Lower Columbia River Ecosystem Monitoring Program

Annual Report for Year 14

BPA Project Number: 2003-007-00
Report covers work performed under BPA contract # 73977
Report was completed under BPA contract # 80237
Report covers work performed from: October, 2017-September, 2018

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Report Created: April 2019

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Suggested Citation:
Kidd, S.A., M.D. Schwartz, R.N. Fuller, R. McNatt, K. Poppe, T.D. Peterson, J.A. Needoba, L. Cook, J. Cordell, M. Ramirez, A.C. Hanson, A.B. Borde, S.A. Zimmerman, S. Hinton, J. Grote, P.M. Chittaro, D. Kuligowski, G.M. Ylitalo, D. Lomax, V.I. Cullinan, L.L. Johnson, H.L., and C.A. Corbett. 2019. Lower Columbia River Ecosystem Monitoring Program Annual Report for Year 14 (October 1, 2017 to September 30, 2018). Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration. Available from the Lower Columbia Estuary Partnership, Portland, OR.

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Abstract

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. In 2018, 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 2018. 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.

Off-channel areas provide important habitat for juvenile salmonids, and the quality of these habitats determines the suitability of these areas for rearing fish. Compared to previous years mainstem flows in 2018 was characterized by an above average winter discharge and a large peak associated with the spring freshet in May. However, river discharge levels dropped below average by mid-June. Water temperatures in 2018 were slightly below past years (2013-2017) in terms of the days exceeding 19°C and similar to 2016 and 2014 in the number of very warm days (exceeding 21, 22, or 23°C), generally being cooler than 2017. Campbell Slough remains the warmest of the trend sites in 2018 with water temperatures exceeding 25°C in August. High connectivity between the mainstem and off-channel areas allows for flushing, fish access, and the flux of organic material in and out of the site. Campbell Slough and Franz Lake Slough are considered to be fairly disconnected from the mainstem (i.e., slow exchange between habitats), whereas Welch Island and Whites Island are well-connected to the mainstem and are typically well-flushed. Ilwaco Slough is highly influenced by tides and marine water, thus has considerably different water quality conditions from the other sites. This difference in connectivity to the mainstem among the sites during low flow periods helps explain warmer water temperatures in Campbell Slough and Franz Lake Slough during the low flow summer months.

Emergent wetland vegetation cover and composition are largely driven by hydrologic patterns. Similar to 2017, 2018 species richness levels were greatest at Welch Island and Whites Island.
Between 2011-2018, across all trend sites, non-native species richness has increased at a slightly greater rate than native species richness. In 2018, non-native reed canarygrass, *Phalaris arundinacea,* cover levels stayed relatively consistent to those observed in 2017. Annual shifts in *P. arundinacea* cover were found strongly correlated with mean % daily inundation for the month of August across Welch Island, Whites Island, Cunningham Lake, and Campbell Slough and with the Columbia River freshet levels (accumulated discharge May-Aug) at Cunningham Lake, Campbell Slough, and Franz Lake. These relationships indicate that daily 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 trend sites. Daily inundation analysis across sites also highlighted an important threshold for *P. arundinacea* growth, *P. arundinacea* abundance diminishing in marsh areas receiving > 30% mean daily inundation (~7 hours a day) in August. Between 2011-2018, *P. arundinacea* mean annual cover was 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 across trend sites. Additionally,

2018 habitat biomass collection efforts identified that *P. arundinacea,* living above ground biomass has higher mean Lignin: Nitrogen and Carbon: Nitrogen ratios than native lyngbe sedge, *Carex lyngbyei,* these data *s*upport the hypothesis that *P. arundinacea* produces lower quality detritus than native *C. lyngbyei,* negatively impacting detrital food web dynamics in the estuary through its continued invasion.

Phytoplankton forms the base of the juvenile salmon food web that feeds zooplankton and macroinvertebrates, which are important juvenile salmon prey items. Chlorophyll *a* concentrations in 2018 varied temporally across the trends sites, peaking in May (before the freshet) at Welch and Whites Island and June and July at Franz Lake Slough and Campbell Slough. Phytoplankton assemblages tend to be dominated by diatoms at Whites Island and Welch Island, while at Campbell Slough and Franz Lake Slough, cyanobacteria and chlorophytes make more substantial contributions to total phytoplankton abundances. Dominant diatom species shift through the monitoring period, with *Asterionella formosa* dominating in the early spring (Feb-Apr) and *Skeletonema potamos* dominating later in the summer. In addition, based on observed dissolved oxygen levels, Whites Island appears to support greater primary productivity earlier in the spring compared to greater primary productivity later in the spring and early summer at Welch Island. Net ecosystem metabolism, the ratio of production to respiration is a useful tool in determining if a system is nutrient enriched and a source or sink for carbon . This wascalculated at two sites that tend to have contrasting patterns in primary productivity (Whites Island and Campbell Slough) and results indicated that both off-channel sites are characterized by the net production of organic matter. Net production of organic matter suggests that these sites are likely exporting/contributing organic matter into the larger lower Columbia food web.

Over multiple monitoring years, the EMP study has shown that juvenile Chinook salmon feed habits 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. The dipterans that juvenile Chinook salmon feed primarily on are the adult and larval stages of chironomids. Chironomids often dominate the composition of juvenile Chinook salmon diets at sites in the upper reaches (Campbell Slough and Franz Lake Slough). Diet composition changes closer to the mouth of the river, shifting to both chironomids and amphipods at Whites and Welch Island and then being dominated almost exclusively by amphipods at Ilwaco Slough. Stable isotope analysis indicated that chironomids most likely consumed organic matter derived from periphyton; however, additional analysis is needed to fully characterize consumption patterns and identify sources of variability in the diets of chironomids. In 2017, mean energy ration (a measure of fitness that reflects stomach fullness and energy consumed) from Campbell Slough was lower than previous years due to an abnormal increase in cladoceran (a relatively energy-poor prey item) consumption. Similarly, in 2017, Franz Lake also showed an increase in copepod consumption (another relatively energy-poor prey item) compared to previous years. While juvenile Chinook salmon at Franz Lake consistently had below average energy rations, these fish also had relatively low metabolic costs, which may offset the quantity and quality of prey consumed.

In 2018, unmarked Chinook salmon were captured at trends sites between February and June (peak densities were observed in April and May) and were abundant at Welch Island and Whites Island. Marked Chinook salmon were captured between April and October and were generally less abundant than 2017 but within the historic range seen across the sites between 2008-2016. The majority of unmarked Chinook salmon were fry, 88.3%; 11.3% were fingerlings, and 0.4% were yearlings. Unmarked fry dominated catches at Welch Island, Whites Island, and Ilwaco Slough, whereas fingerlings and fry were more evenly distributed at Campbell Slough. Marked Chinook salmon were 100% fingerlings across all sites. Chum salmon were the most abundant salmon species captured at Ilwaco Slough, and small numbers were also observed at Welch Island, Whites Island, and Campbell Slough. In 2018, chum salmon were found at the trends sites between March and May with the highest average density in April.

As in previous years, the overall 2018 fish catches at Welch and Whites Islands were composed primarily of native species and were dominated by a single species (threespine stickleback). Catches upriver at Campbell Slough were more evenly distributed among species and included the greatest percentage of non-native fishes (75%). In 2018, Campbell Slough had the highest diversity of species (16) followed by Ilwaco Slough (10) and White Island (9). While relatively high compared to the other EMP sites Campbell Slough's 2018 species richness and diversity were noticeably lower than any previous years and only slightly higher than 2017. Low species richness at Franz Lake in 2018 is likely an artifact of the limited sampling (one month) due to high water temperatures. In 2018, non-native yellow perch comprised 90% of the total number of predatory fish captured and were found across all trend sites.

The EMP provides essential baseline information on ambient environmental conditions and yields insight into the cumulative effects of existing and new management actions and anthropogenic impacts. EMP data are useful for making comparisons to changing conditions, enhancing our understanding of fish habitat use, and determining whether water quality and habitat characteristics are meeting the needs of migrating juvenile salmonids. In addition, the relatively undisturbed conditions at the EMP trends sites should be considered endpoints for the ecological function of habitats undergoing restoration, and findings can inform regional habitat restoration design and translate to additional reference data for comparison to action effectiveness monitoring efforts. Quantifying sources of variability in fish, habitat, and food web metrics allow for increased predictability for how biological components will respond to changes in environmental conditions.

Management Implications

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. 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. 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. Efforts to conduct additional sampling across the tidal range and at high tide may produce results that differ from those derived using traditional methods.

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. Similar to 2017, in 2018, a higher than average number of days where water temperatures exceeded recommended temperature maximums in the mainstem were observed (2015 and 2017 are the only other years that surpass 2018 in this regard). Such a result is concerning since 2018 river discharge patterns were considered to be relatively high compared to other years, yet these higher flows occurred early in the season and did not buffer the warmer temperatures observed. 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 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 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.

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 2019. 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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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. 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
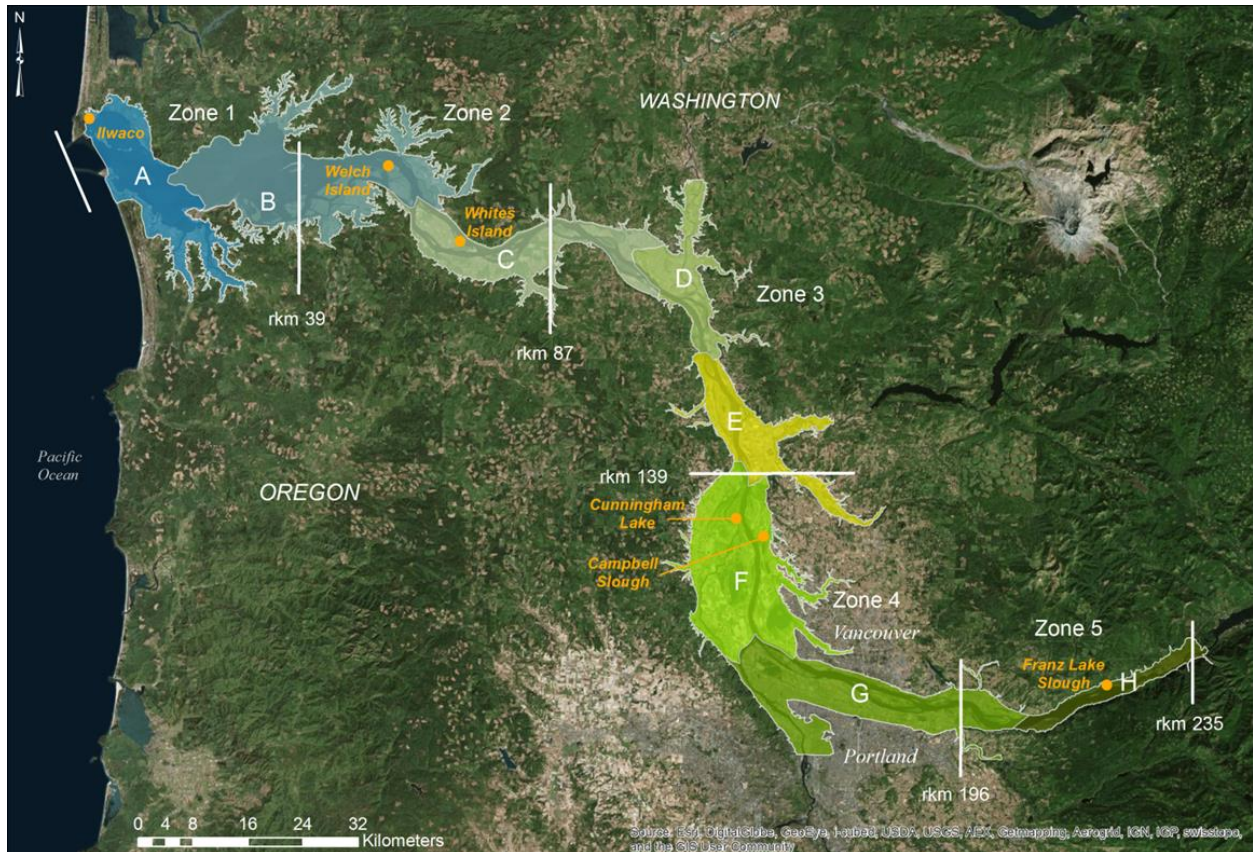
Figure 1. Lower Columbia River and estuary with hydrogeomorphic reaches (A-H) specified by color (Simenstad et al. 2011) and wetland zones (1-5) delineated by white lines (Jay et al. 2016). The 2017 EMP trends sites are shown in orange.

1.3 Characterization of Emergent Wetlands in the Lower Columbia River

1.3.1 Sampling Effort, 2005-2018

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 trends 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 five trends sites are Ilwaco Slough (2010-2018), Secret River (2010-2016), Welch Island (2010-2018), Whites Island (2009-2018), Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2018), and Franz Lake (2008-2009, 2011-2018). In 2018, an additional objective was undertaken by some EMP partners to address a question regarding juvenile salmon presence and food web dynamics. 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. 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 14 Contract (October 1, 2017 – September 30, 2018):

- 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 2018.

Reach	Type of Site	Site Name	Site Code	Vegetation & Habitat[1]	Fish &Prey[5]	Abiotic Conditions	Food Web[4]
A	Trend	**Ilwaco Slough**	BBM	2011-2018	2011-2013, 2015-2018	2011-2013, 2015-2018	2011-2013, 2015-2018
B	Trend	Secret River	SRM	2008[2], 2012-2016	2012, 2013		2012, 2013
	Tributary	Grays River, lower	-		2015		2015
	Trend	**Welch Island**	WI2	2012-2018	2012-2018	2014, 2018	2012-2018
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	2009-2018	2009-2018	2009, 2011-2018	2011-2018
	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	2005-2018	2007-2009		

Reach	Type of Site	Site Name	Site Code	Vegetation & Habitat[1]	Fish &Prey[5]	Abiotic Conditions	Food Web[4]
	Trend	**Campbell Slough**	CS1	2005-2018	2007-2018	2008-2018	2010-2018
G	Status	Water Resources Center	WRC	2006			
	Status	McGuire Island	MIC	2006			
	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**	-			2012-2018	
H	Trend	**Franz Lake (slough)**	FLM	2008-2009, 2011-2018	2008-2009, 2011-2018	2011-2018	2011-2018
	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 – 2018.

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

1.3.2 Site Descriptions

In 2018, 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 2018 are listed in Table 2. The 2018 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 2) 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. This site has been sampled exclusively for habitat and hydrology data.

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 will be updated in 2019 in an attempt to prevent further grazing.

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 trends sites sampled 2017.

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 2016: (a) Ilwaco Slough; (b) Welch Island; (c) Whites Island; (d) Cunningham Lake; (e) Campbell Slough; (f) Franz Lake Slough. Updated site photos were taken in 2017 but were unavailable at the time this report was compiled.

1.3.3 Water Year

Compared to the previous eight years, discharge at Bonneville Dam for 2018 was characterized by an above average winter discharge and a large peak associated with the spring freshet. Discharge rose steadily from early April and peaked in mid-May, with discharge levels during the peak being the highest in 8 years. The high discharge did not result in the lower river reaching significant flood stage, for example, compared to 2017 or 2011 (Figure 3). This may have been due to the relatively short period of high flows compared to previous years. By mid-June, the discharge had dropped to below average levels for the rest of the summer period.
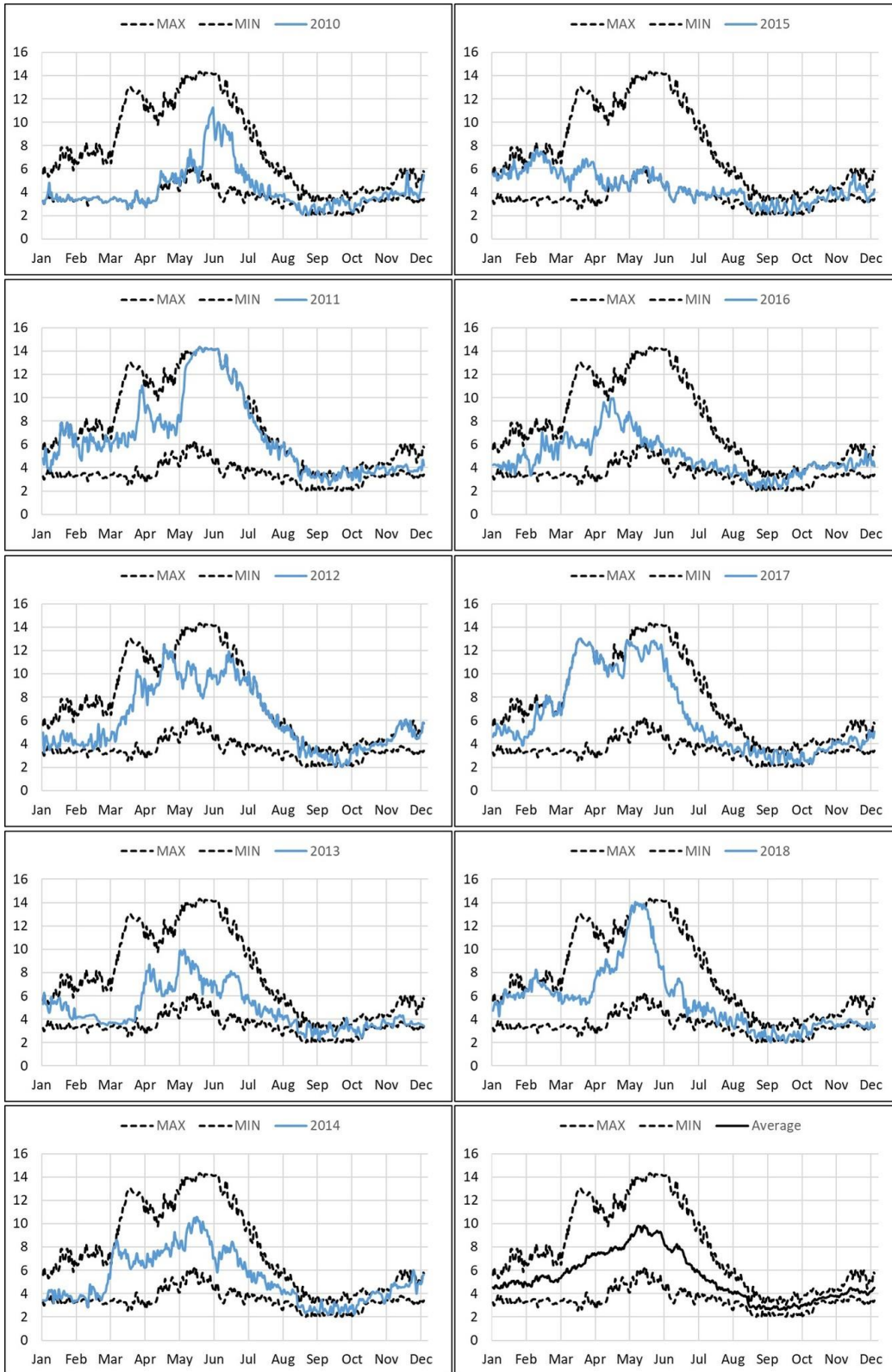
Figure 3. Daily water discharge (m³/s) at Bonneville Dam. Panels show individual years between 2010-2018 (blue lines) and the daily max and min for all years combined. Vancouver gage web page shows recent flood stage years - https://water.weather.gov/ahps2/crests.php?wfo=pqr&gage=vapw1&crest_type=recent

2 Methods

2.1 Mainstem Conditions

2.1.1 Overview

J. Needoba at Oregon Health & Science University (OHSU) operates 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 4). 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 4). 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 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).
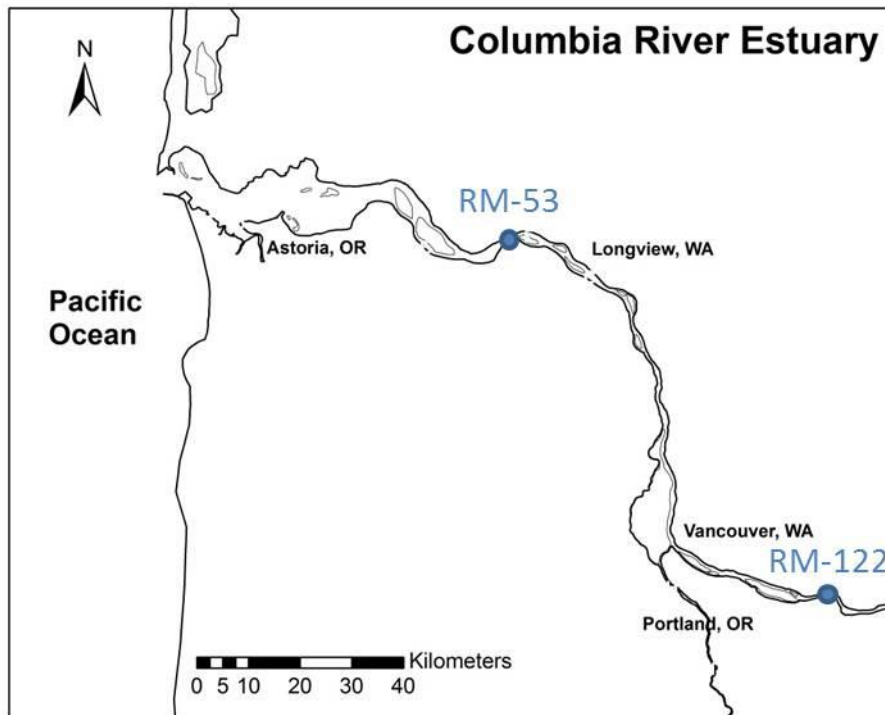
Figure 4. Station locations for the two in-situ water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program. RM-53 (river mile 53) is Beaver Army Terminal, while RM-122 (river mile 122) is located in Camas, WA.

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

The instrument platform was removed for sensor maintenance in October 2017 and deployed in February 2018. The instruments ran continuously between 02/07/18-10/01/2018.

2.1.3 Sensor Configuration

Instruments and sensors common to both platforms are described in Table 3. Sensors are configured to collect a sample and telemeter the data every hour. In addition to the parameters listed in Table 3, 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 3. Description of the components on the sensor platforms located at RM-53 and RM-122.

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 4.

Table 4. 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 2018, water quality was continuously monitored at four of the trends sites, Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake (Table 5). The monitoring protocol can be found on monitoringmethods.org (Method ID 816). Figure 5 shows how the sensors were deployed to ensure ready access for servicing, and data downloads and Figure 6 shows the periods of deployment of in situ sensors between 2011-2018.

Table 5. Locations of water quality monitors (YSI sondes) at trends sites in 2018.

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"	3/16/2018	9/26/2018
Welch Island	461518123285700	Unnamed Slough, Welch Island, Columbia River, OR	B	46° 15' 18.4"	-123° 28' 56.8"	3/13/2018	9/26/2018
Whites Island	460939123201600	Birnie Slough, White's Island, Columbia River, WA	C	46° 09' 39"	-123° 20' 16"	3/13/2018	9/26/2018
Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47' 05"	-122° 45' 15"	3/14/2018	9/248/2017
Franz Lake	453604122060000	Franz Lake Slough Entrance, Columbia River, WA	H	45° 36' 04"	-122° 06' 00"	3/14/18	9/24/2018

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

Figure 5. 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 6 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 2018, the monitors were deployed from late March through mid-September (Table 6). 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 6. 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
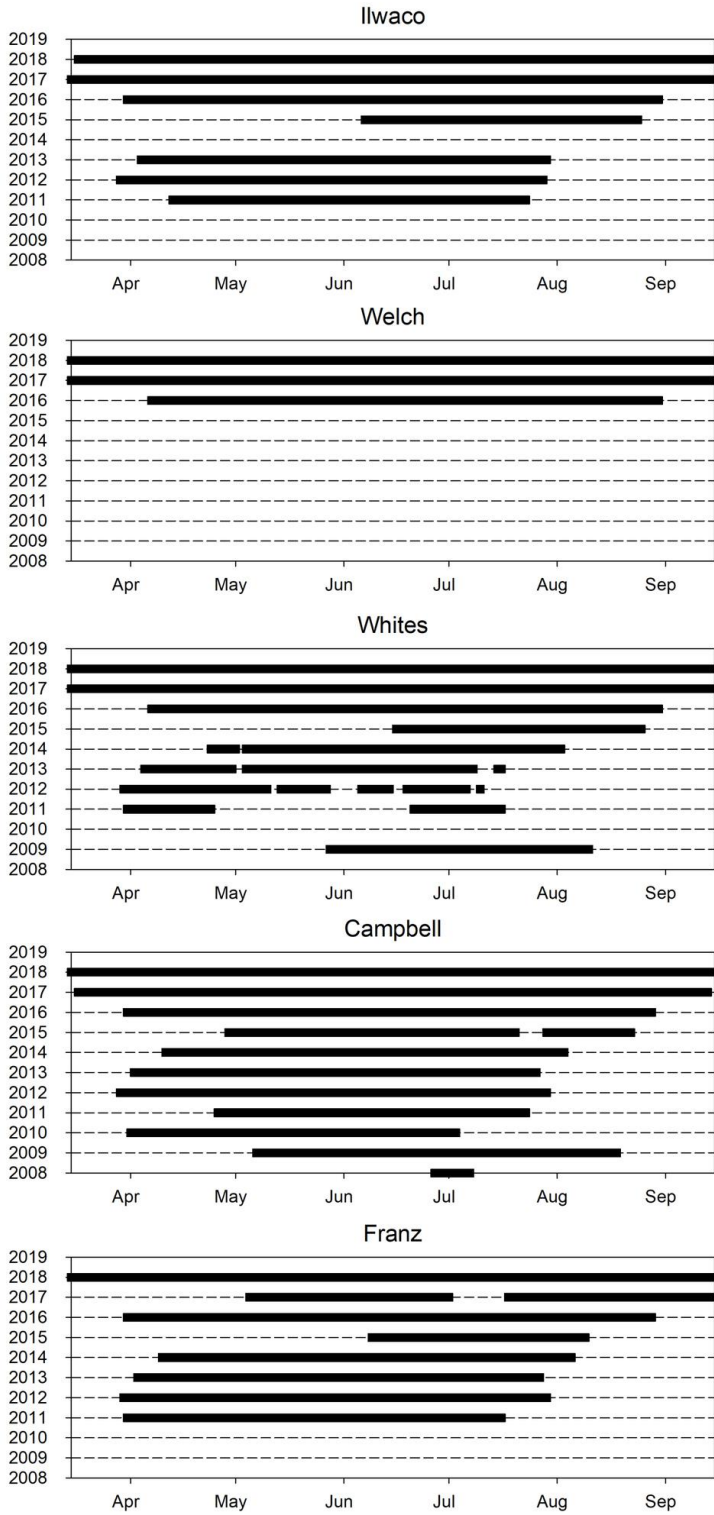
Figure 6. Time periods are corresponding to sensor deployments at five trends sites (2011–2018).

2.2.2 Nutrients (N, P)

Nitrogen and phosphorus are nutrients that are often present at low enough concentrations to limit plant and phytoplankton growth in aquatic environments relative to other growth requirements. To analyze water column nutrient concentrations, two 1 L water grab samples were collected from representative 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 7. 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 7. 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 2018, LCEP and ETG collected field data on vegetation and habitat conditions at the six trends sites (Figure 1). Monitoring dates are provided in Table 8, and detailed maps of the 2018 monitoring sites are presented in Appendix A.

Table 8. 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	7/30/18
Welch Island	WI2	53	Trend	8/1/18
Whites Island	WHC	72	Trend	7/31/18
Cunningham Lake	CLM	145	Trend	8/8/18
Campbell Slough	CS1	149	Trend	8/7/18
Franz Lake	FLM	221	Trend	8/6/18

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 2018, 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); 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. In 2009, pressure transducers (HOBO Water Level Data Loggers, Onset Computer Corporation) were deployed at Whites Island, Cunningham Lake, Campbell Slough, and Franz Lake as a means of continuously logging hourly *in situ* water level data. During the fall of 2010, a sensor was deployed at Ilwaco Slough that turned out to be faulty and was replaced in April 2011. Sensors were deployed at the Welch Island in 2012. 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-placed one meter apart were 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 are measured at deployment then subsequently on an annual basis. The stakes, termed sedimentation stakes or pins, are used to determine gross annual rates of sediment accretion or erosion (Roegner et al. 2009). All previously installed sediment accretion stakes at the trends sites were measured in 2017. The accretion or erosion rate is calculated by averaging the 11 measurements along the one-meter distance from each year and comparing the difference. The monitoring protocol can be found on monitoringmethods.org (Method ID 818). Accretion data from 2017 and 2018 have not yet been processed or reported.

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 2018, 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 2018. 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 (2018) 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 9, Table 10). 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 9: Seasonal data collection schedule Winter 2018-Winter 2019. Winter 2019 data is still under lab processing 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					

Field Methods
From summer 2011 to Spring 2018, aboveground biomass was sampled to estimate the primary productivity at the six 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 1 m^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 1 m^2 biomass plot, a 0.1 m^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 <5ºC 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 5.1 cm and were marked around the outside at 10 cm 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 10. 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 2017-2019 we also sampled at Steamboat Slough, a restoration site located near Whites Island.

Site[1]	Strata	2011-12 S	2011-12 W	2012-13 S	2012-13 F	2012-13 W	2013-14 S	2013-14 W	2015-16 S	2015-16 W	2016-17 S	2016-17 W	2017-18 S	2017-18 W	2017-18 Sp	2018-19 S	2018-19 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	101
WI2	LM			4												9	9	22
WI2	SAV			4		4	6											40
WHC	CALY		1	3		3	3	3	3	3	3	3	3	3	3			31
WHC	HM												1	1		9	9	20
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	18
CLM	ELPA/SALA								6	6	6		5					23
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	18
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											25
FLM	LM															9	9	18
	Total	49	55	73	14	73	114	61	55	55	104	78	52	30	31	54	54	813

[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 500 μm 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
Average dry weight was calculated for various strata and site values. For 2015 to 2018 data (Table 10), 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 *an* 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 11. 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 inverted light microscope following concentration achieved through settling 10–25 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 11. List of samples analyzed (Xs) and data of collection from five trends sites in the Lower Columbia River in 2018.

Site	Zone	Reach	Date	Nutrients	Zooplankton	Phytoplankton
ILWACO SLOUGH	1	A	3/16/18	X	X	X
			4/17/18	X	X	X
			5/14/18	X	X	X
			6/26/18	X	X	X
			7/23/18	X	X	X
			9/26/18	X	X	X
WELCH ISLAND	2	B	3/13/18	X	X	X
			4/13/18	X	X	X
			5/11/18	X	X	X
			6/25/18	X	X	X
			7/25/18	X	X	X
			9/26/18	X	X	X
WHITES ISLAND	3	C	3/13/18	X	X	X
			4/12/18	X	X	X
			5/10/18	X	X	X
			6/25/18	X	X	X
			7/25/18	X	X	X
			9/26/18	X	X	X
CAMPBELL SLOUGH	4	F	3/14/18	X	X	X
			4/19/18	X	X	X
			5/16/18	X	X	X
			6/22/18	X	X	X
			7/26/18	X	X	X
			9/24/18	X	X	X
FRANZ LAKE SLOUGH	5	H	3/14/18	X	X	X
			5/16/18	X	X	X
			6/22/18	X	X	X
			7/26/18	X	X	X
			9/24/18	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 (NMDS$_{total}$) and (ii) only major diatom taxa (NMDS$_{diatom}$). 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 NO2-+NO3-,

NH$_4^+$, PO43-, 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 four trends sites (Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). A list of the collection dates and sampling sites are given above in Table 11.

Abundance

Zooplankton abundances collected via net tow were determined at each of four trends sites (Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). 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:

$$Distance = \frac{Difference\ in\ counts \times Rotor\ Constant}{999999} \tag{1}$$

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:

$$Speed = \frac{Distance\ in\ meters \times 100}{Time\ in\ seconds} \tag{2}$$

The volume is determined as:

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

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 12). 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 12. 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 (δ ^{13}C) and nitrogen (δ ^{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 13). Samples were preserved in 95% ethanol. The monitoring protocol can be found on monitoringmethods.org ([Method ID 1622](#)).

Table 13. The number of invertebrate tow samples (OW and EV) collected at each site per sampling event, 2008-2013, and 2015-2018.

Year	Month	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 12 continued)

Year	Month	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 -80ºF 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 14.

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 14. 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

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		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:

$$IRI_i = (P_{i,\text{numeric}} + P_{i,\text{gravimetric}}) \times FO_i$$

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 2018, NOAA Fisheries monitored habitat use by juvenile Chinook salmon and other fishes at four trends sites, Campbell Slough in Reach F (sampled from 2007-2017), Whites Island site in Reach C (sampled from 2009-2017), Welch Island in Reach B (sampled from 2012-2017), and Ilwaco Slough in Reach A (sampled from 2011-2017), in order to examine year-to-year trends in fish habitat use in the lower river. A fifth trend site, Franz Lake in Reach H (sampled in 2008 – 2016, no sampling in 2017), was only sampled in March then discontinued due to high river levels and warm temperatures. Coordinates of the sampling sites are shown in Table 15.

Fish were collected from February through June/July, and October 2018 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 2018 are shown in Table 15. 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. 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 15. Location of EMP sampling sites in 2018 and the number of beach seine sets per month. ns = not sampled.

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Total
Ilwaco Slough (Reach A) 46.300530° N, 124.045893° W	ns	3	3	2	2	ns	2	ns	ns	3	15
Welch Island (Reach B) 46.255011° N, 123.480398° W	ns	1	1	1	1	ns	ns	ns	ns	3	7
Whites Island (Reach C) 46.159350° N, 123.340133° W	ns	3	2	2	1	ns	1	ns	ns	3	12
Campbell Slough (Reach F) 45.783867° N, 22.754850° W	ns	3	3	3	2	3	ns	ns	ns	3	17
Franz Lake (Reach H)* 45.600583° N, 122.103067W	ns	ns	3	ns	ns	ns	ns	ns	ns	ns	3
Total	0	10	12	8	6	3	3	0	0	12	54

*not sampled due to high water levels and fire

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:

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

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 37-111 mm (mean = 67.5 mm, SD = 13.6 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 (5.58mm) of the regression between fish length and otolith radius ($R^2 = 0.78$, n = 765) and 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 two antennas measuring 4' by 20'. 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 detection system at Campbell Slough 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. During February we installed six antennas measuring 4' x 10'. The antennas were arranged in a vertical "pass-through" configuration (Figure 7) 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 7. 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

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

River discharge at Beaver Army Terminal is shown in Figure 8. Beaver Army Terminal (BAT) discharge includes inputs from tributaries, including the Willamette River, in addition to flows from the Columbia River. Flows at Beaver illustrate the contribution of tributaries; in 2018, early spring flows were moderate to high, both at Bonneville (Figure 3) and at BAT. Flows associated with the freshet at BAT were close to the maximum observed in 2017. 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 3)—flows were low at both sites during the summer and autumn. Thus, winter river discharge at Beaver Army Terminal 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.
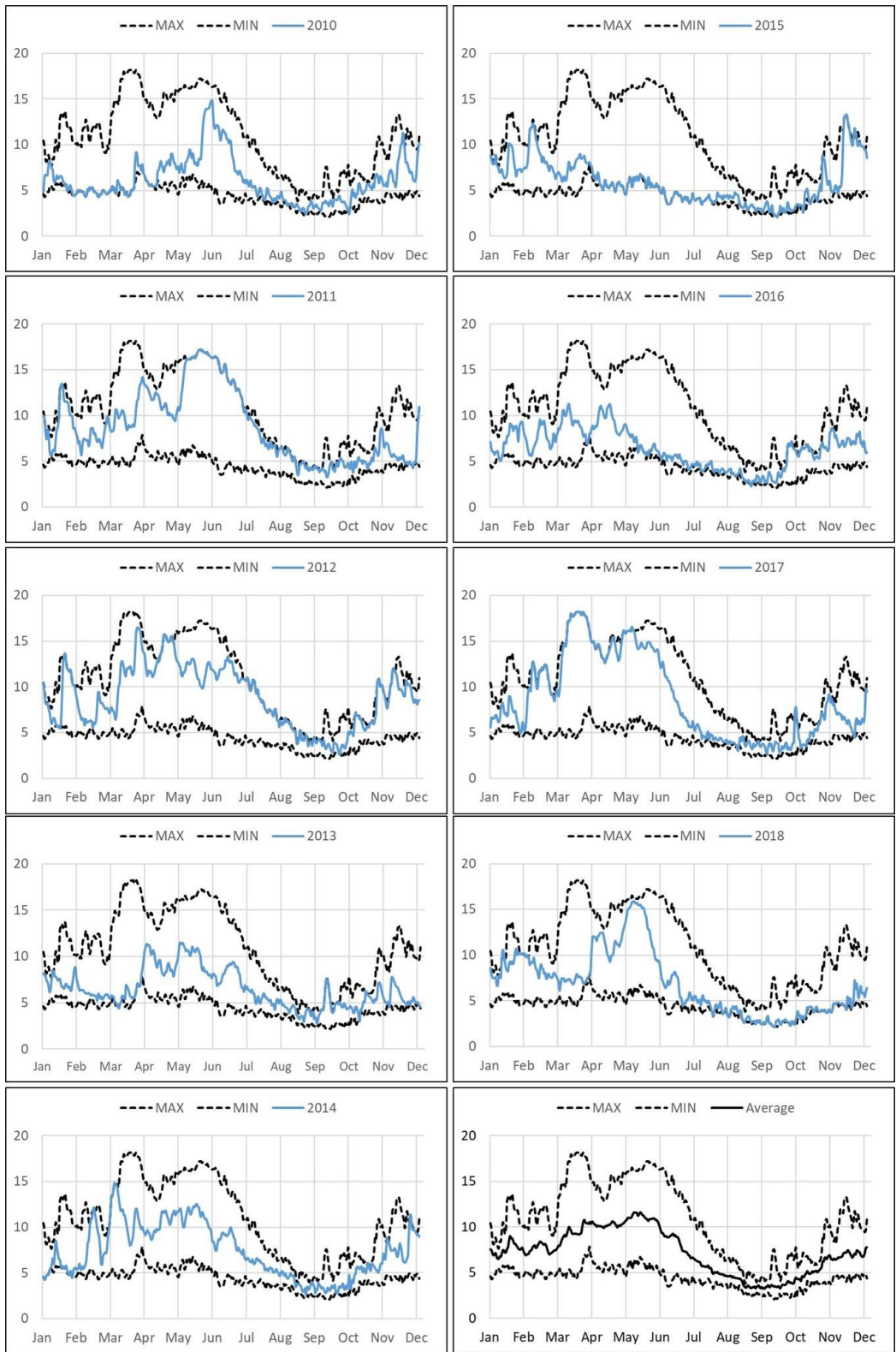
Figure 8. Daily water discharge (m³/s) at Beaver Army Terminal (RM-53) from 2010-2018. Panels show individual years (blue lines) with the daily maximum and minimum indicated (upper and lower dashed lines) in each panel.
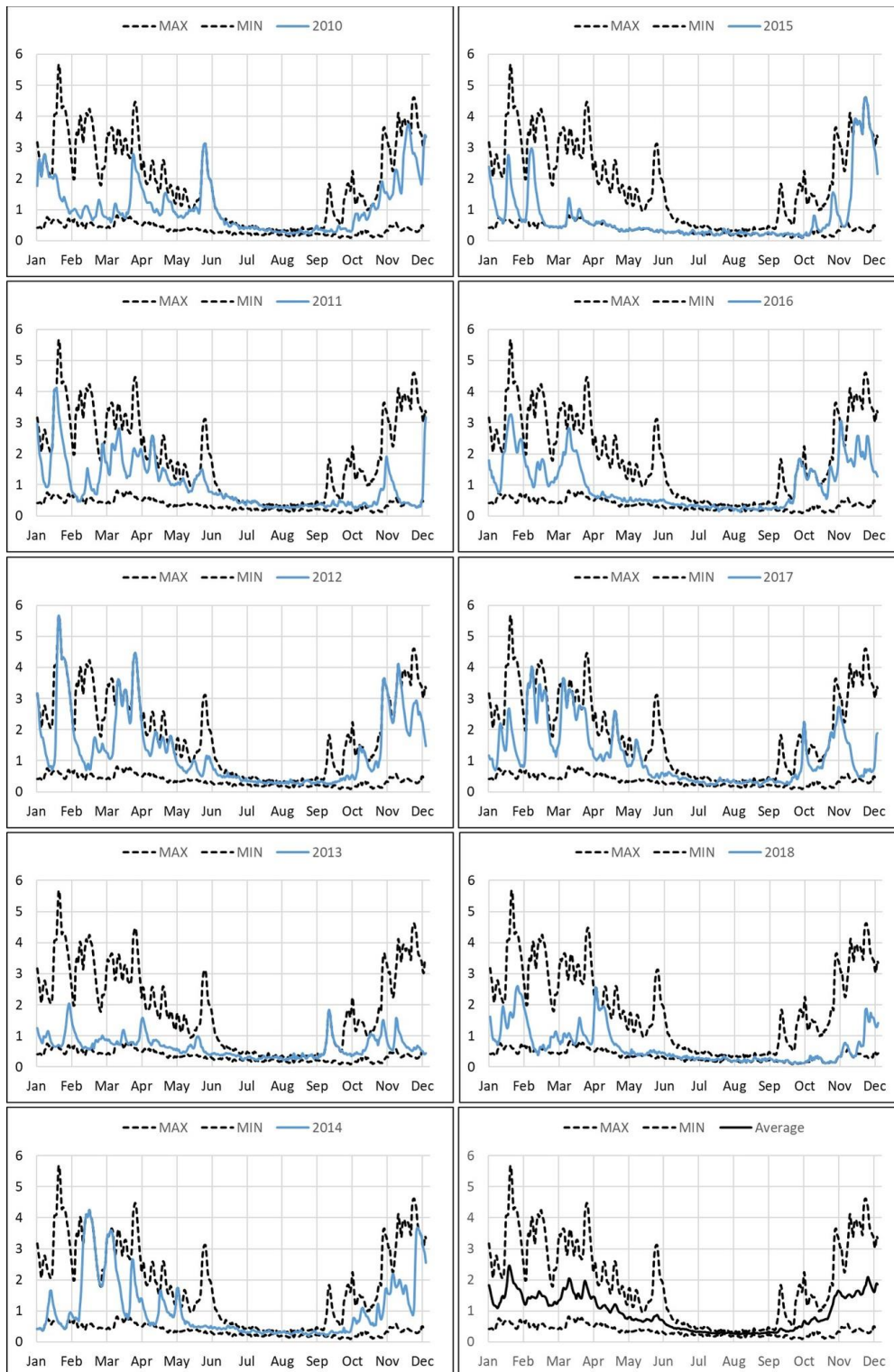
Figure 9. Daily river discharge of the Willamette River measured near the Morrison Bridge for years 2010–2018. Data from USGS 14211720. Also shown are the daily maximum and minimum values for the years 2010–2018.

Time series of river discharge associated with tributaries excluding the Willamette River are shown in Figure 10. 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.
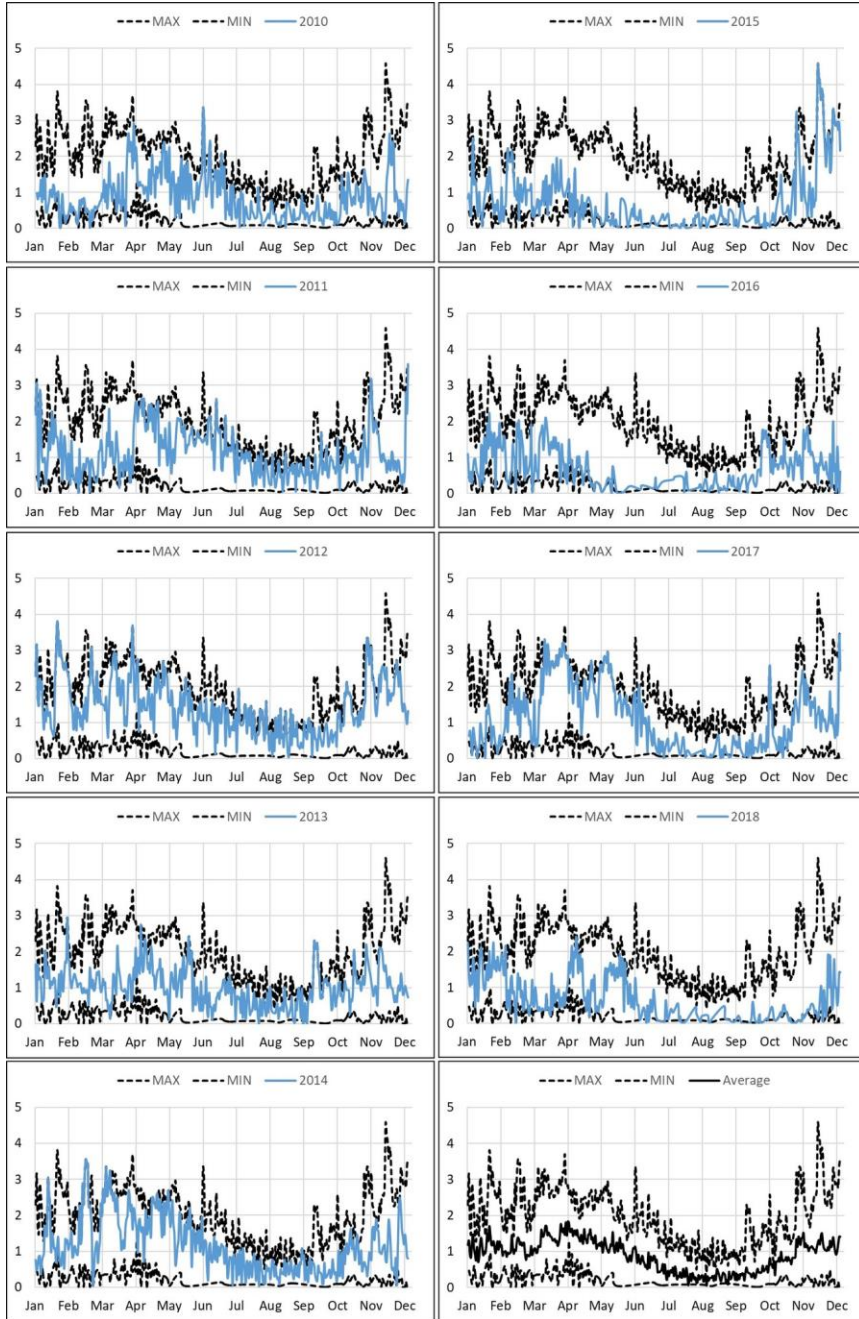
Figure 10. Estimated daily water discharge (m³/s) associated with tributaries (excluding the Willamette River). Discharge fluxes were computed from the difference between observations at Beaver Army Terminal and observations at Bonneville Dam and the Willamette River near the Morrison Bridge (i.e., Tribs = BAT – Bonneville – Willamette). 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 2010–2018 are shown in Figure 11 and Figure 12. 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 12). River discharge in the early spring of 2018 was moderate 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 2017, however, the proportional flow from the Willamette was higher from January through March in 2018. After April, flows were dominated by the Columbia. The first peak at the beginning of the high-discharge spring period had particularly large contributions from the Willamette River and other tributaries. The plots in Figure 12 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 2016; 2017 was closer to average in terms of source contributions; and in 2018, flows were dominated by the Columbia throughout most of the year.

An interesting aspect of the 2018 hydrograph was the combination of relatively high flows during the spring freshet, but low flows during the period preceding and following it. Based on the magnitude, timing, and duration of the freshet as well as flows observed during the rest of the year, 2018 was similar to 2011. However, 2018 differed from 2011 in the proportion of flow attributable to tributaries; the Columbia accounted for a greater proportion of flows during the post-freshet, summer, and autumn periods in 2018 than it did in 2011.
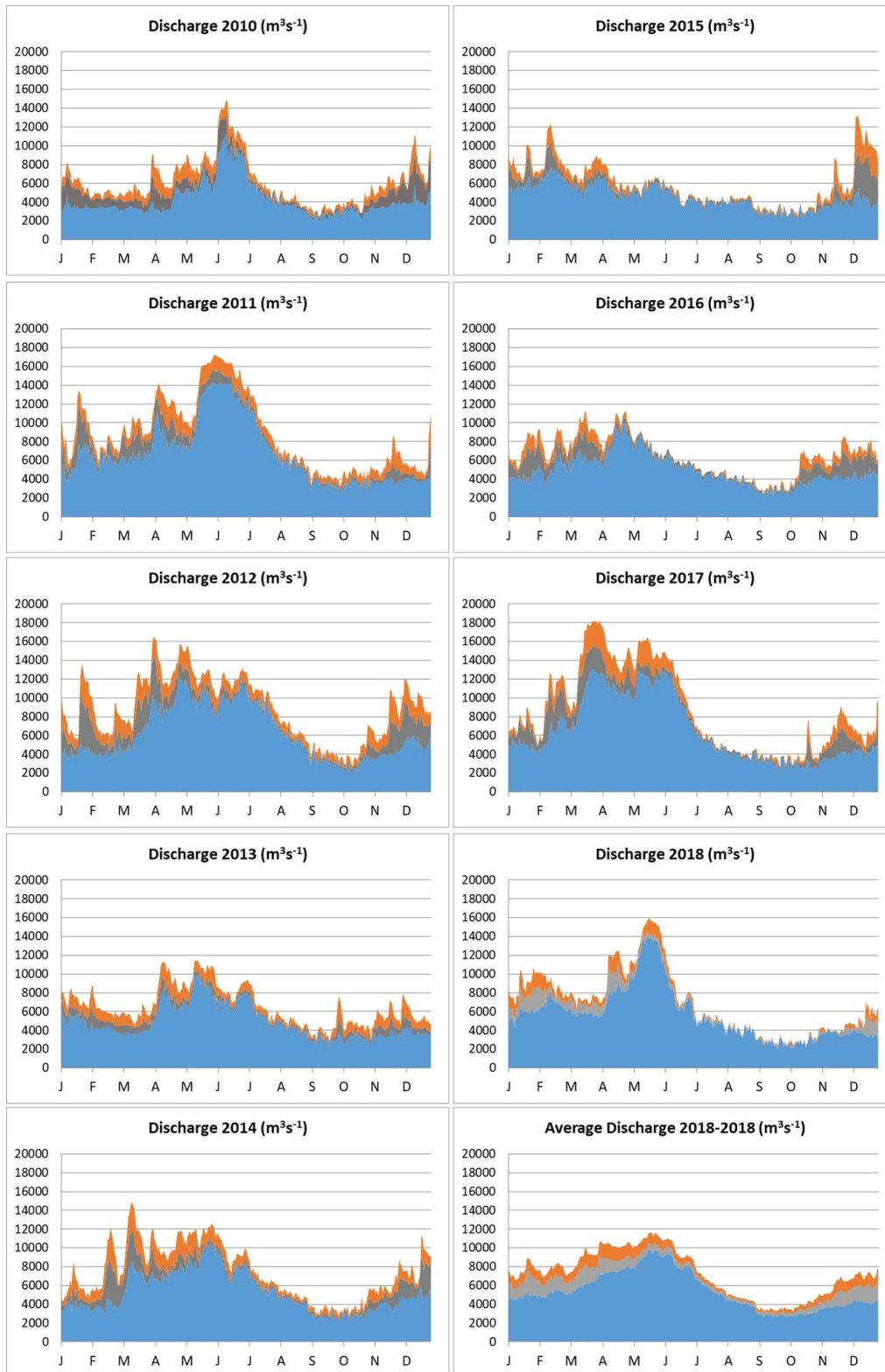
Figure 11. Daily discharge fluxes (m3/s) associated with Columbia River flow (blue), Willamette River flow (orange), and other tributaries (grey). Discharge from the Willamette was determined at the USGS stream gage at the Morrison Bridge; the contribution from other tributaries was computed by subtracting flows observed in the Willamette from those in the Columbia.

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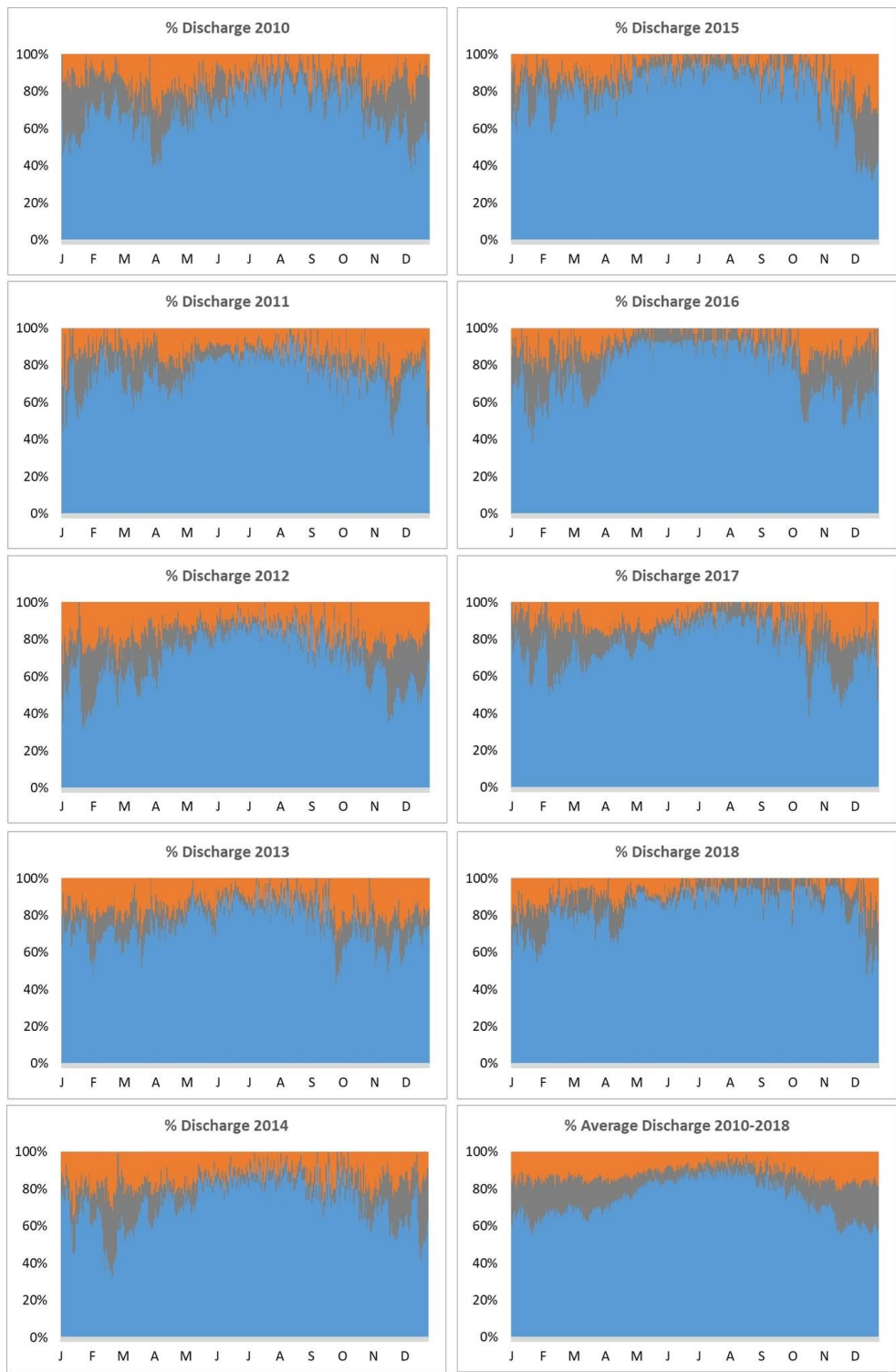
Figure 12. Daily percentage contributions of the Columbia (blue), Willamette (orange), and other tributaries (grey) to total river discharge observed at Beaver Army Terminal (RM 53). Discharge from the Willamette was determined at the USGS stream gage at the Morrison Bridge; the contribution from other tributaries was computed by subtracting flows observed in the Willamette from those in the Columbia.

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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 2018 where temperatures exceeded thresholds associated with reduced performance or physiological stress with the years 2013, 2014, 2015, and 2016 (Figure 13). 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 <16 ºC for optimal performance; rearing and migratory habitats should be <18.0 °C (Oregon standards) or <17.5 °C (Washington standards). The Columbia between the mouth and rkm 497 (Oregon–Washington border) is considered spawning/rearing habitat (Washington State Water Quality Standards). Within the migration corridor, temperatures should be <20 °C (Oregon standards), with a recommendation that water bodies have cold-water refugia having temperatures at least 2 ºC 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 ºC (Kidd, 2011), with an ideal range of 7.2-15.6 ºC for healthy adults and 12.2-13.9 ºC for healthy juveniles (Kidd, 2011).

The most conservative threshold (19 ºC) was exceeded >750 days in 2018, similar but slightly less than in previous years (2013-2017). There were similar numbers of days exceeding 20ºC and 21ºC in 2016; however, 2018 was similar to 2014 in the number of days with temperatures exceeding 22ºC. This pattern reflects the shift from moderate flows early in the water year to low flows after the freshet subsided.
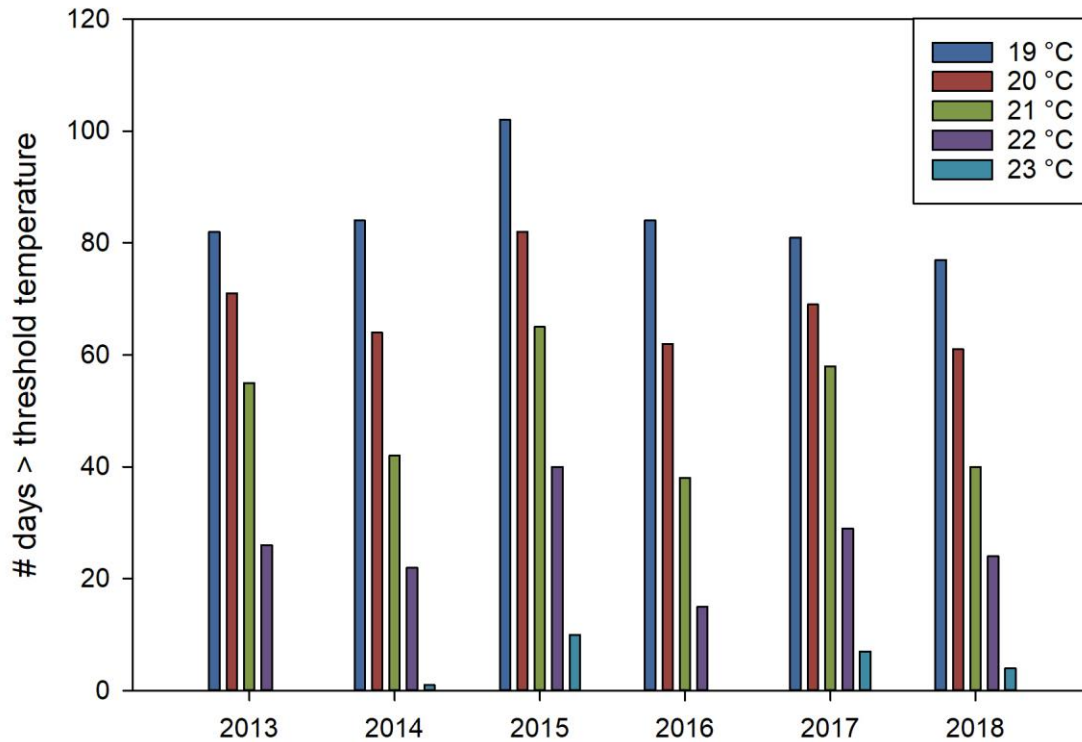
Figure 13. Number of days (computed from hourly data and converted to days, i.e., day-equivalents) where water temperatures exceeded the indicated temperature thresholds in the Columbia mainstem at Camas, WA (RM-122) (19 ºC, 20 ºC, 21 ºC, 22 ºC, 23 ºC).

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 14. 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 2017, the tributaries had a relatively large contribution to total discharge during the peak in April 2018, 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.
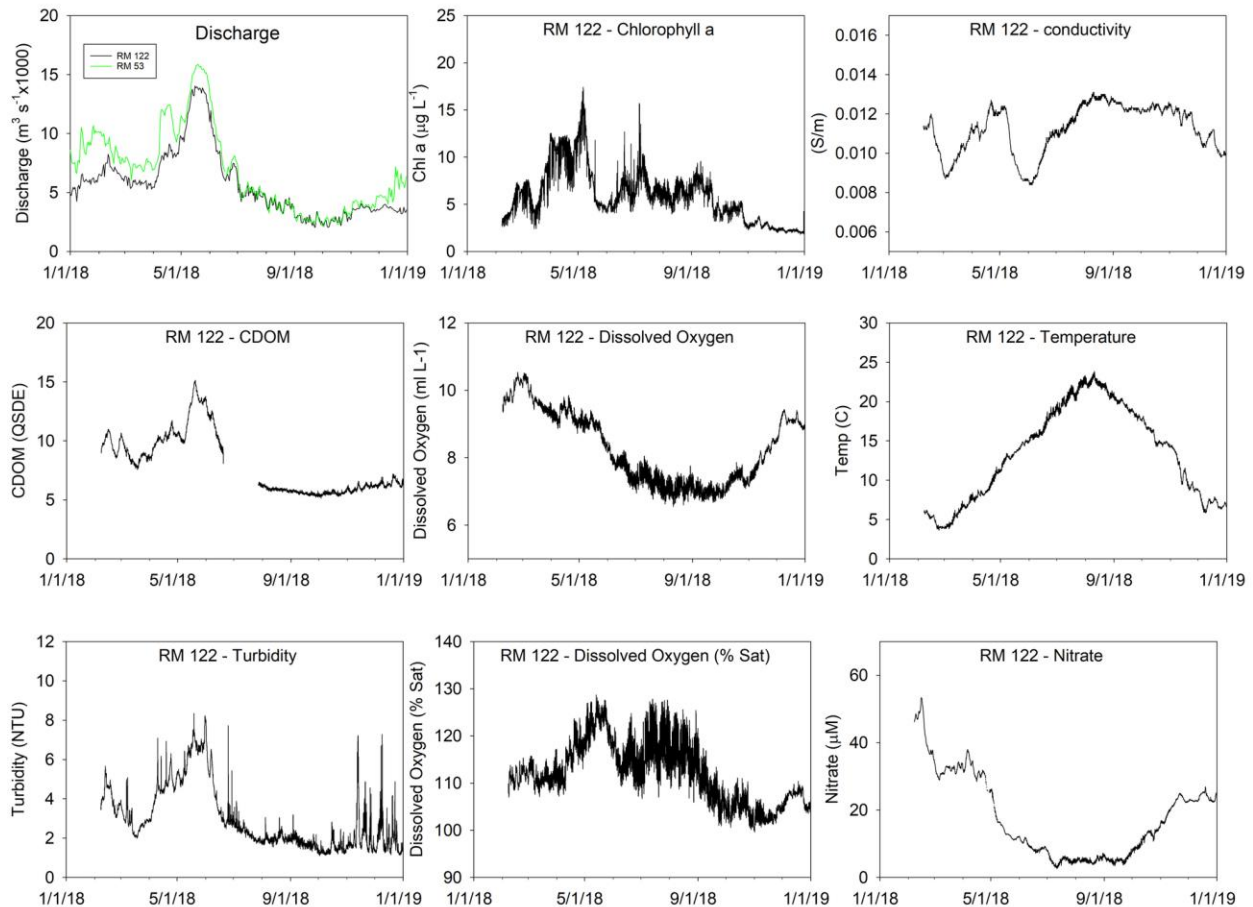
Figure 14. Time series of hourly water quality parameters measured at River Mile 122 (Camas, WA) in 2017.

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 14).

Dissolved nitrate concentrations were lower in 2018 compared to 2017 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*

Ilwaco is strongly influenced by tidal exchange with marine waters from the coastal ocean, particularly in the summer months. Salinity is the clearest indicator of this influence: in late July, waters went from mesohaline (~0.5–5 practical salinity units, PSU) to polyhaline (~5–18 PSU). The influence of coastal water was also evident in the drop in dissolved oxygen saturation, which fell below 50% in June and briefly in July; interestingly, in contrast to 2017 where DO saturation values continued to decline from late June until September, there was a dramatic increase in early July in 2018 and a slower decline until mid-September. While dissolved oxygen percent saturation values relative to the atmosphere were as low as 45% at Ilwaco during the warm months of August and September in 2017, values were between 60 and 70% over the same period. pH values were higher in 2018 than in 2017, with values fluctuating between 7.3 and 7.8. These values fall within the recommended range for good water quality (6.5–8.5; Washington State Water Quality Standards).
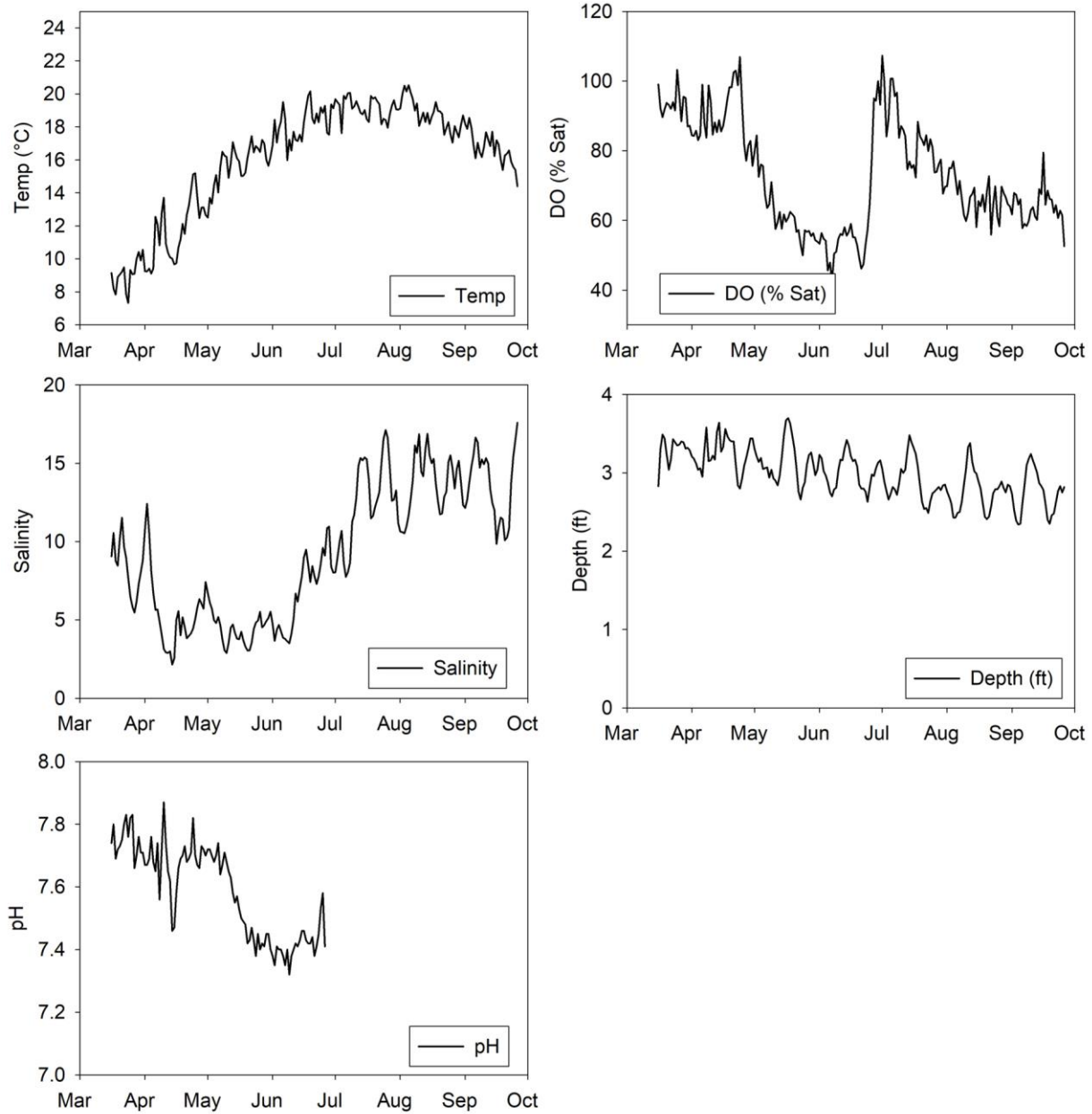
Figure 15. Time series of daily averaged measurements of water quality parameters made at Ilwaco Slough, 2018.

3.2.1.2   *Welch Island*

Maximum summer temperatures at Welch Island reached ~23 °C in August, which was slightly higher than observed at Ilwaco (Figure 16). Dissolved oxygen saturation levels were much higher at Welch Island than at Ilwaco, ranging from ~80% to close to 120%. Conductivity increased during the summer months as river discharge declined and water levels fell. Between March and late September, pH ranged from ~7.5 to 8.2, which falls within the recommended range (6.5–8.5; Washington State Water Quality Standards).
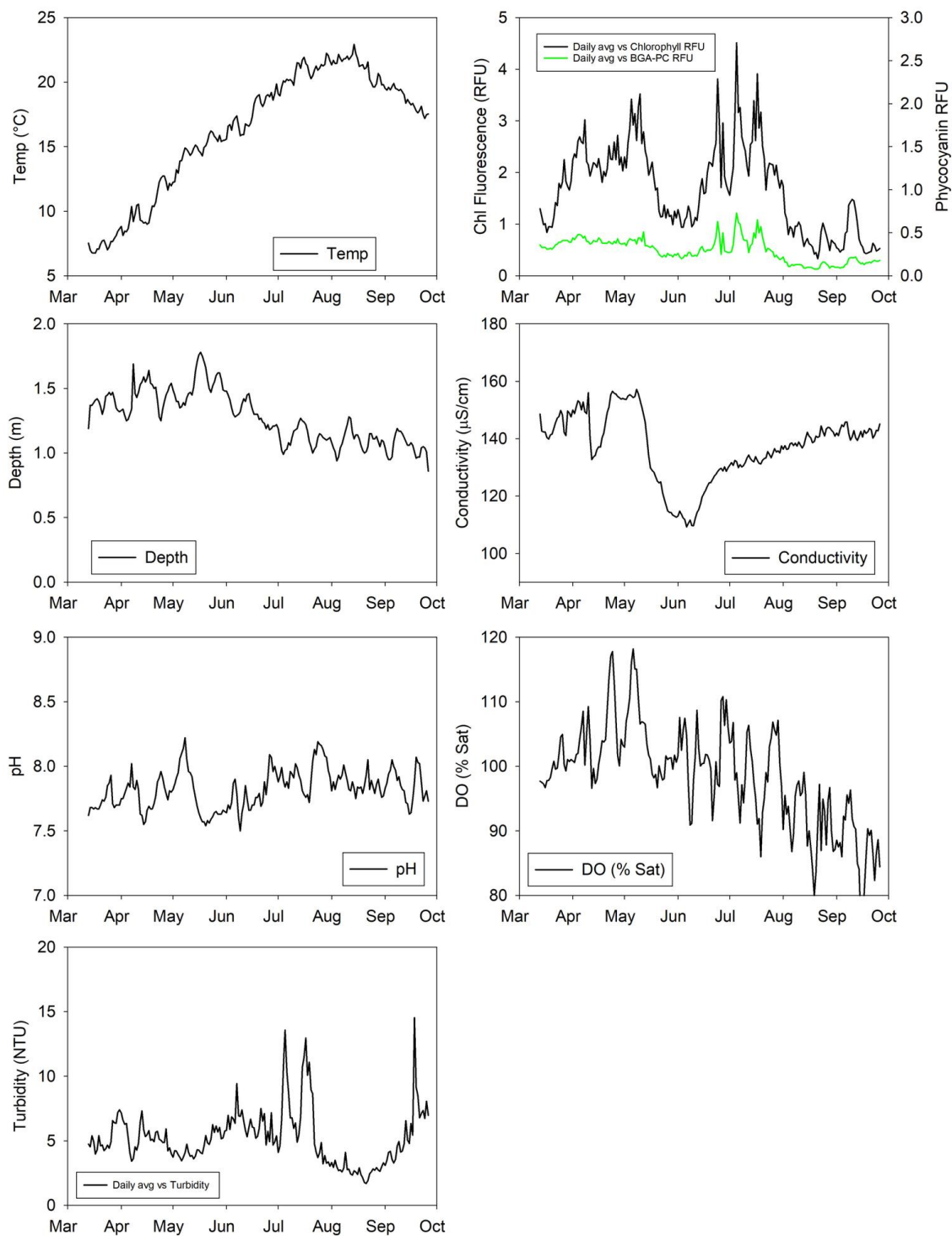
Figure 16. Time series of daily averaged measurements of water quality parameters made at Welch Island, 2018.

3.2.1.3 *Whites Island*

Temporal patterns in water quality parameters at Whites Island were very similar to those observed at Welch Island. In 2018, dissolved oxygen saturation levels were similar between Welch Island and Whites Island prior to the freshet; however, after the freshet subsided, dissolved oxygen saturation was higher at Whites Island, never dipping below 95%.
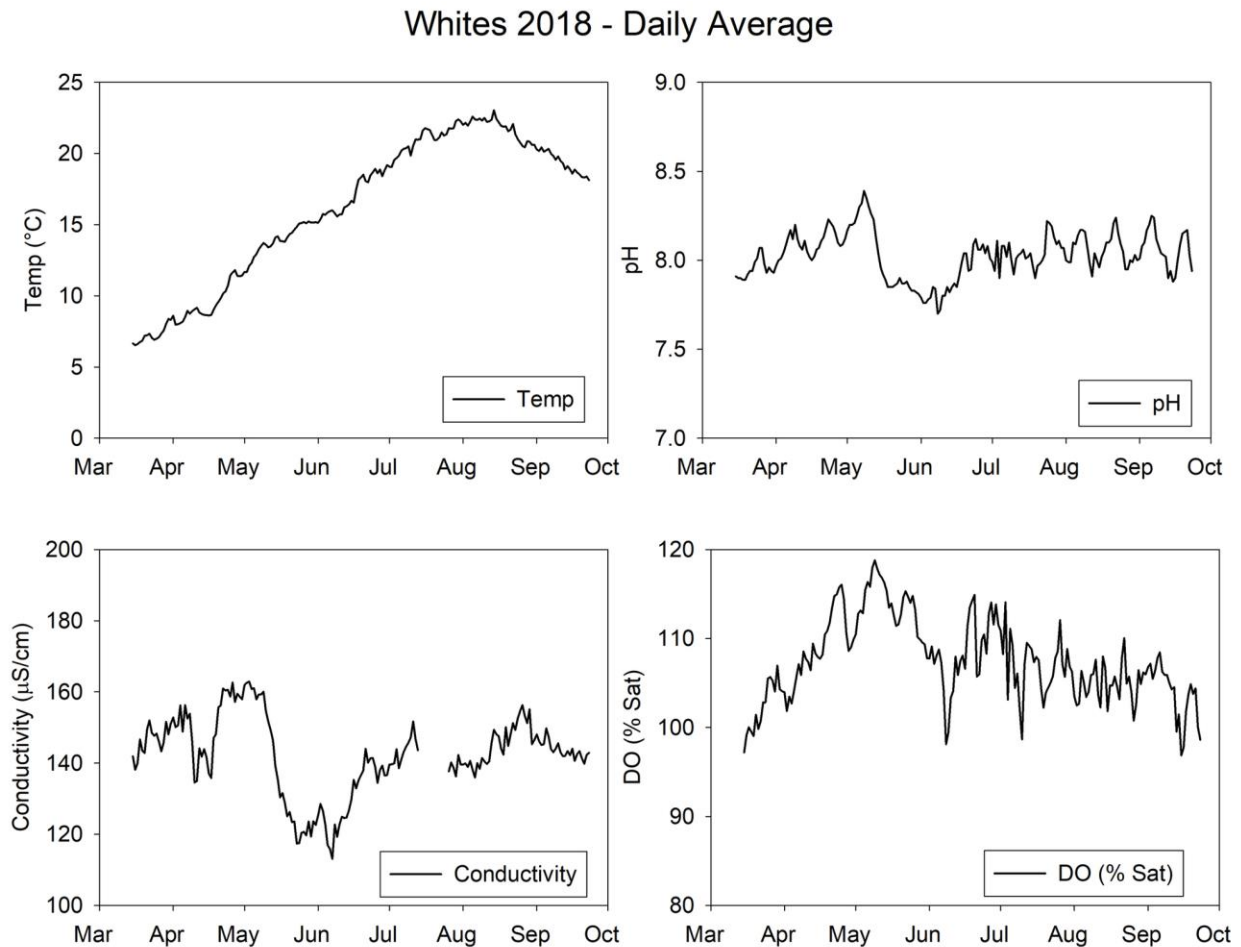
Figure 17. Time series of daily averaged measurements of water quality parameters made at Whites Island, 2018.

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 ºC in August (Figure 18). The influence of river flow on sensor depth is evident in the increased depth that coincides with the spring freshet. All biogeochemical properties reflect the influence of the freshet, with high elevation values, 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).

