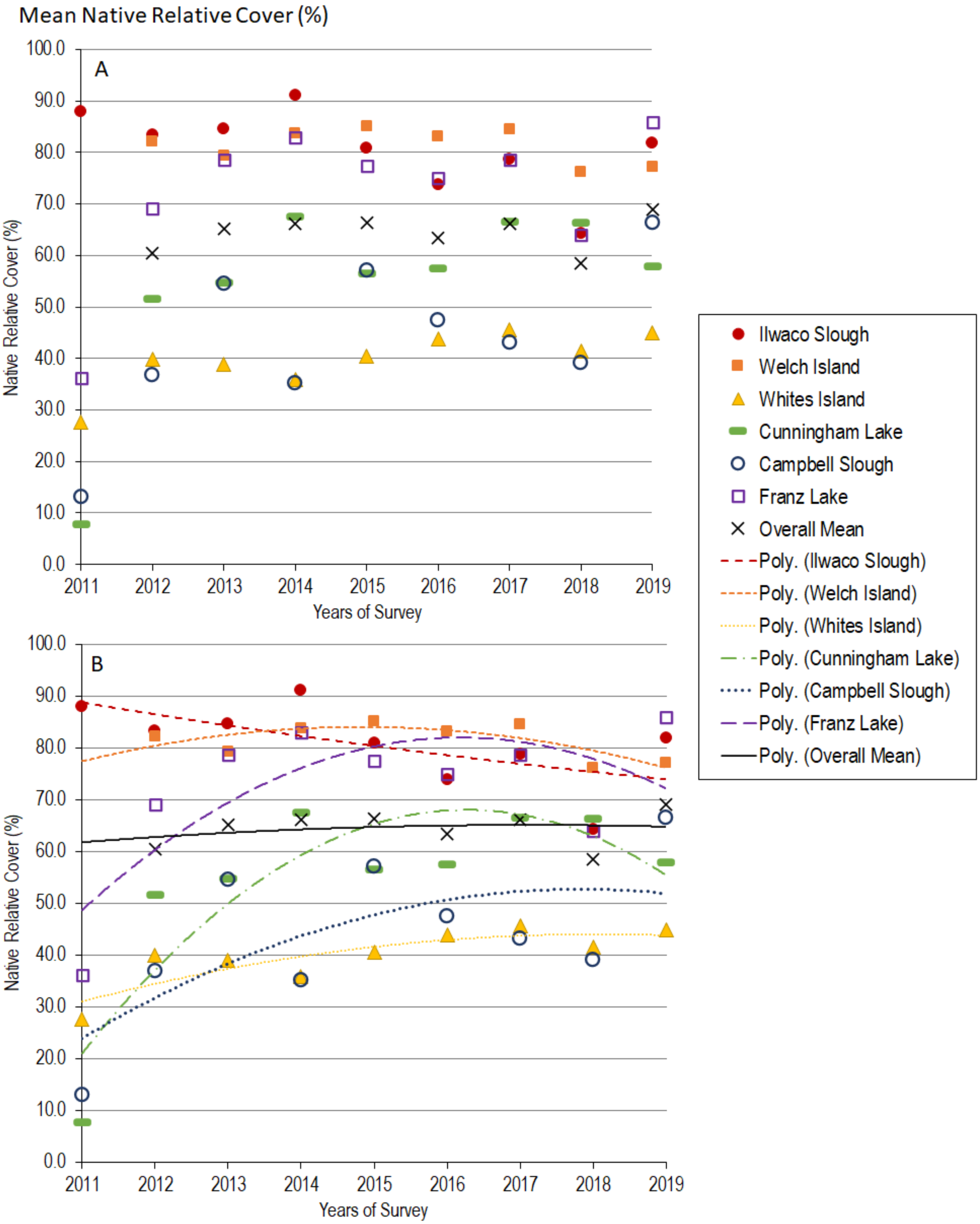


Figure 42. Trends in relative native cover (%) at each site since 2011, when most sites except Welch Island began being monitored. Only the years for which most current trend sites were sampled are shown (2011-2019).

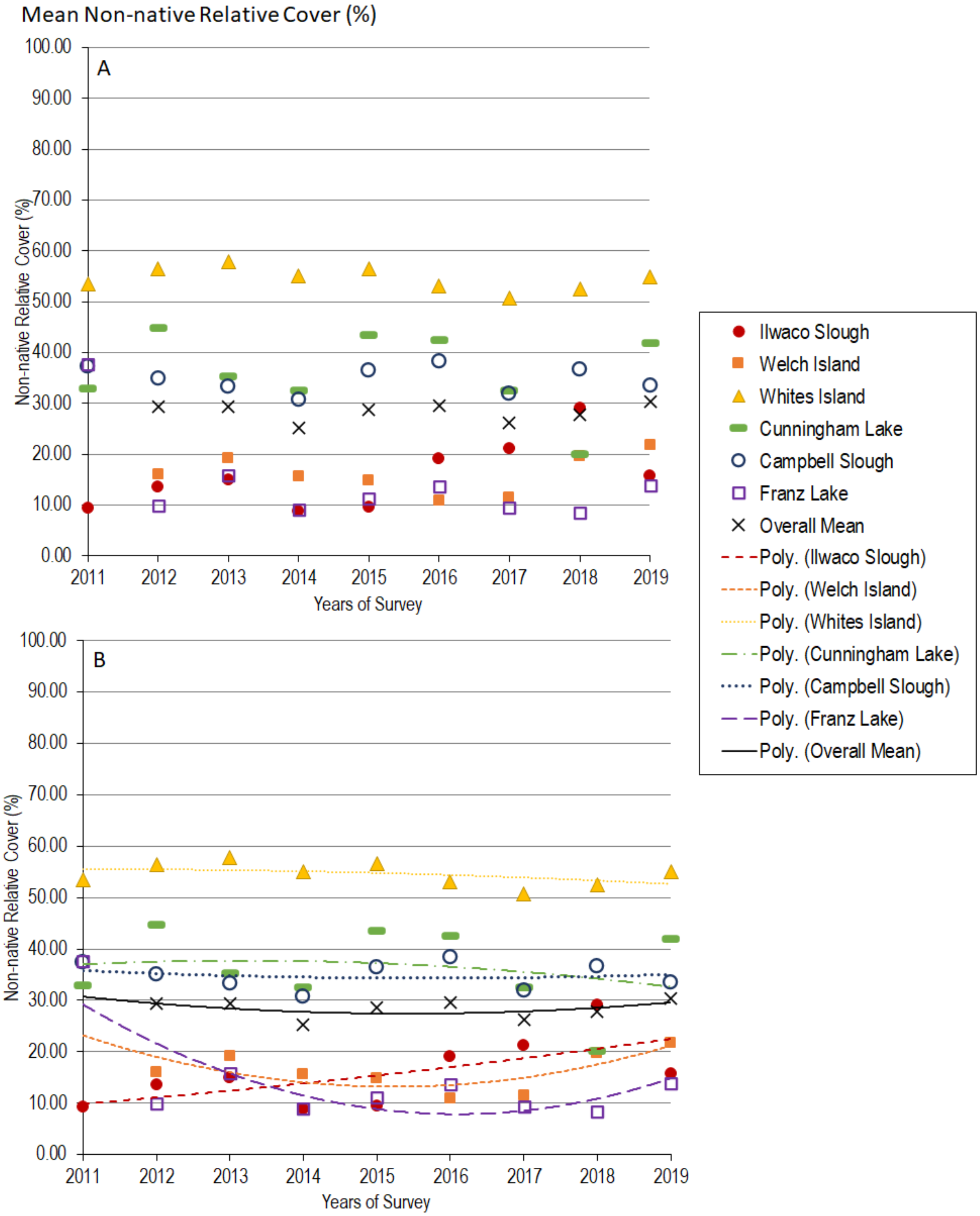


Figure 43: Trends in relative non-native cover (%) at each site since 2011, when most sites except Welch Island began being monitored. Only the years for which most current trend sites were sampled are shown (2011-2019).

Table 24. Changes in average relative % cover of living vegetation and bare ground over time at the six trend sites. Tot=total cover of living plants (not relative cover), Nat=native species relative cover, Non=Non-native species relative cover, OBL=Obligate wetland species relative cover, and BG = bare ground relative cover. Numbers in parentheses are standard deviations.

		Long-term	Year													
Site		**Average**	**19**	**18**	**17**	**16**	**15**	**14**	**13**	**12**	**11**	**10**	**09**	**08**	**07**	**06**	**05**
Ilwaco Slough	Tot	115 (12)	102 (36)	122 (34)	140 (54)	106 (46)	99 (46)	111 (42)	117 (41)	116 (34)	120 (27)						
	BG	8 (7)	5 (7)	6 (14)	3 (6)	21 (25)	20 (27)	3 (8)	6 (15)	5 (9)	2 (5)						
	Nat	81 (8)	82 (20)	64 (25)	79 (21)	74 (26)	81 (26)	91 (15)	85 (21)	83 (19)	88 (13)						
	Non	16 (7)	16 (16)	29 (25)	21 (21)	19 (21)	10 (14)	9 (15)	15 (21)	14 (16)	9 (10)						
	OBL	79 (5)	80 (23)	70 (27)	78 (21)	76 (25)	83 (25)	86 (20)	79 (26)	76 (29)	81 (22)						
Welch Island	Tot	133 (15)	151 (60)	153 (31)	145 (24)	131 (25)	126 (22)	110 (29)	129 (23)	120 (23)							
	BG	3 (2)	0 (1)	0 (1)	1 (3)	6 (7)	3 (3)	6 (10)	3 (5)	4 (8)							
	Nat	81 (3)	77 (23)	76 (19)	84 (20)	83 (22)	85 (23)	84 (24)	79 (26)	82 (20)							
	Non	16 (4)	22 (23)	20 (17)	11 (20)	11 (22)	15 (23)	16 (24)	19 (26)	16 (20)							
	OBL	74 (5)	64 (25)	71 (19)	77 (23)	76 (22)	74 (24)	73 (23)	74 (25)	78 (19)							
Whites Island	Tot	105 (12)	136 (23)	106 (33)	107 (31)	105 (30)	95 (35)	100 (25)	108 (41)	88 (30)	93 (31)	110 (29)	107 (40)				
	BG	7 (4)	2 (5)	3 (10)	3 (6)	8 (14)	5 (11)	9 (17)	8 (19)	16 (23)	14 (24)	9 (18)	5 (13)				
	Nat	40 (5)	45 (28)	41 (21)	46 (28)	44 (32)	40 (32)	36 (25)	39 (33)	40 (29)	28 (28)	39 (29)	47 (33)				
	Non	54 (4)	55 (28)	52 (28)	51 (32)	53 (33)	57 (36)	55 (30)	58 (36)	56 (31)	53 (26)	61 (29)	46 (32)				
	OBL	44 (4)	50 (28)	52 (28)	43 (31)	44 (31)	43 (34)	39 (29)	41 (33)	48 (30)	38 (30)	44 (31)	44 (34)				
Cunningham Lake	Tot	86 (26)	129 (43)	113 (54)	67 (39)	107 (32)	111 (38)	68 (43)	67 (41)	46 (29)	54 (36)	99 (40)	110 (50)	75 (46)	90 (45)	56 (35)	103 (27)
	BG	17 (16)	7 (18)	19 (29)	25 (30)	8 (12)	12 (22)	23 (18)	32 (31)	59 (21)	31 (31)	16 (25)	15 (25)	0 (0)	2 (13)	0 (0)	0 (0)
	Nat	54 (15)	58 (36)	66 (29)	66 (29)	57 (36)	56 (36)	67 (38)	55 (43)	52 (42)	8 (15)	39 (37)	62 (40)	63 (39)	45 (29)	67 (39)	55 (46)
	Non	38 (8)	42 (36)	20 (26)	33 (28)	43 (36)	44 (36)	32 (38)	35 (42)	45 (42)	33 (31)	52 (42)	38 (40)	37 (39)	35 (36)	32 (39)	44 (46)
	OBL	53 (8)	52 (39)	70 (35)	55 (35)	52 (38)	52 (39)	50 (44)	53 (46)	44 (44)	35 (43)	42 (44)	57 (42)	56 (44)	58 (38)	63 (43)	54 (47)
Campbell Slough	Tot	76 (19)	84 (31)	66 (32)	61 (29)	103 (22)	107 (22)	59 (44)	74 (38)	50 (35)	64 (37)	94 (24)	102 (29)	61 (42)	59 (33)	62 (30)	93 (31)
	BG	21 (17)	22 (17)	34 (30)	41 (26)	11 (14)	6 (9)	44 (37)	21 (23)	54 (24)	38 (33)	12 (18)	13 (22)	0 (0)	25 (30)	0 (0)	0 (0)
	Nat	48 (14)	66 (41)	39 (32)	43 (35)	47 (37)	57 (42)	35 (34)	54 (41)	37 (28)	13 (14)	48 (41)	61 (45)	61 (40)	40 (41)	61 (46)	58 (47)
	Non	37 (4)	34 (41)	37 (44)	32 (42)	38 (44)	37 (43)	31 (40)	33 (40)	35 (39)	37 (41)	42 (46)	39 (44)	37 (41)	40 (46)	39 (46)	42 (47)
	OBL	60 (4)	63 (44)	61 (45)	68 (43)	61 (45)	59 (46)	67 (43)	64 (43)	61 (45)	51 (49)	56 (47)	58 (48)	61 (46)	61 (47)	60 (47)	54 (47)
Franz Lake	Tot	99 (21)	105 (68)	95 (62)	128 (55)	123 (48)	119 (53)	92 (47)	81 (44)	60 (35)	73 (35)		106 (51)	105 (54)			
	BG	16 (12)	0 (2)	8 (12)	13 (20)	14 (17)	10 (20)	24 (22)	32 (23)	36 (29)	25 (25)		17 (27)	0 (0)			
	Nat	70 (14)	86 (20)	64 (38)	79 (25)	75 (24)	77 (26)	83 (23)	79 (28)	69 (34)	36 (25)		66 (34)	61 (32)			
	Non	18 (12)	14 (20)	8 (15)	9 (13)	14 (20)	11 (16)	9 (18)	16 (24)	10 (14)	38 (32)		34 (33)	39 (32)			
	OBL	57 (10)	64 (39)	55 (41)	63 (36)	59 (37)	56 (36)	63 (38)	56 (39)	71 (36)	51 (34)		58 (37)	31 (33)			

Trends in plant community composition

Between 2012-2019 the six most common plant species identified throughout the tidal estuary (across the 6 trend sites) in order of overall abundance are *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Carex lyngbyei* (CALY, native), lyngby sedge, *Eleocharis palustris* (ELPA, native), common spikerush, *Sagittaria latifolia* (SALA, native), wapato, *Leersia oryzoides* (LEOR, native), rice cut grass, and *Ludwigia palustris* (LUPA, native), water purslane (Table 25, Figure 44 & Figure 45). While these species are the most common and abundant across all sites over the years, they are not necessarily present at all sites every year. For example, *P. arundinacea* does not grow at Ilwaco, likely due to the saline conditions present at this wetland (Kidd 2017). However it is found growing in abundance at all the other trend sites across the lower river (Table 26 & Table 27, Figure 44). In 2019, *P. arundinacea* cover levels stayed relatively consistent to those observed in 2018, however, at Cunningham, there was a significant increase in *P. arundinacea* levels from 17 to 38% and Franz Lake also experience a small increase from 8.7 to 13.1% (Table 26 & Table 27, Figure 44). *P. arundinacea* frequency (spread across the site) also increased at Cunningham, but only slightly from 75 to 79 plots, and overall *P. arundinacea* frequency dropped at Franz Lake from 70 to 60 plots (Table 26). This shift in *P. arundinacea* levels observed at Cunningham and Franz Lake is likely a product of both low freshet flooding conditions in 2019 (Figure 49) and, at Cunningham Lake, a break from grazing pressure. The last several years cattle have heavily grazed Cunningham Lake wetlands, however reduced grazing activity was observed in 2019. It is well known that cattle pressure can significantly reduce *P. arundinacea* abundance during the growing season (Kidd 2017). Generally, *P. arundinacea* abundance has been found to decrease in years of greater freshet discharge levels, especially in Cunningham Slough, Campbell Slough, and Franz Lake where wetland water levels are tightly correlated with Columbia River discharge conditions, higher water levels making growing conditions less favorable for *P. arundinacea* (Figure 49).

C. lyngbeyi grows in abundance at Ilwaco Slough, Welch Island, and Whites Island but is not found at Cunningham Lake, Campbell Slough, or Franz Lake (Table 27, Figure 44). In 2019, *C. lyngbeyi* levels increased slightly at Ilwaco and Whites, and decreased across Welch Island compared to 2018 levels. All sites remained below long-term average cover conditions. Ilwaco Slough *C. lyngbeyi* cover increased by 3%, Welch Island declined by 7%, and Whites Island increased by 2% (Table 27, Figure 44). Generally, *C. lyngbeyi* abundance has been found to increase in years of greater freshet discharge levels, especially in Ilwaco Slough, where salinity levels are reduced during large discharge years, making growing conditions more favorable for *C. lyngbeyi* (Figure 45).

E. palustris is the only species found growing across all trend sites, however its abundance does range widely, only being found in trace amounts at Ilwaco Slough, and low levels at Whites Island, Welch Island, and Franz Lake, while it is generally found in abundance at Cunningham Lake and Campbell Slough (Table 27, Figure 44). In 2019, *E. palustris* levels were similar to 2018 and close to long-term averages at Ilwaco Slough (<0.1%), Welch Island (5%), and Whites Island (2%), *E. palustris* levels were below the long-term average and 2018 levels at Franz Lake shifting from 2 to 1% cover (long-term average 6±3%) and Campbell Slough shifting from 16 to 23% (long-term average 24±7%) (Table 27, Figure 44). At Cunningham Lake *E. palustris* levels were up compared to 2018 cover levels from 18 to 24% (long-term average 19±12%) (Table 27, Figure 44).

S. latifolia follows a similar trend to *E. palustris*. However, it is not found growing in Ilwaco Slough. In 2019, *S. latifolia* levels were similar to 2018 at Welch Island (4%, 2018 4±5%), Cunningham Lake (8%, 2018 11±9%), and Campbell Slough (5%, 2018 6±10%). Comparatively, *S. latifolia* levels were much lower at Whites Island (8%, 2018 12±16%), Franz Lake (3%, 2018 8±23%) (Table 27, Figure 45). *S. latifolia* has been found to have a delayed reaction to freshet conditions, with lower freshet conditions resulting in an increase in *S. latifolia* abundance the following year. Given this strong reaction to lower freshet conditions it is anticipated that *S. latifolia* levels will increase in 2020 in reaction to the low freshet conditions which occurred in 2019 (lower freshet condition result in lower wetland water levels, which expose more low marsh areas to light and air during the growing season).

L. palustri is a common and consistent species found across all sites (minus Ilwaco Slough), typically averaging less than 5% in overall relative cover. In 2019, *L. palustri* was not found on Welch, Whites, and Franz Lake. Comparatively, *L. palustri* levels were much lower at Cunningham Lake (2%, 2018 16±19%), and increased at Campbell Slough (4%, 2018 0.5±1%) (Table 27, Figure 45).

Leersia oryzoides, is also a common and consistent species found across all sites (minus Ilwaco Slough), typically averaging less than 3% in overall relative cover, 2019 levels were similar to those observed in 2018 across all sites (Table 27, Figure 45).

While not a common species across all the trend sites *Polygonum amphibium* (POAM, native), water knotweed, it is an important species because it has become dominant at Franz Lake. *P. amphibium* levels have significantly increased over the last 8 years, taking over dominance from *P. arundinacea* on the site in 2012 (an extreme high-water year for the site Figure 34) and generally increasing in cover (and *P. arundinacea* declining in cover) every year following (Table 27, Figure 44). In 2019 and 2018, *P. amphibium* level remained (34%) slightly below 2017 levels (38%) but higher than the long-term average (28±10%) (Table 27, Figure 45).

Between 2011 and 2019, *P. amphibium* levels at Franz lake have also be found to follow a similar trend to *S. latifolia* with a one year delayed reaction to decreased freshet conditions, lower freshet conditions (lower water levels across the wetland site) resulting in an increase in *P. amphibium* cover the following growing year (Figure 45). For both species this might be a result of increased rhizome stores from positive growing conditions (low water levels), providing for more robust growth in the following growing season.

The average percent cover of all plant species at each trend site is provided in Appendix D, along with the wetland elevation ranges for each site.

Table 25. The overall long term 6 most commonly occurring plant species across all six trend sites from 2012-2019. Species are listed in order of overall average relative % cover.

Scientific Name Common Name	Wetland Status	Category	Native	Relative Cover (%) between 2012-2018					
				Ilwaco	Welch	Whites	Cunningham	Campbell	Franz
Phalaris arundinacea reed canarygrass	FACW	grass	no	0.0	6.7	39.8	30.2	26.8	9.4
Carex lyngbyei lyngby sedge	OBL	sedge	yes	45.6	38.3	4.3	0.0	0.0	0.0
Eleocharis palustris common spikerush	OBL	sedge	yes	0.0	3.8	2.0	12.0	21.4	4.2
Sagittaria latifolia wapato	OBL	herb	yes	0.0	4.2	9.3	12.7	10.3	3.8
Leersia oryzoides rice cut grass	OBL	Grass	yes	0.0	0.9	1.5	1.9	0.3	2.1
Ludwigia palustris water purslane	OBL	herb	yes	0.0	0.1	0.01	4.2	1.9	0.3

Table 26. ***Phalaris arundinacea*** **(reed canarygrass) average relative % cover and frequency (% of sample plots) at the trend sites between 2005 and 2019. ND indicates No Data.**

Site (Rkm)	**Ilwaco Slough** (6)		**Welch Island** (53)		**Whites Island** (72)		**Cunningham Lake** (145)		**Campbell Slough** (149)		**Franz Lake** (221)	
	Rel Cover	%Freq	Rel Cover	%Freq	Rel Cover	%Freq	Rel Cover	%Freq	Rel Cover	%Freq	Rel Cover	%Freq
2005	ND	ND	ND	ND	ND	ND	44.2	63	41.1	60	ND	ND
2006	ND	ND	ND	ND	ND	ND	31.6	61	38.9	48	ND	ND
2007	ND	ND	ND	ND	ND	ND	35.6	77	29.1	48	ND	ND
2008	ND	ND	ND	ND	ND	ND	35.2	60	33.0	47	38.1	93
2009	ND	ND	ND	ND	36.6	80	34.9	55	35.7	48	29.9	71
2010	ND	ND	ND	ND	44.2	86	50.4	69	40.0	58	ND	ND
2011	0.0	0.0	ND	ND	52.8	88	41.2	75	40.4	62	29.8	81
2012	0.0	0.0	5.8	17	39.8	83	22.1	62	14.0	49	6.6	69
2013	0.0	0.0	8.2	13	43.1	76	31.1	61	28.6	44	11.7	63
2014	0.0	0.0	7.4	38	45.2	88	22.3	64*	24.8	48	7.4	46
2015	0.0	0.0	5.5	25	45.9	77	41.4	67	34.8	50	10.2	48
2016	0.0	0.0	7.2	23	42.5	80	40.1	75	32.4	51	12.0	51
2017	0.0	0.0	6.5	32	41.6	82	24.7	81	24.7	42	8.1	75
2018	0.0	0.0	7.3	63	30.1	89	17.3	75	29.1	54	8.7	70
2019	0.0	0.0	5.9	41	30.1	87	38.6	79	26.1	60	13.1	60

*A different sampling design was used at Cunningham Lake in 2014, so results are not directly comparable to the other years.

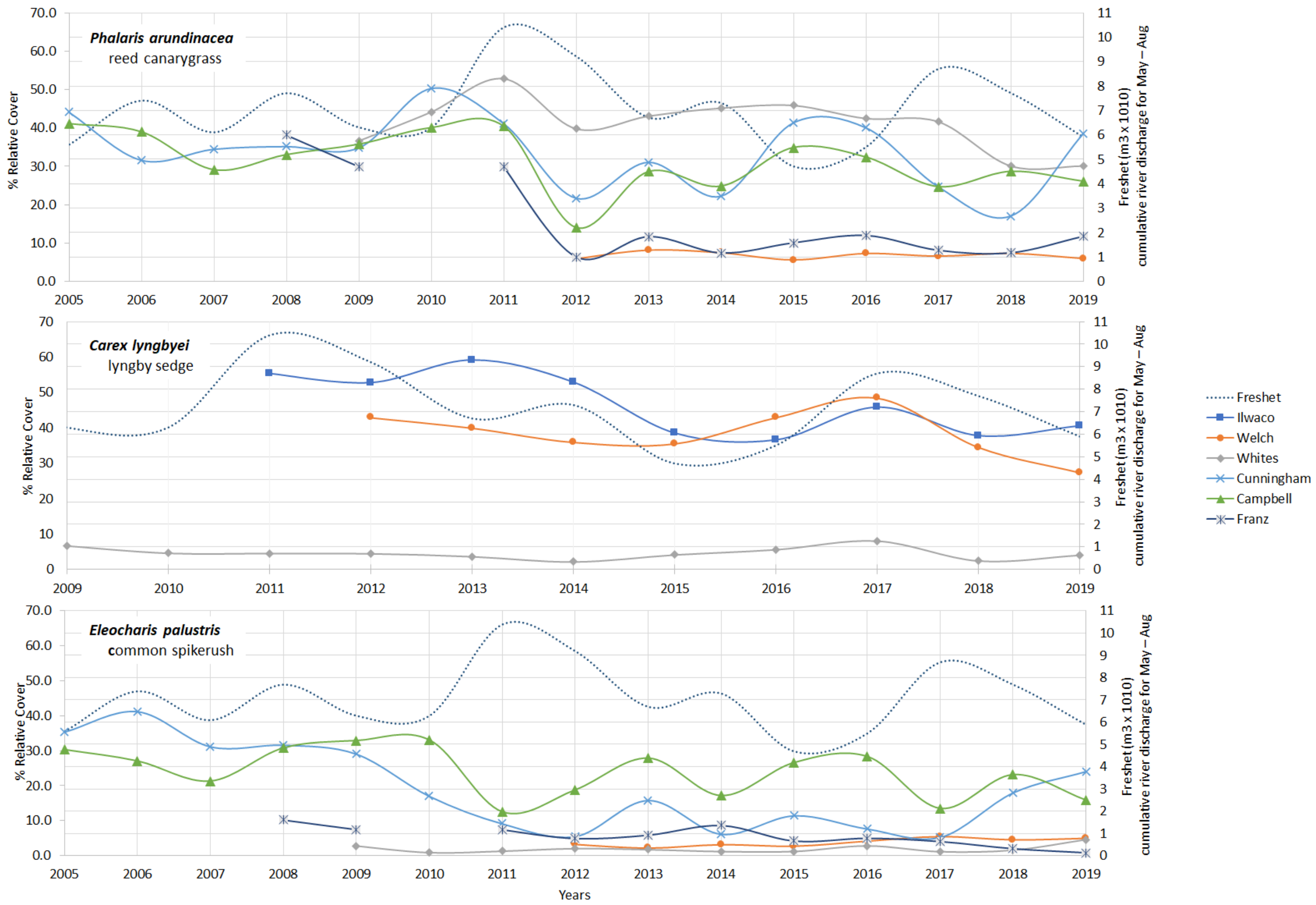


Figure 44: Annual mean relative % cover for *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Carex lyngbyei* (CALY, native), lyngby sedge for all trend sites. Annual cumulative river discharge from May-Aug included for annual water year context (see Section 3.3.1). Relative species cover data can also be found in Table 27.

Figure 45: Annual mean relative % cover for *Eleocharis palustris* (ELPA, native) common spikerush, *Sagittaria latifolia* (SALA, native), wapato, *Leersia oryzoides* (LEOR, native), rice cut grass, *Ludwigia palustris* (LUPA, native), water purslane, and *Polygonum amphibium* (POAM, native), water knotweed, for all trend sites. Annual cumulative river discharge from May-Aug included for annual water year context (see Section 3.3.1). Relative species cover data can also be found in Table 27.

Table 27: Overall total cover of dominant and common plant species found across all six trend sites, standard deviation included in parentheses. *Carex lyngbyei* (CALY, native), lyngby sedge, *Eleocharis palustris* (ELPA, native), common spikerush, *Leersia oryzoides* (LEOR, native), rice cut grass, *Ludwigia palustris* (LUPA, native), water purslane, *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Sagittaria latifolia* (SALA, native), wapato, and *Polygonum amphibium* (POAM, native), water knotweed.

Site	Plant Code	Mean	19	18	17	16	15	14	13	12	11	10	09
Ilwaco Slough	CALY	47 (9)	41 (30)	38 (30)	46 (31)	37 (29)	39 (33)	53 (37)	59 (38)	53 (40)	55 (39)		
	ELPA	0 (0.1)							0.3 (2)				
Welch Island	CALY	38 (7)	27 (17)	34 (20)	48 (26)	43 (26)	36 (25)	36 (27)	40 (27)	43 (25)			
	ELPA	4 (1)	5 (11)	5 (12)	5 (12)	4 (10)	3 (11)	3 (7)	2 (8)	3 (8)			
	LEOR	0.9 (0.6)	1 (2)	2 (3)	0.3 (1)	0.7 (1)	0.8 (1)	0.6 (1)	0.8 (1)	0.9 (2)			
	LUPA	0.1 (0.2)								0.5 (2)			
	PHAR	7 (1)	6 (19)	7 (18)	7 (18)	7 (21)	6 (16)	7 (21)	8 (24)	6 (19)			
	SALA	4 (1)	4 (5)	4 (5)	3 (6)	6 (7)	6 (8)	4 (6)	3 (5)	4 (6)			
Whites Island	CALY	5 (2)	4 (11)	2 (6)	8 (19)	6 (14)	4 (14)	2 (5)	4 (11)	4 (9)	5 (10)	5 (7)	7 (17)
	ELPA	2 (1)	5 (12)	2 (6)	1 (4)	3 (9)	1 (7)	1 (5)	2 (5)	2 (5)	1 (3)	0.9 (2)	3 (8)
	LEOR	1 (1)	2 (4)	2 (4)	2 (8)	2 (4)	0.7 (2)	1 (3)		0.7 (2)	0.5 (2)		
	LUPA	0 (0.1)											0.2 (1)
	PHAR	41 (7)	30 (22)	30 (25)	42 (30)	42 (30)	46 (33)	45 (32)	43 (32)	40 (32)	53 (31)	44 (32)	37 (30)
	SALA	8 (3)	8 (12)	12 (16)	9 (13)	8 (10)	14 (20)	9 (13)	9 (19)	6 (11)	4 (8)	4 (6)	8 (19)
Cunningham Lake	ELPA	19 (12)	24 (27)	18 (18)	5 (7)	8 (12)	11 (17)	6 (12)	16 (24)	6 (8)	9 (19)	17 (21)	29 (27)
	LEOR	1 (2)	2 (4)	3 (4)	0.2 (2)	4 (9)	3 (8)					0.7 (2)	0.8 (3)
	LUPA	4 (4)	2 (4)	16 (19)	4 (7)	2 (5)	2 (5)	2 (5)	1 (3)	0.4 (1)	0.4 (3)	2 (4)	6 (7)
	PHAR	34 (9)	39 (37)	17 (27)	25 (27)	40 (36)	41 (37)	22 (30)	31 (40)	22 (23)	41 (36)	50 (42)	35 (39)
	SALA	11 (7)	8 (8)	11 (9)	20 (19)	22 (21)	20 (17)	1 (3)	3 (5)	5 (5)	7 (11)	6 (7)	7 (7)
Campbell Slough	ELPA	24 (7)	16 (20)	23 (29)	13 (18)	28 (30)	27 (28)	17 (25)	28 (28)	19 (23)	13 (18)	33 (34)	33 (33)
	LEOR	0.2 (0.3)		0.1 (0)		0.7 (2)	0.4 (1)	0.9 (4)	0.1 (0)				0.1 (1)
	LUPA	2 (2)	4 (6)	0.5 (1)	1.1 (2)	1.5 (3)	5 (8)	0.2 (1)	3 (6)	0.2 (1)	1 (3)	1 (2)	0.8 (2)
	PHAR	32 (8)	26 (35)	29 (37)	25 (37)	32 (40)	35 (42)	25 (36)	29 (37)	14 (20)	40 (45)	40 (45)	36 (44)
	POAM	0.1 (0.1)			0.1 (1)		0.1 (1)					0.2 (1)	
	SALA	13 (7)	5 (8)	6 (10)	15 (23)	17 (23)	17 (26)	4 (7)	12 (21)	6 (9)	4 (6)	8 (14)	12 (13)
Franz Lake	ELPA	6 (3)	1 (3)	2 (7)	4 (11)	5 (12)	4 (11)	9 (19)	6 (12)	5 (12)	7 (18)		7 (13)
	LEOR	2 (2)	2 (4)	1 (3)	0.9 (2)	2.9 (5)	6 (9)	2.2 (4)	0.9 (1)	1 (2)	1.4 (3)		3.3 (4)
	LUPA	0.6 (1)		0.1 (1)	1.3 (2)				1.4 (3)		0.7 (2)		3.2 (6)
	PHAR	16 (11)	12 (19)	7 (14)	8 (12)	12 (18)	10 (16)	7 (16)	12 (19)	6 (10)	30 (28)		30 (33)
	POAM	28 (10)	34 (40)	34 (37)	38 (36)	37 (35)	35 (36)	33 (35)	23 (27)	30 (35)	15 (21)		15 (24)
	SALA	4 (2)	3 (8)	8 (23)	6 (17)	3 (8)	3 (10)	2 (8)	2 (13)	2 (6)	1 (4)		6 (17)

3.3.3.2 *Drivers of Plant Community Status and Trends*

3.3.3.2.1 Wetland Hydrology

One of the largest mechanisms of change in the tidal wetlands of the Columbia River estuary is the annual timing, duration, and magnitude of the freshet. The freshet typically occurs in early spring following spring rain events which coincide with the melting of the regional snowpack. Since 2011 the freshet floods have occurred as early as March and as late as June. Timing of these annual flood events can determine the amount of time tidal marshes, especially in the upper estuary, are exposed during the growing season. Across all wetlands, a drawdown of flood waters is essential for plant growth and biomass accumulation. Two metrics represent these dynamic flooding conditions and correlate well with plant community change in the upper estuary fluvial dominated zone; 1) the mean daily marsh inundation (% time) during the month of August (see Section 3.3.1) and 2) the annual freshet accumulated discharge.

Comparing mean marsh elevation and common plant species distributions across all EMP sites (except Ilwaco which only shares *C. lyngbyei* in common), it is clear that within wetland plant community zonation is similar, even if the elevations (m, NAVD88) in which these species are found are different, increasing from the lower river sites to the upper river sites (Figure 46). By evaluating the mean % time these plant communities are inundated daily across all the sites (Figure 47), it is clear that inundation rates are important and consistent drivers of plant community establishment within each site. The graph of common species shows that they are found growing within similar inundation zones across the sites. However, the variability of daily inundation range increases from the lower river sites to the upper river sites (Figure 47). This variability is not surprising as hydrology in the upper river sites can heavily depend on water year and the freshet conditions (see Section 3.3.1).

Combining data from Welch Island, Whites Island, Cunningham Lake, and Campbell Slough, it was found that annual mean % *P. arundinacea* was strongly correlated with mean % daily inundation for the month of August across all sites (Figure 48). These data may provide useful information for future restoration planning, sites with an anticipated >30 % daily inundation (in August) rate may provide improved habitat opportunities for native wetland species such as *Carex lyngbyei* (CALY, native), lyngby sedge, *Eleocharis palustris* (ELPA, native), and *Sagittaria latifolia* (SALA, native), wapato. The 30% daily inundation rate appears to be an important lower threshold in the distribution of *P. arundinacea* observed across multiple sites in the estuary (Figure 47 and Figure 48).

The annual Columbia river freshet accumulated discharge was also found strongly correlated with the year to year variability of % *P. arundinacea* cover within each site, greater freshet levels corresponding with lower % *P. arundinacea* cover at Cunningham Lake ($R^2 = 0.70$), Campbell Slough ($R^2 = 0.75$), and Franz Lake ($R^2 = 0.61$) (Figure 49). Indicating that annual flooding conditions within sites (% daily inundation) and across the river (freshet accumulated discharge) are important mechanisms driving much of the observed annual variability in *P. arundinacea* dominance across the estuary. These data are supporting the hypothesis that annual flooding conditions in the Columbia can dramatically impact year to year shifts in plant community dynamics, especially the non-native species *P. arundinacea* in the upper river sites. *P. arundinacea* mean annual cover was also found to be tightly negatively correlated with native plant community cover across all river zones except the mouth (Ilwaco has no *P. arundinacea* due to high salinity levels), annual increases in *P. arundinacea* resulting in an overall decrease in native plant cover (Figure 50). Summarizing these findings, site level daily inundation patterns in addition to season freshet flooding conditions are important drivers of native and non-native plant species across the EMP sites.

Dominant native wetland species was also found to have significant long-term trends in abundance tied to annual freshet conditions (water levels across sites during the growing season) including *Carex lyngbyei*

(CALY, native), lyngby sedge, *Sagittaria latifolia* (SALA, native), wapato, and *Polygonum amphibium* (POAM, native), water knotweed which has been discussed briefly in the section above (3.3.3.1). Further exploration of these long-term ecological relationships will be reported in the FY21 EMP synthesis report.

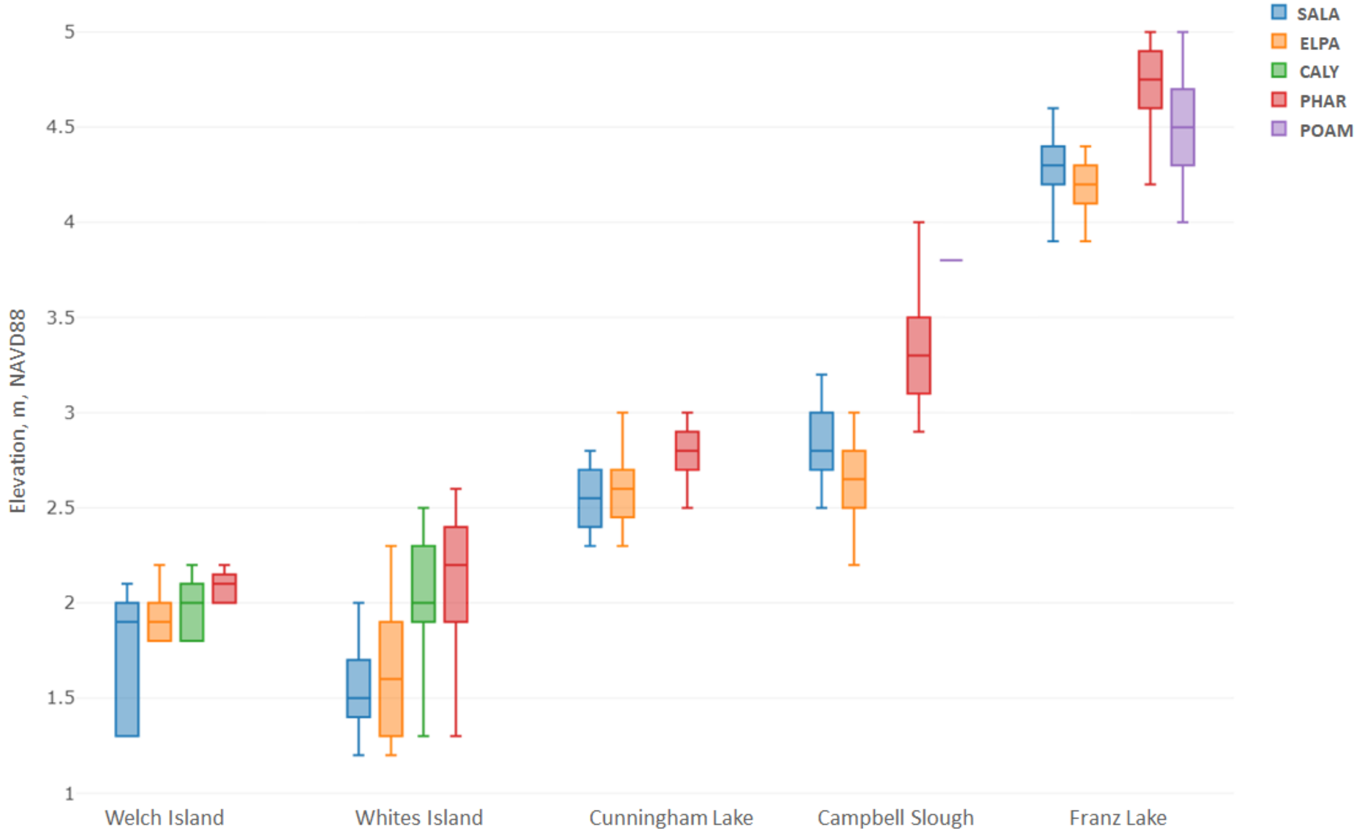


Figure 46: Box plot of elevation range of plant species across each site, sites in order from lower river to upper river from left to right. Species codes: *Carex lyngbyei* (CALY, native), lyngby sedge, *Eleocharis palustris* (ELPA, native), common spikerush, *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Sagittaria latifolia* (SALA, native), wapato, and *Polygonum amphibium* (POAM, native), water knotweed.

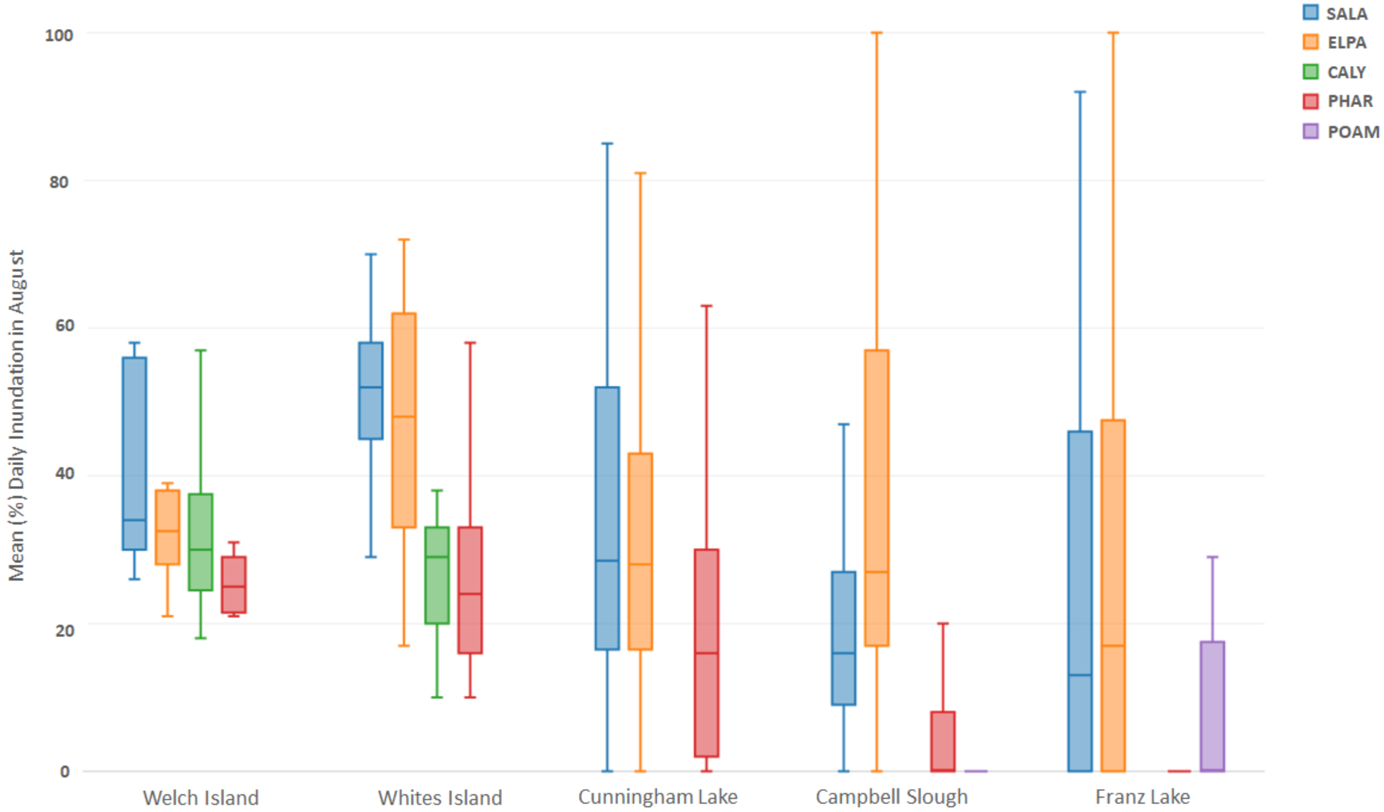


Figure 47: Box plot of daily mean inundation range (%, August - across all years) of plant species across each site, sites in order from lower river to upper river from left to right. Species codes: *Carex lyngbyei* (CALY, native), lyngby sedge, *Eleocharis palustris* (ELPA, native), common spikerush, *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Sagittaria latifolia* (SALA, native), wapato, and *Polygonum amphibium* (POAM, native), water knotweed.

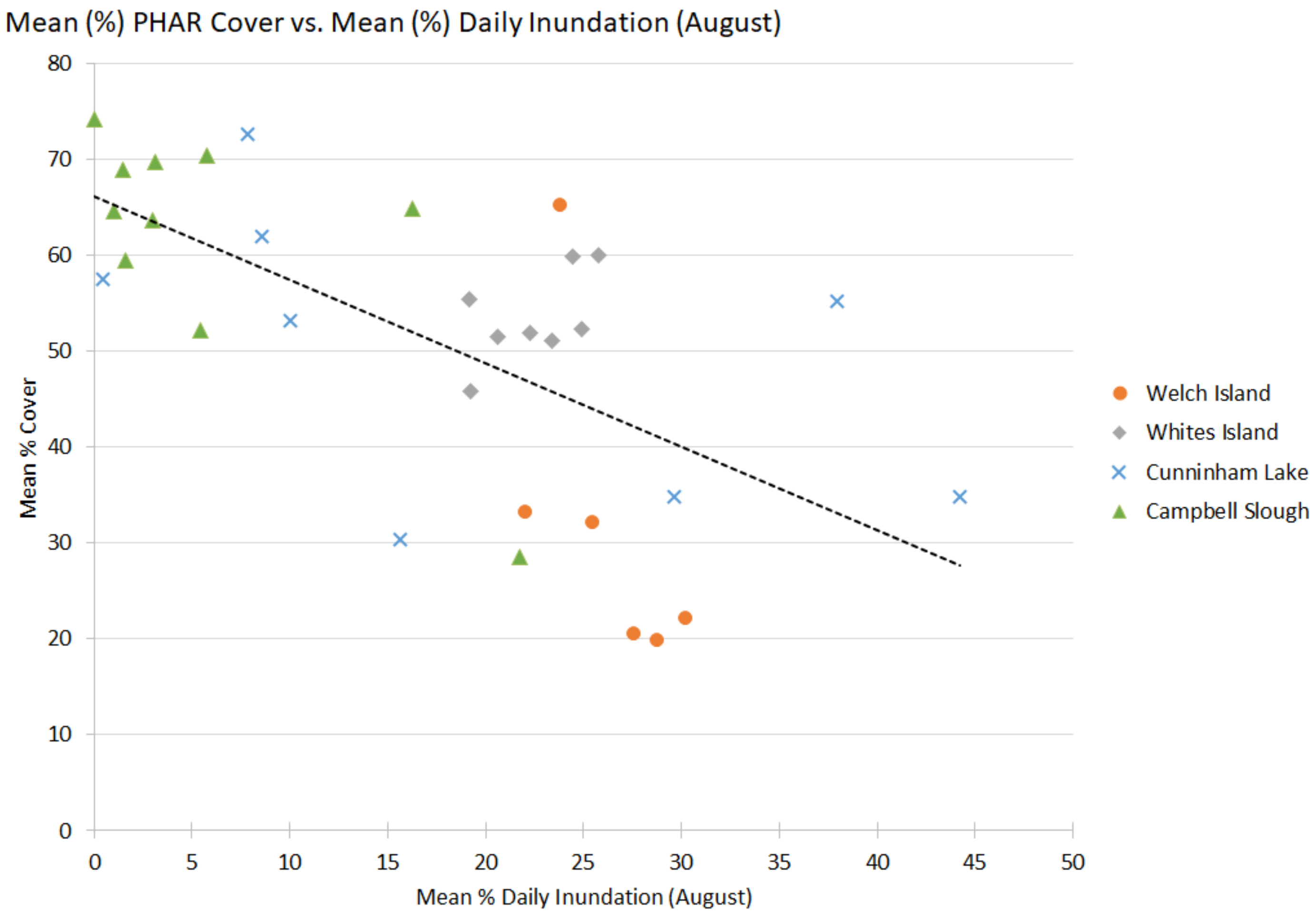


Figure 48: Mean Annual (%) *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, vs. daily (%) inundation (August –2012-2018) of PHAR plots. Linear regression (y = -0.8711x + 66.207), $R^2 = 0.41$, p <0.001.

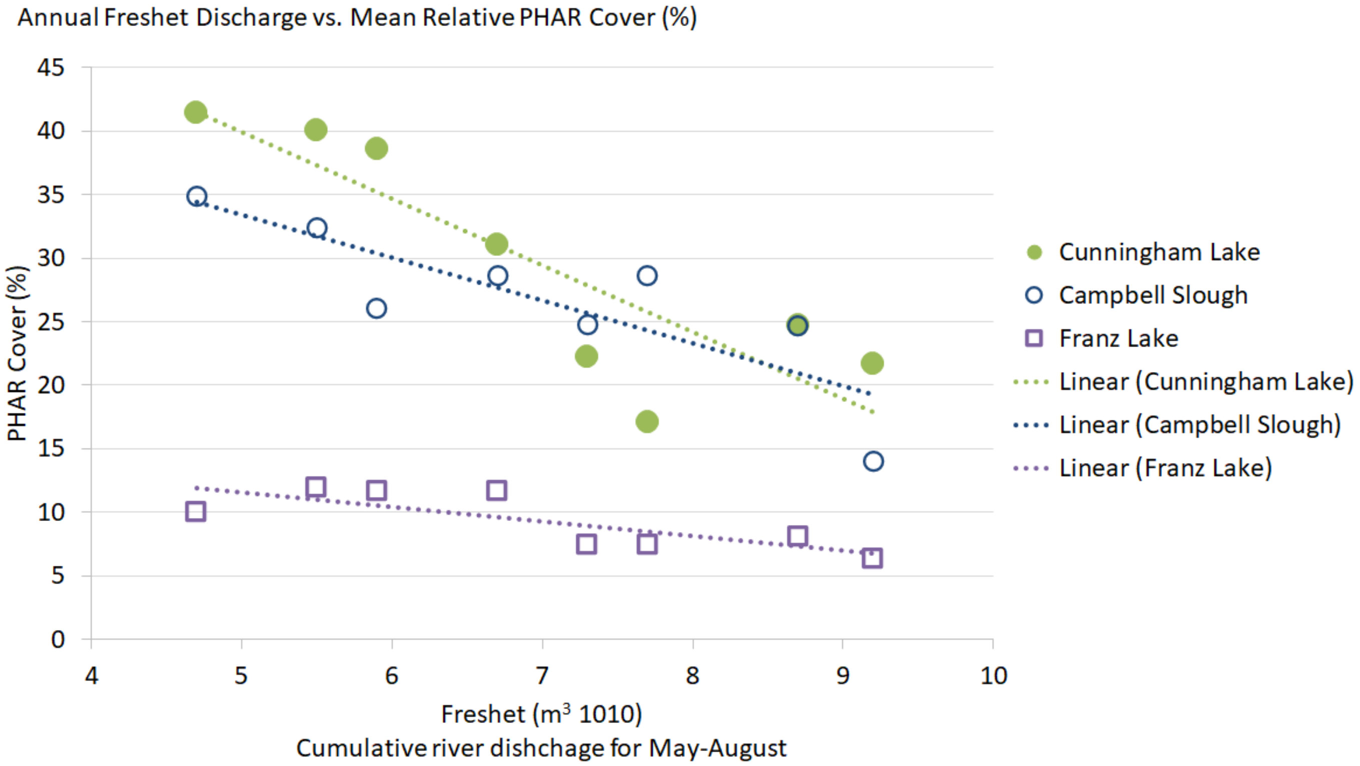


Figure 49: Annual Freshet Cumulative River Discharge (Bonneville Dam, May- August) vs. Mean Annual (%) *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, (2012-2019). Cunningham Lake, linear regression (y = -3.36x + 50.2), R² = 0.70, p <0.001; Campbell Slough linear regression (y = -5.25x + 66.1) R² = 0.75, p <0.001; Franz Lake, linear regression (y -1.13x + 17.2), R² = 0.61, p<0.001.

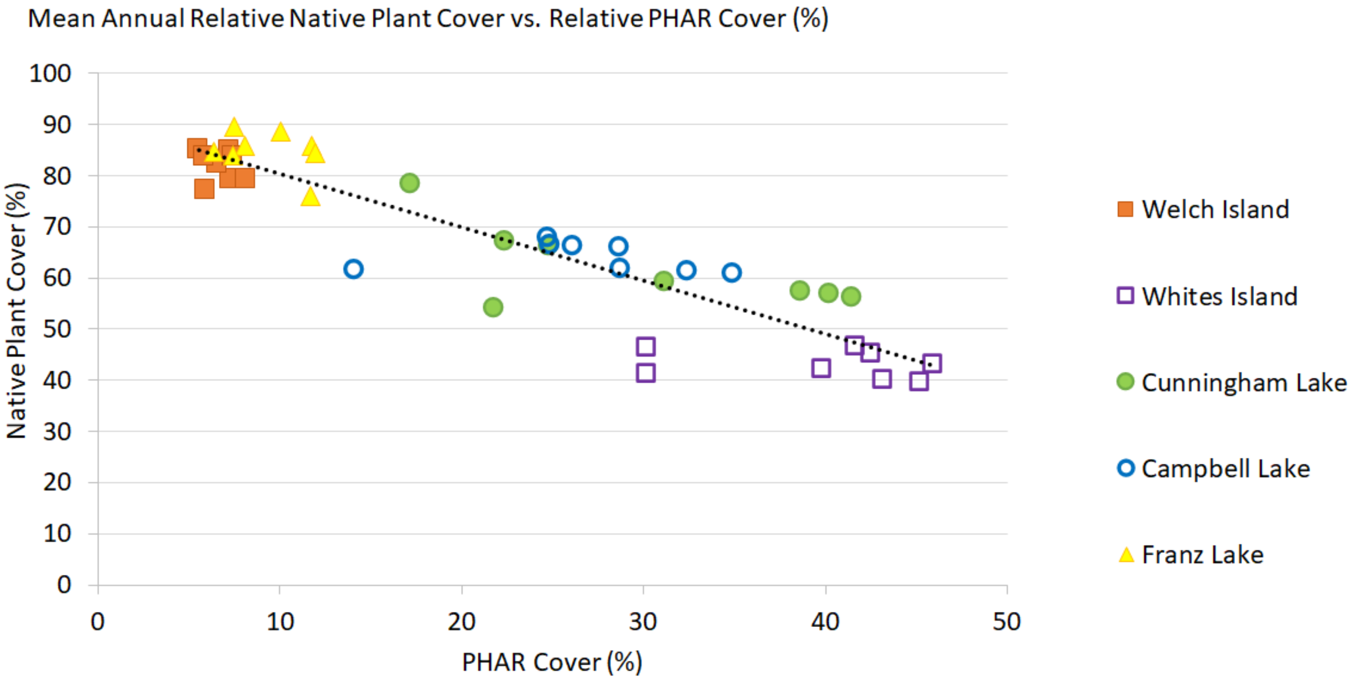


Figure 50: Mean Annual Relative Native Plant Cover vs. Mean (%) *Phalaris arundinacea* (PHAR, non-native), reed canarygrass cover (August –2012-2019). Linear regression ($y = -1.0443x + 90.774$), $R^2 = 0.83$, $p < 0.001$.

3.3.4 Channel Morphology

Channel morphology at the trends sites exhibited low inter-annual variability in years prior to 2016; therefore, only the channel mouth cross section was surveyed in 2016. Channel measurements from previous years are presented with the newly calculated inundation frequency results from 2016 in Table 28. The tidal channels measured at the sites were generally small, with most cross-sectional areas less than 10m^2 (see Appendix A for locations of the measured channels). Five of the tidal channels surveyed were primary channels feeding directly into the Columbia River, while the channels at the Welch and Whites Island sites were secondary channels that feed into a larger tidal channel. The channels varied in width from 1.3 m to 50.1 m; most becoming narrower with increasing elevation, with the exception of the Ilwaco Slough and Whites Island channels, which were slightly wider at the middle than at the mouth. Channel depth ranged from 0.3 m to 2.1 m, with most channels between 0.9 m and 1.2 m in depth. The Thalweg elevation of the channels was generally between 0.0 and 1.0 m and the channel bank between 1.0 and 2.0 m, relative to CRD.

Table 28. Physical channel metrics measured at each site. The channel mouth (indicated with an *) was measured in 2016; the year of full channel cross section measurement is provided in parentheses after the site code. Channel cross-section and hydrology data collected in 2017 are still under analysis and unavailable at the time this report was written, no cross-section data was collected in 2018 and 2019. The text below is adapted from the 2016 report.

		Physical Metrics					
Site (year)	**Cross Section**	**Thalweg Elevation (m, CRD)**	**Bank Elevation (m, CRD)**	**Channel Depth (m)**	**Cross Section Area (m^2)**	**Channel Width (m)**	**Width:Depth Ratio**
Ilwaco Slough (11)	1*	0.87	1.56	0.69	3.3	6.2	9.0
	2	0.70	1.86	1.16	8.94	9.30	8.04
	3	0.90	2.12	1.22	9.73	10.10	8.27
	4	1.01	2.00	0.99	4.33	5.20	5.23
	5	1.17	2.26	1.09	1.58	2.70	2.48
Welch Island (12)	1*	0.30	1.51	1.21	13.0	20.4	16.9
	2	0.36	1.65	1.29	8.75	9.20	7.13
	3	0.71	1.80	1.09	3.96	5.09	4.67
	4	0.78	1.74	0.96	2.07	3.30	3.44
	5	1.31	1.62	0.31	0.42	1.32	4.27
Whites Island (11)	1*	0.42	1.12	0.70	12.1	34.6	49.4
	2	0.34	1.41	1.07	10.8	20.5	19.1
	3	0.61	1.53	0.92	11.1	36.2	39.5
	4	0.92	1.93	1.00	34.0	50.1	50.0
	5	0.44	1.45	1.01	1.90	2.83	2.80
Cunningham Lake (15)	1	0.82	1.26	0.44	5.5	18.3	41.6
Campbell Slough (15)	1	0.80	1.47	0.67	11.7	23.0	34.3
Franz Lake (12)	0*	0.34	2.23	1.89	21.3	23.2	12.2
	3	0.40	1.39	0.99	4.20	14.3	14.4
	4	0.85	1.45	0.60	6.20	13.2	22.0

3.4 Food Web

3.4.1 Primary Production

3.4.1.1 *Emergent Wetland Vegetation*

Starting in the summer of 2017 detritus sampling was included in the biomass sampling and analysis to evaluate detrital production and export. In the winter of 2018 (and all sampling events to follow), biomass sampling protocols changed slightly to accommodate detrital sampling and streamline data collection. This included shifting from "strata" mixed species designations to simple high and low marsh strata descriptions across all sites sampled. This change has also included species biomass weights to be recorded individually to assess species-specific contributions to each high and low marsh stratum (in the past mixes of species were assessed together). In general, these changes will allow for a more detailed understanding of species-specific biomass contributions and still allow for long-term comparisons to overall site, high and low marsh contributions. These shifts in methods (see methods section for full details) should be considered when interpreting the below analysis. Additionally, while the 2019 data are still under analysis general trends follow what has been seen in previous years and are similar to the 2018 results. In depth analysis synthesizing biomass and soil data across all sites is planned for the FY21 synthesis.

Composition of Biomass

At each long-term monitoring site, aboveground biomass was sampled within the high and low marsh vegetation strata to reduce variability associated with sampling across strata. The dominant species (≥ 5% of the total sample by dry biomass) for the strata are identified in Table 29. Within the high marsh strata *Carex lyngbyei*, CALY, native, was the largest contributor to the biomass production at Welch Island, followed by *Myosotis laxa*, MYLA, native, and *Phalaris arundinacea*, PHAR, non-native. *C. lyngbyei* was also a significant contributor to the high marsh strata of Whites Island. However, this was variable between summer 2017 and 2018 with a shift in sampling locations *P. arundinacea* dominating in the summer of 2018. In the high marsh of Steamboat Slough, an AEMR restoration site restored in 2016, *P. arundinacea* and *Juncus effuses*, JUEF, non-native, dominated the biomass contribution. Franz Lake high marsh strata was dominated by *Polygonum amphibium*, POAM, native, and *P. arundinacea*. The low marsh Strata biomass contribution at Welch Island was dominated by *Polygonum hydropiper*, POHY, non-native, *Equisetum fluviatile*, EQFL, native, *P. arundinacea*, and *Eleocharis palustris*, ELPA, native. The low marsh Strata biomass contribution at Whites Island was dominated primarily by SALA, with additional contributions from *E. fluviatile*, *P. arundinacea*, and *P. hydropiper*. Steamboat Slough's low marsh strata changed dramatically from 2017 to 2018 (likely a response from increases in flooding from restoration) shifting from *J. effuses* and *P. arundinacea* to *Schoenoplectus tabernaemontani*, SCTA, native, *E. palustris*, and *Sagittaria latifolia*, SALA, native. The low marsh strata at Franz Lake was composed primarily by *P. amphibium* and *S. latifolia*.

Table 29. The overall proportion of dominant species (≥ 5% of total sample by dry biomass) across high and low marsh stratums sampled between the summer of 2017 and summer 2018. See Appendix D for all plant code information. Welch Island low marsh strata and Franz lake (both strata) were not sampled until summer 2018. Species codes: *Bidens cernua*, BICE, native, *Caltha palustris*, CAPA, native, *Carex lyngbyei*, CALY, native, *Eleocharis palustris*, ELPA, native, *Equisetum fluviatile*, EQFL, native, *Juncus effuses*, JUEF, non-native, *Myosotis laxa*, MYLA, native, and *Phalaris arundinacea*, PHAR, non-native, *Polygonum amphibium*, POAM, native, *Polygonum hydropiper*, POHY, non-native, *Sagittaria latifolia*, SALA, native, *Schoenoplectus tabernaemontani*, SCTA, native.

		Welch Island (Rkm 53)				Steamboat Slough (Rkm 57) (AEMR Site, Restored 2016)				Whites Island (Rkm 72)				Franz Lake (Rkm 221)
	Species Codes	**Sum 17**	**Win 18**	**Spr 18**	**Sum 18**	**Sum 17**	**Win 18**	**Spr 18**	**Sum 18**	**Sum 17**	**Win 18**	**Spr 18**	**Sum 18**	**Sum 18**
High Marsh	**BICE**								6%					
	CALY	56%	49%	52%	32%					31%	14%	11%	1%	
	CAPA			6%	1%									
	JUEF								32%					
	MYLA			17%	2%							1%		
	PHAR				9%	34%	53%	51%	23%	36%	48%	53%	53%	19%
	POAM													46%
	OTHER	44%	51%	23%	11%	1%	8%	1%	2%	24%	34%	29%	11%	
Low Marsh	**SALA**								9%					
	SCTA					5%			18%					
	ELPA				5%				5%				1%	
	EQFL				10%								2%	
	JUEF					28%	20%	10%						
	PHAR				9%	5%	13%	30%			1%	4%	2%	
	POAM										1%			24%
	POHY				12%				1%				2%	
	SALA					1%				5%			9%	6%
	OTHER				1%	21%	6%	5%		4%		2%	2%	2%

Overall proportion of biomass contribution from living, dead, and detritus biomass contribution varied across the seasons, living biomass contributing the most during the summer season, standing dead and detritus contributing the most during the winter, with biomass contributions being more evenly split between living, dead, and detritus in the spring reflecting new plant growth across all sites (except Franz which was not sampled in the summer 2017, winter 2018, or spring 2018) (Table 30, Table 31). This seasonal look at biomass composition shows the largest flux of standing biomass (living + dead) is between the summer and winter time-period, some of this living and dead biomass shifting to detrital material and most being exported from the sampling areas. The largest flux of detritus out of the wetland occurs during the spring-summer time-period, detrital material showing a gradual increase from summer to spring and then a sharp decline between the spring and summer sampling events (Table 30, Figure 51). While the overall amount of biomass contributed is lower coming out of the low marsh compared to the high marsh strata's, they follow similar patterns of living, dead, and detritus biomass contribution over the seasonal shifts. This is with the exception of Steamboat Slough which showed a decline in low marsh and increase in high marsh biomass production between summer 2017 and 2018 (Table 31, Figure 52). Following the shift in species contributions on this site (such as shifting from non-native *J. effusus* and *P.*

arundinacea to native *S. tabernaemontani*, *E. palustris*, and *S. latifolia* in the low marsh zone), this change is likely reflective of plant community shifts stemming from restoration efforts (Table 29).

Table 30. Overall proportion (by dry biomass) of living (live), dead (standing dead), and detritus across high and low marsh stratums sampled between the summer of 2017 and summer 2018. Welch Island low marsh strata and Franz lake (both strata) were not sampled until summer 2018.

		Welch Island (Rkm 53)				**Steamboat Slough (Rkm 57) (AEMR Site, Restored 2016)**				**Whites Island (Rkm 72)**				**Franz Lake (Rkm 221)**
	Species Codes	**Sum 17**	**Win 18**	**Spr 18**	**Sum 18**	**Sum 17**	**Win 18**	**Spr 18**	**Sum 18**	**Sum 17**	**Win 18**	**Spr 18**	**Sum 18**	**Sum 18**
High Marsh	Live	86%	13%	41%	83%	65%	31%	42%	86%	62%	20%	25%	58%	54%
	Standing Dead	7%	16%	14%	5%	8%	18%	22%	0%	3%	33%	21%	13%	17%
	Detritus	7%	72%	45%	6%	27%	51%	36%	7%	34%	47%	54%	15%	15%
Low Marsh	Live				90%	86%	36%	55%	93%	91%	24%	41%	89%	81%
	Standing Dead				1%	5%	12%	19%	3%	6%	11%	23%	2%	14%
	Detritus				9%	9%	52%	27%	3%	4%	65%	36%	9%	6%
Overall	Live	86%	13%	41%	89%	76%	33%	47%	93%	64%	21%	26%	73%	69%
	Standing Dead	7%	16%	14%	4%	6%	16%	21%	1%	3%	32%	21%	12%	18%
	Detritus	7%	72%	45%	8%	18%	51%	32%	7%	33%	48%	53%	15%	14%

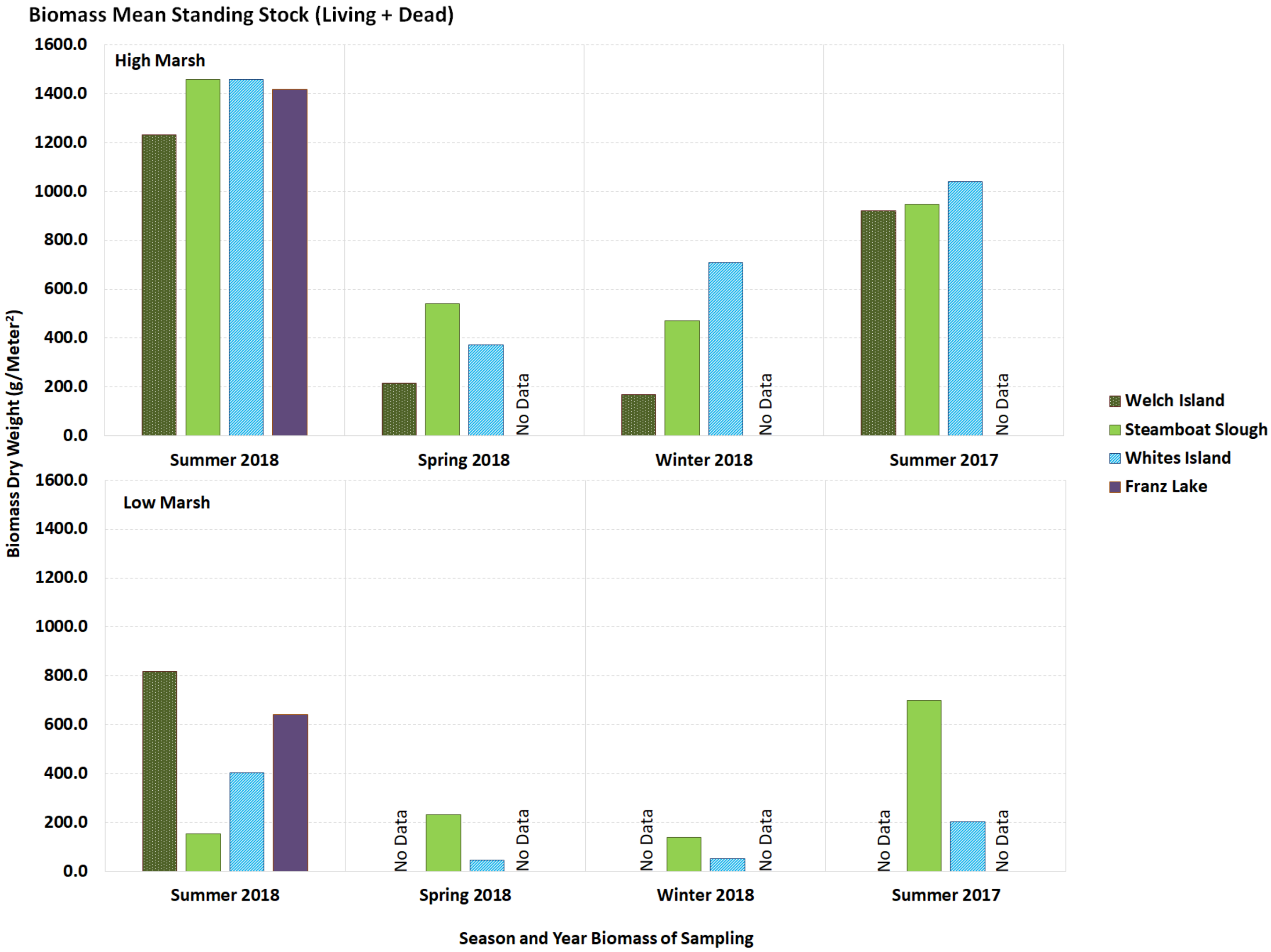


Figure 51: Average aboveground standing stock biomass (living + dead, dry weight g/m^2) for summer 2017, winter 2018, spring 2018, and summer 2018 for both the high and low marsh strata across sites sampled. Sites shown in order of rkm from mouth of the Columbia River to the Bonneville dam. *Welch Island low marsh strata and Franz lake (both strata) were not sampled until summer 2018. See Table 31 for all values and standard deviations.

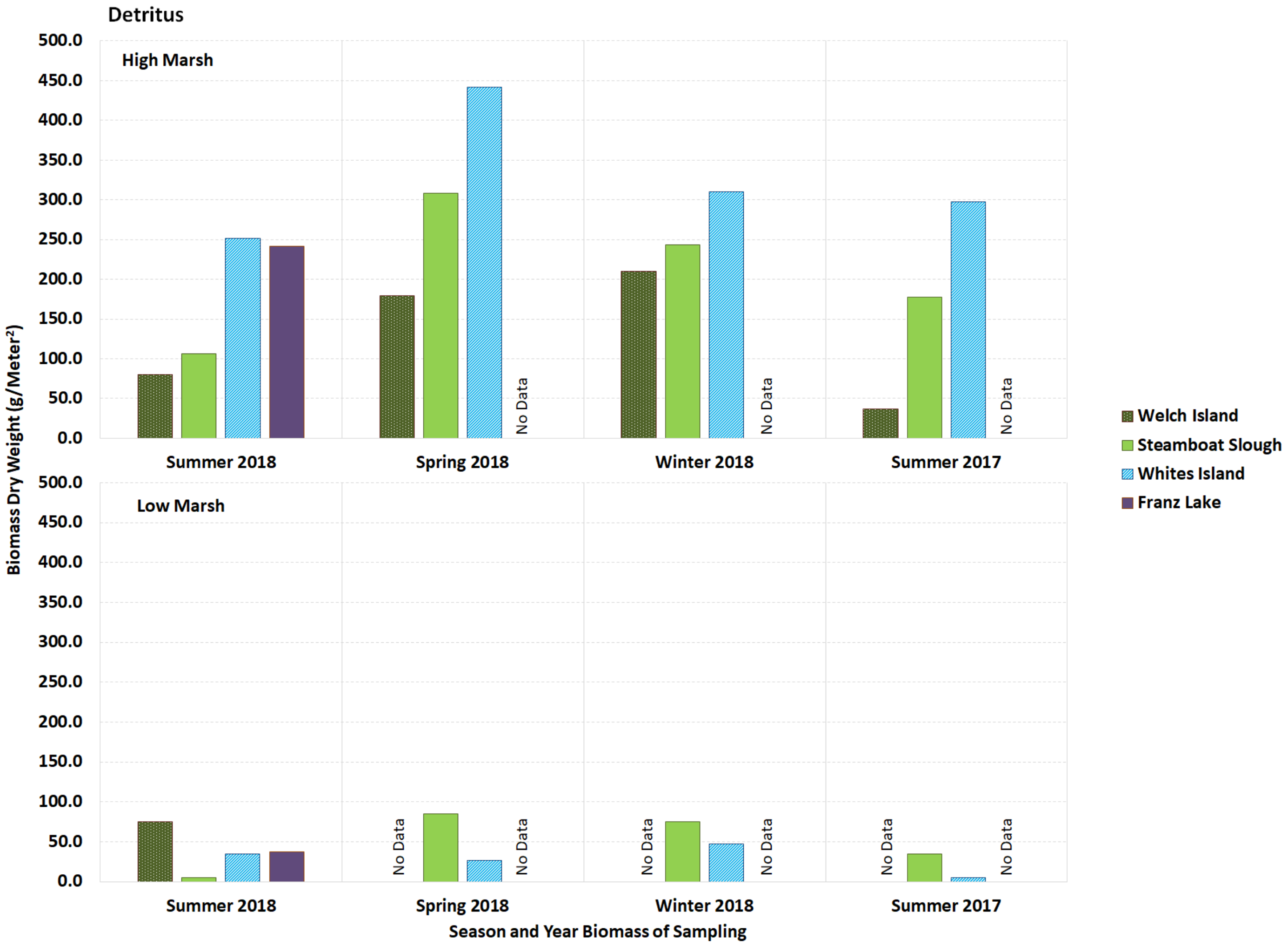


Figure 52: Average detrital biomass (dry weight g/m^2) for summer 2017, winter 2018, spring 2018, and summer 2018 for both the high and low marsh strata across sites sampled. Sites shown in order of rkm from mouth of the Columbia River to the Bonneville dam. Welch Island low marsh strata and Franz lake (both strata) were not sampled until summer 2018. See Table 31 for all values and standard deviations.

Table 31: Average aboveground standing stock biomass (living + dead, dry weight g/m^2) and average detrital biomass (dry weight g/m^2) for summer 2017, winter 2018, spring 2018, and summer 2018 for both the high and low marsh strata across sites sampled. Sites shown in order of rkm from mouth of the Columbia River to the Bonneville dam. n= number of samples, SD = Standard Deviation.

Total Standing Stock g/m^2			Summer 2017			Winter 2018			Spring 2018			Summer 2018		
Site	Strata	Type	n	Mean	SD	n	Mean	SD	n	Mean	SD	n	Mean	SD
Welch Island	High Marsh	Living + Dead	14	922	456	12	167	79	13	215	127	9	1233	374
		Detritus	26	37	28	24	210	60	13	179	125	9	80	65
	Low Marsh	Living + Dead										9	820	638
		Detritus										9	75	103
Steamboat Slough	High Marsh	Living + Dead	7	948	659	6	470	383	6	539	447	9	1459	1099
		Detritus	7	177	225	12	243	148	6	308	141	9	106	225
	Low Marsh	Living + Dead	14	700	1231	13	140	225	13	233	445	9	153	160
		Detritus	14	35	84	26	75	85	13	85	158	9	5	16
Whites Island	High Marsh	Living + Dead	13	1042	527	12	710	586	12	372	199	9	1458	823
		Detritus	24	297	395	24	310	197	12	442	315	9	252	216
	Low Marsh	Living + Dead	6	204	68	6	51	36	6	47	102	10	402	226
		Detritus	9	5	4	12	47	46	6	26	24	10	35	58
Franz Lake	High Marsh	Living + Dead										9	1419	862
		Detritus										9	242	155
	Low Marsh	Living + Dead										9	640	485
		Detritus										9	37	53

Contribution of Biomass

The difference between the plant standing stock in the summer and that remaining in the winter can be considered the amount of organic matter contributed by the plants during that year. Presumably, some material continues to breakdown during the next growing season, but for the purposes of this analysis, we consider the summer-winter difference to represent the annual organic matter contribution. Specifically, the contribution of organic matter is typically calculated as the summer standing stock (live and dead) minus the following winter's standing stock (live and dead). With the addition of detritus data, it can also be calculated as standing stock plus detritus. Including detritus into the equation allows for understanding how much biomass materials leave the sampling area altogether (not just shift from living to the dead to detritus). This calculation shows that not all of the standing stock is exported from the site, some staying onsite as detritus, however the amount of detritus retained on the site varies across the sites, generally being less than 100 g/m^2/year with the exception of Welch Island which retained approximately 173 g/m^2/year between 2017 and 2018 (Table 32). Comparing overall organic matter contributions (g/m^2/year) across sites and years shows that the 2017-2018 biomass production was high for Welch Island at 755 compared to 577 in 2017 and 521 in 2014, but low for Whites Island with overall standing stock 287 g/m^2/year compared to 844 in 2017, 630 in 2016, and similar to 2014 at 226 g/m^2/yr (Table 32). The shift in biomass production on Whites Island appears to be coming from the high marsh strata, as the low marsh strata contributions are similar across all years sampled (Table 32).

Table 32: Mean predicted contribution of organic matter (dry weight g/m^2/year) both including and excluding detrital material. Historic contributions are shown for all sites surveyed between 2013-2018 for reference. Contribution of organic matter is predicted using previous years' average contribution by stratum.

Total Standing Stock g/m^2			Mean Predicted Contribution Dry Wt (g/m^2/year) (= Summer-Winter)			
Site	Strata	Type	2017-2018	2016-2017	2015-2016	2013-2014
Ilwaco Slough	Overall	Living + Dead		618		492
Welch Island	High Marsh	Living + Dead	755	577		521
		Including Detritus	582			
Steamboat Slough	High Marsh	Living + Dead	478			
		Including Detritus	412			
	Low Marsh	Living + Dead	560			
		Including Detritus	520			
	Overall	Living + Dead	503			
		Including Detritus	454			
Whites Island	High Marsh	Living + Dead	332	1184	831	689
		Including Detritus	319			
	Low Marsh	Living + Dead	152	164	228	153
		Including Detritus	110			
	Overall	Living + Dead	287	844	630	226
		Including Detritus	253			
Campbell Slough	Overall	Living + Dead		341	487	142
Franz Lake	Overall	Living + Dead			365	26

Multi-Year Analysis

Above ground biomass data from 2011 – 2018 were analyzed to determine if differences exist in summer biomass (production) and annual organic matter contribution (hereafter termed contribution) between 1) high marsh vs. low marsh, and 2) across survey years. Clear trends in biomass production between high and low marsh strata are evident, low marsh plant communities producing lower weight dry biomass compared to high marsh plant communities across all sites (EMP 2018, Table 32, Figure 53). In 2018, Welch Island low marsh composed approx. 40% of the overall site summer standing stock, Steamboat Slough the low marsh composed approx. 10%, Whites Island 22%, and Franz Lake 31% (Table 32).

Relative to other years 2018 was an average biomass production year for Welch Island and Whites Island, however the low marsh strata on Whites Island exhibited greater levels of biomass production than previous years, but this may be due to slightly higher elevations being included in the "low marsh strata" compared to previous years (Table 33). Franz lake mean biomass production in 2018 was slightly higher than in previous years, but this could also be due to the new method of sampling both the high and low marsh strata on the site, the low marsh strata not historically separated out of samples from Franz Lake when averaging site biomass levels (Table 32, Table 33, Figure 53).

At Whites Island, which has the longest consistent annual record of high and low marsh biomass sampling, the contribution from the low marsh (Summer-Winter) standing stock is relatively consistent from year to year (~150-200 g/m^2/year) compared to the high marsh (~1184-332 g/m^2/year), which is much more variable (Table 32). Across time, differences in summer biomass production between high and low marsh strata follow a consistent pattern with the low marsh producing less biomass than the high marsh (Figure 53 and Figure 54). The high marsh contribution is, however, much more variable from year to year (Figure 54).

The annual freshet magnitude appears to influence biomass production across all sites, greater magnitudes such as in 2011, 2012, and 2017 generally producing less summer biomass (in August, when it is sampled) compared to lower magnitude years (2013, 2015, 2016) (Figure 54, Figure 55). This is likely directly related to the amount of time these marsh areas are exposed during the growing season; a lower magnitude freshet would result in a longer growing season (marsh exposed) resulting in greater plant biomass accumulation. The mean annual summer total standing stock for all sites (Whites, Welch, Franz) shows a strong correlation (R^2 =0.63, p-value <0.001) with the cumulative Columbia River discharge for the month of August, higher discharge correlated with lower biomass production, further supporting this conclusion (Figure 55).

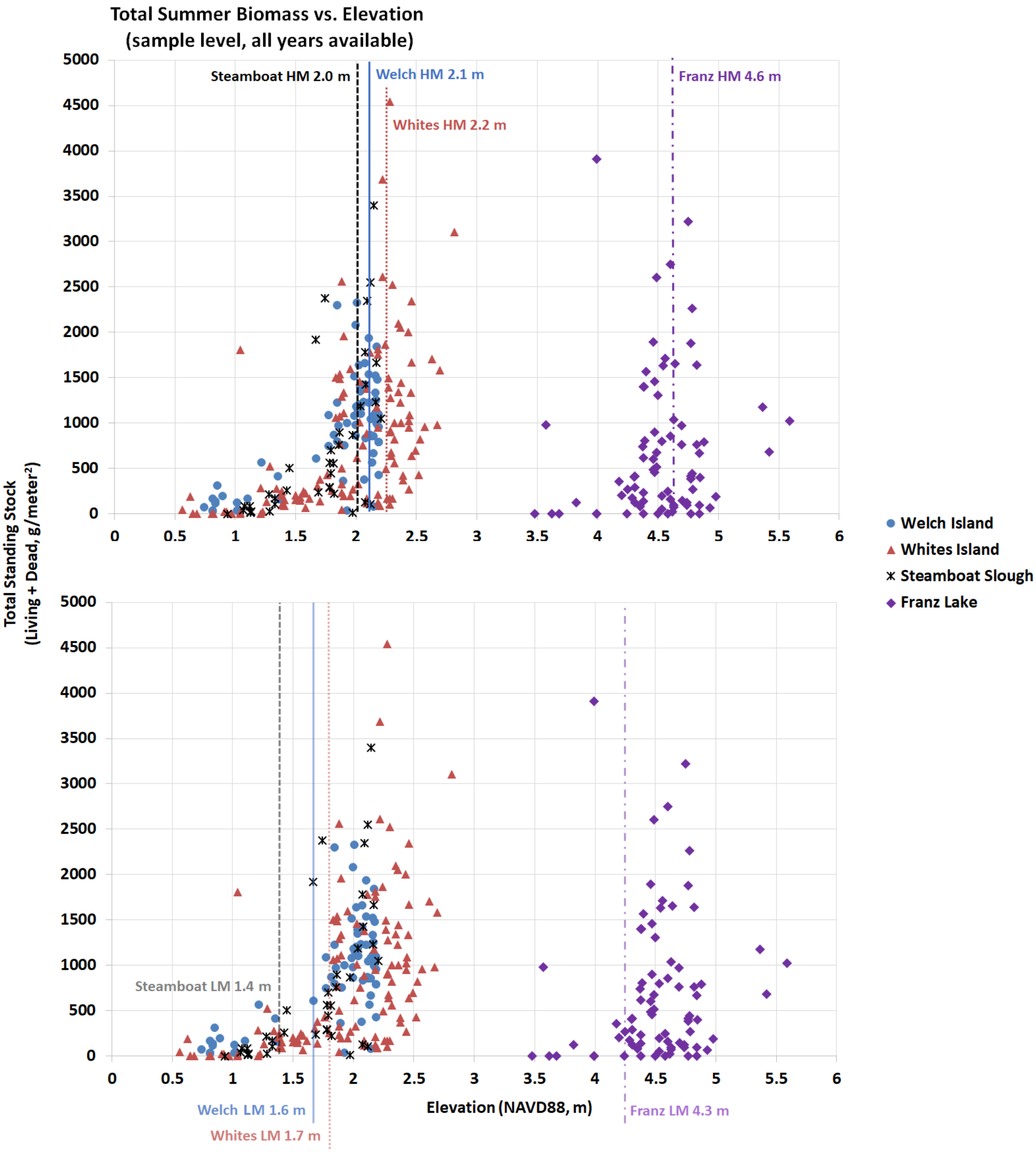


Figure 53: Total summer standing stock biomass (dry weight, g/m^2) data by elevation for all years data was collected at each site. Mean high and mean low marsh sample elevations (averaged across all years) highlighted as vertical lines on graph. A shows HM elevations, B shows LM elevations. HM = High Marsh, LM = Low Marsh.

