Lower Columbia River Ecosystem Monitoring Program

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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. The EMP aims to 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 2017, data were collected on fish and fish prey, habitat, hydrology, food web, abiotic site conditions, and mainstem river conditions at Ilwaco Slough (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 2017. 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. Mainstem flows in 2017 were higher than average in the spring and early summer (characterized by an early and extended freshet) but similar to average flows during the rest of the year. Water temperatures in 2017 were similar to past years in terms of the days exceeding 19°C; however, the number of very warm days (exceeding 21, 22, or 23°C) was greater than all other years except 2015 (even though 2017 had much higher discharge compared to 2015). Campbell Slough was the warmest of the trends sites in 2017 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. Species richness was greatest in Vegetation Zone 2 at Welch Island and Whites Island. In addition, these sites also exhibited the greatest and most variable amount of above ground vegetation biomass collected in summer 2017. Average cover of the native sedge *Carex lyngbyei* at the trends sites increased from 18% in 2016 to 22% in 2017. In contrast, the average cover of non-native reed canarygrass (*Phalaris arundinacea*) across the trends sites decreased from 26% in 2016 to 18% in 2017. *C. lyngbyei*, however, had a lower frequency of occurrence in sampling plots (37%) than *P. arundinacea* (52%). This shift in dominant native and non-native plant cover can be attributed to high spring and early summer flows in 2017 which have been shown to limit *P. arundinacea* growth in the more fluvially influenced sites, Campbell Slough, Cunningham Slough, and Franz Lake.

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 2017 varied temporally across the trends sites, peaking at Campbell Slough in April but not until mid-August at Franz Lake Slough. Phytoplankton communities at the trends sites are typically dominated by diatoms. 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 calculated at two sites that tend to have contrasting patterns in primary productivity (Whites Island and Campbell Slough) 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 has shown that juvenile Chinook salmon feed primarily on 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, with stomach contents collected from fish captured at Ilwaco Slough being dominated almost exclusively by amphipods. Mean energy ration (a measure of fitness that reflects stomach fullness and energy consumed) from Franz Lake Slough was lower than at the other trends sites and Franz Lake Slough was the only site where copepods (a relatively energy-poor prey item) were consumed in large quantities. 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, unmarked Chinook salmon were captured at trends sites between February and June (peak densities were observed in May) and were abundant at Welch Island and Whites Island. Marked Chinook salmon were captured between March and June and were more abundant than unmarked Chinook salmon at Campbell Slough. The majority of unmarked Chinook salmon were fry (57.4%), 42.3% were fingerlings, and 0.3% were yearlings. Unmarked fry dominated catches at Welch Island and Ilwaco Slough, whereas fingerlings were more dominant at Campbell Slough. Marked Chinook salmon were mostly fingerlings (95%) and only 5% yearlings. 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 2017, chum salmon were found at the trends sites in March, April, and May with the highest average density in March.

As in previous years, the overall 2017 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 a greater percentage of non-native fishes. However, in previous years the species richness and diversity of the fish community at Campbell Slough (and Franz Lake) were much greater than the lower river sites. In 2017 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 sample collection concluded in June, whereas in previous years sample collection typically extended into July and August (in 2017 the water temperatures were too warm to sample in July and August). During these months water temperature warms considerably and the proportional species composition of non-native, warm water fishes increases. In June measure of species richness and diversity (mostly non-natives) at Campbell Slough were the highest observed for the year and likely signals an increasing trend for the summer months.

The Ecosystem Monitoring Program 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 end points 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.

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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 affects 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 to 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 allows for greater predictability biotic responses to changing environmental conditions and improves our understanding of ecological function in 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 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 were 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
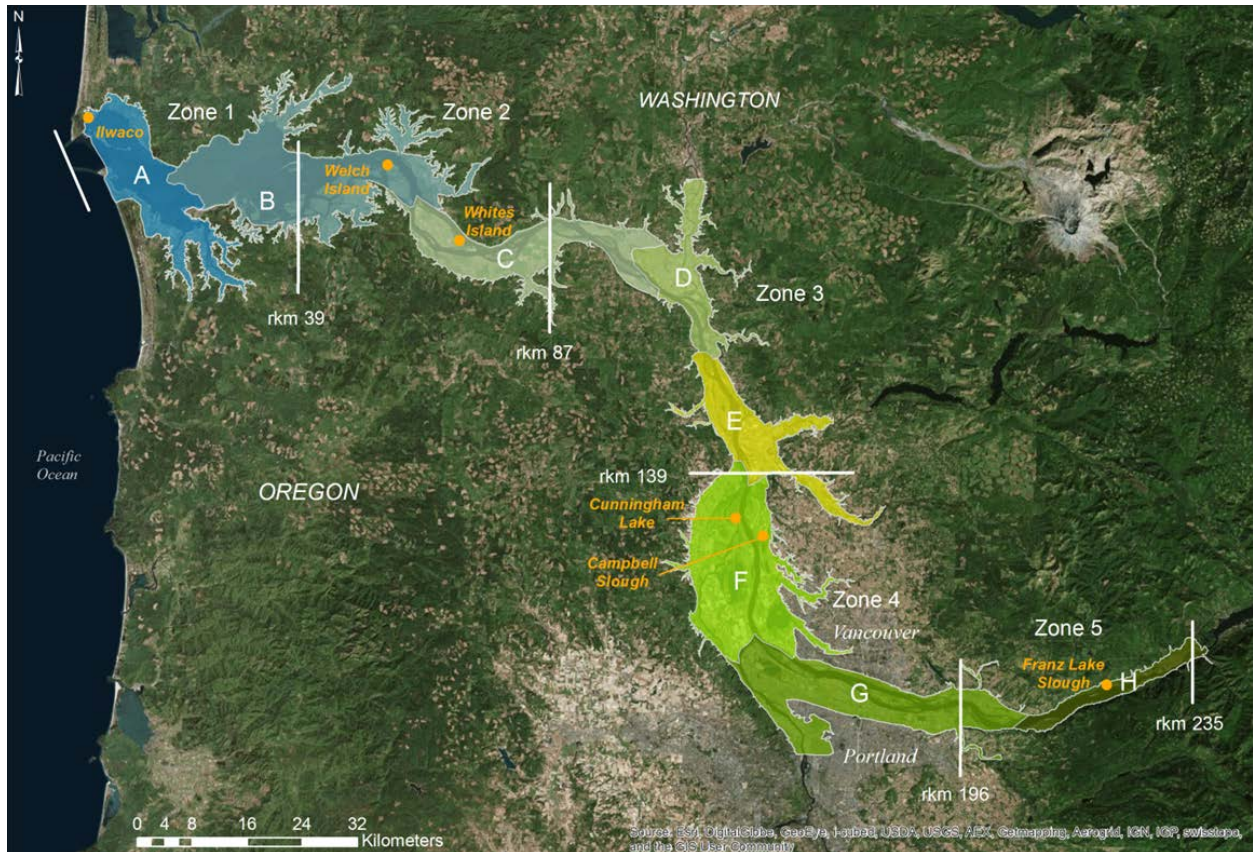
Figure 1. Lower Columbia River and estuary with hydrogeomorphic reaches (A-H) specified by color (Simenstad et al. 2011) and wetland zones (1-5) delineated by white lines (Jay et al. 2016). The 2017 EMP trends sites are shown in orange.

1.3   Characterization of Emergent Wetlands in the Lower Columbia River

1.3.1   Sampling Effort, 2005-2017

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 continued 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.

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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 (2010-2017), Secret River (2010-2016), Welch Island (2010-2017), Whites Island (2009-2017), Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2017), and Franz Lake (2008-2009, 2011-2017). In 2017, 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. 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 13 Contract (October 1, 2016 – September 30, 2017):

- 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 2017.

Reach	Type of Site	Site Name	Site Code	Vegetation & Habitat[1]	Fish &Prey[5]	Abiotic Conditions	Food Web[4]
A	Trend	**Ilwaco Slough**	BBM	2011-2017	2011-2013, 2015-2017	2011-2013, 2015-2017	2011-2013, 2015-2017
B	Trend	Secret River	SRM	2008[2], 2012-2016	2012, 2013		2012, 2013
	Tributary	Grays River, lower	-		2015		2015
	Trend	**Welch Island**	WI2	2012-2017	2012-2017	2014, 2017	2012-2017
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	2009-2017	2009-2017	2009, 2011-2017	2011-2017
	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	2005-2017	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	2005-2017	2007-2017	2008-2017	2010-2017
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**	-			2012-2017	
H	Trend	**Franz Lake (slough)**	FLM	2008-2009, 2011-2017	2008-2009, 2011-2017	2011-2017	2011-2017
	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 – 2017.

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

1.3.2 Site Descriptions

In 2017, 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 2017 are listed in Table 2. The 2017 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 (salinity < 10 Practical Salinity Units, PSU). 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 1880's 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). An electric fence is anticipated to be installed before the 2018 growing season.

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

The 2017 water year was characterized by an early and extended spring freshet, similar in length to 2012 and similar in magnitude to 2011 and 2012 (Figure 3). Peak discharge occurred in late March-early April and began to subside in early June. Early spring peak flows were higher than in previous years (2011–2017), while flows following the freshet were lower than the maximum values recorded in 2011 and 2012. Both early winter (January and February) and summer values (July, August, and September) were between the minimum and maximum values observed in previous years. Thus, river discharge during 2017 could be characterized as higher than average in the spring and early summer and close to average during autumn, winter, and late summer.
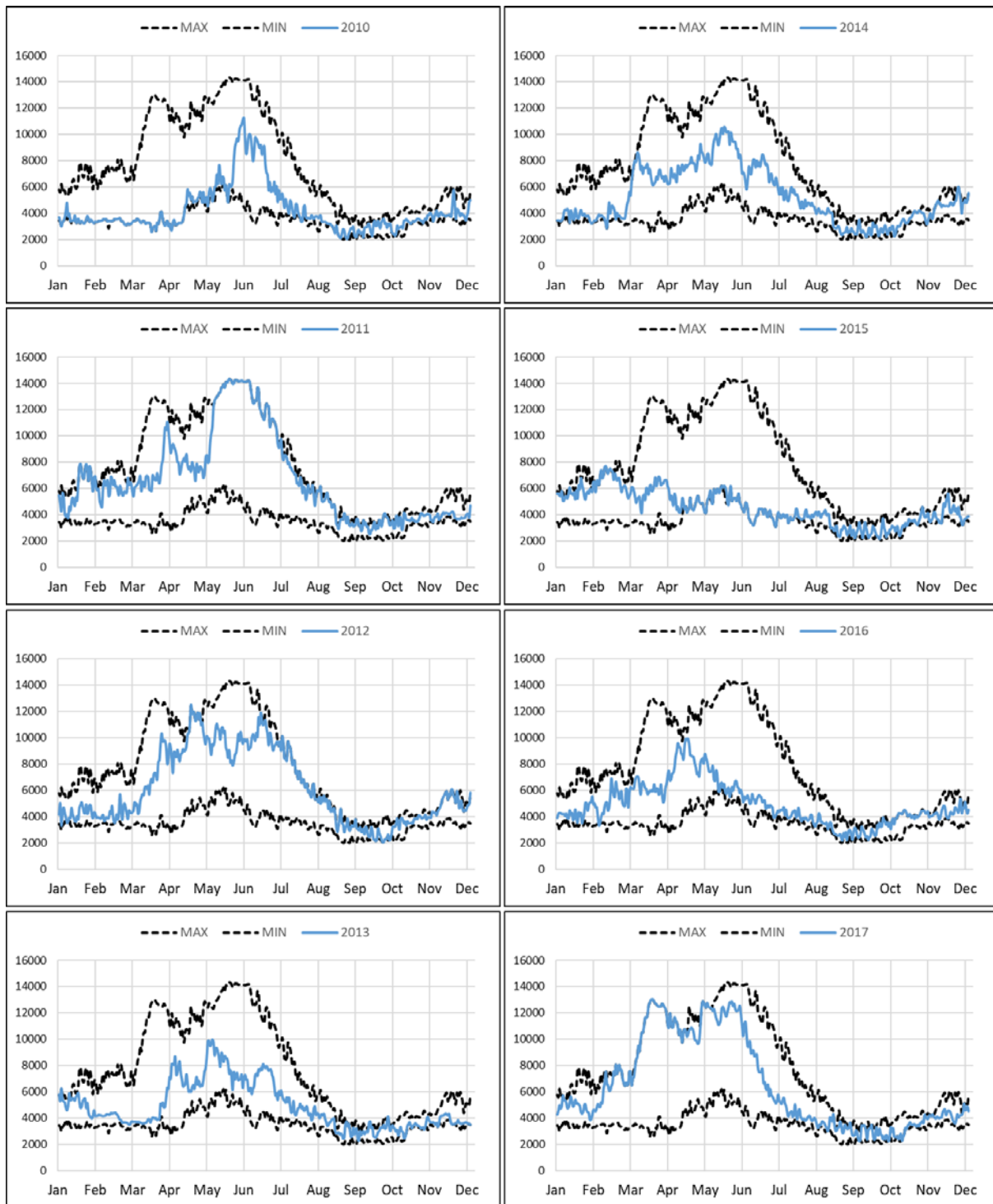
Figure 3. Daily water discharge (m³/s) at Bonneville Dam. Panels show individual years between 2010-2017 (blue lines) and the daily max and min for all years combined.

2 Methods

2.1 Mainstem Conditions

2.1.1 Overview

The Center for Coastal Margin Observation and Prediction (CMOP) 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 also can be accessed as part of the CMOP observation network from the CMOP server (http://www.stccmop.org/datamart/observation_network). 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, 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).
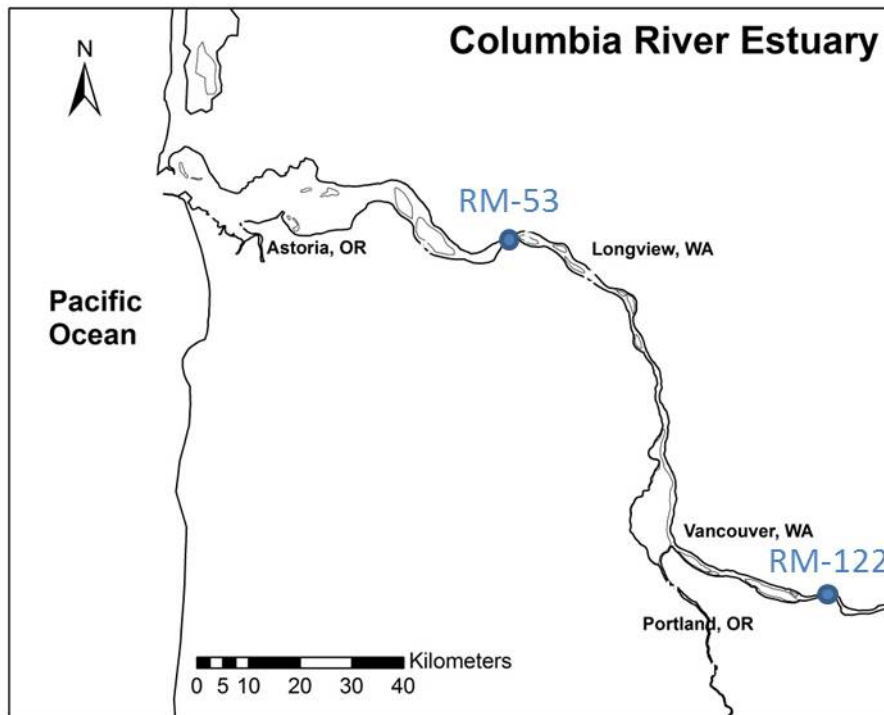
Figure 4. Station locations for the two in-situ water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program. RM-53 (river mile 53) is Beaver Army Terminal, while RM-122 (river mile 122) is located in Camas, WA.

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

The instrument platform ran continuously from October 2015–December 2016. In previous monitoring years, the instrument platform ran continuously from September 2013–December 2013, and from July 2014–December 2014. In December 2013 the instruments were removed for service and calibration and were redeployed in June 2014 (a Yellow Springs Instruments (YSI) 6600 V2 sonde was deployed as a back-up instrument in the interim).

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). The filter size on the instrument is 10 µm, which is significantly higher than traditional filtered samples (0.45 µm). Therefore, data must be compared with caution, since some phosphate removed by traditional sampling is measured by the Cycle-PO4.

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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 during 2016 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. Chlorophyll *a* measurements are used to correct the in situ fluorometer measurements. The discreet 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 2017, 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.

Table 5. Locations of water quality monitors (YSI sondes) at trends sites in 2017.

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"	3/10/2017	9/20/2017
Welch Island	461518123285700	Unnamed Slough, Welch Island, Columbia River, OR	B	46° 15' 18.4"	-123° 28' 56.8"	3/14/2017	9/21/2017
Whites Island	460939123201600	Birnie Slough, White's Island, Columbia River, WA	C	46° 09' 39"	-123° 20' 16"	3/10/2017	9/20/2017
Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47' 05"	-122° 45' 15"	3/16/2017	9/18/2017
Franz Lake	453604122060000	Franz Lake Slough Entrance, Columbia River, WA	H	45° 36' 04"	-122° 06' 00"	5/5/17[†]	9/18/2017

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

[†]Site was underwater until May 2017.

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, a YSI EXO2 with fluorometer was installed at Campbell Slough and Franz Lake Slough capable of detecting and monitoring chlorophyll and phycocyanin, pigments that approximates the biomass of total phytoplankton biomass 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 2017, the monitors were deployed from late March through mid-September, with the exception of Franz Lake Slough, which under too much water to access the pipe in which sensors are deployed. This sonde was deployed in May (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
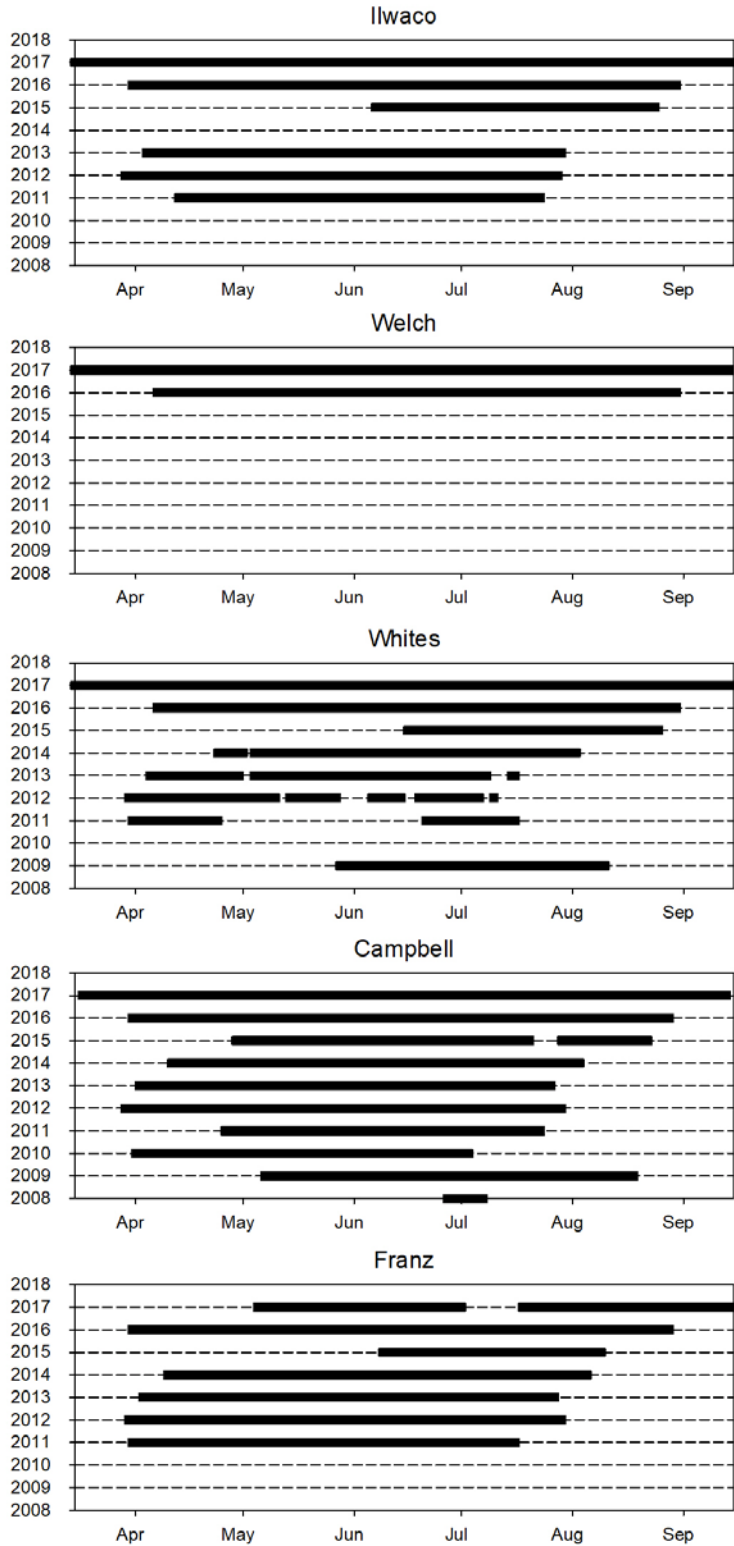
Figure 6. Time periods corresponding to sensor deployments at five trends sites (2011–2017).

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 prior to 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 2017, PNNL collected field data on vegetation and habitat conditions at the six trends sites (Figure 1). Monitoring dates are provided in Table 8**Error! Reference source not found.** and detailed maps of the 2017 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	7/27/17
Welch Island 2	WI2	53	Trend	7/26/17
Whites Island	WHC	72	Trend	7/25/17
Cunningham Lake	CLM	145	Trend	7/30/17
Campbell Slough	CS1	149	Trend	7/28/17
Franz Lake	FLM	221	Trend	7/29/17

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 2017, 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 potential evolution of a site. Evaluating vegetative composition and species cover provides an indication of 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 all of the trends sites, excluding Cunningham Lake. 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
- 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

Hydrology data for 2017 have not yet been processed or reported.

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 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). 2017 salinity data have not yet been processed or reported.

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 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 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. 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).

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 CRD 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 for the entire period of record (approximately one year) and for the growing season, April 22-October 12. The growing season is based on the number of frost-free days for the region, as determined by the Natural Resource Conservation Service (NRCS) in the wetland determination (WETS) table for Clark County, WA (NRCS 2002). The Clark County growing season is used for all the sites in the estuary so that the inundation calculations are standardized to one period. The inundation frequency during the growing season was only calculated during daylight hours (between 0900 and 1700). This limitation was employed primarily for tidal areas where the timing of the daily high tide can be a factor in the amount of time available for plants to photosynthesize.

The percent of time each channel was inundated was calculated for the thalweg and top-of-bank elevations and for two time periods. In order to estimate habitat opportunity for juvenile salmonids, water depth of 50 cm was added to the thalweg elevation of each cross-section as an indicator of the amount of water adequate for fish use of the channel (Nichole Sather, personal communication). Likewise, a 10 cm water depth was added to the top of bank elevation at each cross-section to represent a minimum amount of water needed for fish to access the vegetation at the edge of the bank (Bottom et al. 2005, Kurt Fresh personal communication). The periods assessed were 1) the deployment period (year-round, generally July to the following July) and 2) the period from March 1 through July 31, which represented the peak juvenile Chinook migration period in the lower river, as determined from data collected as part of this Ecosystem Monitoring Program and other studies (Bottom et al. 2005, Sather et al. 2009).

In order to better assess hydrologic patterns and to make sites comparable over time and space, we calculated a single measurement that incorporates magnitude and duration of surface water flooding. Following work conducted in the US and in Europe (Simon et al. 1997, Gowing et al. 2002, Araya et al. 2010) we calculated the sum exceedance value (SEV) using the following equation:

$$ \text{SEV} = \sum_{i=1}^{n} (h_{\text{elev}}) $$

Where *n* is the number of hours present in the time period evaluated, and h_{elev} is the hourly water surface elevation above the marsh elevation. This differs from previous lower river studies (Borde et al. 2011, Sagar et al. 2013) in which the daily mean water surface elevation was used in the calculation rather than the hourly water level elevation used here. The latter was chosen to ensure we captured daily inundation fluctuations that occur in the more tidally dominated sites. The time periods evaluated were the annual deployment period and the growing season. Both periods were standardized to include the same days in each year, as follows:

Growing season: April 22 to June 21 and August 20 to October 12 (115 days)
Annual deployment period: August 20 to June 21 (of the next year; 306 days)

This standardization was necessary because in the past, the deployment and retrieval dates for sensors varied between June 21 and August 20 and the same time periods must be used to compare calculations from past and present data.

For the trends analysis, the SEV was calculated for the average elevation of the three to five species that comprise most of the vegetation cover at the study sites using the water surface elevations measured each year during the growing season. For the years that water surface elevation data were not collected at the sites, we used data from the NOAA tide station with the greatest similarity in hydrologic magnitude and pattern. For Cunningham Lake this was the St. Helens station, for Campbell Slough it was the Vancouver station, and for Franz Lake, it was the USGS Bonneville station. A linear regression model was developed between existing site data and the station data from the same years ($r^2 \geq 0.99$). The model was then applied to the station data to predict the site water surface elevation for missing years. Average water years were used to predict average or low water years and high water years to predict results in higher water years. The monitoring protocol can be found on monitoringmethods.org ([Method ID 954](#)).

Inundation (SEV) was compared to cover data for all years at the Cunningham Lake and Campbell Slough sites using a regression analysis. In some years the cover data was affected by extraneous factors, so the data for those years was plotted, but not included in the analysis. The factors include cattle grazing at Campbell Slough (2007), difficulty distinguishing between live and dead *Phalaris arundinacea* at Campbell Slough (2011), and a different sampling design at Cunningham Lake (2014). Grazing may have also been a factor at Campbell Slough in 2012, however, the data was included in the analysis because the effect was not as evident.

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.

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. 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. 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.

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 contain 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.

2.4   Food Web

2.4.1 Primary Productivity

2.4.1.1   *Emergent Wetland Vegetation*

2.4.1.1.1   Aboveground Vegetation Biomass

Field Methods
From summer 2011 to summer 2017, 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. 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, collection did not continue in subsequent years.

Laboratory Methods

In the laboratory, the biomass samples were stored in a cold room prior to processing. The samples were individually rinsed of all non-organic material and obvious root material was removed. Pre-weighed pieces of tinfoil were used to secure the individual biomass samples, a wet weight was measured, and the samples were placed in an oven set at 90°C for at least four days. When the samples were deemed completely dry, a dry weight was measured for each sample.

Analysis

Average dry weight was calculated for various strata and site values. For 2015 to 2017 data (Table 9), 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.

The average dry weight for each vegetation community strata, for each year, and each site were compared to the average vegetation cover for the same strata, sites, and years. A regression analysis was conducted to determine whether average cover could be used to predict biomass within emergent wetlands.

2.4.1.1.2   Macrodetritus

Field Methods

Macrodetritus samples were collected from the same 0.1 m^2 quadrat after the live and dead aboveground biomass was removed.

Laboratory Methods and Analysis

The detritus samples were rinsed with freshwater over a 500 μm sieve, and all live material, such as roots, were removed from the samples. The type of vegetation in the samples were noted when possible. Samples were air dried in a 30°C oven for one to two weeks and weighed. Samples are then sent to Analytical Laboratory Services, Kelso, Washington where the samples were then analyzed for C and N content using standard EPA method 440.0. After sub-samples were removed for this analysis the samples were analyzed for percent moisture (ASTM method D2974-07a) and a final dry weight were calculated. For each site, the average dry weight, percent C, percent N, and C:N were calculated.

Table 9. Number of samples collected in each year and season (S=summer, F=fall, W=winter) for all sample sites and vegetation strata.

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	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	82
WI2	LM			4											4
WI2	SAV			4		4	6								14
WHC	CALY		1	3		3	3	3	3	3	3	3	3	3	31
WHC	HM												1		1
WHC	PHAR												1	1	2
WHC	PHAR/HM	6	4	5		5	6	6	9	9	9	9	9	8	85
WHC	SALA	2	3	3		3	6	6	6	6	6	6	6	6	59
WHC	SAV	8	8	6		6	6								34
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	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
	Total	49	55	73	14	73	114	61	55	55	104	78	52	30	813

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

2.4.1.2 *Phytoplankton*

Abundance

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

Phytoplankton abundance was also determined by enumeration of individual cells using inverted light microscopy. The dates corresponding to sample collection for determination of nutrient concentrations, zooplankton abundance, and phytoplankton abundance are shown in Table 10. 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 10. List of samples analyzed (Xs) and data of collection from five trends sites in the Lower Columbia River in 2017.

Site	Zone	Reach	Date	Nutrients	Zooplankton	Phytoplankton
ILWACO SLOUGH	1	A	4/12/17	X	X	X
			4/12/17	X	X	X
			5/10/17	X	X	X
			6/22/17	X	X	X
			7/21/17	X	X	X
			8/24/17	X	X	X
			9/20/17	X	X	X
WELCH ISLAND	2	B	3/14/17	X	X	X
			4/12/17	X	X	X
			5/10/17	X	X	X
			6/21/17	X	X	X
			7/20/17	X	X	X
			8/24/17	X	X	X
			9/21/17	X	X	X
WHITES ISLAND	3	C	4/11/17	X	X	X
			5/9/17	X	X	X
			6/21/17	X	X	X
			7/20/17	X	X	X
			8/24/17	X	X	X
			9/20/17	X	X	X
CAMPBELL SLOUGH	4	F	3/16/17	X	X	X
			4/7/17	X	X	X
			6/6/16	X	X	X
			6/20/17	X	X	X
			6/26/17	X	X	X
			7/18/17	X	X	X
			8/25/17	X	X	X
			9/18/17	X	X	X
FRANZ LAKE SLOUGH	5	H	3/16/17	X	X	X
			5/3/16	X	X	X
			6/20/17	X	X	X
			8/25/17	X	X	X
			9/18/17	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 distance between sample points as a measure of similarity of biological communities; short distances represent relatively high similarity between samples, while longer distances represent 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 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 10 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 at 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 10.

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 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 11). 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 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 11. Potential food sources for marked and unmarked juvenile Chinook salmon and invertebrate consumers.

Fish		Invertebrates
Marked Chinook salmon	**Unmarked Chinook salmon**	**Chironomidae, amphipods, zooplankton**
Chironomidae	Chironomidae	Particulate organic matter (POM)
Corophium spp.	*Corophium spp.*	Periphyton
Gammarus spp.*	*Gammarus spp.*	Live vegetation
Zooplankton	Zooplankton	Dead vegetation
Hatchery food		

*Not analyzed in 2017

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 individual; muscle tissue samples from different bodies were not composited.

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

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

2.5 Macroinvertebrates

2.5.1 Salmon Prey Availability Sampling

2.5.1.1 *Open Water and Emergent Vegetation*

To assess the availability of salmon prey at the trends sites, we conducted neuston tows in both open water (OW; in the center of the channel) and emergent vegetation (EV; along 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 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 12). Samples were preserved in 10% formalin until delivered to the laboratory for processing. The monitoring protocol can be found on monitoringmethods.org (Method ID 1622).

Table 12. The number of invertebrate tow samples (OW and EV) collected at each site per sampling event, 2008-2013 and 2015-2017.

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)

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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	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
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 13.

2.5.2.2 *Laboratory Methods*

Organisms in the diets were identified in most cases to 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 13. The number of Chinook salmon diet samples collected at each site per sampling event, 2008-2013, 2015-2017.

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	Pierce Island	Hardy Slough	Total Tow Samples
2008	April													6			13	15	9		43
	May													19				7			26
	June																			13	13
2009	May				9			10		6				10				8			43
	June				10									9							19
2010	April					10	19	16	6					12							63
	May					17	15	14	14					24							84
	June					9	8	18	11					18							64
	July					10		19	11					15							55
	August					8		13													21
2011	May							10			10	13	10	22							65
	June							25													25
	July							2			2										4
2012	February		15	16																	31
	March			14				13							13						40
	April		15	14				10							7	15					61
	May			30				11						18	15	18					92
	June		14	15				15						15	15	36					110
2013	March			9																	9
	May		12	30				15						34							91
	June		1	23				13						9							46
	July		2	25										1							28
2015	April	6																			6
	May			15				15						15				4			49
	June			7				13													20
2016	April			13				13					7					12			45
	May			15				19					13								47
	July			3				8													11

		Ilwaco Slough	Secret River	Welch Island	Ryan Island	Bradwood Slough	Jackson Island	Whites Island	Wallace Island	Lord/Walker Island	Burke Island	Goat Island	Deer Island	Campbell Slough	Lemon Island	Washougal	Sand Island	Franz Lake	Pierce Island	Hardy Slough	Total Tow Samples
	February	2						2													4
	March	1						1													2
2017	April			15				8						1							24
	May			30				30						34							94
	June			32				5						23							60
Total Tow Samples		9	59	306	19	54	42	318	42	6	12	13	30	285	50	69	13	46	9	13	1395

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 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:

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

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 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 weight (R^2=0.972) for use in 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 2017, NOAA Fisheries monitored habitat use by juvenile Chinook salmon and other fishes at four trends sites, Campbell Slough in Reach F (sampled from 2007-2016), Whites Island site in Reach C (sampled from 2009-2016), Welch Island in Reach B (sampled from 2012-2016), and Ilwaco Slough in Reach A (sampled from 2011-2016), 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), was not sampled in 2017 due to high river elevations throughout the season and a forest fire in late summer. Coordinates of the sampling sites are shown in Table 14.

Fish were collected from February through June, and September 2017 using a Puget Sound beach seine (PSBS; 37 x 2.4 m, 10 mm mesh size). PSBS 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 2017 are shown in Table 14. 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 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 14. Location of EMP sampling sites in 2017 and number of beach seine sets per month. ns = not sampled.

Site	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Total
Ilwaco Slough (Reach A) 46.300530° N, 124.045893° W	ns	2	3	3	2	3	ns	ns	3	16
Welch Island (Reach B) 46.255011° N, 123.480398° W	ns	ns	3	1	1	ns	ns	ns	ns	5
Whites Island (Reach C) 46.159350° N, 123.340133° W	ns	2	3	3	2	3	ns	ns	3	16
Campbell Slough (Reach F) 45.783867° N, 22.754850° W	ns	3	3	3	2	3	ns	ns	ns	14
Franz Lake (Reach H)* 45.600583° N, 122.103067W	ns	ns	ns	ns	ns	ns	ns	ns	ns	0
Total	0	7	12	10	7	9	0	0	0	51

*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):

$$H' = -\sum_{i=1}(p_i ln p_i)$$

Where

ni = the number of individuals in species i; the abundance of species i.

N = the total number of all individuals

Pi = 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.

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]).

2.6.2.2 *Lipid Determination and Condition Factor*

As part of our study we determined lipid 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 collection into a set containing 3-5 fish each. Using the composited salmon whole body samples, the total amount of extractable lipid (percent lipid) was determined by Iatroscan and lipid classes were determined by thin layer chromatography with flame ionization detection (TLC/FID), as described in Ylitalo et al. (2005).

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)*

In 2017, otoliths were extracted from juvenile Chinook salmon collected at EMP status and trends sampling sites from February–June in 2017. Franz Lake was not sampled in 2017. Otoliths from fish ranging in fork length from 38-163 mm (mean = 65 mm, SD = 19 mm) were processed for microstructural analysis of recent growth. 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 (11.7mm) of the regression between fish length and otolith radius ($R^2 = 0.74$, n = 615) and where *Lc* represents fork length (mm) at capture. Next, average daily growth rate (mm/day) was calculated for an individuals' last 7 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).

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. 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 presence and to estimate residency of PIT tagged fish in Campbell Slough.

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 10 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 10 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 backup. 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 7. Beaver Army Terminal 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 2017, both spring and late-autumn flows were high relative to other years, which was not observed in the data from Bonneville (Figure 3). Like these data from Bonneville, spring flows were similar to 2011 and 2012. Thus, river discharge at Beaver Army Terminal was high relative to other years during winter, spring, and autumn, but similar to the averages for the late spring and summer.

River Discharge (m³s⁻¹ x 1000)
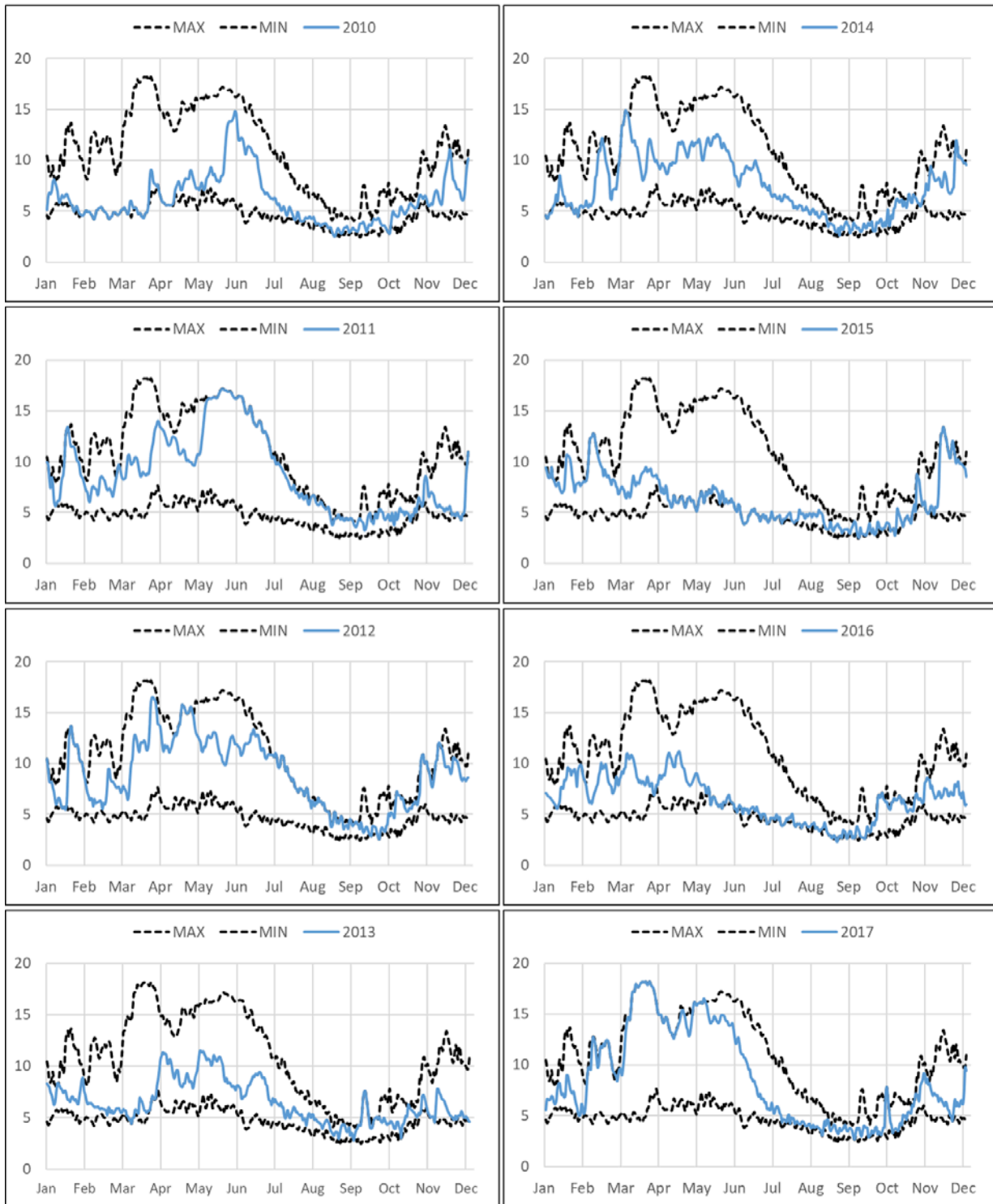
Figure 7. Daily water discharge (m³/s) at Beaver Army Terminal (RM-53) from 2010-2017. Panels show individual years (blue lines) with the daily maximum and minimum indicated (upper and lower dashed lines) in each panel.

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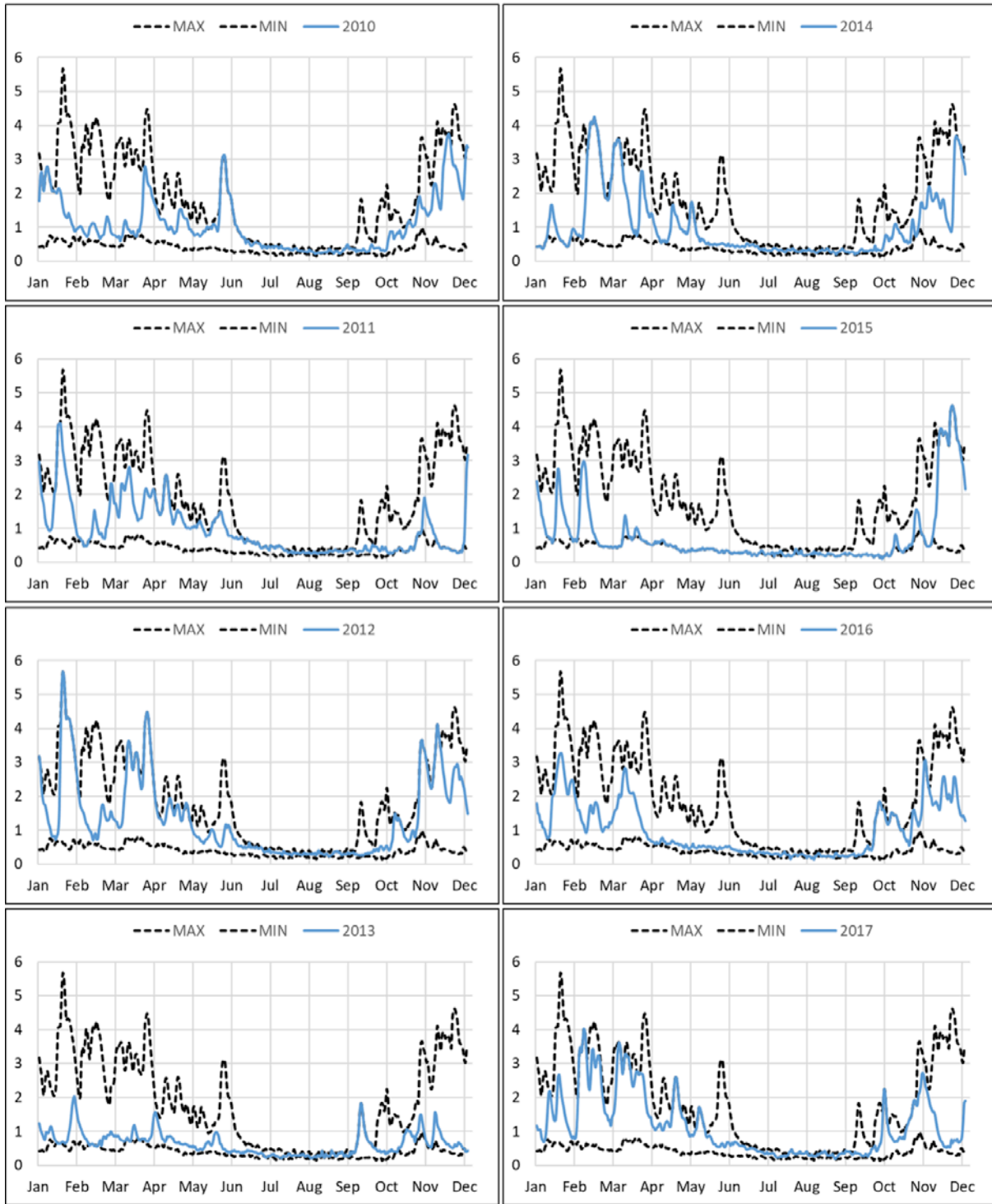
Figure 8. Daily river discharge of the Willamette River measured near the Morrison Bridge for years 2010–2017. Data from USGS 14211720. Also shown are the daily maximum and minimum values for the years 2010–2017.

Time series of river discharge associated with tributaries excluding the Willamette River are shown in Figure 9. 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 higher-than-average flow associated with the early and long-lived freshet and higher-than-average flows during the late-autumn and early winter due to pluvial inputs.

River Discharge (m³s⁻¹ x 1000)
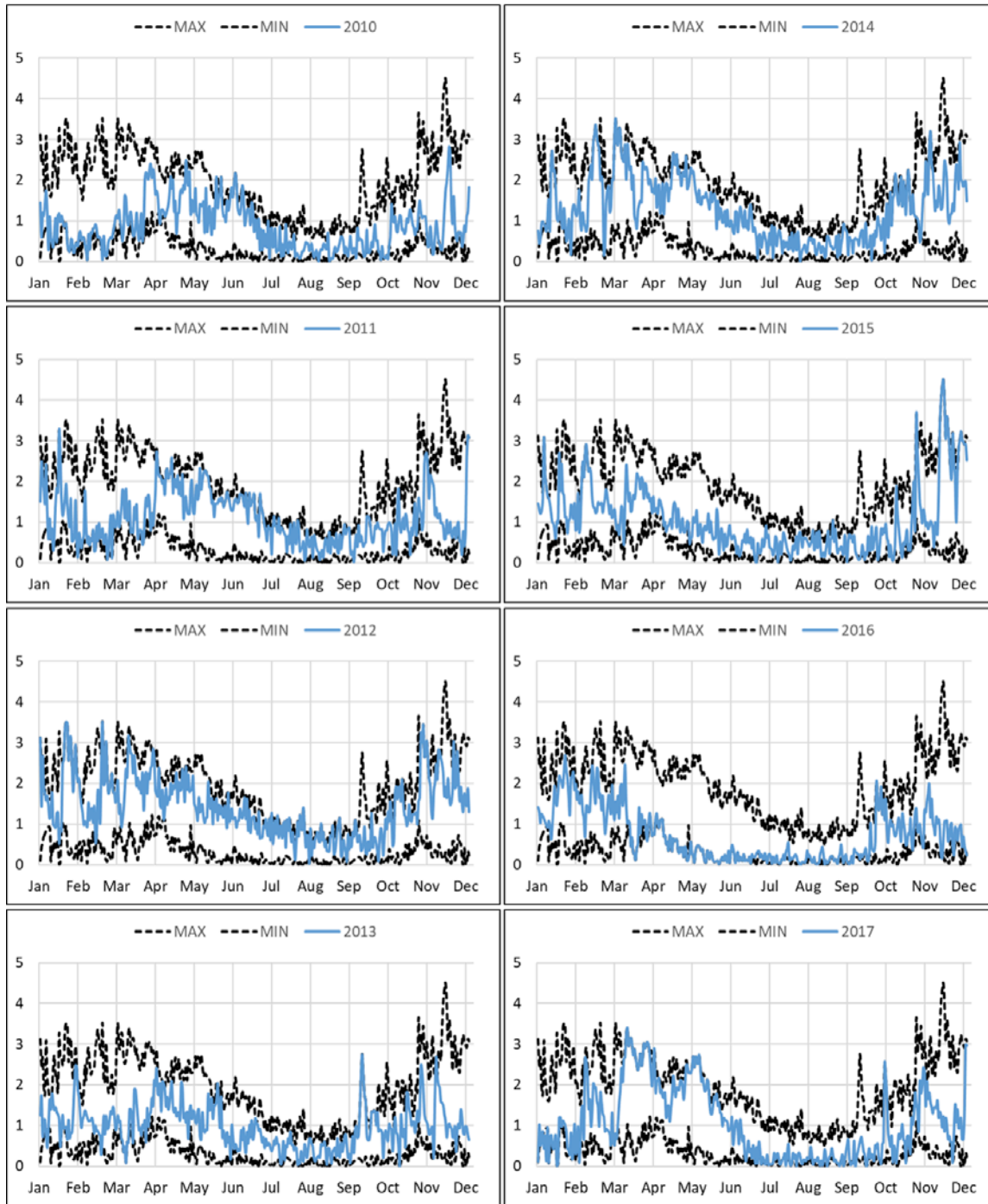
Figure 9. Estimated daily water discharge (m³/s) associated with tributaries (excluding the Willamette River). Discharge fluxes were computed from the difference between observations at Beaver Army Terminal and observations at Bonneville Dam and the Willamette River near the Morrison Bridge (i.e., Tribs = BAT – Bonneville – Willamette). Panels show individual years from 2010–2017 (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 2010–2017 are shown in Figure 14 and 15. Columbia River accounts for the largest proportion of flow throughout the year; however, during the winter months, flows from the Willamette River increase in relative importance, as do flows from other tributaries (Figure 15). High river discharge in early spring 2017 was composed of a relatively large 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). 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 15 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.