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 40% relative to atmospheric values during the spring freshet. Once the freshet subsided and water levels receded, levels

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of dissolved oxygen saturation showed dramatic fluctuations that reflect the period of very low dissolved oxygen that influenced the daily average (
Figure **19**). 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).
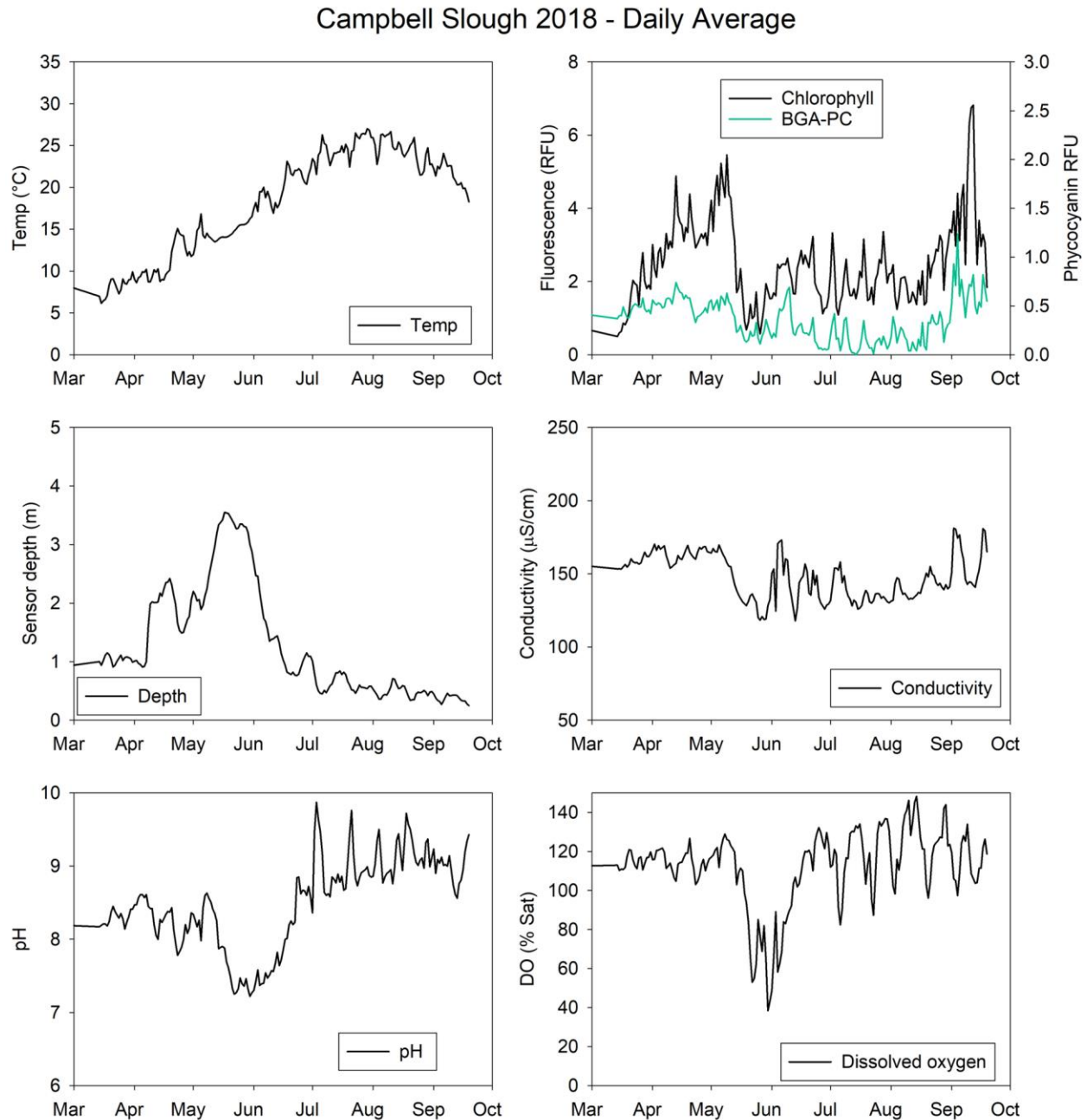
Figure 18. Time series of daily averaged measurements of water quality parameters made at Campbell Slough, 2018.

Campbell Slough 2018 - 30 min intervals
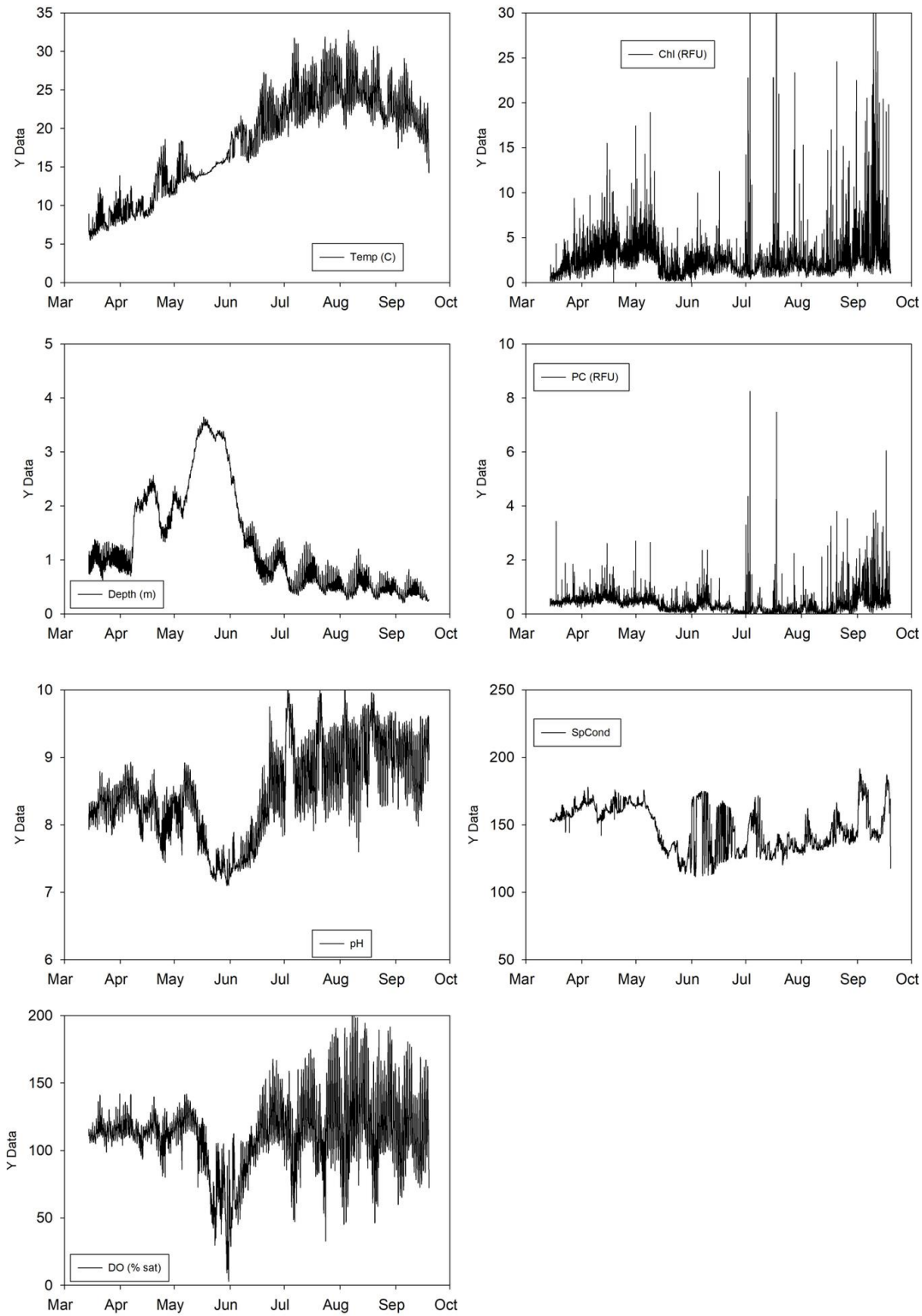
Figure 19. Time series of in situ water quality measurements made every 30 minutes at Campbell Slough in 2018.

3.2.1.5 *Franz Lake Slough*

Similar to Campbell Slough, summer temperatures at Franz exceeded 25 ºC from late-July through mid-August (Figure 20). Peaks in chlorophyll and phycocyanin fluorescence were large from April through August, with a decline accompanying the spring freshet. Spring primary production—as inferred from chlorophyll fluorescence—occurred in a large, long-lived peak lasting from mid-April to mid-May. In contrast, although levels of phytoplankton biomass were high during the post-freshet period, the peaks were shorter lived and punctuated by low values. Conductivity values fluctuated more widely at Franz Lake Slough compared to Campbell Slough, as did pH. Dissolved oxygen saturation values were generally higher than at Campbell Slough; however, short-lived low values were observed in the daily averages, driven by periodic low values occurring during some part of the day (Figure 21).

Unlike in 2017, pH values were within the recommended range for good water quality for the mainstem Columbia (7.0–8.5; Washington State Water Quality Standards). However, a look at the 30-minute measurements shows that pH values exceeding 9.0 occur during some parts of the day (Figure 21), which exceeds water quality thresholds. Phycocyanin peaks were higher at Franz Lake Slough than at the other off-channel sites following the spring freshet; before the spring freshet, both Franz Lake Slough and Campbell Slough had similar peaks in phycocyanin.
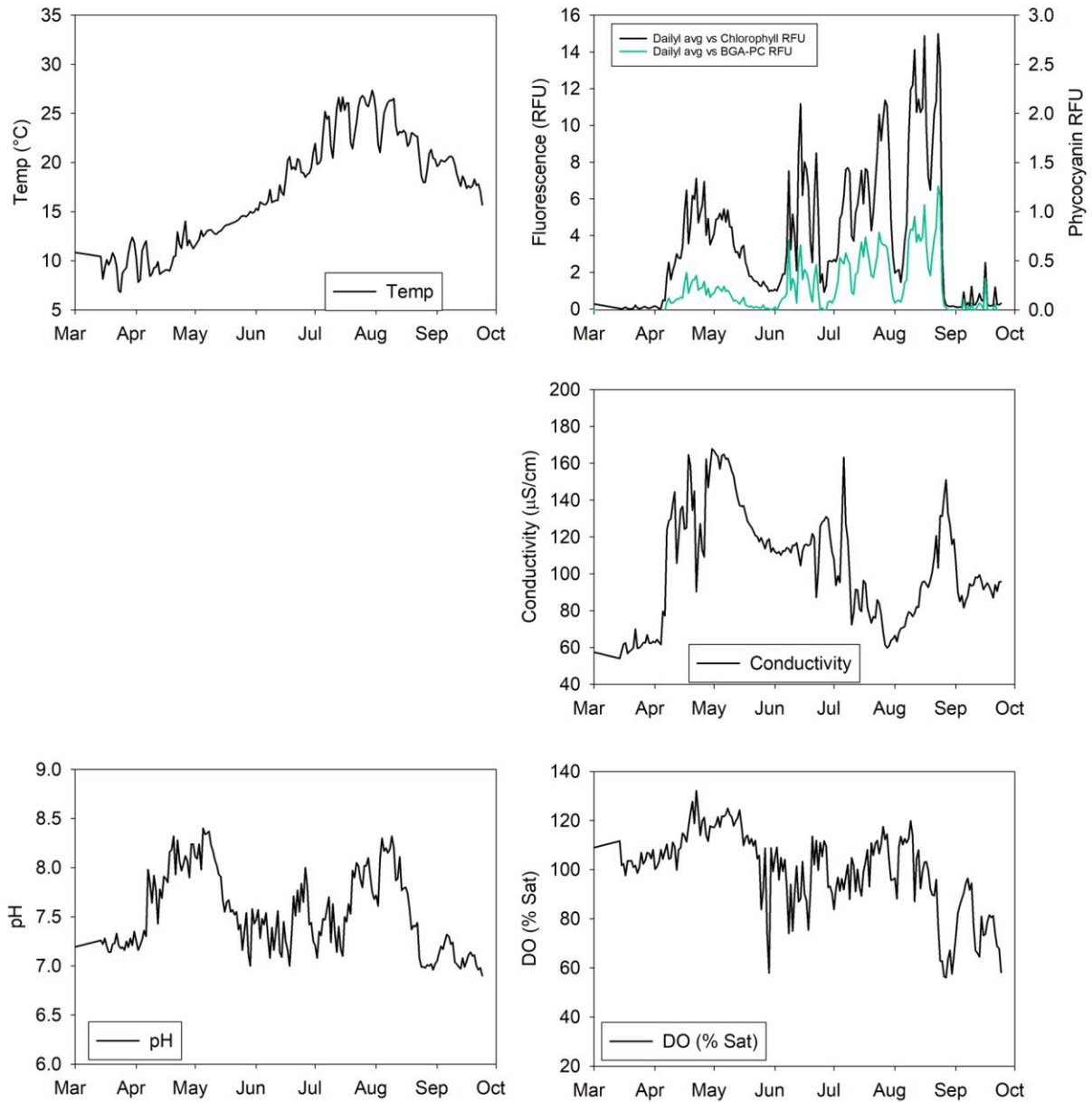
Figure 20. Time series of daily averaged measurements of water quality parameters made at Franz Lake Slough, 2018.
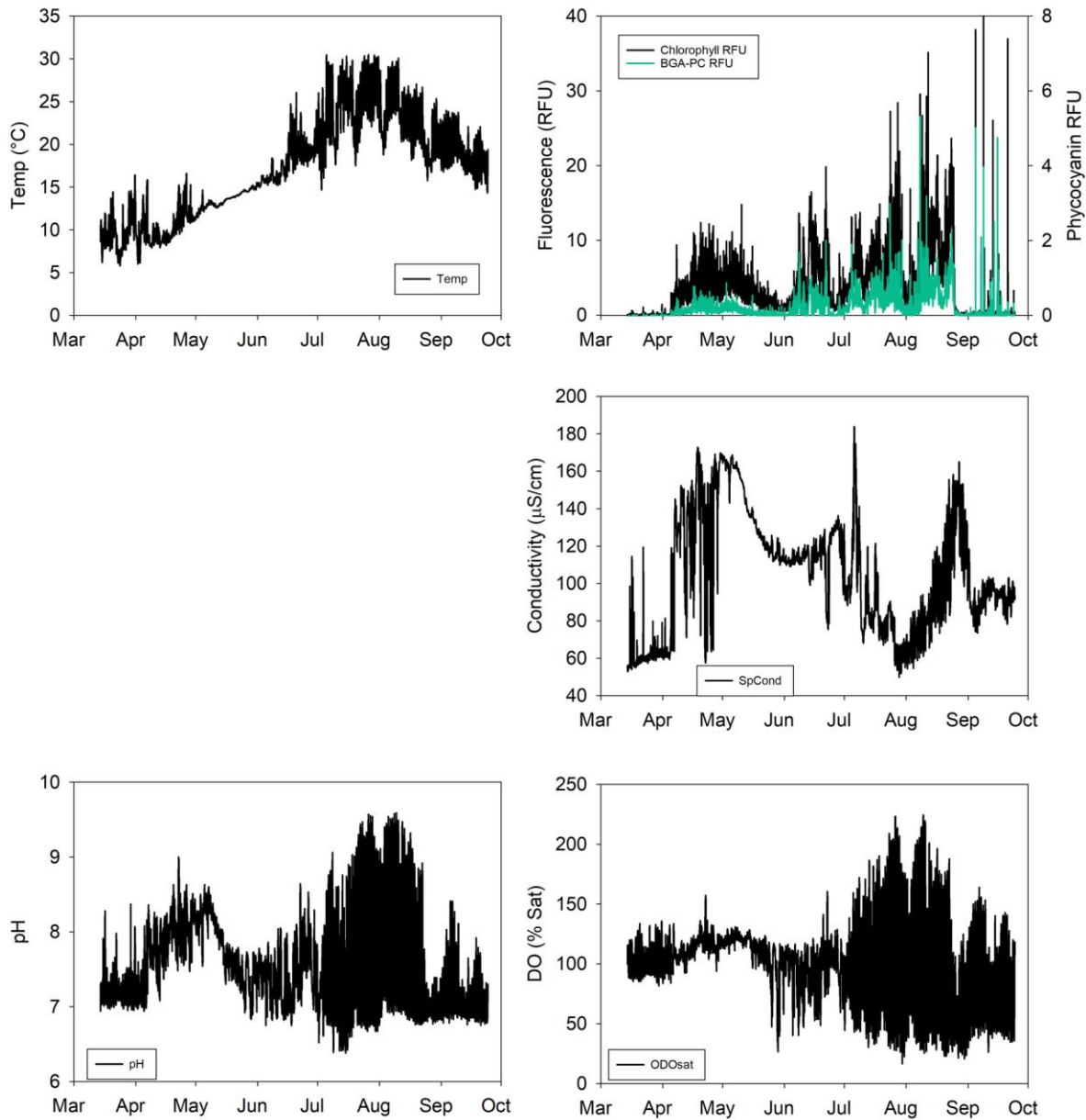
Figure 21. Time series of in situ water quality measurements made every 30 minutes at Franz Lake Slough in 2018.

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 22). Ilwaco had the greatest number of hours where dissolved oxygen concentrations were below each of the three threshold levels (>550 in 2018). This trend was observed for each of the years for which we have data, and the greatest number of hours with sub-optimal dissolved oxygen concentrations was observed in 2018. Dissolved oxygen concentrations below the 2 mg L^{-1} threshold were observed at Ilwaco, Campbell Slough, and to a limited degree, Franz Lake slough in 2018. Dissolved oxygen levels below 6 mg L^{-1} were observed at all of the sites except at Welch Island, with the greatest number occurring in 2016, 2017, and 2018.
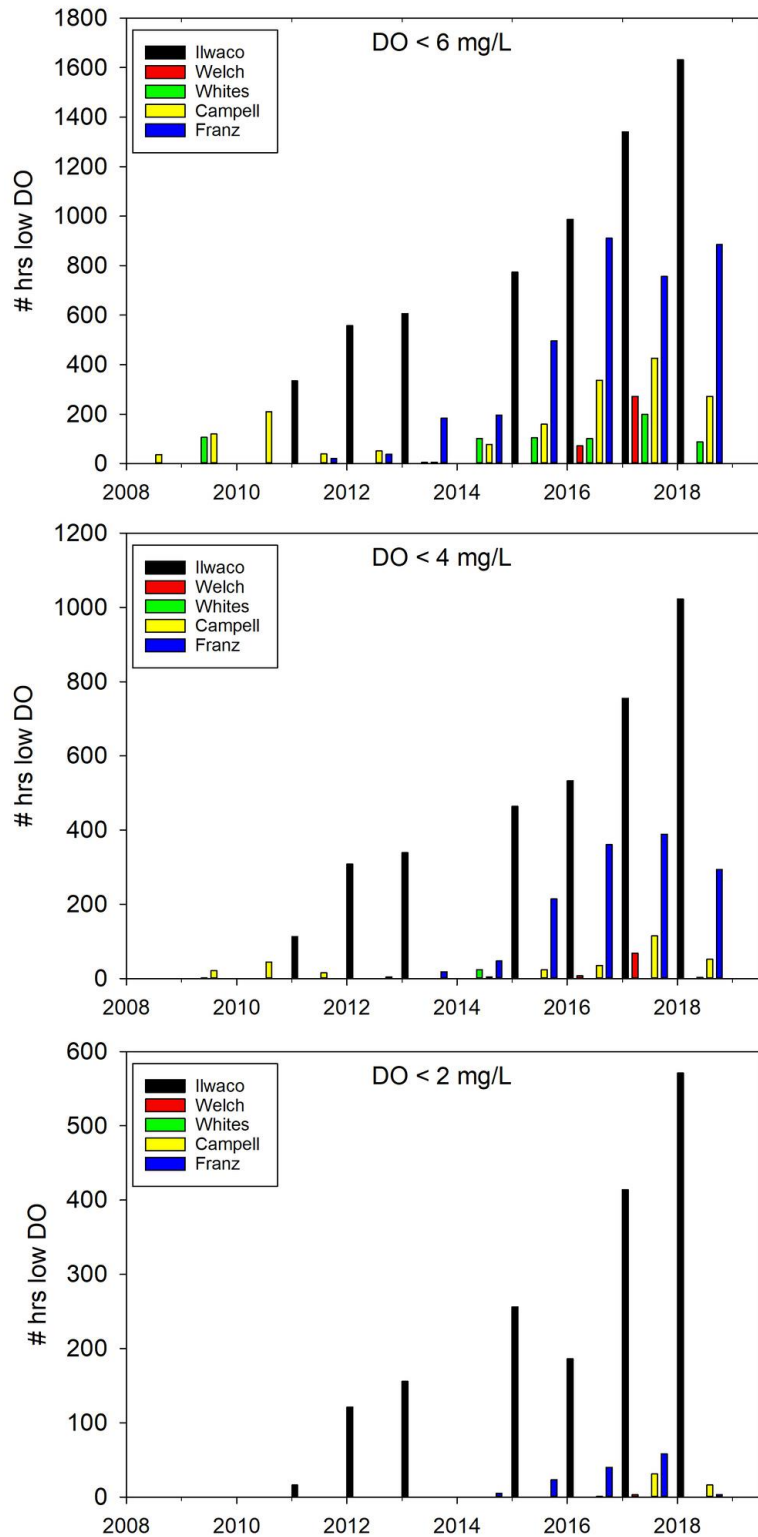
Figure 22. Time series of observations corresponding to three thresholds of dissolved oxygen concentration (mg L^{-1}): top panel, dissolved oxygen concentrations <6 mg L^{-1}; middle panel, dissolved oxygen concentrations <4 mg L^{-1}; bottom panel, dissolved oxygen concentrations <2 mg L^{-1}. The data

show the number of hours where dissolved oxygen concentrations were below each threshold, as determined by in situ sensors. Dissolved oxygen concentrations were below 2 mg L^{-1} for the equivalent of ~23 days (>550 h) between March and September in 2018 at Ilwaco. It should be noted that these hourly measurements are not necessarily consecutive, but they indicate the number of day-equivalents of sub-optimal habitat for juvenile salmonids. Most of the low-oxygen data come from summer observations.

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 where temperatures corresponded to threshold exceedance for five off-channel trends sites (Table 16). The thresholds presented here are >18 ºC, >20 ºC, and >22 ºC. According to these criteria, high temperatures posed a potential problem for salmonids during the summers of 2009, and 2012–2018 at Campbell Slough, based on the high percentages of days where thresholds were exceeded. Frequent high temperatures were observed at Franz Lake Slough from 2013–2018. Whites Island also had a high frequency of days where temperatures exceeded the thresholds in 2015 and 2016-18.

From the data, it is clear that there is variability in terms of the timing of the frequency of high-temperature waters in the late spring and summer, which relates to the timing and magnitude of river discharge. The time series suggests that the period of greatest vulnerability occurs during July and August.

Table 16: Temperatures exceeding 19°C in the EMP sites from 2009-2018. Data are 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	Date	Ilwaco	Welch	Whites	Campbell	Franz
Apr-2009	#N/A	#N/A	#N/A	#N/A	#N/A	Apr-2014	#N/A	#N/A	#N/A	0	0
May-2009	#N/A	#N/A	#N/A	0	#N/A	May-2014	#N/A	#N/A	0	0	0
Jun-2009	#N/A	#N/A	0	40	#N/A	Jun-2014	#N/A	#N/A	0	23	0
Jul-2009	#N/A	#N/A	70	100	#N/A	Jul-2014	#N/A	#N/A	86	100	94
Aug-2009	#N/A	#N/A	100	100	#N/A	Aug-2014	#N/A	#N/A	#N/A	#N/A	#N/A
Sep-2009	#N/A	#N/A	#N/A	#N/A	#N/A	Sep-2014	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2010	#N/A	#N/A	#N/A	0	#N/A	Apr-2015	#N/A	#N/A	#N/A	#N/A	#N/A
May-2010	#N/A	#N/A	#N/A	0	#N/A	May-2015	#N/A	#N/A	#N/A	32	#N/A
Jun-2010	#N/A	#N/A	#N/A	13	#N/A	Jun-2015	17	#N/A	100	90	100
Jul-2010	#N/A	#N/A	#N/A	#N/A	#N/A	Jul-2015	74	#N/A	100	100	100
Aug-2010	#N/A	#N/A	#N/A	#N/A	#N/A	Aug-2015	56	#N/A	100	100	100
Sep-2010	#N/A	#N/A	#N/A	#N/A	#N/A	Sep-2015	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2011	0	#N/A	0	#N/A	0	Apr-2016	0	0	0	0	0
May-2011	0	#N/A	#N/A	0	0	May-2016	0	0	0	19	0
Jun-2011	0	#N/A	0	0	0	Jun-2016	10	13	0	87	55
Jul-2011	13	#N/A	0	58	0	Jul-2016	55	100	87	100	97
Aug-2011	#N/A	#N/A	#N/A	#N/A	#N/A	Aug-2016	23	100	100	100	100
Sep-2011	#N/A	#N/A	#N/A	#N/A	#N/A	Sep-2016	#N/A	#N/A	#N/A	#N/A	#N/A
Apr-2012	0	#N/A	0	0	0	Apr-2017	0	0	0	0	#N/A
May-2012	0	#N/A	0	0	0	May-2017	0	0	0	0	0
Jun-2012	0	#N/A	0	3	0	Jun-2017	7	0	0	33	13
Jul-2012	3	#N/A	#N/A	77	43	Jul-2017	42	87	94	100	93
Aug-2012	#N/A	#N/A	#N/A	#N/A	#N/A	Aug-2017	29	100	100	100	100
Sep-2012	#N/A	#N/A	#N/A	#N/A	#N/A	Sep-2017	0	82	90	100	78
Apr-2013	0	#N/A	0	0	0	Apr-2018	0	0	0	0	0
May-2013	0	#N/A	0	0	0	May-2018	0	0	0	0	0
Jun-2013	23	#N/A	4	53	27	Jun-2018	27	17	3	63	30
Jul-2013	20	#N/A	80	100	96	Jul-2018	68	97	100	100	100
Aug-2013	#N/A	#N/A	#N/A	#N/A	#N/A	Aug-2018	29	100	100	100	90
Sep-2013	#N/A	#N/A	#N/A	#N/A	#N/A	Sep-2018	0	46	57	95	42

3.2.3 Nutrients

3.2.3.1 *Dissolved Inorganic Nutrients (nitrate, phosphate)*

As noted above, dissolved nitrate concentrations reach high levels in the Columbia in the winter. Observations from trends sites begin in March at the end of winter/early spring when nitrate was ~30 μM at all sites except for Ilwaco Slough. At Ilwaco Slough, nitrate was closer to 20 μM. In 2018, nitrate concentrations were higher at all sites in March and April compared to May, June, and July (Figure 23). However, there were spatial differences among the sites; for example, there was a decline in nitrate concentration between March and April at both Campbell Slough and Franz Lake Slough (located upstream of the Willamette-Columbia confluence), which was not observed at Ilwaco Slough, Welch Island, or Whites (Figure 23). There were smaller differences among the monthly observations of nitrate concentration at Ilwaco compared to the other sites due to inputs from the coastal ocean in the summer months. Nitrate concentrations were lowest during the summer months, reaching minimum values in July, particularly at Whites Island and 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.
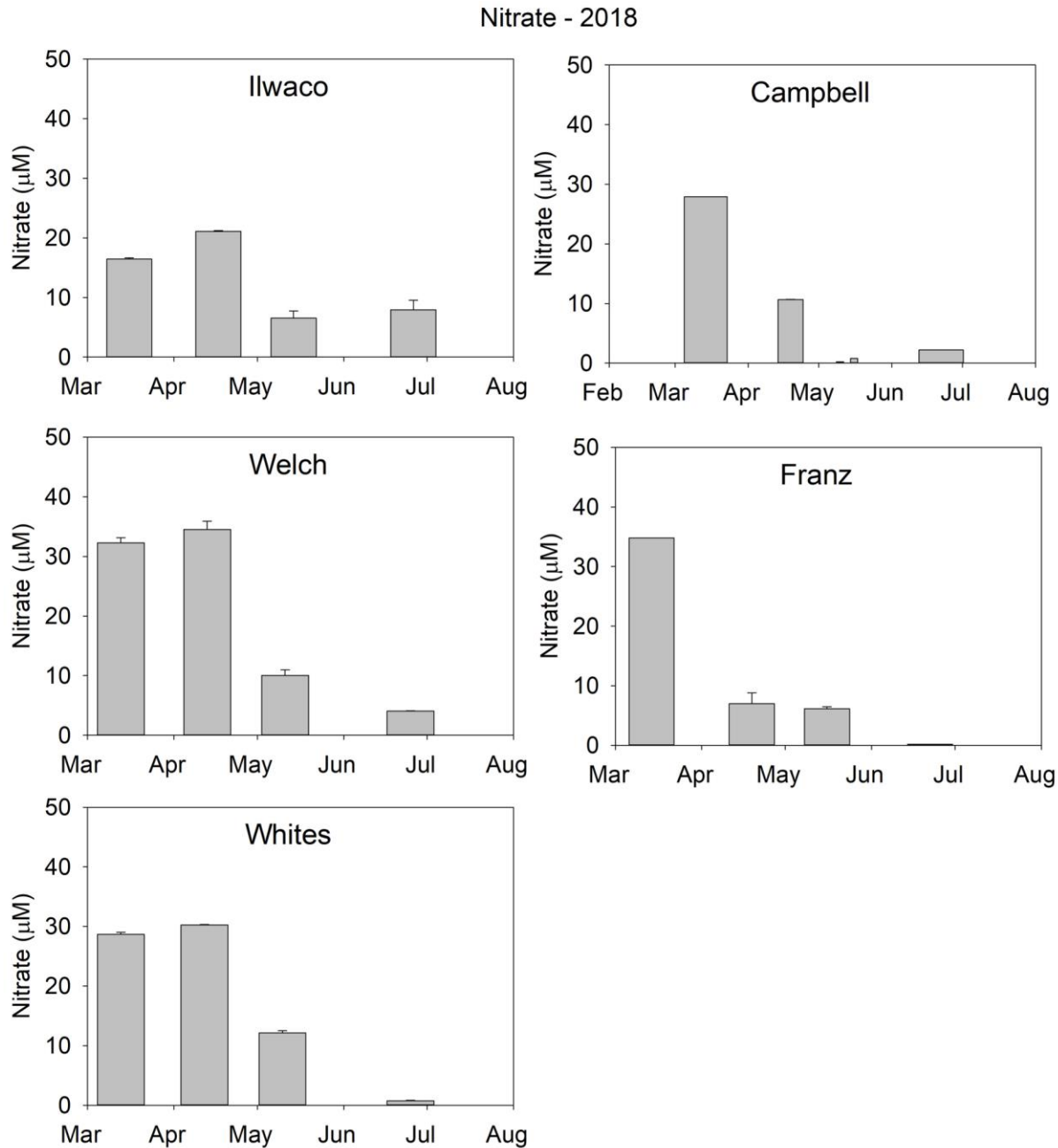
Figure 23. Time series are showing concentrations of dissolved nitrate at the five trends sites in 2018.

Dissolved phosphorus (determined as soluble reactive phosphorus, or ortho-phosphate) showed similar temporal patterns in variation to nitrate concentrations at Ilwaco Slough, Welch Island and Whites Island (Figure 24). At Campbell Slough, phosphate concentrations declined between March and May, consistent with drawdown associated with primary production. However, summer levels were similar to those in March, which reflects inputs from the coastal ocean during the summer at this site. Phosphate concentrations at Franz Lake Slough were higher than at the other sites, with the highest value observed in late June, where levels exceeded 2.5 µM. Phosphate concentrations at Campbell Slough declined between March and May, but increased after the freshet subsided so that levels were similar in March and late June. Seasonal patterns in phosphate concentrations were similar between Welch Island and Whites

Island. 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.

Ammonium concentrations tend to increase as a result of microbial activity. The highest concentrations were observed at Ilwaco, particularly during the summer (Figure 25). At the other sites, ammonium concentrations tended to decrease from spring to summer.
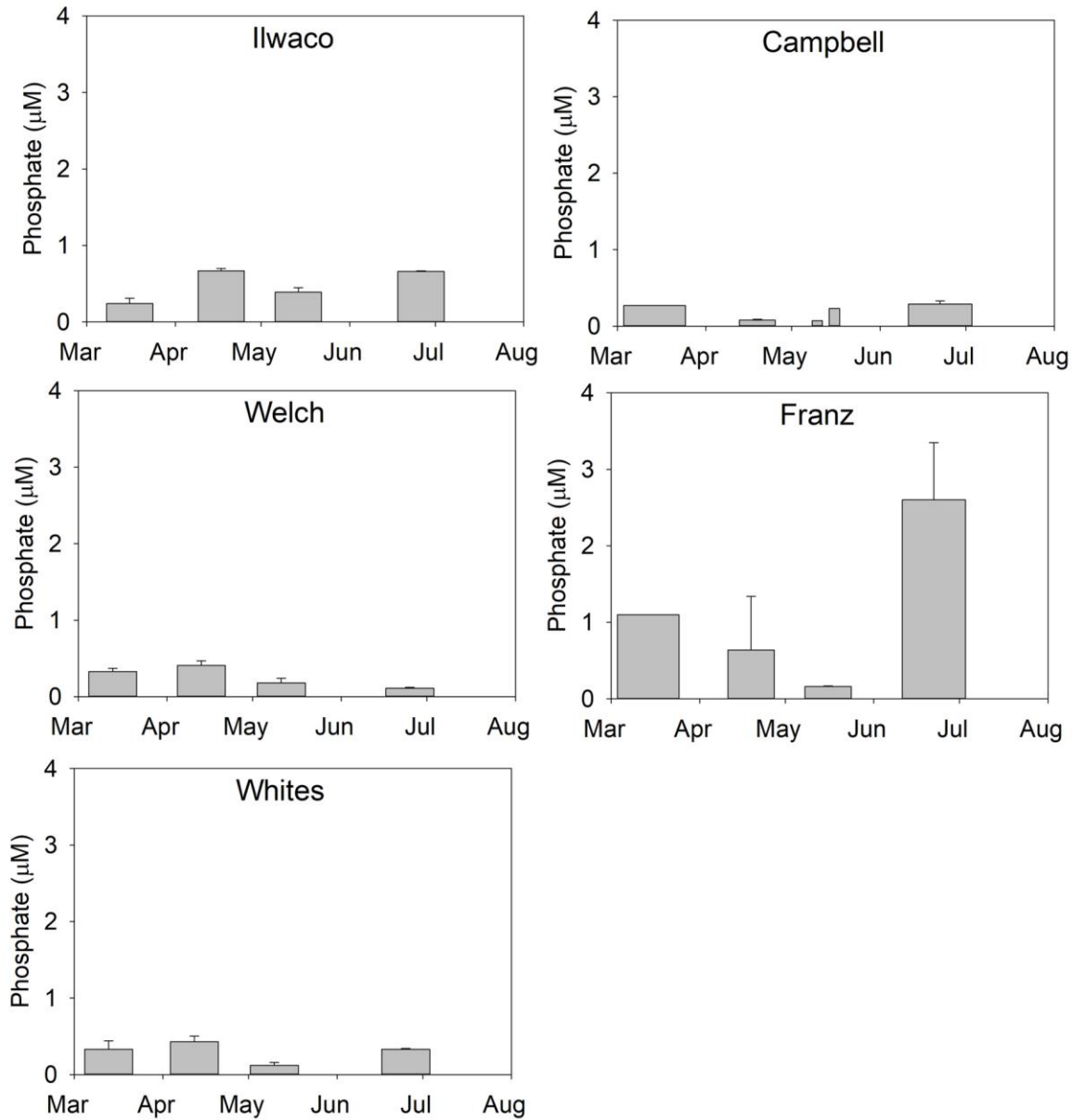
Figure 24. Time series showing concentrations of dissolved phosphate (ortho-phosphate) at the five trends sites in 2018.
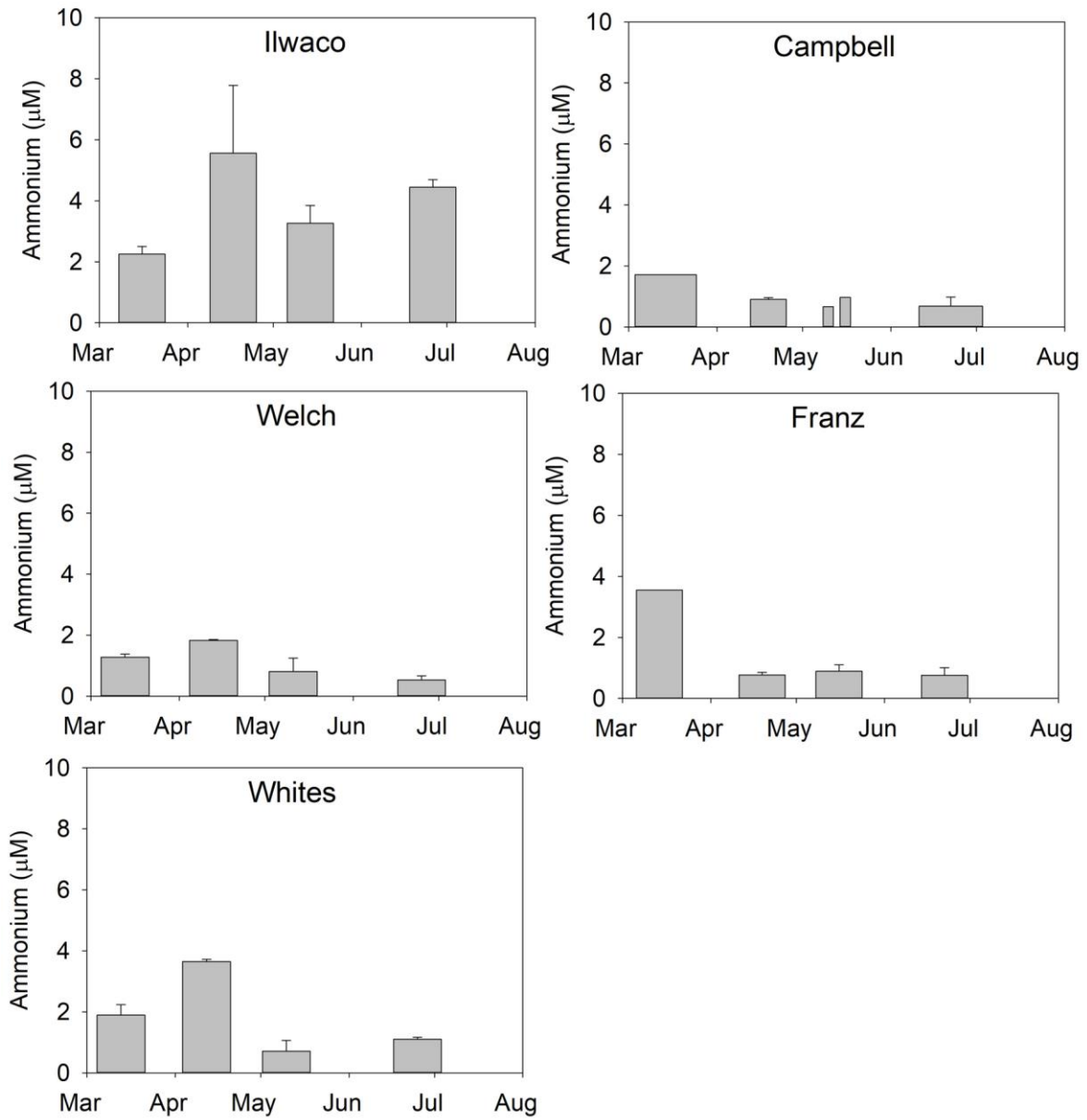
Figure 25. Time series showing concentrations of dissolved ammonium at the five trends sites in 2018.

3.2.3.2 *Assessing Connectivity Between Off-channel Habitats and the Mainstem*

Since off-channel habitats are influenced by water properties in the mainstem river, we can use the degree to which these sites are similar to assess relative connectivity. Since the river stretch between Bonneville Dam and the river mouth includes inputs from tributaries, including the Willamette, we wanted to determine whether using a single mainstem site for comparison to off-channel would be representative. Thus, we compared data from Saturn-04 near RM-17 (closest to sites in Reaches A, B, and C) and Camas, WA (RM-122; closest to sites in Reaches F and H) (Figure 26) to determine the similarity in temperature over the course of a year. The size of the temperature difference was largest when inputs from tributaries are greatest (i.e., during pluvial flow), while differences were much smaller in the summer (Figure 26). However, during the sampling season (March–September), the difference between the two mainstem sites are small. Knowing that observations at Saturn-04 are representative of the mainstem river, we can use these observations to assess connectivity between mainstem and off-channel sites based on temperature differences (Figure 27).
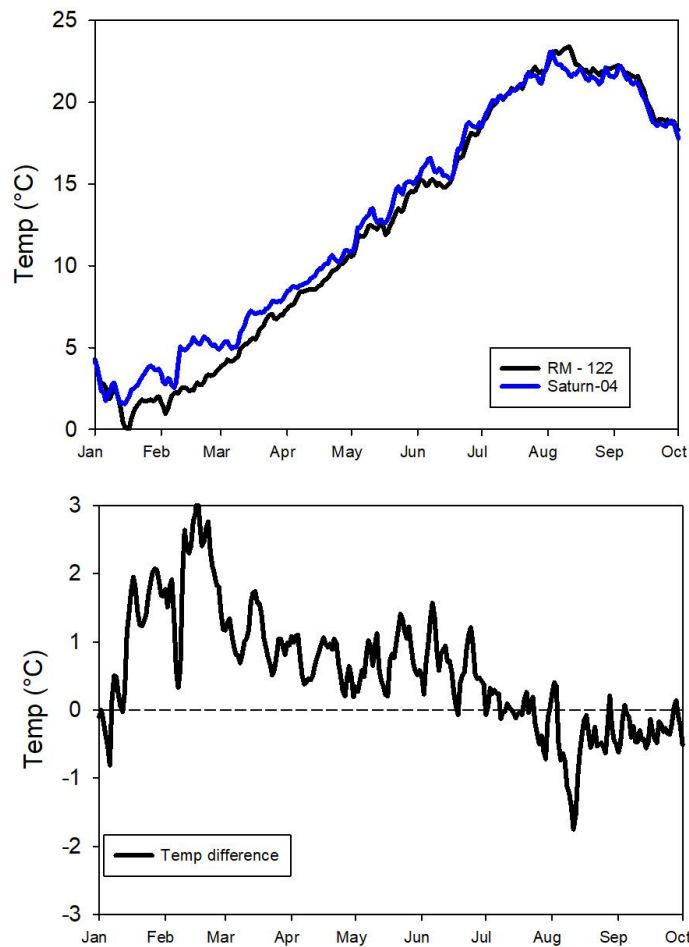
Figure 26. *Upper panel*: **Comparison of hourly water temperature data collected at two sites in the Columbia River mainstem (RM-122, or Camas, WA and RM-17, or Saturn-04). RM-122 is upstream of the Willamette-Columbia confluence, while RM-17 is below the confluence.** *Lower panel*: **Temperature difference (in ºC) between RM-122 and Saturn-04 (i.e., temperature at Saturn-04 – temperature at RM-122). The data indicate that temperature differences between different sites on the mainstem are between 0 and 2 ºC during most of the study period (March–September).**

Most of the off-channel sites were warmer compared to the mainstem prior to the end of June (Figure 27). Since both Welch Island and Whites Island showed temperature elevations that were similar to Saturn-04, these differences from the mainstem are considered non-significant. The temperature data from Whites Island were nearly identical to the data from Saturn-04, indicating that there was little difference from the mainstem. The largest differences were observed at Ilwaco, which was much warmer than the mainstem until the end of June, and then much colder than the mainstem between July and the end of September. Water temperatures at Campbell Slough were higher than in the mainstem prior to and after the period where river discharge began to subside (end of May–mid-June). Based on the differences in temperature between the mainstem and off-channel sites, the site most different from the mainstem is Ilwaco; this reflects the strong tidal exchange and marine influence at this site. Within the tidal freshwater realm, Campbell Slough and Franz Lake Slough could be considered substantially disconnected from the mainstem, which we define as a condition of slow exchange between the habitats. This slow exchange influences how the ecosystems within these habitats develop and function, which in turn affects resource availability and water quality.
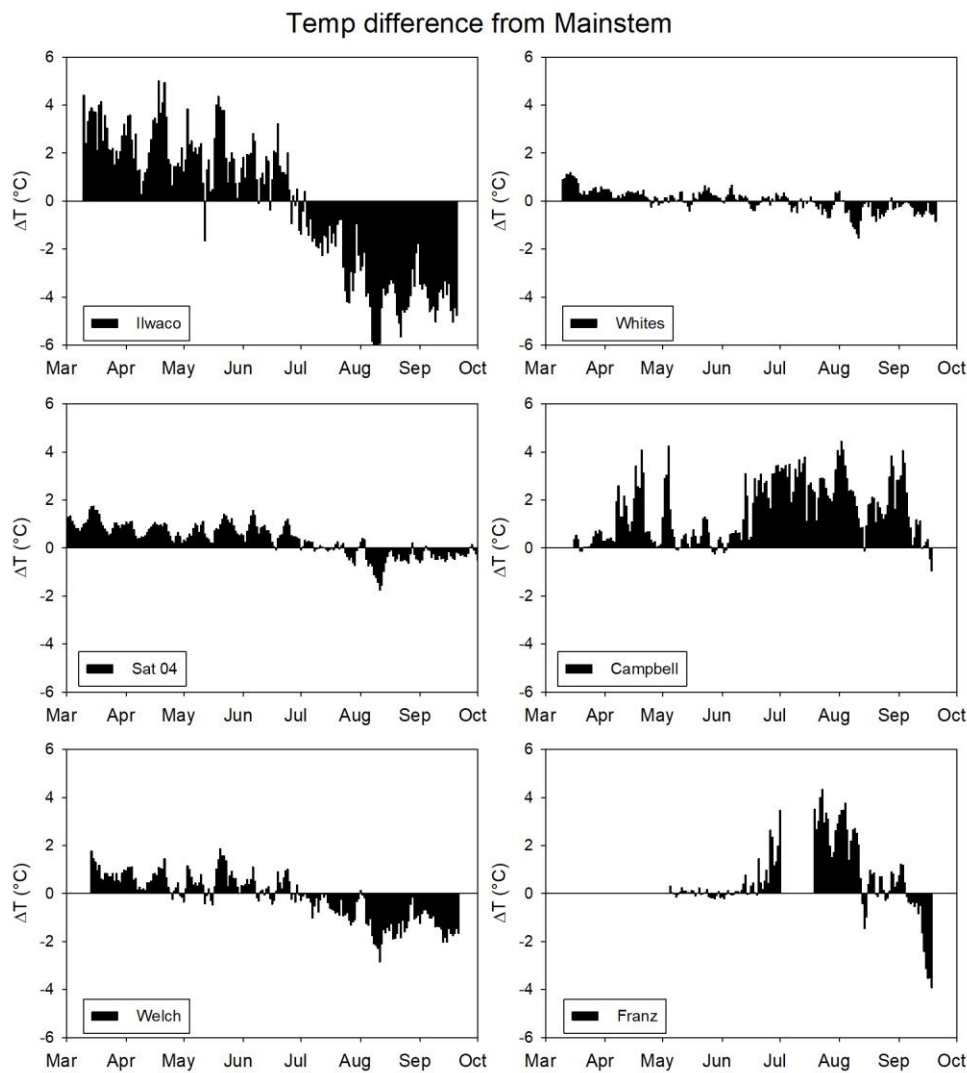
Figure 27. Time series of water temperature differences between the mainstem at Camas, WA (RM-122) and five off-channel trends sites, as well as Saturn-04.

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 2018 water year, are provided in Appendix C. The following observations were made for these sites:

- The WSE at the Ilwaco Slough site (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 at the site was 1.5 m in 2018, 2017, and 2016 (**Table 17**) and surveyed the wetland plant community range was 1.36 m in 2018 (**Table 18**).

- The Welch Island site (rkm 53) is predominantly tidal; however, slightly elevated WSE was detectable during the prolonged spring freshet in 2012, 2014, 2017, and 2018. Winter storms also drive higher water levels at this site, particularly elevating the low tide levels. This site's tidal range has varied from 2.2 to 2.1 m annually (**Table 17**), and the surveyed wetland plant community range was 0.96 m in 2018 (**Table 18**).

- The Whites Island site (rkm 72) exemplifies a mix of hydrologic drivers in the lower river. The average tidal range varied between 1.8-1.6 m (**Table 17**), and the surveyed wetland plant community range was 1.28 m in 2018 (**Table 18**). Elevated water levels occurring during winter storm events and large spring freshets such as in May 2018.

- The Cunningham Lake site (rkm 145) and Campbell Slough site rkm (149), respectively, have similar hydrologic patterns. Historic data indicate that Cunningham Lake has a slightly greater tidal range and slightly lower WSE during flood events compared to Campbell Slough, however the freshet did cause greater flooding at Cunningham Lake in 2018 (**Table 17**). 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. 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 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 18). 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 (on between 0.2- 0.3 m, **Table 17** 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 and 2018 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 WSEs 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 18).

Table 17. 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.

2018	Rkm	MWL	MLLW	MHHW	Avg Tidal Range (m)	Maximum WSE	Date of Maximum WSE	Period of Record	Days (n)
Ilwaco	6	1.5	1.0	2.5	1.5	3.1	Mar 2	Jan-Jul	211
Welch	53	2.4	1.6	3.4	1.8	4.1	May 17	Jan-Jul	212
Whites	72	1.9	0.9	3.0	2.1	4.0	Jan 30	Jan-Feb, May-July	149
Cunningham	145	3.8	3.6	4.1	0.5	7.0	May 17	Jan-Jul	219
Campbell	149	3.6	3.4	3.8	0.4	5.9	May 17	Jan-Jul	220
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 9	Jan-Feb, Aug-Dec	216
Welch	53	2.2	1.5	3.1	1.6	4.0	Dec 3	Jan-Dec	365
Whites	72	2.0	0.9	3.0	2.1	3.9	Feb 9	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	1.9	1.1	2.8	1.7	4.7	May 10	Aug-Dec	153
Whites	72	1.7	0.6	2.8	2.2	3.9	Jan 2	Jan-Dec	366
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 18. 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)	2018	2017	2016
Ilwaco Slough	Plots (n)	40 (1)	40	40	39
	Mean	1.96 (0.04)	1.94 (0.23)	1.91 (0.23)	1.94 (0.17)
	Min	1.08 (0.28)	0.95	0.94	1.44
	Max	2.36 (0.07)	2.31	2.31	2.31
	Range	1.28 (0.29)	1.36	1.38	0.87
Welch Island	Plots (n)	40 (0)	40	41	40
	Mean	2.05 (0.02)	2.00 (0.17)	2.06 (0.16)	2.06 (0.16)
	Min	1.32 (0.05)	1.22	1.34	1.34
	Max	2.19 (0.02)	2.14	2.2	2.2
	Range	0.87 (0.03)	0.92	0.86	0.86
Whites Island	Plots (n)	41 (6)	44	42	42
	Mean	2.06 (0.03)	2.09 (0.36)	2.08 (0.38)	2.08 (0.38)
	Min	1.2 (0.02)	1.21	1.2	1.2
	Max	2.54 (0.04)	2.63	2.53	2.53
	Range	1.34 (0.05)	1.42	1.33	1.33
Cunningham Lake	Plots (n)	56 (16)	67	68	69
	Mean	2.7 (0.07)	2.69 (0.19)	2.78 (0.19)	2.78 (0.19)
	Min	2.32 (0.11)	2.29	2.41	2.41
	Max	2.99 (0.07)	3.07	3.02	3.02
	Range	0.68 (0.08)	0.77	0.60	0.60
Campbell Slough	Plots (n)	57 (14)	60	60	61
	Mean	2.98 (0.05)	3.01 (0.37)	2.95 (0.32)	2.99 (0.36)
	Min	2.46 (0.09)	2.4	2.51	2.51
	Max	3.99 (0.08)	4.01	3.76	4.00
	Range	1.52 (0.13)	1.61	1.25	1.49
Franz Lake	Plots (n)	56 (11)	64	60	61
	Mean	4.59 (0.04)	4.62 (0.29)	4.6 (0.23)	4.6 (0.23)
	Min	3.92 (0.13)	3.87	3.95	3.95
	Max	5.04 (0.04)	5.05	5.01	5.01
	Range	1.11 (0.12)	1.19	1.06	1.06

Inter-annual variation in inundation patterns is much greater at the upper estuary sites (**Table 17**, 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 **Error! Reference source not found.**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).
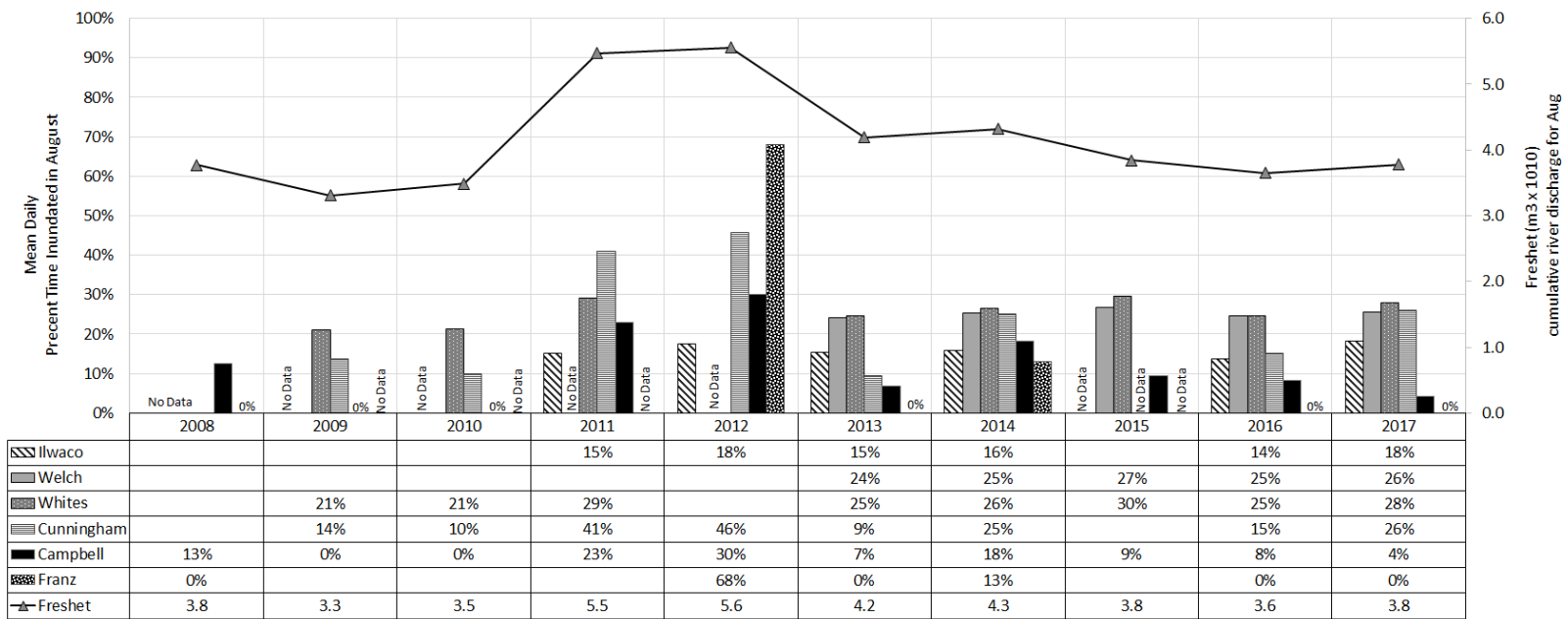
Figure 28: Mean daily % time inundation for the month of August at all five trends sites between 2008-2017; Sites are ordered from left to right starting at the river mouth. Average site elevations are given in Table 18.

	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

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In 2017, the lower river sites, Ilwaco Slough, Welch Island, and Whites Island, showed similar and consistent trends of daily flooding during the month of August compared to previously monitored years (Figure 28-Figure 30). 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).

Ilwaco Slough
Mean Daily % Inundation (August) vs. Marsh Elevation
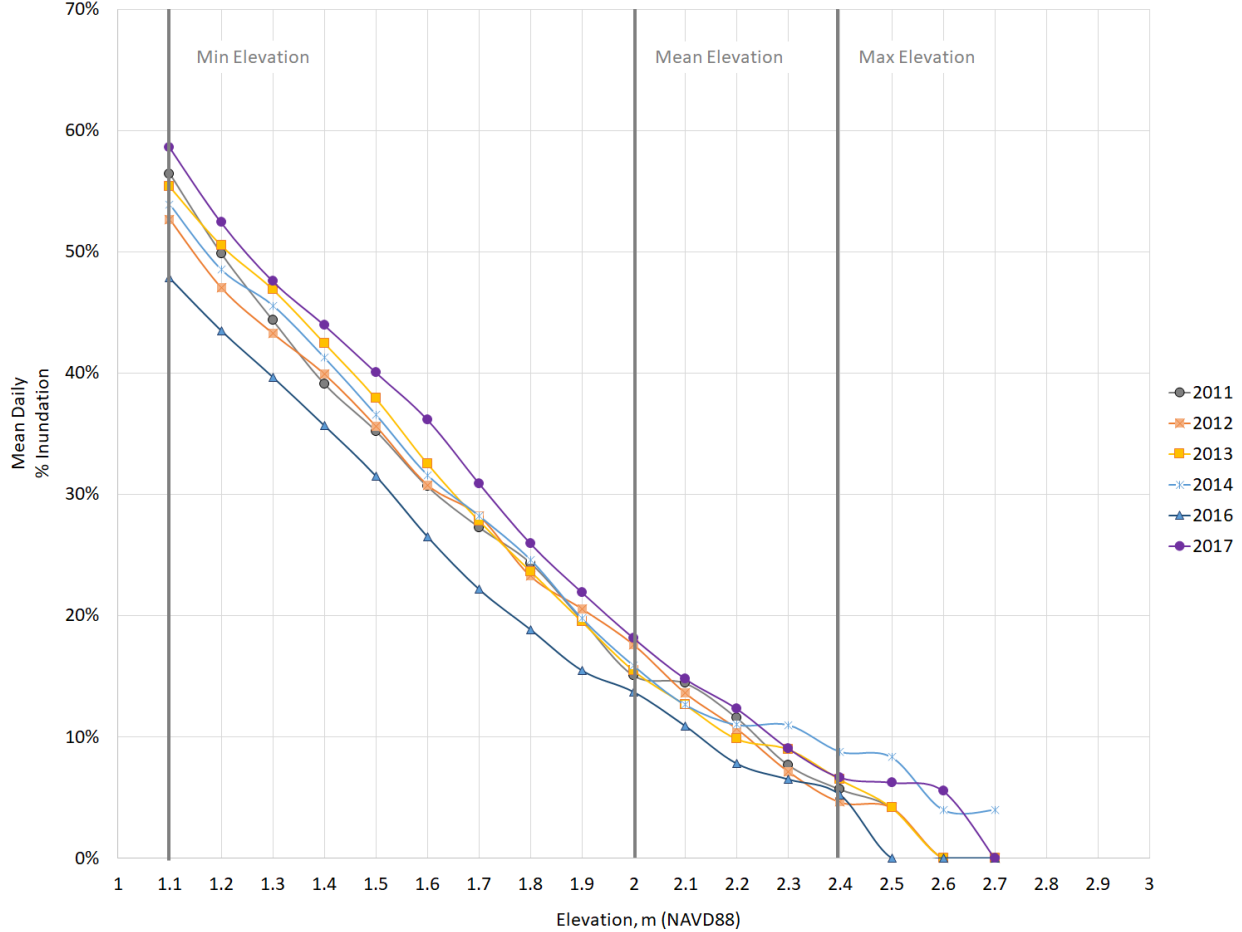
Figure 29: Ilwaco Slough: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2011-2017.
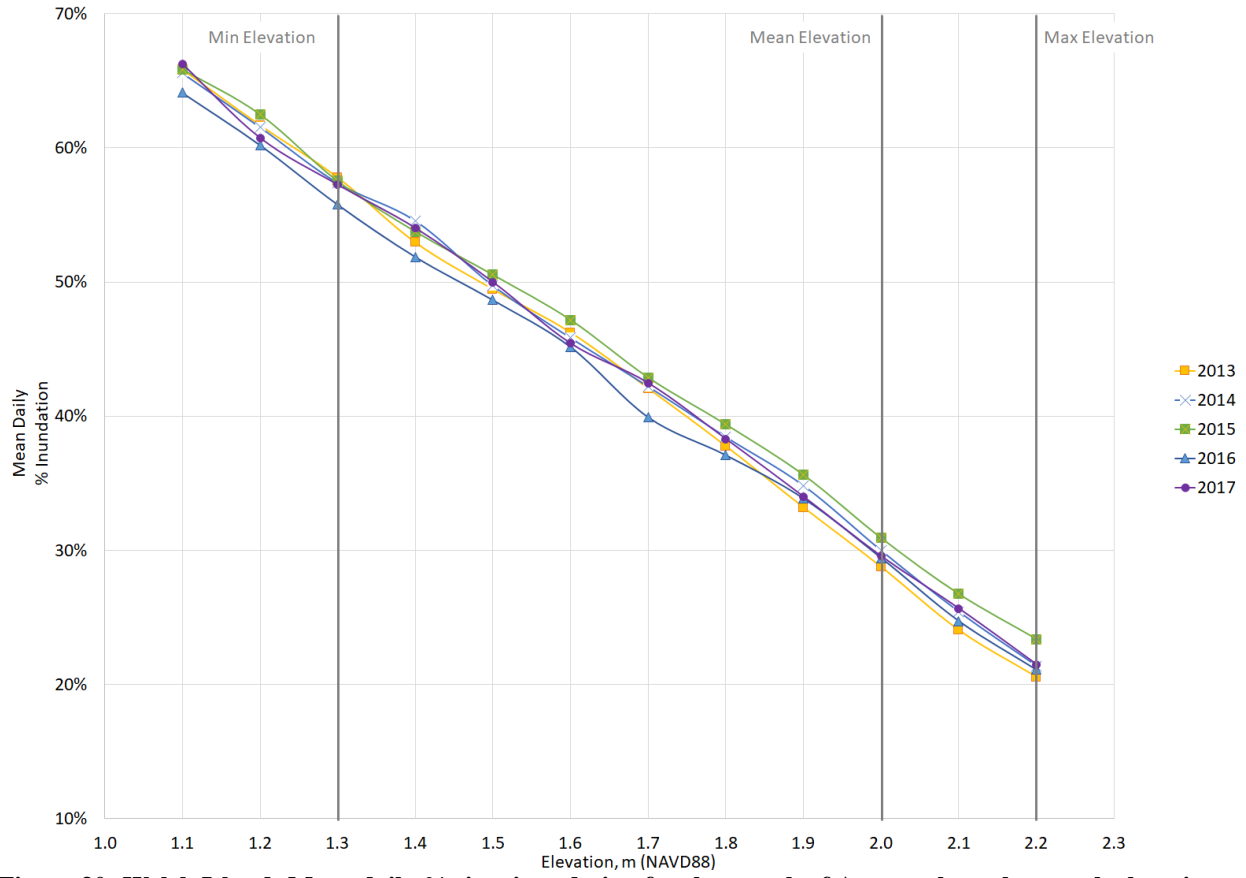
Figure 30: Welch Island: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2013-2017.
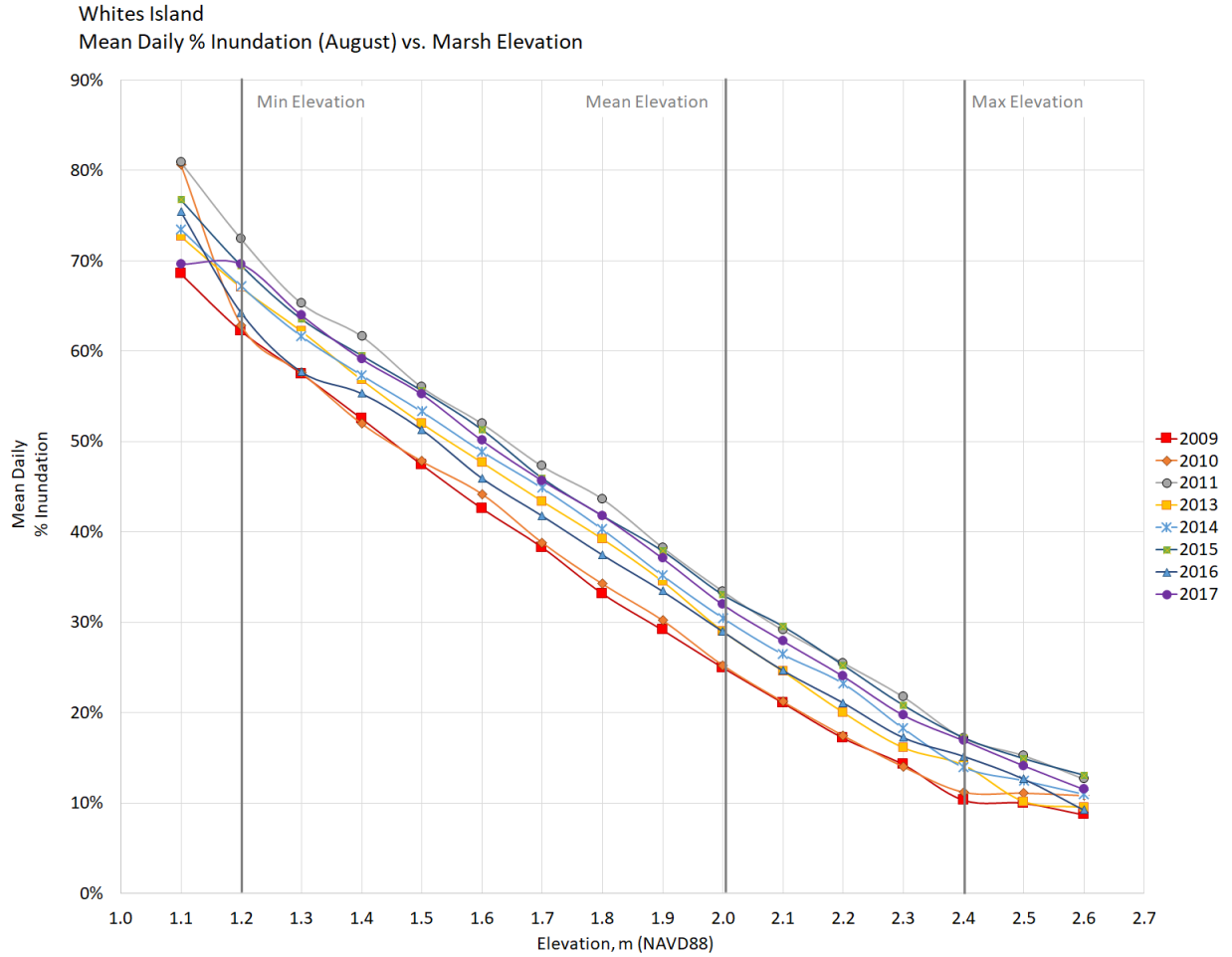
Figure 31: Whites Island: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2009-2017.

In 2017, inundation levels (mean % time flooded daily) at Cunningham Lake, mean marsh elevation of 2.7 meters (NAVD88), 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% (Figure 28, Figure 32, Figure 33). Consistently, for all years monitored, Cunningham Slough experiences greater levels of flooding than Campbell Slough (Figure 28, **Table 17**). 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 2).
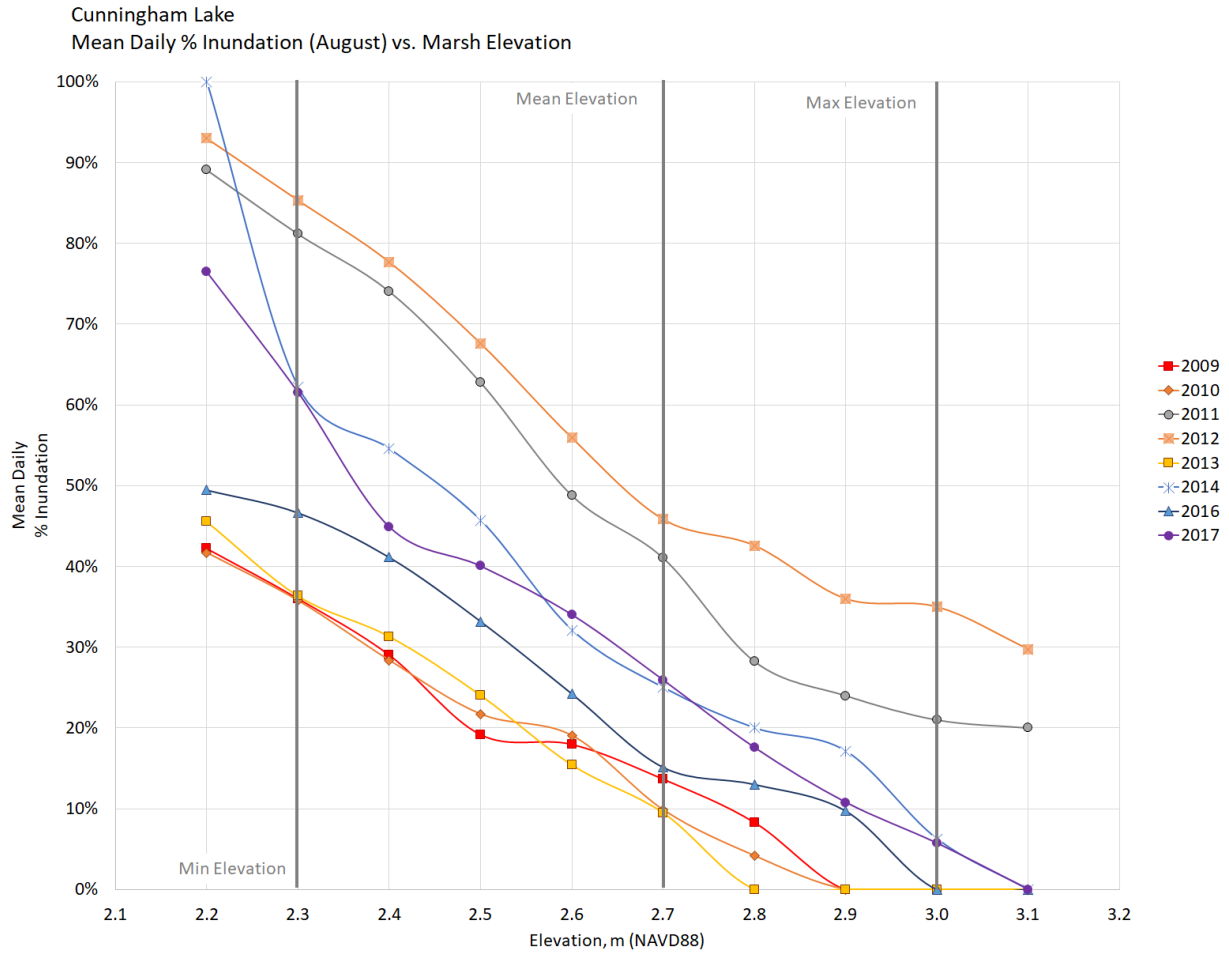
Figure 32: Cunningham Lake: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2009-2017.
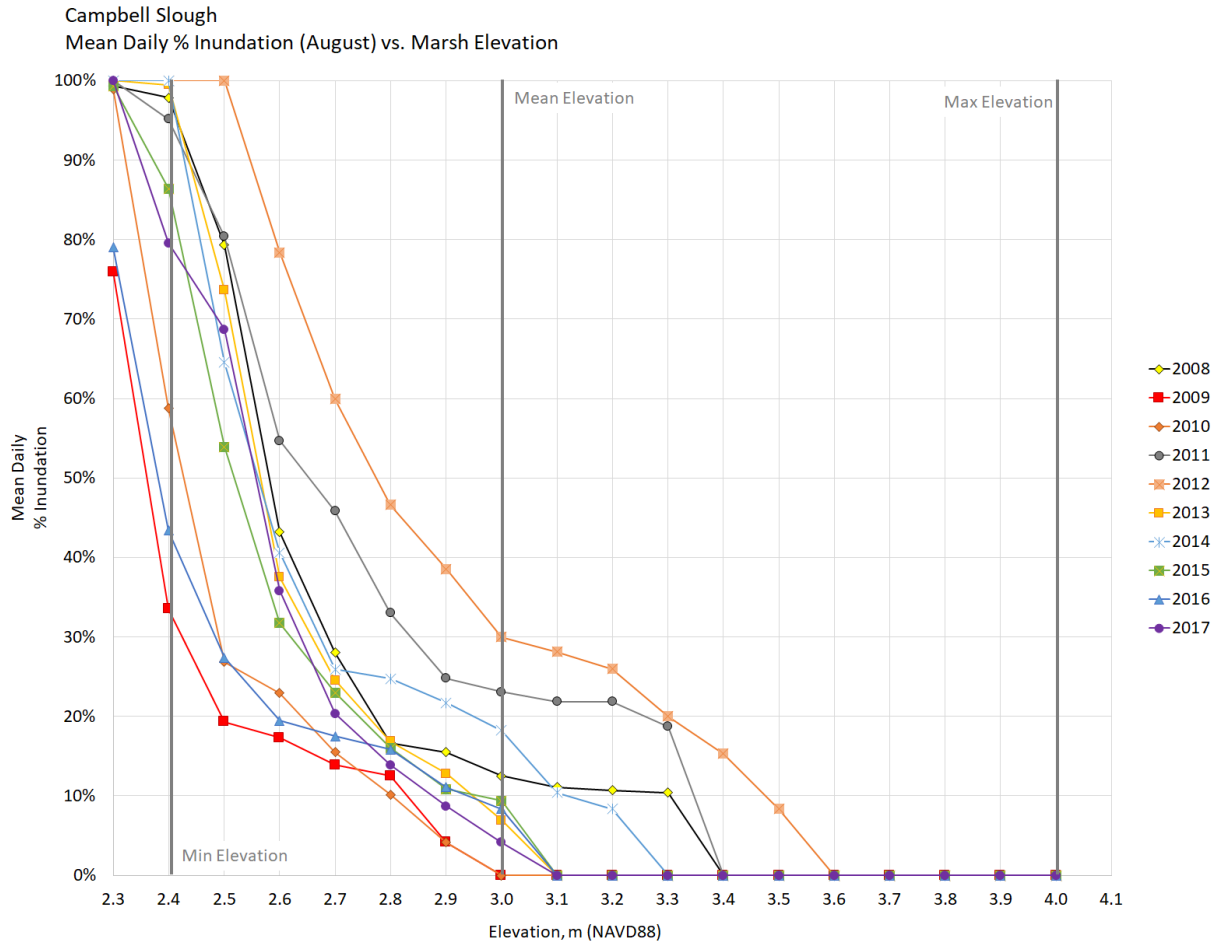
Figure 33: Campbell Slough: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2008-2017.

In 2017, the upper river site, Franz Lake, the mean marsh elevation (4.6 m, NAVD88) was not flooded during the month of August similar to 2008, 2013, and 2016. 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 24).
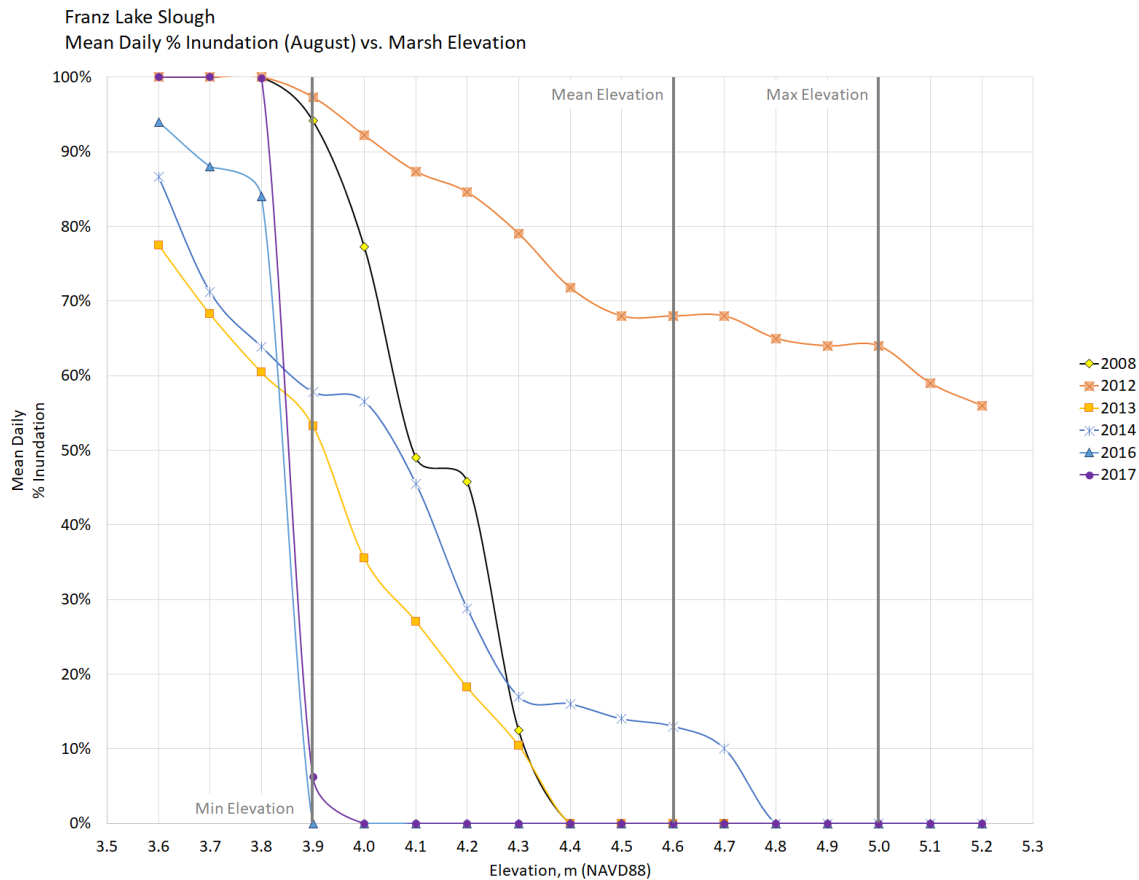
Figure 34: Franz Lake: Mean daily % time inundation for the month of August along the marsh elevation gradient between 2008, 2012-2017.

3.3.2 Sediment Accretion Rates

Sediment accretion data collected in 2017 are still under analysis and unavailable at the time this report was written; these data are required to process the 2018 data. The following details are adapted from the 2016 report.

The sediment accretion measured in 2016 generally followed the trends observed in previous years ranging from 0.0 cm to 2.9 cm per year (Table 19). A notable exception was erosion that was measured at Franz Lake at the original stakes and at a new set closer to the mouth. New stakes were deployed at several sites to measure accretion at additional elevations within site. The data from the new stakes support hypotheses that increased accretion occurs at lower elevations (Cunningham Lake and Campbell Slough), and with proximity to channels (Welch Island).

Average sediment accretion at the trends sites ranges from 0.3 cm to 2.2 cm per year (Table 19) The site with the least variability over time is Welch Island (0.7±0.1 cm), while the greatest variation between years was observed at the Campbell Slough and Franz Lake sites. The Whites Island mid-elevation stake measurements have the highest average accretion of 2.2 cm per year (Table 19). Sediment accretion rates at the trends sites between 2008 and 2015. WHC-M and WHC-H represent mid-elevation and high-elevation marsh locations, respectively.

Table 19. Sediment accretion rates at the trends sites between 2008 and 2015. WHC-M and WHC-H represent mid-elevation and high-elevation marsh locations, respectively.

Site Code:	BBM	BBM-2	WI2	WI2-2	WHC-M	WHC-H	CLM	CLM-2	CS1	CS1-2	FLM	FLM-2
Elevation (m, CRD):	1.81	1.76	1.66	1.69	1.35	1.88	1.54	1.26	1.49	1.91	1.87	2.13
Year						Annual Rate (cm)						
08-09	ND[1]	ND	ND	ND	ND	-1.2	ND	ND	ND	ND	0.5	ND
09-10	ND	ND	ND	ND	ND	1.0	1.9	ND	0.4	ND	ND	ND
10-11	1.7	ND	ND	ND	ND	0.1	1.6	ND	1.7	ND	3.0	ND
11-12	0.1	ND	ND	ND	ND	0.9	1.4	ND	0.9	ND	-0.4	ND
12-13	0.6	ND	0.8	ND	1.2	0.2	1.3	ND	0.2	ND	3.0	ND
13-14	0.3	ND	0.6	ND	2.3	0.8	0.5	ND	1.5	ND	0.7	ND
14-15	1.0	ND	0.7	ND	2.7	0.0	-0.5	ND	-2.4	ND	1.2	ND
15-16	0.0	0.3	ND	0.97	2.6	ND	0.9	2.9	1.4	0.8	-0.6	-2.3
16-17												
17-18												
Average	0.6		0.7		2.2	0.3	1.0		0.5		1.0	
Std Dev	0.6		0.1		0.7	0.8	0.8		1.4		1.5	

ND No data.