Table 33: Total mean summer standing stock biomass (dry weight, g/m^2) data split by high and low marsh plant community strata by survey year. Mean elevation of strata, and standard deviation (SD) of standing stock biomass also presented. Data also depicted in Figure 54.

Site	Strata	Biomass (g/m^2)	2018	2017	2016	2015	2013	2012	2011
Welch Island	High Marsh	Elevation	2.09	2.07	2.12		2.13	1.99	
		n	9	14	12		9	5	
		Mean	1233	922	1095		1361	1142	
		SD	374	456	320		647	322	
	Low Marsh	Elevation	1.59					1.64	
		n	9					4	
		Mean	820					401	
		SD	638					362	
Steamboat Slough	High Marsh	Elevation	2.03	2.05					
		n	9	7					
		Mean	1459	948					
		SD	1099	659					
	Low Marsh	Elevation	1.27	1.57					
		n	9	14					
		Mean	153	700					
		SD	160	1231					
Whites Island	High Marsh	Elevation	2.54	2.18	2.19	2.10	2.14	2.22	2.27
		n	9	13	12	12	9	8	6
		Mean	1458	1042	1802	1372	1359	739	1152
		SD	823	527	1161	462	834	623	844
	Low Marsh	Elevation	1.97	1.51	1.53	1.73	1.78	1.78	1.16
		n	10	6	6	6	6	3	2
		Mean	402	204	198	261	163	114	88
		SD	226	68	32	152	126	102	89
Franz Lake	High Marsh	Elevation	4.99		4.41	4.51	4.60	4.63	4.61
		n	9		12	12	9	7	8
		Mean	1419		1287	893	434	672	203
		SD	862		1205	719	317	557	152
	Low Marsh	Elevation	4.32						
		n	9						
		Mean	640						
		SD	485						

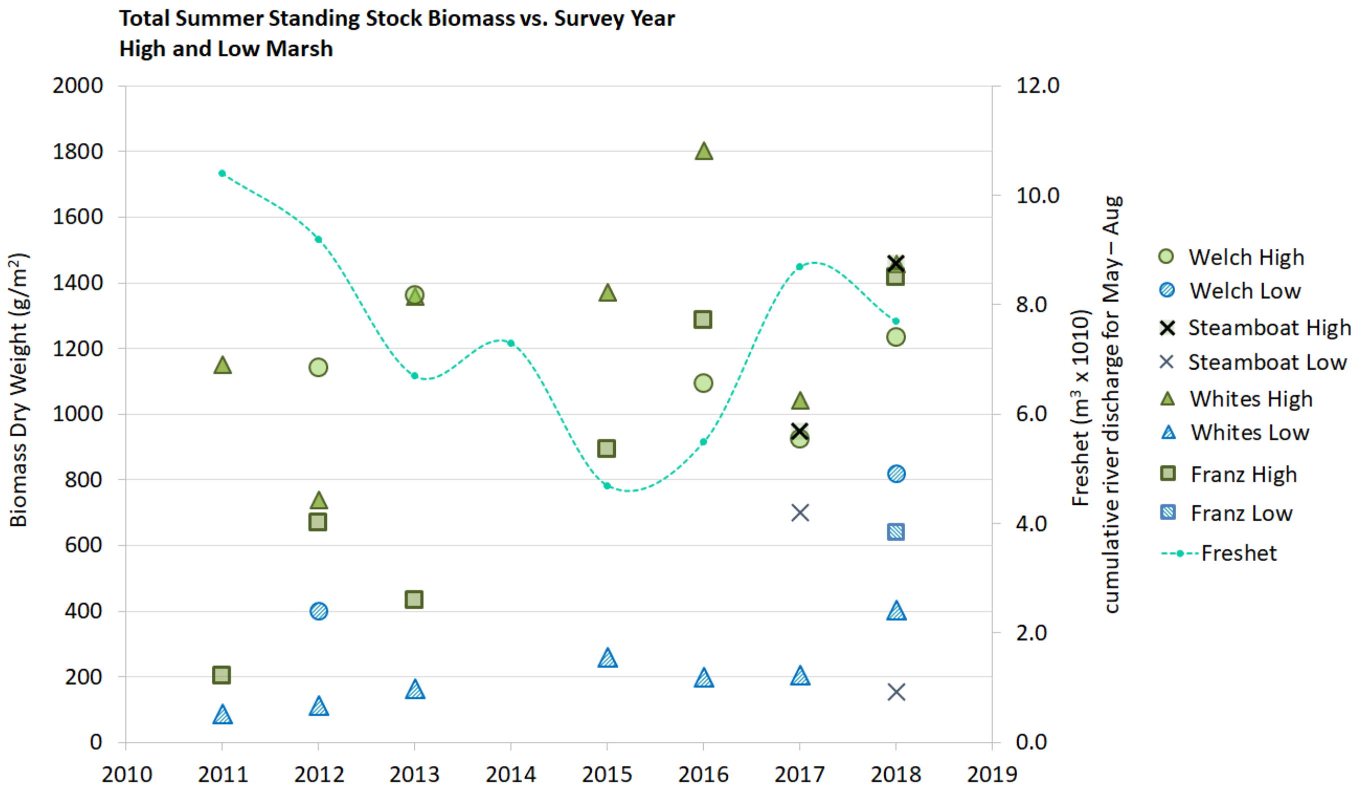


Figure 54: Total summer standing stock biomass (dry weight, g/m^2) data split by high and low marsh plant community strata by survey year. Freshet magnitude is shown as the right axis for reference (see section 3.3.1. for details). Biomass data and standard deviation is shown in Table 33.

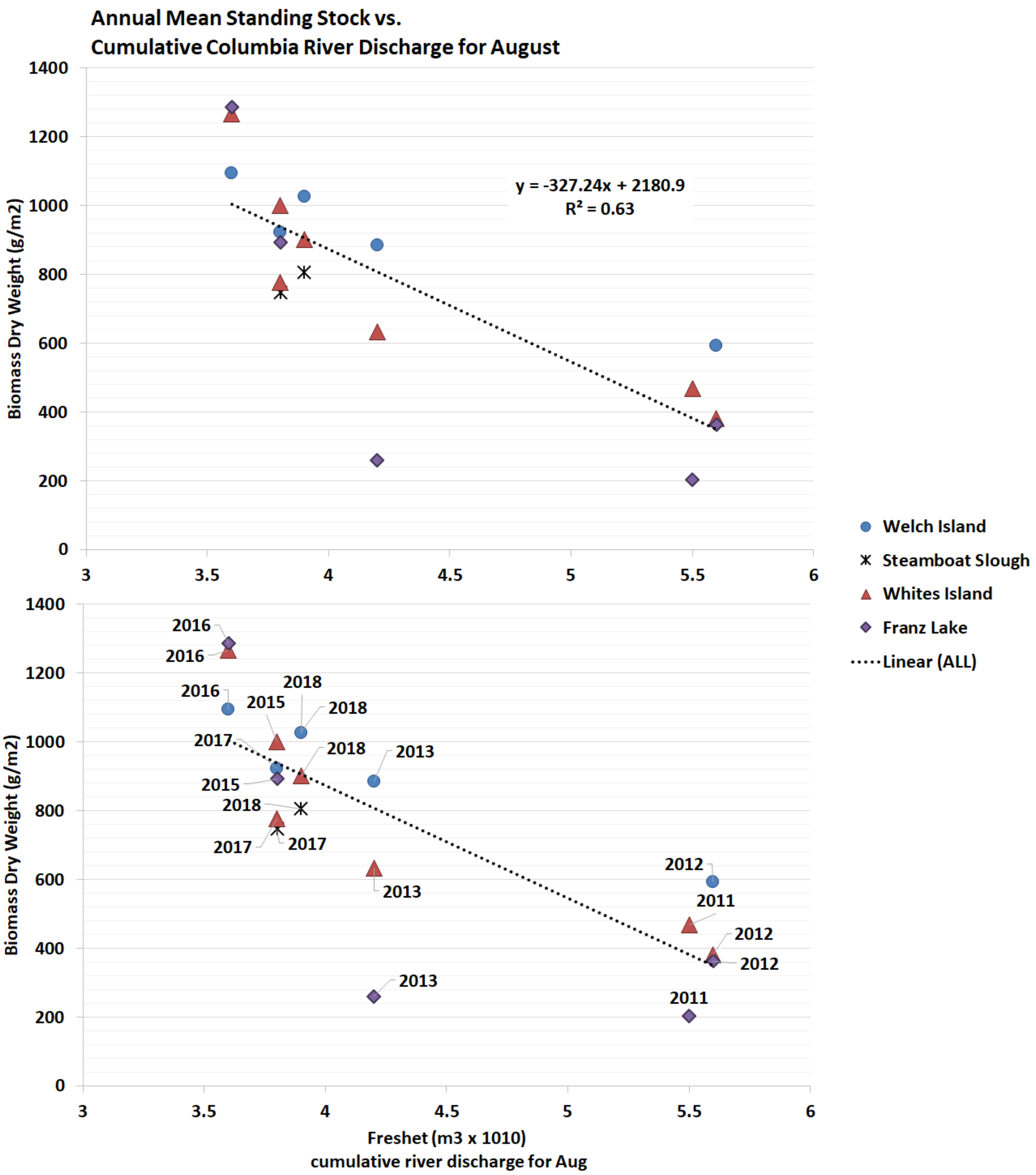
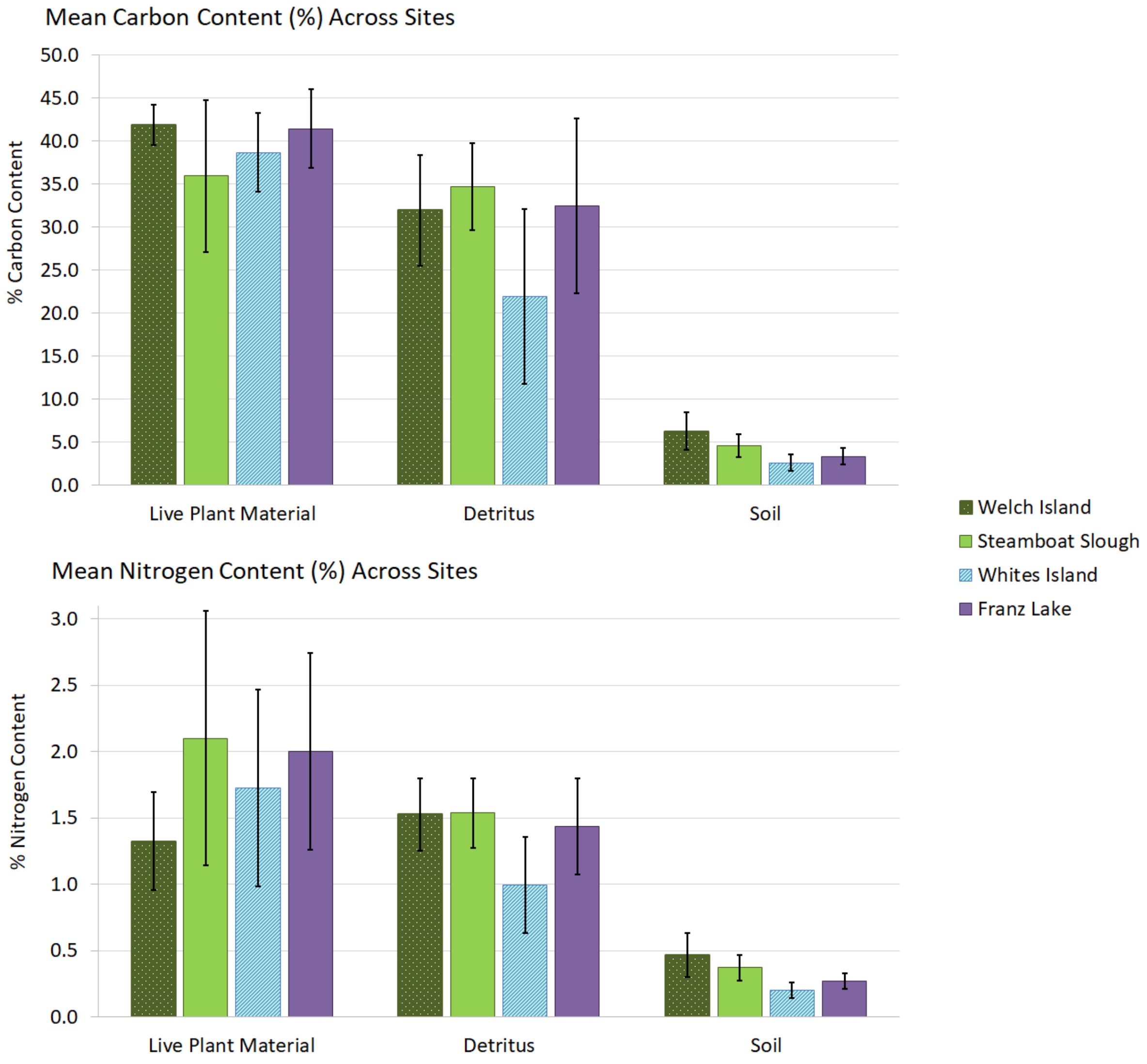


Figure 55: Overall annual mean total standing stock biomass (dry weight, g/m^2/year) vs. the Freshet cumulative river discharge for the month of August. Second graph has annual labels for emphasis.

Biomass and Soil Nutrient Composition

In 2018, a sub-sample of live biomass and detritus samples were tested for nitrogen, carbon, and ADF lignin content. These data provide information on the nutrient content of the living biomass and decomposing detritus. Additionally, the soil nutrient content and texture were analyzed at the location of the biomass samples. Overall, Carbon (C) content was greatest in the living above ground biomass across all sites, followed by detritus, and soil (Figure 56, Table 34). In the living aboveground biomass mean C content ranging from 41.9 ± 2.3% at Welch Island to 36.0 ± 8.8% at Steamboat Slough (restoration site). Generally, the mean above ground living biomass C content was similar across sites, with Steamboat Slough showing the overall lowest C content (Figure 56, Table 34). Mean detritus C content was more variable than living biomass C content and ranged from 34.7 ± 5.1% at Steamboat Slough to 21.9 ± 10.2% at Whites Island. Mean soil C content ranged from 6.2 ± 2.2% at Welch Island to 2.6 ± 0.9% at Whites Island.

Nitrogen (N) content was greatest in the living above ground biomass across all sites, followed by detritus, and soil (Figure 56, Table 34). In the living aboveground biomass mean N content ranging from 2.1 ± 1.0% at Whites Island to 1.3 ± 0.4% at Welch Island. Generally, the mean above ground living biomass N content was similar across sites, with Welch Island showing the overall lowest N content (Figure 56, Table 34). Mean detritus N content was more variable than living biomass N content and ranged from 1.5 % at Steamboat Slough and Welch Island to 1.0 ± 0.4% at Whites Island. Mean soil N content ranged from 0.5 ± 0.2% at Welch Island to 0.2 ± 0.1% at Whites Island.


Mean Carbon Content (%) Across Sites

Mean Nitrogen Content (%) Across Sites

Figure 56: Mean % Carbon and Nitrogen content of above ground living biomass, detritus, and soil across sites, samples collected in the Summer of 2018. Data provided in Table 34.

The C:N ratio was greatest in the living above ground biomass across all sites, followed by detritus, and soil (Figure 57, Table 34). In living aboveground biomass mean C:N content ranging from 34.7 ± 11.1% at Welch Island to 23.1 ± 7.9% at Franz Lake. Mean detritus C:N content was less variable than living biomass C:N content and ranged from 21.3 ± 4.7% at Welch Island to 23.6 ± 10.2% at Franz Lake. Mean soil C:N content ranged from 13.9 ± 2.0 % at Welch Island to 12.3 ± 1.7% at Steamboat Slough.

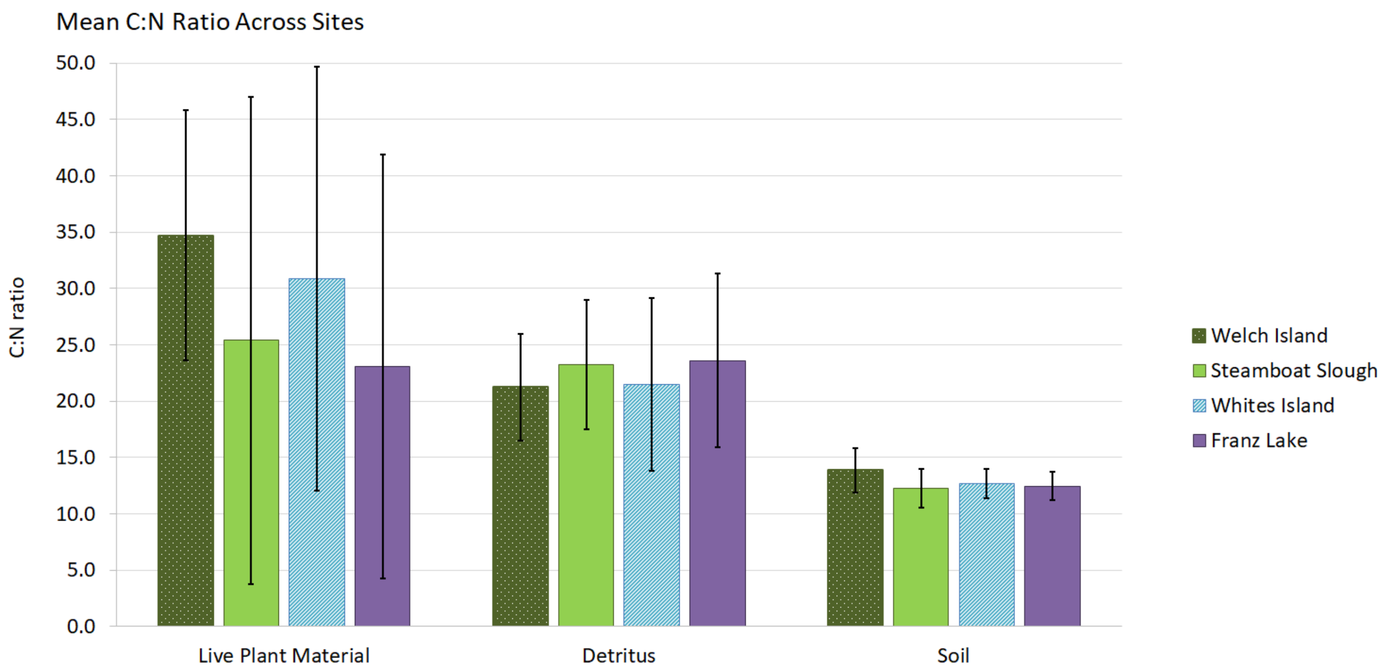


Figure 57: Mean Carbon and Nitrogen ratio of above ground living biomass, detritus, and soil across sites, samples collected in the Summer of 2018. Data provided in Table 34.

Table 34: Mean % Carbon and Nitrogen content of above ground living biomass, detritus, and soil across sites, samples collected in the Summer of 2018. For comparative graphs see Figure 56 and Figure 57.

Site		Carbon (%)			Nitrogen (%)			C:N Ratio		
		Live Plant Material	Detritus	Soil	Live Plant Material	Detritus	Soil	Live Plant Material	Detritus	Soil
Welch Island	n	12	16	10	12	16	10	12	16	10
	Mean	41.9	32.0	6.2	1.3	1.5	0.5	34.7	21.3	13.9
	SD	2.3	6.5	2.2	0.4	0.3	0.2	11.1	4.7	2.0
Steamboat Slough	n	17	7	10	17	7	10	17	7	10
	Mean	36.0	34.7	4.6	2.1	1.5	0.4	25.4	23.3	12.3
	SD	8.8	5.1	1.4	1.0	0.3	0.1	21.6	5.7	1.7
Whites Island	n	19	16	11	19	16	11	19	16	11
	Mean	38.7	21.9	2.6	1.7	1.0	0.2	30.9	21.5	12.7
	SD	4.5	10.2	0.9	0.7	0.4	0.1	18.8	7.7	1.3
Franz Lake	n	18	16	10	18	16	10	18	16	10
	Mean	41.5	32.4	3.4	2.0	1.4	0.3	23.1	23.6	12.5
	SD	2.7	8.1	0.9	0.6	0.5	0.1	7.9	6.2	0.9

At Steamboat Slough, a strong positive correlation was identified between C content in living above ground biomass and sample elevation within the wetland, higher elevations having greater C content (Figure 58). This relationship was not identified across the other EMP sites and may be a factor of the Steamboat's recent re-flooding through restoration and this impact on wetland plant development along the elevation gradient within this site (Figure 58). No relationship in detrital carbon content and elevation was identified across the research sites, while strong positive correlation relationships in soil C content and wetland elevation were found across all the EMP sites (Figure 58). Steamboat Slough did not show this trend in soil content, again, likely due to the recent soil biochemical changes occurring on that site from the restoration.

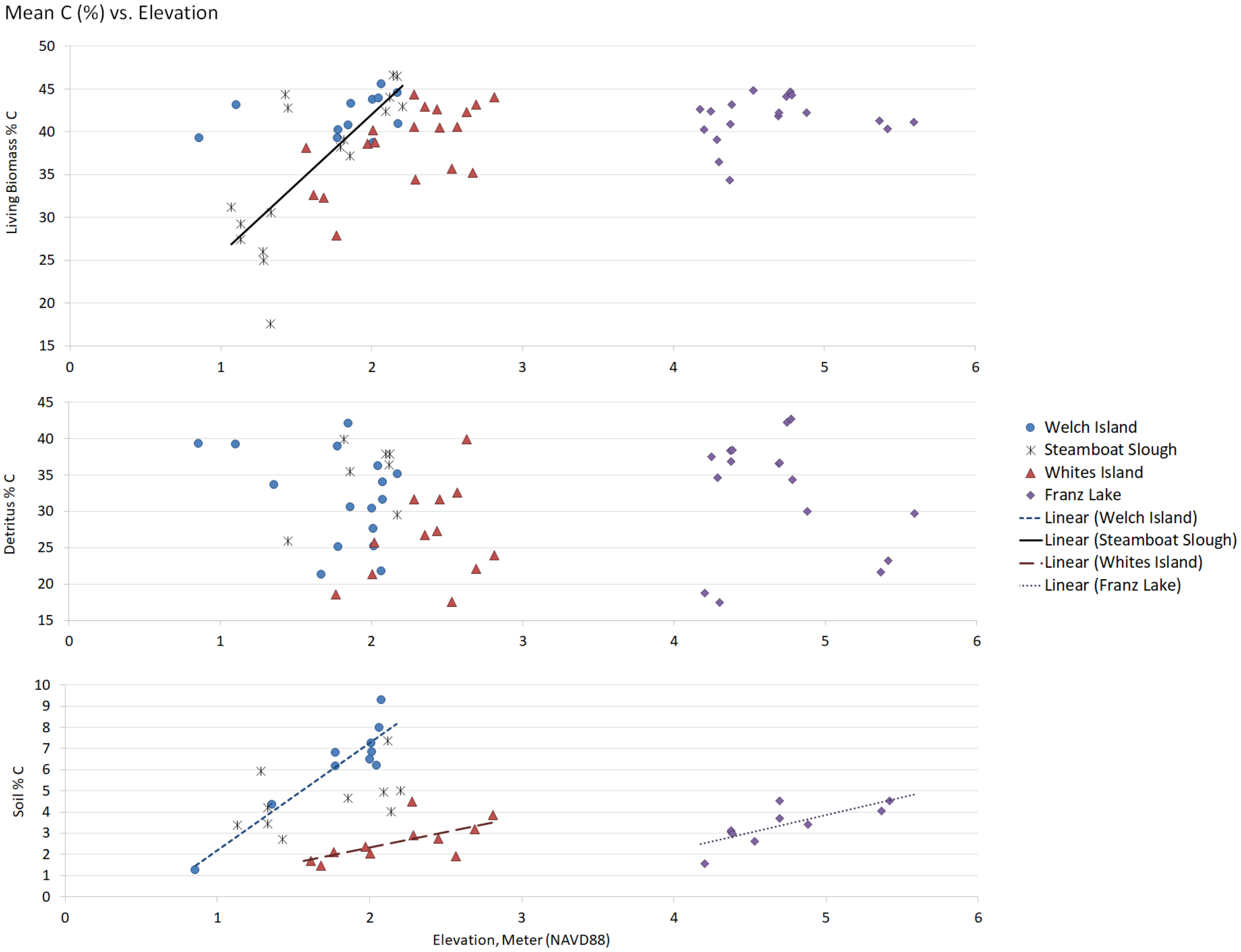


Figure 58: Carbon content (%) of above ground living biomass, detritus, and soil (graphs from top to bottom) vs. sample elevations in meters NAVD88. Significant within site correlations shown in each graph, site summary data provided in Table 34.

All sites including Steamboat Slough showed a strong negative correlation between N content in living above ground biomass and sample elevation within the wetland, lower elevations having greater N content (Figure 59). No relationships in detrital N content and elevation were identified across the research sites, while strong positive correlations in soil N content and wetland elevation were found across all the sites (Figure 59).

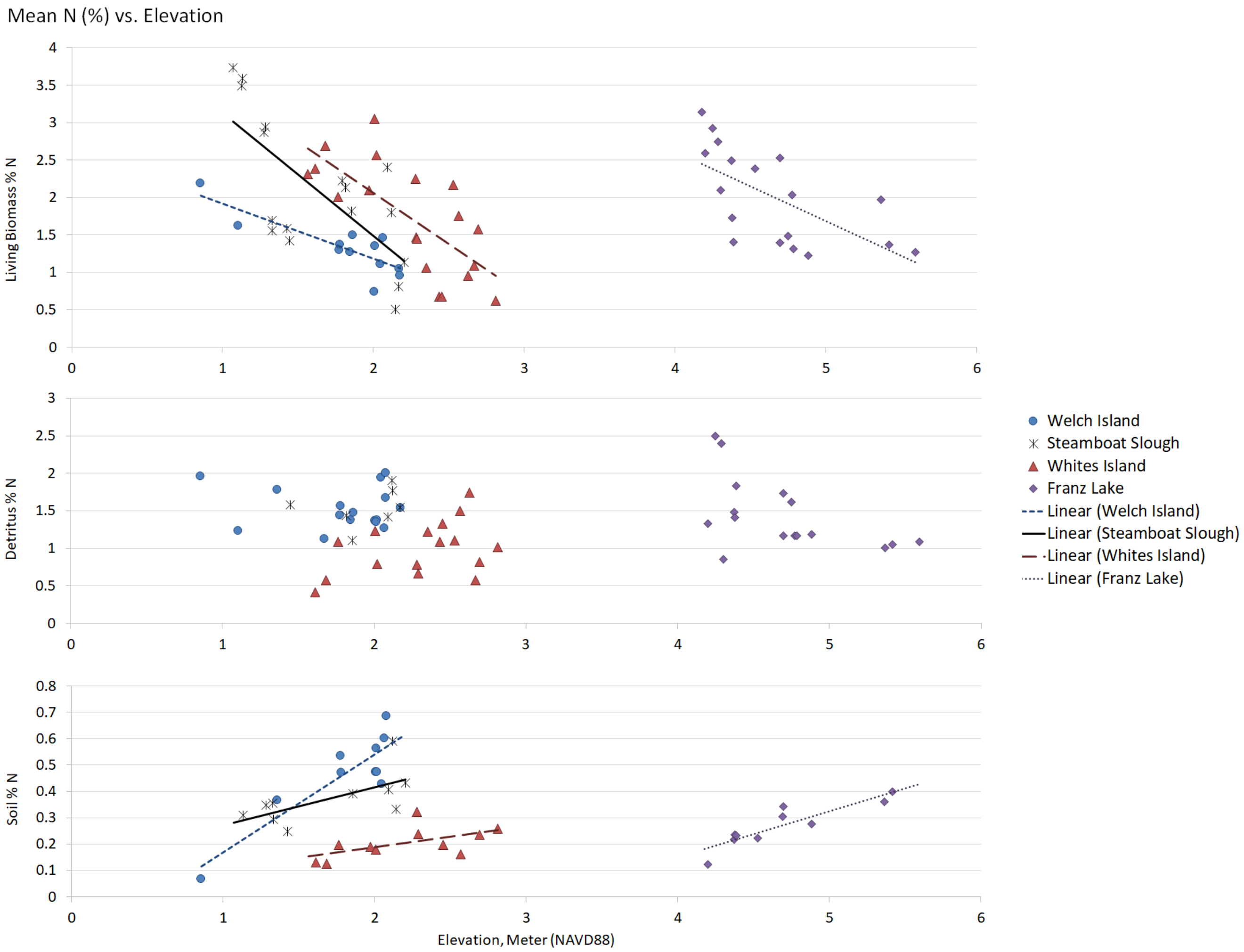


Figure 59: Nitrogen content (%) of above ground living biomass, detritus, and soil (graphs from top to bottom) vs. sample elevations in meters NAVD88. Significant within site correlations shown in each graph, site summary data provided in Table 34.

All sites including Steamboat Slough showed a strong positive correlation between C:N ratio in living above ground biomass and sample elevation within the wetland, lower elevations having lower C:N ratios (Figure 60). No relationships in detrital C:N ratio and elevation were identified across the research sites (Figure 60). Strong positive correlations in soil C:N ratios and wetland elevation were found across all the sites, except Franz Lake where a negative correlation in soil C:N ratio and wetland elevation was identified (Figure 60). This is particularly interesting, as the soil N content was lowest in the lower wetland elevation and living plant biomass in these elevations had the highest N content. This potentially reflects the shift in plant species and plant species nutrient use along the high to low marsh gradient. The low marsh species having lower carbon content, and lower C:N ratios overall, indicating less decomposition time required for the plant species found in the low marsh zone; C:N Ratio under 25 indicating no N limitation to decomposition (Wang et al. 2016). These results have potential implication for decomposition differences in the high and low marsh plant biomass corresponds to the overall differences found in detritus accumulation between the high and low marsh zone across sites, less detritus accumulation occurring in the low marsh zone (Figure 52). The above ground living biomass Lignin:

Nitrogen (L:N) ratio is also known as a good predictor of plant biomass decomposition rates, smaller ratios indicate more N and less Lignin, and quicker decomposition (Taylor et al. 1989, Talbot et al. 2011). L:N ratios across the wetlands were found to also correlate with elevation, following the N content trend, with smaller ratios in the lower marsh zones across sites (Figure 65).

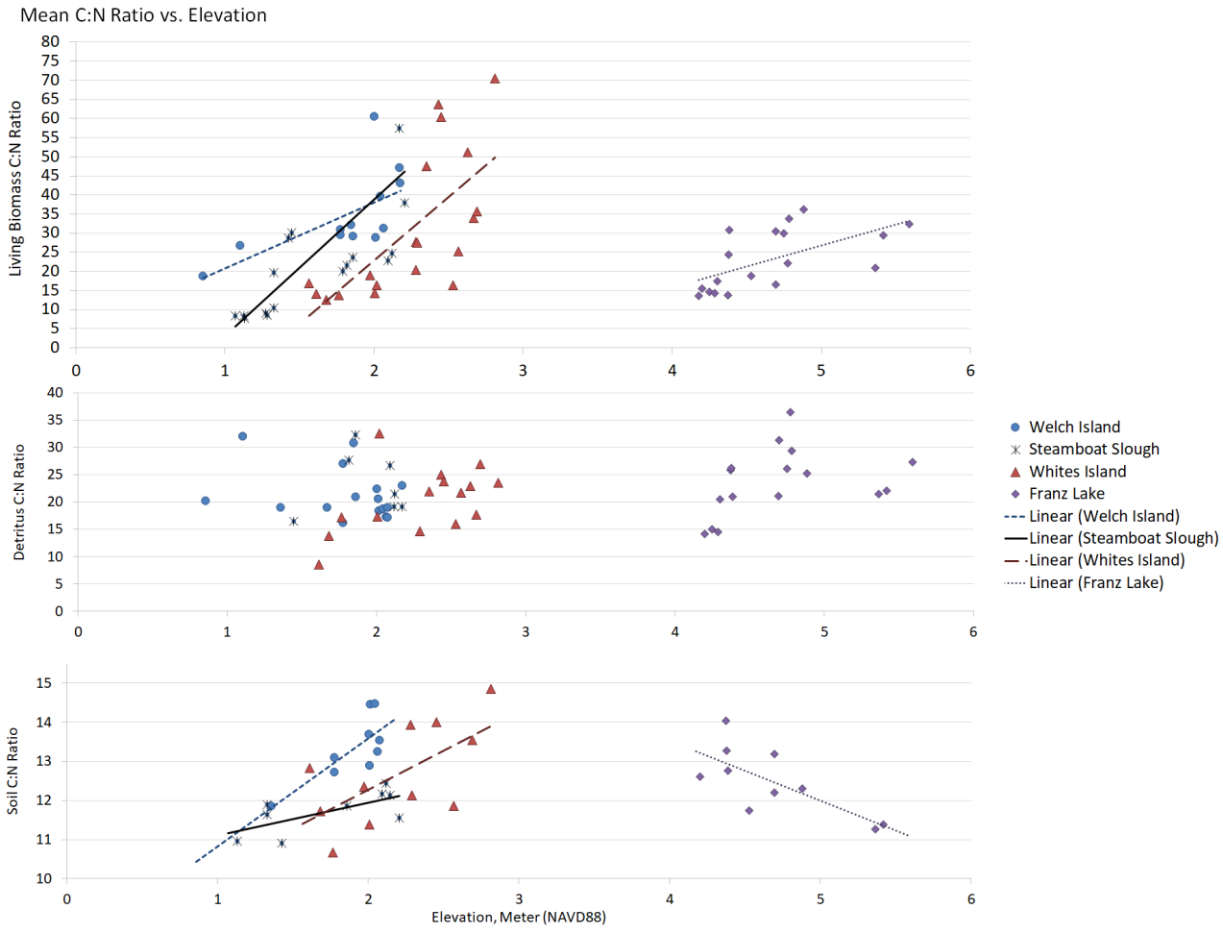


Figure 60: Carbon and Nitrogen ratio of above ground living biomass, detritus, and soil (graphs from top to bottom) vs. sample elevations in meters NAVD88. Significant within site correlations shown in each graph, site summary data provided in Table 34.

The mean soil N and C content showed a strong positive correlation, increases in soil C content corresponding to higher levels of N content (Figure 61). This relationship was also found in the detritus, with detrital C and N having a positive correlation across all sites (Figure 64). No relationship was found between mean living above ground biomass C and N content, indicating that this relationship becomes clearer once decomposition begins (detritus) and the decaying plant matter and associated microbial communities are incorporated into the soil within these sites.

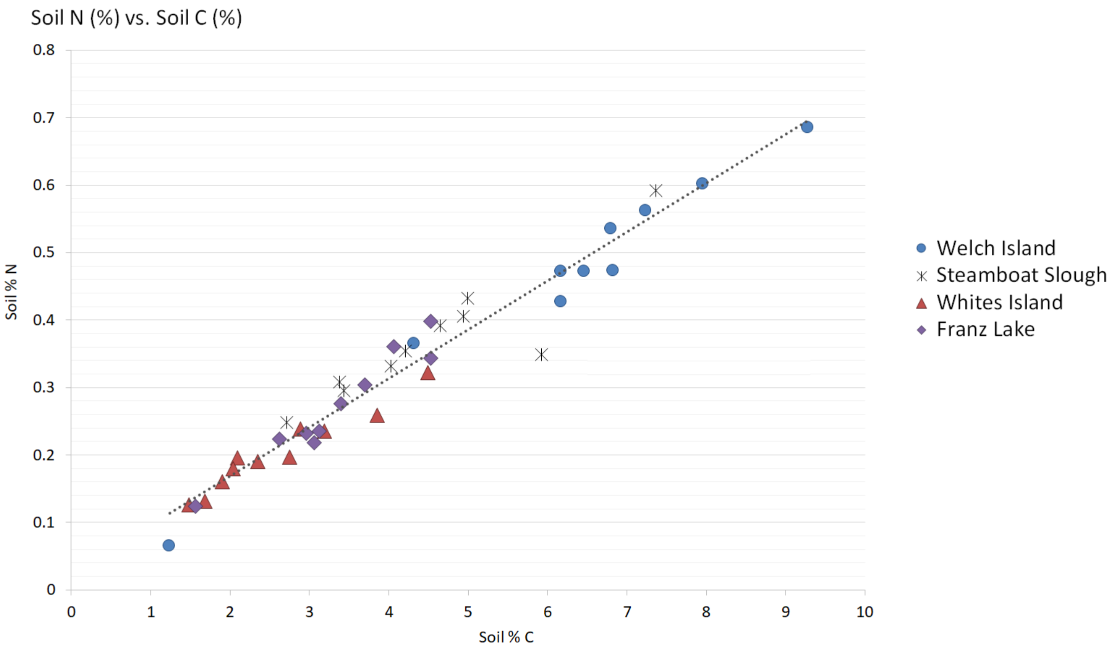


Figure 61. Soil N content (%) vs. Soil C content (%), a strong correlation was shown across all sites (r^2 = 0.95, p<0.001). Summary data provided in Table 34.

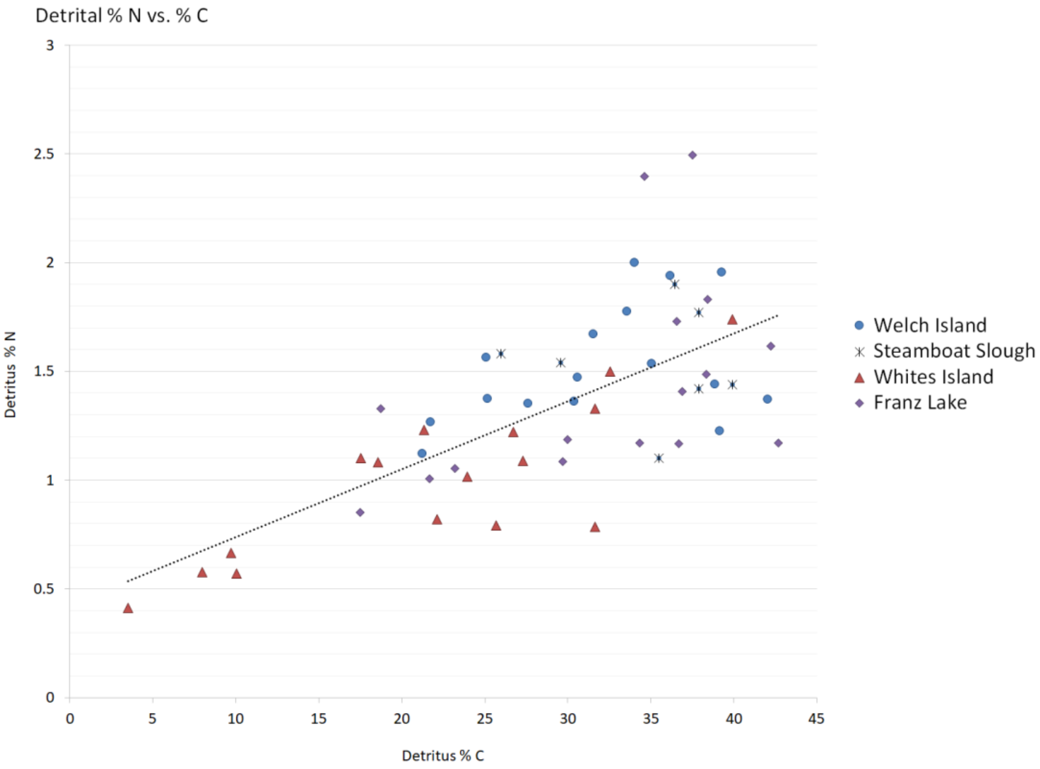


Figure 62: Detritus N content (%) vs. Detritus C content (%), a strong correlation was shown across all sites (r^2 = 0.47, p<0.001). Summary data provided in Table 34.

Overall, mean summer ADF Lignin content was greatest in the detritus samples compared to the living plant biomass (Figure 63, Table 35), this follows the expected trend of ADF Lignin concentrations increasing in the detritus as decomposition occurs (lignin and associated compounds resisting decomposition (Taylor et al. 1989, Talbot et al. 2011). The greatest ADF Lignin content was found in the detritus from Franz Lake, 19.8 ± 7.0%, and the living above ground biomass from Franz Lake, 12.9 ± 6.6%, followed by the detritus, 15.4 ± 9.0%, and biomass at Welch Island, to 9.0 ± 15.4%. The lowest ADF Lignin levels were identified within the living above ground biomass at Steamboat Slough, 5.8 ± 1.7%, and the lowest detritus ADF Lignin levels were identified at Whites Island, 6.6 ± 8.8%. Detrital ADF Lignin content was found to be positively correlated with detrital carbon content, greater carbon levels within the detritus corresponding with greater levels of lignin. Similarly, detritus L:N ratio was also positively correlated with detritus carbon content, higher levels of Lignin and lower levels of N corresponding with greater levels of carbon (Figure 64, Figure 65). This result is expected, as others have found that as the biomass breaks down, the ratio of Lignin and C will increase compared to N (Taylor et al. 1989, Talbot et al. 2011). This relationship is essentially showing N limitation in the long-term breakdown of organic matter with high C and Lignin content (Taylor et al. 1989, Talbot et al. 2011).

Table 35: Mean % ADF Lignin content of above ground living biomass, and detritus across sites, samples collected in the Summer of 2018.

Site		ADF Lignin (%)	
		Live Plant Material	Detritus
Welch Island	n	12	17
	Mean	9.0	15.4
	SD	5.1	6.0
Steamboat Slough	n	17	7
	Mean	5.8	13.0
	SD	1.7	2.2
Whites Island	n	19	16
	Mean	6.6	8.8
	SD	2.2	7.4
Franz Lake	n	18	17
	Mean	12.9	19.8
	SD	6.6	7.0

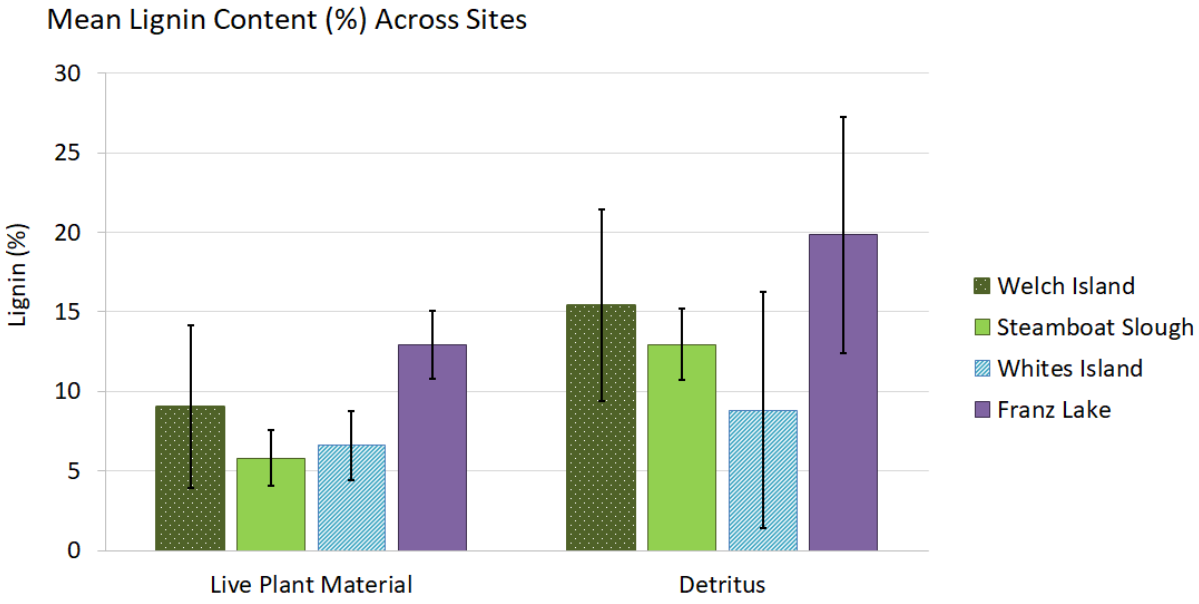


Figure 63: Mean % ADF Lignin content of above ground living biomass, and detritus across sites, samples collected in the Summer of 2018. For summary data see Table 35.

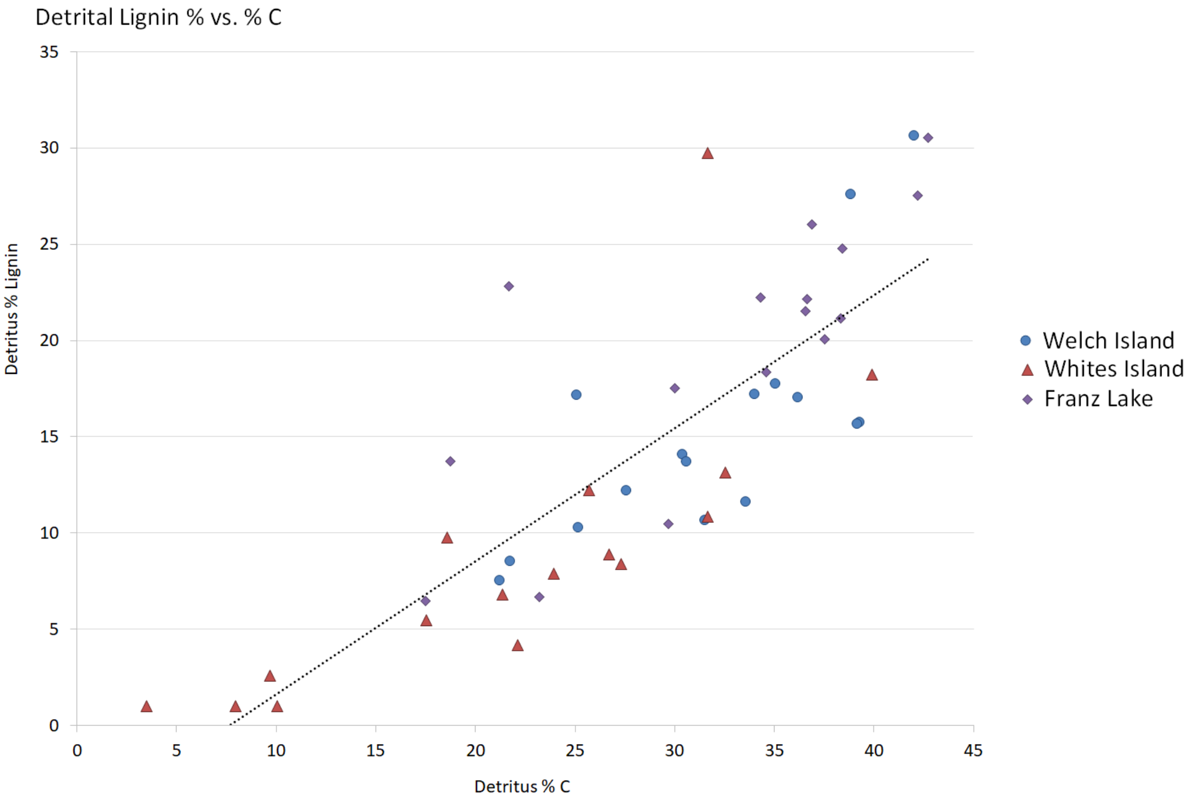


Figure 64: Detritus ADF Lignin content (%) vs. Detritus C content (%), a strong correlation was shown across all sites (r^2 = 0.67, p<0.001). Summary data provided in Table 35.

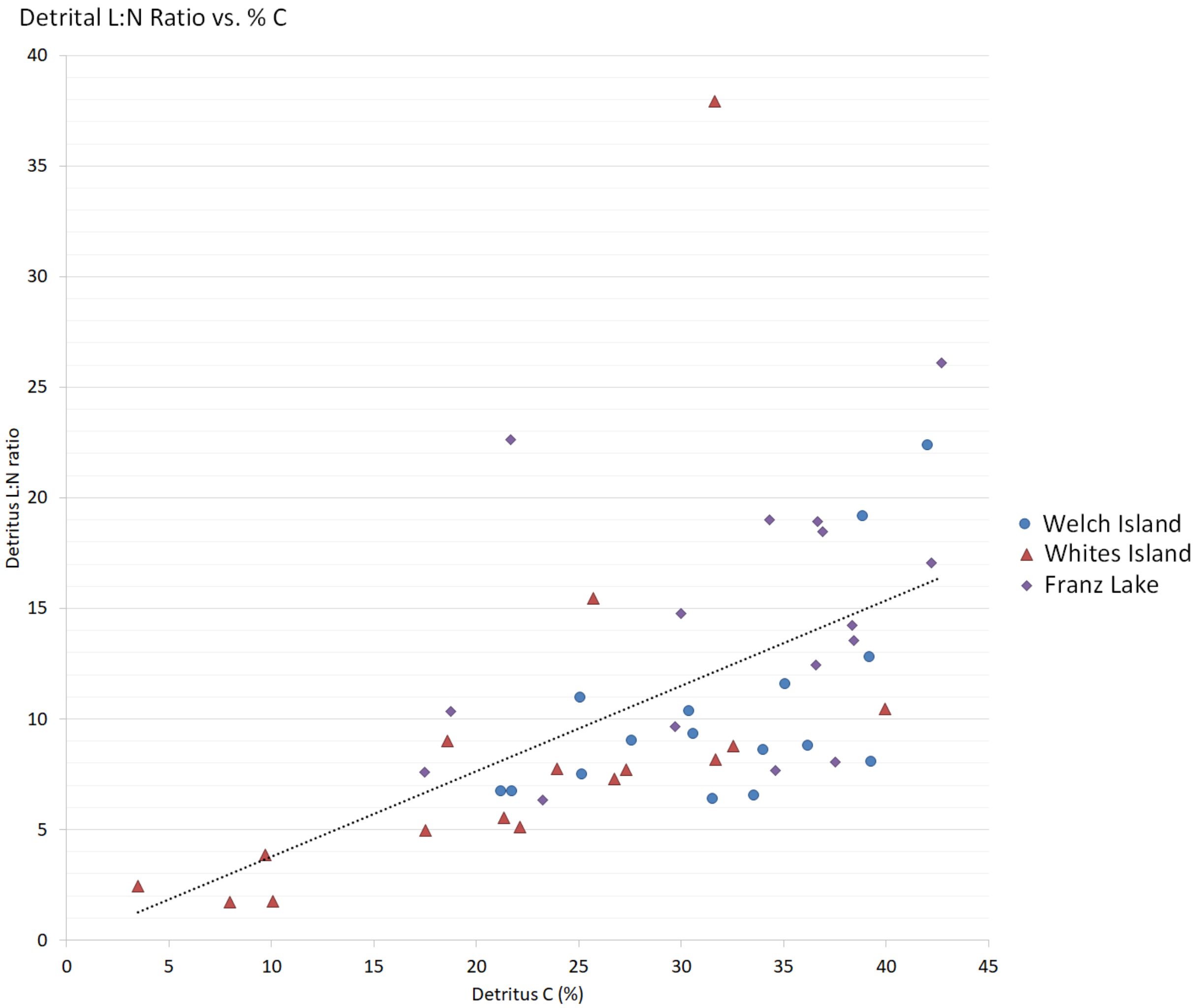


Figure 65: Detritus ADF Lignin:Nitrogen (L:N) content (%) vs. Detritus C content (%), a strong correlation was shown across all sites ($r^2 = 0.30$, $p<0.05$). Summary data provided in Table 35.

General variability in the mean living above ground biomass and detritus carbon, nitrogen, and lignin content can be attributed to several factors including the general variability of these components across different wetland plant species (Table 36, Table 37) and variability between leaf and stem composition. Additionally, due to how the biomass data were collected and the degree of decay associated with detritus samples, detritus could not be associated specifically with any one species. This can make comparing across living and detritus samples challenging because no direct species to species comparison can be made. In general, the living above ground biomass results reflect a mix of stem and leaf materials and it is assumed the detritus samples are also a mix of decaying leaf and stem components.

Comparison among species of the above ground living biomass carbon, nitrogen, and lignin content shows that there is a large range of variability, however, species-specific trends were generally found to be consistent across all sites sampled (Table 36, Table 37). Of the most common species, *Polygonum amphibium* and *Polygonum hydropiperoides* were found to have the highest overall lignin to nitrogen ratios (L:N), 11.7±3.3 and 14.7±4.9, this is not particularly surprising as these species have woody (high in lignin) perennial stems (especially when compared to the other common wetland grass and herb species) that persist throughout the winter months. These species were followed in L:N content by *P. arundinacea*, 6.7±2.8, and *C. lyngbyei*, 6.0±3.2 for the high marsh species, and *S. latifolia*, 2.9±0.6 and *E.*

palustris, 2.4±0.9, for the low marsh species. These L:N ratios mirror observations of decomposition in the field with *P. arundinacea* generally being retained on the site as standing dead biomass, followed in abundance by *C. lyngbyei*, and low marsh species, *S. latifolia,* and *E. palustris,* generally not found as standing dead due to its quick state of decay and location in the low marsh which is exposed to more active hydrologic flushing compared to the high marsh. *P. amphibium* and *P. hydropiperoides* are an interesting comparison to the other marsh species because they do lose their leaves annually without much dead leaf accumulation, but their stems tend to fall dormant (not actually standing dead), indicating that their L:N ratios may vary dramatically between the two plant structures (more in the perennial stems and less in the leaves). Additionally, the C:N ratio of the *P. amphibium, 24.8 7.0,* and *P. hydropiperoides, 29.8 6.3*, species were found to be below that of *P. arundinacea*, 46.2±19.2, and *C. lyngbyei*, 38.2±9.1, and above that of *S. latifolia*, 15.7±3.1, and *E. palustris,* 21.2±4.9. Further testing and distinction between leaves and stems of all species will help us better understand these functional plant traits and how they inform plant decomposition and detrital production within these sites moving forward.

Table 36: Plant species-specific mean (± SD) living above ground biomass elevation (m, NAVD88), ADF Lignin, C:N ratio, %C, %N, and dry biomass (g/m^2) across all sites sorted by mean elevation within each site (low to high marsh). Y = Native, N = Non-native. Data from summer 2018 biomass data collection.

Site	Sp. Code	Scientific Name	Native (Y/N)	n	Elevation	ADF Lignin %	C:N	%C	%N	Biomass g/m^2
Welch Island	ELPA	*Eleocharis palustris*	Y	2	0.98 (0.18)	3.8 (0.1)	24.8 (2.5)	42.3 (1.1)	1.7 (0.1)	122 (160)
	POHY	*Polygonum hydropiperoides*	Y	3	1.8 (0.04)	22 (1.3)	32.6 (3.4)	42.4 (2.1)	1.3 (0.1)	244 (265)
	EQFL	*Equisetum fluviatile*	Y	3	1.88 (0.12)	4 (1)	28.6 (3.1)	37.2 (1.4)	1.3 (0.1)	332 (374)
	PHAR	*Phalaris arundinacea*	N	3	2.01 (0.16)	6.5 (0.7)	52.4 (20.5)	43.6 (0.3)	1 (0.5)	629 (322)
	CALY	*Carex lyngbyei*	Y	4	2.07 (0.07)	7.4 (1.8)	39.3 (10.1)	44.6 (0.9)	1.2 (0.3)	368 (358)
	MYLA	*Myosotis laxa*	Y	1	2.18	11.5	43.1	40.9	1	289
Steamboat Slough	POHY	*Polygonum hydropiperoides*	Y	1	1.33	9.6	19.6	30.6	1.6	14
	ELPA	*Eleocharis palustris*	Y	2	1.44 (0.01)	6.2 (2.7)	26.9 (4.6)	43.2 (0.5)	1.6 (0.3)	165 (148)
	BICE	*Bidens cernua*	Y	3	1.82 (0.03)	11.7 (2.4)	19.7 (4.6)	39 (3.1)	2 (0.3)	12 (5)
	PHAR	*Phalaris arundinacea*	N	5	1.95 (0.18)	3.8 (1.7)	33.9 (3.8)	40.6 (2.4)	1.2 (0.2)	420 (520)
	SALA	*Sagittaria latifolia*	Y	2	1.97 (0.17)	7.3 (1)	15.2 (4.1)	41.9 (3.5)	2.9 (1)	405 (361)
	SCTA	*Schoenoplectus tabernaemontani*	Y	1	2.12	1.5	22.8	45.6	2	1579
	JUEF	*Juncus effusus*	N	2	2.16 (0.02)	5.5 (0.6)	74.9 (24.7)	46.6 (0.1)	0.7 (0.2)	2164 (1404)
Whites Island	POHY	*Polygonum hydropiperoides*	Y	1	1.97	15.5	21.4	82.3	1.9	126
	SALA	*Sagittaria latifolia*	Y	8	1.98 (0.31)	7.5 (1.4)	16.2 (3.8)	43.1 (17.1)	2.4 (0.4)	123 (147)
	ELPA	*Eleocharis palustris*	Y	2	2.01 (0.01)	6 (2.1)	21.6 (0.6)	44.1 (3)	2 (0.2)	59 (45)
	EQFL	*Equisetum fluviatile*	Y	1	2.29	4.2	28.7	36.7	1.3	88
	PHAR	*Phalaris arundinacea*	N	9	2.54 (0.17)	5.9 (0.7)	54 (17.2)	40.8 (2.5)	0.9 (0.5)	648 (550)
	CALY	*Carex lyngbyei*	Y	1	2.63	4.3	33.7	44.8	1.3	61

Franz Lake	SALA	*Sagittaria latifolia*	Y	4	4.24 (0.06)	6 (0.8)	14.8 (1)	39.1 (3.4)	2.7 (0.4)	204 (86)
	ELPA	*Eleocharis palustris*	Y	1	4.25	5.4	13.2	41.3	3.1	22
	POAM	*Polygonum amphibium*	Y	11	4.55 (0.23)	20.1 (2.7)	24.8 (7)	46.4 (14.2)	1.8 (0.5)	679 (654)
	PHAR	*Phalaris arundinacea*	N	5	5.19 (0.38)	6.3 (1.2)	28.5 (10.6)	40.8 (0.5)	1.6 (0.6)	475 (376)

Table 37: Overall plant species-specific mean (± SD) living above ground biomass elevation (m, NAVD88), Lignin:Nitrogen (L:N) content, C:N ratio, %C, %N, ADF Lignin %, and dry biomass (g/m^2). Y = Native, N = Non-native. Data from summer 2018 biomass data collection. Summary of common plant species combining all EMP site data (Welch Island, Whites Island, and Franz Lake).

General Elevation	Sp. Code	Scientific Name	Native (Y/N)	Count	L:N	C:N	C%	N%	ADF Lignin %	Biomass g/m^2
High Marsh	PHAR	*Phalaris arundinacea*	N	17	6.7 (2.8)	46.2 (19.2)	41.3 (2.1)	1.1 (0.6)	6.1 (0.9)	598 (438)
	CALY	*Carex lyngbyei*	Y	5	6.0 (3.2)	38.2 (9.1)	44.6 (0.8)	1.2 (0.3)	6.8 (2.1)	302 (270)
	POAM	*Polygonum amphibium*	Y	11	11.7 (3.3)	24.8 (7.0)	46.4 (14.2)	1.8 (0.5)	20.1 (2.7)	639 (570)
Low Marsh	POHY	*Polygonum hydropiperoides*	Y	4	14.7 (4.9)	29.8 (6.3)	52.4 (20.0)	1.5 (0.3)	20.4 (3.4)	220 (245)
	SALA	*Sagittaria latifolia*	Y	12	2.9 (0.6)	15.7 (3.1)	41.8 (13.9)	2.5 (0.4)	7.0 (1.4)	125 (137)
	ELPA	*Eleocharis palustris*	Y	5	2.4 (0.9)	21.2 (4.9)	42.8 (2.0)	2.1 (0.6)	5.0 (1.6)	77 (105)

Mean soil texture across of the sites (Welch Island, Steamboat Slough, Whites Island, and Franz Lake) can generally be described as silt loam (Table 38, Figure 66). The exact amount of silt, sand, and clay did vary within and among sites, with Welch Island having the highest levels of sand, 33.3 ± 19.2%, followed by Steamboat Slough, 27.7 ± 7.3%, Whites Island, 24.7 ± 9.1%, and Franz Lake, 11.0 ± 6.4%. Correspondingly, Silt levels were greatest at Franz Lake, 77.2 ± 5.8%, followed by Whites Island, 69.9 ± 6.7%, Welch Island, 59 ± 15.4%, and Steamboat Slough 57.5 ± 8%. Steamboat Slough showed the greatest soil clay content, 14.8 ± 4.8%, followed by Franz Lake, 11.7 ± 1.3%, Welch Island 7.6 ± 4.3%, and Whites Island, 5.3 ± 3.3%. Generally, it appears sand and silt soil content are correlated with a location within the estuary, greater silt levels in the upper river site (Franz Lake) and greater sand content at the site located closest to the mouth of the Columbia River (Welch Island) (Figure 66). Soil bulk density was found similar across sites, with the highest bulk density found at Whites Island, followed by Steamboat Slough, Franz Lake, and Welch Island (Figure 66, Table 35).

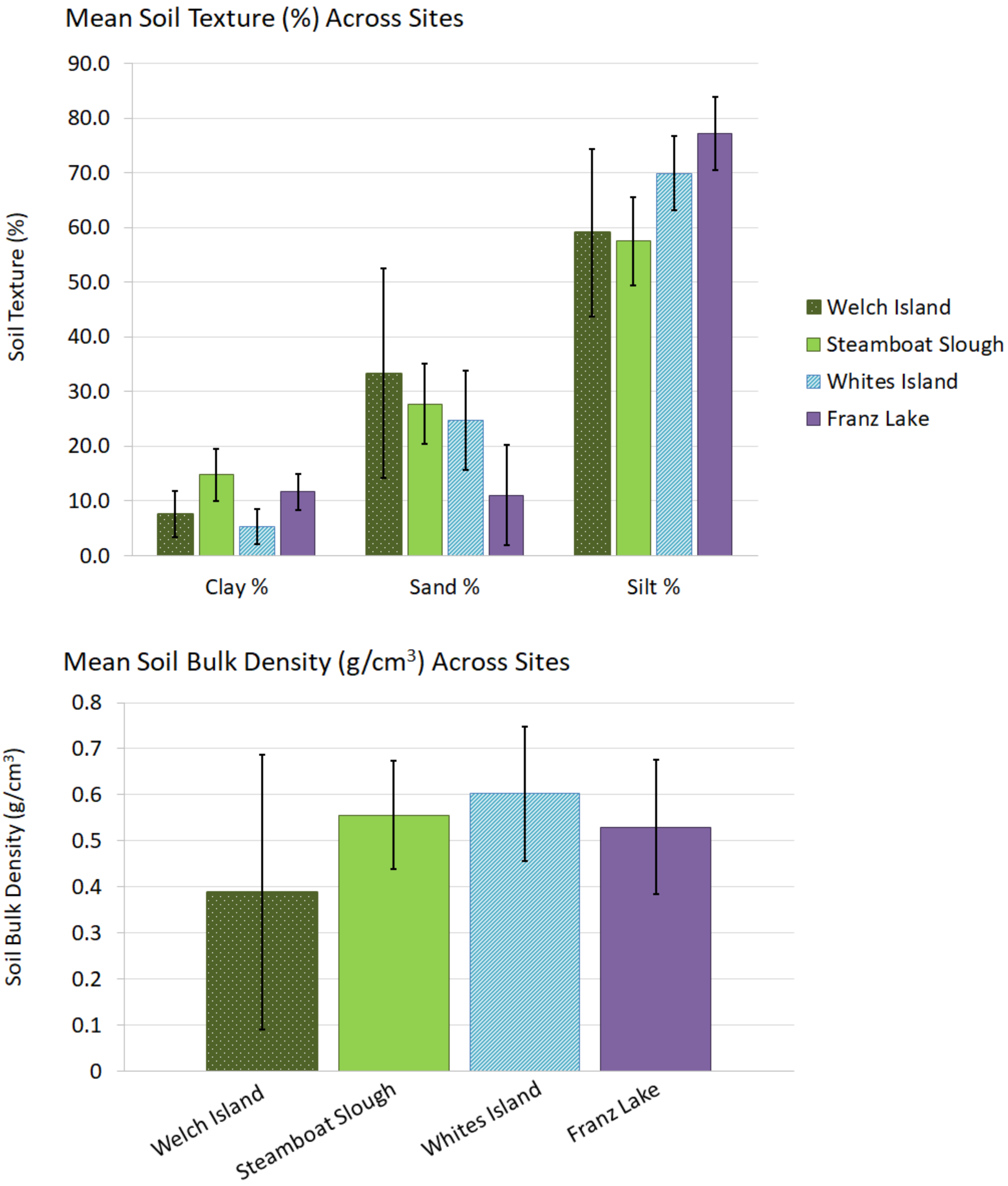


Figure 66. Mean soil texture composition (%) and bulk density (g/cm^3) across sites, samples collected in the Summer of 2018. For summary data see Table 35.

Table 38: Mean soil texture composition (%) and bulk density (g/cm^3) across sites, samples collected in the Summer of 2018. For comparative graphs see Figure 66.

Site		Soil Characteristics			
		Clay (%)	Sand (%)	Silt (%)	Bulk Density (g/cm^3)
Welch Island	n	10.0	10.0	10.0	10
	Mean	7.6	33.3	59.0	0.39
	SD	4.3	19.2	15.4	0.30
Steamboat Slough	n	10.0	10.0	10.0	10
	Mean	14.8	27.7	57.5	0.56
	SD	4.8	7.3	8.0	0.12
Whites Island	n	11.0	11.0	11.0	11
	Mean	5.3	24.7	69.9	0.60
	SD	3.3	9.1	6.7	0.15
Franz Lake	n	10.0	10.0	10.0	10
	Mean	11.7	11.0	77.2	0.53
	SD	1.3	6.4	5.8	0.15

3.4.1.2 *Pelagic*

Primary production contributed by phytoplankton in the water column was estimated by the concentration of the pigment, chlorophyll *a*, which is found in all photosynthetic organisms. In addition to hourly fluorescence-based measurements of chlorophyll *a*, whole water samples were analyzed by extracting the chlorophyll *a* pigment from particulate matter collected on filters. This step is necessary to validate fluorescence data from in situ sensors. Together these data provide information about the amount of biomass associated with fluvial phytoplankton.

In 2019, algal biomass was moderate in March and low in April at trend sites (Figure 67). Following the relatively brief freshet in May, high biomass was observed in Franz Lake Slough, in particular. This site is prone to the development of algal blooms in the summer months, which often discolor the water. Similar to previous years, the lowest chlorophyll *a* values were observed at Ilwaco Slough. Only two measurements exceeded 25 µg L^{-1} in 2019 (Figure 67): Whites Island in March and Franz Lake Slough in May. If a benchmark of 15 µg L^{-1} is used, five observations were above the recommended threshold suggesting poor water quality (Oregon State Water Quality Standards). However, since a body of water is only considered impaired when the threshold is exceeded in observations from three consecutive months, no site met this criterion.

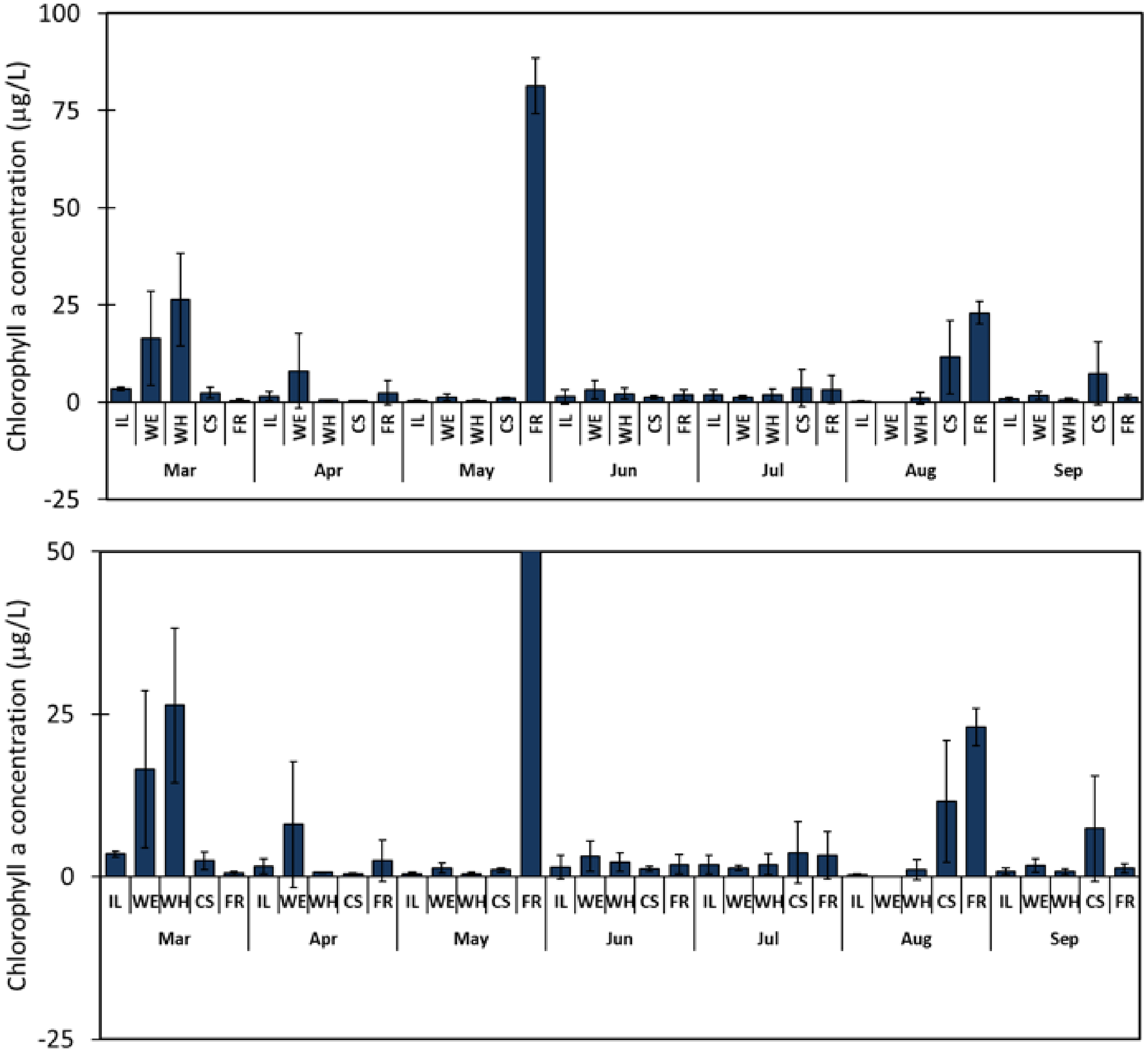


Figure 67: Chlorophyll a concentration at five off-channel trends sites in 2019 (top and bottom). The largest peas were observed at Franz Lake Slough May. The bottom graph shows the same data at a reduced vertical scale to show variation in chlorophyll concentrations at the other sites more clearly.

Aside from the very high value observed in May at Franz Lake Slough, primary production, as approximated by chlorophyll *a* concentration, was highest in March at the lower river sites (Welch Island and Whites Island) and in August at the more upriver sites (Campbell Slough and Franz Lake Slough), 2019 (Figure 67**Error! Reference source not found.**).

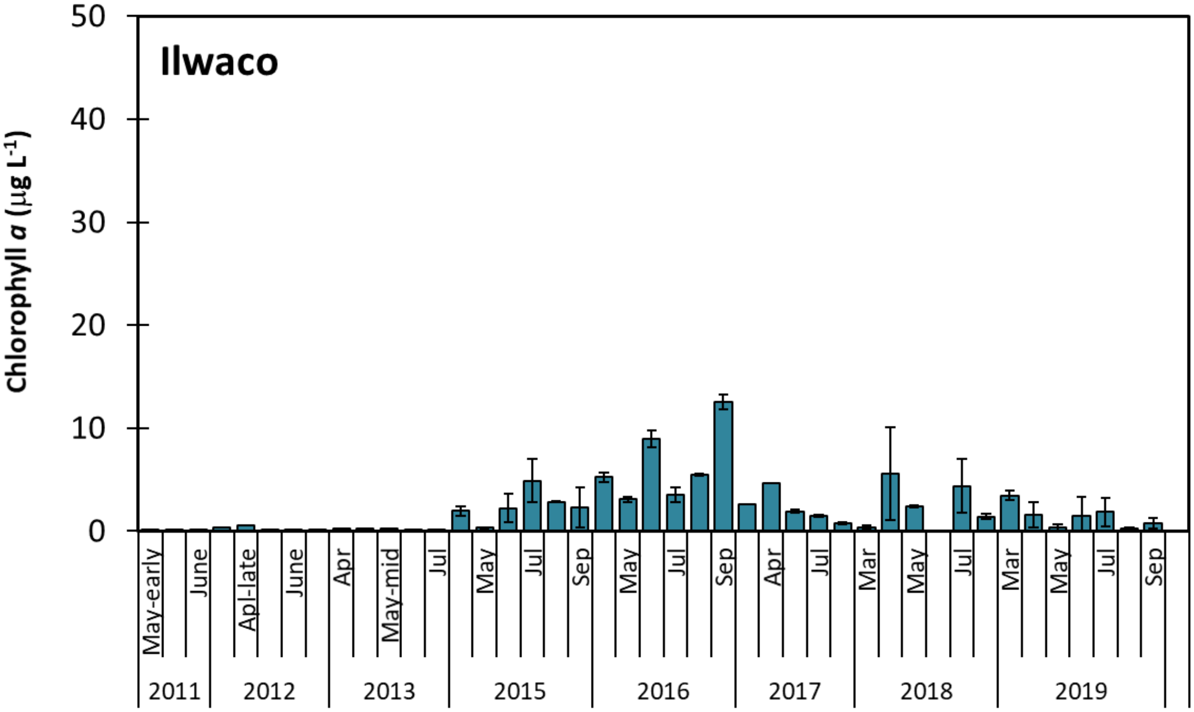


Figure 68. Chlorophyll *a* concentration in discrete samples collected from Ilwaco (2011–2019).

At Ilwaco in 2019, magnitudes of primary production were lower than in most other years (Figure 68**Error! Reference source not found.**); chlorophyll values were highest in March and lowest in May and August. Values were always less than at the other sites and generally <10 µg L^{-1} throughout the time series. At Welch Island, chlorophyll *a* values were relatively high in March and April and relatively low thereafter (Figure 69**Error! Reference source not found.**).

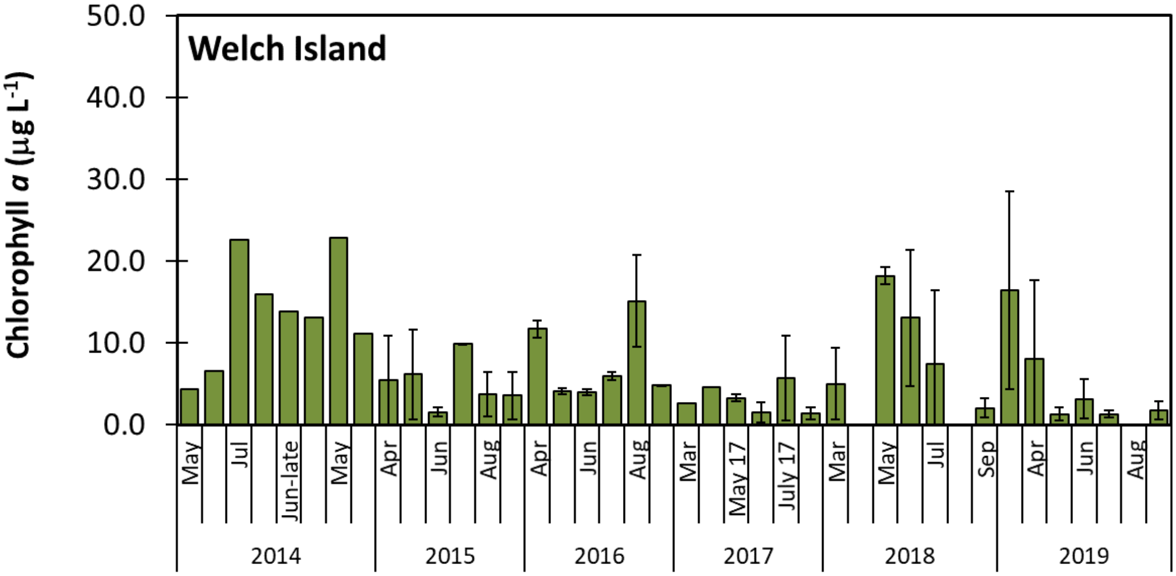


Figure 69. Chlorophyll *a* concentration in discrete samples collected from Welch Island (2014–2019).

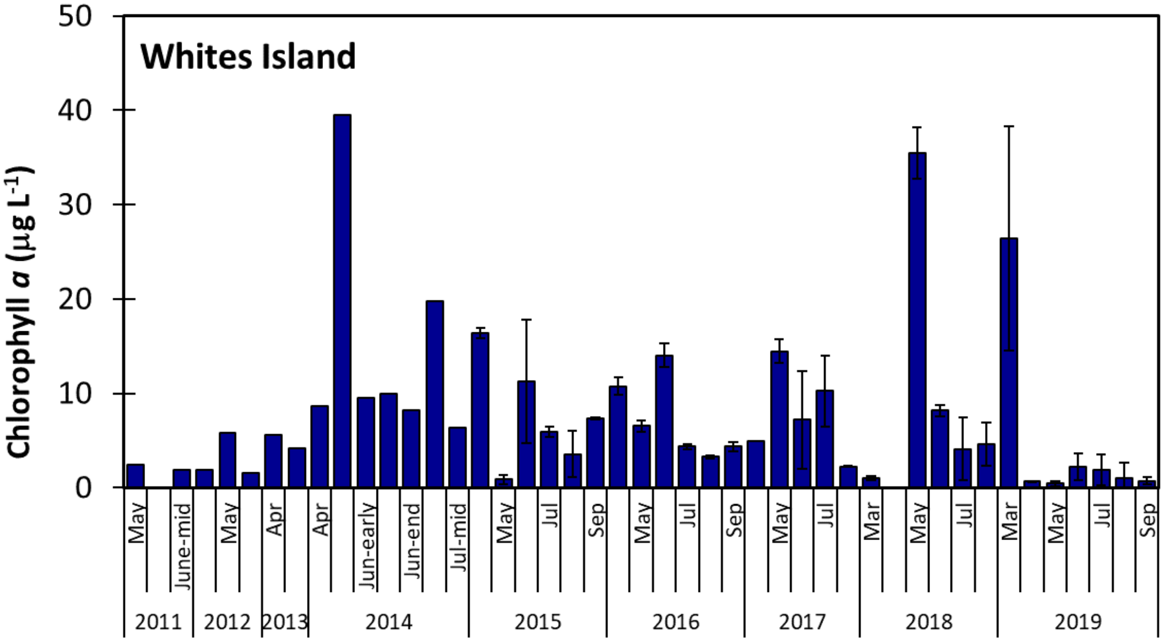


Figure 70. Chlorophyll *a* concentration in discrete samples collected from Whites Island (2011–2019).

Pre-freshet primary production in 2019 was among the highest values of the time series at Whites Island (Figure 70**Error! Reference source not found.**). After that, however, chlorophyll *a* values were lowest of all monitoring years.

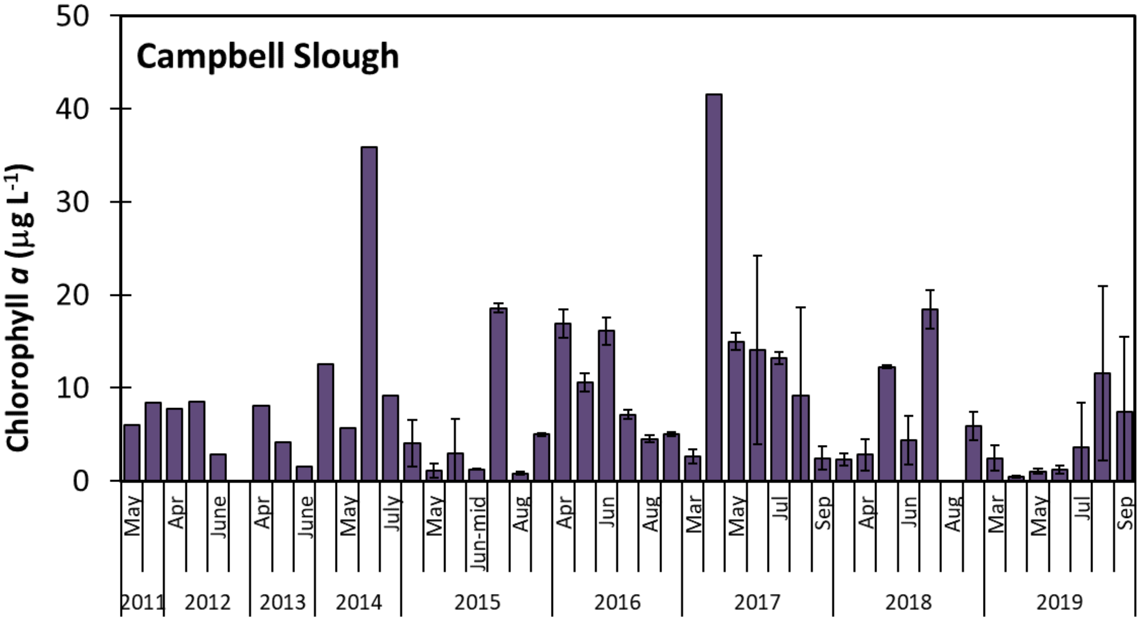


Figure 71. Chlorophyll *a* concentrations in discrete samples collected from Campbell Slough (2011–2019).

In 2019, at Campbell Slough, chlorophyll *a* values were lower than the long-term average in the early spring (March-April) and as the freshet was subsiding in June (Figure 71**Error! Reference source not found.**). Chlorophyll a values in 2019 at Franz Lake Slough were similar to 2016 and 2017 (except the very high value observed in June 2017) (Figure 72**Error! Reference source not found.**). Franz Lake Slough includes more outliers than the other sites, indicating that high-density phytoplankton blooms are more likely to form at this site, particularly in the summer months.

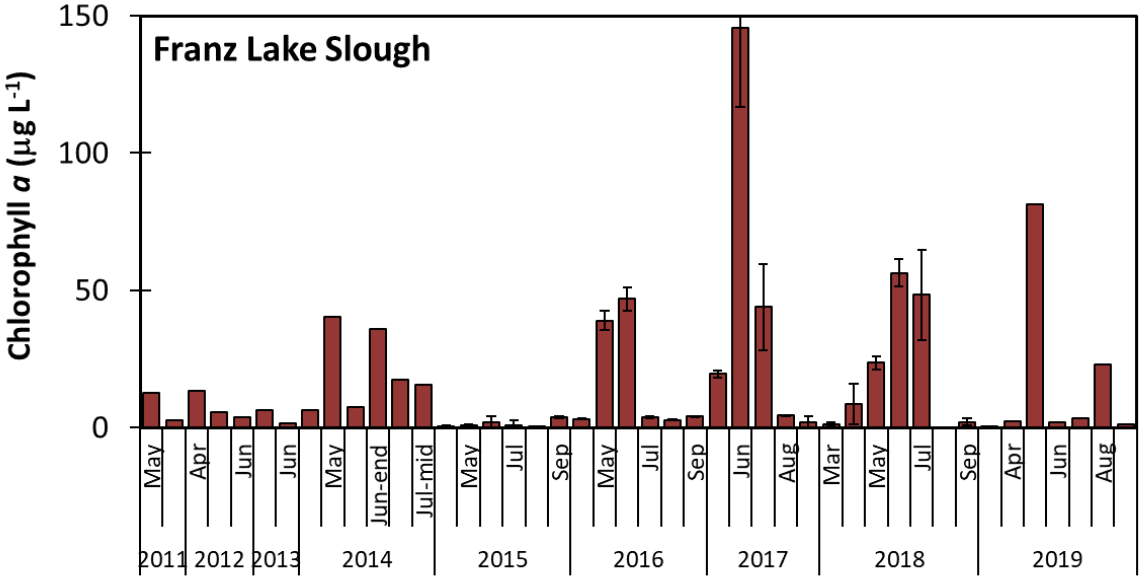


Figure 72. Chlorophyll *a* concentrations in discrete samples collected from Franz Lake Slough (2011–2019). From the time series, it is apparent that the high chlorophyll *a* concentration observed at Franz Lake Slough in June 2017 was exceptionally compared to observations in other years.