Plots showing the average discharge over the 2010-2017 period at Beaver Army Terminal (RM 53; Figure 10), Bonneville Dam (Figure 11), the Willamette River (at Morrison Bridge; Figure 12), and the sum of tributaries other than the Willamette River (Figure 13) demonstrate the strong seasonality of snowmelt-driven flows at Bonneville Dam, which drive similar patterns at Beaver Army Terminal. In contrast, the episodic pluvial flows in the Willamette River and other tributaries contribute to flows throughout year; however, the largest contributions come during the winter when Columbia River flow is less influenced by snowmelt and in the summer when snowmelt-driven flows subside.
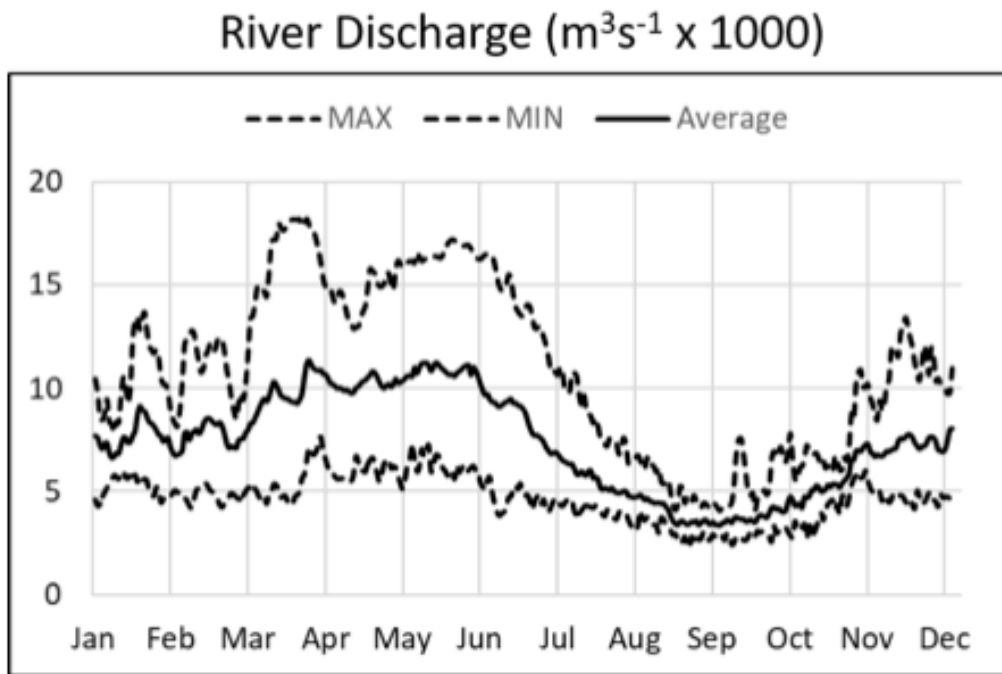
Figure 10. Maximum, minimum, and average daily discharge over the period 2010–2017 at river mile 53 USGS dolphin near the Beaver Army terminal.
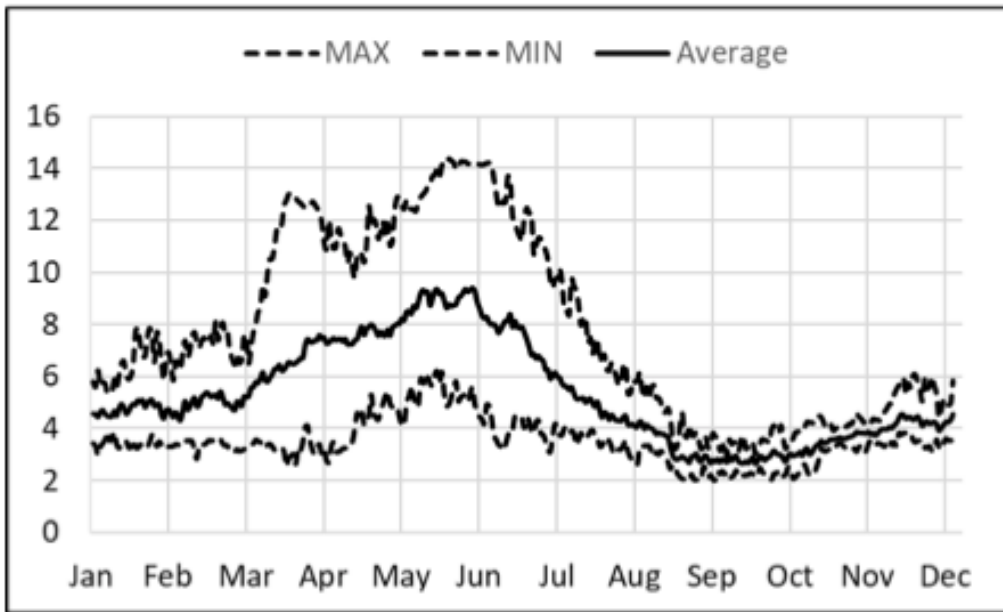
Figure 11. Maximum, minimum, and average daily discharge over the period 2010 – 2017 at Bonneville Dam.
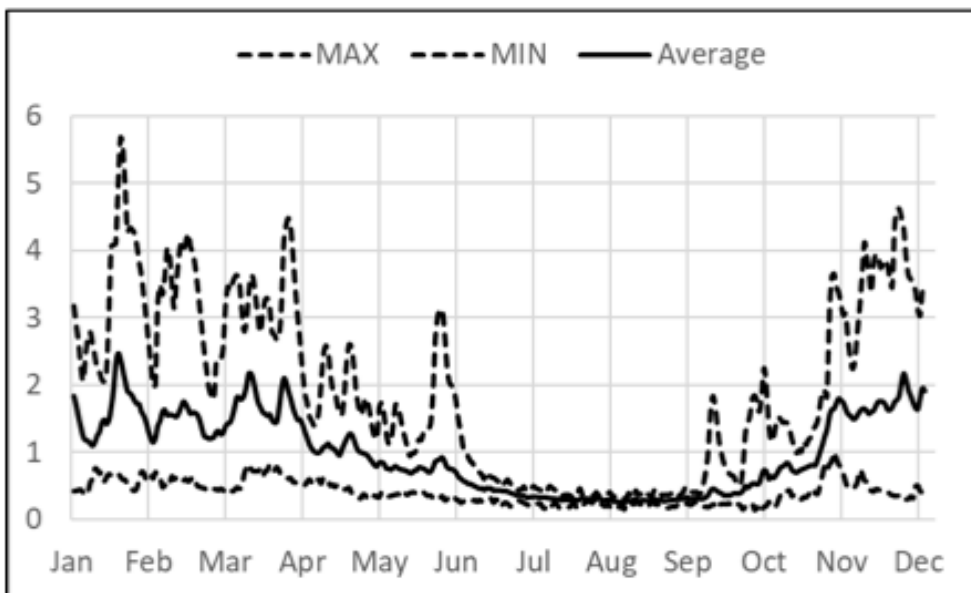
Figure 12. Maximum, minimum, and average daily discharge over the period 2010–2017 of the Willamette river measured near the Morrison Bridge.
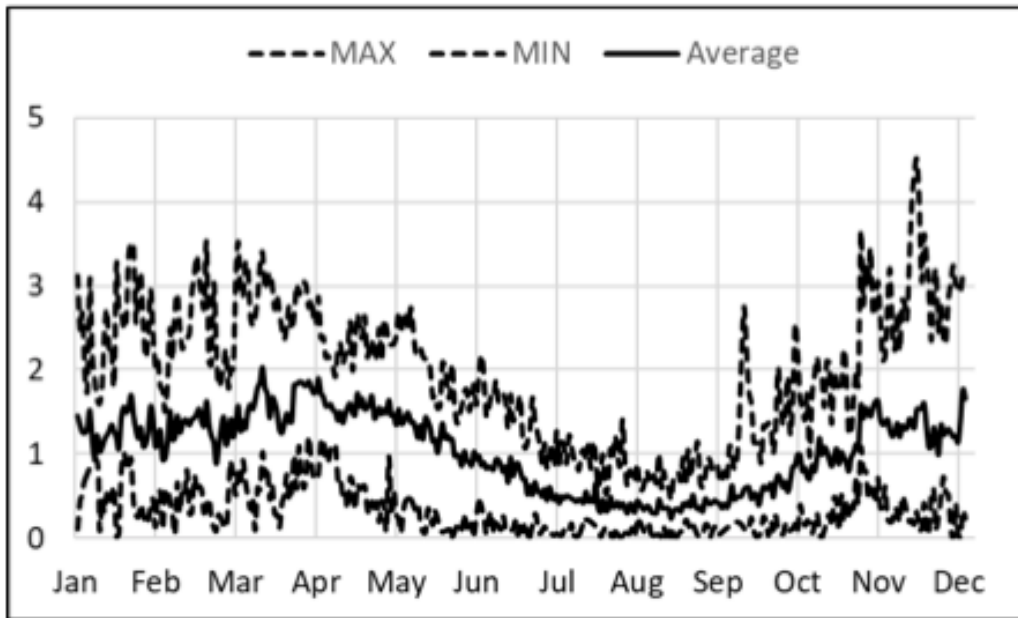
Figure 13. Maximum, minimum, and average daily discharge over the period 2010 – 2017 of the lower Columbia river tributaries except the Willamette river. Data calculated as follows: Tribs = (BAT)-(Bonneville)-(Willamette).
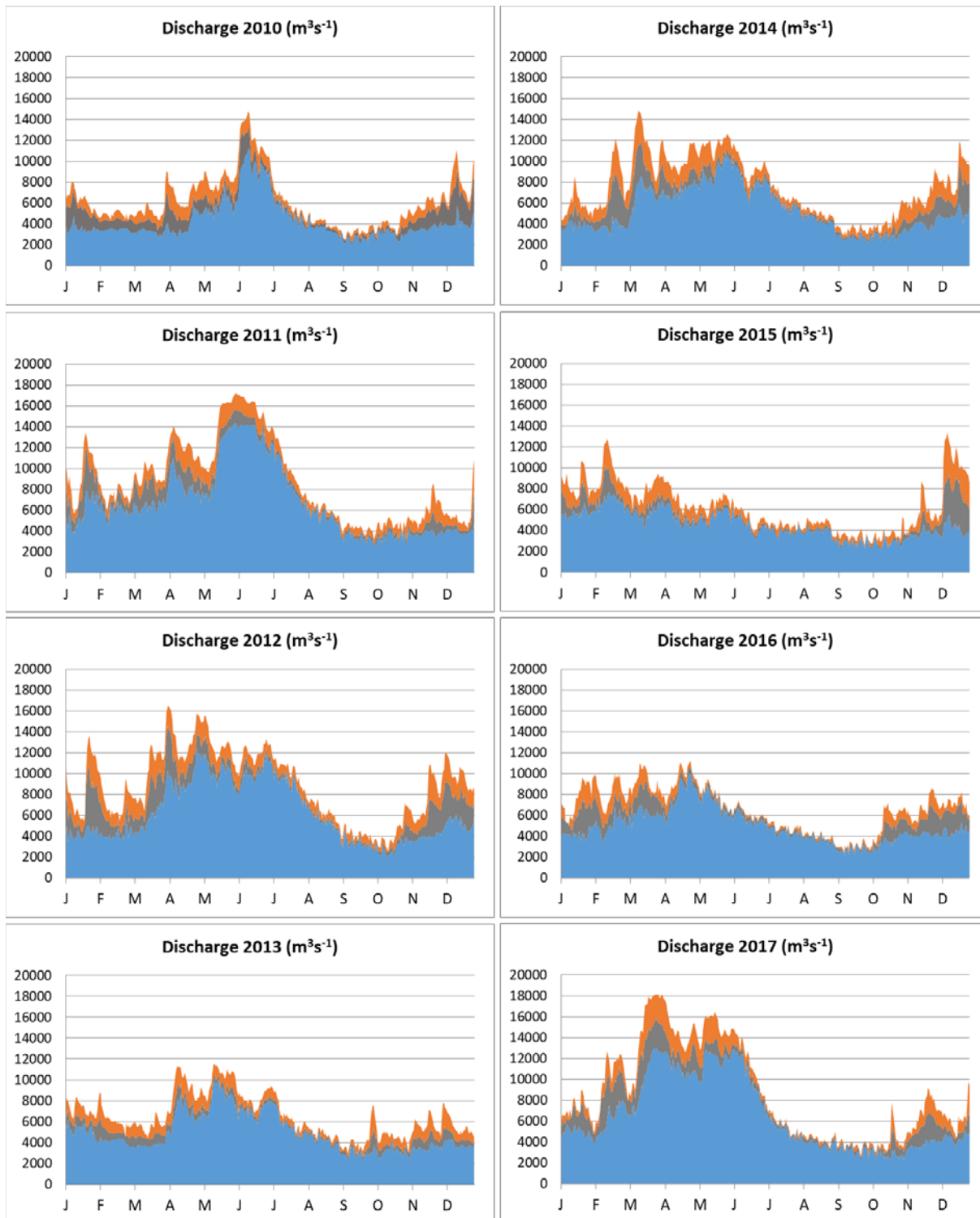
Figure 14. Daily discharge fluxes (m3/s) associated with Columbia River flow (blue), Willamette River flow (orange), and other tributaries (grey). Discharge from the Willamette was determined at the USGS stream gage at the Morrison Bridge; the contribution from other tributaries was computed by subtracting flows observed in the Willamette from those in the Columbia.
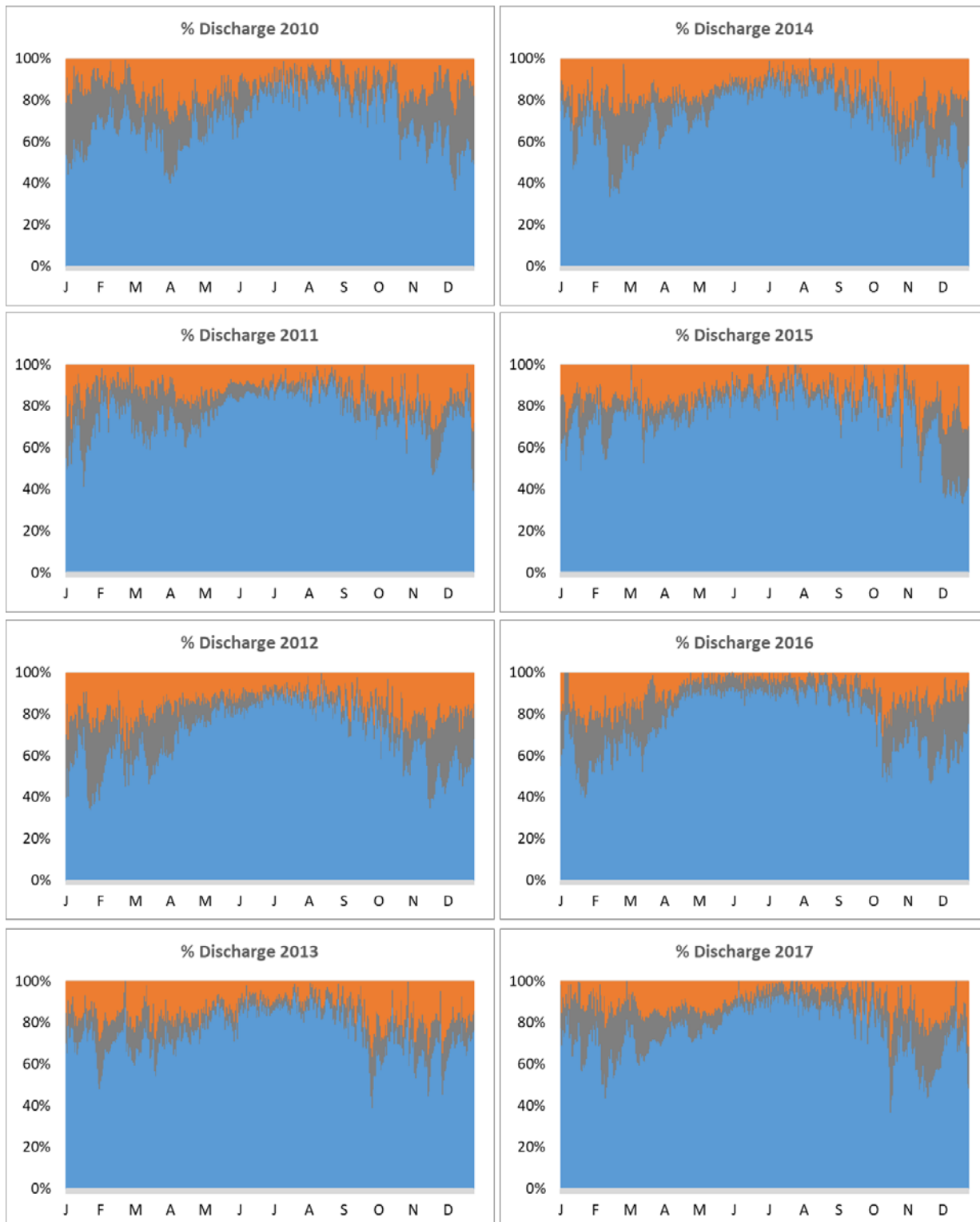
Figure 15. Daily proportional contributions of the Columbia (blue), Willamette (orange), and other tributaries (grey) to total river discharge observed at Beaver Army Terminal (RM 53). Discharge from the Willamette was determined at the USGS stream gage at the Morrison Bridge; the contribution from other tributaries was computed by subtracting flows observed in the Willamette from those in the Columbia.

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 2017 where temperatures exceeded thresholds associated with reduced performance or physiological stress with the years 2013, 2014, 2015, and 2016 (Figure 16). 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 >80 days in 2017, similar to 2013, 2014, and 2016. Only 2015 had more days above this threshold. 2017 was similar to other years in terms of the days exceeding 20 ºC; however, the number of very warm days (exceeding 21, 22, or 23 ºC) was greater than all other years except 2015. This is interesting since 2017 had much higher discharge fluxes compared to 2015; in 2017 the higher water volumes did not result in a buffer to the warming of mainstem river waters.
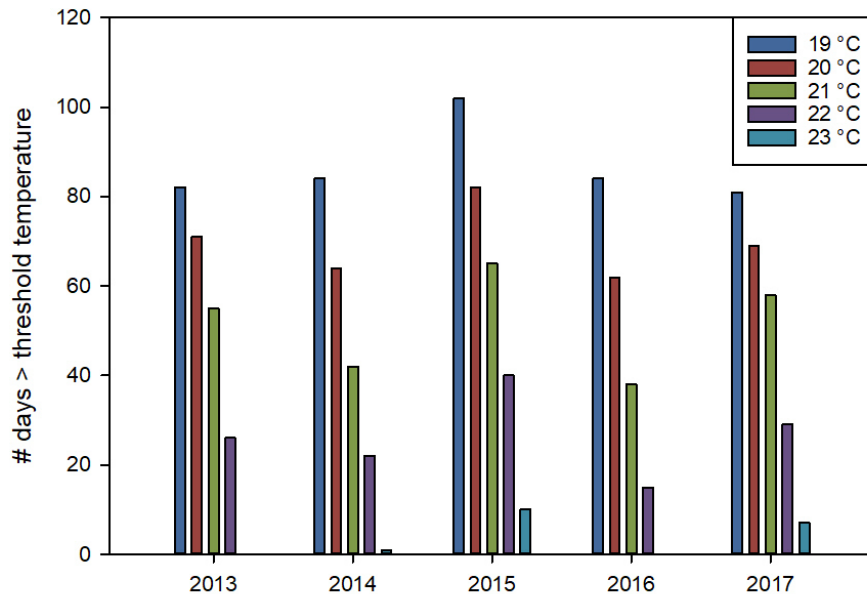
Figure 16. Number of days (computed from hourly data and converted to days, i.e., day-equivalents) where water temperatures exceeded the indicated temperature thresholds (19 ºC, 20 ºC, 21 ºC, 22 ºC, 23 ºC).

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 17. 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. The tributaries had a relatively large contribution to total discharge during the peak in April 2017, which was associated with peaks in colored dissolved organic matter (CDOM), turbidity, and nitrate and demonstrate 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 late February-early March, with smaller peaks in late May and in early July. The latter was associated with a decline in river discharge.
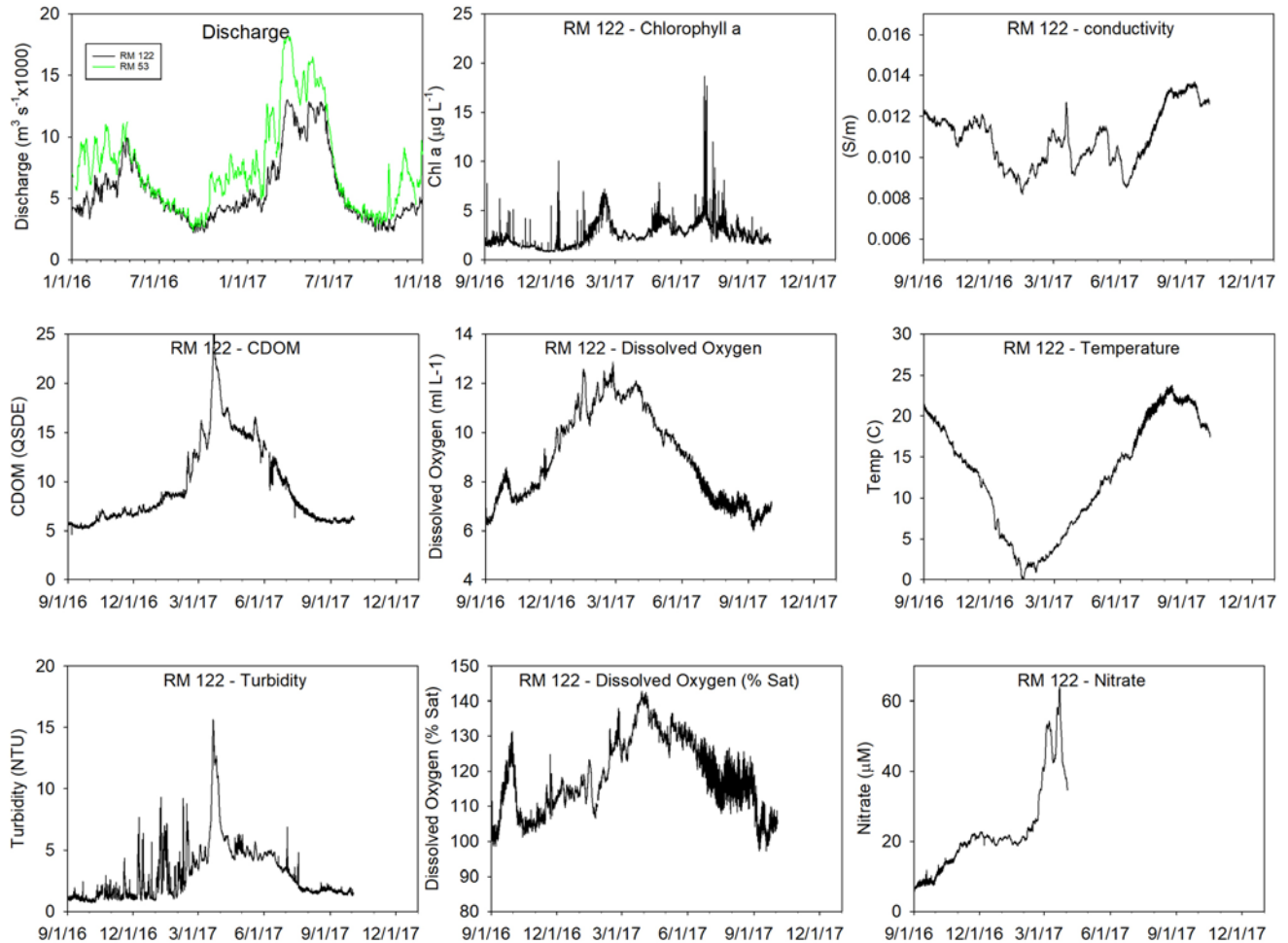
Figure 17. Time series of hourly water quality parameters measured at River Mile 122 (Camas, WA) in 2017.

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 17).

Dissolved nitrate concentrations exceeded 50–60 μM during peak flows 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). The strong growth of phytoplankton resulted in high percent saturation of dissolved oxygen in March–April. 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).

3.2 Abiotic Site Conditions

3.2.1 Continuous Water Quality

Hourly measurements of water quality parameters were made at five trends sites (Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough) using sensor packages moored at fixed depths.

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 70% in mid–late July and continued to decline until September. Dissolved oxygen percent saturation relative to the atmosphere was as low as 45% at Ilwaco during the warm months of August and September. pH fluctuated between 6.8 and 7.2, which falls within the recommended range (6.5–8.5; Washington State Water Quality Standards).
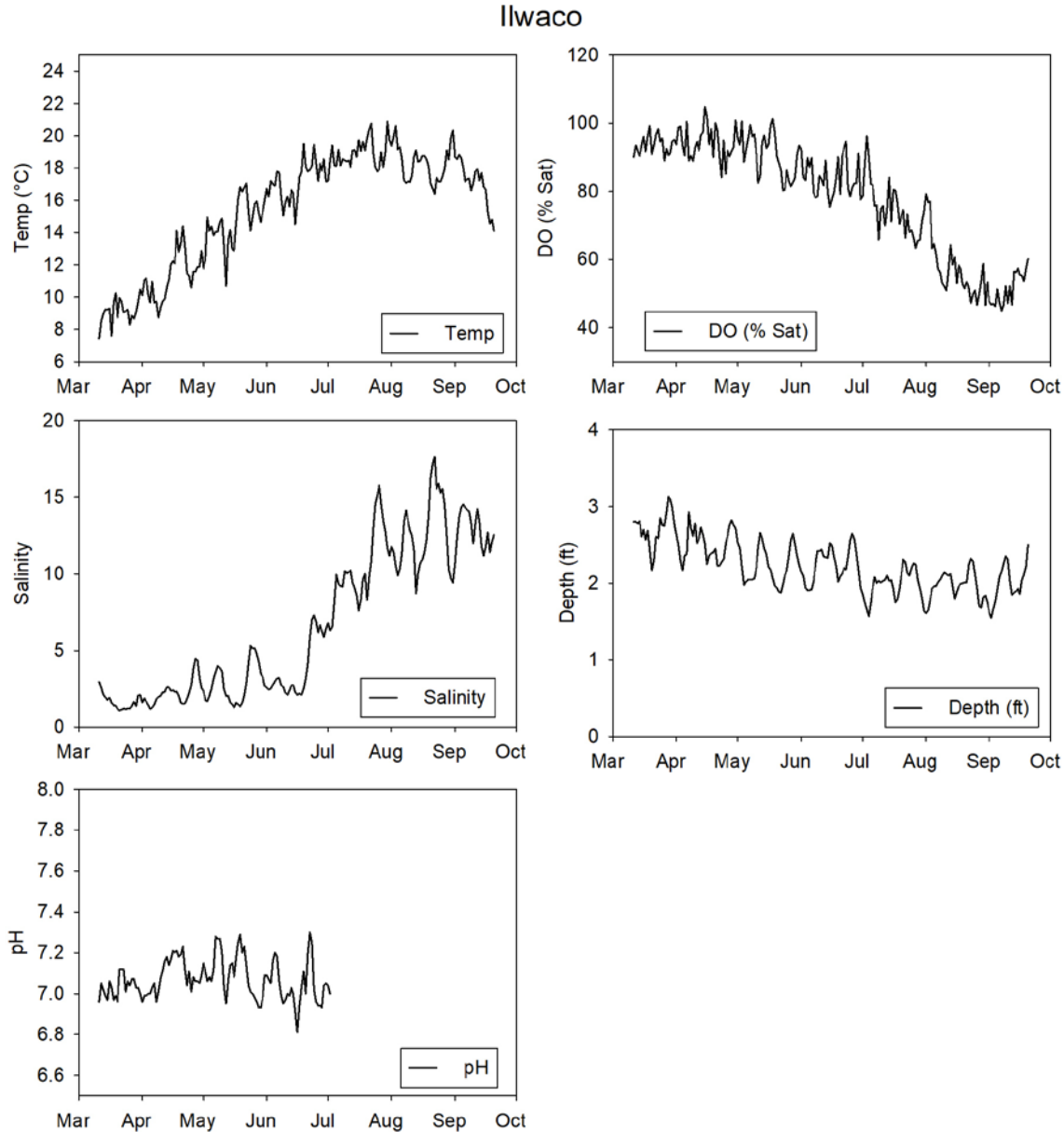
Figure 18. Time series of hourly measurements of water quality parameters made at Ilwaco Slough, 2017.

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 19). Dissolved oxygen saturation levels were much higher at Welch Island than at Ilwaco, ranging from ~85% 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 ~6.5 to 7.4, which falls within the recommended range (6.5–8.5; Washington State Water Quality Standards). There was a sharp increase in pH at the end of June as river discharge declined (Figure 19); since the instruments were calibrated regularly and laboratory testing did not reveal any instrument drift relative to standard solutions, we consider this observation to be robust.
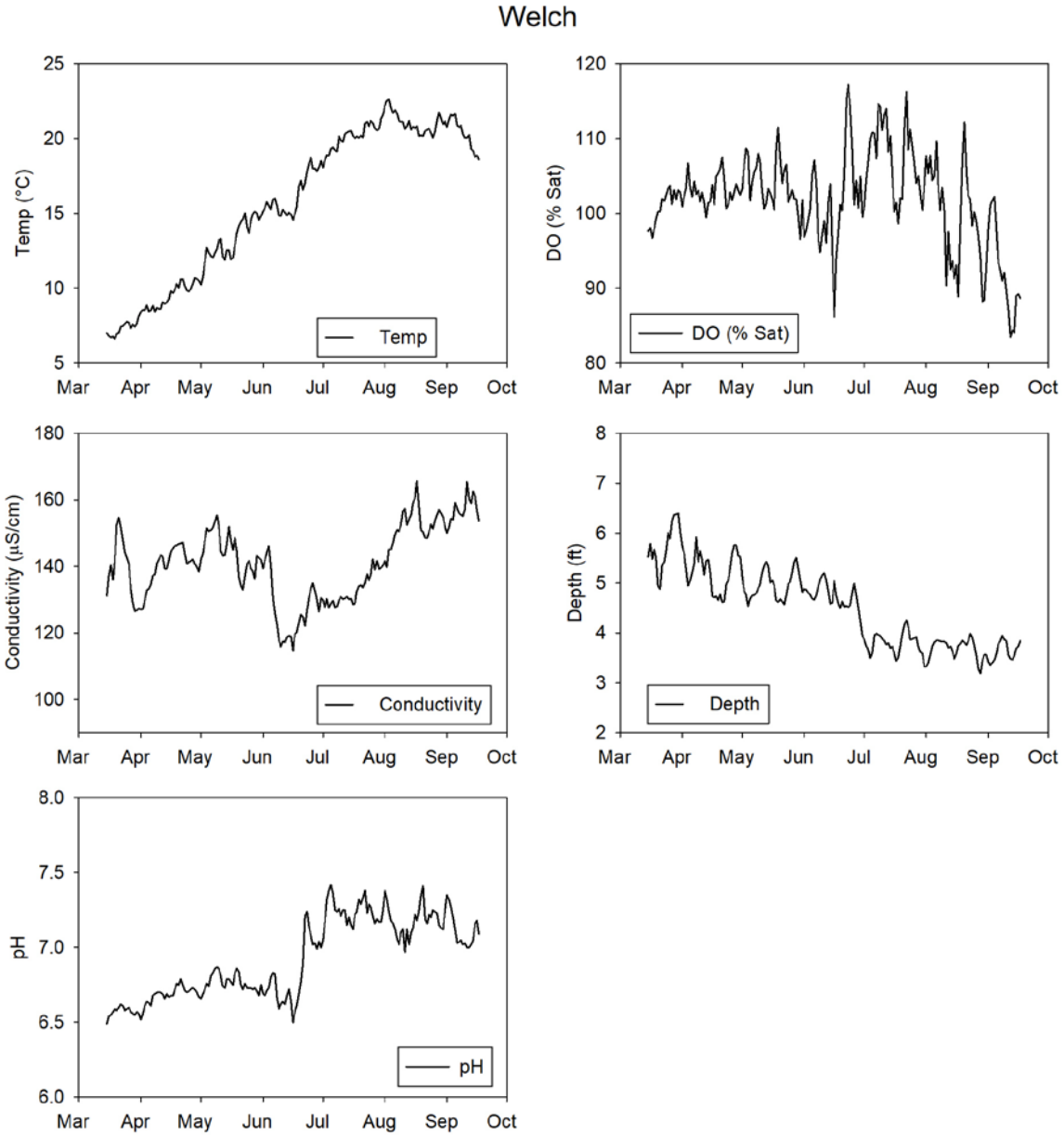
Figure 19. Time series of hourly measurements of water quality parameters made at Welch Island, 2017.

3.2.1.3 *Whites Island*

Temporal patterns in water quality parameters at Whites Island were very similar to those observed at Welch Island, although the specific range values different somewhat (Figure 20). pH was slightly higher at Whites (~7.0–7.6) compared to Welch, and the percent saturation of dissolved oxygen ranged from 85% in late summer to ~110% between April and June. Based on patterns in dissolved oxygen saturation, Whites Island appears to have greater primary productivity earlier in the spring, while Welch has greater primary productivity later in the spring and into the summer. The range of conductivity values was small at Whites Island compared to Welch.
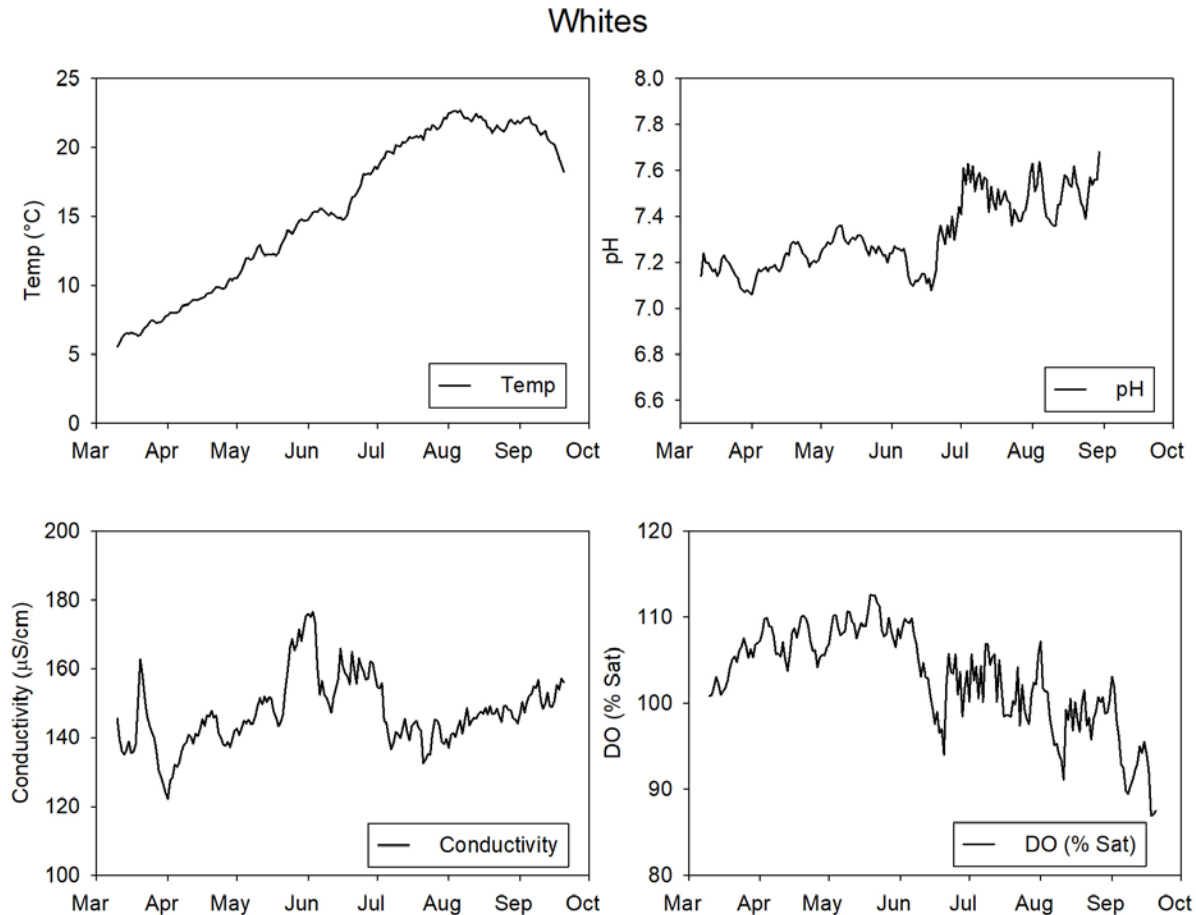
Figure 20. Time series of hourly measurements of water quality parameters made at Whites Island, 2017.

3.2.1.4 *Campbell Slough*

Of the five off-channel trends sites, Campbell Slough had the highest summer water temperatures, with values exceeding 25 °C in August (Figure 21). There was a dramatic change in water elevation, as shown by the change in sensor depth from spring to summer when the freshet subsided in June. As the water level dropped, conductivity and pH increased. pH increased to values close to 9 by late August-September, 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 in April when chlorophyll levels were high; similar peaks in chlorophyll were associated with low percent saturation of dissolved oxygen in the spring (April through June). However, it is unclear why chlorophyll peaks of similar magnitude to spring values did not lead to such dramatic declines in dissolved oxygen in the summer months; it is possible that increased water-air gas exchange resulting from reduced water levels could account for the observations, as could the longer daylight hours which would lead to more hours in the day when primary productivity exceeds respiration. Since chlorophyll concentrations were similar during both high and low night-time dissolved oxygen saturation levels, the latter hypothesis suggests another control on phytoplankton biomass, likely zooplankton grazing. Chlorophyll concentrations observed at Campbell Slough were below the recommended

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benchmark of 15 µg L^{-1} (based on three samples collected over three consecutive months; Washington State Water Quality Standards).

Campbell
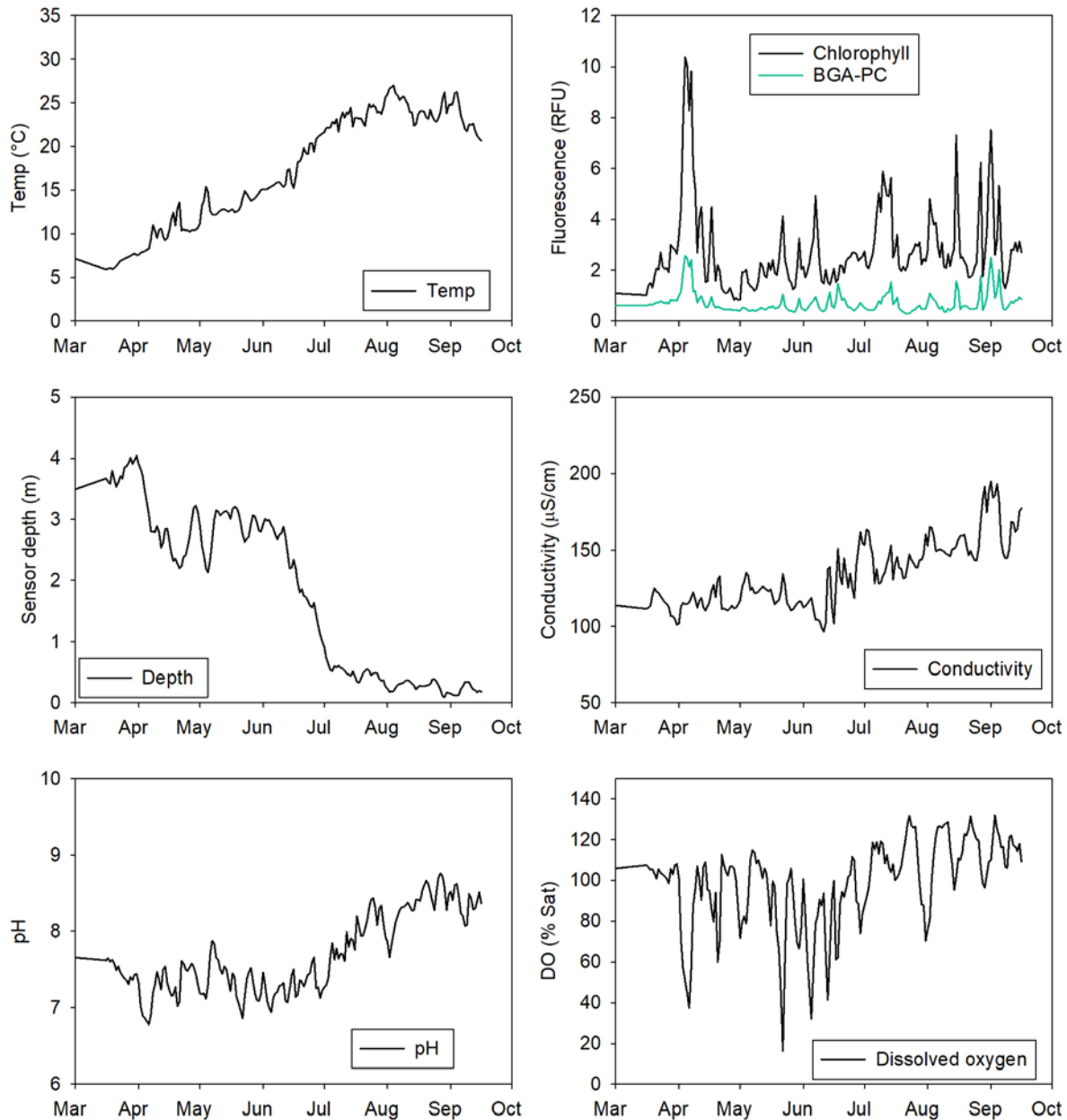
Figure 21. Time series of hourly measurements of water quality parameters made at Campbell Slough, 2017.

3.2.1.5 *Franz Lake Slough*

Water levels were too high to access Franz Lake Slough during the early part of 2017. Although not quite as high as observed at Campbell Slough, summer temperatures at Franz exceeded 25 °C from late-July through mid-August (Figure 22). There was a longer-lasting spring decline in conductivity compared to observations from some of the other sites (e.g., Whites and Welch Islands). Similar to Campbell Slough and the other sites, conductivity increased during the summer as river discharge fluxes decreased.