3.3.3 Vegetation Species Assemblage

3.3.3.1 *2018 Vegetation Patterns Across Estuary Zones*

Trends in species richness and diversity
In 2018, the highest overall total species richness was found at Whites Island (51 species, Table 20), though the highest average richness per plot was at Welch Island (10.3 species, Table 20). Both sites occur in wetland zone 2, in the lower river. As in previous years, the lowest species richness was at the brackish Ilwaco Slough (19 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, 32 respectively), though Cunningham Lake had substantially higher average richness per plot (5.7) than Campbell Slough (2.9). Franz Lake, wetland zone 5, the upper river site, was intermediate in both total richness (27) and average plot richness (6.9). 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 2018 can be found in Table 20 and the long-term total species richness data can be found in Table 21 and Table 22.

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 36). 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 35- Figure 37). 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 (2016-2018) 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 22). 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 22). Indicating, while species compositions may be shifting slightly year to year the general distribution of species dominance have not changed substantially across the sites. Welch and Whites Islands having the greatest evenness of 0.7, Ilwaco Slough and Cunningham Lake 0.6, and Campbell Slough and Franz Lake having the lowest at 0.4 (Table 22).

Table 20. 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 21-Table 23.

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**	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 21. 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.

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

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Table 22. 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 21).

Site		Long-term Average	Year 18	17	16	15	14	13	12	11	10	09	08	07	06	05
Ilwaco Slough	TSR	4.7 (0.7)	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	3.9 (0.6)	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	0.3 (0.4)	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)	1.1 (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	0.5 (0.1)	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	11.7 (1)	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)	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	1.8 (0.5)	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)	1.8 (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)	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	7.0 (1.0)	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	4.5 (0.7)	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	2.2 (0.5)	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 (0.1)	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)	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	3.8 (1.3)	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 (1.5)	3.8 (1.5)	3 (1.2)	3.5 (2.1)
	NSR	2.8 (1.0)	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)	1.2 (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)	1.2 (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 (0.2)	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	3.3 (0.9)	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	2.2 (0.6)	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	0.9 (0.3)	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 (0.1)	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)	0.4 (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	5.3 (1.2)	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 (0.9)	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)	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)	0.9 (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)	0.5 (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)			
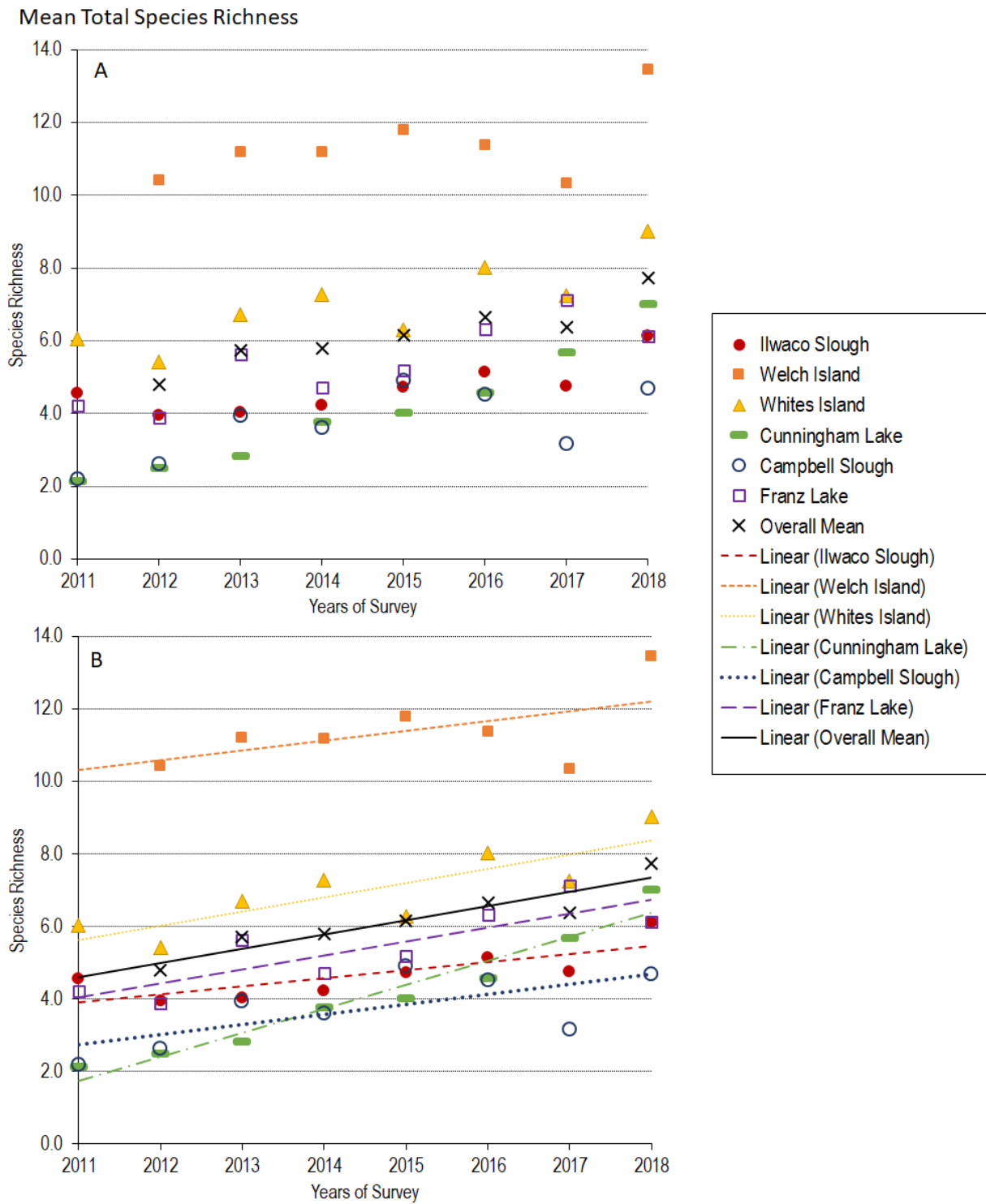
Figure 35. Changes in mean total species richness over time at each trend site. Only the years for which most current trend sites were sampled are shown (2011-2018).

Mean Non-native Species Richness
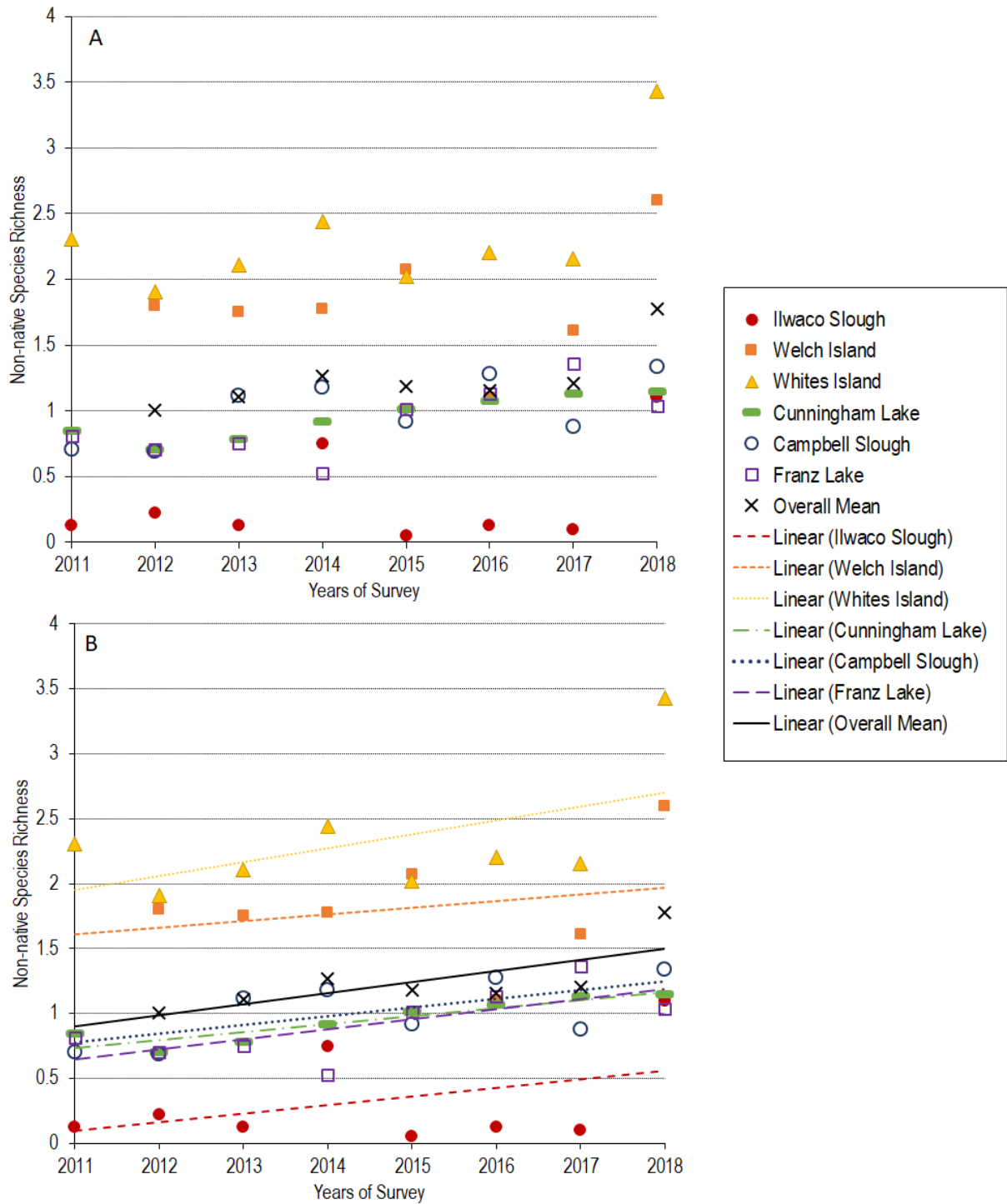
Figure 36. Changes in mean non-native species richness over time at each trend site. Only the years for which most current trend sites were sampled are shown (2011-2018).

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Mean Native Species Richness
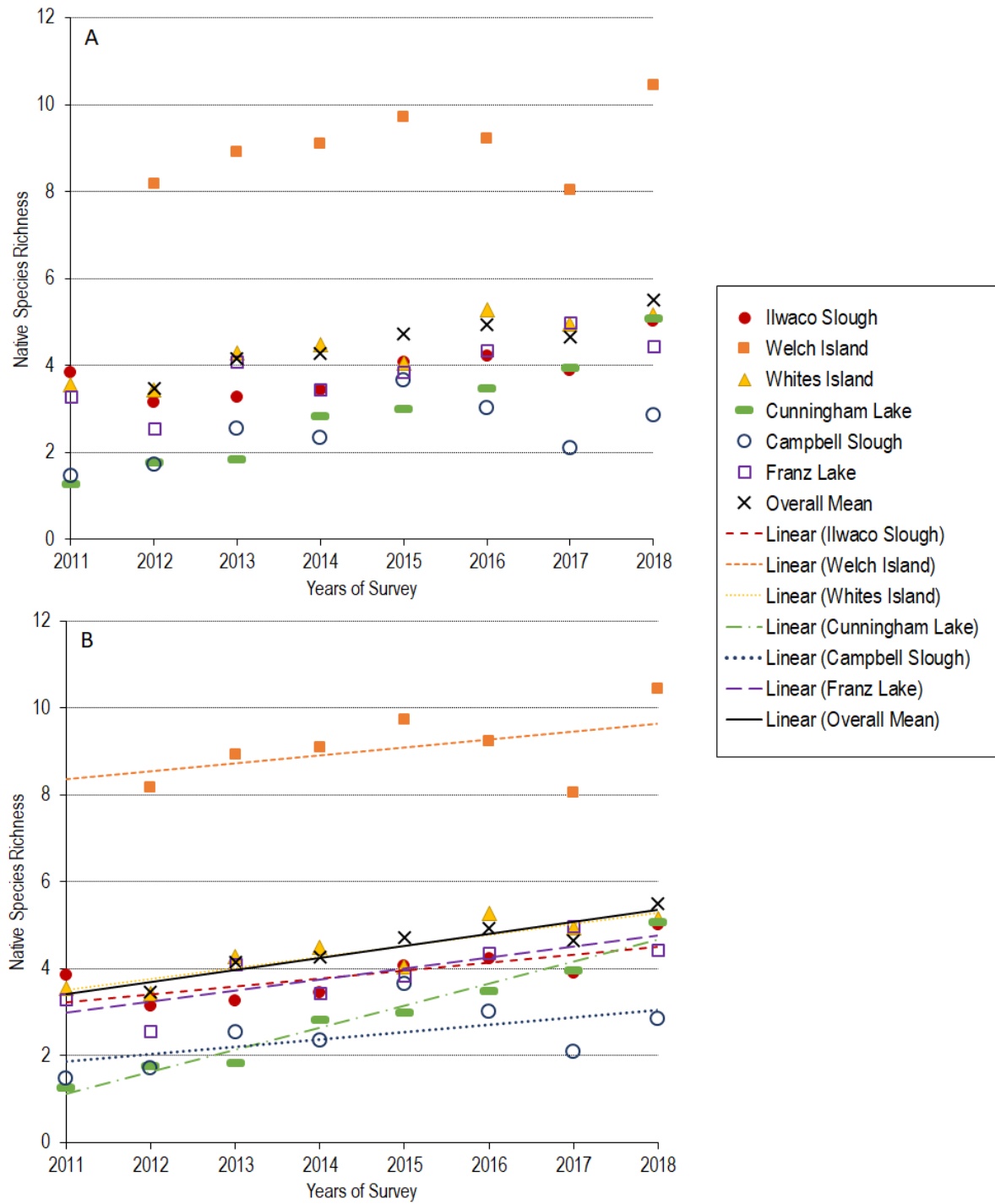
Figure 37. Changes in mean native species richness over time at each trend site. Only the years for which most current trend sites were sampled are shown (2011-2018).

Trends in plant cover
When comparing overall % total cover in 2018, Campbell Slough had substantially lower % vegetative cover (66%, similar to 2017) than all other sites (95-153%) (Table 23, Figure 38-Figure 39). 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 and 2018 from 67% to 113% which was also grazed in 2017 but not in 2018. Welch Island had the highest cover (153%), while the sites at the two ends of the tidal estuary, Ilwaco Slough (122%) and Franz Lake (95%) 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 21, Table 23).

Relative cover of native and non-native plant communities across the sites have followed a less linear trend than total cover overtime (Figure 40). 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 40 & Figure 41), 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 2018, mean relative native cover was greatest at Welch Island (76±19%, long-term average 82±3%), followed by Cunningham Lake (66±29%, long-term average 54±16%), Ilwaco Slough and Franz Lake (both 64%, long-term averages 81±9% & 69±14%), Whites Island (41±21%, long-term averages 40±6%), and Campbell Slough (39±32%, long-term averages 47±14%) (Table 23). Compared to long-term averages, the native cover was generally lower in 2018 across all sites except Cunningham Lake (66±29%, long-term average 54±16%) and Whites Island (41±21%, long-term averages 40±6%) (Table 23). Overtime, there has been a slight decrease in relative native cover at Ilwaco Slough and Welch Island, the native cover has seen both an overall increase and then recent slight decrease across time at Franz Lake, Cunningham Lake, and Campbell Slough (Table 23, Figure 40 & Figure 41). Relative native cover at Whites Island has remained relatively stable only fluctuating around 10% between 2012-2018.

Compared to the long-term site averages, non-native cover has increased in 2018 at Ilwaco Slough (long-term average 16±7%, 2018 average 29±25%), Welch Island (long-term average 15±3%, 2018 average 20±17%) and decreased at Cunningham Lake (long-term average 37±8%, 2018 average 20±26%), and Franz Lake (long-term average 19±13%, 2018 average 8±15%) (Table 23). Whites Island (long-term average 54±4%, 2018 average 52±28%) and Campbell Slough (long-term average 37±4%, 2018 average 37±44%), 2018 averages are similar to their long-term trends (Table 23). Shifts in relative native and non-native cover seen between 2017 and 2018 at Ilwaco Slough and Welch Island are likely partially due to the positive identification of a previously common grass species *Agrostis sp.* (no native or non-native status identified in past years) to the non-native *Agrostis stolonifera*.

Mean Total Living Cover (%)
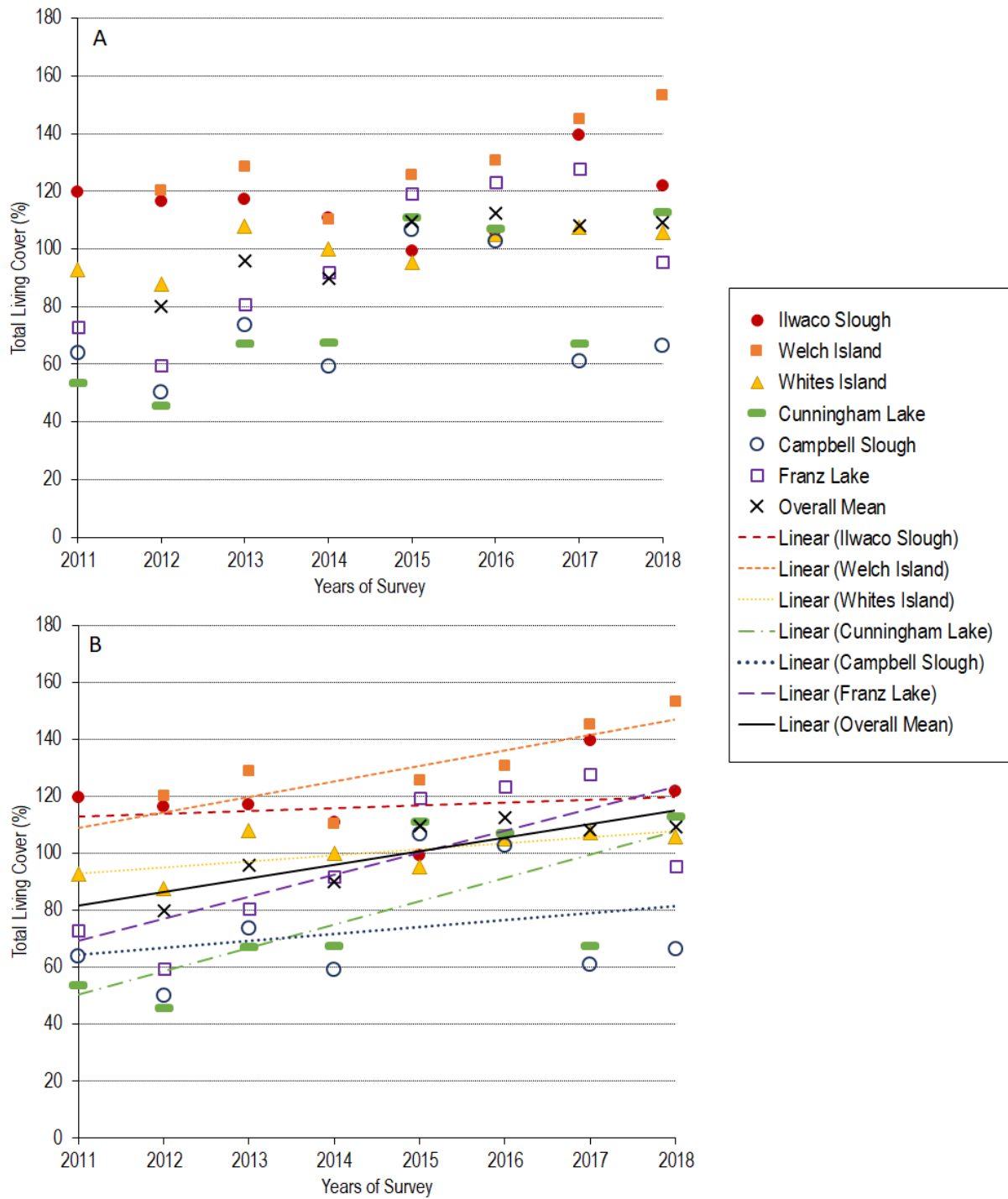
Figure 38. Trends in Total Living % Cover at each site since 2011, when most sites except Welch Island began being monitored. Only the years for which most current trend sites were sampled are shown (2011-2018).

Mean Total Living Cover 2011-2018
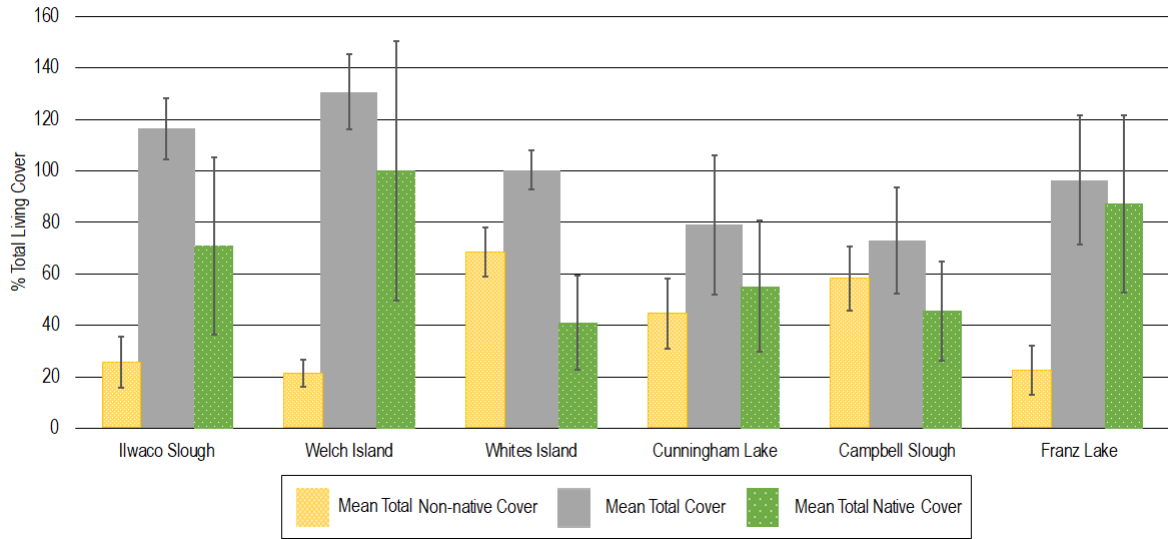
Figure 39. Average % total cover (not relative cover) of vegetation at each trend site, 2011-2018, which is the period during which all six current trend sites were being monitored. Error bars are standard deviations.

Mean Native Relative Cover (%)
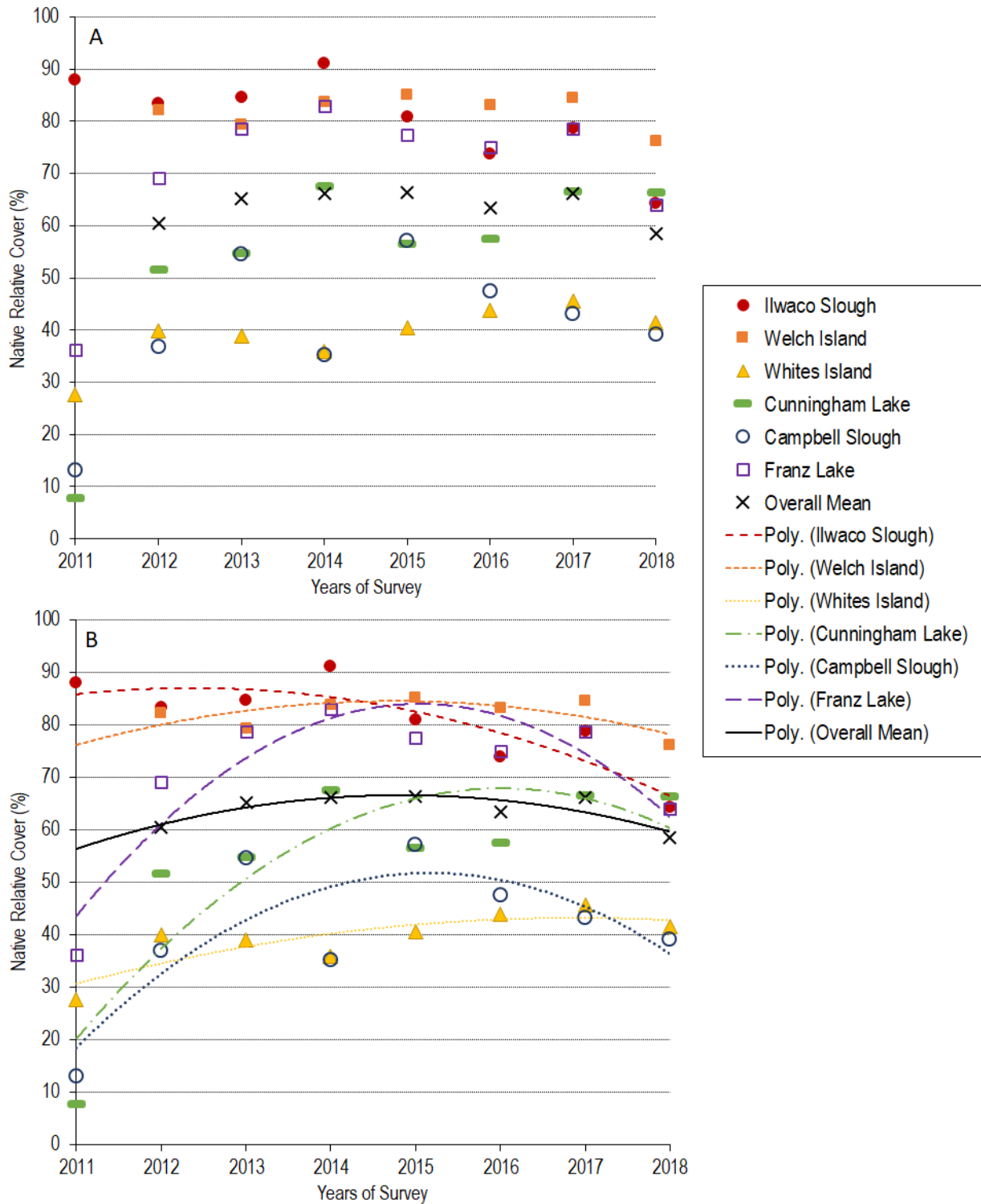
Figure 40. Trends in relative native cover (%) at each site since 2011, when most sites except Welch Island began being monitored. Only the years for which most current trend sites were sampled are shown (2011-2018).
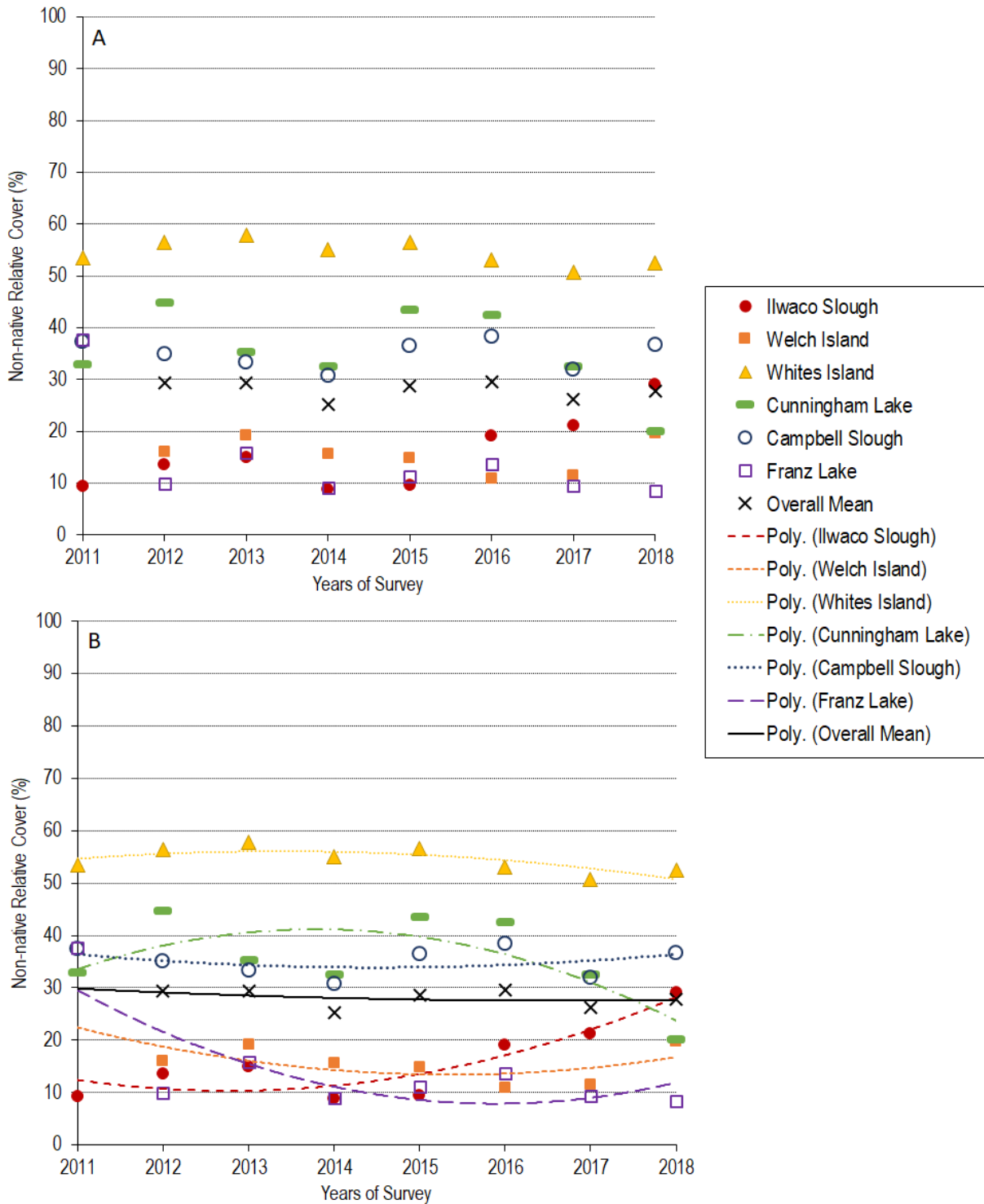
Figure 41: Trends in relative non-native cover (%) at each site since 2011, when most sites except Welch Island began being monitored. Only the years for which most current trend sites were sampled are shown (2011-2018).

Table 23. 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.

Site		Long-term Average	Year 18	17	16	15	14	13	12	11	10	09	08	07	06	05
Ilwaco Slough	Tot	116(12)	122 (34)	140 (54)	106 (46)	99 (46)	111 (42)	117 (41)	116 (34)	120 (27)						
	BG	8 (8)	6 (14)	3 (6)	21 (25)	20 (27)	3 (8)	6 (15)	5 (9)	2 (5)						
	Nat	81 (9)	64 (25)	79 (21)	74 (26)	81 (26)	91 (15)	85 (21)	83 (19)	88 (13)						
	Non	16 (7)	29 (25)	21 (21)	19 (21)	10 (14)	9 (15)	15 (21)	14 (16)	9 (10)						
	OBL	79 (5)	70 (27)	78 (21)	76 (25)	83 (25)	86 (20)	79 (26)	76 (29)	81 (22)						
Welch Island	Tot	131(15)	153 (31)	145 (24)	131 (25)	126 (22)	110 (29)	129 (23)	120 (23)							
	BG	3 (2)	0 (1)	1 (3)	6 (7)	3 (3)	6 (10)	3 (5)	4 (8)							
	Nat	82 (3)	76 (19)	84 (20)	83 (22)	85 (23)	84 (24)	79 (26)	82 (20)							
	Non	15 (3)	20 (17)	11 (20)	11 (22)	15 (23)	16 (24)	19 (26)	16 (20)							
	OBL	75 (3)	71 (19)	77 (23)	76 (22)	74 (24)	73 (23)	74 (25)	78 (19)							
Whites Island	Tot	102 (8)	106 (33)	107 (31)	105 (30)	95 (35)	100 (25)	108 (41)	88 (30)	93 (31)	110 (29)	107 (40)				
	BG	8 (4)	3 (10)	3 (6)	8 (14)	5 (11)	9 (17)	8 (19)	16 (23)	14 (24)	9 (18)	5 (13)				
	Nat	40 (6)	41 (21)	46 (28)	44 (32)	40 (32)	36 (25)	39 (33)	40 (29)	28 (28)	39 (29)	47 (33)				
	Non	54 (4)	52 (28)	51 (32)	53 (33)	57 (36)	55 (30)	58 (36)	56 (31)	53 (26)	61 (29)	46 (32)				
	OBL	43 (4)	52 (28)	43 (31)	44 (31)	43 (34)	39 (29)	41 (33)	48 (30)	38 (30)	44 (31)	44 (34)				
Cunningham Lake	Tot	83 (24)	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)	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 (16)	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	37 (8)	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 (9)	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	75 (20)	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 (18)	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	47 (14)	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)	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 (5)	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	98 (22)	95 (62)	128 (55)	123 (48)	119 (53)	92 (47)	81 (44)	60 (35)	73 (35)		106 (51)	105 (54)			
	BG	18 (11)	8 (12)	13 (20)	14 (17)	10 (20)	24 (22)	32 (23)	36 (29)	25 (25)		17 (27)	0 (0)			
	Nat	69 (14)	64 (38)	79 (25)	75 (24)	77 (26)	83 (23)	79 (28)	69 (34)	36 (25)		66 (34)	61 (32)			
	Non	19 (13)	8 (15)	9 (13)	14 (20)	11 (16)	9 (18)	16 (24)	10 (14)	38 (32)		34 (33)	39 (32)			
	OBL	56 (10)	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 2012-2018 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 24, Figure 42 & Figure 43). 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 25 & Table 26, Figure 42). In 2018, *P. arundinacea* cover levels stayed relatively consistent to those observed in 2017, however, at Whites Island and Cunningham, there was a significant drop in *P. arundinacea* levels from 42 to 30% and 25 to 17%, respectively (Table 25 & Table 26, Figure 42). This drop in cover was not translated into a drop in *P. arundinacea* frequency (spread across the site) which increased across all sites (Table 25). *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 26, Figure 42). In 2018, *C. lyngbeyi* levels declined across the sites compared to 2017 levels, and all sites remained below long-term average cover conditions. Ilwaco Slough *C. lyngbeyi* cover declined by 8%, Welch Island declined by 14%, and Whites Island declined by 6% (Table 26, Figure 42).

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 26, Figure 42). In 2018, *E. palustris* levels were similar to 2017 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 2017 levels at Franz Lake shifting from 4 to 2% cover (long-term average 6±3%) (Table 26, Figure 42). At Campbell Slough and Cunningham Lake *E. palustris* levels were up compared to 2017 cover level and came in similar to long-term average cover levels in 2018, with 23% cover at Campbell Slough (long-term average 24±7%) and 18% cover at Cunningham Lake (long-term average 19±12%) (Table 26, Figure 42).

S. latifolia follows a similar trend to *E. palustris*. However it is not found growing in Ilwaco Slough. In 2018, *S. latifolia* levels were similar to 2017 at Welch Island (4%, 2017 3±6%) and Franz Lake (8%, 2017 6±17%). However *S. latifolia* levels were higher than 2017 at Whites Island (12%, 2017 9±13%) (Table 26, Figure 43). Comparatively, *S. latifolia* levels were much lower at Campbell Slough (6%, 2017 15±23%) and Cunningham Lake (11%, 2017 20±19%) compared to 2017 levels but falling within the long-term average range at Cunningham Lake (11±7%) (Table 26, Figure 43). *L. palustri* is a common and consistent species found across all sites (minus Ilwaco Slough), typically averaging less than 5% in overall relative cover, however, in 2018 its cover has increased to 16% at Cunningham Lake (Table 26). Similarly, *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, however, in 2018 its cover has increased from 0.2 to 3% at Cunningham Lake (Table 26, Figure 43).

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 26, Figure 42). In 2018, *P. amphibium* levels

(34%) slightly below 2017 levels (38%) but higher than the long-term average (27±10%) (Table 26, Figure 43).

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 24. The overall long term 6 most commonly occurring plant species across all six trend sites from 2012-2018. 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	6.9	41.2	28.7	26.9	8.9
Carex lyngbyei lyngby sedge	OBL	sedge	yes	46.3	40.1	4.3	0.0	0.0	0.0
Eleocharis palustris common spikerush	OBL	sedge	yes	0.0	4.3	9.5	13.4	11.1	3.9
Sagittaria latifolia wapato	OBL	herb	yes	0.0	3.7	1.7	9.9	22.2	4.8
Leersia oryzoides rice cut grass	OBL	Grass	yes	0.0	0.9	1.4	1.9	0.3	2.2
Ludwigia palustris water purslane	OBL	herb	yes	0.0	0.1	0.04	4.6	1.6	0.4

Table 25. *Phalaris arundinacea* (reed canarygrass) average relative % cover and frequency (% of sample plots) at the trend sites between 2005 and 2018. 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

*A different sampling design was used at Cunningham Lake in 2014, so results are not directly comparable to the other years.
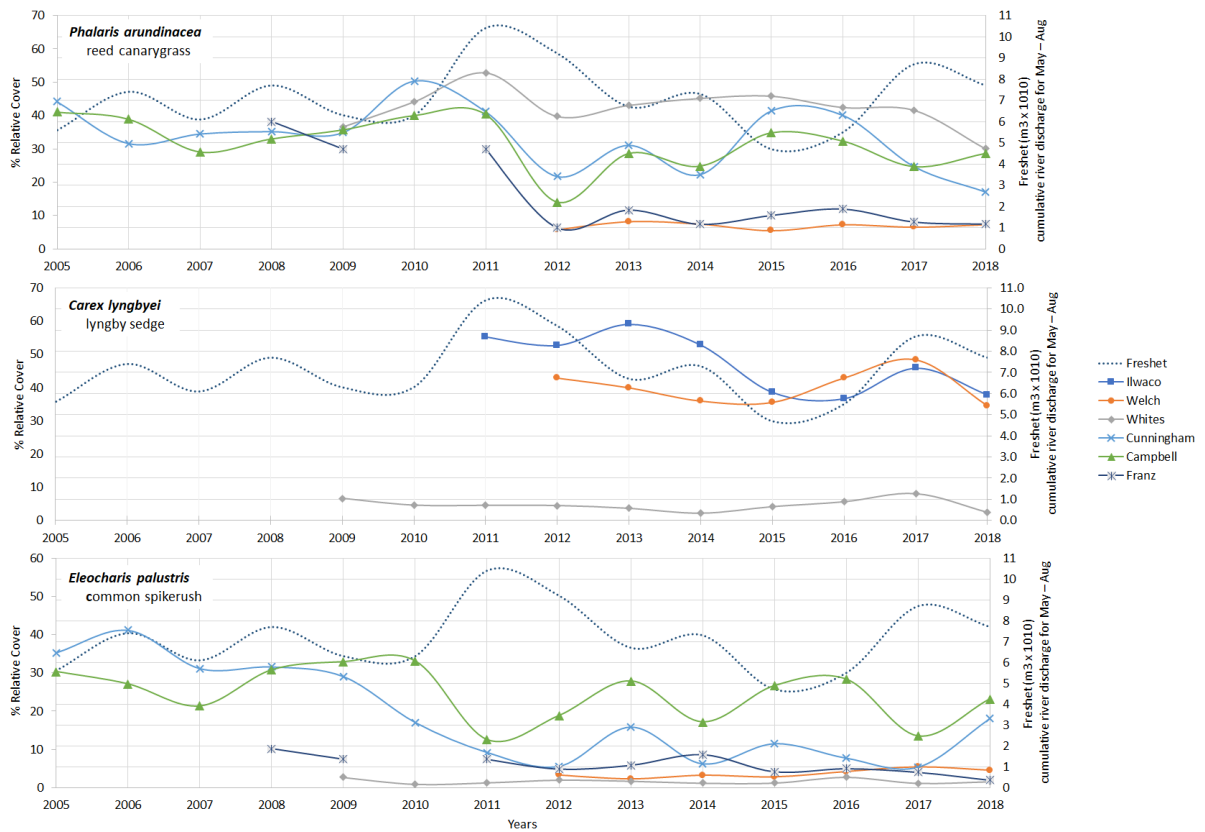
Figure 42: Annual mean relative % cover for *Phalaris arundinacea* (PHAR, non-native), reed canarygrass, *Carex lyngbyei* (CALY, native), lyngby sedge for all trend sites. Annual cumulative river discharge from May-Aug included for annual water year context (see Section 3.3.1). Relative species cover data can also be found in Table 26.
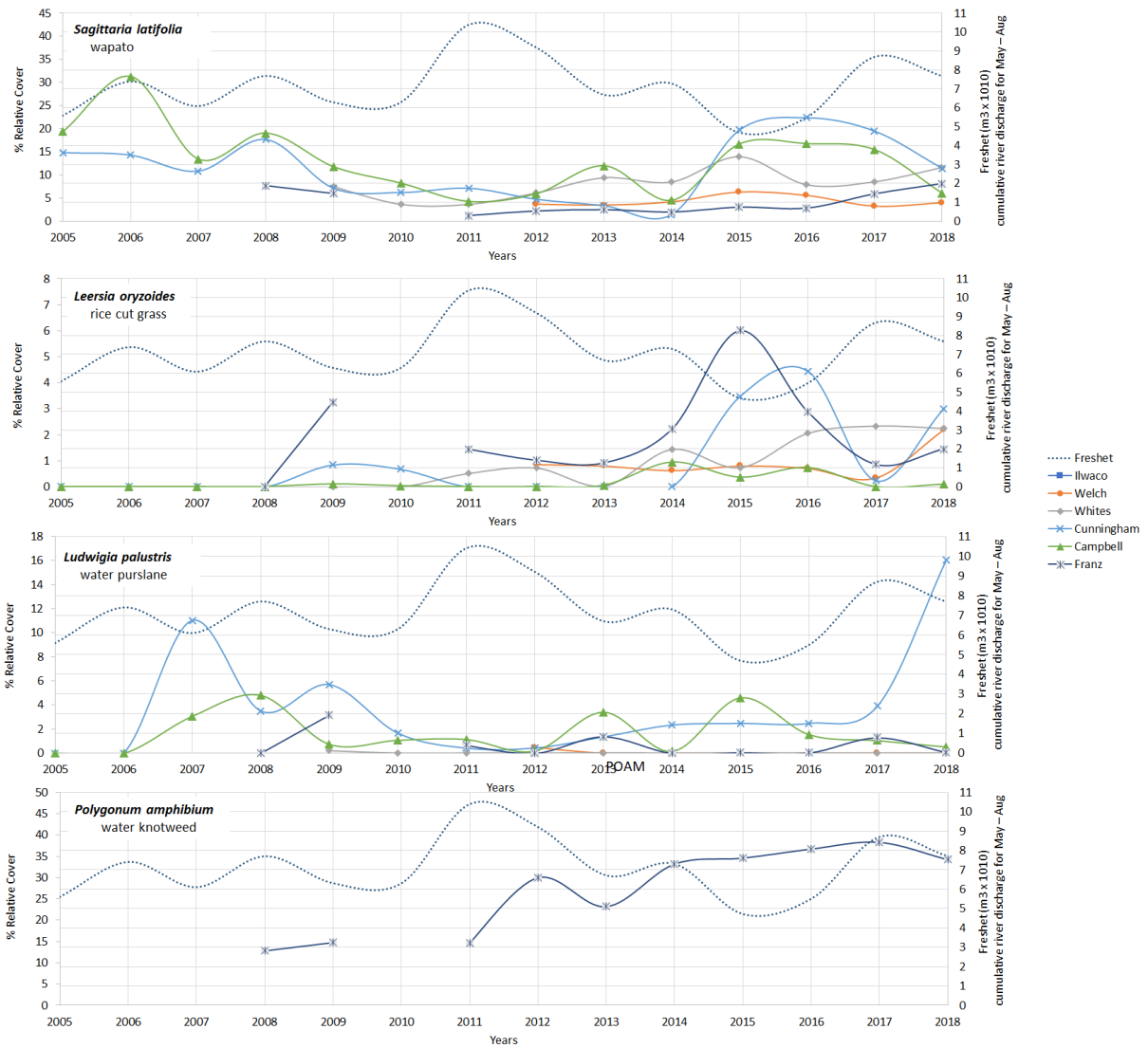
Figure 43: Annual mean relative % cover for *Eleocharis palustris* (ELPA, native) common spikerush, *Sagittaria latifolia* (SALA, native), wapato, *Leersia oryzoides* (LEOR, native), rice cut grass, *Ludwigia palustris* (LUPA, native), water purslane, and *Polygonum amphibium* (POAM, native), water knotweed, 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 26.

Table 26: 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	Long-term Average	18	17	16	15	14	13	12	11	10	09
Ilwaco Slough	CALY	47 (9)	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	40 (5)	34 (20)	48 (26)	43 (26)	36 (25)	36 (27)	40 (27)	43 (25)			
	ELPA	4 (1)	5 (12)	5 (12)	4 (10)	3 (11)	3 (7)	2 (8)	3 (8)			
	LEOR	0.9 (0.6)	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)	7 (18)	7 (18)	7 (21)	6 (16)	7 (21)	8 (24)	6 (19)			
	SALA	4 (1)	4 (5)	3 (6)	6 (7)	6 (8)	4 (6)	3 (5)	4 (6)			
Whites Island	CALY	5 (2)	2 (6)	8 (19)	6 (14)	4 (14)	2 (5)	4 (11)	4 (9)	5 (10)	5 (7)	7 (17)
	ELPA	2 (1)	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 (8)	2 (4)	0.7 (2)	1 (3)		0.7 (2)	0.5 (2)		
	LUPA	0 (0.1)										0.2 (1)
	PHAR	42 (6)	30 (25)	42 (30)	42 (30)	46 (33)	45 (32)	43 (32)	40 (32)	53 (31)	44 (32)	37 (30)
	SALA	8 (3)	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)	18 (18)	5 (7)	8 (12)	11 (17)	6 (12)	16 (24)	6 (8)	9 (19)	17 (21)	29 (27)
	LEOR	1 (2)	3 (4)	0.2 (2)	4 (9)	3 (8)					0.7 (2)	0.8 (3)
	LUPA	4 (5)	16 (19)	4 (7)	2 (5)	2 (5)	2 (5)	1 (3)	0.4 (1)	0.4 (3)	2 (4)	6 (7)
	PHAR	34 (10)	17 (27)	25 (27)	40 (36)	41 (37)	22 (30)	31 (40)	22 (23)	41 (36)	50 (42)	35 (39)
	SALA	11 (7)	11 (9)	20 (19)	22 (21)	20 (17)	1 (3)	3 (5)	5 (5)	7 (11)	6 (7)	7 (7)
Campbell Slough	ELPA	25 (7)	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)	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)	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)	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)	2 (7)	4 (11)	5 (12)	4 (11)	9 (19)	6 (12)	5 (12)	7 (18)		7 (13)
	LEOR	2 (2)	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	0.7 (1)	0.1 (1)	1.3 (2)				1.4 (3)		0.7 (2)		3.2 (6)
	PHAR	16 (12)	7 (14)	8 (12)	12 (18)	10 (16)	7 (16)	12 (19)	6 (10)	30 (28)		30 (33)
	POAM	27 (10)	34 (37)	38 (36)	37 (35)	35 (36)	33 (35)	23 (27)	30 (35)	15 (21)		15 (24)
	SALA	4 (3)	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*

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 (Figure 3). 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 (NAVD88) in which these species are found are different, increasing from the lower river sites to the upper river sites (Figure 44). By evaluating the mean % time these plant communities are inundated daily across all the sites (Figure 45), 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 45). 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 46). 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 45 and Figure 46).

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 ($R^2 = 0.73$), Campbell Slough ($R^2 = 0.79$), and Franz Lake ($R^2 = 0.58$) (Figure 47). 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 48). 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.
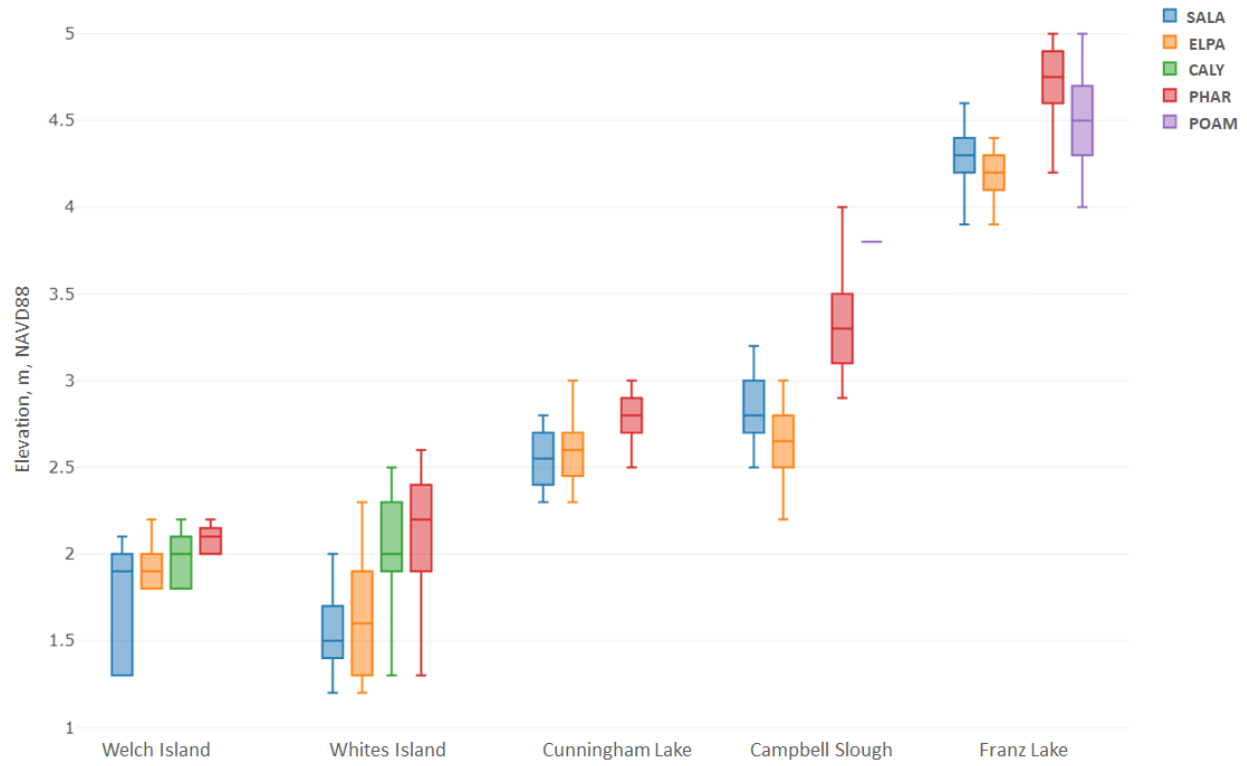
Figure 44: Box plot of elevation range of plant species across each site, sites in order from lower river to upper river from left to right. Species codes: *Carex lyngbyei* **(CALY, native), lyngby sedge,** *Eleocharis palustris* **(ELPA, native), common spikerush,** *Phalaris arundinacea* **(PHAR, non-native), reed canarygrass,** *Sagittaria latifolia* **(SALA, native), wapato, and** *Polygonum amphibium* **(POAM, native), water knotweed.**
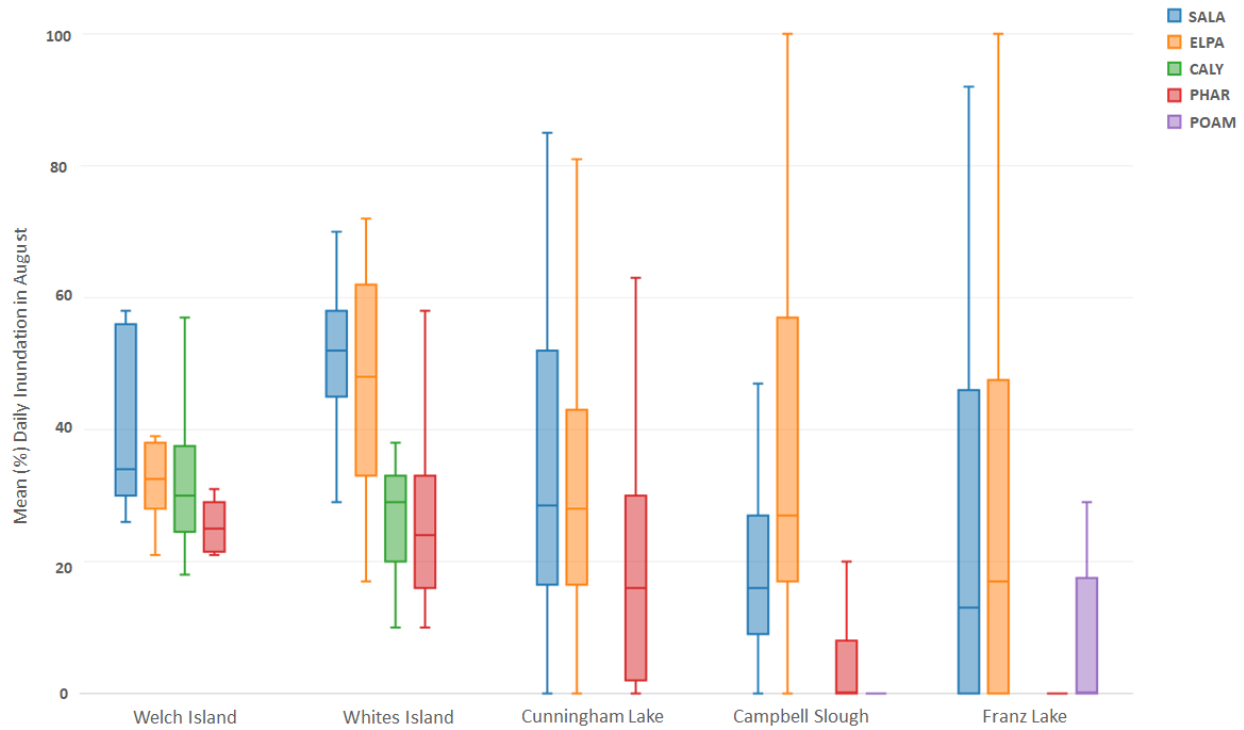
Figure 45: Box plot of daily mean inundation range (%, August - across all years) of plant species across each site, sites in order from lower river to upper river from left to right. Species codes: *Carex lyngbyei* **(CALY, native), lyngby sedge,** *Eleocharis palustris* **(ELPA, native), common spikerush,** *Phalaris arundinacea* **(PHAR, non-native), reed canarygrass,** *Sagittaria latifolia* **(SALA, native), wapato, and** *Polygonum amphibium* **(POAM, native), water knotweed.**
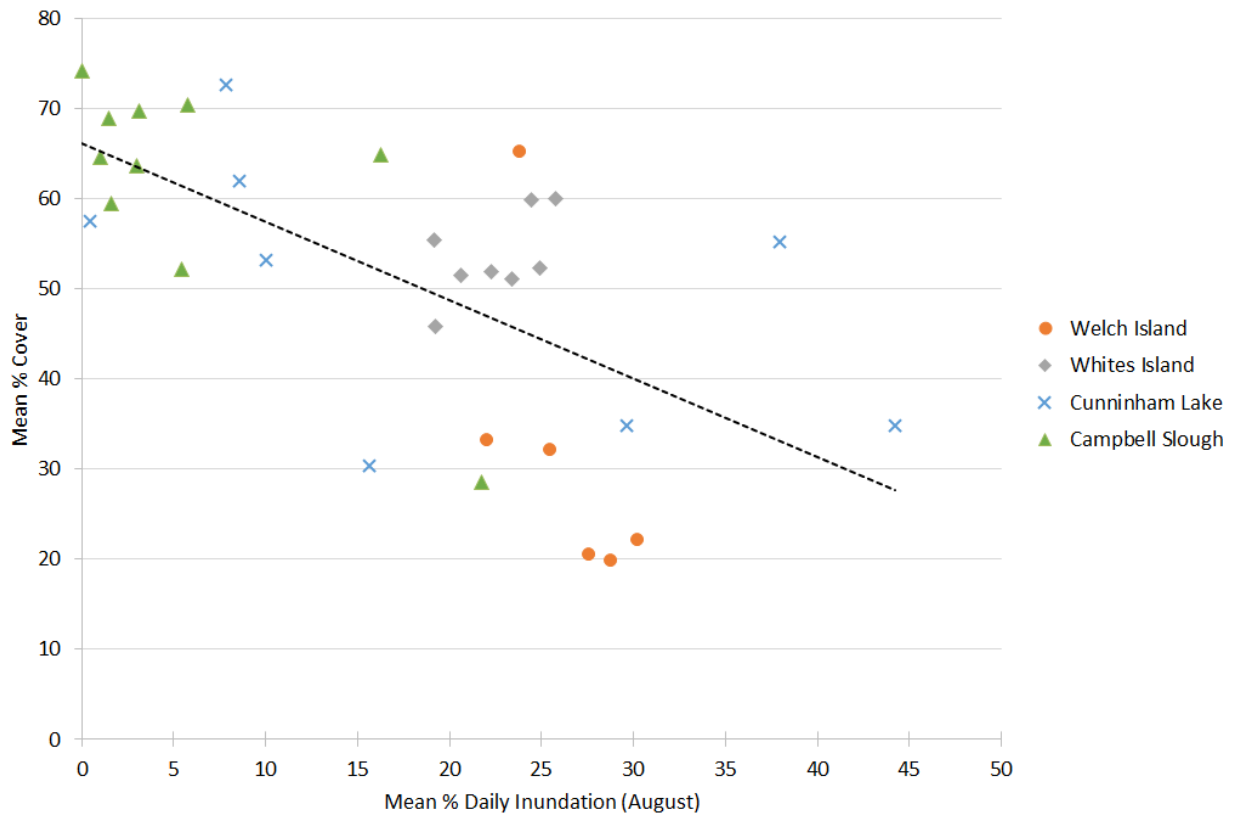
Figure 46: Mean Annual (%) *Phalaris arundinacea* **(PHAR, non-native), reed canarygrass, vs. daily (%) inundation (August –2012-2018) of PHAR plots. Linear regression (y = -0.8711x + 66.207), R² = 0.41, p <0.001.**

Annual Freshet Discharge vs. Mean Relative PHAR Cover (%)
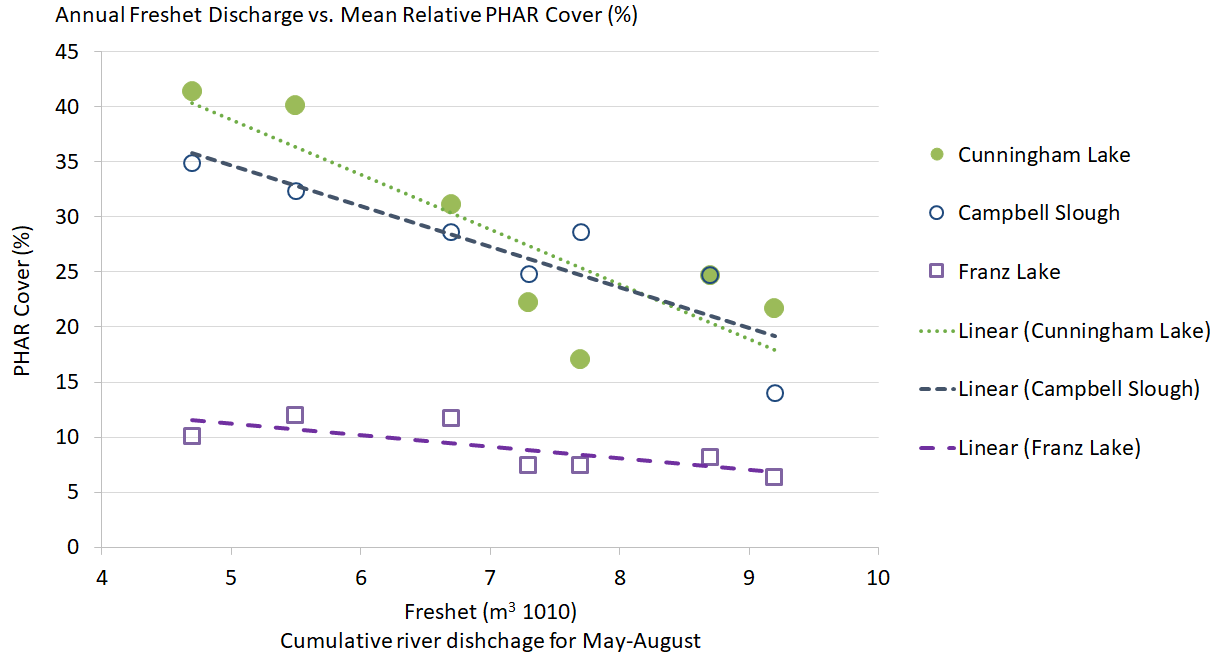
Figure 47: Annual Freshet Cumulative River Discharge (Bonneville Dam, May- August) vs. Mean Annual (%) *Phalaris arundinacea* **(PHAR, non-native), reed canarygrass, (2012-2018). Cunningham Lake, linear regression (y = -4.98x + 63.79), R² = 0.73, p <0.001; Campbell Slough linear regression (y = -3.69x + 53.09) R² = 0.79, p <0.001; Franz Lake, linear regression (y = -1.05x + 16.45), R² = 0.58, p<0.00.**
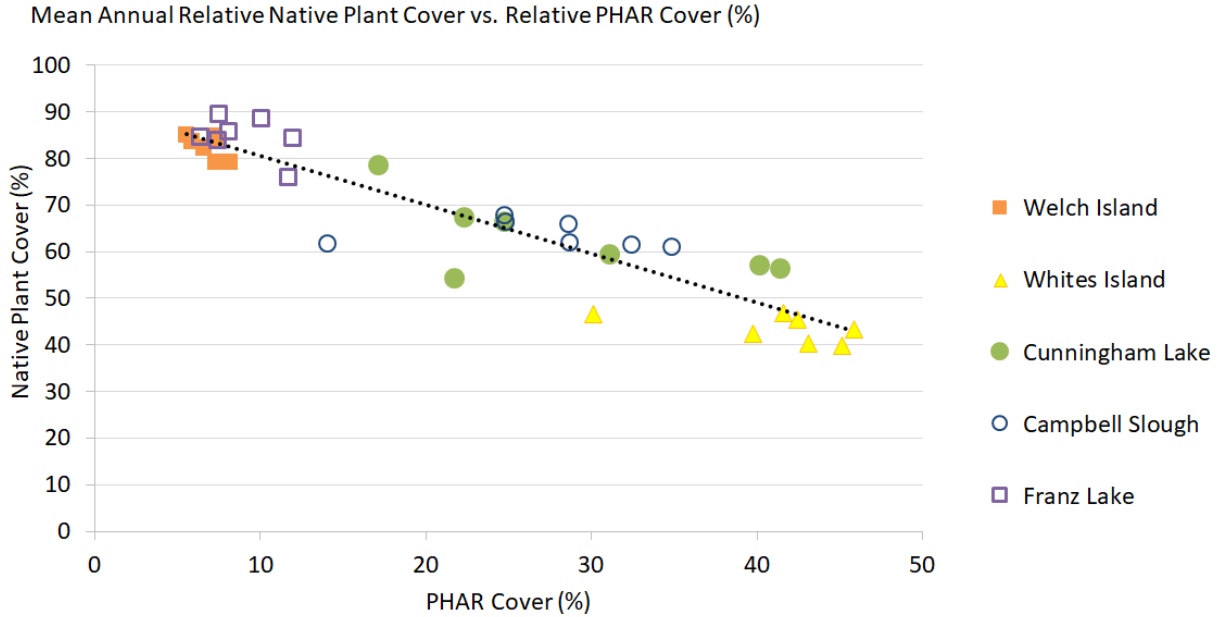
Figure 48: **Mean Annual Relative Native Plant Cover vs. Mean (%)** *Phalaris arundinacea* **(PHAR, non-native), reed canarygrass cover (August –2012-2018). Linear regression (y = -1.0497x + 91.114), R² = 0.87, p <0.001.**

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[7]. Channel measurements from previous years are presented with the newly calculated inundation frequency results from 2016 in Table 24. The tidal channels measured at the sites were generally small, with most cross-sectional areas less than 10 m² (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.

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[7] 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. The text below is adapted from the 2016 report.

Table 27. 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. The text below is adapted from the 2016 report.

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
				Physical Metrics			
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.

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 28. 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 28. 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.

	Species Codes	Welch Island (Rkm 53)				Steamboat Slough (Rkm 57) (AEMR Site, Restored 2016)				Whites Island (Rkm 72)				Franz Lake (Rkm 221)
		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%

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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 29, Table 30). 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 29, Figure 49). 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 30, Figure 50). 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 28).

Table 29. 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.

	Species Codes	Welch Island (Rkm 53)				Steamboat Slough (Rkm 57) (AEMR Site, Restored 2016)				Whites Island (Rkm 72)				Franz Lake (Rkm 221)
		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%
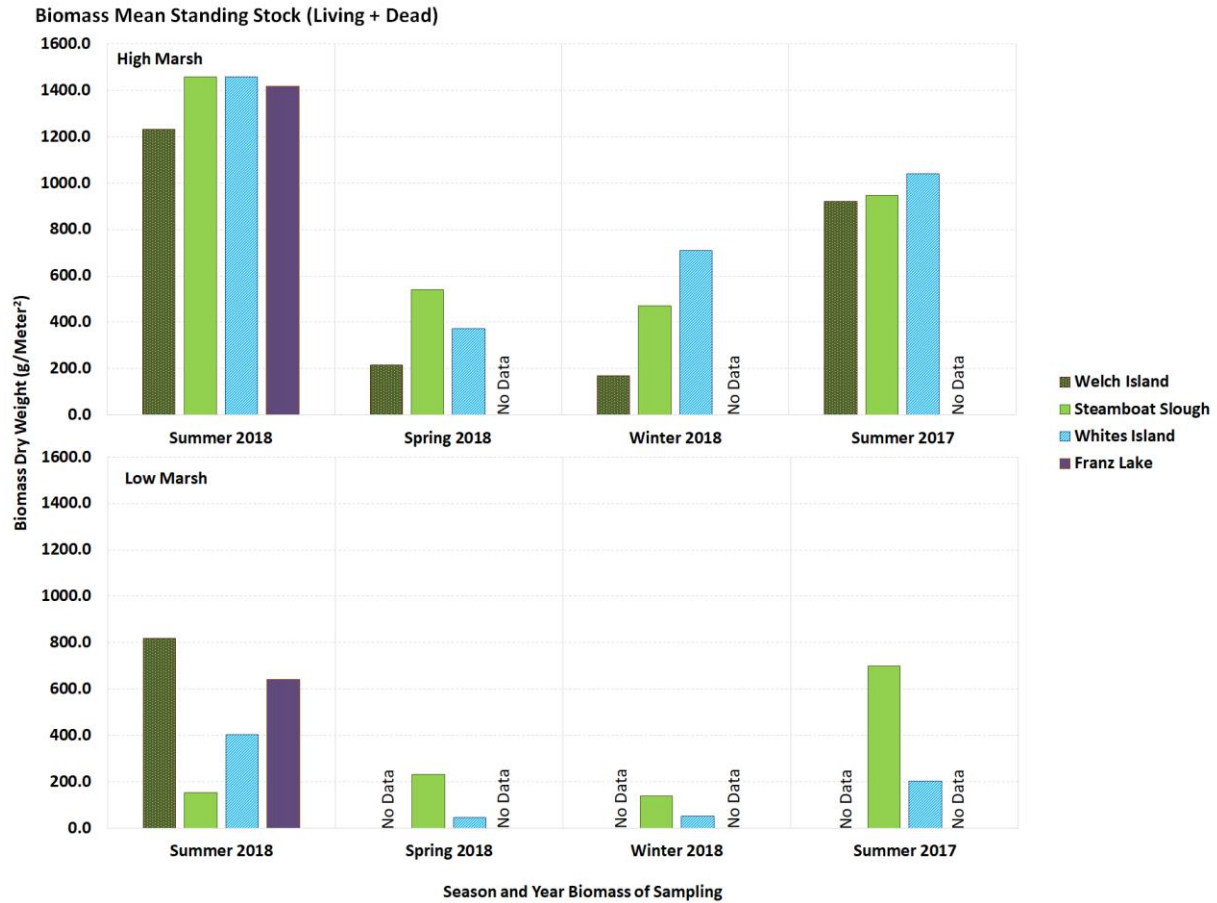
Figure 49: Average aboveground standing stock biomass (living + dead, dry weight g/m²) for summer 2017, winter 2018, spring 2018, and summer 2018 for both the high and low marsh strata across sites sampled. Sites shown in order of rkm from mouth of the Columbia River to the Bonneville dam. *Welch Island low marsh strata and Franz lake (both strata) were not sampled until summer 2018. See Table 30 for all values and standard deviations.
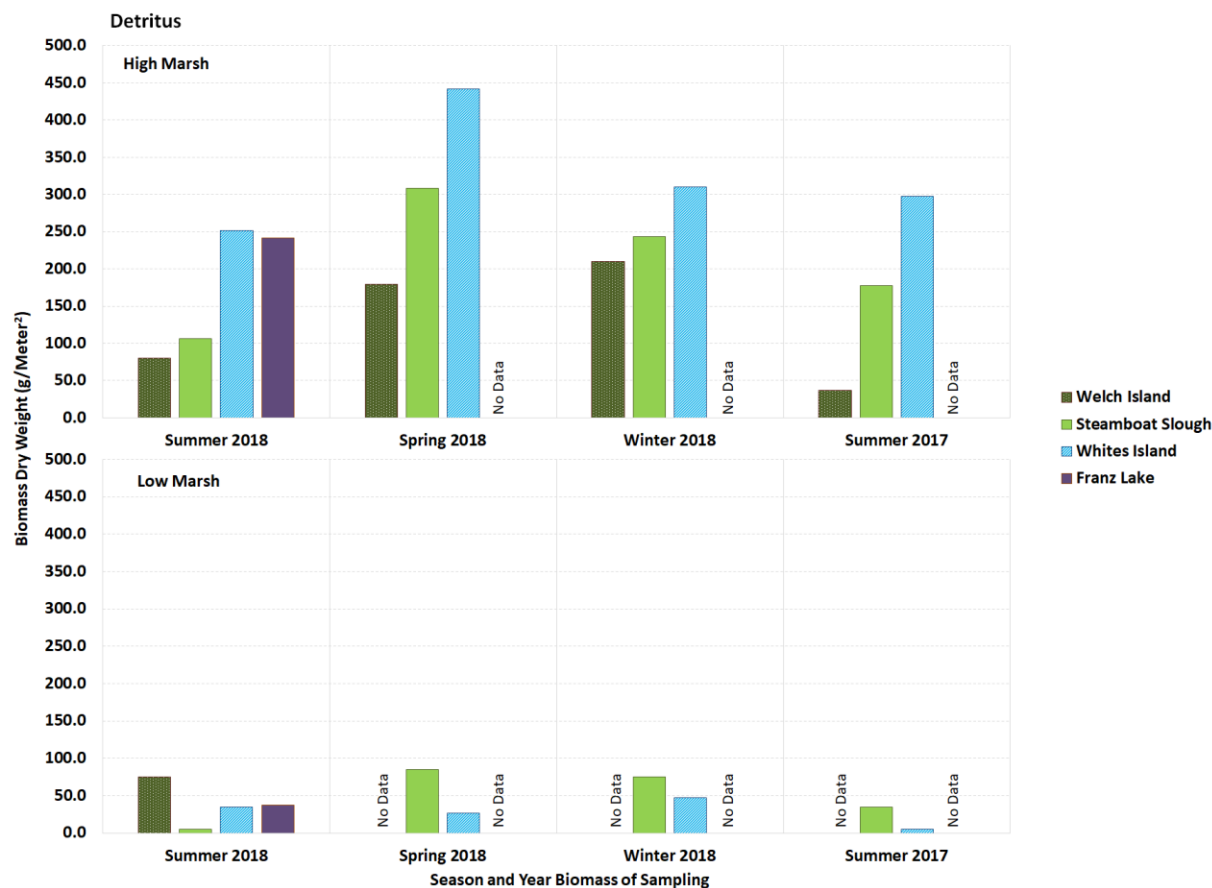
Figure 50: Average detrital biomass (dry weight g/m²) for summer 2017, winter 2018, spring 2018, and summer 2018 for both the high and low marsh strata across sites sampled. Sites shown in order of rkm from mouth of the Columbia River to the Bonneville dam. Welch Island low marsh strata and Franz lake (both strata) were not sampled until summer 2018. See Table 30 for all values and standard deviations.

Table 30: Average aboveground standing stock biomass (living + dead, dry weight g/m²) and average detrital biomass (dry weight g/m²) 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/m²			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 31). 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 31). 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 31).

Table 31: 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 31, Figure 51). 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 31).

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 32). 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 31, Table 32, Figure 51).

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/m2/year) compared to the high marsh (~1184-332 g/m2/year), which is much more variable (Table 31). 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 51 and Figure 52). The high marsh contribution is, however, much more variable from year to year (Figure 52).

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 52, Figure 53). 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 53).
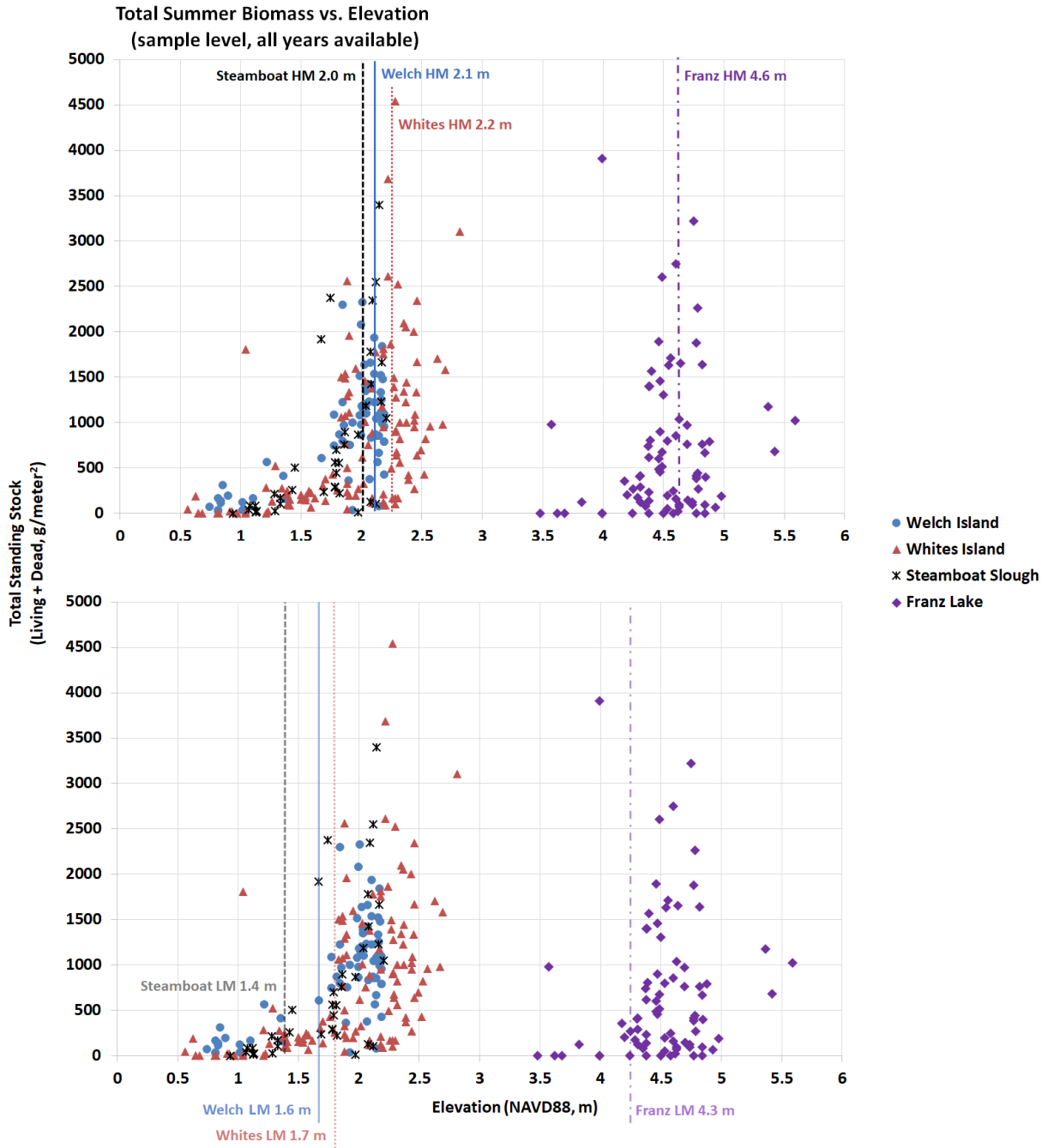
Figure 51: Total summer standing stock biomass (dry weight, g/m²) data by elevation for all years data was collected at each site. Mean high and mean low marsh sample elevations (averaged across all years) highlighted as vertical lines on graph. A shows HM elevations, B shows LM elevations. HM = High Marsh, LM = Low Marsh.

Table 32: 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 52.