Looking at a summary of chlorophyll data from the five trend sites, it is clear that the highest episodic values are seen at Franz Lake Slough (Figure 73**Error! Reference source not found.**). The distribution of data is similar at Campbell Slough, Welch Island, and Whites Island, although Campbell Slough. The fewest high values are observed at Ilwaco Slough, where the bulk of the chlorophyll concentrations are low and consistent.

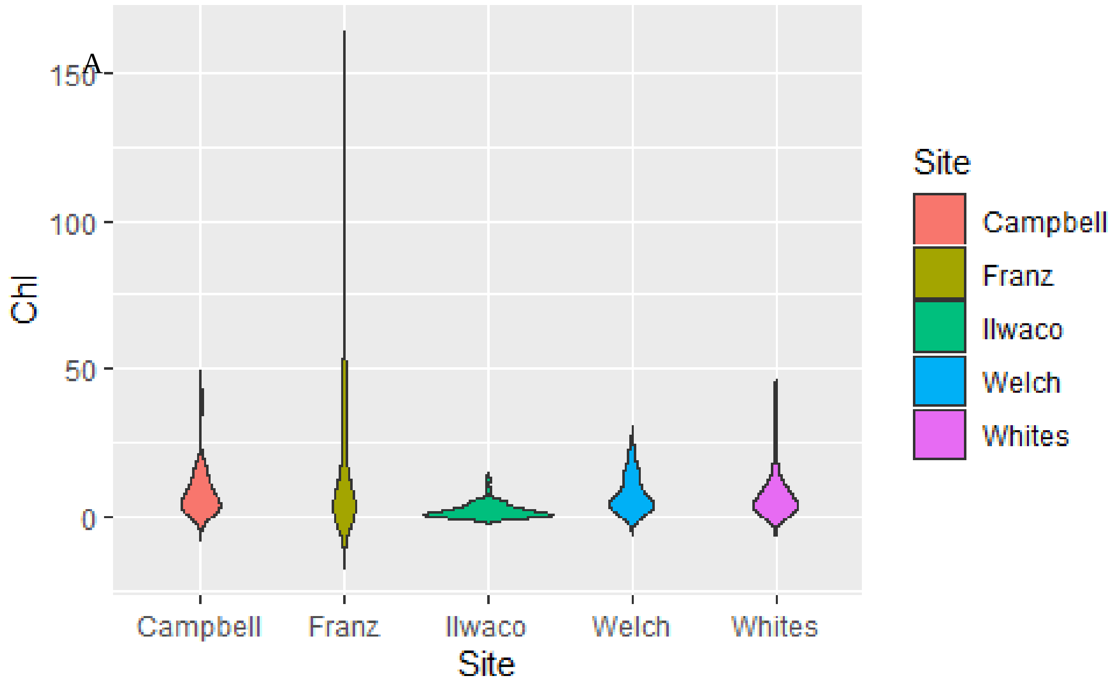


Figure 73. Violin boxplots showing chlorophyll concentrations determined at the five off-channel trends sites between 2011 and 2019.

3.4.1.3 *Phytoplankton Species Composition*

Sampling for phytoplankton species composition took place between March and September from 2011-2019, although sampling was not performed during each of these months for all years. Phytoplankton taxa were placed in the following groupings: diatoms (Class Bacillariophyceae), chlorophytes (Class Chlorophyceae), chrysophytes (Class Chrysophyceae), cryptophytes (Class Cryptophyceae), cyanobacteria (Class Cyanophyceae), and dinoflagellates (Class Dinophyceae). Also, ciliates were enumerated, since there are some species that can be photosynthetic (e.g., *Mesodinium rubrum*; Lindholm et al., 1985, Herfort et al., 2011a, 2011b), particularly at Ilwaco Slough. However, ciliate abundances were relatively low and therefore were not included in plots. The following sections report out on long-term species composition at trend sites between 2011-2019.

Overview

At all sites, diatoms (Class Bacillariophyceae) tend to dominate the phytoplankton assemblage in the spring, prior to the annual snowmelt-driven freshet. During the freshet, populations decline due to dilution by the increased water volume. In the summer, a shift from diatoms to green algae and unicellular flagellates tends to occur at the off-channel sites, with the exception of Ilwaco, which becomes increasingly influenced by tidal exchange with the coastal ocean. At Campbell Slough and Franz Lake Slough, cyanobacteria abundance increases to high levels in the summer months, particularly from July through September.

Over the years of study, we observed a similar succession pattern among the diatoms, with early spring assemblages dominated by small centrics (mainly *Cyclotella* spp.). Following the small centrics is *Asterionella formosa*, which occurs either in advance of or coincident with *Aulacoseira* spp. Next comes *Fragilaria* (*F. crotonensis and F. intermedia*), and finally *Skeletonema potamos*. In some years, *Tabellaria* spp. occurs at the same time as *Skeletonema*. In different years, we observe these key species at different times, depending on environmental conditions and when we were able to collect samples.

Ilwaco Slough

The phytoplankton assemblage at Ilwaco varies seasonally, with riverine conditions dominating until after the spring freshet subsides. Assemblages look more similar to the other sites in early to mid-spring (March-May), with diatoms dominating the assemblage (Figure 74). During the transition to summer, the assemblage includes a greater proportion of cryptophytes and other flagellate species (e.g., chrysophytes). There was a higher abundance of cyanobacteria species at Ilwaco in 2019 compared to previous years in both April-May and August (Figure 75).

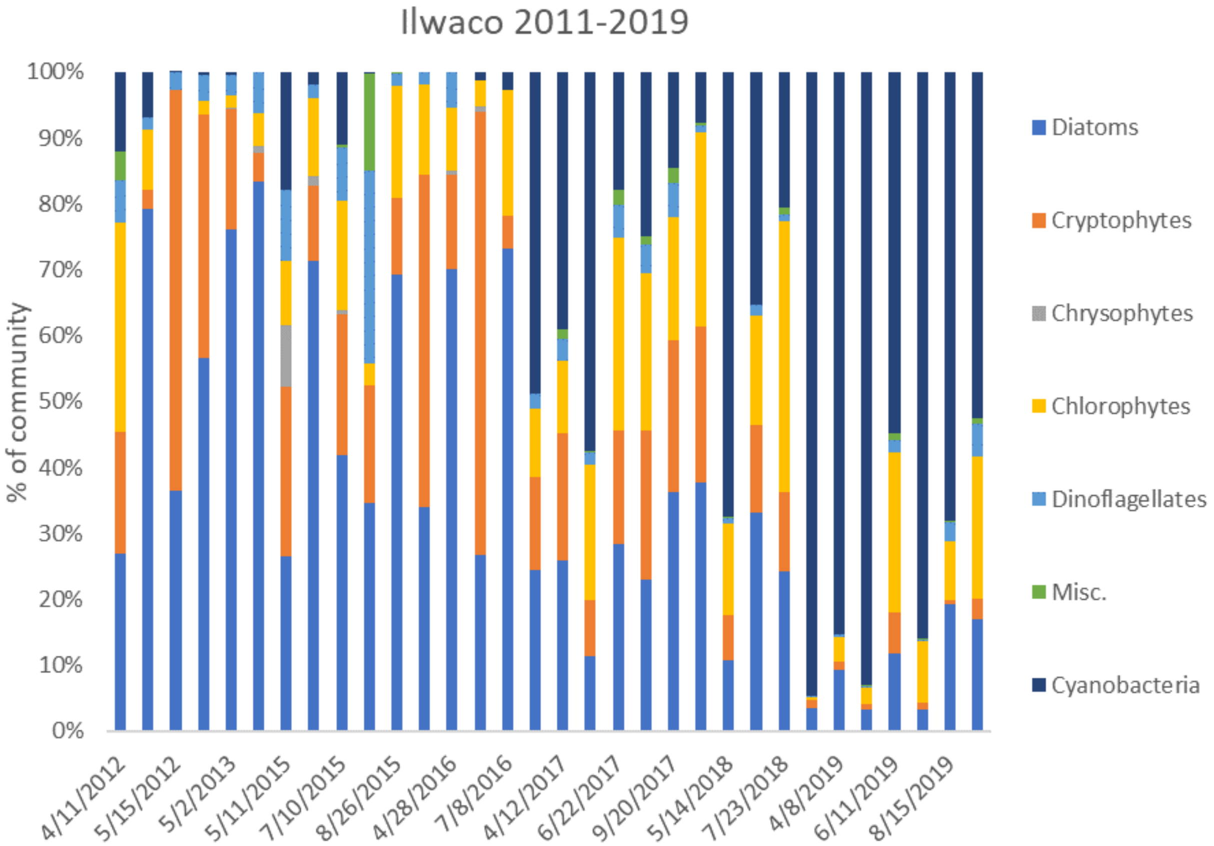


Figure 74. Relative percentages of different phytoplankton classes at Ilwaco observed during the spring and summer months between 2011 and 2019.

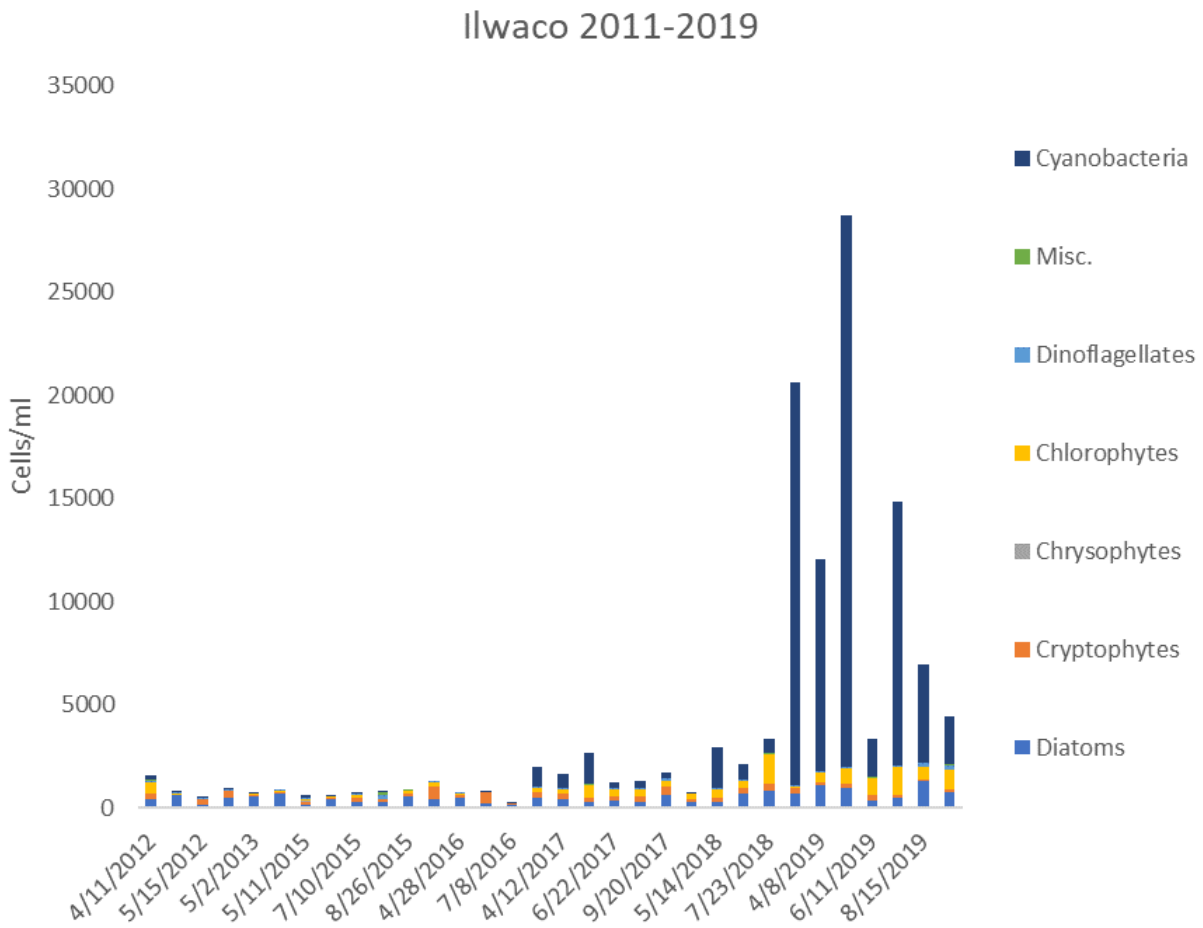


Figure 75. Absolute abundances of phytoplankton classes at Ilwaco from 2011 to 2019.

Welch Island **and Whites Island**

The phytoplankton assemblages at Welch Island and Whites Island was dominated by diatoms, particularly in early spring (Figure 76 and Figure 78). Abundances of cyanobacteria were higher in 2017, 2018 and 2019 than in previous years, particularly in the summer months at Welch Island (Figure 76). Abundances of Cyanobacteria were higher at Whites Island in 2017 and 2018, and had since reduced in 2019 (Figure 79).

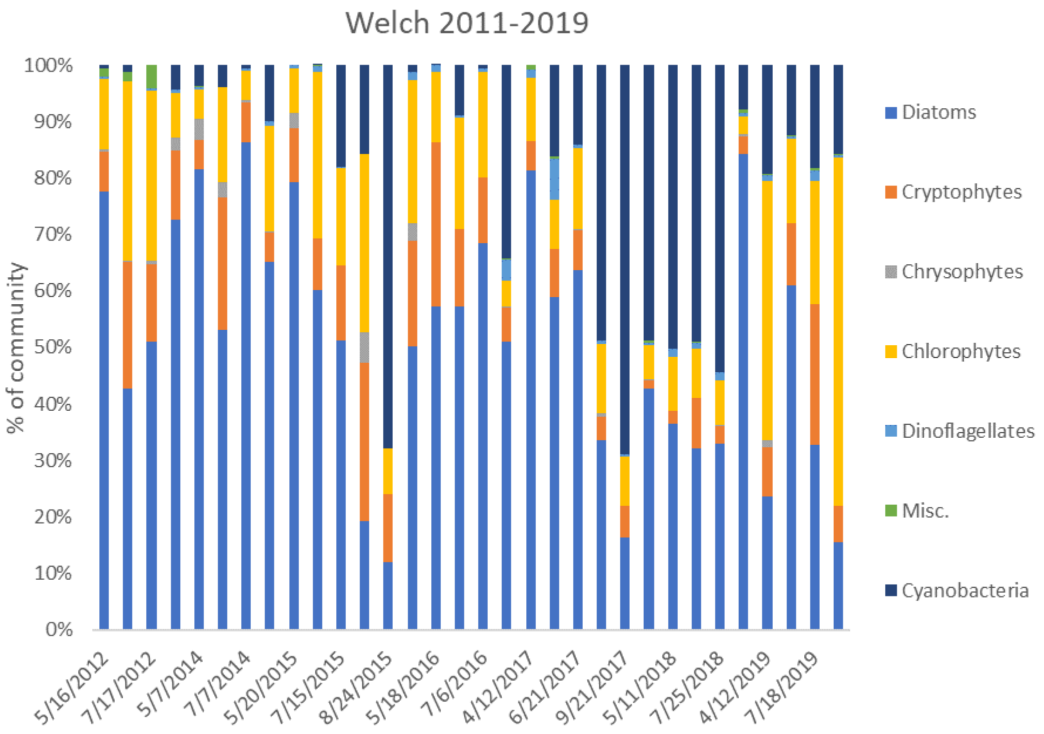


Figure 76. Relative percentages of different phytoplankton classes at Welch Island from 2011 to 2019.

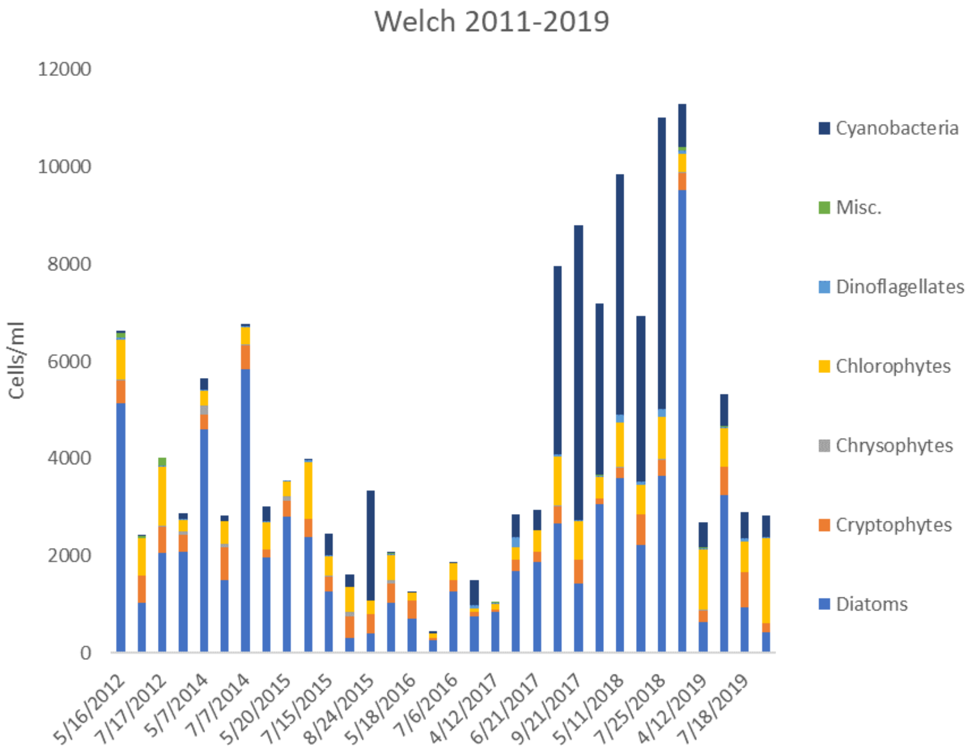


Figure 77. Absolute abundances of phytoplankton classes at Welch Island from 2011 to 2019.

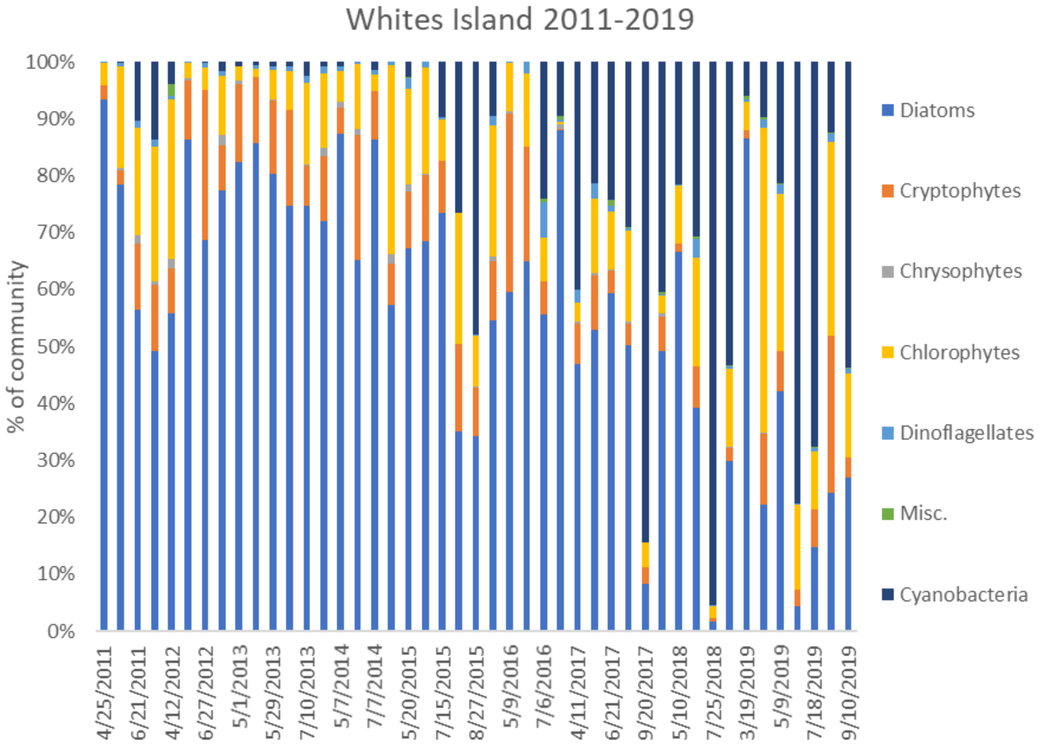


Figure 78. Relative percentages of different phytoplankton classes at Whites Island from 2011 to 2019

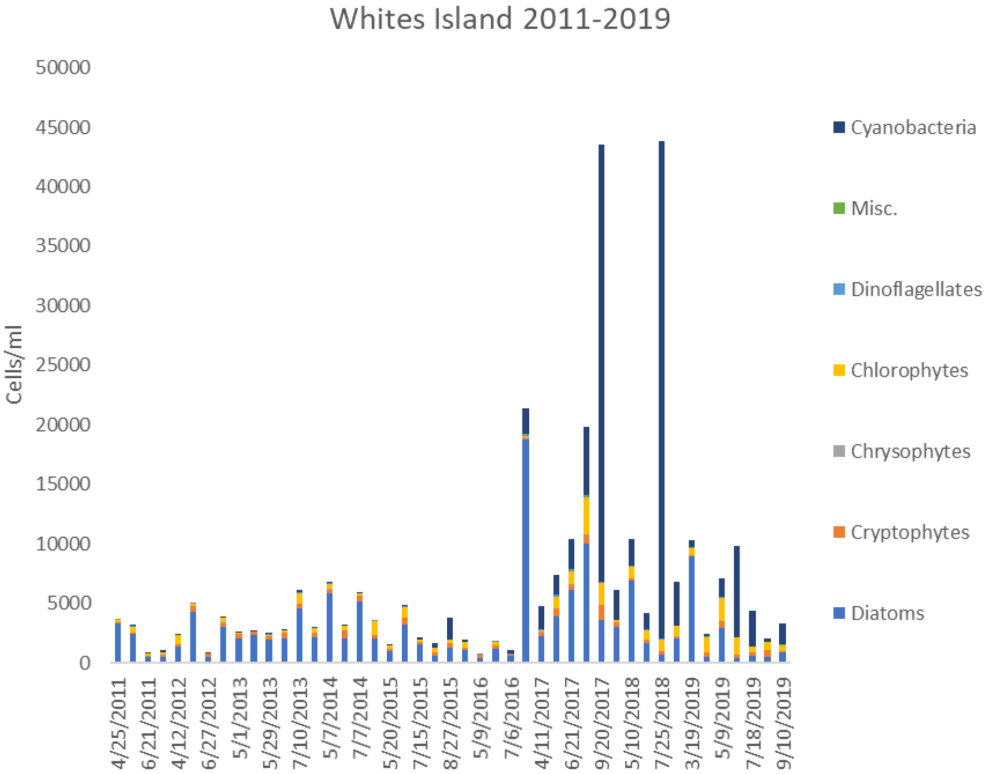


Figure 79. Abundances of different phytoplankton classes (in cells/mL) at Whites Island from 2011 to 2019.

Campbell Slough and Franz Lake Slough

Phytoplankton assemblages in the upriver sites have evolved since summer 2015, showing a growing dominance of cyanobacteria at both Campbell and Frank Lake Sloughs, with a short hiatus in 2016 (Figure 80 and Figure 82). Abundance of cyanobacteria were greater in 2018 than in 2019 at Campbell (Figure 81). Cyanobacteria abundance in 2019 at Franz Lake slough was significantly lower than previous years (Figure 83).

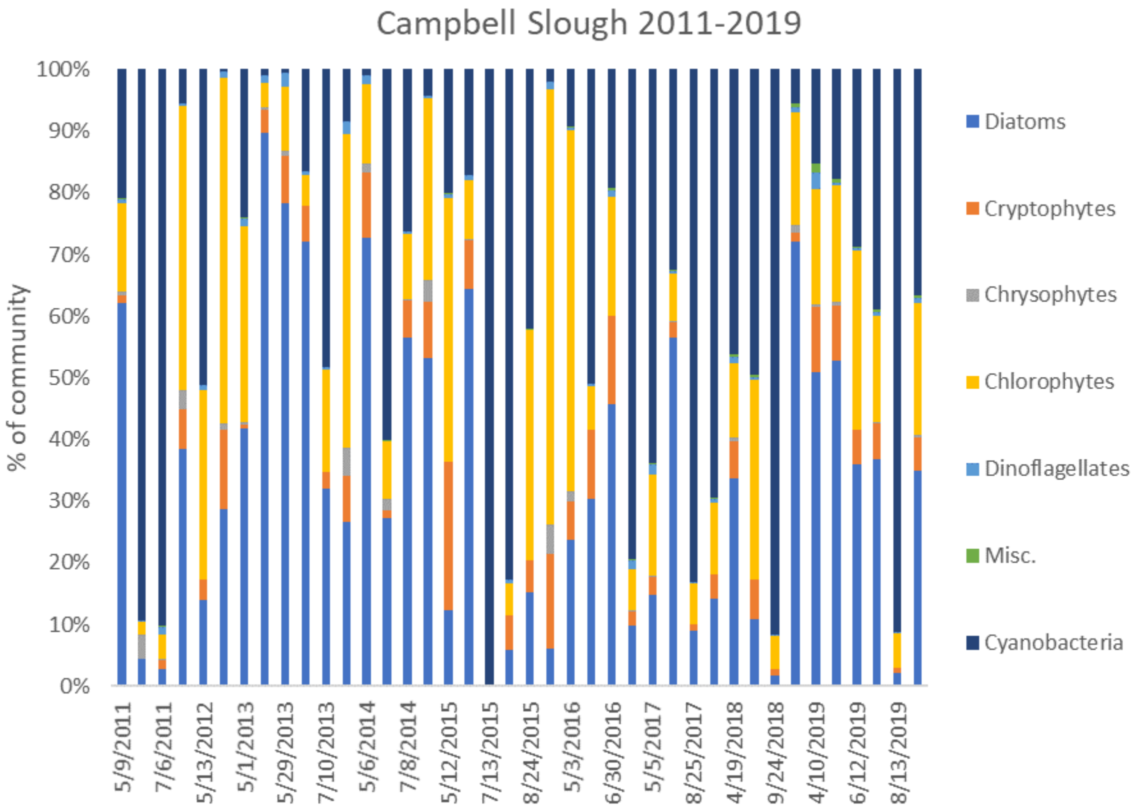


Figure 80. Relative percentages of different phytoplankton classes at Campbell Slough from 2011 to 2019.

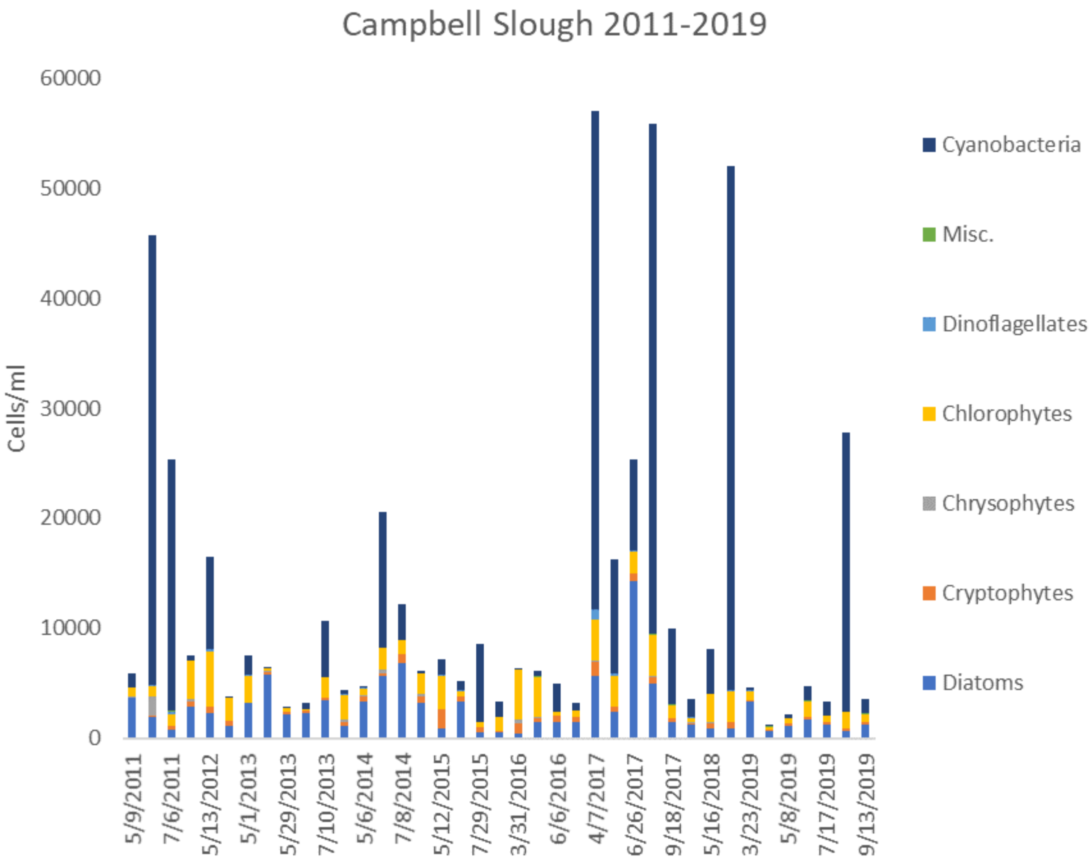


Figure 81 Abundances of different phytoplankton classes (in cells/mL) at Campbell Slough from 2011 to 2019.

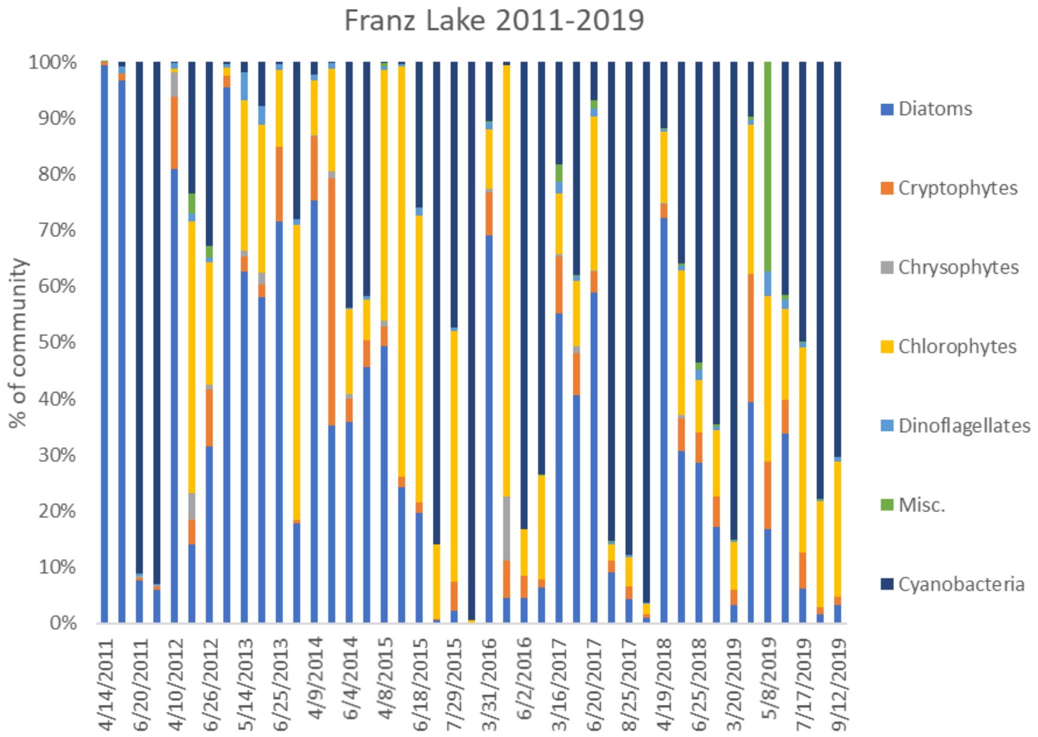


Figure 82. Relative percentages of different phytoplankton classes at Franz Lake Slough from 2011 to 2019.

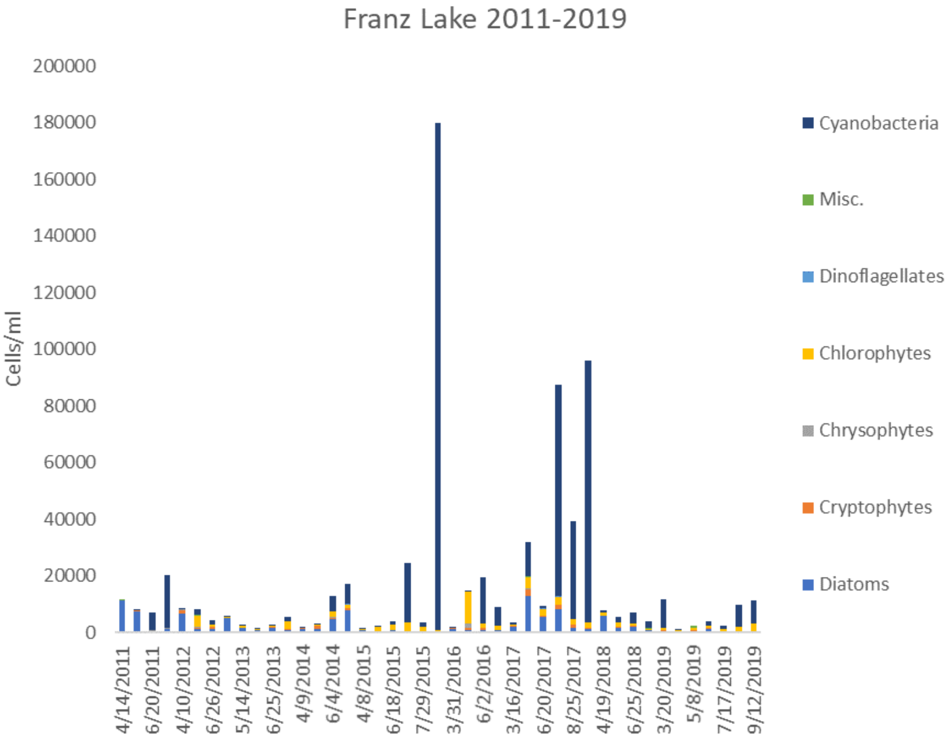


Figure 83. Abundances of different phytoplankton classes (in cells/mL) at Franz Lake Slough from 2011 to 2019.

3.4.1.4 *Phytoplankton and environmental variables*

To look for relationships between environmental variables and phytoplankton assemblages, we performed canonical correspondence analysis on phytoplankton and environmental data collected between 2012 and 2019 (Figure 84). The plot shows a greater spread along the second axis of variability, which was most closely related to nitrate concentration, temperature, and percentage of maximum discharge. Positive associations were observed between temperature and abundance of chlorophytes, dinoflagellates, and cryptophytes, while negative associations were observed between temperature and the abundance of diatoms and chrysophytes. The first axis was most closely associated with variation in phosphate concentration, which was most closely associated with variations in the abundance of cyanobacteria and negatively associated with the abundance of diatoms. The CCA also showed that high levels of nitrate and the highest percentages of maximum yearly discharge were associated with diatoms, while cryptophytes, dinoflagellates, and chlorophytes were negatively related to nitrate concentration.

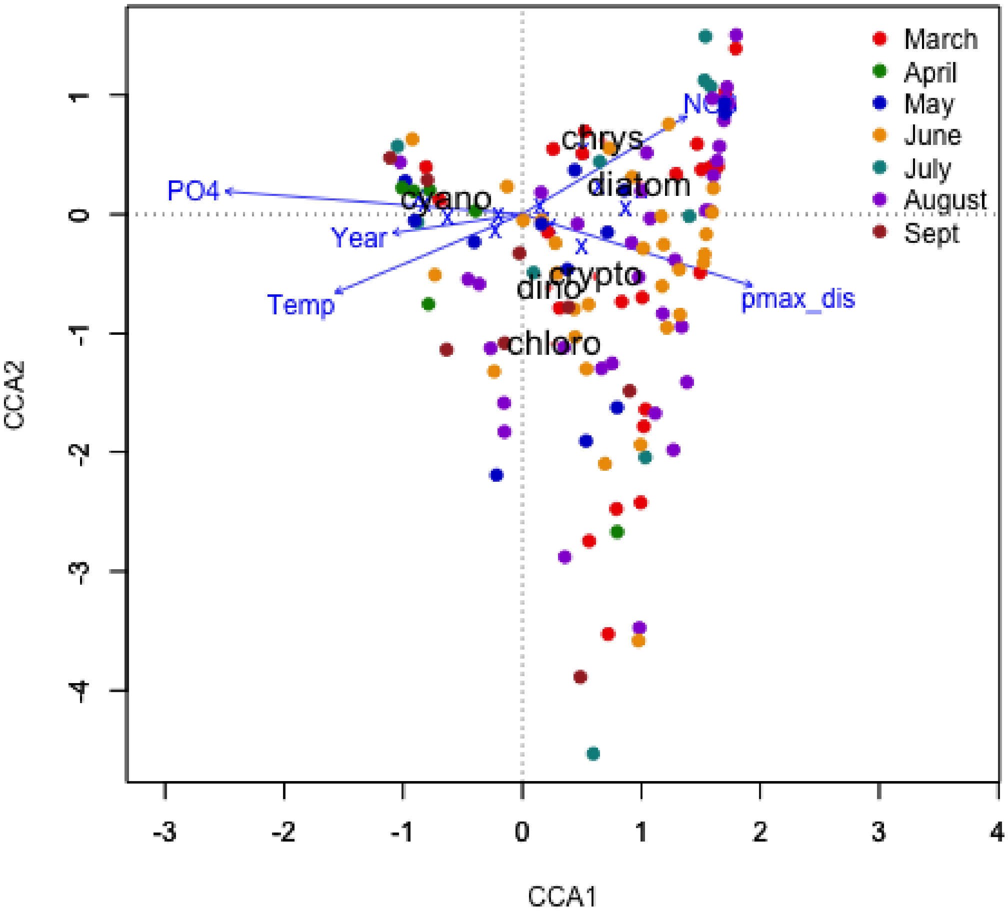


Figure 84. Biplot generated from canonical correspondence analysis relating phytoplankton taxonomic groups (labeled according to month sampled) to environmental variables: PO_4 = phosphate, NO_3 = nitrate, Temp = river temperature

3.4.2 Spring Zooplankton Assemblages

In 2019, at the time of writing, data are available for Campbell Slough and Whites Island, but not the other sites. Hence, we have provided an in-depth view of the changes in zooplankton assemblages between years 2017 to 2019.

In 2017, copepods and cladocerans accounted for the greatest proportion of mesozooplankton assemblages across sites and months. The greatest density and diversity of cladocerans was observed at Campbell Slough, particularly in 2017. Aside from Ilwaco in Reach A, where copepods dominated the zooplankton assemblage, there was a shift from dominance by copepods to cladocerans from spring to summer in 2017, 2018, and 2019.

Among the copepods, cyclopoids generally dominated the zooplankton assemblages across sites, except in a few cases where either harpacticoids or calanoids co-dominated alongside the cyclopoids (e.g., Whites Island in June 2019 and Franz Lake Slough in 2017). There was less of a dominance of cyclopoid taxa at Ilwaco, where harpacticoid and calanoid taxa were co-dominant in May/June 2018. Cyclopoid densities peaked earlier in 2017 (May) than 2018 (June-July) at Whites Island, Campbell Slough, and Franz Lake Slough. At Ilwaco, harpacticoid abundances peaked in May in both 2017 and 2018.
In 2017, dipteran larvae were present early in the season at Ilwaco, while they appeared at Whites and Franz during the summer months.

2017

In 2017, copepods dominated the mesozooplankton assemblages at all sites (Figure 85). Densities of copepods were highest at Campbell Slough, where cyclopoids were approximately 10 times denser than at the other sites. Franz Lake Slough had the lowest copepod densities, which consisted mainly of calanoid and harpacticoid species. The greatest proportional abundance of copepods was observed at Ilwaco Slough, where assemblages were dominated by harpacticoid and cyclopoid species. The zooplankton assemblage at Welch Island in Reach B was also composed mainly of harpacticoid and cyclopoid copepods early in the season, followed by cladocerans later in the season. Cladoceran assemblages at Welch included *Bosmina* spp. (with and without eggs) and *Daphnia* spp. In general, juvenile copepods made up most of the zooplankton assemblage at Whites Island. The highest zooplankton densities were observed in May (i.e., during the spring freshet), when abundances of both copepods (consisting primarily of cyclopoids with a high proportion of juveniles) and cladocerans (dominated by *Bosmina* spp.) peaked.

Among the cladocerans, *Bosmina* species dominated at all sites in 2017, except for Ilwaco where *Daphnia* spp. were the most abundant cladocerans taxa. The density of cladocerans at Campbell Slough was nearly 20 times higher than at any other site in 2017, with the highest abundances occurring in April and June (i.e., before and after the spring freshet). While *Bosmina* spp. dominated at Campbell Slough like the other sites throughout the year, *Ceriodaphnia, Chydorus* and *Daphnia* were also present in high abundance in June when total cladoceran densities were highest. The greatest diversity of cladocerans in terms of richness, or the number of species observed, was also observed at Campbell Slough in June. Similar to Campbell Slough, cladocerans were most abundant in June at Welch Island, where the assemblage was dominated by *Bosmina* spp. Cladocerans made up half of the zooplankton assemblage at Welch Island in June. In June, dipteran larvae were most abundant at Whites and Campbell compared to other sites.

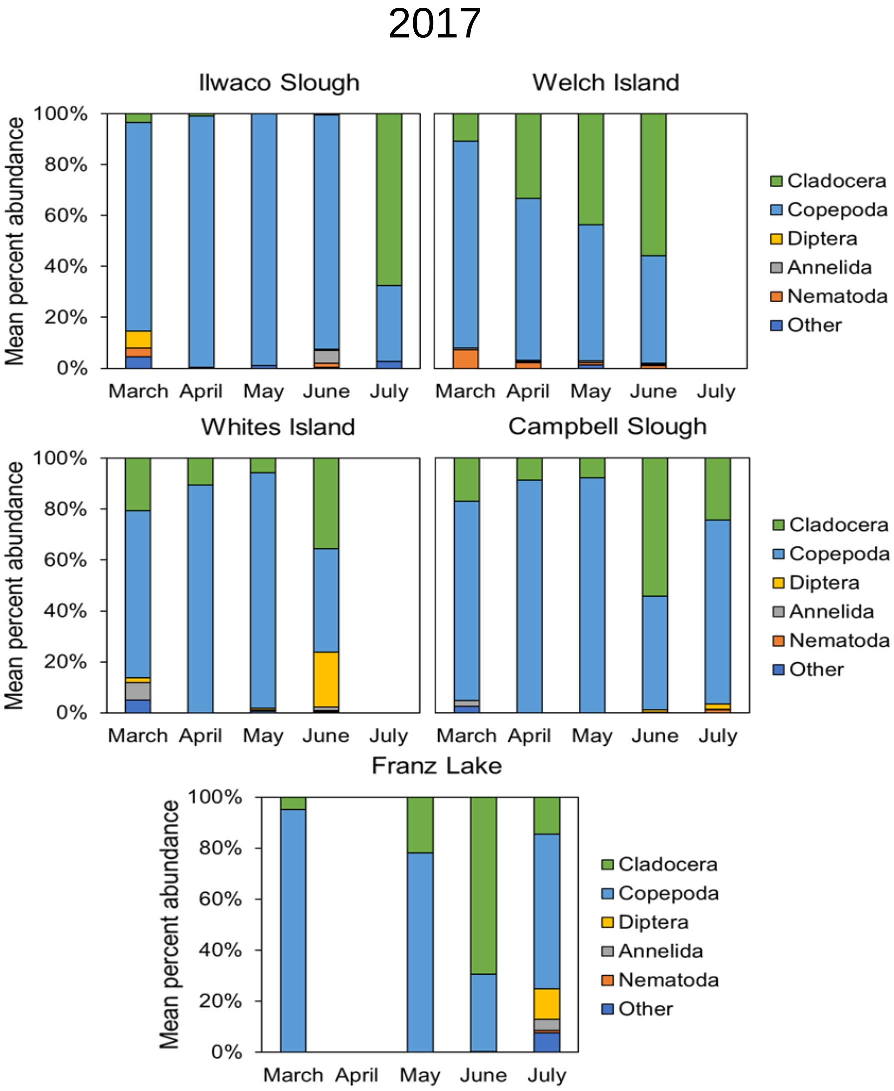


Figure 85. Mean percent abundance for different zooplankton taxonomic groups at the five trends sites in 2017.

2018

Copepods and cladocerans accounted for most of the diversity and abundance of mesozooplankton assemblages at all sites in 2018 (Figure 86). In 2018, Ilwaco Slough had the greatest abundances of zooplankton, with densities ten times more than the other sites. Similar to previous years, zooplankton assemblages were dominated by copepods in the spring (March–May) except at Campbell Slough, where cladocerans were already dominant by May. Among the copepods, cyclopoid and calanoid species dominated at all sites except for Ilwaco, where calanoid and harpacticoid species together accounted for the greatest proportion of copepod taxa in all months sampled, with abundances peaking in June. Aside from the high densities of copepods at Ilwaco in May and June, copepod species at Welch Island in April were two to three times denser than any other site in April.

Cladocerans increased in abundance from spring to summer at Welch Island and Whites Island, but not at Campbell Slough, Franz Lake Slough, or Ilwaco (Figure 88). Similar to 2017, *Bosmina* spp. dominated the assemblage of cladocerans at all sites in 2018. The highest diversity of cladocerans was observed at Campbell Slough, where—in addition to *Bosmina* spp—*Chydorus* and *Ceriodaphnia* species were also present at high relative abundance. *Bosmina* spp. dominated the zooplankton assemblage at Campbell Slough in May. In 2018, ephippiums, or resting eggs, were abundant in May at Whites Island. Cladoceran species were rarely found at Ilwaco Slough and when they did occur, diversity was low. *Daphnia* spp. made up abundances in June, while *Bosmina* spp. were present in April and May.

Interestingly, although copepods and cladocerans were most abundant in 2018, similar to 2017, annelids were found in high numbers at Ilwaco in June and polychaete larvae accounted for nearly 5% of the zooplankton assemblage. By July, polychaete larvae made up 67% of the zooplankton assemblage at Ilwaco. Nematodes made up >10% of the zooplankton composition March at Ilwaco, and they accounted for ~5% of the zooplankton assemblage at Whites Island in May at Whites Island and in July at Franz Lake Slough. Dipterans were present in low numbers across all sites, making up less than 5% of composition at most.

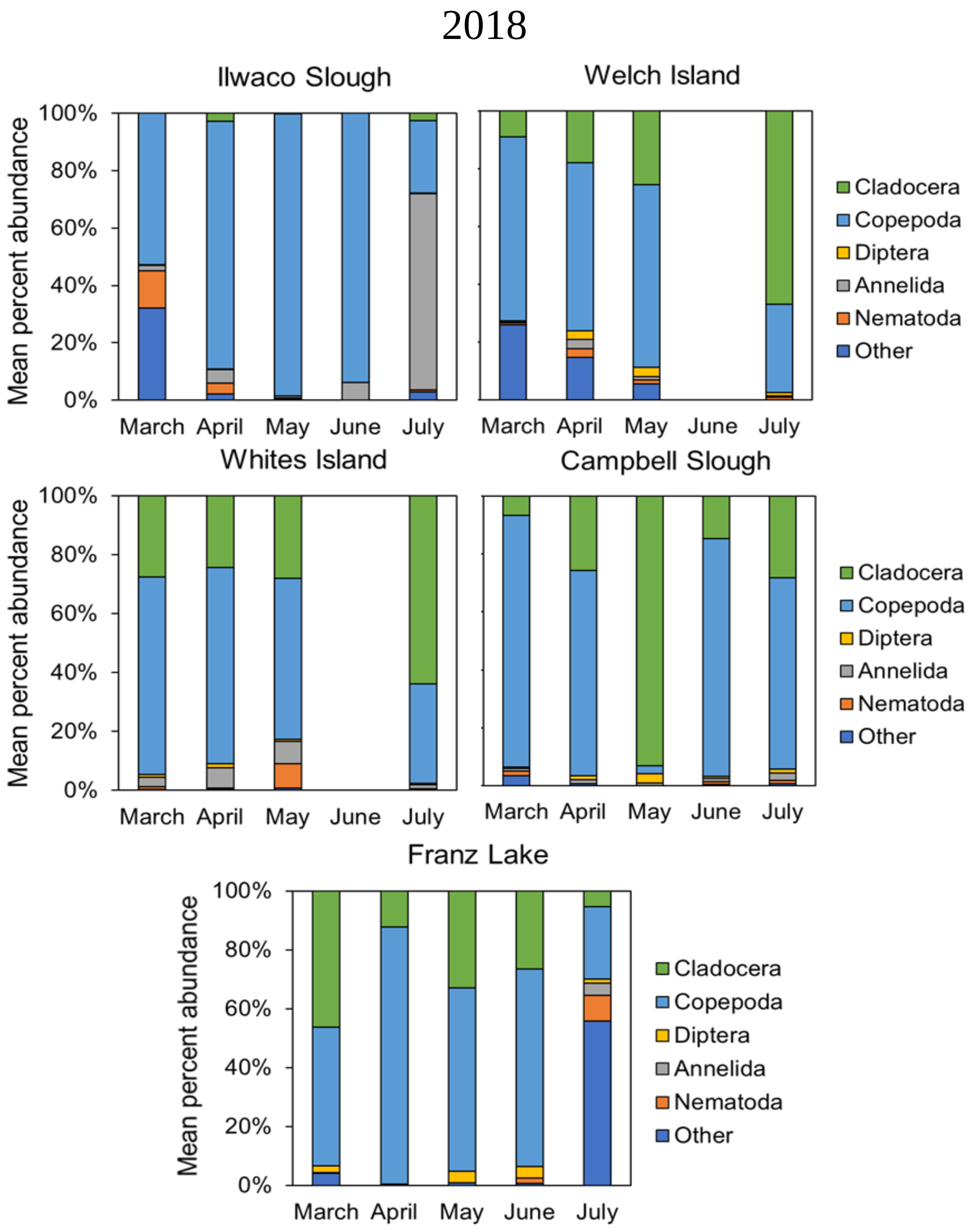


Figure 86. Mean percent abundance of different zooplankton taxonomic groups at the five trends sites in 2018.

2019

At the time of writing, data are available for Campbell Slough and Whites Island, but not the other sites. Similar to previous years, copepods and cladocerans dominated the mesozooplankton assemblages during all months sampled at Whites Island and Campbell Slough (Figure 87). Overall, Whites Island had higher mesozooplankton densities compared to Campbell Slough. Although copepod densities peaked in June (when the peak total zooplankton density was observed), they were found at similar relative proportions at Whites Island during all months, dominated by cyclopoids and harpacticoids; however, there was greater variability in the relative abundance of copepods at Campbell Slough due to increases in the abundance of cladocerans.

Among the copepods, cyclopoid species dominated at Campbell Slough; at Whites Island there was a co-dominance of cyclopoid and harpacticoids. Among the cladocerans, *Bosmina* species dominated at Whites Island while, in addition to *Bosmina* spp., *Chydoridae* spp. dominated at Campbell Slough; *Alona* and *Ceriodaphnia* were also found at relatively high proportions at Campbell Slough in June. Similar to copepods, the peak abundance of cladocerans was observed in June.

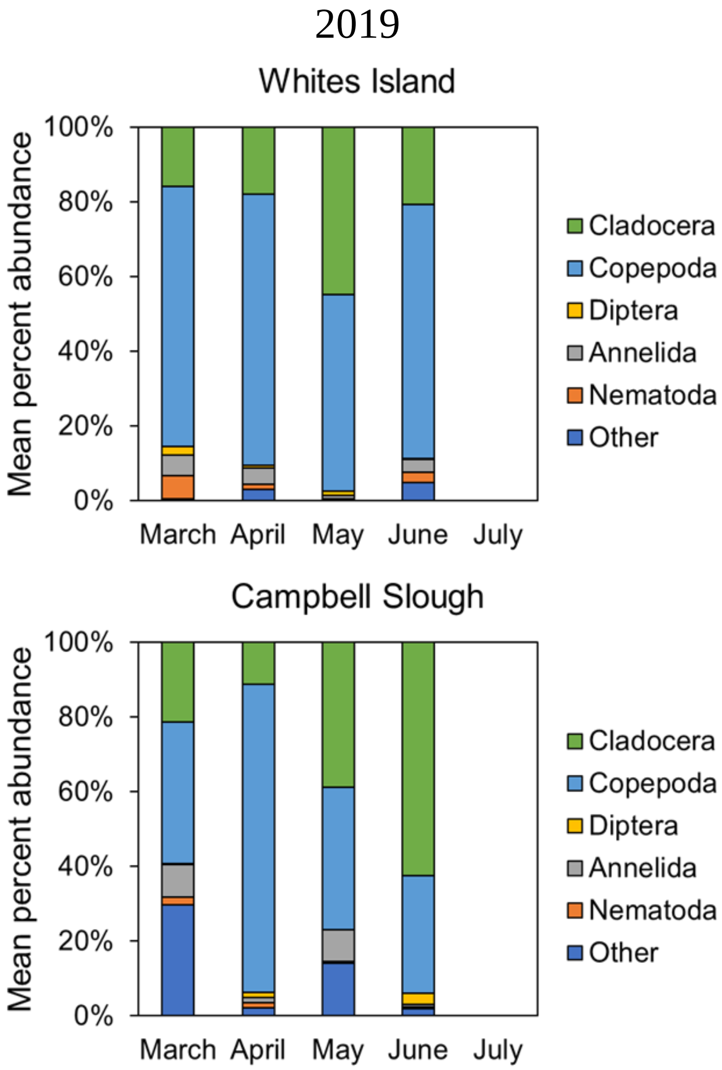


Figure 87. Mean percent abundance of different zooplankton taxonomic groups at two of the trends sites in 2019.

In March, the zooplankton assemblages at both Campbell Slough and Franz Lake Slough were dominated by copepod nauplii and rotifers, particularly *Keratella* spp. and *Brachionus* spp. Aside from nauplii, copepods and cladocerans were present at similar densities (Figure 88), with the former dominated by cyclopoid species and the latter dominated by *Bosmina* spp. and *Chydoridae* spp. By early May, rotifers dominated (especially *Asplanchna* spp. and *Notholca squamula*) and copepod assemblages were dominated by cyclopoid copepodites. Cladocerans were dominated by *Chydoridae* spp., which were present at much lower densities than the copepods or rotifers. Other taxa, including unidentified eggs and ostracods accounted for ~25% of zooplankton assemblages at Campbell Slough in March.

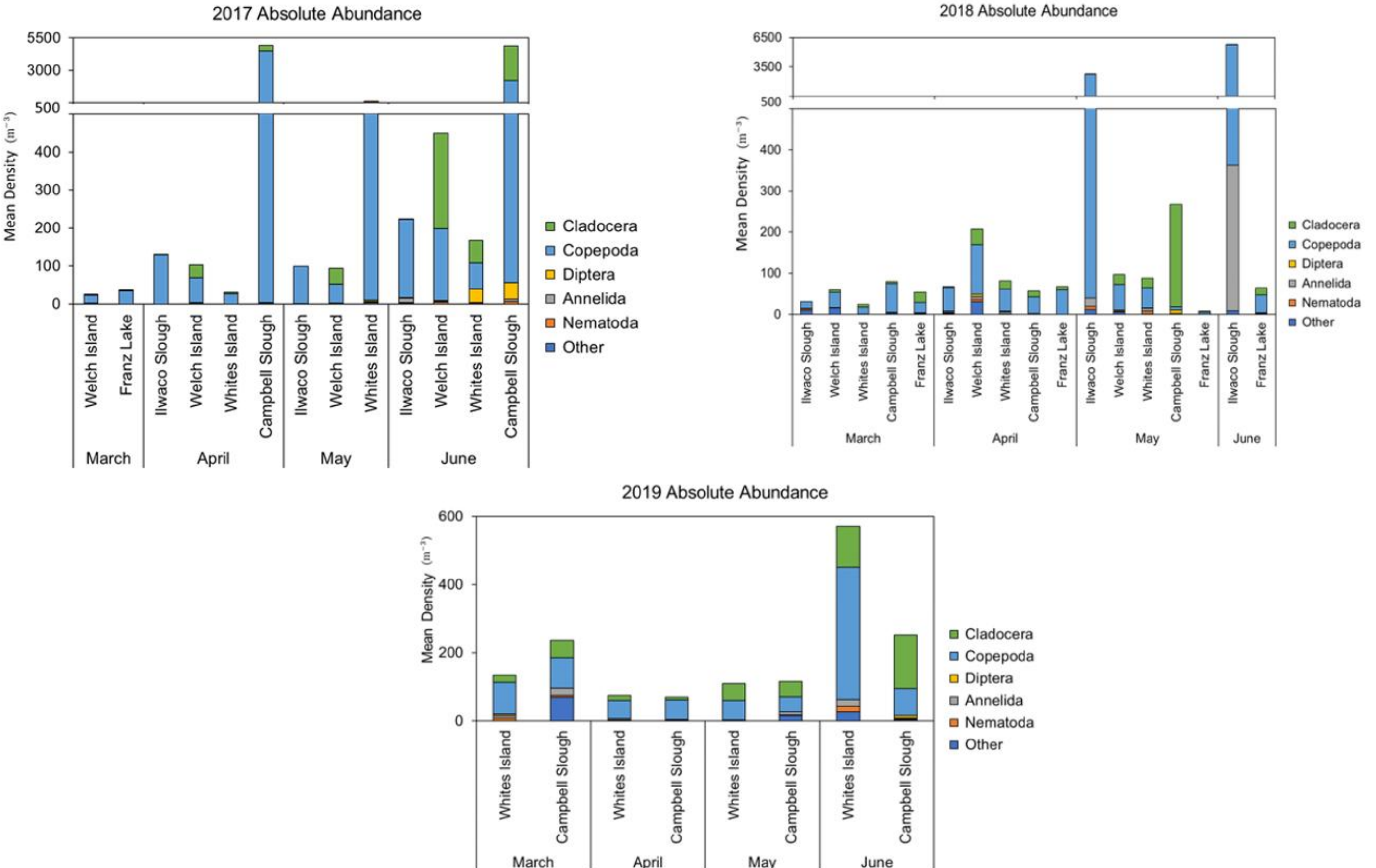


Figure 88. Mean densities (cells/m^3) of different zooplankton taxonomic groups at trends sites in 2017, 2018, and 2019.

3.4.3 Stable Isotope Ratios of Carbon and Nitrogen

Carbon isotopes can be used to determine the source of organic matter to a consumer, while nitrogen isotopes can be used to determine the trophic level of a consumer. Most terrestrial plants have δ^{13}C values between -24 and -34‰, seaweeds, and marine plants between -6 and -19‰, and algae and lichens -12 to -23‰. According to Cloern (2002), δ^{13}C values for freshwater phytoplankton are between -29.5 and -27.5‰, which overlaps with emergent vascular plant matter, which typically has δ^{13}C values ranging from -28.1 to -27.2‰. Sediments and soils tend to have heavier isotopic signatures, while runoff can have lighter values. In 2019, Isotopic values of carbon in particulate organic matter (δ^{13}C-POM) collected onto filters revealed δ^{13}C signatures in the range of freshwater phytoplankton most of the time, with values closer to terrestrial vascular plants in May and June at Campbell Slough and Franz Lake Slough. δ^{13}C-POM at Ilwaco was closer to marine values. It is interesting that the signatures of δ^{13}C-POM would be close that that of vascular plants, since POM was collected by filtering whole water onto glass fiber filters and much of the material collected onto the filters was composed of phytoplankton (the same collection method is used to determine chlorophyll *a* concentrations).

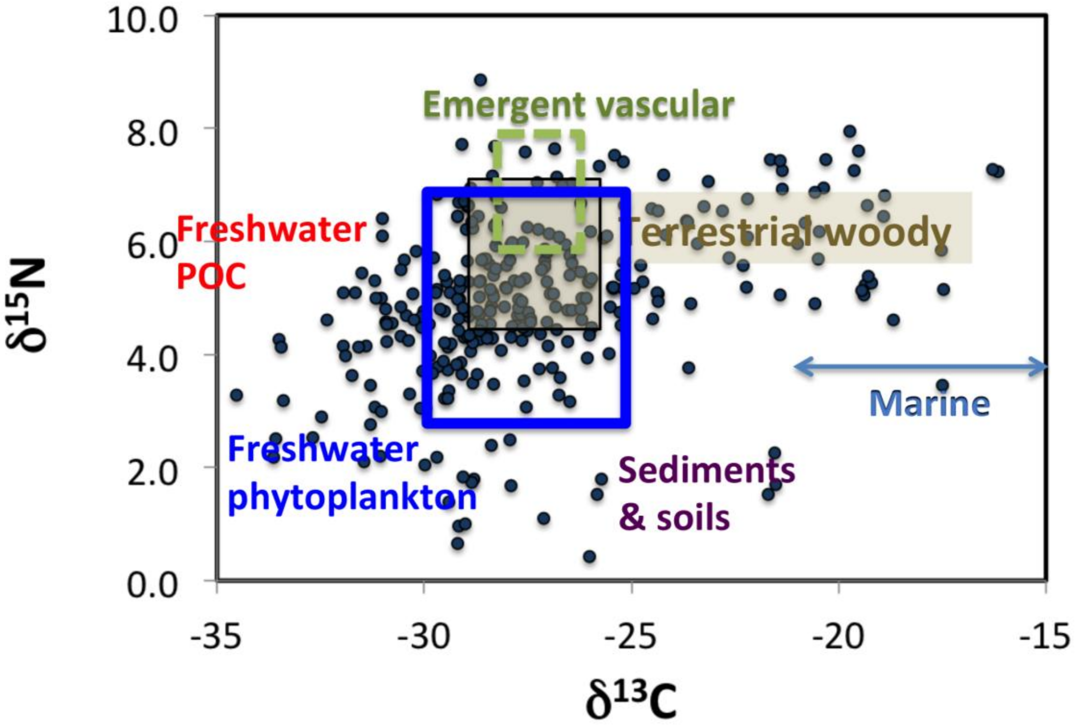


Figure 89. Plot of particulate organic matter (POM) data from off-channel trends sites (circles) in isospace; typical isotopic signature ranges for different organic matter sources are shown (derived from Cloern, 2002). The contribution of various sources to measured POM in the lower Columbia is evident in the data spread.

When all the data from particulate organic matter are placed in context with typical values for different sources (e.g., sediments, marine phytoplankton, woody debris), the spread in isotopic signatures (Figure 89) suggests that there are several sources contributing to POM in the lower Columbia, with several samples resembling freshwater phytoplankton, while others appeared to include some woody debris or soil organic compounds. There is considerable overlap in the stable isotope signatures of emergent vascular plants, freshwater phytoplankton, and terrestrial woody debris, with a narrower range for emergent vascular plant material.

3.4.3.1 *Stable Isotope Ratios of Carbon and Nitrogen of Primary Producers*

All primary producers

When stable isotope signatures for carbon and nitrogen associated with all primary producers is combined, two broad patterns emerge (Figure 90). The average $\delta^{13}C$ (ratio of ^{13}C to ^{12}C) is slightly higher in very dry years (for example, 2015) as well as very wet years (for example, 2017), and lower for more moderate years. In the case of nitrogen, this effect is more pronounced with bi-modal features during moderate, dry, and very dry years.

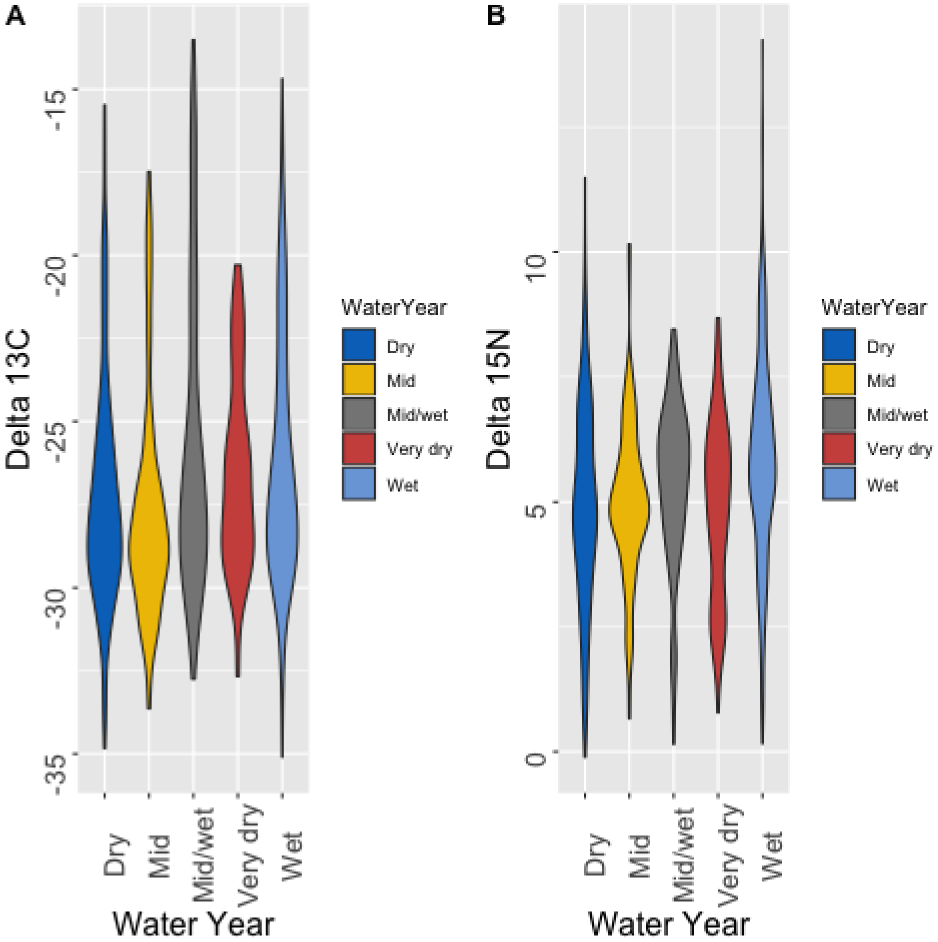


Figure 90. Plots showing stable isotope signatures of all primary producers according to the type of water year (Dry, moderate (“mid”), wet-to-moderate (“Mid/wet)”, Very dry, and Wet). The data spread for $^{13}C/^{12}C$ was greatest for moderately wet, wet, and dry years, while for $^{15}N/^{14}N$ was greatest for wet and dry years.

Pelagic primary producers: Particulate Organic Matter (POM)

The data suggest that heavier carbon isotope signatures in particulate organic matter (POM) were associated with dry years, while wet years were associated with lighter carbon isotope values in POM (Figure 91). In very dry years, like 2015, the average $\delta^{13}C$ was -25.6‰, which was significantly heavier than in dry, moderately wet, and wet years ($p < 0.05$). Wet and moderately wet years did not differ in the carbon isotopic signature of POM ($p = 0.99$). In contrast, there were no significant differences in the stable isotope signatures of nitrogen ($^{15}N/^{14}N$) in POM, with the exception of the difference between very dry (4.75‰) and dry (4.14‰) years ($p = 0.05$).

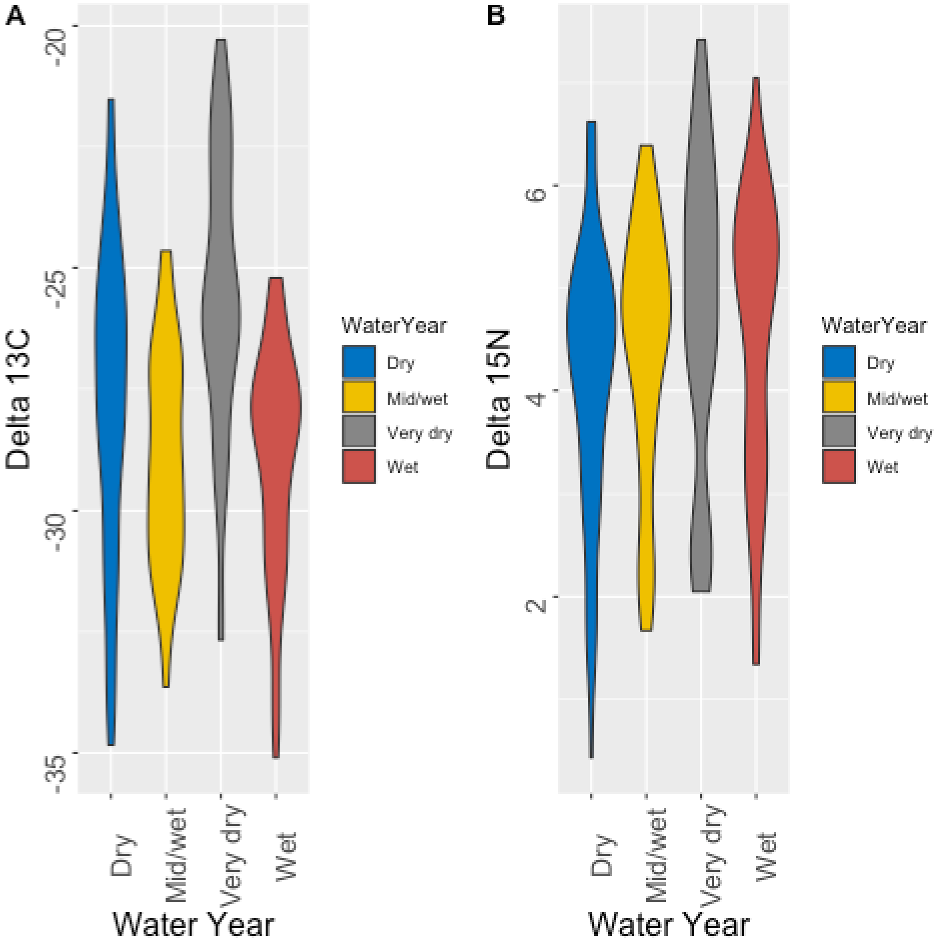


Figure 91. Plots showing average stable isotope signatures ($^{13}C/^{12}C$, or Delta 13C, and $^{15}N/^{14}N$, or Delta ^{15}N) for particulate organic matter (POM) in different water years. (Dry years, n=63; very dry years, n = 63; wet years, n = 51; moderately wet years, n=77).

Since there were more data points from samples collected at Campbell Slough and Whites, the average values of delta C and delta N for POM over the 2011-2019 time series were compared (Figure 92). The carbon data showed a wider spread at Campbell Slough compared to Whites Island, although the mode values were similar. Similarly, delta N values showed a wider spread at Campbell Slough compared to Whites Island with similar mode values at the two sites.

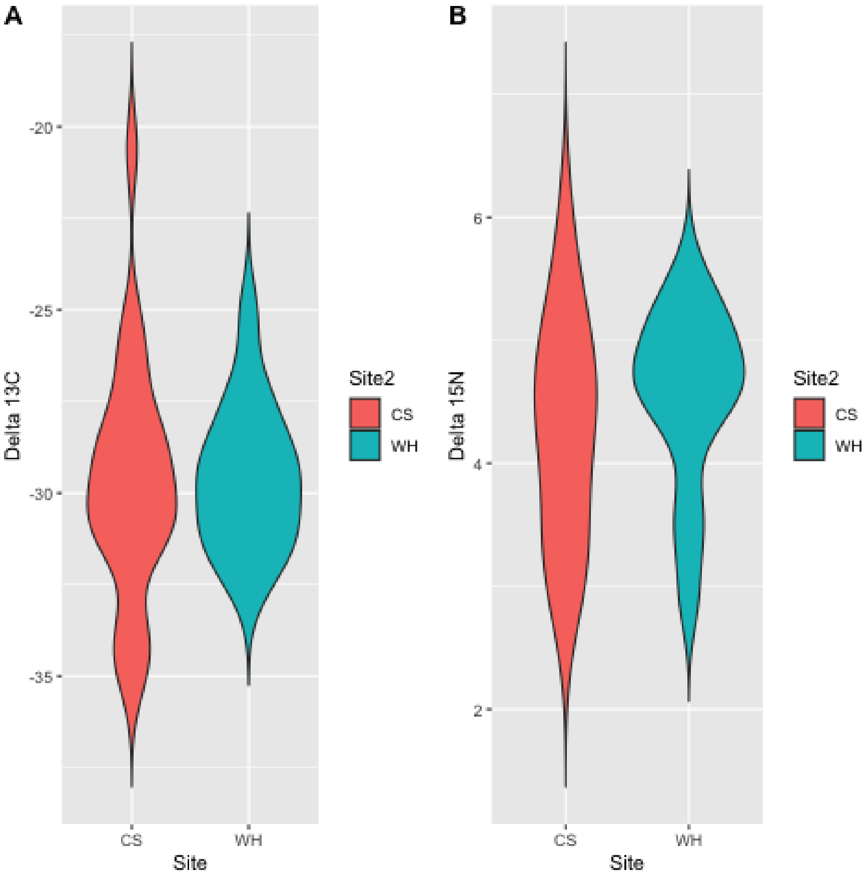


Figure 92. Violin boxplots showing the δ^{13}C and δ^{15}N signatures of particulate organic matter at Campbell Slough and Whites Island. The data included samples collected between 2011-2019.

Stable isotope signatures of periphyton

The stable isotope signatures of periphyton collected across the trends sites between 2011 and 2019 varied widely across the data set. When the samples were divided to compare wet vs. dry years, there were a few notable observations. Average values of δ^{13}C were higher during moderately wet and wet years (Figure 93), although there was considerable spread in the data. There was an increase in the average δ^{15}N over a gradient of dry to wet years, with the largest spread in data observed for wet years.

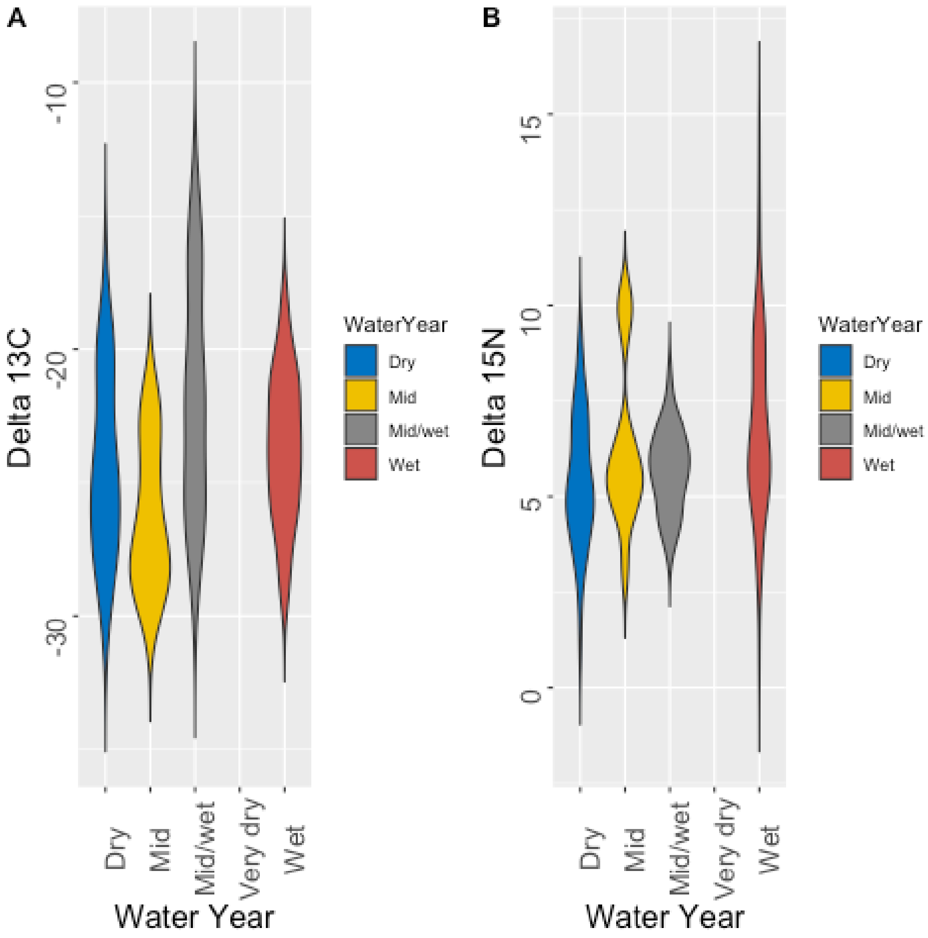


Figure 93. Plots showing stable isotope signatures (^{13}C/^{12}C, or δ13C, and ^{15}N/^{14}N, or δ15N`) associated with periphyton in different water years. The data from all sites were pooled.

Stable isotope signatures of vegetation

Comparison among sites

When the stable isotope signatures of carbon associated with vegetation (Figure 94) were compared among the sites five EMP sites according to a one-way analysis of variance, values from Ilwaco were different from all other sites ($p<0.01$). Similarly, δ^{15}N values differed at Ilwaco compared to the other sites. We performed a second set of ANOVAs excluding the data from Ilwaco followed by Tukey post hoc multiple comparisons tests. These revealed that for δ^{13}C, there were significant differences between Franz Lake Slough and Campbell Slough, Whites Island and Campbell Slough, and between Welch

Island and Franz Lake Slough, as well as Welch Island and Whites Island. Whites Island and Franz Lake Slough also differed significantly. Interestingly, Welch Island and Campbell Slough were not different (p=0.90).

In the case of $\delta^{15}N$, signatures at Campbell Slough and Franz Lake Slough were similar while $\delta^{15}N$ signatures at Welch Island and Whites Island were similar. The two groups differed significantly from each other.

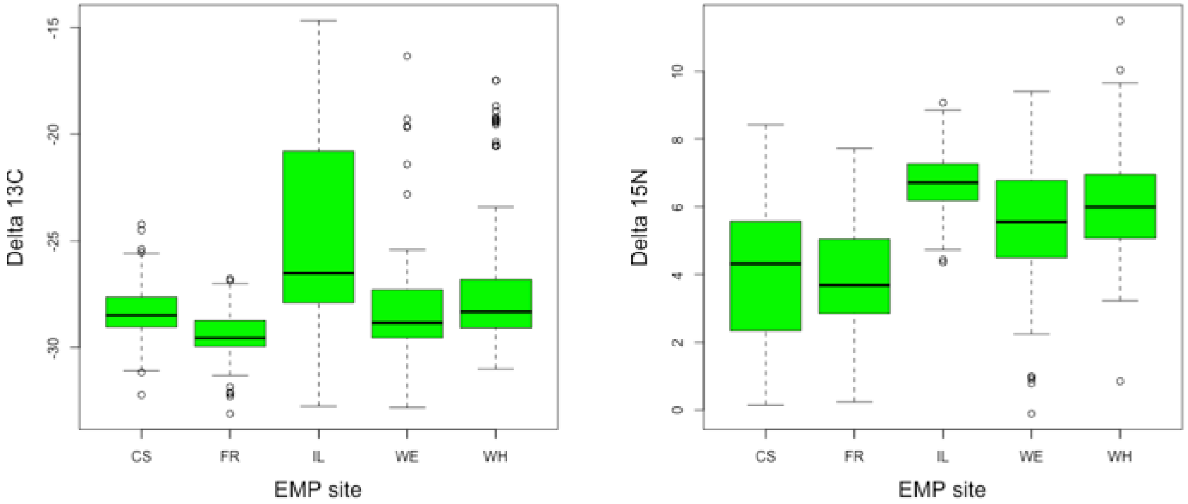


Figure 94. Boxplots showing the distribution of data for stable isotope of carbon ($\delta^{13}C$) and nitrogen ($\delta\ ^{15}N$) from vegetation at EMP sites, CS = Campbell Slough, FR = Franz Lake Slough, IL = Ilwaco, WE = Welch Island, WH = Whites Island.

Comparison among years

When the samples were grouped according to whether they came from years with low, moderate, or high cumulative discharge (very dry, dry, moderate, wet), there were significant differences in average $\delta^{13}C$, but not in $\delta^{15}N$ (Figure 95). The only non-significant differences among year types was between moderately wet and wet years.

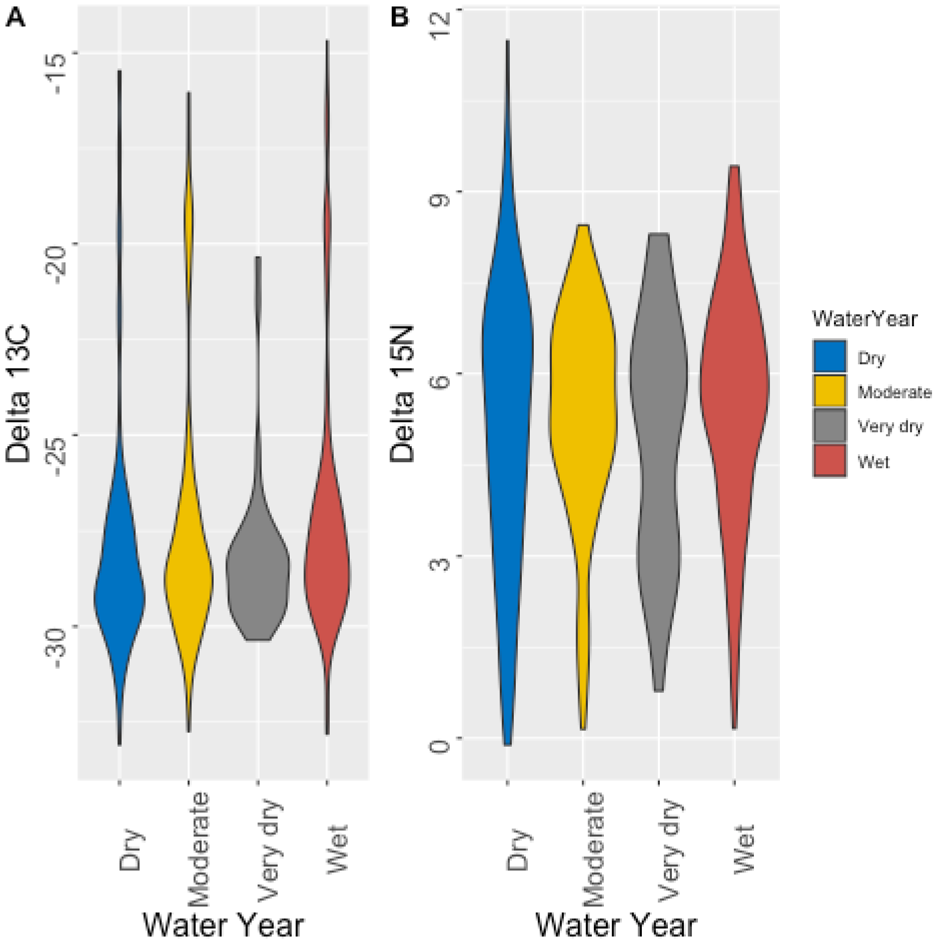


Figure 95. Plots showing average stable isotope signatures ($^{13}C/^{12}C$, or $\delta^{13}C$, and $^{15}N/^{14}N$, or $\delta^{15}N$`) for vegetation in different water years. Data from all sites were pooled.

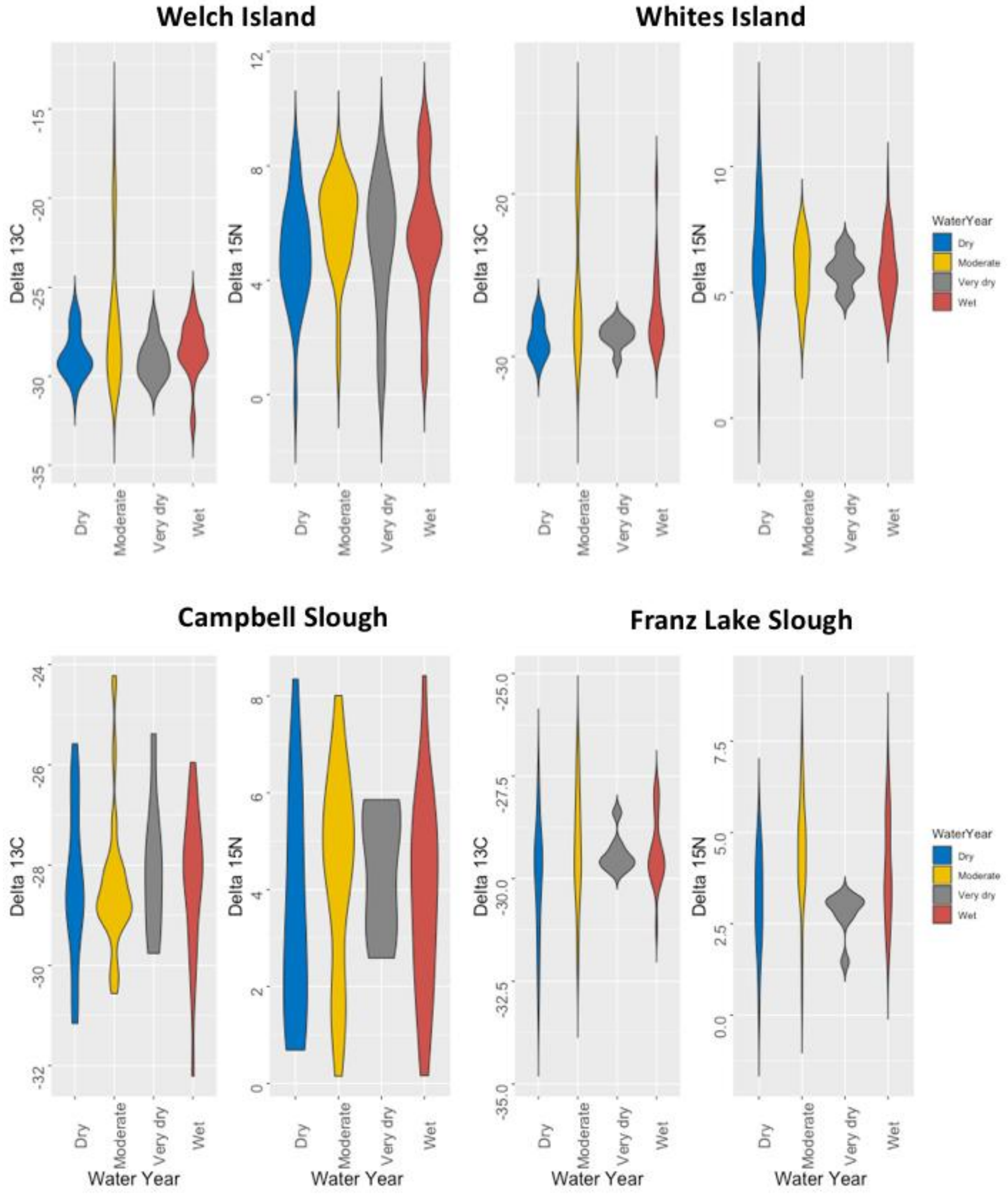


Figure 96. Stable isotope ratios of carbon and nitrogen for vegetation tissues at four of the EMP sites: Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough.