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There was a wide range of dissolved oxygen percent saturation values at Franz Lake Slough, ranging from ~30-40% in mid-August and mid-September to as high as 120% in May. Chlorophyll concentrations were of similar magnitude to observations at Campbell Slough; however, the highest concentrations were observed in late summer in contrast to Campbell Slough where peak concentrations were observed in April. Similar to Campbell Slough, chlorophyll concentrations did not exceed the recommended benchmark of 15 μg L^{-1} in three observations over three consecutive months.

pH values ranged from <5 to >9 at Franz Lake Slough. Some of these data points were outside of the recommended range for good water quality for the mainstem Columbia (7.0–8.5; Washington State Water Quality Standards). pH exceeded water quality benchmarks in early July during a peak in chlorophyll a that coincided with a peak in the cyanobacteria pigment, phycocyanin (Figure 22). The largest chlorophyll peak coincided with a smaller phycocyanin peak in mid-August. This peak was associated with increasing conductivity and low percent saturation of dissolved oxygen, likely due to high biological oxygen demand during the phytoplankton bloom.
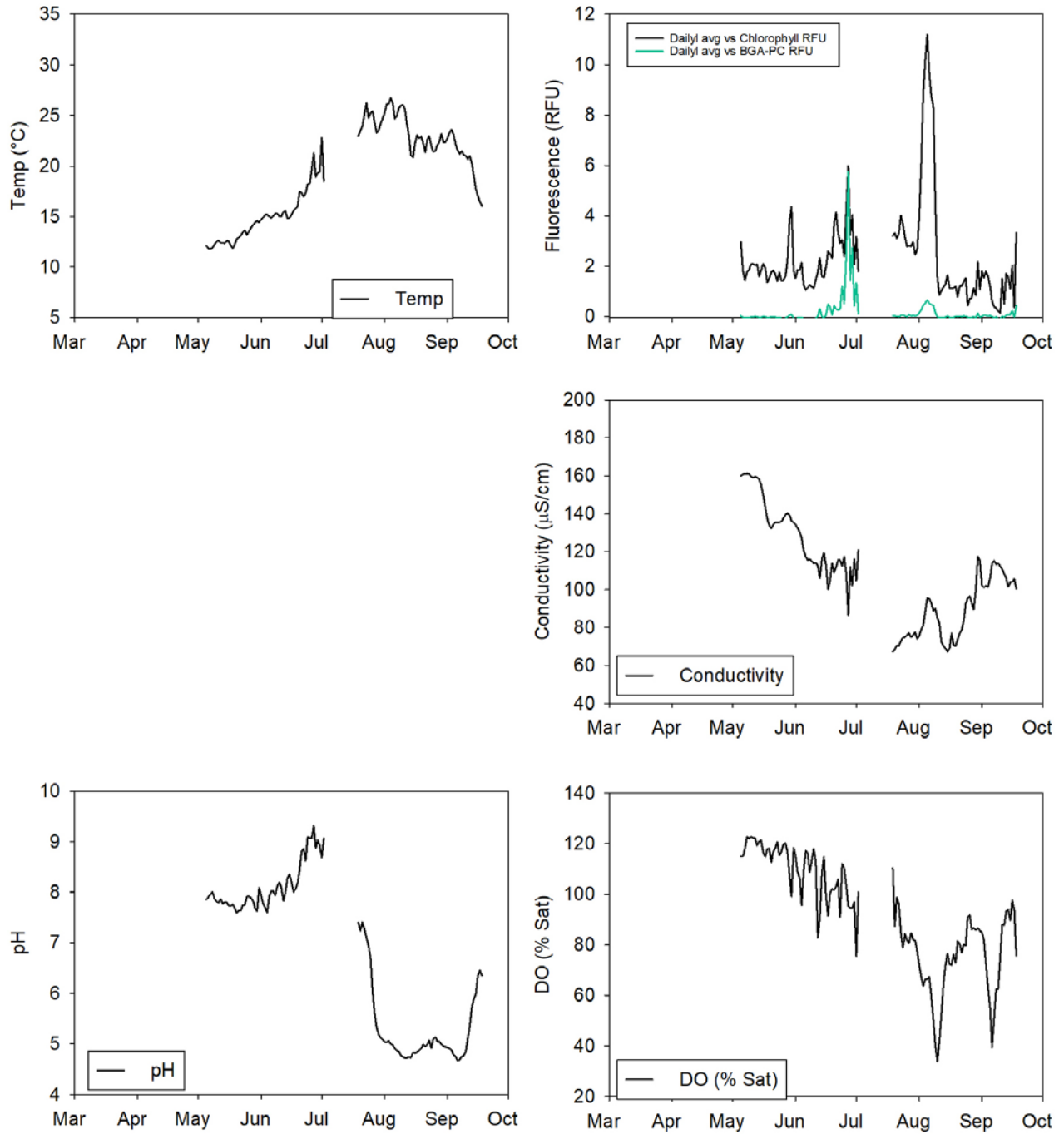
Figure 22. Time series of hourly measurements of water quality parameters made at Franz Lake Slough, 2017.

3.2.2 Dissolved Oxygen at Trends Sites

There was a wide range of values corresponding to dissolved oxygen saturation at the off-channel trends sites. It is recommended that dissolved oxygen should not fall below 6.0 mg L^{-1} for cold-water species, including native salmon (Oregon State Water Quality Standards); lower concentrations (4 and 2 mg L^{-1}) are considered to be increasingly detrimental to aquatic life. Using these thresholds to estimate stress associated with suboptimal levels of dissolved oxygen, we computed the number of hours below 6, 4, and 2 mg L^{-1} for each of the five trend sites (Figure 23). Ilwaco had the greatest number of hours where dissolved oxygen concentrations were below each of the three threshold levels. This trend was observed for each of the years for which we have data. Dissolved oxygen concentrations below the 2 mg L^{-1} threshold were observed at Ilwaco, Franz Lake Slough, Campbell Slough, and to a limited degree, Welch Island in 2017. Dissolved oxygen levels below 6 mg L^{-1} were observed at all of the sites, with the greatest number occurring in 2016 and 2017. Observations of dissolved oxygen <4 mg L^{-1} increased over the time series.
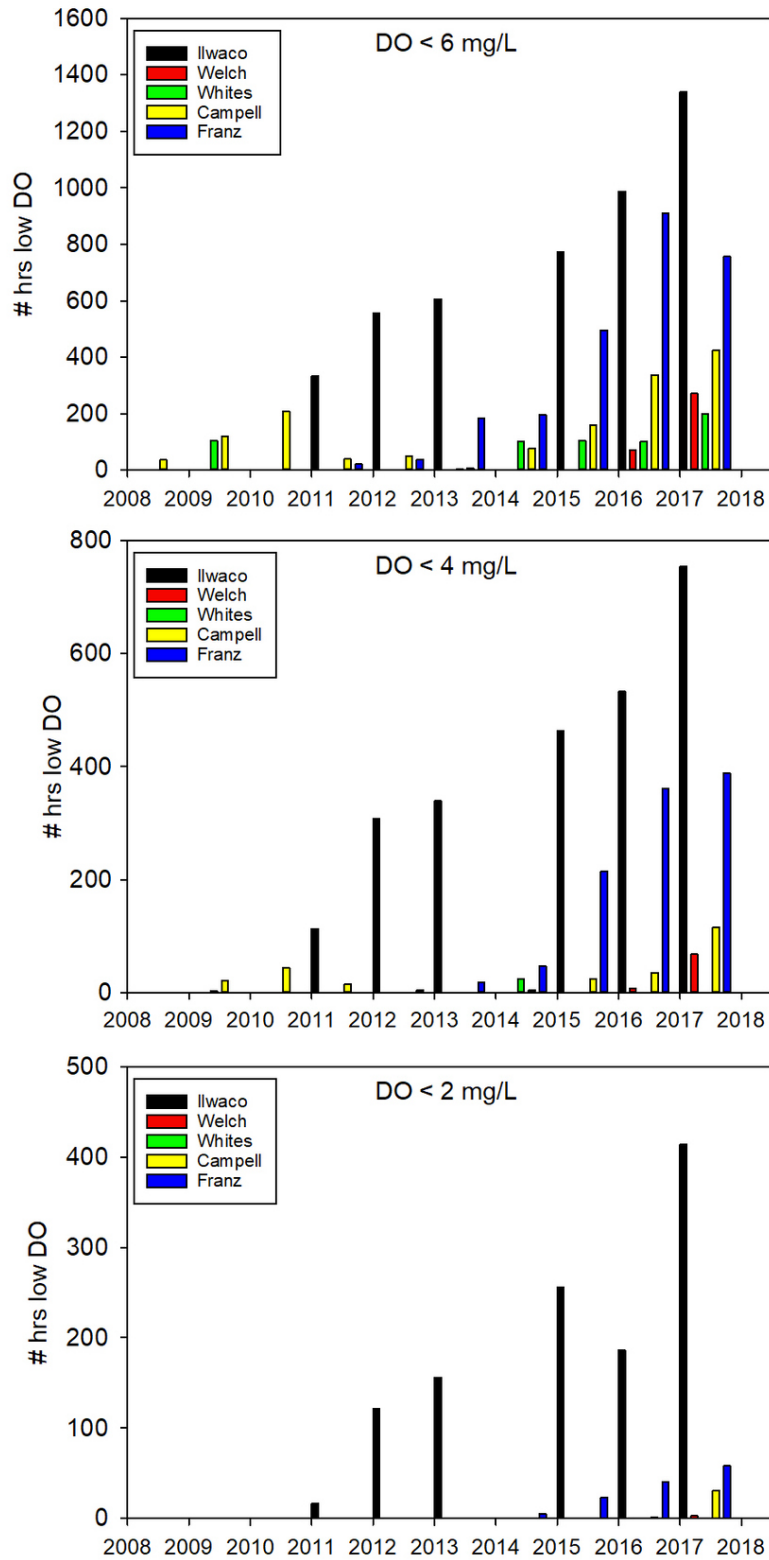
Figure 23. Time series of observations corresponding to three thresholds of dissolved oxygen concentration (mg L^{-1}): top panel, dissolved oxygen concentrations <6 mg L^{-1}; middle panel, dissolved oxygen concentrations <4 mg L^{-1}; bottom panel, dissolved oxygen concentrations <2 mg L^{-1}.

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 (Figure 24). The thresholds presented here are >18 ℃, >20 ℃, and >22 ℃. According to these criteria, high temperatures posed a potential problem for salmonids during the summers of 2009, and 2012–2016 at Campbell Slough, based on the high percentages of days where thresholds were exceeded. Frequent high temperatures were observed at Franz Lake Slough from 2013–2016. Whites Island also had a high frequency of days where temperatures exceeded the thresholds in 2015 and 2016.

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.
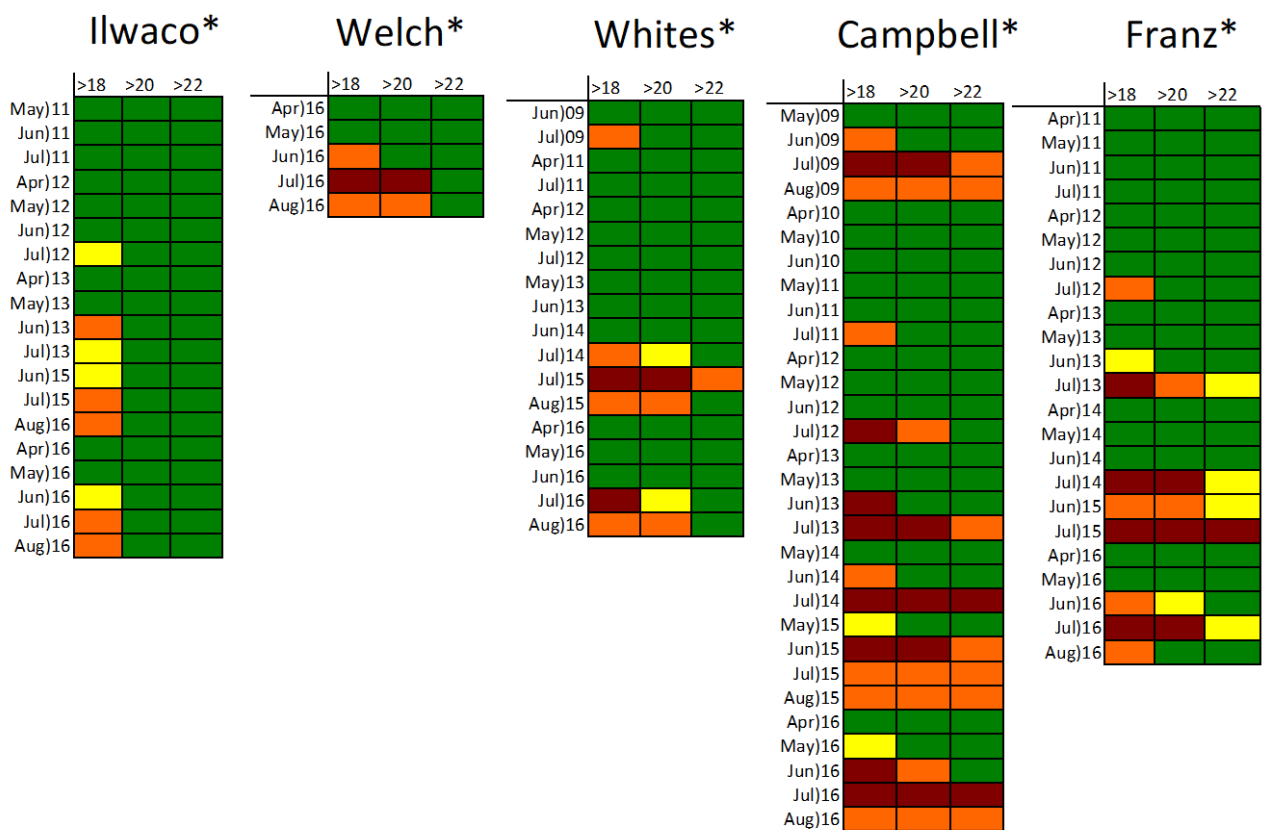
Figure 24. Plots showing time series of the percent of days in a month with average temperature exceeding one of three temperature thresholds (>18, >20, >22 ℃) at five off-channel trends sites (Ilwaco, Welch Island, Whites Island, Campbell Slough, Franz Lake Slough). Green = <25%; yellow = 25-49%; orange = 50-80%; red = >80%.

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. In 2017, nitrate concentrations were highest at Whites Islands in March, where values reached 40 μM (Figure 25). Although not quite as high as concentrations at Whites, nitrate levels at Ilwaco and Welch were also high (>30 μM). Nitrate concentrations were lowest during the summer months, reaching minimum values between July and September, depending on the site. An exception was at Ilwaco, where summer nutrients are supplied from the coastal ocean. There was a nearly complete drawdown of nitrate at Franz Lake Slough between March and June, with a >four-fold decline in nitrate concentration in the water column between March and May. 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.
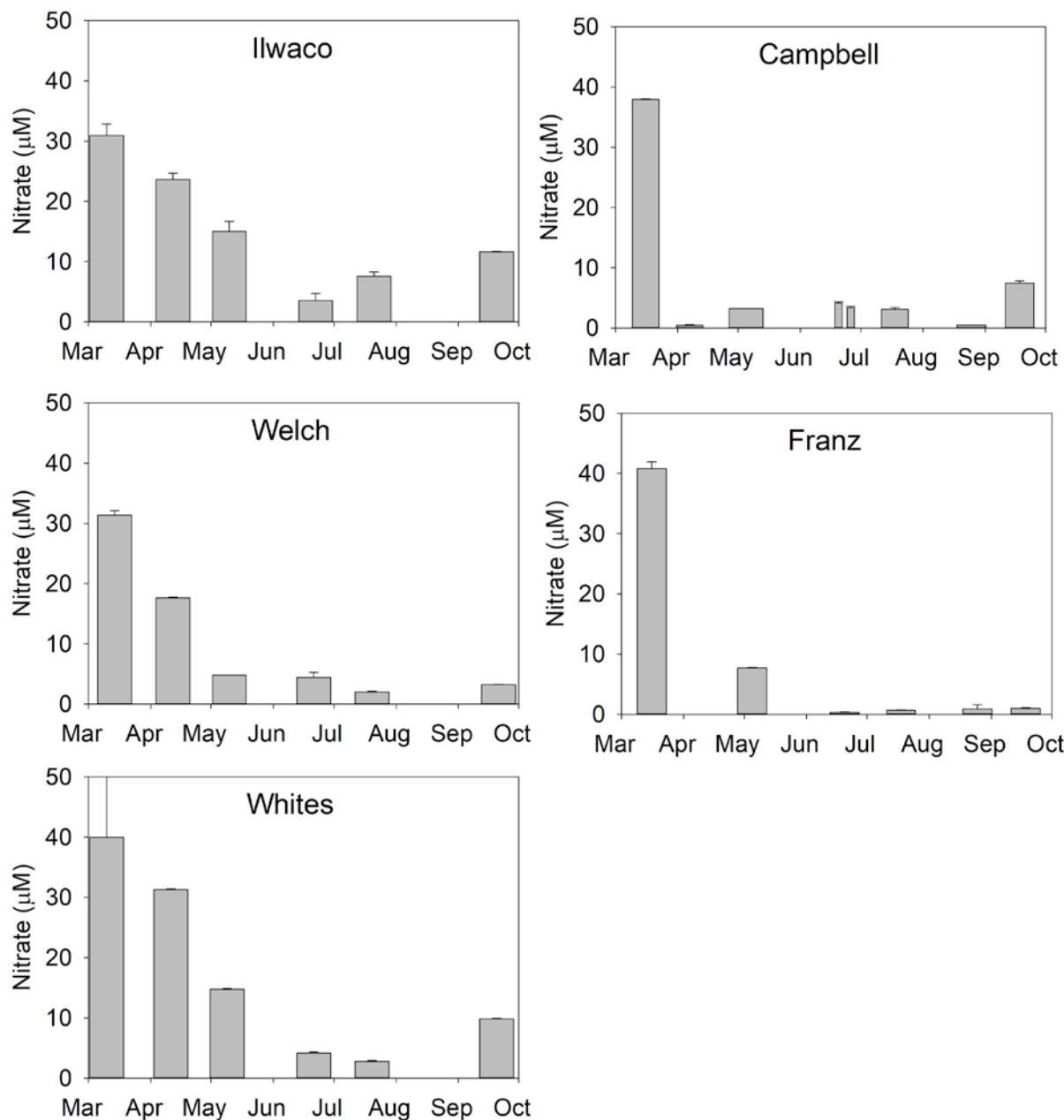
Figure 25. Time series showing concentrations of dissolved nitrate at the five trends sites in 2017.

Dissolved phosphorus (determined as soluble reactive phosphorus, or ortho-phosphate) exceeded 2 μM at Ilwaco Slough after April and from mid-July through September at Franz Lake Slough; at the latter site, concentrations exceeded 3 μM in mid-September. 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. Seasonal patterns in phosphate concentrations were similar between Welch Island and Whites Island, although there was a slightly larger dynamic range observed at Whites Island (i.e., higher highs and lower lows). Phosphate concentrations at Campbell were relatively constant throughout the sampling season; in contrast, there was a summer increase in phosphate at Franz Lake Slough; concentrations increased from < 1 μM to almost 3 μM between late June and the end of July (Figure 26).

Ammonium concentrations tend to increase as a result of microbial activity. High concentrations were observed at Ilwaco, particularly during the summer, where values as high as 18 µM were measured (Figure **27**). Ammonium concentrations were <2 µM at Welch Island, Whites Island, Campbell Slough (with a few exceptions were closer to 3 µM), and Franz Lake Slough.
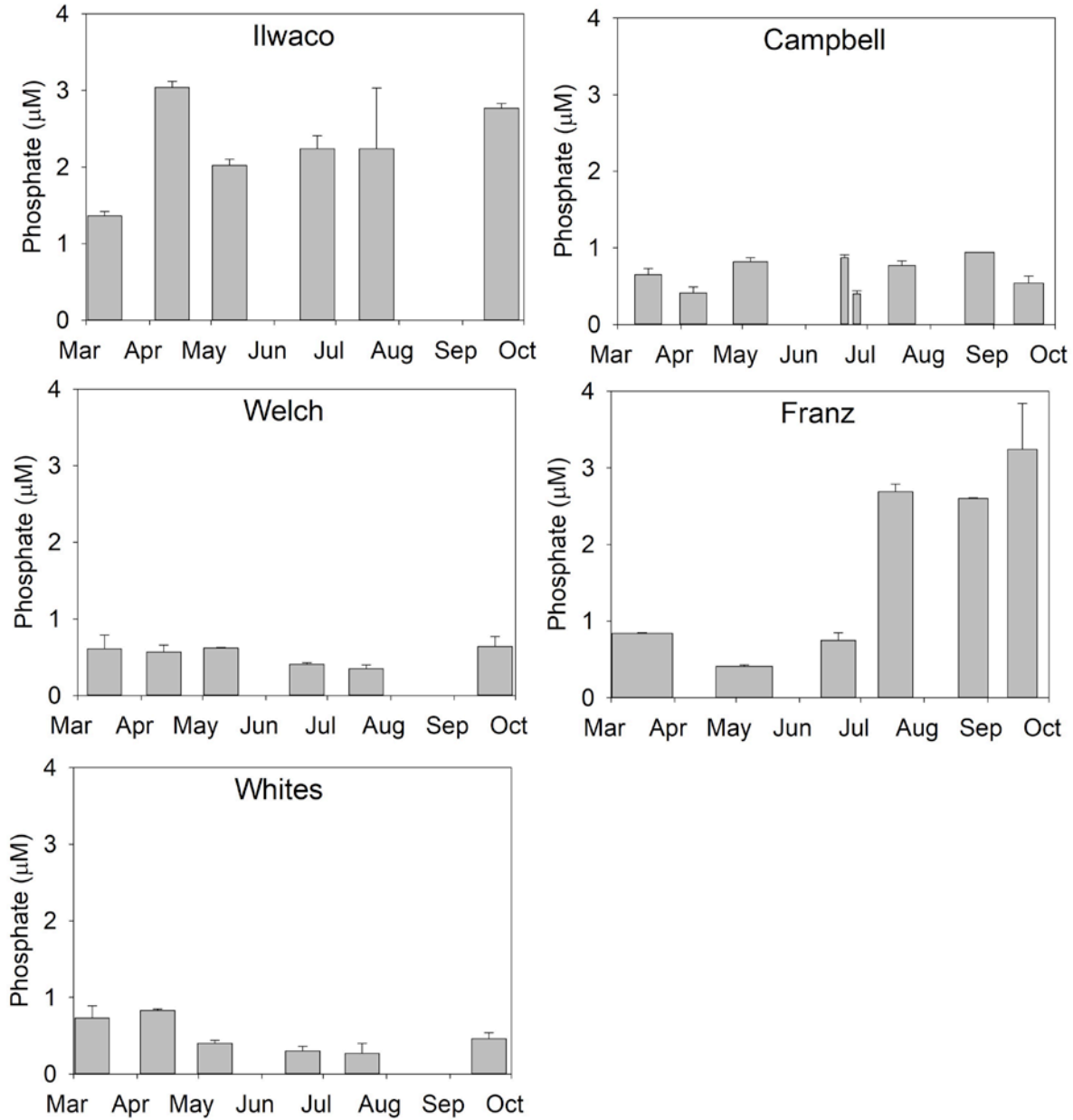
Figure 26. Time series showing concentrations of dissolved phosphate (ortho-phosphate) at the five trends sites.
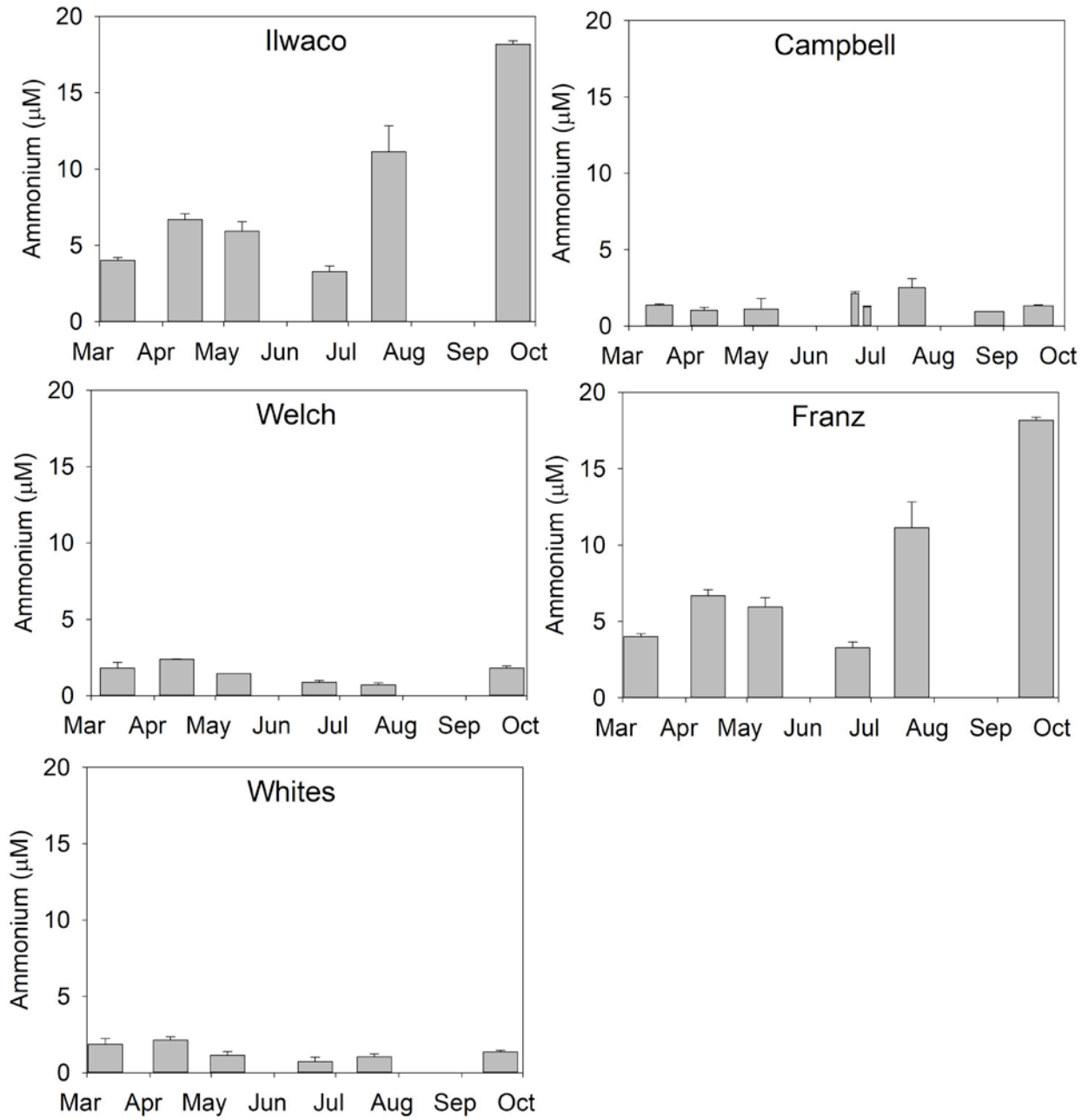
Figure 27. Time series showing concentrations of dissolved ammonium at the five trends sites.
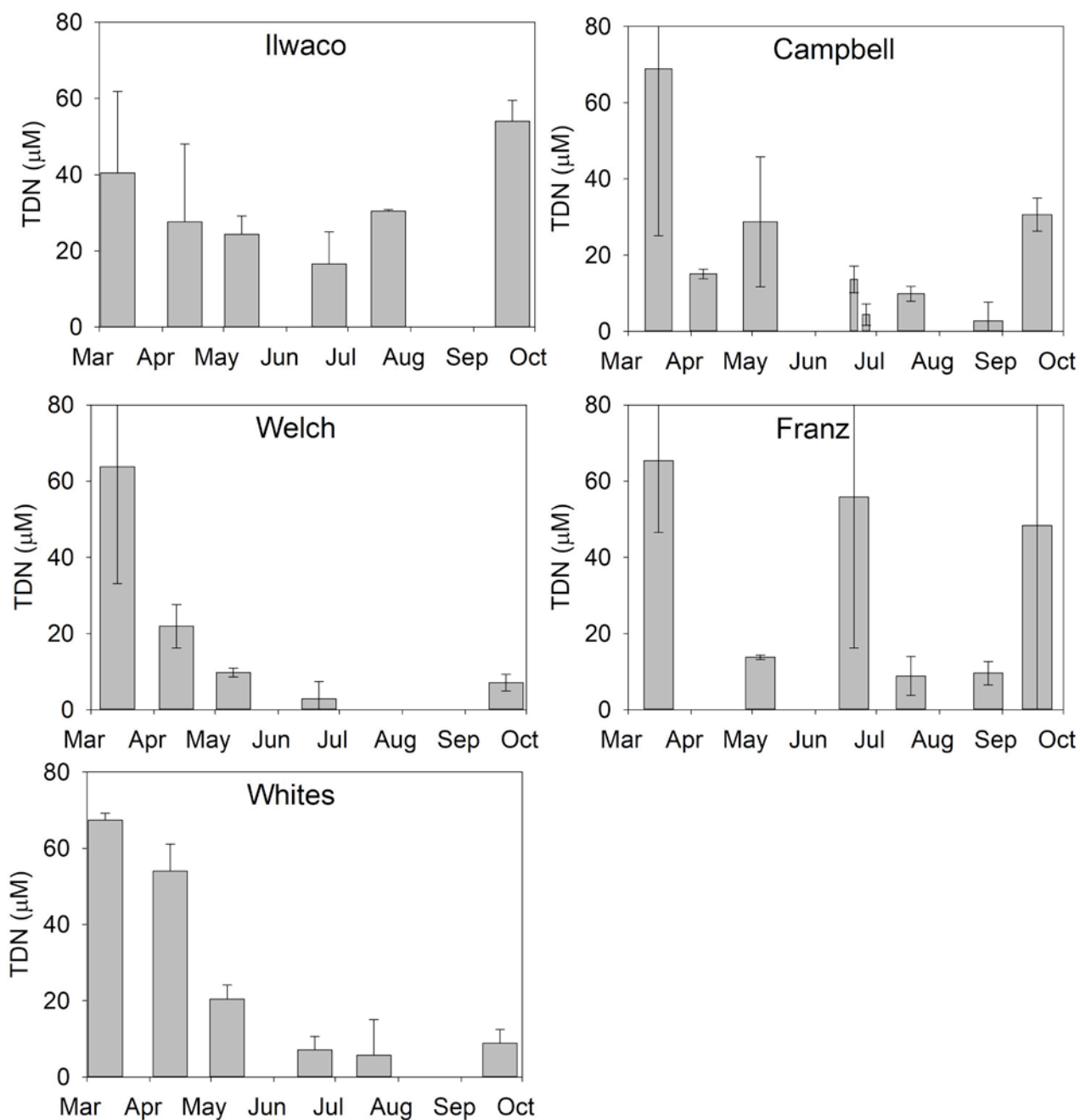
Figure 28. Time series showing concentrations of total dissolved nitrogen at the five trends sites.

Total dissolved nitrogen (TDN) was high at all sites in the early spring, declining during the period of strong phytoplankton growth (March-May) at Whites Island and Welch Island (Figure 28). In contrast to Whites and Welch, TDN did not decline to the same degree at Campbell Slough and Franz Lake Slough; in particular, Franz Lake Slough had high concentrations of TDN in late June and September, reflecting a high contribution of organic nitrogen to the total nitrogen at the site.

Concentrations of dissolved organic nitrogen (DON) were higher in the early spring (March and April) than in the summer at Welch Island and Whites Island (Figure 29); at Campbell Slough, higher concentrations of DON were sustained between March and May before declining. In conjunction with other dissolved nitrogen species (nitrate and ammonium), DON increased in the summer at Ilwaco. Franz Lake Slough had the highest concentrations of DON, particularly in late June when chlorophyll *a* concentrations and pH were elevated.
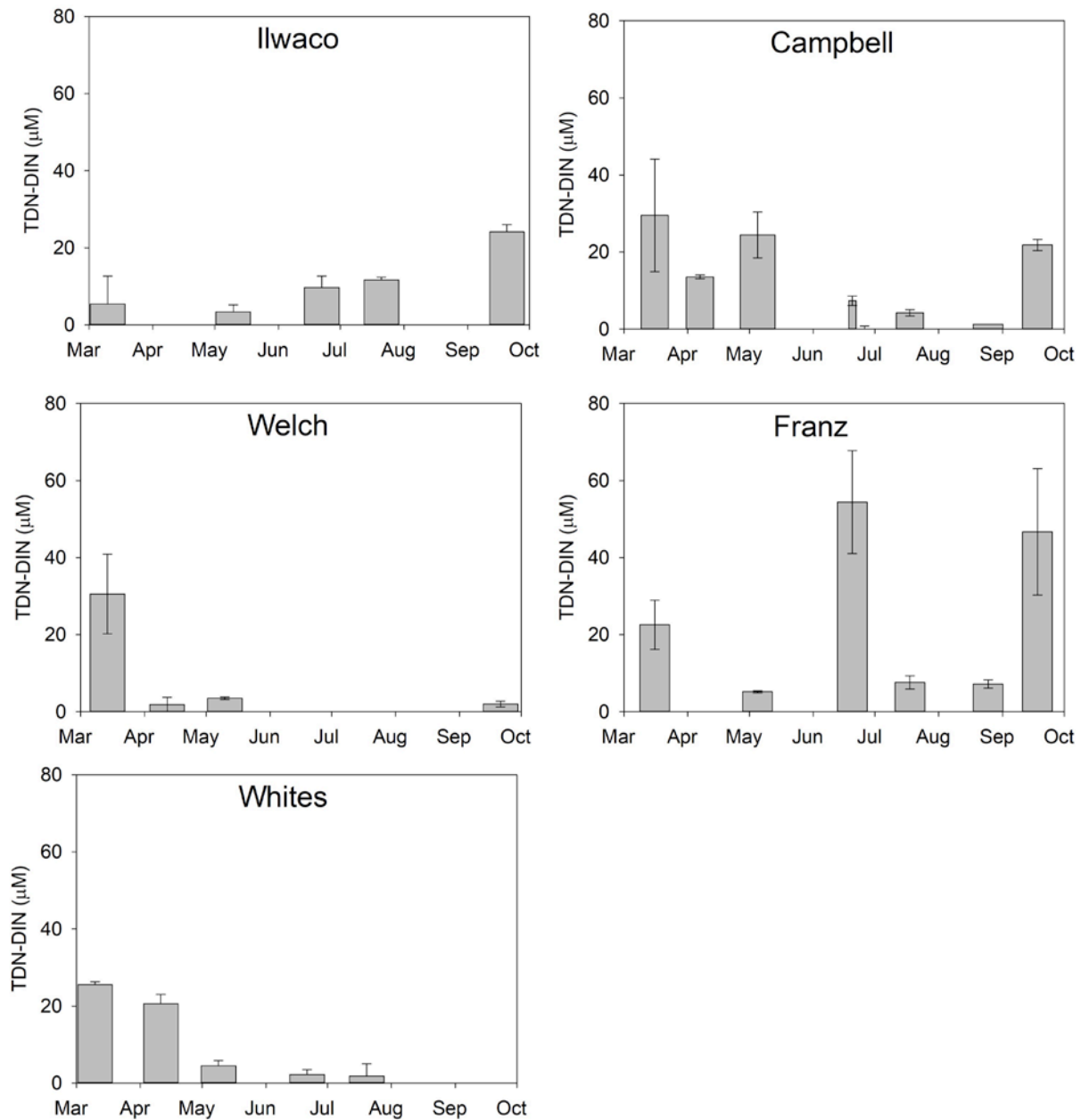
Figure 29. Time series showing concentrations of total dissolved nitrogen at the five trends sites. Dissolved organic nitrogen is derived by subtracting dissolved inorganic nitrogen from total dissolved nitrogen (TDN-DIN, labeled on the y-axis).

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Ratios of nitrogen to phosphorus (N:P) reflect the nutrient status of primary producers. A typical N:P ratio of 16:1 indicates nutrient-replete growth; ratios exceeding 16:1 suggest phosphorus limitation of growth, while ratios less than 16:1 suggest N-limitation. At Ilwaco, N:P ratios in the dissolved and particulate phases were generally close-to or less-than 16:1; after March, ratios were ~10 for most subsequent observations, although there was indication of N-limitation of growth in late June (Figure 30). All sites appeared to be P-limited in the early spring (March); however, only Welch Island and Whites Island demonstrated clear P-limitation after mid-March. At Whites Island, N:P ratios exceeded 16:1 until June; ratios were close to 16:1 in July and August, increasing to >20 by the end of the summer. At Welch Island, N:P ratios were ~10 from June-August, increasing to >20 by the end of the summer. Thus for each of these two sites, there was a general pattern of P-limited spring growth, nutrient replete late-spring/early summer growth, and mildly P-limited autumn growth.

There were more dramatic changes in nutrient ratios at Campbell Slough and Franz Lake Slough. Early spring growth was very likely P-limited, based on high N:P ratios; however, by April, N:P ratios were <10 (for both dissolved and particulate fractions) at Campbell and remained so until late September when they approached 16:1. At Franz, dissolved N:P ratios were even lower than at Campbell during the spring and summer, suggest very strong N-limitation of growth.

In general, N:P ratios in the particulate fraction were similar to the dissolved fraction, indicating that phytoplankton were assimilating nutrients at ratios similar to their availability. The exception was observed at Franz Lake Slough, where the particulate fraction had higher N content than would be expected from source nutrients. This likely reflects the fixation of N_2 by cyanobacteria, which were often abundant at this site.
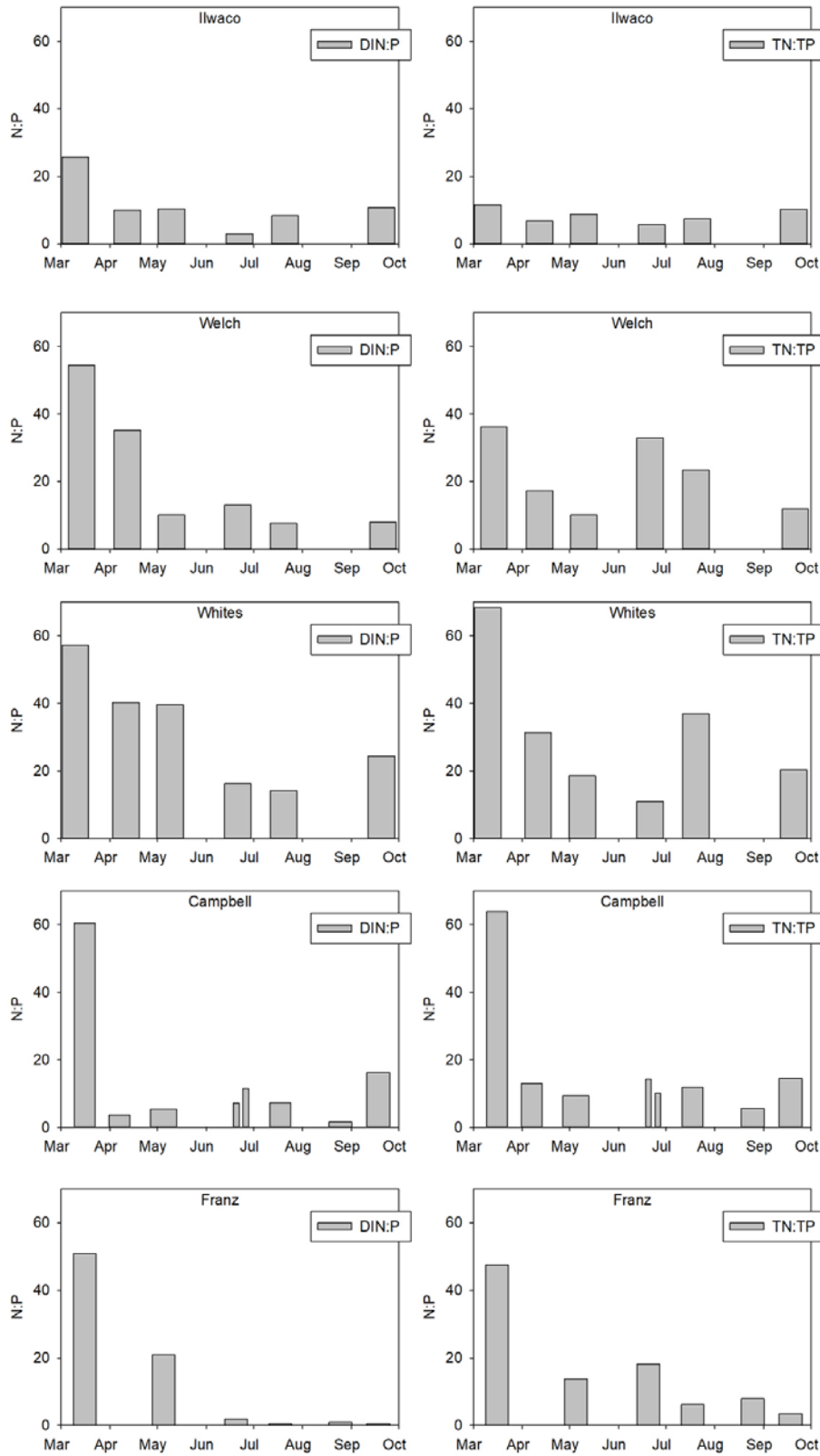
Figure 30. Ratios of dissolved and particulate nitrogen to phosphorus. DIN = dissolved inorganic nitrogen (=nitrate + ammonium); P = orthophosphate. TN = total nitrogen; TP = total phosphorus.

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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 31) 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 31). However, during the sampling season (March–September), 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 32).
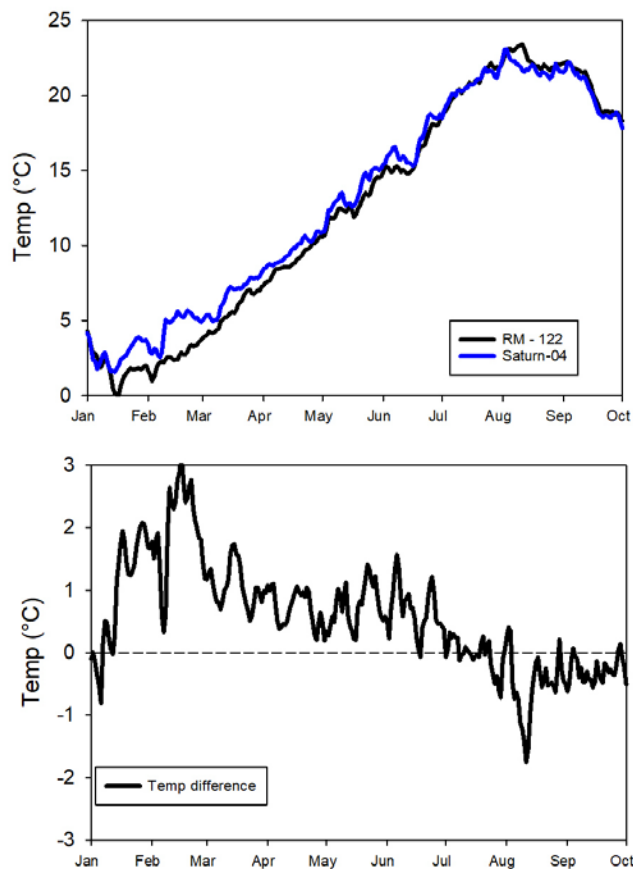
Figure 31. *Upper panel*: **Comparison of hourly water temperature data collected at two sites in the Columbia River mainstem (RM-122, or Camas, WA and RM-17, or Saturn-04). RM-122 is upstream of the Willamette-Columbia confluence, while RM-17 is below the confluence.** *Lower panel*: **Temperature difference (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 32). 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.
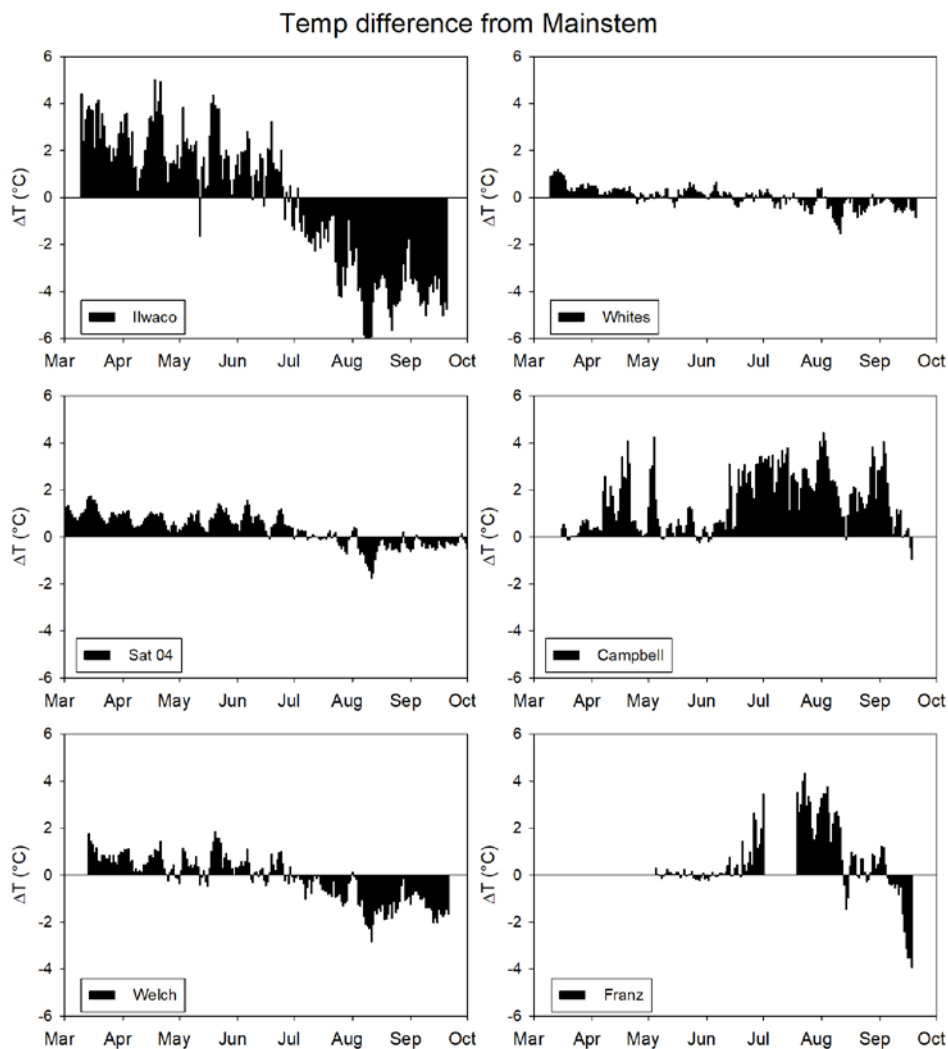
Figure 32. Time series of water temperature differences between the mainstem at Camas, WA (RM-122) and five off-channel trends sites, as well as Saturn-04.

3.3 Habitat Structure

3.3.1 Hydrology

Hydrology data collected in 2017 are still under analysis and unavailable at the time of the writing of this report. The following details are adapted from the 2016 report.

Hydrologic patterns vary from year to year at all but the most tidal sites. In 2016, water levels were elevated at the upriver sites between December 2015 and June 2016. Mean water levels (MWL) over the year were similar between all sites, ranging from 1.4 m CRD at Welch Island to 1.7 m CRD at Campbell Slough (Table 15); MWL was likely higher at Franz Lake in 2016 however sensor loss precluded data collection at that site). Hydrographs from all the years in which water surface elevation (WSE) was sampled at the trends sites, including the 2016 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. Average tidal range at the site was 1.51 m in 2016 (Table 15).

- The Welch Island site, located at rkm 53, is predominantly tidal; however, slightly elevated WSE was detectable during the prolonged spring freshet in 2012 and 2014. Winter storms also drive higher water levels at this site, particularly elevating the low tide levels. The tidal range of 2.22 m was greatest at this site in 2016 (Table 15) compared to other sites, primarily due to the depth of the tidal channel below the extreme low water level.

- The depth sensor at Whites Island malfunctioned in 2016; however, observations from previous years indicate that the hydrologic pattern at the site (rkm 72) exemplifies the mix of hydrologic drivers in the lower river. The average tidal range was 1.72 m in 2015 (Hanson et al. 2016), with elevated water levels occurring during winter storm events. In previous years, elevated water levels were also observed during the spring freshet.

- The depth sensor at Cunningham Lake malfunctioned in 2016; however hydrologic patterns at Cunningham Lake and Campbell Slough, at rkm 145 and 149, respectively, have similar hydrologic patterns. Data from 2015 indicate that Cunningham Lake has a slightly greater tidal range and slightly lower WSE during flood events compared to Campbell Slough (Hanson et al. 2016). 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 2016 winter storms also increased the WSE at these sites.