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						
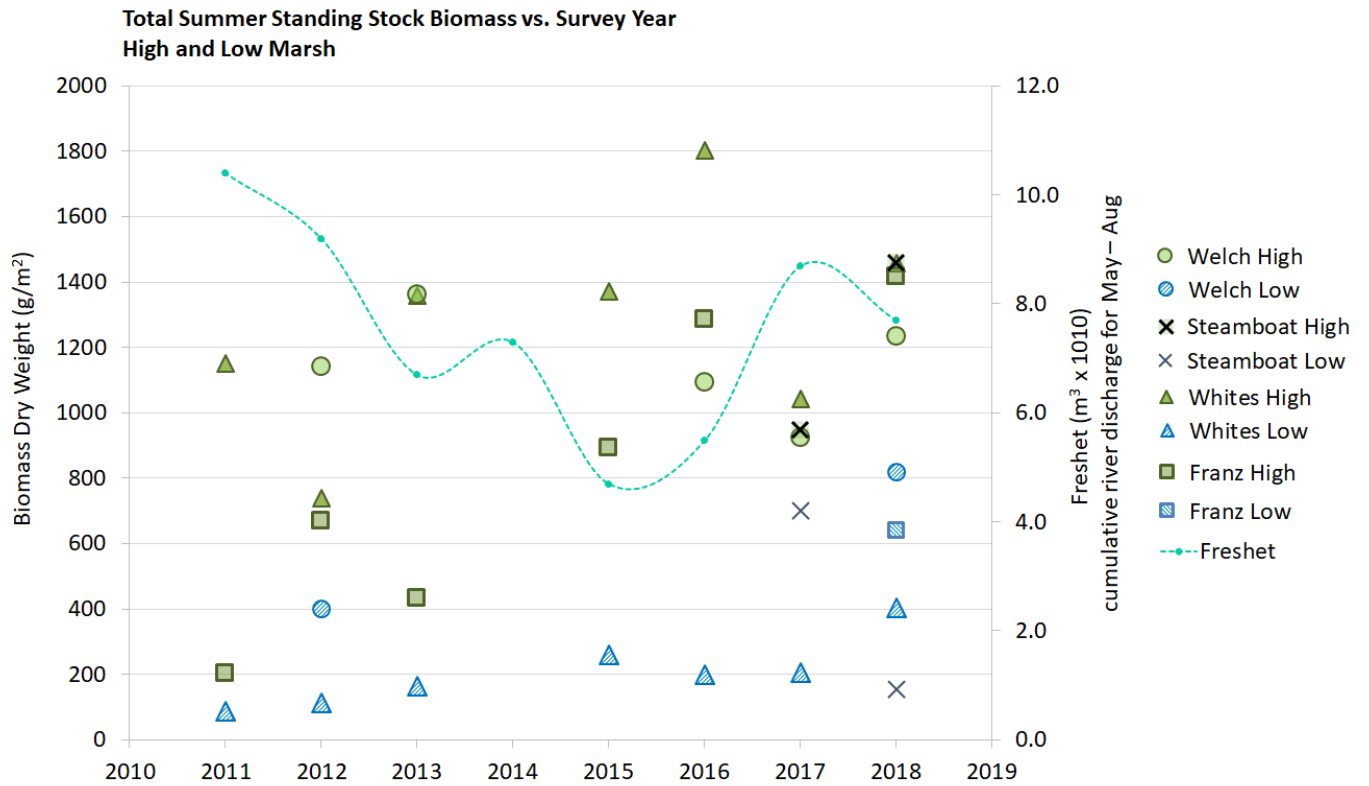
Figure 52: Total summer standing stock biomass (dry weight, g/m^2) data split by high and low marsh plant community strata by survey year. Freshet magnitude is shown as the right axis for reference (see section 3.3.1. for details). Biomass data and standard deviation is shown in Table 32.
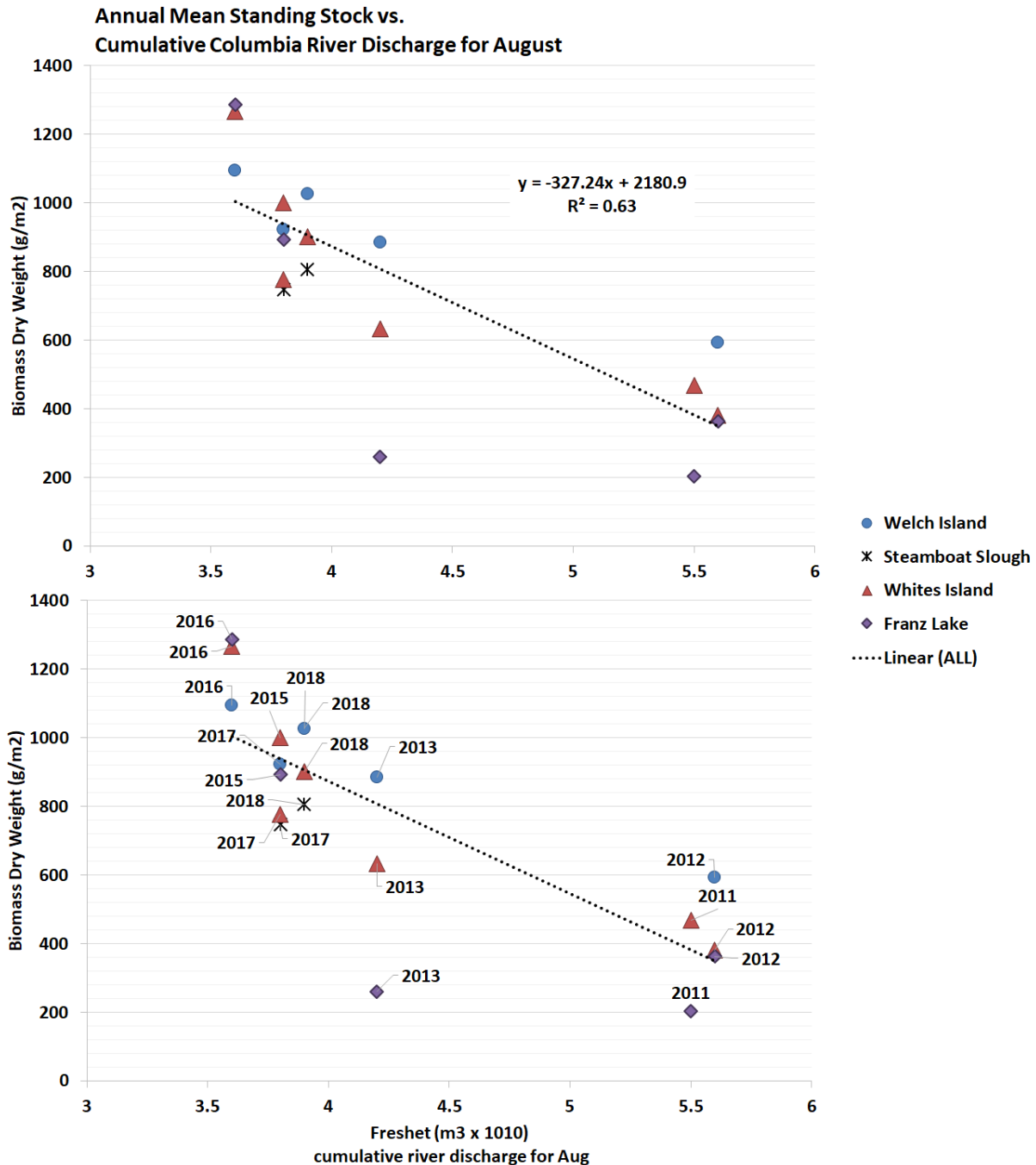
Figure 53: Overall annual mean total standing stock biomass (dry weight, g/m²/year) vs. the Freshet cumulative river discharge for the month of August. Second graph has annual labels for emphasis.

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 54, Table 33). 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 54, Table 33). 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 54, Table 33). 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 54, Table 33). 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.
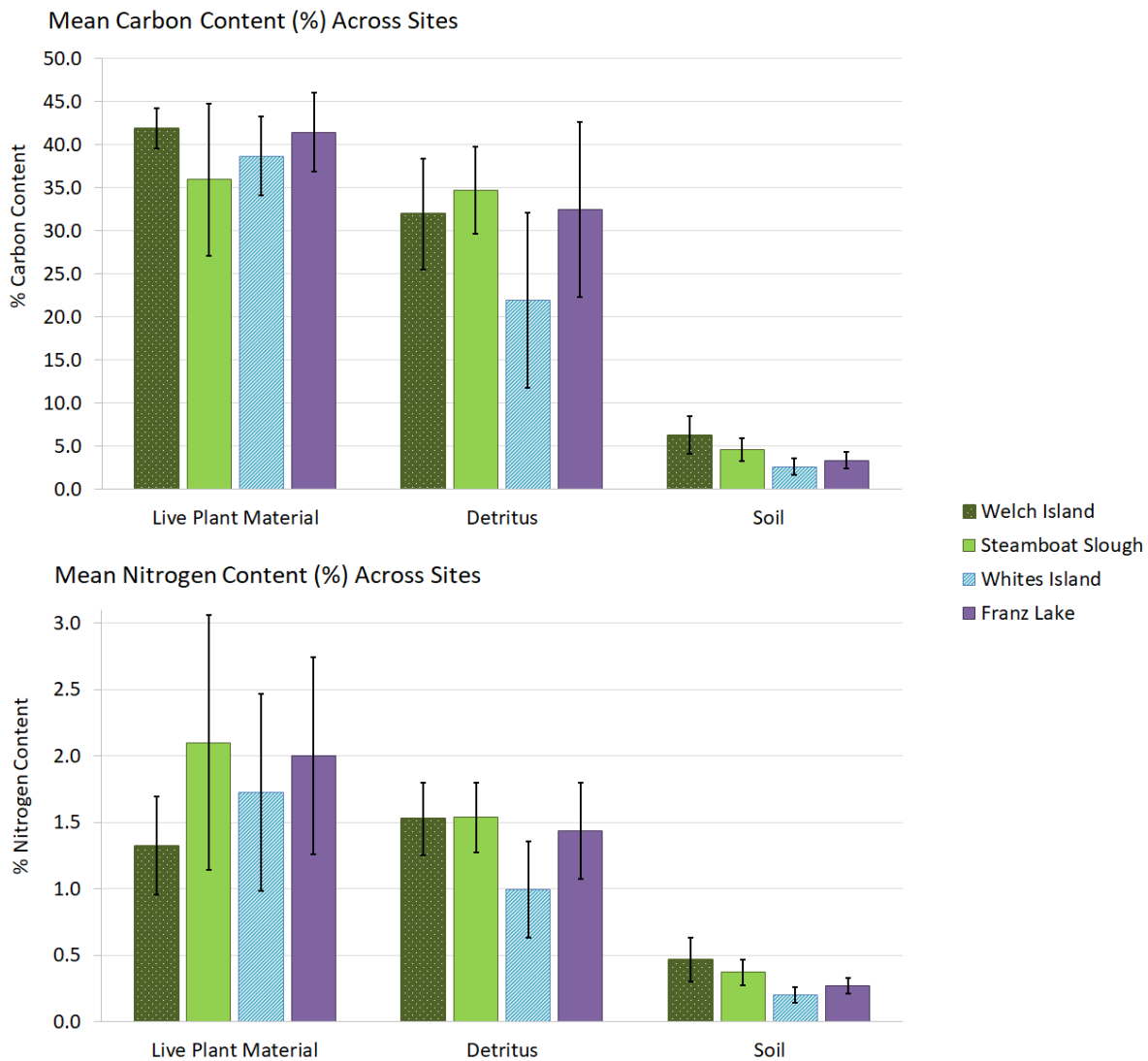
Figure 54: Mean % Carbon and Nitrogen content of above ground living biomass, detritus, and soil across sites, samples collected in the Summer of 2018. Data provided in Table 33.

The C:N ratio was greatest in the living above ground biomass across all sites, followed by detritus, and soil (Figure 55, Table 33). In living aboveground biomass mean C:N content ranging from $34.7 \pm 11.1\%$ at Welch Island to $23.1 \pm 7.9\%$ at Franz Lake. Mean detritus C:N content was less variable than living biomass C:N content and ranged from $21.3 \pm 4.7\%$ at Welch Island to $23.6 \pm 10.2\%$ at Franz Lake. Mean soil C:N content ranged from $13.9 \pm 2.0\%$ at Welch Island to $12.3 \pm 1.7\%$ at Steamboat Slough.
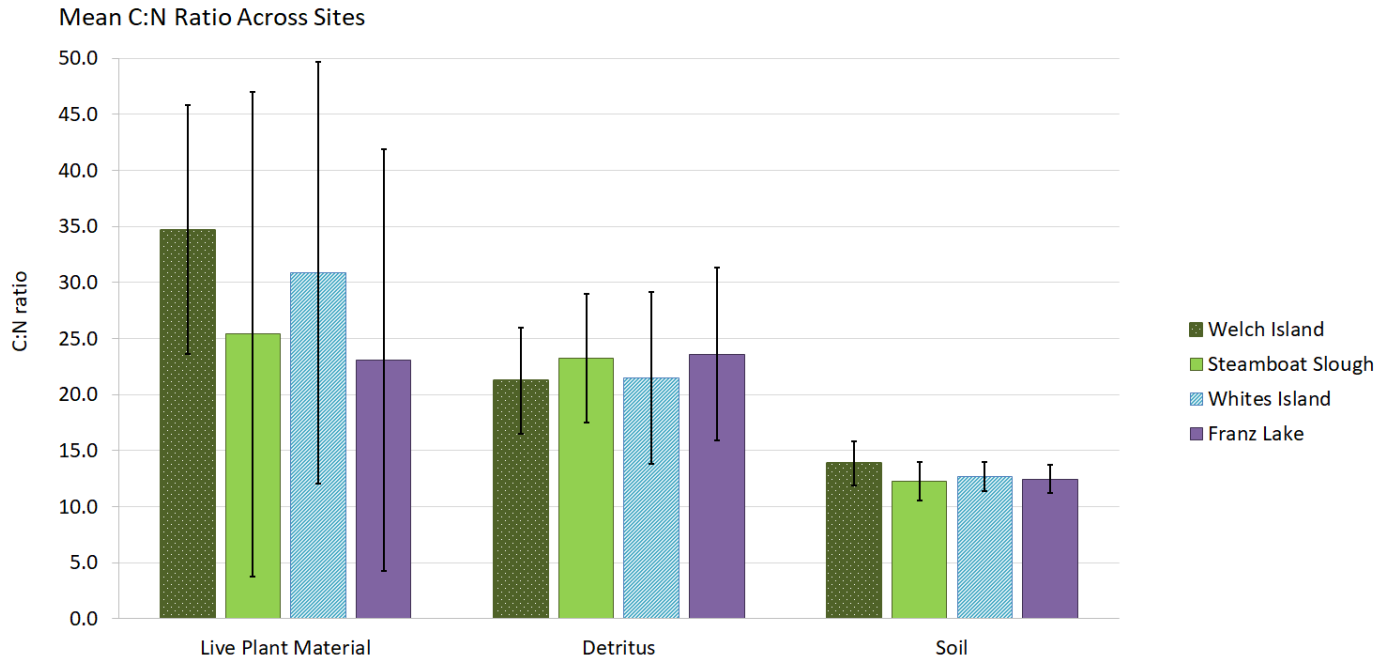
Figure 55: Mean Carbon and Nitrogen ratio of above ground living biomass, detritus, and soil across sites, samples collected in the Summer of 2018. Data provided in Table 33.

Table 33: 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 Error! Reference source not found. **and Figure 55.**

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 56). 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 56). 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 56). 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.
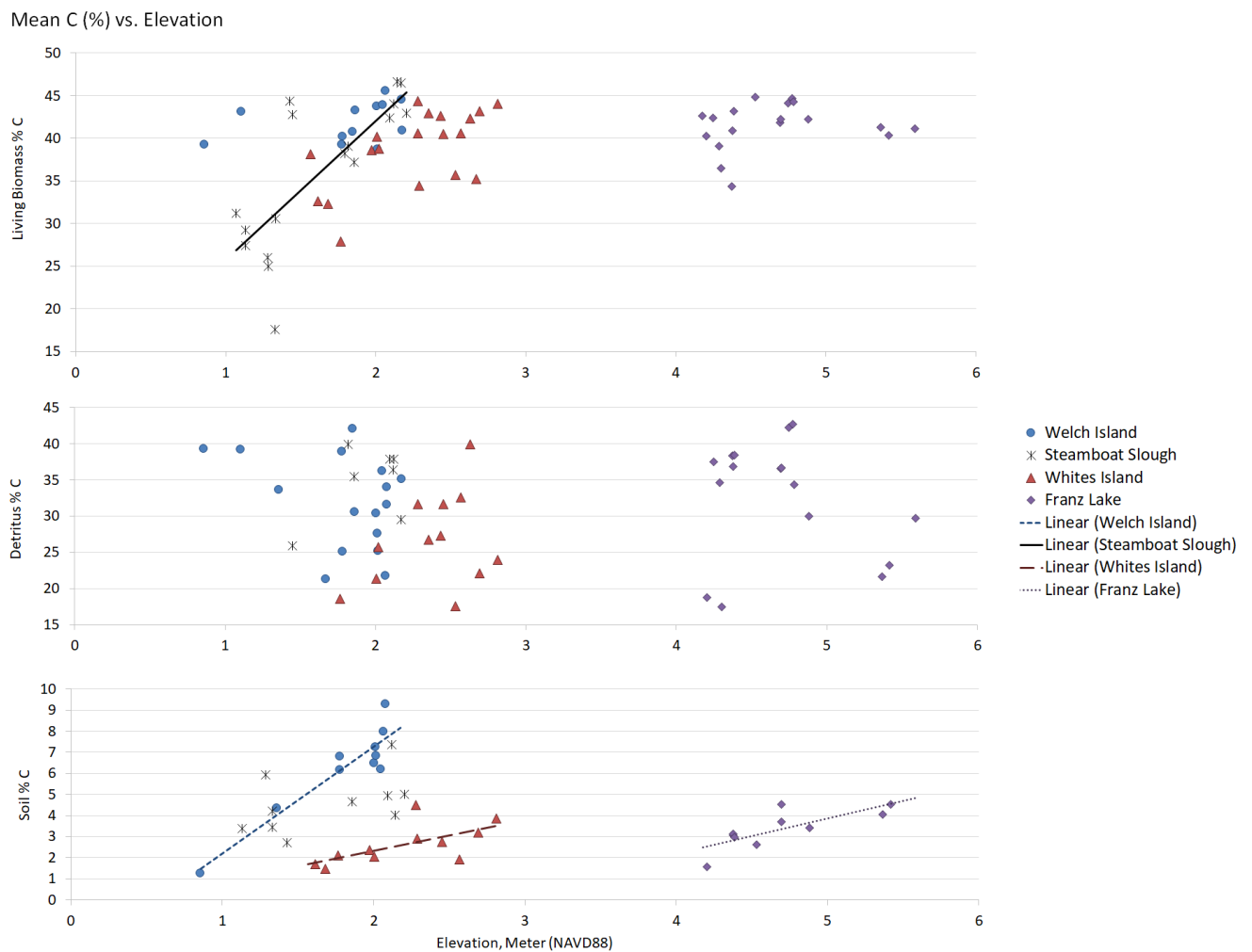
Figure 56: Carbon content (%) of above ground living biomass, detritus, and soil (graphs from top to bottom) vs. sample elevations in meters NAVD88. Significant within site correlations shown in each graph, site summary data provided in Table 33.

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 57). 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 57).
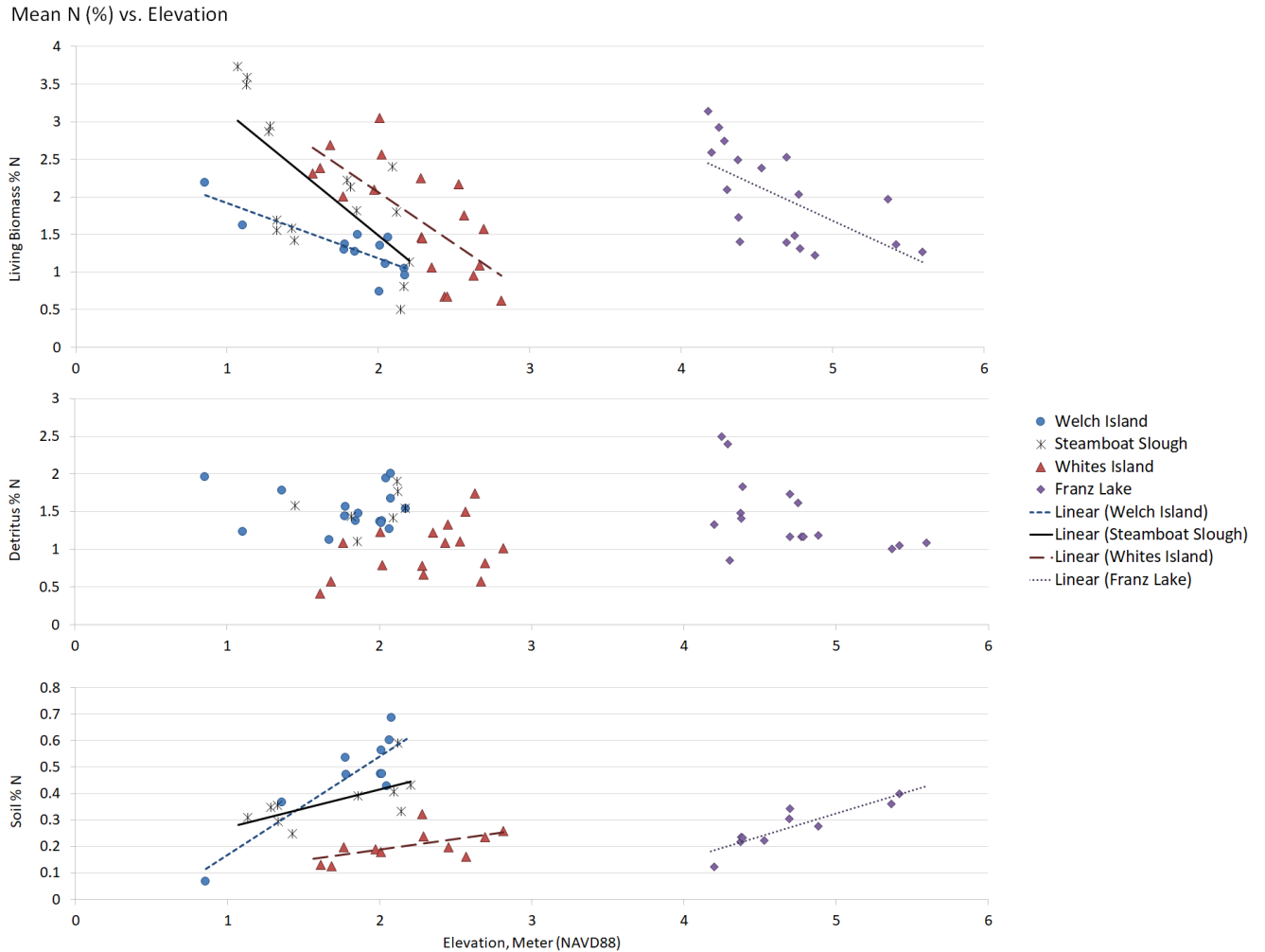
Figure 57: Nitrogen content (%) of above ground living biomass, detritus, and soil (graphs from top to bottom) vs. sample elevations in meters NAVD88. Significant within site correlations shown in each graph, site summary data provided in Table 33.

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 58). No relationships in detrital C:N ratio and elevation were identified across the research sites (Figure 58). 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 58). 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 50). 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 63).
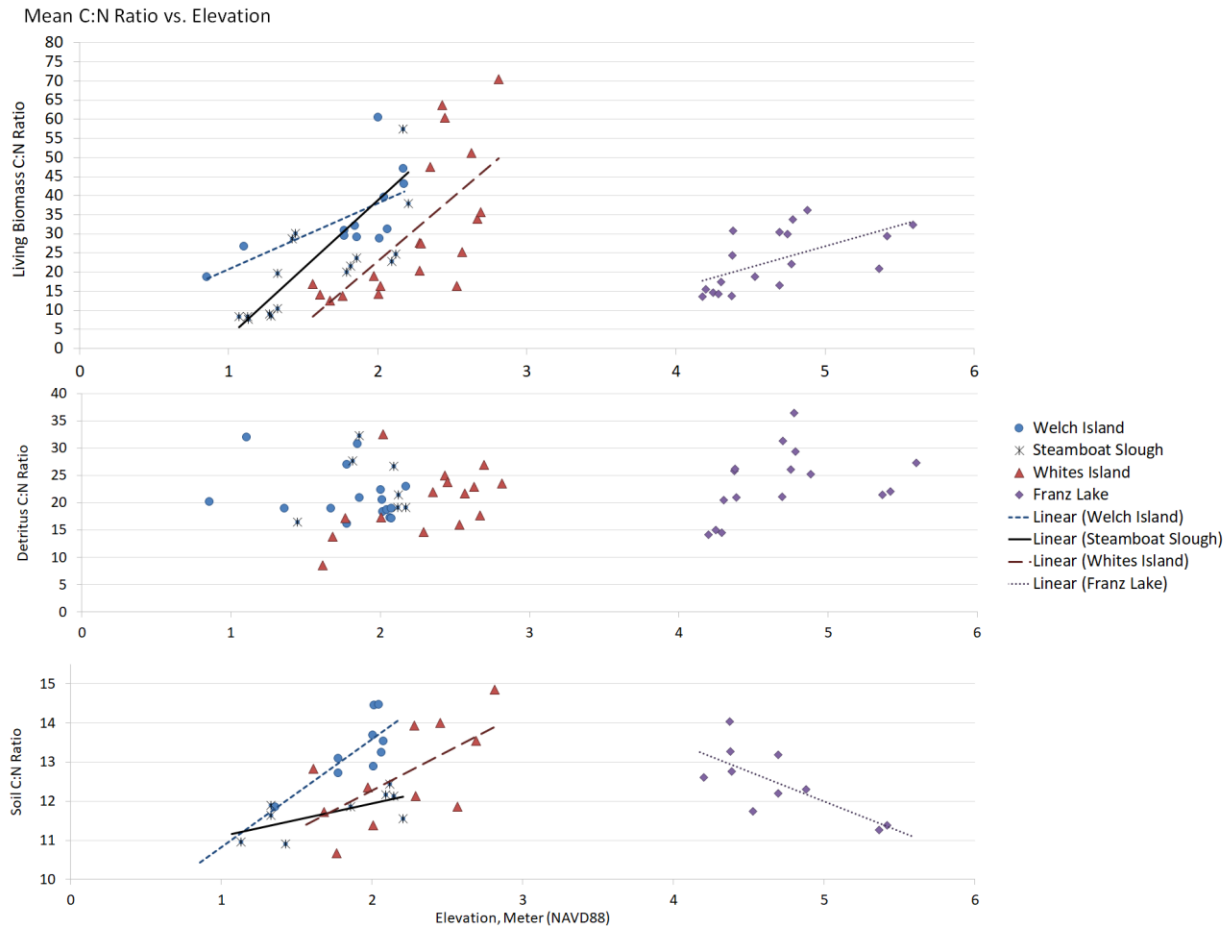
Figure 58: Carbon and Nitrogen ratio of above ground living biomass, detritus, and soil (graphs from top to bottom) vs. sample elevations in meters NAVD88. Significant within site correlations shown in each graph, site summary data provided in Table 33.

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 59**). This relationship was also found in the detritus, with detrital C and N having a positive correlation across all sites (Figure 62). 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.
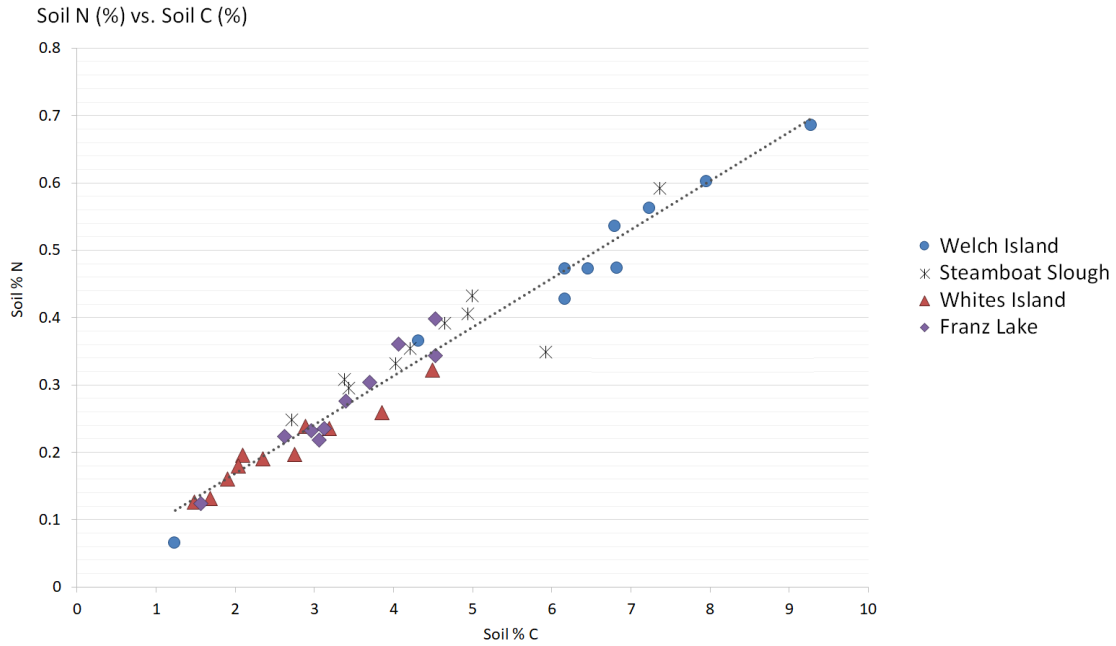
Figure 59. Soil N content (%) vs. Soil C content (%), a strong correlation was shown across all sites ($r^2 = 0.95$, p<0.001). Summary data provided in Table 33.
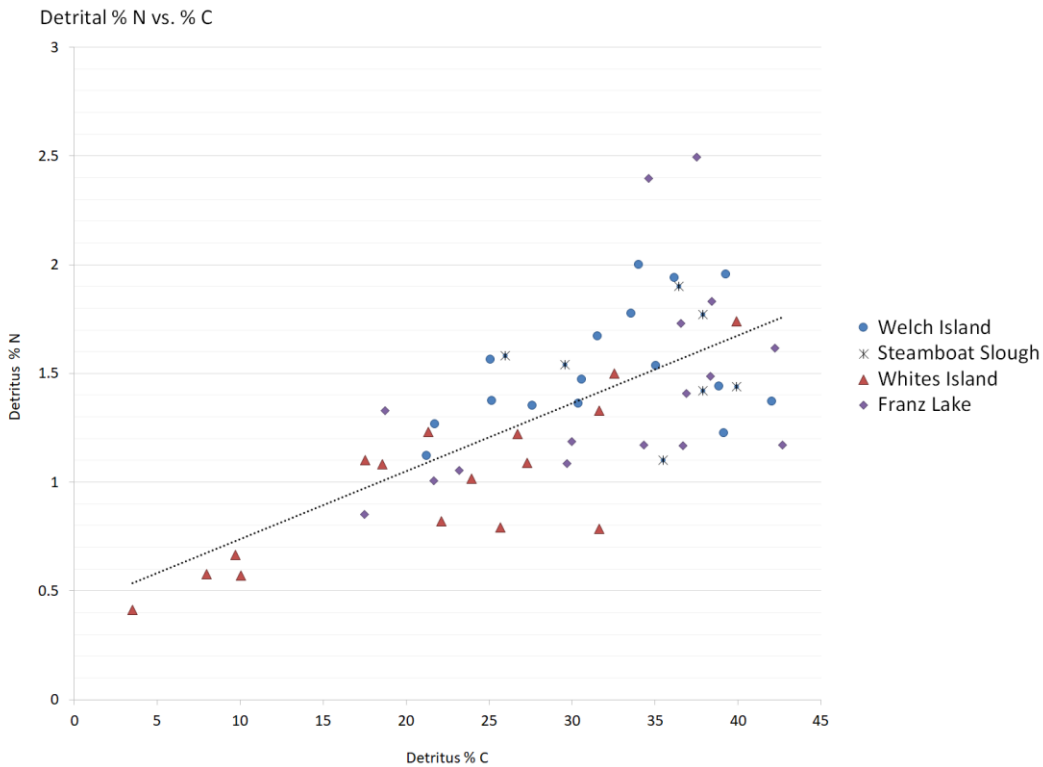
Figure 60: Detritus N content (%) vs. Detritus C content (%), a strong correlation was shown across all sites ($r^2 = 0.47$, p<0.001). Summary data provided in Table 33.

Overall, mean summer ADF Lignin content was greatest in the detritus samples compared to the living plant biomass (Figure 61, Table 34), 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, $19.8 \pm 7.0\%$, and the living above ground biomass from Franz Lake, $12.9 \pm 6.6\%$, followed by the detritus, $15.4 \pm 9.0\%$, and biomass at Welch Island, to $9.0 \pm 15.4\%$. The lowest ADF Lignin levels were identified within the living above ground biomass at Steamboat Slough, $5.8 \pm 1.7\%$, and the lowest detritus ADF Lignin levels were identified at Whites Island, $6.6 \pm 8.8\%$. 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 62, Figure 63). 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 34: Mean % ADF Lignin content of above ground living biomass, and detritus across sites, samples collected in the Summer of 2018. For comparative graphs see Table 34.

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 61: 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 34 Error! Reference source not found.**.**

Figure 62: 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 34.

Detrital L:N Ratio vs. % C

Figure 63: 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 34.

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 35, Table 36) 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 35, Table 36). 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 35: 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/m²
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 36: 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/m²). Y = Native, N =

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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/m²
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 37, Figure 64). The exact amount of silt, sand, and clay did vary within and among sites, with Welch Island having the highest levels of sand, $33.3 \pm 19.2\%$, followed by Steamboat Slough, $27.7 \pm 7.3\%$, Whites Island, $24.7 \pm 9.1\%$, and Franz Lake, $11.0 \pm 6.4\%$. Correspondingly, Silt levels were greatest at Franz Lake, $77.2 \pm 5.8\%$, followed by Whites Island, $69.9 \pm 6.7\%$, Welch Island, $59 \pm 15.4\%$, and Steamboat Slough $57.5 \pm 8\%$. Steamboat Slough showed the greatest soil clay content, $14.8 \pm 4.8\%$, followed by Franz Lake, $11.7 \pm 1.3\%$, Welch Island $7.6 \pm 4.3\%$, and Whites Island, $5.3 \pm 3.3\%$. 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 64). 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 64, Table 34).

Mean Soil Texture (%) Across Sites

Mean Soil Bulk Density (g/cm³) Across Sites
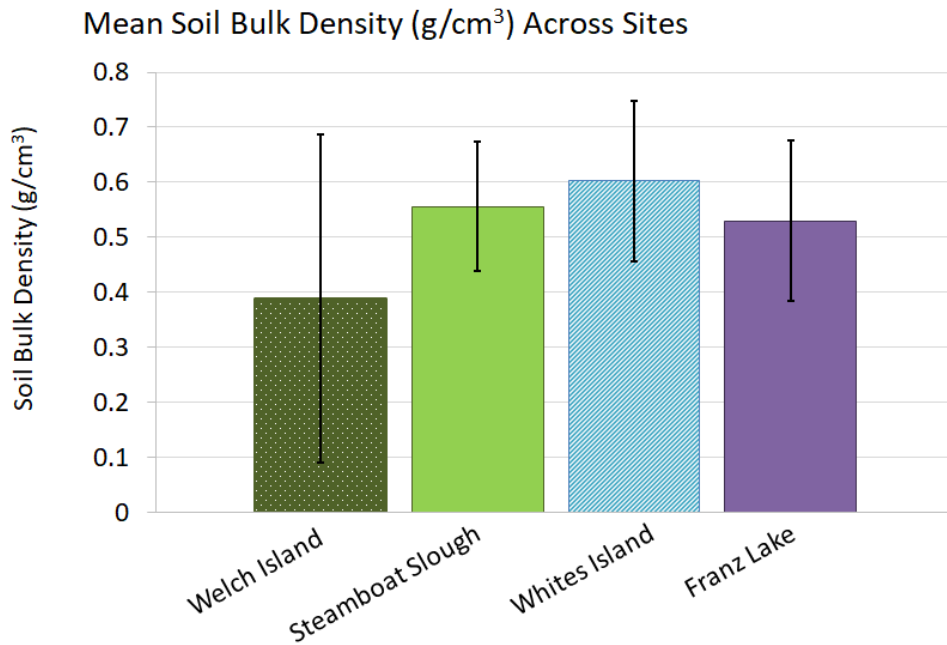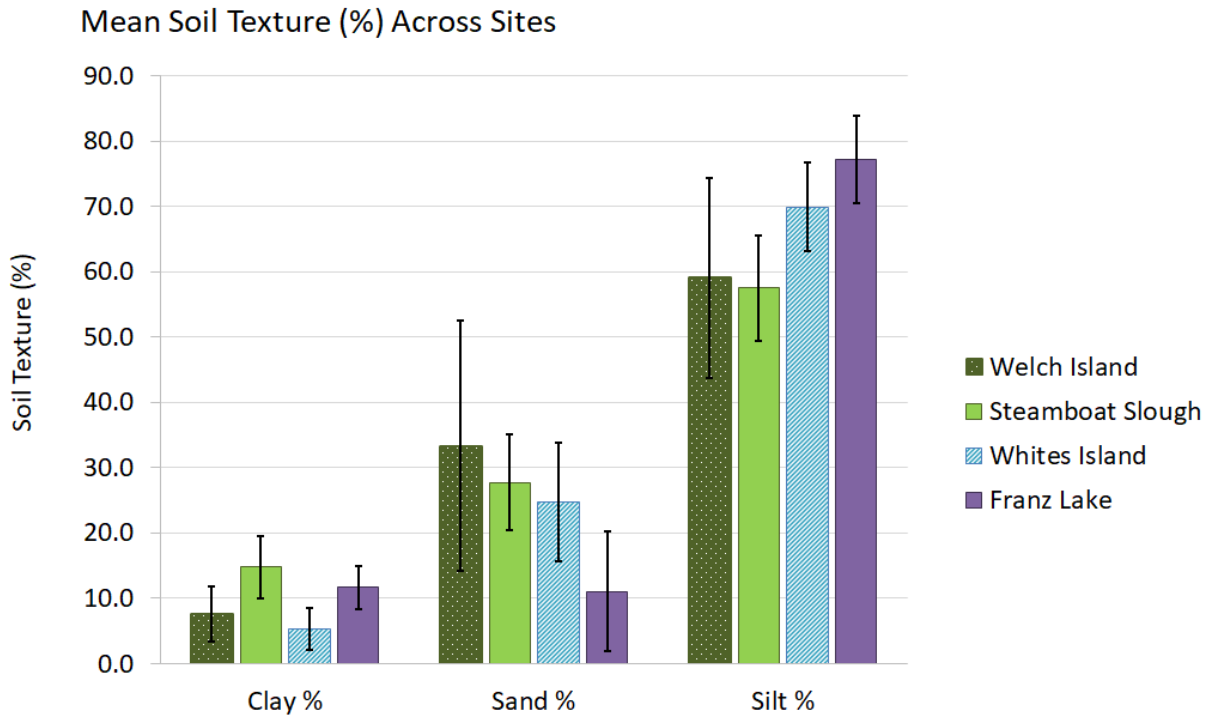
Figure 64. Mean soil texture composition (%) and bulk density (g/cm³) across sites, samples collected in the Summer of 2018. For summary data see Table 34. Error! Reference source not found.

Table 37: Mean soil texture composition (%) and bulk density (g/cm³) across sites, samples collected in the Summer of 2018. For comparative graphs see Figure 64.

Site		Soil Characteristics			
		Clay (%)	Sand (%)	Silt (%)	Bulk Density (g/cm³)
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 2018, algal biomass was moderate to low in March and April (Figure 65). 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 (Figure 66). Chlorophyll a values were moderately high in May at both Whites Island and Welch Island, which suggest strong growth, given that flows were >6000 m³/s. Similar to previous years, the lowest chlorophyll *a* values were observed at Ilwaco Slough. Only three measurements exceeded 25 µg L⁻¹ in 2018 (Figure 65). If a benchmark of 15 µg L⁻¹ is used, six observations were above the recommended threshold suggesting poor water quality (Oregon State Water Quality Standards). However, since a body of is only considered impaired when the threshold is exceeded in observations from three consecutive months, no site met this criterion.
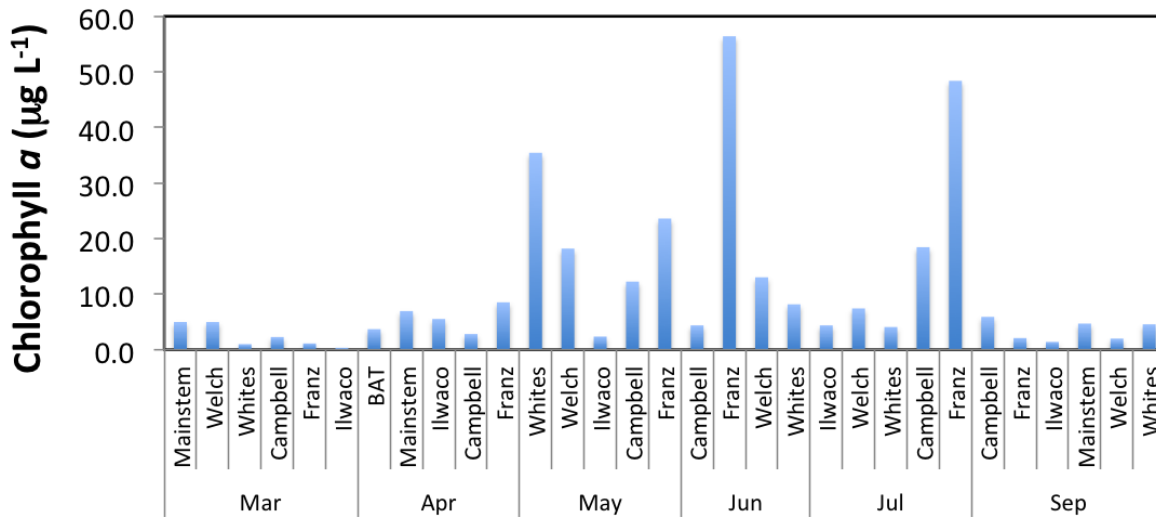
Figure 65. Chlorophyll *a* concentrations at five off-channel trends sites in 2018. The largest peaks were observed at Franz Lake Slough in June and July.
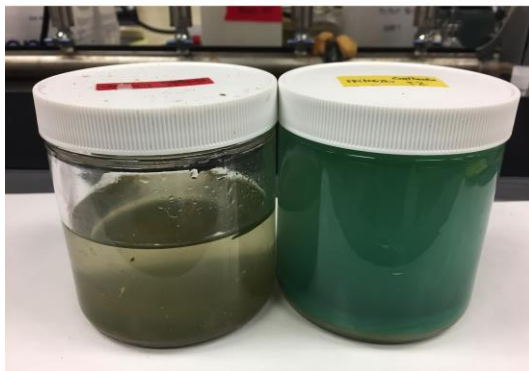
Figure 66. Image showing contents from net tows at Whites Island (left) and at Franz Lake Slough (right). The Franz Lake sample was dominated by a mixed assemblage of cyanobacteria, including *Dolichospermum* spp., *Microcystis* spp., and *Aphanizomenon* spp.

Primary production, as approximated by chlorophyll *a* concentration, was highest in the June and July at Franz Lake Slough in 2018, where peaks reached 50 μg L^{-1} or higher (Figure 65). High values were also observed in May at Whites Island and Welch Island. Unlike at Franz Lake Slough, chlorophyll concentrations at Campbell Slough were not higher than at Whites Island or Welch Islands in 2018, except in July when concentrations were slightly higher. In 2017, chlorophyll *a* concentrations were high at Campbell Slough prior to the spring freshet. Overall, the highest chlorophyll concentrations tended to occur later in the year, either as the freshet was subsiding or during the warm summer months (e.g., at Franz) when cyanobacteria biomass can reach very high levels (e.g., Figure 66).
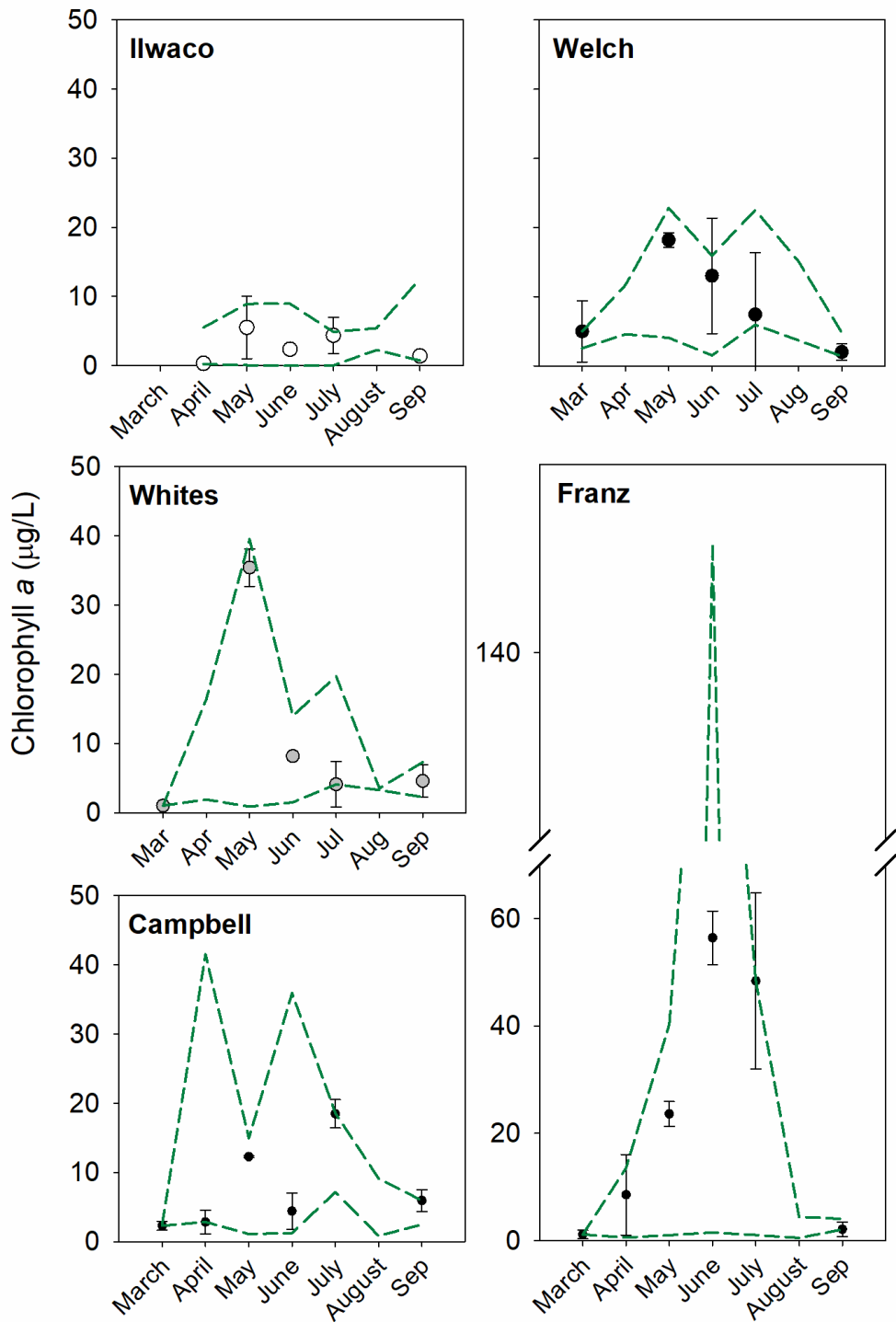
Figure 67. Measurements (data points with error bars representing +/- 1 standard deviation of the mean) of chlorophyll *a* at the five trends sites in 2018 (Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough). Dashed lines show the minimum and maximum values observed between 2011 and 2018.

Based on a comparison to the maximum and minimum chlorophyll *a* values observed between 2011 and 2018, primary production in 2018 was high prior to the freshet at Welch Island, Whites Island, and Franz Lake Slough; values at Campbell Slough were low compared to the long-term average. At Ilwaco values prior to the freshet were low or moderate compared to those observed between 2011 and 2018.
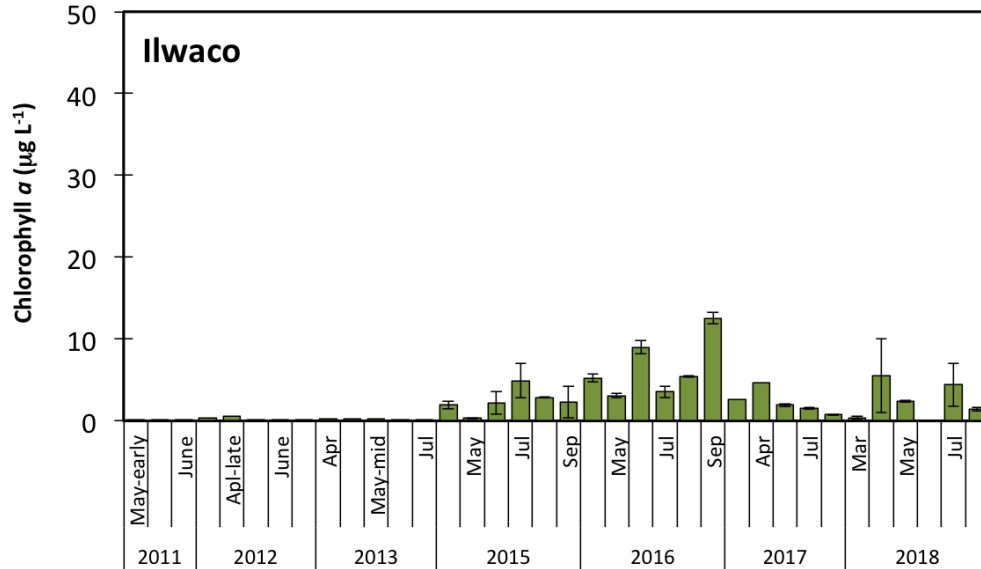
Figure 68. Chlorophyll *a* concentrations in discrete samples collected from Ilwaco (2011–2018).

At Ilwaco, magnitudes of primary production were similar in 2018 to values observed in 2015; values were higher than in 2017 but lower than in 2016 (Figure 68). Values were always less than at the other sites and generally <10 μg L⁻¹ throughout the time series.
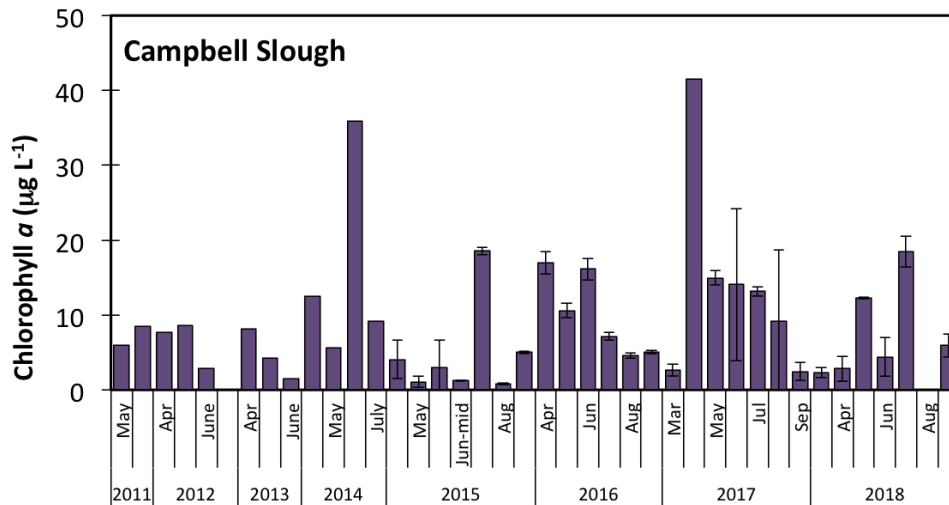
Figure 69. Chlorophyll *a* concentrations in discrete samples collected from Campbell Slough (2011–2018).

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 (June) (Figure 69). Month-to-month variability was higher in 2018 than in previous years. Chlorophyll a values at Franz Lake Slough were similar to 2016 and 2017 (except the very high value observed in June 2017) (Figure 70). 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.
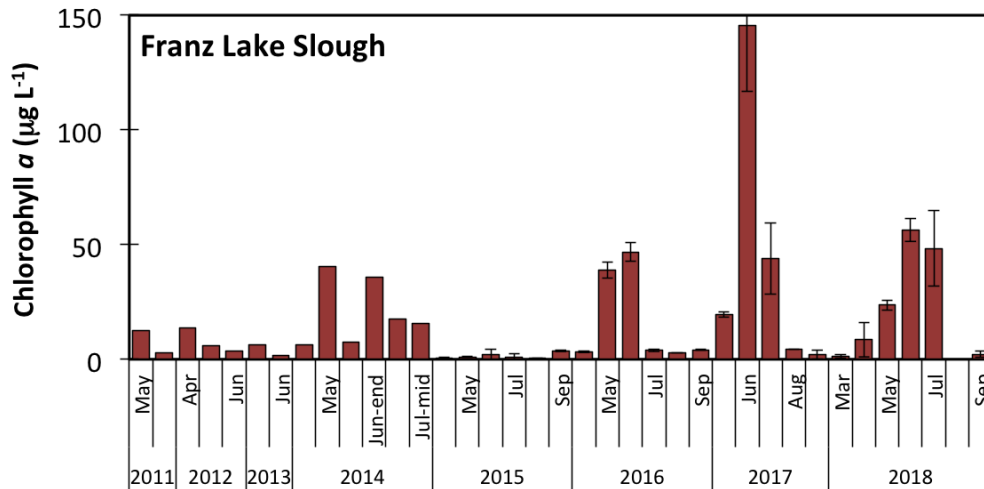
Figure 70. Chlorophyll *a* concentrations in discrete samples collected from Franz Lake Slough (2011–2018). From the time series, it is apparent that the high chlorophyll *a* concentration observed at Franz Lake Slough in June 2017 was exceptionally compared to observations in other years.
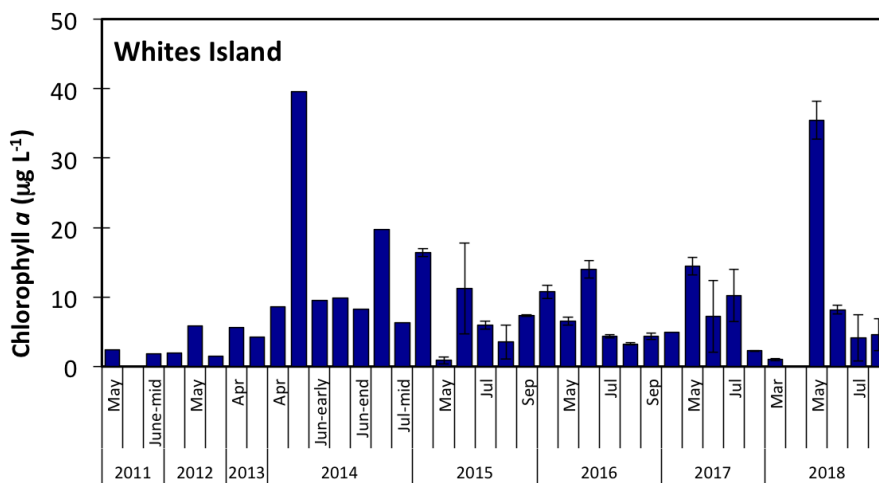
Figure 71. Chlorophyll *a* concentrations in discrete samples collected from Whites Island (2011–2018).

Pre-freshet primary production in 2018 was among the highest values of the time series (Figure 71). After that, however, chlorophyll *a* values were similar to those observed in 2015 and 2016.

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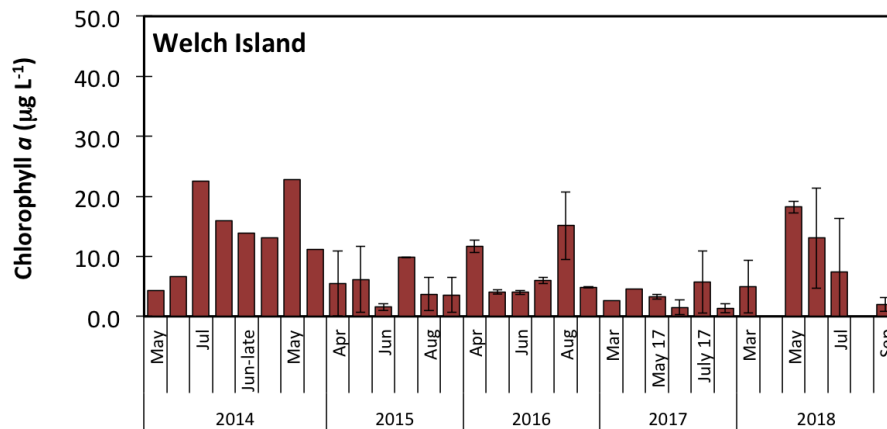
Figure 72. Chlorophyll *a* concentrations in discrete samples collected from Welch Island (2014–2018).

At Welch Island, chlorophyll *a* values were relatively high in May and June, with a decline as the freshet subsided (Figure 72). By September, chlorophyll values were low.
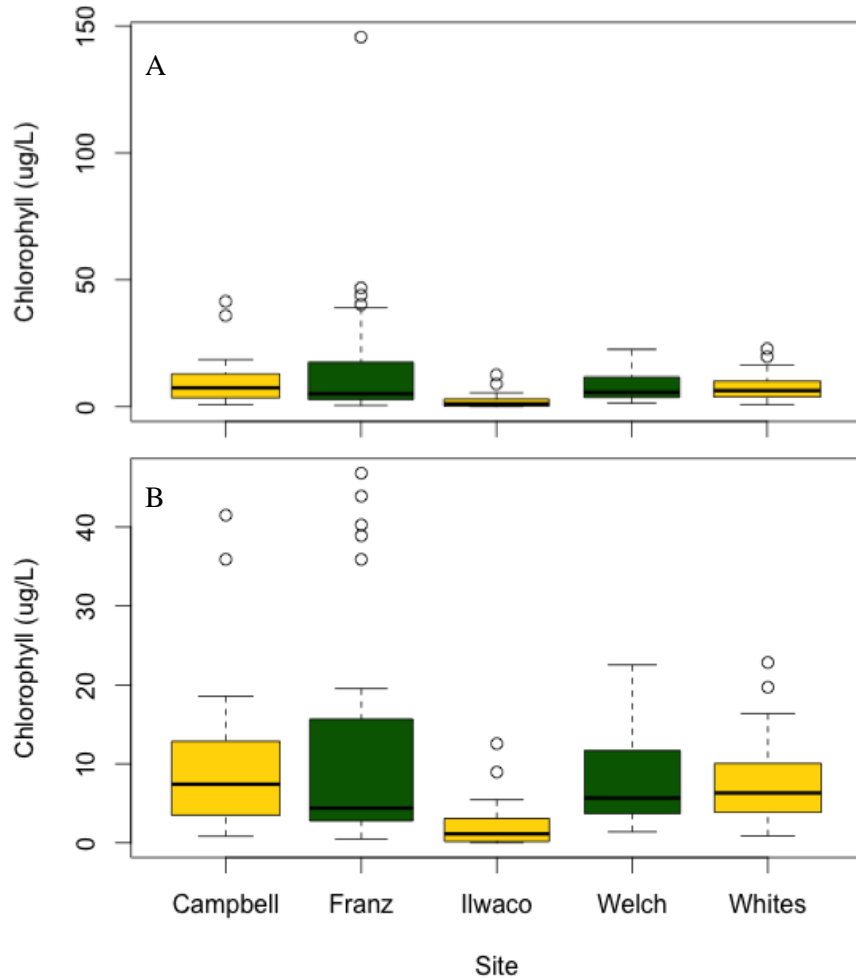
Figure 73. Boxplots are showing chlorophyll concentrations determined at the five off-channel trends sites between 2011 and 2017. (A) The data include the extreme outlier at Franz Lake Slough in 2017; (B) the data excluding the outlier at Franz Lake Slough in 2017.

3.4.1.3 *Phytoplankton Species Composition*

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.

Abundances determined from settled samples are still being analyzed for 2018; however, images showing the phytoplankton assemblage during spring of 2018 were collected using a flow microscope (Flow Cytometer And Microscope, FlowCAM) at Welch Island (Figure 74, Figure 75, Figure 76), Whites Island (Figure 77), and Campbell Slough (Figure 78) and are shown below. Between March and April, large, colonial diatoms became more prevalent at Welch Island; by June, the diatom assemblage was dominated by single cells (Figure 76). The type of phytoplankton in the assemblage is important since taxonomic groups differ in their nutritional value to grazers and because large colonies are often inaccessible to grazers, including copepods, cladocerans, and rotifers.

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As in previous years, abundances of green algae and cryptophytes were higher at Campbell Slough compared to Welch Island and Whites Island (Figure 78). Preliminary analysis of 2018 data indicate that *A. formosa* dominated the phytoplankton assemblage at Whites Island and Welch Island in early to mid-spring, while in late spring and summer--when phytoplankton abundance and chlorophyll *a* concentration was highest at Franz Lake Slough—the assemblages were dominated by the diatoms *Aulacoseira* spp., *Fragilaria crotonensis*, *Cyclotella*, and small pennates, with a diverse assemblage of chlorophytes and cryptophytes present.
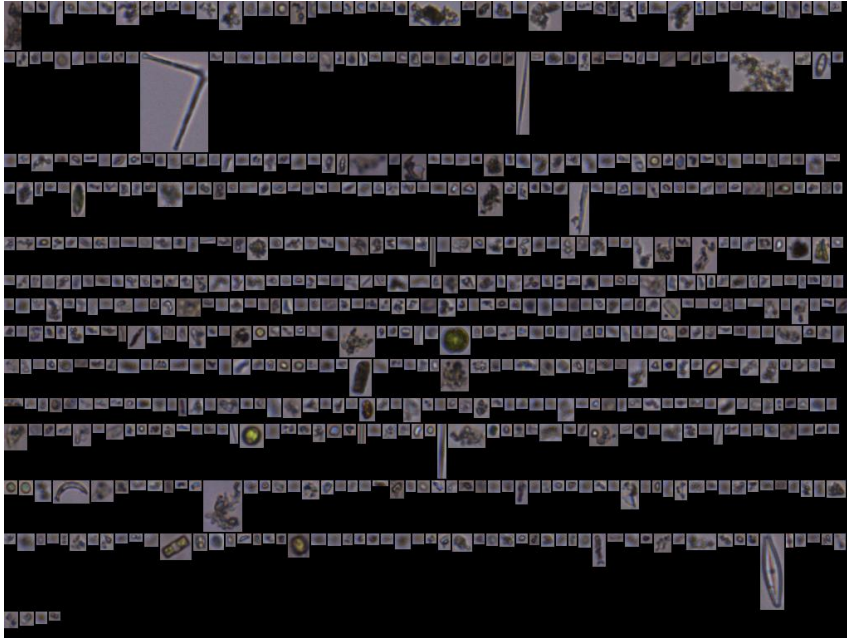
Figure 74. The phytoplankton assemblage at Welch Island in March 2018 (pre-freshet). Most of the *A. formosa* cells present were found in colonies. Data analysis also revealed the presence of cyanobacteria filaments in March.
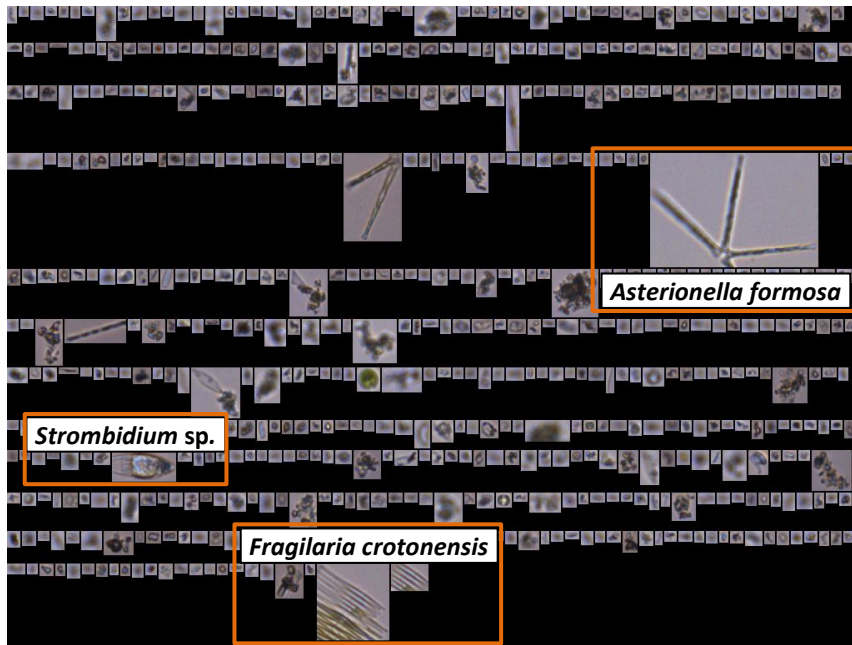
Figure 75. The phytoplankton assemblage at Welch Island in April 2018 (pre-freshet). The dominant diatoms were *Asterionella formosa* (in colonies) and *Fragilaria crotonensis*. Strombidoid ciliates (small microzooplankton grazers) were relatively abundant in April.

Figure 76. The phytoplankton assemblage at Welch Island was dominated by *Asterionella formosa* and other pennate diatoms in June, 2018. Most of the *A. formosa* cells were not found in colonies.

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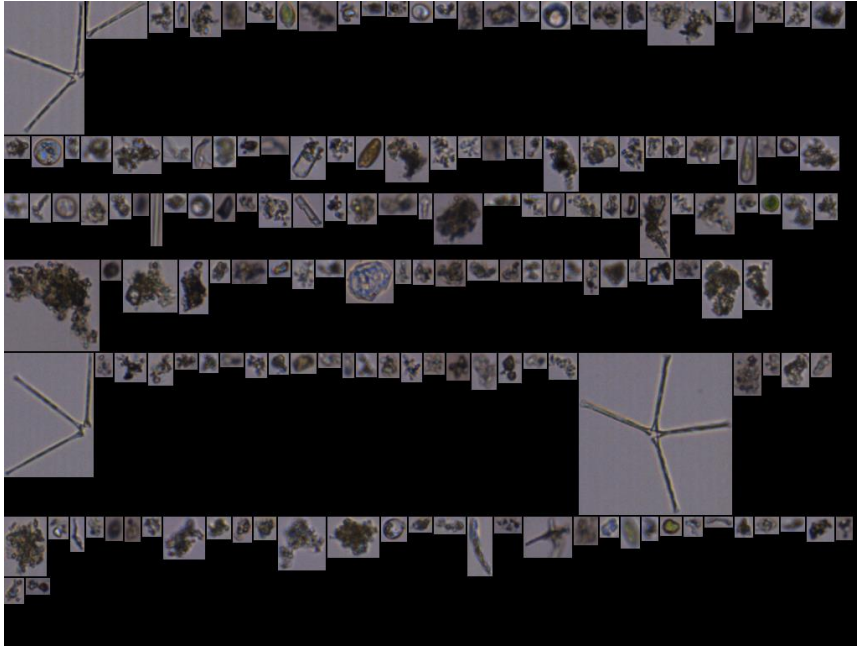
Figure 77. Example of phytoplankton assemblage at Whites Island in March 2018. Star-shaped colonies are *Asterionella formosa***.**
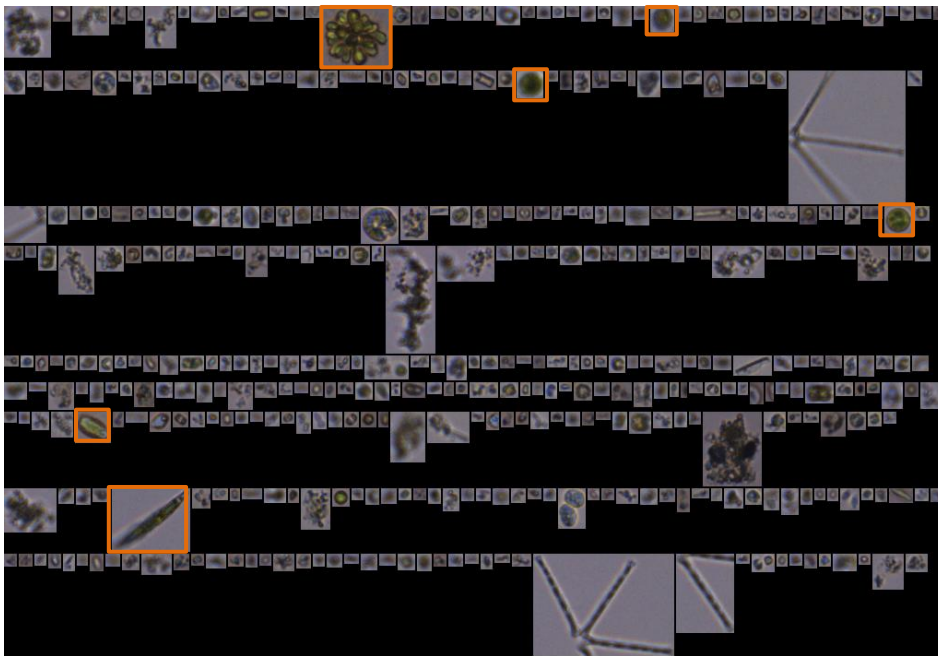
Figure 78. Phytoplankton assemblage at Campbell Slough in March 2018 showing colonies of *Asterionella formosa* **as well as examples of green algae (some of them are indicated by the orange boxes) that are typically present at higher abundances compared to other sites.**

Phytoplankton assemblages tend to be dominated by diatoms at Whites Island and Welch Island, while at Campbell Slough and Franz Lake Slough, cyanobacteria and chlorophytes make more substantial contributions to total phytoplankton abundances. Observations in 2017 were consistent with this trend (**Error! Reference source not found.**, Figure 80, Figure 81, Figure 82).
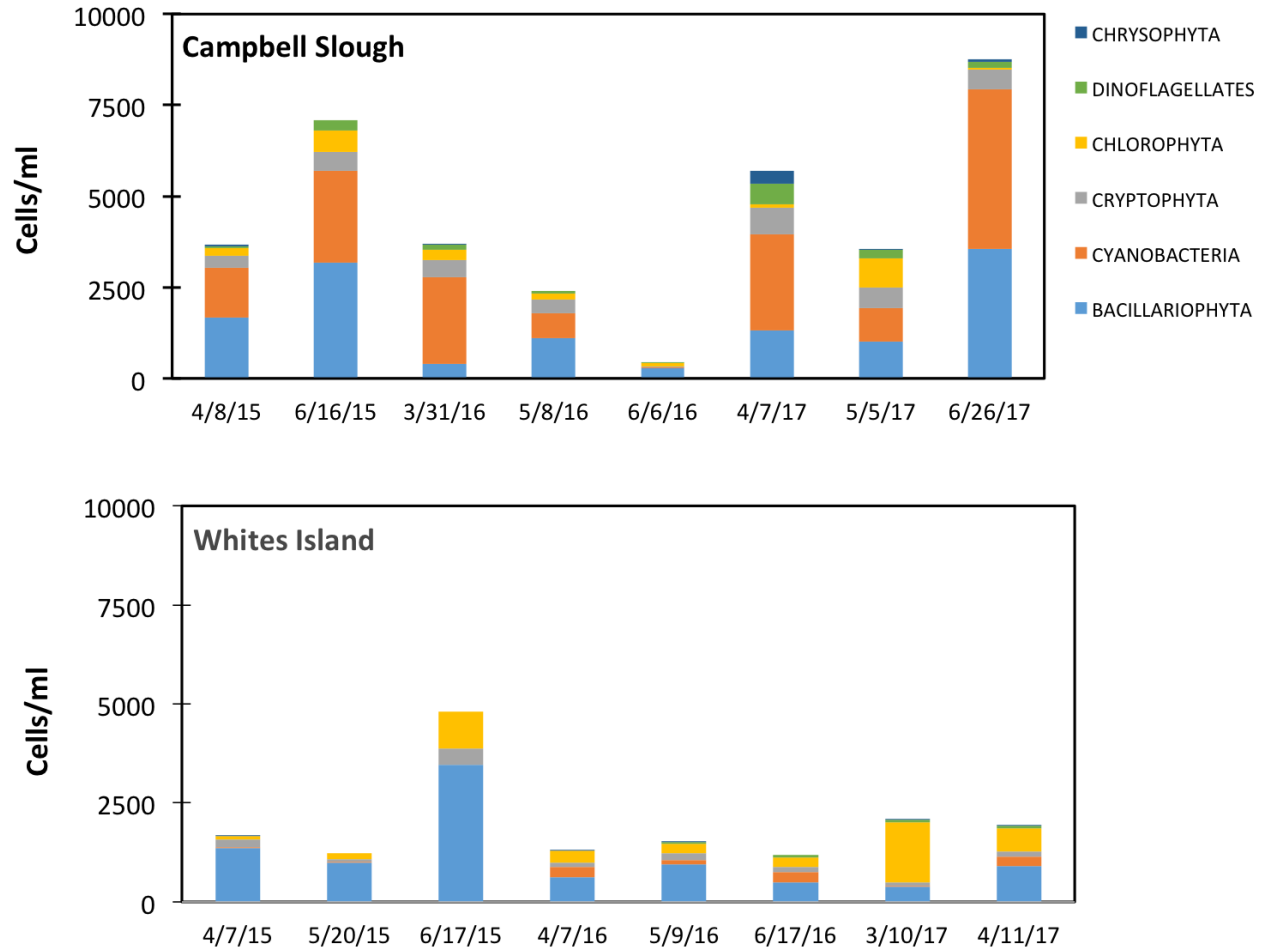
Figure 79. Cell densities of phytoplankton at Campbell Slough (upper panel) and Whites Island (lower panel) during the spring (2015-2017). To improve graph readability, the abundance of cyanobacteria at Campbell Slough on 4/7/2017 is 10 times the number shown.
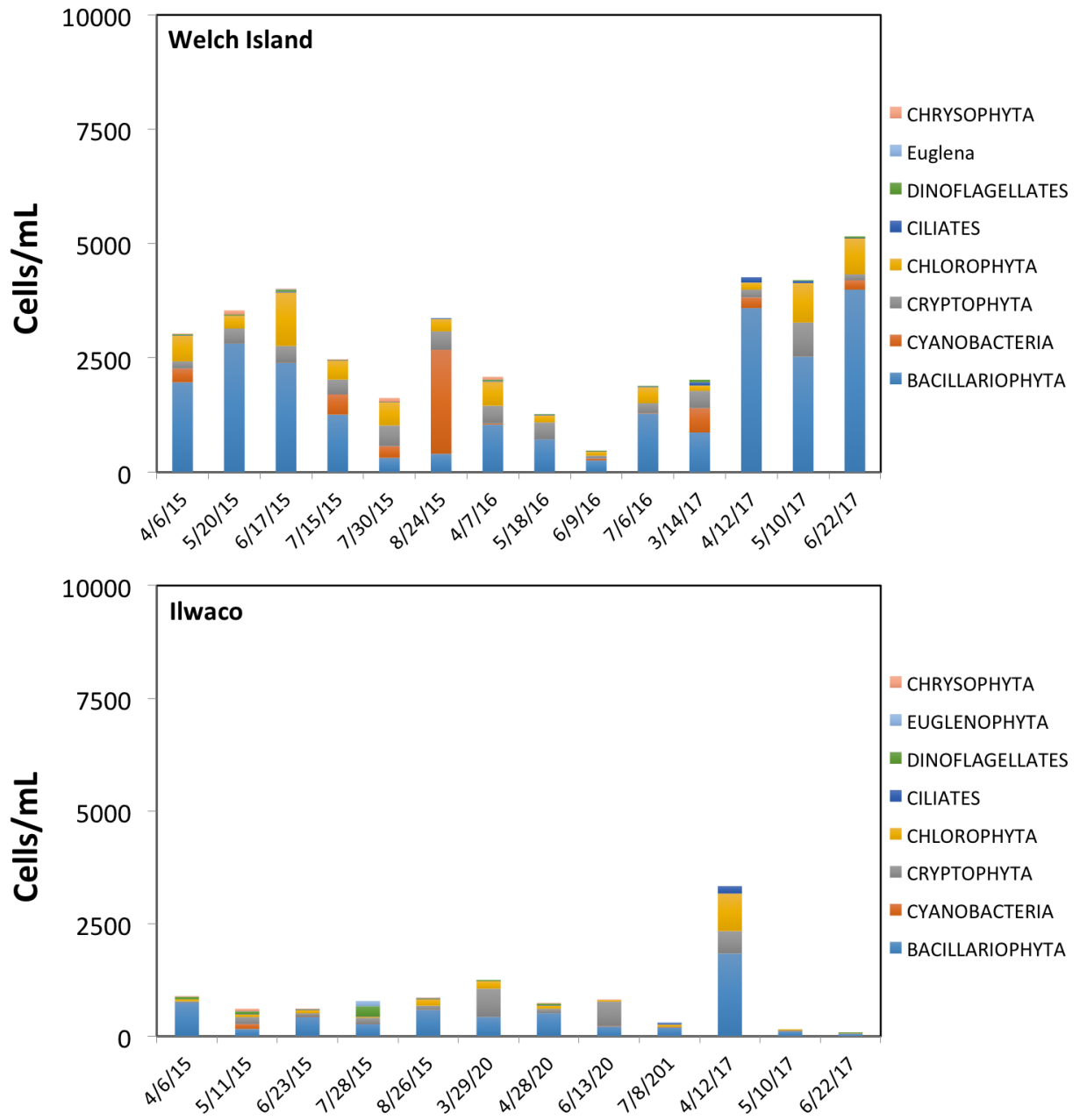
Figure 80. Cell densities of phytoplankton at Welch Island (upper panel), and Ilwaco (lower panel) during the spring (2015-2017).
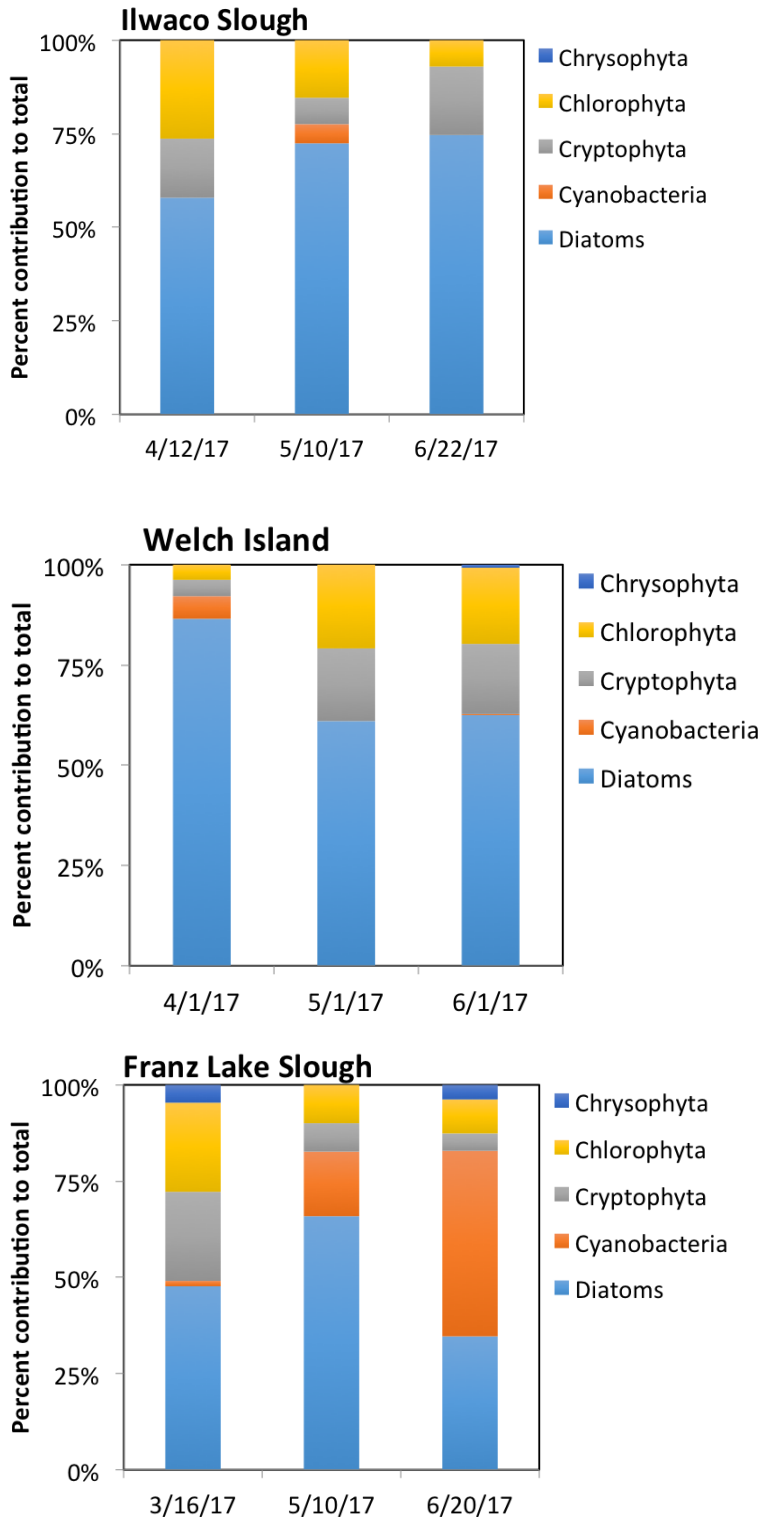
Figure 81. The relative proportion of different classes of phytoplankton at Ilwaco Slough, Welch Island, and Franz Lake Slough in spring 2017.