3.4.3.2 *Stable Isotope Ratios Associated with Potential Salmon Prey*

According to direct observations from fish stomach contents, juvenile salmonids in the lower Columbia appear to rely primarily on chironomids and amphipods in their diets. However, the energy ration associated with other taxa, particularly the large, terrestrial insects, can be higher and might contribute to growth of juvenile salmonid, so we explored how the isotopic signatures of potential prey items varied with habitat. For this analysis, prey were divided into four different habitat types: benthic, pelagic, terrestrial, or ‘other’ (i.e., uncertain). Pelagic prey had the largest spread in $\delta^{13}C$ values, while the spread of $\delta^{15}N$ values was similar for all habitat types. The mode of $\delta^{15}N$ values for terrestrial prey was lighter than that for benthic or pelagic prey (Figure 97).

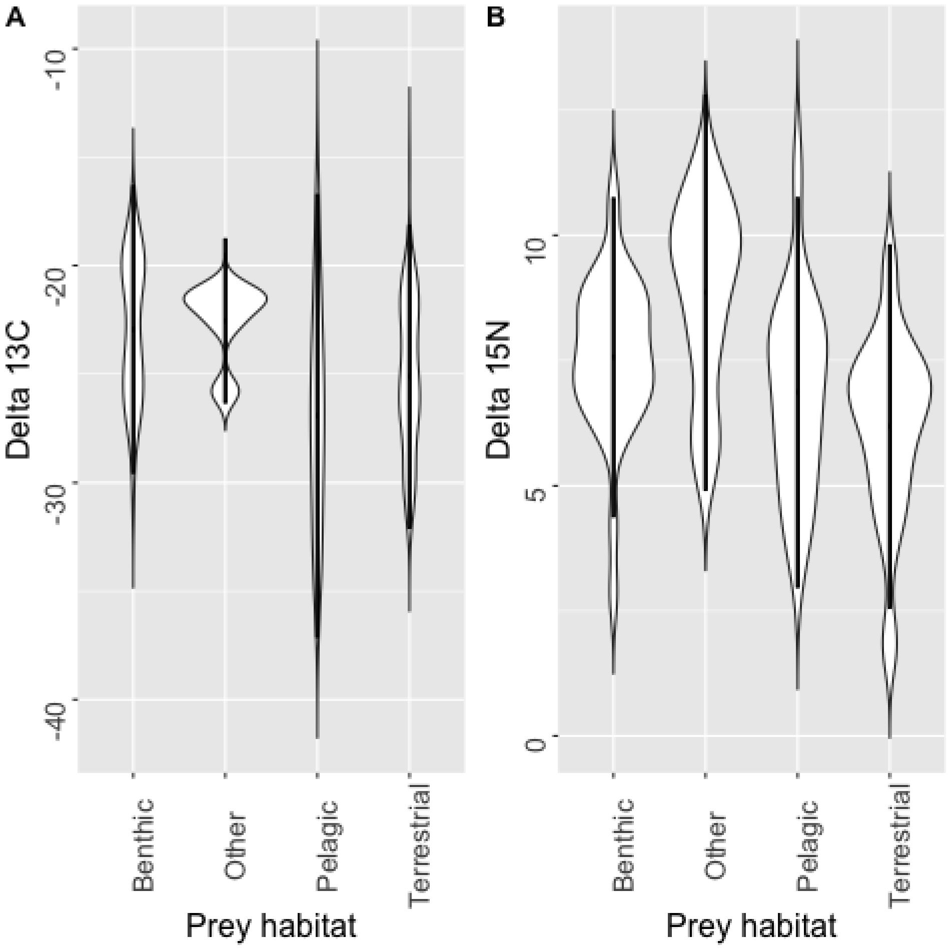


Figure 97. Stable isotope ratios of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) for prey sources divided according to their typical habitat (benthic, pelagic, terrestrial, or ‘other’ [unknown, or mixed]). The graphs show the data distribution over the range of observed values, with narrow shapes indicating few data points per observed value (relative to the total number of observations), and wider shapes indicating more data points per observed value. For benthic prey, n=60; for pelagic prey (copepods and cladocerans), n = 16; for terrestrial prey, n =100; for ‘Other’, n = 5.

We were also interested in investigating how the stable isotope signatures of organic matter (i.e., primary producers) related to those of the two primary salmonid prey taxa (chironomids and amphipods). When the δ^{13}C and δ^{15}N values of vascular plants primary producers (VegA and VegB; Veg A groups those plant tissues with heavier δ^{13}C and δ^{15}N, while Veg B includes those having light δ^{13}C and δ^{15}N), particulate organic matter (a proxy for pelagic phytoplankton), and periphyton were compared with salmon prey (chironomids), there was overlap with periphyton, POM, and the VegA group (Figure 98). Some of the chironomid tissues, however, were lighter in terms of nitrogen isotopes than any of the organic matter sources (i.e., <5‰) and one was lighter than any of the carbon isotope values observed in organic matter sources (ie., <-29‰).

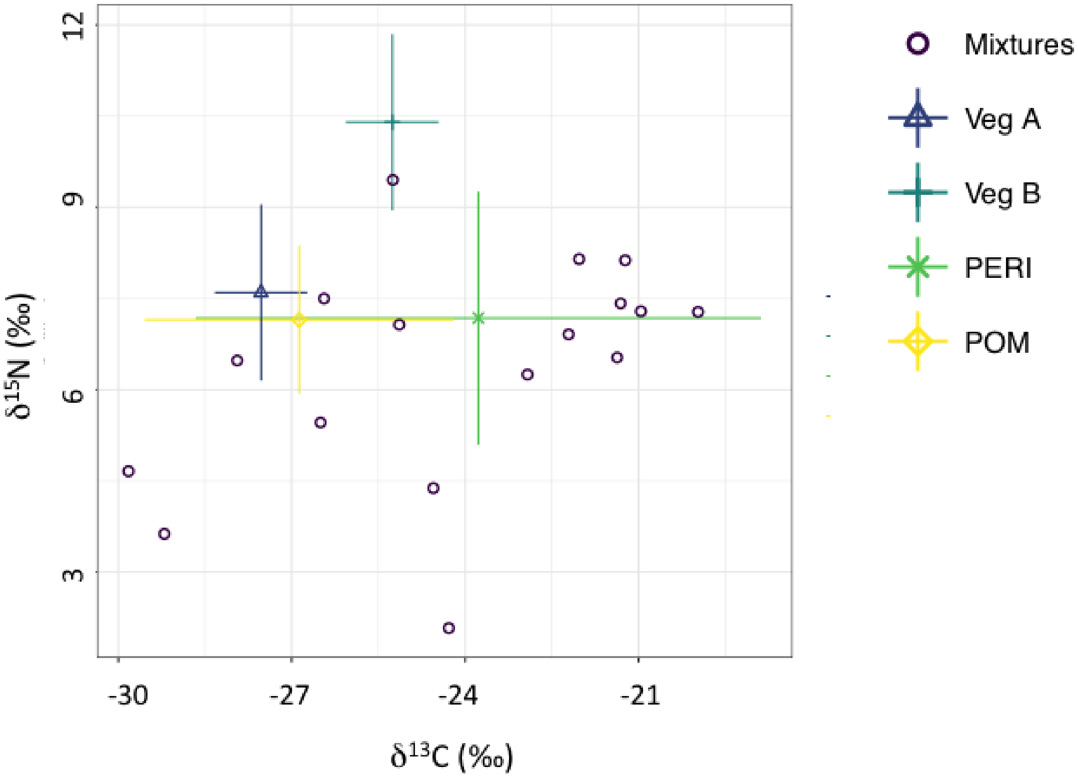


Figure 98. Isospace plot showing stable isotope signature of chironomids ("mixtures") compared to vascular plant matter with heavier δ^{13}C and δ^{15}N (Veg A) and those having light δ^{13}C and δ^{15}N (Veg B) as well as to periphyton (PERI) and particulate organic matter (POM).

3.4.3.3 *Stable Isotope Ratios Associated with Salmon*

Among salmonid muscle tissues, there was a positive relationship between δ^{13}C and δ^{15}N (Figure 99). Some of the observed values associated with salmon were heavier in both C and N than were the prey items, which suggests that not all prey sources were accounted for.

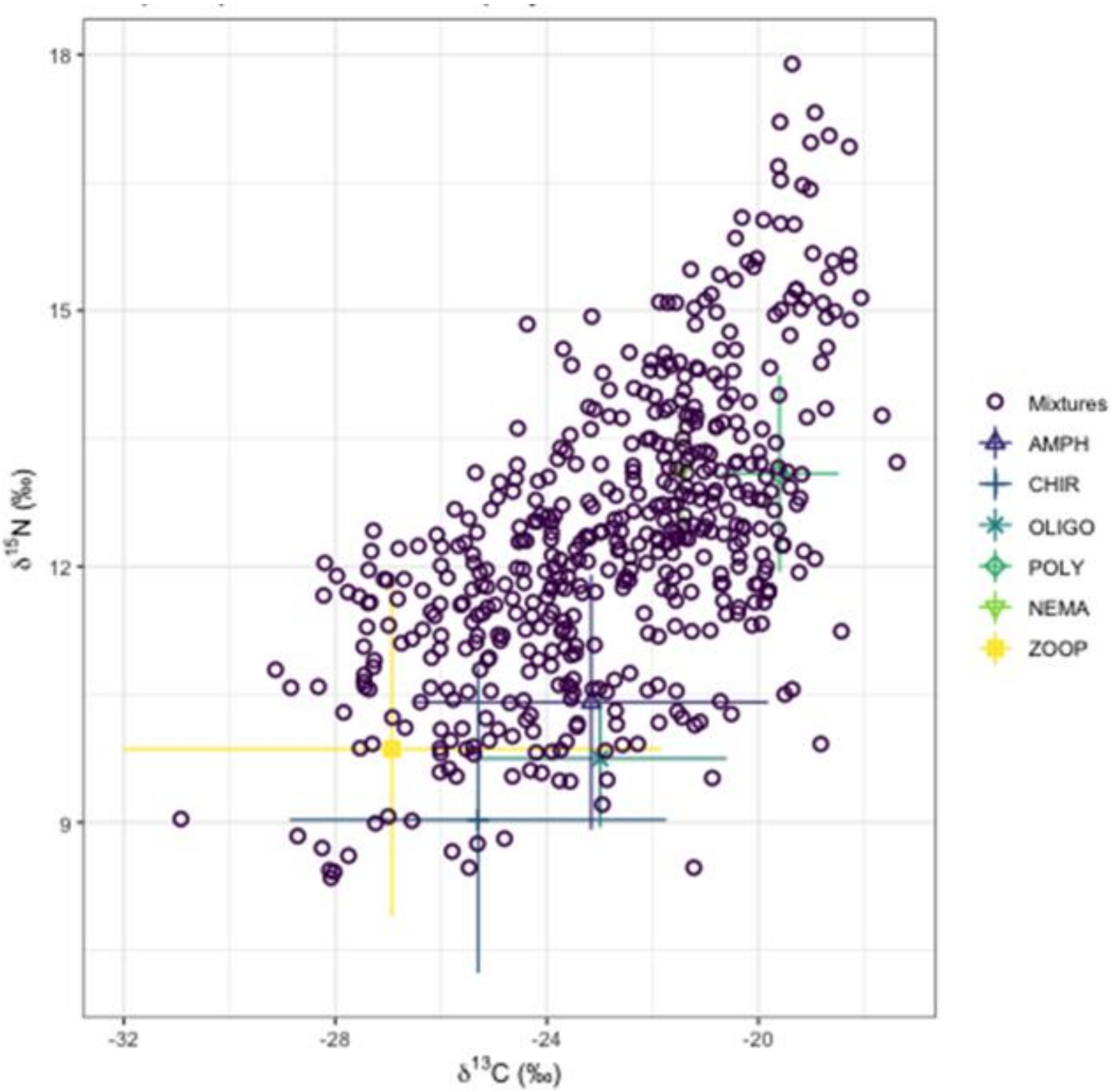


Figure 99. Isospace plot showing isotopic signatures for salmon tissue ("Mixtures") as well as prey sources (AMPH = amphiphods, CHIR = chironomids, OLIGO = oligochaetes, POLY = polychaetes, NEMA = nematodes, and ZOOP = copepods and cladocerans).

When isotope signatures from juvenile salmonid tissues were divided according to water year (i.e., wet vs. dry), the average $^{13}C/^{12}C$ and $^{15}N/^{14}N$ values were heaviest in years of moderate flow (Figure 100), with wide ranges around the average in all years observed. The lowest $\delta^{13}C$ values were observed during wet years; interestingly, there was a biomodal pattern in the $\delta^{13}C$ value during the wet years, with fewer data points occurring between lower and higher values.

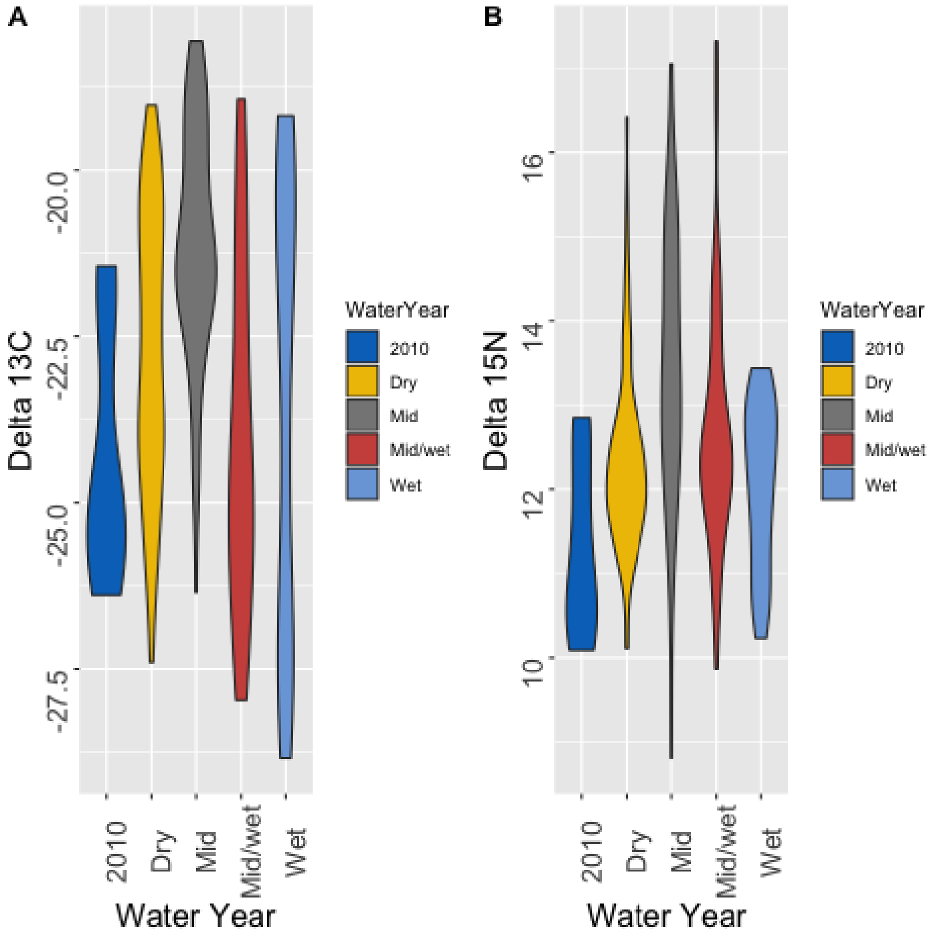


Figure 100. Isotope ratios (delta 13C and delta 15N) of juvenile salmonid tissues pooled according to years categorized by variations in river flow.

When the isotopic signatures were grouped according to site, the largest differences in $\delta^{13}C$ and $\delta^{15}N$ were observed between Franz Lake Slough and all the other sites (Figure 99). In addition, the $\delta^{15}N$ values were lower at Campbell Slough than the other sites. This is exemplified in the data from 2014 (Figure 100), a year when data were available from multiple sites, which is not always the case (i.e., in some years and at some sites, fish catches are not large enough to provide samples for isotope analysis of fish tissue). The 2014 data show a wider spread in isotopic values of both carbon and nitrogen, with a lower average than at the other sites.

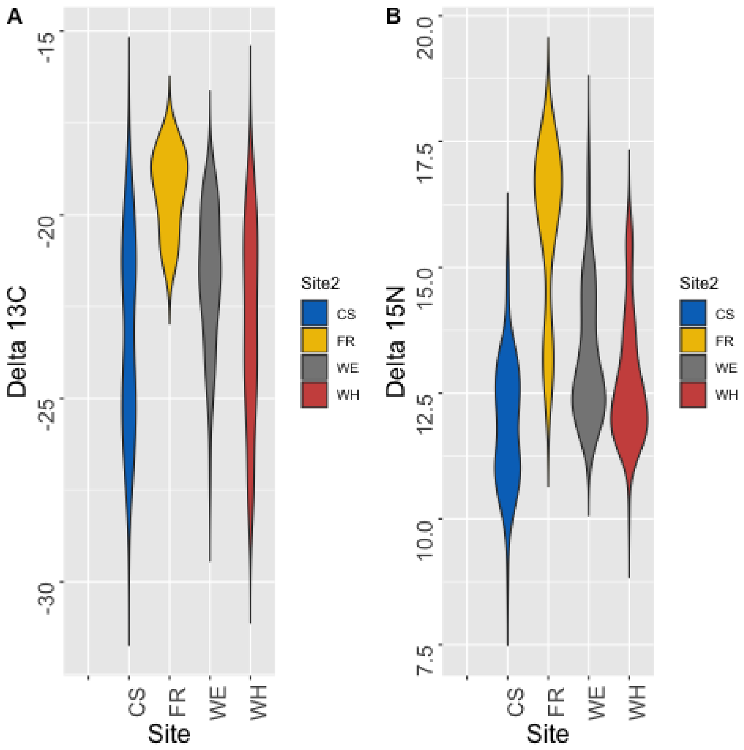


Figure 101. Violin plots showing (A) delta 13C ($^{13}C/^{12}C$) in juvenile salmon tissue collected from Campbell Slough (CS), Franz Lake Slough (FR), Welch Island (WE), and Whites Island (WH).

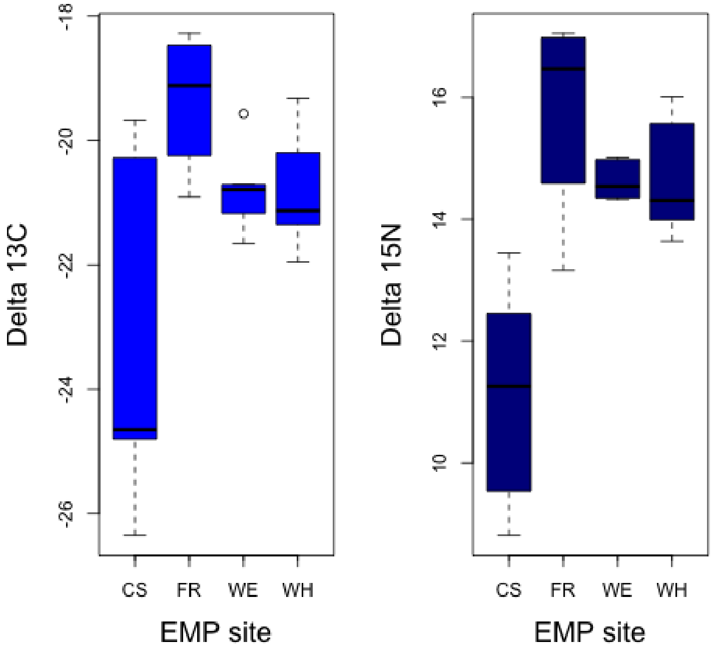


Figure 102. Boxplot showing delta 13C and 15N values for salmon muscle tissue in April 2014 at four EMP sites: CS = Campbell Slough, FR = Franz Lake Slough, WE = Welch Island, and WH = Whites Island. According to a one-way anova with Tukey's HSD post hoc testing, signatures of tissues at Campbell Slough were significantly different than those at the other sites.

3.5 Macroinvertebrates

3.5.1 Salmon Prey Availability

Salmon prey availability data collected in 2019 are still under analysis, the 2018 data is reported below.

3.5.1.1 *Benthic*

The average numeric composition of taxa groups in benthic cores was significantly different between sites, except between Welch Island and Whites Island, and Campbell Slough and Franz Lake (Figure 103).

Oligochaete worms were typically the most numerically abundant taxa group in benthic core samples from all sites, though nematode worms dominated counts at Franz Lake in 2018 (Figure 104**Error! Reference source not found.**). Nematode worms and dipterans were also numerically abundant at Welch Island, Whites Island, Campbell Slough, and Franz Lake, while Ilwaco Slough had higher numbers of polychaete worms and amphipods relative to other sites. While not always numerically abundant, the large body size of amphipods and bivalves made a relatively large contribution to the gravimetric composition when they were present (Figure 104**Error! Reference source not found.**)

Amphipods were relatively abundant at Ilwaco Slough (Figure 105). Average amphipod abundance declined upstream from Ilwaco Slough, with low densities at Welch Island and White Island (Figure 105) and almost none at Campbell Slough and Franz Lake (one individual recorded, Figure 106). Dipterans, including Chironomidae, were collected from all sites between April and July (when sampled), except in July 2015 and April 2016 at Campbell Slough and July 2017 and July 2018 at Franz Lake (Figure 107, Figure 108). Highest average dipteran densities were observed at Ilwaco Slough in April 2017 and Welch Island in May 2016, though occurrence in benthic cores was highly variable. The average density of dipterans tended to be lowest at Franz Lake.

Hydrologic condition varied across the years that benthic cores were collected; however, average abundance of benthic taxa did not correlate with changing conditions. Figure 109 shows an example from Whites Island and Campbell Slough of the distribution of average monthly benthic densities in an NMDS plot. Within a site the composition of benthic densities was not significantly different between years.

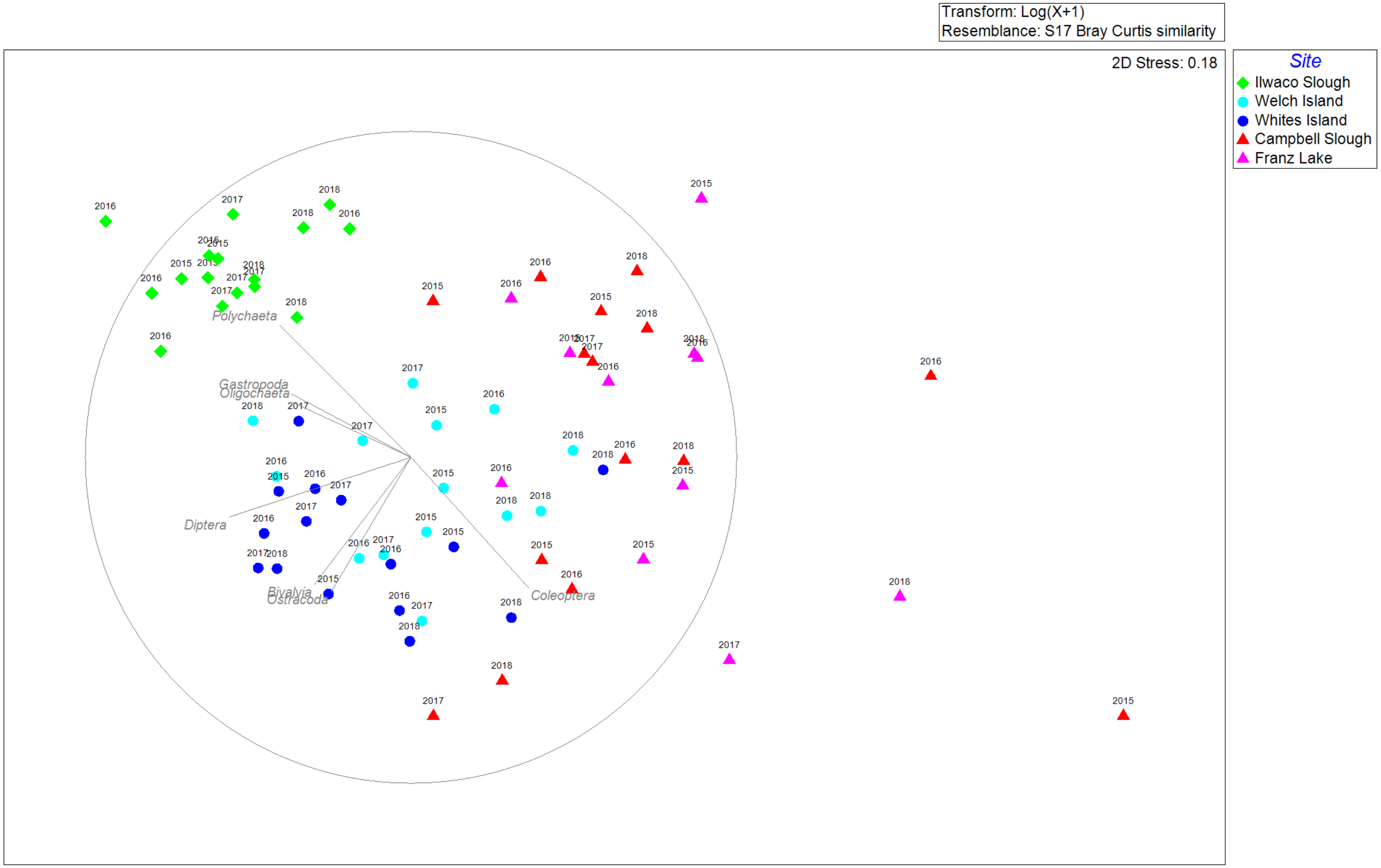


Figure 103. Two-dimensional NMDS plot based on Bray-Curtis similarities between log transformed numeric abundances of taxa collected in benthic cores between 2015 and 2018. Each point represents the composition of the average monthly abundance of taxa collected between April and July within a site and year. Correlation with taxa (Pearson R > 0.4) are represented as gray vectors.

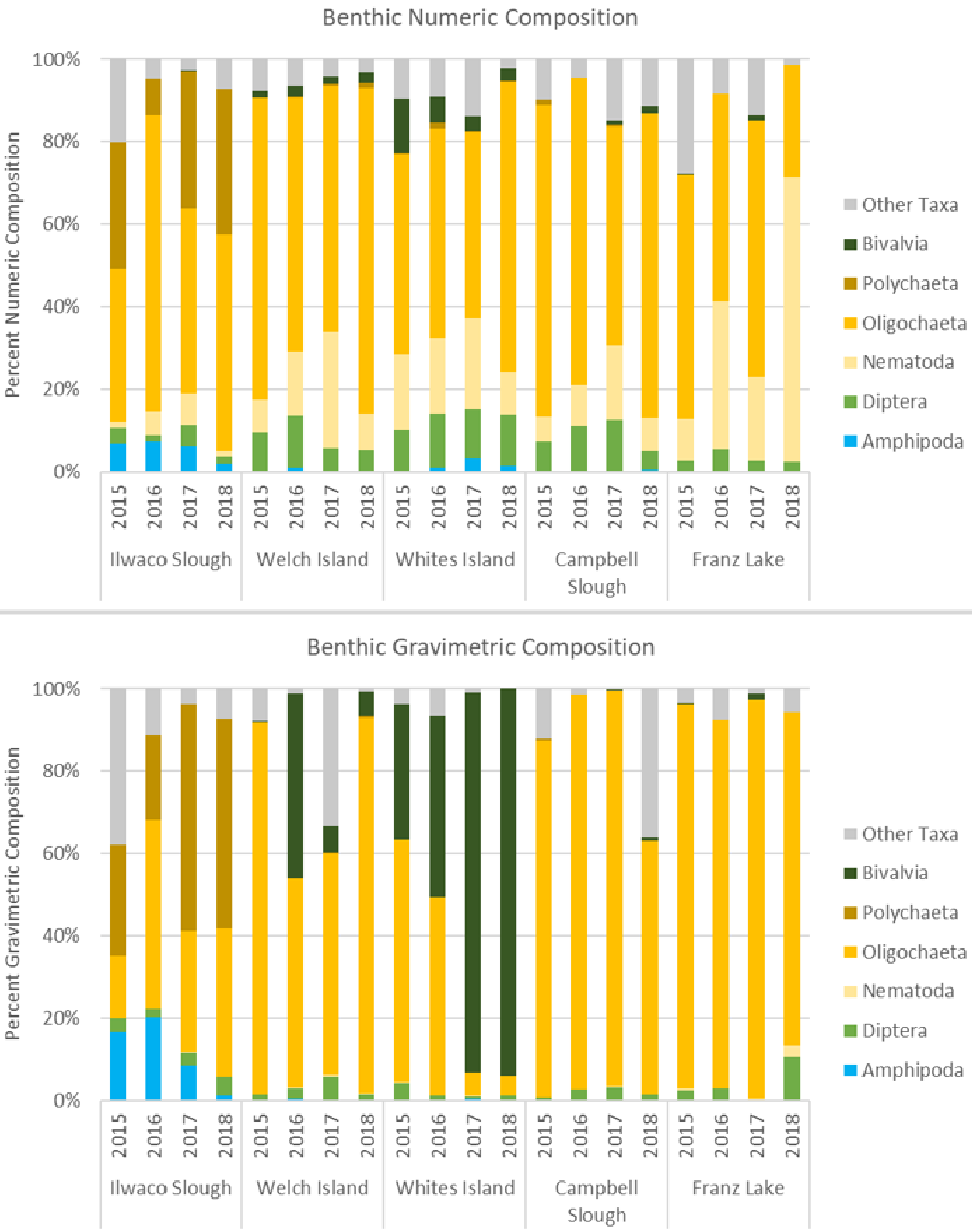


Figure 104. Percent numeric (above) and gravimetric (below) composition of benthic core samples collected between April-July, arranged by site and year.

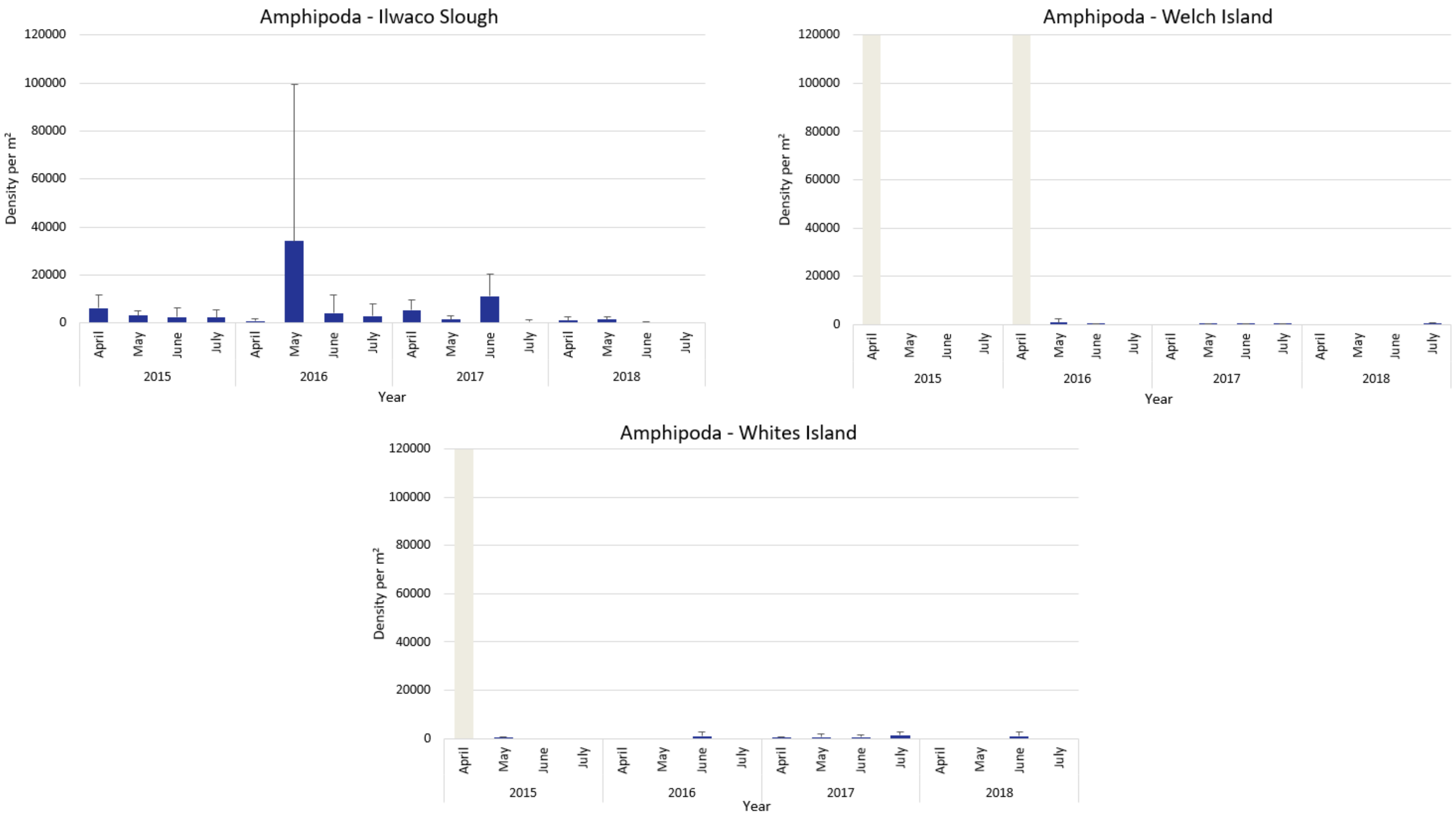


Figure 105. Average density (count per m^2) of Amphipoda collected by benthic core at Ilwaco, Welch and Whites. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected.

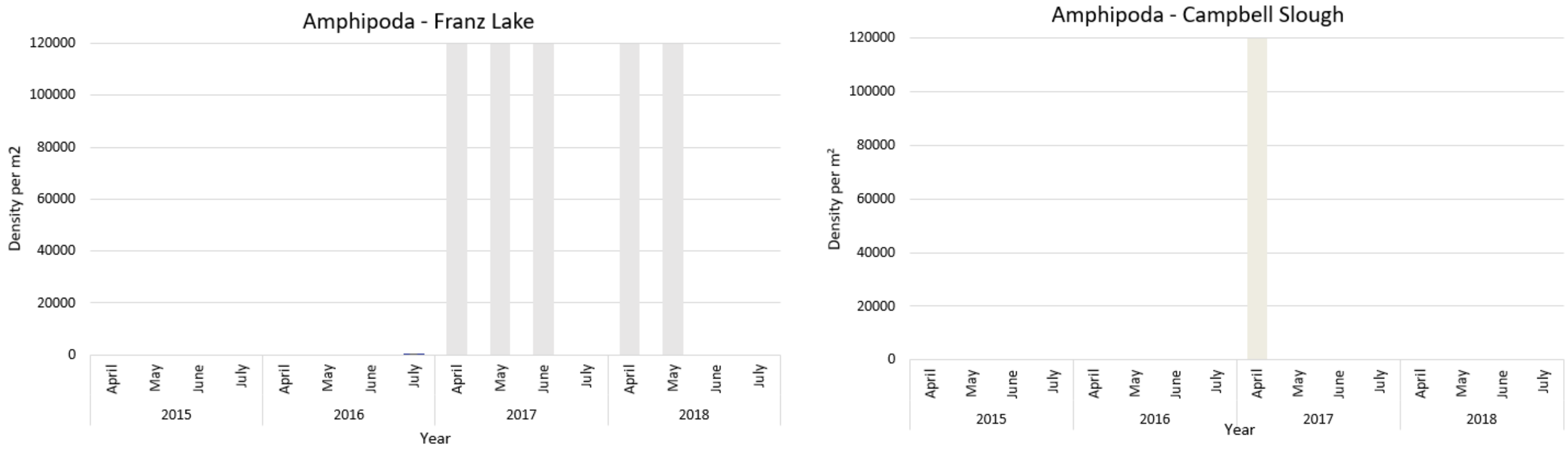


Figure 106: Average Density (count per m^2) of Amphipoda collected by benthic cores at Campbell Slough and Franz Lake. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected.

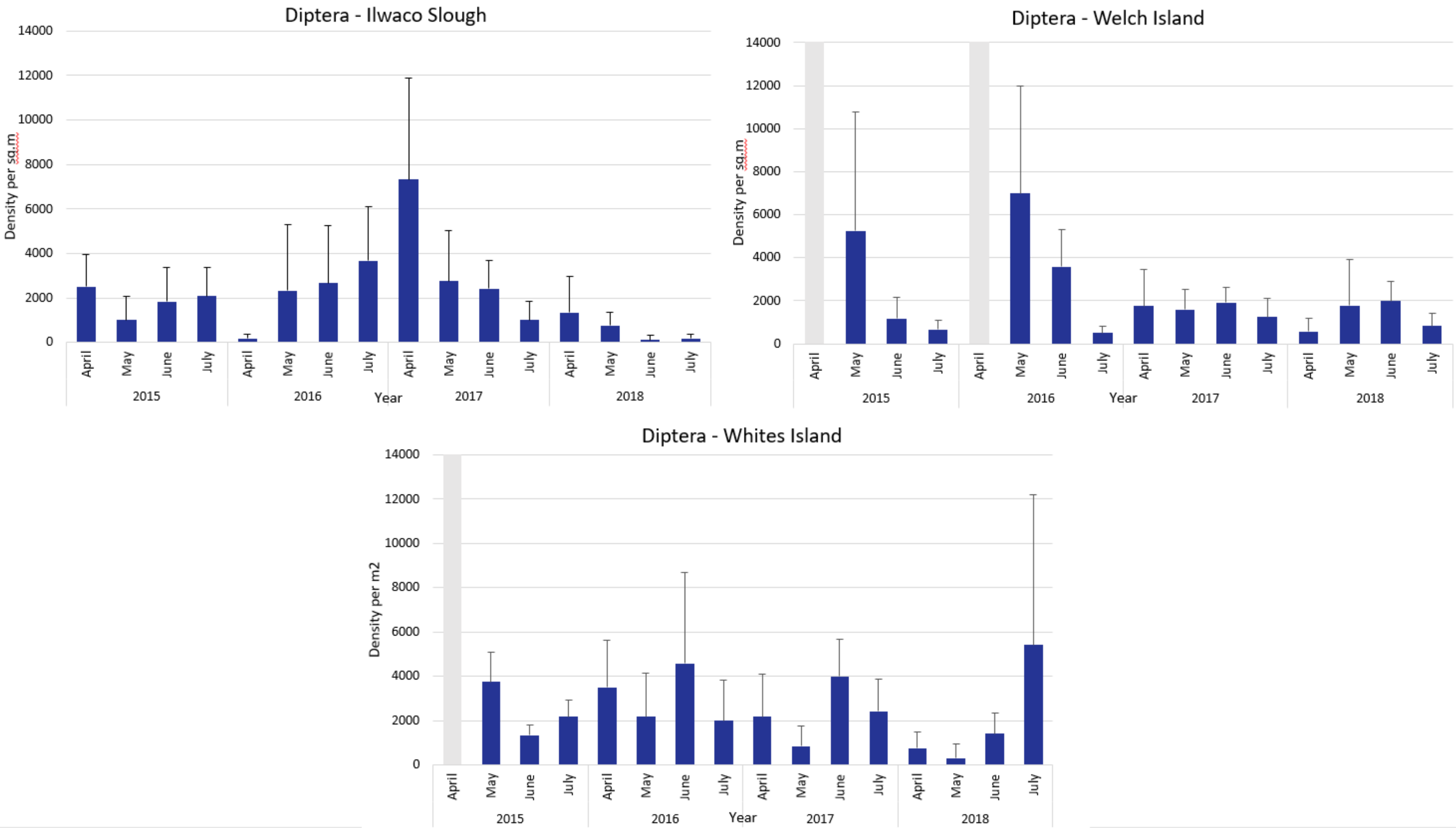


Figure 107. Average density (count per m^2) of Diptera collected by benthic core, arranged by site, year and month. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected

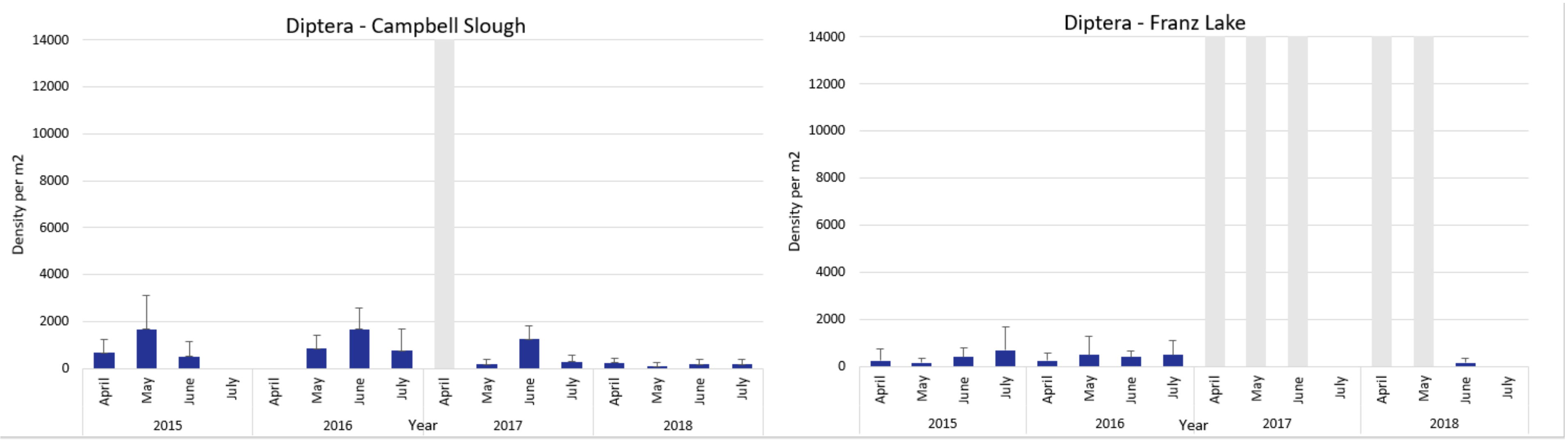


Figure 108 Average density (count per m^2) of Diptera collected by benthic core, arranged by site, year and month. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected.

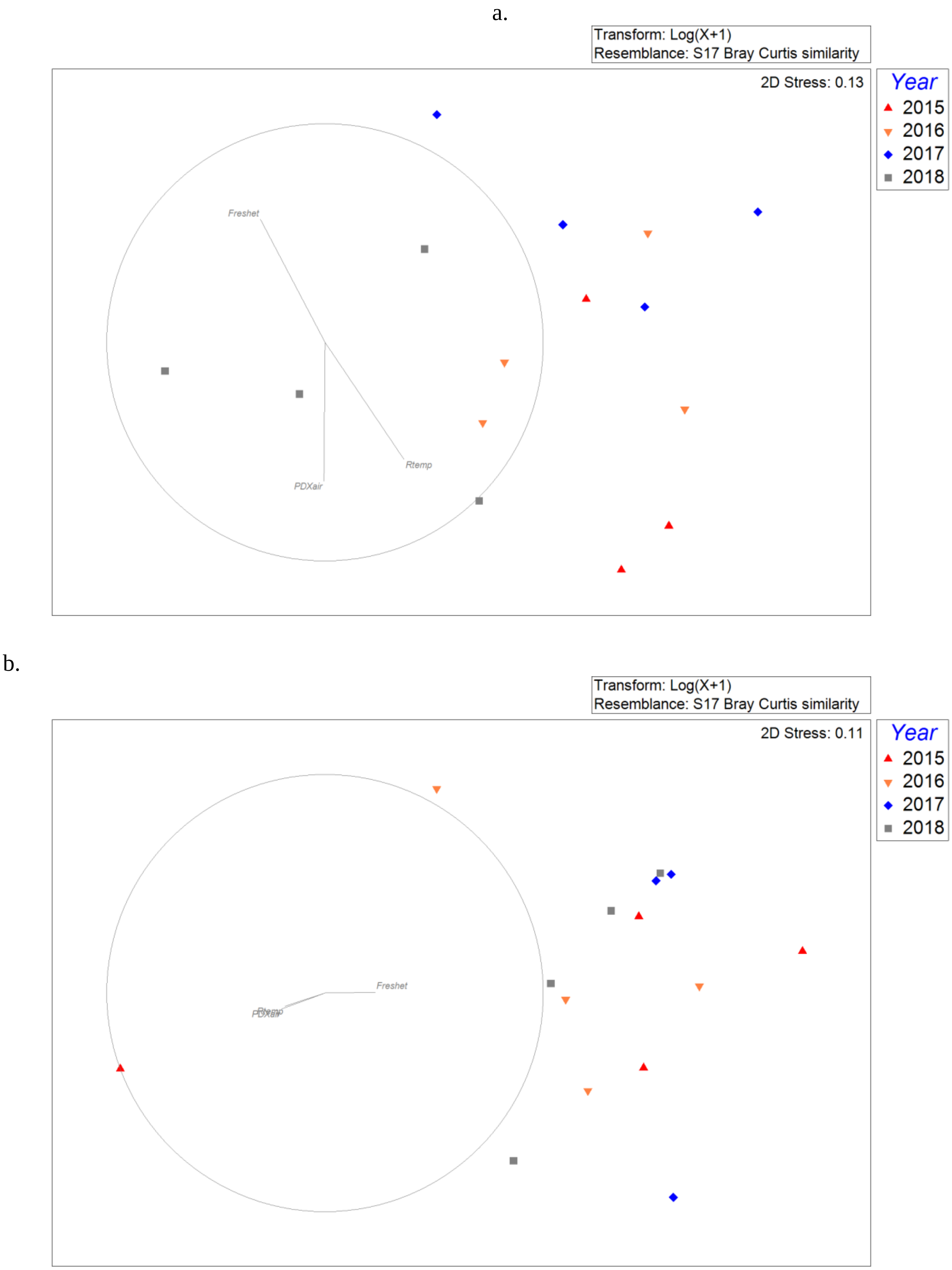


Figure 109. Two-dimensional NMDS plot based on Bray-Curtis similarities of log transformed average monthly density in benthic cores from a) Whites Island and b) Campbell Slough. Correlation with three annual environmental variables (Freshet, PDXair, Rtemp) are represented as gray vectors.

3.5.1.2 *Neuston*

Neuston samples were composed of a diverse array of benthic/epibenthic, terrestrial riparian, and planktonic taxa (Figure 110**Error! Reference source not found.**). Small planktonic taxa, such as copepods and cladocerans, were numerically abundant in both the open water and emergent vegetation but made less contribution to the gravimetric composition. While not always numerically abundant, the large body size of amphipods, bivalves, gastropods, hemipterans, odonates, and isopods made relatively large contributions to the proportional biomass when they were present.

The highest densities of Amphipoda were observed from the emergent vegetation at Welch Island in May 2016 and 2017 (Figure 111**Error! Reference source not found.**). Diptera, including Chironomidae, were collected from the emergent vegetation and open water at all sites sampled between April and July, 2015 through 2018 (Figure 112). The highest dipteran densities were observed at Whites Island, Campbell Slough, and Franz Lake in the emergent vegetation. The average density of dipterans was relatively low at all sites sampled in 2017 and 2018, except for emergent vegetation densities at Campbell Slough in 2017. Counts were variable, however, and we did not observe a clear temporal trend in dipteran abundance.

Cladocera and Copepoda were collected from the emergent vegetation and open water at all sites sampled between April and July, 2015 through 2018 (Figure 113, Figure 114). The density of Cladocera and Copepoda peaked in May 2017 at Campbell Slough from the emergent vegetation. At this time, counts of cladocerans exceeded 1200 individuals on average per meter towed, and counts of copepods were nearly 1800 individuals on average per meter towed. While we have observed other peaks in these taxa (e.g., Cladocera in April 2015 at Campbell Slough, and Copepoda in April 2017 at Campbell Slough and April 2015 at Franz Lake), densities in May 2017 were much greater than previous observations. By June 2017, the average density of both Cladocera and Copepoda at Campbell Slough had returned to typical levels, approximately 35 and 2 individuals per meter towed, respectively.

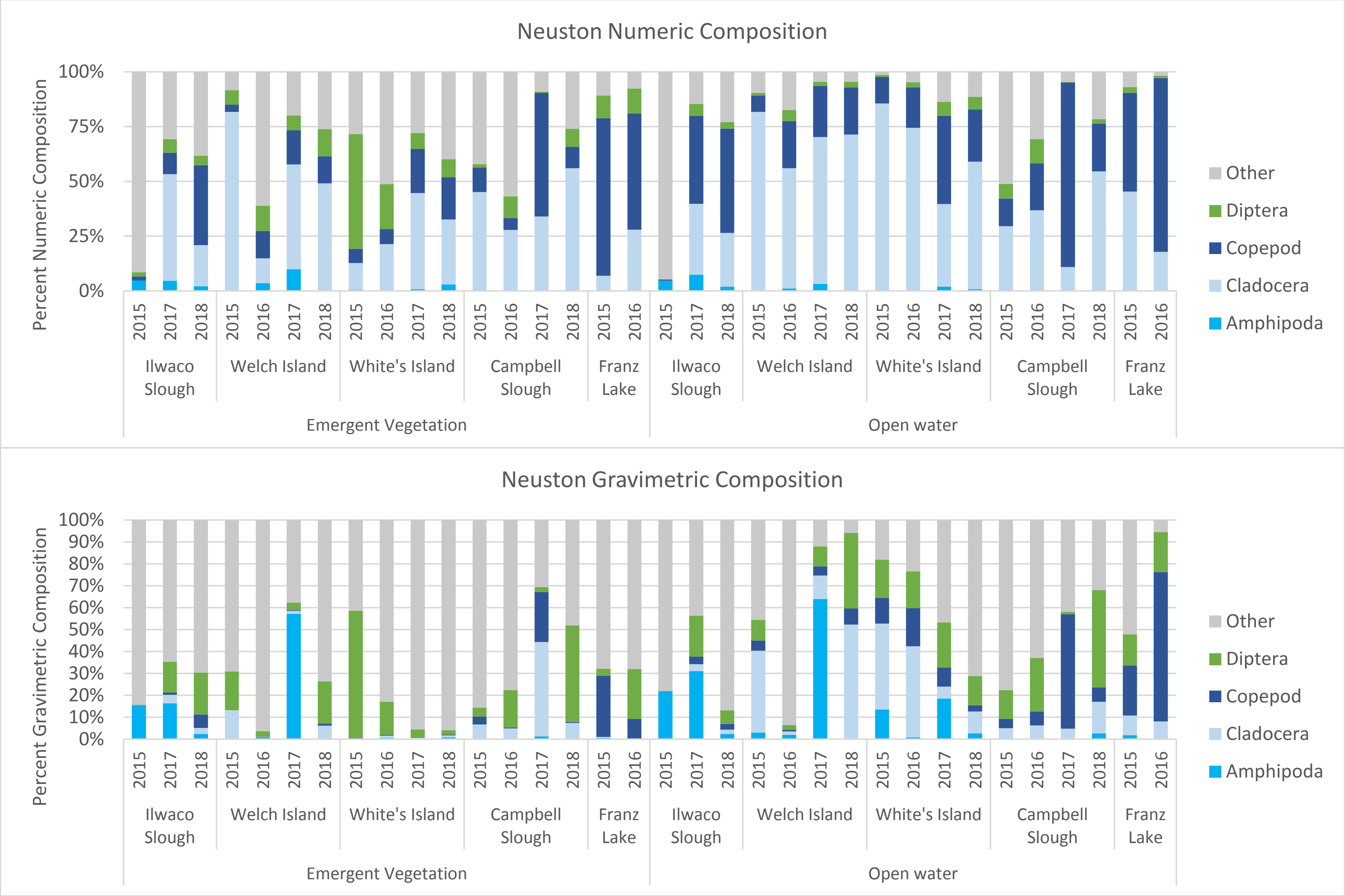


Figure 110. Percent numeric (above) and gravimetric (below) composition of neuston samples collected between April-July, arranged by habitat (emergent vegetation and open water), site and year

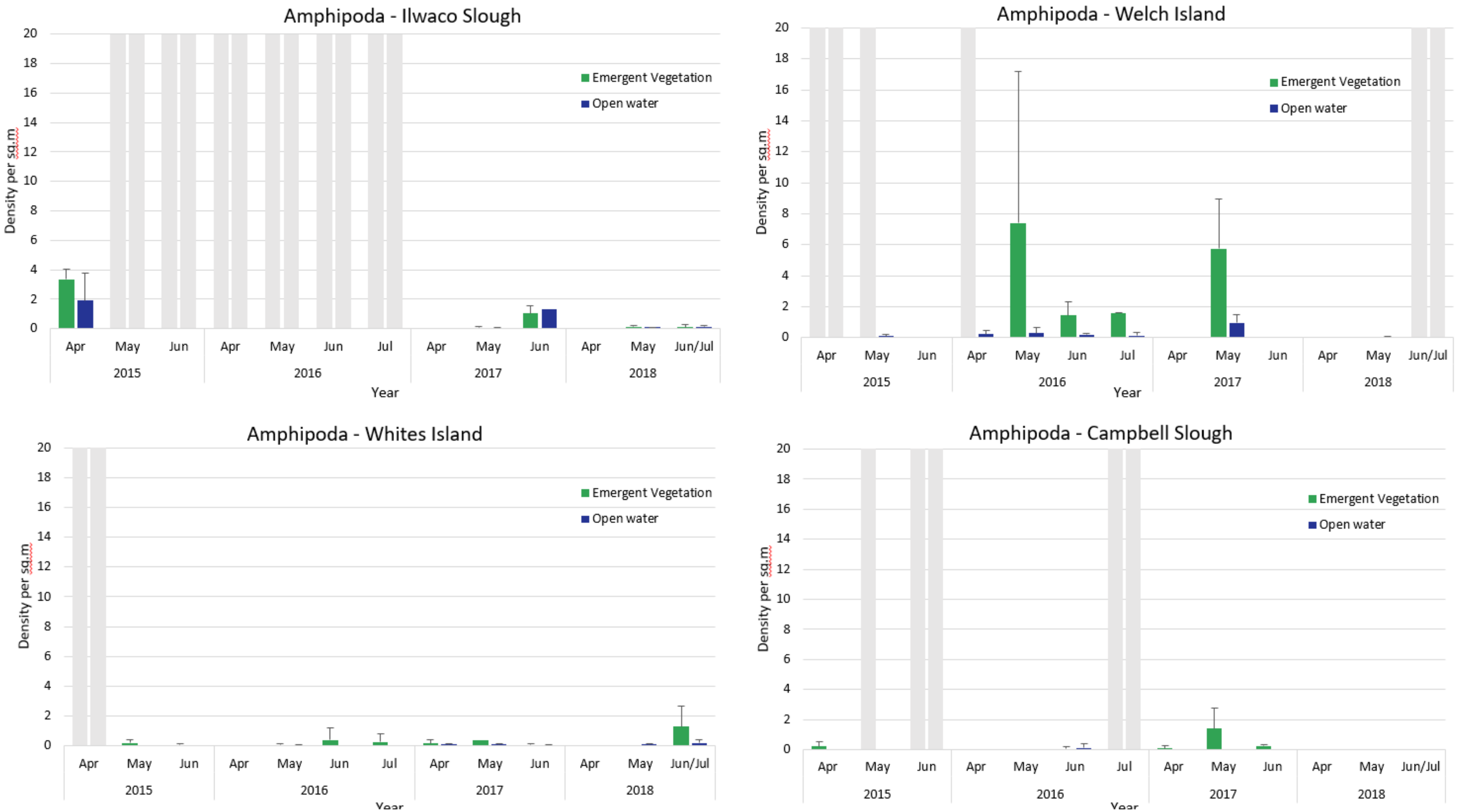


Figure 111. Average density (count per meter towed) of Amphipoda by neuston tow from emergent vegetation (green) and open water (blue), by year, month, and site. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected.

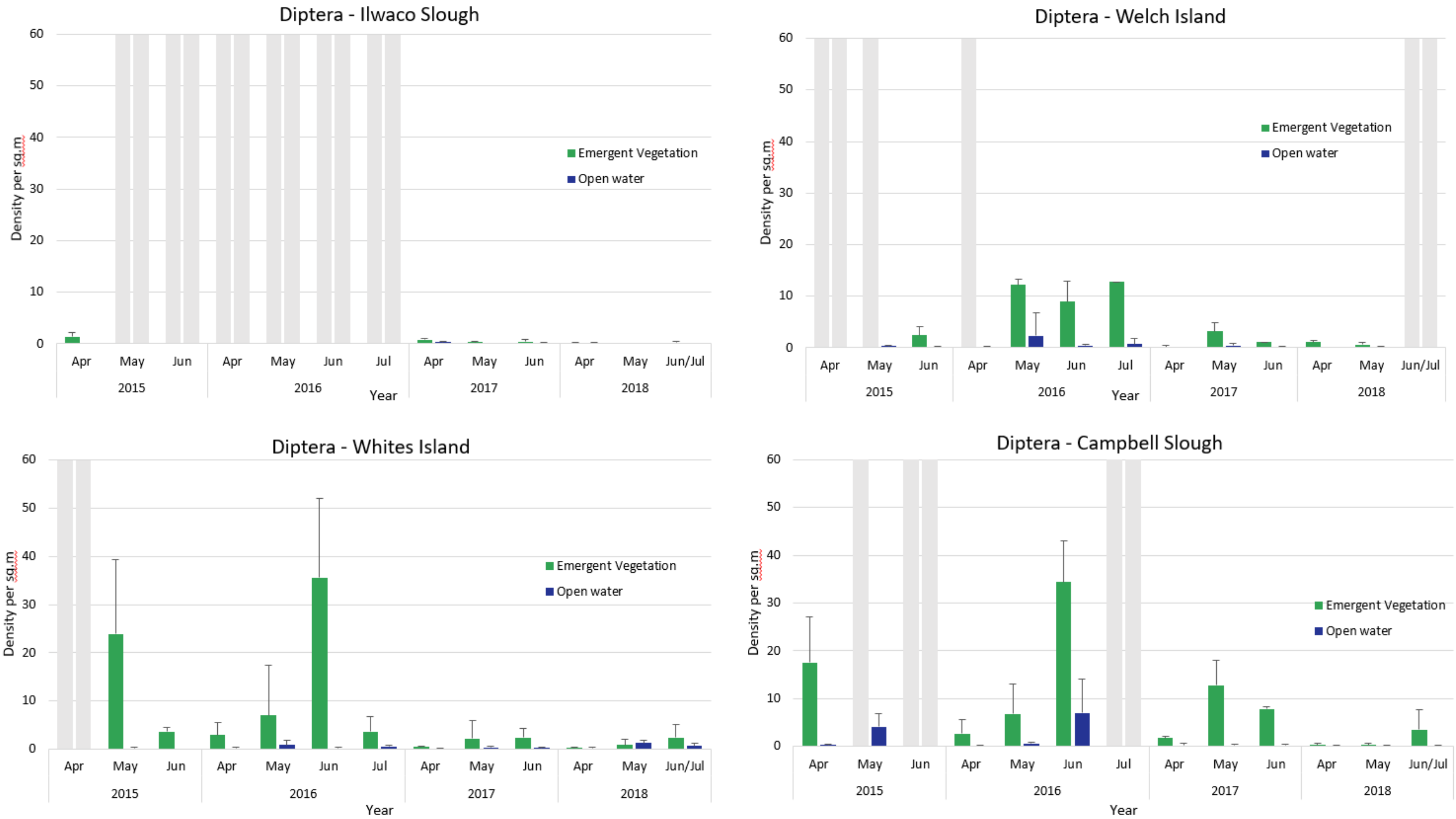


Figure 112. Average density (count per meter towed) of Diptera by neuston tow from emergent vegetation (green) and open water (blue), by year, month, and site. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected.

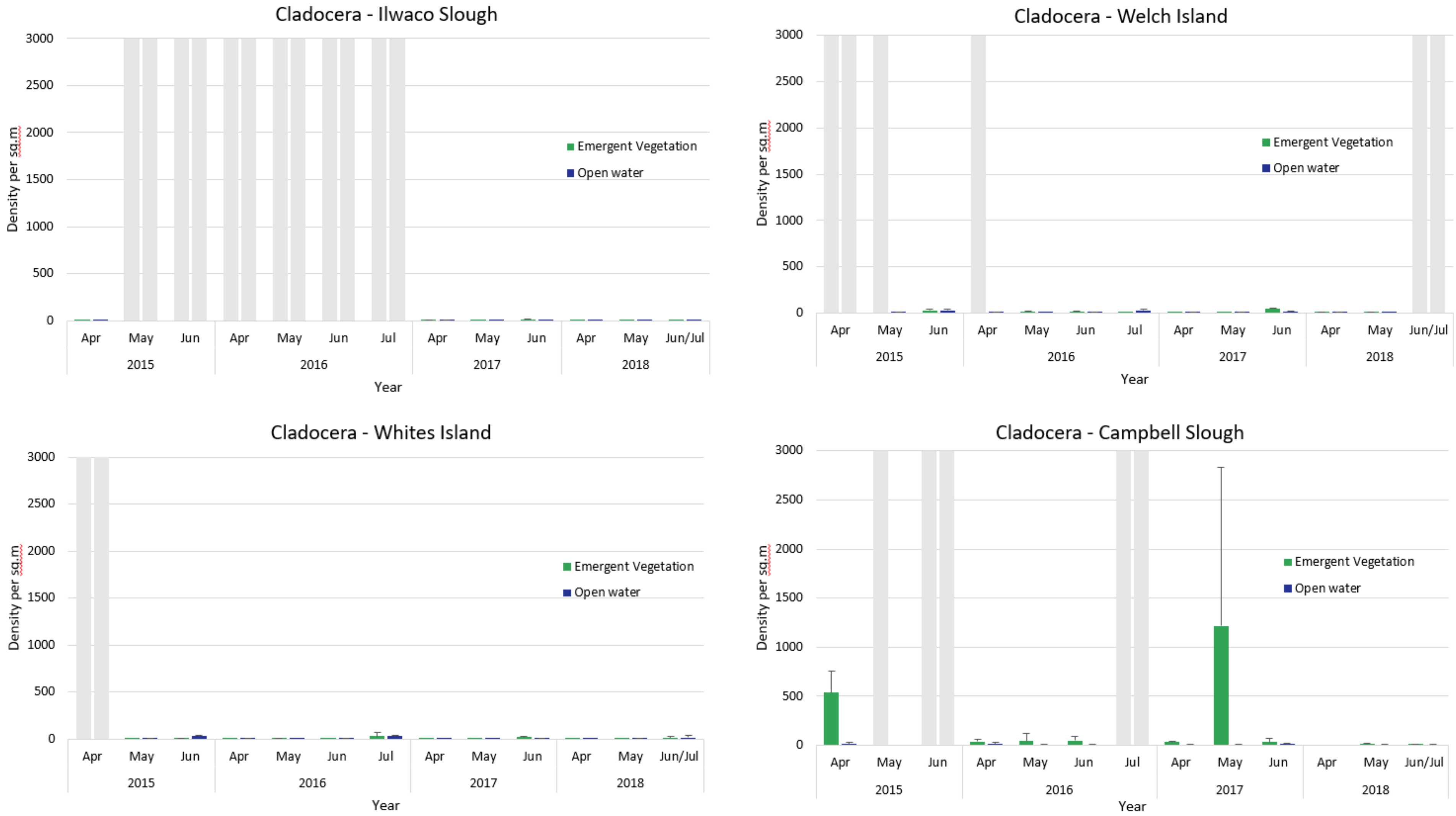


Figure 113. Average density (count per meter towed) of Cladocera by neuston tow from emergent vegetation (green) and open water (blue), by year, month, and site. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected.

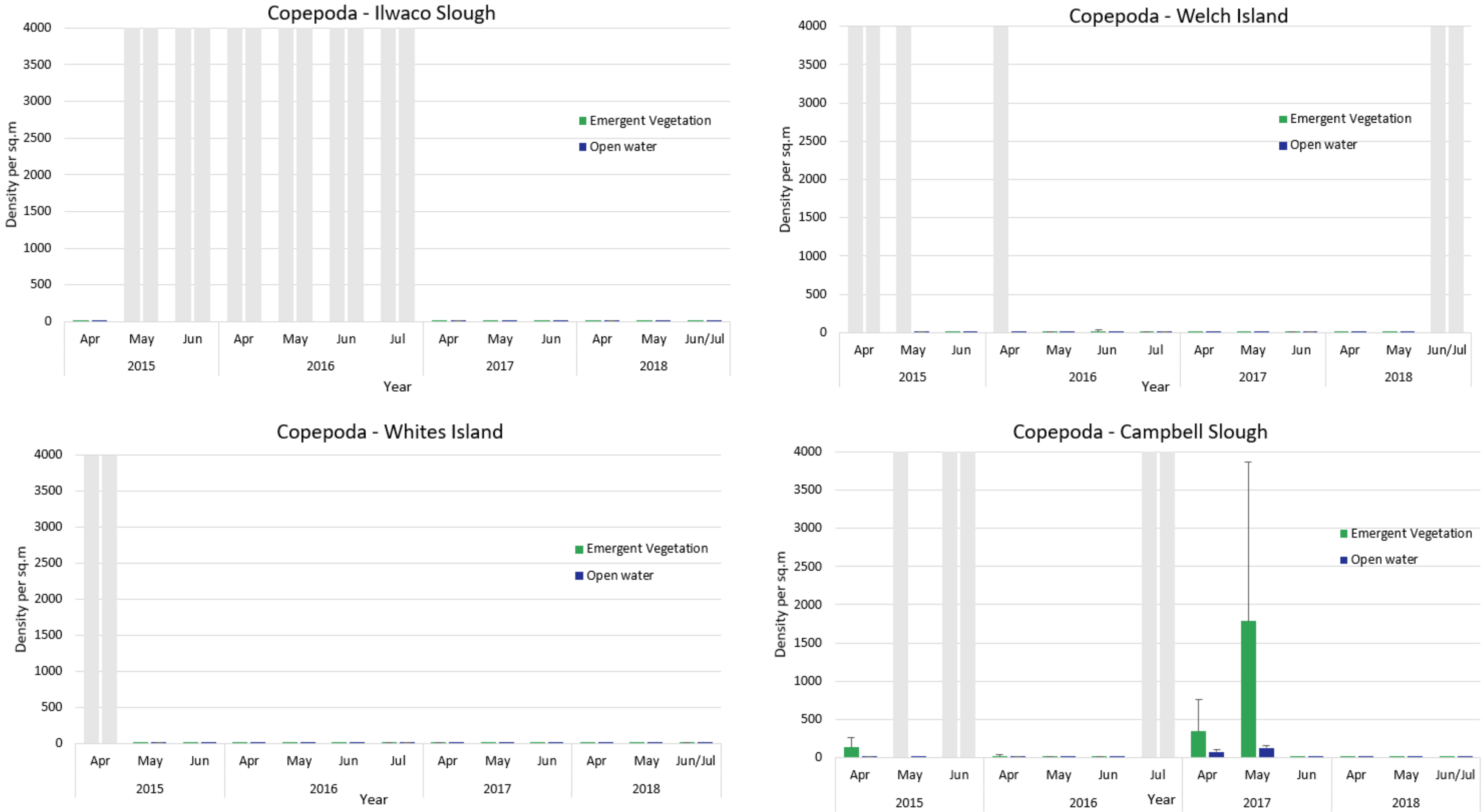


Figure 114. Average density (count per meter towed) of Copepoda by neuston tow from emergent vegetation (green) and open water (blue), by year, month, and site. Error bars represent the 95% confidence interval. Gray bars indicate no sample collected.

3.5.2 Salmon Diet

Salmon diet data collected in 2019 are still under analysis, 2018 results are reported below.

3.5.2.1 *Salmon Diet*

Juvenile Chinook diets measured by the index of relative importance (IRI), were typically dominated by amphipods and dipterans, particularly chironomid dipterans. The NMDS plot in Figure 115**Error! Reference source not found.** shows a clear distinction in diet composition of fish collected from the upriver sites (Campbell Slough and Franz Lake) which had higher contributions by dipterans, to the downriver sites (Ilwaco Slough, Welch Island, Whites Island), which had higher contributions by amphipods.

The EMP study has observed a consistent transition from the consumption of primarily dipterans and other wetland insects at Franz Lake and Campbell Slough, to the consumption of both dipterans and amphipods at Whites Island and Welch Island, to the consumption of primarily amphipods at Ilwaco Slough near the estuary mouth (Figure 117**Error! Reference source not found.**). While this trend occurs over most sample years (2008-2013, 2015-2018), there are also sometimes exceptions to the pattern. For example, the relative contribution of amphipods and dipterans to diets at Whites Island varies between years. Amphipods comprised approximately 85% of the IRI at Whites Island in 2018 and nearly 90% in 2017, but only 15% in 2012. In contrast, dipterans comprised only 10% of the IRI at Whites Island in 2018 and 8% in 2017, but nearly 80% in 2012. The percent IRI of amphipods at both Welch Island and Whites Island was highest in 2017 relative to previous sample years. Juvenile Chinook diets at Franz Lake were different in 2016 from previously sampled years, with a significant increase in the presence of copepods, and in 2017, diets collected from Campbell Slough were dominated by cladocerans. In previous sample years, dipterans had consistently comprised at least 70% of the IRI at Campbell Slough, but in 2017, dipterans made up only 10% and cladocerans accounted for nearly 90% of the IRI (Figure 117**Error! Reference source not found.**). In 2018, Campbell Slough diets had the typical composition with approximately 70% of the IRI comprised of dipterans overall, though the smallest size class of fish still consumed a significant portion of cladocerans (Figure 116).

In order to study the influence of cumulative discharge from the mainstem on salmon diets, 3 trend sites (Welch Island, Whites Island and Campbell Slough) were selected. These sites had the most consistent dataset over the monitoring years. Cumulative discharges years were grouped based on size of the spring freshet (hereby called "freshet condition") as 'wet' in 2011, 2012, and 2017, 'mid-range' in 2013 and 2018, 'dry' in 2016, and 'very dry' in 2015. Hydrologic condition varied across the years that juvenile salmon diets were analyzed; however, diet composition did not correlate with changing conditions (Figure 118, Figure 119, Figure 120). Within a site, there was no significant difference in the IRI between freshet conditions.

Combining measures of energy ration (ER) and maintenance metabolism (J_M) provides a general assessment of habitat quality and juvenile Chinook salmon growth potential at a given site and month. Figure 121 and Figure 122**Error! Reference source not found.** provide a graphic representation of this, plotting mean J_M against mean ER for fish sampled in May and June from Welch Island, Whites Island, and Campbell Slough under different cumulative mainstem discharges or "Freshet condition" ('dry' and 'very dry' years were combined in quadrant charts). Samples with high energy assimilation and low metabolic costs (lower right quadrant) reflect conditions especially conducive to juvenile salmon growth. Samples with low energy assimilation and high metabolic costs (upper left quadrant) reflect relatively stressful conditions that are not conducive to juvenile salmon growth. Maintenance metabolism in May and June was similar for the smallest fish (30-59mm length range) at the three sites shown in the quadrant charts and did not appear to correlate with freshet condition (Figure 121). Maintenance metabolism for medium sized juvenile Chinook salmon (60-79mm length range) was generally above average at Campbell Slough in May and June, but again did not appear to correlate with freshet condition (Figure 122).

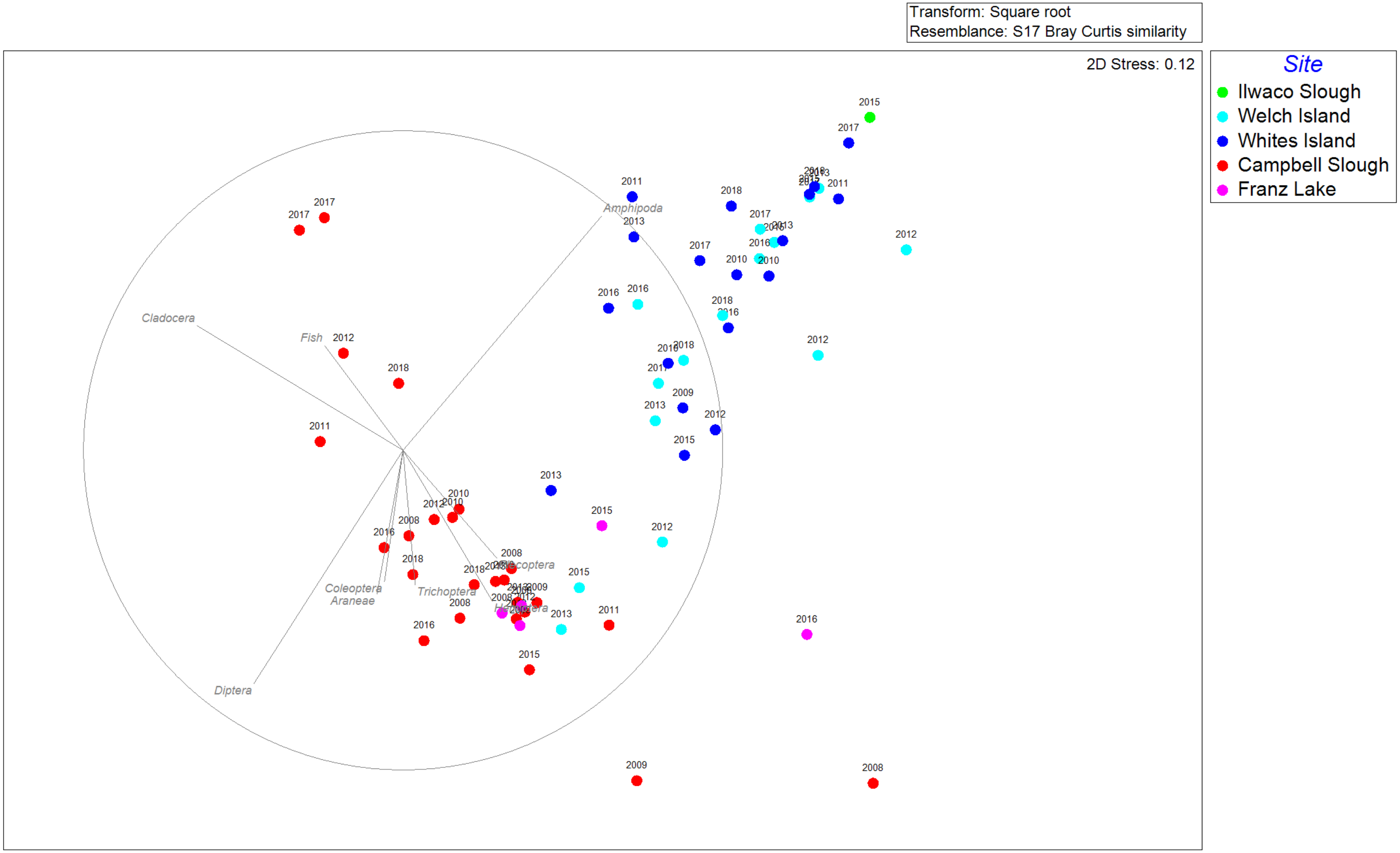


Figure 115. Two-dimensional NMDS plot based on Bray-Curtis similarities between transformed percent IRI of major prey in juvenile Chinook diets sampled between 2008 and 2018. Each point represents all fish collected between April and June within a size class (30-59, 60-79, 80-99 fish fork length in mm), site, and year. Correlation with prey taxa (Pearson R > 0.4) are represented as gray vectors.

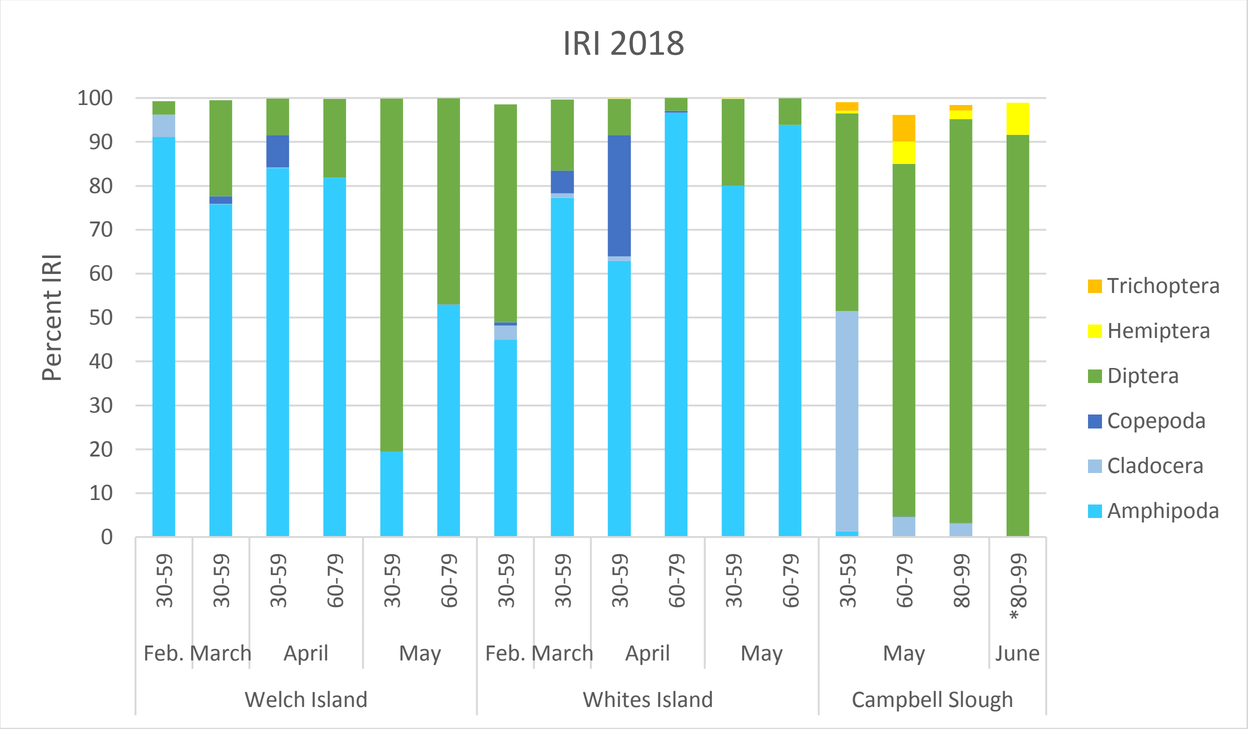


Figure 116. Percent IRI in 2018 by site, month, and size class (fish fork length in mm) for juvenile Chinook salmon collected between February and June. *Only one fish collected. All other sample sizes range between 3-17 individuals.

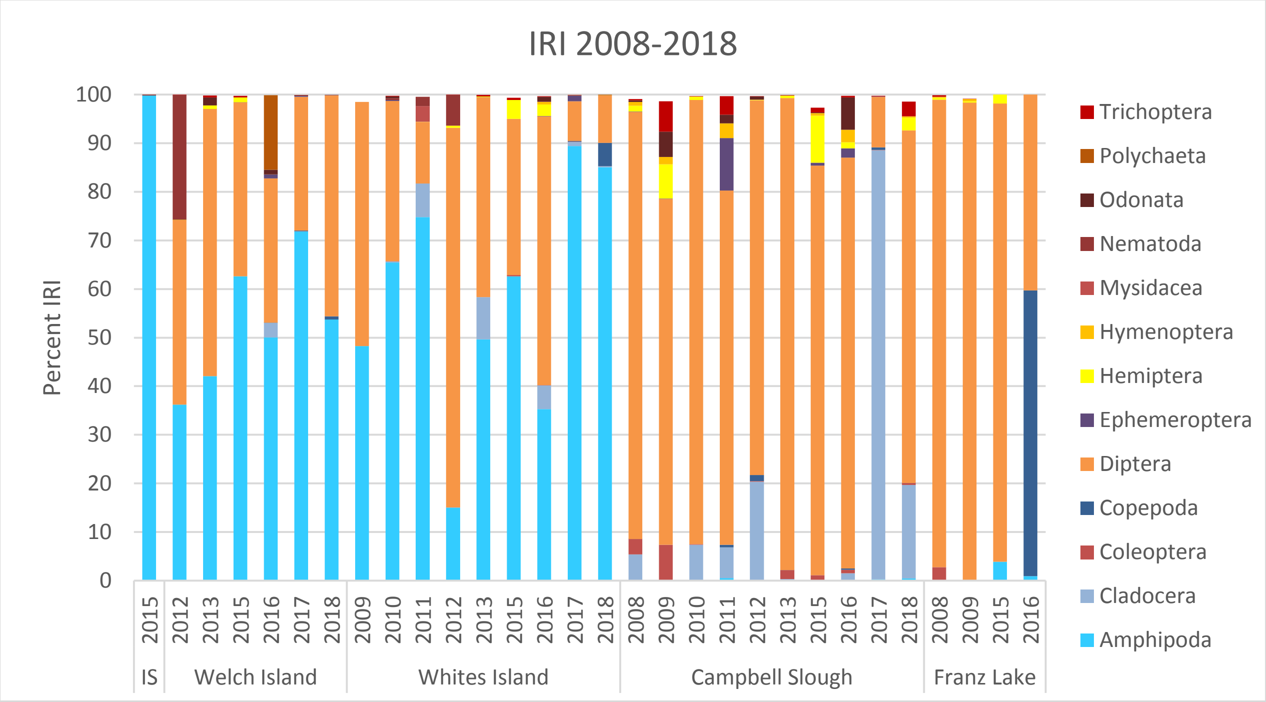


Figure 117. Percent IRI by site and year for juvenile Chinook salmon collected between April and June.

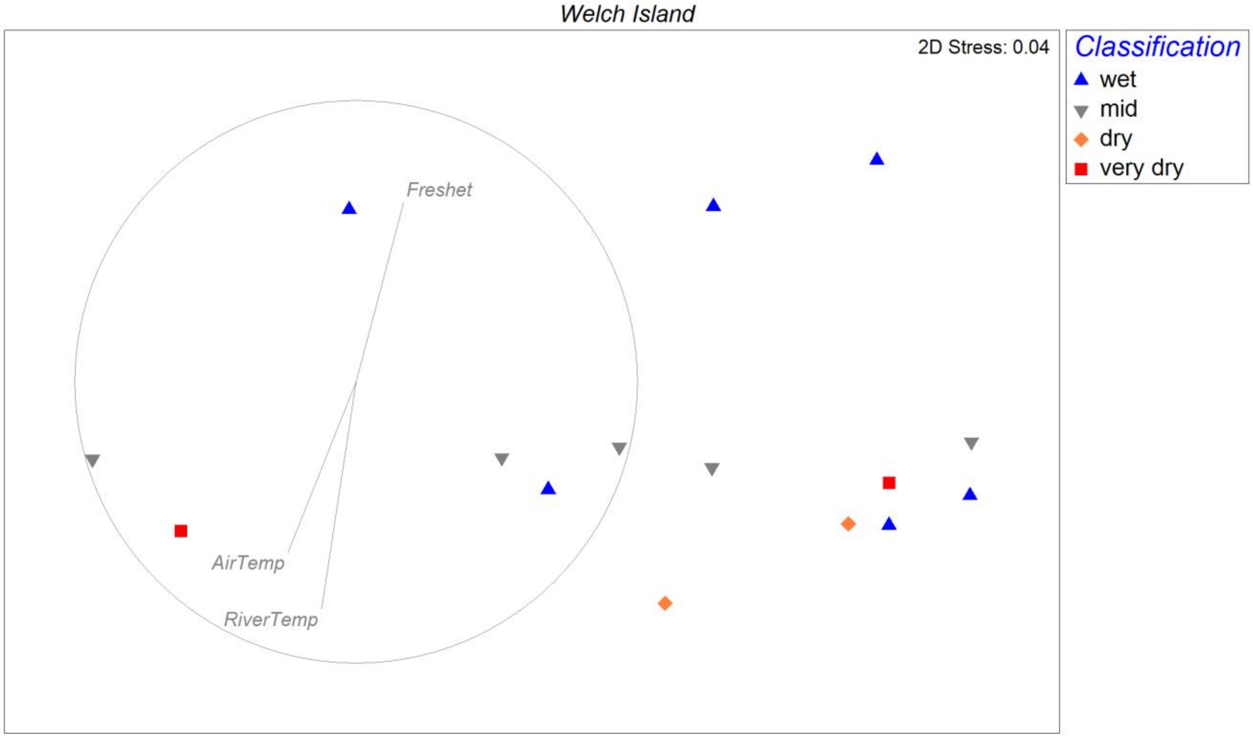


Figure 118: Two-dimensional NMDS plot based on Bray-Curtis similarities between transformed percent IRI of major prey in juvenile Chinook diets from a. Welch Island, Each point represents all fish collected between April and June within a size class and year. Years are classified as 'wet' (2011, 2012, 2017), 'mid-range' (2013, 2018), 'dry' (2016), and 'very dry' (2015).