- The depth sensor at Franz Lake was lost during 2016, however observations from previous years indicate there is a small tidal signal (on average 0.24 m in 2015; Hanson et al. 2016) 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. In most years, the winter and spring high WSEs are both discernable, however, the spring levels are usually considerably higher than those in winter.

Based on data from the nearest water level gauge (Cascade Island, below Bonneville Dam), the site was inundated above the marsh surface most of the winter and spring through June in 2016.

Table 15. Water surface elevation (WSE) metrics calculated at each site for the sensor deployment period ending in 2016. All metrics are in meters, relative to the Columbia River Datum (CRD). MWL = mean water level; MLLW = mean lower low water; MHHW = mean higher high water.

Site	Rkm	MWL	MLLW	MHHW	Avg Tidal Range (m)	Maximum WSE	Date of Maximum WSE	Period of Record
Ilwaco[1]	6	1.49	0.94	2.45	1.51	3.42	3/10/2016	Aug 2015 - Aug 2016
Welch	53	1.42	0.32	2.54	2.22	3.69	12/10/2015	Aug 2015 - Aug 2016
Whites	72	NA	NA	NA	NA	NA	NA	NA
Cunningham	145	NA	NA	NA	NA	NA	NA	NA
Campbell	149	1.71	1.56	1.93	0.37	3.68	12/10/2015	Aug 2015 - Aug 2016
Franz	221	NA	NA	NA	NA	NA	NA	NA

The frequency of inundation at each site is dependent on the elevation, the position along the tidal and riverine gradient, and the seasonal and annual hydrologic conditions. The frequency of inundation at the average elevation of the sites in 2016 is shown in Figure 33. At all sites in 2016, the percent of time that the high marshes were inundated was greater over the whole year, ranging from 22 to 44 percent, than it was during the growing season, driven by higher winter water levels. In 2016, Campbell Slough had growing-season inundation frequency of 23 percent in contrast to a frequency of 3 percent the previous year. The lower-river high-marsh sites had slightly lower frequencies compared to previous years, ranging from 10 to 20 percent compared to 15 to 25 percent during the growing season.
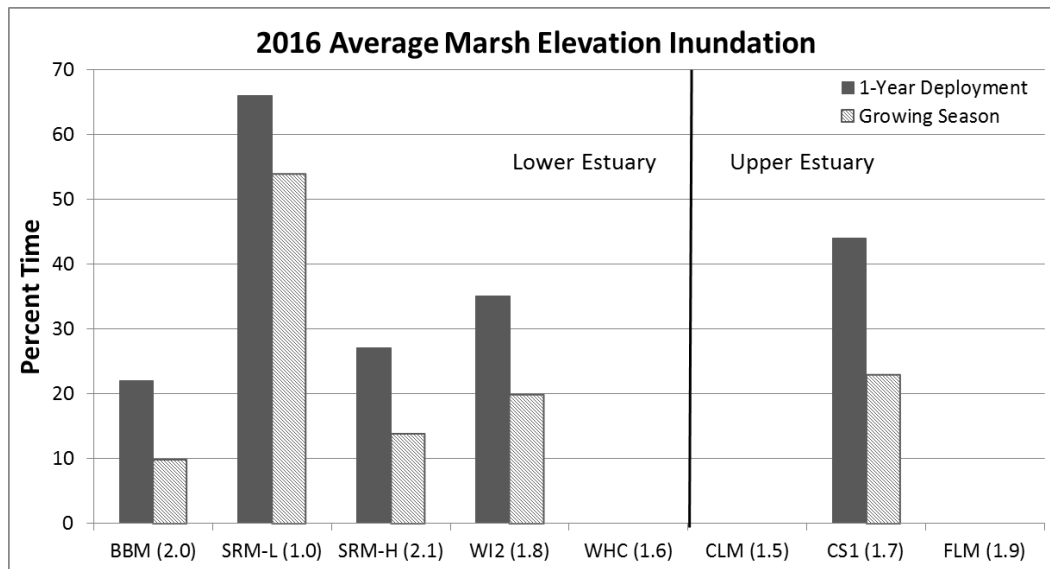
Figure 33. Inundation frequency at four of the trends sites in 2016; one-year deployment is from July 2015 to July 2016 and the growing season is from April -October. Site codes are defined in Table 8. Sites are ordered from left to right starting at the river mouth. Average site elevations are given in parentheses after the site codes. All sites are high marshes with the exception of Secret River Low Marsh (SRM-L), where the highest inundation occurred. Sensor loss or failure occurred at WHC, CLM, and FLM in 2016.

The cumulative inundation during the growing season, as measured by the sum exceedance value (SEV), is a means of comparing sites to each other and over time. In most years, cumulative inundation increases up-estuary, with the highest inundation at Franz Lake. However, in 2016 the SEV was similar at all four trends sites measured (Figure 34). Most of the inundation in 2016 occurred in a prolonged series of low-level flood events over the winter and spring.
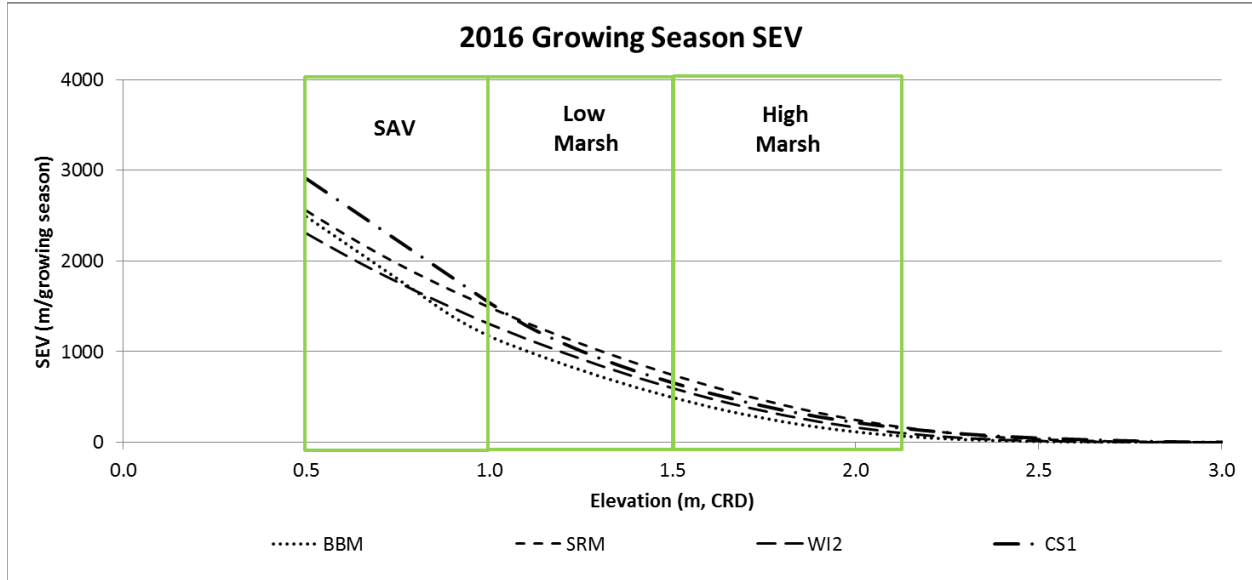
Figure 34. 2016 growing season sum exceedance values (SEVs) for the trends monitoring sites. Site codes are defined in Table 8. Plotted lines represent the calculated SEVs for 2016 at typical wetland elevations; the vertical lines represent the approximate boundaries between vegetation communities (SAV = submerged aquatic vegetation).

Inter-annual variation in inundation patterns is much greater at the upper estuary sites (Figure 35), 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. At Whites Island, the impact of high water during the 2011 and 2012 spring freshets is slightly discernable in the SEV at the average marsh elevation, whereas the up-estuary sites have large differences in the SEV between years. At Campbell Slough, the SEV in 2016 was the second lowest since data collection began in 2010, and it was most similar to 2013. At the lower estuary sites, inundation in 2016 was similar to other years.
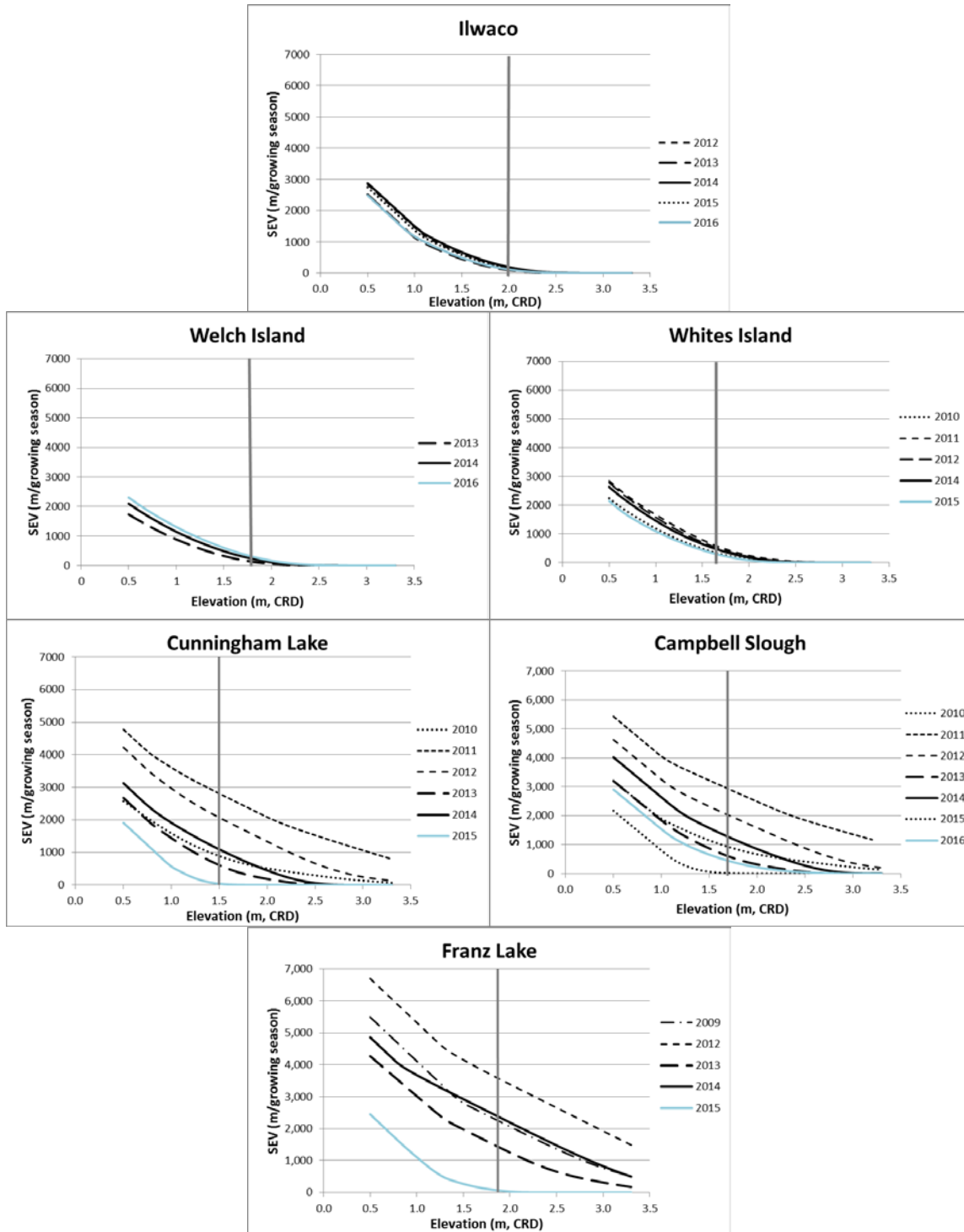
Figure 35. Annual growing season sum exceedance values (SEVs) for the trends monitoring sites. Plotted lines represent the calculated SEVs for a given year at typical wetland elevations; the blue lines indicate the most recent year available. The vertical line represents the average elevation at each site. Sites are ordered with the site nearest the Columbia River mouth at the top.

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. 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 16). 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 the 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 16) 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 16). 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 16. 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
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 *2017 Vegetation Patterns*

In 2017, the highest overall species richness was found at Whites Island (49 species, Table 17), though the highest average richness per plot was at Welch Island (10.3 species, Table 17). Both sites occur in wetland zone 2. As in previous years, the lowest species richness was at the brackish Ilwaco Slough (15 species). In wetland zone 4, Campbell Slough and Cunningham Lake both had similar richness (19), though Cunningham Lake had substantially higher average richness per plot (5.7) than Campbell Slough (3.2). Franz Lake (wetland zone 5) was intermediate in both total richness (26) and average plot richness (6.9). Zone 4 wetland sites (Cunningham and Campbell) had substantially lower species richness than zone 2, despite having about 50% more sample plots than the zone 2 sites (Welch and Whites).

Wetland zone 2, Welch and Whites Islands, had the highest total richness, as well as both the highest native and non-native species richness (Table 17Table 17). Ilwaco Slough had lower overall richness, but the highest proportion of species that were native (87%), while Campbell Slough had the lowest proportion of natives (63%). However, although Whites Island had both the highest total species richness and native species richness, it had the lowest proportion of total plant cover that was native (49%). The highest proportion of total plant cover that was native was at Welch Island and Franz Lake (85% each).

Cunningham Lake, in wetland zone 4, had the second lowest non-native species count (3), after Ilwaco Slough, while Campbell Slough was intermediate at 6 non-native species. However, Campbell Slough has low total richness to begin with, so it had proportionally more non-native species (32%) than any other site. Franz Slough in wetland zone 5 was intermediate in both number (6) and proportion (23%) of non-native species.

Table 17. 2017 tidal marsh species richness and areal cover at the trend sites. OBL species are obligate wetland indicator species, or species that are found in wetlands more than 99% of the time (Reed 1988).

	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
Rkm	6	53	72	145	149	221
# plots	40	41	44	69	65	61
Site Species Richness	15	38	49	19	19	26
Avg. Rich. Per Plot	4.8	10.3	7.2	5.7	3.2	6.9
Avg. % Cover Per Plot (SD)	114 (31)	140 (23)	103 (31)	67 (39)	61 (30)	112 (42)
Native Species						
Richness n (%)	13 (87)	29 (76)	38 (78)	15 (79)	12 (63)	19 (73)
Relative Cover (SD)	75 (22)	85 (32)	49 (37)	63 (32)	59 (29)	85 (37)
Non-Native Species						
Richness n (%)	2 (13)	9 (24)	10 (20)	3 (16)	6 (32)	6 (23)
Relative Cover (SD)	25 (28)	15 (26)	51 (34)	36 (24)	41 (35)	14 (23)
OBL Wetland Species						
Richness n (%)	10 (67)	19 (50)	29 (59)	12 (63)	11 (58)	11 (42)
Relative Cover (SD)	74 (20)	78 (35)	44 (37)	45 (26)	56 (28)	54 (33)

Tidal marshes in wetland zone 4 (Campbell Slough and Cunningham Lake) had substantially lower % vegetative cover (61-67%) than all other sites (103-140%). Welch Island had the highest cover (140%), while the sites at the two ends of the tidal estuary, Ilwaco Slough (114%) and Franz Lake (112%) were similar to each other, with relatively high total cover.

Most observed plant species have been assigned a wetland indicator value, reflecting how dependent they are on wetland hydrology (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. We can examine the proportion of plant cover that falls within the different wetland indicator categories to understand how wetland communities respond to variability in hydrology and to track change over time

and space. Additional analysis of these patterns is described later, but in 2017 OBL wetland species dominated the vegetative cover in the tidal marshes at the two most downstream sites, Ilwaco Slough (74% of total cover was OBL, Table 17), and Welch Island (78% OBL). The lowest OBL cover was at Whites Island (44%), while the other sites ranged from 45-56% OBL. One of the most important drivers of this pattern is likely *Phalaris arundinacea* (reed canary grass), which is a common non-native species classified as FACW. Whites Island had the highest relative % cover of *P. arundinacea* (42%) and as discussed later, as the relative cover of *P. arundinacea* increases, the relative cover of OBL species tends to decline.

The six most common plant species throughout the tidal estuary are shown in Table 18. *Carex lyngbyei* increased slightly in relative cover to 18% from its 2016 level of 17%. In contrast, *Phalaris arundinacea* (reed canarygrass) decreased to 22% relative cover from its 2016 value of 25%. However, *C. lyngbyei* had a lower frequency (37%) than *P. arundinacea* (52%). The latter was present at all sites except Ilwaco Slough, while *C. lyngbyei* was common only in the three most downstream estuary sites. 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 18. **The most common plant species in tidal marshes across all six trend sites. Species are listed in order of 2017 average relative % cover. Numbers in parentheses are standard deviations (SD). Average relative % cover estuary-wide was calculated by taking the average of all years (through 2016) at each site then averaging all sites. The 2017 data was averaged by site then all sites averaged. % frequency is the % of sample plots in which the species occurred. The "number of sites common" is the number of trend sites where the species was at least 5% relative % cover in 2017.**

Scientific Name Common Name	Wetland Status	Category	Native	# sites common	Average Rel % Cover		% Frequency	
					All Years	2017	All Years	2017
Phalaris arundinacea reed canarygrass	FACW	grass	no	5	27.5 (19.1)	22.0 (16.9)	47.9 (28.1)	51.9 (30.3)
Carex lyngbyei Lyngby sedge	OBL	sedge	yes	3	17.0 (22.8)	17.7 (22.2)	35.5 (40.5)	37.4 (42.6)
Sagittaria latifolia wapato	OBL	herb	yes	3	6.9 (4.7)	11.6 (11.0)	40.1 (22.3)	51.2 (27.7)
Eleocharis palustris common spikerush	OBL	sedge	yes	3	11.3 (12.1)	7.2 (8.1)	38.7 (21.3)	38.1 (22.1)
Polygonum amphibium water smartweed	OBL	herb	yes	1	5.1 (11.2)	6.6 (14.7)	11.1 (23.9)	14.2 (31.1)
Agrostis sp. bentgrass	FAC	grass	no	1	2.1 (4.6)	3.8 (8.4)	10.8 (23.3)	12.5 (28.0)

Sagittaria latifolia (wapato) was as frequent as *P. arundinacea*, though had an average relative % cover of only 11.6%. *Polygonum amphibium* (water smartweed) was common only at Franz Lake, where its abundance gave it an overall average relative % cover of 7%. *Eleocharis palustris* (common spikerush) was common at Welch Island, Cunningham Lake, and Campbell Slough, reaching an average relative % cover of 7%.

Overall, *Carex lyngbyei, Sagittaria latifolia, Polygonum amphibium,* and *Agrostis sp.* all had higher relative cover and higher frequency in 2017 than their long-term average values. *Phalaris arundinacea* was 5.5% lower in relative cover than its long-term average, however its frequency increased by 4%. *Eleocharis palustris* was 4% lower in relative cover and declined slightly in frequency from its long term average.

P. arundinacea decreased in cover in 2017, despite the fact that most sites had higher total vegetation percent cover in 2017 than in 2016. At five out of the six trend sites, reed canarygrass decreased in relative % cover, but increased in frequency (Table 19). Frequency is the percent of sample plots that contained reed canarygrass. The sixth site, Campbell Slough, showed the opposite trend, with an increase in relative cover but a decrease in frequency. Although the overall average relative cover of reed canarygrass decreased, the fact that frequency increased suggests that actual cover in plots that already had reed canarygrass likely didn't decline much if at all, and that the decline in average relative cover may be an artefact of low cover in newly invaded plots. At Welch Island, relative % cover of reed canarygrass declined by more than 1% but frequency increased 9%. At Whites Island, which typically has the highest proportion of reed canarygrass, relative cover declined by about 5%, but frequency increased 2%. Cunningham Lake had the greatest decline in relative % cover from 44 to 34%, but frequency increased 6% from 75% to 81%. At Franz Lake, cover decreased by 2% while frequency increased a substantial 24%.

Table 19. *Phalaris arundinacea* (reed canarygrass) average relative % cover and frequency (% of sample plots) at the trend sites between 2005 and 2017. 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	43.1	63	39.9	60	ND	ND
2006	ND	ND	ND	ND	ND	ND	33.2	61	50.1	48	ND	ND
2007	ND	ND	ND	ND	ND	ND	41.2	77	41.9	48	ND	ND
2008	ND	ND	ND	ND	ND	ND	48.4	60	47.8	47	40.0	93
2009	ND	ND	ND	ND	40.2	80	37.9	55	37.9	48	35.0	71
2010	ND	ND	ND	ND	47.2	86	57.2	69	44.3	58	ND	ND
2011	0.0	0.0	ND	ND	60.9	88	54.2	75	59.2	62	41.8	81
2012	0.0	0.0	5.0	17	48.3	83	49.2	62	29.2	49	9.1	69
2013	0.0	0.0	7.6	13	50.4	76	55.1	61	45.0	44	18.4	63
2014	0.0	0.0	7.6	38	49.3	88	35.9*	64*	45.9	48	11.1	46
2015	0.0	0.0	6.6	25	50.3	77	45.4	67	30.1	50	13.2	48
2016	0.0	0.0	7.3	23	47.0	80	44.0	75	35.9	51	14.4	51
2017	0.0	0.0	5.9	32	42.4	82	33.6	81	39.3	42	11.0	75

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

3.3.3.2 *Vegetation Community Change Analysis*

Total species richness across all six trend sites declined in 2017 (93 species) from the high values seen in the previous two years (100-101 species, Table 20). The number of non-native species declined more than the native, with the proportion of species that were native actually increasing from 73% in 2016 to 75% in 2017 (Table 20). A similar pattern was seen with % cover, with an overall decline in 2017 of 5% total vegetation cover, but a gain in the relative cover provided by native species (Table 21).

At all sites, species richness reached maximum values during at least one of the years 2014-2016 (Table 20). Four of six sites reached or tied their maximum values in 2015 which is when the mean site richness peaked at 34 species. Most of the current trend sites have been sampled annually since 2011, with the

exception of Welch Island which started in 2012. Since 2011, mean site species richness increased from 20 (StdDev 9) to its peak of 34 (StdDev 10) in 2015, after which it declined by 1 in 2016 and then dropped to 28 (StdDev 12) in 2017 (Table 20). Comparing the sites since 2011, three sites have generally increasing trends in species richness, two sites peaked in the middle years and have decreased substantially in the past one to two years, while the last site has not varied much (Figure 36). Ilwaco varied the least in species richness ranging 15-20 species (StdDev 1.5), and reaching its lowest value in 2017. At the other end of the spectrum, Campbell Slough has varied the most, ranging 15-38 species (StdDev 8.7). Campbell had the biggest change in richness in 2017, dropping to 19 species from its peak of 38 in 2016 (Table 20). The other site that has declined substantially recently is Welch Island where richness dropped to 38 species from a high of 48 in 2014-2015 (Figure 36, StdDev 3.99). The three sites that have generally increasing trends in species richness since 2011 (Figure 36) are Whites Island (ranging 32-49, StdDev 5.44, peaking in 2017), Franz Lake (ranging 17-33, StdDev 5.7, peaking in 2015), and Cunningham Lake (ranging 11-23, StdDev 4.36, peaking in 2015-2016).

Similar to species richness, the average total % cover of vegetation declined in 2017 from the previous two years. The patterns varied widely among the sites, with four sites actually increasing in 2017, and two sites decreasing substantially. In the seven years since most sites have been monitored annually, average % cover peaked in 2016 at 104%, declining to 99% in 2017 (Table 21). However, the three most downstream sites (Ilwaco, Welch, Whites) as well as the most upstream site (Franz Lake) all achieved or tied their maximum cover values (for the period 2011-2017) in 2017. In contrast, the wetland zone 4 sites (Campbell and Cunningham) both saw their % cover decrease by about 40% in 2017 compared to 2016, and that decline was enough to pull the average cover for all six sites down below the 2016 value.

Table 20. Species richness over time at the six trend sites. R=richness (# of species). Native R=native species richness, Non-Native R=non-native species richness.

Site		Year												
		17	**16**	**15**	**14**	**13**	**12**	**11**	**10**	**09**	**08**	**07**	**06**	**05**
Ilwaco Slough	# of plots	40	39	39	40	40	41	40						
	Total Site R	15	17	20	17	18	16	17						
	Native R	13	14	18	13	15	13	14						
	Non-Native R	2	2	2	3	3	3	3						
	Avg R per plot	4.8	4.6	4.1	4.1	4.0	3.9	4.3						
Welch Island	# of plots	41	40	40	40	40	41							
	Total Site R	38	42	48	48	45	39							
	Native R	29	32	36	34	34	30							
	Non-Native R	9	10	12	12	10	9							
	Avg R per plot	10.3	11.4	11.8	11.2	11.2	10.2							
Whites Island	# of plots	44	45	47	43	46	42	42	35	25				
	Total Site R	49	46	46	47	44	32	39	37	34				
	Native R	38	31	32	29	30	21	25	26	24				
	Non-Native R	10	13	13	17	12	11	14	11	9				
	Avg R per plot	7.3	7.8	6.3	7.2	6.5	5.4	6.0	7.4	6.5				
Campbell Slough	# of plots	65	61	62	61	61	61	61	62	61	64	62	62	25
	Total Site R	19	38	36	27	37	22	15	26	23	19	17	13	9
	Native R	12	24	25	16	21	14	11	16	13	11	8	9	7
	Non-Native R	6	13	11	10	15	6	3	9	9	7	9	4	2
	Avg R per plot	3.2	4.5	4.9	3.4	3.8	2.6	2.2	3.2	3.2	3.0	2.4	2.2	2.2
Cunningham Lake	# of plots	69	69	69	36	98	61	59	62	65	63	65	64	30
	Total Site R	19	23	23	15	16	11	13	21	19	13	13	12	15
	Native R	15	17	16	11	12	8	9	14	12	11	10	9	13
	Non-Native R	3	6	6	3	3	3	3	5	7	2	3	3	2
	Avg R per plot	5.7	4.6	4.0	3.8	3.0	2.5	2.1	3.7	3.7	3.2	3.7	2.9	3.3
Franz Lake	# of plots	61	72	79	61	60	64	58		38	40			
	Total Site R	26	31	33	24	21	17	18		18	11			
	Native R	19	23	24	18	15	14	15		15	8			
	Non-Native R	6	7	7	5	3	2	2		2	3			
	Avg R per plot	6.9	5.3	4.5	4.6	5.6	3.9	4.2		4.4	3.6			
Trend Site Annual Summary	# Sites	6	6	6	6	6	6	5	3	4	3	2	2	2
	Tot. System R	93	100	101	92	97	78	66	56	56	27	23	20	16
	Mean Site R (SD)	28 (12)	33 (10)	34 (10)	30 (13)	30 (12)	23 (10)	20 (9)	28 (7)	24 (6)	14 (3)	15 (2)	13 (1)	12 (3)
	Native R (%)	72 (77)	73 (73)	76 (75)	64 (70)	65 (67)	57 (73)	48 (73)	38 (68)	38 (68)	18 (67)	13 (57)	14 (70)	13 (81)
	Non-Native R (%)	20 (22)	25 (25)	23 (23)	26 (28)	29 (30)	19 (24)	17 (26)	16 (29)	17 (30)	8 (30)	10 (43)	6 (30)	3 (19)
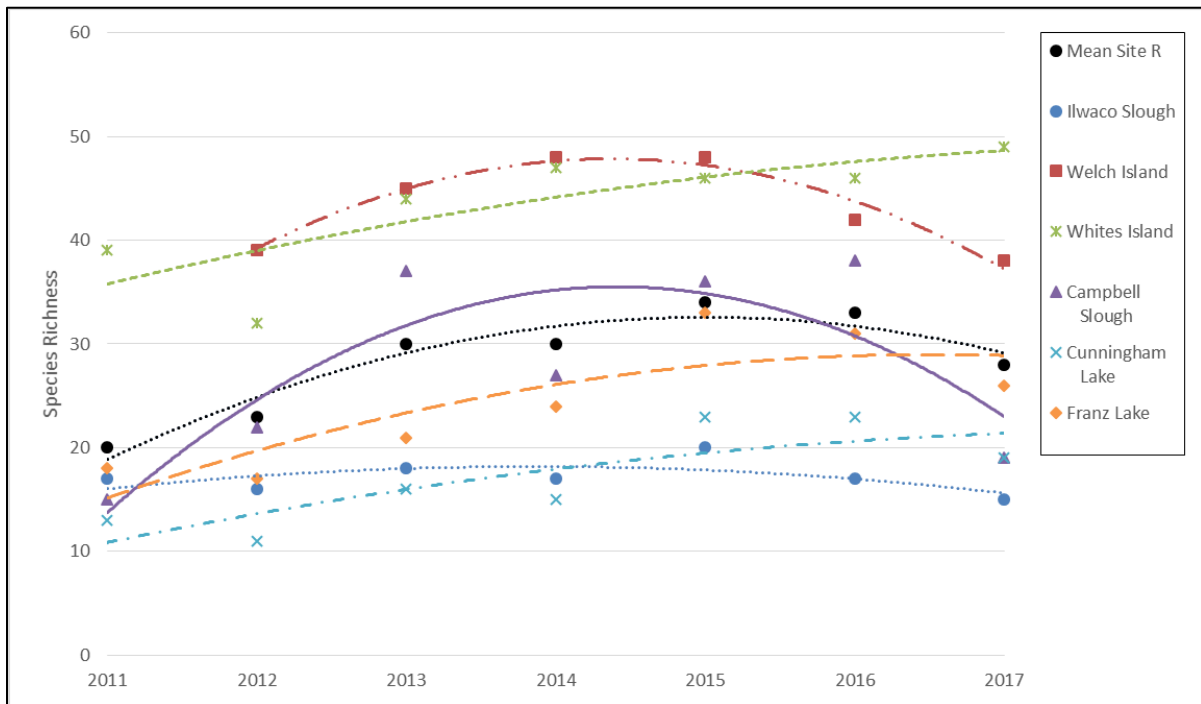
Figure 36. Changes in species richness over time at each trend site. Only the years for which all 6 current trend sites were sampled are shown. Trend lines are second order polynomials.

If we compare the patterns of change among the sites since 2011, which is when most sites were being monitored, we see that three sites (Welch, Whites, and Franz) appear to be on a generally increasing trend for % cover, one site (Ilwaco) declined substantially in 2015 but bounced back in 2017, and two sites (Campbell and Cunningham) peaked in 2015-2016 but then declined substantially in 2017 (Figure 37).
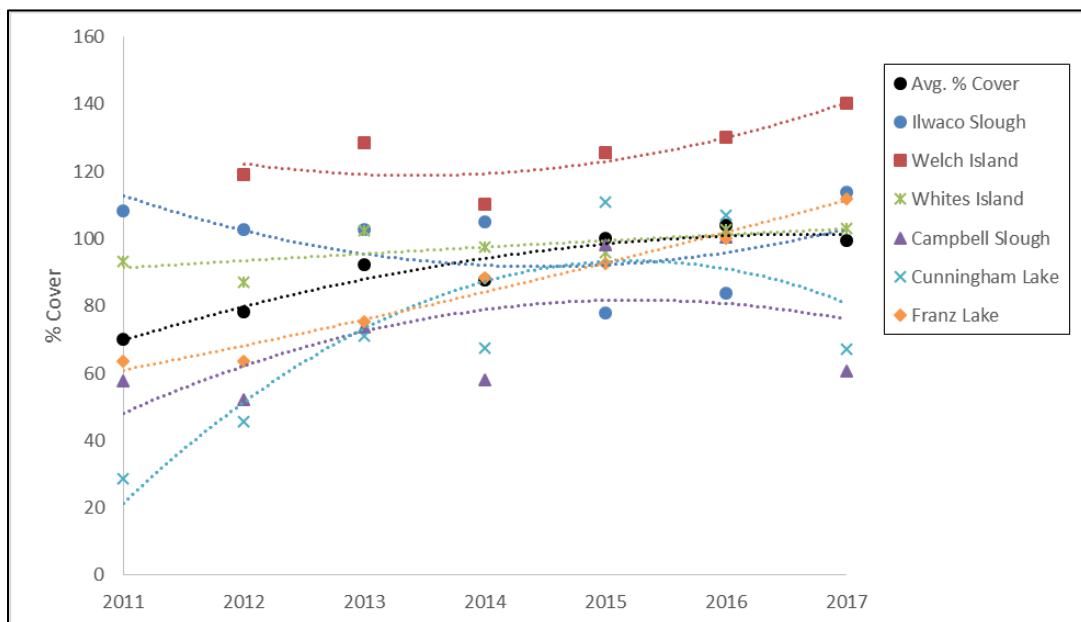
Figure 37. Trends in Total % Cover at each site since 2011, when most sites except Welch Island began being monitored. Trend lines are 2nd order polynomials.

Table 21. Changes in average relative % cover of vegetation over time at the six trend sites. Tot=total cover, Nat=native species relative cover, Non=Non-native species relative cover, OBL=Obligate wetland species relative cover. Numbers in parentheses are standard deviations.

Site		**Year** 17	16	15	14	13	12	11	10	09	08	07	06	05
Ilwaco Slough	Tot	114 (31)	84 (30)	78 (37)	105.2 (40)	103 (27)	103 (20)	108 (19)						
	Nat	75 (22)	77 (26)	87 (36)	91 (28)	84 (28)	84 (18)	90 (15)						
	Non	25 (28)	23 (23)	13 (15)	9 (12)	16 (22)	16 (20)	10 (13)						
	OBL	74 (20)	72 (27)	80 (30)	82 (29)	76 (26)	72 (24)	80 (23)						
Welch Island	Tot	140 (23)	130 (25)	126 (22)	110 (29)	129 (23)	119 (27)							
	Nat	85 (32)	85 (34)	86 (36)	86 (39)	80 (39)	84 (34)							
	Non	15 (26)	15 (26)	14 (25)	15 (22)	20 (31)	16 (32)							
	OBL	78 (35)	75 (30)	74 (34)	76 (38)	75 (36)	80 (32)							
Whites Island	Tot	103 (31)	103 (31)	96 (35)	97 (23)	103 (43)	87 (29)	93 (30)	110 (29)	107 (40)				
	Nat	49 (37)	42 (35)	38 (33)	37 (25)	31 (26)	35 (18)	26 (17)	36 (25)	41 (27)				
	Non	51 (34)	57 (40)	61 (40)	63 (32)	68 (46)	65 (34)	74 (33)	64 (36)	59 (38)				
	OBL	44 (37)	41 (33)	38 (32)	36 (23)	33 (25)	41 (21)	32 (20)	40 (27)	37 (25)				
Campbell Slough	Tot	61 (30)	100 (22)	98 (34)	58 (44)	74 (39)	52 (46)	58 (35)	94 (23)	100 (27)	61 (40)	44 (23)	61 (31)	89 (35)
	Nat	59 (29)	59 (42)	67 (55)	47 (30)	50 (25)	64 (39)	38 (27)	53 (43)	59 (50)	47 (30)	54 (25)	50 (31)	59 (50)
	Non	41 (35)	41 (46)	33 (38)	53 (41)	50 (45)	34 (24)	61 (39)	46 (46)	40 (46)	51 (38)	46 (25)	51 (39)	42 (45)
	OBL	56 (28)	57 (43)	64 (54)	43 (26)	46 (25)	54 (28)	32 (22)	52 (44)	56 (51)	39 (23)	54 (25)	49 (31)	55 (50)
Cunningham Lake	Tot	67 (39)	107 (31)	111 (38)	68 (42)	71 (43)	46 (29)	29 (21)	100 (40)	101 (39)	68 (35)	88 (33)	50 (22)	97 (36)
	Nat	63 (32)	55 (39)	53 (42)	63 (37)	48 (36)	50 (27)	45 (22)	41 (39)	61 (46)	50 (27)	57 (34)	65 (26)	56 (52)
	Non	36 (24)	45 (40)	47 (42)	36 (32)	51 (34)	50 (23)	54 (15)	59 (44)	40 (42)	50 (37)	43 (37)	34 (22)	44 (45)
	OBL	45 (26)	47 (39)	47 (44)	35 (29)	40 (30)	29 (14)	25 (11)	33 (38)	54 (46)	39 (26)	53 (32)	57 (24)	54 (52)
Franz Lake	Tot	112 (42)	100 (56)	93 (58)	89 (37)	75 (37)	64 (34)	63 (32)		93 (45)	79 (29)			
	Nat	85 (37)	83 (49)	85 (50)	84 (36)	77 (33)	83 (36)	58 (25)		64 (46)	59 (31)			
	Non	14 (23)	15 (24)	14 (22)	11 (22)	19 (22)	9 (8)	42 (28)		36 (36)	41 (27)			
	OBL	54 (33)	51 (40)	49 (39)	58 (36)	46 (28)	71 (35)	42 (19)		54 (43)	25 (24)			
Overall Average Total Cover		99 (28)	104 (14)	100 (15)	88 (19)	92 (21)	78 (27)	70 (28)	101 (7)	100 (5)	69 (8)	66 (22)	55 (6)	93 (4)
Overall Native Relative % Cover		69 (13)	67 (16)	69 (19)	68 (20)	62 (20)	66 (19)	51 (22)	43 (7)	56 (9)	52 (5)	55 (2)	57 (8)	57 (1)
Overall Non-native Relative % Cover		30 (14)	33 (16)	30 (19)	31 (21)	37 (20)	32 (20)	48 (22)	56 (7)	35 (20)	47 (4)	44 (2)	42 (8)	43 (1)
Overall OBL Relative % Cover		58 (13)	57 (13)	59 (15)	55 (19)	53 (17)	58 (18)	42 (19)	41 (8)	50 (8)	34 (7)	53 (1)	53 (4)	55 (1)

Across the 2011-2017 time period, the three most downstream sites (Ilwaco Slough, Welch Island, and Whites Island) had an overall higher average cover and lower variability (107%, SD 9.1, Figure 38) than the three most upstream sites (Campbell Slough, Cunningham Lake, and Franz Lake; 76%, SD 21.1). However Franz Lake has the strongest increasing trend in cover, and for the past two years has been similar in cover to the three most downstream sites (Figure 37).

The substantially higher variability in the cover at Cunningham Lake and Campbell Slough is also evident in Figure 39 which shows the change in cover at each site throughout the entire period of monitoring.
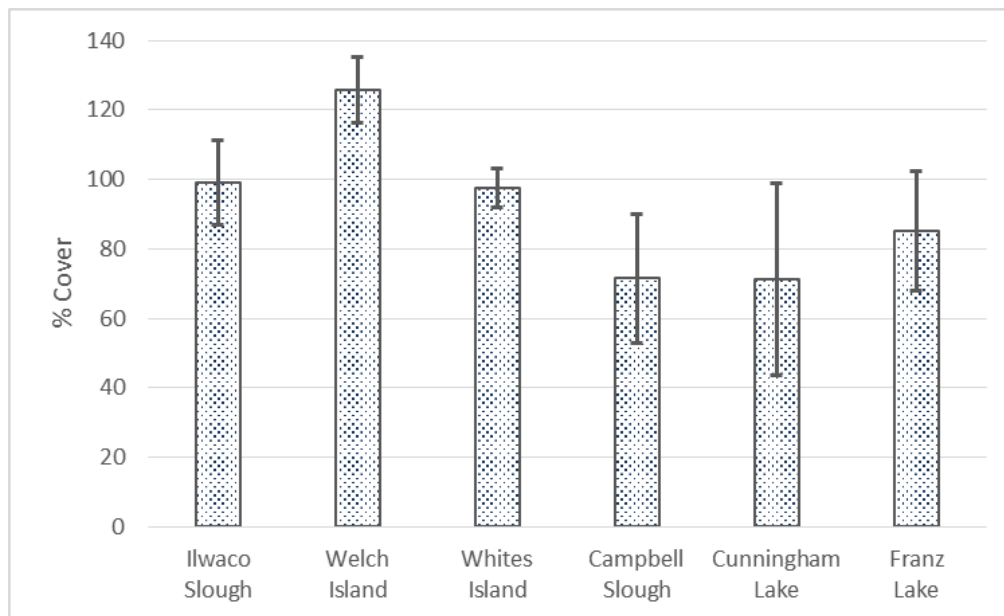
Figure 38. Average % Cover of vegetation at each trend site, 2011-2017, which is the period during which all six current trend sites were being monitored. Error bars are standard deviations.

Native vs. Non-native species
Across all sites, non-native species richness declined substantially more than native species did in 2017. Across all six trend sites, the total number of non-native species declined from 25 in 2016 to 20 in 2017 (Table 20), while native species only declined from 73 to 72. All sites saw a decline in non-native species richness except Ilwaco Slough which held steady at 2 species. The greatest proportional declines in non-native species richness were seen at Campbell Slough and Cunningham Lake which both saw declines of about 50%. Campbell Slough dropped from 13 non-native species in 2016 to six in 2017, while Cunningham Lake declined from six non-native species in 2016 to three in 2017 (Table 20).

Native species increased slightly in relative % cover from 2016 to 2017, while non-native species decreased slightly. The patterns at each site for % cover of non-native species were not the same as with species richness. Overall, the average relative percent cover of non-native species declined from 33 to 30% between 2016 and 2017 (Table 21). However, at Ilwaco, the relative % cover of non-natives increased from 23 to 25%. The proportion of non-native species has been increasing steadily over time at Ilwaco (Figure 40). In terms of the relative % cover of non-natives, Welch Island (15%) and Campbell Slough (41%) held steady from 2016 to 2017 (Figure 40). Sites where there was a decline in the proportion of non-native vegetation (Table 21, Figure 40) included Whites Island (6% decline), Cunningham Lake (9% decline), and Franz Lake (1% decline).

Overall, native species dominate the vegetation at Ilwaco Slough, making up 75% of the total plant cover (Table 21, Figure 40), as well as at Welch Island (85%) and Franz Lake (85%). At Campbell Slough

(59%) and Cunningham Lake (63%), natives predominate though not as strongly. And at Whites Island, natives (49%) pull in slightly behind the non-natives. However at Whites Island there has been a steady decline in dominance of non-native species cover since 2011, reaching its all-time low of 51% of total cover in 2017.
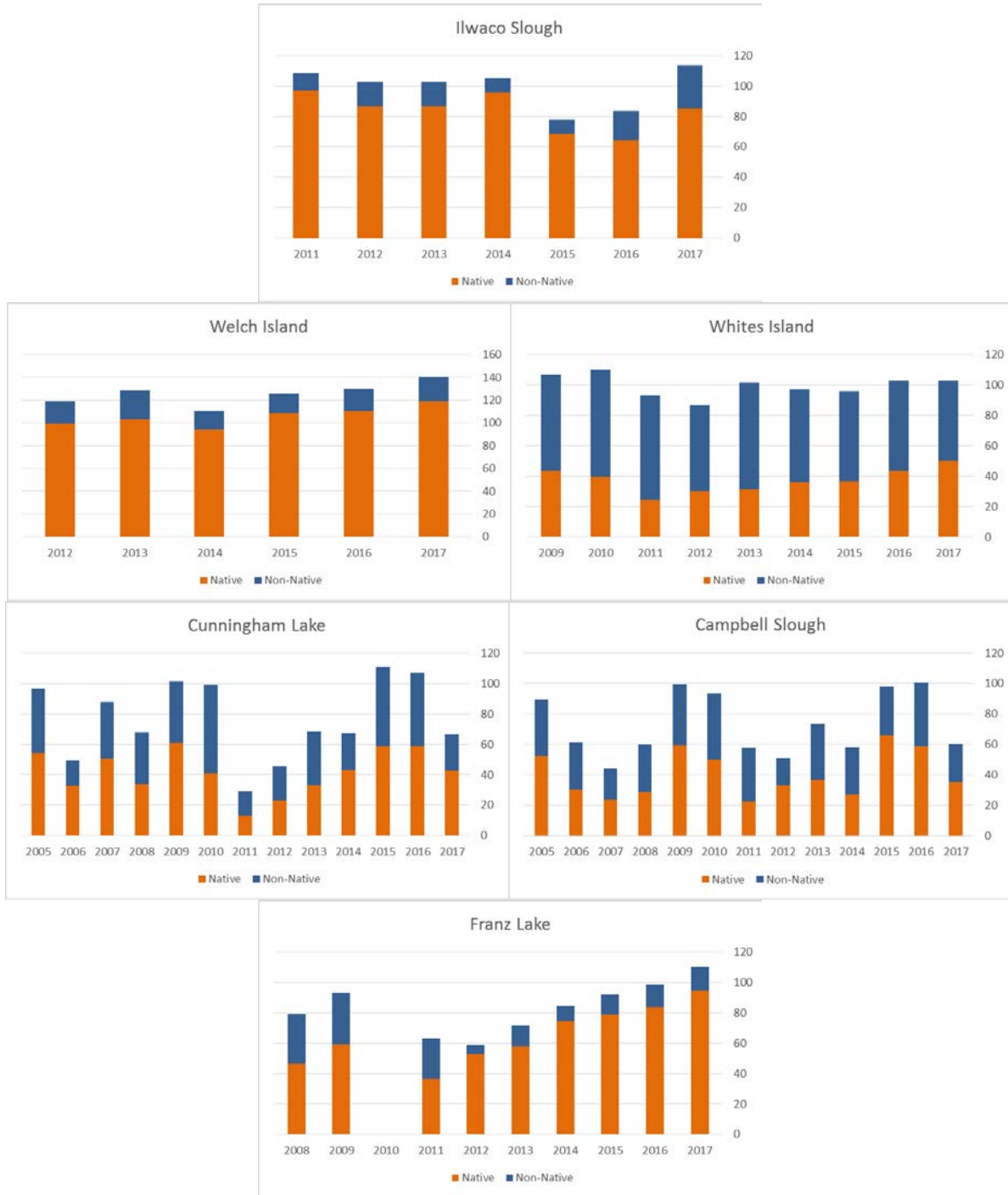
Figure 39. Changes in % Cover of vegetation at the trend sites over time. Native species are shown in orange, non-native in blue. Sites are in order from the most downstream to the most upstream site.
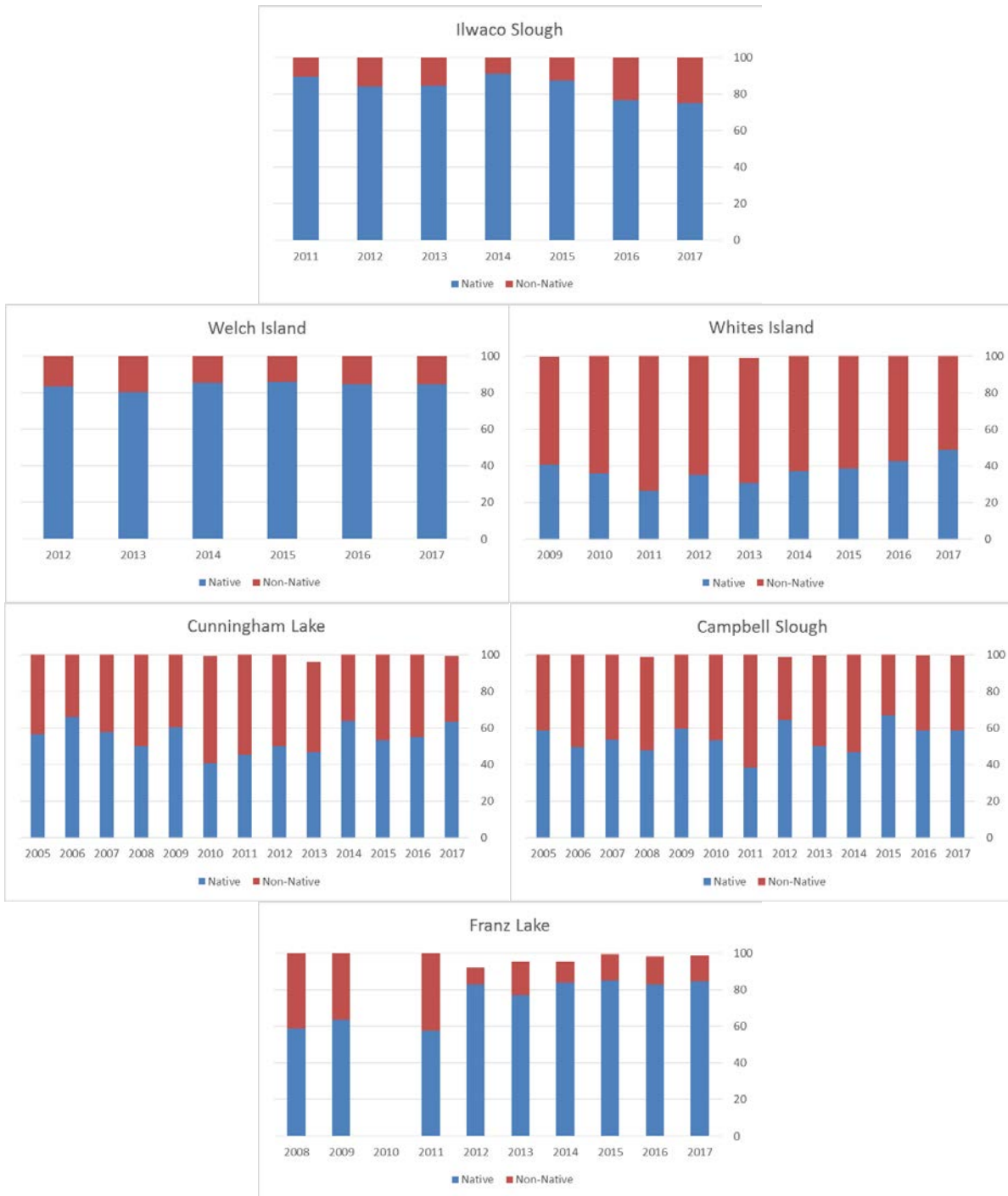
Figure 40. Changes in the relative % cover of native and non-native species at the trend sites over time. Blue is the relative cover of native species. Red is the relative cover of non-native species.