As a proportion of the total phytoplankton assemblage, small cells (including cyanobacteria and small chlorophytes) can dominate when considered numerically, particularly at Campbell Slough (Figure 82). Diatoms constitute a greater relative proportion of the assemblage at Whites Island compared to Campbell Slough.
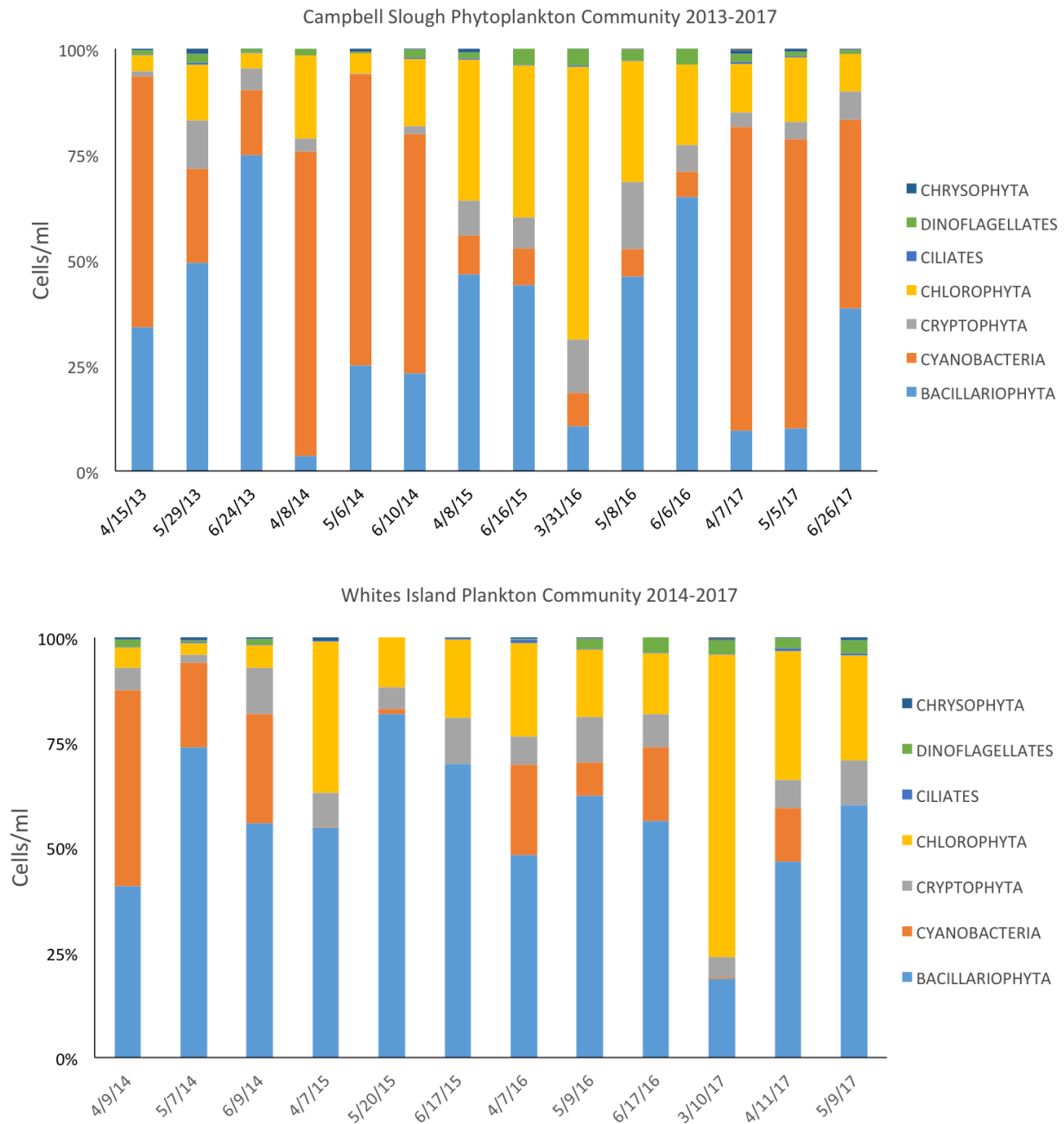
Figure 82. Time series showing percent contribution to total phytoplankton assemblages by various algal groups at Campbell Slough (*top panel*) and Whites Island (*bottom panel*) for spring periods in 2014, 2015, 2016, and 2017.

3.4.1.4 *Phytoplankton carbon*

The amount of carbon associated with unicellular primary producers can be computed from cell volumes; this information is important for determining the potential carbon biomass entering the system since chlorophyll content per cell can vary depending on physiological processes and environmental conditions. Therefore, we report the amount of carbon associated with phytoplankton at Whites Island and Campbell Slough from 2012 to 2017. Between 2012 and 2017, phytoplankton carbon was almost entirely attributable to diatoms at Whites Island, with the highest total cellular carbon concentrations observed early in 2012 and in 2013 (Figure 83). The lowest carbon values associated with phytoplankton were observed in 2015 and 2016. At Campbell Slough, the highest particulate organic carbon concentrations associated with phytoplankton were observed in 2012 and 2017 when river levels were highest (Figure 84). The cellular carbon data provide a slightly different picture than the chlorophyll data presented above.

Whereas at Whites Island, the phytoplankton cellular carbon was strongly dominated by diatoms, at Campbell Slough other taxa made substantial contributions to total phytoplankton carbon, including cryptophytes, euglenophytes, and chlorophytes (Figure **84**).
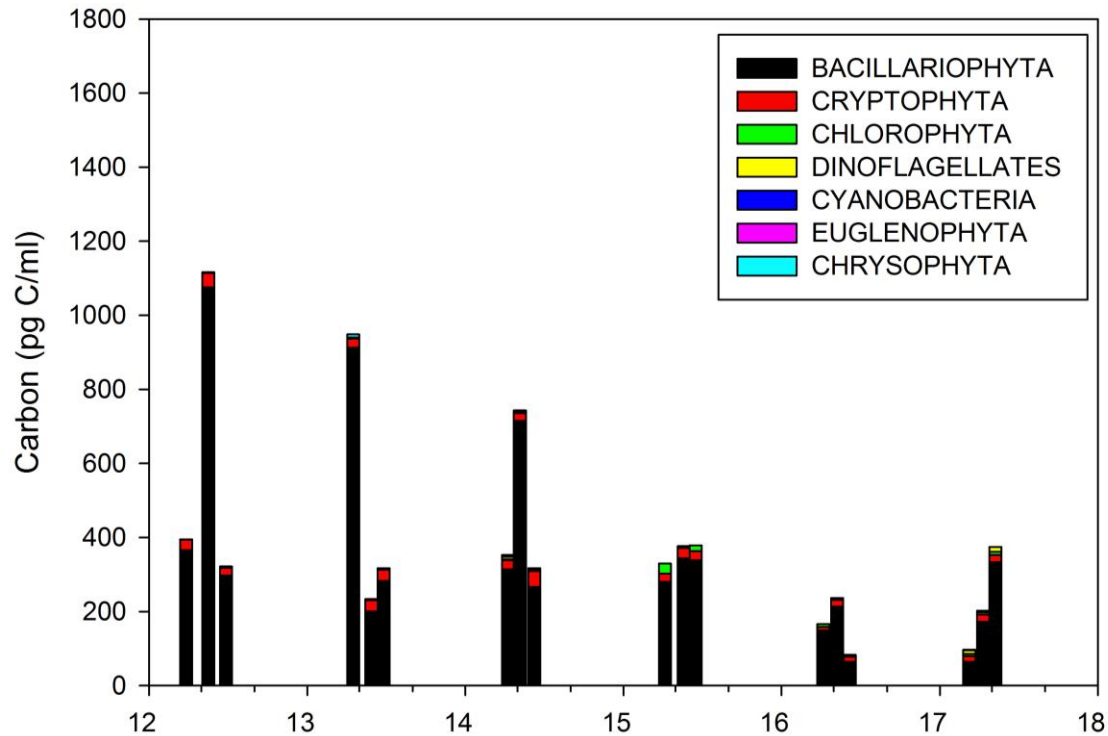
Figure 83. Amount of particulate organic carbon associated with different phytoplankton taxa at Whites Island. Phytoplankton carbon is dominated by diatoms at this site.
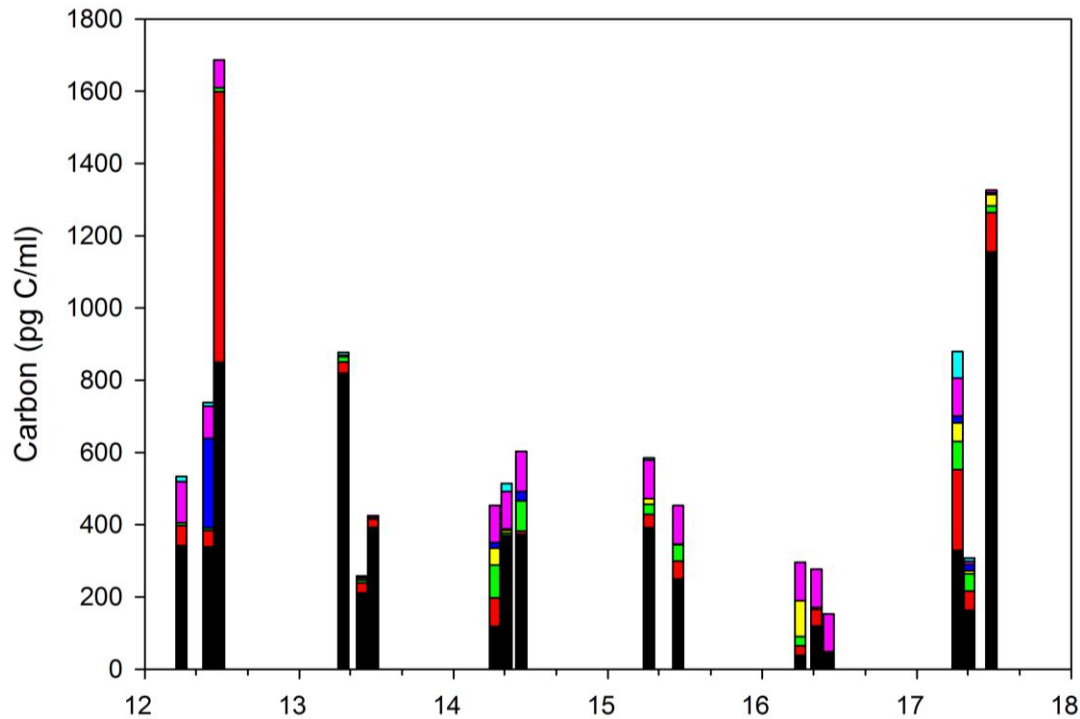
Figure 84. Amount of particulate organic carbon associated with different phytoplankton taxa at Campbell Slough. Phytoplankton carbon is not as strongly dominated by diatoms compared to Whites Island; at Campbell Slough, important contributions to total phytoplankton carbon come from cryptophytes, euglenophytes, green algae, and chrysophytes.

3.4.1.5 *Phytoplankton and environmental variables*

To look for predictive relationships between environmental variable and phytoplankton assemblages, we performed generalized additive modeling on data collected between 2012 and 2017. Significant relationships are shown in Figure 85. There was a negative relationship between the abundance of flagellates and pH; a positive relationship between the abundance of flagellates and nitrate; and a peak in the abundance of flagellates at moderately high river discharge. To examine the influence of phytoplankton species composition on environmental variables such as pH and dissolved oxygen which are influenced by the process of photosynthesis, we performed gam modeling with pH and dissolved oxygen as dependent variables. In this case, there was a positive relationship between dissolved oxygen and the abundance of diatoms, indicating that diatoms are likely responsible for most of the primary productivity at the trends sites in the lower Columbia.
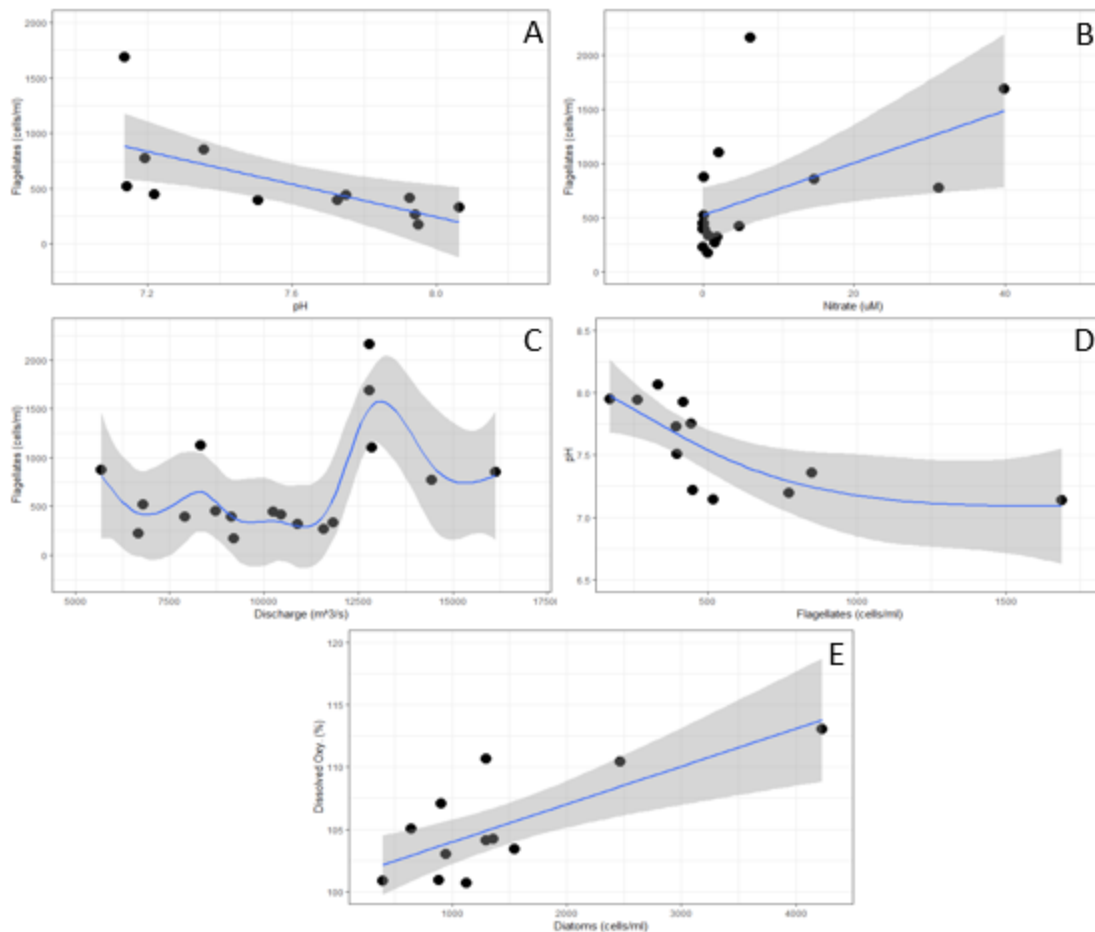
Figure 85. GAM models showing relationships between cellular abundances of phytoplankton at Whites Island vs. environmental variables. (A) Abundance of flagellates (cells ml^{-1}) vs. pH; (B) flagellate abundance (cells ml^{-1}) vs. nitrate (µM); (C) flagellate abundance (cells ml^{-1}) vs. river discharge (m³/s); (D) pH vs. flagellate abundance (cells ml^{-1});(E) dissolved oxygen (%) vs. diatom abundance (cells ml^{-1}).

3.4.1.6 *Parasitism of dominant fluvial primary producers by chytrid fungi*

Infection of diatoms by chytrid fungi can result in the transformation of organic carbon from large, inedible colonial taxa that are essentially unavailable to zooplankton into small, nutritious zoospores, which are easily consumed. Chytrid infections were consistently observed in several diatom taxa, including *Asterionella formosa*, *Aulacoseria* sp., *Cyclotella* sp. and *Thalassosira* sp., *Melosira* sp., nitzschioid diatoms, and *Tabellaria* sp, between 2012 and 2017 (Figure 86, Figure 87). During warmer years, infections in *Synedra* sp. and *Stephanodiscus* sp. were not observed. The prevalence of infection (i.e., the number of hosts infected relative to the total number of hosts in the population) of *A. formosa* was ~20% in all samples, regardless of environmental conditions. Infections were greater and spread across more taxa during 2012 and 2013 when conditions were cooler, and discharge fluxes were high, especially for groups like *Aulacoseira*, nitzchioid pennates, and *Tabellaria* spp. There were few infections outside of *A. formosa* in 2016 and 2017 when river discharge fluxes were both low (2016) and high (2017). The highest (≥80%) infection prevalence in *A. formosa* was observed in 2016 when host abundances were very low; this suggests that the samples may have been collected during the

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latter phase of the bloom. Similar observations of high infection prevalence were seen at low host abundances for *Skeletonema potamos*, *Tabellaria*, and *Melosira*.
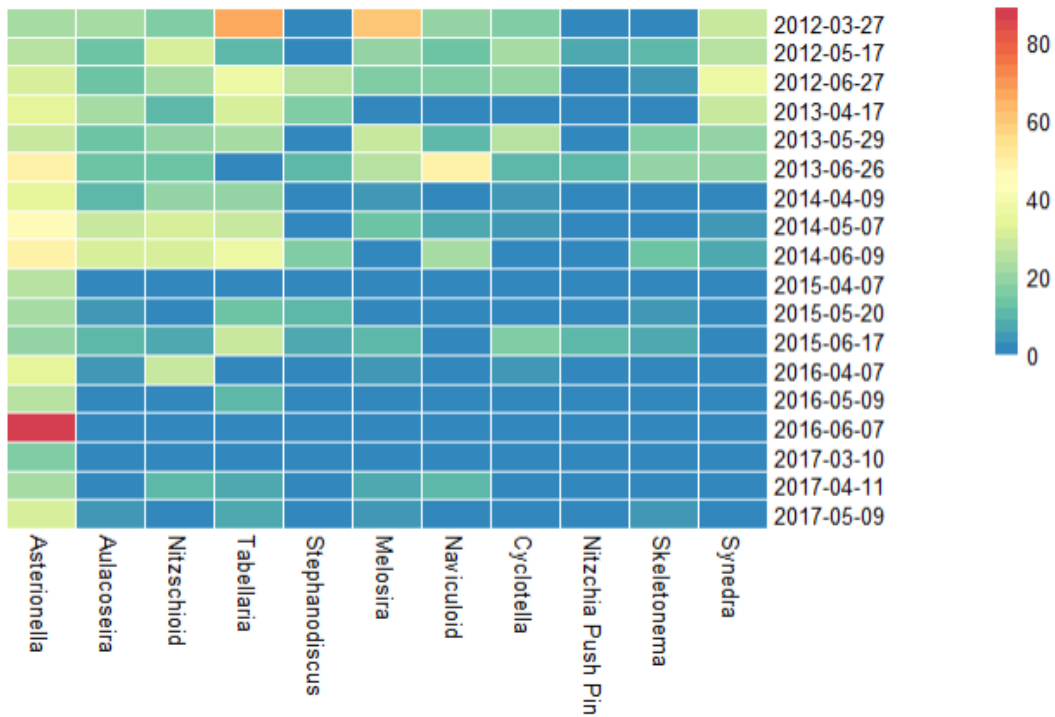
Figure 86. Heat map showing the prevalence of infection (color scale bar) among different diatom taxa (indicated below the grid) at Whites Island from 2012 to 2017. Low prevalence is represented by blue and high prevalence by yellow/orange/red. Prevalence refers to the percent of the population showing evidence of infection by chytrid parasites.
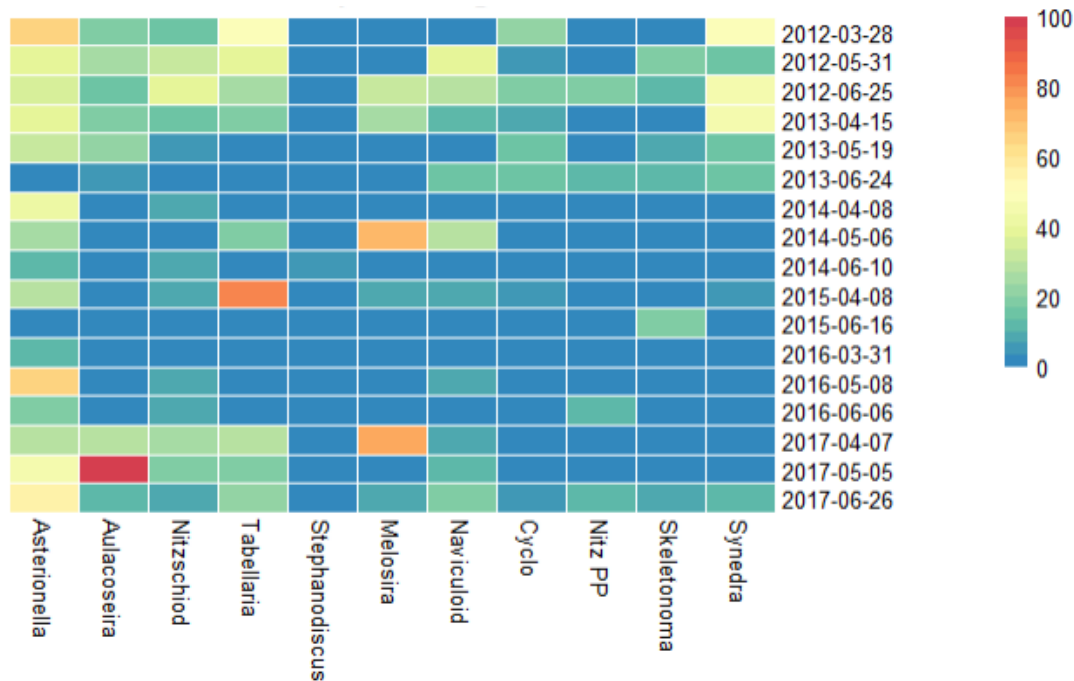
Figure 87. Heat map showing the prevalence of infection (color scale bar) among different diatom taxa (indicated below the grid) at Campbell Slough from 2012 to 2017. Low prevalence is represented by blue and high prevalence by yellow/orange/red. Prevalence refers to the percent of the population showing evidence of infection by chytrid parasites.

3.4.2 Spring Zooplankton Assemblages

The zooplankton assemblages were dominated by a variety of rotifers in the spring (March through May) at all sites in 2018, with higher densities at Campbell Slough than the other sites (Figure 88). Similar to previous years, zooplankton assemblages at Ilwaco were dominated by copepods, with spring samples heavily dominated by nauplii. The zooplankton assemblages at both Campbell Slough and Franz Lake Slough were dominated by copepod nauplii and rotifers in March, particularly *Keratella* spp. and *Brachionus* spp. Aside from nauplii, copepods and cladocerans were present at similar densities, 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, which were present at much lower densities than the copepods or rotifers.

Zooplankton assemblages at Welch and Whites Island differed from those at Campbell Slough and Franz Lake Slough but were similar to each other. Welch and Whites shared similar rotifer assemblages, which were dominated by *Keratella* spp. and *Gastropus* spp. in the spring; copepod assemblages were dominated by cyclopoid taxa. In contrast, the zooplankton assemblages at Ilwaco were dominated by rotifers (*Notholca* sp.), copepod nauplii, and harpacticoid copepods.

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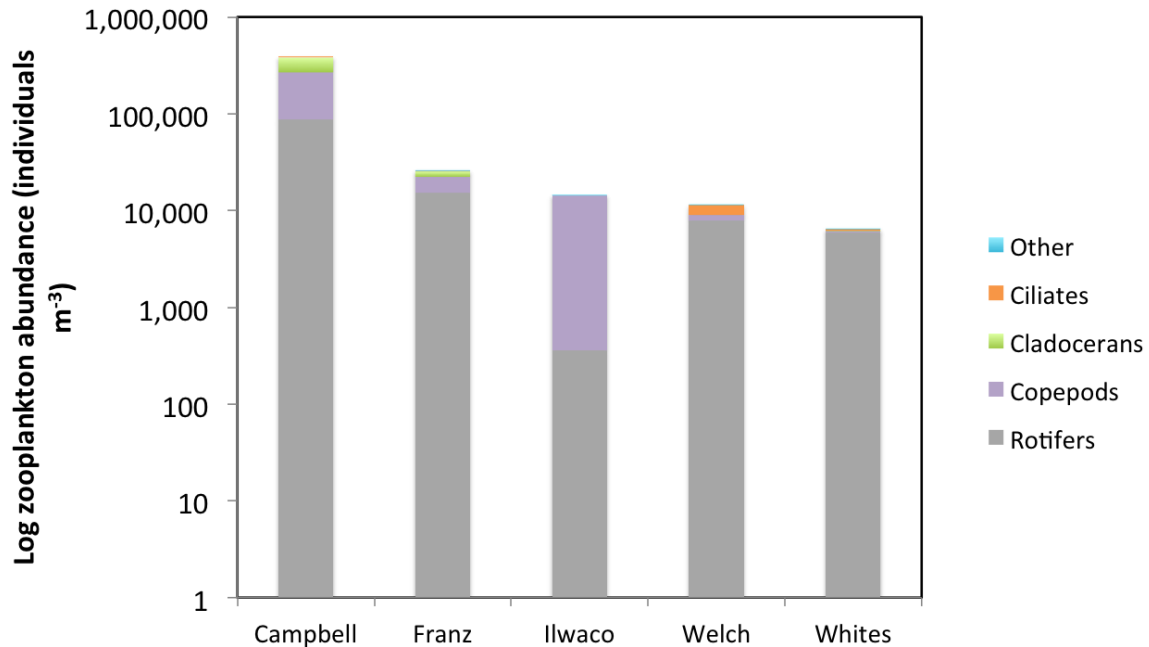
Figure 88. Spring zooplankton assemblage composition at five off-channel trends sites.

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 $\delta^{13}C$ values between -24 and -34‰, seaweeds, and marine plants between -6 and -19‰, and algae and lichens -12 to -23‰. According to Cloern (2002), $\delta^{13}C$ values for freshwater phytoplankton are between -29.5 and -27.5‰, which overlaps with emergent vascular plant matter, which typically has $\delta^{13}C$ values ranging from -28.1 to -27.2‰. Sediments and soils tend to have heavier isotopic signatures, while runoff can have lighter values. 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. It is interesting that the signatures of $\delta^{13}C$-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).

Nitrogen isotopic signatures determined from particulate organic matter were more variable compared to carbon (Figure 89). The very light value for nitrogen isotopes observed at Franz Lake Slough in June (2 ‰) coincided with high biomass of cyanobacteria; light $\delta^{15}N$ values are indicative of atmospheric fixation of nitrogen gas (N_2), which cyanobacteria carry out under conditions of low availability of other N sources (e.g., nitrate).

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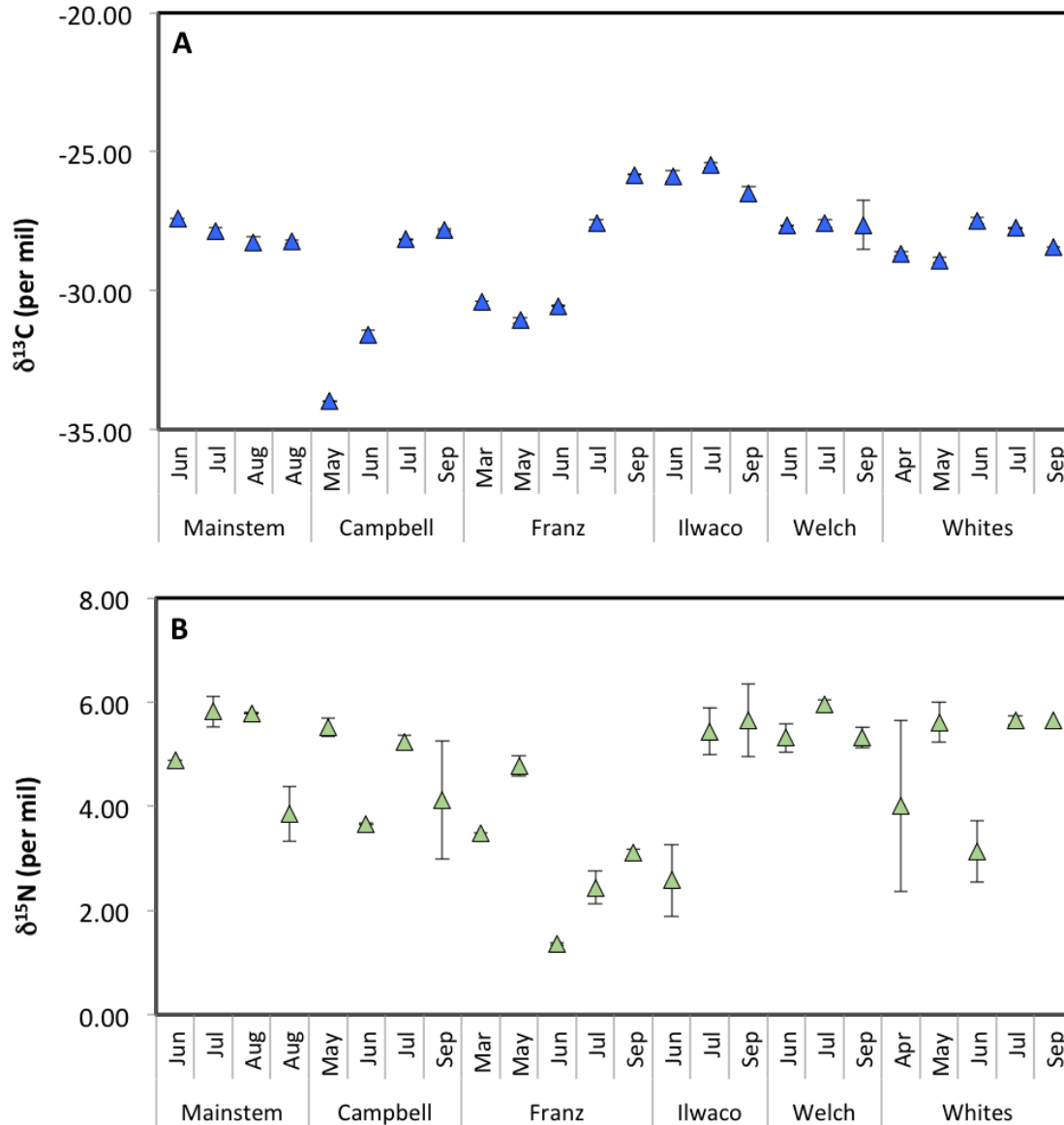
Figure 89. Carbon (A) and nitrogen (B) isotope data collected from particulate organic matter (POM) at the five off-channel trends sites as well as the river mainstem in 2017. Data from 2018 were not available at the time of this report.

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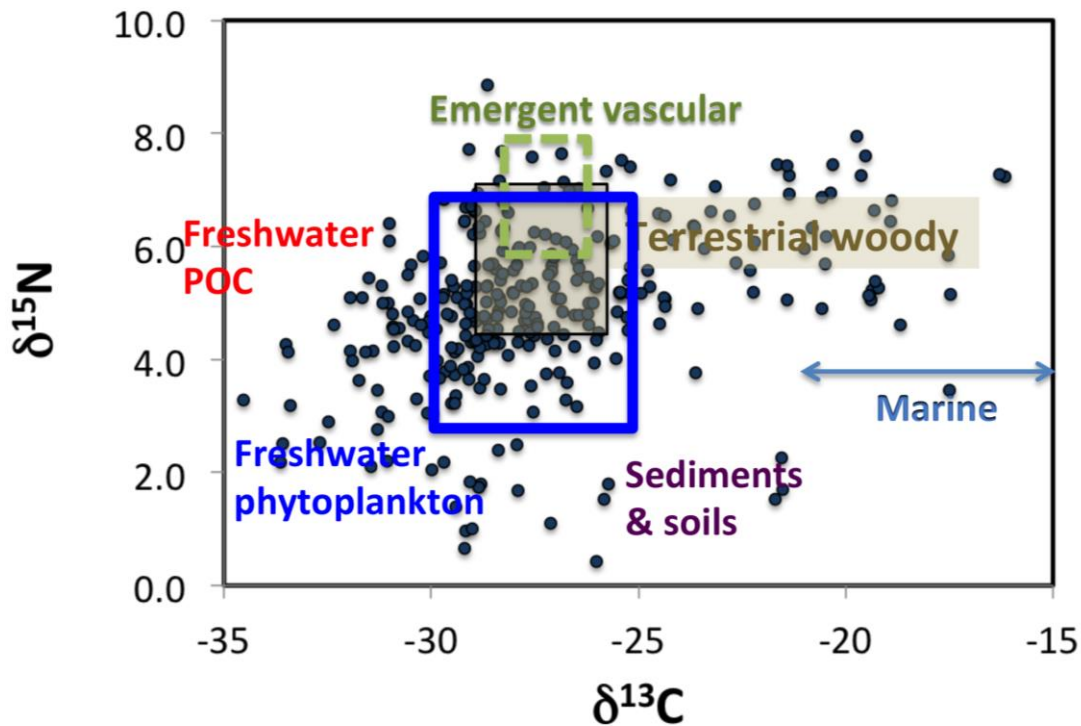
Figure 90. Plot of particulate organic matter (POM) data from off-channel trends sites (circles) in isospace; typical isotopic signature ranges for different organic matter sources are shown (derived from Cloern, 2002). The contribution of various sources to measured POM in the lower Columbia is evident 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 90) 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 Associated with Potential Salmon Prey*

When isotope signatures were used in a Bayesian mixing model to determine the contributions of various organic matter sources to the diet of chironomids, the wide range of values associated with not only POM but also periphyton (Figure 91) led to large uncertainties in the estimated dietary proportions (Figure 94).
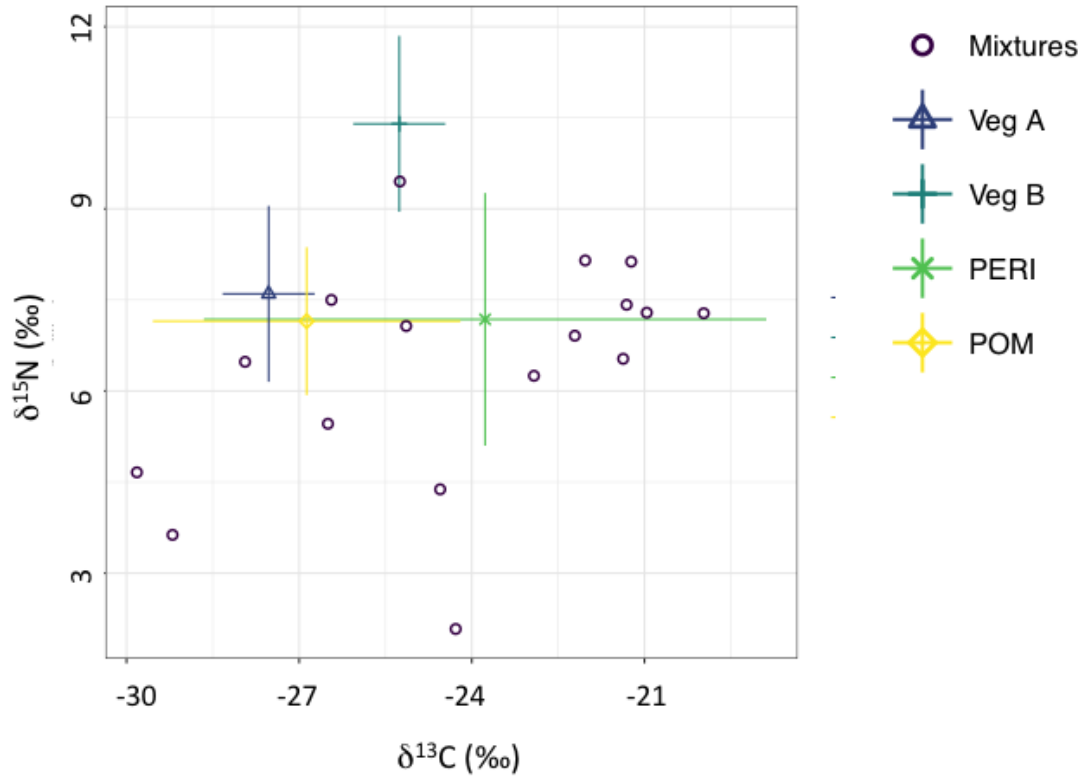
Figure 91. Isospace plot showing stable isotope signature of chironomids ("mixtures") compared to vascular plant matter with heavier δ¹³C and δ¹⁵N (Veg A) and those having light δ¹³C and δ¹⁵N (Veg B) as well as to periphyton (PERI) and particulate organic matter (POM).
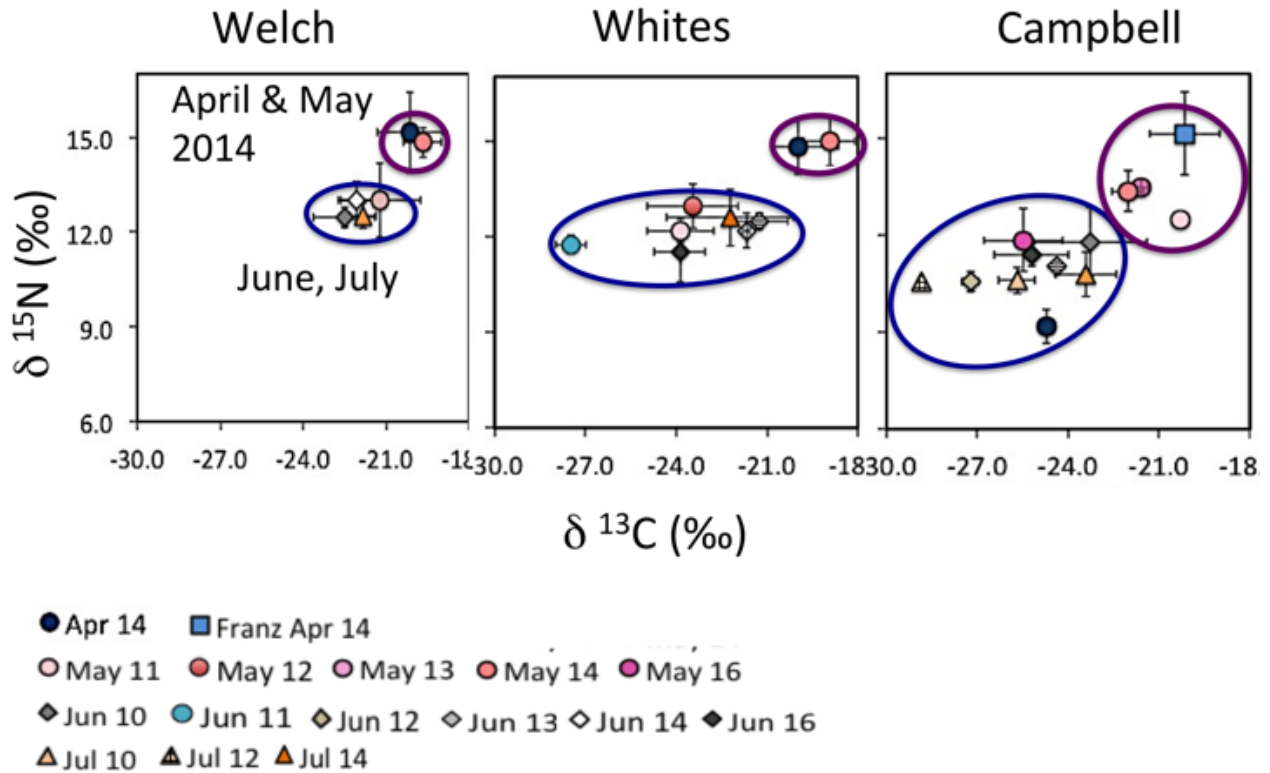
Figure 92. Isospace plots are showing stable isotope signatures (C, N) from muscle tissue in juvenile salmonids collected between 2011 and 2016. Tissues with similar signatures were grouped by eye. Groups with higher δ^{15}N and δ^{13}C were observed in April and May 2014 (Welch and Whites) and mostly May with an observation from April at Franz Lake Slough.
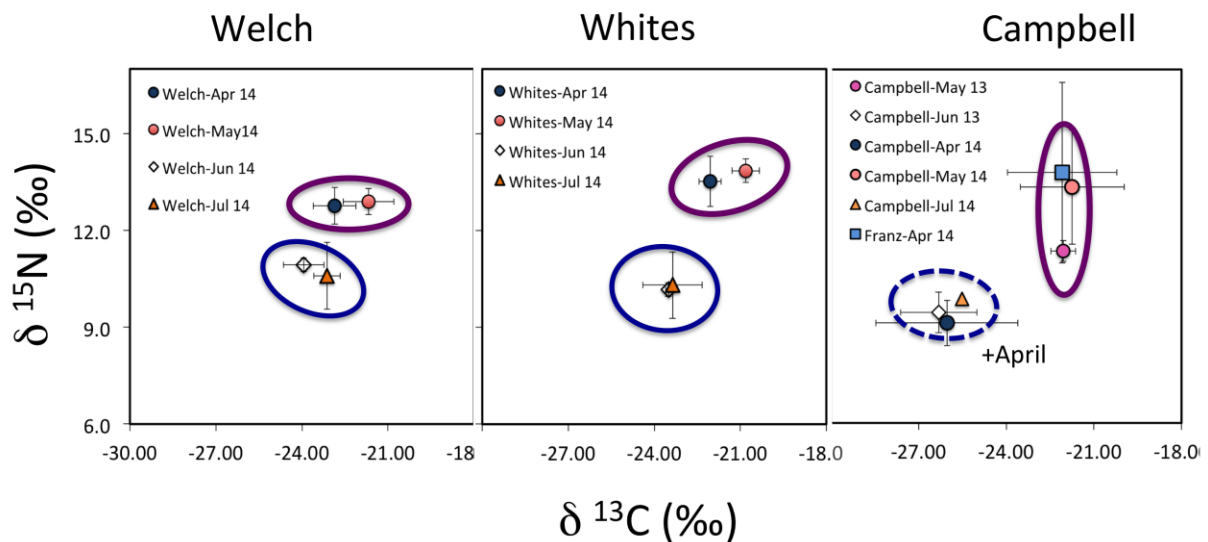
Figure 93. Isospace plots are showing stable isotope signatures associated with fish liver tissues (2013-2014). Grouping of data (by eye) shows differences in δ^{15}N signatures of samples from April and May vs. June and July at Welch Island and Whites Island and in d13C between samples that included April and May (high δ^{13}C and June/July (with an April sample).
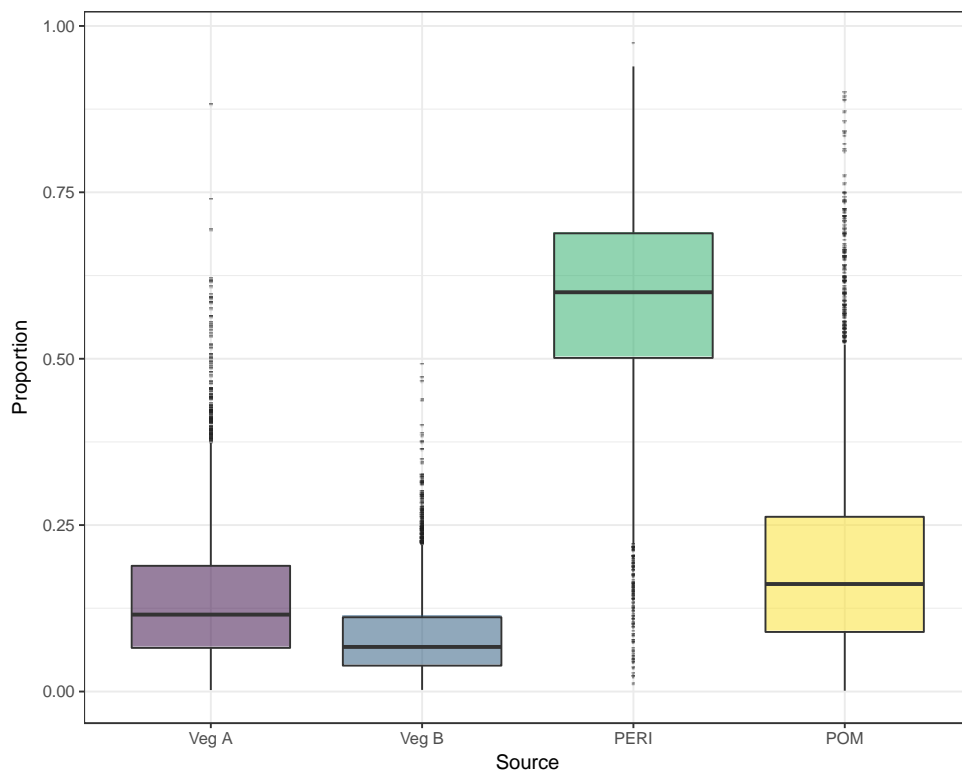
Figure 94. Dietary proportions of various organic matter sources for chironomids estimated by Bayesian stable isotope mixing model. Sources include vegetation (Veg A and Veg B), periphyton (PERI), and particulate organic matter (POM).

According to the estimates produced by the Bayesian mixing model, it is likely that chironomids were consumers of organic matter derived from periphyton (Figure 94), although the widespread in isotopic signatures of sources renders it impossible to derive a robust estimate of the exact proportion until more data are collected to more completely characterize source variability in space and time.
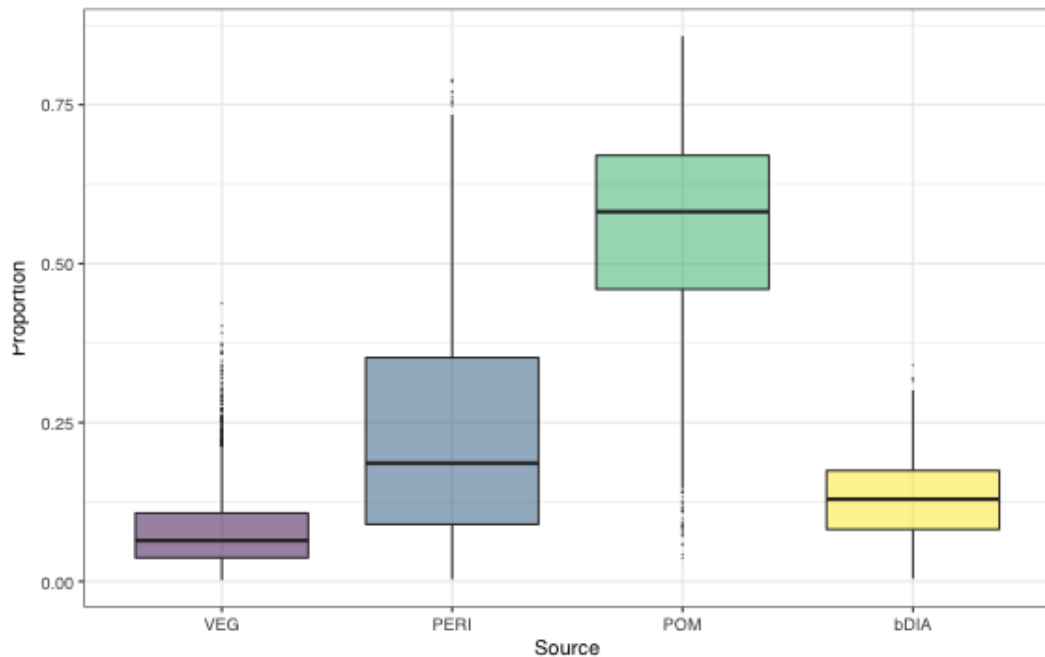
Figure 95. Dietary proportions of various organic matter sources for amphipods estimated by Bayesian stable isotope mixing model. Sources include vegetation (Veg A and Veg B), periphyton (PERI), and particulate organic matter (POM).

3.4.3.2   *Unmarked vs. Hatchery Reared Fish*

Muscle tissue from marked (hatchery raised) and unmarked (a mixture of unmarked hatchery fish and wild fish) juvenile Chinook salmon differed in average isotopic signatures of carbon and nitrogen, with a greater spread in the data in the unmarked fish (Figure 96Figure 95**Error! Reference source not found.**). Carbon was heavier while nitrogen was lighter in marked fish relative to unmarked fish. This resulted in differences in the estimated dietary proportions from various organic matter sources (Figure 96). Because of the large spread in isotopic values measured in organic matter sources, there was large uncertainty in estimates of dietary proportions, with amphipods, chironomids, copepods, and "other" all accounting for >20% of the total in the unmarked fish. In marked, or hatchery reared fish, the largest contribution came from "other"; it is not known what constitutes this category, but the data indicate that there is a food source that is not being accounted for in the source inputs to the model.
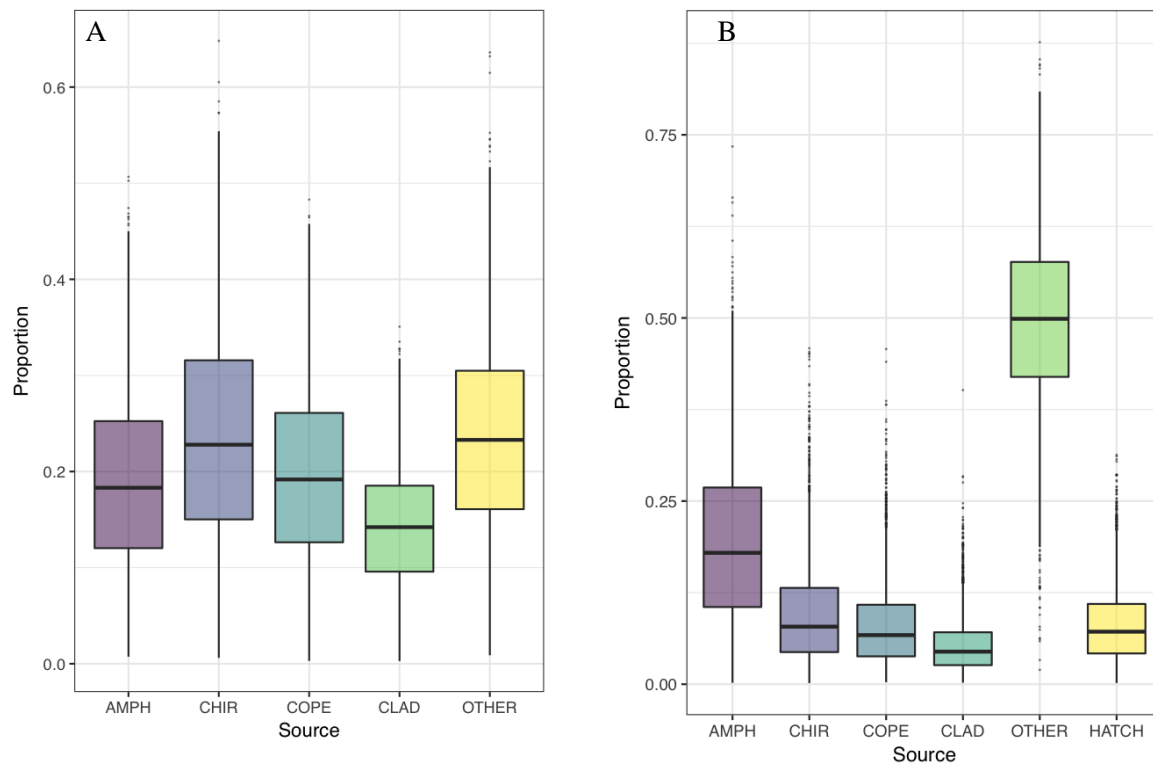
Figure 96. Dietary proportions of different organic matter sources supporting juvenile Chinook salmon. A: unmarked; B: marked. Sources include: amphipods (AMPH), chironomids (CHIR), copepods (COPE), cladocerans (CLAD), hatchery food (HATCH), and "other" sources (includes polychaetes, oligochaetes, and other insects).

3.5 Macroinvertebrates

3.5.1 Salmon Prey Availability

Salmon prey availability data collected in 2018 are still under analysis, the 2017 data is reported below.

3.5.1.1 *Benthic*

The numeric composition of benthic core samples from all sites were consistently dominated by oligochaete worms (Figure 97). Nematode worms and dipterans are 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, bivalves, hirudinea (leeches), and isopods, made a relatively large contribution to the gravimetric composition when they were present (Figure 97)

Amphipods were relatively abundant at Ilwaco Slough (Figure 97, **Figure 98**). Amphipod abundance declined upstream from Ilwaco Slough, with low densities at Welch Island and White Island and almost none at Campbell Slough (zero recorded) and Franz Lake (one individual recorded). Dipterans, including Chironomidae, were collected from all sites between April and June (when sampled), except in April 2016 at Campbell Slough (**Figure 98**). Highest 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.
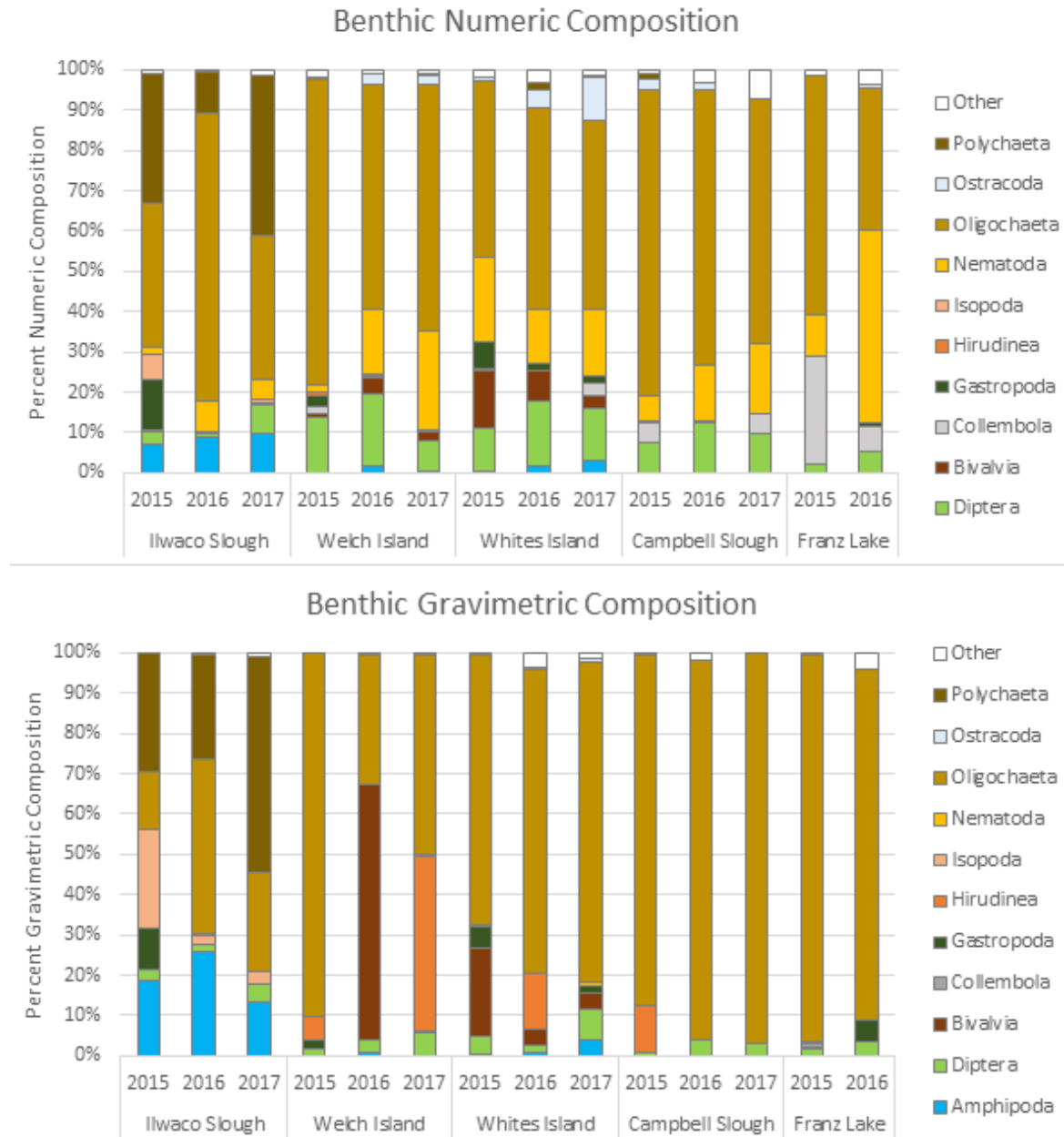
Figure 97. Percent numeric (above) and gravimetric (below) composition of benthic core samples collected between April-July, arranged by site and year.
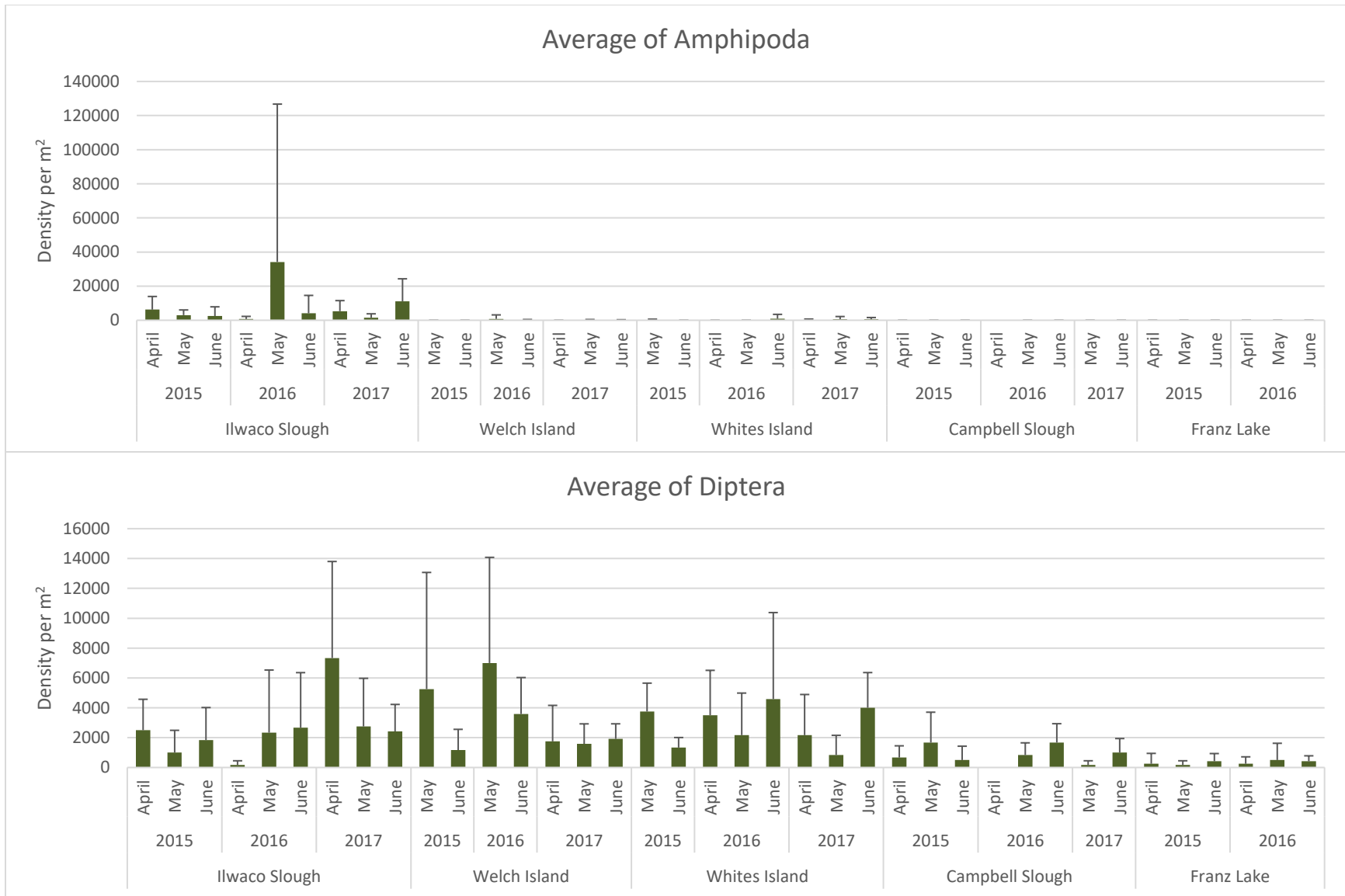
Figure 98. Average density (count per m^2) of major salmonid prey items (above: Amphipoda; below: Diptera) collected by benthic core, arranged by site, year, and month. Error bars represent the 95% confidence interval.

3.5.1.2 *Neuston*

Neuston samples were composed of a diverse array of benthic/epibenthic, terrestrial riparian, and planktonic taxa (Figure 99). 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 100). Diptera, including Chironomidae, were collected at all sites between April and June in 2015-2017 (Figure 100). The average density of dipterans was relatively low at Ilwaco Slough and Welch Island. The highest dipteran densities were observed at Whites Island, Campbell Slough, and Franz Lake in the emergent vegetation. Counts were variable, however, and we did not observe a clear temporal trend in dipteran abundance.

The density of Cladocera and Copepoda peaked in May 2017 at Campbell Slough from the emergent vegetation (Figure 101). 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 99. Percent numeric (above) and gravimetric (below) composition of neuston samples collected between April-June, arranged by habitat, site and year.
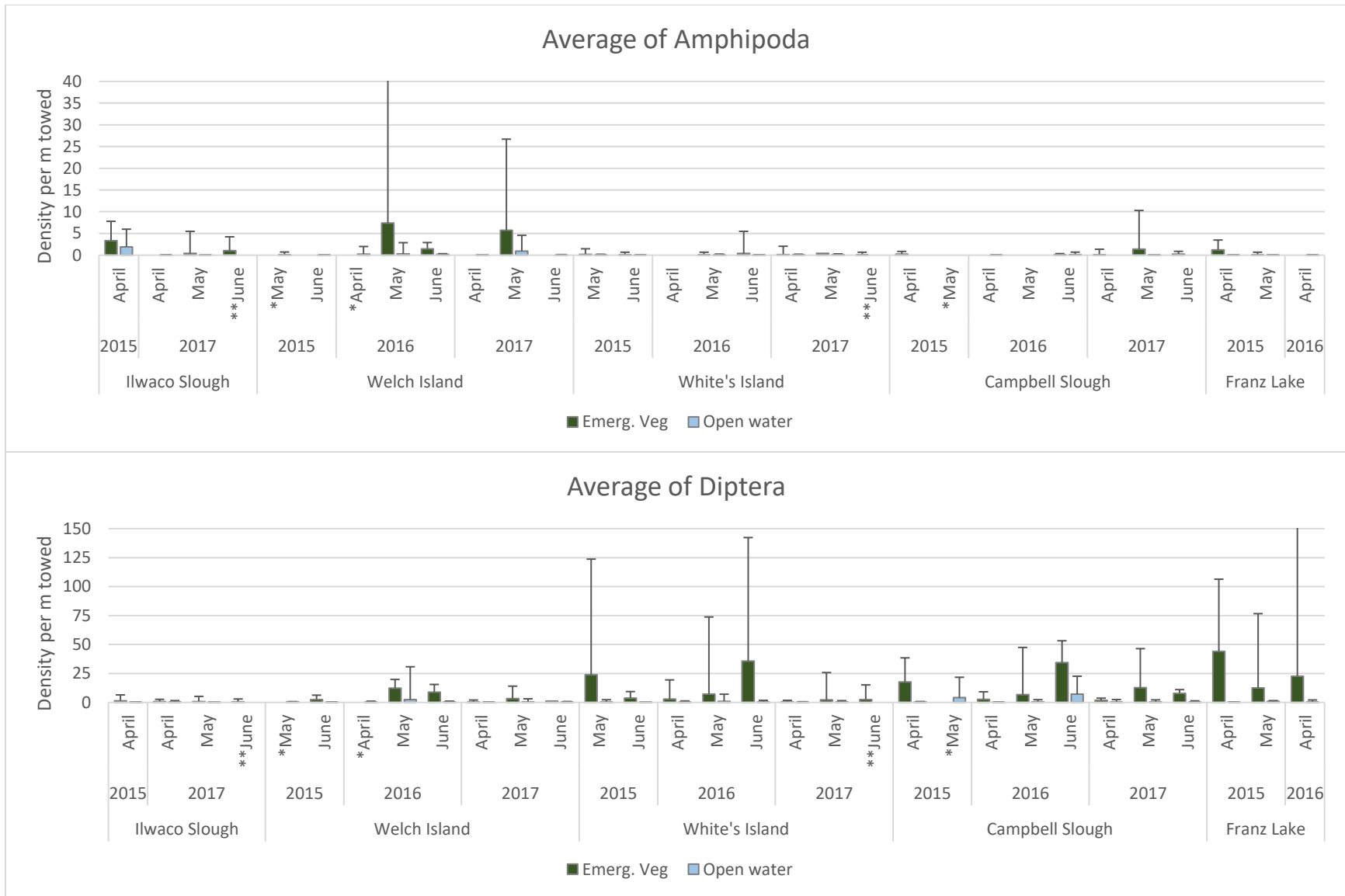
Figure 100. Average density (count per meter towed) of major salmonid prey items (above: Amphipoda; below: Diptera) collected by neuston tow, arranged by site, year, and month. Error bars represent the 95% confidence interval. *Emergent vegetation, **Open water not sampled.
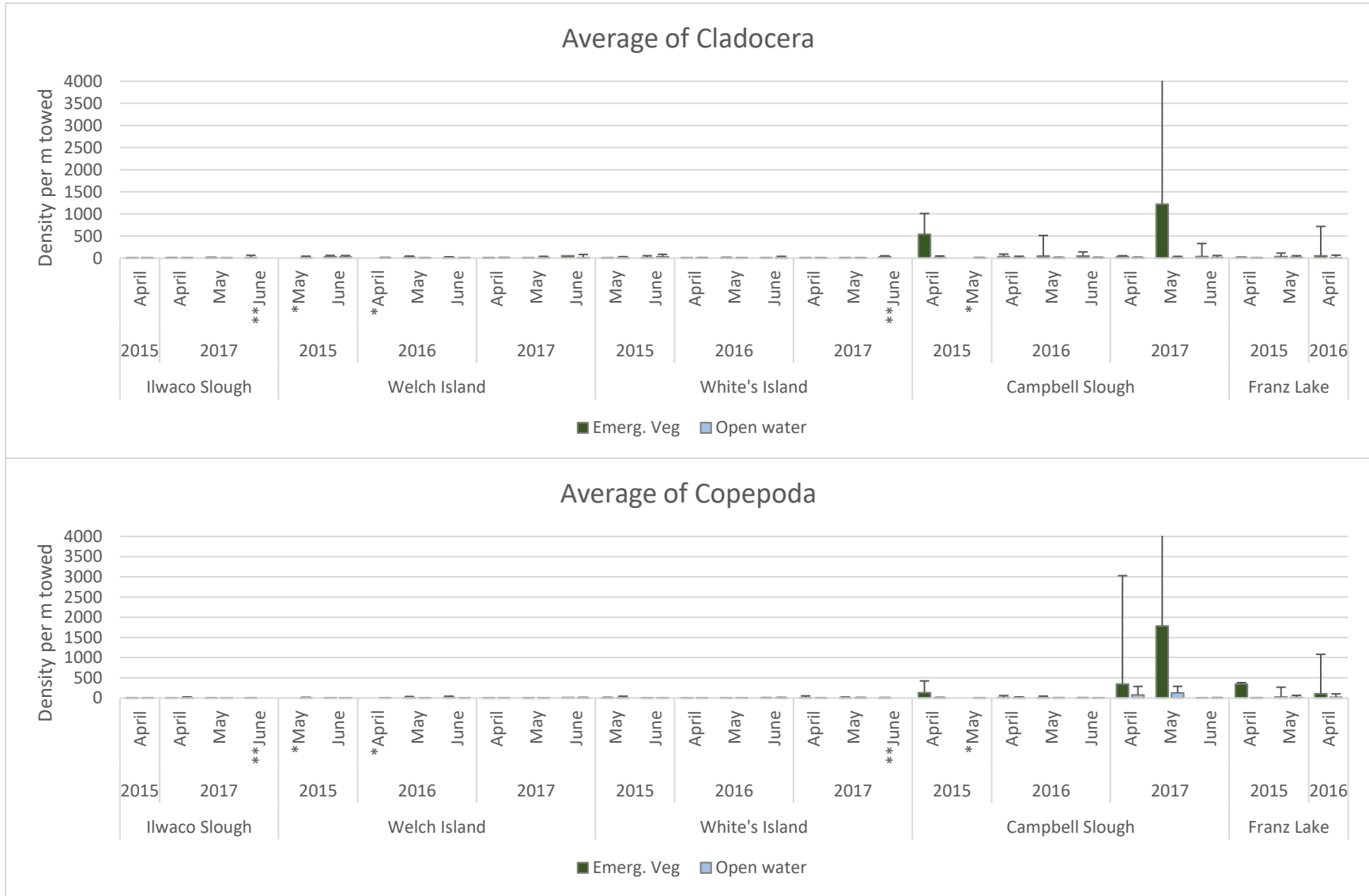
Figure 101. Average density (count per meter towed) of major salmonid prey items (above: Cladocera; below: Copepoda) collected by neuston tow, arranged by site, year, and month. Error bars represent the 95% confidence interval. *Emergent vegetation, **Open water not sample

3.5.2 Salmon Diet

Salmon diet data collected in 2018 are still under analysis, 2017 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 102 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 103). While this trend occurs over most sample years (2008-2013, 2015-2017), 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 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. 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 103). The large increase in cladocerans in 2017 Campbell Slough diets was observed in both May and June over all fish size classes (Figure 104).

Instantaneous and energy ration in 2017 was typically higher at Welch Island and Whites Island where juvenile Chinook salmon consumed dipterans and amphipods and lower at Campbell Slough where fish primarily consumed cladocerans (Figure 105). A comparison of the instantaneous ration (IR) and energy ration (ER) of fish collected in 2017 to those collected in previous sample years within the same site Figure 106. Shows that IR at Welch Island was higher overall in 2017 relative to previous years, and similar to previous years at Whites Island and Campbell Slough. Similarly, ER at Welch Island was higher overall in 2017 relative to previous years, and similar to previous years at Whites Island. In 2017 energy ration at Campbell Slough was similar to or lower than in previous years.