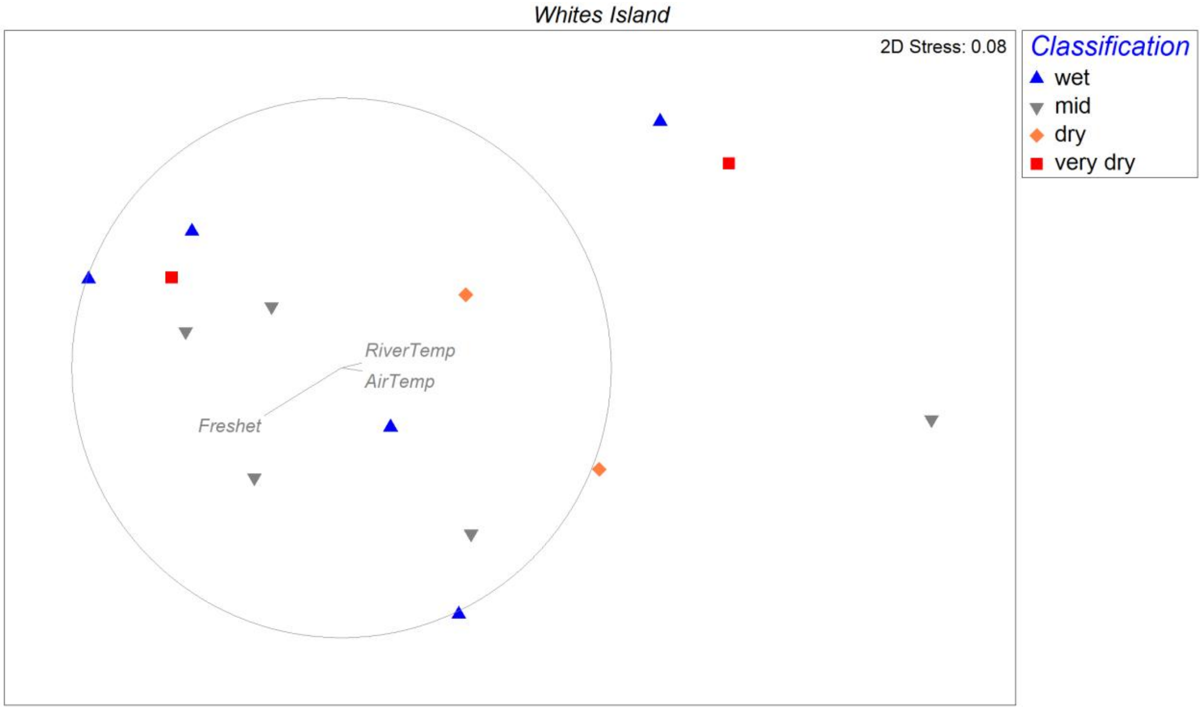


Figure 119: Two-dimensional NMDS plot based on Bray-Curtis similarities between transformed percent IRI of major prey in juvenile Chinook diets from Whites Island. . Each point represents all fish collected between April and June within a size class and year. Years are classified as 'wet' (2011, 2012, 2017), 'mid-range' (2013, 2018), 'dry' (2016), and 'very dry' (2015).

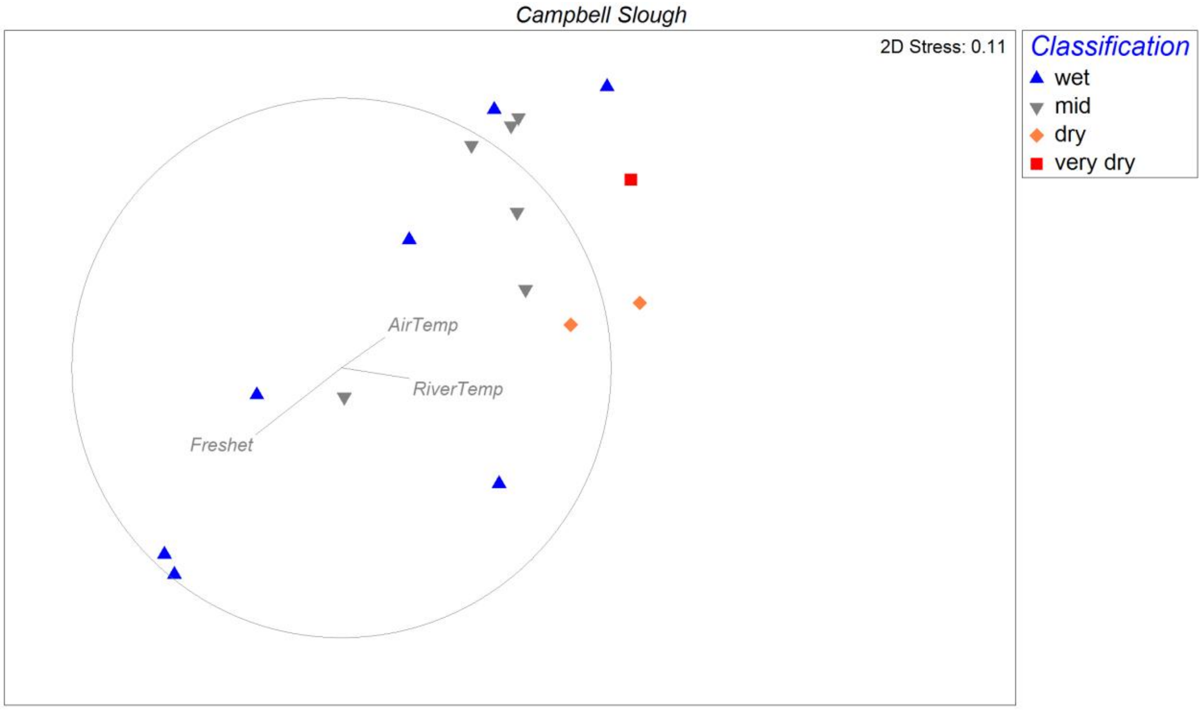
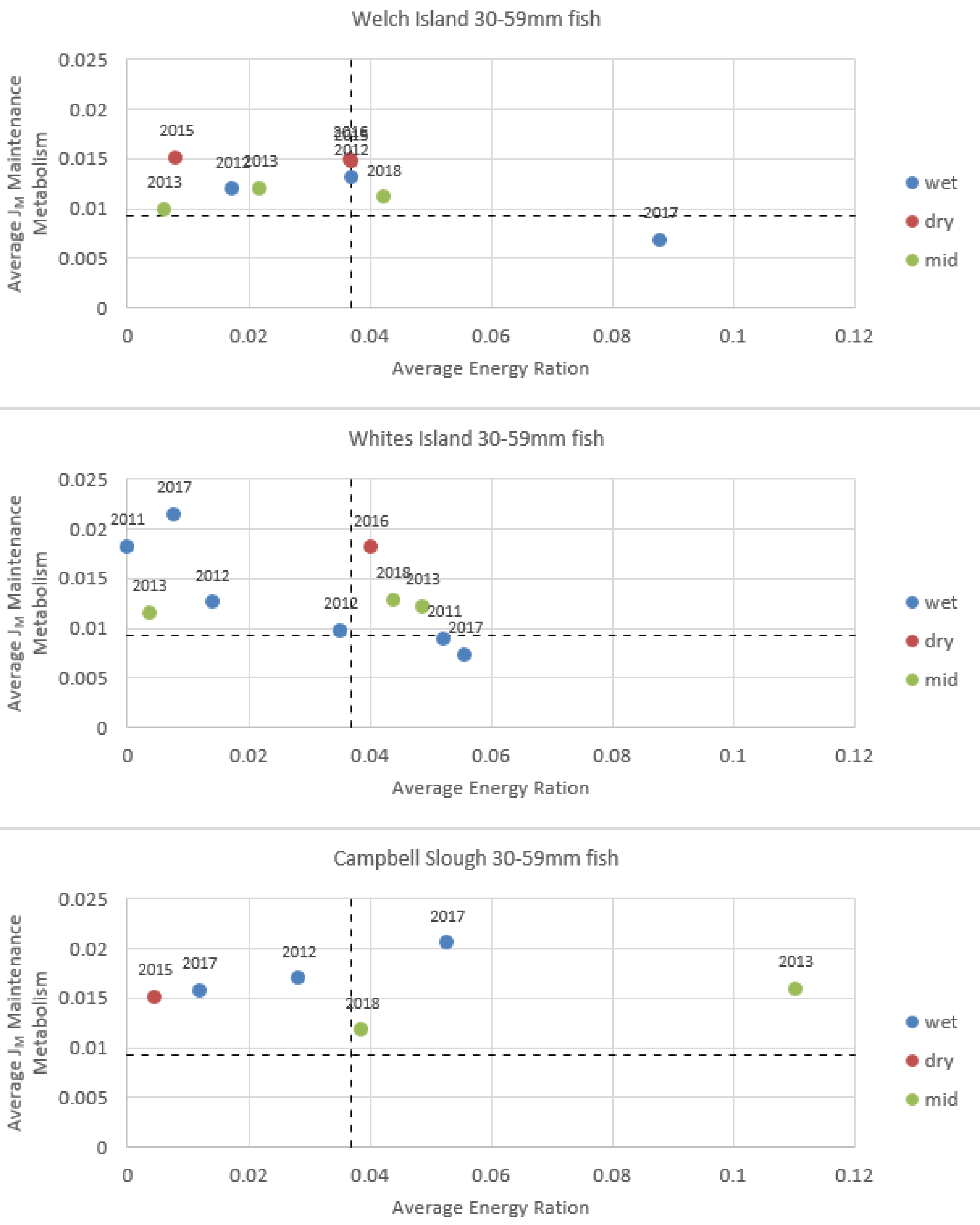


Figure 120. Two-dimensional NMDS plot based on Bray-Curtis similarities between transformed percent IRI of major prey in juvenile Chinook diets from Campbell Slough. Each point represents all fish collected between April and June within a size class and year. Years are classified as ‘wet’ (2011, 2012, 2017), ‘mid-range’ (2013, 2018), ‘dry’ (2016), and ‘very dry’ (2015).


Welch Island 30-59mm fish

Whites Island 30-59mm fish

Campbell Slough 30-59mm fish

Figure 121. Quadrant charts of monthly average maintenance metabolism (J_M) and average energy ration (ER), for juvenile Chinook salmon (30-59mm fork length) collected in May and June. Symbols distinguish freshet condition for wet (2011, 2012, 2017), dry (2015, 2016), and mid-range (2013, 2018) years. The dashed line is the 50th percentile of J_M and ER for 30-59mm fish collected across all sites and dates.

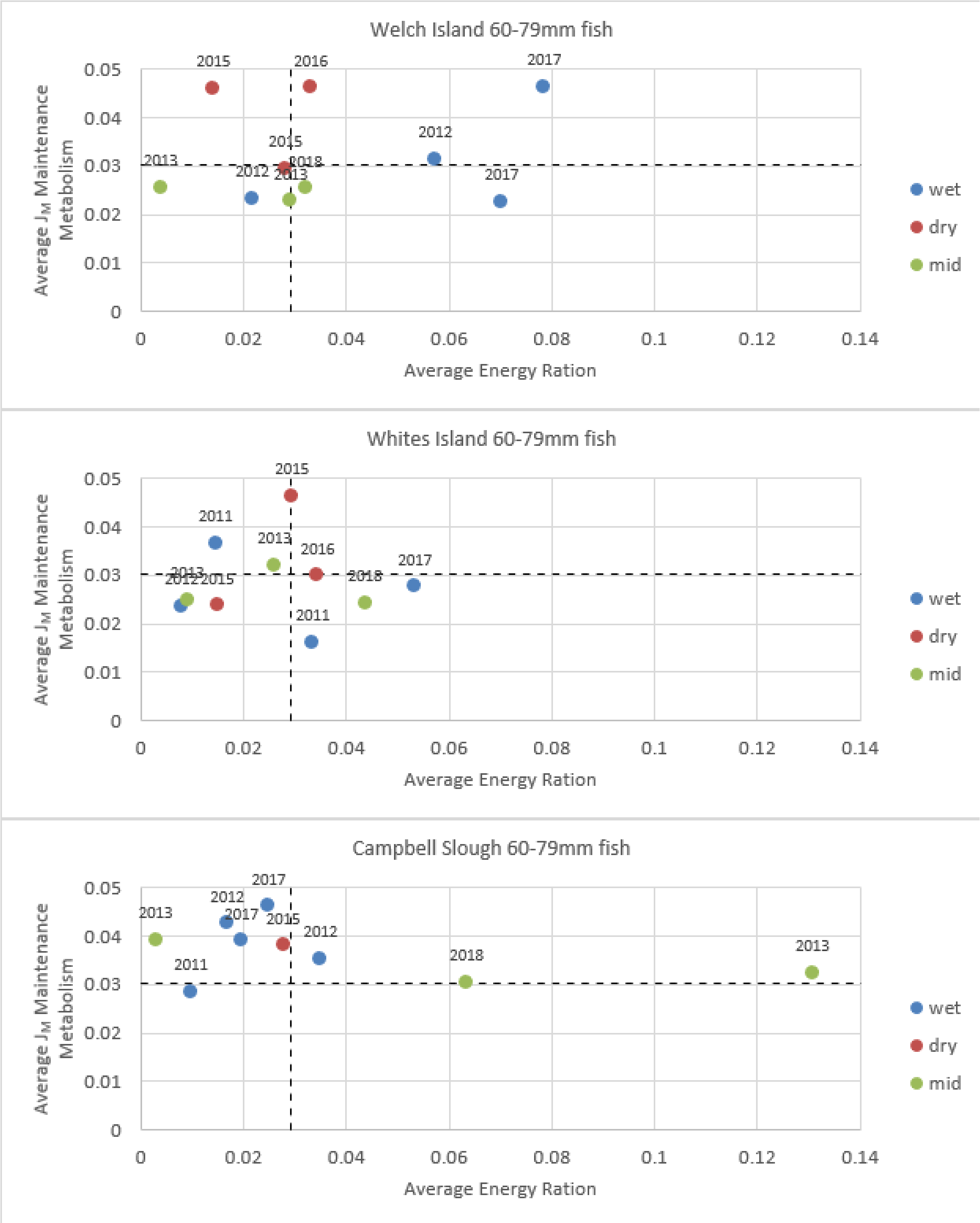


Figure 122. Quadrant charts of monthly average maintenance metabolism (JM) and average energy ration (ER), for juvenile Chinook salmon (60-79mm fork length) collected in May and June. Symbols distinguish freshet condition for wet (2011, 2012, 2017), dry (2015, 2016), and mid-range (2013, 2018) years. The dashed line is the 50th percentile of JM and ER for 60-79mm fish collected across all sites and dates.

3.6 Fish

3.6.1 Fish Community Composition

In 2019, fish communities were sampled at four of the five trend sites. Ilwaco Slough, Whites Island and Campbell Slough April through June and in October, and Welch Island March through June and in October. Sampling was attempted but unsuccessful in Franz Lake during high river levels in April then discontinued for the remainder of the year due to water levels (too high and/or low) and high temperature.

Thirteen different families of fishes were present and within those families 21 different taxonomic categories which included a combination of specific species and/or unidentified species within family/genus categories (Figure 123**Error! Reference source not found.**, and Appendix Table E-2). Threespine stickleback and banded killifish were the only species that were consistently present at all four sites sampled. Three salmon species were captured, with Chinook the most common and often the most abundant species, followed by chum which were captured at three sites but were not always numerically dominant. Campbell Slough had the highest diversity of species (14) followed by Welch Island (10) and Whites Island (9). Campbell Slough also had the highest percent of introduced fishes in the total catch (35%) followed by Welch Island (6%). The dominant species at the four sites was Threespine stickleback; IS=78%, WEL=84%, WHI=83% and CS=53%. Other commonly captured species for each site were; IS= Pacific Staghorn Sculpin 14%, Shiner perch 6% and chum salmon 1%; WEI= banded killifish 11%, Chinook salmon 4%; WHI= Chinook salmon 11%, banded killifish 4%; and CS= yellow perch 30%, Chinook salmon 9%.

Although relative abundances of fish species varied among sites, patterns of community structure can be discerned through multivariate analyses. The ANOSIM test on species abundance indicated a significant difference in fish community structure among trend sites from 2008-2019 (global $R = 0.268$, $P = 0.001$). Pairwise tests show that the community structure at all sites were significantly different from other sites except for Welch and Whites Islands ($P = 0.651$). The NMDS plot of community structure by site (Figure 124**Error! Reference source not found.**) shows the separation and overlap of sites and the driving factors. There is high overlap between Welch and Whites Islands, which is driven by Threespine sticklebacks and banded killifish. Ilwaco Slough tends to separate from Welch and Whites Islands based on the presence of marine species. Campbell Slough and Franz Lake are more similar with each other than with the lower river sites.

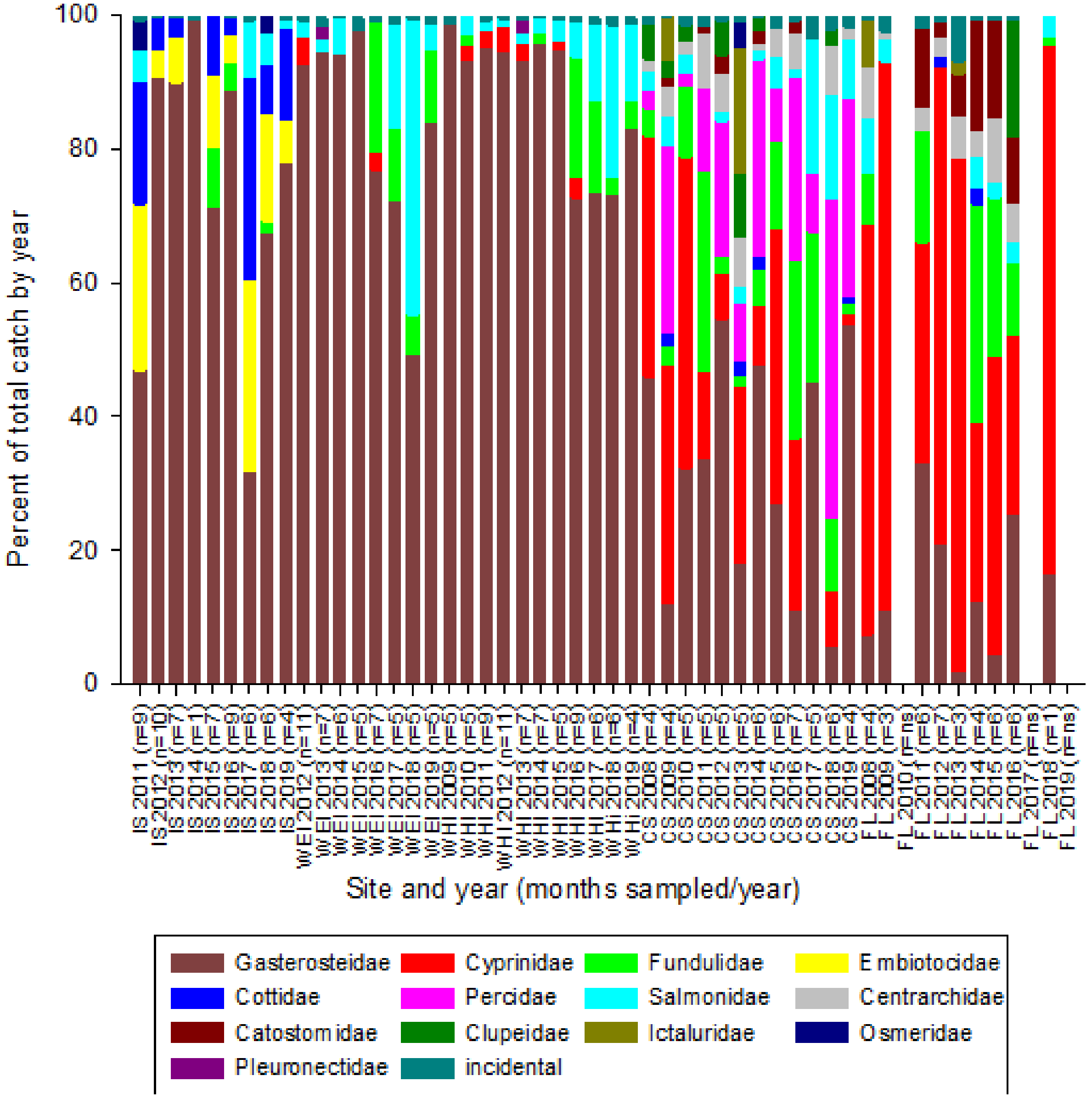


Figure 123. Fish community composition at EMP trend sites sampled from 2008-2019, presented by Family. For each year, the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

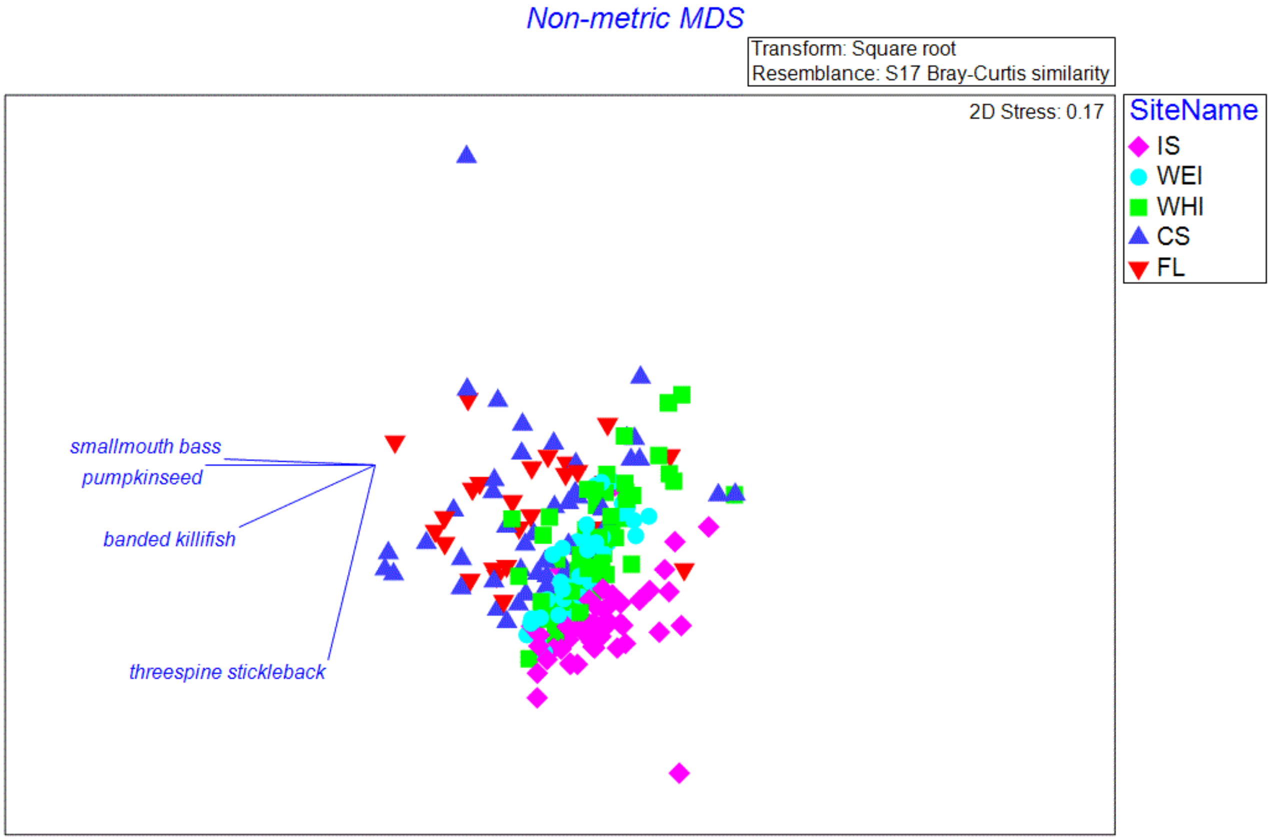


Figure 124. Nonmetric multidimensional scaling (NMDS) plot based on square-root transformed species abundance at five trend sites, 2008-2019. Significant correlation with variables (Pearson R > 0. 5) are represented as blue vectors. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

Mean species richness and associated ranges for each year of sampling at Ilwaco Slough, Welch and Whites Islands all fall between 0-10 species and remain similar from year to year (Figure 125**Error! Reference source not found.**). Campbell Slough and Franz Lake reflect a more diverse community (ranging from 0-18 species) that is dominated by introduced species that can tolerate warm water. The same trend is observed in the Shannon-Weiner diversity indices (Figure 126**Error! Reference source not found.**), where the values can vary minimally, yet, tend to be the highest at Campbell Slough and Franz Lake. Species richness at Campbell Slough remains lower than what had been observed from 2008-2016 and is likely an result of sampling protocol change that occurred in 2017 whereby sampling in July-September was suspended. However, the species diversity at Campbell Slough in 2019 was similar to values observed from 2008-2016.

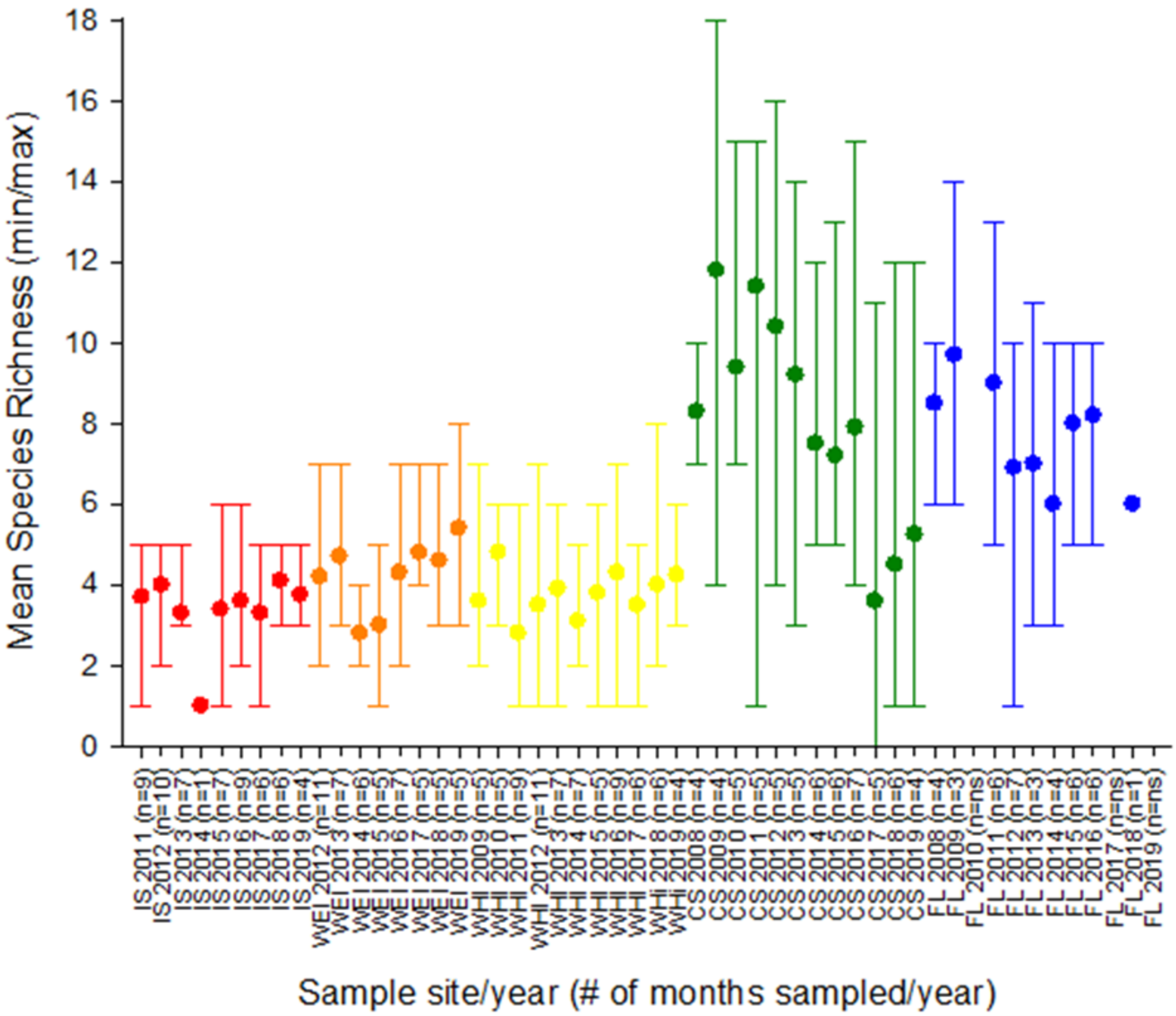


Figure 125. Mean species richness with minimum/maximum ranges for EMP trend sites sampled from 2008-2019. For each year, the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

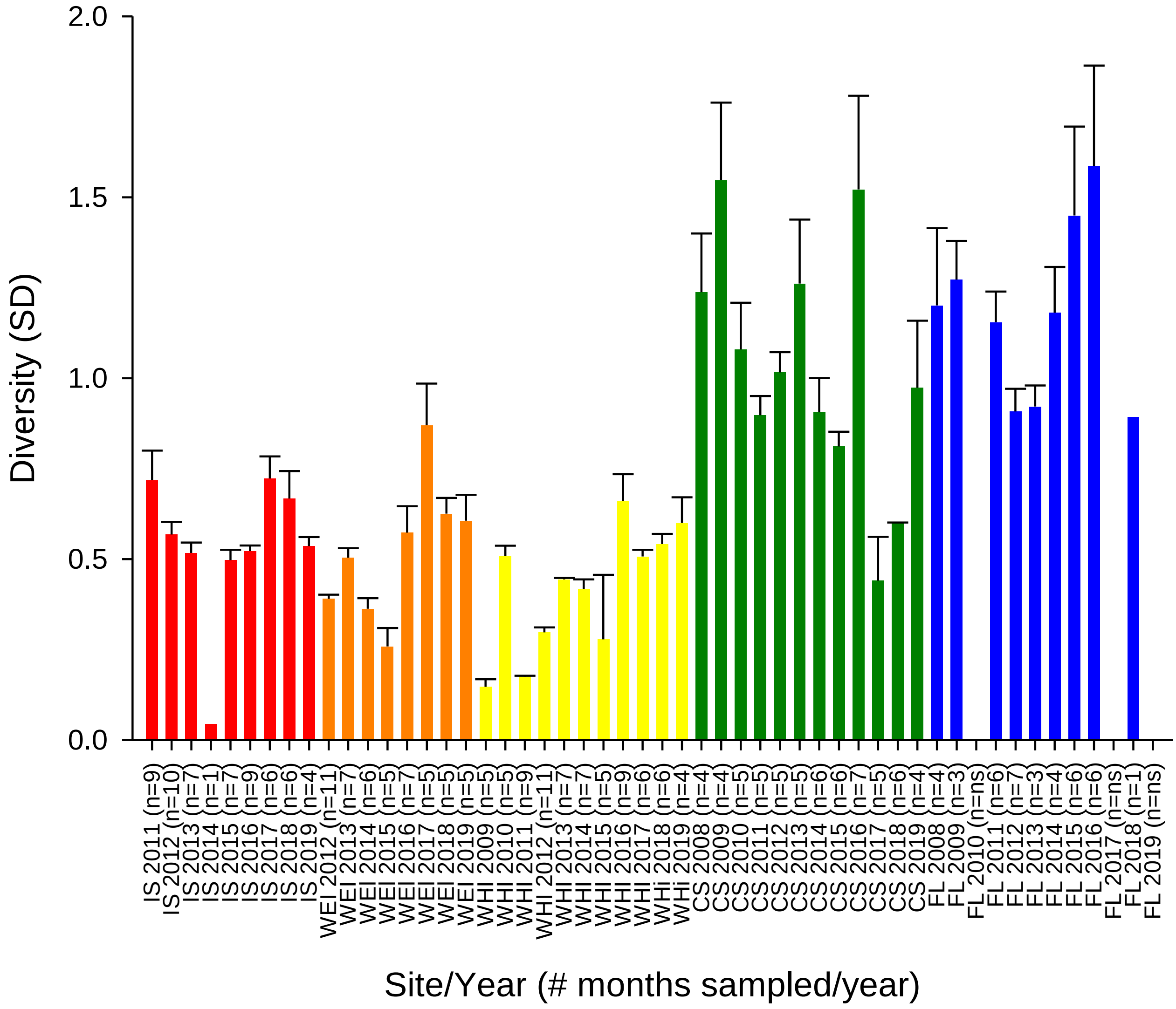


Figure 126. Mean Shannon-Weiner diversity index with standard deviation from EMP trend sites sampled from 2008-2019. For each year, the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

In 2019, the monthly sampling at each site did not show any distinct pattern of species richness and diversity (Figure 127). The number of species each month typically ranged from 1-6, the only exception occurred at Campbell Slough in June where the total number of species reached 12. Campbell Slough also had the greatest range in diversity (0.0-1.1) which reflects the higher number of species and the overall total catch of those species.

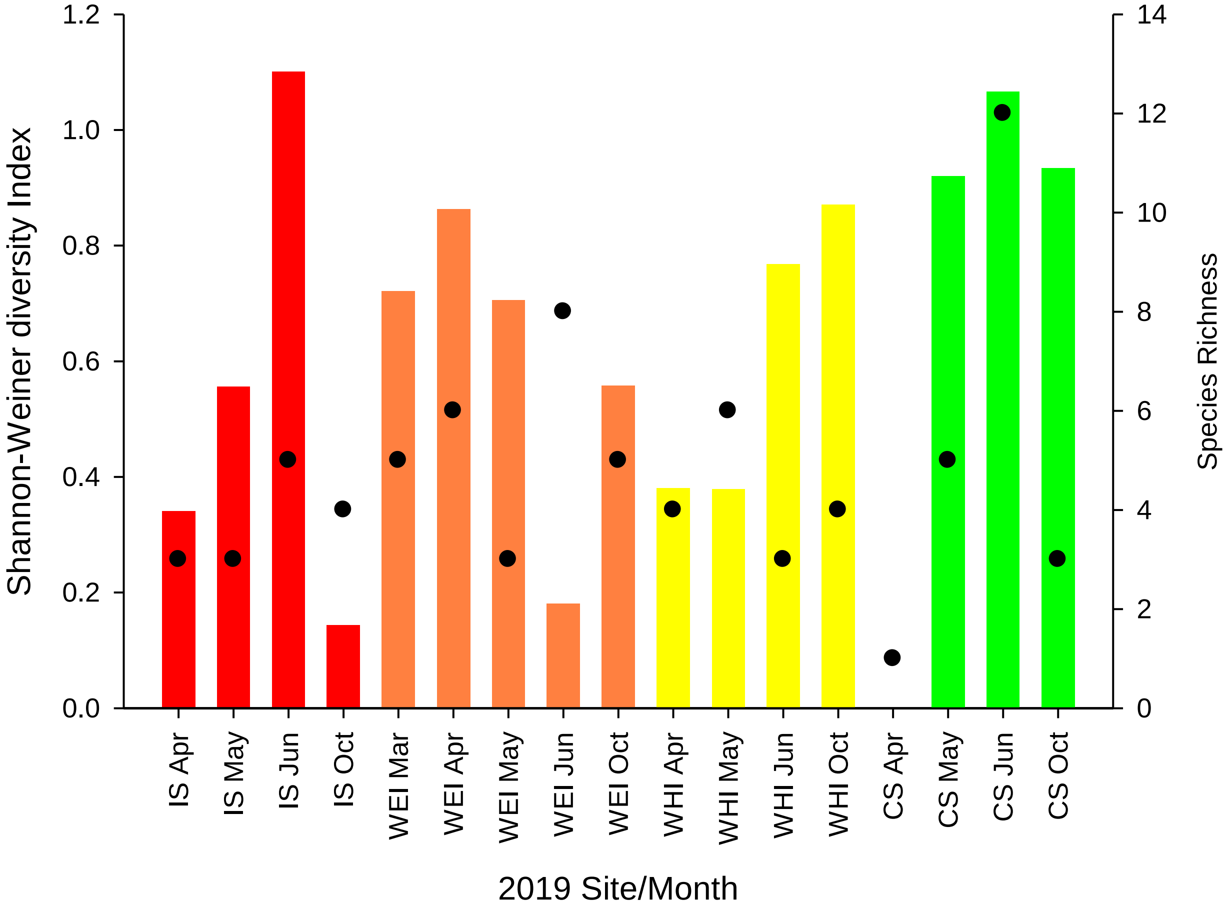


Figure 127. Shannon-Weiner diversity index (bars) and species richness (closed circles) for EMP trend sites sampled monthly in 2019. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough.

Non-native fish species occur at all five trend sample sites over all sampling years; their presence is highly variable and likely very dependent on water levels and temperature (Figure 128). The highest number of non-native fishes occur at Campbell Slough where the catch rates have exceeded 50% (range from 40-76%) for seven out of the last eleven years. At Campbell Slough, banded killifish, yellow perch, and unidentified juvenile carp comprise the majority of the non-native species. Franz Lake has the and second highest numbers of non-native species exceeding 20% for six out of the last nine years of sampling and ranging from 6-54%. At Franz Lake, banded killifish, unidentified juvenile carp, and yellow bullhead are the predominant non-native species.

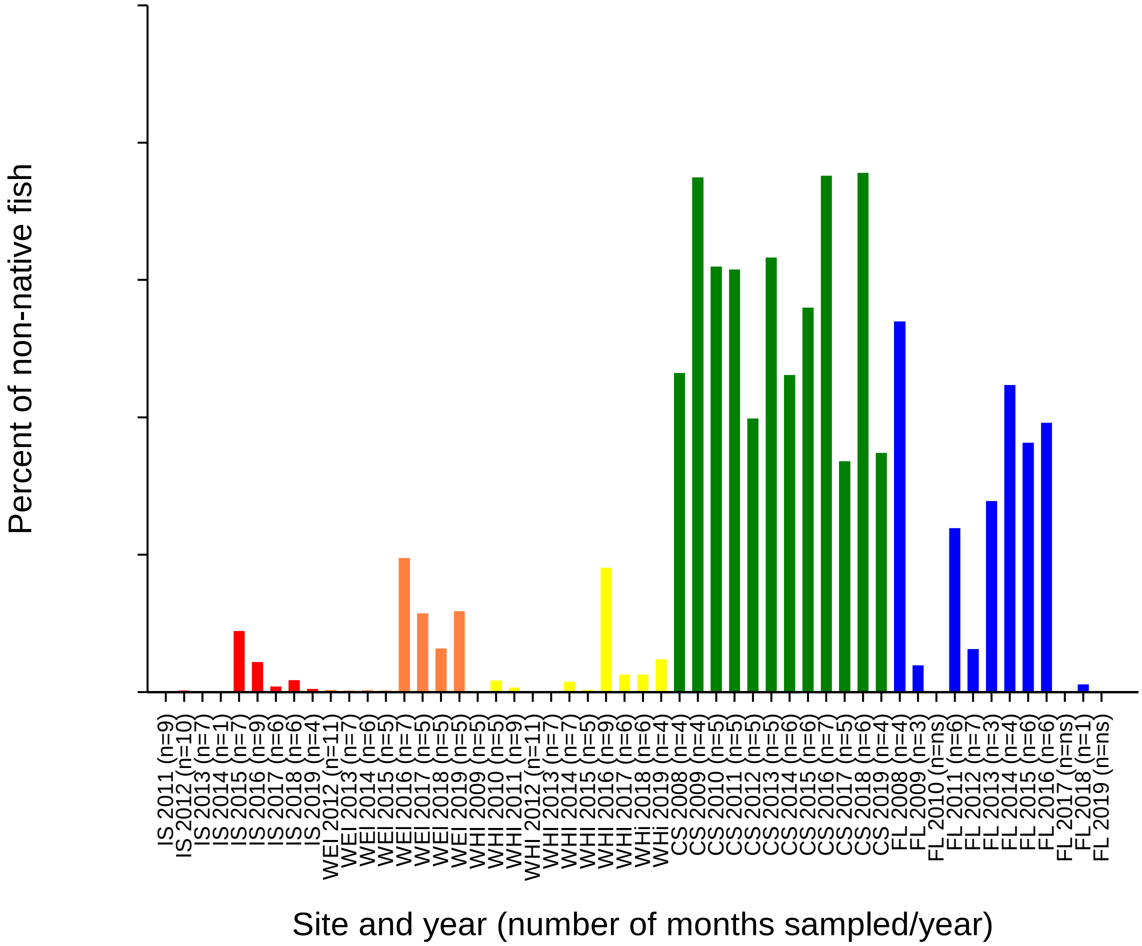


Figure 128. Percent of total fish catches per year that are non-native species for EMP trend sites sampled in 2008-2019. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

There are five non-native (small and largemouth bass, walleye, warmouth, and yellow perch) and one native (northern pikeminnow) fish species that produce mature stages that can prey on juvenile salmon. These fish are freshwater species that primarily occur at Campbell Slough and Franz Lake and minimally occur at Welch and Whites Islands (Figure 129). Yellow perch are the most common species followed by northern pikeminnow and are found at the four sites. Smallmouth bass is the third most common species has been captured at Campbell Slough and Franz Lake exclusively. In 2019, yellow perch comprised 81% of the total number of predatory fish captured.

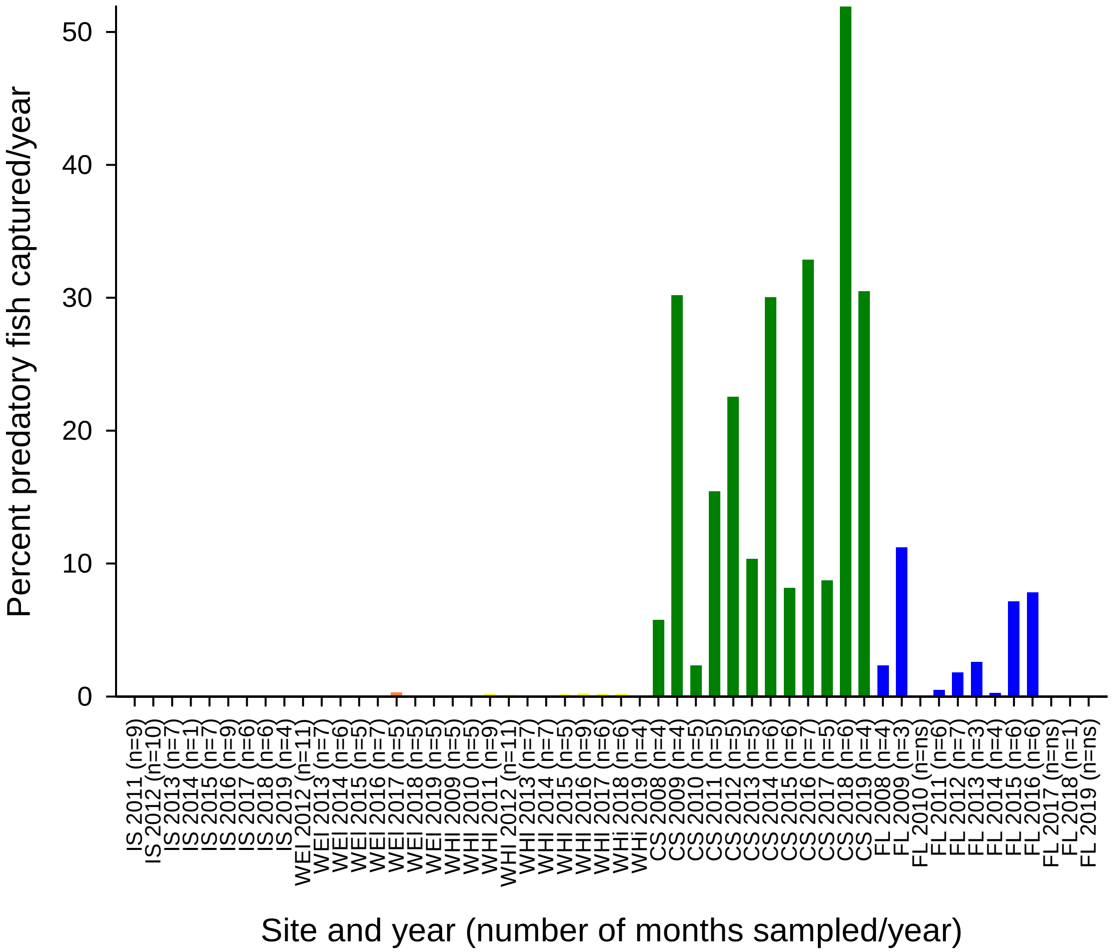


Figure 129. Total percentage of the yearly (2008-2018) catch of fish species that have mature stages that could be predatory toward juvenile salmon. Species include small and largemouth bass, northern pikeminnow, walleye, warmouth, and yellow perch. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.2 Salmon Species Composition

Similar to previous sampling years, 2019 salmon species composition varied by site, showing distinct patterns associated with hydrogeomorphic reach (Figure 130**Error! Reference source not found.**). In 2019, Chinook salmon were caught at three of the four sampled sites and were the dominant salmon species at Welch Island in Reach B, Whites Island in Reach C and Campbell Slough in Reach F. At these sites, Chinook salmon comprised 90 to 100% of salmonid catches. In 2019, unmarked (presumably wild) Chinook were more abundant at all three sites where Chinook were sampled than marked hatchery Chinook (Figure 131**Error! Reference source not found.**). In addition to Chinook salmon, small numbers of chum salmon were found at Welch Island and Whites Island. This pattern is typical for Welch and Whites Islands and has been evident since 2012. Chum salmon were the most abundant and only salmon species captured at Ilwaco Slough in Reach A. Only one unmarked Coho salmon was collected at Whites Island. No trout or sockeye salmon were caught in 2019 (Figure 130 and Table E-3 in Appendix E).

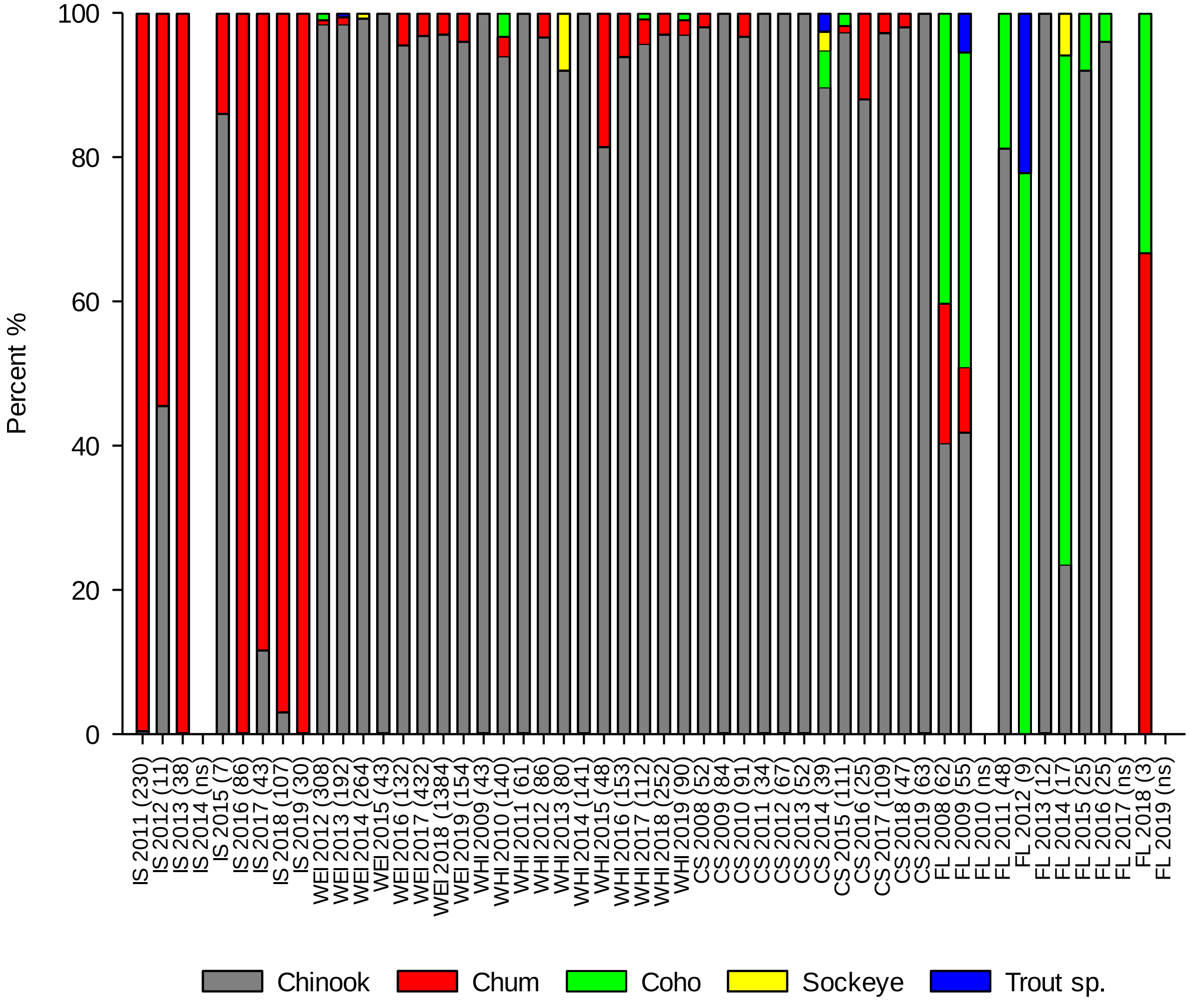


Figure 130. Percentage of salmonid species collected at EMP trends sites in 2019, as compared to percentages collected in previous sampling years. Total number of salmonids captured at a given site and year are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

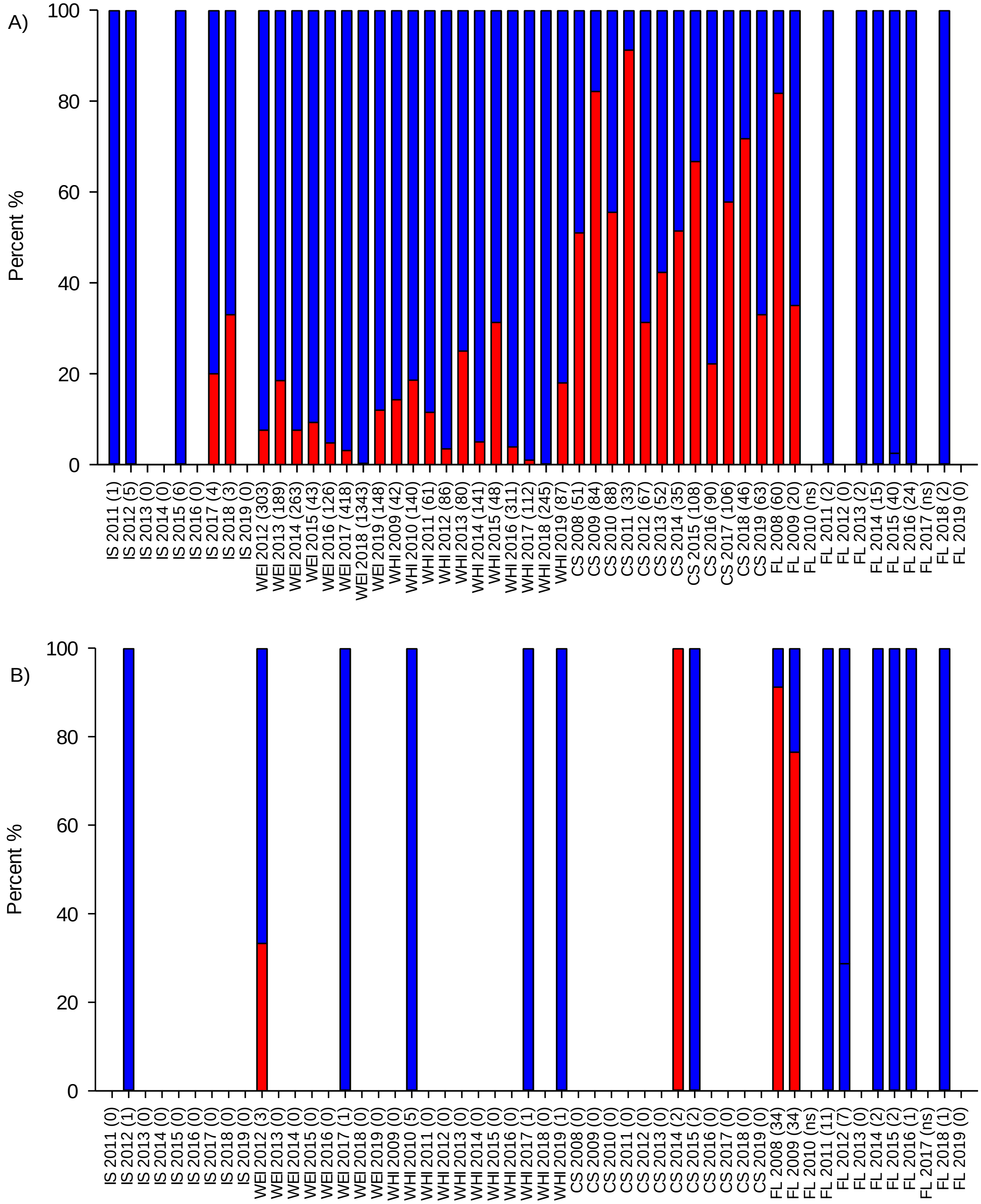


Figure 131. Percentage of marked (red) and unmarked (blue) A) Chinook salmon and B) Coho salmon captured at the EMP sampling sites in 2019, as compared to previous sampling years. Horizontal Axis represents site and year of sampling. Total number of the specified salmon species captured at a given site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.2.1 *Salmon Density*

Chinook salmon

In 2019, unmarked Chinook salmon were captured at the EMP trend sites from March (Welch only) through June and again in October. The highest average densities of unmarked juvenile Chinook salmon were 69.5 per 1000 m^2 in May. Marked Chinook salmon were captured from March through June and again in October as well, with the highest average densities of 17 fish per 1000 m^2 in May (Figure 132a). Mean Chinook salmon densities by site and year are shown in Figure 133. In 2019 the density of unmarked Chinook salmon was highest at Welch Island (54.60 fish per 1000 m^2) and Whites Island (35.89 fish per 1000 m^2) and lowest at Campbell Slough (19.26 fish per 1000 m^2) with no Chinook captured at Ilwaco Slough and Franz Lake in 2019. Densities of marked Chinook salmon in 2019 were greatest at Campbell Slough and similar to catches in the past three previous years. The densities of marked Chinook salmon in 2019 were generally within the same ranges as 2008-2018 at all sites (Figure 133).

Coho salmon

In 2019, only one unmarked coho salmon was collected at Whites Island in May (Figure 130). This marks only the third time in the past eleven years in which coho salmon have been sampled at Whites Island. Coho salmon densities by site and year are listed in (Appendix E). Coho salmon have been captured only sporadically at Ilwaco Slough, Campbell Slough and Welch Island, so their absence in 2019 was not unusual compared to previous years. No sampling was conducted at Franz Lake in 2019 due to high water and warm summer water conditions. Franz Lake is traditionally the only site where coho salmon have been consistently collected. Coho salmon density at Franz Lake was at its lowest reported level in 2016 and has shown a consistent decline since 2011. However, low sampling efforts in 2017, 2018 and no sampling in 2019 have made it difficult to determine any recent trends in coho abundance levels at Franz Lake. Marked coho salmon, which were common at Franz Lake in 2008 and 2009, have not been observed since 2012.

Chum salmon

In 2019, chum salmon were found at the trends sites in March, April and May with the highest average density in May (11.14 fish per 1000 m^2; Figure 131b). Chum salmon were present at Ilwaco Slough, Welch Island and Whites Island in 2019 (Figure 130). In 2019 the density of chum salmon was highest at Ilwaco Slough (11.08 fish per 1000 m^2) and lowest at White Island (1.50 fish per 1000 m^2). However, no chum were captured at Campbell Slough and Franz Lake was not sampled in 2019. Chum salmon have been found at all the sampling sites at varying densities, although not consistently. Chum salmon have not been observed at Franz Lake since 2009.

Sockeye salmon and trout species

In 2019, as in the past 4 years, Sockeye salmon and trout were not caught. Historic densities for sockeye salmon and trout for at all sampling have been extremely low (Appendix E, Table E-3).

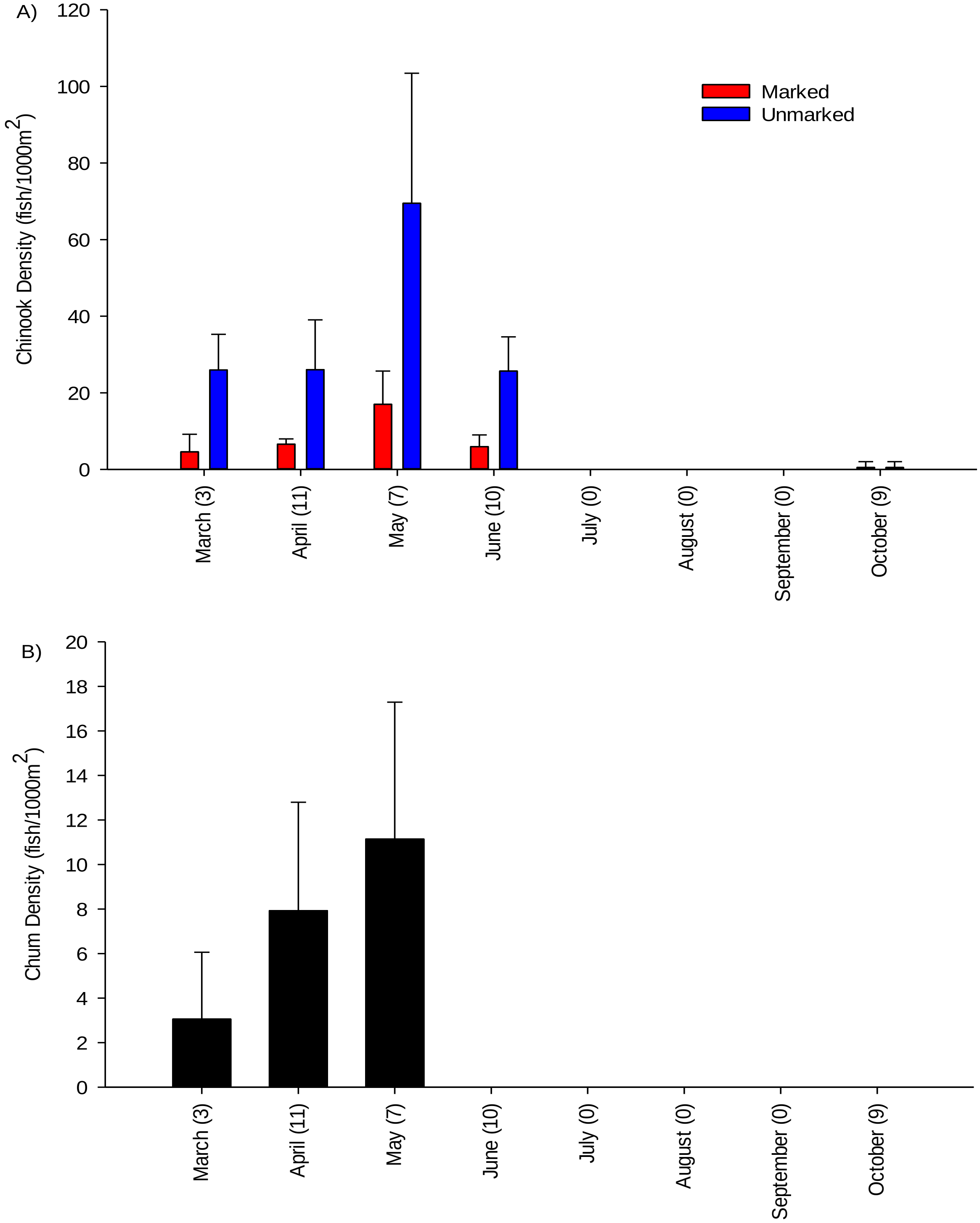


Figure 132. Monthly Mean (SE) densities (fish per 1000 m^2) of a) marked (red bars) and unmarked (blue bars) juvenile Chinook salmon, b) chum salmon in 2019 (all sites combined). Total number of sampling efforts per month are presented in parentheses. Only one coho salmon was captured at all sites in 2019 therefore no monthly density for coho salmon is shown.

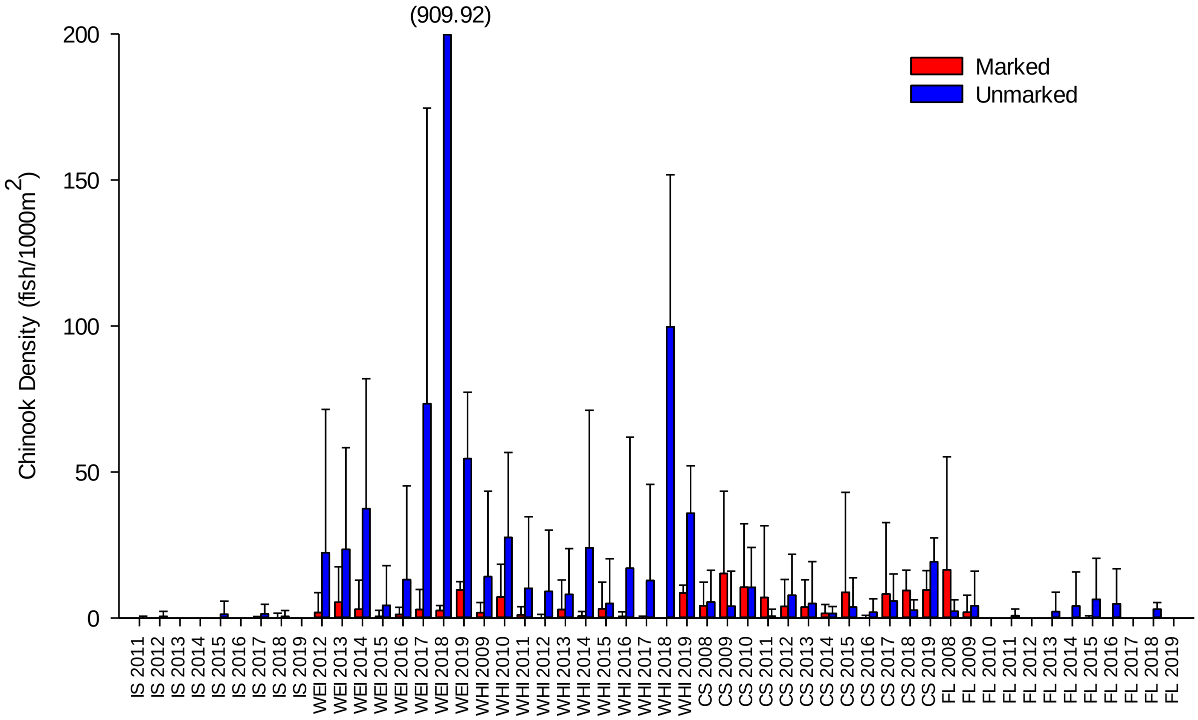


Figure 133. Marked (red bars) and unmarked (blue bars) juvenile Chinook salmon densities (fish per 1000 m^2) by site and year. Welch 2018 was truncated for ease of viewing and the number in parentheses is the upper limit of the error (SE). IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.3 Salmon Metrics

3.6.3.1 *Genetic Stock Identification*

In 2019, genetics data were collected from Chinook salmon at Welch Island, Whites Island, and Campbell Slough, . To maintain the highest level of confidence in stock assignments, we only reported stock assignments for fish that had an assignment probability greater than or equal to 0.90. We applied this criterion across all reporting years. On average, 86% of genetic samples assigned at 0.90 or greater.

Among unmarked fish in 2019 (Figure 134a), West Cascade fall Chinook were the predominant stock at sites in Reaches B and C: Welch Island and Whites Island. This trend has persisted since 2010 and 2012 when Whites and Welch Islands were first sampled by this study. The percentages of upper Columbia summer/fall stocks at both Welch and Whites Islands, 21 and 17%, respectively, were the highest yet reported at those sites. Similar to previous years, the unmarked stock composition at Campbell Slough was more diverse than the lower river sites with West Cascade fall stocks comprising only 16%-- the lowest observed contribution of West Cascade fall stock at Campbell Slough except for 2011 and 2015 when no West Cascade fall were collected. In contrast, upper Columbia summer/fall comprised 66% of the sample at Campbell Slough in 2019, the highest percentage observed at that site. Spring Creek fall stock represented 12.5% and West Cascade spring stock comprised 6.3%.

Substantially fewer marked fish were collected and subsequently analyzed for genetics at trend sites in 2019 (Figure 134b). Changes in stock composition occurred at Welch and Whites Islands in 2019, and spring stocks represented a larger percentage of the sample than had been observed in previous years. 2019 was the first year that West Cascade spring stocks were observed at Welch and Whites, and the percentage of Willamette River spring stock increased at both sites, as well. At Campbell Slough, 85% of marked Chinook assigned to the Spring Creek Group, yet 15% assigned to the mid and upper Columbia spring reporting group. This is the first time the mid and upper Columbia spring reporting group, which includes ESUs listed as endangered, has been observed at any of the trend sites.

The seasonal distribution of stocks in 2019 (Figure 135a) reveals that West Cascade fall Chinook are present throughout the lower Columbia River and estuary during spring and summer. The increase in upper Columbia summer/fall stock in 2019 is evident in May and June. May and June are also the months when the greatest diversity of stocks is observed with the mid and upper Columbia spring present at Campbell Slough in May and Snake River fall present in June. Willamette River spring fish were present in the early spring and in June. Interior stocks occurred earlier at upper reaches and were not present in lower reaches until June.
The seasonal trend of stocks in 2019 varied from previous years due to higher percentages of upper Columbia summer/fall stock and at all sites and mid and upper Columbia spring stock at Campbell Slough(Figure 135b).

Differences in stock composition at trend sites from 2008-2019 were analyzed using non-metric multidimensional scaling and ANOSIM. The NMDS plot (Figure 136) shows similarities in stock composition at Welch and Whites Islands and separation of Campbell Slough, Franz Lake, and Ilwaco Slough. ANOSIM analysis indicates a significant difference in stock composition among sites (global $R = 0.387$, $P = 0.001$), however pairwise tests show there is no difference in stock composition at Welch and Whites Islands or between Franz Lake and Ilwaco Slough ($P = 0.66$ and $P = 0.09$, respectively).The lack of difference in stock composition between Franz Lake and Ilwaco Slough is likely due to small sample sizes at both locations. The similarities between Welch and Whites Island is driven by West Cascade fall stock. The separation between Welch and Whites Islands and other sites is driven by upper Columbia summer/fall and Spring Creek group fall stocks.

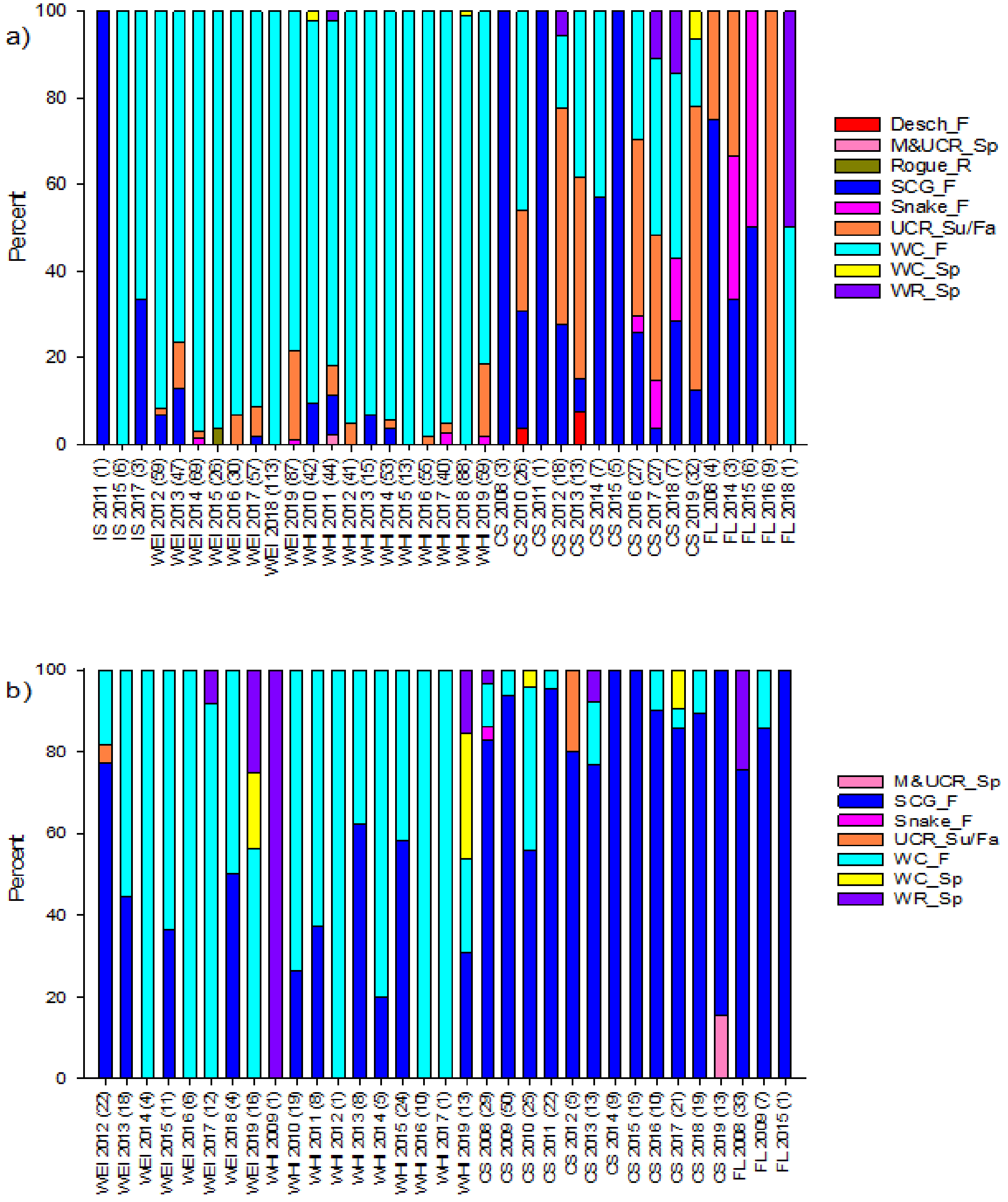


Figure 134. Genetic stock composition of a) unmarked and b) marked Chinook Salmon at trend sites from 2008–2019. Genetic sample sizes for each site is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake. Chinook salmon stocks: Desch_F = Deschutes River fall, M&UCR_Sp = mid and upper Columbia River spring, Rogue_R = Rogue River, SCG_F = Spring Creek Group fall, Snake_F = Snake River fall, Snake_Sp/Su = Snake River spring/summer, UCR_Su/Fa = Upper Columbia River summer/fall, WC_F = West Cascade fall, WC_Sp = West Cascade spring, WR_Sp =Willamette River Spring.

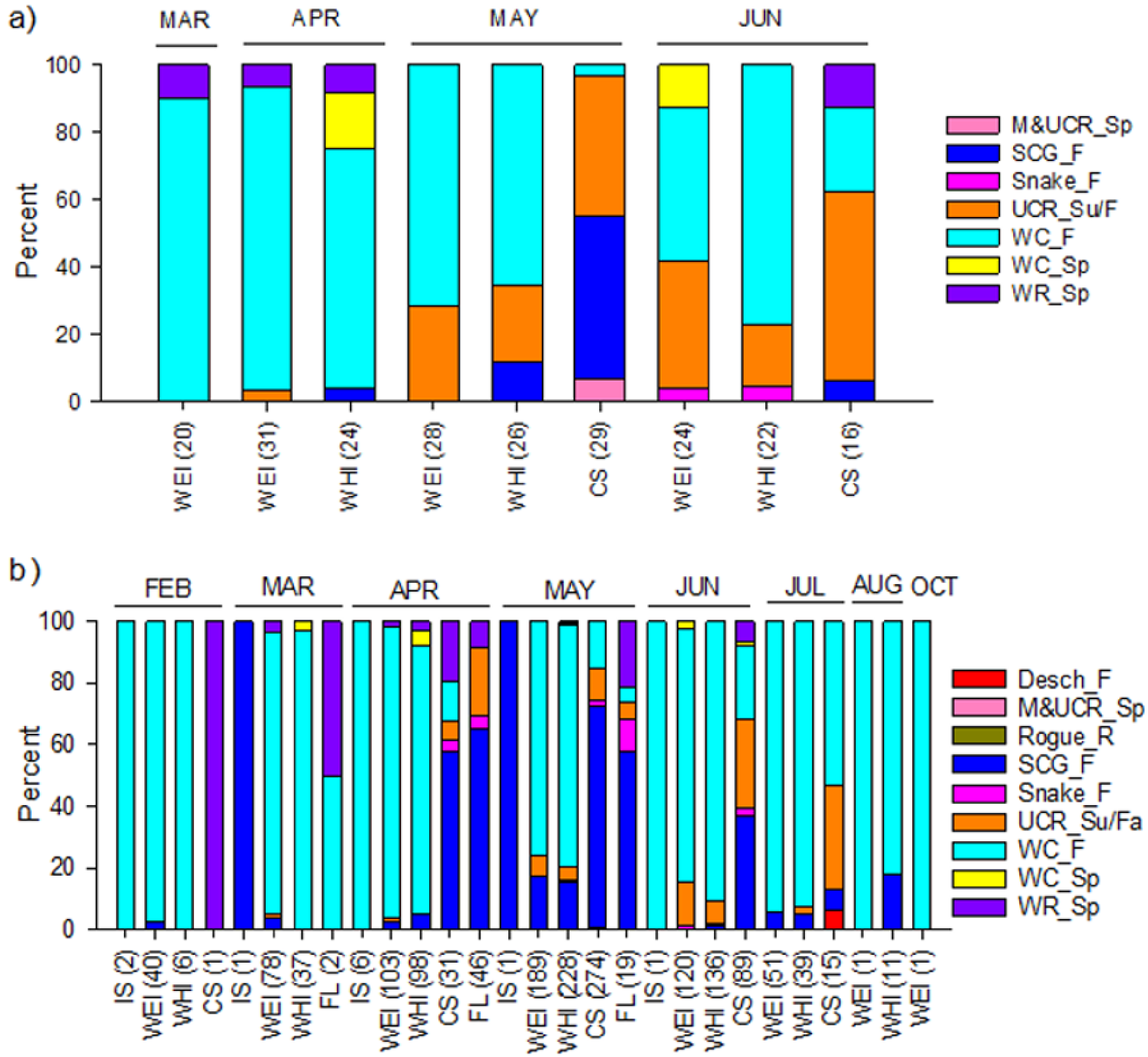


Figure 135. Seasonal percent stock composition per site for Chinook Salmon collected in a) 2019 and b) 2008–2019. Plots include both unmarked and marked Chinook Salmon. Genetic sample sizes for each site is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake. Chinook salmon stocks: Desch_F = Deschutes River fall, M&UCR_Sp = mid and upper Columbia River spring, Rogue_R = Rogue River, SCG_F = Spring Creek Group fall, Snake_F = Snake River fall, UCR_Su/Fa = Upper Columbia River summer/fall, WC_F = West Cascade fall, WC_Sp = West Cascade spring, WR_Sp =Willamette River Spring.

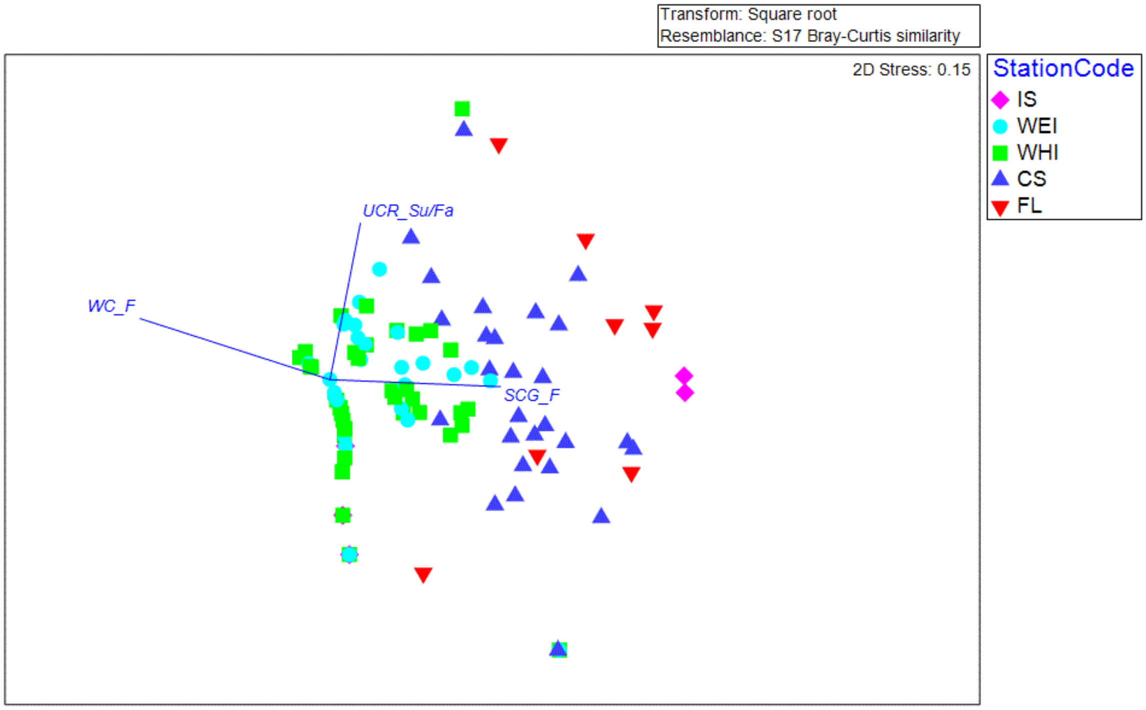


Figure 136. Nonmetric multidimensional scaling (NMDS) plot based on square-root transformed genetic stock abundance at five trend sites, 2008-2019. Significant correlation with variables (Pearson R > 0.4) are represented as blue vectors. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake. SCG_F = Spring Creek Group fall, UCR_Su/Fa = Upper Columbia River summer/fall, WC_F = West Cascade fall.

3.6.3.2 *Salmon Size and Condition*

Chinook salmon

Length, weight, and condition factor

Chinook salmon were caught at all sampled locations in 2019. In 2019, the average length, weight and condition factor for unmarked Chinook captured at Welch Island, Whites Island and Campbell Slough are presented in Table 39. No average length, weight and condition factor was calculated for Ilwaco due to no Chinook being sampled and no effective sampling was conducted at Franz Lake in 2019. The length, weight, and condition of unmarked Chinook salmon in 2019 showed similar patterns, with the largest fish typically captured at Campbell Slough (Figure 137**Error! Reference source not found.**). Within sites, there was some variation among years, though no clear increasing or decreasing trends. Unmarked Chinook sampled at Whites Island in 2019 appeared to have a slightly larger average weight than seen in 2018 and 2017. However, the condition appears to follow the overall trend on Whites Island over the past 5 years. Unmarked Chinook sampled at Welch Island in 2019, had a larger average fork length, weight and condition than previously observed in the 2018 sample. However, this is likely due to no June sampling at Welch Island in 2018, which would have likely contributed to the decreased averages of all measured indices.

In 2019, marked Chinook salmon were caught at three of the five sampled locations. Welch Island, Whites Island and Campbell Slough experienced significant enough catches to examine marked Chinook size and condition. The average length, weight and condition factor for marked Chinook captured are presented in Table 39. Campbell Slough shows little variation in length, weight, and condition across the past three sampled years (Figure 138). Similar to unmarked Chinook sampled at Welch Island in 2019, marked Chinook size, weight and condition were all greater than previous seen in the 2018 sampling season (Figure 138).