Since 2011, the proportion of total plant cover that is non-native has been declining at Whites Island, Campbell Slough, and Cunningham Lake (Figure 41). All three sites started out with more than 50% of cover non-native, though Whites Island has declined the most from 74% non-native to 51%. Variability is highest at Campbell Slough (SD 9.66) compared to the other two sites (SD 6.7). The proportion of non-native cover is increasing at Ilwaco, having more than doubled from 10% in 2011 to 25% in 2017 (Figure 41). Welch Island and Franz Lake have changed little since 2012. Franz lake began in 2011 with 42% non-native cover, but dropped precipitously in 2012 to 9% and has been bouncing between 11-19% since then.
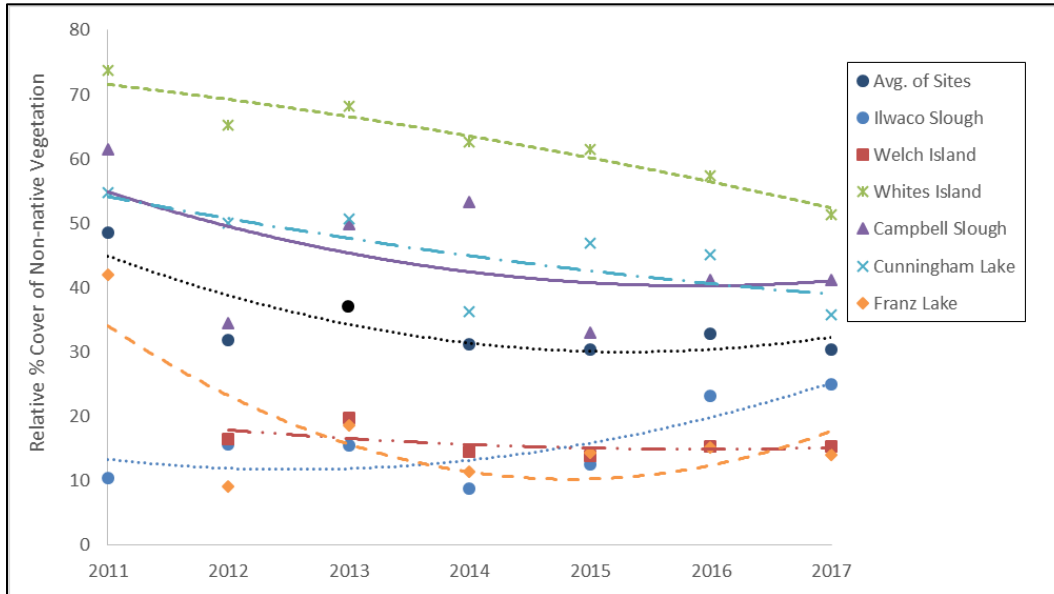
Figure 41. Comparison of trends in relative % cover of non-native plants across the six sites since 2011, the time period during which most sites were being monitored. Trend lines are 2nd order polynomials.

3.3.4 Channel Morphology and Inundation

Channel cross-section and hydrology data collected in 2017 are still under analysis and unavailable at the time this report was written. The text below is adapted from the 2016 report.

Channel morphology at the trends sites exhibited low inter-annual variability in years prior to 2016; therefore, only the channel mouth cross section was surveyed in 2016. Channel measurements from previous years are presented with the newly calculated inundation frequency results from 2016 in Table 24. The tidal channels measured at the sites were generally small, with most cross-sectional areas less than 10 m^2 (see Appendix A for locations of the measured channels). Five of the tidal channels surveyed were primary channels feeding directly into the Columbia River, while the channels at the Welch and Whites Island sites were secondary channels that feed into a larger tidal channel. The channels varied in width from 1.3 m to 50.1 m; most becoming narrower with increasing elevation, with the exception of the Ilwaco Slough and Whites Island channels, which were slightly wider at the middle than at the mouth. Channel depth ranged from 0.3 m to 2.1 m, with most channels between 0.9 m and 1.2 m in depth. The thalweg elevation of the channels was generally between 0.0 and 1.0 m and the channel bank between 1.0 and 2.0 m, relative to CRD.

Table 22. 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.

Site (year)	Cross Section	Thalweg Elevation (m, CRD)	Bank Elevation (m, CRD)	Channel Depth (m)	Cross Section Area (m²)	Channel Width (m)	Width:Depth Ratio
Ilwaco Slough (11)	1*	0.87	1.56	0.69	3.3	6.2	9.0
	2	0.70	1.86	1.16	8.94	9.30	8.04
	3	0.90	2.12	1.22	9.73	10.10	8.27
	4	1.01	2.00	0.99	4.33	5.20	5.23
	5	1.17	2.26	1.09	1.58	2.70	2.48
Welch Island (12)	1*	0.30	1.51	1.21	13.0	20.4	16.9
	2	0.36	1.65	1.29	8.75	9.20	7.13
	3	0.71	1.80	1.09	3.96	5.09	4.67
	4	0.78	1.74	0.96	2.07	3.30	3.44
	5	1.31	1.62	0.31	0.42	1.32	4.27
Whites Island (11)	1*	0.42	1.12	0.70	12.1	34.6	49.4
	2	0.34	1.41	1.07	10.8	20.5	19.1
	3	0.61	1.53	0.92	11.1	36.2	39.5
	4	0.92	1.93	1.00	34.0	50.1	50.0
	5	0.44	1.45	1.01	1.90	2.83	2.80
Cunningham Lake (15)	1	0.82	1.26	0.44	5.5	18.3	41.6
Campbell Slough (15)	1	0.80	1.47	0.67	11.7	23.0	34.3
Franz Lake (12)	0*	0.34	2.23	1.89	21.3	23.2	12.2
	3	0.40	1.39	0.99	4.20	14.3	14.4
	4	0.85	1.45	0.60	6.20	13.2	22.0

Inundation frequency of the channel thalweg and the bank are generally less in the lower estuary than in the upper estuary due to reduced riverine flooding effects. Tidal inundation results in bank frequency inundation from 3 to 63 percent in the lower estuary except in high water years (2011 and 2012) when slightly greater frequencies were observed. Additionally, most of the lower estuary sites have decreasing inundation frequencies higher up the channel gradient. Bank inundation frequencies increase moving up the estuary except for low river flow years (2015) when they are at times less than 10 percent.

Site specific differences in channel morphology and hydrology affect the inundation frequencies. At Ilwaco Slough the channel is somewhat perched and as a result, frequencies are lowest and generally less than 35%. Frequencies were highest in 2011–12, and 2014, while the lowest frequencies occurred in 2013 and 2016, not 2015 the lowest river flow year. Variability is low between years on the same cross section at Ilwaco Slough, less than 10 percent. Differences between the Cunningham Lake and the Campbell Slough channel morphology result in very different inundation frequencies. Campbell Slough has a higher bank but the thalweg elevations are very similar, however inundation frequency differs due to the weir at mouth in lower water years (2013, 2015). The channel hydrology at Franz Lake is affected by beaver dams and the channel below the dams (e.g., cross sections 0 and 3) was probably not actually inundated 98-100 percent of the time. The beaver dam appears to wash out every year then is gradually built up from an elevation of approximately 0.8 m to 1.5 m CRD (see Appendix C for annual hydrographs of Franz Lake).

Table 23. Inundation frequency for channel thalweg (+50 cm) and channel bank (+10 cm) during peak juvenile chinook migration period (March – July).

		Inundation Frequency																		
		Percent of time water level greater than thalweg +50cm										Percent of time water level greater than bank +10cm								
Site	Cross Section	2008	2009	2010	2011	2012	2013	2014	2015	2016		2008	2009	2010	2011	2012	2013	2014	2015	2016
Ilwaco Slough	1	nd	nd	nd	51	50	41	47	45	45		nd	nd	nd	38	34	24	32	33	32
	2	nd	nd	nd	59	59	51	58	54	51		nd	nd	nd	27	27	17	25	22	18
	3	nd	nd	nd	51	51	43	49	46	44		nd	nd	nd	14	14	8	13	10	8
	4	nd	nd	nd	46	46	38	44	42	39		nd	nd	nd	20	20	12	18	15	12
	5	nd	nd	nd	39	39	31	38	35	32		nd	nd	nd	9	9	5	9	6	5
Welch Island	1	nd	nd	nd	nd	nd	66	75	83	73		nd	nd	nd	nd	nd	24	32	38	40
	2	nd	nd	nd	nd	nd	59	67	79	71		nd	nd	nd	nd	nd	21	29	43	34
	3	nd	nd	nd	nd	nd	45	51	63	55		nd	nd	nd	nd	nd	15	23	35	27
	4	nd	nd	nd	nd	nd	41	49	60	53		nd	nd	nd	nd	nd	18	25	38	30
	5	nd	nd	nd	nd	nd	19	26	39	31		nd	nd	nd	nd	nd	22	31	44	35
Whites Island	1	nd	nd	79	94	94	nd	88	71	nd		nd	nd	46	79	76	nd	50	58	nd
	2	nd	nd	71	92	95	nd	88	72	nd		nd	nd	28	63	61	nd	52	40	nd
	3	nd	nd	65	83	84	nd	75	58	nd		nd	nd	34	56	54	nd	46	35	nd
	4	nd	nd	47	68	66	nd	56	44	nd		nd	nd	21	32	33	nd	26	16	nd
	5	nd	nd	64	89	91	nd	83	67	nd		nd	nd	42	60	58	nd	50	38	nd
Cunningham Lake	1	nd	nd	67	98	98	73	96	51	nd		nd	nd	64	98	98	80	97	59	nd
Campbell Slough	1	nd	nd	67	97	98	86	98	74	85		nd	nd	23	96	96	68	96	6	64
Franz Lake	0	nd	100	nd	nd	100	100	100	100	nd		nd	51	nd	nd	91	52	82	9	nd
	3	nd	100	nd	nd	100	100	100	100	nd		nd	80	nd	nd	96	75	99	53	nd
	4	nd	93	nd	nd	100	76	100	84	nd		nd	71	nd	nd	95	74	98	49	nd

3.4 Food Web

3.4.1 Primary Production

3.4.1.1 *Emergent Wetland Vegetation*

Sampling was conducted within vegetation strata at each of the long term monitoring sites. Results presented here are divided into those for Summer 2017 and those from a multi-year analysis conducted on all data collected 2011 - 2017. Species composition was the same in all analyses and is presented first.

Species Composition

At each long-term monitoring site, aboveground biomass was sampled within vegetation strata to reduce variability associated with sampling across strata. The dominant species for the strata are identified in Table Table 24. The strata generally covered an entire site as identified in the vegetation community maps (see Appendix A). In 2015-2017 we divided each sample by the dominant species present within the

sample to determine the proportion comprised of certain species. The samples from single species strata were generally comprised of greater than 90 percent of that species (Table 24). When the stratum was a mix of two dominant species the results were mixed. The strata that were a high or low marsh mix were generally comprised of lower proportions of a single species, with the dominant species comprising as little as 44 percent of the samples (Table 24).

Table 24. Average proportion of dominant species within the samples of each stratum and average proportion of samples that are live and dead in summer 2017.

| Site | Stratum Code | Stratum | Species Present in Stratum Samples | | | | | | | Summer | |
			CALY	ELPA	ELPA/SALA	LEOR	PHAR	SALA	Other	Dead	Live
WEI	HM	High marsh mix	0.44	0.10		0.07	0.07	0.09	0.22	0.13	0.87
WHI	CALY	*C. lyngbyei*	0.91				0.03		0.06	0.04	0.96
WHI	HM	High marsh mix	1.00							0.00	1.00
WHI	PHAR	*P. arundinacea*					0.83		0.17	0.05	0.95
WHI	PHAR/ HM	*P. arundinacea/* High marsh mix	0.10				0.57		0.33	0.06	0.94
WHI	SALA	*S.latifolia/* Low marsh mix					0.09	0.58	0.33	0.07	0.93
CL	ELPA/ SALA	*E. palustris/ S.latifolia*		0.98			0.01		0.01	0.06	0.94
CL	PHAR	*P.arundinacea*		0.02			0.76	0.14	0.02	0.59	0.41
CL	SALA	*S. latifolia*						1.00		0.05	0.95
CS	SALA	*S. latifolia*						1.00		0.02	0.98

To more accurately represent the annual contribution of organic matter from marsh plants, we collect the live and dead standing stock in the summer (late July) and the dead, previous year's standing stock in the winter (February). We divided the sample into live and dead in both seasons, which allowed us to calculate the proportion of each sample that is live and dead. In the summer, most strata have less than 10 percent standing dead; two sites had strata with greater than 10 percent standing dead (Table 24).

Summer 2017 Results
Annual biomass sampling results for summer 2017 are summarized in Figure 42 and Table 25. Similar to summer 2016 results, the highest amount of summer 2017 biomass and the greatest variability was in the samples from the Whites Island *P. arundinacea*/high marsh stratum. The second highest sample biomass was also found at Whites Island from the *C. lyngbei* stratum.

110
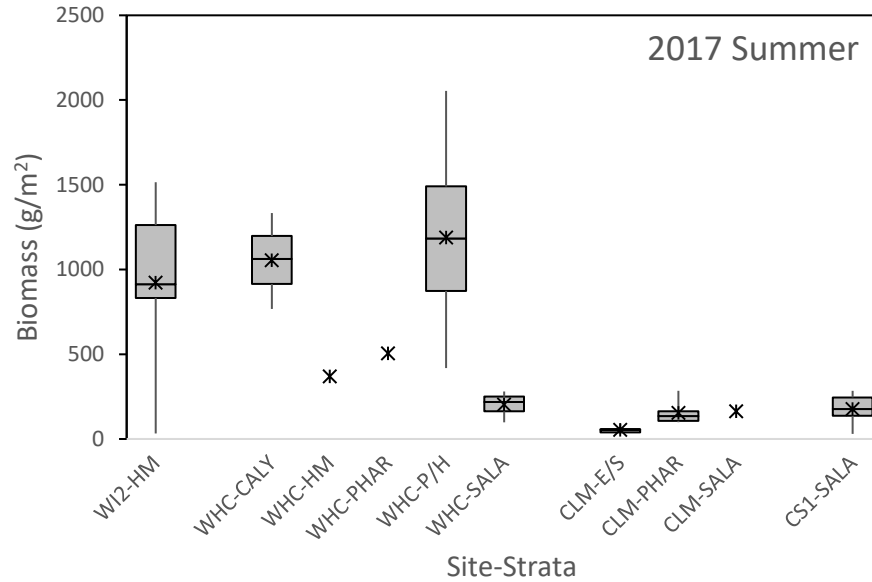
Figure 42. Aboveground biomass (dry weight g/m^2) for all sites from summer 2017 sampling.

The contribution of organic matter is typically calculated as the summer standing stock (live and dead) minus the following winter's standing stock (dead), however the winter 2018 biomass sample processing has not yet been completed at the time of this report, therefore the organic matter contribution is predicted based on the average percentage of summer stock contributing to organic matter from previous years. The previous years' organic matter contributions were averaged by stratum, ranging from 45 to 93 percent of the summer stocks (Table 25). Applying these stratum-specific proportions to the summer 2017 biomass values resulted in predicted organic matter contributions ranging from 50.0 g/m^2/yr from the Cunningham Lake *E. palustris/S. latifolia* stratum, to 843.4 g/m^2/yr from the Whites Island *C. lyngbei* strata (Table 25).

Table 25. Aboveground biomass (dry weight g/m2) for all sites from summer 2017 sampling. Contribution of organic matter is predicted using previous years' average contribution by stratum. Sites are ordered starting at the mouth and strata are ordered by elevation lowest to highest.

Site	Vegetation Strata	n	Summer 2017 (live + dead standing stock) Average Dry wt (g/m^2)	StdDev Dry wt (g/m^2)	Average Proportion Contributed by Stratum (2016-17)	Predicted Contribution Dry Wt (2017-18) (g/m^2/year)
WEI	HM	14	922.5	455.6	0.75	691.9
WHI	SALA	6	203.8	68.4	0.93	189.5
WHI	CALY	3	1054.3	283.0	0.80	843.4
WHI	PHAR	1	504.1	na	0.72	363.0
WHI	PHAR/HM	9	1188.3	568.3	0.45	534.7
WHI	HM	1	367.5	na	0.75	275.6
CL	SALA	1	161.9	na	0.93	150.6
CL	ELPA/SALA	5	54.9	21.9	0.91	50.0
CL	PHAR	6	154.2	69.3	0.72	111.0
CS	SALA	6	176.9	92.9	0.93	164.5

The live fraction was the largest of all summer 2017 biomass components (live, dead, and detritus) at all sites except at Cunningham Lake, where detritus formed the largest component (Figure 43). The dead and detrital components were minimal at Campbell Slough. With these three components averaged within vegetation strata, all high marsh strata contained more live biomass than in the low marsh strata. The *C. lyngbei* strata contained the most live biomass at 1037 ± 379 g dry weight/m^2. The *P. arundinacea* strata contained the most detritus at 347 ± 350 g dry weight/m^2. Live, dead, and detrital biomass components for all sites are included in Table 26.
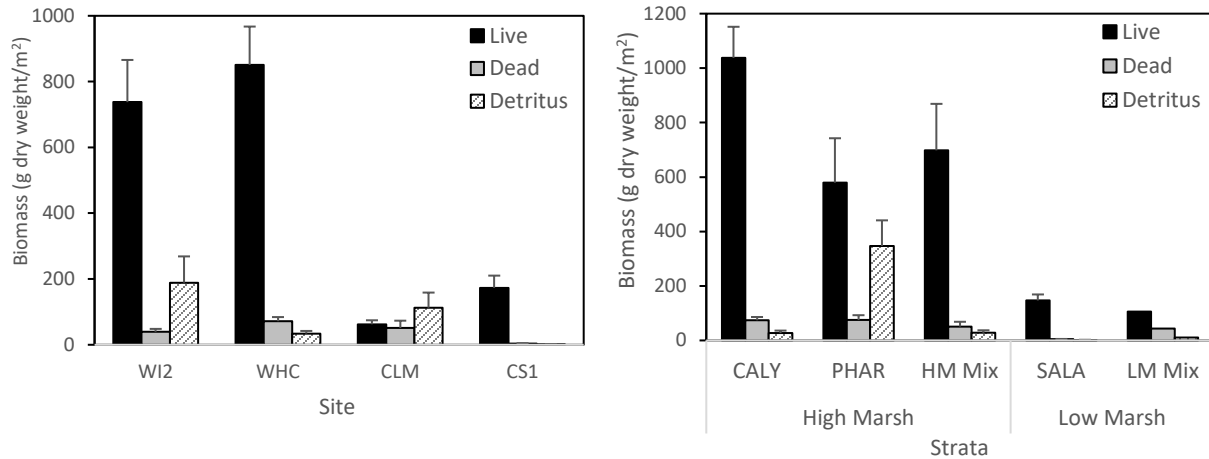
Figure 43. Summer 2017 biomass (dry weight g/m^2) separated into live, dead, and detrital components, shown by site (left) and by strata (right). Error bars represent standard error.

Table 26. Live, dead, and detrital components of biomass (dry weight g/m2) for all sites from summer 2017 sampling. Sites are ordered starting at the mouth and strata are ordered by elevation lowest to highest.

Site	Vegetation Strata	n	Live Average Dry wt (g/m^2)	Live StdDev Dry wt (g/m^2)	Dead Average Dry wt (g/m^2)	Dead StdDev Dry wt (g/m^2)	Detritus Average Dry wt (g/m^2)	Detritus StdDev Dry wt (g/m^2)
WEI	HM	14	850.7	436.2	71.8	46.5	34.1	29.6
WHI	SALA	6	192.0	76.3	11.8	16.8	4.8	4.8
WHI	CALY	3	1004.8	256.4	49.5	27.8	8.7	8.8
WHI	PHAR	1	480.4	na	23.7	na	115.8	na
WHI	PHAR/HM	9	1125.0	542.2	63.3	42.5	428.9	442.1
WHI	HM	1	367.5	na	0	na	0	na
CL	SALA	1	153.0	na	8.9	na	0.6	na
CL	ELPA/SALA	5	51.8	21.0	3.1	2.2	2.5	2.5
CL	PHAR	6	56.1	41.5	98.1	87.4	223.3	162.2
CS	SALA	6	173.1	91.1	3.8	2.4	0.7	1.7

Multi-Year Analysis

Above ground biomass data from 2011 – 2017 were analyzed to determine if differences exist in summer biomass (production) and annual organic matter contribution (hereafter termed contribution) between 1) broad vegetation strata (high marsh [HM] vs. low marsh [LM]), 2) years, and 3) location in the river.

More specific vegetation strata were also analyzed to determine if there were differences in production and contribution between the strata at individual sites and more generally between the strata estuary wide.

Production – Summer Biomass
Data for individual sites and years are summarized in Appendix E and statistical analysis results of the summer biomass data collected at the trend sites between 2011 and 2017 are summarized here. The results from the general linear model (GLM) indicated that the covariate rkm (p<0.001) and the main effect of broad strata (High Marsh and Low Marsh; p<0.001) were significantly different (Figure 44). High marsh summer biomass is consistently higher than low marsh biomass (Figure 45). Regression analysis of the High Marsh and Low Marsh summer biomass by rkm indicate that the fit for this relationship is very low (R^2 of 0.36 and 0.05, respectively), however, biomass in the High Marsh strata is consistently greater in the lower estuary and more variable in the upper estuary (Figure 46).
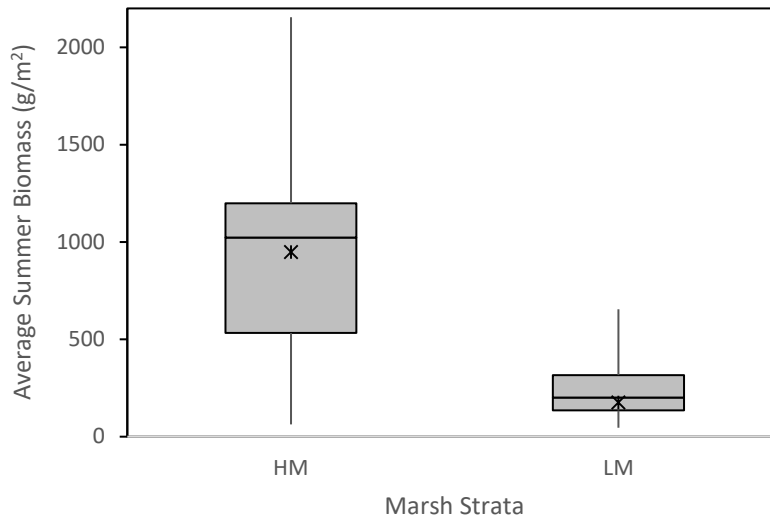
Figure 44. Average summer biomass (g dry weight/m^2) from the high marsh (HM) and low marsh (LM) strata.
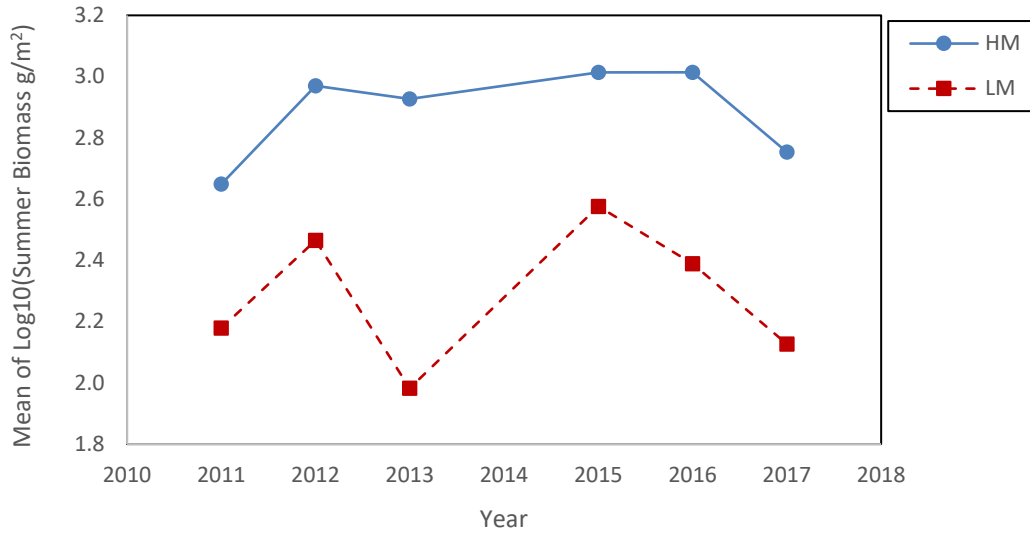
Figure 45. Average annual low marsh and high marsh summer biomass (g dry weight /m^2) by year. Results are transformed by Log10 for statistical analysis.
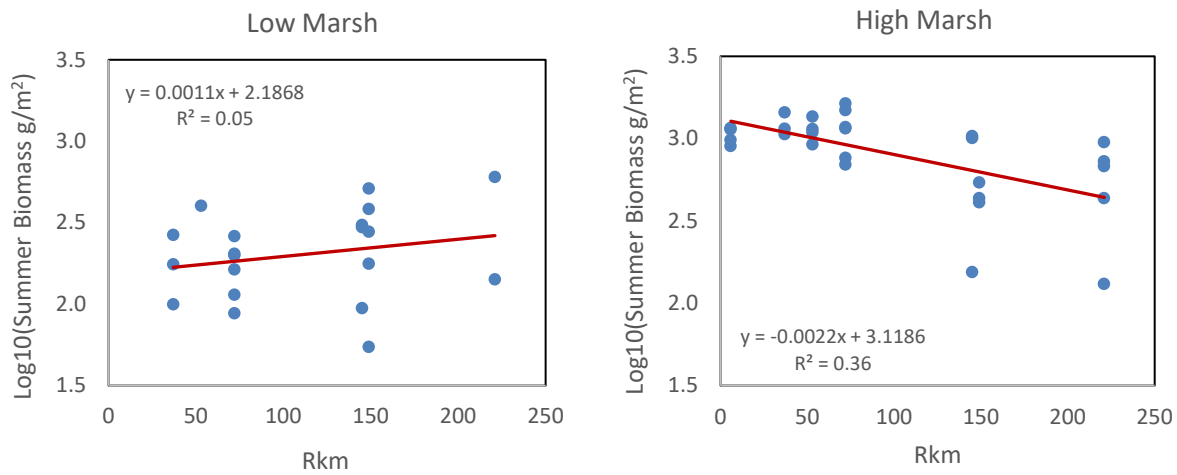
Figure 46. Average annual summer biomass (g dry weight/m^2) compared to rkm for low marsh (left) and high marsh (right) strata. Results are transformed by Log10 for statistical analysis.

A one-way ANOVA indicated no significant difference in summer biomass between the Low Marsh site specific strata (Figure 47), Low Marsh averaged strata (Figure 48), and High Marsh averaged strata (Figure 48). Pairwise comparisons (ANOVA, Tukey) for the High Marsh site specific strata indicated that the summer biomass from the PHAR/HM stratum at Franz Lake was significantly less than the PHAR/HM stratum at Whites Island (p=0.03, Figure 47). No other High Marsh site specific strata were significantly different. Biomass production was highest and most consistent in the strata dominated by *C. lyngbyei*, averaging 1193 ±277 g dry weight/m^2, compared to the strata dominated by *P. arundinacea* which averaged 584 ± 324 g dry weight/m^2.

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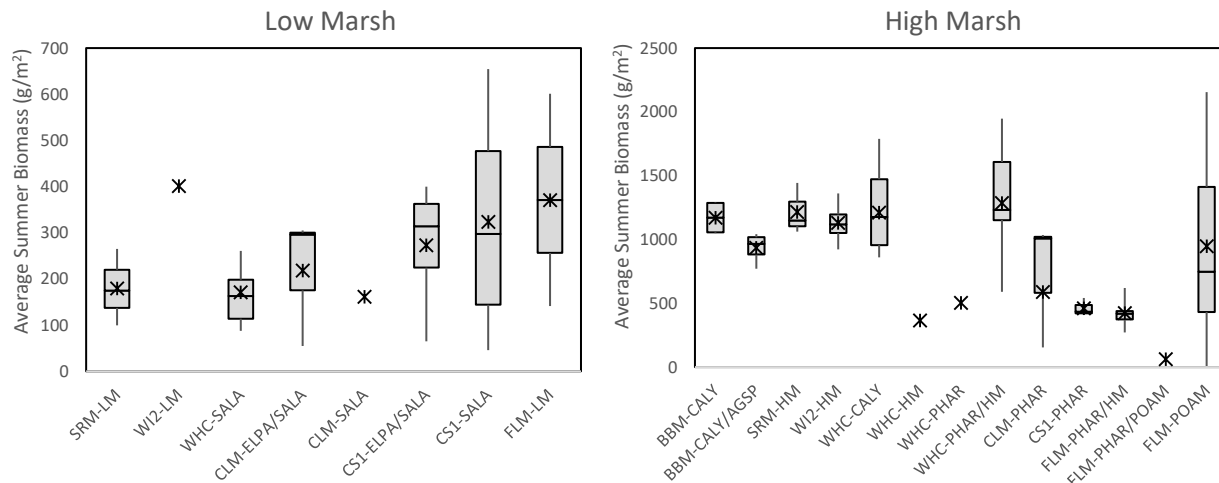
Figure 47. Average summer biomass (g dry weight/m^2) from site specific low marsh (left) and high marsh (right) strata for all years. See Table 246 for definition of site and strata codes. Note different scales on the two plots.
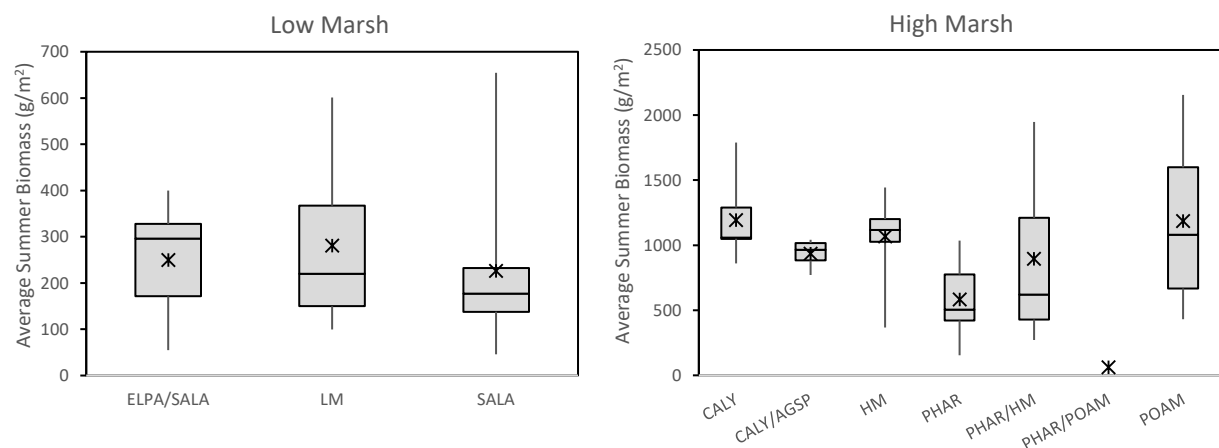
Figure 48. Average summer biomass (g dry weight/m^2) by low marsh (left) and high marsh (right) vegetation strata for all sites and all years. See Table 24 for definition of strata codes. Note different scales on the two plots.

Annual Organic Matter Contribution

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. Statistical results for the analysis of the annual contribution are summarized below and data for individual sites and years are summarized in Appendix E. This analysis covers the period from 2011-2012 to 2016-2017, as the 2017-2018 organic matter contribution cannot be calculated yet.

A GLM indicated that there was a significant difference between the contribution of the low marsh strata and the high marsh strata when averaged across all sites and all years (ANOVA, p=0.03; Figure 49). The

contribution from the high marsh is more variable, with higher contribution levels occurring in some years; however, the difference is not statistically significant (Figure 50). Regression analysis of the LM and HM summer biomass by rkm indicated that the slopes were significant (p=0.040 and p=0.000, respectively), however the fit was low (R^2=0.26 and R^2=0.33, respectively; Figure 51). Contribution from the LM stratum is variable between years; however, it is generally higher from sites at rkm 150 than other sites. Contribution from the HM stratum is consistently greater in the lower estuary and more variable in the upper estuary. The organic matter contribution was highest and most consistent in the strata dominated by *C. lyngbyei*, averaging 882 ± 277 g dry weight/m^2 compared to the strata dominated by *P. arundinacea* which averaged 425 ± 381 g dry weight/m^2.
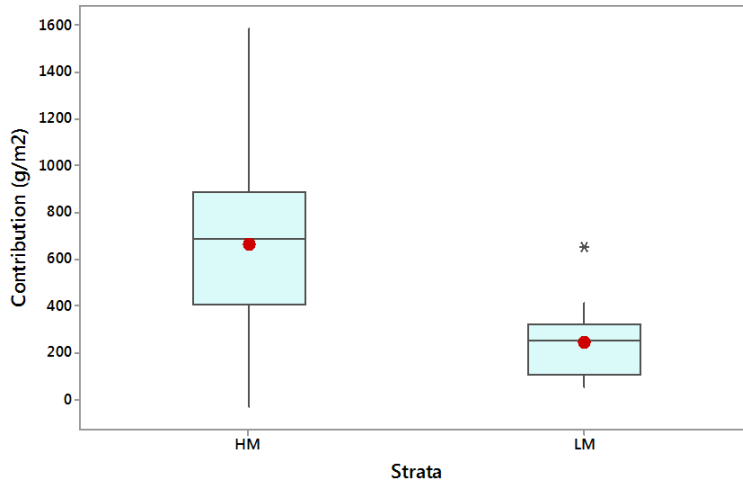
Figure 49. Average annual organic matter contribution (g dry weight/m^2) from the low marsh (LM) and high marsh (HM) strata, from 2011-12 to 2016-17.
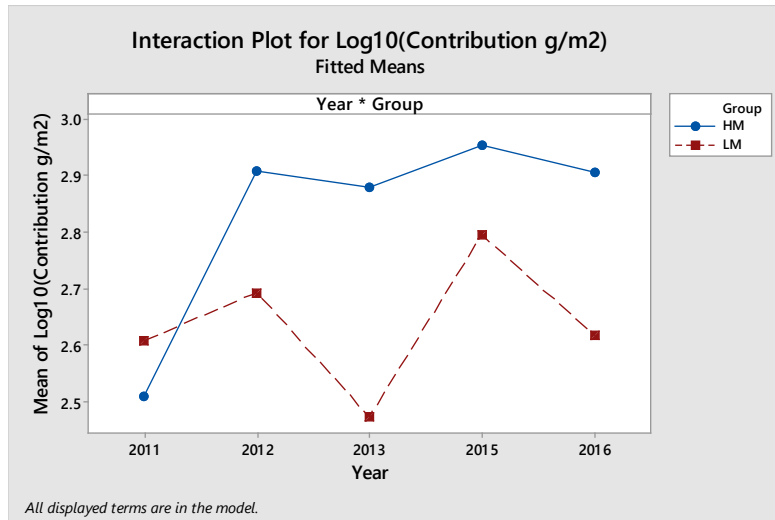
Figure 50. Average annual low marsh and high marsh organic matter contribution (g dry weight /m^2) by year, from 2011-12 to 2016-17. Results are transformed by Log10 for statistical analysis.
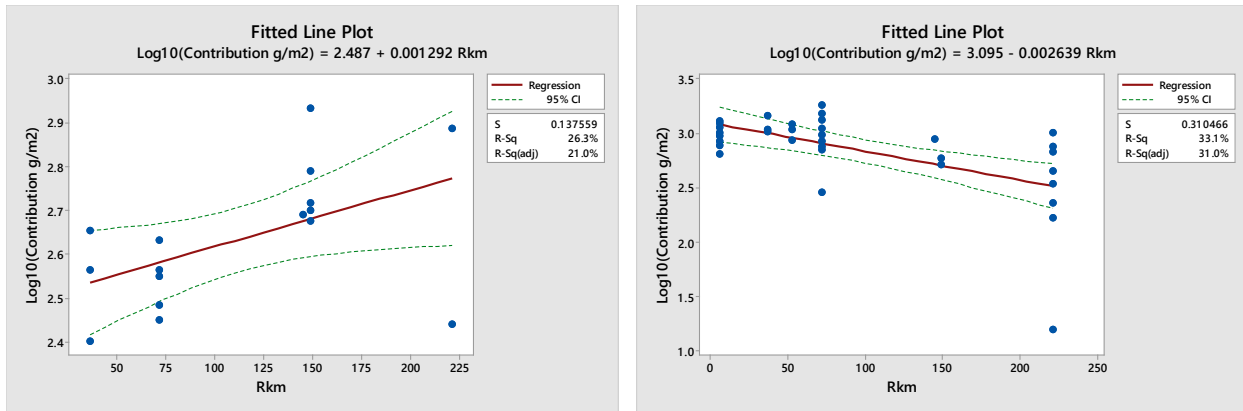
Figure 51. Average annual organic matter contribution (g dry weight/m²) compared to rkm for low marsh (left) and high marsh (right) strata, from 2011-12 to 2016-17. Results are transformed by Log10 for statistical analysis. Note the different scales on the y-axes.

Pairwise comparisons (ANOVA) indicate no significant difference in the contribution between sites, the LM site specific strata (Figure 52), or LM averaged strata (Figure 53). Pairwise comparisons (ANOVA) for the contribution from the HM strata indicated that Franz Lake was significantly lower than Ilwaco Slough and Whites Island (ANOVA, Tukey p=0.033). In addition, contribution from the site specific strata was significantly different (ANOVA, Tukey p=0.000; Figure 52). When the high marsh strata for all sites were averaged CALY was significantly greater than PHAR/HM (ANOVA, Tukey p=0.000; Figure 53) and the PHAR/POAM stratum was significantly lower than all other strata.
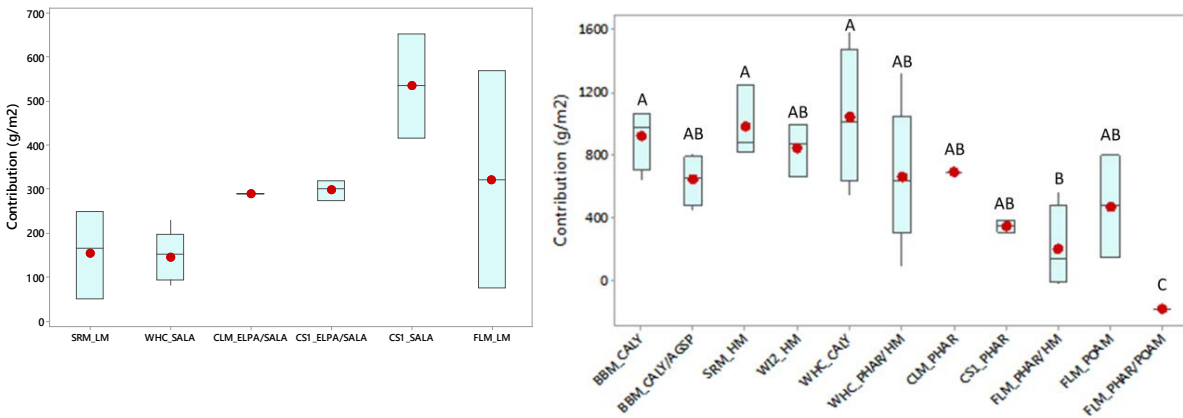
Figure 52. Average annual organic matter contribution (g dry weight/m²) from site specific low marsh (left) and high marsh (right) strata for all years from 2011-12 to 2016-17. Different letters above bars indicate means that were significantly different (p=0.000, Tukey pairwise comparison; there was no significance between low marsh strata). See Table 246 for definition of site and strata codes. Note different scales on the two plots.
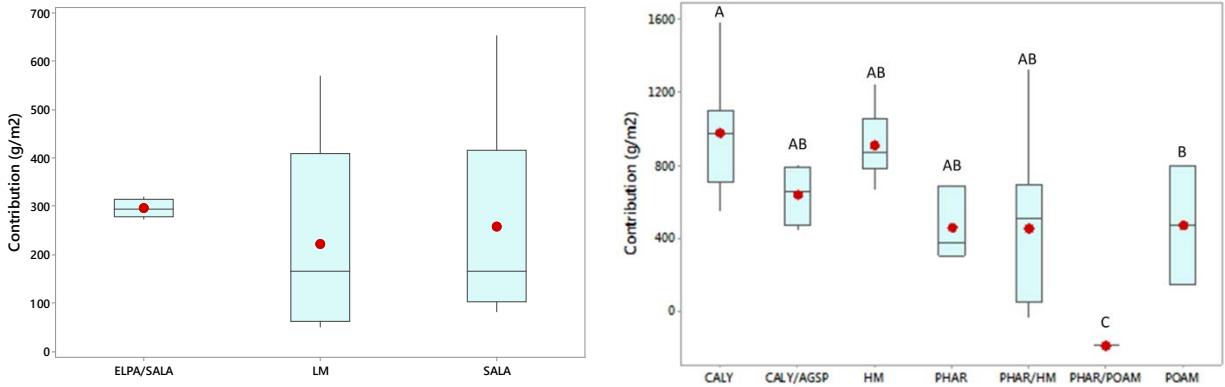
Figure 53. Average annual organic matter contribution (g dry weight/m²) by low marsh (left) and high marsh (right) vegetation strata for all sites and all years from 2011-12 to 2016-17. Different letters above bars indicate means that were significantly different (p=0.000, Tukey pairwise comparison; there was no significance between low marsh strata). See Table 24 for definition of strata codes. Note different scales on the two plots.

The proportion of summer biomass that is broken down and contributed to the organic matter pool is somewhat consistent between years within vegetation strata regardless of the quantity of summer biomass produced (Figure 54). Nearly all of the low emergent marsh biomass is broken down over the winter (Table 27). The proportion is lowest in the low marsh mix primarily because of the 2016 samples in which summer biomass was very low and 48 percent of the vegetation remained in the winter, whereas in previous years very little remained. An estimated 80 percent of *C. lyngbyei* breaks down over the winter, however, this amount is reduced when additional species are present in the strata (i.e., CALY/AGSP, HM). The PHAR/HM stratum was split between Whites Island and Franz Lake because, although *P. arundinacea* was the dominant species (Table 24), the vegetation community assemblage and the processes driving the plant material breakdown are different at the two sites. The quantity of organic matter contributed by plants in this stratum was the lowest of all the strata and variability was greatest within the PHAR/HM strata at FLM. In contrast, the *P. arundinacea* stratum, with fewer other species present (Table 24), had much lower variability (0.72 ± 0.04, Table 27). The *P. amphibium* stratum also had a high proportion of the plant standing stock remaining and was variable between years.