Combining measures of 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 1Figure 107 provides a graphic representation of this, plotting mean J_M against mean ER for all fish sampled monthly between 2008-2013, and 2015-2017, arranged by size (fork length) bins of fish. 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 was low for the smallest fish (30-59mm length range) at all sites. The highest observation of J_M occurred at Whites Island during the summer when the fish also had relatively high rates of energy consumption. Maintenance metabolism for medium sized juvenile Chinook salmon (60-79mm length range) was also highest at Whites Island during the summer, but again energy consumption was above average at that time. Medium sized fish from Franz Lake that were collected in spring over four years had below average energy assimilation, but also below average metabolic costs. Large sized

juvenile Chinook salmon (80-99mm length range) from Campbell Slough experienced a wide range of growing conditions. Some of these Campbell Slough fish collected in spring had high energy assimilation and low metabolic costs, reflecting favorable growing conditions. However, at other times, Campbell Slough fish collected in spring had high metabolic costs coupled with low energy assimilation, reflecting poor growing conditions. Large fish were collected from Franz Lake in the spring months. Like the other size classes, they had relatively low metabolic costs but did not consume energy dense prey. As shown for medium and small sized fish, some of the large fish from Whites Island and Welch Island in summer months may be experiencing a tradeoff of relatively higher temperatures (and metabolic needs) coinciding with peak emergence of energy dense prey.
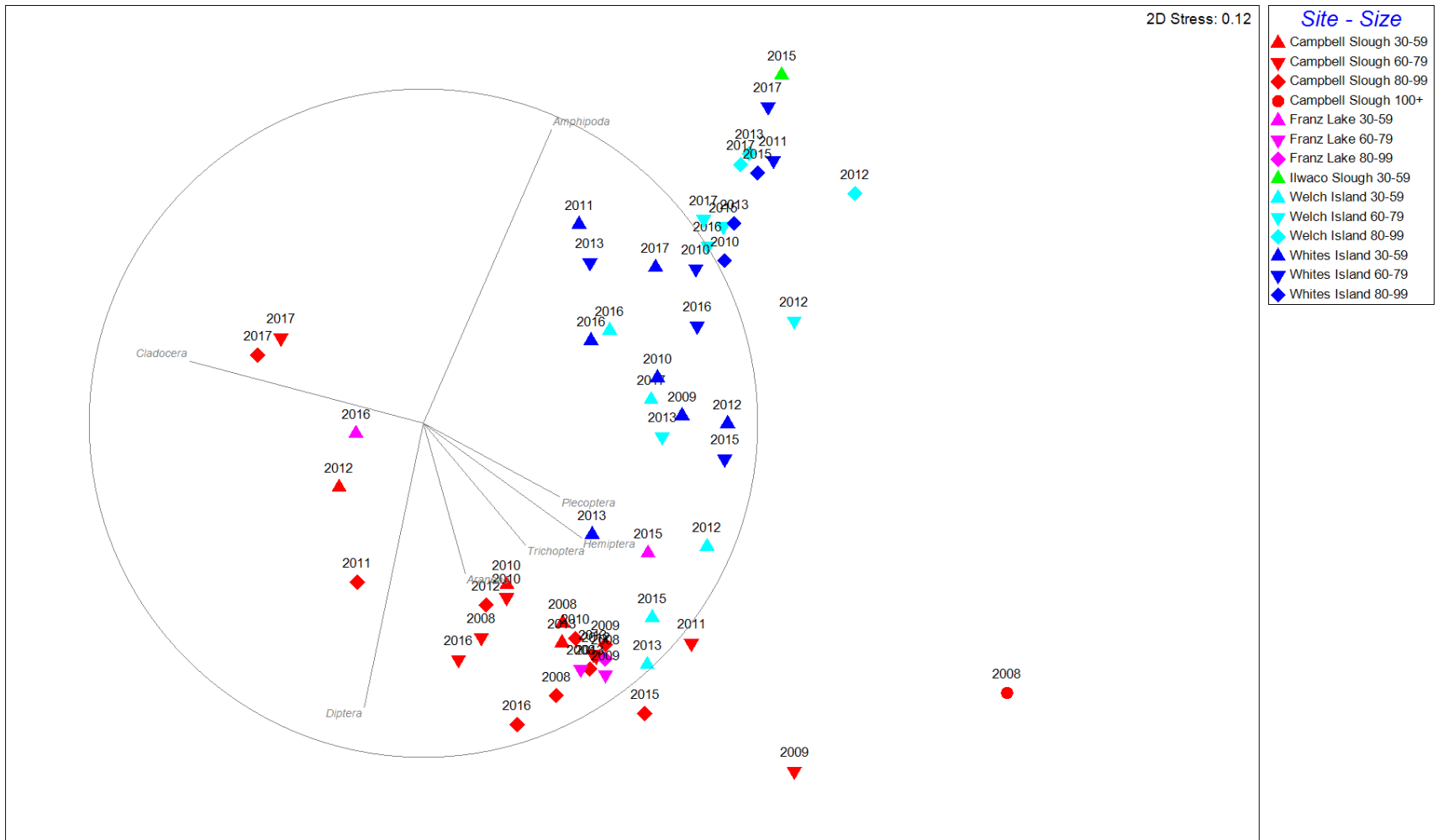
Figure 102. Two-dimensional nonmetric multidimensional scaling (NMDS) plot based on Bray-Curtis similarities between transformed percent IRI of major prey in juvenile Chinook diets sampled between 2008 and 2017. Each point represents all fish collected between April and June within the defined size class (fish fork length in mm), site, and year. Significant correlation with variables (Pearson R > 0.45) are represented as gray vectors.
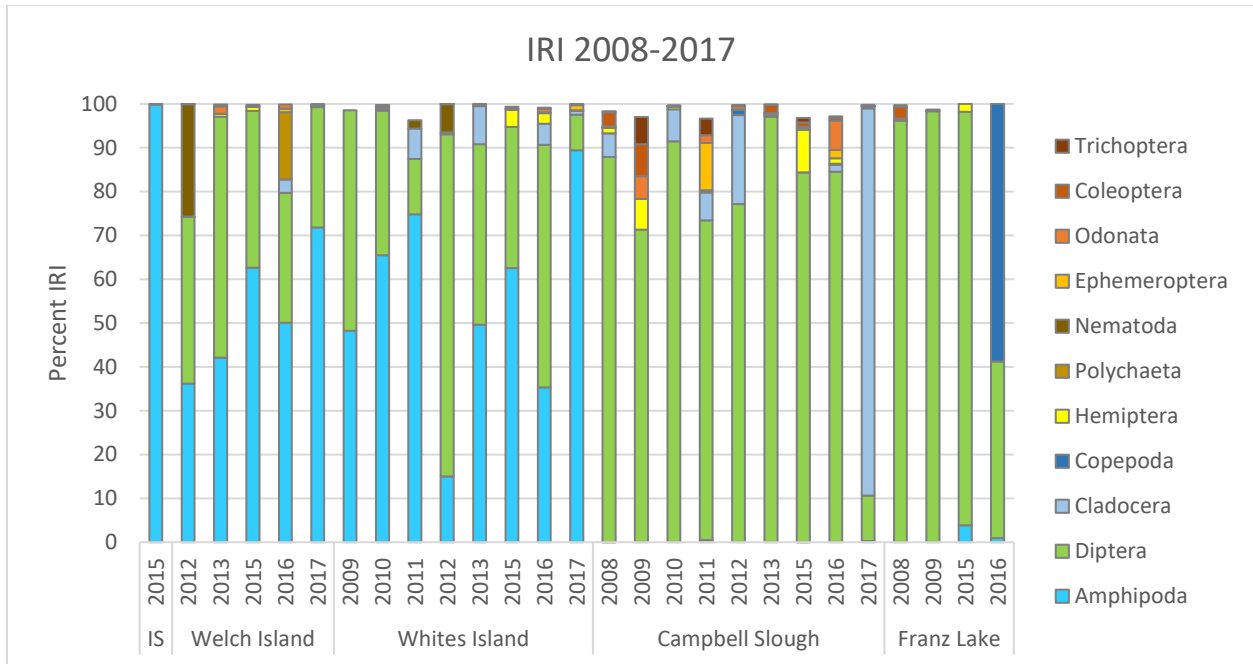
Figure 103. Percent IRI by site and year for juvenile Chinook salmon collected between April and June.
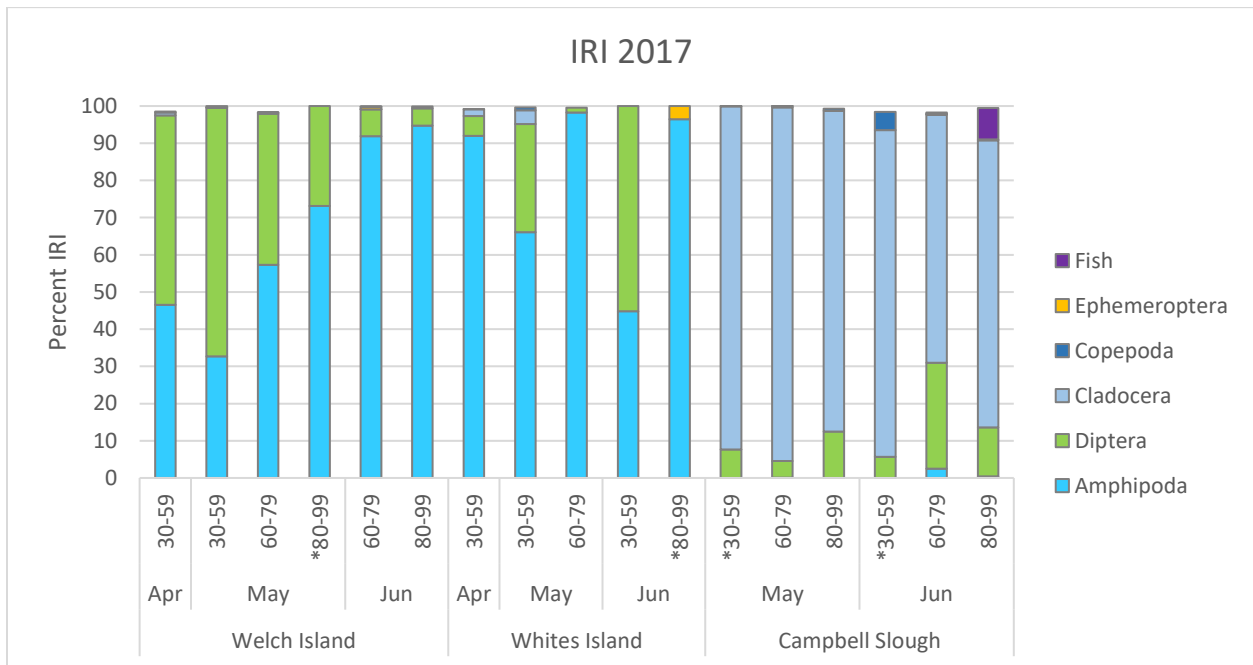
Figure 104. Percent IRI in 2017 by site, month, and size class (fish fork length in mm) for juvenile Chinook salmon collected between April and June. *Only one fish collected. All other sample sizes range between 4-24 individuals.
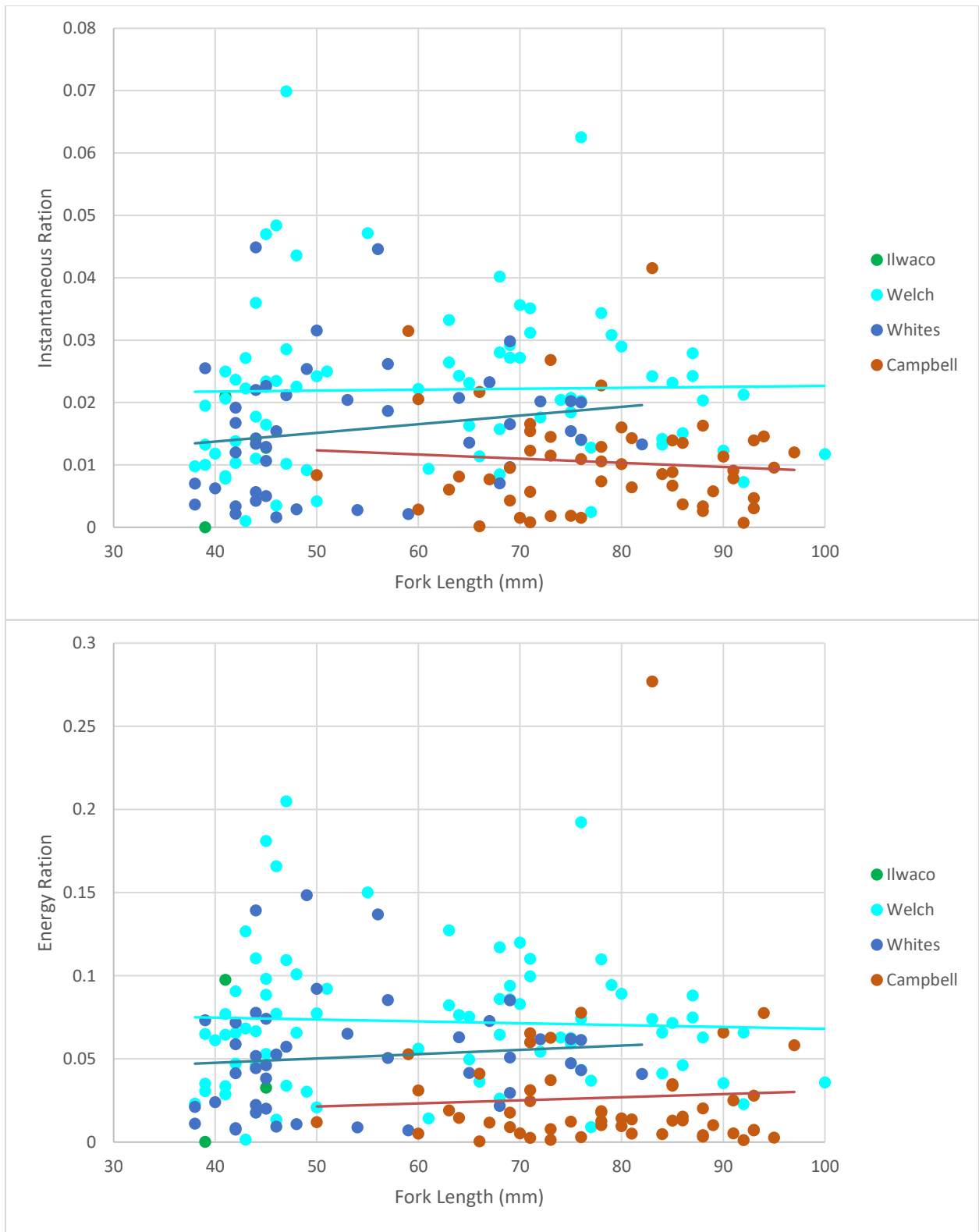
Figure 105. Instantaneous ration (above) and energy ration (below) by fish fork length for individual juvenile Chinook salmon collected in 2017. Solid line is the linear trendline. Trendline was not drawn for Ilwaco Slough where only 3 fish were collected.

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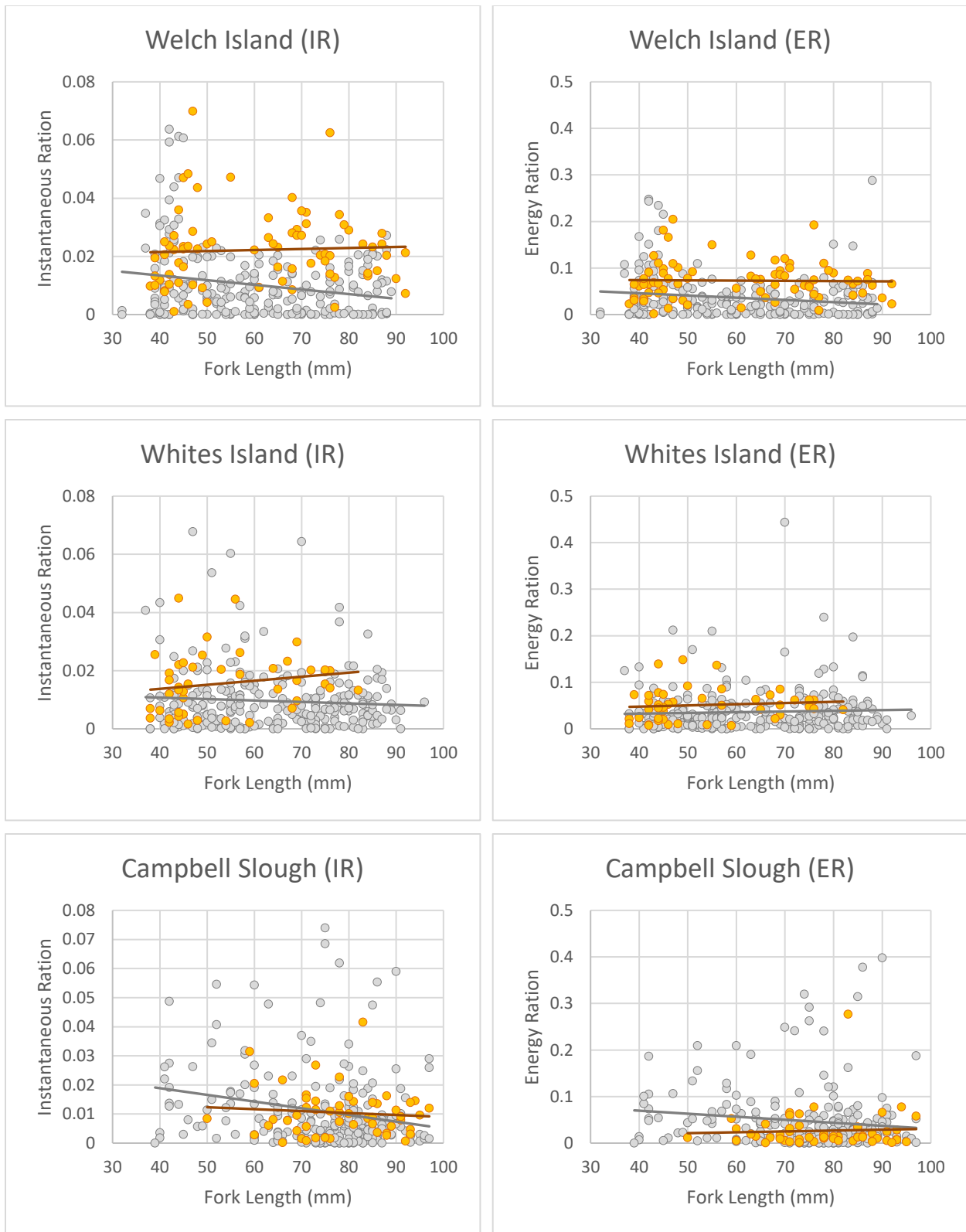
Figure 106. Instantaneous ration (IR) and energy ration (ER) by fish fork length, for juvenile Chinook salmon collected in 2017 (orange) and between 2008-2013, 2015-2016 (gray). The solid line is the linear trendline.
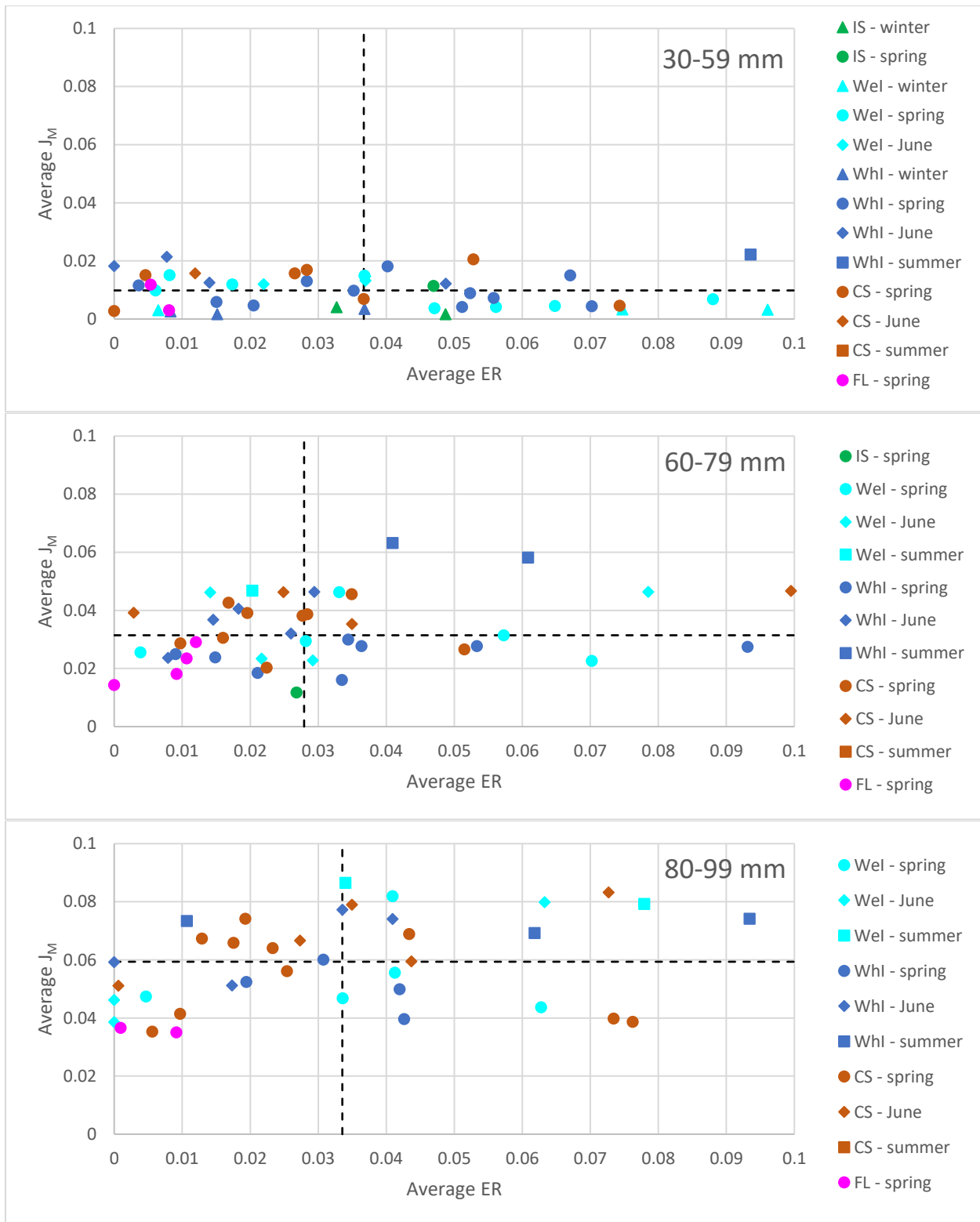
Figure 107. Quadrant charts of monthly average maintenance metabolism (J_M) and average energy ration (ER), arranged by length bin of juvenile Chinook salmon sampled between 2008-2013, and 2015-2017. Symbols separated by winter (February, March), spring (April, May), early summer (June), and summer (July, August) months. The dashed line is the 50th percentile of J_M and ER.

3.6 Fish

3.6.1 Fish Community Composition

In 2018, fish communities were sampled at five trend sites (Ilwaco Slough, Welch Island, Whites Island, and Campbell Slough), on a monthly basis February-July and October, Franz Lake was sampled only in March and was discontinued for the remainder of the year due to water levels (too high and/or low) and temperature (too high). Fourteen different families of fishes were present and within those families 32 different categories which included a combination of specific species and/or unidentified species within family/genus categories (Figure 108, and Appendix Table F-2). Chinook salmon, threespine stickleback and banded killifish were the only species that occurred at all five sites. Four salmon species were captured, with Chinook the most common and often the most abundant species, followed by Chum which were captured at four sites but were not always numerically dominant. Campbell Slough had the highest diversity of species (16) followed by Ilwaco Slough(10) and White Island (9). Campbell Slough also had the highest percent of introduced fishes in the total catch (75.3%) followed by Welch Island at 6.3%. Ilwaco Slough, the only site with marine influence was dominated by threespine sticklebacks (67.6%), shiner perch (16%) and Pacific Staghorn sculpin (6.6%). Welch and White Islands were dominated by the same species; threespine stickleback (49.4 and 73.3%), Chinook salmon (42.9 and 22.1%) and banded killifish (5.9 and 2.5%). At Campbell Slough, the most abundant species were Chinook salmon (15.8%), banded killifish (10.7), common carp (6.9%), and threespine stickleback (5.8%). The one sampling effort in March at Franz Lake captured primarily peamouth (72.5%) and threespine stickleback (16.5%).

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 sites (global $R = 0.241$, $P = 0.001$). Pairwise tests show that the community structure at all sites were significantly different from other sites except for Welch and Whites Islands ($P = 0.098$). The NMDS plot of community structure by site (Figure 109) 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 Pacific staghorn sculpin. Campbell Slough and Franz Lake tend to be more separated from the lower river sites, yet do not show much overlap emphasizing the differences in fish community structure at both sites.

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 **110**). 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 111), where the values can vary minimally, yet, tend to be the highest at Campbell Slough and Franz Lake. Within sites, the only standout difference was at Campbell Slough where the 2018 species richness and diversity were noticeably lower than any previous years and only slightly higher than 2017. Low species richness at Franz Lake is likely an artifact of the limited sampling (one month)

In 2018, the monthly sampling at each site did not show any distinct pattern of species richness and diversity (Figure 112). The number of species each month ranges from 0-8, with the exception of Campbell Slough in June where the total number of species reached 12. Campbell Slough also had the greatest range in diversity (0.0-1.3) which reflects the higher number of species and the overall total catch of those species.

Non-native fish species occur at all five trend sample sites (all years combined); their presence is highly variable and likely very dependent on water levels and temperature (Figure 113). The highest number of

non-native fishes occur at Campbell Slough where the catch rates have exceeded 50% seven out of the last 11 years, and range from 40-76%. At Campbell Slough, banded killifish, yellow perch, and unidentified juvenile carp comprise the majority of the non-native species. Franz Lake has the second highest numbers of non-native species exceeding 20% six out of the last eight years that sampling occurred and ranging from 6-54%. At Franz Lake, banded killifish, unidentified juvenile carp, and yellow bullhead are the predominant non-native species.

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 114). 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 and has been captured at Campbell Slough and Franz Lake exclusively. In 2018, yellow perch comprised 90.2% of the total number of predatory fish captured.
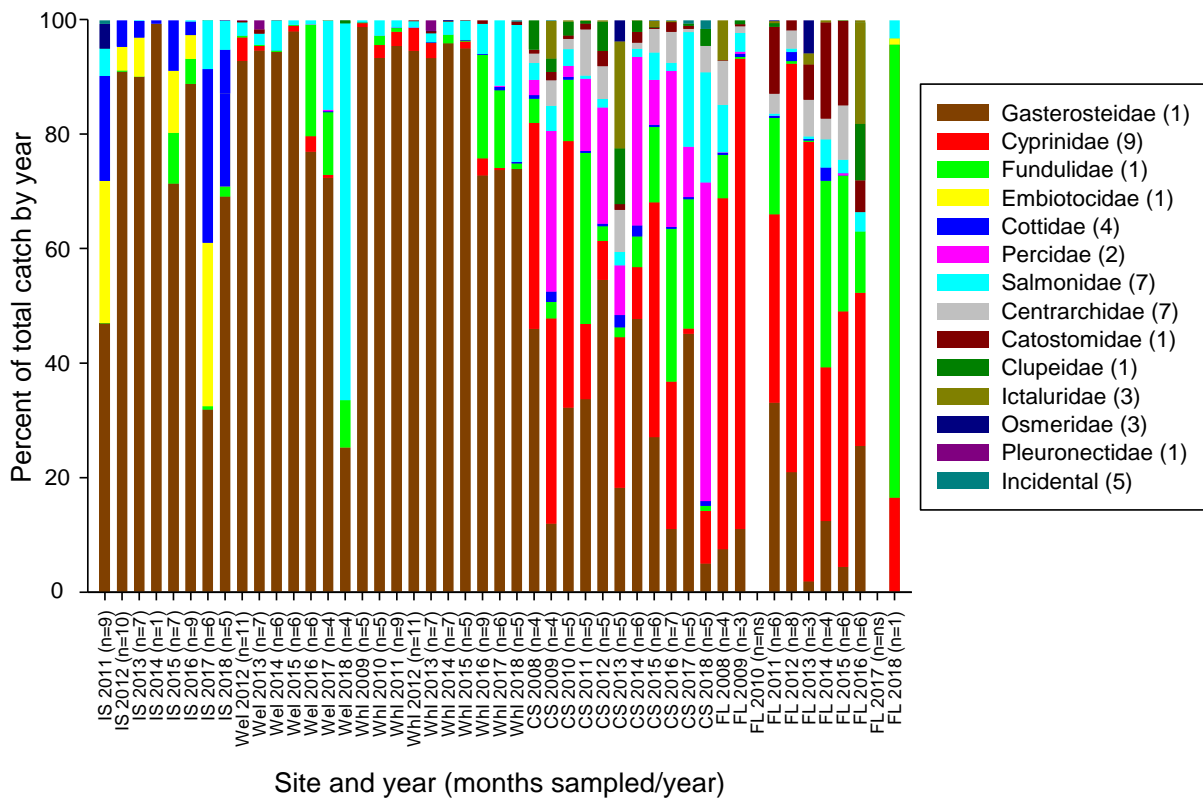
Figure 108. Fish community composition at EMP trend sites sampled from 2008-2018, presented by Family with the number of major taxonomic categories in parentheses in the legend. For each year the total number of sampling months is presented in parentheses in the legend. For each year the total number of sampling months is presented in parentheses on the X axis. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
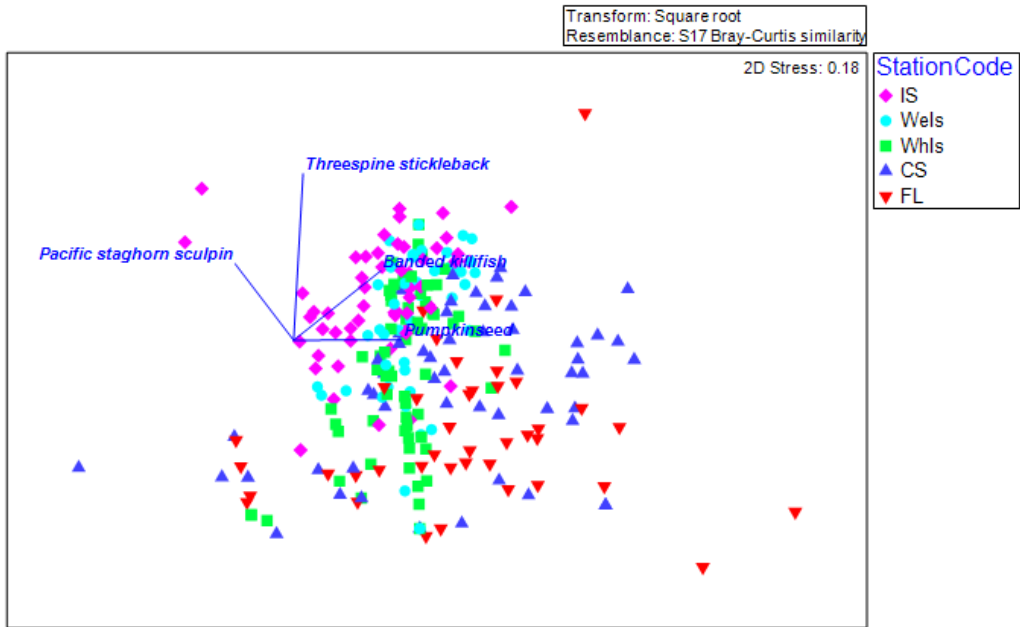
Figure 109. Nonmetric multidimensional scaling (NMDS) plot based on square-root transformed species abundance at five trend sites, 2008-2018. Significant correlation with variables (Pearson R > 0.4) are represented as blue vectors. IS = Ilwaco Slough, WeIs = Welch Island, WhIs = Whites Island, CS = Campbell Slough, FL = Franz Lake.
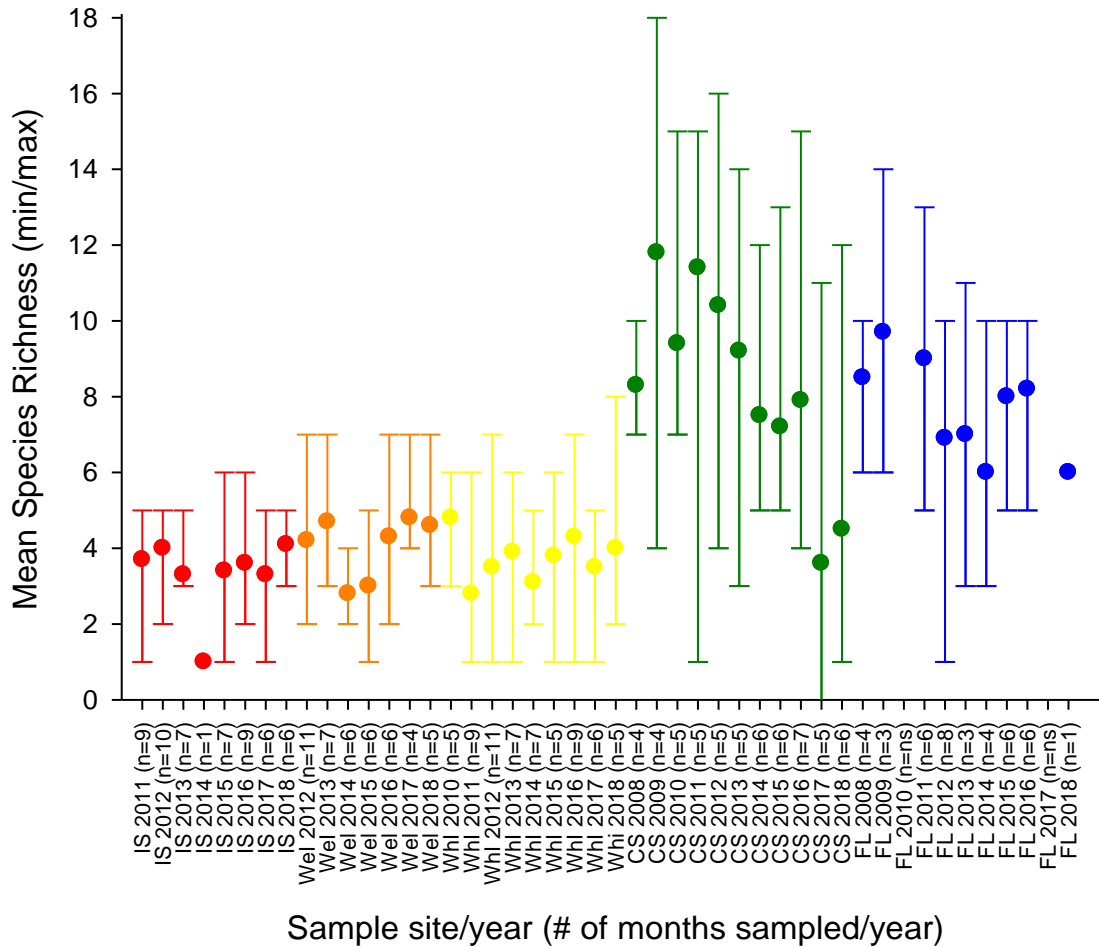
Figure 110. Mean species richness with minimum/maximum ranges for EMP trend sites sampled from 2008-2018. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
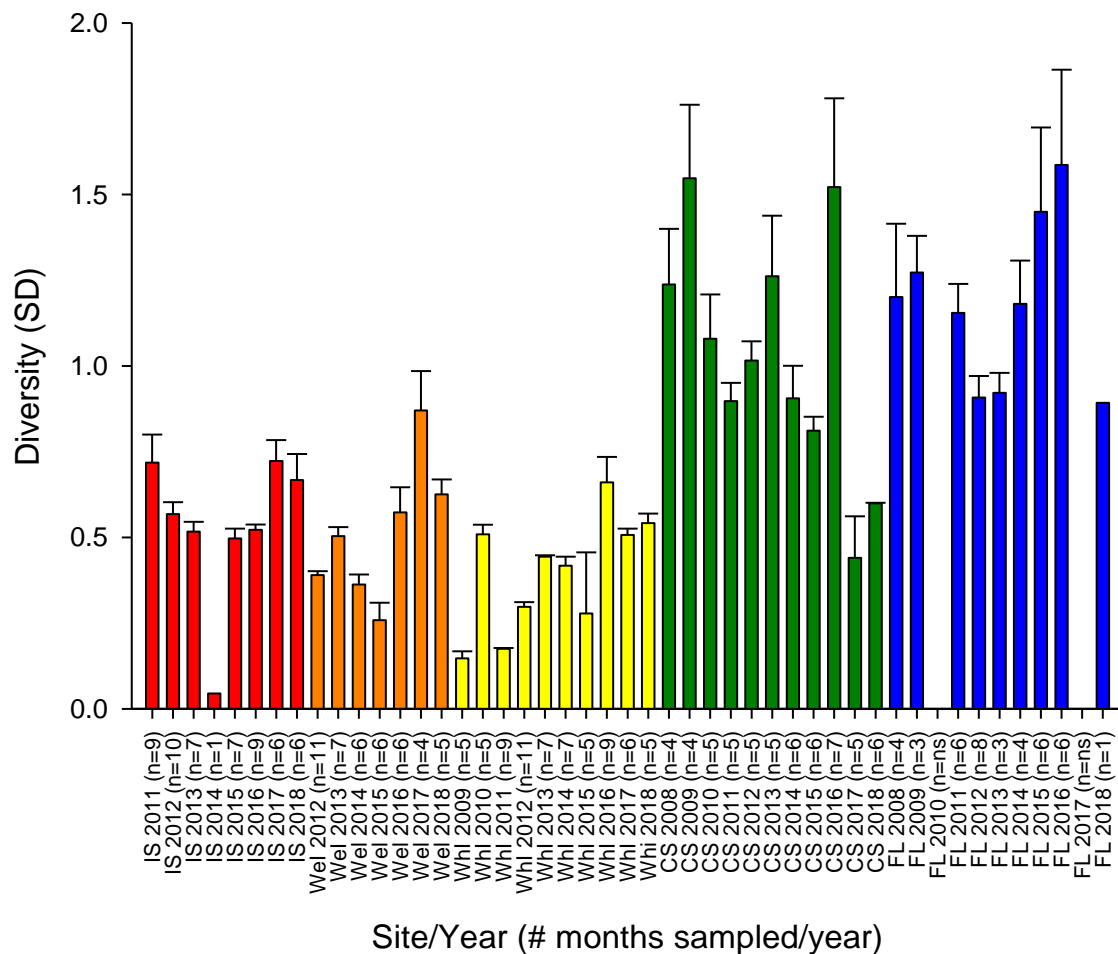
Figure 111. Mean Shannon-Weiner diversity index with standard deviation from EMP trend sites sampled from 2008-2018. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
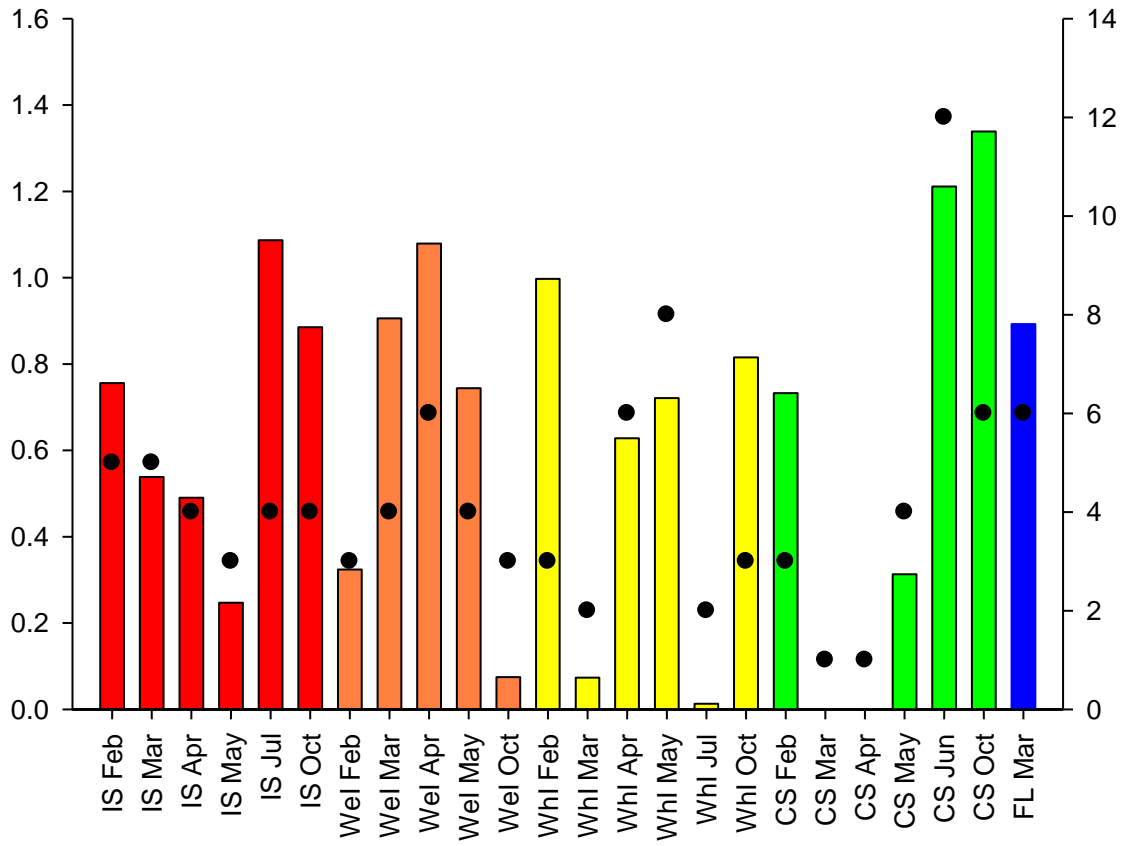
Figure 112. Shannon-Weiner diversity index (bars) and species richness (closed circles) for EMP trend sites sampled monthly in 2018. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough.
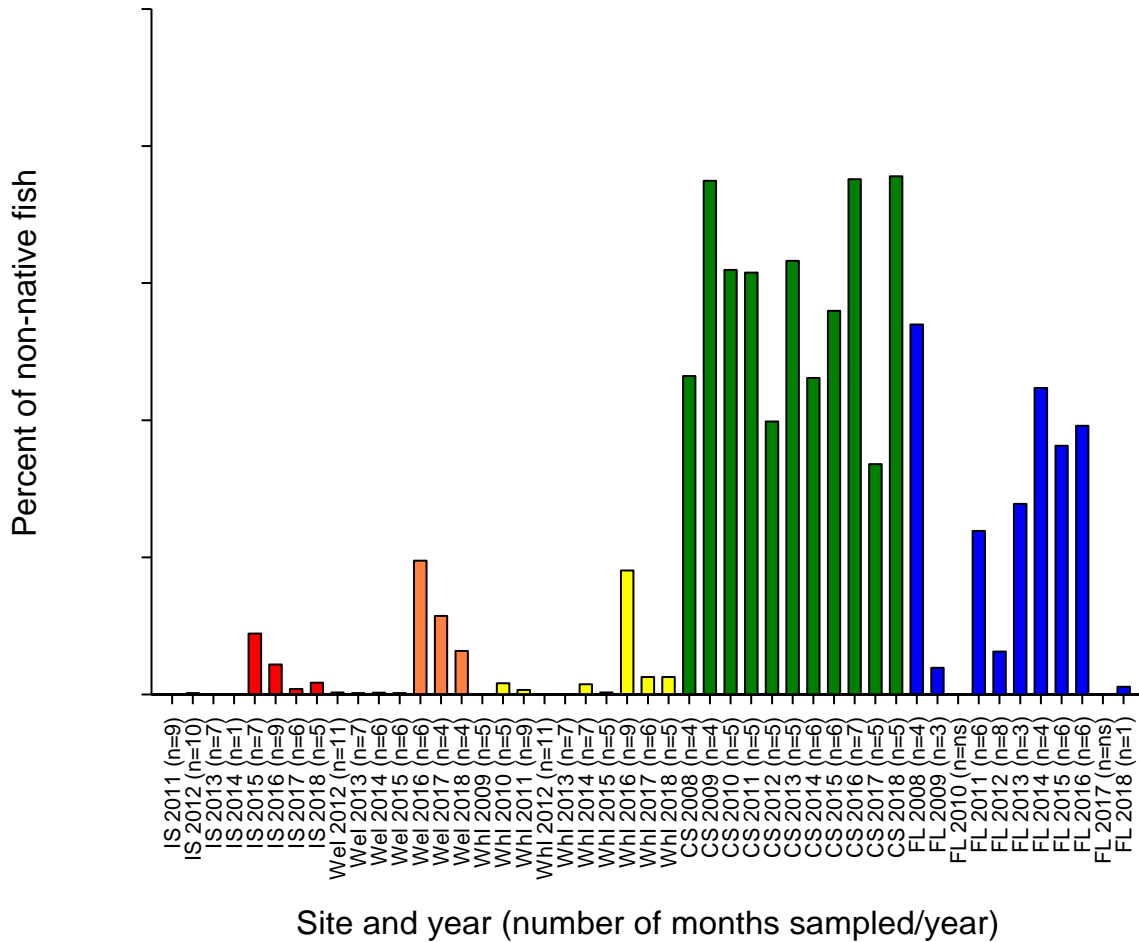
Figure 113. Percent of total fish catches per year that are non-native species for EMP trend sites sampled in 2008-2018. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
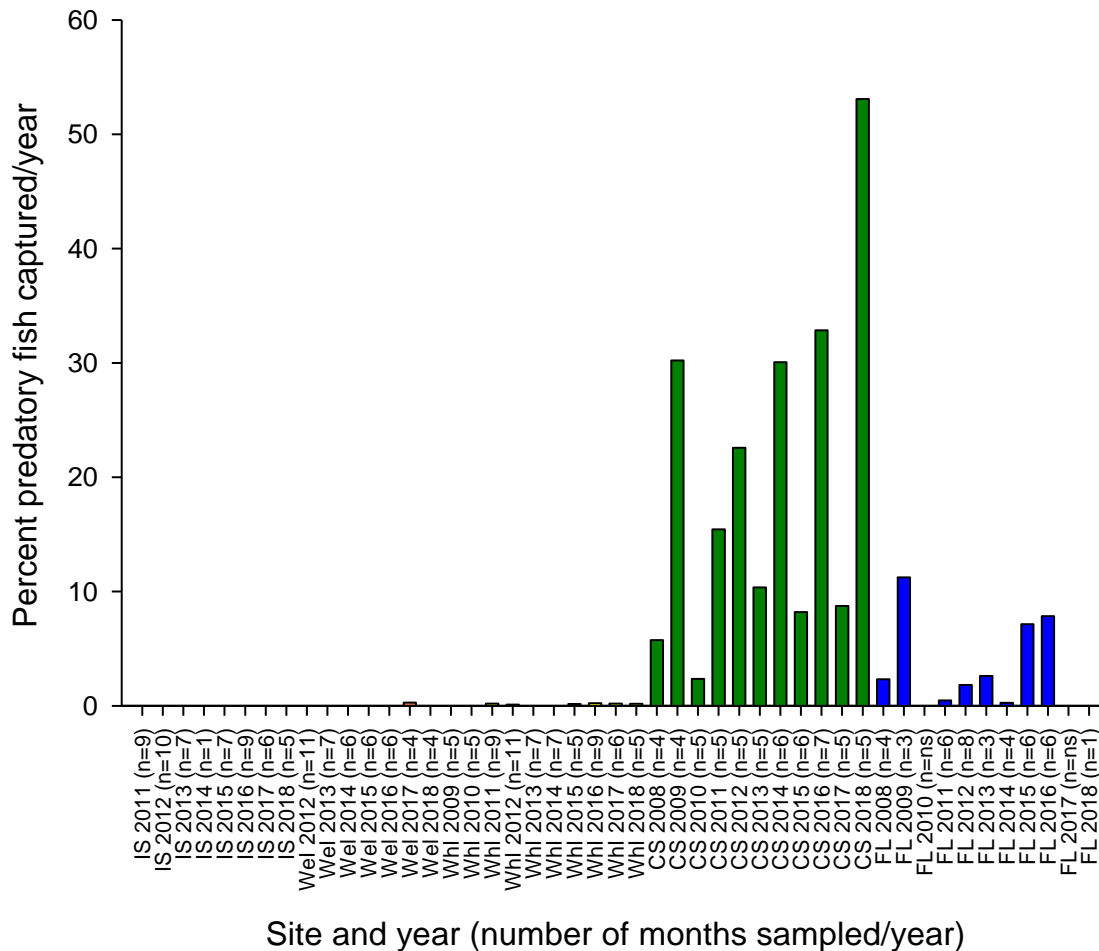
Figure 114. Total percentage of the yearly (2008-2018) catch of fish species that have mature stages that could be predatory toward juvenile salmon. Species include small and largemouth bass, northern pikeminnow, walleye, warmouth, and yellow perch. For each year the total number of sampling months is 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, 2018 salmon species composition varied by site, showing distinct patterns associated with hydrogeomorphic reach (Figure 115). In 2018, Chinook salmon were caught at all five sampled sites and were the dominant salmon species at Welch Island in Reach B, Whites Island in Reach C, Campbell Slough in Reach F and Franz Lake in Reach H. At these sites, Chinook salmon comprised 90 to 100% of salmonid catches. In 2018, unmarked (presumably wild) Chinook were more abundant at Welch, Whites Island, Ilwaco and Franz Lake than marked hatchery Chinook, however, marked Chinook were more abundant than unmarked Chinook at Campbell Slough (Figure 116). In addition to Chinook salmon, small numbers of chum salmon were found at Welch Island, Whites Island, and Campbell Slough. This pattern is typical for Welch and Whites Islands and has been evident since 2011. Chum salmon were the most abundant salmon species captured at Ilwaco Slough in Reach A. Only one unmarked coho salmon was collected at Franz lake. No trout or sockeye salmon were caught in 2018 (Figure 115 and Table E-3 in Appendix E).
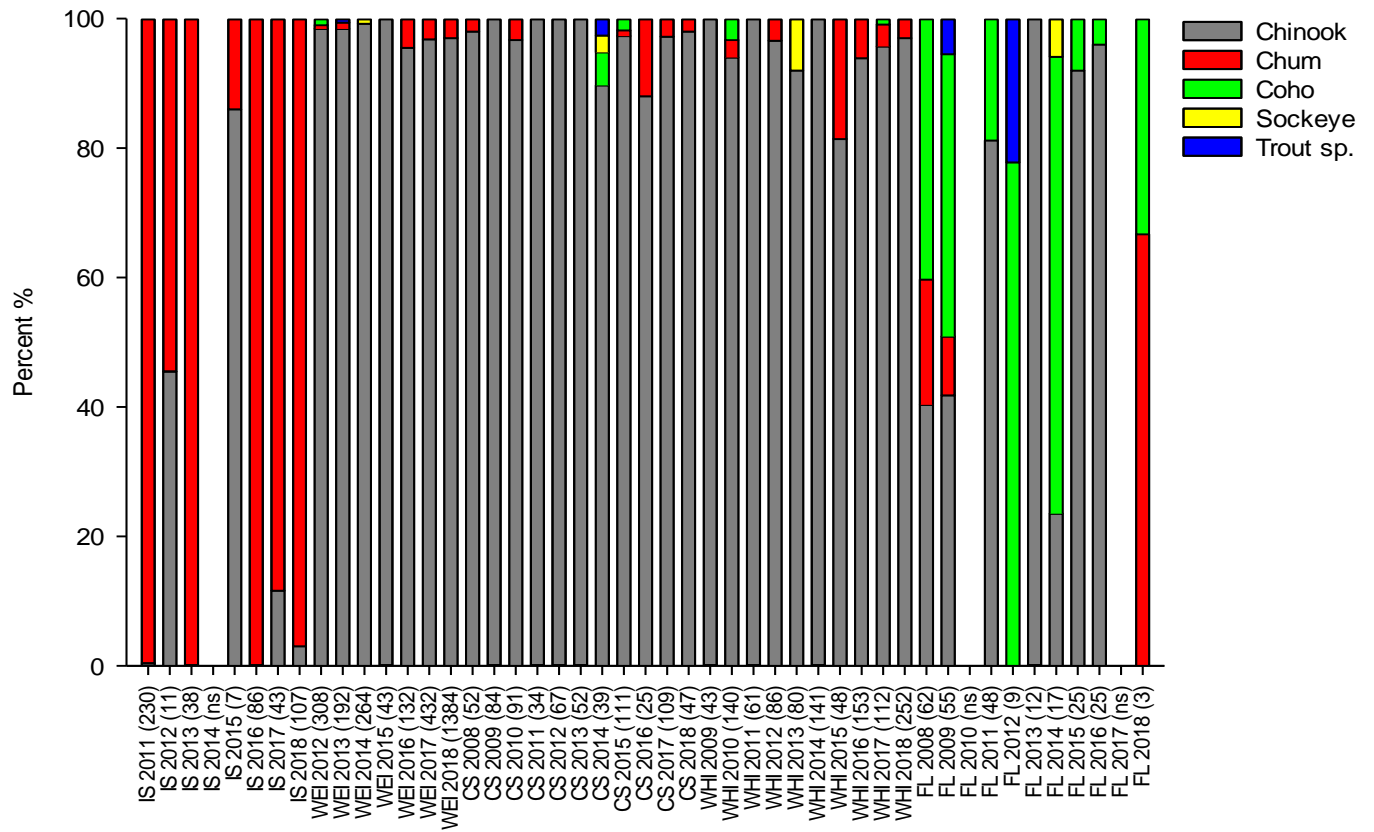
Figure 115. Percentage of salmonid species collected at EMP trends sites in 2018, as compared to percentages collected in previous sampling years. Total number of salmonids captured at a given site and year are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
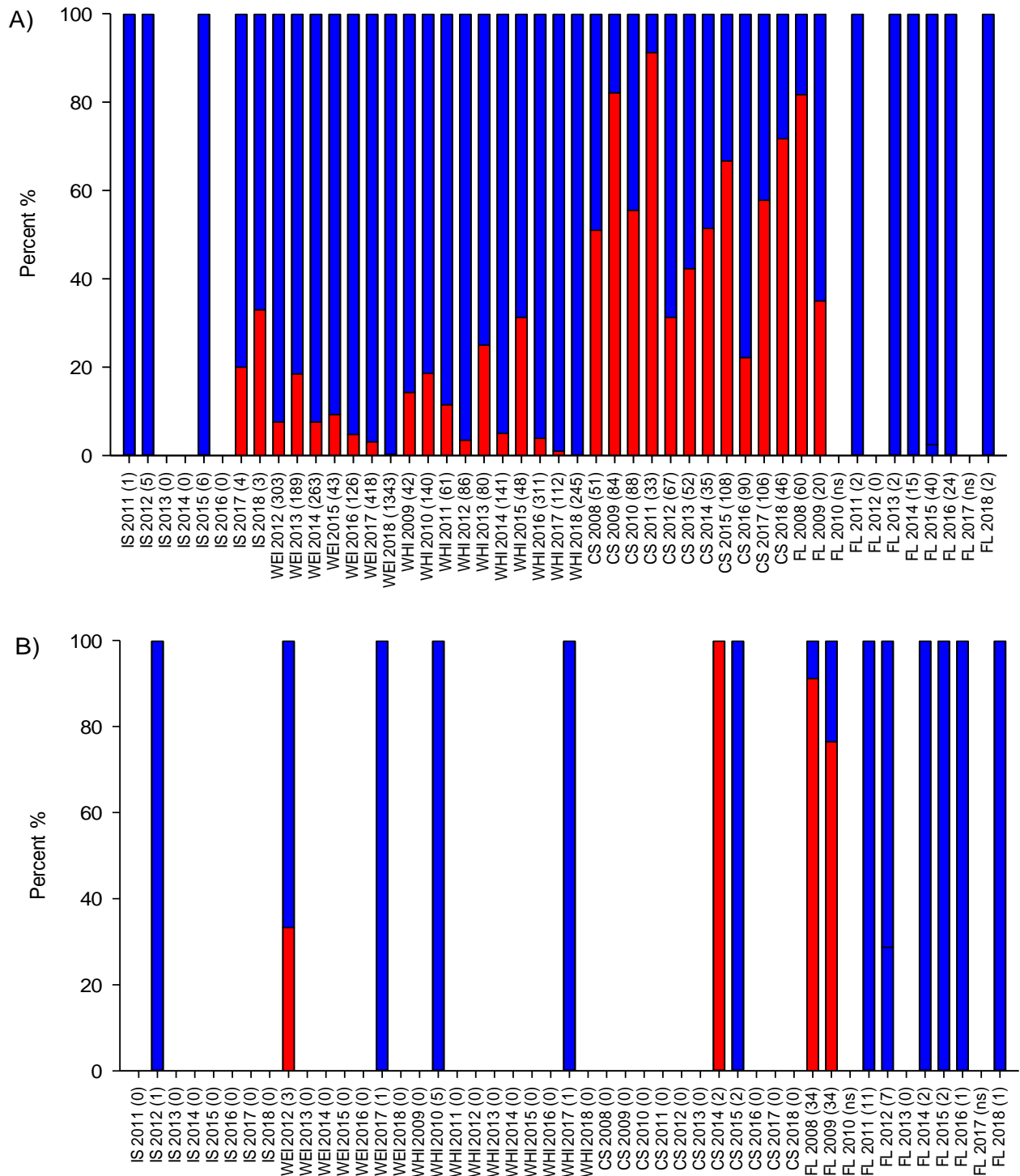
Figure 116. Percentage of marked (red) and unmarked (blue) a) Chinook salmon and b) Coho salmon captured at the EMP sampling sites in 2018, as compared to previous sampling years. Total number of the specified salmon species captured at a given site and year are presented in parentheses. **IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.**

3.6.2.1 *Salmon Density*
Chinook salmon

In 2018, unmarked Chinook salmon were captured at the EMP trends sites from February through June. The highest average densities of unmarked juvenile Chinook salmon were 301 and 648 per 1000 m^2 in April and May, respectively. Marked Chinook salmon were captured from April to October, with the highest average densities of 26 fish per 1000 m^2 in May (Figure 117a). Mean Chinook salmon densities by site and year are shown in Figure 118. In 2018 the density of unmarked Chinook salmon was highest at Welch Island (200.01 fish per 1000 m^2) and Whites Island (99.77 fish per 1000 m^2) and lowest at Ilwaco Slough (0.61 fish per 1000 m^2) with densities in the 2-4 fish per 1000 m^2 range at Campbell Slough and Franz Lake. Densities of unmarked Chinook salmon in 2018 were greater in Welch and White Island and within the same ranges as previous years at all other sites. The densities of marked Chinook salmon in 2018 were generally within the same ranges similar 2008-2016 at all sites, but less than those in 2017. (Figure 118).

Coho salmon

In 2018, only one unmarked coho salmon was collected at Franz Lake in March (Figure 115). Coho salmon densities by site and year are listed in Appendix E. Coho salmon have been captured only sporadically at Ilwaco Slough, Campbell Slough, Welch Island, and Whites Island, so their absence in 2018 was not unusual compared to previous years. Only one sampling date was conducted at Franz Lake in 2018 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 and 2018 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 2018, chum salmon were found at the trends sites in March, April and May with the highest average density in March (43.05 fish per 1000 m^2; Figure 117b). Chum salmon were present at Ilwaco Slough, Welch Island, Whites Island and Campbell Slough in 2018 (Figure 115). In 2018 the density of chum salmon was highest at Ilwaco Slough (32.11 fish per 1000 m^2) and lowest at Campbell Slough (0.27 fish per 1000 m^2). However, no chum were captured at Franz Lake in 2018. 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 2018, as in 2017 and 2016, 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).
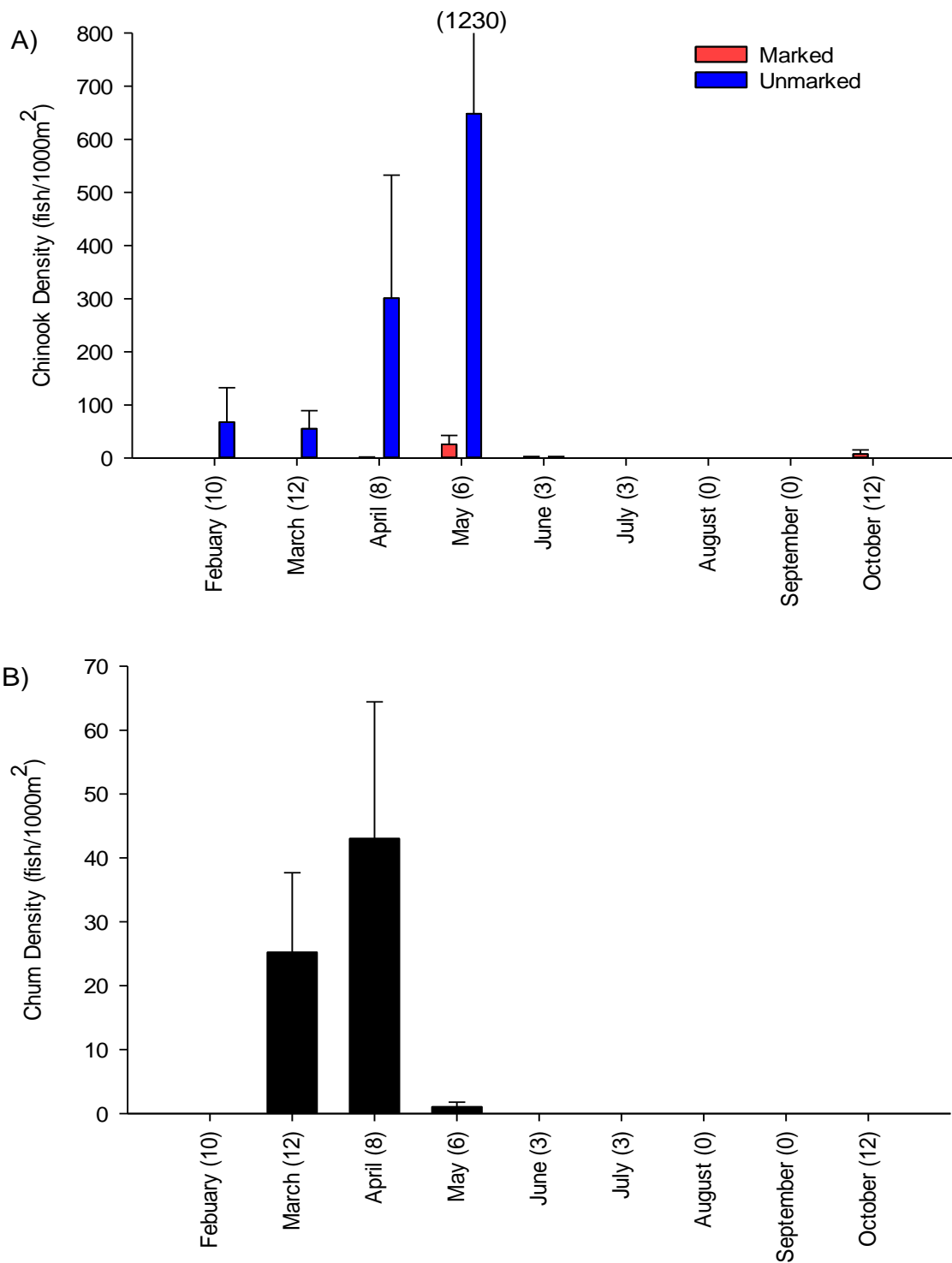
Figure 117. Mean (SE) densities (fish per 1000 m2) of marked (red bars) and unmarked (blue bars) juvenile a) Chinook salmon, b) chum salmon by month during the 2018 sampling year (all sites combined). Total number of sampling efforts per month are presented in parentheses. May 2018 was truncated for ease of viewing and the number above the error bar in parentheses is the upper limit of the error (SE). Only one Coho salmon was captured at all sites in 2018 therefore no monthly density for Coho salmon is shown.
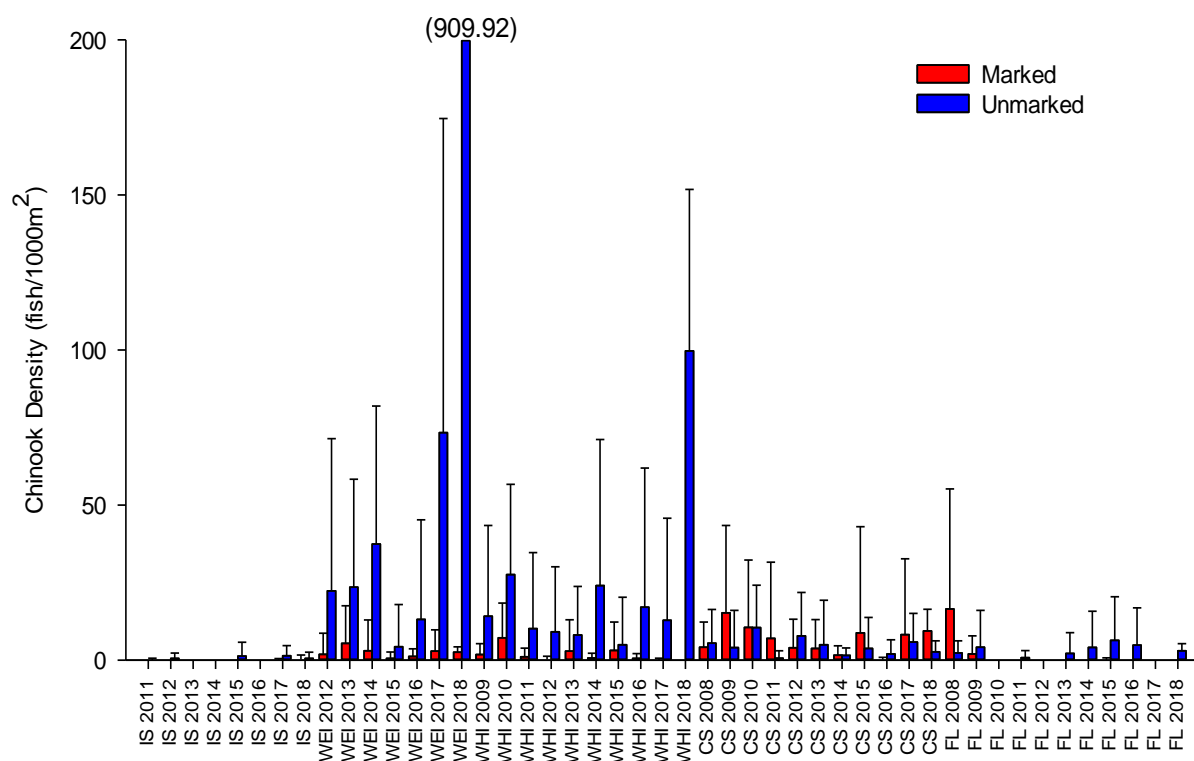
Figure 118. Marked (red bars) and unmarked (blue bars) juvenile Chinook salmon densities (fish per 1000 m2) by site and year. Welch 2018 was truncated for ease of viewing and the number in parentheses is the upper limit of the error (SE). IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.3 Salmon Metrics

3.6.3.1 *Genetic Stock Identification*

In 2018, genetics data were collected from Chinook salmon at Welch Island, Whites Island, Campbell Slough, and Franz Lake. 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 2018 (Figure 119a), 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. 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 43%, followed by Snake River fall stock at 29%. However, the genetics sample size of unmarked Chinook in Campbell Slough was small in 2018 (N=7), so the difference of just one fish in a stock category has a large impact on the proportions. Two unmarked fish were collected at Franz Lake in 2018. One individual assigned to West Cascade fall stock while the other assigned to Willamette River spring stock. This is the first instance of unmarked West Cascade fall and Willamette River spring reporting groups at Franz Lake which is typically represented by unmarked Spring Creek Group, Upper Columbia summer/fall, and Snake River fall stocks.

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Substantially fewer marked fish were collected and subsequently analyzed for genetics at trend sites in 2018 (Figure 119b), with the exception of Campbell Slough. At Campbell Slough, 90% of marked Chinook assigned to the Spring Creek Group, while 10% assigned to the West Cascade fall reporting group. Welch Island was the only other site where marked juvenile Chinook salmon were collected. The individuals were equally represented by West Cascade fall and Spring Creek reporting groups.

The seasonal distribution of stocks in 2018 (Figure 120a) reveals that West Cascade fall Chinook are present throughout the lower Columbia River and estuary during spring and summer. Willamette River spring fish were present in the early spring while Snake River fall fish were present in late spring/early summer. Interior stocks occurred earlier at upper reaches and were not present in lower reaches until June. The seasonal trend of stocks in 2018 was similar to seasonal trends observed in previous years (Figure 120b).

Differences in stock composition at trend sites from 2008-2018 were analyzed using non-metric multidimensional scaling and ANOSIM. The NMDS plot (Figure 121) 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 = 0.384, 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 = 0.74 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, and to a lesser extent, West Casacade spring stock. The separation between Welch and Whites Islands and other sites is driven by upper Columbia summer/fall, Spring Creek group fall, Willamette River spring, and, to a lesser extent, Snake River fall stocks.
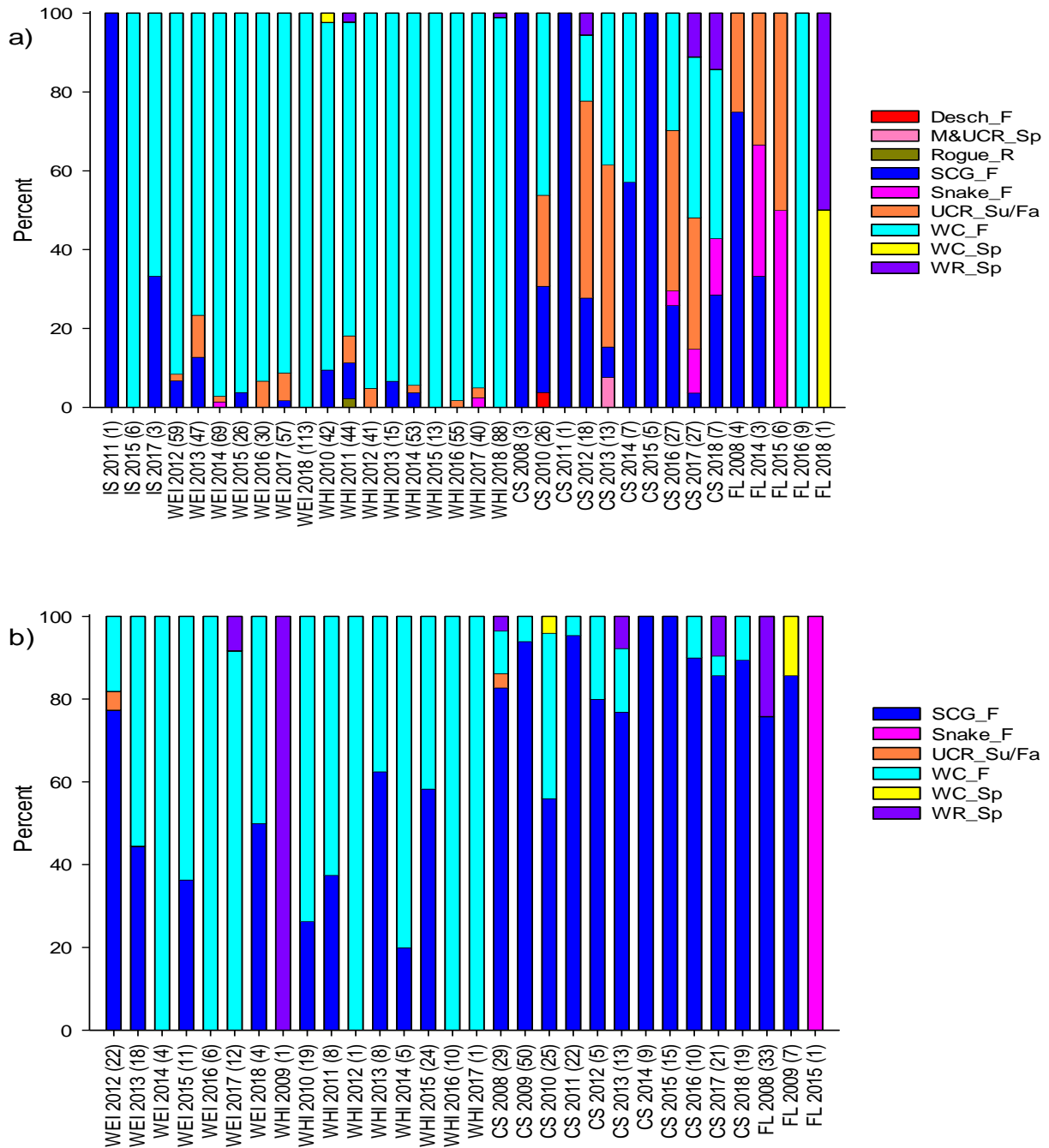
Figure 119. Genetic stock composition of a) unmarked and b) marked Chinook Salmon at trend sites from 2008–2018. Genetic sample sizes for each site is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake. Chinook salmon stocks: Desch_F = 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.
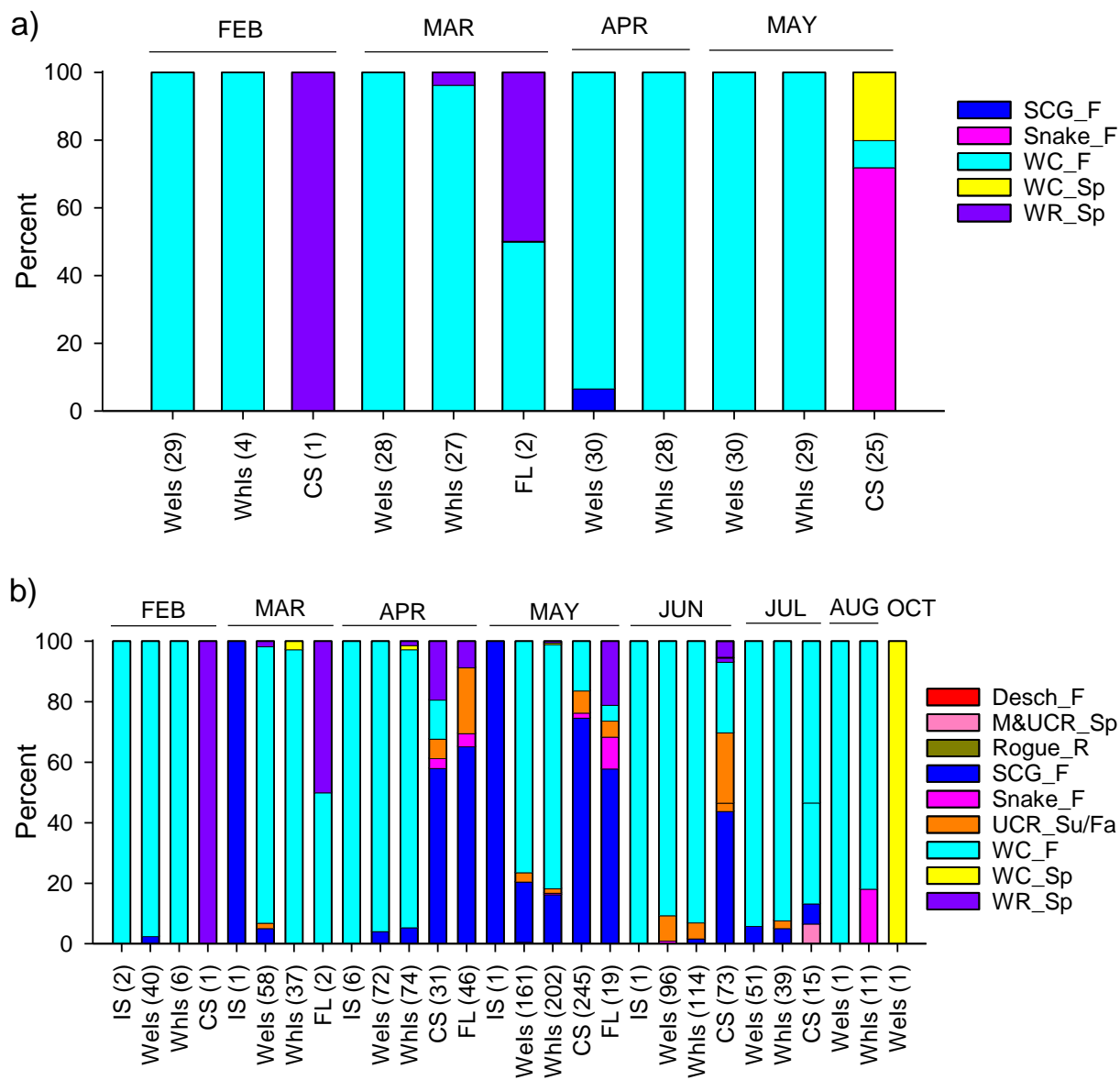
Figure 120. Seasonal percent stock composition per site for Chinook Salmon collected in a) 2018 and b) 2008–2018. Plots include both unmarked and marked Chinook Salmon. Genetic sample sizes for each site is presented in parentheses. IS = Ilwaco Slough, WeIs = Welch Island, WhIs = Whites Island, CS = 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.
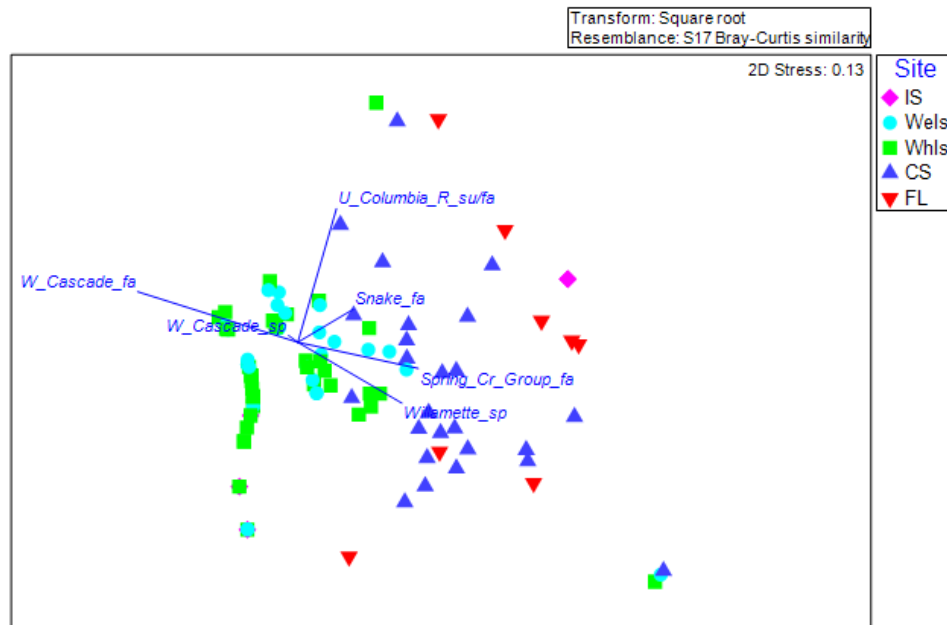
Figure 121. Nonmetric multidimensional scaling (NMDS) plot based on square-root transformed genetic stock abundance at five trend sites, 2008-2018. Significant correlation with variables (Pearson R > 0.4) are represented as blue vectors. IS = Ilwaco Slough, WeIs = Welch Island, WhIs = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.3.2 *Salmon Size and Condition*

Chinook salmon

Length, weight, and condition factor. Chinook salmon were caught at all sampled locations in 2018. In 2018, the average length, weight and condition factor for unmarked Chinook captured at the following sampled sites were: Welch Island 46.29 ± 8.47 mm; 1.02 ± 0.78 g; and 0.87 ± 0.15, Whites Island 54.73 ± 9.35 mm; 1.87 ± 1.09 g; and 1.02 ± 0.11, and Campbell Slough 64.77 ± 14.29 mm; 3.55 ± 2.06 g; and 1.09 ± 0.10, respectively (Figure 122). No average length, weight and condition factor was calculated for Ilwaco and Franz Lake due to very low sample size, which is a similar pattern seen over a number of years at both sites. The length, weight, and condition of unmarked Chinook salmon in 2018 showed similar patterns, with the largest fish typically captured at Campbell Slough (Figure 122). Within sites, there was some variation among years, though no clear increasing or decreasing trends. Unmarked Chinook sampled at Whites Island in 2018 appeared to have a similarly low average weight as seen in 2017 and 2016. However, the condition appears to follow the overall trend on Whites Island over the past 5 years. Unmarked Chinook sampled at Welch Island in 2018, had the lowest average fork length, weight and condition observed over the past seven years of sampling at the site. However, this is likely due to no June sampling at Welch Island in 2018, which would have likely contributed to an increased average of all measured indices. Unlike previous years (2011- 2017) where June catches played a role in the contribution to unmarked Chinook indices, May was the last month where unmarked and marked Chinook were captured at Welch Island before warm river temperatures reached sampling limitations in mid-June to early July.

In 2018, marked Chinook salmon were caught at three of the five sampled locations. Only Welch Island and Campbell Slough experienced significant enough catches to examine marked Chinook size and condition. In 2018 the average length, weight and condition factor for marked Chinook captured at the

following sampled sites were: Welch Island (± SD) 68.5 ± 2.13 mm; 3.35 ± 0.91g; and 1.03 ± 0.19, and Campbell Slough 86.04 ± 5.56 mm; 6.70 ± 1.42 g; and 1.04 ± 0.05, respectively (Figure 123). Campbell Slough shows little variation in length, weight, and condition across sampled years. Similar to unmarked Chinook sampled at Welch Island in 2018, marked Chinook size, weight and condition were lower than previous years.

Life History. At the trend sites in 2018, the majority of unmarked Chinook salmon were fry, 88.3%; 11.3% were fingerlings, and 0.4% were yearlings (Figure 124a). At Welch, Whites, and Ilwaco fry dominated catches, making up 92.0 %, 85.0%, and 100% of unmarked Chinook salmon, respectively. Campbell Slough showed a more even distribution of fry and fingerling with 38% and 54% of unmarked Chinook salmon, respectively. In comparison to previous years, the percentage of fry at all of the trend sites was slightly greater than in recent years.

A total of 38 (100%) marked Chinook salmon caught at the trends sites in 2018 were fingerlings (Figure 124b). In comparison to previous sampling years, the proportion of yearlings encountered varies, but generally remains low and does not appear to differ from the overall trend. Contrary to 2017, no fry marked Chinook were observed in 2018 at any of the sampled locations.

Other salmon species

A total of 154 chum salmon were captured in 2018, 105 at Ilwaco Slough, 41 at Welch Island, 7 at Whites Island, and one at Campbell Slough. All chum were caught between March and May. Unfortunately, no weights were taken on the 105 chum salmon caught at Ilwaco in 2018 due to the difficulties of measuring weight (g) in the field during adverse weather conditions. In 2018 the average length, weight, and condition factor of chum salmon captured at the following sampled sites were: Ilwaco Slough (± SD) 44.52 ± 4.41 mm, no average weight and condition index (k), Welch Island 47 ± 7.80 mm; 0.86 ± 0.62 g; and 0.72 ± 0.11, Whites Island 52.57 ± 9.51 mm; 1.27 ± 0.07 g; and 0.78 ± 0.09, and Campbell Slough 70 mm; 3.3 g; and 0.96, respectively (Figure 125). The chum salmon collected in 2018 were comparable in size to those that have been collected in previous years, not especially large or small. Similarly, the mean 2018 value for condition factor (0.73) was intermediate, between a high of 1.10 in 2008 and a low of 0.58 in 2013 and similar to 2017 (0.71). The largest fish, in terms of length and weight, were generally found at Whites Island and Campbell Slough, while condition factor tended to be highest at and Campbell Slough, this site is located second farthest upstream of all current sampled sites. Although chum salmon were captured sporadically, some variation by year was 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 2018, an unmarked coho found at Franz Lake in March. Franz Lake is the only site where coho salmon have been caught consistently enough to compare size measurements by sampling year. However, due to high water throughout the 2017 sampling season, Franz Lake was not sampled; therefore no comparisons can be made to last year's catch. Overall size and condition of unmarked and marked coho sampled at Franz Lake are shown below in FiguresFigure 126 andFigure 127.