Table 39: Average length, weight and Fulton's Index (k) for unmarked and marked chinook in 2019

Site	Length (mm)	SD	Weight (g)	SD	Fulton's Index	SD
Welch Island						
Unmarked	54.67	2.19	2.69	0.65	0.93	0.02
Marked	80.15	4.27	6.2	5.12	1.04	0.02
Whites Island						
Unmarked	61.67	4.44	3.46	0.69	1.01	0.11
Marked	82.14	3.81	6.65	0.89	1.14	0.03
Campbell Slough						
Unmarked	63.13	2.64	3.46	0.43	1.05	0.03
Marked	85	5.27	7.65	1.77	0.99	0.01

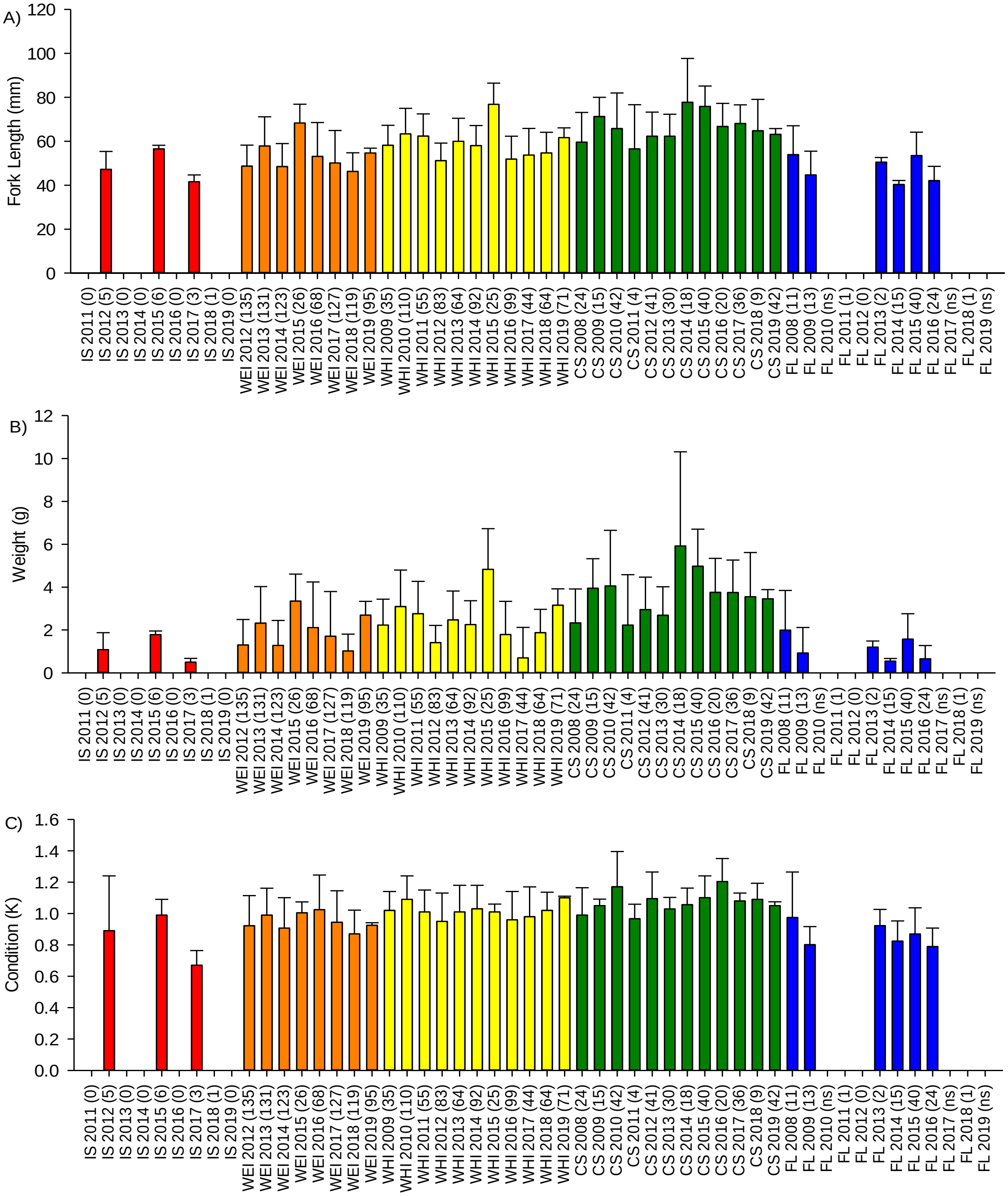


Figure 137. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of unmarked juvenile Chinook salmon at trends sites in 2019 as compared to previous years. Total number of Chinook salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

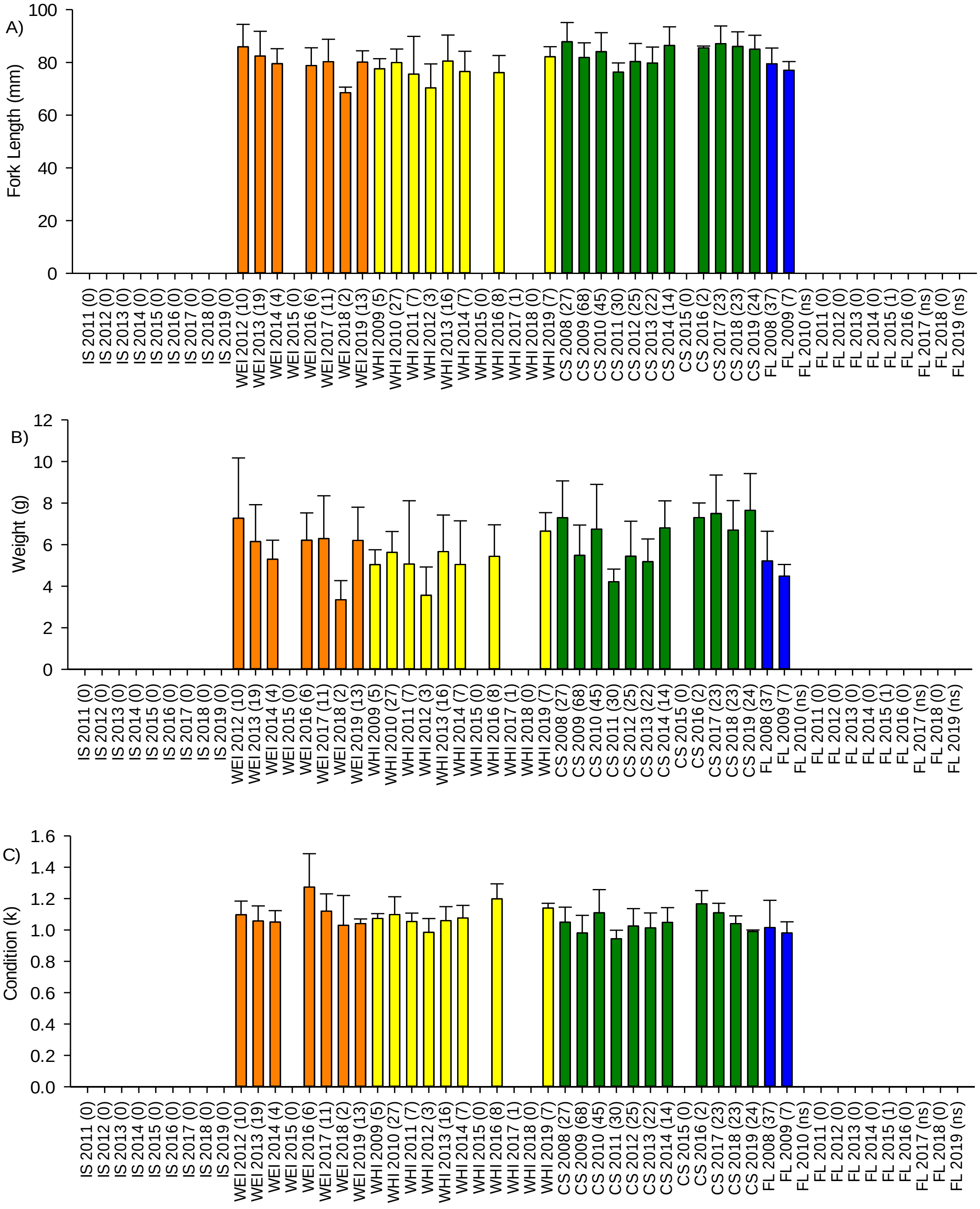


Figure 138: Mean (SD) a) length (mm), b) weight (g) and c) condition factor of marked Chinook salmon at trends sites in 2019 compared to previous sampling years. Total number of Chinook salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

Life History

At the trend sites in 2019, the majority of unmarked Chinook salmon were fry, 64%, 32% were fingerlings, and 4 % were yearlings (Figure 139a). Only at Welch Island did fry dominate catches, making up 77% of unmarked Chinook salmon. Campbell Slough and Whites Island showed a more even distribution of fry and fingerling of unmarked Chinook salmon. In comparison to previous years, the percentage of fry at all of the trend sites was slightly lower than in recent years.

A total of 37 (67 %) marked Chinook salmon caught at the trends sites in 2019 were fingerlings (Figure 139b). In comparison to previous sampling years, the proportion of yearlings encountered in 2019 was greater (18%). This does appear to differ from the overall trend; however, sample size is relatively low. Similar to all previous years, no fry marked Chinook were observed in 2019 at any of the sampled locations.

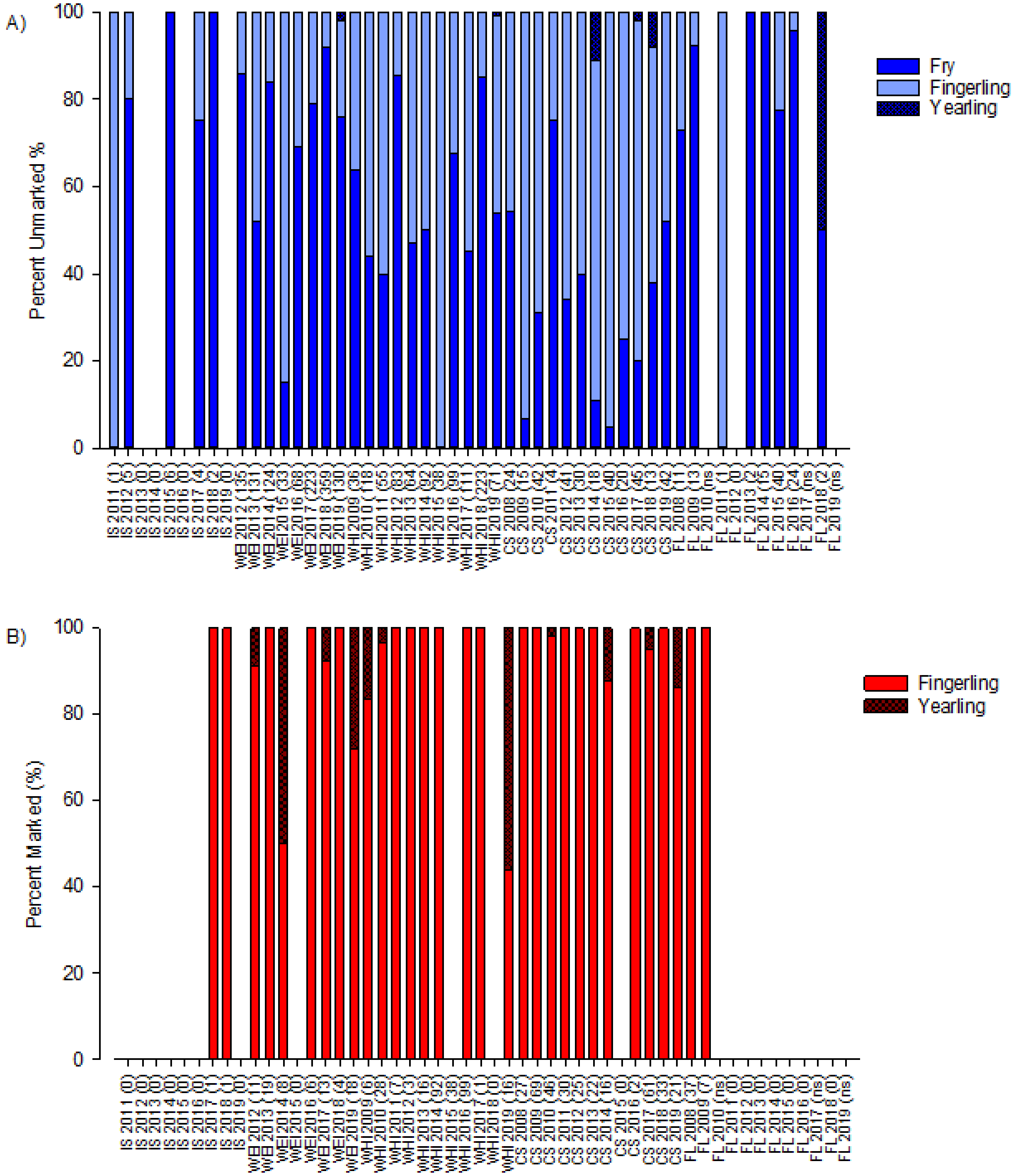


Figure 139. Percentages of life history types of a) unmarked and b) marked juvenile Chinook salmon captured at trends sites in 2019 and in previous sampling years. Total numbers of Chinook salmon captured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

Other salmon species

A total of 38 chum salmon were captured in 2019, 30 at Ilwaco Slough, 6 at Welch Island and 2 at Whites Island. All chum were caught between March and May. Unfortunately, no weights were taken on the 30 chum salmon caught at Ilwaco in 2019 due to the difficulties of measuring weight (g) in the field during adverse weather conditions. In 2019 the average length, weight, and condition factor of chum salmon captured at the following sampled sites were: Ilwaco Slough (± SD) 46.97 ± 1.08 mm, no average weight and condition index (k), Welch Island 45.63 ± 3.41 mm; 0.39 ± 0.14 g; and 0.72 ± 0.16, Whites Island no average fork length, weight and condition index (k) due to small sample size (Figure 140). The chum salmon collected in 2019 were comparable in size to those that have been collected in previous years, not especially large or small. Similarly, the mean 2019 value for condition factor was intermediate, between a high of 1.10 in 2008 and a low of 0.58 in 2013 and the same as 2017 (0.72). However 2018 and 2019 had very low sample sizes due to no weights taken in the field at Ilwaco Slough where the majority of chum salmon are caught each year. The largest fish, in terms of length and weight, are generally found at Whites Island and Campbell Slough, while condition factor tended to be highest at Campbell Slough. This site is located second farthest upstream of all current sampled sites. However, no chum salmon were captured at Campbell slough in 2019. Although chum salmon were captured sporadically, some variation by year is found at Ilwaco Slough, Welch Island, and Whites Island (Franz Lake is not considered in this comparison, as no chum salmon have been captured at the site since 2009).

Only one coho salmon was caught in 2019, an unmarked coho found at Whites Island in May. Franz Lake is the only site where coho salmon have been caught consistently enough to compare size measurements by sampling year. However, low sampling efforts in 2017, 2018 and no sampling in 2019 have made it difficult to determine any recent trends in coho at Franz Lake. Overall size and condition of unmarked and marked coho sampled at Franz Lake are shown below in Figure 141 and Figure 142.

Sockeye salmon and trout were not caught at any of the trends sites in 2019. Sockeye salmon were last sampled at Welch Island in 2014 and trout were last caught at Welch Island in 2013.

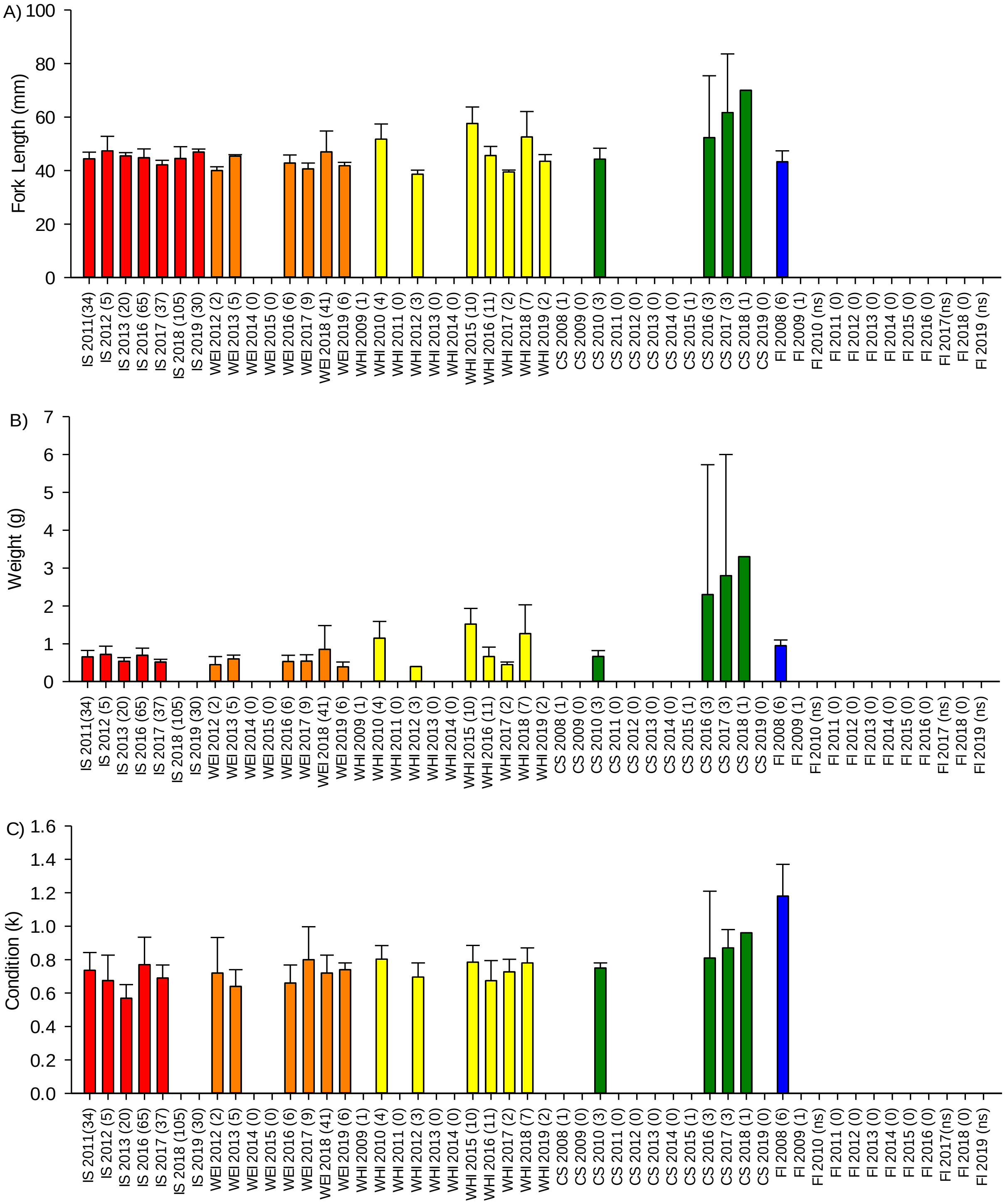


Figure 140. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of chum salmon at trends sites in 2019 compared to previous sampling years. In 2019, due to difficult weather conditions no chum weights were recorded at Ilwaco Slough (IS), therefore no Condition index (k) was calculated. Total number of chum salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

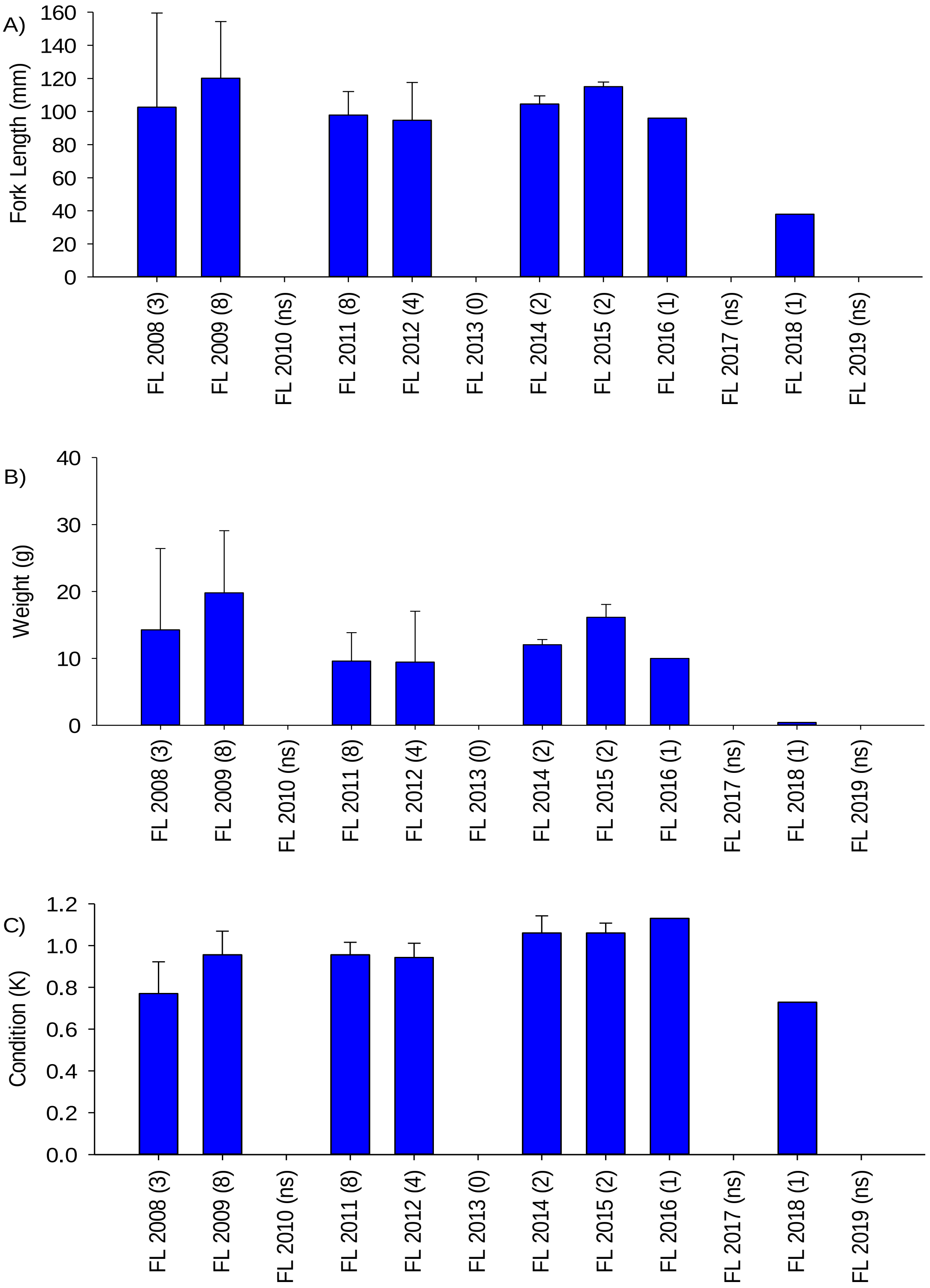


Figure 141. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of unmarked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses.

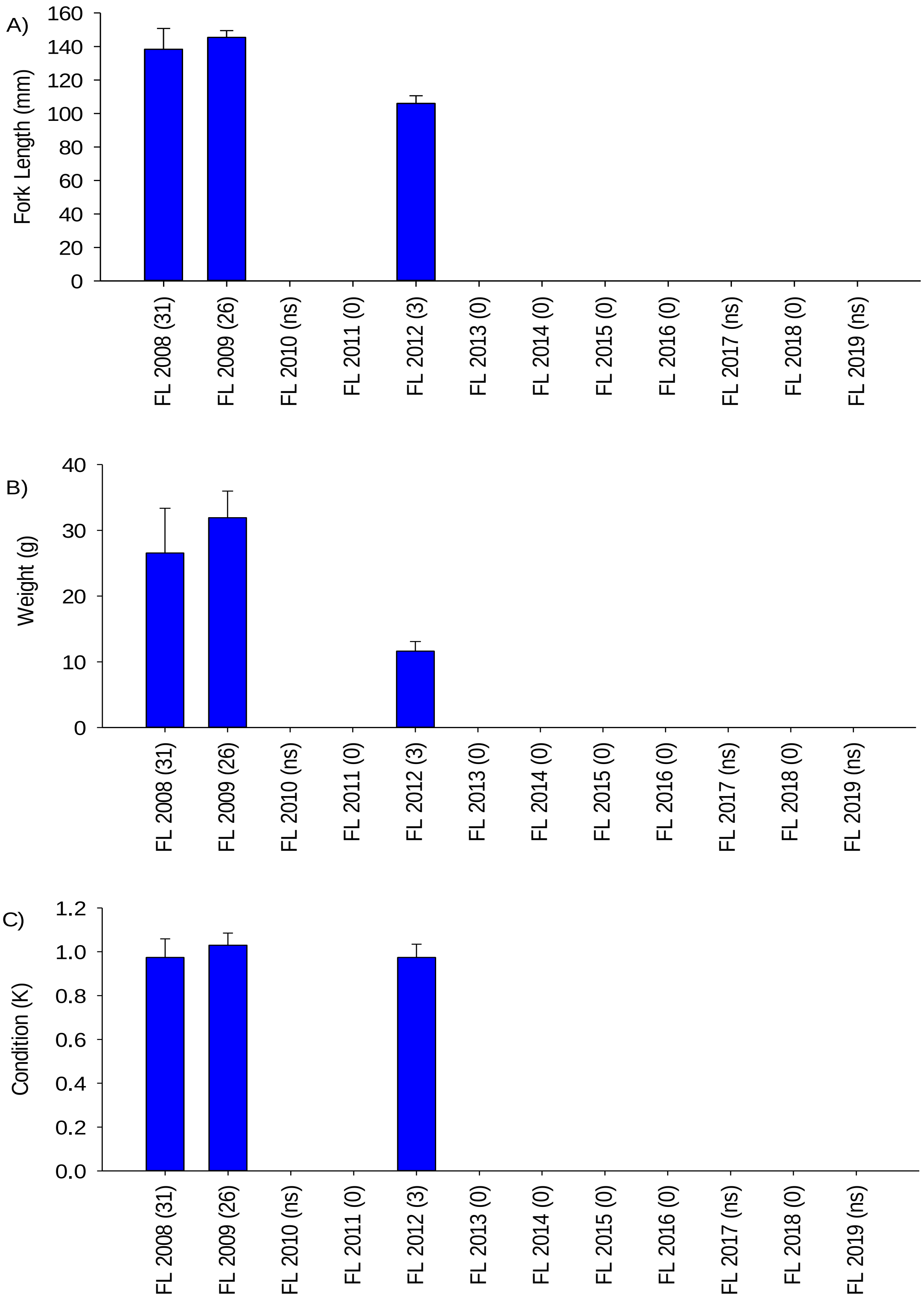


Figure 142. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of marked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses.

3.6.3.3 *Somatic Growth Analyses*

Otoliths are used to estimate somatic growth rates in fish. 2019 otolith data were still under analysis at the time of thos report. Patterns in somatic growth rate can represent variations in growth in response to genetic stocks or environmental conditions. Previous analyses of otolith data find that somatic growth rate in Chinook salmon ranged from 0.31 to 0.87 mm/day with an average of 0.54 mm/day (Chittaro et al. 2018). These analyses included sites representing mainstem and off-channel habitats that have been sampled as part of the Ecosystem Monitoring Program from 2005–2018. In 2018 we ran separate analyses for mainstem and off-channel habitat to better align with the reporting of trend site data. Sample sizes were large enough at Welch, Whites, and Campbell sites to accommodate site-specific analyses. Specific years used in each analysis is provided below. See Chittaro et al. 2018 for locations of non-trend site sampling locations.

Main stem: Fish (n=100) collected in main stem habitat were obtained across several years (2005, 2008, 2013) and sites (Beaver, Columbia city, confluence, Point Adams, & Warrendale). Our GLM analysis indicated that the model that best explained variability in growth rate (mm/day) only included discharge. Specifically, we detected a significant ($p < 0.05$) negative relationship between growth rate and discharge (Figure 143**Error! Reference source not found.**).

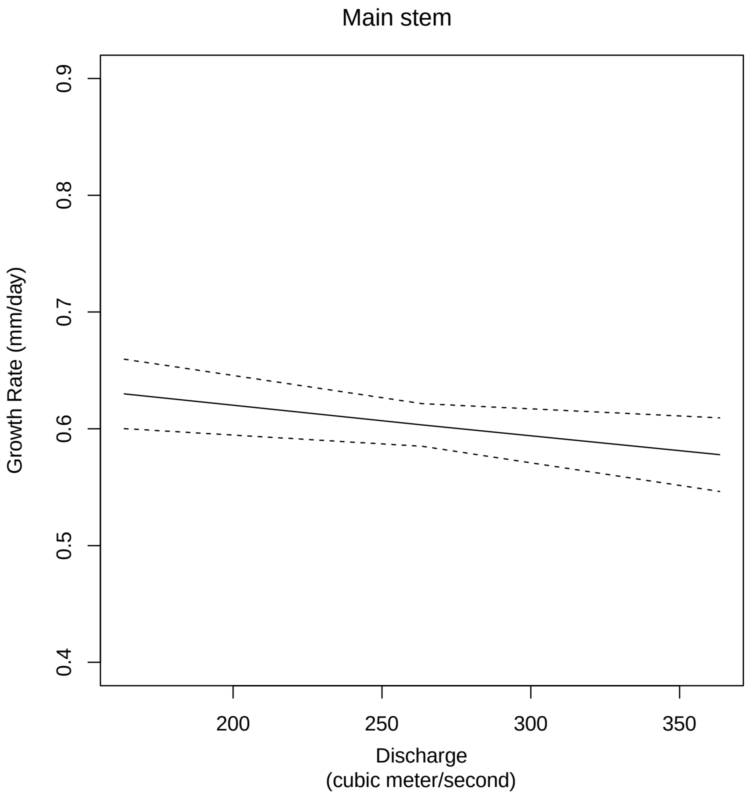


Figure 143. Relationship between growth rate and discharge for juvenile Chinook Salmon collected at mainstem Columbia River sites, 2005, 2008, and 2013.

Off-channel: Fish (n=652) collected in off-channel habitat were obtained across several years (2007-2014, 2016-2018) and sites (Beacon, Bradwood, Burke, Campbell, Deer, Franz, Goat, Jackson, Lemon, Lord/Walker, Mirror Lake 1 & 4, Pierce, Ryan, Sand, Secret, Wallace, Washougal, Welch, & Whites). Our GLM analysis indicated that the model that best explained variability in growth rate (mm/day) included year, river kilometer, stock, hatchery/unmarked, and fork length. Specifically, we observed a significant ($p< 0.01$) (curvilinear) positive relationship to river km and fork length. Also, fish collected in 2007 had significantly faster growth than those from 2011 and 2014 (Figure 144).

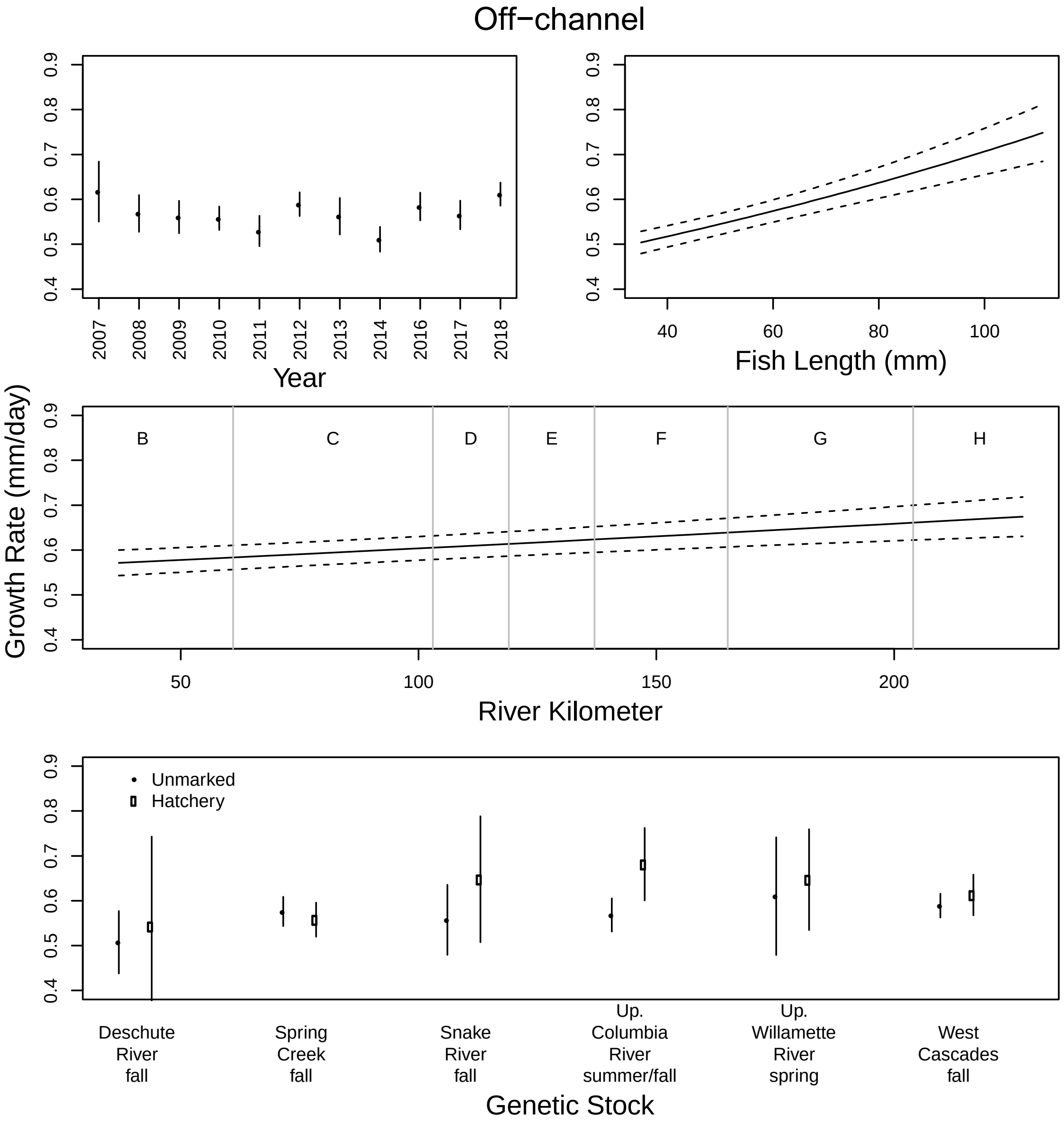


Figure 144. Relationships between growth rate and a suite of variables for juvenile Chinook Salmon collected at off-channel Columbia River sites, 2007–2014, and 2016–2018.

Welch Island: Fish (n=141) collected from Welch Island were obtained across several years (2012-2014, 2016-2018). Our GLM analysis indicated that the model that best explained variability in growth rate (mm/day) consisted of year and fork length. Specifically, we detected a significant ($p < 0.05$) positive relationship between growth rate and fork length. Also, fish collected in 2014 had significantly ($p<0.01$) slower growth than those from 2012, 2016, & 2017 (Figure 145).

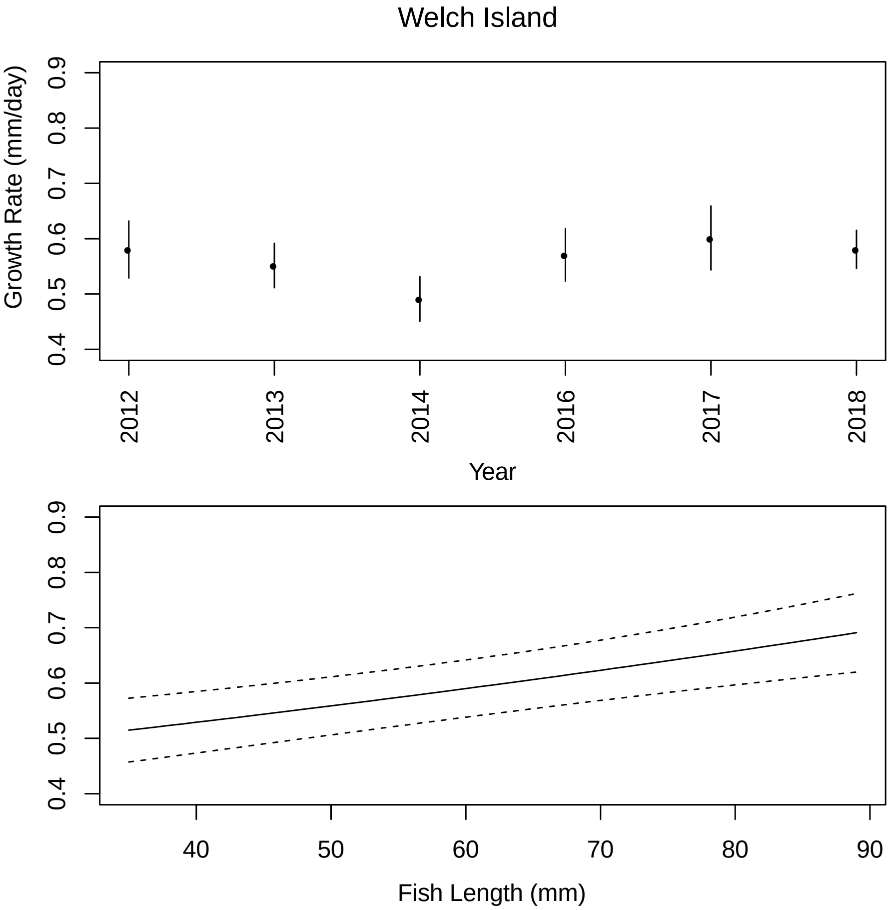


Figure 145. Relationships between growth rate and a suite of variables for juvenile Chinook Salmon collected at Welch Island, 2012–2014, and 2016–2018.

Whites Island: Fish (n=136) collected from Whites Island were obtained across several years (2009-2014, 2016-2018). Our GLM analysis indicated that the model that best explained variability in growth rate (mm/day) consisted of year and fork length. Specifically, we detected a significant ($p < 0.05$) positive relationship between growth rate and fork length. Also, fish collected in 2012, 2013, 2016, and 2018 had significantly ($p<0.05$) faster growth than those from 2014, and those from 2013 grew significantly faster than those from 2011, 2017 (Figure 146).

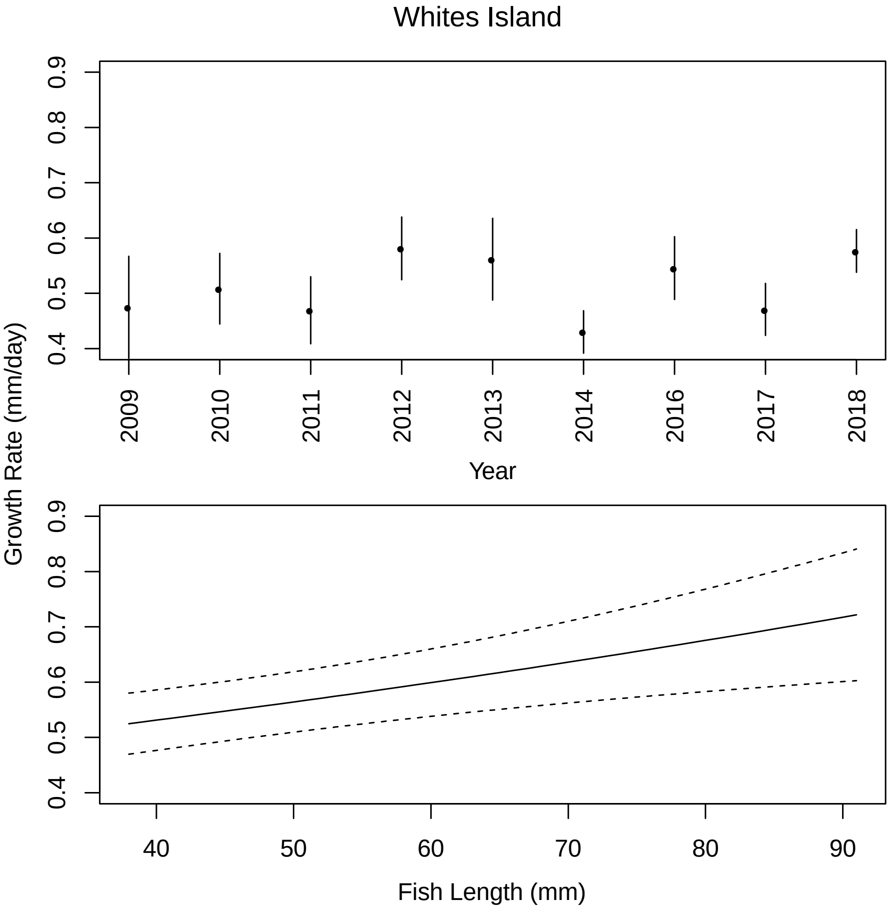


Figure 146. Relationships between growth rate and year and fish length for juvenile Chinook Salmon collected at Whites Island, 2009–2014, and 2016–2018.

Campbell Slough: Fish (n=112) collected from Campbell Slough were obtained across several years (2007-2014, 2016-2018). Our GLM analysis indicated that the model that best explained variability in growth rate (mm/day) consisted of fork length and the interaction between stock and hatchery/wild. Specifically, we detected a significant ($p< 0.05$) positive relationship between growth rate and fork length (Figure 147**Error! Reference source not found.**).

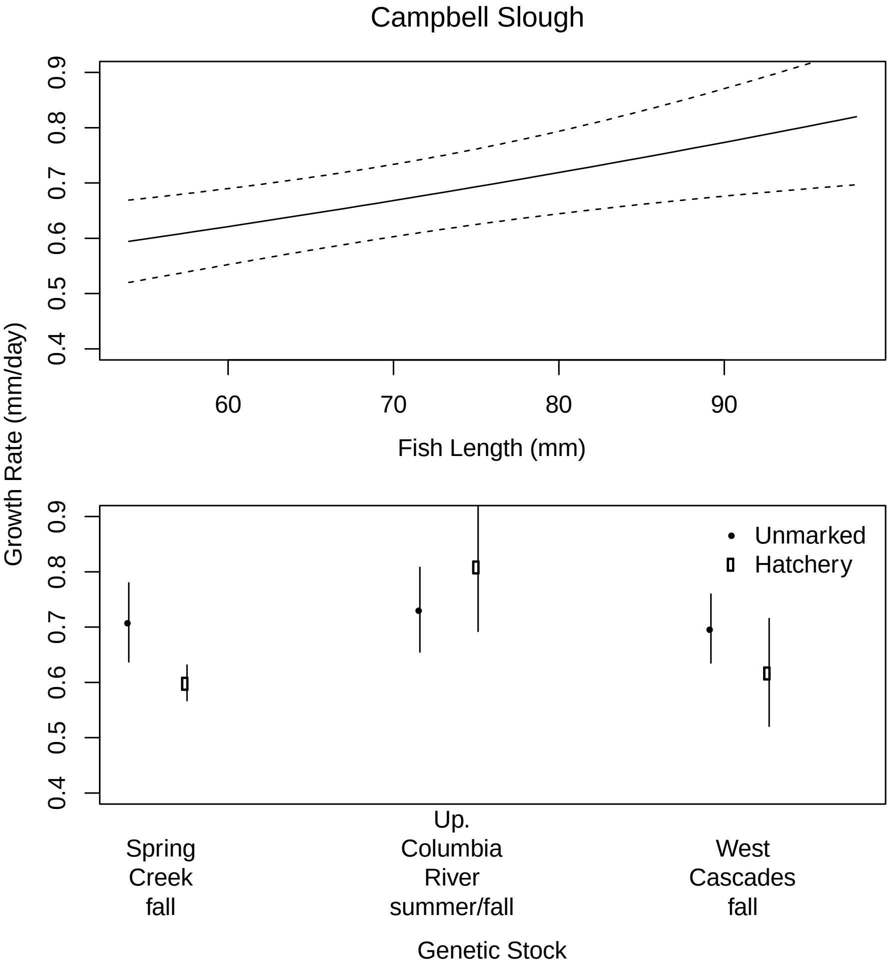


Figure 147. Relationships between growth rate and fish length and genetic stock for juvenile Chinook Salmon collected at Campbell Slough, 2007–2014, and 2016–2018.

3.6.3.4 *Lipid Content of Juvenile Chinook Salmon*

Lipid analyses for fish collected in 2019 has not been completed. The following are the results up to 2018. Lipid content can be a useful indicator of salmon health such that higher content is associated with improved survival (Biro et al. 2004). In this report we present data on percent lipid content and the percent of lipid present as triglycerides in juvenile Chinook salmon between 2007 and 2018 (Figure 148). Because these measures did not differ significantly between marked and unmarked fish, samples from both groups of fish were pooled, increasing sample size. Juvenile Chinook whole body samples collected in 2014 for lipid determinations were compromised due to a freezer failure, so no data are reported for that sampling year.

Overall, we observed considerable overlap in both percent lipid and triglycerides through space (i.e., among sites and within years) and time (i.e., among years and within sites) suggesting little change in the relative health of Chinook during this study. Some of the lowest percent lipid and triglyceride values were found in 2009-2013, which might indicate reductions in health during these years. In addition, median percent triglycerides in 2018 increased with respect to river kilometer, yet this pattern was not present in previous years.

Significant differences ($p < 0.05$) in percent lipid content among years were observed at each sampling site. Only at Welch Island did Tukey's post hoc pairwise comparisons indicate significant differences in percent lipids between 2018 and any of the previous sampling years. Specifically, fish collected in 2018 had significantly lower percent lipid content relative to those from 2015.

Significant differences ($p < 0.05$) in percent triglycerides were observed among years for Campbell Slough, Welch Island, and Whites Island, but not Franz Lake. At Welch Island, Tukey's post hoc pairwise comparisons indicated significantly lower percent triglycerides in fish collected in 2018 relative to those collected in 2013 and 2015. At Whites Island, percent triglycerides for fish collected in 2018 were significantly lower than those collected in 2013. In contrast, fish from Campbell Slough in 2018 had significantly higher percent triglycerides compared to the fish collected in 2007. To date, only one composite sample from Ilwaco Slough has been analyzed, so trends at this site cannot be evaluated.

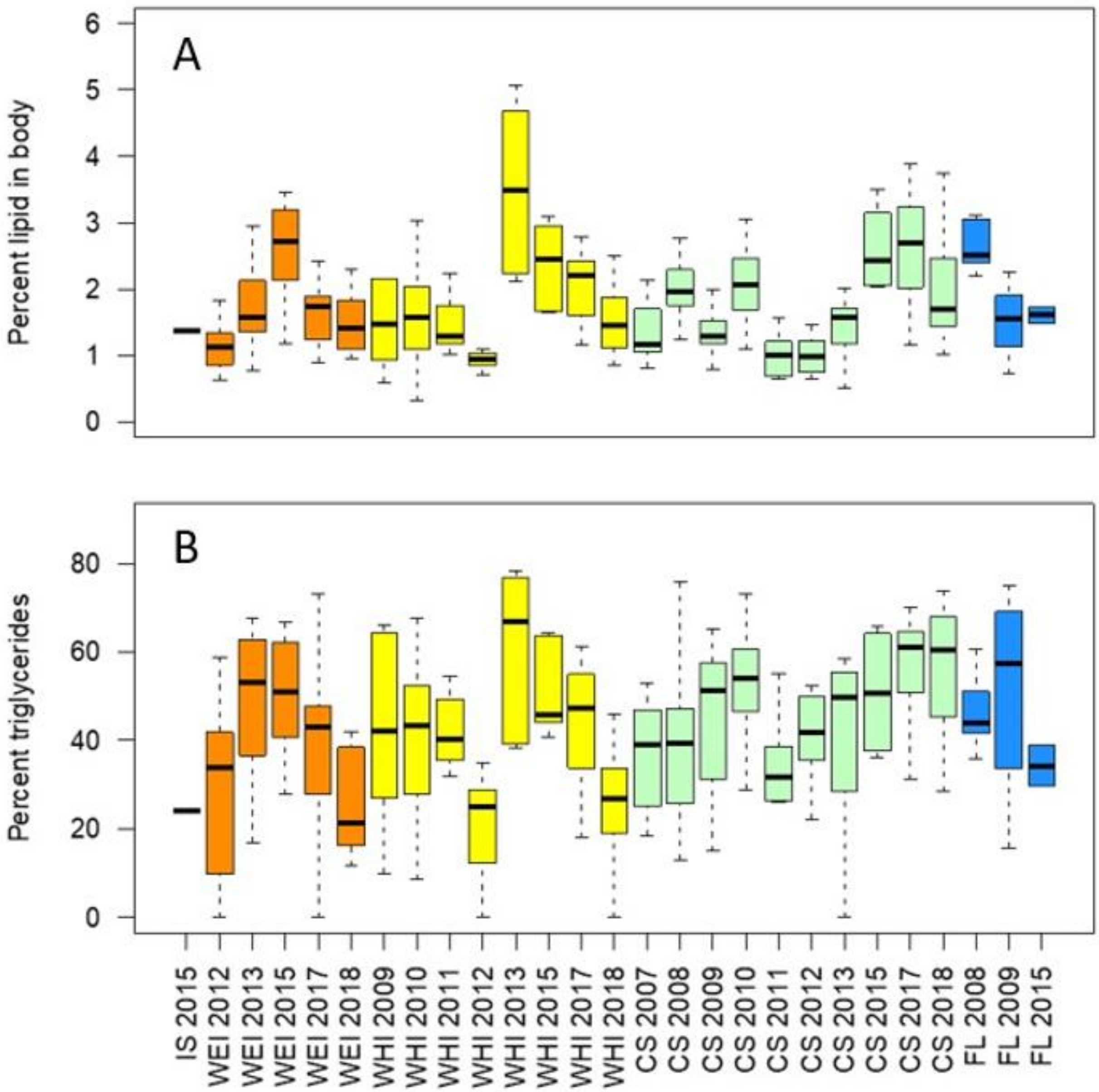


Figure 148. Percent lipid content (A) and percent total lipids that were triglycerides (B) determined in whole bodies of juvenile Chinook salmon collected from the trend sites in 2018 compared to previous sampling years. Unlike letters indicate 2018 values within each site that differ significantly from those determined in other years (Kruskal-Wallis, Tukey's post hoc test, $p < 0.05$). Sites are organized in increasing distance from the mouth of the Columbia River. Site abbreviations: IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.4 PIT-Tag Array Monitoring of Juvenile Salmon Residence

The PIT tag detection system at Campbell Slough collected data from March 20, 2019 to August 30, 2019. Forty-nine individual fish were detected from April 10–August 4. Eighty percent (N=39) of the fish detected were hatchery fall Chinook salmon with all but one originating from Spring Creek NFH located above Bonneville Dam at rkm 269. The other hatchery fall Chinook salmon originated from Willard Hatchery on the Little White Salmon River which enters the Columbia at rkm 261. Hatchery spring Chinook and northern pikeminnow (known predators of juvenile salmon) were almost equally represented at 3 and 2 individuals, respectively. Two of the spring Chinook originated from hatcheries in the middle Columbia River Basin, and one was released from Rapid River Hatchery in the Snake Basin. The two northern pikeminnow were released in the Columbia River between the Lewis River and Bonneville Dam in 2014 and 2016. Five detected tags were unknown (no tag data in the regional database; www.ptagis.org). Detection numbers and residence times are listed in Table 40**Error! Reference source not found.**.

Residence times were measured by quantifying elapsed time from first to last detection. It is important to note that reported residence times are merely estimates because we do not know if the fish remained in the vicinity undetected. Overall, salmon residence times at Campbell Slough were short with median residence times of 2 seconds for both spring and fall Chinook. Typically, we observe greater variation in the amount of time that fall Chinook reside in tidal side channels and sloughs, like Campbell Slough, but in 2019, the residence times for most fall Chinook were less than one minute. Only three of 39 fall Chinook resided in Campbell Slough for greater than a day, but all three had residence times of greater than two weeks (16, 18, and 20 days). Northern pikeminnow are also PIT tagged in the Columbia Basin as part of the Northern Pikeminnow Sport-Reward Program (www.pikeminnow.org). In Campbell Slough 2 individuals were detected and they demonstrated the longest residence times of all fish, with one fish residing for 46 days and the other for 91 days.

Table 40. Number and residence time (max and median) of fish detected at Campbell Slough PIT array in 2019. Residence time is a measure of elapsed time from first to last overall detection, not a measure of time spend upstream of the array.

		Residence time		
	N	Max	Median	Average
Fall Chinook	39	20.4 d	2 s	1.4 d
Spring Chinook	3	25 s	2 s	9 s
Northern Pikeminnow	2	90.8 d	46.4 d	46.4 d
Unknown	5	41.3 d	3.3 d	16.6 d

The Horsetail Creek PIT detection array collected data from March 20 to October 13, 2019. Although not all 10 antennas were operating, we had coverage of four antennas on both the downstream and upstream sides of the culvert.

Fifteen individual fish were detected between April 26 and August 14. Forty-seven percent (N=7) of fish detected were hatchery fall Chinook, one of which originated from the Snake River. All other fall Chinook salmon originated from the middle Columbia Basin. The second most prevalent category was hatchery spring Chinook at 27% (N=4). One hatchery Coho released in the Umatilla River was also detected. Additionally, two northern pikeminnows and one unknown (no tag data in the regional database; www.ptagis.org) fish were detected. Detection numbers and residence times are listed in Table 41.

Residence times were measured by quantifying elapsed time from first to the last detection. It is important to note that reported residence times are merely estimates because we do not know if the fish remained in the vicinity undetected. Residence times at Horsetail Creek were relatively short in 2019. Spring Chinook salmon had the longest median residence time of 11.7 h, however, the longest maximum residence time was observed in fall Chinook at 5.4 d. The single Coho detected at Horsetail Creek resided for 1.7 d. Two northern pikeminnow were detected at Horsetail Creek over a short period of time in early June.

Table 41: Number and residence time (max and median) of fish detected at Horsetail Creek PIT array in 2018. Residence time is a measure of elapsed time from first to last overall detection, not a measure of time spent upstream of the array. Numbers in parentheses represent the number of known wild origin fish in the total.

		Residence time		
	N	Max	Median	Average
Spring Chinook	4	1.07 d	11.7 h	12.3 h
Fall Chinook	7	5.4 d	37 s	18.5 h
Hatchery Coho	1	1.7 d	--	--
Northern Pikeminnow	3	2.2 d	1.1 d	1.1 d
Unknown	2	1.75 m	--	--

4 Status and Trends Discussion

4.1 Mainstem Conditions

The 2019 hydrograph resembled that of 2018, but winter flows were much lower. Together with values observed in 2010, winter flows in 2019 were the lowest of the 2011-2019 record. The freshet lasted from early April through late May, with peak flows that were above average in the early part of the freshet (when flows from the tributaries and Columbia were both considered); peak flow in early April was the highest observed during that period. There was a rapid change in flow between March and April in 2019, with a 2.5-fold increase in discharge at Bonneville dam over the span of a week. Once the freshet subsided by early June, river flows were again among the lowest of the 2011-2019 record and, similar to the case in 2018, the contribution of total flow by the Willamette River and other tributaries was low in 2019. The low-flow summer conditions observed in are consistent with hydrologic models that predict intensified late summer drought in the Pacific Northwest (Hamlet and Lettenmaier, 2007, Hamlet and Littell, 2012, Lutz et al., 2012) due to earlier snowpack melt (Cayan et al., 2001, Nayak et al., 2010, Stewart et al., 2005). Hydrologic changes are rooted in increased air temperatures observed throughout the Pacific Northwest (Littell et al., 2011), which has changed the size of annual snow packs (Hamlet et al., 2005, Mote, 2003). As a result of these changes, high water temperatures have been recorded in the lower Columbia over the last few years, particularly in 2015 when the atmosphere was very warm (Gentemann et al., 2016).

In 2019, river water temperatures peaked at 24^{o}C in August. While there was a similar number of days where temperatures exceeded thresholds of 19^{o}C, 20^{o}C, and 21^{o}C in 2019 compared to the previous few years (~80 and 62-65, 42-45, respectively), there were fewer days with temperatures exceeding higher thresholds shown to be deleterious to juvenile salmonids in 2019 compared to 2018 or 2017: whereas in 2018 there were ~25 day-equivalents with temperatures >22^{o}C, there were <10 days in 2019 that were this warm, and there were no day-equivalents with temperatures exceeding 23^{o}C, whereas there were a few days in each of 2018 and 2017 that were this warm. Climate change, generally manifested through warmer ocean temperatures over a sustained period of time, has been linked to shifts in survival, distribution, and biomass of marine organisms (Schwing et al., 2010, Doney et al., 2012, Chust et al., 2014, Cheung et al., 2015). In addition, recent work has shown that temperature strongly influences food consumption by juvenile salmonids, with consumption increasing during warm periods (Daly and Brodeur, 2016). This is significant since decreased survival of juvenile Chinook salmon has been linked to higher temperatures, which is thought to occur due to reduced food availability (Burke et al., 2013, Daly et al., 2013).

4.1 Abiotic Site Conditions

Water quality parameters determined at the five fixed trends sites revealed similar patterns to previous years (Sagar et al. 2016, Hansen et al. 2017). Ilwaco stands out for the low dissolved oxygen saturation values, which were below 100% (relative to equilibrium with the atmosphere) throughout the spring and summer. Franz Lake Slough also had low dissolved oxygen saturation levels in the summer, while the other sites had minimal observations of low dissolved oxygen levels in 2019. Temperatures were high in 2019 during the late spring and summer, similar to observations from 2015 and 2016; the number of day-equivalents greater than thresholds of relevance to salmonid growth and survival was higher in 2019 than in 2018 or 2017 in the summer. The high water temperatures following the spring freshet reflect the fact that aside from high flows occurring during the freshet, river flow was very low in 2019, making summer conditions more like those observed during dry years like 2015.

With the exception of Campbell Slough, pH values were in the target range for good water quality throughout the sampling season in 2019. The short-term fluctuations in pH occasionally fell outside the range for good water quality according to the Washington Department of Ecology's acceptable limits (7–8.5), but the daily averages were acceptable. In contrast, at Campbell Slough, pH values exceeded 8.5 in late June, and remained higher from June through September compared to the March-June period. Levels that exceed 9 lead to a shift in the speciation of ammonium, from its ionic form to the toxic gas, ammonia.

Similar to previous years, nitrate concentrations were highest early in the sampling season (i.e., March and April) at Welch Island, Whites Island, Campbell Slough in 2019. Following the period of spring growth of phytoplankton nitrate concentrations tend to decline, and fluxes of nitrate associated with the spring freshet are small; combined, levels of nitrate tend to be lower from May through the end of the summer. In 2019, the concentration of nitrate exceeded the recommended benchmark for good water quality (<0.399 mg L^{-1}, or 28.5 μM; Oregon's National Rivers and Streams Assessment 2008-2009) at all sites except Ilwaco Slough. High nitrate values exceeding benchmarks (>30 μm) were observed in March and April at Whites Island, Campbell Slough, and Franz Lake Slough; high values were observed at Welch Island in March only. Nitrate concentrations at all sites declined after April. However, at both Campbell Slough and Franz Lake Slough, phosphorus concentrations were high in the summer months (July, August); coincidentally, there were high proportional contributions of cyanobacteria to the total phytoplankton assemblage at these two sites, although the absolute cell densities were not as high as in previous years (e.g., 2018).

4.2 Habitat Structure

4.2.1 Hydrology and Sediment Dynamics

Marsh Hydrology

Hydrologic processes are the primary environmental driver dictating wetland sediment accretion and erosion dynamics, soil biogeochemistry, plant species assemblages, vegetation productivity, and overall wetland condition. Understanding hydrologic processes and variability across tidal wetland sites in the lower Columbia River is critical to informing conservation and restoration efforts throughout the estuary.

Similar to 2018, maximum flood levels in 2019 occurred across the upper river sites: Franz Lake, Campbell Slough, and Cunningham Lake in April during the first peak of the freshet in 2019. As we move down-river, these high flow conditions were less apparent during the period of the freshet at the mid-river sites: Whites Island, and Welch Island (Appendix C). The mid-river sites see maximum flood conditions in the winter when high king tides correspond with winter storm events. Due to sensor failure, no data was available for Ilwaco Slough in 2019. These trends were similar to those observed in 2017; however, the maximum flood levels occurred earlier in March-April in 2017 following the record-breaking freshet levels (Figure 5) that year.

In general, we have found that inter-annual variation in inundation patterns is much greater at the upper river sites, Franz Lake, Campbell Slough, and Cunningham Lake where seasonal flooding (winter and freshet) can result in months of continuous inundation during high-water years. In contrast, at the mid and lower estuary sites, Whites Island, Welch Island, and Ilwaco Slough, dominated by tidal patterns, inundation lasts just a few hours during high tide, but occurs frequently, usually one to two times daily. The average inundation daily, as measured by the average number of hours a day (converted to a %) the water surface level is above the marsh elevation, is a means of comparing sites to each other over time. The average inundation daily at each site is dependent on the elevation, the position along the tidal and riverine gradient, and the seasonal and annual hydrologic conditions. The average % of the day the mean marsh elevation is inundated for the month of August is critical for plant development in the upper river sites, as the freshet draws down and exposes the marsh surface. Generally, the trends in % time inundated identified in August correlate well with average % daily inundation for the year (unpublished data), it does not, however, always correlate with the overall magnitude of the annual freshet. This is because the timing of the freshet can vary from year to year, in some years like 2011 and 2012 the high flows from the freshet have lasted into August, resulting in significantly greater daily inundation patterns at the upper river sites (Figure 29-Figure 34), while other years like 2017 freshet levels were high but receded quickly resulting in low inundation levels in August and generally more of the growing season. These shifts in daily inundation are critical for plant community development and can have major implications for not only plant species composition by also biomass production. Lower water years (in August) such as 2015, 2017 (and it appears 2018), producing greater plant biomass than high August water years (Figure 54). We hypothesize that the annual timing, magnitude, and duration of the freshet may also impact the long-term status and trends of tidal wetland fish utilization, macroinvertebrate assemblages, and plankton productivity.

Sediment Dynamics

Sediment accretion rates are variable within the Columbia River estuary and within individual sites, likely due to variation in elevation, sediment loading, and flood inundation frequency (Kadlec & Robbins, 1984, Chmura et al., 2003, Woods & Kennedy, 2011) and may even be affected by the vegetation present (Larsen et al., 2010; Mudd et al., 2010 , Marani et al., 2013). In 2019, sediment accretion and erosion rates at the five trend sites and Cunningham lake ranged between -0.4cm and 2.3cm per year. The greatest sediment accretion rates have been measured at the Whites Island site in a patch of *C. lyngbyei* located at a mid- to low-marsh elevation (2.46 m, NAVD88) very close to the primary tidal channel at the site (<10 m from marsh edge). This is a good example of conditions conducive to high accretion rates: proximity to the tidal channel, high inundation frequency (about 50 percent), and vegetation that produces high amounts of organic material and effectively traps mineral and organic material, both important sources of sediment accretion in tidal freshwater marshes (Neubauer 2008).

Erosion rates at Campbell slough and Cunningham lake can be attributed to constant cattle grazing trampling, which affects soil compaction and removal of above ground biomass (Trimble, 1994; Nolte et al., 2013). In 2019, large variation in accretion and erosion rates were observed at all five trend sites and Cunningham lake. The 2017 and 2018 sediment accretion and erosion data for the EMP were included with the 2019 sediment dataset, and were included into the long-term dataset. This inclusion is one possible cause of change in trends observed in Table 20. Similar observations were noticed in a study researchers in the marshes along Gulf of Mexico (Callaway et al., 1997).

When combining long-term sediment accretion rates across sites we found that marsh elevation (m, CRD) was negatively correlated with sediment accretion rates, meaning that low-marsh zones accrete more sediment than high marsh zones. This pattern of sediment accretion is well supported by other studies (Harrison & Bloom, 1977; Cahoon et al., 1996), locally, Kidd (2017) found similar patterns within wetland sites in Young Bay. The mechanism diving these observations are the differences in daily and seasonal tidal ranges that can manifest in differences in sediment loading across the marsh elevation gradient. Sediment depositions being more pronounced in low marshes near marsh channels, which receive more daily inundation and sediment exposure than high marsh zones (Hassan et al., 2005; Larsen et al., 2010; Jay et al., 2015).

While we have found a significant trend in sediment accretion across the EMP sites, in general our study of sediment dynamics at the trend sites is has limitations. Firstly, the overall lack of sufficient sedimentation stakes prevents us from making definitive connections with inundation and flooding frequency, effects of vegetation of accretion rates as well as studying the influence of cumulative discharge. Secondly, Table 20 shows standard deviation much greater than the mean of accretion at all the trend sites and Cunningham lake, suggesting low reliability of the dataset.

There are still several questions that need to be answered. The interplay of mineral sediment accretion and the accumulation of organic material is important in determining the rates of sediment accretion and also the rates of carbon sequestration (Craft 2007). In Tidal Freshwater marshes, carbon accumulation in the sediment comes from organic material associated with mineral sediments in the water column and from in situ biomass production and breakdown (Neubauer 2008). Similar to sediment accretion variability, carbon density and accumulation rates are likely variables in the Tidal Freshwater marshes of the LCRE. Carbon density is often greater at higher marsh elevations with lower flooding frequency and lower sediment loading; however, the inverse may be true of carbon accumulation rates (Chmura et al., 2003). Overall, in LCRE marshes, carbon in the surface sediments (~10 cm) accounts for approximately 3 to 10 percent of the sediment (Borde et al., 2011; Sagar et al., 2013). This carbon content is similar to those amounts found in a prograding riverine brackish marsh with high mineral sediment accretion rates (Thom 1992), but lower than some other Tidal Freshwater marsh sediments (Craft 2007; Thom 1992) where organic material may account for more of the accretion. In general, Tidal Freshwater wetlands store more carbon and have higher carbon accumulation rates than salt marshes (Craft 2007), but understanding the variability of this process in the LCRE will be important to gain a better understanding of the overall storage capacity of these wetlands now and in the future.

In the future, it may be informative to relate site hydrology and sediment dynamics between and among sites. This effort may require more detailed tracking of sediment accretion and erosion rates within and across sites due to the high level of variability seen in the historic data and generally inherent to monitoring sediment dynamics (Takekawa et al. 2010).

4.2.2 Vegetation Community Condition and Dynamics

Overall trends in plant community composition

Overall, 2019 total plant cover was relatively stable across Ilwaco Slough, Welch Island, Whites Island, and Franz Lake compared to levels observed in 2018 and 2017. Cunningham Lake total cover has continued to increase between 2017 and 2019, rebounding from the heavy cattle grazing observed in 2017, and Campbell Slough has exhibited a small increase in total cover levels from 2018 to 2019, however the overall cover at Campbell is still low compared to non-grazed conditions. Cattle grazing has continued at Campbell Slough since 2017, with fencing efforts failing to keep the cattle out of the wetland both in 2018 and 2019.

Generally, native and non-native cover are more similar from year to year in the zone 1 and 2 sites (Ilwaco, Welch, Whites) compared to the zone 4 and 5 sites (Cunningham, Campbell, and Franz) (Figure 42 & Figure 43), this is likely due to the general hydrology of these sites, inundation patterns being much more stable from year to year in the tidally drive lower river, zone 1 and 2, sites compared to the fluvially dominated mid and upper river, zone 4 and 5, sites (see section 3.3.1). These trends were generally observed in 2019, with Whites and Welch Islands retaining similar native cover conditions as to previous years. Franz Lake and Ilwaco Slough, however, did both see an approximately 20% increase in native cover compared to 2018. At Ilwaco Slough this was account for through a general reduction in non-native *Agrostis stolonifera* cover and a corresponding increase in native *Carex lyngbyei* cover, in addition to increases in other natives such as *Argentina egedii ssp. Egedii*, *Deschampsia cespitosa*, *Lilaeopsis occidentalis*, and *Symphyotrichum subspicatum*. At Franz Lake, this shift was a result of a general mixed increase in native species cover including *Argentina egedii ssp. Egedii*, *Fraxinus latifolia*, *Salix lucida*, and *Helenium autumnale*.

Comparatively, Campbell Slough also showed a 27% increase in native relative cover (Table 24, Figure 42 & Figure 43). This shift can be accounted for by an increase in native herbs such as *Eleocharis ovata*, *Helenium autumnale*, *Lindernia dubia*, and *Ludwigia palustris* which were found growing in the plots heavily disturbed by grazing. This shift, caused by grazing, indicates that these native species are found in the seed bank but are normally (under no grazing) suppressed by more dominate non-native species such as *P. arundinacea* (Kidd 2015). Comparatively, Cunningham lake, which has not experienced heavy grazing since 2017, had a decrease in native cover from 66 in 2018 to 58% in 2019 (Table 24, Figure 42 & Figure 43).This decrease in native cover was accompanied by a general increase in non-native cover including at 22% increase in *P.arundinacea* cover between 2018 and 2019 (Table 21). This increase in *P.arundinacea* is to be expected both because of the reduced grazing pressure and because of the lower water conditions experienced during the 2019 growing season, which favors *P. arundinacea* growth (see more on this below).

Between 2012-2019 the six most common plant species identified throughout the tidal estuary (across the 6 trend sites) in order of overall abundance are *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Carex lyngbyei* (CALY, native), lyngby sedge, *Eleocharis palustris* (ELPA, native), common spikerush, *Sagittaria latifolia* (SALA, native), wapato, *Leersia oryzoides* (LEOR, native), rice cut grass, and *Ludwigia palustris* (LUPA, native), water purslane (Table 25, Figure 44 & Figure 45). While these species are the most common and abundant across all sites over the years, they are not necessarily present at all sites every year. For example, *P. arundinacea* does not grow at Ilwaco, likely due to the saline conditions present at this wetland (Kidd 2017). However it is found growing in abundance at all the other trend sites across the lower river (Table 26 & Table 27, Figure 44).

Trends in P. arundinacea abundance

In 2019, *P. arundinacea* cover levels stayed relatively consistent to those observed in 2018, however, at Cunningham, there was a significant increase in *P. arundinacea* levels from 17 to 38% and Franz Lake also experience a small increase from 8.7 to 13.1% (Table 26 & Table 27, Figure 44). *P. arundinacea* frequency (spread across the site) also increased at Cunningham, but only slightly from 75 to 79 plots, and overall *P. arundinacea* frequency dropped at Franz Lake from 70 to 60 plots (Table 26). This shift in *P. arundinacea* levels observed at Cunningham and Franz Lake is likely a product of both low freshet flooding conditions in 2019 (Figure 49) and, at Cunningham Lake, a break from grazing pressure. The last several years cattle have heavily grazed Cunningham Lake wetlands, however reduced grazing activity was observed in 2019. It is well known that cattle pressure can significantly reduce *P. arundinacea* abundance during the growing season (Kidd 2015). Generally, *P. arundinacea* abundance has been found to decrease in years of greater freshet discharge levels, especially in Cunningham Slough, Campbell Slough, and Franz Lake where wetland water levels are tightly correlated with Columbia River discharge conditions, higher water levels making growing conditions less favorable for *P. arundinacea* (Figure 49).

Water year conditions and impacts of plant community composition

In 2019, data continued to support our findings that annual shifts in *P. arundinacea* cover are strongly correlated with Columbia River discharge levels and site water levels during the growing season (Figure 48), with lower water levels (and lower discharge levels) favoring *P. arundinacea* growth and observed abundance. These findings indicate that annual flooding conditions within sites (% daily inundation) and across the river (freshet accumulated discharge) are important mechanisms driving much of the observed annual variability in *P. arundinacea* dominance across the estuary. Additionally, these data support the hypothesis that annual flooding conditions in the Columbia can dramatically impact year to year shifts in plant community dynamics, especially the non-native species *P. arundinacea* in the upper river sites. *P. arundinacea* mean annual cover was also found to be tightly negatively correlated with native plant community cover across all river zones except the mouth (Ilwaco has no *P. arundinacea* due to high salinity levels), annual increases in *P. arundinacea* resulting in an overall decrease in native plant cover (Figure 50).

The long-term trends in the abundance of native species *C. lyngbeyi, S. latifolia, P. amphibium* have also been found to be strongly (and significantly) linked to annual river discharge conditions. Generally, *C. lyngbeyi* abundance has been found to increase in years of greater freshet discharge levels, especially in Ilwaco Slough, where salinity levels are reduced during large discharge years, making growing conditions more favorable for *C. lyngbeyi* (Figure 45). *S. latifolia* has been found to have a delayed reaction to freshet conditions, with lower freshet conditions resulting in an increase in *S. latifolia* abundance the following year. Given this strong reaction to lower freshet conditions it is anticipated that *S. latifolia* levels will increase in 2020 in reaction to the low freshet conditions which occurred in 2019 (lower freshet condition result in lower wetland water levels, which expose more low marsh areas to light and air during the growing season). Additionally, *P. amphibium* levels at Franz lake have also be found to follow a similar trend to *S. latifolia* with a one year delayed reaction to decreased freshet conditions, lower freshet conditions (lower water levels across the wetland site) resulting in an increase in *P. amphibium* cover the following growing year (Figure 43). For both species, this might be a result of increased rhizome stores from positive growing conditions (low water levels), providing for more robust growth in the following growing season.

Summarizing these findings, site level daily inundation patterns in addition to season freshet flooding conditions are important drivers of native and non-native plant communities across the estuary. Publication of these data and further investigations of these relationships will be explored in the FY21 synthesis report.

4.3 Food Web

4.3.1 Primary Production

4.3.1.1 *Emergent Wetland Vegetation*

Net aboveground primary productivity (NAPP) is the rate of storage of organic matter in aboveground plant tissues exceeding the respiratory use by the plants during the period of measurement (Odum 1971). Many methods exist to estimate NAPP; however, for our ecosystems in which there is a clear seasonality, a good method is a single harvest at peak biomass (Sala and Austin 2000). Our analysis of the proportion of live versus dead material indicated that for most species the live proportion of the summer samples averaged greater than 90 percent; a confirmation that we indeed were sampling at or near the biomass peak. Starting in the summer of 2017 detritus sampling was included in the biomass sampling and analysis to evaluate detrital production and export. In the winter of 2018 (and all sampling events to follow), biomass sampling protocols changed slightly to accommodate detrital sampling and streamline data collection. This included shifting from “strata” mixed species designations to simple high and low marsh strata descriptions across all sites sampled. This change has also included species biomass weights to be recorded individually to assess species specific contributions to each high and low marsh stratum (in the past mixes of species were assessed together). In general, these changes will allow for a more detailed understanding of species-specific biomass contributions and still allow for long-term comparisons to overall site, high and low marsh contributions.

Generally, productivity in the high marsh strata has been very high and similar in quantity to the most productive North American marshes (Brinson et al. 1981; Bernard et al. 1988; Windham 2001). Average summer biomass of 1000 to 1500 g dry weight/m^2 in the high marsh strata is not an uncommon observation throughout the estuary (Kidd et al. 2018). In 2018, the highest average summer biomass was observed at Whites Island high marsh strata, with 1,458 g/m^2, however, the multi-year analysis of the summer biomass revealed high variability between years at Whites Island. Across sample sites, year to year variability in overall total biomass contribution was found to be negatively correlated with cumulative river discharge for August, indicating the importance of river conditions on annual wetland biomass production and export, even at the lower river wetland locations, Whites and Welch Island (Figure 55).

Overall proportion of biomass contribution from living, dead, and detritus varied across the seasons, living biomass contributing the most during the summer season, standing dead and detritus contributing the most during the winter, with biomass contributions being more evenly split between living, dead, and detritus in the spring, reflecting new spring plant growth across all sampled sites (Table 30, Table 31). This seasonal look at biomass composition shows the largest flux of standing biomass (living + dead) out of these wetlands is between the summer and winter time-period, some of this living and dead biomass shifting to detrital material and most being exported from the sampling areas altogether. The largest flux of detritus out of the wetland occurs during the spring-summer time-period, detrital material showing a gradual increase from summer to spring and then a sharp decline between the spring and summer sampling events (Table 30, Figure 51). While the overall amount of biomass contributed is lower coming out of the low marsh compared to the high marsh strata, they were found to follow similar patterns of living, dead, and detritus biomass contribution over the seasonal shifts.

The EMP biomass sampling efforts continue to highlight the significant organic plant material contribution from these wetland sites to the estuary ecosystem annually; however, this contribution relative to the energy needs of the estuary food web is unknown. Overall, across sites the high marsh strata dominated by a mix of native sedge *C. lyngbyei*, native herb *P. amphibium*, and the non-native grass *P. arundinacea* contributed the highest and most consistent amount of organic material, signifying the importance of the high marsh plant community complex to the estuary food web. The low marsh strata dominated by a mix of native *P. hydropiperoides, S. latifolia, and E. palustris* also contributes a consistent flux of organic material, while much lower in overall biomass weight, these low marsh contributions are generally less variable than the high marsh on a site to site and year to year basis. If organic material from marsh plants is indeed a limiting factor for the detrital based food web in the lower river, the restoration of additional marsh area dominated by native high and low marsh species could improve those conditions.

4.3.1.2 *Emergent Wetland Vegetation Nutrient Dynamics*

One factor new to the EMP biomass analysis in 2018 was the evaluation of the living above ground biomass, detritus, and soil nutrients Carbon (C), Nitrogen (N), and ADF lignin (L, lignin) composition. These data provide insight into the quality and nutrient dynamics of the biomass contributions and highlight the variability in nutrient composition between plant species and the high and low marsh strata. Species-specific functional plant traits such as C:N ratio and L:N ratio can also provide insight into the potential decomposition rates of species, with low C:N and L:N ratio species having greater decomposition potential than species with higher C:N and L:N ratios. The C:N ratio is commonly used to define the N immobilization-mineralization gradient, a greater C:N ratio promoting N up take by microbes (immobilization) and detrital accumulation, while a lower C:N ratio promotes N mineralization (release) by microbes and detrital decomposition (Reddy and Delaune 2008). The quality and rate of decomposition provides insight into the direct food web contributions provided by different species found in the high and low marsh stratas across wetlands.

Nutrient Conditions Observed Across Strata

Comparing C, N, and C:N ratios between the above ground living biomass, detritus, and soil across the elevation gradient can provide insight into plant species nutrient use efficiency and decomposition.

Trends in C, N, and C:N ratios across the elevation gradient within wetlands were particularly interesting with living above ground biomass and soil C content both increasing along the elevation gradient; low in the low marsh strata and high in the high marsh strata. Soil N followed a similar pattern being higher in the high marsh strata and lower in the low marsh strata. Living above ground biomass N content followed a reverse trend with lower N levels in the high marsh strata and higher levels in the lower marsh. These results generally translated into greater C:N ratios in the high marsh soil and living above ground biomass and lower C:N ratios in the low marsh soil and living above ground biomass. These results potentially reflect both a shift in plant species and plant species nutrient use efficiency along the high to low marsh gradient. The low marsh species having lower carbon content, and lower C:N ratios overall, indicating less decomposition time required for the plant species found in the low marsh zone, C:N Ratio under 25 indicating no N limitation to decomposition (Wang et al. 2016). The high to low marsh shift in C:N ratios also corresponds to the overall differences found in detritus accumulation between the high and low marsh zone across sites, less detritus accumulation occurring in the low marsh zone (Figure 52). L:N ratios across the wetlands were found to also correlate with elevation, following the N content trend, with smaller ratios in the lower marsh zones across sites (Figure 65). The above ground living biomass L:N ratio is also known as a good predictor of plant biomass decomposition rates, smaller ratios indicate more N and less lignin, and quicker decomposition (Taylor et al. 1989, Talbot et al. 2011).

Overall, mean summer lignin content was greatest in the detritus samples compared to the living plant biomass (Figure 63, Table 35), this follows the expected trend of lignin concentrations increasing in the detritus as decomposition occurs, lignin and associated compounds resisting decomposition (Taylor et al. 1989, Talbot et al. 2011). Detrital lignin content was found to be positively correlated with detrital carbon content, greater carbon levels within the detritus corresponding with greater levels of lignin. Similarly, detritus L:N ratio was also positively correlated with detritus carbon content, higher levels of lignin and lower levels of N corresponding with greater levels of carbon (Figure 64, Figure 65). This result is expected, as others have found that as the biomass breaks down, the ratio of lignin and C will increase compared to N (Taylor et al. 1989, Talbot et al. 2011). This relationship is essentially showing N limitation in the long-term breakdown of organic matter with high C and Lignin content (Taylor et al. 1989, Talbot et al. 2011).

The mean soil N and C content showed a strong positive correlation, increases in soil C content corresponding to higher levels of N content (Figure 61). This relationship was also found in the detritus, with detrital C and N having a positive correlation across all sites (Figure 64). No relationship was found between mean living above ground biomass C and N content, indicating that this relationship becomes clearer once decomposition begins (detritus) and the decaying plant matter and associated microbial communities are incorporated into the soil within these sites.

Incorporating these nutrient dynamics into the long-term status and trends monitoring will provide added insight and confidence in our understanding of the detrital and nutrient flux within these sites and their contributions to the greater estuary food web.

Species Specific Traits Observed

Specific species analysis of the above ground living biomass C, N, and lignin content showed a large range of variability in these traits from species to species, however, species specific trends were generally found consistent across all sites sampled (Table 36, Table 37). Species specific C:N and L:N ratio results have provided insight into the quality of biomass and detritus being produced by dominant species. It has long been hypothesized that non-native grass *P. arundinacea* produces lower quality biomass (higher L:N and C:N ratios) than the native sedge *C. lyngbyei*, preliminary results from summer biomass sampling in 2018 support this hypothesis (Hanson et al. 2016). The common high marsh non-native species, *P. arundinacea*, was found to have a higher mean L:N and C:N ratios than *C. lyngbyei*. These differences in L:N and C:N ratios mirror observations of decomposition in the field with more *P. arundinacea* being retained on the site as standing dead biomass than *C. lyngbyei* (Hanson et al. 2016).

Common native low marsh species *S. latifolia,* and *E. palustris,* were found to have much lower L:N and C:N ratios than the high marsh species (Table 37), indicating these species have more N in their living above ground biomass than *P. arundinacea* and *C. lyngbyei,* aiding fast decomposition rates. *S. latifolia,* and *E. palustris,* are not generally found as standing dead due this faster decomposition and location in the low marsh which is exposed to more active hydrologic flushing (exporting the dead biomass) compared to the high marsh.

Other common species, *Polygonum amphibium* (Franz Lake, High Marsh) and *Polygonum hydropiperoides* (Whites and Welch Island, Low Marsh) were found to have the highest overall L:N ratios, this is not particularly surprising as these species have woody (high in lignin) perennial stems (especially when compared to the other common wetland grass and herb species) that persist throughout the winter months. *P. amphibium* and *P. hydropiperoides* are an interesting comparison to the other marsh species because they do lose their leaves annually without much dead leaf accumulation, but their stems tend to fall dormant (not actually standing dead), indicating that their L:N ratios may vary dramatically between the two plant structures (more in the perennial stems and less in the leaves). Further testing and distinction between leaves and stems of all species will help us better understand these functional plant traits and how they inform plant decomposition and detrital production within these sites moving forward.

4.3.1.3 *Pelagic*

In 2019, total algal biomass, as estimated by concentrations of chlorophyll *a*, was highest in March, prior to the spring freshet, at Welch Island and Whites Island; in contrast, the highest algal biomass at Campbell Slough and Franz Lake Slough was observed in August, with an exceptional peak in biomass at Franz Lake Slough in May. While there were a number of years between 2011 and 2019 where samples were not obtained in March, the values observed in March 2019 are relatively high for the early spring period. The low river flows in winter 2019 coincided with relatively high algal biomass, consistent with previous analyses that showed a negative correlation between river flow and algal biomass during the winter and spring months (Maier, 2014). The low levels of chlorophyll *a* observed after the freshet subsided and flows were reduced to some of the lowest rates in the 2011-2019 time series is also consistent with previous observations that in the summer months, river flow is positively associated with algal biomass (Maier, 2014).

The contrast in timing of maximum pelagic algal biomass between Welch and Whites Islands compared to Campbell Slough and Franz Lake Slough reflects the differences in species responsible for the bulk of the pelagic primary production. Whereas at Welch and Whites Islands the assemblages were dominated by diatoms, peak biomass at Campbell Slough and Franz Lake Slough was dominated by cyanobacteria and chlorophytes. This is an important distinction due to the differences in nutritional quality among the different groups of phytoplankton; studies have shown that feeding rates of zooplankton are very low on cyanobacteria (Schmidt and Jonasdottir, 1997). However, that same study showed that a supplementation of diatom diets by cyanobacteria can lead to an increase in feeding rates among copepods, as indicated by egg production rates; they observed that a 3:1 ratio of diatoms to small, unicellular cyanobacteria could result in an elevated feeding rate relative to diatoms alone (Schmidt and Jonasdottir, 1997). It is possible that the higher diversity in algal taxonomic classes observed at Campbell Slough and Franz Lake Slough contributes to the high densities of zooplankton there, in addition to slower presumed flushing rates that prevent dilution of standing stocks.

Over the last number of years, pelagic productivity has been high at Franz Lake Slough, which reached a peak >100 μg chl *a* L^{-1} in 2017. In 2019, cyanobacteria accounted for a large proportion of the phytoplankton assemblage in the summer; however, the cell densities (of cyanobacteria and other phytoplankton) were not as high as in 2017 or 2018 at most of the trends sites, with the exception of Campbell Slough, which had high abundances of cyanobacteria in August. There were also relatively high densities of cyanobacteria in 2019 at Ilwaco. Total phytoplankton biomass (as estimated by chlorophyll *a*) was highest in early spring at Ilwaco, Welch Island, and Whites Island (i.e., March, April); in contrast, peak biomass occurred after June at Campbell Slough. Phytoplankton biomass was high both before and after the freshet at Franz Lake Slough. At Franz Lake Slough the first peak coincided with high nitrate concentrations while the second peak (after the freshet) coincided with high phosphate concentrations. The species composition of the first peak was dominated by diatoms and chlorophytes, whereas the second peak was dominated by cyanobacteria, where the assemblage was dominated by *Anabaena* spp. and *Microcystis* spp. Anabaena was also abundant at Campbell Slough, in addition to Merismopedia spp., in August. *Microcystis* and *Pseudo-anabaena* were the most abundant cyanobacteria taxa at Whites Island, Welch Island, and Ilwaco.

The availability of phosphorus without available nitrate tends to stimulate the predominance of cyanobacteria (Andersson et al., 2015) since many of them are able to fix atmospheric nitrogen (Vahtera et al., 2007). The in situ fluorescence data showed a peak in phycocyanin, a pigment associated with cyanobacteria, in August, when phosphate concentrations were high. Although the proportional contributions by cyanobacteria to the phytoplankton assemblages was high, the cell densities were not as high as those observed in 2017 or 2018. Cyanobacteria blooms have been regularly observed in off-channel habitats during the mid to late summer months throughout the duration of the Ecosystem Monitoring Program (Sagar et al., 2016, Hansen et al., 2017), and each year of observations contributes to a better understanding of factors that control the initiation and development of blooms in these habitats. It is interesting to note that although cyanobacteria blooms tend to be associated with high temperatures (Paerl and Huisman, 2009, Paerl et al., 2013), the blooms observed during the warmest of recent years (2015) was associated with species that were not toxin-producing (i.e., *Merismopedia*, Tausz, 2015, Peterson et al., in prep.). In 2019, river flows and nutrient concentrations were not as high as in previous years; thus, while temperatures were favorable for the development of cyanobacteria blooms-and the proportional contributions to the total assemblage were high-the absolute densities were likely limited by low fluxes of nutrients to the system. This highlights the interplay between species composition and environmental conditions that influence the development of blooms, especially nutrient supply, temperature, and transport and colonization of organisms. Since nutrient supply to the lower Columbia River appear to come from different sources, including particulate matter (phosphorus), direct inputs from tributaries (nitrogen; especially from the Willamette), and the ocean (nitrogen or phosphorus, depending on the season; especially at Ilwaco), it is important to better understand how temporal patterns in nutrient supply influence the timing and magnitude of phytoplankton blooms, especially when they are dominated by noxious species such as toxin-producing cyanobacteria.

Outside of the warm summer months, the phytoplankton assemblage at Whites Island and Welch Island tends to be dominated by diatoms, with *Asterionella formosa* repeatedly being most abundant in the early part of the period of spring growth, while other diatoms, including *Skeletonema potamos* increased in abundance later in the year. *S. potamos* is a species typically associated with warmer waters; this species was present in high abundance in 2015, and was observed during the summer months during most years. In 2019, *S. potamos* was observed in relatively high abundance in May at Whites Island and in June at Welch Island and Campbell Slough, but was not observed at Franz Lake Slough, nor at Ilwaco. In each of the years between 2009 and 2019, *A. formosa* has constituted the early succession species that initiates the spring bloom in the river (Maier, 2014, Maier and Peterson, 2017, Maier et al., in review). This species is prone to heavy parasitism by flagellated chytrid fungi in the river mainstem (Maier and Peterson, 2014); the degree to which shallow water habitats with longer residence time influence rates and prevalence of parasitism upon primary producers that fuel aquatic food webs is currently being investigated (Cook and Peterson, unpubl. data). Since parasitism is often dependent on temperature (Ibelings et al., 2011), it is likely that periods of higher temperature would have a different prevalence of parasitism and thus influence carbon cycling and transfer through the lower food web.