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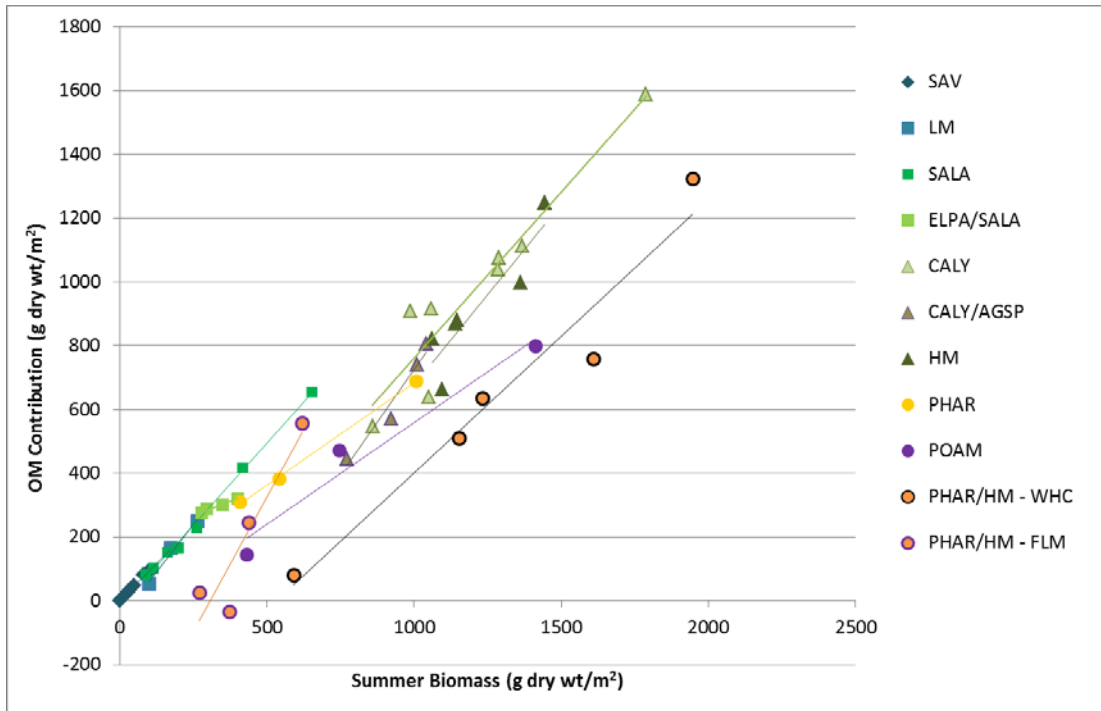
Figure 54. Linear regression between summer biomass and annual organic matter (OM) contribution for different wetland vegetation strata, from 2011-12 to 2016-17. Each dot represents the average amount for each strata within a year based on 3-12 replicates. Codes are defined and regression equations are provided in Table 27.

Table 27. Average proportion of summer 2016 biomass that was broken down over the winter and contributed to the organic matter pool for each vegetation stratum. Stratum are ordered by elevation low to high. Regression equations and R^2 values are provided based on Figure 54.

Vegetation Stratum Code	Vegetation Stratum	Average Proportion Contributed	Standard Deviation Proportion	Regression R^2 (Summer vs. Contribution)	Regression Equation
SAV	Submerged aquatic vegetation	0.90	0.27	1.00	y=0.94x + 0.83
LM	Low marsh mix	0.80	0.25	0.98	y=1.19x – 57.8
SALA	*S. latifolia*	0.93	0.06	1.00	y=1.03x - 19.6
ELPA/ SALA	*E. palustris/ S.latifolia*	0.91	0.09	0.97	y=0.35x + 182
CALY/ AGSP	*C. lyngbyei/ Agrostis* spp	0.68	0.09	0.95	y=1.32x - 595
CALY	*C. lyngbyei*	0.80	0.11	0.90	y=1.04x - 282
HM	High marsh mix	0.75	0.08	0.82	y=1.14x - 470
PHAR/HM - WHC	*P. arundinacea/* High marsh mix	0.45	0.20	0.95	y=0.85x - 455
PHAR/HM - FLM	*P. arundinacea/* High marsh mix	0.37	0.45	0.86	y=1.96x - 520
PHAR	*P. arundinacea*	0.72	0.04	1.00	y=0.64x + 41.7
POAM	*P. amphibium*	0.51	0.27	0.96	y=0.64x - 79.5

3.4.1.2 *Pelagic*

Primary production contributed by phytoplankton in the water column was estimated by the concentration of 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. Campbell Slough and Franz Lake Slough had the highest chlorophyll *a* values, particularly in April, June and July. Chlorophyll *a* concentrations in May were similar at Campbell Slough, Franz Lake Slough, and Whites Island. Otherwise, all sites had lower chlorophyll *a* than Campbell Slough and Franz Lake Slough.

Most chlorophyll *a* concentrations were <50 µg L^{-1}; however, one very high value was noted in June 2017 at Franz Lake Slough. This sample was dominated by a mixed assemblage of cyanobacteria that included *Dolichospermum* spp., *Microcystis* spp., and *Aphanizomenon* spp. (Figure 56).
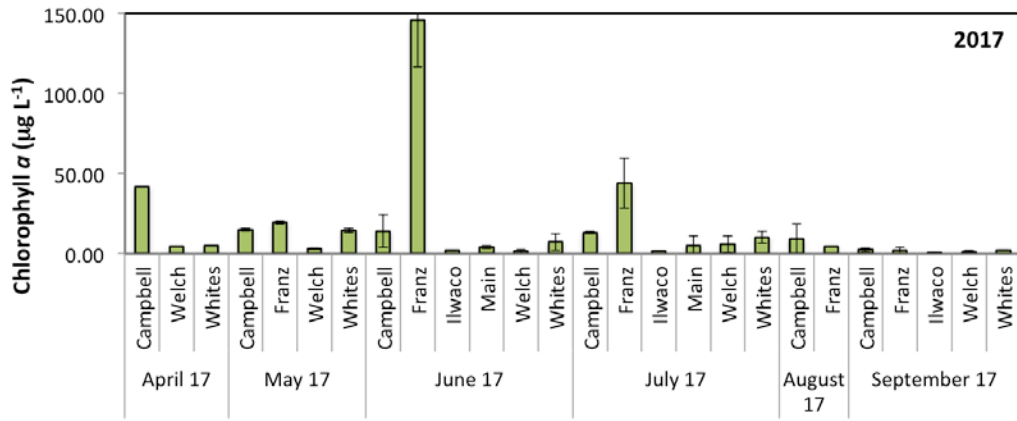
Figure 55. Chlorophyll *a* concentrations at five off-channel trends sites in 2017.
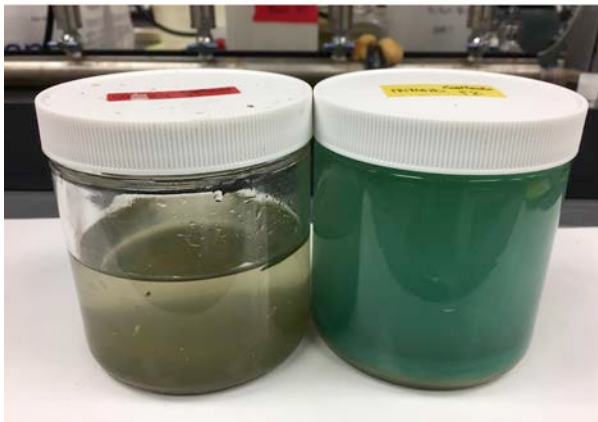
Figure 56. Image showing contents from net tows performed in June 2017 at Whites Island (left) and at Franz Lake Slough (right). The Franz Lake sample was dominated by a mixed assemblage of cyanobacteria, including *Dolichospermum* spp., *Microcystis* spp., and *Aphanizomenon* spp.

Time series of extracted chlorophyll *a* measurements revealed that Campbell Slough (Figure 58) and Franz Lake Slough (Figure 59) had levels exceeding benchmark water quality standards, that is three

observations in at least three consecutive months where chlorophyll *a* concentration exceeded 15 µg L^{-1} (Oregon State Water Quality Standards). The other three sites had levels that did not exceed water quality standards.

Primary production, as approximated by chlorophyll *a* concentration, was highest in the early spring prior to the freshet at Whites Island and Welch Island. Chlorophyll concentrations in 2017 at Ilwaco were lower than in 2016, which had the highest values observed in the 2011-2017 data set Figure 57. Primary production at this site was greatest in the late summer in 2017, with chlorophyll concentrations similar to previous years). The highest chlorophyll concentrations at Campbell Slough and Franz Lake Slough tend to occur later in the year, either as the freshet is subsiding (e.g., at Campbell) or during the warm summer months (e.g., at Franz) when cyanobacteria biomass can reach very high levels (e.g., in June 2017). However, chlorophyll *a* concentrations can also be high at Campbell Slough prior to the spring freshet, as was observed in 2017 (Figure 32, Figure 58).
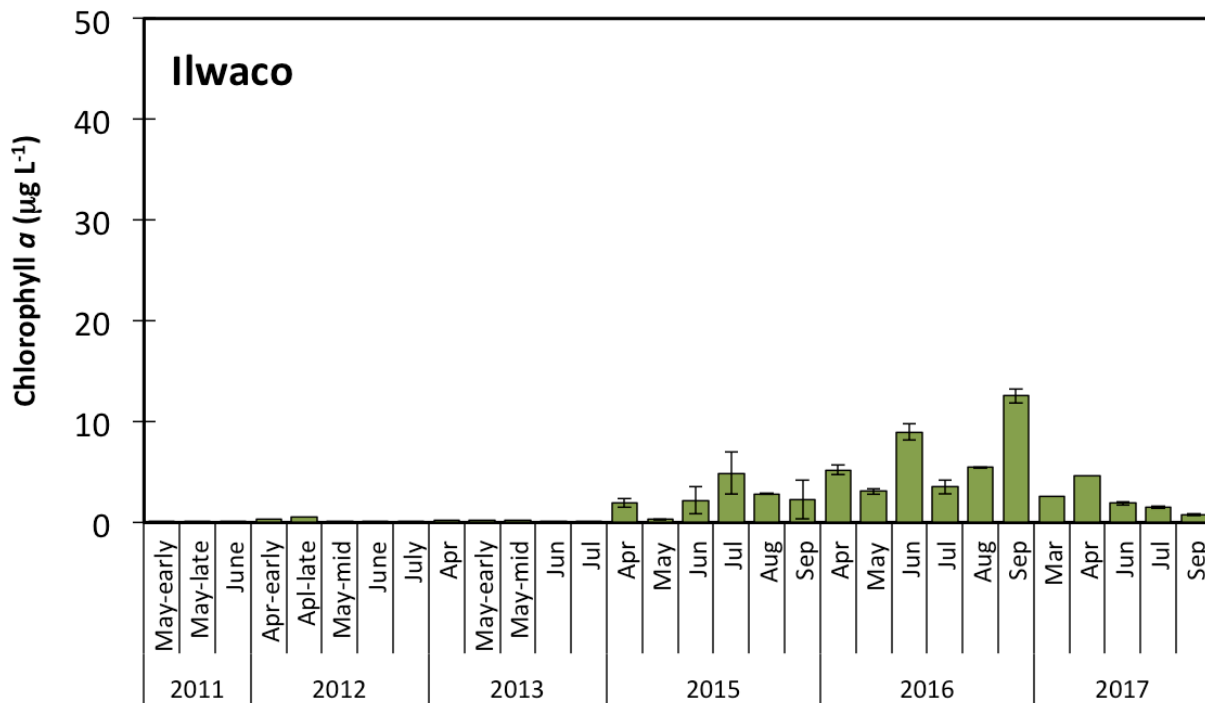
Figure 57. Chlorophyll *a* concentrations in discrete samples collected from Ilwaco (2011–2017).
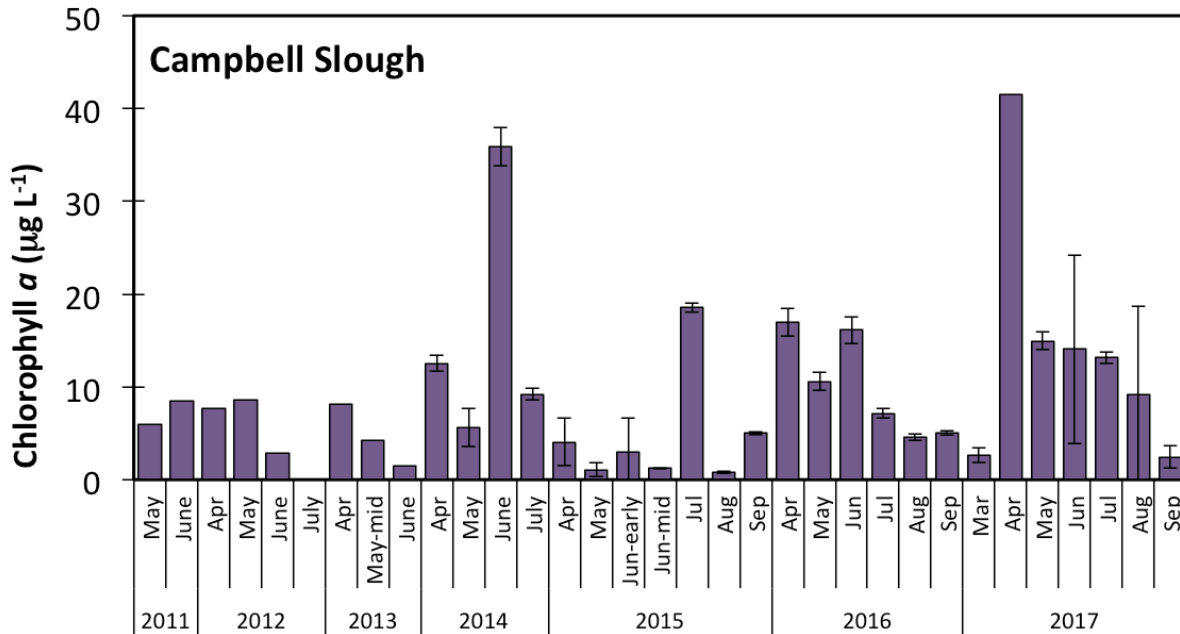
Figure 58. Chlorophyll *a* concentrations in discrete samples collected from Campbell Slough (2011–2017).
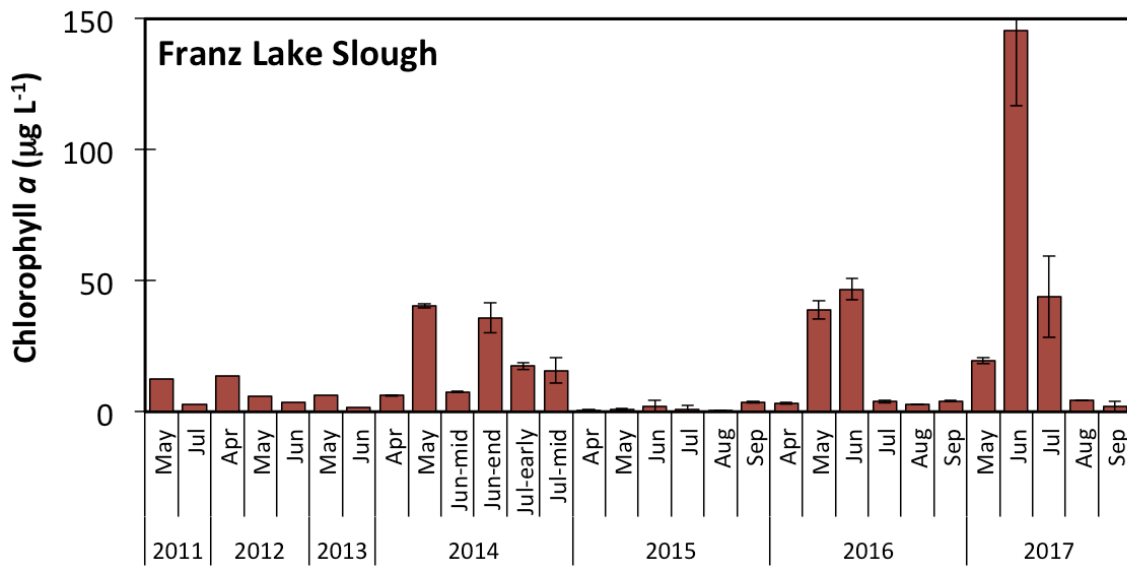
Figure 59. Chlorophyll *a* concentrations in discrete samples collected from Franz Lake Slough (2011–2017). From the time series, it is apparent that the high chlorophyll *a* concentration observed at Franz Lake Slough in June 2017 was exceptionally compared to observations in other years.
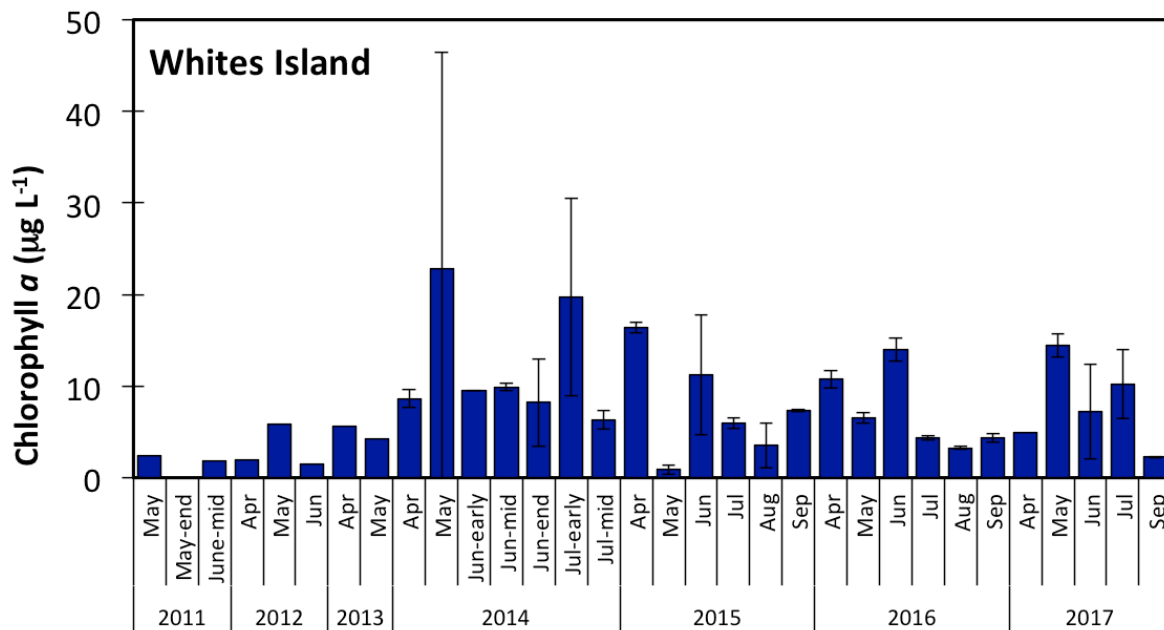
Figure 60. Chlorophyll *a* concentrations in discrete samples collected from Whites Island (2011–2017).
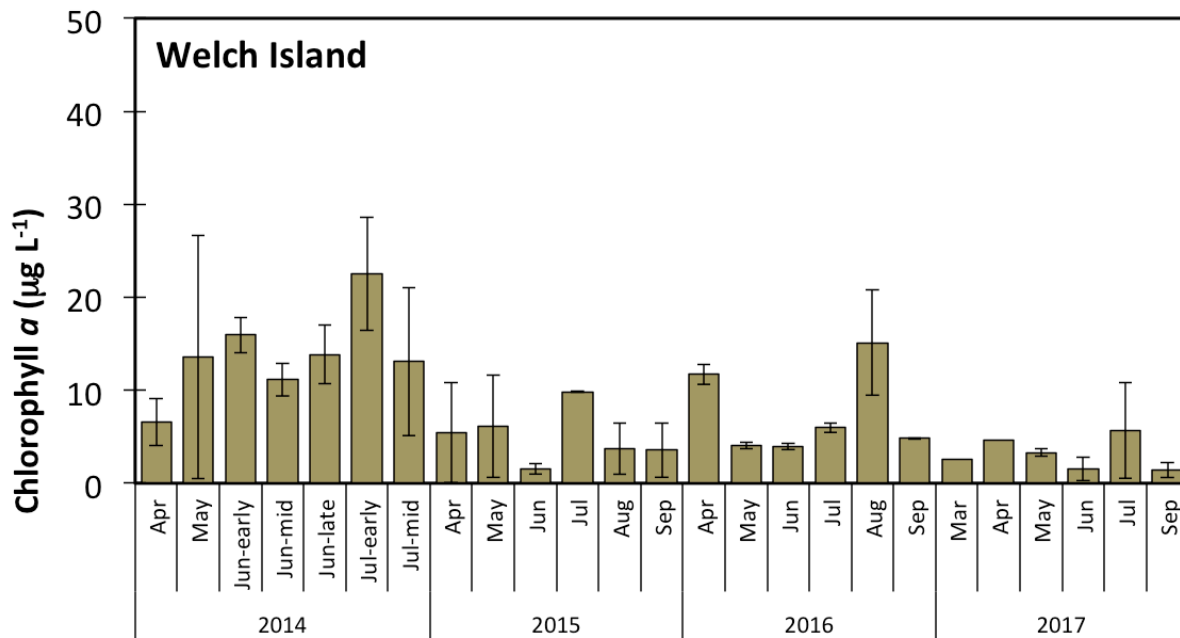
Figure 61. Chlorophyll *a* concentrations in discrete samples collected from Welch Island (2014–2017).

When all available data are plotted (2011-2017), the very highest value observed at Franz in 2017 stands out as an extreme outlier (Figure 62). Based on the data set, 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.
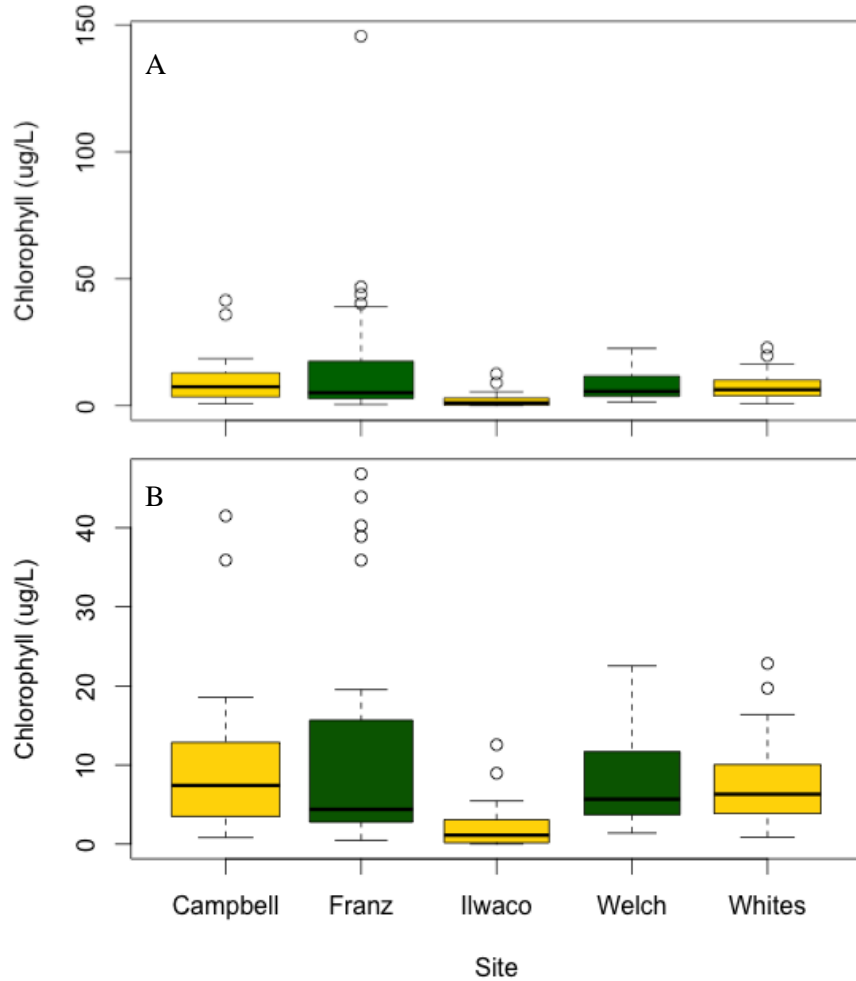
Figure 62. Boxplots showing chlorophyll *a* concentrations determined at the five off-channel trends sites between 2011 and 2017. (A) The data include the extreme outlier at Franz Lake Slough in 2017; (B) the data excluding the outlier at Franz Lake Slough in 2017.

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). In addition, 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). However, ciliate abundances were very low and therefore were not included in plots.

As in previous years, and consistent with pigment data, phytoplankton were present at higher densities at Campbell Slough (Reach F) compared to Whites Island (Reach C) (Figure 63). Similar to previous years,

there was also a greater contribution to the phytoplankton assemblage by cyanobacteria at Campbell Slough compared to Whites Island (Figure 63, Figure 64).
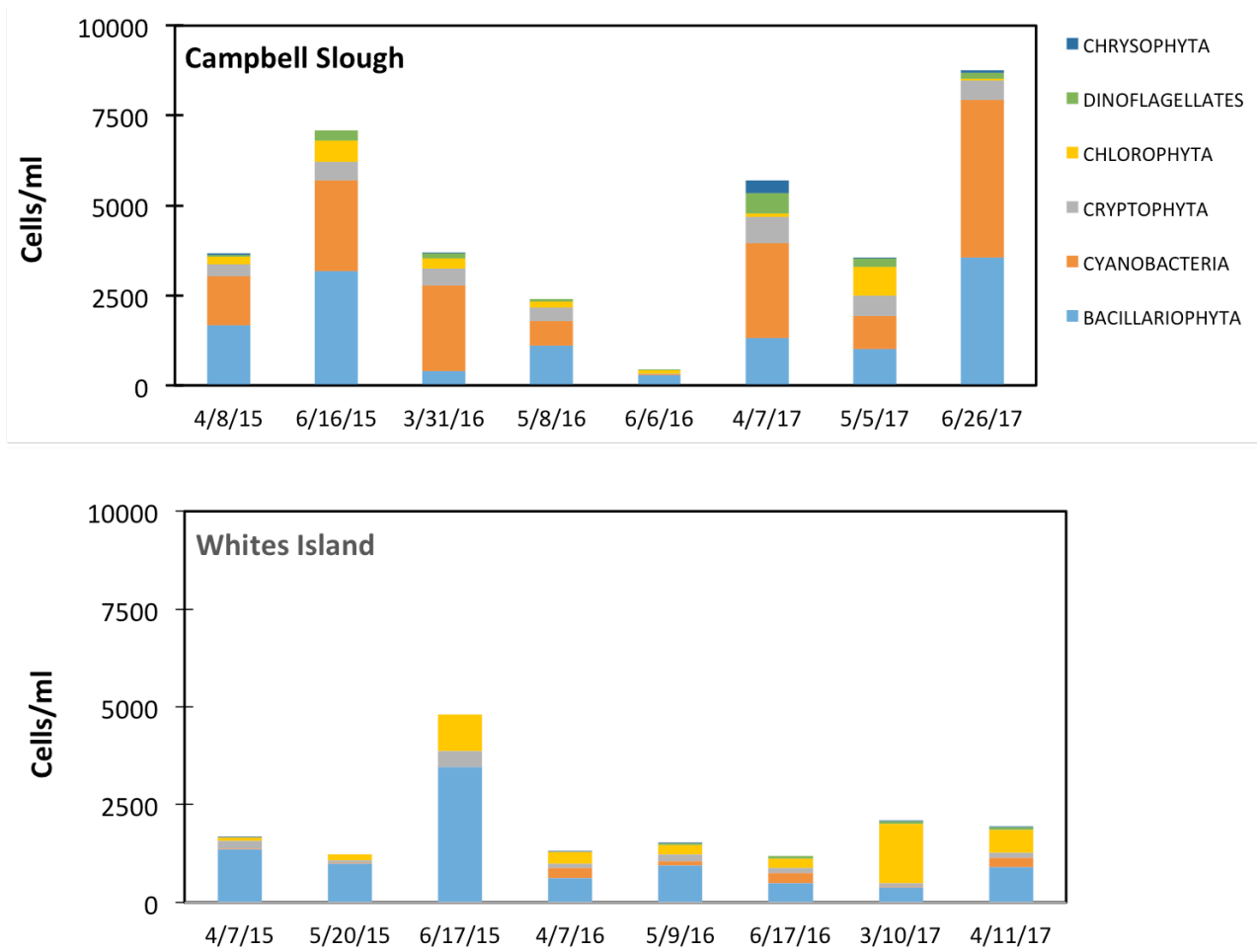
Figure 63. Cell densities of phytoplankton at Campbell Slough (upper panel) and Whites Island (lower panel) during the spring (2015-2017). To improve graph readability, the abundance of cyanobacteria at Campbell Slough on 4/7/2017 is 10 times the number shown.
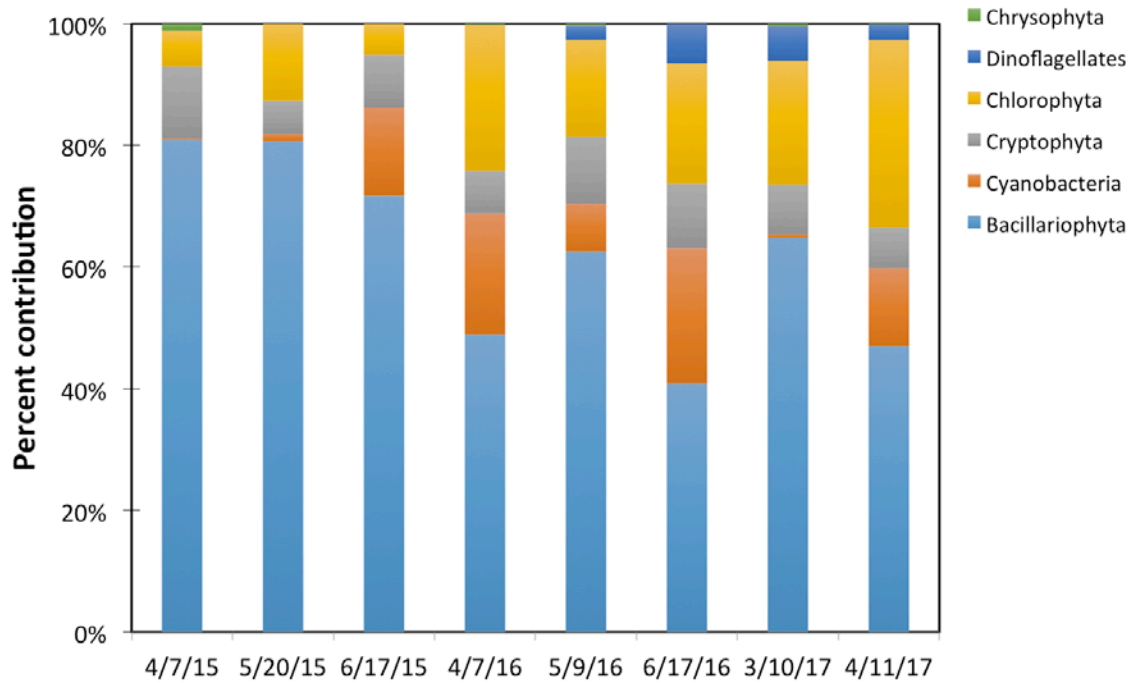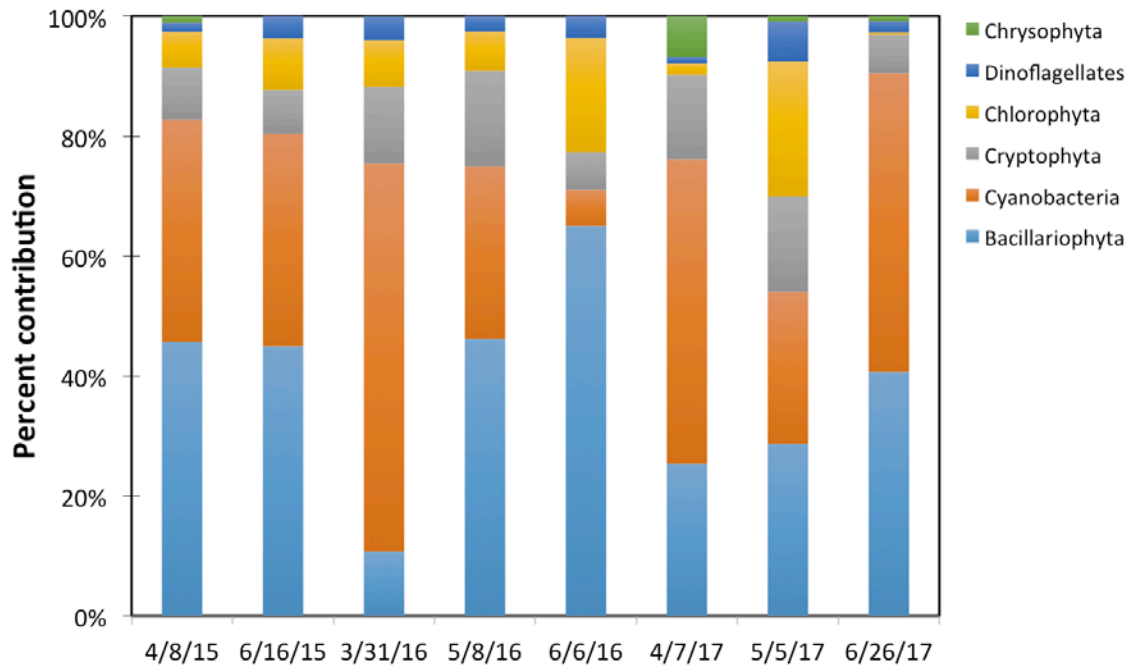
Figure 64. Time series showing percent contribution to total phytoplankton assemblages by various algal groups at Campbell Slough (*top panel*) and Whites Island (*bottom panel*) for spring periods in 2015, 2016, and 2017.

After performing a Canonical Analysis of Principal Coordinates (CAP), a small number of diatom species were found to account for significant proportions of variability in the phytoplankton density data. Two of the species contributing the most to variability (as determined by eigenvector values) were *Asterionella formosa* and *Skeletonema potamos*. Both *A. formosa* and *S. potamos* have been shown to contribute to blooms of phytoplankton in the spring and summer (Maier 2014, Tausz, 2015, Maier and Peterson 2017). *A. formosa* tends to be more abundant and dominant in the early spring (February–April); that is, it is a "spring-type" species, while *S. potamos* tends to become abundant later in the season (i.e., a "summer-type" species). Relationships between species abundance and environmental characteristics support these classifications (Figure 65).
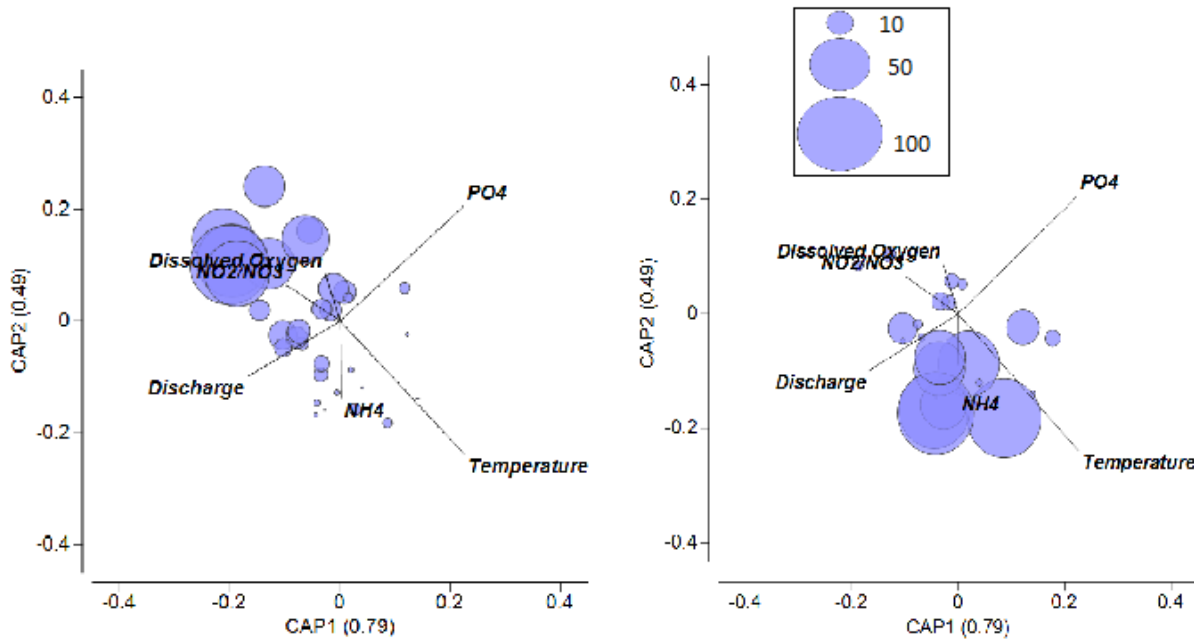
Figure 65. Results from canonical correspondence analysis (CCA) showing "spring type" phytoplankton species (left is *Asterionella formosa*) and "summer type" phytoplankton species (right is *Skeletonema potamos*); bubble size indicates relative abundance from 0% to 100% of the phytoplankton assemblage.

3.4.1.4 *Trends in Net Ecosystem Metabolism within Off-channel Sites*

Primary productivity – the fixation of inorganic carbon through the process of photosynthesis – can be estimated from changes in dissolved oxygen, since one of the products of photosynthesis is oxygen. Using a time series of hourly dissolved oxygen data, daytime oxygen production and night-time oxygen removal (through respiration) can be computed and compared to yield net ecosystem metabolism, the difference between production and utilization of oxygen in the water column. Daily NEM was computed at Whites Island (Figure 66) and Campbell Slough (Figure 67), two sites with contrasting patterns in primary productivity. In these figures, positive NEM values indicate that primary production dominates, while negative values indicate that respiration dominates. In the years examined (2011, 2013, 2014, and 2016), NEM was almost universally positive at both sites, rarely approaching zero. These data show that the off-channel sites were characterized by net autotrophy, or net production of organic matter.

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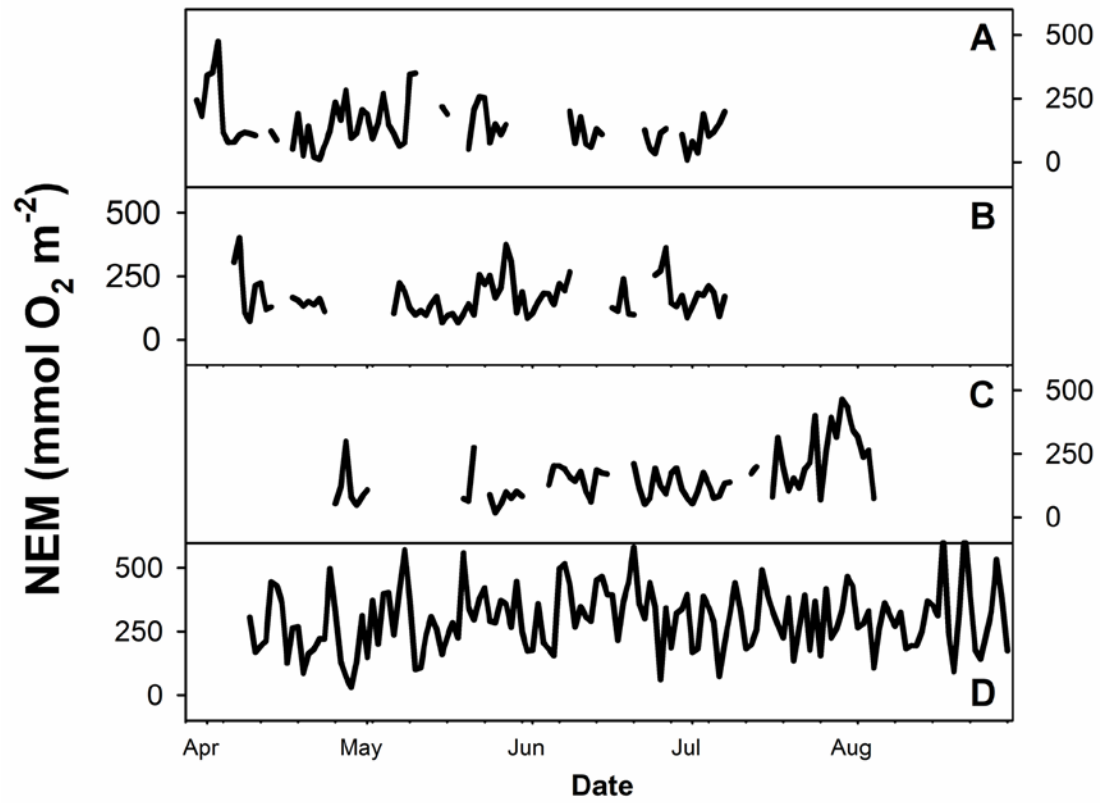
Figure 66. Daily net ecosystem metabolism values computed for Whites Island in A: 2011, B: 2013, C: 2014, D: 2016
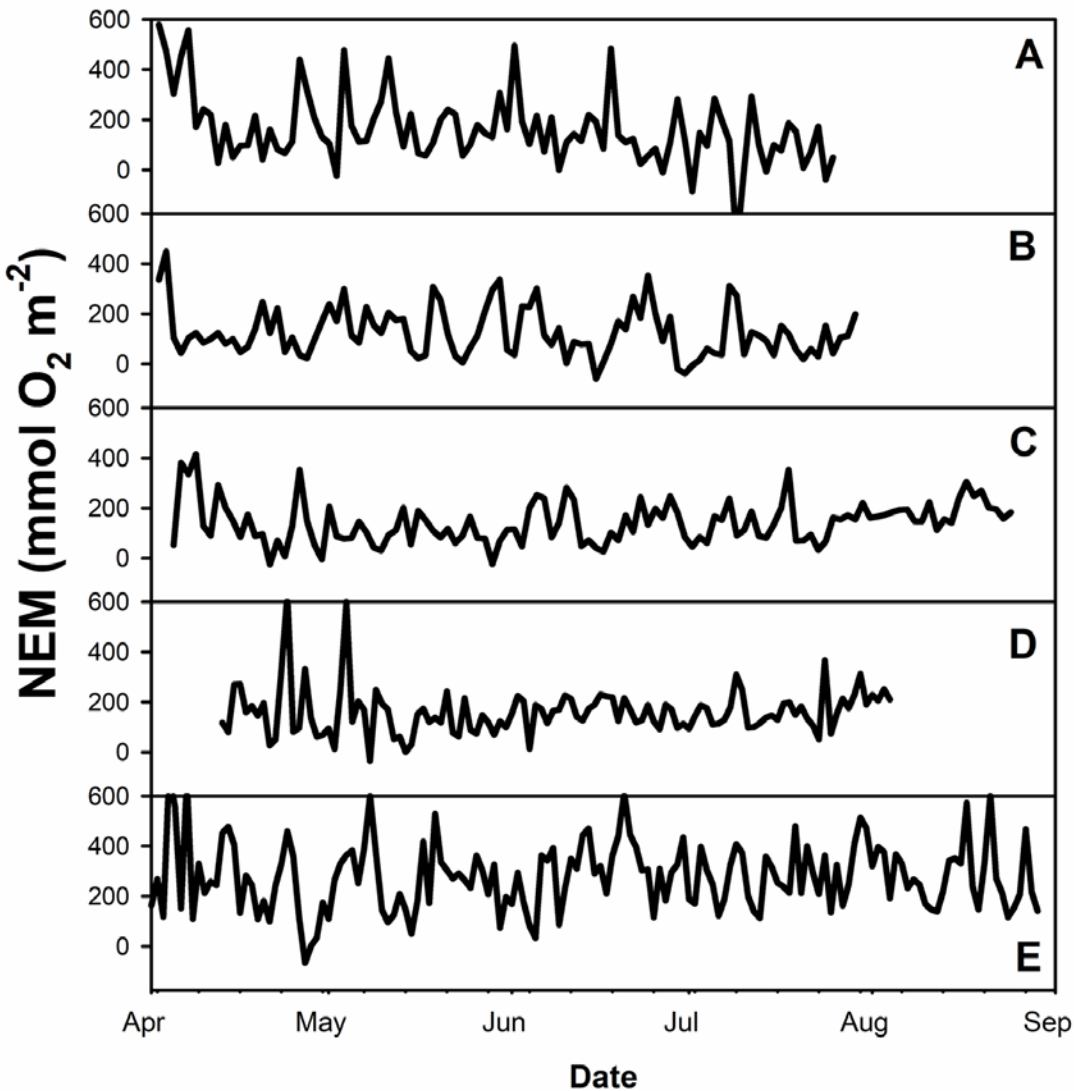
Figure 67. Daily net ecosystem metabolism at Campbell Slough. A: 2010, B: 2012, C: 2013, D: 2014, E: 2016

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, with higher densities at Campbell Slough than the other sites (Figure 68). Similar to previous years, zooplankton assemblages at Ilwaco were dominated by copepods. The zooplankton assemblages at both Campbell Slough and Franz Lake Slough were dominated by copepod nauplii and rotifers in March, particularly *Keratella* 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 domianted 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 (*Nothalca* sp.), copepod nauplii, and harpacticoid copepods.
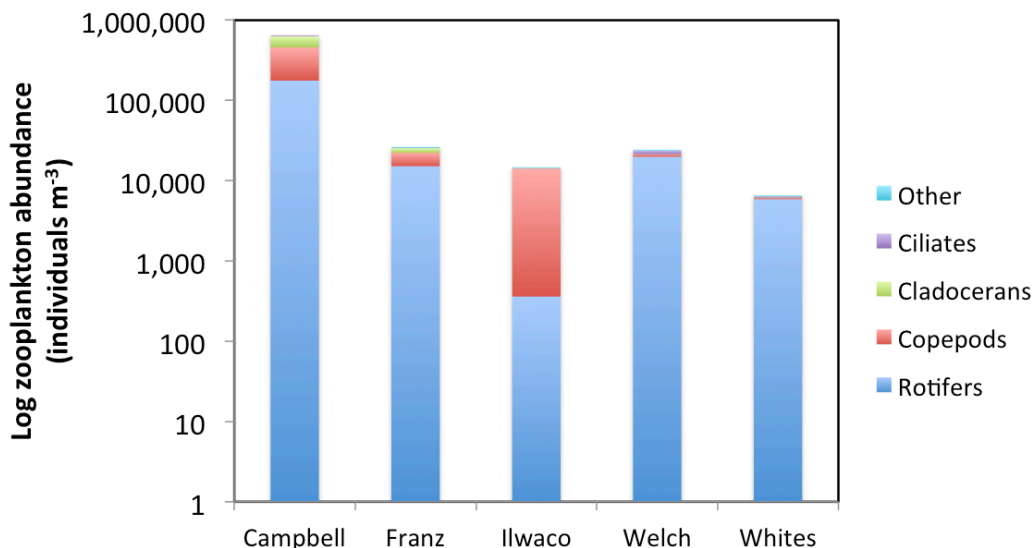
Figure 68. Spring zooplankton assemblage composition at five off-channel trends sites.