Sockeye salmon and trout were not caught at any of the trends sites in 2018. Sockeye salmon were last sampled at Welch Island in 2014 and trout were last caught at Welch Island in 2013.
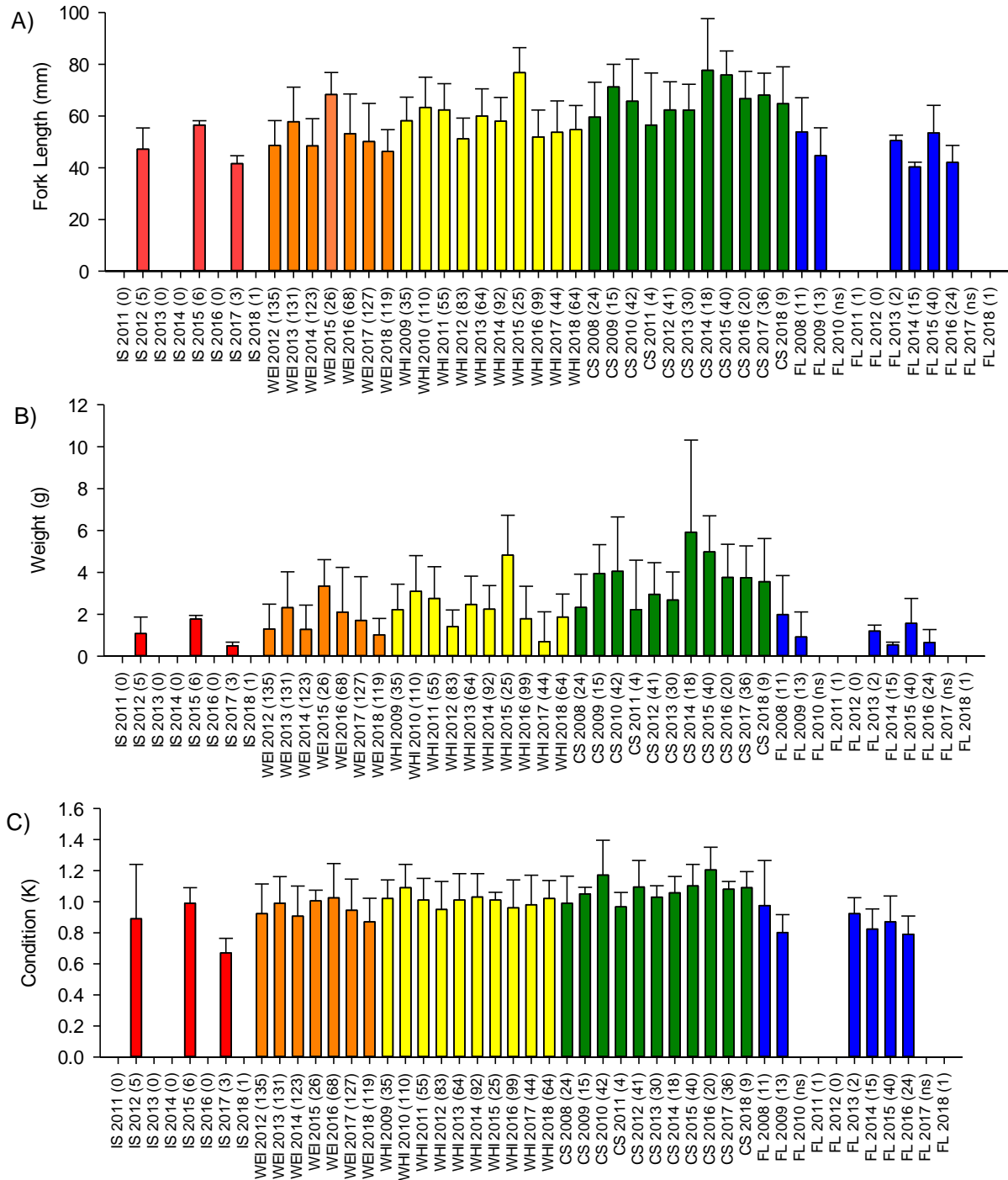
Figure 122. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of unmarked juvenile Chinook salmon at trends sites in 2018 as compared to previous years. Total number of Chinook salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
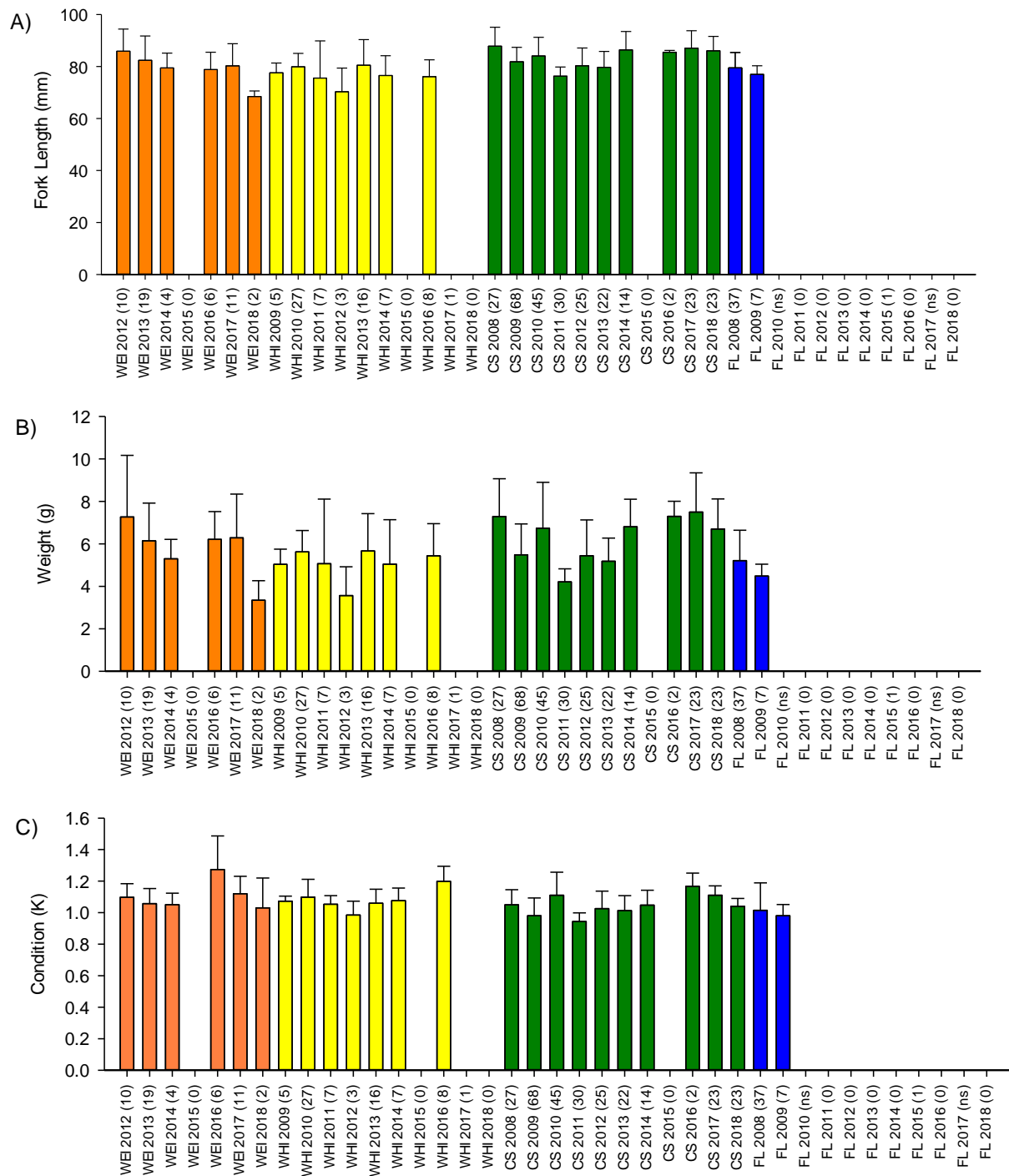
**Figure 123. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of marked Chinook salmon at trends sites in 2018 compared to previous sampling years. Total number of Chinook salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.**
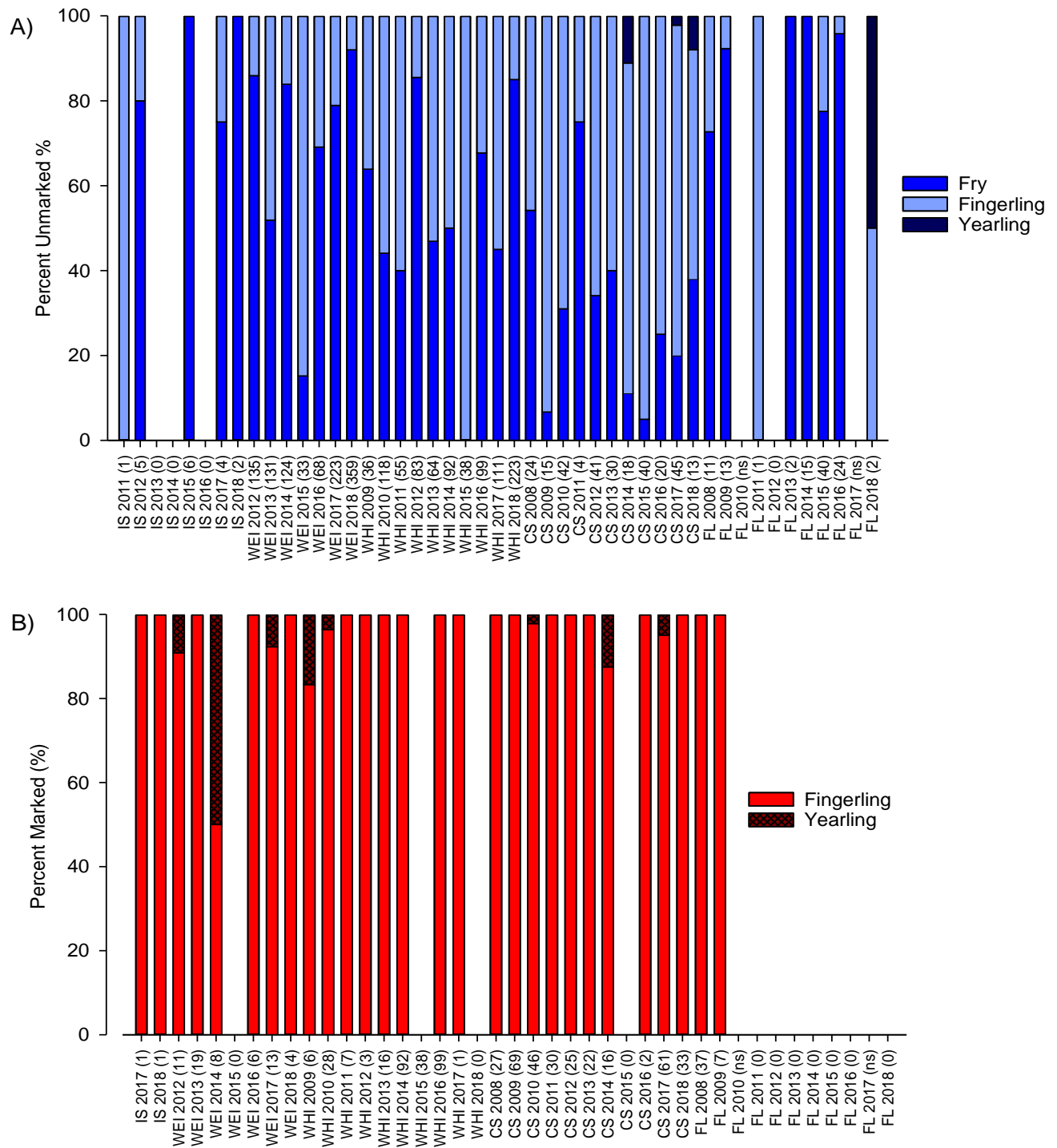
Figure 124. Percentages of life history types of a) unmarked and b) marked juvenile Chinook salmon captured at trends sites in 2018 and in previous sampling years. Total numbers of Chinook salmon captured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
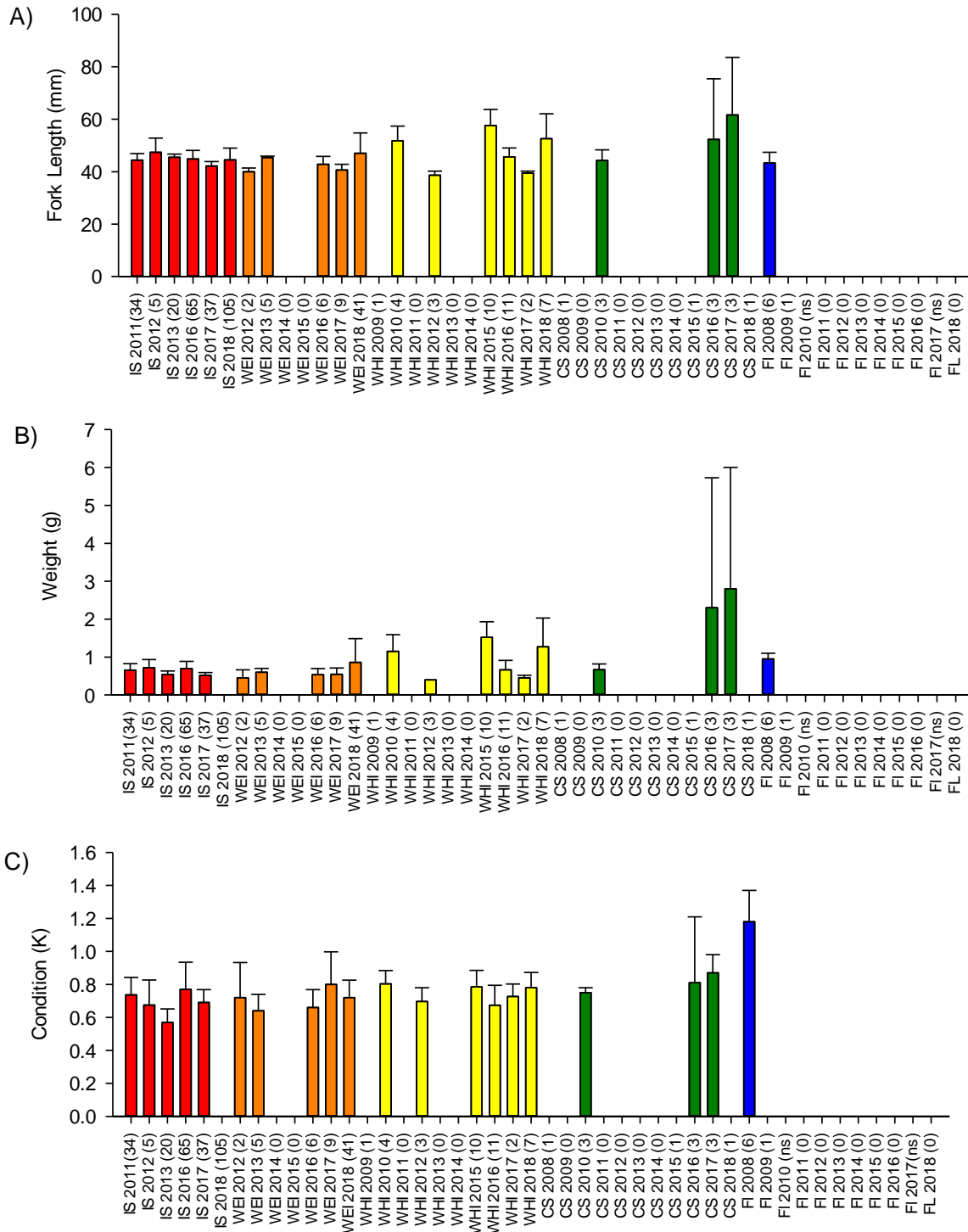
Figure 125. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of chum salmon at trends sites in 2018 compared to previous sampling years. In 2018, due to difficult weather conditions no Chum weights were recorded at Ilwaco Slough (IS), therefore no Condition index (k) was calculated. Total number of chum
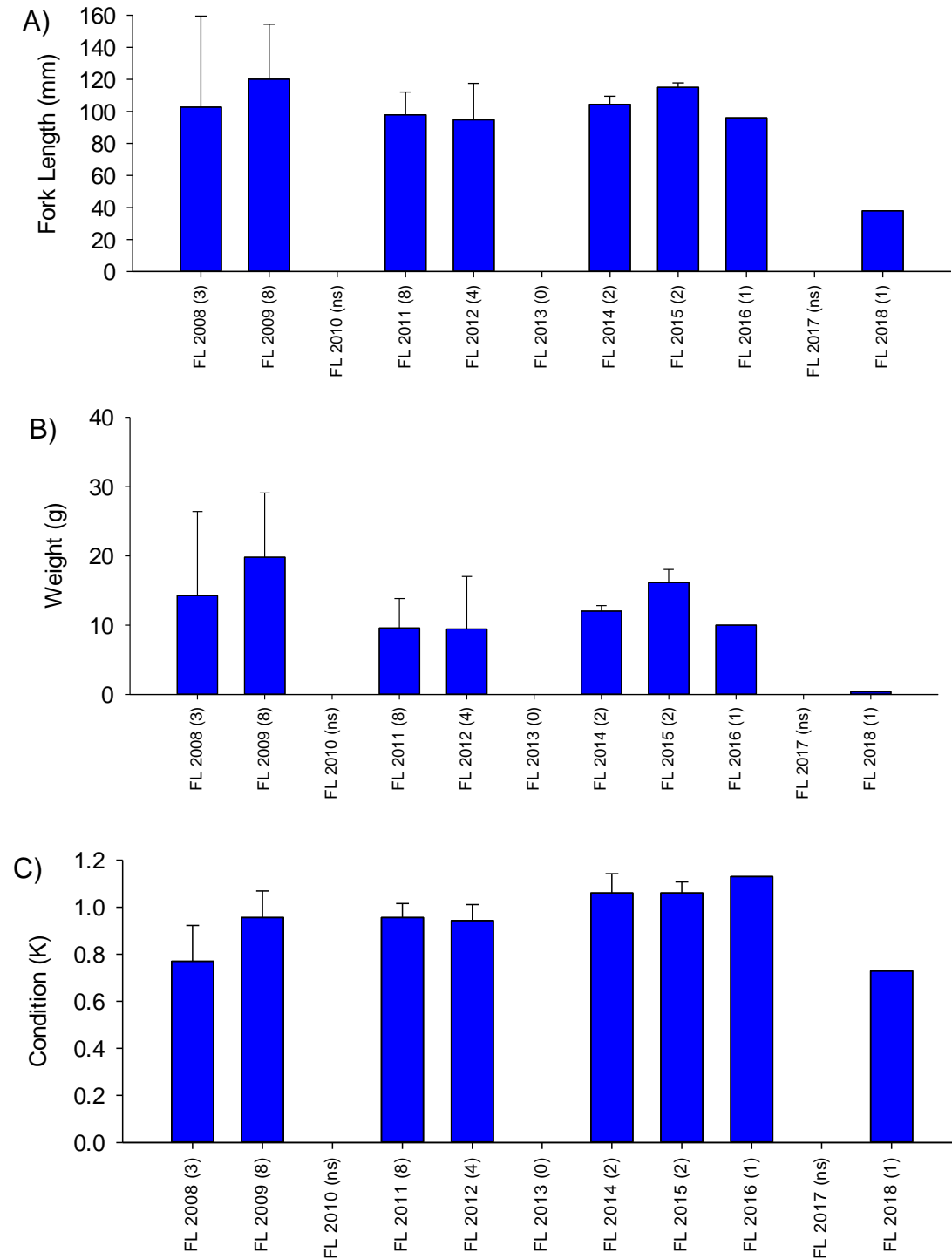
Figure 126. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of unmarked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses.
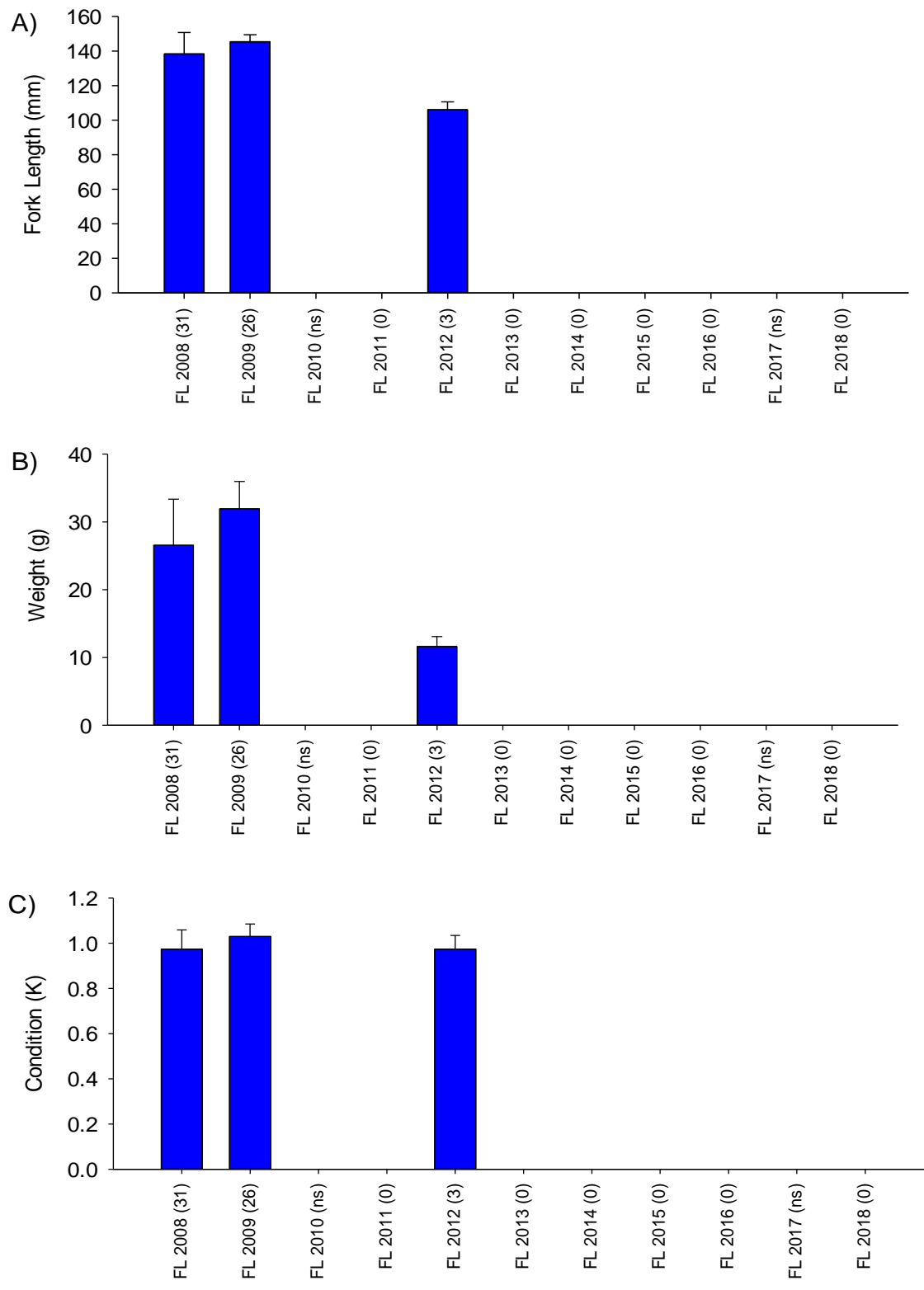
Figure 127. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of marked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses.

3.6.3.3 *Somatic Growth Analyses*

Otoliths are used to estimate somatic growth rates in fish. 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 2005–2013. 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 128).
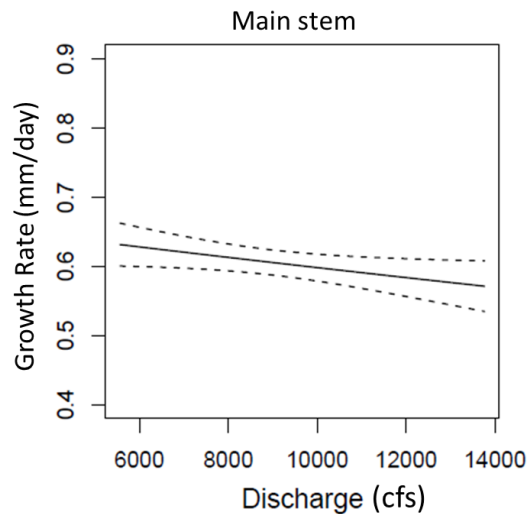
Figure 128. Relationship between growth rate and discharge for juvenile Chinook Salmon collected at mainstem Columbia River sites, 2005, 2008, and 2013.

Off-channel: Fish (n=535) collected in off-channel habitat were obtained across several years (2007-2014, 2016-2017) 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 day of year, year, river kilometer, stock, hatchery/unmarked, & fork length. Specifically, we observed a significant (p< 0.01) negative (curvilinear) relationship between growth rate and day of year, a significant positive relationship to river km and fork length. Also, fish collected in 2007 had significantly faster growth than those from 2011 and 2014 (Figure 129).
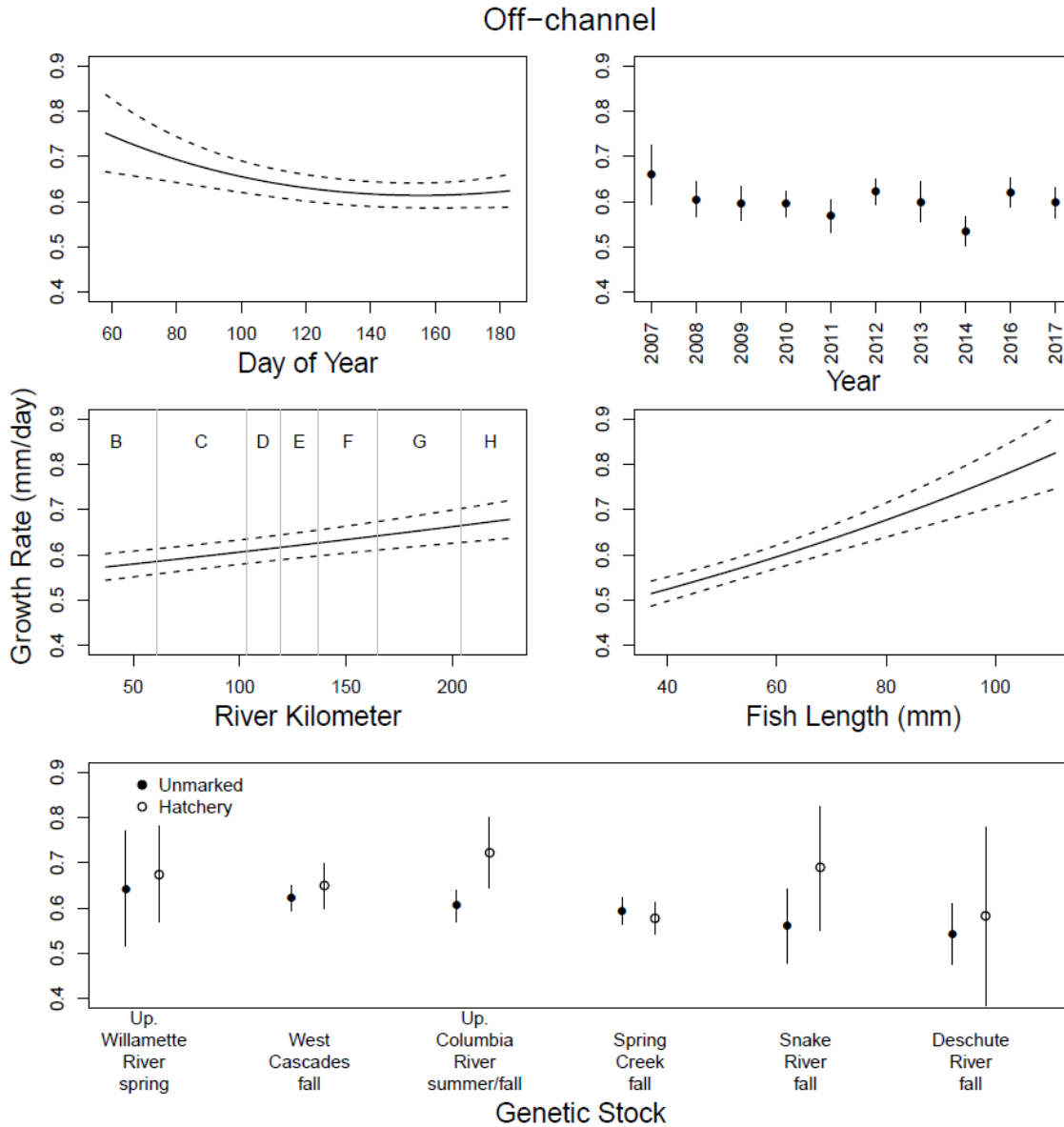
Figure 129. Relationships between growth rate and a suite of variables for juvenile Chinook Salmon collected at off-channel Columbia River sites, 2007–2014, and 2016–2017.

Welch Island: Fish (n=97) collected from Welch Island were obtained across several years (2012-2014, 2016-2017). Our GLM analysis indicated that the model that best explained variability in growth rate (mm/day) consisted of year, discharge, and fork length. Specifically, we detected a significant (p< 0.05) negative and positive relationship between growth rate and discharge and fork length, respectively. Also, fish collected in 2012 & 2017 had significantly (p<0.01) faster growth than those from 2013, 2014, & 2016 (Figure 130).

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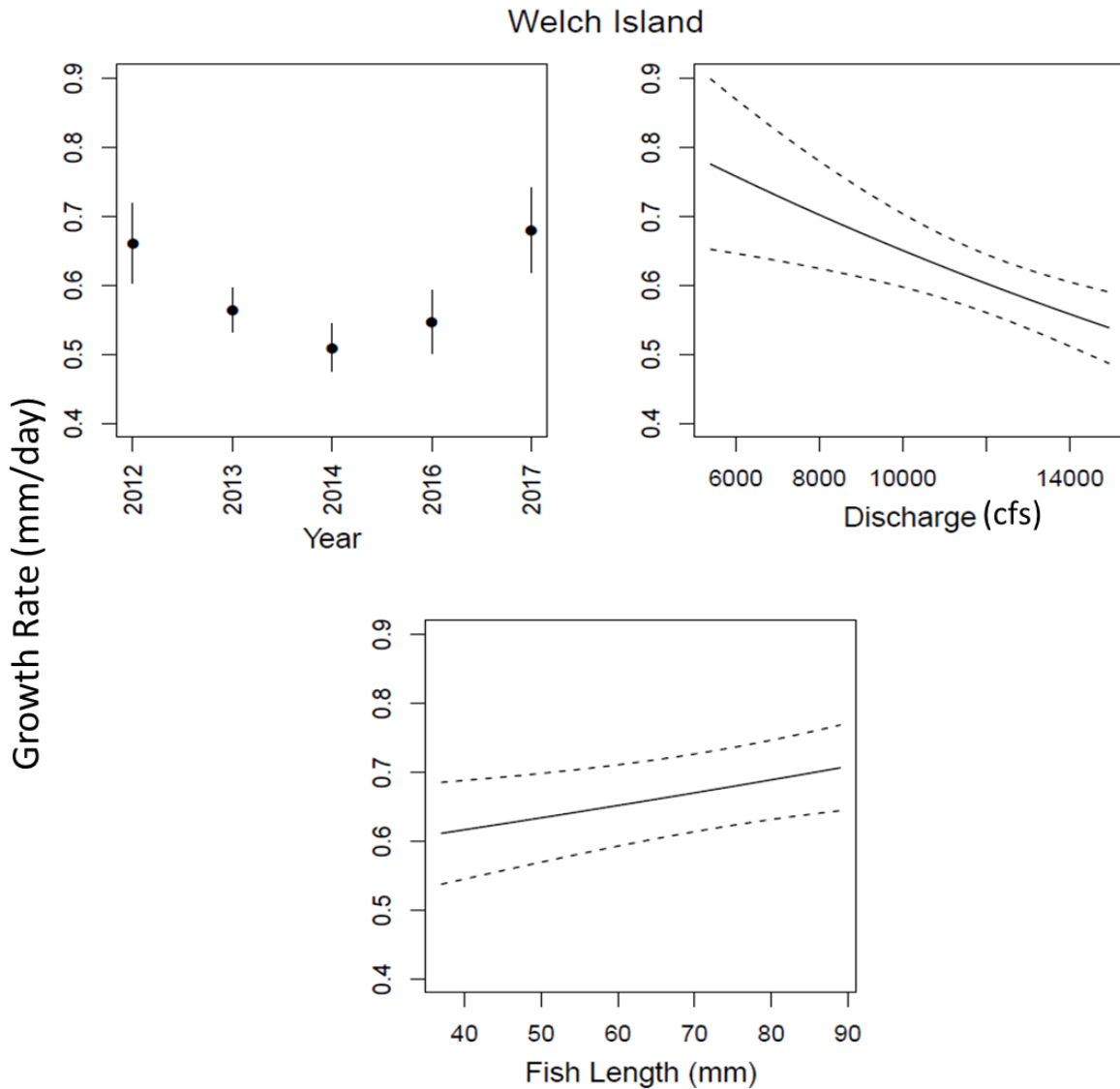
Figure 130. Relationships between growth rate and a suite of variables for juvenile Chinook Salmon collected at Welch Island, 2012–2014, and 2016–2017.

Whites Island: Fish (n=102) collected from Whites Island were obtained across several years (2009-2014, 2016-2017). 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 had significantly (p<0.05) faster growth than those from 2009, 2014, & 2017 (Figure 131).

Whites Island
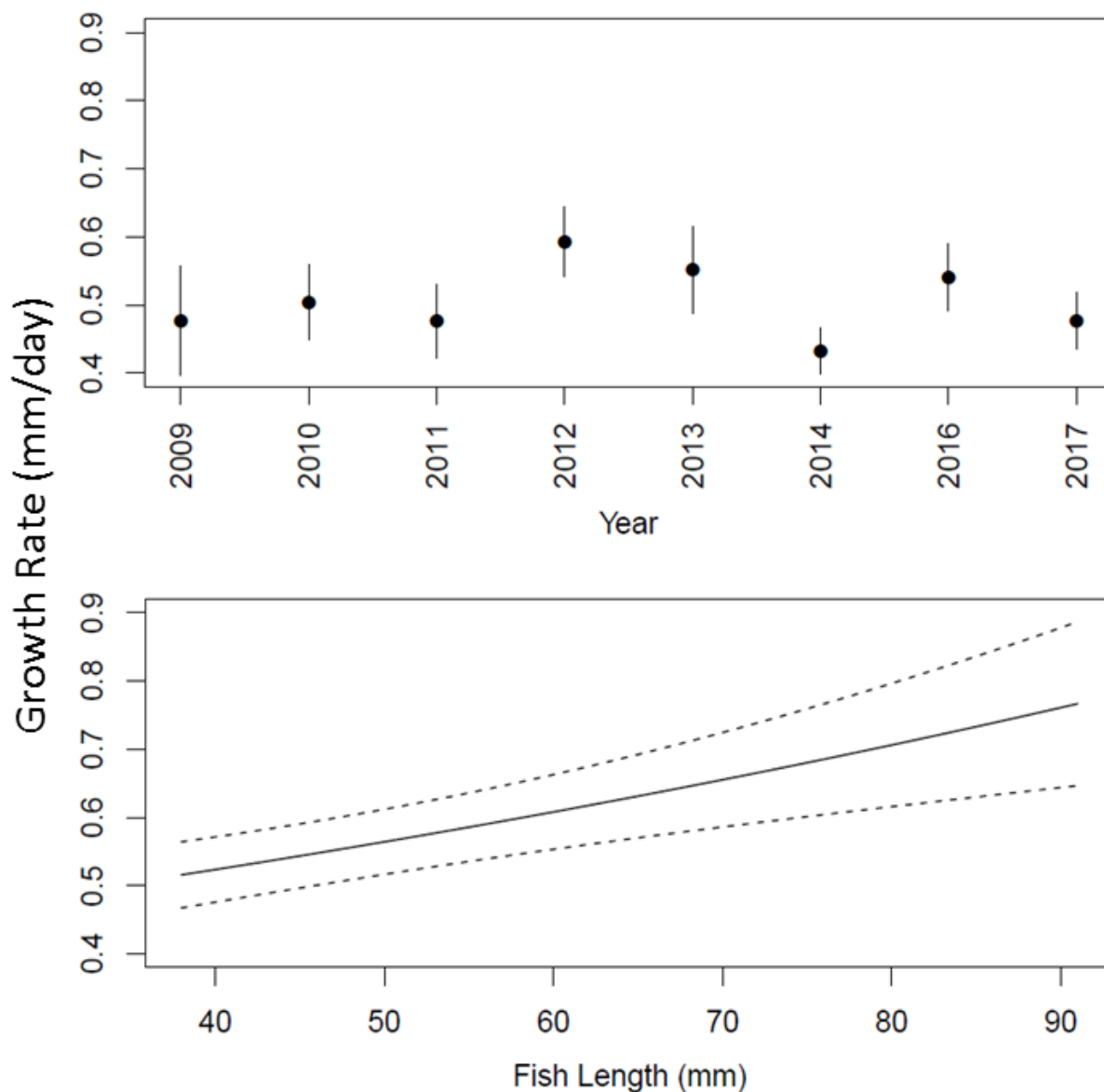
Figure 131. Relationships between growth rate and year and fish length for juvenile Chinook Salmon collected at Whites Island, 2009–2014, and 2016–2017.

Campbell Slough: Fish (n=102) collected from Campbell Slough were obtained across several years (2007-2014, 2016-2017). 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 132).
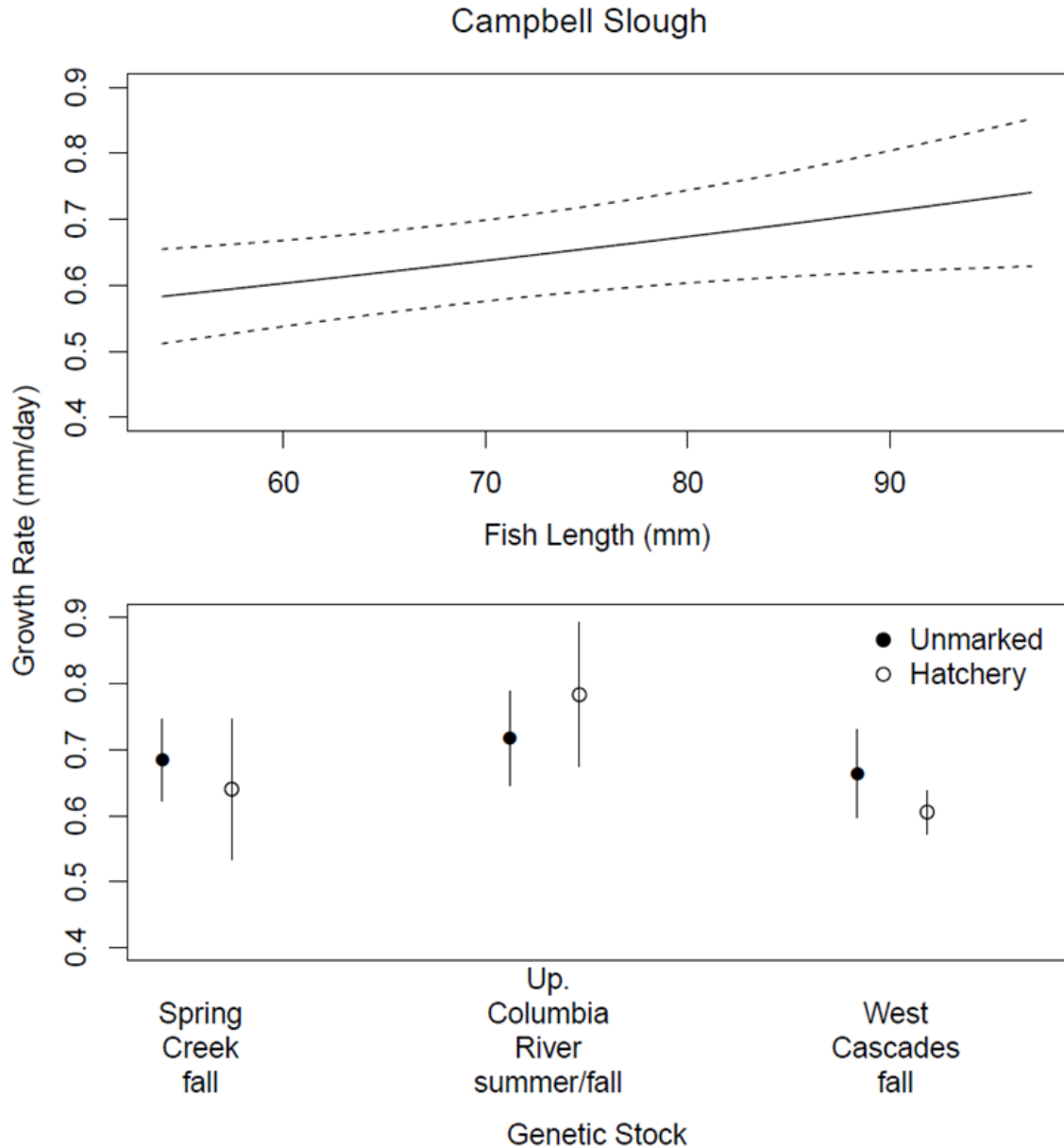
Figure 132. Relationships between growth rate and fish length and genetic stock for juvenile Chinook Salmon collected at Campbell Slough, 2007–2014, and 2016–2017.

3.6.3.4 *Lipid Content of Juvenile Chinook Salmon*

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 133). 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.
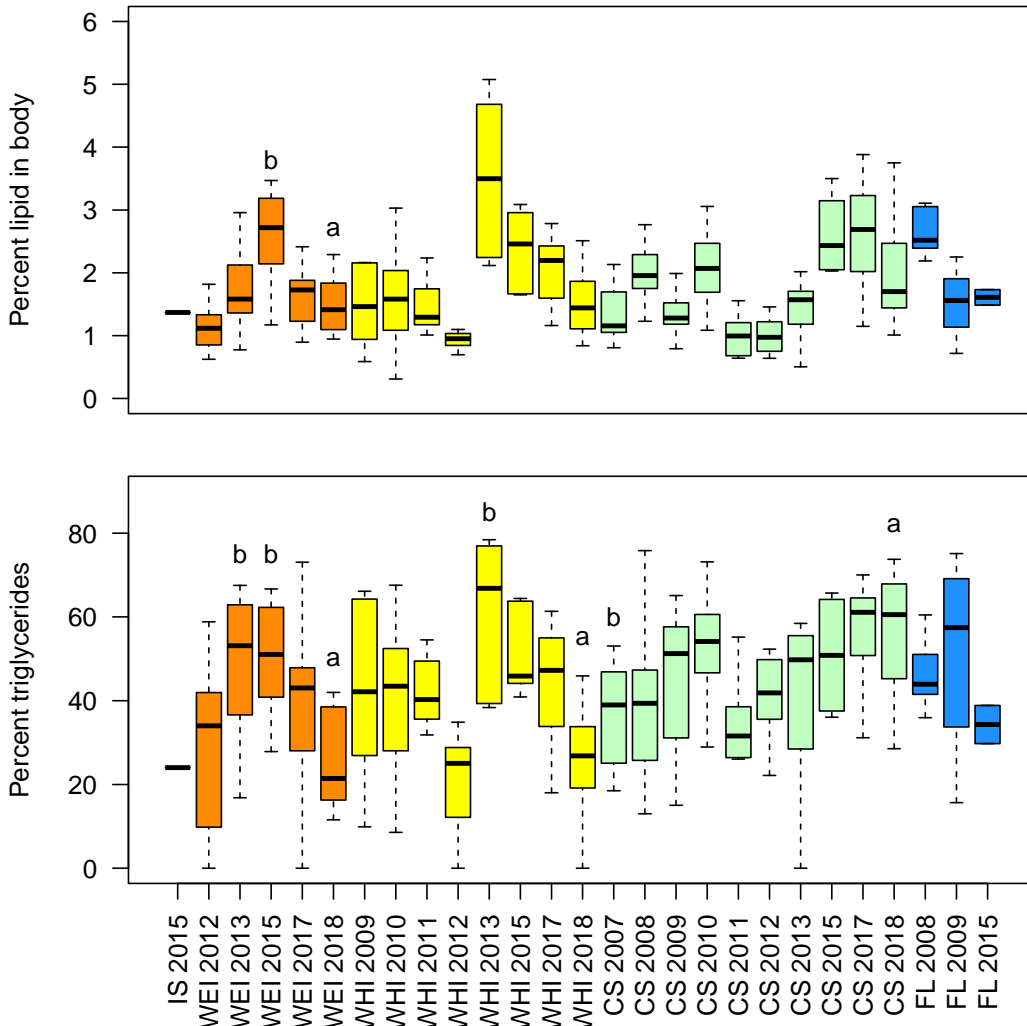
Figure 133. Percent lipid content (A) and percent total lipids that were triglycerides (B) determined in whole bodies of juvenile Chinook salmon collected from the trend sites in 2018 compared to previous sampling years. Unlike letters indicate 2018 values within each site that differ significantly from 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 newly revamped PIT tag detection system at Campbell Slough began operating on February 28, 2018, and was continuously collecting data until November 2, 2018. Thirty-two individual fish were detected from April 12–July 17. Fifty percent of the fish detected were hatchery fall Chinook salmon. One hatchery fall Chinook, originated from Lyons Ferry NFH at Snake River rkm 95, the remaining fall Chinook originated from Spring Creek NFH located above Bonneville Dam at rkm 269. The second most prevalent fish category detected was northern pikeminnow at 31%. Nine percent of fish detected were hatchery spring Chinook from the middle Columbia River Basin. Two fish (6%) were Chinook from the Snake River Basin which were barged to below Bonneville Dam, and one (3%) was a wild coho, which

was released in the Lewis River and moved upstream to Campbell Slough. Detection numbers and residence times are listed in Table 38.

Residence times were measured by quantifying elapsed time from first to last detection. Overall, residence times at Campbell Slough were variable, but stock/species-specific patterns were observed. Fall chinook exhibited the longest maximum residence time of 34 days, however, residence times were so variable that half of fall chinook detected resided for less than 30 minutes. Numbers of other salmon detected were low and those fish did not show much site fidelity. 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 10 individuals were detected and they demonstrated the longest residence times of all fish. The median residence time was 22 hours and one individual resided for almost three months.

Table 38. Number and residence time (max and median) of fish detected at Campbell Slough PIT array in 2018. Residence time is a measure of elapsed time from first to last overall detection, not a measure of time spend upstream of the array. Numbers in parentheses represent the number of known wild origin fish in the total.

		Residence time	
	N	Max	Median
Fall Chinook	16	34.4 d	29 m
Spring Chinook	3	2 s	1 s
Chinook, unknown run	2 (2)	6 h	3 h
Coho	1 (1)	1 s	--
Northern Pikeminnow	10	84.8d	22 h

The Horsetail Creek PIT detection array was operational from February 28–November 21, 2018. Although not all 10 antennas were operating, we had coverage of three antennas on the downstream side and two antennas on the upstream side of the culvert.

Forty individual fish were detected from April 10–August 29. Thirty-eight percent of fish detected were hatchery spring Chinook, one of which originated from the Clearwater River in the Snake Basin. All other spring Chinook salmon originated from the middle Columbia Basin. The second most prevalent category was hatchery fall Chinook at 23%. Hatchery summer steelhead represented 15% of fish detected. One of the hatchery summer steelhead originated from the middle Columbia Basin while the remaining steelhead originated from the Snake River Basin. Three wild steelhead from the middle Columbia Basin were also detected. One hatchery coho and one hatchery summer Chinook, both from the Snake River Basin, were also detected. Additionally, three northern pikeminnows and two unknown (no tag data in the regional database; www.ptagis.org) fish were detected. Detection numbers and residence times are listed in Table 39.

Residence times were measured by quantifying elapsed time from first to the last detection. Similar to Campbell Slough, residence times at Horsetail Creek were variable, but greater salmonid residence times were observed. The longest residence time was observed in summer steelhead which had a maximum residence time of 67 days. Several other groups of salmon resided for days rather than hours or minutes as observed at Campbell Slough. These groups included spring Chinook, fall Chinook, and steelhead of the unknown run. Northern pikeminnow were also present at Horsetail Creek though in lesser quantities and duration.

Table 39. 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.

	N	Residence time	
		Max	Median
Spring Chinook	15	10.7 d	28 s
Fall Chinook	9	19.1 d	4.65 m
Summer Chinook	1	6.6 h	--
Summer Steelhead	6	67.4 d	8.75 m
Steelhead, unknown run	3 (3)	12.3 d	9.5 m
Hatchery Coho	1	6.9 h	--
Northern Pikeminnow	3	28.2 d	10.9 d
Unknown	2	4.9 h	4.4 h

4 Status and Trends Discussion

4.1 Mainstem Conditions

River discharge fluxes in the Columbia River were relatively low in the late winter through spring prior to the freshet as well as after the freshet and into the summer; however, flows during the freshet reached high levels in 2018, similar to 2017. Unlike in 2017, however, the period of high flow was short in 2018. The low-flow summer conditions 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). There were similar numbers of days with temperatures exceeding thresholds shown to be deleterious to juvenile salmonids; close to 80 days had temperatures exceeding 19℃, ~60 days had temperatures of 20℃, and ~40 days had temperatures at 21℃. The number of day-equivalents for these thresholds has been similar from 2015 onward. 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).

The contribution of total flow by the Willamette River was lower in 2018 compared to 2017, with a brief period of relatively high contribution in early-mid April. Similar to other years, this contribution was evident in water quality parameters in the mainstem, particularly colored dissolved organic matter, which tends to be higher in association with pluvial rather than snowmelt-driven flows (Aiken et al., 2011).

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). Coastal influences in Ilwaco Slough were evident in low dissolved oxygen concentrations (~40% saturation relative to the atmosphere) observed between July and September when salinity increased; similarly low dissolved oxygen saturation levels were also observed at Campbell Slough and Franz Lake Slough. Dissolved oxygen concentrations have been decreasing at Ilwaco in recent years, with the greatest number of observations of low-DO water (at all three thresholds examined, <6 mg/L, <4 mg/L, and <2 mg/L) occurring in 2018. The timing of low dissolved oxygen at the latter two sites coincided with peak freshet flows at the end of May when pH and chlorophyll *a* concentrations declined and water levels dropped. 2018 was somewhat unusual in that primary production was not exceptionally high at Campbell Slough compared to previous years. It is possible that phytoplankton populations were washed out during the relatively brief period of high flow, underscoring the importance of timing of high and low flows for water quality parameters.

pH values were in the target range for good water quality at Ilwaco, Welch Island, Whites Island, and Franz Lake Slough throughout the sampling season in 2018. 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. Levels that exceed 9 lead to a shift in the speciation of ammonium, from its ionic form to the toxic gas, ammonia. High concentrations

of ammonium, which tend to increase as a result of microbial activity, were observed at Ilwaco during the summer of 2017, where values as high as 18 µM were measured; however, at that time pH at Ilwaco did not approach 9, so the observed concentrations were not harmful. In 2018, ammonium concentrations did not exceed ~6 µM, and concentrations were generally <2 µM at the other trends sites.

Water temperatures in 2018 were warmest at Campbell Slough and Franz Lake Slough, where they reached 25°C in August. Similar to 2017, a 20°C threshold was exceeded at Welch Island (~23°C) and Whites Island (~23°C) from mid-July through August, while at Campbell Slough and Franz Lake Slough temperatures exceed 20°C at the end of June. Franz Lake Slough, in particular, showed a sharp increase in temperature at the end of June. Assuming that temperature differences between off-channel sites reflect the degree of mixing and exchange with the mainstem, Campbell Slough and Franz Lake Slough had much lower connectivity to the mainstem than the other sites during periods not associated with high water levels. The percentage of hours with temperatures exceeding 19°C was highest in 2015, where percentages were close to 100 from June through August (September data were not available for comparison). In 2018, the percentage of hours with high temperature was restricted to July and August, with cooler temperatures in September than in 2017.

Similar to 2017 concentrations of nutrients in 2018 were highest in early spring and declined rapidly during the period of spring growth of phytoplankton. Nitrate concentrations were high in March and April at Welch Island, Whites Island, and Ilwaco, while concentrations decline after March at both Campbell Slough and Franz Lake Slough. Interestingly, at Franz Lake Slough, elevated phosphorus concentrations were observed in June. A source of phosphorus to this site could explain the high abundances of cyanobacteria observed there during the summer months since many cyanobacteria species are able to fix atmospheric nitrogen (thus requiring only phosphorus, inorganic minerals and trace elements, and carbon for growth). 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 Welch Island, Whites Island, and Franz Lake Slough in the spring, but high concentrations were only observed during a window of approximately one month (March–April). Between April and May, nitrate declined by ~20 µM, reflecting a combination of drawdown by phytoplankton growth and an increasing proportion of snowmelt-driven discharge.

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.

In 2018 maximum flood levels occurred across the upper river sites, Franz Lake, Campbell Slough, and Cunningham Lake in May during the freshet, these high flow conditions were less apparent at the mid-river sites, Whites Island, Welch Island, and not detected at the lower river site, Ilwaco Slough (Appendix C). Ilwaco Slough typically seeing maximum flood conditions in the winter when high king tides correspond with winter storm events. 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 3) 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 52). 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 and erosion rates within the EMP reference wetlands could provide critical information about wetland succession, carbon accumulation, and resilience to future changes anticipated from sea level rise. The 2017 and 2018 sediment accretion and erosion data for the EMP sites are still under analysis; however past results discussion have been provided below. 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).

 From the FY2016 EMP Report:
"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 (Chmura et al., 2003) and may even be affected by the vegetation present (Marani et al., 2013). We installed sediment accretion stakes at different elevations within a site in an effort to capture this variability and generally found that accretion rates decreased with increasing elevation, however, proximity to the tidal channel also seems to be an important variable. For example, a natural levee occurs along the channel at the Secret River site caused by increased sedimentation at the channel bank (Cazanacli and Smith 1998) and, sedimentation continues to be higher there than at the marsh surface farther from the channel. Sediment accretion rates in the low marsh at the Secret River site have consistently been low or erosional since we started measuring in 2012. This could in part be due to decreased sediment deposition during times when wave action takes the sediment higher into the marsh at high tide (Davidson-Arnott et al., 2002). The site is likely often exposed to wave action due to the large fetch coming from the west, the prevailing wind direction in the lower estuary from March to October when vegetation is present and more likely to trap sediments (https://www.windfinder.com/windstatistics/astoria). 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 (1.3 m, CRD) 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 TFW marshes (Neubauer 2008).

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 TFW 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 TFW 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 TFW marsh sediments (Craft 2007; Thom 1992) where organic material may account for more of the accretion. In general, TFW 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."

4.2.2 Vegetation Community Condition and Dynamics

Overall, 2018 total plant cover was similar to 2017, however, site to site variability in cover was greater than 2018. Ilwaco Slough had lower total cover in 2018 compared to 2017, but slightly higher than the long-term average, Welch Island and Whites Island total cover was similar to 2017, Cunningham Lake total cover increased between 2017 and 2018, rebounding from the heavy cattle grazing observed in 2017, and Campbell Slough had similar low total cover levels from 2018 to 2017, cattle grazing occurring both years. Franz lake total cover in 2018, was much lower than in 2017, but near long-term average levels. Between 2012-2018 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 24, Figure 42 & Figure 43).

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 40 & Figure 41), 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). In 2018, mean relative native cover was greatest at Welch Island, followed by Cunningham Lake, Ilwaco Slough, Franz Lake, Whites Island, and lowest at Campbell Slough (Table 23). Compared to long-term averages, native cover was generally lower in 2018 across all sites except Cunningham Lake (66±29%, long-term average 54±16%) (Table 23). Over time, there has been a slight decrease in relative native cover at Ilwaco Slough and Welch Island, native cover has seen both an overall increase and then recent slight decrease across time at Franz Lake, Cunningham Lake, and Campbell Slough (Table 23, Figure 40 & Figure 41). Relative native cover at Whites Island has remained relatively stable only fluctuating around 10% between 2012-2018.

Compared to the long-term site averages, non-native cover has increased in 2018 at Ilwaco, Welch Island, and decreased at Cunningham Lake and Franz Lake (Table 23). 2018 averages are similar to their long-term trends at Whites Island and Campbell Slough (Table 23). Shifts in relative native and non-native cover seen between 2017 and 2018 at Ilwaco Slough and Welch Island are likely partially due to the positive identification of a previously common grass species *Agrostis sp.* (no native or non-native status identified in past years) to the non-native *Agrostis stolonifera*.

In 2018, non-native *P. arundinacea* cover levels stayed relatively consistent to those observed in 2017, however, at Whites Island and Cunningham there was a significant drop in *P. arundinacea* levels from 42 to 30% and 25 to 17%, respectively (Table 24 & Table 25, Figure 47). This drop in cover was not translated into a drop in *P. arundinacea* frequency (spread across the site) which increased across all sites (Table 24). Shifts in *P. arundinacea* cover at Cunningham Lake may partially be in response to the heavy grazing the site received in 2017, which is known to reduce *P. arundinacea* abundance (Kidd & Yeakley 2015). Additionally, annual shifts in *P. arundinacea* cover were found strongly correlated with mean % daily inundation for the month of August across Welch Island, Whites Island, Cunningham Lake, and Campbell Slough and correlated with the Columbia River freshet levels (accumulated discharge May-Aug) at Cunningham Lake, Campbell Slough, and Franz Lake (Figure 46). 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 48). 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.

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 53).

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 29, Table 30). 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 29, Figure 49). 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 50). 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 63). 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 61, Table 34), 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 62, Figure 63). 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 59). This relationship was also found in the detritus, with detrital C and N having a positive correlation across all sites (Figure 62). 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 35, Table 36). 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 36), 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*

Over the last number of years, pelagic productivity has been high at Franz Lake Slough, which reached a peak >100 µg L^{-1} in 2017. During 2017, there was a peak of ~40,000 cells/mL at Franz Lake Slough, which was dominated by both diatoms and cyanobacteria. Values in 2018 were not that extreme, but they were higher at Franz Lake Slough compared to other sites. In contrast to previous years, chlorophyll a values at Campbell Slough were similar to those observed at Welch Island and Whites Island. As in previous years, the phytoplankton assemblages differed among the fixed sites, with those at Ilwaco being the most different from the others due to the stronger marine influence and higher tidal range. The site most similar to the mainstem in terms of broad taxonomic representation was Whites Island, similar to other years (Tausz, 2014). Give that it is situated in Reach C, designated a transport reach (Simenstad et al., 2011); this similarity is not surprising.

Peak biomass of phytoplankton occurred during the spring at Whites Island and at Welch Island; in contrast, primary production at Franz Lake Slough was highest in June. The phytoplankton assemblage consisted of small pennate diatoms, chlorophytes, cryptophytes, and cyanobacteria. This peak coincided with increased dissolved phosphorus concentrations and a decline in river discharge flux. 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 indicated that the blooms included cyanobacteria, since peaks in chlorophyll coincided with peaks in phycocyanin, a pigment found mainly in cyanobacteria. While cyanobacteria were present in the sample from June, abundances were not as high as was observed in 2017 when visual inspection confirmed that bloom material consisted of colonial cyanobacteria including Dolichospermum, Aphanizomenon, Microcystis and others. 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). The ecological and health implications of these blooms are not well known for the lower Columbia; 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.). 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 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, which was a warmer year than 2017. In each of the years between 2009 and 2017, 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.

4.1 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. Similar to other observations from reservoirs in the Columbia-Snake Rivers (e.g., Emerson et al., 2015), the assemblages at Welch Island and Whites Island tend to be dominated by rotifers early in the spring, after which time copepods increase in abundance. Cladocerans increase in abundance into the summer months. Both Campbell Slough and Franz Lake Slough tend to support high abundances of both copepods and cladocerans.

There is a large size range in the rotifer species that are prevalent in spring samples at the trends sites, with most falling in a size class that would be unable to consume the very large phytoplankton colonies that are observed during the period of strong spring growth (e.g., *Aulacoseira granulata*, *Fragilaria crotonensis*). For this reason, we tracked the prevalence of parasitism by these colonial diatom species by chytrid fungi, since parasitism effectively repackages the carbon associates with large, inedible taxa into small, nutritious zoospores which can be consumed by smaller grazers from copepod nauplii to the smaller rotifers and ciliates. Analysis of the carbon likely made available by parasitism and the parasitic shunt is ongoing at this time.

4.2 Macroinvertebrates

Macroinvertebrate data collected in 2018 are still under analysis and unavailable at the time of the writing of this report. This report presents results of the 2017 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).

Prey availability and selection has energetic consequences for the foraging fish. Cladocera rank as the least energy dense prey item juvenile Chinook will typically consume in the Columbia River estuary (David et al. 2016, Kidd et al. 2017). While measures of instantaneous ration were similar from Campbell Slough in 2017 to previous years, energy ration was lower in 2017 when fish were consuming primarily cladocerans. This means that fish were eating enough of the small-bodied cladocerans to maintain stomach fullness, but were not acquiring the same amount of calories as in previous years. In fact, in 2017, Campbell Slough fish ate on average 118 individual prey, compared to an average 40 individual prey in previous years. There may have been increased energetic costs associated with the capture and consumption of these small prey, but this may have been offset to some extent by the very high prey densities (e.g., expending less search time for abundant zooplankton prey vs. rarer insect prey).

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). This study examined the interaction of maintenance metabolism, driven by water temperature and fish size, with energy ration, driven by prey availability and quality. Most sites early in the season (February to May), had at least one occurrence where both high energy assimilation and low metabolic cost occurred in juvenile Chinook salmon. Few fish experienced both low energy assimilation and high metabolic costs during a single sample event. Rather, a trade-off often occurred in the warmer months where metabolic costs were relatively high, but energy dense prey, such as hemipterans, hymenopterans, and other insect taxa, were consumed. Juvenile Chinook salmon at Franz Lake consistently had below average energy rations. Though samples were only obtained in April and May, these fish also had relatively low metabolic costs, which may offset, to an unknown degree, the quantity and quality of prey consumed.

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.

4.3   Fish

In 2018, fish community composition was sampled at five trend sites—Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake— from February through June/July, and October. Franz

Lake was sampled in March only due to high water levels or high water temperatures during other sampling months.

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 and Franz Lake were more evenly distributed among species and included a greater percentage of non-native fishes. However, in previous years the species richness and diversity of the fish community at Campbell Slough were much greater than the lower river sites. In 2017-2018 the species diversity and richness at Campbell Slough was similar to the lower river sites. The timing of sample collection may explain this apparent decrease in diversity. In 2017 and 2018 sample collection concluded in June or early July, whereas in previous years sample collection typically extended into late July and August. During these months water temperature warms considerably and the proportional species composition of non-native, warm water fishes increases. Indeed, the June and later, October, measures of species richness and diversity at Campbell Slough in 2018 were much greater than earlier in the year and indicates an increasing trend for the summer and fall months. 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 on incoming tides. Coho salmon have frequently been observed at Franz Lake. In the recent low flow years of 2015-2016, they were less abundant, but in 2018 there was an increased proportion of coho, though the overall catch numbers were small. 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 were the predominant stock of Chinook observed at the lower river sites: Ilwaco, 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.

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.

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). In 2018 we saw huge increases in densities of juvenile chinook salmon at Welch and Whites Island. These increases were driven by large catches during May, and might be related to the change in gear type from a Puget Sound style beach seine to a marsh style beach seine with deeper wings. 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 (2005-2013, 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 and day of year correlated with growth rates. Larger fish grew faster than smaller fish and fish collected in March experienced higher growth rates than fish samples later in the season. 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.

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 have 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 134). 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.
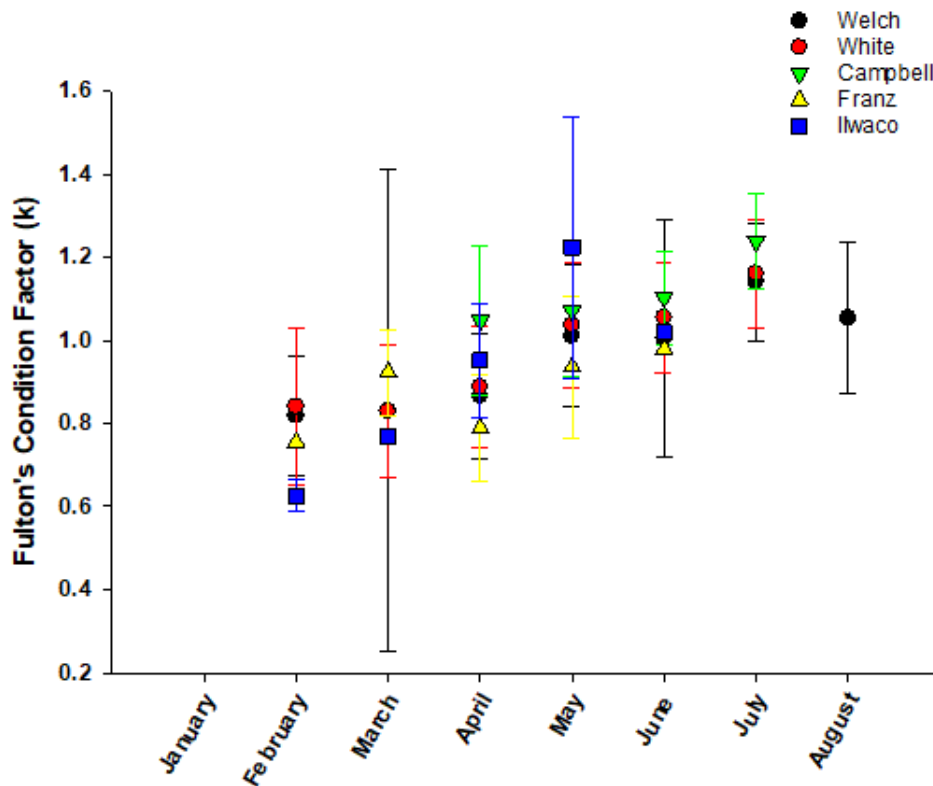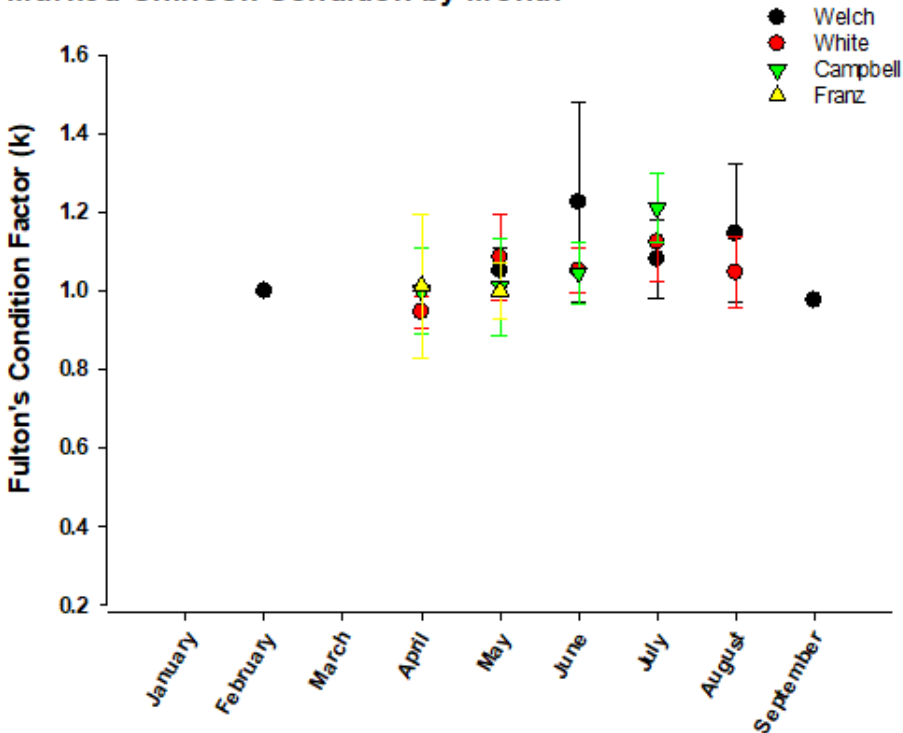
Figure 134. Monthly mean (SD) Fulton's condition factor of unmarked and marked Chinook Salmon, 2008-2017.

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 135). 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.
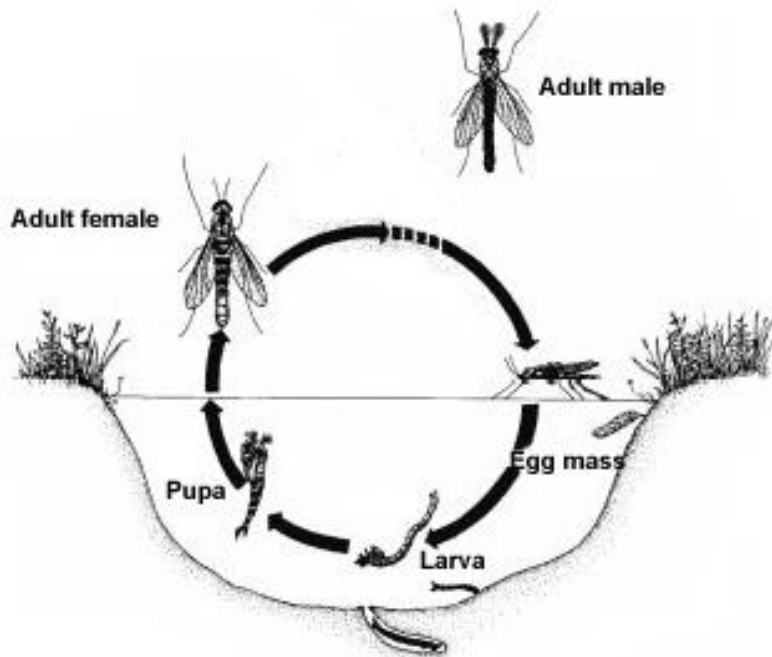
Figure 135. Life cycle of chironomid flies (image from North Carolina State University, Cooperative Extension).

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 136). 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).

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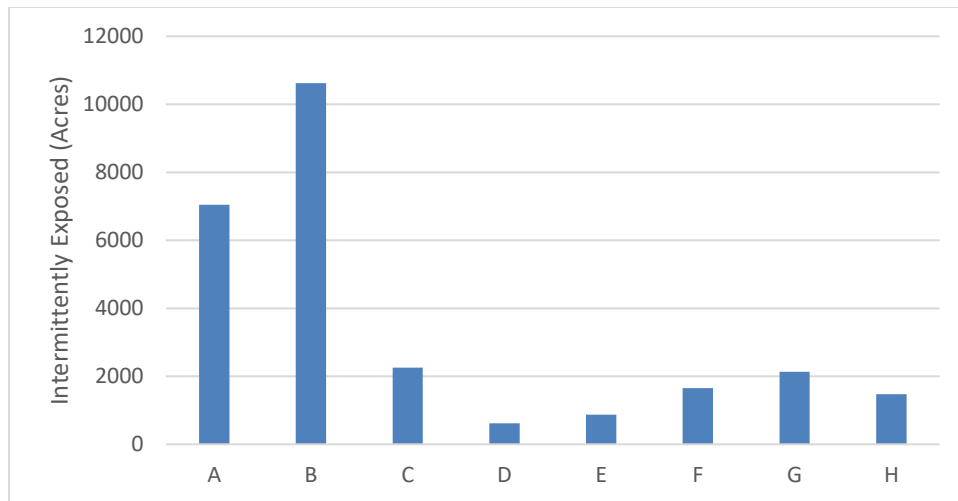
Figure 136. Area (acres) of the Intermittently Exposed Geomorphic Catena by Hydrogeomorphic Reach, as mapped in the Columbia River Estuary Ecosystem Classification.

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.

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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 137). 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).
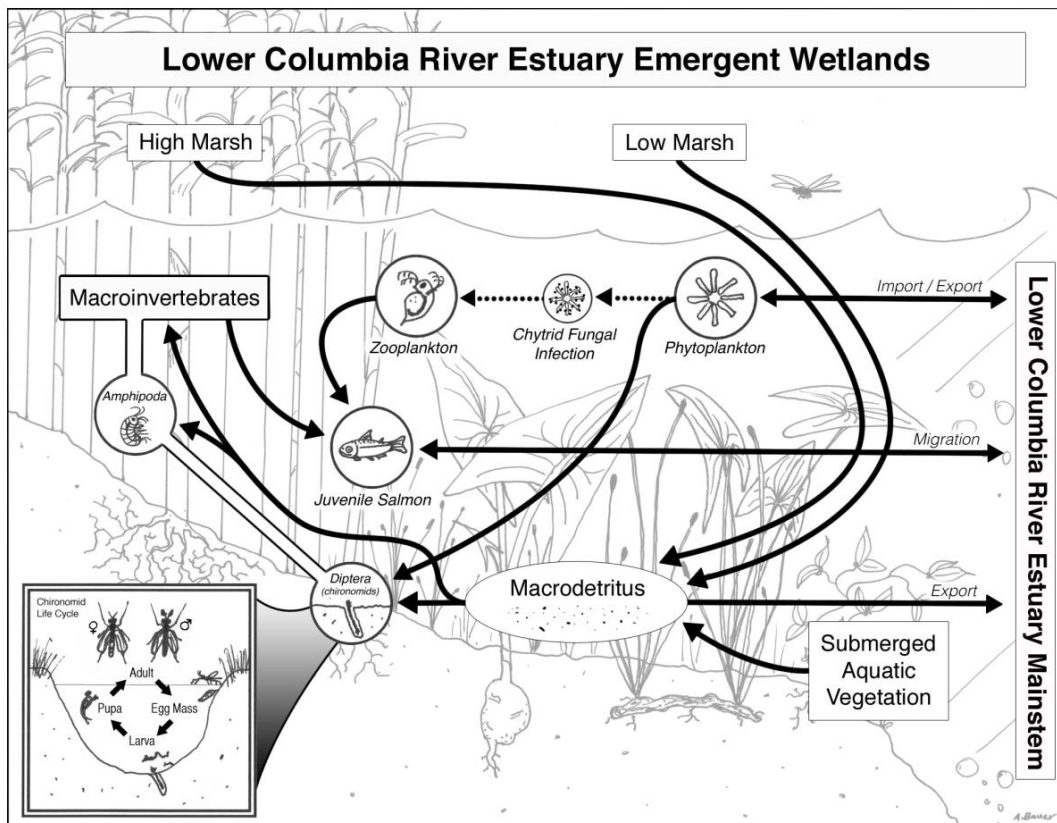
Figure 137: Conceptual model of food web interactions within Lower Columbia River emergent wetlands.

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 40. 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 40. 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 138, 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.
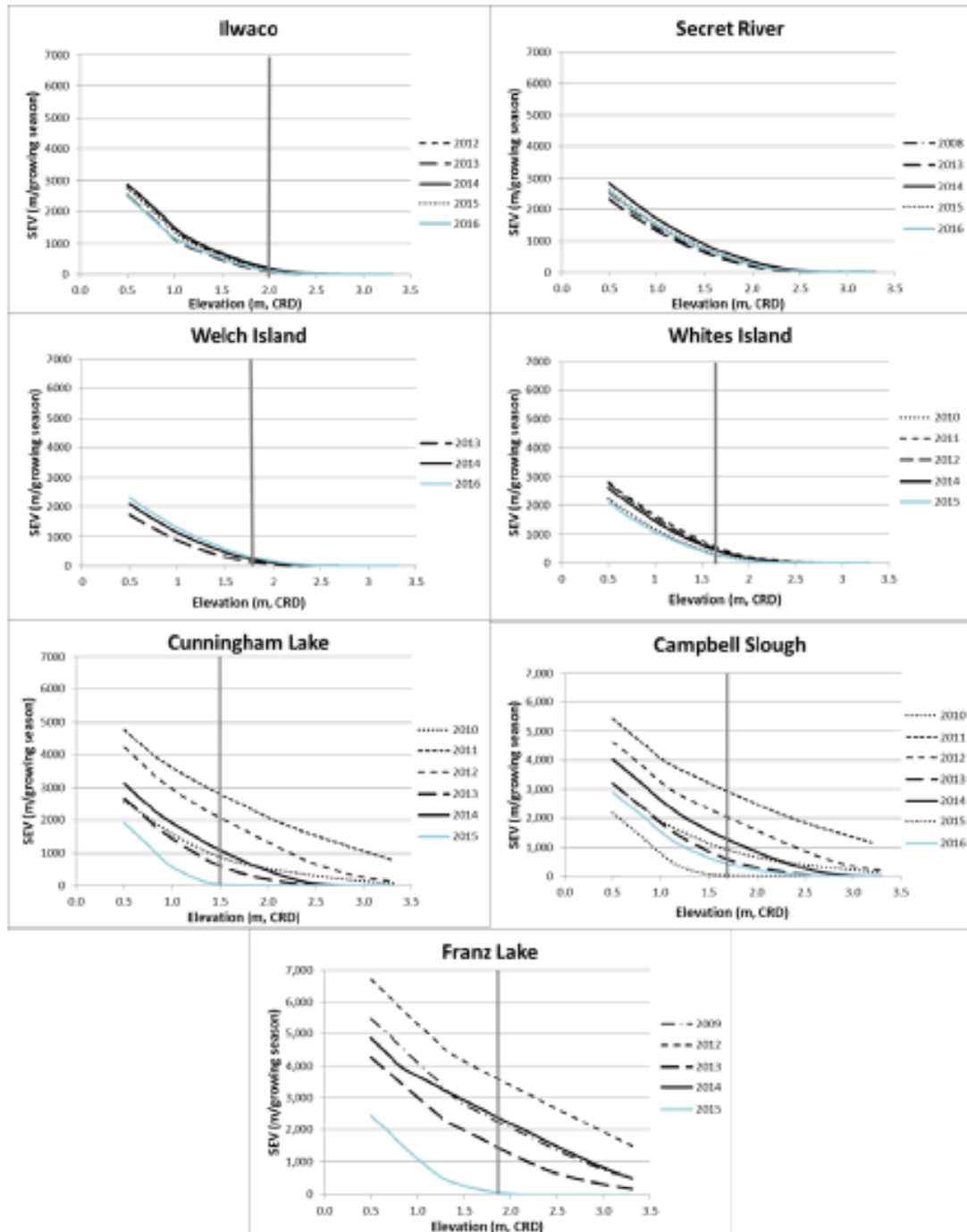
Figure 138. Cumulative inundation of wetlands during the annual growing season, as measured by sum exceedance values (SEVs). Plotted lines represent the calculated SEVs for a given year; the blue lines indicate the most recent year available. The vertical line represents the average elevation at each site. Sites are ordered with the site nearest the Columbia River mouth in the upper left panel.

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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 139). 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 140). 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.
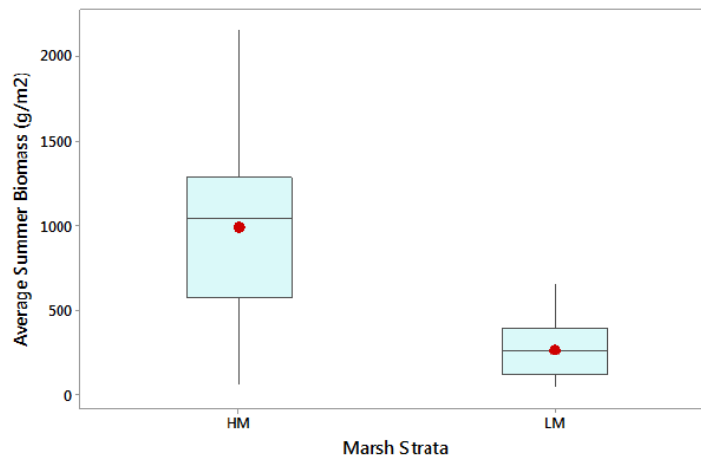
Figure 139. Overall average summer biomass (g dry weight/m^2) from the high marsh (HM) and low marsh (LM) strata.
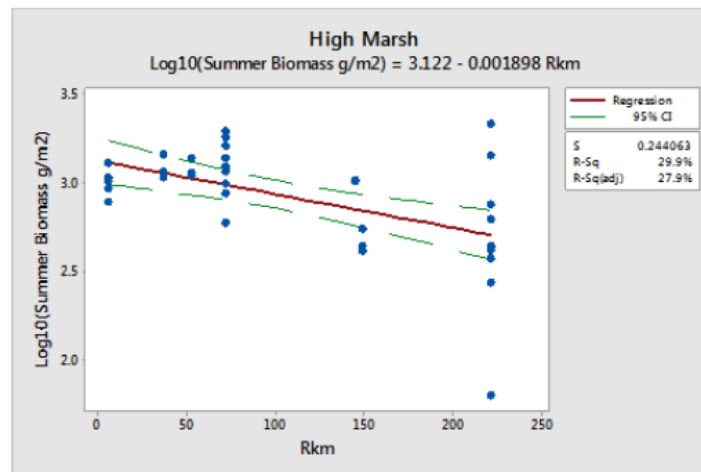
Figure 140. Average annual summer biomass (g dry weight/m^2) compared to river km for the high marsh strata. Results are transformed by Log10 for statistical analysis.