Analysis of relationships between environmental variables and phytoplankton assemblages revealed that high relative proportions of diatoms are associated with high concentrations of dissolved oxygen and high dissolved oxygen saturation relative to the atmosphere. Diatom growth is also associated with a reduction in nutrient concentrations (accomplished through drawdown associated with growth). High dissolved oxygen saturation and low-to-moderate nutrient concentrations are indicative of good water quality. Diatoms tend to dominate in the spring months, where populations can get quite large; most of the annual growth of phytoplankton occurs in the spring and is accomplished by diatoms (Maier and Peterson, in prep.).

According to a Bayesian Inference stable isotope mixing model, phytoplankton carbon contributes to the juvenile salmonid food web as part of the diet of chironomid prey, based on stable isotope signatures of carbon; this carbon is incorporated as particulate organic matter and as periphyton (attached organisms). Models looking at how different sources of primary production contribution to additional prey sources are being investigated as more data are gathered, but analysis thus far suggests that periphyton constitutes an important source of organic matter for the preferred prey of juvenile salmonids (i.e., amphipods and chironomids). Estimates of dietary contributions from different prey items inferred from stable isotope mixing models suggest that juvenile salmonid growth is supported by amphipods, chironomids, and other crustacean prey, which is consistent with observations derived from stomach analysis.

4.3.2 Zooplankton

Zooplankton assemblages differ along the spatial gradient from Ilwaco Slough to Franz Lake Slough and over time from early spring to summer. Ilwaco Slough is consistently dominated by copepods, with inputs from rotifers, but very few cladoceran taxa. At the other sites, copepods generally dominated the zooplankton assemblages. At Welch Island and Whites Island, there was an increase in the proportional contribution by cladocerans from spring to summer in each of 2017, 2018, and 2019. At Campbell Slough and Franz Lake Slough, an increase in the proportional contribution of cladocerans was observed from March to June; however, by July, the relative proportions of cladocerans decreased at both sites in 2017 and 2018.

4.4 Macroinvertebrates

Macroinvertebrate data collected in 2019 are still under analysis and unavailable at the time of the writing of this report. This report presents results of the 2018 sample season.

We examined trends in the availability of major juvenile Chinook salmon prey taxa, including amphipods, dipteran flies, cladocerans, and copepods. Amphipod abundance in benthic core samples was greatest at Ilwaco Slough. Relatively few amphipods were collected from Welch Island and Whites Island, and typically no amphipods were collected from the furthest upriver sites, Campbell Slough and Franz Lake. The distribution of benthic invertebrates in the environment is not uniform, and high variation occurred among benthic samples. Regardless, the pattern of declining abundance in amphipods upriver is consistent over time and is also reflected in the diets of juvenile Chinook salmon.

Similar to benthic amphipods, benthic dipteran larvae abundances were variable, but densities tended to be greatest from the three lower river sites, Ilwaco Slough, Welch Island, and Whites Island. In contrast, counts of Diptera (including larvae, pupae, and adults) collected by neuston tow were often highest from the upriver sites. Campbell Slough and Franz Lake have lower connectivity to the mainstem, especially during low water periods, and aquatic insects, like chironomids and other dipterans, may be retained more within these sites than at more open sites like Ilwaco Slough and Welch Island. The extent of invertebrate export from tidal marsh systems is influenced by the size and geomorphology of wetland channels as well as the energy associated with oscillating water levels and velocities. Connectivity to the mainstem is likely a factor in the potential for fluvial export of wetland insects and may help explain the disconnect between our benthic and neuston sampling. Continued monitoring of patterns in benthic and neuston dipteran densities at the trend sites will help inform the complexity of prey availability in these tidal wetlands.

Spring of 2017 saw a spike in Chlorophyll a concentration at Campbell Slough (Kidd et al. 2017). This was followed by substantial increases in zooplankton, with cladoceran and copepod densities peaking in May. At this time, counts of cladocerans exceeded 1200 individuals on average per meter towed, and counts of copepods were nearly 1800 individuals on average per meter towed. While we have observed other peaks in these taxa (e.g., Cladocera in April 2015 at Campbell Slough, and Copepoda in April 2017 at Campbell Slough and April 2015 at Franz Lake), densities at Campbell Slough in May 2017 were far greater than in previous observations.

In Pacific Northwest estuaries, including the Columbia River estuary, juvenile Chinook salmon diet composition is typically dominated by amphipods and dipterans (Simenstad et al. 1982, Lott 2004, David et al. 2016). The EMP study has consistently described a dietary transition from wetland insects to amphipods as juvenile Chinook salmon grow and move toward the estuary mouth. In 2017, however, we observed a very different prey composition juvenile Chinook salmon diets from Campbell Slough than previously seen. They typically consume Chironomidae and other dipteran taxa at this site but switched to preying primarily on Cladocera during the spike in zooplankton abundance. Campbell Slough diets in May and June across all fish size classes were dominated by cladocerans, with this group accounting for nearly 90% of the IRI overall (ranging between 67 and 95% of the IRI depending on size class and month).

Conditions affecting the growth potential of juvenile Chinook salmon varied over both spatial and temporal scales in the estuary. For the fish, habitat opportunity metrics such as site accessibility, temperature, water depth, and salinity interacts with habitat capacity metrics such as prey availability, competition, and predation to determine salmon feeding success, growth, and survival (Bottom et al. 2005). Calculating and examining average metabolic costs and energy assimilation experienced by fish may be a useful tool to allow us to evaluate habitat quality across various time scales. For example, it could tell us how habitat changes at the scale of a single juvenile Chinook out-migration season or at scales of years or decades that experience large scale differences in climate. The method may also be useful in comparing among different sites to understand where salmon experience relatively good or poor growing conditions. For example, salmon sampled from a new restoration site could be plotted along with the long-term averages from the trend sites to provide an evaluation of the new habitat relative to other areas in the estuary.

This study examined the interaction of maintenance metabolism, driven by water temperature and fish size, with energy ration, driven by prey availability and quality. The interaction of energy assimilation and metabolic cost was plotted for fish collected in May and June by annual freshet condition; however, these metrics did not appear to correlate with each other. While hydrologic condition varied across the four-year period between 2015 and 2018, this study did not observe a difference in prey availability or prey consumption between years. Comparatively, a study of juvenile Chinook prey resources and diet composition in a California river floodplain did find significant effects of extreme hydrologic regimes on both prey availability and Chinook diet (Goertler et al. 2018). The study in California examined conditions that persisted over multiple years and represented environmental extremes. It is possible, the hydrologic conditions described here in the Columbia River estuary did not persist long enough or fluctuate to the extent that would significantly impact prey resources or necessitate adaptive behaviors by the fish. We recommend continued monitoring of the relationship between salmon food webs and river temperature and flow to better understand these interactions and implications for future scenarios.

4.5 Fish

In 2019, fish community composition was sampled at four trend sites—Ilwaco Slough, Welch Island, Whites Island, and Campbell Slough. Due to permits arriving late (an unfortunate result of the federal government shutdown in January 2019) sampling did not commence until April, though Welch was able to be sampled in March.—Franz Lake was unsuccessfully sampled in April, but by that point water level were already too high and remained high until May. Water temperatures at Franz Lake were too high during the sampling windows in June and October.

There is much overlap in overall species composition at all five trend sites with specific attributes that either separate or link sites in terms of similarity. Ilwaco stands apart with a greater influence of marine species while Welch and Whites Islands tend to resemble each other and overlap Ilwaco and the upriver sites at Campbell Slough and Franz Lake. As in previous years, the catches at Welch and Whites Islands were composed primarily of native species and were dominated by a single species (Threespine stickleback). Catches at Campbell Slough were more evenly distributed among species and included a greater percentage of non-native fishes. In 2017-2018, the species richness and diversity of the fish community at Campbell Slough were much lower than observed in previous years. In 2019 the species diversity increased markedly at Campbell Slough, while the species richness remained similar to 2017-2018. The increased species diversity in the upper reaches of the estuary is primarily driven by non-native species, many of which have mature stages that could prey upon juvenile salmon. The greater proportion of non-natives species in this part of the estuary and river is likely due to several factors including reduced marine influence, summer water temperatures, and the predominance of back water sloughs connected to the mainstem through tide gates and water control structures. Studies have shown that these areas can be hotspots for non-native species and foster environmental conditions, such as high temperature and low dissolved oxygen, which many non-native species can tolerate (Scott et al. 2016, McNatt et al. 2017).

Patterns of salmon species composition vary by year and more strongly by site. Chum salmon dominate catches at Ilwaco Slough, but Chinook salmon are the most prevalent salmonid observed at all other sites. The lack of Chinook at Ilwaco Slough is consistent across years yet difficult to explain. It is possible that prevailing currents cause smolts to bypass the area or that the site’s location adjacent to a vast mud flat limits juvenile salmon access to later stages of incoming tides. The majority of Chinook caught at all sites are unmarked fry and fingerlings except at Campbell Slough where the proportion of unmarked and marked fish varies. These findings support the results of other studies of juvenile salmon use in the lower river and estuary (Bottom et al. 2011, McNatt et al. 2016, Sather et al. 2016).

Site-specific trends in the stock composition are evident. Unmarked West Cascade fall are the predominant stock of Chinook observed at Welch and Whites Islands. These sites are located downstream of tributaries such as the Lewis, Kalama, and Cowlitz Rivers which produce large numbers of West Cascade fall stock. Franz Lake is located upstream of West Cascade fall tributaries, and this is reflected in the higher percentage of interior and Spring Creek Group stocks observed there. The greatest diversity of stocks is located at Campbell Slough in Reach F, where salmon from interior Columbia Basin, Willamette River, and lower river stocks converge. Results from this study support the findings of Teel et al. (2014) who sampled hydrogeomorphic reaches throughout the estuary and found the greatest diversity of stocks in Reaches E and F. Specific to 2019, upper Columbia summer/fall stocks were observed at Welch, Whites, and Campbell Slough at the highest percentages ever observed at those locations. The reason for this increase is unknown, but patchiness in the distribution of the stock coupled with coarse temporal sampling frequency (monthly) could play a role, whereby in previous years we may have missed large numbers of this stock.

Spring Creek group stock dominate catches of marked Chinook at Campbell Slough and Franz Lake in the upper portions of the estuary. This is likely due to the close proximity to and a large number of hatchery fish of this stock released from hatcheries just above and below Bonneville Dam. Spring Creek Group stock comprise a larger percentage of marked than unmarked Chinook at Welch and Whites Islands, but West Cascade fall stocks remain the predominant stock of both unmarked and marked fish at these sites. Specific to 2019, tissue samples were collected from two ad-clipped yearling Chinook that were captured and released at Campbell Slough in May. Both fish assigned with high probability to the mid and upper Columbia spring Chinook reporting group, which includes ESUs listed as endangered.

The seasonal distribution of stocks is similar to what has been found in previous studies (Roegner et al. 2012, Teel et al. 2014). West Cascade fall stock are present throughout the year. Spring Creek group stock tend to increase in proportion during April–May, concurrent with large hatchery releases, and interior stocks tend to show up beginning in April and through summer.

The temporal distributions of Chinook and chum salmon indicate separation in the timing of estuary use. Chum salmon densities peak in March or early April, whereas Chinook salmon densities increase through April, peak in May, and then start to decline. This pattern of estuary use is similar to patterns of abundance found by Roegner et al. (2012). The consistency with which juvenile salmon are captured at EMP trend sites demonstrates the importance of tidal wetlands to juvenile Chinook salmon. Chinook are rearing in these areas during times of low and high flows. The predominance of Chinook salmon in tidal wetland habitats is consistent with findings of other studies within the Columbia River estuary and elsewhere (Levy and Northcote 1982, Healey 1991, Bottom et al. 2011, Hanson et al. 2017).

The abundance of food resources in tidal wetlands is a likely attractant of juvenile Chinook. This study and others have demonstrated that prey items originating from tidal wetlands are an important part of Chinook diet (Lott 2004, Maier and Simenstad 2009, Hanson et al. 2017, Weitkamp et al. 2018) and Chinook have been observed entering wetland channels against water flow during times of peak diel prey abundance (McNatt et al. 2016). Condition factors at EMP trend sites are consistent, with little variability over the years. Measures of percent lipids and triglycerides are variable over time and across sites. The value ranges of lipid content for juvenile Chinook within the Columbia River estuary (1.4–2.3%) are consistent with values observed in Chinook salmon shortly after ocean entry. Daly et al. (2010) measured percent lipid of juvenile Chinook salmon in May and June off the coast of the Columbia River and southern Washington and found average (SD) values of 1.3% (0.7), whereas other marine fishes tended to have much higher values, e.g., Liparidae = 5.8% (0.5) and Cottidae = 6.8% (1.5).

Somatic growth analyses from otoliths indicate that fish collected in this study (2005-2018, over a range of sites mainstem and off-channel sites, historically sampled) are growing at rates similar to or greater than what other studies in the Columbia River estuary have observed (this study: 0.54 mm/d, Chittaro et al. 2018; 0.41 mm/d, Campbell 2010; 0.23 mm/d, Goertler et al. 2016; 0.53 mm/d, McNatt et al. 2016). At off-channel sites, fish length correlated with growth rates, as larger fish grew faster than smaller fish. Chittaro et al. (2018) also found that fish collected in the upper reaches of the estuary grew at faster rates than those collected at lower reaches of the estuary. This pattern seems contrary to conventional thinking that growth rates increase as the salmon move from colder tributary waters to warmer estuarine habitats with large capacities of prey production. A number of factors could contribute to this observation. The transition from freshwater to saltwater environments and maintaining position in an increasingly tidal habitat may require additional energetic resources. At off-channel sites, growth rates have been consistent from 2007–2018, implying that differences in flow regimes from year-to-year have little impact on the growth rates of juvenile Chinook that utilize tidal wetlands. Of note is that otoliths sampled from 2015, which was an extreme low flow and high temperature year, were lost and not processed, so any impacts from such extreme conditions cannot be ascertained.

Additionally, as juvenile salmon pass through lower reaches of the river, the input of highly estuary-dependent stocks such as West Cascade falls increases. This could lead to density-dependent impacts on fish utilizing tidal wetlands. Given that 70% of vegetated tidal wetlands in the Columbia River estuary have been lost (Marcoe and Pilson 2017) the reduced capacity of the estuary to produce adequate prey resources may exacerbate increased competition for food.

Data from off-channel PIT detection arrays indicate that off-channel habitat is used by a wide variety of stocks and species including Chinook and coho salmon, as well as steelhead. The extent of use varies among stock. Fall Chinook typically are the most abundant in these areas and reside longer than other stocks. However, at Horsetail Creek individual steelhead have been shown to reside for several months. One caveat to off-channel use is that northern pikeminnow, a known predator of juvenile Chinook salmon has also been detected in these habitats and tend to reside for weeks to months. Thus, extended use of these habitats could increase juvenile salmon vulnerability to predation.

The ecological trade-off between predation risk and foraging opportunity in tidal wetlands, as in tributaries and the ocean, is the mechanistic driver of survival. Increases in foraging opportunities through habitat restoration and efforts to decrease predators (especially non-native predators) may help tilt the scale towards improved salmon survival.

5 Juvenile Chinook Salmon Food Web Synthesis Discussion

5.1 Introduction

The EMP has been collecting ecosystem condition data in the lower Columbia since 2005, focusing its efforts on collecting on-the-ground data from relatively undisturbed emergent wetlands to provide information about habitat structure, fish use, abiotic site conditions, salmon food web dynamics, and river mainstem conditions to assess the biological integrity of the lower river, enhance our understanding of estuary function, and support recovery of threatened and endangered salmonids. The creation and maintenance of long-term datasets are vital for documenting the history of change within important resource populations. Therefore, through this program, we aim to assess the status (i.e., spatial variation) and track the trends (i.e., temporal variation) in the overall condition of the lower Columbia River, provide a better basic understanding of ecosystem function, provide a suite of reference sites for use as endpoints in regional habitat restoration actions, and place findings from other research and monitoring efforts (e.g., action effectiveness monitoring) into context with the larger ecosystem. The synthesis below is a summary of *juvenile salmon food web* information which has been developed as part of this program from the past 12 years of data collection in the lower Columbia River.

5.2 Characterization of Salmonids in the lower Columbia River

5.2.1 Salmon Tidal Wetlands Use Patterns

All anadromous salmonids common in the Columbia River basin have been observed in tidal emergent wetland and backwater slough sites typical of the Lower Columbia Estuary Partnership's Ecosystem Monitoring Program (EMP) sites. The degree of wetland utilization varies with species and life history type. For example, species with yearling life histories, such as sockeye salmon, steelhead, and cutthroat trout, are rarely observed. However, coho salmon, which also has a yearling life history strategy, are caught in Reach H closest to Bonneville Dam. Chum salmon, which have a subyearling life history, are the second most frequent species observed. Chum have been seen at all sites, and their use of tidal wetlands peaks in April and is limited temporally from March-May.

Chinook salmon, which have both yearling and subyearling life histories, are described as the most estuary-dependent species (Healey 1982, Levy and Northcote 1982). Indeed, Chinook salmon are the predominant species observed in tidal wetlands by this study and others. Bottom et al. (2011) sampled tidal wetlands and mainstem beaches in Reaches A-B, from 2002-2008 and found distinct size-related patterns of juvenile Chinook salmon use. Wetlands were dominated by fry (<60 mm fork length) and fingerlings rarely larger than 90 mm, while a larger range of sizes were sampled at beach sites. Further upriver in Reaches D-E, Sather et al. (2016) also saw a wider range of size classes in mainstem beach sites versus wetland channels.

Residence data indicates that subyearling Chinook salmon may reside in tidal wetlands for extended periods. McNatt et al. (2016) marked and PIT tagged juvenile Chinook salmon in an emergent wetland near rkm 36. While there was a wide range of residence times recorded, 33% of recaptured individuals resided for more than one week, despite being forced to exit the wetland during low tides. However, PIT tag data from tidal wetlands suggest that the size-related patterns of wetland use may be biased by gear type. McNatt et al. (2015) have observed that larger salmon from interior stocks tend to enter wetlands at high tide, implying that yearlings may be under-represented by traditional sampling methods.

5.2.2 Fish Condition and Growth

Foraging opportunities likely attract juvenile Chinook salmon to tidal wetlands. Wetlands are productive habitats that provide a variety of ecological services (Nixon 1980, Boesch and Turner 1984, Peterson et al. 2008) In the lower Columbia River it has been estimated that up to 1 million macroinvertebrates are exported from a single wetland channel during an ebb tide and that the majority of the exported taxa are chironomids (C. Roegner, pers. comm.). Chironomids are known to be an important component of juvenile Chinook salmon diets. In the first comprehensive diet study of Columbia River juvenile Chinook salmon collected in tidal wetlands, Lott (2004) found that chironomids accounted for 85.3% of the total IRI. Since that time chironomids have been documented as a major element of juvenile Chinook salmon diets throughout the lower Columbia River and estuary (Kidd et al. 2018, Goertler et al.2016, Sather et al.2009). Juvenile Chinook salmon have been observed entering wetland channels against water flow during times of peak diel prey abundance (McNatt et al.2016) further supporting the tenet that foraging opportunities attract juvenile Chinook salmon to these habitats.

Growth and fish condition are used to measure the capacity of wetland habitats to support salmon populations. Growth rates for juvenile Chinook salmon in the estuary indicate that fish collected in this study (2005-2013) are growing at rates similar to or greater than what other studies in the Columbia River estuary have observed (this study: 0.54 mm/d, Chittaro et al. 2018; 0.41 mm/d, Campbell 2010; 0.23 mm/d, Goertler et al. 2016; 0.53 mm/d, McNatt et al. 2016). Subyearling Chinook salmon that reside in tidal wetlands can achieve substantial growth. McNatt et al. (2016) measured increases of 20 mm for individuals that resided in a tidal wetland for 15 days or more. The condition of unmarked Chinook salmon is more variable than that of marked fish, yet, in both groups, condition increases over the course of the migration period at all trend sites (Figure 149). Limited data indicated that after July condition starts to decline, likely due to high water temperature.

Measures of performance such as condition factor and growth, coupled with residence time, indicate that tidal wetlands are productive and beneficial habitats for juvenile salmon. The abundance of prey items and refuge from piscine predators and high flows creates beneficial rearing habitat to allow juvenile salmon to grow and adjust to an increasingly marine environment as they migrate seaward.

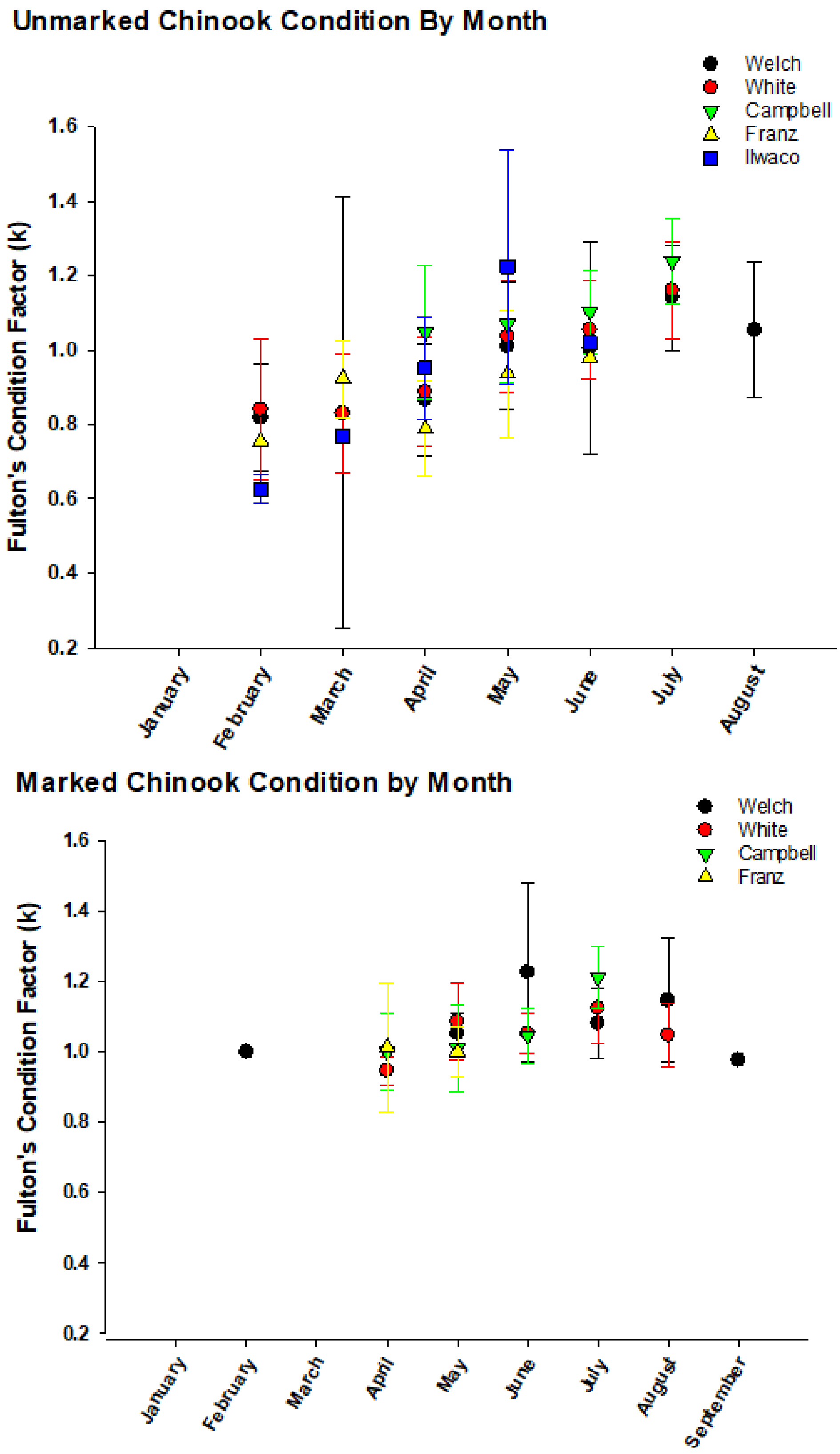


Figure 149. Monthly mean (SD) Fulton's condition factor of unmarked and marked Chinook Salmon, 2008-2017.

5.3 Characterization of Salmonid Prey Conditions in the lower Columbia River

5.3.1 Juvenile Salmon Prey and Diet

While there may be considerable diversity of prey taxa in juvenile Chinook diets overall, the EMP study has consistently identified a propensity for two particular prey items consumed by juvenile Chinook salmon collected at the emergent wetland and backwater slough trends sites. These taxa, Chironomidae (Diptera) and Amphipoda, dominate consumption patterns according to the Index of Relative Importance (IRI). Amphipoda is a diverse order of soft-bodied epibenthic crustaceans. Amphipods consumed by fish in 2015-2017 were identified to a lower taxonomic level than in previous years and were primarily from the genus *Americorophium* in the family Corophiidae. *Americorophium* spp. are estuarine amphipods, commonly found in brackish to freshwater environments. They build tubes in sand and mud flats and adjoining shallow water habitats that are intermittently exposed with the tide along larger channels in emergent marshes and along the mainstem river. *Americorophium* becomes available as prey for juvenile salmon and other fish when they leave their burrows to migrate or as part of reproductive behavior (e.g., males looking for mates, Davis 1978, Wilson 1983).

Chironomidae is a ubiquitous family of small dipteran flies, commonly known as midges, which provide food for a wide range of predators throughout all stages of their life cycle (Armitage 1995). The family is regularly reported as the dominant insect group from wetland and estuarine systems, including the lower Columbia River (Stagliano et al. 1998, Williams and Williams 1998, Lott 2004, Ramirez 2008). They are the most widespread of all aquatic insect families, occurring on all continents (Ferrington 2008). Chironomids are also exceptionally diverse, with estimates as high as 15,000 species, and some taxa can tolerate diverse climates and conditions, including a wide array of water quality (Cranston 1995, Ferrington 2008). A study on chironomid distribution in an emergent marsh of the Columbia River estuary showed that abundance peaked in mid-June with temporal and microhabitat patterns in distribution driven by three dominant genera (Ramirez 2008). These genera exhibited microhabitat preferences within a tidal channel, but not habitat specialization. As non-specialists, these insects can adapt to a variety of conditions, a characteristic consistent with the ubiquitous nature of the family (Cranston 1995, Ferrington 2008).

There are four stages in the chironomid life cycle (Figure 150). For aquatic insects, the metamorphosis and subsequent emergence to adult form are accompanied by a switch from the benthic growth period of the larvae to the terrestrial reproductive and dispersal stage of the adults. Lott (2004) found that emerging adults were the dominant life history stage appearing in the diets of juvenile Chinook salmon in shallow water wetland habitats of the Columbia River estuary. The EMP study, however, found that juvenile Chinook salmon fed primarily on the adult and larval stages of chironomids. Emergent chironomids, as well as those in the pupal stage, were regularly consumed by fish, but less frequently than the adults and larvae.

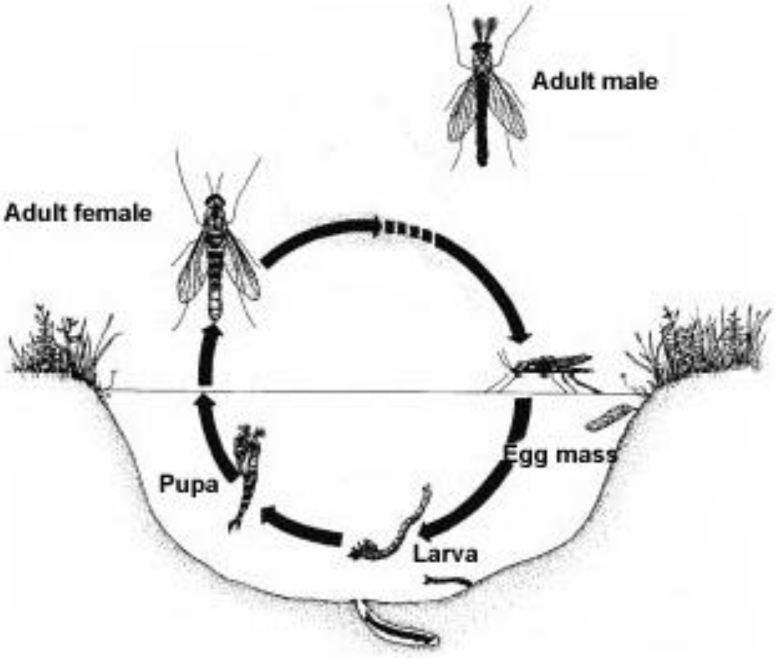


Figure 150. Life cycle of chironomid flies (image from North Carolina State University, Cooperative Extension).

Several studies have described a dietary transition from wetland insects to amphipods as juvenile Chinook salmon grow and move toward the estuary mouth (McCabe et al. 1986, Lott 2004, Bottom et al. 2011). Juvenile Chinook salmon diets from the trends sites located further upriver and less connected sites (Campbell Slough and Franz Lake), are dominated by chironomids and other wetland insects. Fish collected from Welch and Whites Island, located in Reach B and C, respectively, mainly consume a combination of amphipods and chironomids or other dipteran flies. At Ilwaco Slough in Reach A near the estuary mouth, juvenile Chinook feed almost exclusively on amphipods. According to stable isotope signatures of carbon and nitrogen (Peterson and Fry 1987, Phillips et al. 2014), the organic matter source supporting chironomids appears to be primarily periphyton, similar to a study in which grazing larval chironomids fed on periphyton and diatoms in a shallow, hypertrophic lake in Poland (Tarkowska-Kukuryk 2013). Corophiid amphipods bore carbon isotopic signatures that were heavier on average than those of vascular plants or particulate organic matter (presumed to be dominated by fluvial phytoplankton), indicating that their primary dietary source of organic matter is heavier than either of those two sources. A likely candidate is benthic diatoms (Maier and Simenstad 2009), although there were times when periphyton also appeared to be an important food source for corophiids.

The current Columbia River estuarine landscape and distribution of habitats may explain some of the patterns seen in salmon diets across sites. Reach A and Reach B, both subject to coastal influences, have broad sand and mud flats (Reach A) and successional development of emergent marshes on sand and mud flats (Reach B; Simenstad et al. 2011). Reach B contains complex channel islands with extensive networks of distributary and tidal channels. The widespread surge plain in these lower reaches supports relatively large areas of intermittently exposed shallow-water habitats, suitable for both juvenile salmon utilization and corophiid amphipod colonization (Figure 151). Farther up the estuary in Reach C, the river valley is more constricted, and areas of intermittently exposed habitat are typically limited to narrow sandy beaches (Simenstad et al. 2011).

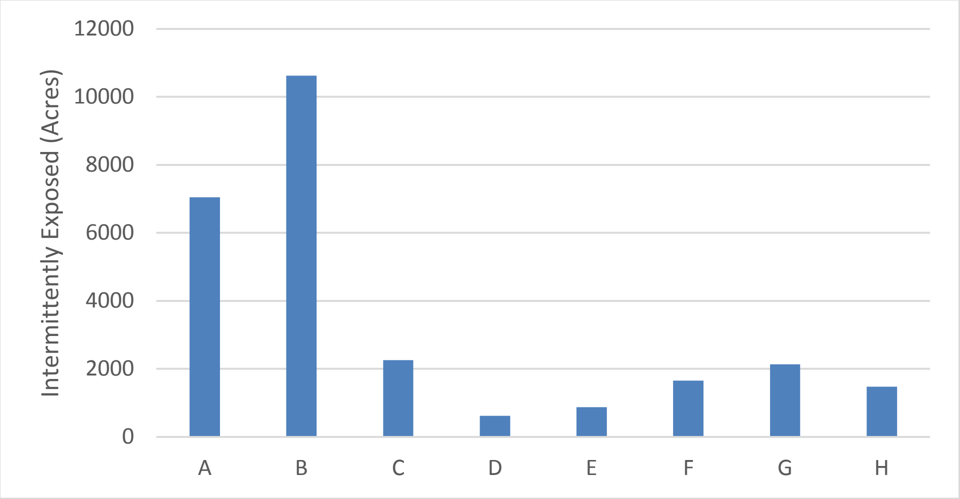


Figure 151. Area (acres) of the Intermittently Exposed Geomorphic Catena by Hydrogeomorphic Reach, as mapped in the Columbia River Estuary Ecosystem Classification.

Amphipods occurred regularly in monthly benthic core samples at Ilwaco Slough in Reach A between April and July in 2015-2017. Densities of amphipods at the other trends sites have been nearly non-existent (Campbell Slough and Franz Lake) or too inconsistent and low (Welch Island and Whites Island) to establish any patterns of occurrence. Prey sampling methods at EMP trends sites included benthic cores and neuston tows. These methods may not be optimal for sampling *Americorophium* spp. which occur mainly within the sediment and are patchy in distribution. While the distribution of amphipods has not been effectively assessed across trends sites, the sampling locations at Welch and Whites Island are within large distributary channels adjacent to intermittently exposed shallow-water habitats. These areas would presumably support greater abundances of amphipods than the backwater sloughs further upriver at Campbell Slough and Franz Lake where flats are not as well developed. Bottom et al. (2011) describe differences in amphipod distribution and consumption that they observed between fish collected monthly from small interior channels of the emergent marsh surge plain at Russian Island in Cathlamet Bay, and those collected by beach seine for a diel consumption study in a large distributary channel (similar to the EMP trends site at Welch and Whites Island) just downstream of the monthly surveys:

> "*The incidence of Americorophium spp. in the diets of Chinook salmon collected during diel sampling was considerably higher than that of salmon collected during monthly trap-net surveys. This may reflect some microhabitat differences in invertebrate distributions. Although Americorophium spp. rarely occurred in benthic samples collected at the trap-net site, dense colonies of the amphipod could be seen in the larger adjacent tidal channels where the diel studies were conducted.*"

Thus, it may be informative in future research to conduct distributional studies of *Americorophium* spp. at the trends sites, to determine which sub-habitats they use at each site. This would allow for refinement of sample locations for benthic cores within each site.

Similar to Bottom and Levy's findings for *Americorophium* in the Columbia River estuary, Levy et al. (1979) found that in the Fraser River estuary chironomids were more prominent in the diets of fish in tidal channels within the marsh complex than those of mudflat and adjoining shallow-water habitats. The results from both of these studies highlight the importance in associating what fish are eating with where they are feeding, because even small shifts in habitat can significantly affect macroinvertebrate availability and consequently diet composition.

While a predictable shift in prey consumption along the estuarine gradient is evident in results from the EMP study, we do report at times annual variation in consumption patterns within a site. This variation ranges from moderate to significant when comparing IRI metrics across years. For example, while juvenile Chinook at Welch Island and Whites Island consistently consume a combination of both amphipods and chironomids or other dipteran flies, the relative contribution of these two taxa groups will vary from year to year. Amphipods comprised nearly 75% of the IRI at Whites Island in 2011 and nearly 90% in 2017, but only 15% in 2012. In contrast, dipterans comprised only 12% of the IRI at Whites Island in 2011 and 8% in 2017, but nearly 80% in 2012. At Welch Island the contribution of amphipods to the IRI ranged between 36% in 2012 to over 70% in 2017.

More significant deviations from the typical consumption pattern have been observed at the upriver sites, likely driven by site differences in prey supply and availability. Franz Lake was the only site where copepods (a relatively energy-poor prey item) were consumed in large quantities, comprising up to 60% of the IRI in April 2016 for the smallest size class, and never more than one percent at other sites. Dipteran densities in neuston tows at Franz Lake were variable, but high relative to other sites in April 2016. Also, while copepods were very abundant in April 2016 neuston tows from Franz Lake, they also occurred in neuston tows from all other sites but were not consumed by juvenile Chinook salmon at other sites.

Another significant shift in consumption patterns was observed in 2017 at Campbell Slough. In previous sample years, dipterans had consistently comprised at least 70% of the IRI at Campbell Slough, but in 2017 dipterans made up only 10% and cladocerans accounted for nearly 90% of the IRI. Spring of 2017 saw a spike in Chlorophyll a concentration at Campbell Slough (Kidd et al. 2017), followed by substantial increases in zooplankton, with cladoceran and copepod densities peaking in May. While measures of instantaneous ration were similar from Campbell Slough in 2017 to previous years, energy ration did appear to be depressed in 2017 when fish were consuming primarily energy-poor cladocerans. This means that fish were eating enough of the small bodied taxa to maintain stomach fullness, but were not acquiring the same amount of calories as in previous years. This result suggests there may be impacts to juvenile Chinook growth potential when zooplankton abundance spikes and fish shift their prey selection. In a study of juvenile Chinook salmon prey selection in a floodplain-tidal slough complex of the Sacramento River, California, Goertler et al. (2018) found that fish fed primarily on aquatic-riparian insects during flood conditions and zooplankton during drought conditions. In contrast, Columbia River mainstem flows in 2017 were higher than average in the spring and early summer when zooplankton were abundant and heavily consumed at Campbell Slough. The EMP may benefit from a more rigorous analysis of the environmental factors driving zooplankton abundance and distribution as well as the energetic consequences of shifts in prey selection.

Zooplankton densities tend to be highest at Campbell Slough (Reach F) compared to other trends sites. Rotifers are very abundant early in the season (i.e., before the freshet) throughout the lower estuary, with the exception of Ilwaco (Reach A), while cladocerans and copepods are more abundant after the spring freshet. Similar to the spatial gradient in prey consumption by juvenile salmonids, there is a downstream gradient in zooplankton composition. In Reach A, at Ilwaco, the zooplankton community is always dominated by copepods, while upstream, the community transition from one dominated by rotifers in the early spring and shifting to copepods and cladocerans after the freshet. Seasonality in river discharge and water elevation is associated with changes in zooplankton abundance and composition and abundance, which has also been found to be reflected in the stomach contents of salmonids sampled across the sites. Zooplankton abundance increases throughout the lower estuary following spring growth of phytoplankton. During the spring freshet, abundances of zooplankton decrease due to dilution; during the summer, abundances increase substantially once water levels recede, particularly at Campbell Slough (Reach F), where connectivity to the mainstem is relatively low.

5.4 Characterization of Food Web Primary Productivity in the lower Columbia River

5.4.1 Marsh Plants Fuel the Salmon Food Web

The energy that supports a food web, and constrains its productivity, is provided by the system's primary producers, including plants, phytoplankton, and benthic microalgae. The productivity of invertebrate prey for salmon depends in part on the volume, quality, and timing of delivery of biomass from the marsh (Hanson et al. 2016a, Figure 152). Marsh plants provide more biomass and are a higher source of energy than phytoplankton or microalgae (Hanson et al. 2016 b). The productivity of marsh plants varies over both space and time, in response to changes in key biophysical drivers like water levels, sediment dynamics, invasive species, and other sources of stress. When plant biomass production or its quality declines, there is less food to fuel the invertebrate food web that supports salmon. For this reason, it is important to understand the biophysical interactions that drive variation in plant productivity and how this productivity feeds into the salmonid food web. Stable isotope signatures of carbon within salmonid muscle and prey items do not provide a direct match to that of marsh plants, and therefore it is possible that degradation of plant material through microbial or fungal processing may be important to the assimilation of this high-energy material. These processes should be characterized to identify the mechanisms by which energy flows from primary production to salmonids and other ecosystem components. Elsewhere, the lability of carbon compounds within vascular plants was significantly altered through decomposition processes, increasing carbon assimilation rates later in the growing season when higher temperatures lead to higher growth rates of decomposers that include bacteria and fungi (Campeau et al., 1994).

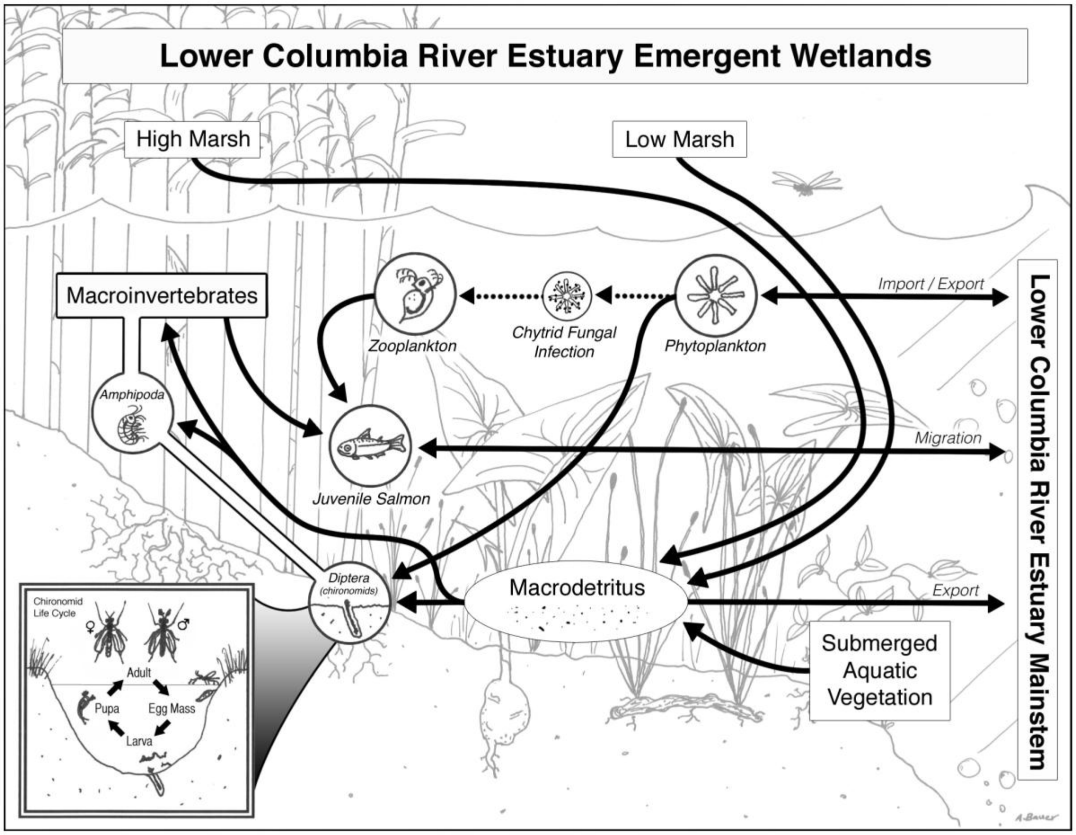


Figure 152: Conceptual model of food web interactions within Lower Columbia River emergent wetlands.

In addition to overall biomass productivity, the quality of biomass varies in ways that may affect its contribution as food for salmon prey. Low marsh plants contribute 80-93% of their annual aboveground biomass to the detrital food web, with particularly high values for *Sagittaria latifolia*, *Eleocharis palustris*, and submerged aquatic vegetation. Within the high marsh, communities that are dominated by the native sedge, *Carex lyngbyei,* contribute 68-80% of their annual aboveground biomass to the food web each year. In contrast, communities dominated by the non-native reed canarygrass, *Phalaris arundinacea,* contribute only 37-72% of their annual biomass to the food web in the same year. In addition to contributing less of its annual biomass to the detrital food web that supports salmon prey, *P. arundinacea*'s contribution is also substantially more variable. Overall, wetlands dominated by the native sedge *C. lyngbyei* contributed the highest and most consistent amount of organic material, signifying the importance of this species to the food web in the estuary.

Furthermore, there is some evidence that the non-native plant species, *P. arundinacea,* produces biomass with a higher concentration of lignin which is difficult to decompose and may reduce the proportion of annual biomass that enters the detrital food web. This potential difference in biomass quality may reduce the food available to support salmon prey. Biomass quality is a new area of investigation and may lead to new insights about the importance of adjusting restoration and management strategies to favor native wetland species.

5.4.1.1 *Plant Assemblages in Columbia River Tidal Marshes*

Many different vegetation assemblages occur in tidal marshes, but for simplicity, we generalize the major groupings into three main strata: high marsh, low marsh, and submerged aquatic vegetation (SAV). Within a site, those three categories occupy different places in the elevation spectrum, from high to low. For the purposes of this discussion, we'll focus on just the high and low marsh assemblages.

Species richness in tidal marshes ranged from 16 – 42 species in 2016, with the lower value at the downstream-most site at Ilwaco Slough and the highest value at Whites Island. In the same year, total percent cover of plants ranged from a low of 84% at Ilwaco Slough to a high of 130% at Welch Island. The dominant plant species throughout the estuary are shown in Table 42. The two most common species are the native sedge, *Carex lyngbyei*, and the non-native grass, *Phalaris arundinacea*, which appear to have different effects on the detrital food web that supports salmon prey.

Table 42. Cover (%) of native and non-native species at the monitoring sites.

Species Code	Scientific Name	Common Name	Wetland Status	Category	Native	All Years Avg. % Cover (SD)	2016 Avg. % Cover (SD)
CALY	*Carex lyngbyei*	Lyngby sedge	OBL	Sedge	yes	20.4 (25.4)	19.1 (24.2)
PHAR	*Phalaris arundinacea*	Reed canarygrass	FACW	Grass	no	20.4 (17.4)	22.7 (19.8)
ELPA	*Eleocharis palustris*	Common spikerush	OBL	Sedge	yes	7.3 (8.7)	7.5 (10.8)
SALA	*Sagittaria latifolia*	Wapato	OBL	Herb	yes	3.5 (3.9)	8.7 (10.0)
POAM	*Polygonum amphibium*	Water smartweed	OBL	Herb	yes	2.8 (7.8)	6.4 (17.0)
OESA	*Oenanthe sarmentosa*	Water parsley	OBL	Herb	yes	2.8 (4.6)	1.0 (1.4)

5.4.1.2 *Key Drivers of Marsh Productivity*

There are several key biophysical drivers of marsh productivity including water levels, sediment dynamics, salinity, herbivores, and invasive species. Of these, variations in water level are the biggest driver of both the distribution and abundance of different plant species.

The depth of inundation strongly affects plant growth, as does the timing, frequency, and duration of inundation. All of these hydrologic characteristics vary annually and seasonally as the river flows change. Tidal fluctuations add a daily variable to inundation periods. At sites closer to the river mouth (Ilwaco Slough and Welch Island), tidal influence and winter storms have a stronger influence on water level dynamics than the spring freshet. The influence of the freshet increases farther upstream and contributes to a mixed set of tidal and freshet drivers at Whites Island. At Cunningham Lake and Campbell Slough, the primary driver shifts to the freshet. And finally, at the farthest upstream trend site at Franz Lake, the tidal signal is difficult to discern from the influence of dam operations. There, the marsh surface is inundated much of the time during high river levels in winter and spring. Beavers also play a role in Franz Lake water levels, elevating them in years with an active dam and lowering them when the dam is absent.

In general, marsh surface inundation times increase upriver as the influence of the winter and spring high flows increases. In addition, the inter-annual variability in inundation increases along with the relative influence of the river flows. This can be illustrated by looking at the variability in the cumulative inundation experienced by the wetlands over the course of a growing season. We do this using Sum Exceedance Values (SEV) which measure of the cumulative inundation. As can be seen in Figure 153, downstream trends sites have much lower cumulative inundation periods and those periods are experienced as daily tidal flooding that lasts for a few hours at a time. These sites also show limited variability among years, regardless of river flow volume because their hydrology is dominated by the tidal signal. Farther upstream, cumulative inundation periods increase substantially as river flow comes to dominate hydrological patterns and the freshet keeps marsh surfaces flooded for days or weeks at a time. In addition, the cumulative inundation varies greatly from year to year, depending on climate impacts on river flows. These differences in inundation patterns have significant implications for plant biomass production.

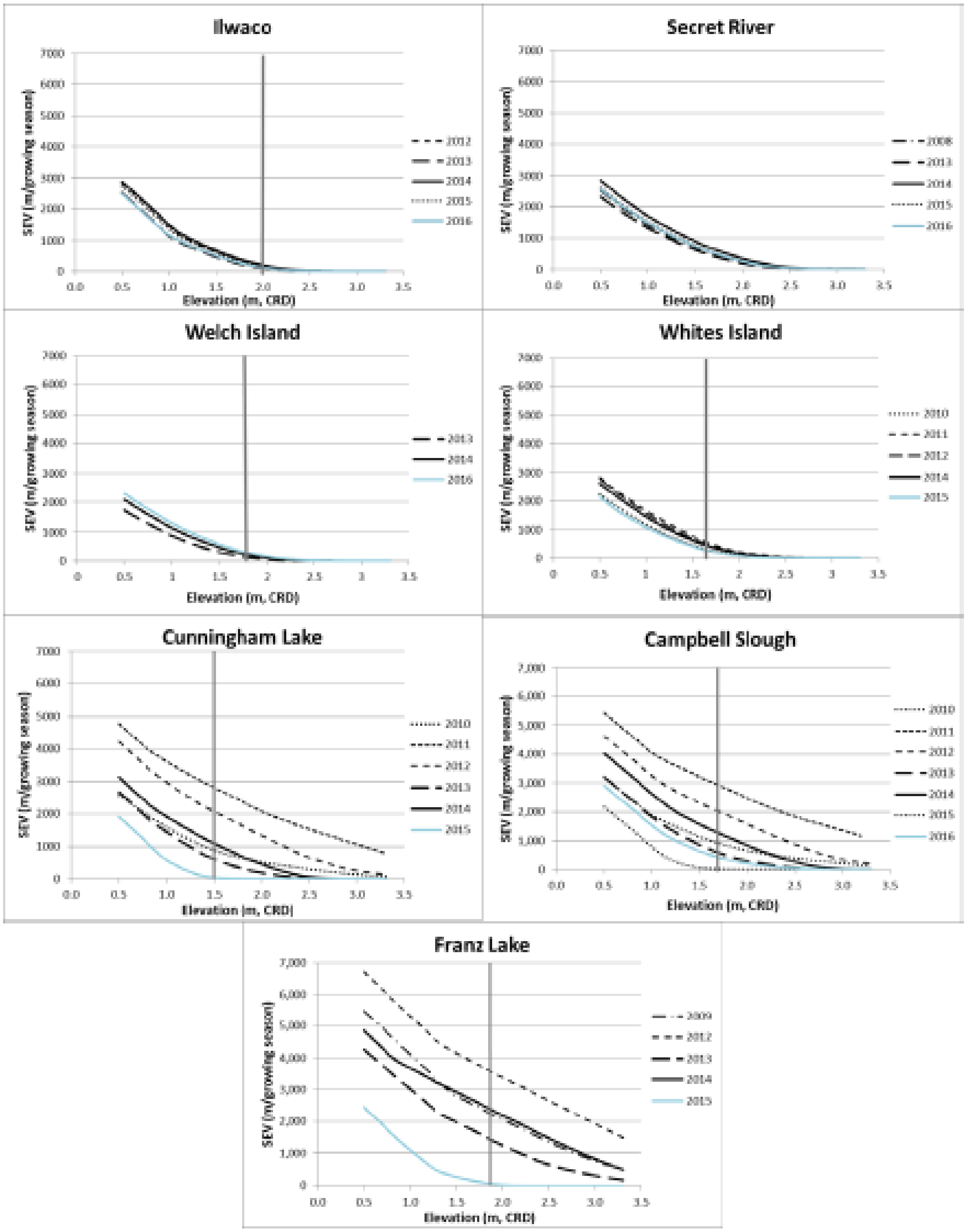


Figure 153. Cumulative inundation of wetlands during the annual growing season, as measured by sum exceedance values (SEVs). Plotted lines represent the calculated SEVs for a given year; the blue lines indicate the most recent year available. The vertical line represents the average elevation at each site. Sites are ordered with the site nearest the Columbia River mouth in the upper left panel.

5.4.1.3 *Inundation Periods Affect Marsh Productivity*

Plant species composition and productivity responds to inundation periods and to the amount of variability in inundation. In the Columbia estuary, the species composition and % cover increase in variability at upper river sites, just as inundation periods increase in variability. Furthermore, the range of variability increases with time, with the sites that have been monitored the longest having changed the most. This is a clear indicator of the importance of long-term monitoring since data from any single year tells only a small part of the story of how marsh dynamics may affect the larger food web.

In general, plant productivity declines as the inundation period increases. High marsh generally produces greater biomass than low marsh (Figure 154). This pattern is consistent in the lower estuary but becomes more variable in the upper estuary. In the upper estuary, freshet flows can inundate high marsh for extended periods of time, which can reduce productivity compared to sites closer to the river mouth (Figure 155). Low marsh is consistently flooded more often than high marsh, regardless of location in the upper or lower estuary, and there was no statistical difference in productivity in the low marsh strata between the lower and upper estuary sites.

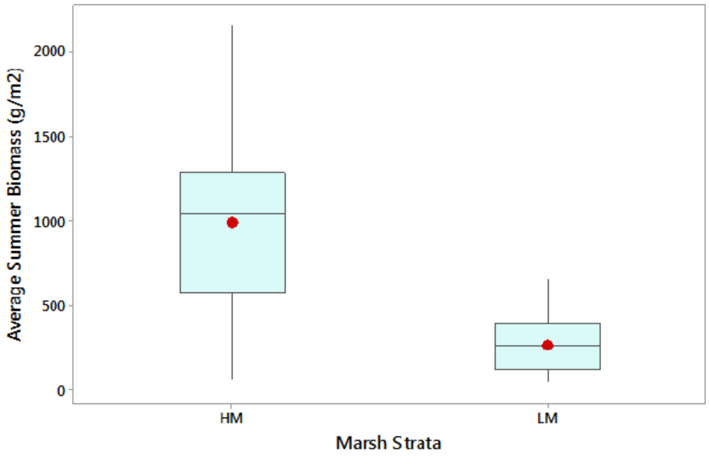


Figure 154. Overall average summer biomass (g dry weight/m^2) from the high marsh (HM) and low marsh (LM) strata.

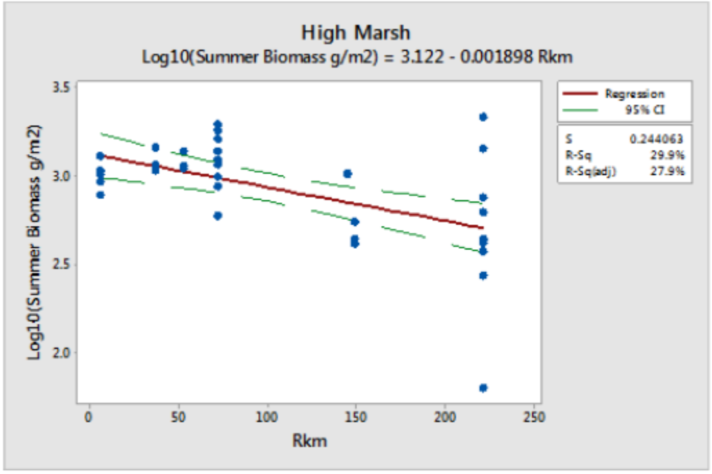


Figure 155. Average annual summer biomass (g dry weight/m^2) compared to river km for the high marsh strata. Results are transformed by Log10 for statistical analysis.

Plant species differ not only in their annual biomass production, but also in the proportion of their annual production that enters the detrital food web. We estimate the organic matter contribution to the annual food web by subtracting the winter standing stock from the summer standing stock and calculating the proportion of summer production that has been contributed. Low marsh plants contribute 80–93% of their annual aboveground biomass to the detrital food web, with particularly high values for *Sagittaria latifolia*, *Eleocharis palustris*, and SAV. Within the high marsh, communities that are dominated by the native sedge, *Carex lyngbyei*, contribute 68–80% of their annual aboveground biomass to the food web each year, averaging 882 ± 277 g dry weight/m^2. In contrast, communities dominated by the non-native reed canarygrass, *Phalaris arundinacea*, contribute only 37–72% of their annual biomass to the food web in the same year, averaging just 425 ± 381 g dry weight/m^2. In addition to contributing less of its annual biomass to the detrital food web that supports salmon prey, *P. arundinacea*'s contribution is also substantially more variable. Overall, wetlands dominated by the native sedge *C. lyngbyei* contributed the highest and most consistent amount of organic material, signifying the importance of this species to the food web in the estuary.

Macroinvertebrates have carbon isotopic signatures more similar to periphyton and the fluvial-phytoplankton-dominated particulate organic matter than to vascular and aquatic plants, suggesting that other roles in addition to direct consumption of plant material may be important in plant communities.

5.4.1.4 Phytoplankton Distribution

Fluvial phytoplankton distributions are strongly influenced by the hydrograph, with high flows being characterized almost exclusively by colonial diatoms in the mainstem Columbia upstream of the salt-influenced estuary (Maier 2014, Breckenridge et al. 2015). Lower in the estuary, seasonality in phytoplankton abundance and composition comes from river discharge and the seasonality in ocean influence. In general, the system is dominated by diatoms throughout much of the year and throughout most of the river (Lara-Lara et al. 1990). Through the Ecosystem Monitoring Program, we have documented the dominance of diatoms in phytoplankton assemblages at sites downstream of the Willamette-Columbia confluence; in contrast, at Campbell Slough and Franz Lake Slough, phytoplankton communities are not as strongly dominated by diatom and instead include greater proportions of chlorophytes, cryptophytes, cyanobacteria, and euglenophytes. Compared to diatoms, these latter groups tend to be smaller in size, with lower quality fatty acids that make them less nutritious to consumers than diatoms.

Although diatoms are more nutritious than other phytoplankton groups, colonial diatoms are often considered inaccessible to many grazers due to their large size; if they are not consumed, diatoms may be rapidly exported from the system in the mainstem, or else they may become deposited in back channels and sloughs within the system. An alternative pathway, however, arises when they become infected with lethal fungal parasites, call chytrids. These infections repackage carbon from inedible, large colonial diatoms into small, nutritious zoospores which are easily consumed by zooplankton grazers. The presence of chytrid zoospores has been demonstrated in the Columbia (Maier and Peterson, 2016) and downstream increases in infection of diatom hosts demonstrates that infections actively occur during downstream transit toward the river mouth (Maier and Peterson, 2017). This process reduces export losses of carbon associated with fluvial phytoplankton and has the potential to shunt carbon into zooplankton and the salmonid food web.

Through the EMP, we have also shown that dissolved oxygen levels are influenced by the quantity of diatoms, suggesting that they have an important effect on water quality in salmonid habitats. When other types of phytoplankton dominate the assemblages and are present in high abundance, pH levels and dissolved oxygen concentrations can become unsuitable for salmonids and other aquatic organisms, underscoring the important role that aquatic microbiota play in determining water quality and habitat characteristics.

Prior to the spring freshet, colonial diatoms dominate the phytoplankton assemblage, with high similarity among all sites except Ilwaco Slough (Hanson et al. 2016, Hanson et al. 2017). At Ilwaco Slough, the phytoplankton assemblage contains a significant proportion of benthic diatoms, which have been resuspended in the water column. At the other sites, the spring freshet dilutes populations of phytoplankton, leading to lower abundances during that period. Once water levels begin to decrease, phytoplankton populations once again increase, and the loss of connectivity to the mainstem at Campbell Slough and Franz Lake Slough result in the development of distinct phytoplankton assemblages characterized by higher proportions of flagellate taxa, including chlorophyte, cryptophyte, and chrysophyte algae. These algal groups are less nutritious than are diatoms, likely resulting in a less high-quality organic matter source supporting consumers such as microzooplankton and macrozooplankton. In addition, at both of these sites, cyanobacteria populations increase as temperatures rise, often resulting in noxious blooms (Sagar et al. 2015, Tausz 2015, Hanson et al. 2016, Hanson et al. 2017).

5.5 Conclusions

Despite the number of research studies completed in the Columbia River Estuary that provided valuable habitat data (focused mainly in Reaches A and B), the Ecosystem Monitoring Program is currently the only long-term monitoring program that consistently collects long-term habitat data in the lower river from the mouth to the upper, freshwater reaches. Data collected under the EMP provides context for action effectiveness monitoring results and EMP sites often act as reference sites to which habitat restoration sites are compared. These long-term observations are valuable for capturing the range of annual variability of environmental conditions, and the longer the monitoring program is implemented, the more descriptive the dataset becomes. This long-term data set provides a basis for evaluating how future environmental fluctuations predicted to be associated with climate change may impact salmonid habitat and food web dynamics. Future EMP research will focus on synthesizing these environmental observations and identifying how shifting climatic, and habitat conditions will impact the salmonid food web.

6 Adaptive Management & Lessons Learned

Habitat restoration practitioners look to the best available science to inform restoration design. Despite the number of research studies completed in the Columbia River Estuary that provided valuable habitat data (focused mainly in Reaches A and B), the Ecosystem Monitoring Program is currently the only long-term monitoring program that consistently collects long-term habitat data in the lower river from the mouth to the upper, freshwater reaches. Information provided under the EMP provides context for action effectiveness monitoring results and EMP sites often act as reference sites to which habitat restoration sites are compared. Long-term observations are essential for capturing the range of and potential drivers of annual variability in environmental conditions, and the longer a monitoring program is implemented, the more descriptive the dataset becomes.

The lower river and estuary provide rearing and refugia habitat for juvenile salmonid stocks originating from across the Columbia River basin. Long-term monitoring of the various stocks that use lower river habitats, migration timing through the lower river, and the extent to which salmonids use these habitats is valuable information for resource managers. Tracking fish habitat use in conjunction with abiotic variables at reference sites provides information about conditions necessary for juvenile salmon survival and, in turn, can inform habitat restoration design. In addition, EMP data track annual patterns in fish presence, size, condition, growth, and diet of juvenile salmon during their migration period. These patterns vary according to genetic stock, life history type, and whether the fish is marked or unmarked (e.g., marked fish catches correspond to the timing of hatchery releases). Such monitoring data can be used to track how fish from these different groups utilize lower river habitats during this critical time of their life cycle. However, new data suggest that the current sampling methods (specifically the timing of fish collection with respect to the tidal cycle) may not be fully inclusive of all life history types, with yearlings potentially being underrepresented in catches. The lack of new sampling methods also results in low to no catches in Franz Lake, which is a unique EMP site based on its abiotic conditions and plankton assemblages. Efforts to conduct additional sampling across the tidal range and at high tide may produce results that differ from those derived using traditional methods, and provide additional information to further explore the influence of tidal ranges.

Non-native fish species are consistently caught throughout all reaches of the lower river and estuary. It is unclear to what degree non-natives compete with juvenile salmon for resources such as food and space. Juvenile Chinook salmon consume a wide range of prey functional groups from benthic to pelagic to terrestrial-derived. As such, there is a high likelihood that prey items consumed by juvenile Chinook salmon overlap with prey items consumed by non-native species. A comprehensive examination of diet contents of non-native fish that overlap spatially and temporally with juvenile Chinook salmon would help illuminate some of these interactions that may have a substantial impact of juvenile salmon foraging success. Additionally, some non-native fish species, such as smallmouth bass and yellow perch, are predators of juvenile salmon in their adult form. Management options for controlling the numbers of these predators need to be explored.

Non-native species can pose risks to native species (e.g., increasing competition for resources, predation, the introduction of disease, reducing biodiversity, altering ecosystem function). For example, reed canarygrass (*Phalaris arundinacea*) is known to out-compete native wetland plants, and above-ground biomass data indicate that this species does not contribute the same quantity and quality of macrodetritus to the system as native species. Wetland plant distribution is highly dependent on elevation and hydrology, thus vegetation community structure and % cover can vary from year-to-year based on river discharge patterns. Long-term vegetation monitoring in emergent wetlands offers valuable information to managers seeking to control non-native plant species by helping them predict how vegetation at a recently restored site will respond to annually fluctuating river flows. These data are especially critical when trying to evaluate if restoration actions used to control *P. arudinacea* have been successful or if *P. arudinacea* abundances are changing due to natural variability.

Physical, biogeochemical, and ecological habitat characteristics across varied hydrologic years may offer insight into how environmental factors (e.g., water temperature, dissolved oxygen levels) play into the survival success of juvenile salmon. Unsuitable conditions in off-channel habitats can have negative implications for rearing juvenile salmon. Water temperatures in 2019 were higher than 2018 during late spring and summer; so were the average number of days where water temperatures exceeded relevant thresholds for salmon survival. Similar observations were made in 2015 and 2016, which were dry years, with low discharge freshets. River discharge for 2019 were generally low, similar to 2015, except for high freshet flows observed in April. These conditions, in combination with patterns observed over the past decade indicate a shift in climate patterns, which needs to be explored further.

Water quality can vary within a watershed based on season and location. Even though the EMP sites are considered to be relatively undisturbed, our results indicate that water quality values sometimes exceed water quality standards and could pose a risk to aquatic organisms. In addition, connectivity between off-channel areas and the mainstem river is important for flushing and exchange of biotic and abiotic material. In poorly flushed sites, water chemistry characteristics such as very low dissolved oxygen and high chlorophyll concentrations may cause hypoxic conditions that are harmful to aquatic life, as well as nutrient inputs that can trigger further algae growth, including the proliferation of cyanobacteria.

Based on EMP data collected over the last several years, there are a number of potential threats to the survival and growth of salmonids associated with poor water quality. For example, over the last several years, the tidal intrusion of ocean waters in Baker Bay at Ilwaco Slough in the summer months has led to increasing poor water quality in terms of dissolved oxygen saturation and pH; 2018 had the greatest number of observations of hours with low dissolved oxygen over the last several years. In 2019, while pH at Ilwaco slough were largely in the range of good water quality, in contrast, Campbell slough exceeded standard in June, and remained high through September. In some years, pH fluctuations have been outside of the range for good water quality, and chlorophyll concentrations have exceeded water quality standards, particularly at Franz Lake Slough (e.g., in 2017). High abundances of cyanobacteria have been consistently observed at both Campbell Slough and Franz Lake Slough during the summer months, with high abundances occurring occasionally in the spring as well. In general, these threats to water quality mainly occur in the summer months when water temperatures are highest.

To some extent, the threats can be mitigated through increase water volume and flushing; however, as atmospheric temperatures increase and snowpack declines with global climate change, high flows do not necessarily provide as strong a temperature buffer as they have in the past. Flows in 2017 were high relative to the long-term average; yet, there was a higher number of days with temperatures exceeding recommended values for salmonid growth and survival compared to all years but 2015, which had both low flows and high atmospheric temperatures. When water temperatures are high despite relatively high flows, cold water refugia become extremely important for salmonids. Monitoring the water quality in the lower river provides contextual information that identifies critical times periods and locations that should be targeted for management.

Water volume and quality (temperature, dissolved oxygen, pH, nutrients, chlorophyll) are driven by river flows under the influence of climatic factors that include atmospheric temperature and precipitation patterns. Biological production at the base of aquatic food webs depends directly on some these features (e.g., water residence time, temperature, nutrients) and also influences some of these features (e.g., pH, dissolved oxygen). The growth and survival of salmonids depend on food availability—which is directly tied to primary and secondary production—and to water quality parameters that influence growth and physiology (e.g., dissolved oxygen, pH, and temperature). We are developing models to infer the diet of juvenile salmon so that we can relate hydrologic characteristics to components of the food web to improve our ability to predict how hydrology will influence salmon production and survival. In particular, habitat restoration efforts should consider how interventions influence water retention time and volume; EMP data show that when waters have long retention times during warm periods, they are vulnerable to the proliferation of noxious phytoplankton blooms, which impairs water quality in terms of dissolved oxygen, temperature, and pH. Additionally, it is important for managers to consider future fluctuations predicted to be associated with climate change and the consequences of rising water temperatures when planning habitat projects.

There are a number of questions that emerge based on several years of observations in the lower Columbia. Some of these have been presented below. Based on the long-term dataset available, we recommend an EMP Synthesis study addressing some of these questions:

- ***How important are biogeochemical processes upstream of Bonneville Dam for the tidal freshwater estuary?*** It is unclear how conditions above Bonneville Dam influence water chemistry and plankton stocks observed downstream. Measurements of water quality and food web components from above the dam would help to determine the degree to which advection is important versus in situ processes such as growth and gas equilibration with the atmosphere.
- ***What is the importance of decomposition of organic matter by microbial organisms in determining its quality for salmon prey?*** Microbial decomposition often results in "trophic upgrading", whereby less labile compounds are transformed through microbial metabolism to compounds that are more easily assimilated. How are these processes influenced by water chemistry, temperature, and nature of the organic matter (e.g., non-native vs. native plant species)?
- ***What factors contribute to cyanobacteria blooms in Franz Lake Slough? Do these blooms pose a problem for wildlife, and if so, what is the extent of the problem?*** Over the last few years, elevated phosphorus concentrations have been observed at Franz Lake Slough in advance of cyanobacteria blooms, although the source is unknown.
- ***How do pulses in primary production from different sources vary in space and time, and how does this influence secondary production and salmon food webs?*** The timing of availability of different sources of organic matter produced through primary production varies between pelagic phytoplankton and marsh vegetation. It would be helpful to compare the magnitude of these stocks to identify patterns that could inform food web models. In addition, pulse events, such as the production and deposition of pollen, could produce reservoirs of organic matter originating from vascular plants in the water column that is independent of detritus transport.
- ***How does prey quality and quantity vary spatially and temporally across the estuary?*** While studies have shown that emergent wetlands are important for prey production and export, accurate assessments of information on prey source in the mainstem and floodplain habitats are yet to be made in the lower Columbia river. The spatial and temporal variation of energy densities of chironomids and amphipods in these undisturbed sites of the lower Columbia river would provide an important functional tool for restoration design.
- ***How does mainstem cumulative discharge effect prey availability and juvenile salmon health and habitat use?*** Additional information is needed to explore the effect of different mainstem hydrologic conditions on the food web and habitat structure for the EMP. Since many EMP sites serve as reference sites for restoration projects, additional information about changes in habitat use and structure under various freshet conditions would help determine crucial actions in restoration design, and mitigate effects of climate change.
- ***How much do specific environmental factors impact growth, fish condition, residence time, age at maturation and survival of anadromous salmonids in the estuary?*** Habitat use in the lower Columbia depends on a myriad of abiotic conditions, and a closer look into specific

characteristics such as temperature, DO, discharge, etc. would provide critical information about juvenile salmonid behavior which can be used to inform landscape principles in restoration planning.

- ***How does sediment carbon interact with Greenhouse gases in EMP Trend Sites?*** In order to understand the effects of climate change on the EMP sites, another aspect that needs to be explored further are the exchanges between carbon and greenhouse gases in emergent wetlands. While some data is available from sediment analysis, further exploration is required in terms of accretion and nutrients and carbon sequestration.

The Estuary Partnership shares results from the monitoring program with other resource managers in the region and results from this multi-faceted program are applied to resource management decisions. Results from the EMP are presented and discussed at an annual Science Work Group meeting. The Science Work Group is composed of over 60 individuals from the lower Columbia River basin representing multiple regional entities (i.e., government agencies, tribal groups, academia, and private sector scientists) with scientific and technical expertise who provide support and guidance to the Estuary Partnership. In addition, EMP results will also be shared with regional partners at the Salmon Recovery Conference in April 2021. Data are often provided to restoration practitioners for use in restoration project design and project review templates (e.g., ERTG templates). Finally, data from the EMP are used to compare and contextualize results from the Action Effectiveness Monitoring Program (see Kidd et al. 2020). A regional database to store and share these data would be a valuable contribution to this work and allow for wider data dissemination and use.

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8 Appendices

Appendix A. Site Maps and Habitat Change Analysis

Site maps and habitat change analysis were conducted in 2015.

Contents:

Baker Bay, 2015

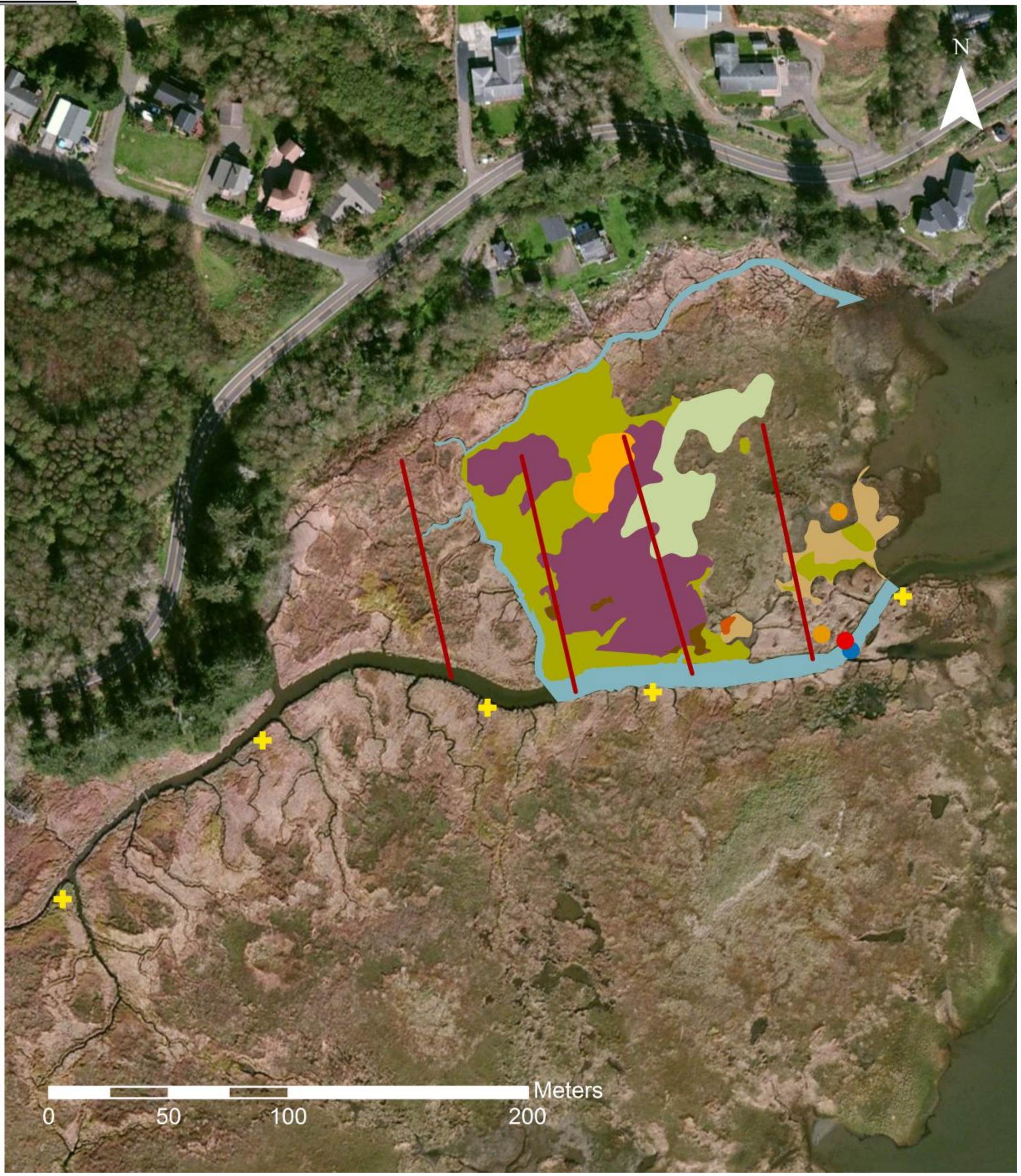
GPS Mapping

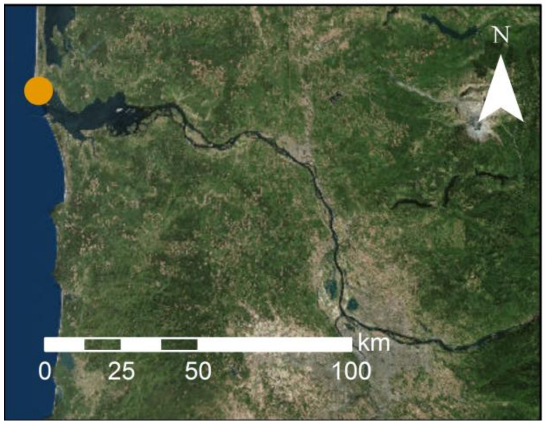
Vegetation Communities

- 🟩 Carex lyngbyei
- 🟧 Isolepis cernua
- 🟪 Scirpus americanus, Agrostis spp., C. lyngbyei (stunted)
- 🟨 Typha spp.
- 🟫 Bare ground
- 🟦 Channel
- 🟩 Mixed grass
- 🟫 Pan

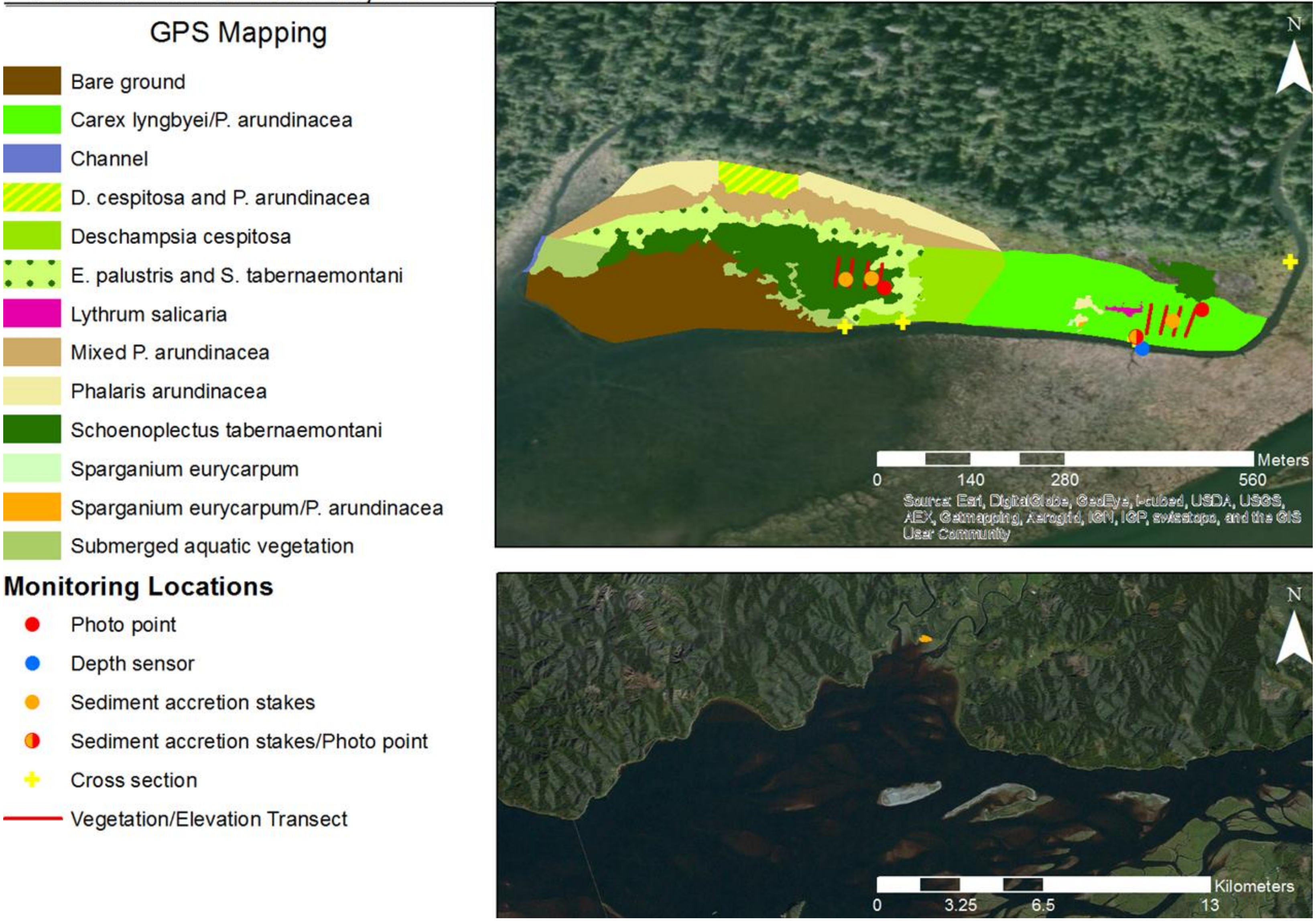
Monitoring Locations

- ✚ Cross section endpoints
- ● Depth sensor
- ● Photo point
- ● Sediment accretion stakes
- — Vegetation/Elevation