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 δ^{13}C values between -24 and -34‰, seaweeds and marine plants between -6 and -19‰, and algae and lichens -12 to -23‰. According to Cloern (2002), δ^{13}C values for freshwater phytoplankton are between -29.5 and -27.5‰, which overlaps with emergent vascular plant matter, which typically has δ^{13}C values ranging from -28.1 to -27.2‰. Sediments and soils tend to have heavier isotopic signatures, while runoff can have lighter values. Isotopic values of carbon in particulate organic matter (δ^{13}C-POM) collected onto filters revealed δ^{13}C signatures in the range of freshwater phytoplankton most of the time, with values closer to terrestrial vascular plants in May and June at Campbell Slough and Franz Lake Slough. δ^{13}C-POM at Ilwaco was closer to marine values. It is interesting that the signatures of δ^{13}C-POM would be close that that of vascular plants, since POM was collected by filtering whole water onto glass fiber filters and much of the material collected onto the filters was composed of phytoplankton (the same collection method is used to determine chlorophyll *a* concentrations)(Figure 69).

Nitrogen isotopic signatures determined from particulate organic matter were more variable compared to carbon (Figure 69). The very light value for nitrogen isotopes observed at Franz Lake Slough in June (2 ‰) coincided with high biomass of cyanobacteria; light δ^{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).
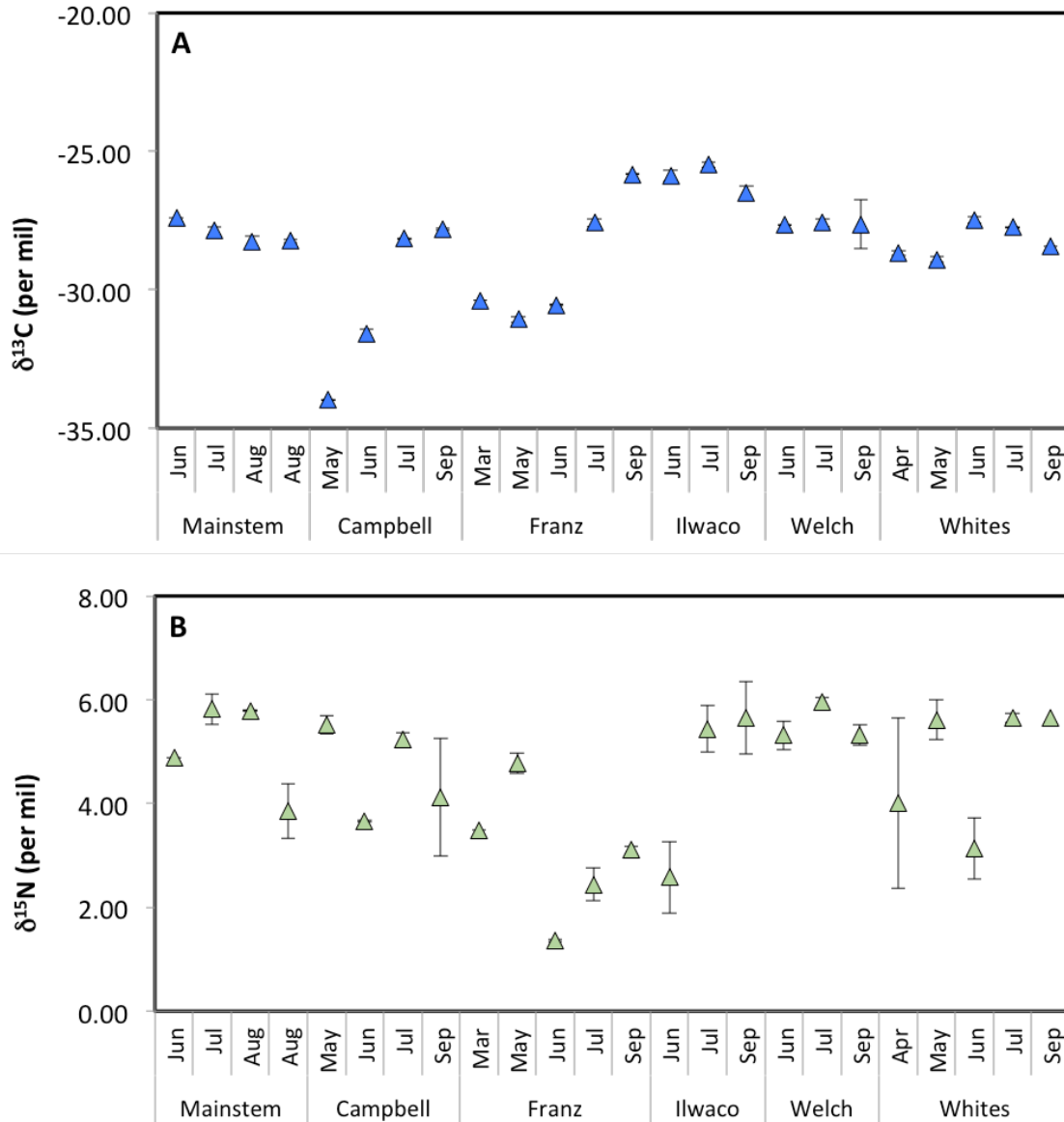
Figure 69. Carbon (A) and nitrogen (B) isotope data collected from particulate organic matter (POM) at the five off-channel trends sites as well as the river mainstem in 2017.
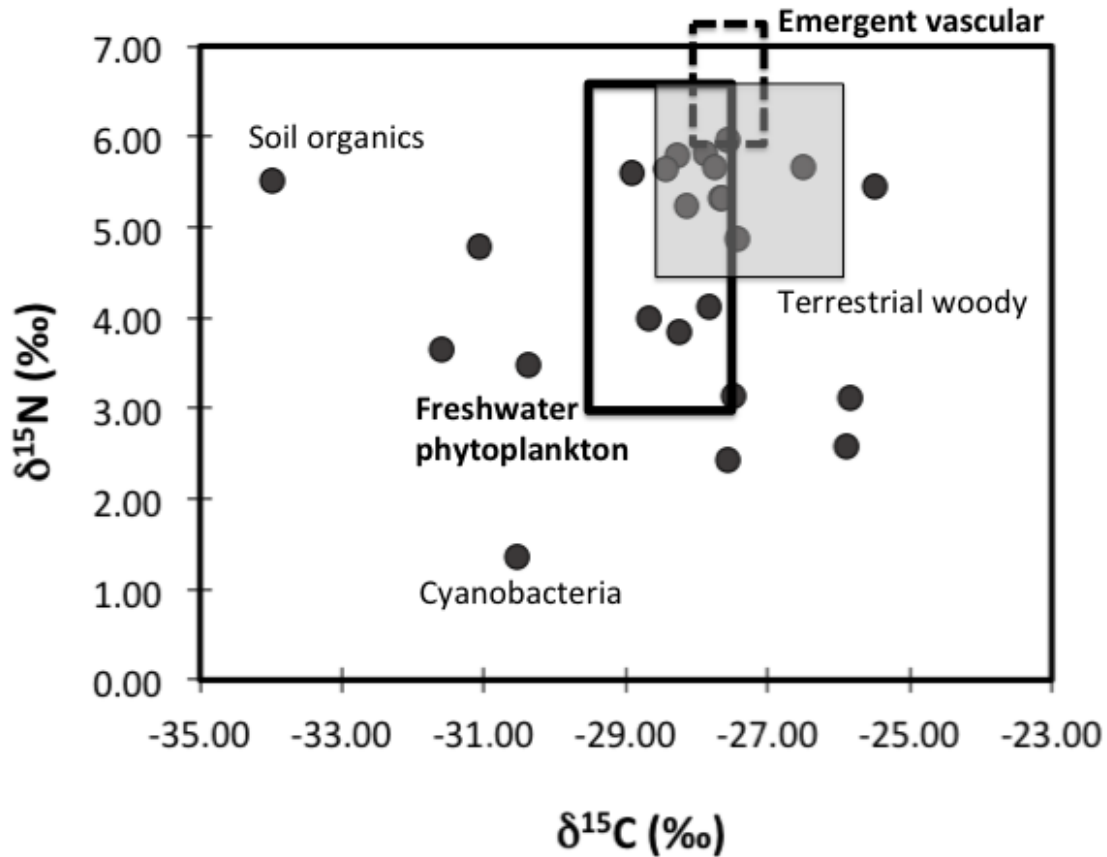
Figure 70. Plot of particulate organic matter (POM) data from off-channel trends sites (circles) in isospace; typical isotopic signature ranges for different organic matter sources are shown (derived from Cloern, 2002). The contribution of various sources to measured POM in the lower Columbia is evident in the data spread.

The spread in isotopic signatures (Figure 70) 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.

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 71) led to large uncertainties in the estimated dietary proportions (Figure 72).
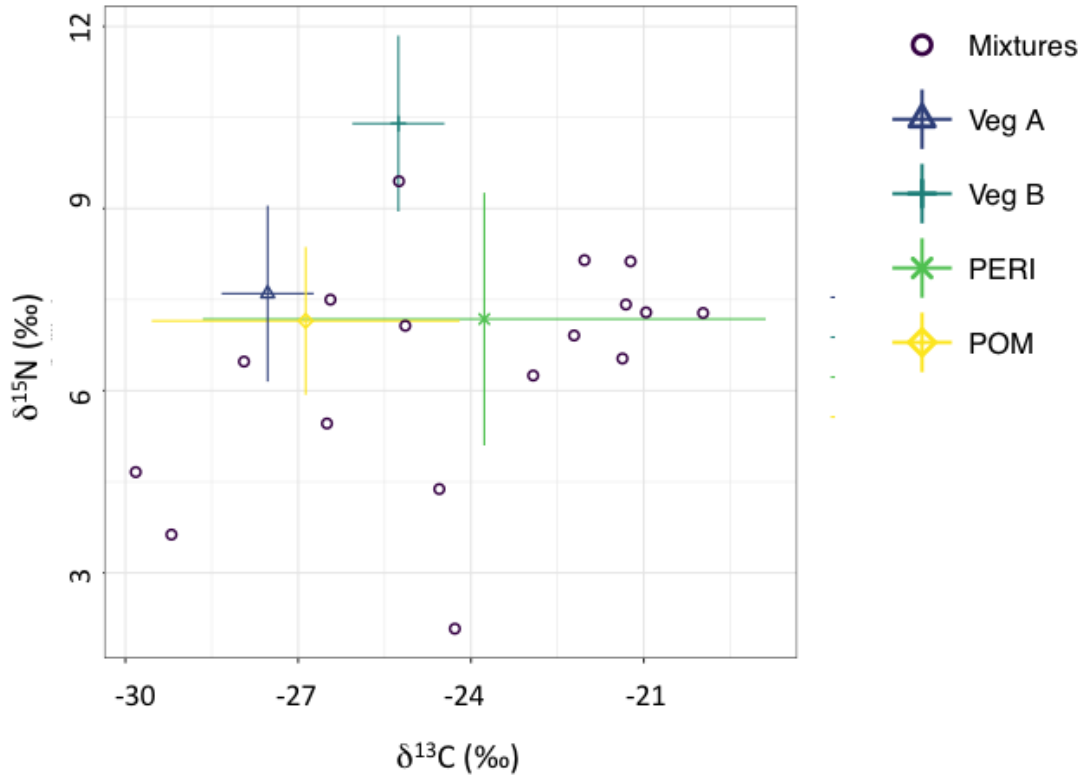
Figure 71. Isospace plot showing stable isotope signature of chironomids ("mixtures") compared to vascular plant matter with heavier δ¹³C and δ¹⁵N (Veg A) and those having light δ¹³C and δ¹⁵N (Veg B) as well as to periphyton (PERI) and particulate organic matter (POM).
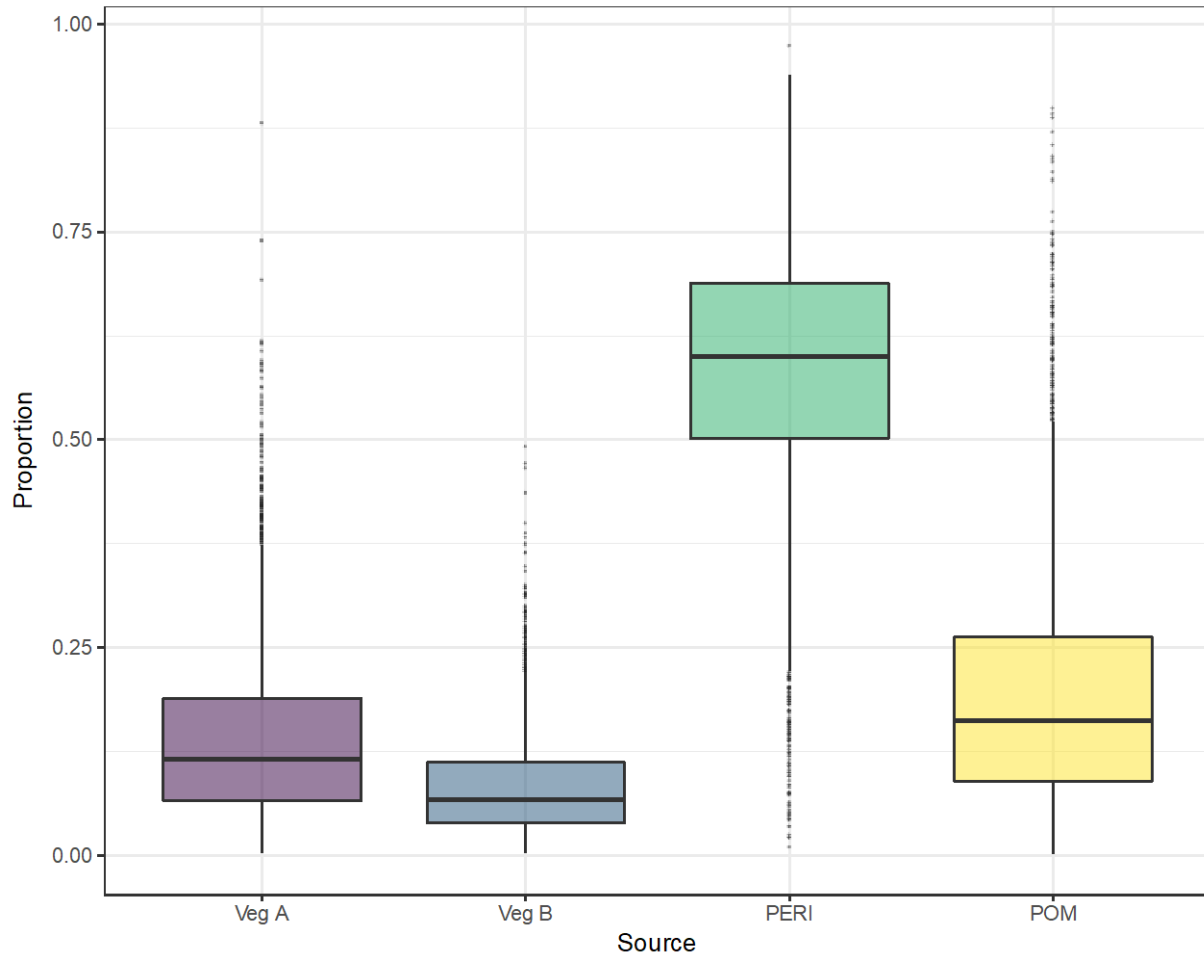
Figure 72. Dietary proportions of various organic matter sources for chironomids estimated by Bayesian stable isotope mixing model. Sources include vegetation (Veg A and Veg B), periphyton (PERI), and particulate organic matter (POM).

According to the estimates produced by the Bayesian mixing model, it is likely that chironomids were consumers of organic matter derived from periphyton (Figure 72), although the wide spread 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.

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 greater spread in the data in the unmarked fish (Figure 73). Carbon was heavier while nitrogen was lighter in hatchery fish relative to unmarked fish. This resulted in differences in the estimated dietary proportions from various organic matter sources (Figure 74). 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.
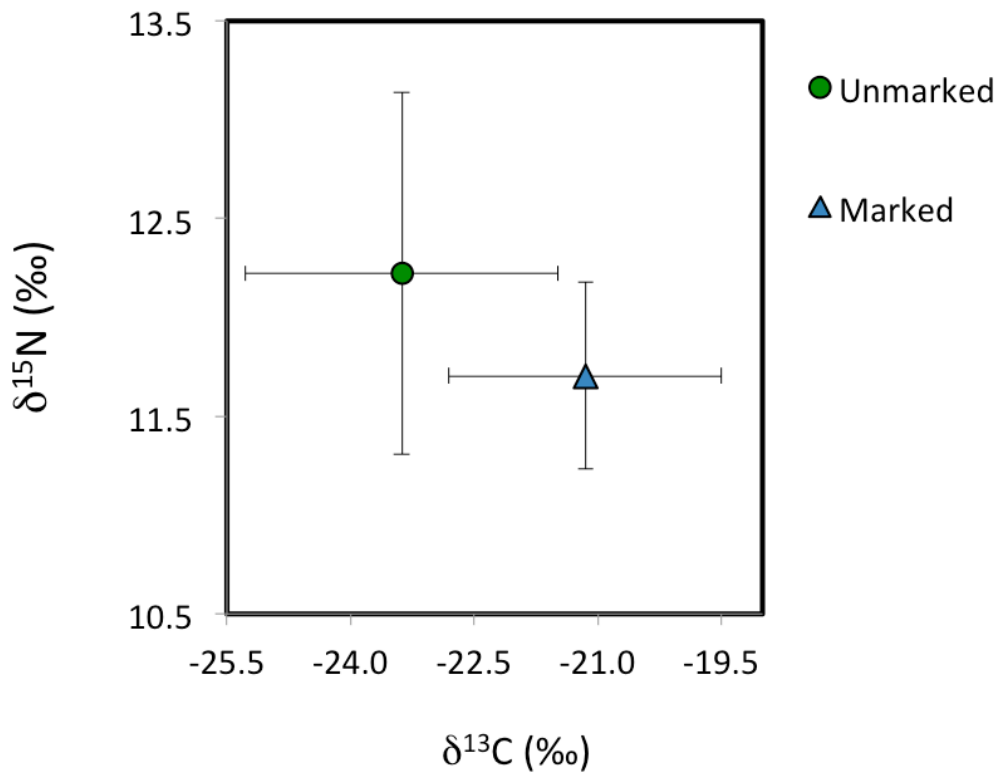
Figure 73. Average stable isotope ratios of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) from juvenile Chinook salmon muscle associated with marked and unmarked individuals.
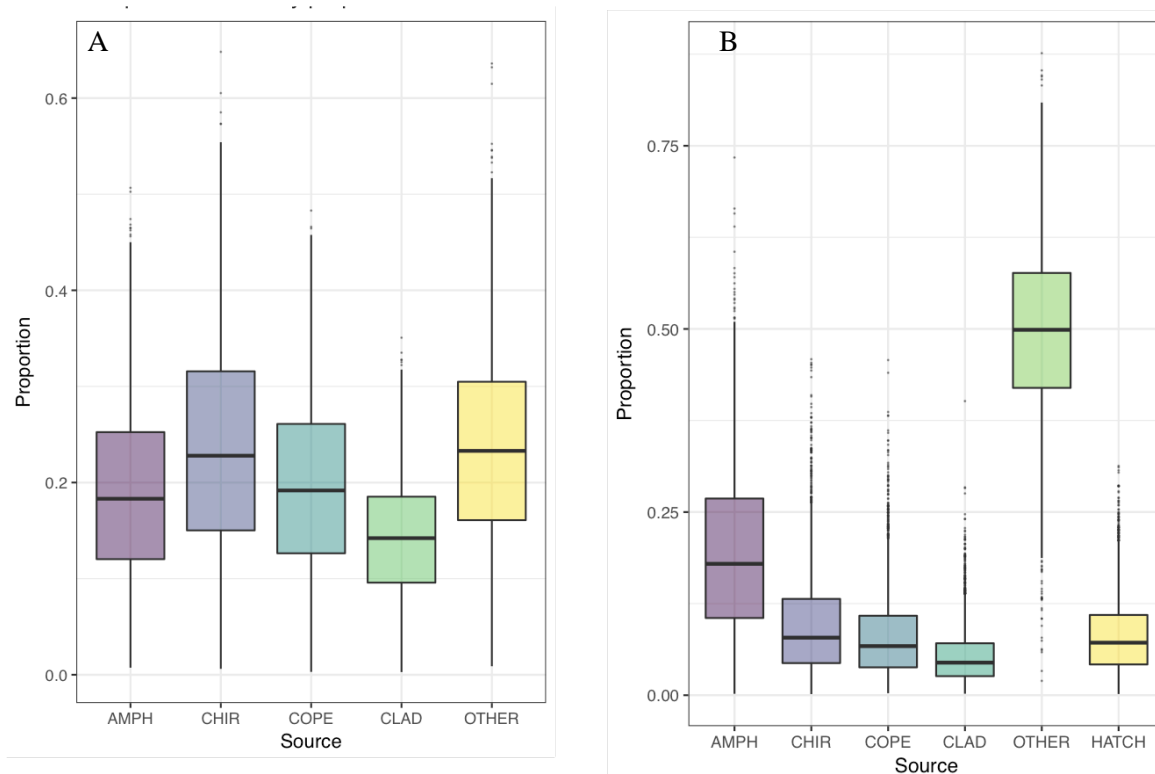
Figure 74. Dietary proportions of different organic matter sources supporting juvenile Chinook salmon. A: unmarked; B: marked. Sources include: amphipods (AMPH), chironomids (CHIR), copepods (COPE), cladocerans (CLAD), hatchery food (HATCH), and "other" sources (includes polychaetes, oligochaetes, and other insects).

3.5 Macroinvertebrates

3.5.1 Salmon Prey Availability

Salmon prey availability data collected in 2017 are still under analysis and unavailable at the time of the writing of this report. The following details are adapted from the 2016 report.

3.5.1.1 *Benthic*

Benthic core samples from all sites were dominated by nematode and oligochaete worms (Figure 75). Together, these two groups accounted for on average 82 percent of benthic invertebrate counts and 74 percent of biomass. Chironomids and other dipterans were also consistently collected in benthic cores; Campbell Slough in April was the only occasion where no flies were collected. Flies contributed on average eight percent of counts and four percent of biomass. Small-bodied invertebrates, such as chironomids, collembola, and nematodes, made larger contributions to density counts than to total biomass. While not always numerically abundant, the large body size of amphipods, bivalves, gastropods, isopods, and unidentified invertebrate egg cases, made a relatively large contribution to the proportional biomass when they were present.
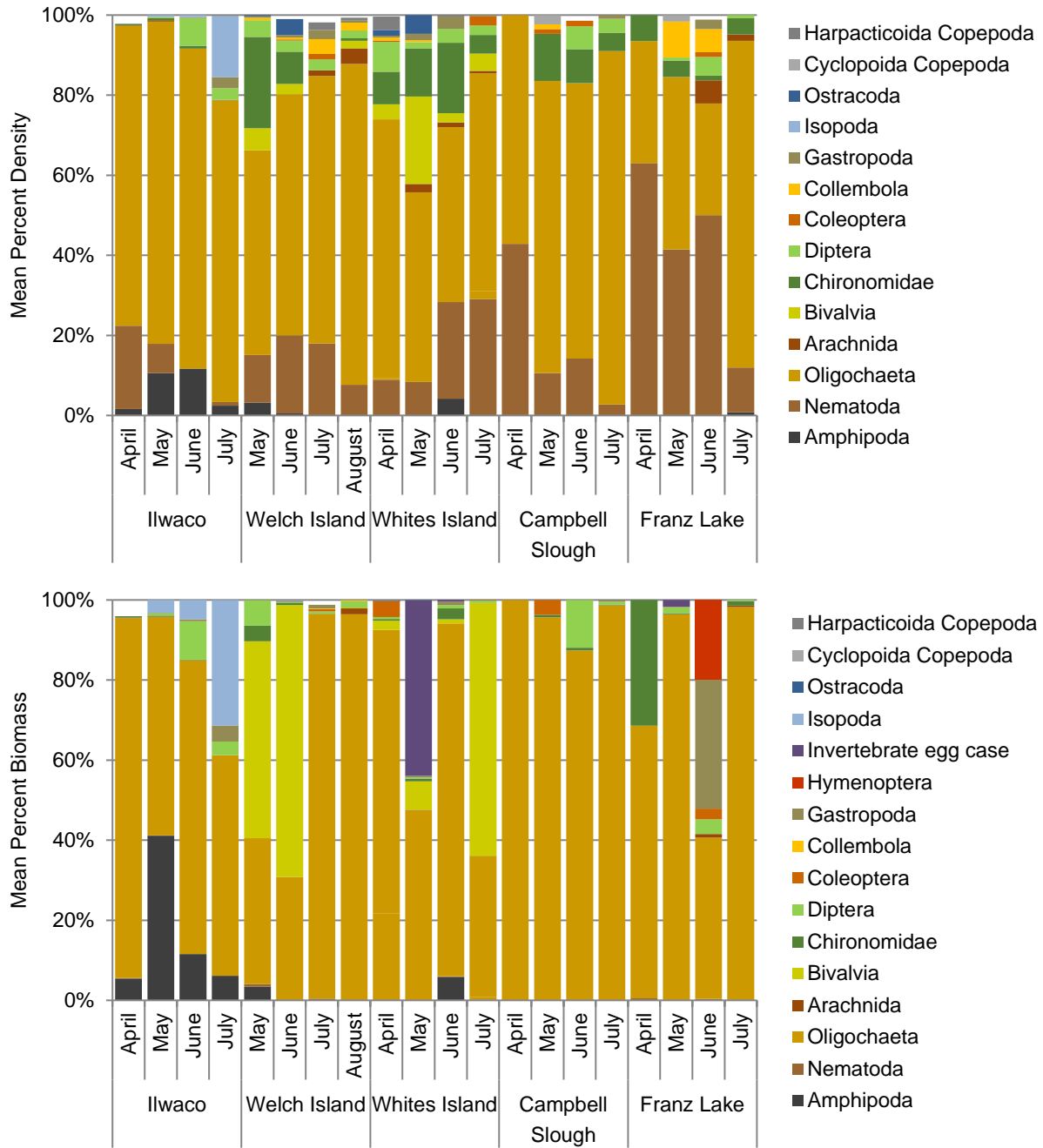
Figure 75. Composition of benthic core samples by mean percent abundance (top) and biomass (bottom) in 2016.

Average counts of all invertebrate taxa were greatest from Ilwaco Slough in all months compared to other sites, and peaked in May with over four million individuals per cubic meter of benthic sediment (Figure 76). Consistent with 2015 samples, average densities were similar from sites in Reach C (Welch Island and Whites Island) as were densities from the upper estuary sites (Campbell Slough and Franz Lake) which were lower on average relative to other sites. Average biomass of the benthos was similar among the lower estuary sites (Ilwaco Slough, Welch Island, and Whites Island). Campbell Slough and Franz Lake consistently had a lower average biomass in monthly comparisons to other sites. A seasonal pattern in density or biomass was not apparent in the benthic samples.
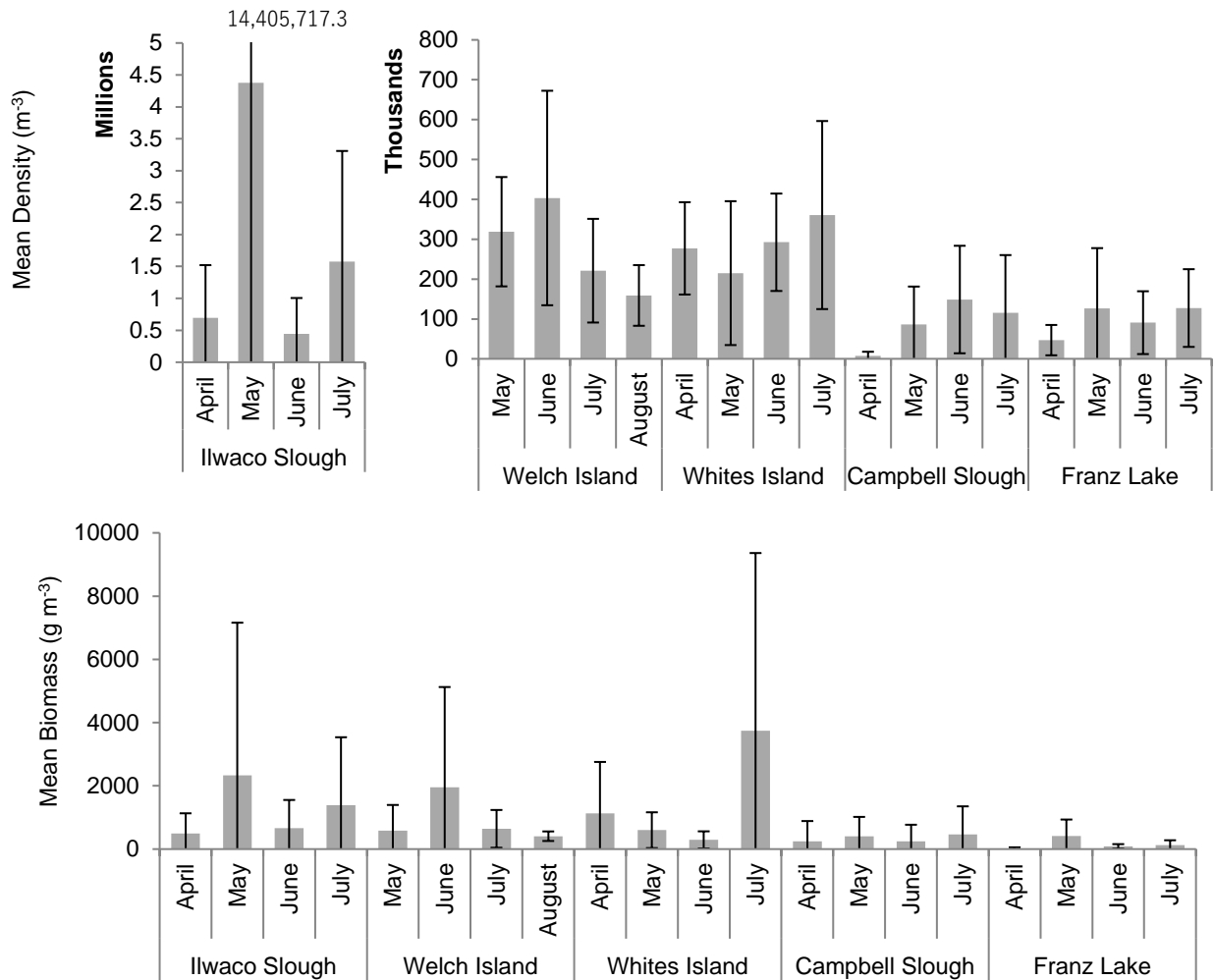
Figure 76. Average density (top) and biomass (bottom) per cubic meter of all invertebrate taxa collected by benthic core in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted. Densities from Ilwaco Slough are shown on a different scale than other sites.

Dipterans were collected from all sites in each month sampled, except Campbell Slough in April (Figure 77). Trends in the average biomass tended to follow those of the average density, with peaks occurring in May, June, or July depending on the site.
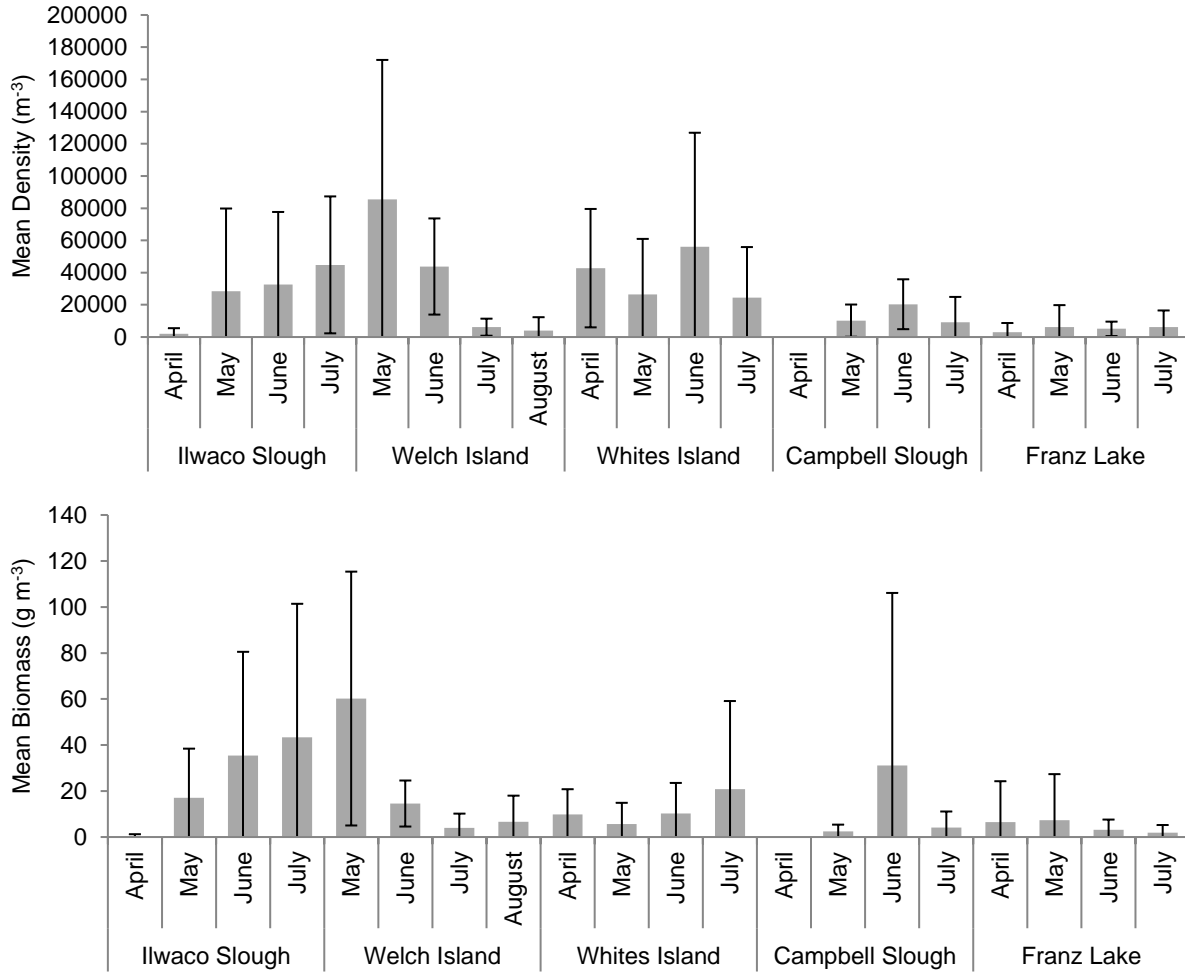
Figure 77. Average density (top) and biomass (bottom) per cubic meter of Diptera (including Chironomidae) collected by benthic core in 2016. Error bars are 95% confidence intervals (CI) around the mean.

Amphipods were abundant at Ilwaco Slough in May, though their occurrence at that time was highly variable (Figure 78). Other sample events saw much lower densities and biomass of amphipods, with zero collected from Campbell Slough and only one individual collected from Franz Lake in July.
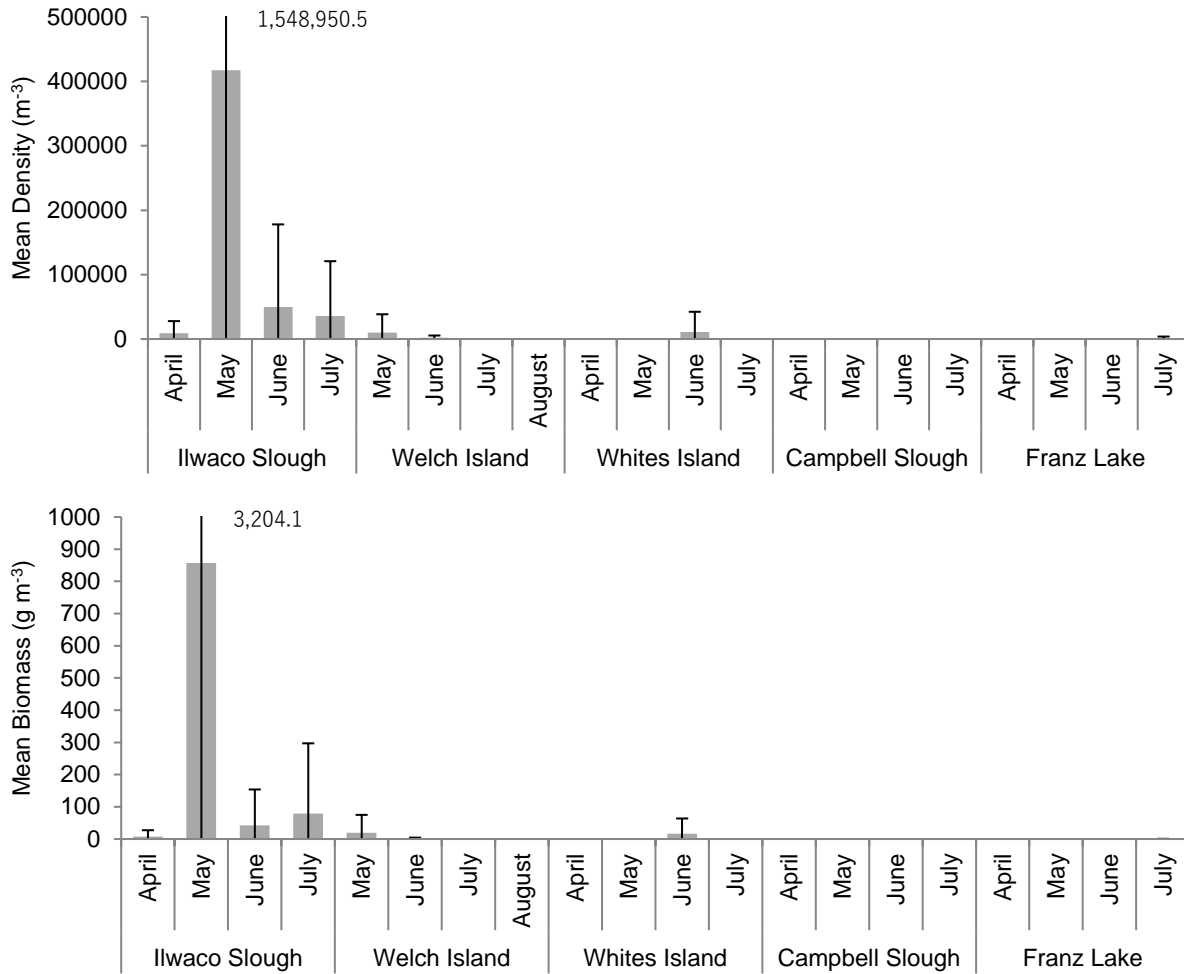
Figure 78. Average density (top) and biomass (bottom) per cubic meter of Amphipoda collected by benthic core in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted.

The average density of all taxa was compared to benthic samples from previous years (Figure 79). Average density in 2016 was greater than that from previous years at Ilwaco Slough in May and July. However, there was considerable variation within these 2016 samples and the average from previous years is within the 95 percent confidence interval. Other samples in 2016 were similar to the previous years' average.
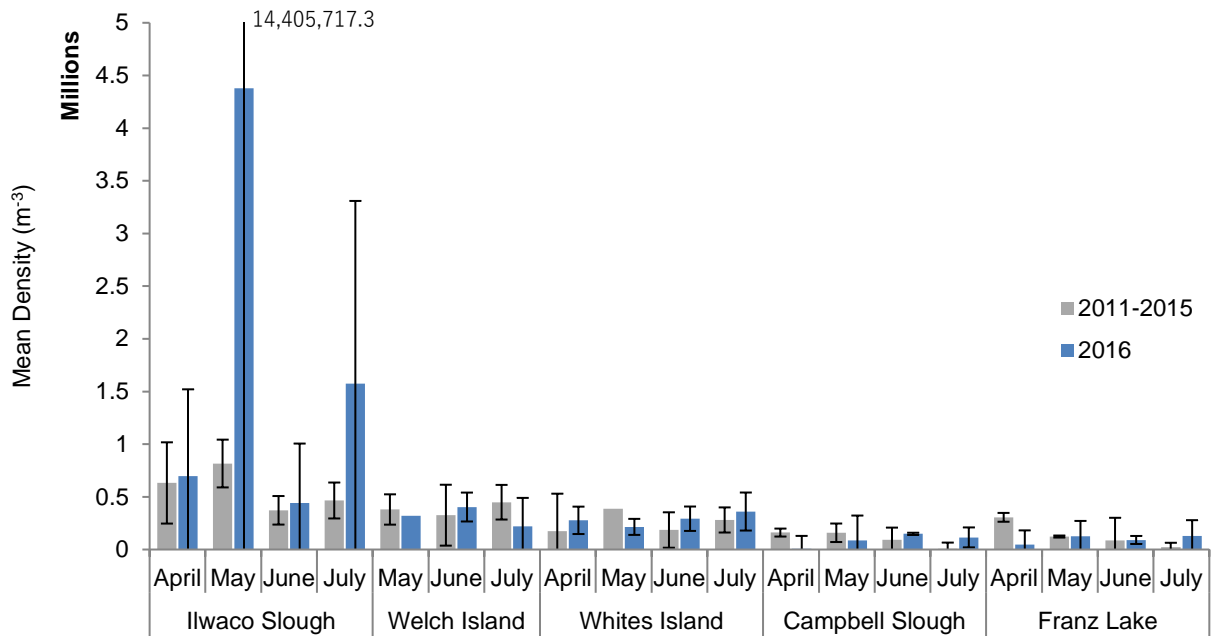
Figure 79. Comparisons between average density (count per cubic meter) summarized between 2011 and 2015 to average density in 2016 of all benthic invertebrate taxa by sample site and month. Error bars represent 95% confidence intervals. Where CI extends off the chart, the maximum range is noted.

3.5.1.2 *Neuston*

Neuston samples were composed of a diverse array of benthic/epibenthic, terrestrial riparian, and planktonic taxa (Figure 80). Composition varied both within sites across months, as well as across sites. However, given the variation within categories, ANOSIM tests for differences between sites, months, and habitats found no significant differences in average abundance or biomass. Planktonic taxa, such as copepods and cladocerans, tended to comprise a higher proportion of abundance and biomass in the open water habitat. Dipterans, hemipterans, gastropods, and other terrestrial riparian taxa tended to account for greater proportions of the emergent vegetation composition. While not always numerically abundant, the large body size of amphipods, bivalves, gastropods, hemipterans, odonates, and mysids made a relatively large contribution to the proportional biomass when they were present.

Figure 80. Composition of neuston tows in emergent vegetation (EV) and open water (OW), by mean percent abundance (top) and biomass (bottom) in 2016.

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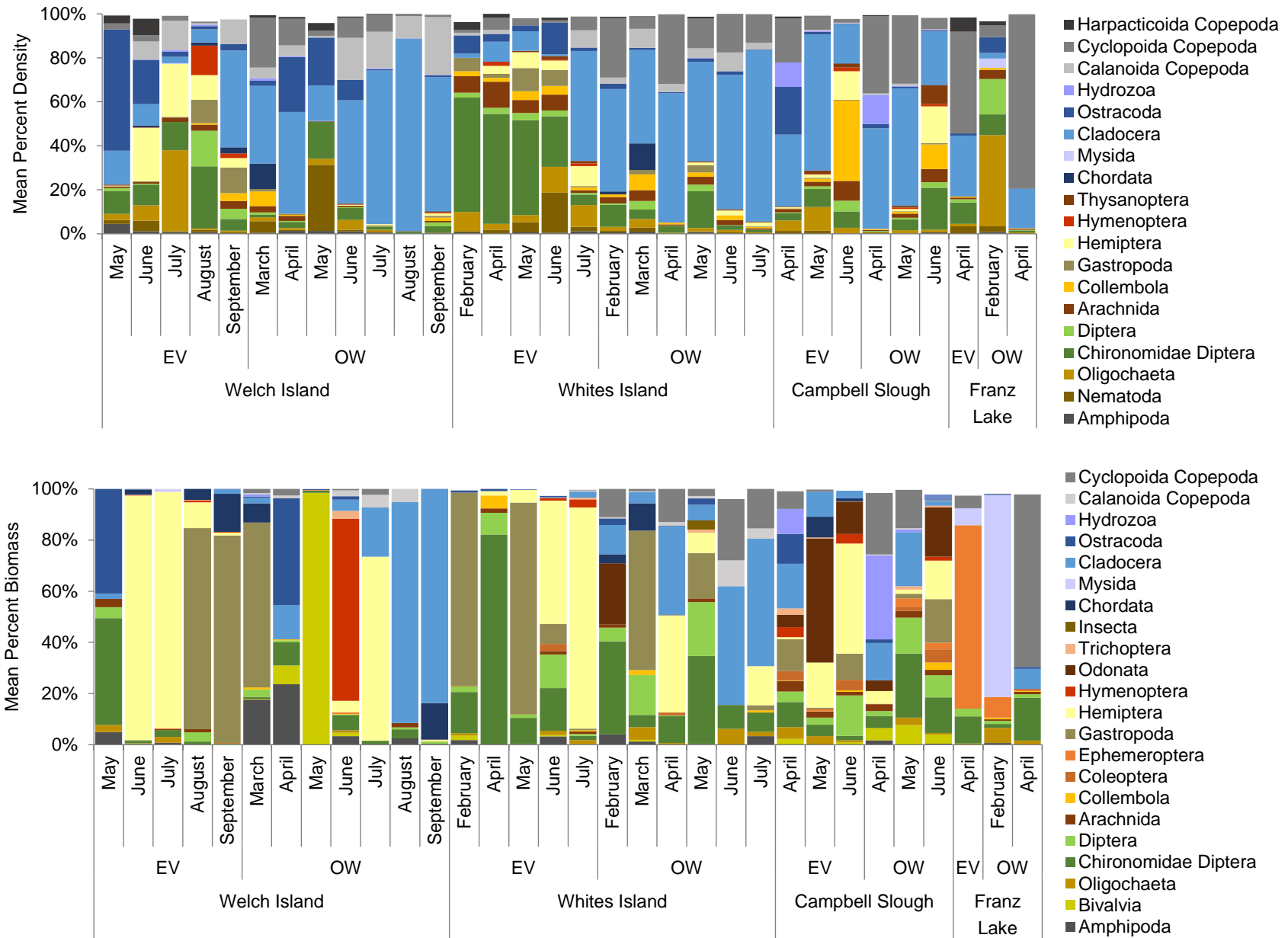
The average density of all invertebrate taxa collected by neuston tows was greater in emergent vegetation habitats than in open water habitats, except from Welch Island in August and Whites Island in April (Figure 81). Open water densities peaked at Welch Island in August at approximately 55 individuals per meter towed, equaling over six times the number of individuals collected concurrently at that site in the emergent vegetation. Cladocerans accounted for 90 percent of the open water counts at Welch Island in August. Emergent vegetation densities were greatest at Campbell Slough in June with approximately 275 individuals per meter towed, equaling just over 8.5 times the number of individuals collected concurrently at that site in the open water. This peak in density coincides with the greatest abundances of Collembola (springtails). The emergent vegetation at Franz Lake was only sampled in April; however, average density that month was at least twice as high from Franz Lake compared to other sites (200 individuals per meter towed). These counts were dominated by Cyclopoid copepods and cladocerans.