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. 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. 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. Efforts to conduct additional sampling across the tidal range and at high tide may produce results that differ from those derived using traditional methods.

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. Similar to 2017, in 2018, a higher than average number of days where water temperatures exceeded recommended temperature maximums in the mainstem were observed (2015 and 2017 are the only other years that surpass 2018 in this regard). Such a result is concerning since 2018 river discharge patterns were considered to be relatively high compared to other years, yet these higher flows occurred early in the season and did not buffer the warmer temperatures observed. 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 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 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.

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 2019. 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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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

N

0   25   50   100   km

0   50   100   200   Meters
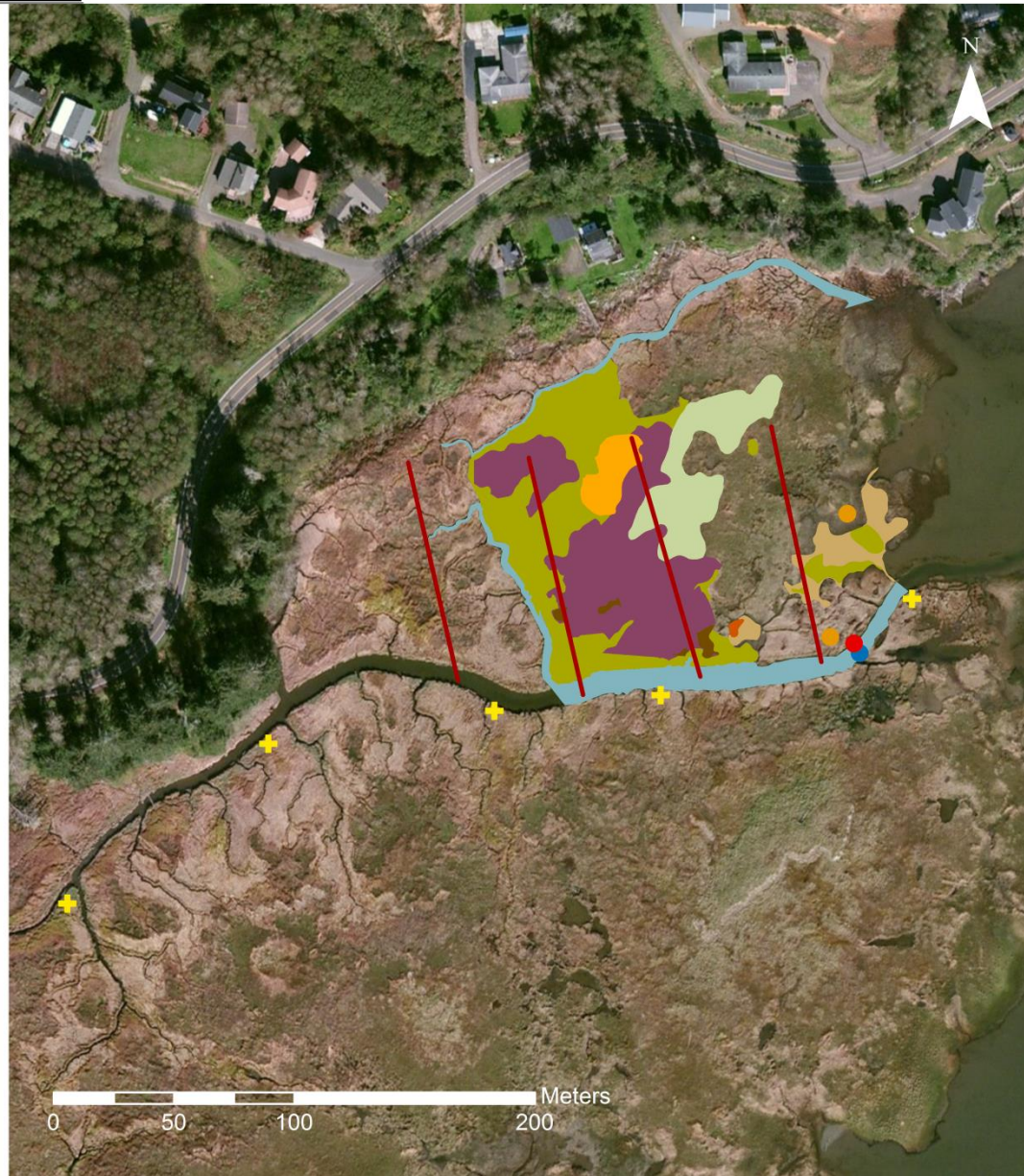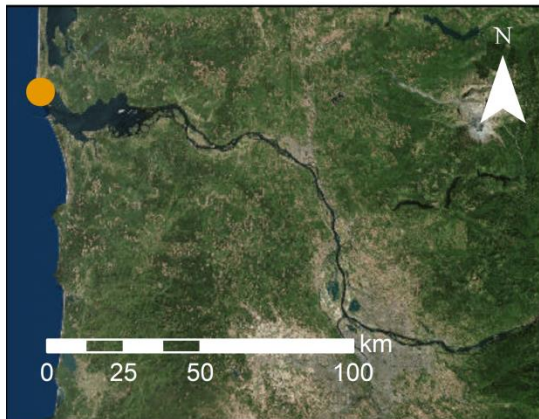
Secret River Marsh, 2013

GPS Mapping

- Bare ground
- Carex lyngbyei/P. arundinacea
- Channel
- D. cespitosa and P. arundinacea
- Deschampsia cespitosa
- E. palustris and S. tabernaemontani
- Lythrum salicaria
- Mixed P. arundinacea
- Phalaris arundinacea
- Schoenoplectus tabernaemontani
- Sparganium eurycarpum
- Sparganium eurycarpum/P. arundinacea
- Submerged aquatic vegetation
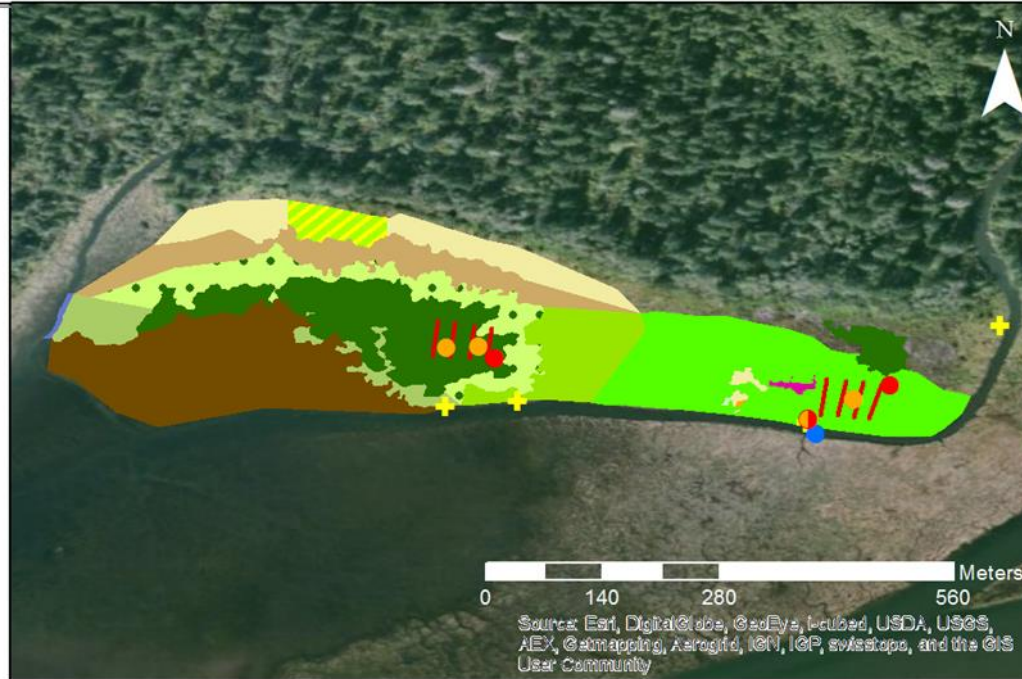
Monitoring Locations

- Photo point
- Depth sensor
- Sediment accretion stakes
- Sediment accretion stakes/Photo point
- Cross section
- Vegetation/Elevation Transect

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

Meters
0   140   280   560

Kilometers
0   3.25   6.5   13

276

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
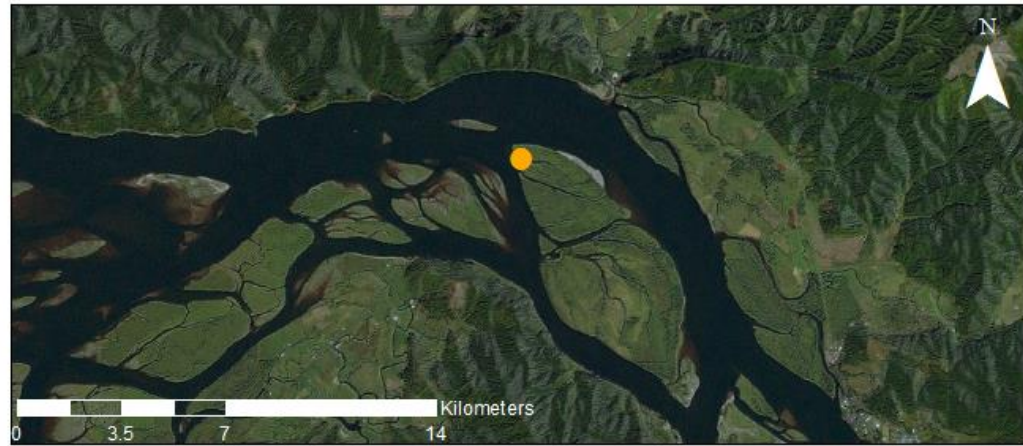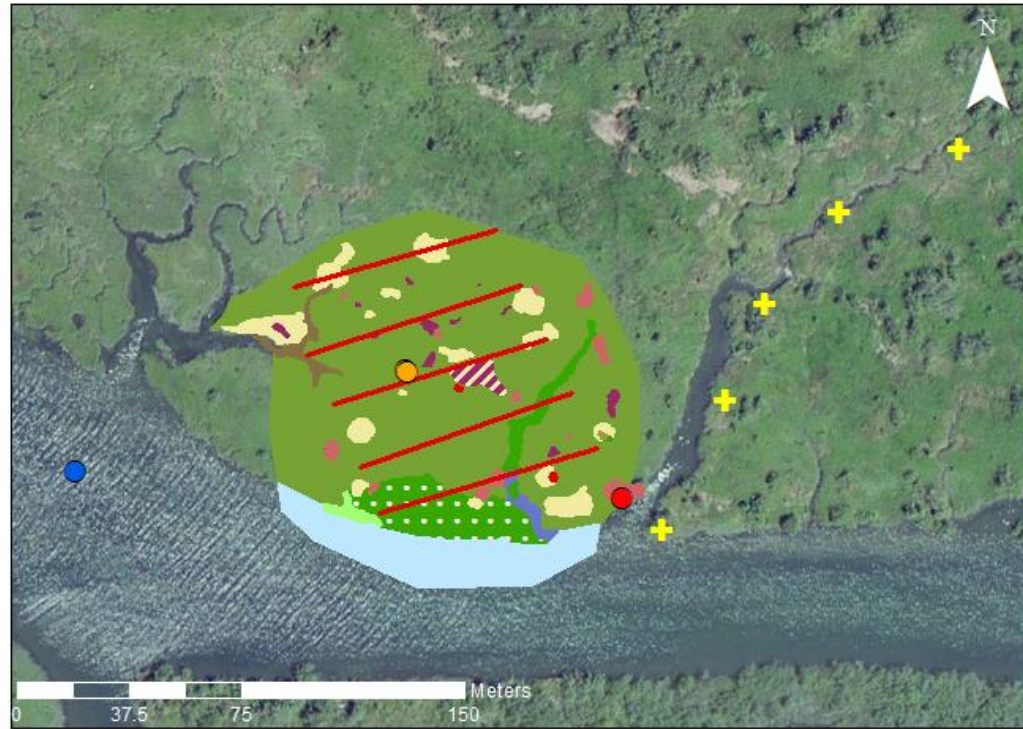
White's Island, 2015
GPS Mapping

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

278
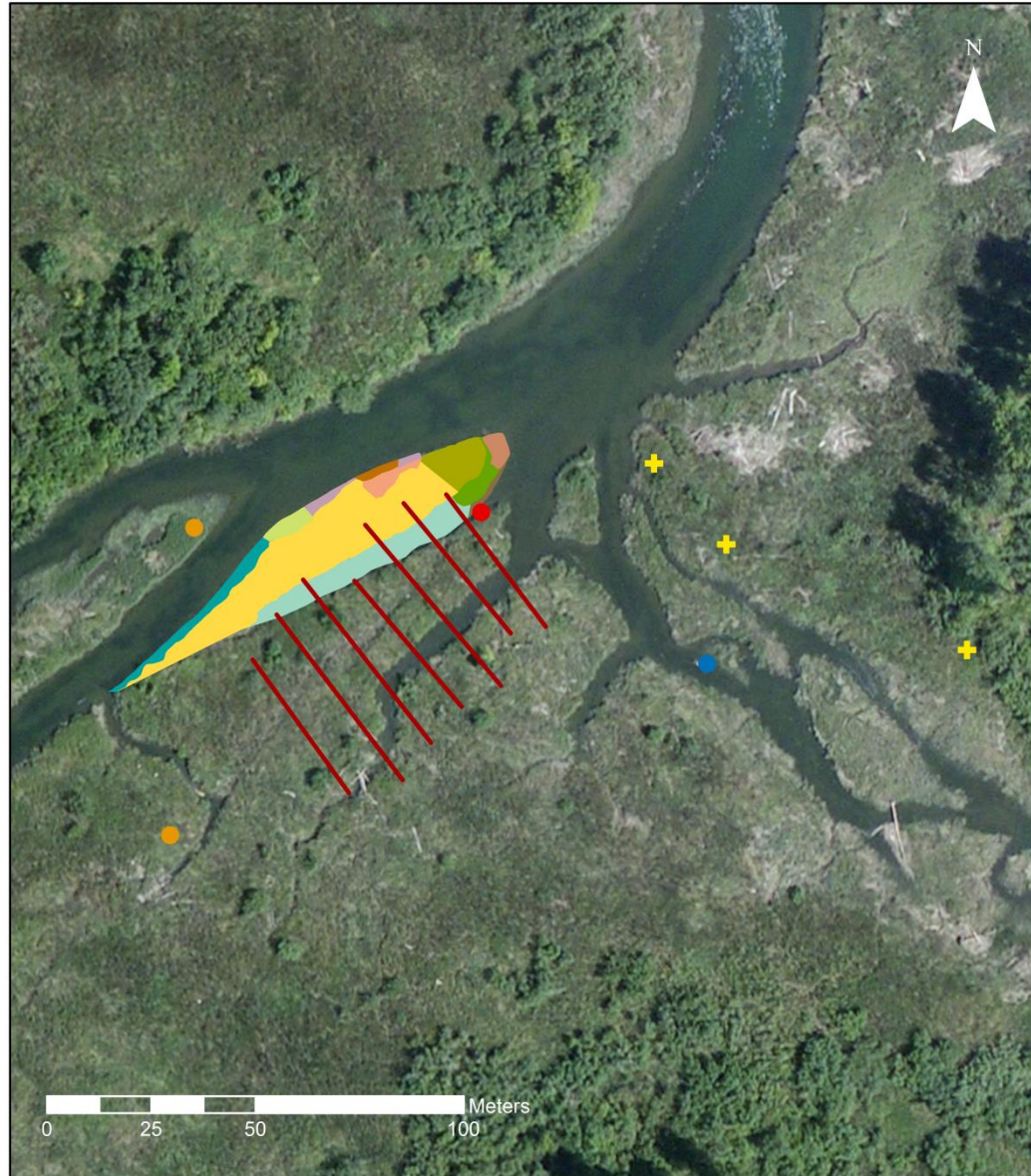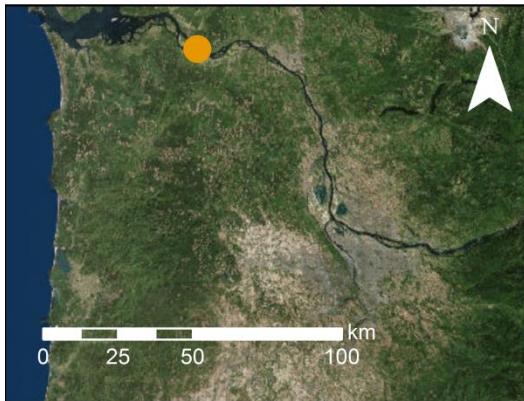
Cunningham Lake, 2015
GPS Mapping

Vegetation Communitites

- Eleocharis palustris, S. latifolia
- Phalaris arundinacea
- P. arundinacea, S. latifolia
- Sagittaria latifolia
- Salix spp.
- Mud

Monitoring Locations

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

279
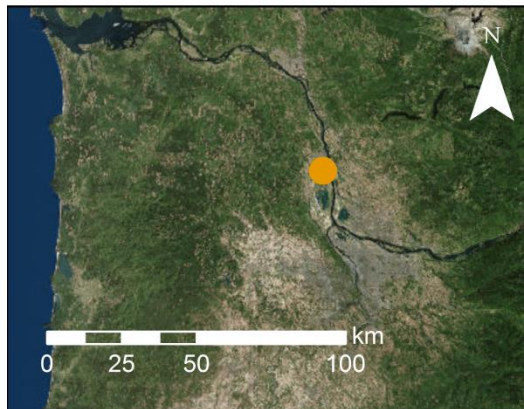
Campbell Slough, 2015
GPS Mapping

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

280
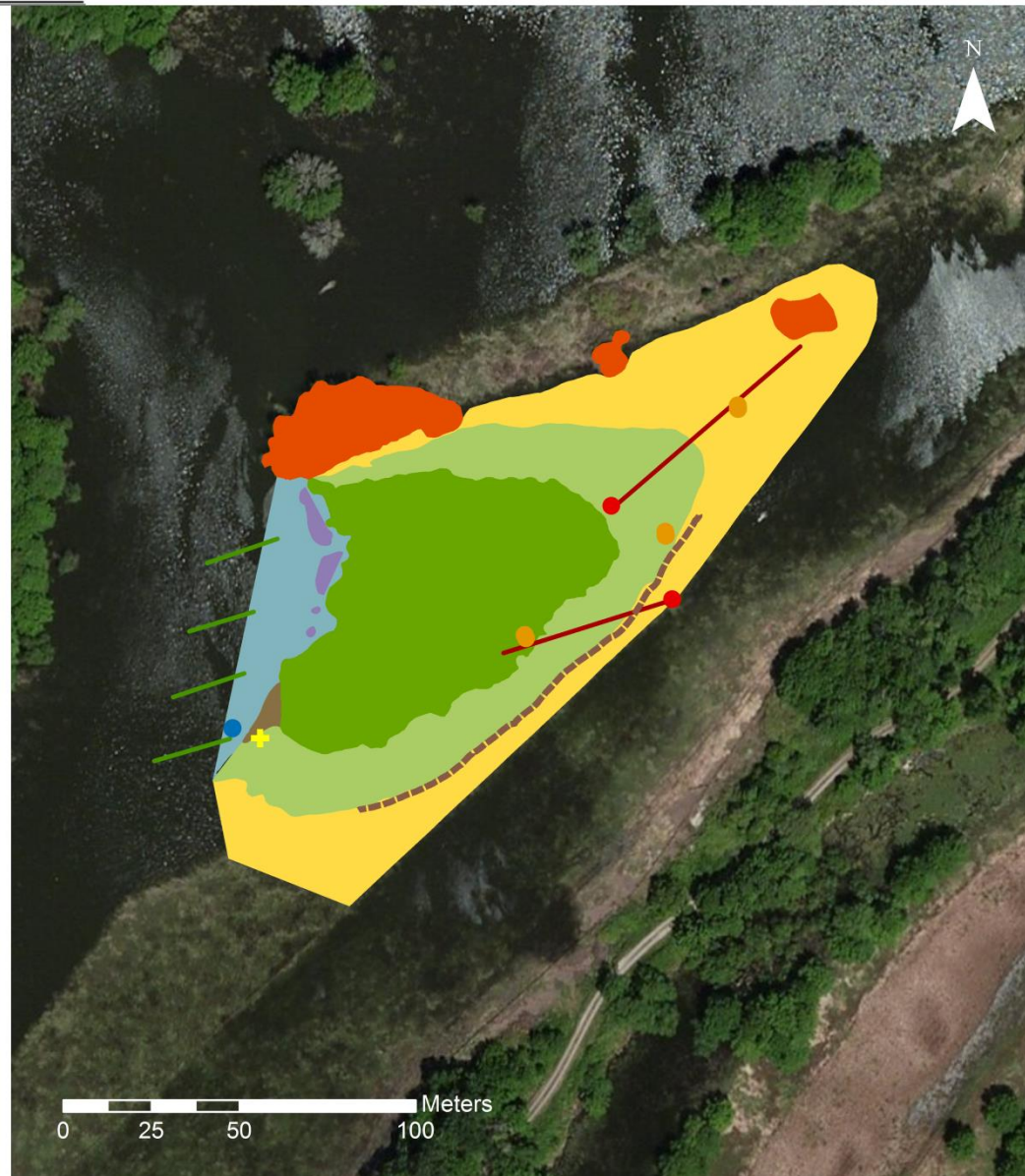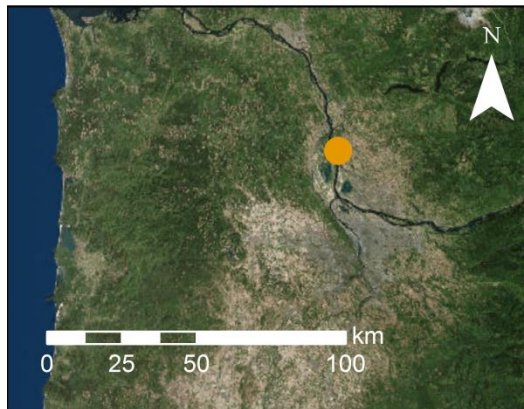
Franz Lake, 2015

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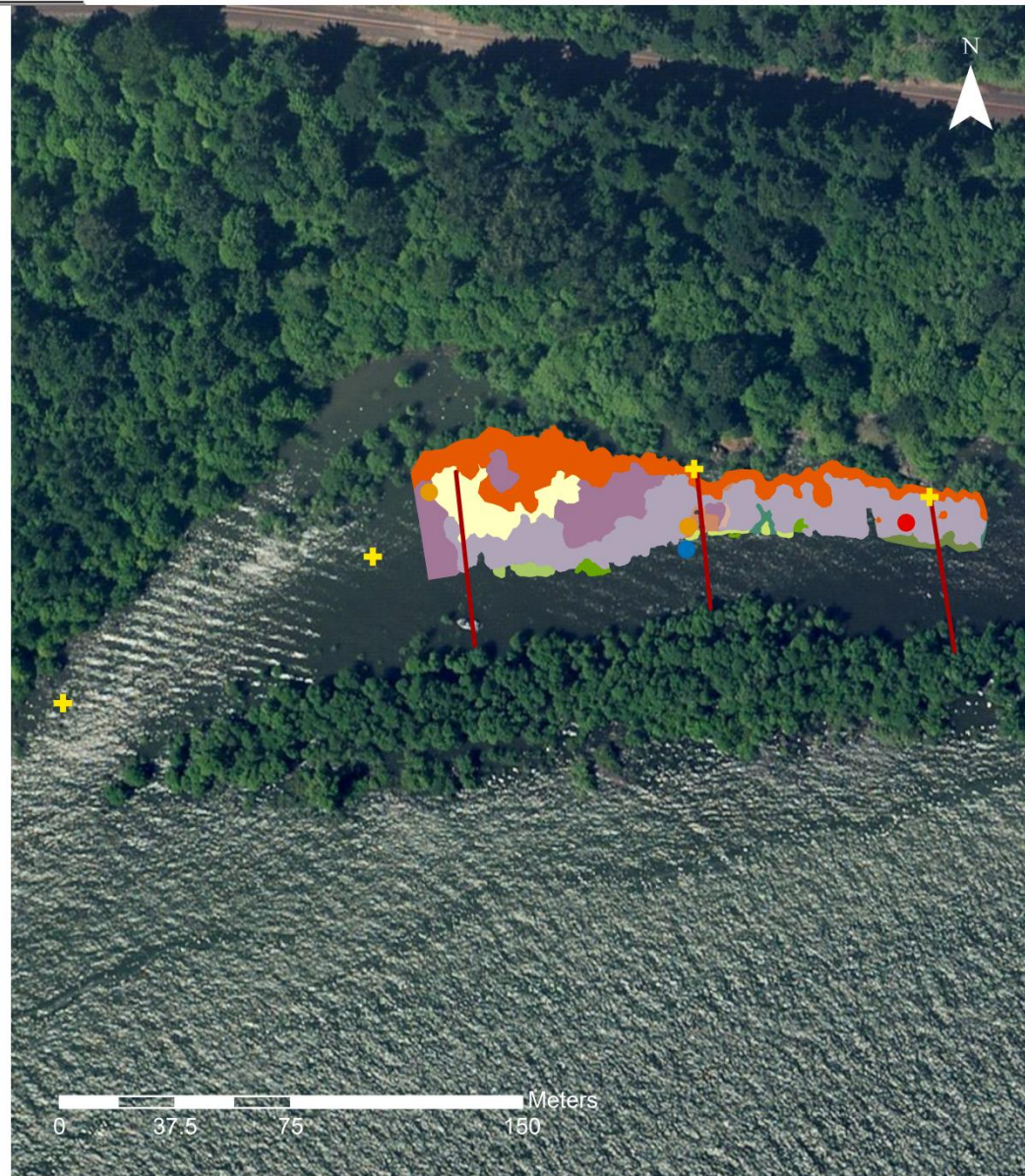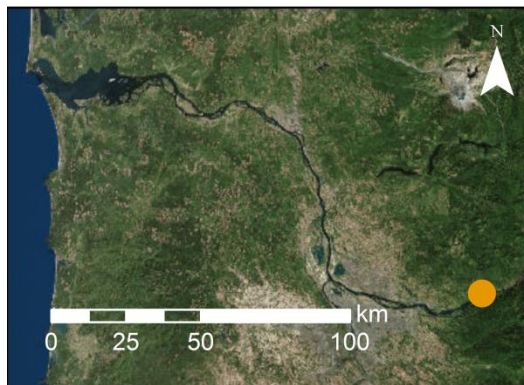
Table A1. Habitat change analysis of vegetation communities at the trend sites; comparison of overlapping areas for the earliest year mapped and the latest year mapped. All area units are square meters. Vegetation communities are ordered from the lowest elevation to the highest elevation at a site; species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

Ilwaco

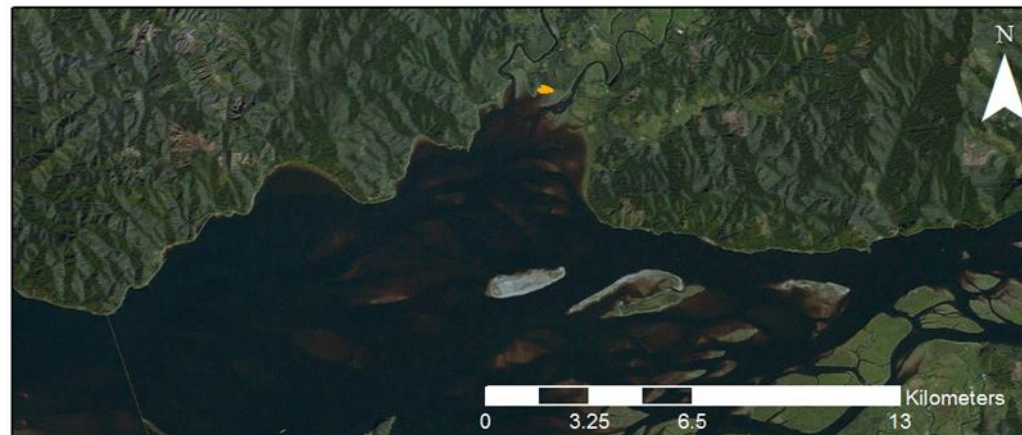|			2011 Vegetation Community				
Area Compared:	13312						
			Channel, ZAPA	Pan	CALY	AGSP, CALY	TYSP
Area Changed:	6416						
No Change:	6895		1558	383	6455	4792	134
2015 Vegetation Community Channel	2164		1548				
Pan	804			241	559	4	
Bare	127				68	58	
CALY	3898		10	142	3045	85	
AGSP, CALY	4048	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

Welch Island

			2012 Vegetation Community						
Area Compared:	1126								
			Channel	SALA	CALY, high marsh	PHARSALA	PHAR	PHAR LYSA	LYSA
Area Changed:	603								
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						
					ELPA, SALA	ELPA, SCAM	CALY	PHAR
Area Changed:	729	Channel	SALA	ELPA				
No Change:	855	163	252	115	18	82	191	763
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

2015 Vegetation Community (row axis label)

Cunningham Lake

Area Compared:	4033	2006 Vegetation Community					
			ELPA, SALA		PHAR, ELPA		
Area Changed:	1800	SALA		ELPA		PHAR	SASP
No Change:	2232	1059	1041	44	278	492	1118
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

2015 Vegetation Community (row axis label)

Campbell Slough

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

2015 Vegetation Community (row axis label)

Franz Lake

	Area Compared:	1762	2008 Vegetation Community					
	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	Channel, SALA	ELPA, SALA	ELPA	PHAR, POAM	SASP	Rock
Area Changed:	4336						
No Change:	1384	**1417**	**24**	**488**	**2177**	**1579**	**35**
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

(2012 Vegetation Community — row labels)

Franz Lake

		2012 Vegetation Community												
Area Compared:	5203	Bare Ground	Beaver Activity	Channel	Channel, SALA	SALA	ELPA	CASP	POAM, SALA	POAM	PHAR	SASP, PHAR	SASP, CASP	SASP
Area Changed:	2344													
No change:	2859	9	14	194	50	5	193	192	25	2159	196	279	20	1865
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

(2015 Vegetation Community — row labels)

Baker Bay
Vegetation Change 2011 to 2015

2011

2015

N

Meters
0 25 50 100

N

Meters
0 25 50 100

N

Meters
0 25 50 100

Vegetation Communities

- AGSP, CALY
- AGSP, DECE, GLSP
- Bare
- CALY
- Channel
- Channel, ZAPA
- Pan
- TYSP

Vegetation Change

- AGSP, CALY to Bare
- AGSP, CALY to CALY
- AGSP, CALY to AGSP, DECE, GLSP
- AGSP, CALY to Pan
- AGSP, CALY to TYSP
- CALY to AGSP, CALY
- CALY to Bare
- CALY to Pan
- CALY to TYSP
- Channel, ZAPA to CALY
- Channel, ZAPA to Channel
- Pan to CALY

286

Welch Island
Vegetation Change 2012 to 2015
PHAR Specific

2012

2015

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

287
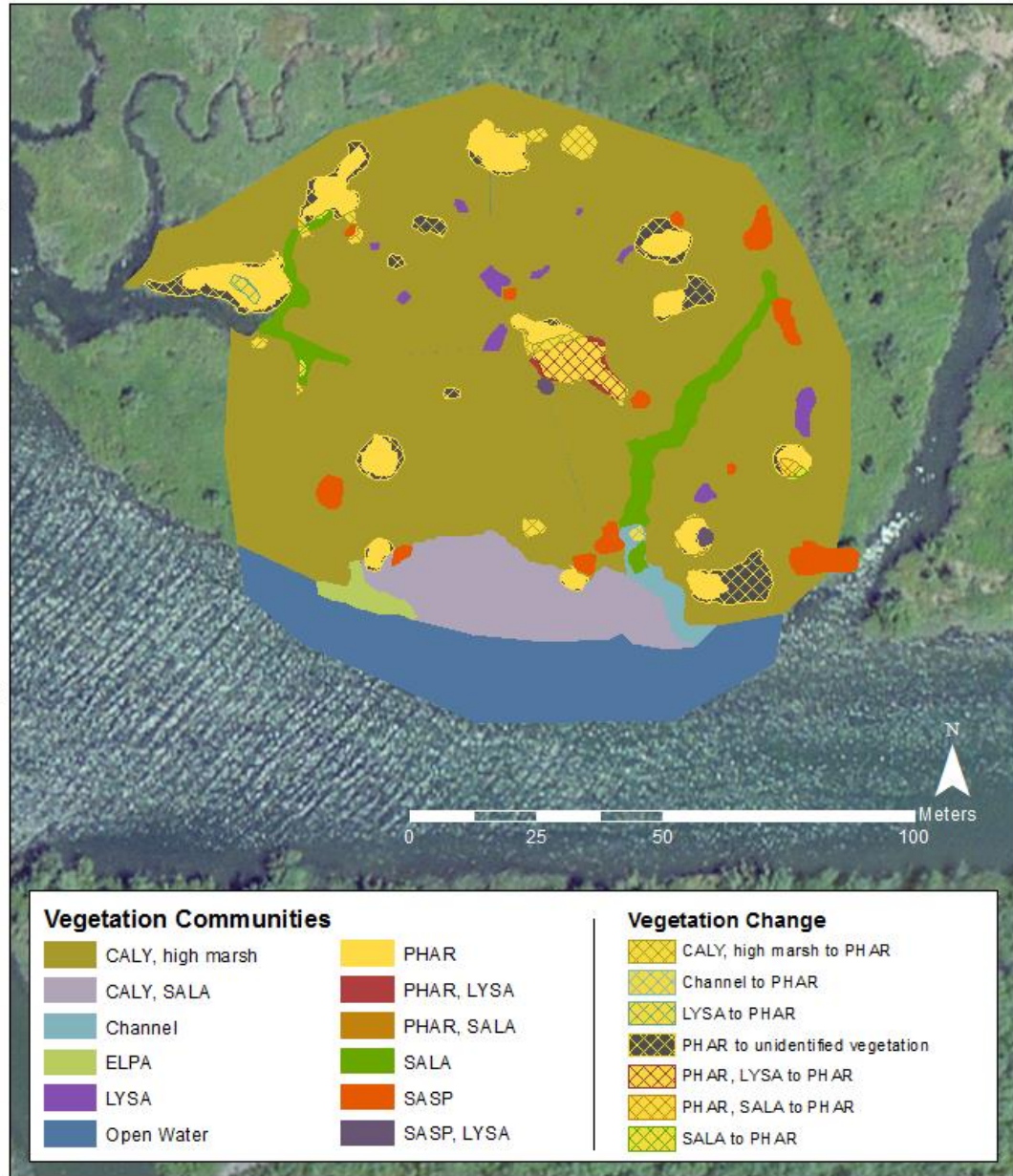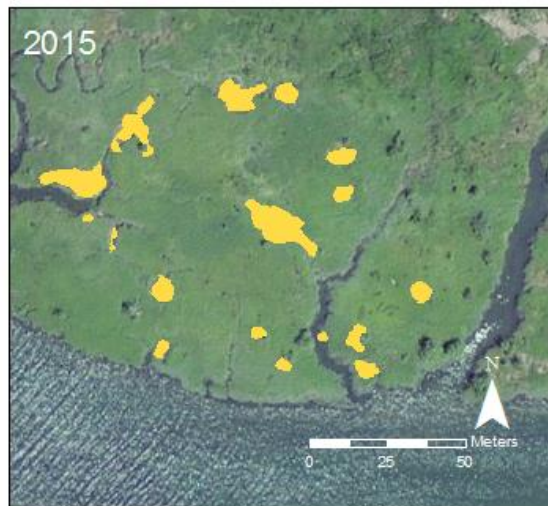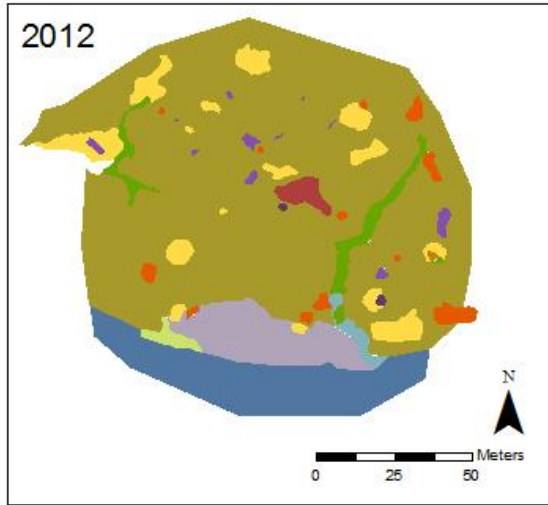
White's Island
Vegetation Change 2009 to 2015

2009

2015

Vegetation Communities
- ALPL, BICE, SALA
- BICE, ALPL
- CALY
- Channel
- ELPA
- ELPA, SALA
- ELPA, SCAM
- Mud
- PHAR
- SALA
- SALA, ELPA, BICE
- SCAM
- SCAM, CALY
- SCAM, MIGU

Vegetation Change
- CALY to PHAR
- CALY to SALA
- CALY to SCAM, CALY
- Channel to ALPL, BICE, SALA
- Channel to BICE, ALPL
- Channel to ELPA
- Channel to Mud
- Channel to PHAR
- Channel to SCAM, CALY
- Channel to SCAM, MIGU
- ELPA to PHAR
- ELPA to SALA
- ELPA to SALA, ELPA, BICE
- ELPA to SCAM
- ELPA, SALA to PHAR
- ELPA, SALA to SALA, ELPA, BICE
- ELPA, SCAM to ALPL, BICE, SALA
- ELPA, SCAM to BICE, ALPL
- ELPA, SCAM to ELPA
- ELPA, SCAM to PHAR
- ELPA, SCAM to SCAM, MIGU
- PHAR to ALPL, BICE, SALA
- PHAR to SALA, ELPA, BICE
- PHAR to SCAM, CALY
- SALA to Mud
- SALA to SALA, ELPA, BICE

288

Cunningham Lake
Vegetation Change 2006 to 2015

2006

2015

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

289
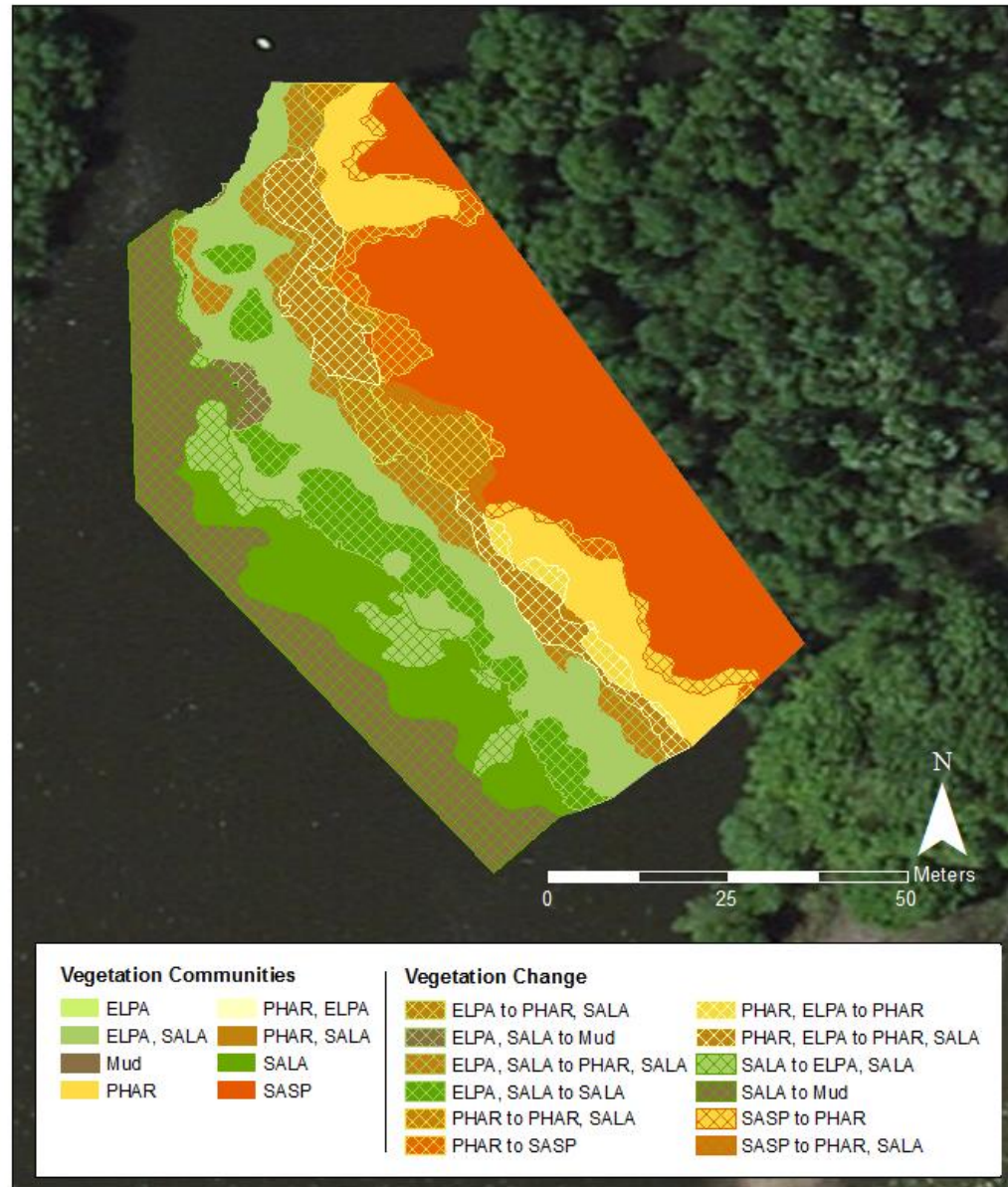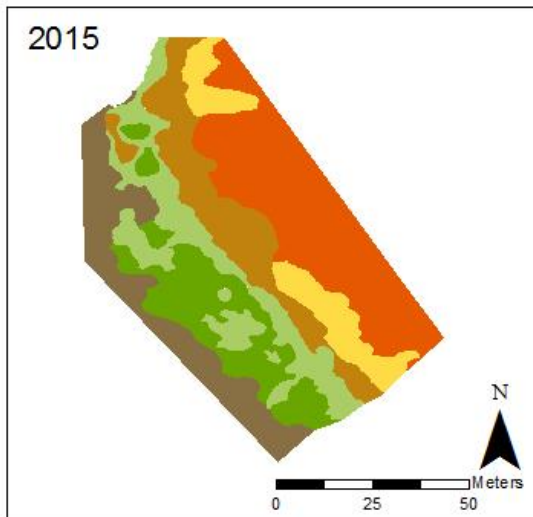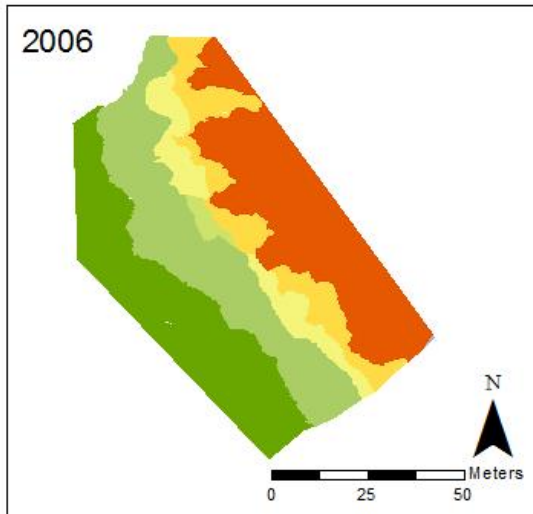
Campbell Slough
Vegetation Change 2005 to 2015

2005

2015

N

Meters
0   25   50

N

Meters
0   25   50   100

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

290
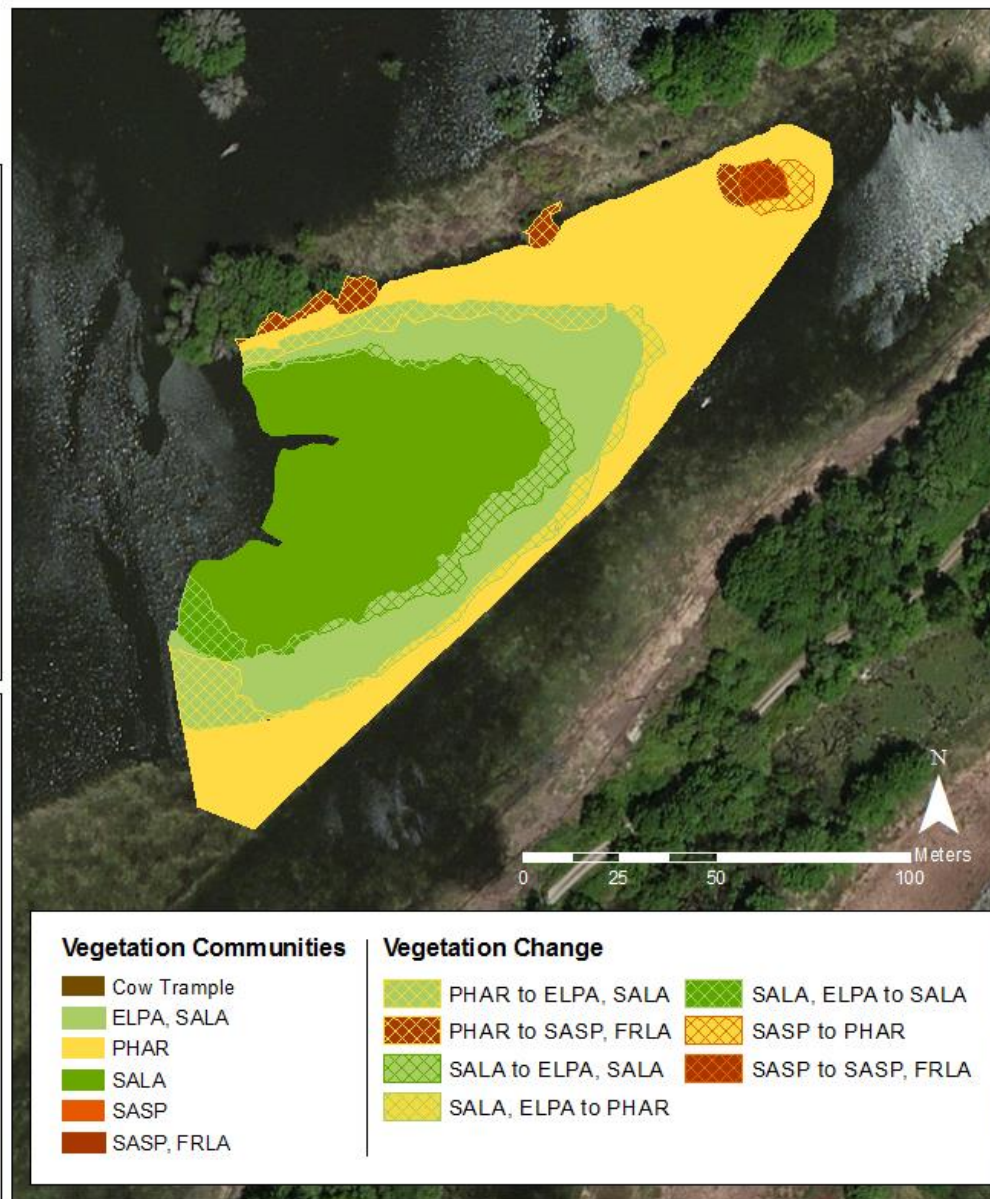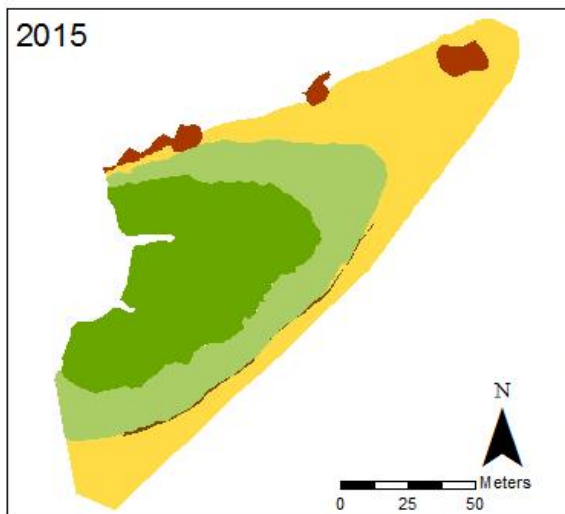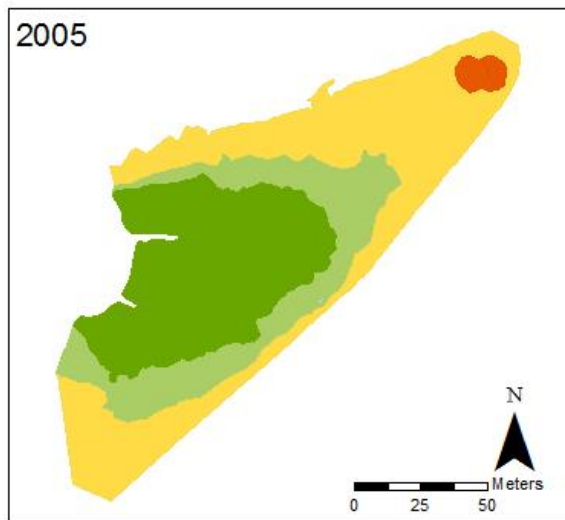
Franz Lake
Vegetation Change 2008, 2012 & 2015

2008

2012

2015

2008 to 2012

2008 to 2015

2012 to 2015
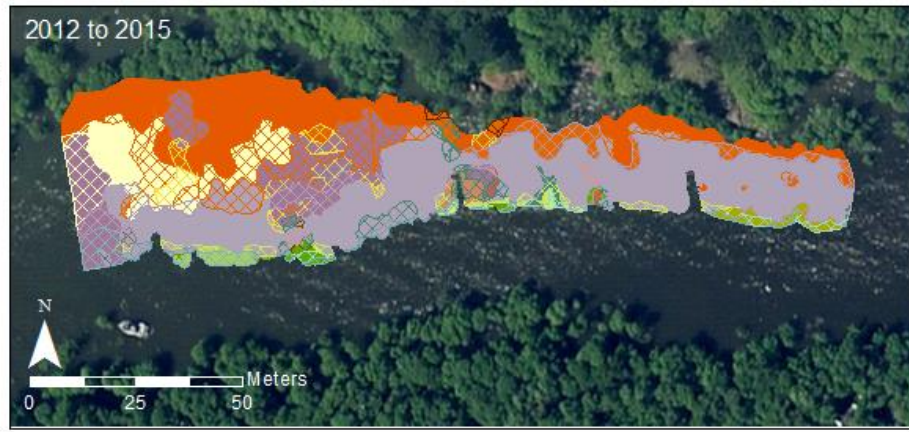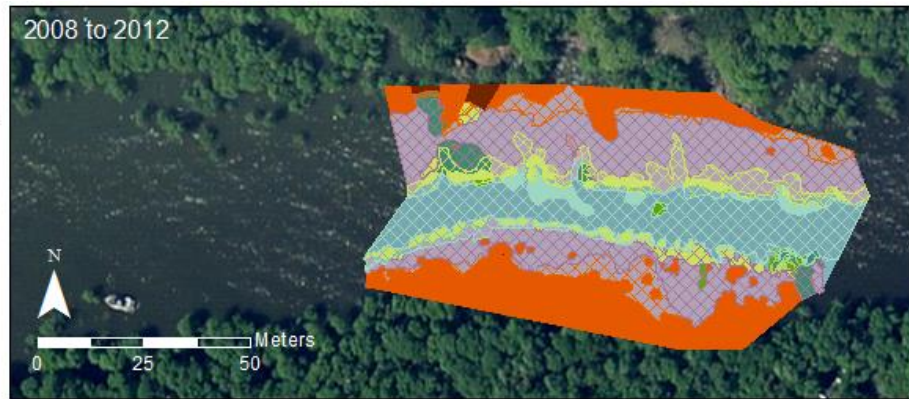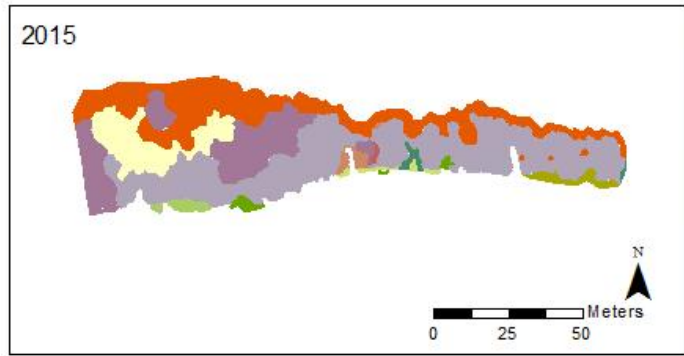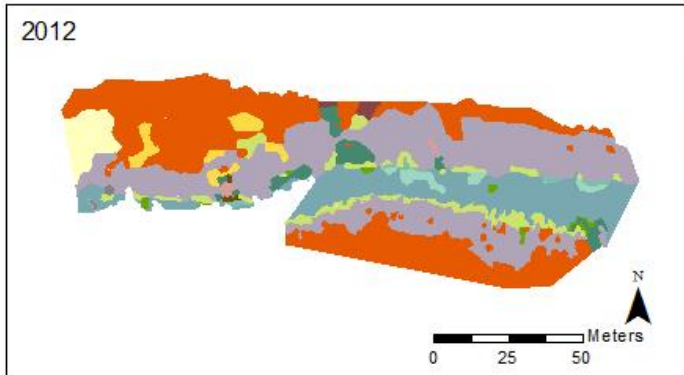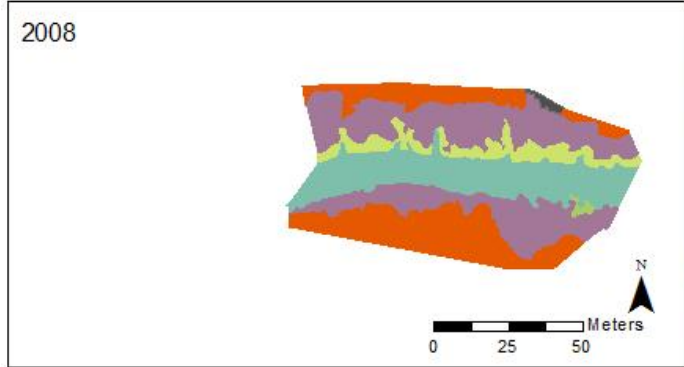
291

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

5 February 2013

22 July 2013

Whites Island – PP1

4 February 2014

31 July 2014

30 July 2015

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.
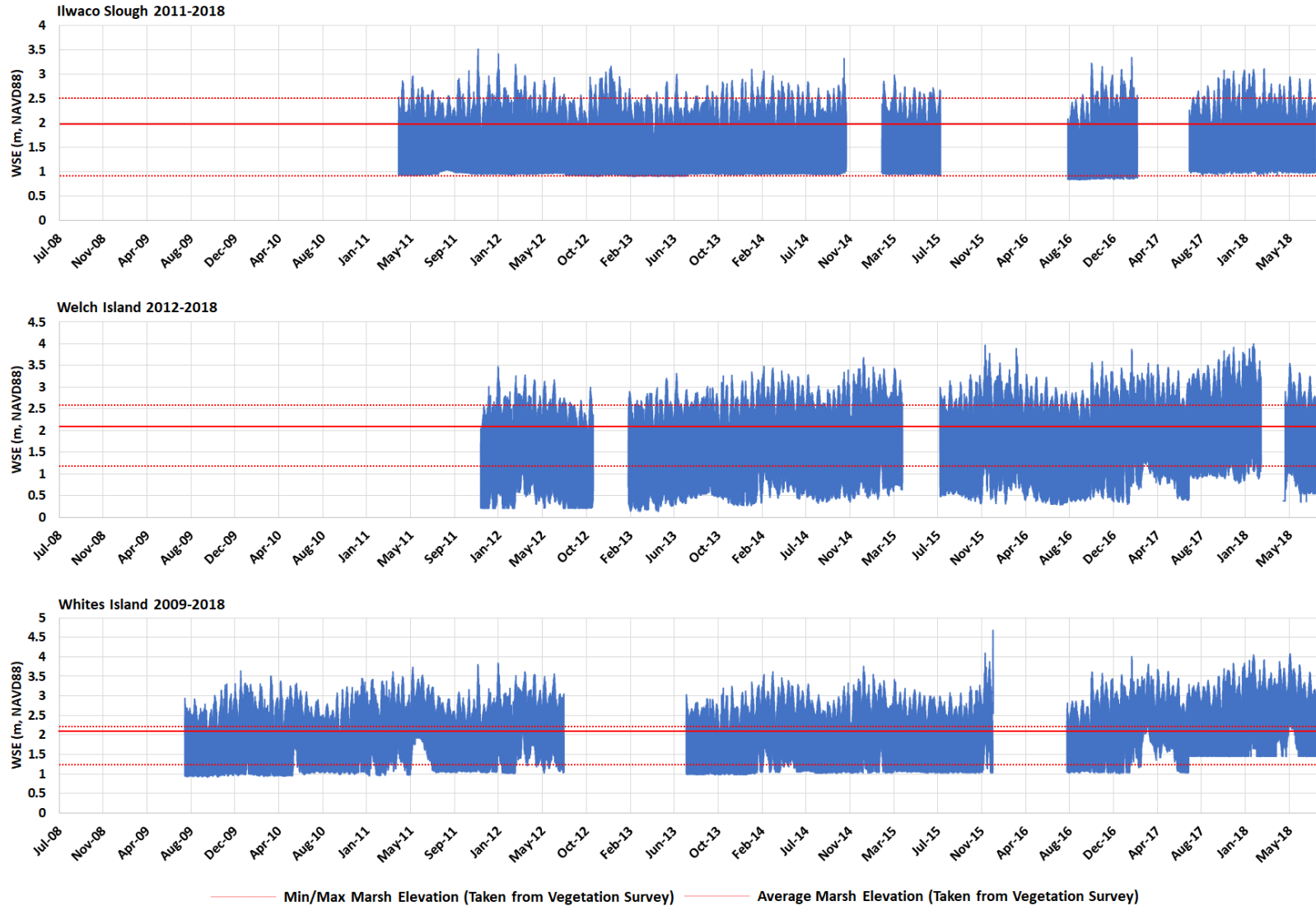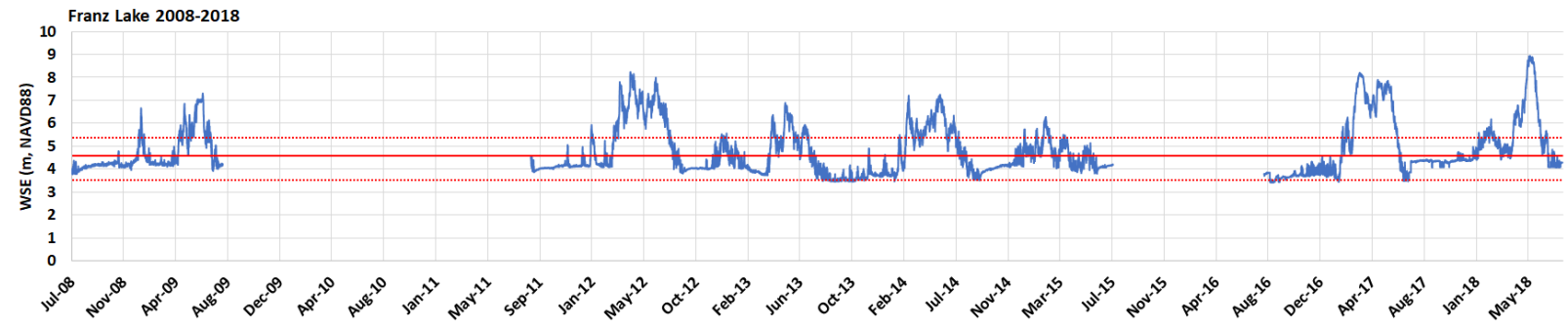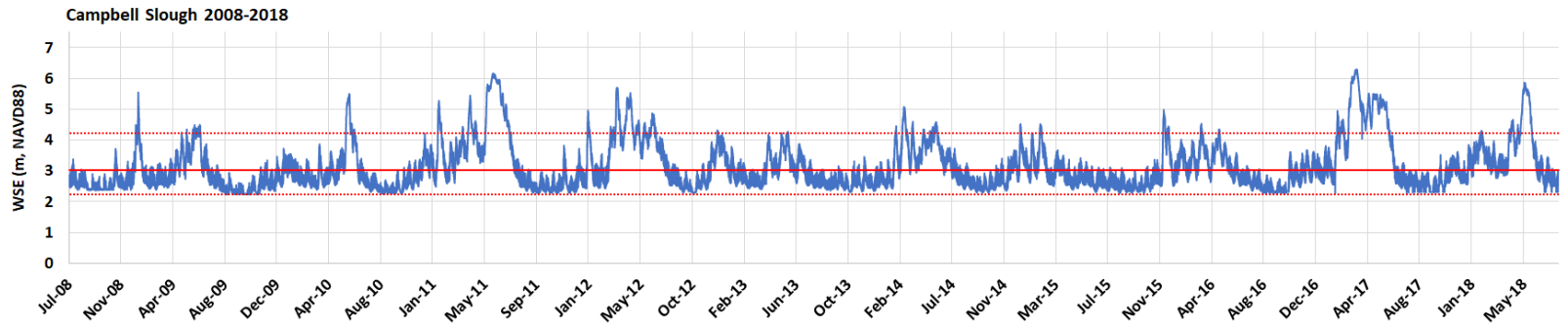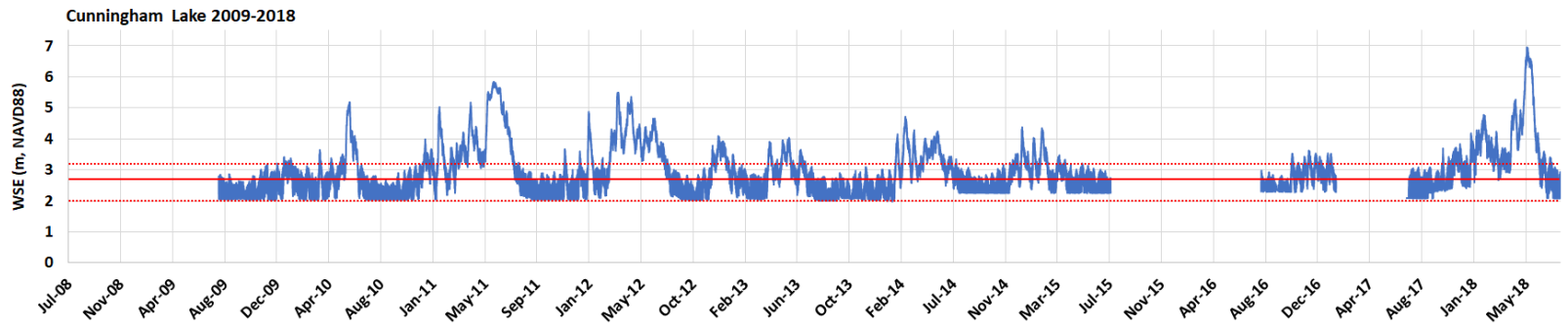
Ilwaco Slough 2011-2018

Welch Island 2012-2018

Whites Island 2009-2018

Min/Max Marsh Elevation (Taken from Vegetation Survey) — Average Marsh Elevation (Taken from Vegetation Survey)

Cunningham Lake 2009-2018

Campbell Slough 2008-2018

Franz Lake 2008-2018

——— Min/Max Marsh Elevation (Taken from Vegetation Survey)   ——— Average Marsh Elevation (Taken from Vegetation Survey)

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

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

Site	Metric	1	2	3	4	5	6	7	8	9	10	11
Cunningham Lake	Duration	Jan-Jul	Jan, Aug-Dec	Aug-Dec	Jan-Jul	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Aug-Dec	
	Days	219	193	152	209	365	365	366	365	365	160	
	MHHW	4.1	3.0	2.9	3.1	3.2	2.9	3.5	3.5	3.0	2.7	
	MLLW	3.6	2.4	2.5	2.6	2.7	2.4	3.1	3.1	2.5	2.1	
	MWL	3.8	2.7	2.7	2.8	2.9	2.6	3.3	3.3	2.7	2.4	
	Annual Range	0.5	0.6	0.4	0.5	0.4	0.5	0.4	0.5	0.5	0.6	
	Annual Max	7.0	4.2	3.6	4.3	4.7	4.0	5.5	5.8	5.2	3.2	
Campbell Slough	Duration	Jan-Jul	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Aug-Dec
	Days	220	365	362	365	365	365	366	364	365	365	164
	MHHW	3.8	3.8	3.2	3.1	3.4	3.1	3.7	3.7	3.1	3.1	2.8
	MLLW	3.4	3.4	2.8	2.7	3.0	2.7	3.3	3.3	2.8	2.7	2.5
	MWL	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	0.3	0.4	0.4	0.3
	Annual Max	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-Jul	Jan-Dec	Aug-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec	Sep-Dec		Jan-July	Aug-Dec
	Days	218	365	152	208	365	365	366	129		209	163
	MHHW	5.8	5.3	3.9	4.8	4.9	4.5	5.4	4.2		5.2	4.2
	MLLW	5.5	5.1	3.7	4.5	4.7	4.2	5.2	4.1		4.9	4.1
	MWL	5.6	5.2	3.8	4.6	4.8	4.3	5.3	4.1		5.0	4.2
	Annual Range	0.3	0.2	0.2	0.3	0.3	0.3	0.3	0.1		0.3	0.1
	Annual Max	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)	2018	2017	2016	2015	2014	2013	2012	2011	2010	2009	2008	2007	2006	2005
Ilwaco Slough	Plots (n)	40 (1)	40	40	39	37	40	40	40	40						
	Mean	1.96 (0.04)	1.94	1.91	1.94	2.00	2.03	1.96	1.96	1.92						
	SD		0.23	0.23	0.17	0.16	0.24	0.23	0.23	0.23						
	Min	1.08 (0.28)	0.95	0.94	1.44	1.61	0.93	0.94	0.95	0.89						
	Max	2.36 (0.07)	2.31	2.31	2.31	2.38	2.53	2.37	2.36	2.31						
	Range	1.28 (0.29)	1.36	1.38	0.87	0.77	1.60	1.44	1.41	1.42						
Welch Island	Plots (n)	40 (0)	40	41	40	40	40	40	40							
	Mean	2.05 (0.02)	2.00	2.06	2.06	2.06	2.06	2.06	2.06							
	SD		0.17	0.16	0.16	0.16	0.16	0.16	0.16							
	Min	1.32 (0.05)	1.22	1.34	1.34	1.34	1.34	1.34	1.34							
	Max	2.19 (0.02)	2.14	2.20	2.20	2.20	2.20	2.20	2.20							
	Range	0.87 (0.03)	0.92	0.86	0.86	0.86	0.86	0.86	0.86							
Whites Island	Plots (n)	41 (6)	44	42	42	47	43	45	42	42	35	25				
	Mean	2.06 (0.03)	2.09	2.08	2.08	2.03	2.08	2.06	2.10	2.09	2.02	2.01				
	SD		0.36	0.38	0.38	0.40	0.38	0.38	0.33	0.34	0.38	0.45				
	Min	1.2 (0.02)	1.21	1.20	1.20	1.20	1.20	1.20	1.21	1.16	1.18	1.25				
	Max	2.54 (0.04)	2.63	2.53	2.53	2.53	2.53	2.53	2.52	2.50	2.56	2.49				
	Range	1.34 (0.05)	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).

Site		Mean (SD)	2018	2017	2016	2015	2014	2013	2012	2011	2010	2009	2008	2007	2006	2005
Cunningham Lake	Plots (n)	56 (16)	67	68	69	69	36	31	59	59	61	62	62	64	62	20
	Mean	2.7 (0.07)	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		0.19	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	2.32 (0.11)	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)	3.07	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	0.68 (0.08)	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)	60	60	61	62	59	61	61	60	62	61	64	62	61	8
	Mean	2.98 (0.05)	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	0.32	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)	2.40	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)	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	1.52 (0.13)	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)	64	60	61	67	61	59	62	58		35	37			
	Mean	4.59 (0.04)	4.62	4.60	4.60	4.56	4.62	4.60	4.54	4.64		4.60	4.51			
	SD		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)	3.87	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	5.01	5.01	5.01	5.05	5.05	5.00	5.07		5.13	5.00			
	Range	1.11 (0.12)	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 2018, 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	26.6 (24.1)	1.5 (2.9)	1.1 (2.8)			
AREG	*Argentina egedii ssp. Egedii*	Yes	1.5 (4.5)	2.9 (4.6)				0.3 (1.6)
CAAP	*Carex aperta*	Yes						3.3 (9.5)
CALY	*Carex lyngbyei*	Yes	37.8 (30)	34.5 (19.8)	2.4 (5.6)			
CAPA	*Caltha palustris*	Yes		3.1 (3.9)	0.2 (1.6)			
CARE	*Carex retrorsa*	Yes						2.0 (11.4)
CASE	*Calystegia sepium*	No			3.4 (9)		0.5 (3.3)	
DECE	*Deschampsia cespitosa*	Yes	1.0 (2.5)	0.3 (1.3)				
ELAC	*Eleocharis acicularis*	Yes			0.01 (0.1)	1.2 (4.7)		
ELCA	*Elodea canadensis*	Yes		0.1 (0.3)	5.5 (16.7)			
ELPA	*Eleocharis palustris*	Yes		4.5 (11.6)	1.6 (6.4)	18 (18.4)	23.2 (28.9)	2.0 (7.2)
ELPAR	*Eleocharis parvula*	Yes	3.6 (10.4)					
EQFL	*Equisetum fluviatile*	Yes		3.8 (6.5)	6.5 (7.8)	0.01 (0.2)	0.2 (1.1)	
FRLA	*Fraxinus latifolia*	Yes					0.2 (1.9)	1.0 (6.2)
FUDI	*Fucus distichus*	Yes	1.6 (3.9)					

Sp. Code	Scientific Name	Native (Y/N)	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
GATR	*Galium trifidum L. spp. columbianum*	Yes		0.7 (2)	1.0 (2.8)			
GLMA	*Glaux maritima*	Yes	1.0 (3.4)					
IRPS	*Iris pseudacorus*	No		0.7 (2.8)	2.1 (4.8)			
ISCE	*Isolepis cernua*	Yes	2 (3.9)					
JUOX	*Juncus oxymeris*	Yes		1.8 (2.5)	0.3 (1)		0 (0.2)	
LEOR	*Leersia oryzoides*	Yes		2.2 (3.3)	2.2 (4.1)	3 (4.5)	0.1 (0.3)	1.4 (3.2)
LIDU	*Lindernia dubia*	Yes				0.6 (1.6)	1.6 (4.1)	2.3 (6.6)
LIOC	*Lilaeopsis occidentalis*	Yes	2.6 (3.8)					
LOCO	*Lotus corniculatus*	No		0.3 (1.7)	2.6 (8.4)			
LUPA	*Ludwigia palustris*	Yes				16 (18.6)	0.5 (1.0)	0.1 (0.7)
LYAM	*Lysichiton americanus*	Yes		1.7 (3.9)				
LYAM2	*Lycopus americanus*	Yes			2.9 (4.8)			
LYNU	*Lysimachia nummularia L.*	No				0.01 (0.1)	2.4 (5.0)	
LYSA	*Lythrum salicaria*	No		0.6 (2.4)	1.3 (3.6)			
MEAR	*Mentha arvensis*	Yes		1.1 (2.0)	0.01 (0.2)	0.01 (0.1)	0.01 (0.2)	0.3 (1)
MIGU	*Mimulus guttatus*	Yes		3.1 (3.1)	0.5 (1.4)			
MYLA	*Myosotis laxa*	Yes		3.2 (7.8)	0.01 (0.1)	0.1 (0.5)		

Sp. Code	Scientific Name	Native (Y/N)	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
MYSC	*Myosotis scorpioides*	No		8.2 (7.7)	4.2 (7.3)			
OESA	*Oenanthe sarmentosa*	Yes		3.7 (4.5)	5.1 (8.4)			
PHAR	*Phalaris arundinacea*	No		7.3 (17.7)	30.1 (24.6)	17.1 (26.7)	28.7 (36.5)	7.5 (14)
POAM	*Polygonum amphibium*	Yes						34.3 (37.1)
POHY	*Polygonum hydropiper, P. hydropiperoides*	Mixed		1.3 (3.2)	0.6 (2.3)	0.01 (0.2)	0.1 (0.2)	
POPE	*Polygonum persicaria*	No		1.4 (4.2)	0.1 (0.6)	2.4 (5.1)	0.1 (0.2)	
SALA	*Sagittaria latifolia*	Yes		4.0 (5.0)	11.6 (15.8)	11.4 (8.7)	6.0 (10.1)	8.1 (23.4)
SALU	*Salix lucida*	Yes		0.1 (0.8)		5.8 (17.6)		12.3 (19.2)
SCAM	*Schoenoplectus americanus*	Yes	1.9 (3.9)	0.01 (0.1)	0.01 (0.2)			
SISU	*Sium suave*	Yes		1.4 (3.5)	0.4 (1.6)	0.01 (0.1)		
SMH	*small mixed herbs*	Unknown		0.2 (0.6)	0.4 (0.9)	0.8 (1.1)	0.4 (0.5)	1.5 (3.7)
SYSU	*Symphyotrichum subspicatum*	Yes	0.4 (2.7)	1.2 (3.1)	0.1 (0.5)			
TRMA	*Triglochin maritima*	Yes	3.7 (5.6)					
TYAN	*Typha angustifolia*	No	2.2 (7.1)	0.01 (0.1)	5.0 (9.4)			
ULLA	*Ulva lactuca*	Yes	5.6 (11.4)					
ZAPA	*Zannichellia palustris*	Yes	1.2 (6.4)					

Appendix E. Fish catch summaries, 2008–2018

Appendix Table F-1. Species list of fishes captured during the EMP study, by year, 2008-2018 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
Clupeidae	American shad[i]	x	x	x	x	x	x	x	x	x	x	X
Salmonidae	brown trout[i]					x						
	Chinook salmon	x	x	x	x	x	x	x	x	x	x	x
	chum salmon	x	x	x	x	x	x		x	x	x	x
	coho salmon	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
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
	peamouth	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
	unid carp[i]	x	x	x	x	x	x	x	x	x		

Appendix Table F-1, cont.

Family	Species	1	2	3	4	5	6	7	8	9	10	11
	unid Cyprinidae[i]		x		x							x
Catostomidae	largescale sucker	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
Poeciliidae	gambusia[i]				x						x	x
Gasterosteidae	threespine stickleback	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
	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
	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
Embiotocidae	shiner perch				x	x	x		x	x	x	
Ammodytidae	sandlance				x					x		
Gobiidae	Amur goby[i]									x	x	x
	unid goby[i]		x						x	x		
Cottidae	mottled sculpin		x									
	Pacific staghorn sculpin				x	x	x	x	x	x	x	x
	Prickly sculpin										x	x
	unid sculpin	x	x	x	x	x	x	x	x	x	x	x
Pleuronectidae	starry flounder		x	x		x	x	x	x			x

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Appendix Table E-2. Species list of fishes captured at the five EMP trend sites in 2018.

Family	Species_Name	Ilwaco Slough	Welch Island	White Island	Campbell Slough	Franz Lake
Clupeidae	American shad[i]	x	x		x	
Salmonidae	Chinook salmon	x	x	x	x	x
	chum salmon	x	x	x	x	
	coho salmon					x
	pink salmon	x				
Osmeridae	larval smelt	x				
Cyprinidae	golden shiner[i]					
	peamouth			x		x
	common carp				x	
	Northern pikeminnow			x	x	
	redside shiner					x
	Unidentified cyprinid		x			
Catostomidae	largescale sucker			x	x	
Fundulidae	banded killifish[i]	x	x	x	x	x
Poeciliidae	gambusia[i]					
Gasterosteidae	threespine stickleback	x	x	x	x	x
Centrarchidae	largemouth bass[ip]				x	
	Black crappie				x	
	Bluegill				x	
	Unidentified crappie[i]				x	
	pumpkinseed				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	

Family	Species_Name	Ilwaco Slough	Welch Island	White Island	Campbell Slough	Franz Lake
	unidentified sculpin	x				
Pleuronectidae	Starry flounder		x	x		

Table E-3. Juvenile Coho (unmarked and marked), chum salmon, sockeye salmon, and trout densities (fish per 1000 m2) (SD), by year captured at trends sites 2008 - 2018. 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
Ilwaco Slough											
Coho (unmarked) (1)	ns	ns	ns	0	0.12 (0.67)	0	0	0	0	0	0
Coho (marked) (0)	ns	ns	ns	0	0	0	0	0	0	0	0
Chum (501)	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)
Sockeye (0)	ns	ns	ns	0	0	0	0	0	0	0	0
Trout sp (0)	ns	ns	ns	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
Coho (marked) (1)	ns	ns	ns	ns	0.11 (0.54	0	0	0	0	0	0
Chum (64)	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)
Sockeye (1)	ns	ns	ns	ns	0	0	0.12 (0.41)	0	0	0	0
Trout sp (1)	ns	ns	ns	ns	0	0.27 (1.05)	0	0	0	0	0
White Island											
Coho (unmarked) (6)	ns	0	1.12 (2.03)	0	0	0	0	0	0	0.12 (0.46)	0
Coho (marked) (0)	ns	0	0	0	0	0	0	0	0	0	0
Chum (40)	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)
Sockeye (7)	ns	0	0	0	0	1.32 (5.77)	0	0	0	0	0
Trout sp (0)	ns	0	0	0	0	0	0	0	0	0	0

Continued from Table E-3

Site/Species/#	Year										
	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
Campbell Slough											
Coho (unmarked) (2)	0	0	0	0	0	0	0	0.18 (0.70)	0	0	0
Coho (marked)	0	0	0	0	0	0	0.18 (0.51)	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)
Sockeye (1)	0	0	0	0	0	0	0.09 (0.38)	0	0	0	0
Trout sp. (1)	0	0	0	0	0	0	0.78 (0.33)	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)
Coho (marked) (60)	5.75 (10.72)	7.92 (17.99)	ns	0	0.51 (2.17)	0	0	0	0	ns	0
Chum (7)	2.11 (5.23)	0.32 (1.09)	ns	0	0	0	0	0	0	ns	0
Sockeye (2)	0	0	ns	0	0	0	0.55 (1.56)	0	0	ns	0
Trout sp. (5)	0	0.95 (2.30)	ns	0	1.43 (6.54)	0	0	0	0	ns	0

Table E-4. Total number of unmarked and marked Chinook salmon captured by year at each site 2008 – 2018, ns = site not sampled.

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