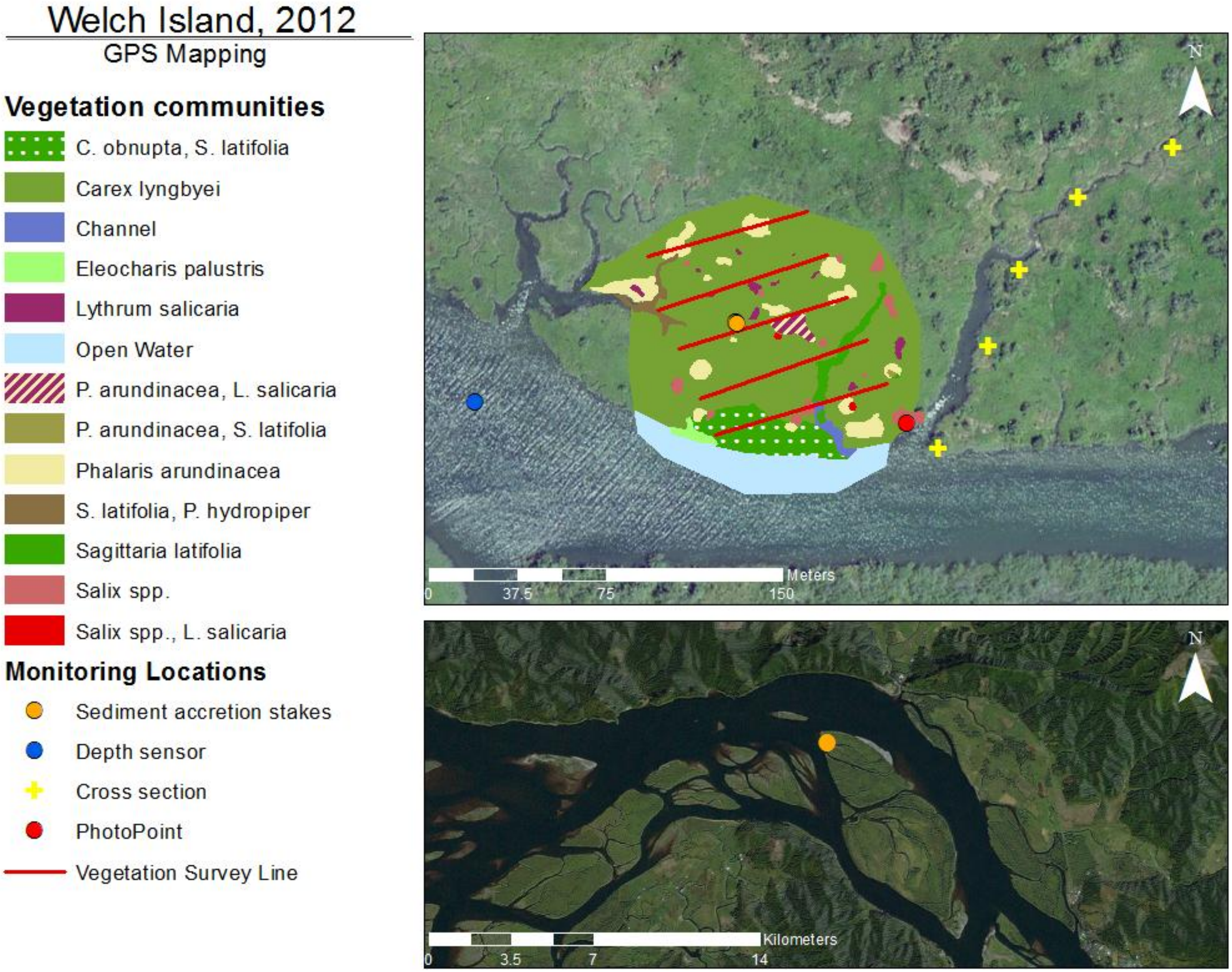


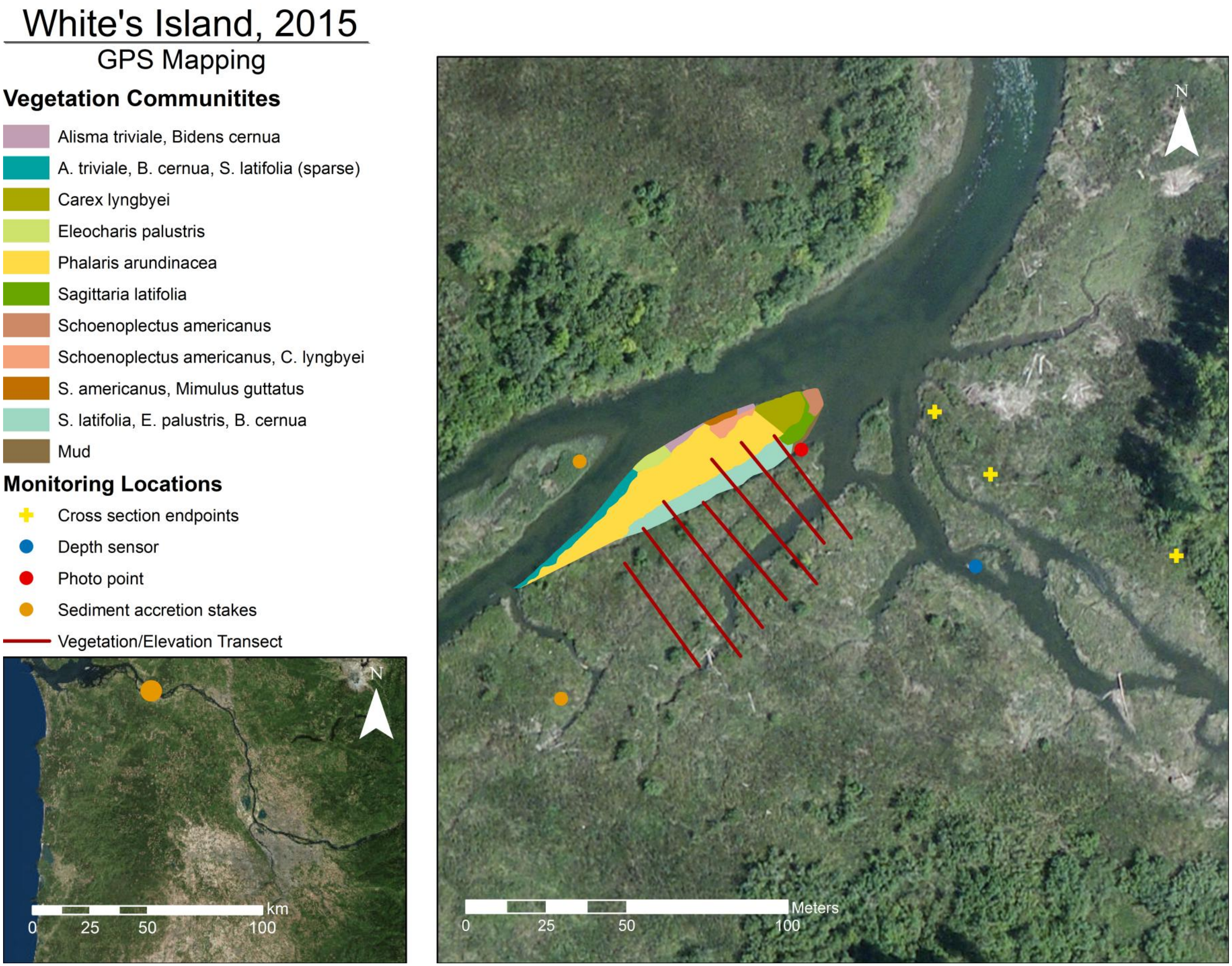


Secret River Marsh, 2013



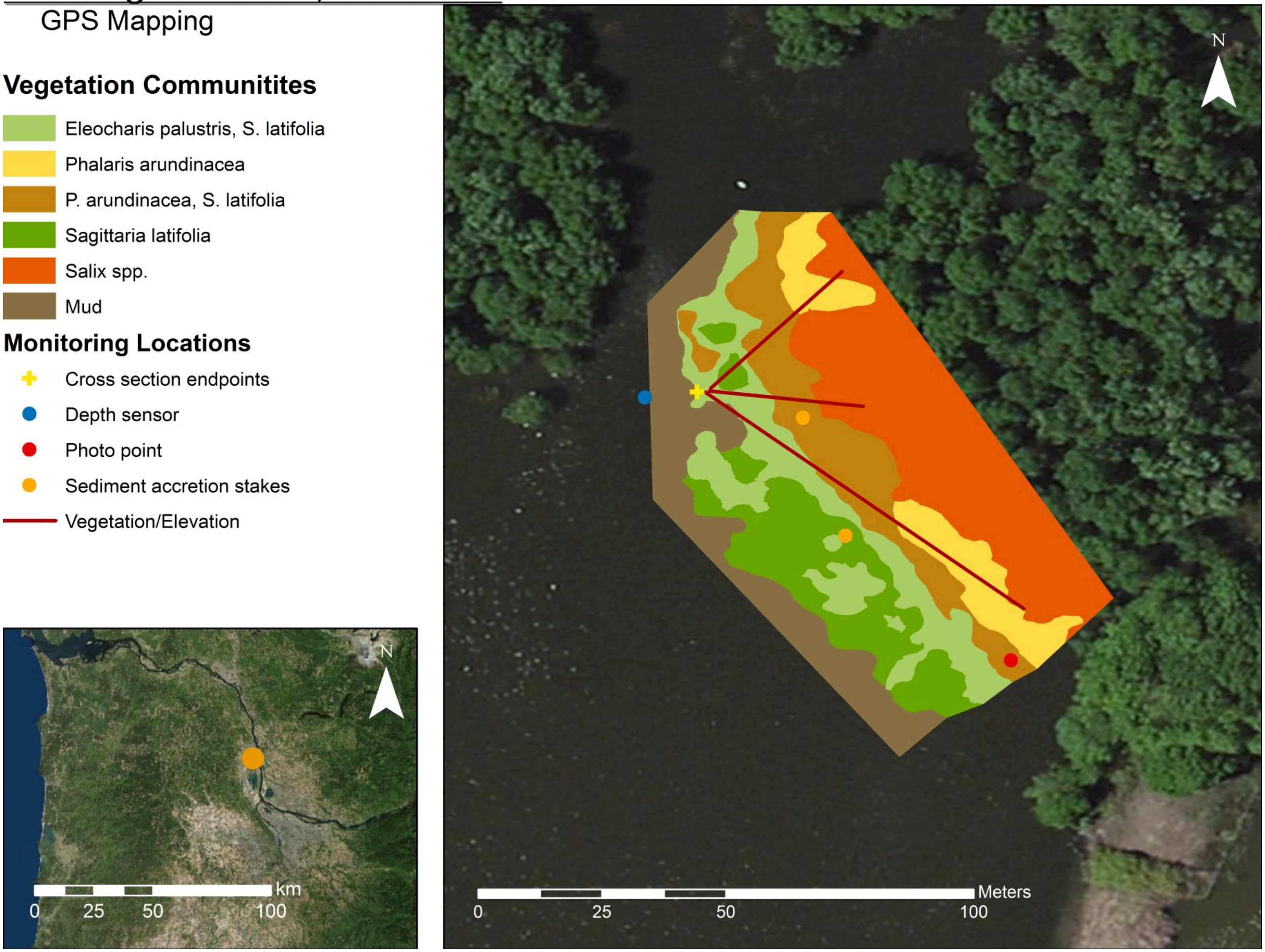
Source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community


Welch Island, 2012
GPS Mapping
Vegetation communities
C. obnupta, S. latifolia
Carex lyngbyei
Channel
Eleocharis palustris
Lythrum salicaria
Open Water
P. arundinacea, L. salicaria
P. arundinacea, S. latifolia
Phalaris arundinacea
S. latifolia, P. hydropiper
Sagittaria latifolia
Salix spp.
Salix spp., L. salicaria
Monitoring Locations
Sediment accretion stakes
Depth sensor
Cross section
PhotoPoint
Vegetation Survey Line
N
Meters
0
37.5
75
150
N
Kilometers
0
3.5
7
14


White's Island, 2015
GPS Mapping
Vegetation Communitites
Alisma triviale, Bidens cernua
A. triviale, B. cernua, S. latifolia (sparse)
Carex lyngbyei
Eleocharis palustris
Phalaris arundinacea
Sagittaria latifolia
Schoenoplectus americanus
Schoenoplectus americanus, C. lyngbyei
S. americanus, Mimulus guttatus
S. latifolia, E. palustris, B. cernua
Mud
Monitoring Locations
Cross section endpoints
Depth sensor
Photo point
Sediment accretion stakes
Vegetation/Elevation Transect
N
km
0
25
50
100
N
Meters
0
25
50
100

Cunningham Lake, 2015

GPS Mapping



Campbell Slough, 2015

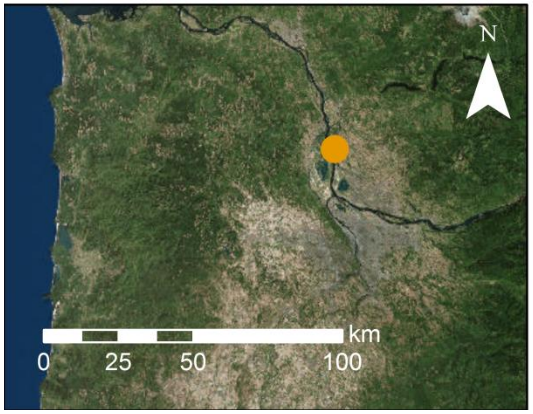
GPS Mapping

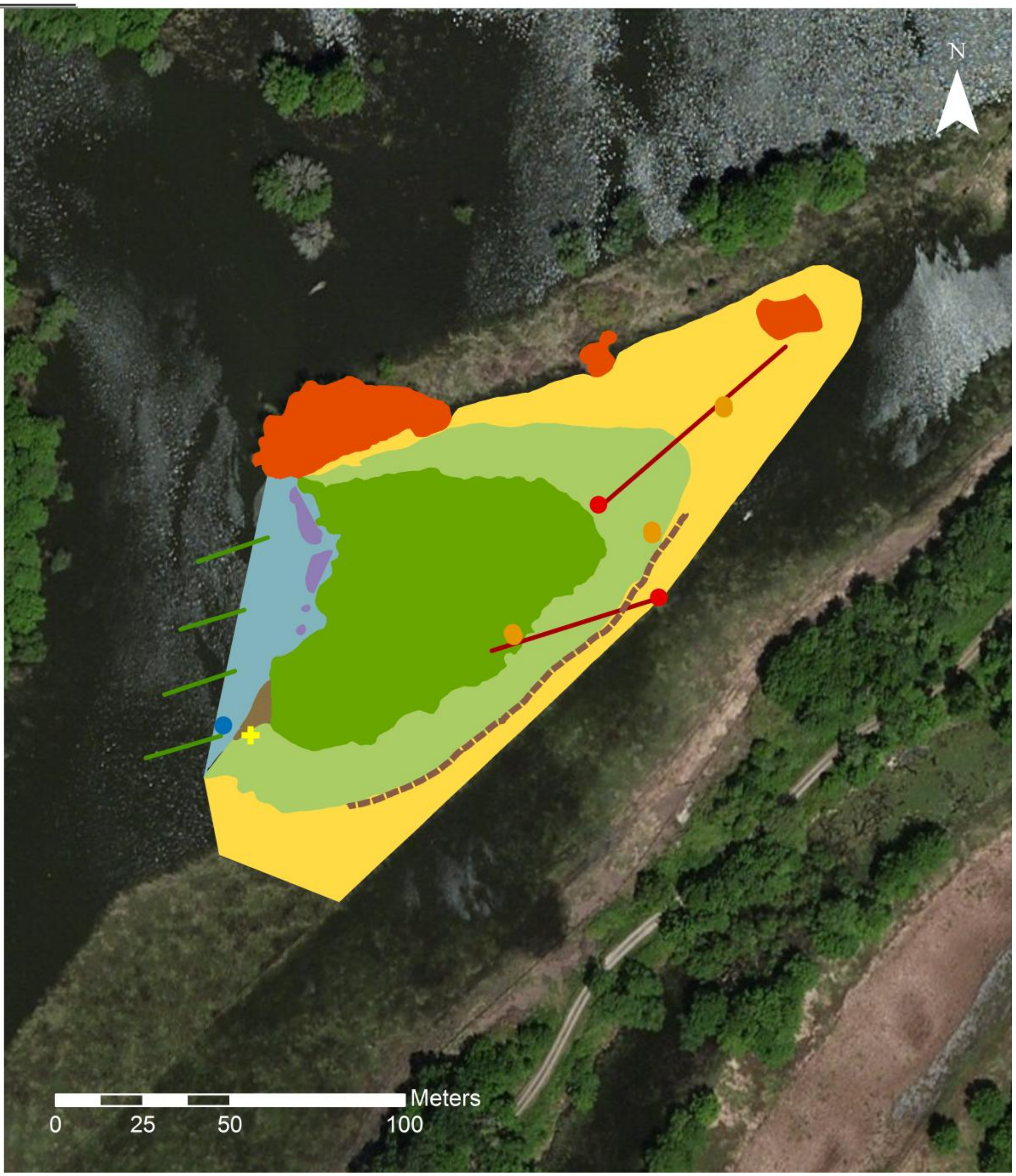
Vegetation Communities

- 🟩 E. palustris, S. latifolia
- 🟨 Phalaris arundinacea
- 🟪 Potamogeton natans
- 🟩 Sagittaria latifolia
- 🟧 Salix spp., Fraxinus latifolia
- 🟦 Channel
- 🟫 Mud
- ---- cow trample

Monitoring Locations

- ✚ Cross section end point
- ● Depth sensor
- ● Photo point
- ● Sediment accretion stakes
- — SAV/Elevation
- — Vegetation/Elevation





Franz Lake, 2015

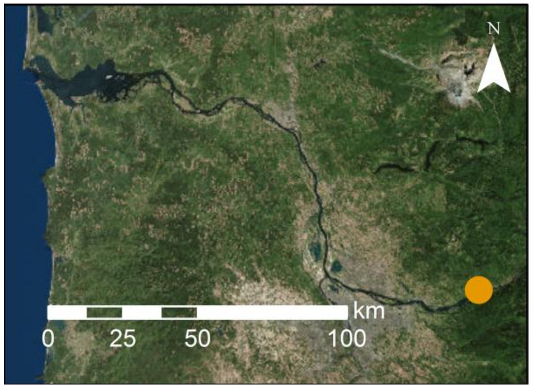
GPS Mapping

Vegetation Communities

- 🟩 Carex spp.
- 🟩 Eleocharis palustris
- 🟩 E. palustris, S. latifolia
- 🟫 Phalaris arundinacea, Helenium autumnale
- 🟪 Polygonum amphibium
- 🟫 P. amphibium, Salix spp.
- 🟪 P. arundinacea, P. amphibium
- 🟩 Sagittaria latifolia
- 🟧 Salix spp.
- 🟨 Salix spp., P. arundinacea
- 🟩 S. latifolia, E. palustris, Carex spp.

Monitoring Locations

- ✚ Cross section endpoints
- 🔵 Depth sensor
- 🔴 Photo point
- 🟠 Sediment accretion stakes
- — Vegetation/Elevation



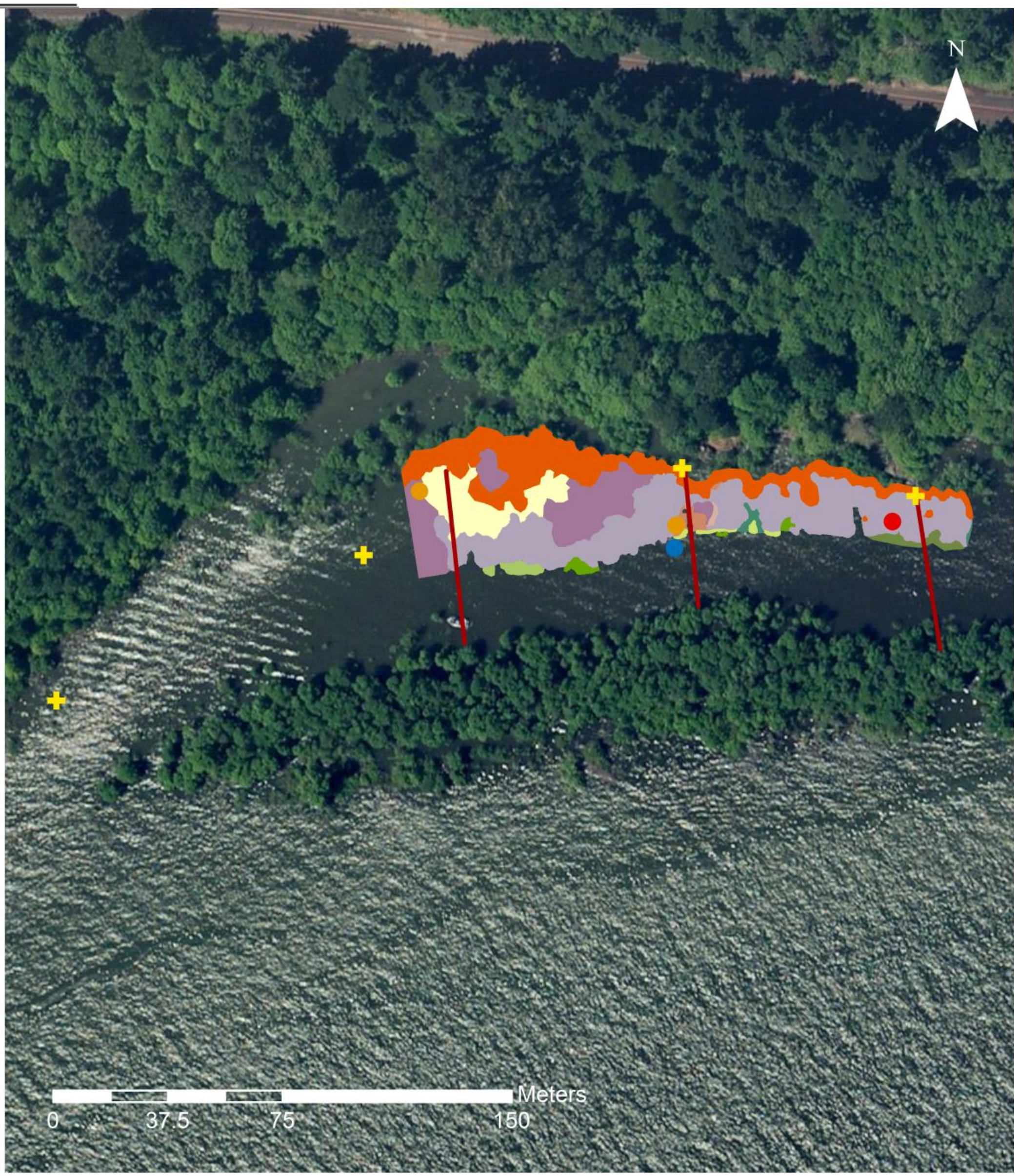


Table A1. Habitat change analysis of vegetation communities at the trend sites; comparison of overlapping areas for the earliest year mapped and the latest year mapped. All area units are square meters. Vegetation communities are ordered from the lowest elevation to the highest elevation at a site; species codes are provided in Appendix D. Sites are ordered in the table starting at the mouth of the Columbia River and moving upstream.

Ilwaco

			2011 Vegetation Community				
	Area Compared:	**13312**					
	Area Changed:	**6416**	**Channel, ZAPA**	**Pan**	**CALY**	**AGSP, CALY**	**TYSP**
	No Change:	**6895**	**1558**	**383**	**6455**	**4792**	**134**
2015 Vegetation Community	**Channel**	**2164**	1548				
	Pan	**804**		241	559	4	
	Bare	**127**			68	58	
	CALY	**3898**	10	142	3045	85	
	AGSP, CALY	**4048**			1188	2860	
	AGSP, DECE, GLSP	**1754**				1754	
	TYSP	**517**			356	28	134

Welch Island

			2012 Vegetation Community						
	Area Compared:	**1126**							
	Area Changed:	**603**	**Channel**	**SALA**	**CALY, high marsh**	**PHARSALA**	**PHAR**	**PHAR LYSA**	**LYSA**
	No Change:	**523**	**8**	**15**	**126**	**8**	**838**	**116**	**15**
2015 Veg Community	**PHAR**	**812**	8	15	126	8	523	116	15
	Un-mapped Vegetation	**314**					314		

Whites Island

	Area Compared:	1585	2009 Vegetation Community						
	Area Changed:	729	Channel	SALA	ELPA	ELPA, SALA	ELPA, SCAM	CALY	PHAR
	No Change:	855	163	252	115	18	82	191	763
2015 Vegetation Community	Mud	15	6	8					
	ALPL, BICE	43	31				12		
	SALA	55		14	31			10	
	SALA, ALPL, BICE	107	86				14		8
	SALA, ELPA, BICE	297		230	36	13			17
	SCAM	34			34				
	SCAM, MIGU	19	7				12		
	SCAM, CALY	39	5					23	11
	ELPA	40	13				26		
	CALY	114						114	
	PHAR	823	14		14	5	18	44	727

Cunningham Lake

	Area Compared:	4033	2006 Vegetation Community					
	Area Changed:	1800	SALA	ELPA, SALA	ELPA	PHAR, ELPA	PHAR	SASP
	No Change:	2232	1059	1041	44	278	492	1118
2015 Vegetation Community	Mud	587	557	30				
	SALA	634	366	269				
	ELPA, SALA	673	136	537				
	PHAR, SALA	612		205	44	234	107	21
	PHAR	381				44	285	52
	SASP	1145					100	1045

Campbell Slough

	Area Compared:	13476	2005 Vegetation Community			
	Area Changed:	2551	SALA	ELPA, SALA	PHAR	SASP
	No Change:	10925	4719	2905	5636	216
2015 Vegetation Community	SALA	4434	4133	301		
	ELPA, SALA	3632	586	2276	770	
	PHAR	4955		328	4441	111
	SASP, FRLA	381			276	105
	Cow Trample	74			74	

Franz Lake

			2008 Vegetation Community				
	Area Compared:	1762					
	Area Changed:	1430	Channel, SALA	ELPA	PHAR, POAM	SASP	Rock
	No Change:	**331**	**25**	**372**	**1047**	**303**	**15**
2015 Vegetation Community	SALA	**5**		5			
	SALA, ELPA, CASP	**81**	16	65			
	ELPA	**35**		35			
	CASP	**34**	9	25			
	POAM	**1097**		216	848	34	
	POAM, SASP	**10**			10		
	PHAR, HEAU	**27**		27			
	PHAR, POAM	**28**			28		
	SASP	**445**			161	269	15

Franz Lake

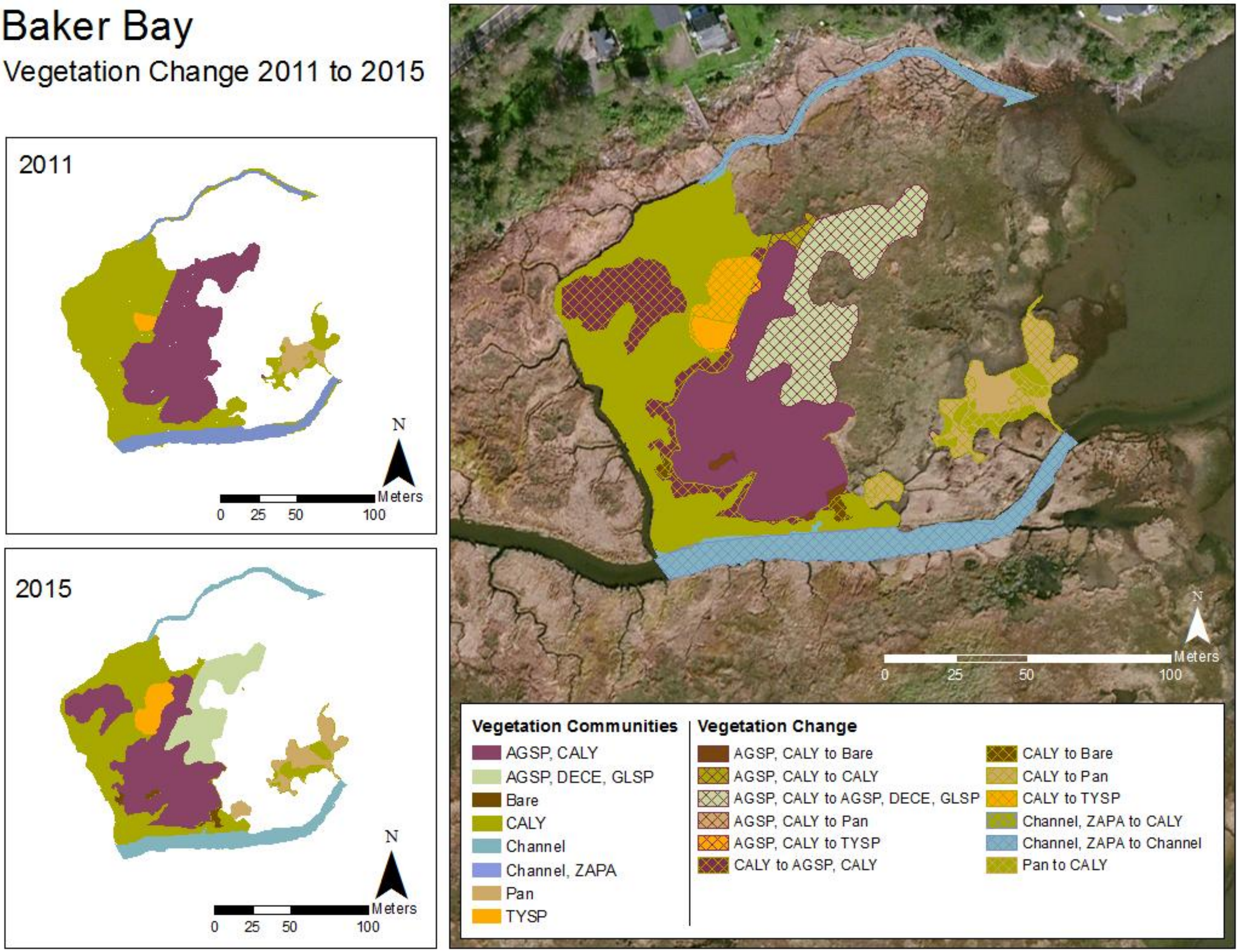
			2008 Vegetation Community					
	Area Compared:	5720						
	Area Changed:	4336	Channel, SALA	ELPA, SALA	ELPA	PHAR, POAM	SASP	Rock
	No Change:	1384	**1417**	**24**	**488**	**2177**	**1579**	**35**
2012 Vegetation Community	**Channel**	**1182**	1081	4	69	28		
	Channel, SALA	**147**	112		35			
	SALA	**37**	15	9	4	9		
	ELPA	**344**	133	5	92	106	8	
	CASP	**178**	6	6	39	121	6	
	POAM, SALA	**7**				7		
	POAM	**2310**	71		249	1645	344	
	SASP, CASP	**41**					41	
	SASP	**1475**				260	1180	35

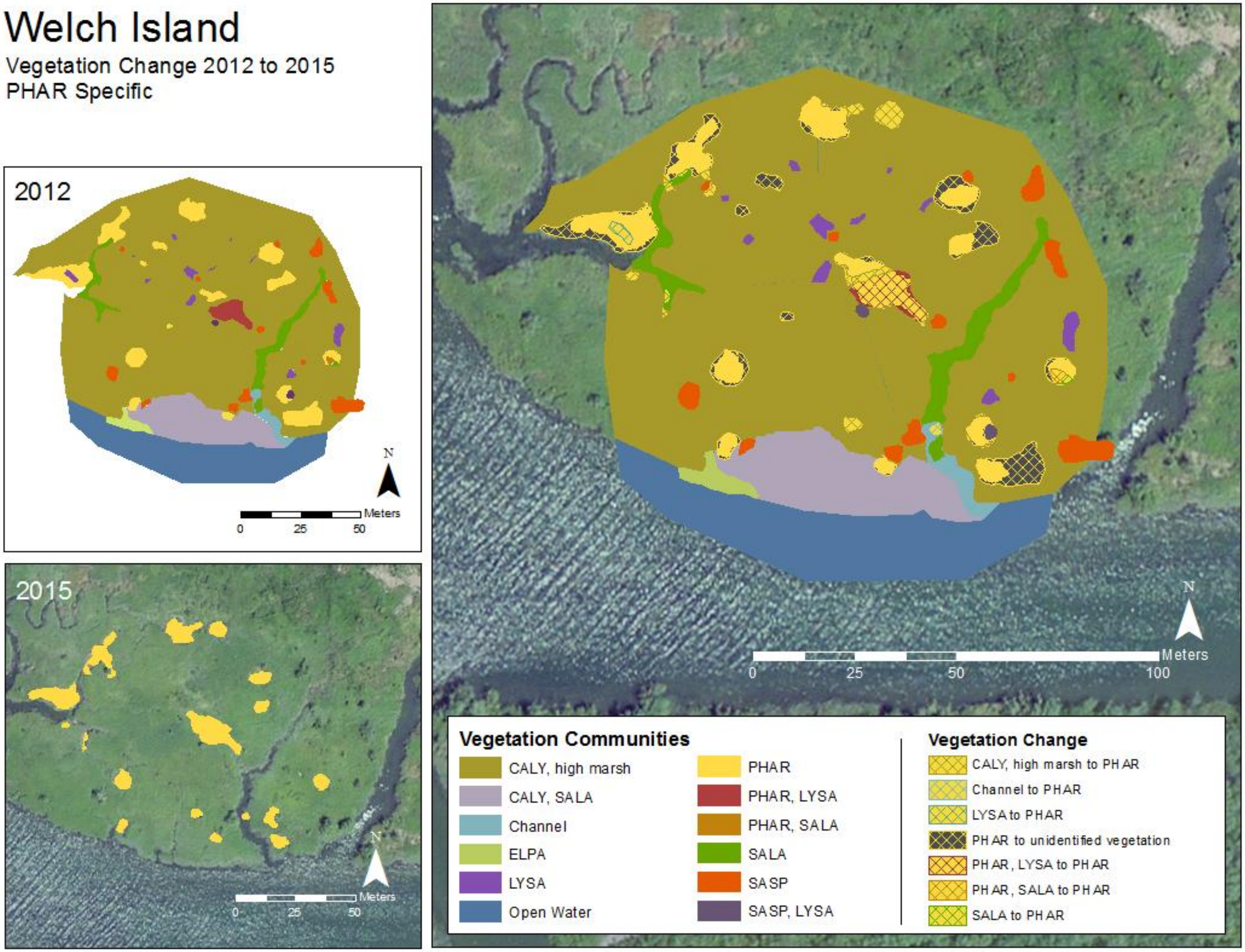
Franz Lake

			2012 Vegetation Community												
	Area Compared:	5203													
	Area Changed:	2344	Bare Ground	Beaver Activity	Channel	Channel, SALA	SALA	ELPA	CASP	POAM, SALA	POAM	PHAR	SASP, PHAR	SASP, CASP	SASP
	No change:	**2859**	**9**	**14**	**194**	**50**	**5**	**193**	**192**	**25**	**2159**	**196**	**279**	**20**	**1865**
2015 Vegetation Community	**SALA**	**41**	9		11	13					8				
	SALA, ELPA, CASP	**79**			19	27		9			24				
	ELPA	**25**			7	6		11							
	ELPA, SALA	**58**			45		5	8							
	CASP	**37**				4		14			18				
	PHAR, HEAU	**49**						18	31						
	POAM	**2131**		8	40			69	93	25	1748	12			137
	POAM, PHAR	**867**		6	72			47	27		152	103	135		324
	POAM, SASP	**10**							10						
	SASP	**1387**						18	31		170	39	8	20	1100
	SASP, PHAR	**522**									39	43	136		304

Baker Bay

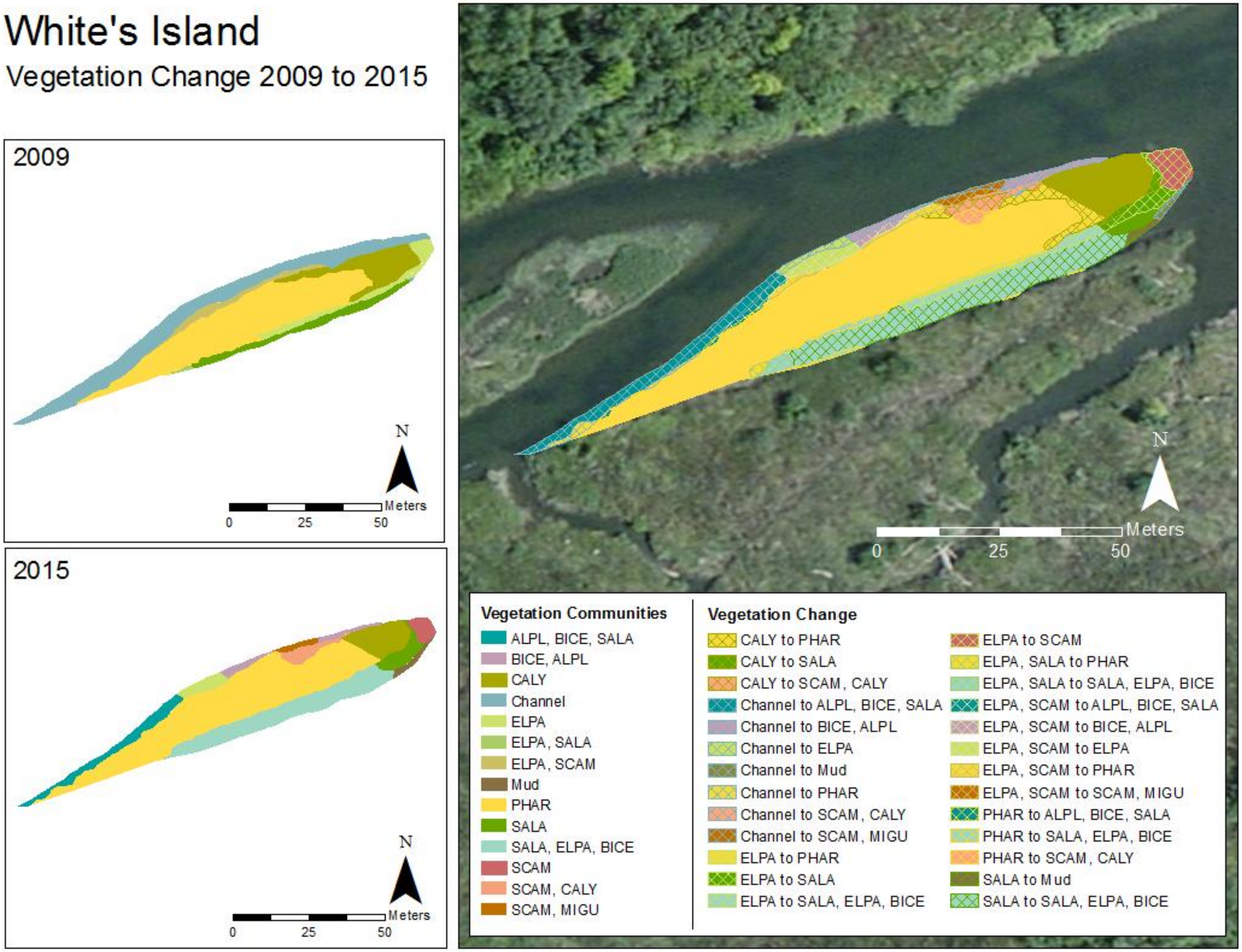
Vegetation Change 2011 to 2015

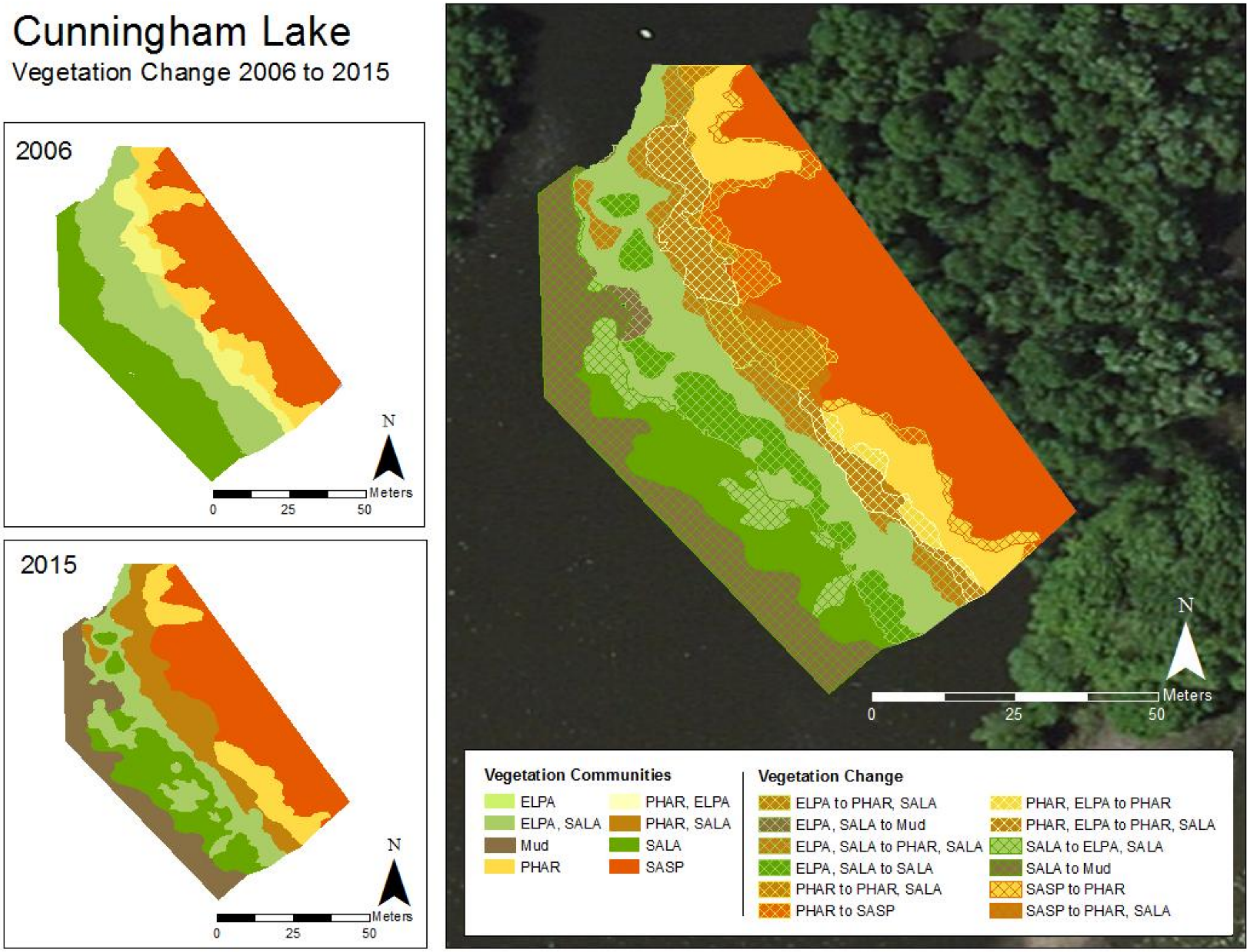


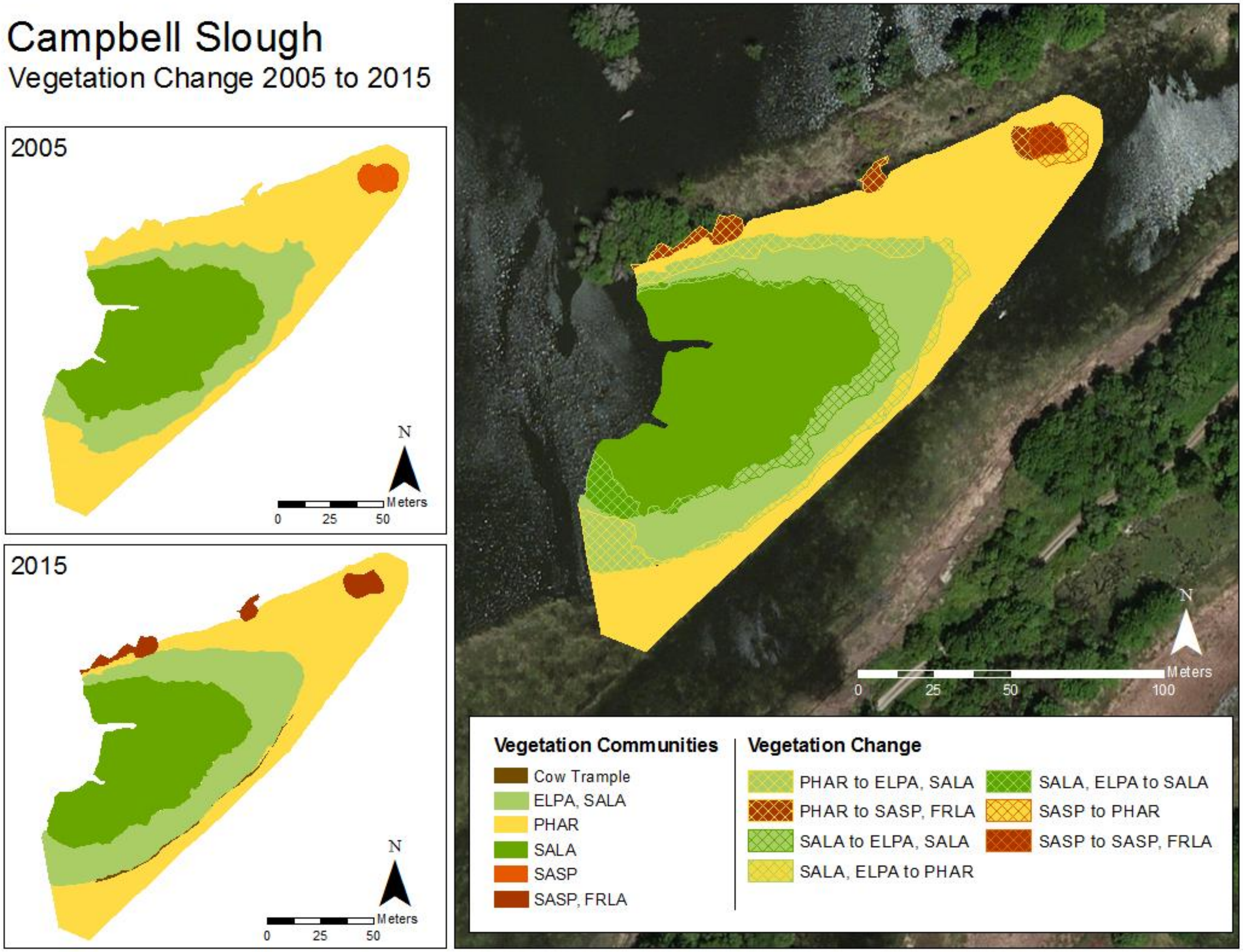

Welch Island
Vegetation Change 2012 to 2015
PHAR Specific
2012
N
Meters
0
25
50
2015
N
Meters
0
25
50
N
Meters
0
25
50
100
Vegetation Communities
CALY, high marsh
CALY, SALA
Channel
ELPA
LYSA
Open Water
PHAR
PHAR, LYSA
PHAR, SALA
SALA
SASP
SASP, LYSA
Vegetation Change
CALY, high marsh to PHAR
Channel to PHAR
LYSA to PHAR
PHAR to unidentified vegetation
PHAR, LYSA to PHAR
PHAR, SALA to PHAR
SALA to PHAR

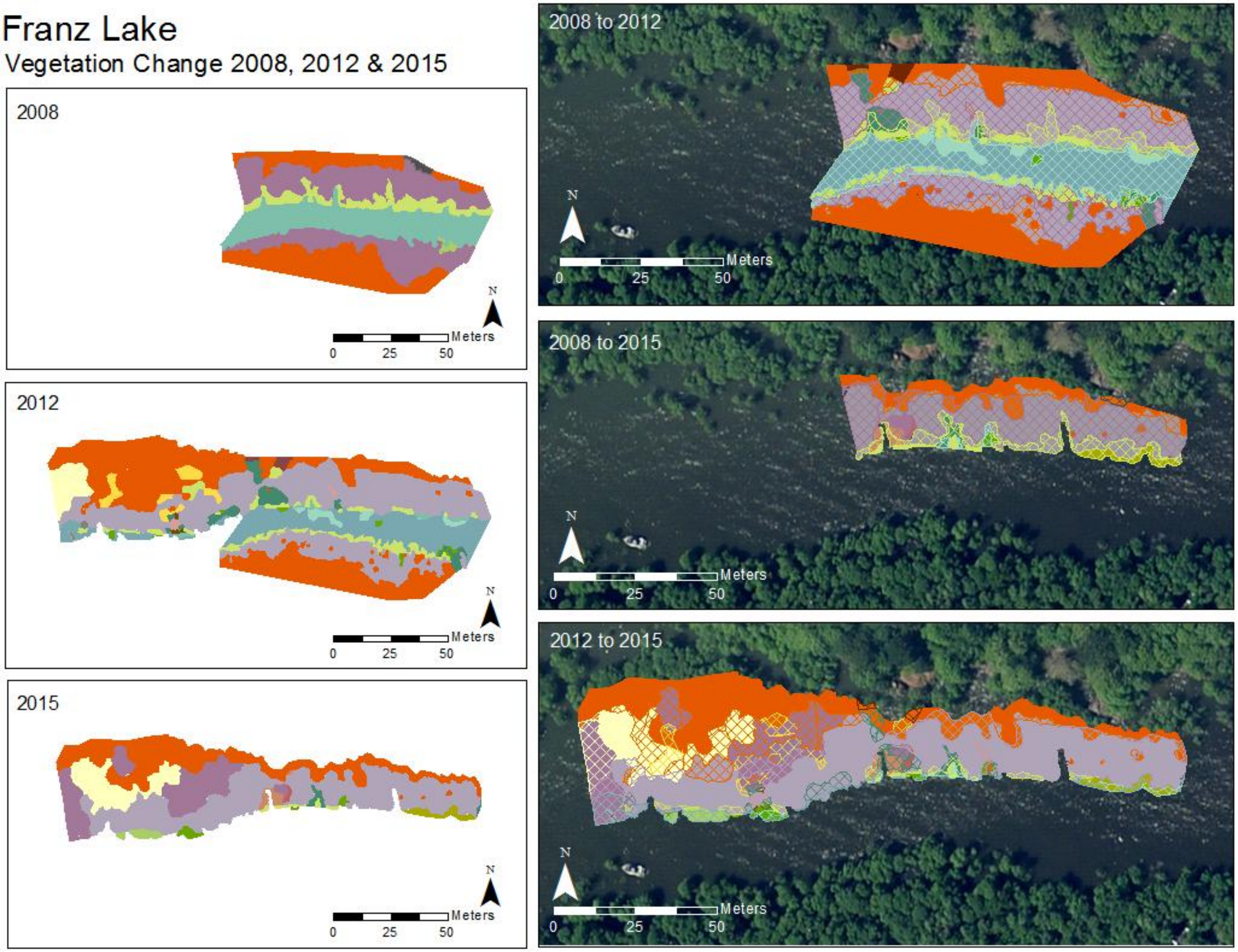
White's Island

Vegetation Change 2009 to 2015




Cunningham Lake
Vegetation Change 2006 to 2015
2006
N
Meters
0
25
50
2015
N
Meters
0
25
50
N
Meters
0
25
50
Vegetation Communities
ELPA
ELPA, SALA
Mud
PHAR
PHAR, ELPA
PHAR, SALA
SALA
SASP
Vegetation Change
ELPA to PHAR, SALA
ELPA, SALA to Mud
ELPA, SALA to PHAR, SALA
ELPA, SALA to SALA
PHAR to PHAR, SALA
PHAR to SASP
PHAR, ELPA to PHAR
PHAR, ELPA to PHAR, SALA
SALA to ELPA, SALA
SALA to Mud
SASP to PHAR
SASP to PHAR, SALA


Campbell Slough
Vegetation Change 2005 to 2015
2005
N
0 25 50 Meters
2015
N
0 25 50 Meters
N
0 25 50 100 Meters
Vegetation Communities
Cow Trample
ELPA, SALA
PHAR
SALA
SASP
SASP, FRLA
Vegetation Change
PHAR to ELPA, SALA
PHAR to SASP, FRLA
SALA to ELPA, SALA
SALA, ELPA to PHAR
SALA, ELPA to SALA
SASP to PHAR
SASP to SASP, FRLA


Franz Lake
Vegetation Change 2008, 2012 & 2015
2008
2012
2015
2008 to 2012
2008 to 2015
2012 to 2015
N
Meters
0
25
50

Baker Bay – PP1

Appendix B. Annual photo points from EMP trends sites

Photo points collected in 2017 are still under analysis and unavailable at the time of the writing of this report.

31 July 2011

15 February 2012

4 August 2012

4 February 2013

Baker Bay – PP1

26 July 2013

20 September 2013

3 February 2014

27 June 2014

Baker Bay – PP1

2 August 2015

7 August 2016

Secret River – PP1 [HIGH MARSH]

5 February 2010

2 August 2012

9 August 2013

1 August 2015

Secret River – PP1 [HIGH MARSH]

6 August 2016

Secret River – PP2 [LOW MARSH]

1 December 2011

2 August2012

15 July 2014

1 August 2015

Secret River – PP2 [LOW MARSH]

6 August 2016

Secret River – PP3 [CHANNEL]

1 December 2011

……

15 July 2014

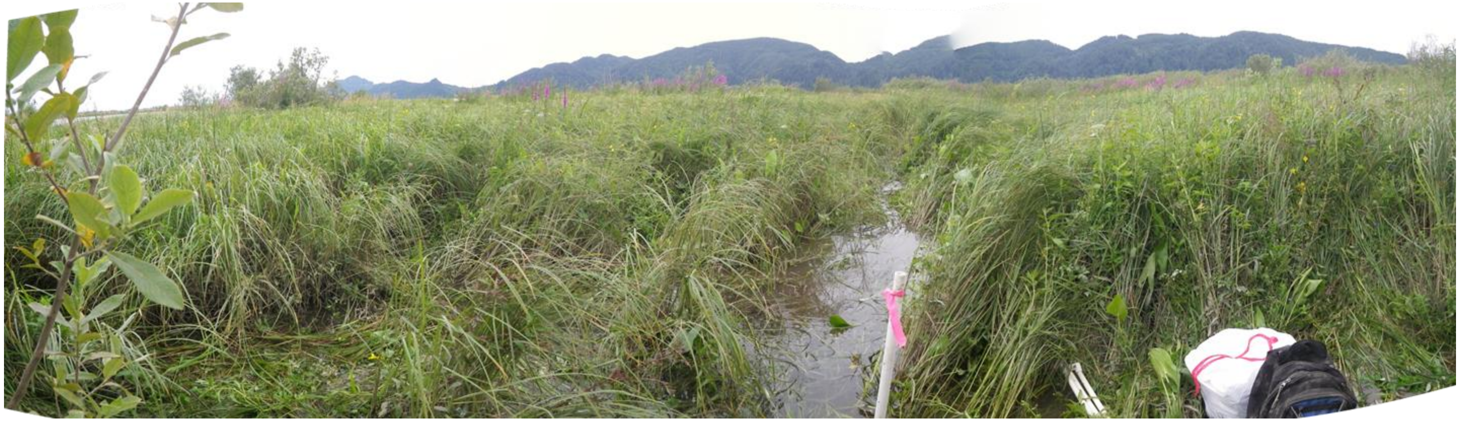
Secret River – PP3 [CHANNEL]

1 August 2015

6 August 2016

Welch Island – PP1

1 August 2012



3 February 2013

23 July 2013

Welch Island – PP1

1 August 2014

31 July 2015

5 August 2016

Whites Island – PP1

22 July 2009

13 July2010

2 August 2011

15 February 2012

Whites Island – PP1

31 July 2012

5 February 2013

22 July 2013

Whites Island – PP1

4 February 2014

31 July 2014

30 July 2015

Whites Island – PP1

4 August 2016

Cunningham Lake – PP1

26 July 2005

18 July 2007

21 July 2008

Cunningham Lake – PP1

25 July 2009

17 May 2010

28 July 2010

Cunningham Lake – PP1

30 July 2011

8 August 2012

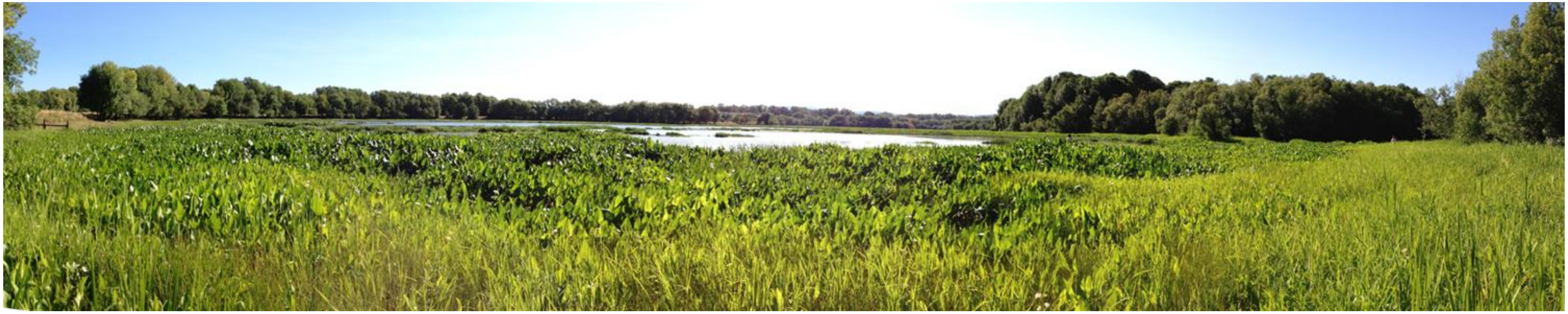
29 July 2013

18 July 2014

Cunningham Lake – PP1

28 July 2015

1 August 2016



Campbell Slough – PP1

29 July 2005

---

15 July 2006

---

5 September 2006

Campbell Slough – PP1

17 July 2007

---

26 July 2010

29 July 2011

Campbell Slough – PP1

15 February 2012

21 July 2012

10 August 2012

Campbell Slough – PP1

27 July 2013

18 July 2014

29 July 2015

Campbell Slough – PP1

3 August 2016

Campbell Slough – PP2

25 July 2005

----

27 July 2009

26 July 2010

29 July 2011

10 Aug 2012

Campbell Slough – PP2

27 July 2013

18 July 2014

29 July 2015

3 August 2016

22 July 2008

28 July 2009

25 August 2011

14 February 2012

21 July 2012

30 August 2012

11 October 2012

6 February 2013

31 July 2013

12 February 2014

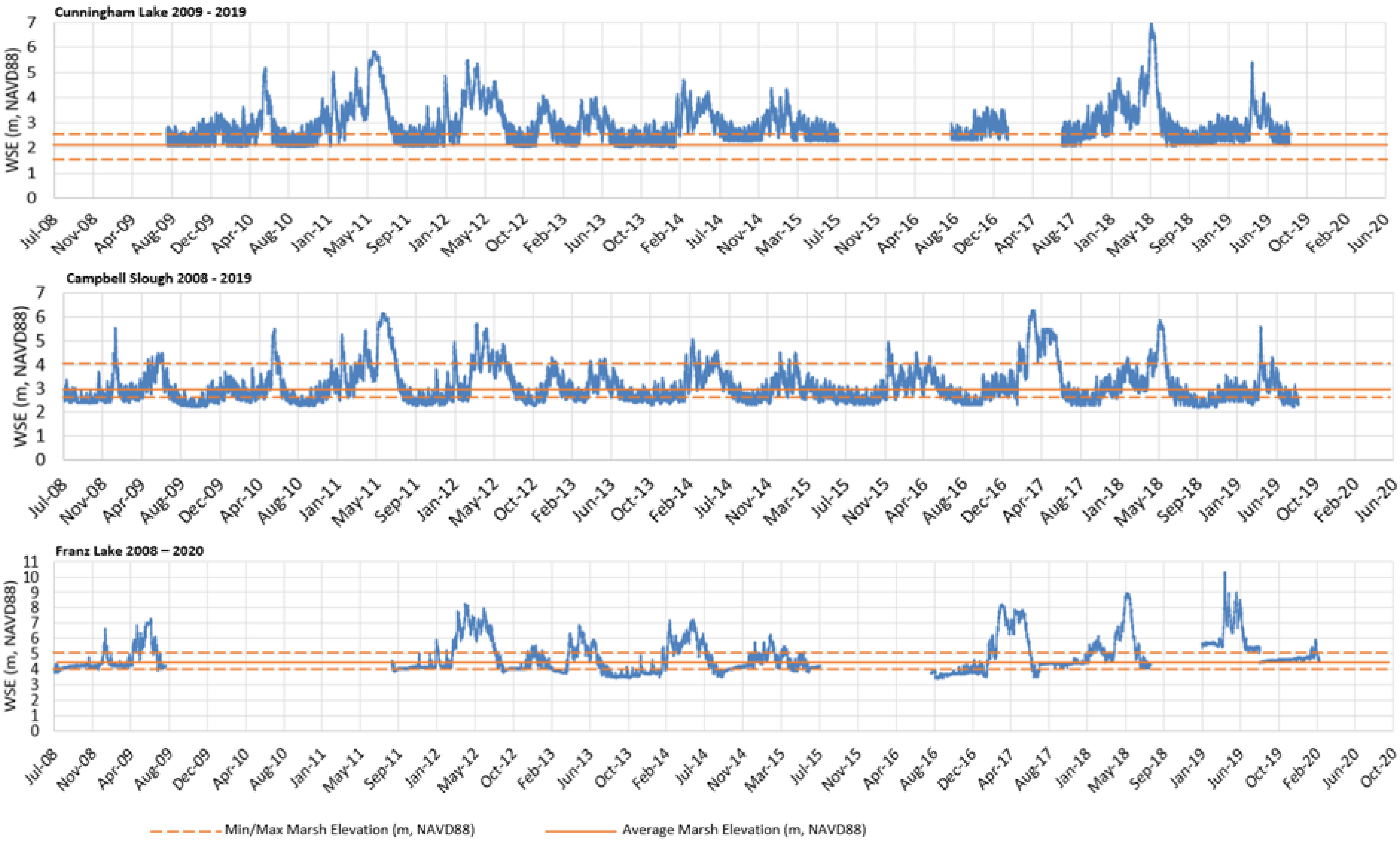
7 August 2014

27 July 2015

2 August 2016

Appendix C. Site Hydrographs

Hydrographs are in order by site location in the River, starting at the mouth. Followed by hydrology summary statistics for each site. *2012 data for Welch Island is of questionable quality due to movement of logger during deployment.


Cunningham Lake 2009 - 2019

Campbell Slough 2008 - 2019

Franz Lake 2008 – 2020

Table C.1. Hydrologic summary statistics for each site and year. See methods section 2.3.2.1 for definitions and calculations.

	Years	2019	2018	2017	2016	2015	2014	2013	2012	2011	2010	2009	2008
Ilwaco Slough	Duration	NA	Jan-Jul	Jan-Feb, Aug-Dec	Aug-Dec	Feb-Jul	Jan-Oct	Jan-Dec	Jan-Dec	Apr-Dec			
	Days	NA	211	216	147	172	304	365	366	270			
	MHHW	NA	2.5	2.4	2.4	2.4	2.4	2.3	2.5	2.4			
	MLLW	NA	1.0	1.0	0.9	1.0	1.0	0.9	1.0	1.0			
	MWL	NA	1.5	1.5	1.4	1.4	1.5	1.4	1.5	1.4			
	Annual Range	NA	1.5	1.5	1.5	1.4	1.5	1.4	1.5	1.4			
	Annual Max	NA	3.1	3.3	3.2	3.0	3.3	3.0	3.4	3.5			
Whites Island	Duration	Jan-Dec	Jan-Jul	Jan-Dec	Aug-Dec	Jan-Dec	Jan-Dec	Jul-Dec	Jan-Jul	Jan-Dec	Jan-Dec	July-Dec	
	Days	365	212	365	153	365	365	163	213	365	365	163	
	MHHW	2.8	3.4	3.1	2.8	2.8	2.9	2.6	3.0	2.9	2.8	2.6	
	MLLW	1.1	1.6	1.5	1.1	1.1	1.1	1.0	1.3	1.2	1.1	1.0	
	MWL	1.8	2.4	2.2	1.9	1.9	1.9	1.7	2.1	1.9	1.8	1.6	
	Annual Range	1.7	1.8	1.6	1.7	1.7	1.7	1.6	1.7	1.6	1.7	1.6	
	Annual Max	3.7	4.1	4.0	4.7	4.1	3.8	3.2	3.8	3.8	3.6	3.3	
Welch Island	Duration	Sept-Dec	Jan-Feb, May-July	Jan-Dec	Jan-Dec	Jan-Apr, Aug-Dec	Jan-Dec	Feb-Dec	Jan-Oct	Dec			
	Days	113	149	365	366	257	365	332	298	31			
	MHHW	2.8	3.0	3.0	2.8	2.9	2.8	2.6	2.6	2.4			
	MLLW	0.6	0.9	0.9	0.6	0.7	0.6	0.4	0.4	0.3			
	MWL	1.7	1.9	2.0	1.7	1.7	1.7	1.5	1.5	1.3			
	Annual Range	2.1	2.1	2.1	2.2	2.2	2.2	2.2	2.2	2.1			
	Annual Max	3.3	4.0	3.9	3.9	4.0	3.7	3.3	3.5	3.0			

Cunningham Lake	**Duration**	**Jan-Aug**	**Jan-Jul**	**Jan, Aug-Dec**	**Aug-Dec**	**Jan-Jul**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Aug-Dec**	
	Days	**223**	**219**	**193**	**152**	**209**	**365**	**365**	**366**	**365**	**365**	**160**	
	MHHW	**3.1**	**4.1**	**3.0**	**2.9**	**3.1**	**3.2**	**2.9**	**3.5**	**3.5**	**3.0**	**2.7**	
	MLLW	**2.6**	**3.6**	**2.4**	**2.5**	**2.6**	**2.7**	**2.4**	**3.1**	**3.1**	**2.5**	**2.1**	
	MWL	**2.8**	**3.8**	**2.7**	**2.7**	**2.8**	**2.9**	**2.6**	**3.3**	**3.3**	**2.7**	**2.4**	
	Annual Range	**0.5**	**0.5**	**0.6**	**0.4**	**0.5**	**0.4**	**0.5**	**0.4**	**0.5**	**0.5**	**0.6**	
	Annual Max	**5.4**	**7.0**	**4.2**	**3.6**	**4.3**	**4.7**	**4.0**	**5.5**	**5.8**	**5.2**	**3.2**	
Campbell Slough	**Duration**	**Jan-Aug**	**Jan-Jul**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Aug-Dec**
	Days	**225**	**220**	**365**	**362**	**365**	**365**	**365**	**366**	**364**	**365**	**365**	**164**
	MHHW	**3.2**	**3.8**	**3.8**	**3.2**	**3.1**	**3.4**	**3.1**	**3.7**	**3.7**	**3.1**	**3.1**	**2.8**
	MLLW	**2.7**	**3.4**	**3.4**	**2.8**	**2.7**	**3.0**	**2.7**	**3.3**	**3.3**	**2.8**	**2.7**	**2.5**
	MWL	**2.9**	**3.6**	**3.6**	**3.0**	**2.9**	**3.2**	**2.8**	**3.4**	**3.5**	**2.9**	**2.9**	**2.6**
	Annual Range	**0.4**	**0.4**	**0.4**	**0.4**	**0.4**	**0.4**	**0.4**	**0.4**	**0.3**	**0.4**	**0.4**	**0.3**
	Annual Max	**5.6**	**5.9**	**6.3**	**4.5**	**5.0**	**5.1**	**4.3**	**5.7**	**6.2**	**5.5**	**5.6**	**3.7**
Franz Lake	**Duration**	**Jan-Dec**	**Jan-Jul**	**Jan-Dec**	**Aug-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Jan-Dec**	**Sep-Dec**		**Jan-July**	**Aug-Dec**
	Days	**341**	**218**	**365**	**152**	**208**	**365**	**365**	**366**	**129**		**209**	**163**
	MHHW	**5.6**	**5.8**	**5.3**	**3.9**	**4.8**	**4.9**	**4.5**	**5.4**	**4.2**		**5.2**	**4.2**
	MLLW	**5.4**	**5.5**	**5.1**	**3.7**	**4.5**	**4.7**	**4.2**	**5.2**	**4.1**		**4.9**	**4.1**
	MWL	**5.5**	**5.6**	**5.2**	**3.8**	**4.6**	**4.8**	**4.3**	**5.3**	**4.1**		**5.0**	**4.2**
	Annual Range	**0.3**	**0.3**	**0.2**	**0.2**	**0.3**	**0.3**	**0.3**	**0.3**	**0.1**		**0.3**	**0.1**
	Annual Max	**10.3**	**8.9**	**8.2**	**4.6**	**6.3**	**7.2**	**6.9**	**8.2**	**5.1**		**7.3**	**4.9**

Appendix D. Vegetation Species Cover

Table D.1. Site marsh elevation range in meters based on the vegetation plot elevation (with ≥5% absolute living plant cover), relative to the North American Vertical Datum of 1988 (NAVD88). Mean number of plots, mean elevation, standard deviation (SD), minimum elevation (Min), and maximum elevation (Max).

		Mean (SD)	2019	2018	2017	2016	2015	2014	2013	2012	2011	2010	2009	2008	2007	2006	2005
Ilwaco Slough	Plots (n)	40 (1)	40	40	40	39	37	40	40	40	40						
	Mean	1.95 (0.05)	1.86	1.94	1.91	1.94	2.00	2.03	1.96	1.96	1.92						
	SD		0.30	0.23	0.23	0.17	0.16	0.24	0.23	0.23	0.23						
	Min	1.04 (0.29)	0.70	0.95	0.94	1.44	1.61	0.93	0.94	0.95	0.89						
	Max	2.35 (0.08)	2.28	2.31	2.31	2.31	2.38	2.53	2.37	2.36	2.31						
	Range	1.31 (0.29)	1.58	1.36	1.38	0.87	0.77	1.60	1.44	1.41	1.42						
Welch Island	Plots (n)	41 (1)	43	40	41	40	40	40	40	40							
	Mean	2.04 (0.03)	1.98	2.00	2.06	2.06	2.06	2.06	2.06	2.06							
	SD		0.21	0.17	0.16	0.16	0.16	0.16	0.16	0.16							
	Min	1.31 (0.06)	1.22	1.22	1.34	1.34	1.34	1.34	1.34	1.34							
	Max	2.19 (0.03)	2.14	2.14	2.20	2.20	2.20	2.20	2.20	2.20							
	Range	0.88 (0.03)	0.92	0.92	0.86	0.86	0.86	0.86	0.86	0.86							
Whites Island	Plots (n)	41 (6)	45	44	42	42	47	43	45	42	42	35	25				
	Mean	2.09 (0.09)	2.33	2.09	2.08	2.08	2.03	2.08	2.06	2.10	2.09	2.02	2.01				
	SD		0.45	0.36	0.38	0.38	0.40	0.38	0.38	0.33	0.34	0.38	0.45				
	Min	1.21 (0.03)	1.29	1.21	1.20	1.20	1.20	1.20	1.20	1.21	1.16	1.18	1.25				
	Max	2.57 (0.12)	2.92	2.63	2.53	2.53	2.53	2.53	2.53	2.52	2.50	2.56	2.49				
	Range	1.36 (0.10)	1.63	1.42	1.33	1.33	1.33	1.33	1.33	1.31	1.35	1.39	1.24				

Table D.1. Site marsh elevation range in meters based on the vegetation plot elevation (with ≥5% absolute living plant cover), relative to the North American Vertical Datum of 1988 (NAVD88). Mean number of plots, mean elevation, standard deviation (SD), minimum elevation (Min), and maximum elevation (Max).

		Mean (SD)	2019	2018	2017	2016	2015	2014	2013	2012	2011	2010	2009	2008	2007	2006	2005
Cunningham Lake	Plots (n)	57 (15)	69	67	68	69	69	36	31	59	59	61	62	62	64	62	20
	Mean	2.7 (0.07)	2.67	2.69	2.78	2.78	2.76	2.52	2.83	2.67	2.66	2.67	2.68	2.69	2.71	2.70	2.71
	SD		0.20	0.19	0.19	0.19	0.19	0.24	0.16	0.17	0.17	0.17	0.17	0.17	0.17	0.16	0.14
	Min	2.31 (0.11)	2.30	2.29	2.41	2.41	2.40	2.00	2.48	2.28	2.28	2.28	2.28	2.25	2.31	2.33	2.41
	Max	2.99 (0.07)	3.01	3.07	3.02	3.02	3.02	2.79	3.05	2.96	2.96	3.01	2.96	3.01	2.95	3.02	2.98
	Range	0.67 (0.07)	0.71	0.77	0.60	0.60	0.62	0.78	0.56	0.68	0.68	0.73	0.68	0.76	0.65	0.68	0.56
Campbell Slough	Plots (n)	57 (14)	62	60	60	61	62	59	61	61	60	62	61	64	62	61	8
	Mean	2.98 (0.05)	2.96	3.01	2.95	2.99	2.98	3.02	3.00	2.98	2.98	2.98	3.10	2.95	2.95	2.97	2.86
	SD		0.37	0.37	0.32	0.36	0.36	0.36	0.36	0.37	0.37	0.38	0.38	0.42	0.41	0.38	0.57
	Min	2.46 (0.09)	2.50	2.40	2.51	2.51	2.51	2.53	2.53	2.47	2.46	2.45	2.59	2.22	2.41	2.41	2.45
	Max	3.99 (0.08)	3.97	4.01	3.76	4.00	4.00	4.02	4.02	3.96	4.00	4.02	4.17	4.01	4.01	4.00	3.98
	Range	1.53 (0.11)	1.47	1.61	1.25	1.49	1.49	1.48	1.48	1.49	1.53	1.57	1.58	1.78	1.60	1.59	1.53
Franz Lake	Plots (n)	56 (11)	66	64	60	61	67	61	59	62	58		35	37			
	Mean	4.59 (0.04)	4.63	4.62	4.60	4.60	4.56	4.62	4.60	4.54	4.64		4.60	4.51			
	SD		0.30	0.29	0.23	0.23	0.27	0.23	0.23	0.33	0.23		0.27	0.21			
	Min	3.92 (0.13)	3.97	3.87	3.95	3.95	3.85	4.00	4.04	3.63	4.09		3.90	3.96			
	Max	5.04 (0.04)	5.05	5.05	5.01	5.01	5.01	5.05	5.05	5.00	5.07		5.13	5.00			
	Range	1.11 (0.11)	1.08	1.19	1.06	1.06	1.15	1.05	1.01	1.36	0.98		1.22	1.04			

Table D.2. Average percent cover of each plant species at trend sites in 2019, only species with ≥1% cover shown below. Species are sorted by their four-letter code (1st two letters of genus and 1st two letters of species).

Sp. Code	Scientific Name	Native (Y/N)	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
AGST	*Agrostis stolonifera L.*	No	10.7 (9.9)	0 (0.1)	0.2 (0.7)			0 (0.1)
AREG	*Argentina egedii ssp. Egedii*	Yes	1.7 (4.9)	2.1 (3)				0.5 (3.3)
BICE	*Bidens cernua*	Yes		1.5 (3.3)	0.6 (2.1)	0 (0.1)	0 (0.1)	
CAAP	*Carex aperta*	Yes						1.6 (5.2)
CAHE	*Callitriche heterophylla*	Yes		0.1 (0.3)	1.6 (3.9)		0.1 (0.4)	
CALY	*Carex lyngbyei*	Yes	40.6 (31.7)	27.3 (17.9)	4 (10.6)			
CAPA	*Caltha palustris*	Yes		3.2 (4.7)	0.1 (0.4)			
CASE	*Calystegia sepium*	No			3.1 (4.5)			
DECE	*Deschampsia cespitosa*	Yes	2.8 (10.1)	0.2 (0.9)				
ELAC	*Eleocharis acicularis*	Yes				2.5 (8.1)	0.1 (0.4)	
ELCA	*Elodea canadensis*	Yes		0.1 (0.3)	2.4 (8.5)			
ELOV	*Eleocharis ovata*	Yes				0.3 (2.4)	5.8 (9)	
ELPA	*Eleocharis palustris*	Yes		5 (11.4)	4.5 (12.4)	24.1 (27)	15.8 (20)	0.8 (2.5)
ELPAR	*Eleocharis parvula*	Yes	1.8 (7.7)		0.1 (0.9)			
EQFL	*Equisetum fluviatile*	Yes		4.5 (7)	7 (7.6)			
EQPA	*Equisetum palustre*	Yes				0.5	0 (0.2)	0.4

Sp. Code	Scientific Name	Native (Y/N)	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
						(2.7)		(1.2)
FGA	*Filamentous green algae*		2.2 (12.5)		0.4 (1.9)		0.4 (1.6)	
FRLA	*Fraxinus latifolia*	Yes						1.8 (10.3)
FUDI	*Fucus distichus*	Yes	1.4 (4.7)					
GATR	*Galium trifidum L. spp. columbianum*	Yes		3.9 (13.9)	0.4 (1.1)			
GLMA	*Glaux maritima*	Yes	1 (3.4)					
HEAU	*Helenium autumnale*	Yes					2 (5.7)	0.7 (2.6)
IMCA	*Impatiens capensis*	Yes		1.2 (2.1)	0.9 (2.7)			
IRPS	*Iris pseudacorus*	No		0.6 (2.6)	3 (6.1)	0.2 (1.7)		
ISCE	*Isolepis cernua*	Yes	2.1 (5.2)					
JUOX	*Juncus oxymeris*	Yes		2.3 (3.3)	0.8 (2.5)		0 (0.2)	
LEOR	*Leersia oryzoides*	Yes		1.2 (2.2)	2.2 (3.7)	2 (3.9)		1.7 (3.5)
LIDU	*Lindernia dubia*	Yes				0.2 (1)	3.4 (4.9)	0.9 (1.9)
LIOC	*Lilaeopsis occidentalis*	Yes	6 (6.8)					
LOCO	*Lotus corniculatus*	No		0.3 (1.7)	1.9 (4.8)		0.3 (1)	
LUPA	*Ludwigia palustris*	Yes				2.3 (4.4)	3.8 (6.5)	
LYAM	*Lysichiton americanus*	Yes		1.3 (3.5)				
LYAM2	*Lycopus americanus*	Yes		1.3 (8.5)	2.7 (5.2)			

Sp. Code	Scientific Name	Native (Y/N)	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
LYNU	*Lysimachia nummularia L.*	No				0.1 (0.5)	2.8 (5.2)	
LYSA	*Lythrum salicaria*	No		1.6 (6.4)	2 (6.6)			
MEAR	*Mentha arvensis*	Yes		2.4 (8.6)			1 (2.6)	0.3 (0.9)
MIGU	*Mimulus guttatus*	Yes		1.8 (2.4)	0.6 (1.6)			
MYLA	*Myosotis laxa*	Yes		2.5 (5.5)		0 (0.1)		
MYSC	*Myosotis scorpioides*	No		9.3 (11.5)	5.7 (9.7)			
OESA	*Oenanthe sarmentosa*	Yes		5 (6.2)	2.3 (2.8)			
POAM	*Polygonum amphibium*	Yes					0 (0.1)	34.2 (39.5)
PHAR	*Phalaris arundinacea*	No		5.9 (18.7)	30.1 (22.4)	38.6 (37)	26.1 (35.1)	11.7 (18.6)
POHY	*Polygonum hydropiper, P. hydropiperoides*	Mixed		0.3 (1)	0.6 (2.2)	0.2 (1.4)	0.2 (0.9)	
PONA	*Potamogeton natans*	Yes				1.7 (6.4)	1.6 (5)	
POPE	*Polygonum persicaria*	No		1.4 (2.3)	0.3 (1.1)	1.8 (3.1)	0 (0.2)	
SALA	*Sagittaria latifolia*	Yes		3.6 (4.7)	7.6 (12.4)	8.1 (7.7)	4.5 (8.3)	2.8 (8.3)
SALU	*Salix lucida*	Yes				5.8 (16.9)		11.9 (21.1)
SASI	*Salix sitchensis*	Yes			1.9 (7.5)			
SCAM	*Schoenoplectus americanus*	Yes	1.8 (5)		0 (0.1)			
SISU	*Sium suave*	Yes		1.5 (3.6)	0.3 (0.9)			

Sp. Code	Scientific Name	Native (Y/N)	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
SYSU	*Symphyotrichum subspicatum*	Yes	0.7 (4.7)	1 (2.8)				
TRMA	*Triglochin maritima*	Yes	6.2 (10.9)					
TYAN	*Typha angustifolia*	No	2.3 (11.5)	0 (0.1)	6.5 (10.5)			
ULLA	*Ulva lactuca*	Yes	4.7 (9.3)					

Appendix E. Fish catch summaries, 2008–2019

Appendix Table E-1. Species list of fishes captured during the EMP study, by year, 2008-2019 at the five trend sites, Ilwaco Slough, Welch Island, White Island, Campbell Slough and Franz Lake.

Family	Species	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
Clupeidae	American shad[i]	x	x	x	x	x	x	x	x	x	x	x	x
	Pacific herring												x
Salmonidae	brown trout[i]					x							
	Chinook salmon	x	x	x	x	x	x	x	x	x	x	x	x
	chum salmon	x	x	x	x	x	x		x	x	x	x	x
	coho salmon	x	x	x	x	x		x	x	x	x	x	x
	cutthroat trout		x				x	x					
	mountain whitefish					x		x			x		
	sockeye salmon						x	x					
	steelhead (rainbow trout)		x			x						x	
	Pink salmon											x	
	unid whitefish					x							
Osmeridae	eulachon						x	x					
	longfin smelt						x						
	unid smelt				x	x	x		x	x		x	x
Cyprinidae	chiselmouth	x	x	x	x	x	x	x	x	x			
	golden shiner[i]				x	x					x	x	
	northern pikeminnow[p]	x	x	x	x	x	x	x	x	x		x	x
	peamouth	x	x	x	x	x	x	x	x	x	x	x	x
	redside shiner									x		x	
	speckled dace					x							
	tui chub		x	x	x	x	x	x	x	x			
	Common Carp											x	x

	unid carp[i]	x	x	x	x	x	x	x	x	x			
	unid Cyprinidae[i]		x		x							x	
Catostomidae	largescale sucker	x	x	x	x	x	x	x	x	x	x	x	x
Cobitidae	oriental weatherfish[i]			x						x			
Ictaluridae	brown bullhead[i]	x	x						x				
	channel catfish	x											
	yellow bullhead	x	x	x	x	x	x	x	x	x			
Fundulidae	banded killifish[i]	x	x	x	x	x	x	x	x	x	x	x	x
Poeciliidae	gambusia[i]			x							x	x	
Gasterosteidae	threespine stickleback	x	x	x	x	x	x	x	x	x	x	x	x
Centrarchidae	bluegill[i]		x	x	x	x	x	x	x			x	
	largemouth bass[ip]		x							x	x	x	x
	pumpkinseed[i]	x	x	x	x	x	x	x	x	x		x	
	smallmouth bass[ip]	x	x	x	x	x	x	x	x	x			
	Black crappie											x	x
	unid crappie[i]		x	x	x	x	x	x	x	x		x	
	Unidentified centrarchid[i]										x		
	warmouth[ip]				x								
Percidae	walleye[ip]	x											
	yellow perch[ip]	x	x	x	x	x	x	x	x	x	x	x	x
Embiotocidae	shiner perch				x	x	x		x	x	x		x
Ammodytidae	sandlance				x					x			
Gobiidae	Amur goby[i]									x	x	x	x
	unid goby[i]		x						x	x			
Cottidae	mottled sculpin		x										
	Pacific staghorn sculpin				x	x	x	x	x	x	x	x	x
	Prickly sculpin										x	x	x
	unid sculpin	x	x	x	x	x	x	x	x	x	x	x	x
Pleuronectidae	starry flounder		x	x		x	x	x	x			x	x

Appendix Table E-2. Species list of fishes captured at the five EMP trend sites in 2019.

Family	Species_Name	Ilwaco Slough	Welch Island	Whites Island	Campbell Slough	Franz Lake
Clupeidae	American shad[i]		x	x	x	
	Pacific herring	x				
Salmonidae	Chinook salmon		x	x	x	
	chum salmon	x	x	x		
	coho salmon			x		
Osmeridae	larval smelt	x				
Cyprinidae	peamouth		x		x	
	common carp				x	
	Northern pikeminnow				x	
Catostomidae	largescale sucker		x	x	x	
Fundulidae	banded killifish[i]	x	x	x	x	
Gasterosteidae	threespine stickleback	x	x	x	x	
Centrarchidae	largemouth bass[ip]				x	
	Black crappie				x	
Percidae	yellow perch[ip]				x	
Embiotocidae	shiner perch	x				
Gobiidae	Amur goby[i]				x	
Cottidae	Pacific staghorn sculpin	x				
	Prickly sculpin		x	x	x	
	unidentified sculpin		x			
Pleuronectidae	Starry flounder		x	x	x	

Table E-3. Juvenile coho (unmarked and marked), chum salmon, sockeye salmon, and trout densities (fish per 1000 m^2) (SD), by year captured at trends sites 2008 - 2019. Total number of salmonids captured at a site is presented in parentheses under site/species/#.

						Year						
Site/Species/#	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Ilwaco Slough												
Coho (unmarked) (1)	ns	ns	ns	0	0.12 (0.67)	0	0	0	0	0	0	0
Coho (marked) (0)	ns	ns	ns	0	0	0	0	0	0	0	0	0
Chum (531)	ns	ns	ns	27.60 (139.62)	0.71 (2.49)	25.01 (108.97)	0	0.24 (1.10)	11.19 (32.39)	11.70 (26.16)	32.11 (54.55)	11.08 (5.51)
Sockeye (0)	ns	ns	ns	0	0	0	0	0	0	0	0	0
Trout sp (0)	ns	ns	ns	0	0	0	0	0	0	0	0	0
Welch Island												
Coho (unmarked) (2)	ns	ns	ns	ns	0.15 (0.78)	0	0	0	0	0	0	0
Coho (marked) (1)	ns	ns	ns	ns	0.11 (0.54	0	0	0	0	0	0	0
Chum (70)	ns	ns	ns	ns	0.14 (0.71)	0.26 (1.01)	0	0	0.65 (2.68)	2.50 (5.02)	26.87 (36.55)	2.28 (1.40)
Sockeye (1)	ns	ns	ns	ns	0	0	0.12 (0.41)	0	0	0	0	0
Trout sp (1)	ns	ns	ns	ns	0	0.27 (1.05)	0	0	0	0	0	0
Whites Island												
Coho (unmarked) (7)	ns	0	1.12 (2.03)	0	0	0	0	0	0	0.12 (0.46)	0	0.51 (0.50)
Coho (marked) (0)	ns	0	0	0	0	0	0	0	0	0	0	0
Chum (42)	ns	0.14 (0.59)	1.03 (2.55)	0	0.34 (1.23)	0	0	1.55 (5.62)	1.13 (3.80)	0.45 (1.23)	2.93 (7.21)	1.50 (1.81)
Sockeye (7)	ns	0	0	0	0	1.32 (5.77)	0	0	0	0	0	0
Trout sp (0)	ns	0	0	0	0	0	0	0	0	0	0	0

Continued from Table E-3

	Year											
Site/Species/#	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Campbell Slough												
Coho (unmarked) (2)	0	0	0	0	0	0	0	0.18 (0.70)	0	0	0	0
Coho (marked)	0	0	0	0	0	0	0.18 (0.51)	0	0	0	0	0
Chum (10)	0.29 (0.99)	0	0.67 (2.24)	0	0	0	0	0.09 (0.35)	0.35 (1.09)	0.13 (0.48)	0.26 (1.11)	0
Sockeye (1)	0	0	0	0	0	0	0.09 (0.38)	0	0	0	0	0
Trout sp. (1)	0	0	0	0	0	0	0.78 (0.33)	0	0	0	0	0
Franz Lake												
Coho (unmarked) (32)	0.56 (1.01)	2.09 (4.85)	ns	4.86 (13.36)	2.29 (7.15)	0	1.17 (2.29)	0.20 (0.87)	0.27 (1.06)	ns	1.53 (2.65)	ns
Coho (marked) (60)	5.75 (10.72)	7.92 (17.99)	ns	0	0.51 (2.17)	0	0	0	0	ns	0	ns
Chum (7)	2.11 (5.23)	0.32 (1.09)	ns	0	0	0	0	0	0	ns	0	ns
Sockeye (2)	0	0	ns	0	0	0	0.55 (1.56)	0	0	ns	0	ns
Trout sp. (5)	0	0.95 (2.30)	ns	0	1.43 (6.54)	0	0	0	0	ns	0	ns

Table E-4. Total number of unmarked and marked Chinook salmon captured by year at each site 2008 – 2019, ns = site not sampled.

	Years											
Site	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**
Ilwaco Slough												
Unmarked	ns	ns	ns	1	5	0	0	6	0	4	2	0
Marked	ns	ns	ns	0	0	0	0	0	0	1	1	0
Welch Island												
Unmarked	ns	ns	ns	ns	280	154	241	39	121	398	1339	130
Marked	ns	ns	ns	ns	23	35	20	4	6	13	4	18
Whites Island												
Unmarked	ns	36	114	54	83	64	134	33	144	111	245	71
Marked	ns	6	26	7	3	20	7	15	6	1	0	16
Campbell Slough												
Unmarked	25	15	40	3	46	30	17	36	20	45	13	42
Marked	26	69	48	31	21	22	18	72	2	61	33	21
Franz Lake												
Unmarked	11	13	ns	2	0	2	15	40	24	ns	2	ns
Marked	49	7	ns	0	0	0	0	1	0	ns	1	ns