The average biomass of all invertebrate taxa collected by neuston tows was considerably greater in emergent vegetation habitats than in open water habitats, with only a few exceptions (Figure 81). Open water biomass was greatest in May at Welch Island at approximately 17 mg per meter towed. This was mainly due to a small number of bivalves (0.08 individuals per m towed). Average biomass was relatively low at Whites Island in April from both habitats. Otherwise, average biomass ranged between 7 times (Campbell Slough in April) and nearly 250 times (Whites Island in February) greater in the emergent vegetation than in the open water. Also, June biomass at Whites Island was over 200 times greater, and June and July biomass at Welch Island was over 100 times greater in the emergent vegetation than in the open water. As with average densities, average biomass in April was greatest from Franz Lake at 14.4 mg per meter towed.

While macroinvertebrate production is generally expected to increase seasonally along with warming temperatures, average biomass at Whites Island in February was much greater than in April and similar to averages in May through June. Average density at Whites Island was also higher in February than in April and May, though a significant increase in average density occurred by June. Neuston was not sampled from the emergent vegetation in March and April at Welch Island, March at Whites Island, and February at Franz Lake; therefore, comparisons between habitat types were not possible for these samples.
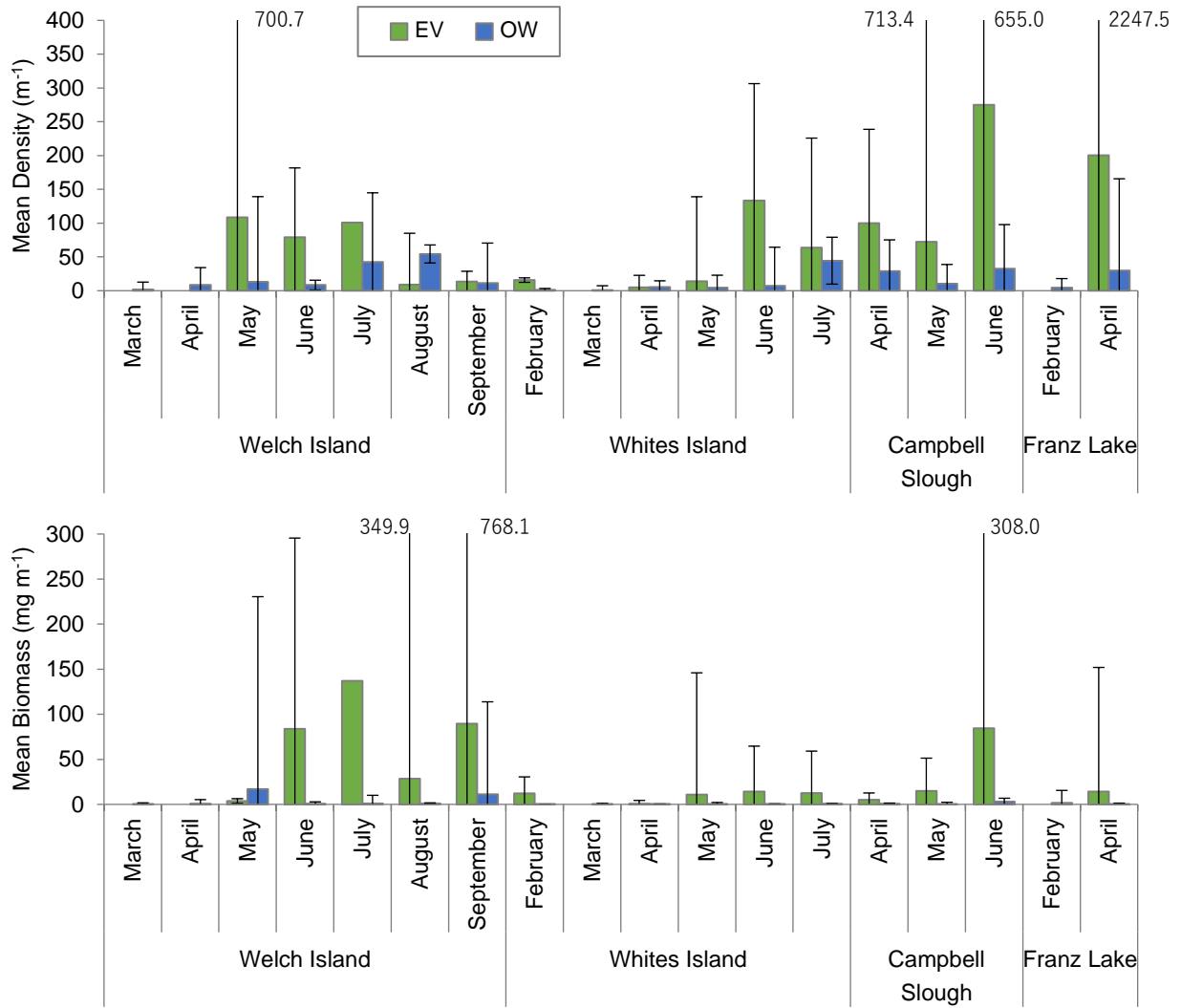
Figure 81. Average density (top) and biomass (bottom) per meter towed of all invertebrate taxa collected by neuston tow in emergent vegetation (green) and open water (blue) in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted. Emergent vegetation was not sampled in March and April at Welch Island, March at Whites Island, and February at Franz Lake. Open water was sampled at all site/month events.

Diptera, including Chironomidae, occurred at all sites on each date sampled. The average density and biomass of dipterans was consistently greater in emergent vegetation habitats than in open water habitats (Figure 82). Whites Island and Campbell Slough had the highest average dipteran density, approximately 35 individuals per meter towed in emergent vegetation, both occurring in June. Dipterans accounted for approximately 26 and 13 percent of emergent vegetation invertebrate abundances in June at Whites Island and Campbell Slough, respectively. The highest average dipteran density in open water also occurred in June at Campbell Slough at just over 7 individuals per meter towed. Dipteran densities in April were greatest from Franz Lake with an average of 22.6 individuals collected per meter towed, representing approximately 10 percent of the site's total invertebrate abundance. As was described for all macroinvertebrate taxa, February values of the average density and biomass of Diptera in emergent vegetation at Whites Island was relatively high, exceeding all other months sampled at that site, except for June.
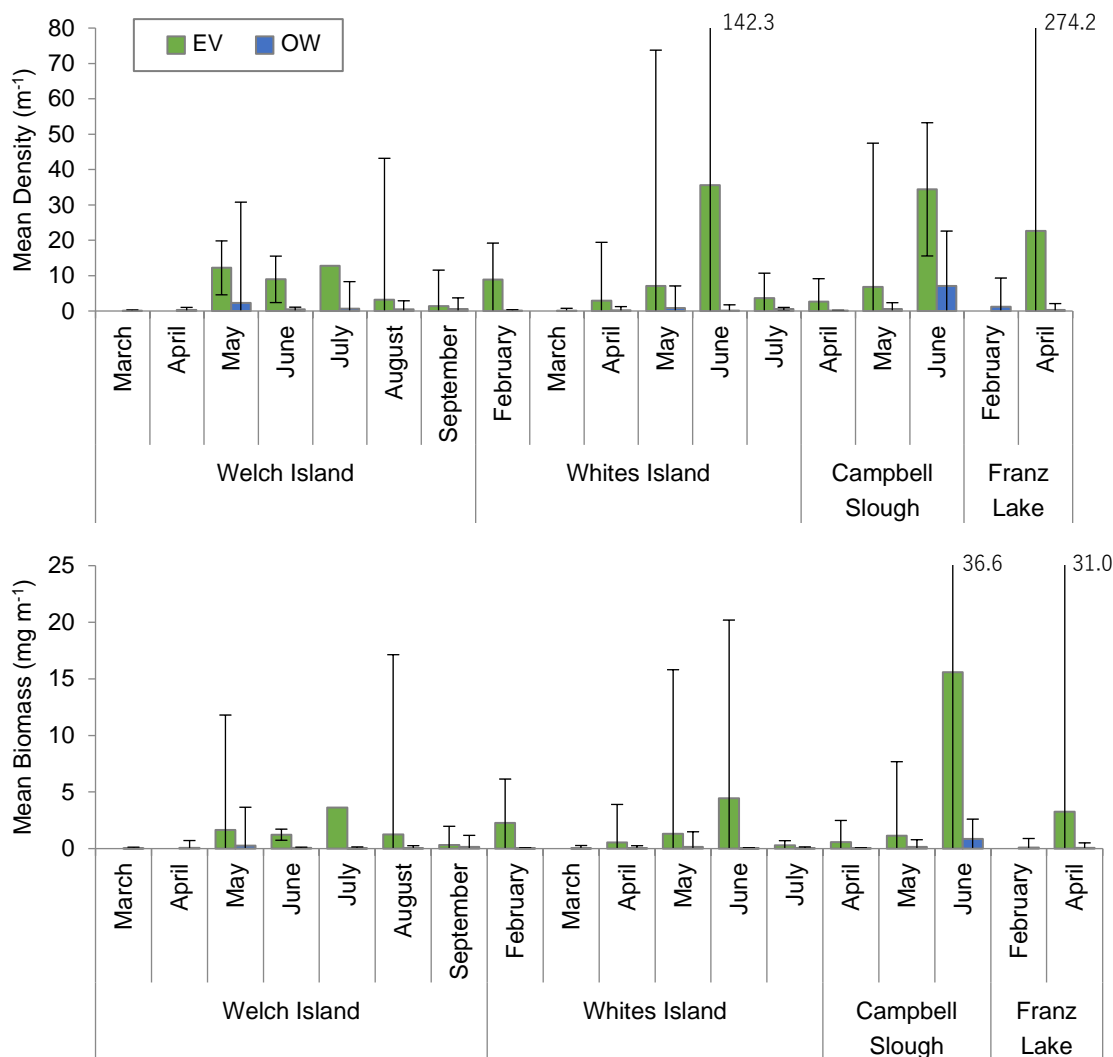
Figure 82. Average density (top) and biomass (bottom) per meter towed of Diptera (including Chironomidae) collected by neuston tow in emergent vegetation (green) and open water (blue) in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted. Emergent vegetation was not sampled in March and April at Welch Island, March at Whites Island, and February at Franz Lake. Open water was sampled at all site/month events.

Amphipods were most abundant from the emergent vegetation habitat at Welch Island (Figure 83). Even at peak occurrences in the emergent vegetation, densities of amphipods were relatively low: 7.4, 1.4, and 1.6 individuals per meter towed from Welch Island in May, June, and July, respectively. These counts represent roughly 6.8, 1.8, and 1.6 percent of the total macroinvertebrates collected from Welch Island in May, June, and July, respectively. While densities were low, amphipods were collected in the open water from all sample events, except Whites Island in May and Campbell Slough in May.

Average biomass of amphipods was greatest in July at Welch Island. Amphipods were most abundant in May however; individuals were much larger in July.
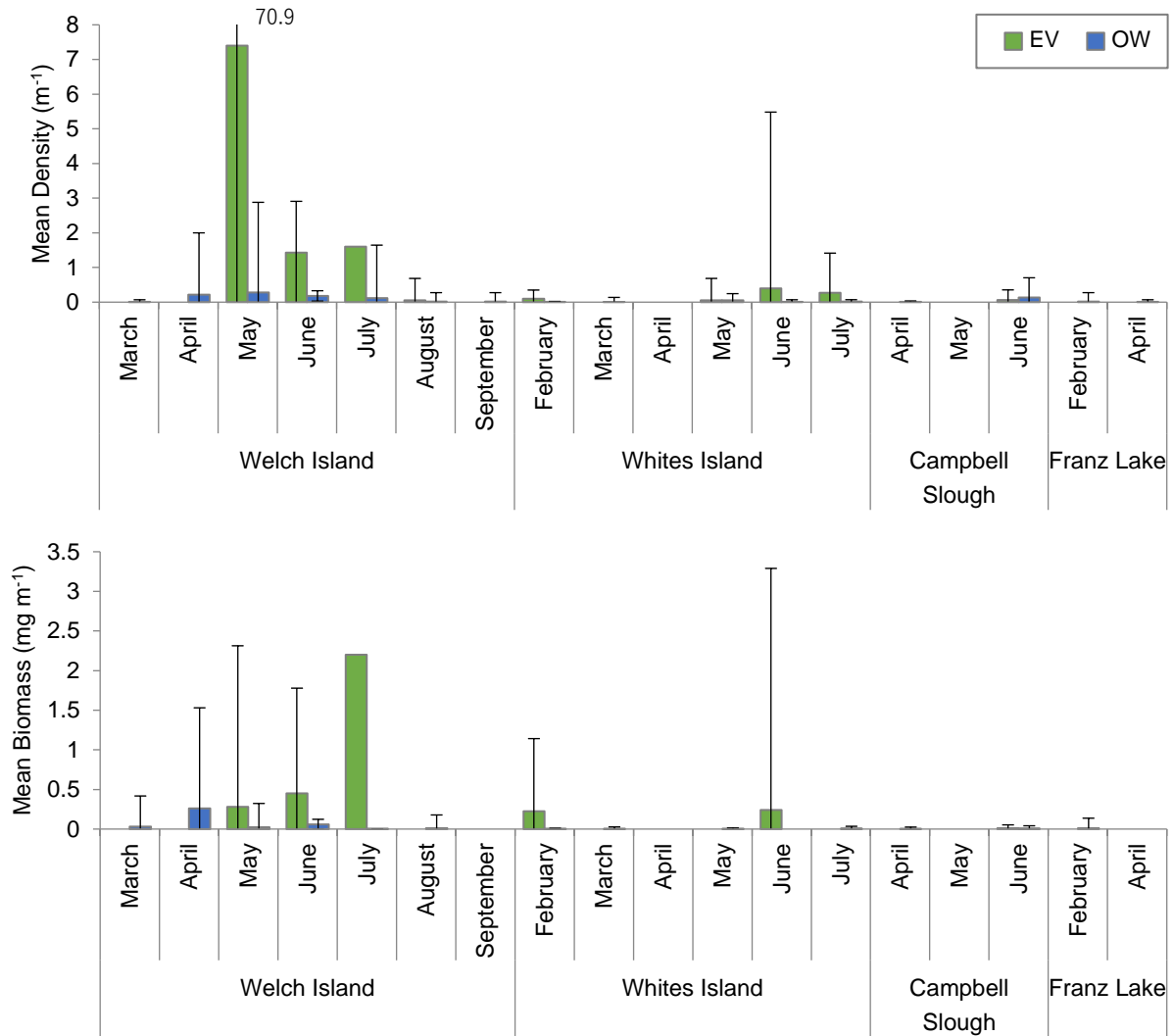
Figure 83. Average density (top) and biomass (bottom) per meter towed of Amphipoda collected by neuston tows in emergent vegetation (green) and open water (blue) in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted. Emergent vegetation was not sampled in March and April at Welch Island, March at Whites Island, and February at Franz Lake. Open water was sampled at all site/month events.

Welch Island, Whites Island, and Campbell Slough have the most comprehensive neuston tow dataset and were therefore selected for comparison. Average monthly densities (log-transformed) of all macroinvertebrates collected by neuston tow were compared among study years (2008-2013, 2015, 2016; Figure 84). Due to unequal variances and sample sizes, a Welch's ANOVA test was used to compare average densities from samples taken between April and July. Significant differences were found among years for open water samples (p=0.000). The Games-Howell post hoc test concluded average densities from 2015 and 2016 were significantly greater than all other years sampled (p=0.000). Average densities did not differ between other years, and 2015 and 2016 did not differ from each other (p>0.05). Significant differences were also found between years for emergent vegetation samples (p=0.001), though results were not as extensive as for open water densities. The Games-Howell post hoc test concluded the average density from 2015 was significantly greater than that from 2012, and average density from 2016 was significantly greater than that from 2010-2013 (p<0.05), but not significantly different than 2008, 2009, or 2015.
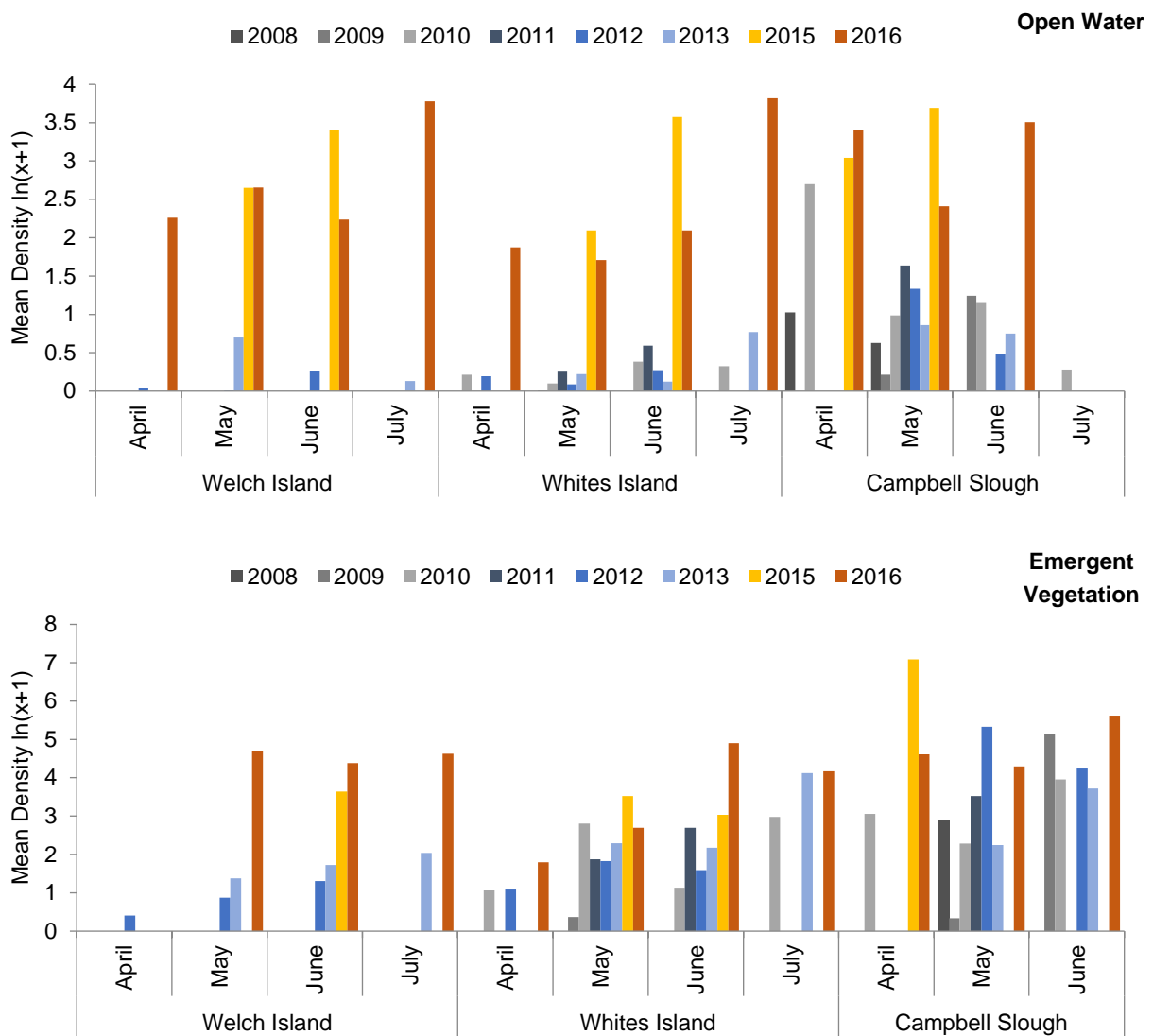
Figure 84. Comparisons across years of log-transformed average density per meter towed of all invertebrate taxa collected by neuston tow in open water (top) and emergent vegetation (bottom).

Yearly comparisons were also examined for average dipteran and cladoceran densities from Welch Island, Whites Island, and Campbell Slough samples taken between April and July. There was no significant difference in dipteran densities from open water or emergent vegetation habitats between study years (Figure 85). Significant differences in cladoceran densities were found between years for open water samples (p=0.001; Figure 86). The Games-Howell post hoc test concluded average densities from 2015 and 2016 were significantly greater than all other years sampled (p<0.05). Average density did not differ between other years, and 2015 and 2016 did not differ from each other. There was no significant difference in cladoceran densities from emergent vegetation habitats between study years.
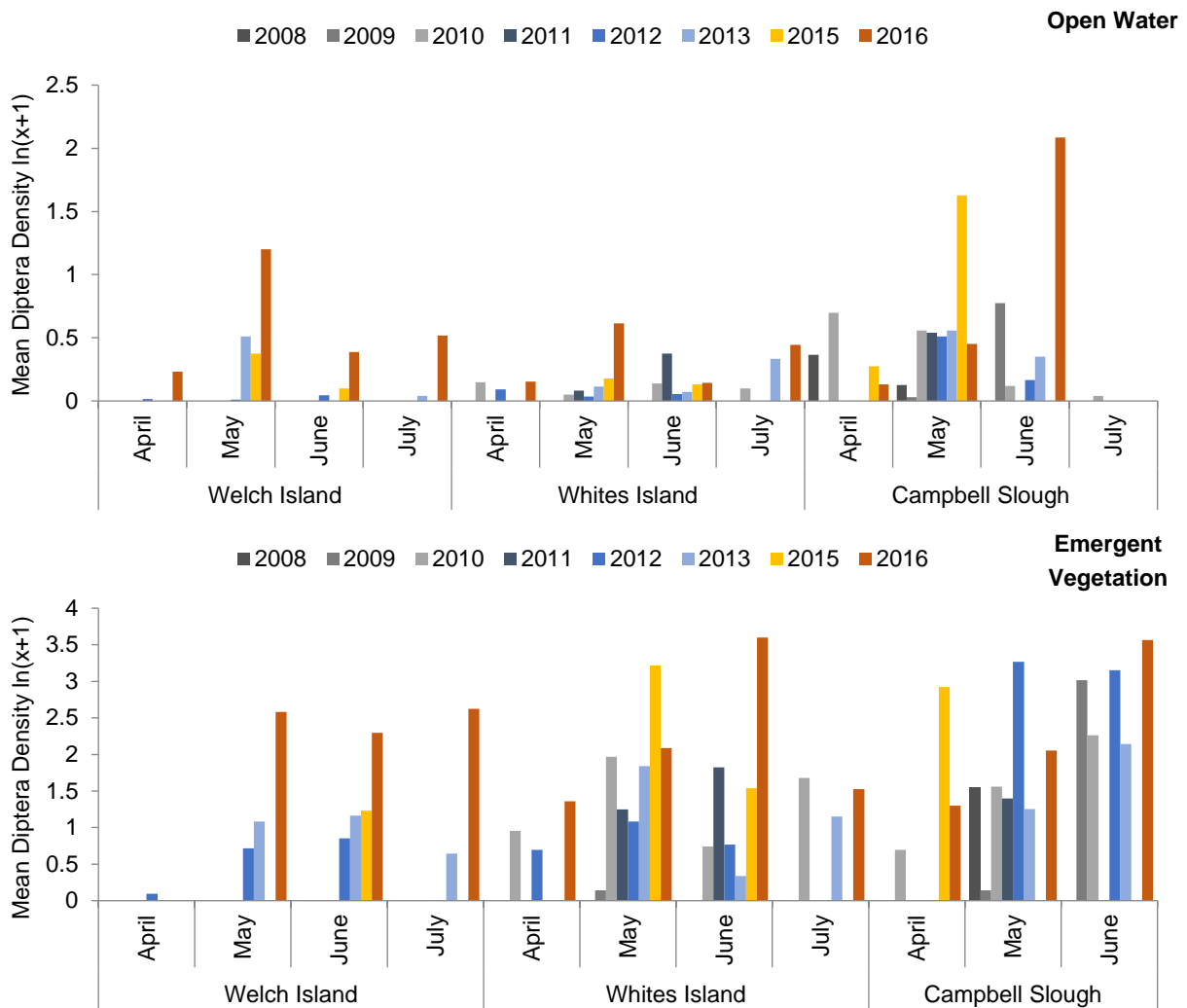
Figure 85. Comparisons across years of log-transformed average density per meter towed of all Diptera taxa collected by neuston tow in open water (top) and emergent vegetation (bottom).

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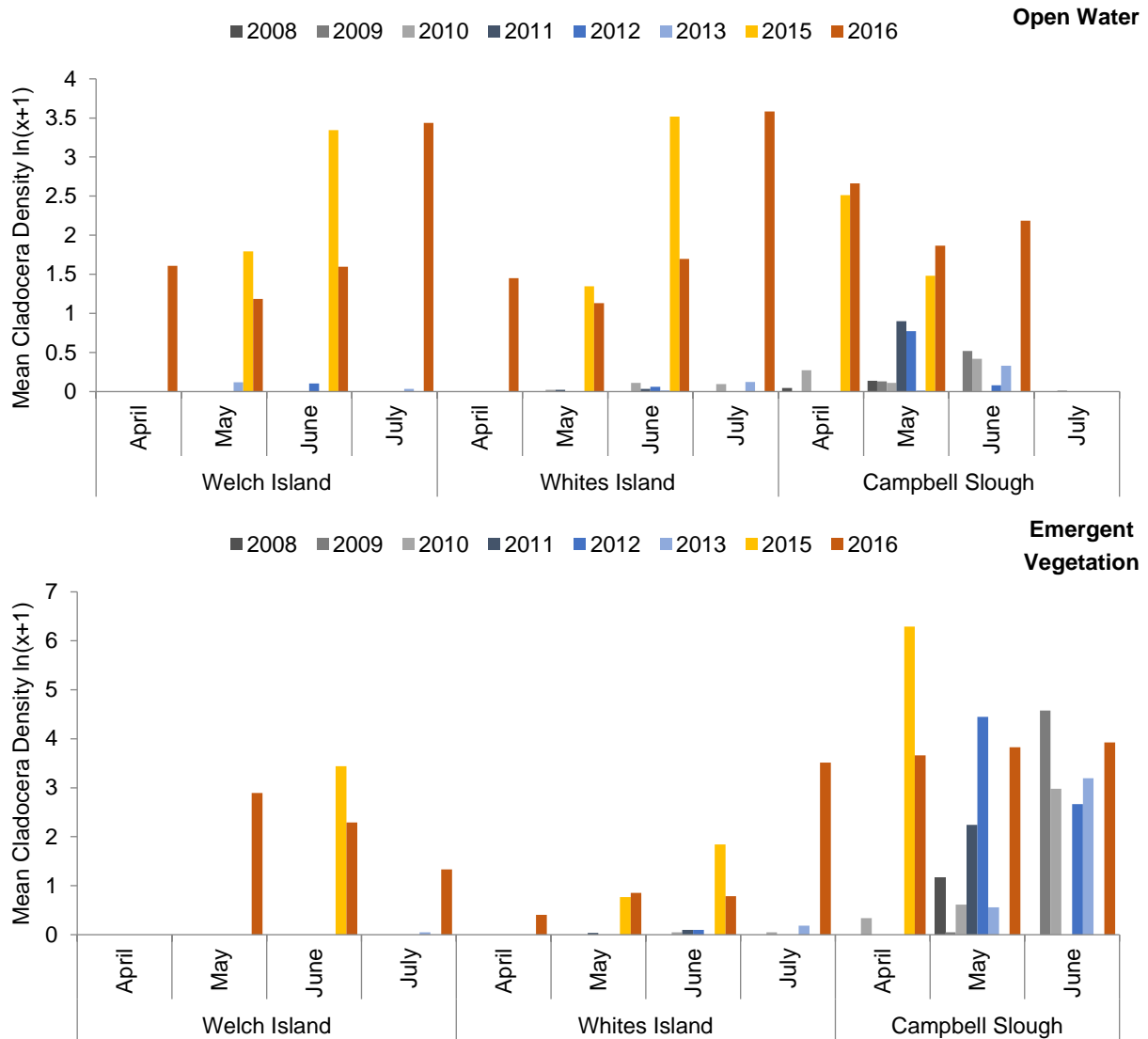
Figure 86. Comparisons across years of log-transformed average density per meter towed of all Cladocera taxa collected by neuston tow in open water (top) and emergent vegetation (bottom).

An ANOSIM test confirmed a statistical difference in the composition (log-transformed average density) of neuston tow samples between study years in the open water (R=0.459, p<0.01). Pairwise comparisons of open water samples subsequently show a significant difference in the composition of 2016 samples from all other study years except 2015 (p<0.01), as well as 2015 from all other study years except 2016 (p<0.05). Differences were primarily driven by a greater average abundance of cladocerans and copepods in the 2015 and 2016 samples (Figure 87).

An ANOSIM test also showed a significant difference in the composition (log-transformed average density) of neuston tow samples between study years in the emergent vegetation, though the difference was more moderate relative to that from open water samples (R=0.367, p<0.01). Again, there was not a significant difference between 2015 and 2016 emergent vegetation samples, but 2016 (p<0.01) and 2015 (p<0.05) did differ significantly from all other study years. Differences were primarily driven by a greater

average abundance of a number of taxa, including dipterans, cladocerans, copepods, and oligochaetes, in the 2015 and 2016 samples (Figure 88).
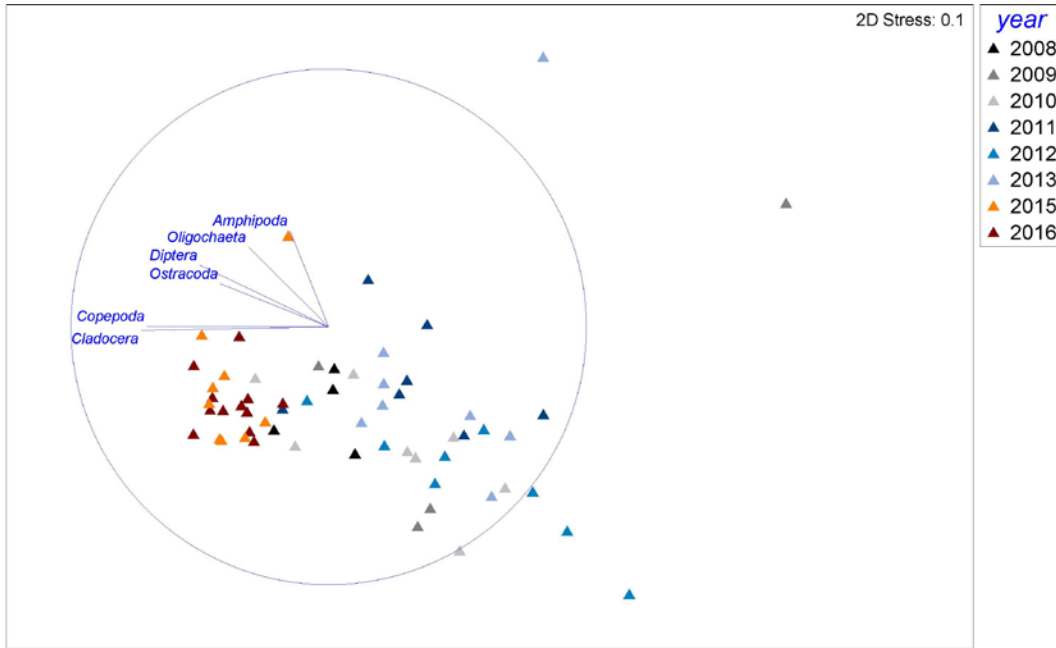
Figure 87. Nonmetric multidimensional scaling (NMDS) plot based on Bray-Curtis similarities between log transformed average monthly densities of taxa collected by open water neuston tows between 2008 and 2016. Significant correlation with variables (Pearson R > 0.4) are represented as blue vectors.
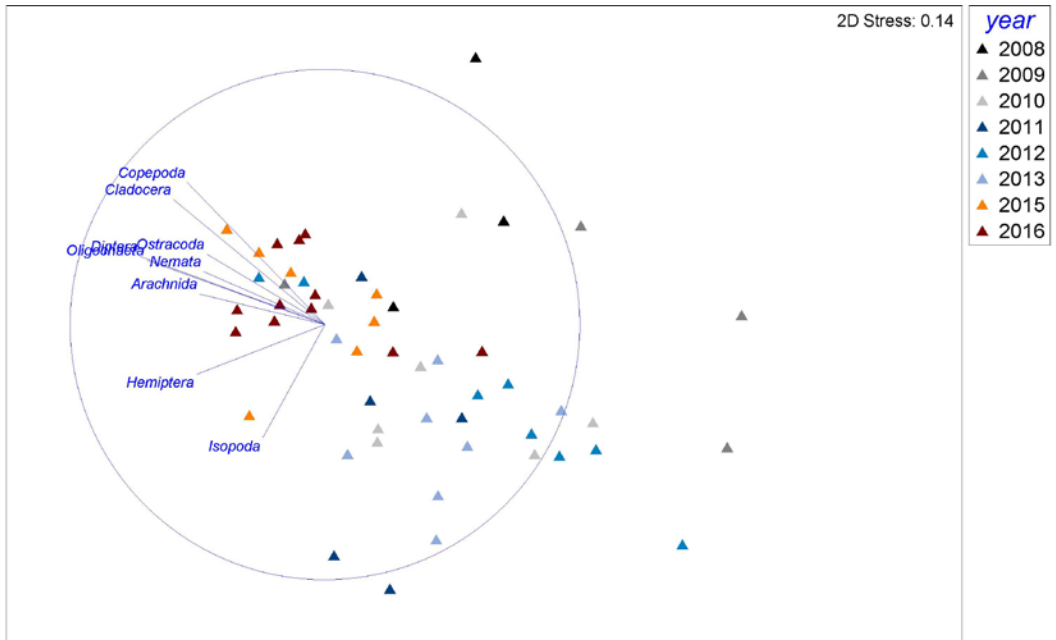
Figure 88. Nonmetric multidimensional scaling (NMDS) plot based on Bray-Curtis similarities between log transformed average monthly densities of taxa collected by emergent vegetation neuston tows between 2008 and 2016. Significant correlation with variables (Pearson R > 0.5) are represented as blue vectors.

3.5.2 Salmon Diet

Salmon diet data collected in 2017 are still under analysis and unavailable at the time of the writing of this report. The following details are adapted from the 2016 report.

3.5.2.1 *Salmon Diet*

Salmon diet composition, as measured by an index of relative importance (IRI), varied within site across months as well as across sites (Figure 89). Amphipods and dipterans (primarily Chironomidae) comprised most of the index at Welch Island in all months and made significant contributions at Whites Island. Cladocerans were also common in diets at Welch Island and Whites Island in April, while hemipterans (plant hoppers) were common at Whites Island in July. The increase in importance of hemipterans at Whites Island coincides with their greatest contribution to the percent density and biomass of neuston samples in July.

Campbell Slough diets were dominated by insect taxa, particularly odonates (damselfly nymphs) in April and chironomids in May. Juvenile Chinook were only collected from Franz Lake in April; these diets were dominated by Cyclopoid copepods.
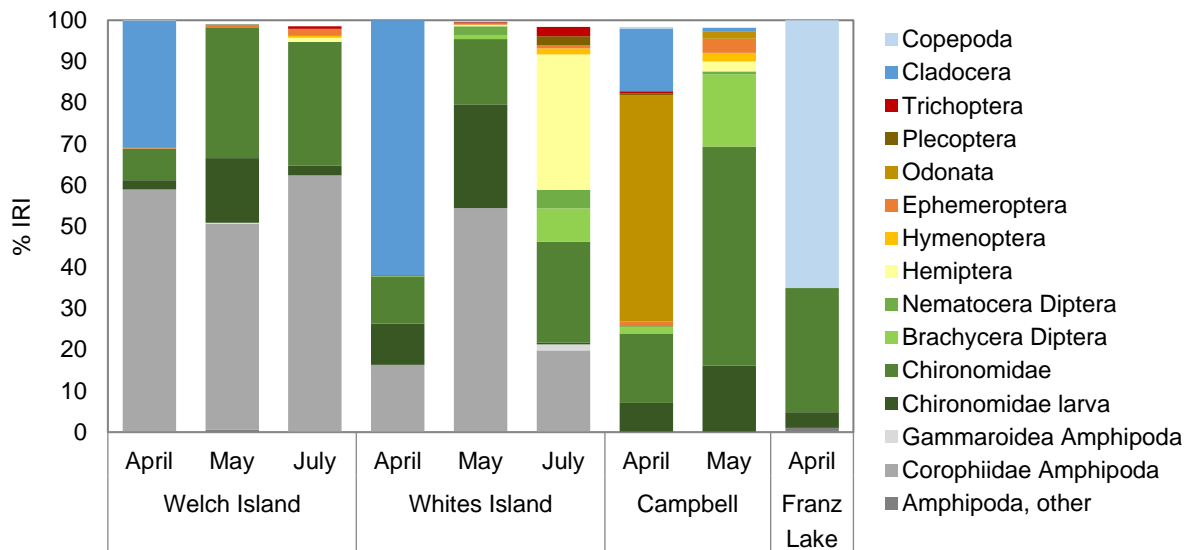
Figure 89. Contribution of prey to juvenile Chinook salmon diets by percent IRI in 2016.

Percent IRI in 2016 was compared to juvenile Chinook diets sampled in previous years (2008-2013, and 2015; Figure 90). Distribution of these observations in an NMDS plot shows a separation between each of the April 2016 samples and previous years. Welch Island and Whites Island samples both have greater contributions from cladocerans in April 2016, though are similar to diets from Whites Island in May 2013. April 2016 diets from Campbell Slough are distinguished by odonates and other insects, while a high proportion of copepods distinguish Franz Lake diets from previous years. The NMDS plot also illustrates a clear separation of the more downriver sites (Welch Island and Whites Island, blue symbols) and the more upriver sites (Campbell Slough and Franz Lake, green symbols). An ANOSIM test confirms a statistical difference in prey consumption between sites (R=0.416, p<0.001). Pairwise comparisons subsequently show a significant difference between upriver and downriver sites (p<0.001), but no significant difference within the two downriver sites or within the two upriver sites. Diptera, including Chironomidae, contribute to diets from all of the sites analyzed, though the percent contribution is greater

on average from Campbell Slough and Franz Lake. Amphipods, on the other hand, were strongly associated with Welch Island and Whites Island.
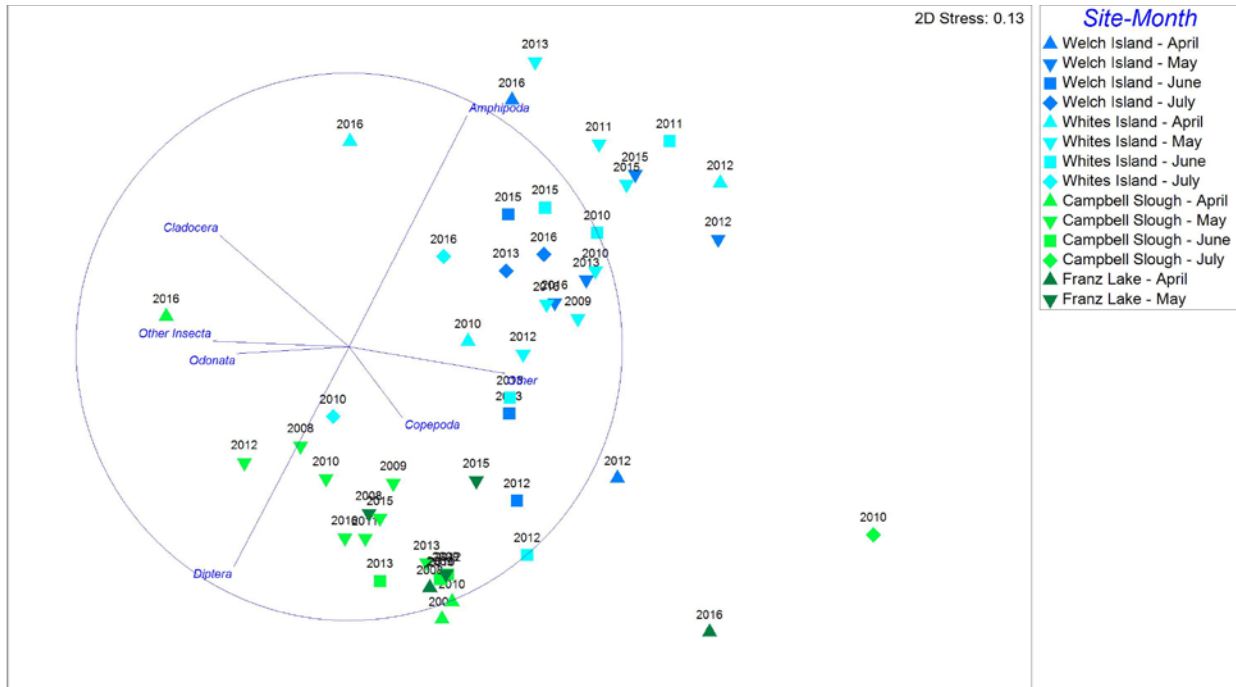
Figure 90. Two-dimensional nonmetric multidimensional scaling (NMDS) plot based on Bray-Curtis similarities between transformed percent IRI of major prey groups in diets sampled between 2008 and 2016. Significant correlation with variables (Pearson R > 0.2) are represented as blue vectors.

The energy density of major prey items ranges from a high of 11.7 kJ g^{-1} wet mass for hymenopterans (bees, wasps, ants) to a low of 1.4 kJ g^{-1} wet mass for cladocerans (Figure 91). The Corophiidae family of amphipods made up the greatest total mass consumed by juvenile Chinook salmon in 2016, followed by Chironomidae and Odonata. Even though the energy density of Corophiidae is typically less than that of adult and emergent insects, juvenile Chinook overall gained the most energy from this prey item given the amount of biomass consumed.
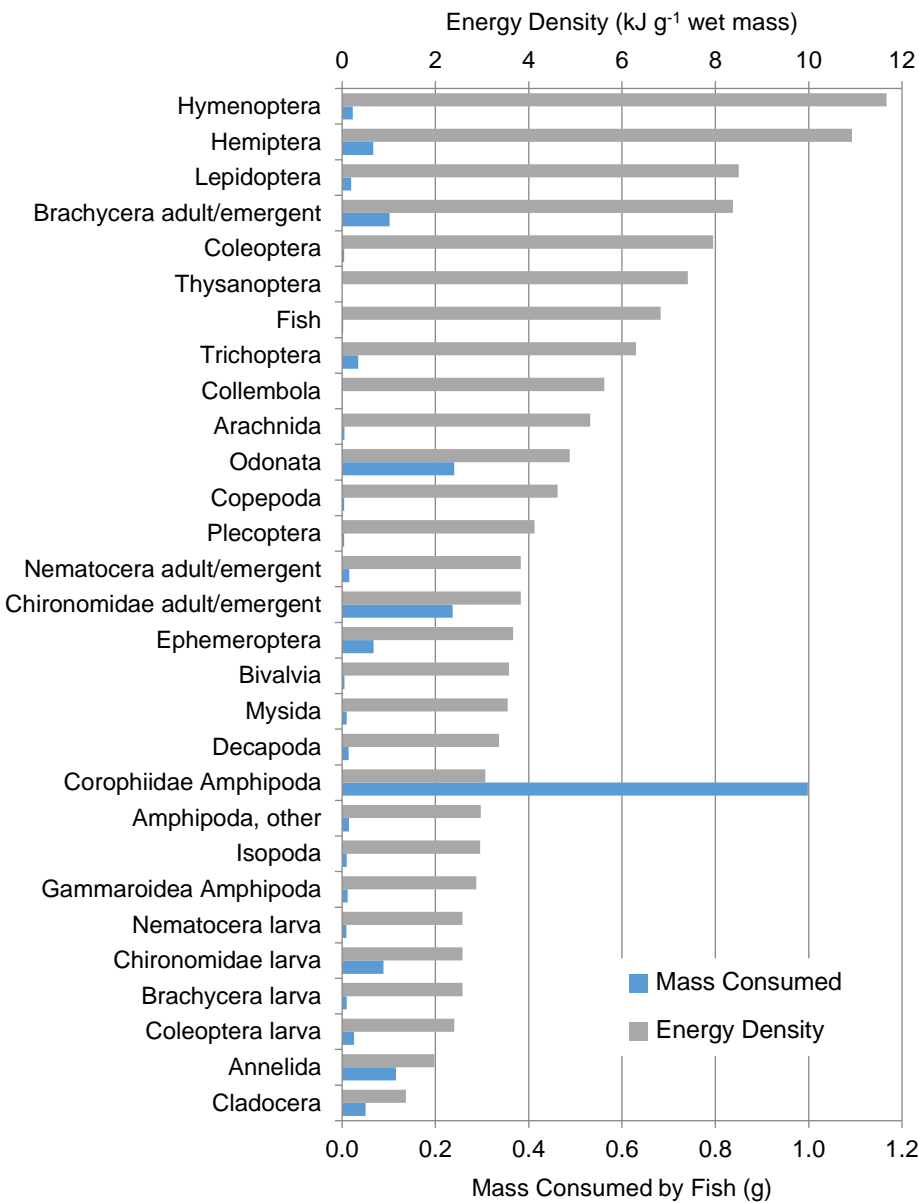
Figure 91. Energy density of individual prey taxa and total mass consumed by all juvenile Chinook salmon sampled in 2016. Energy densities were acquired from the literature and compiled in David et al. (2016).

Figure 92 compares the composition of prey in regards to their contribution to the IRI and to the energy density consumed by juvenile Chinook salmon. While cladocerans and copepods are at times numerically abundant in the diets and therefore contribute substantially to the IRI, they typically comprise a much lower proportion of the gravimetric composition. This results in a reduced contribution to the total energy density. For example, copepods made up nearly 65 percent of the IRI for juvenile Chinook salmon at Franz Lake in April, but just over 20 percent of the energy consumed. While not the dominant prey item, chironomids provided the majority of energy to fish feeding at Franz Lake. Many insect taxa, on the other hand, are typically not numerically abundant, but are relatively large-bodied and energy dense, and therefore contribute a greater proportion to the energy consumed than to the IRI.
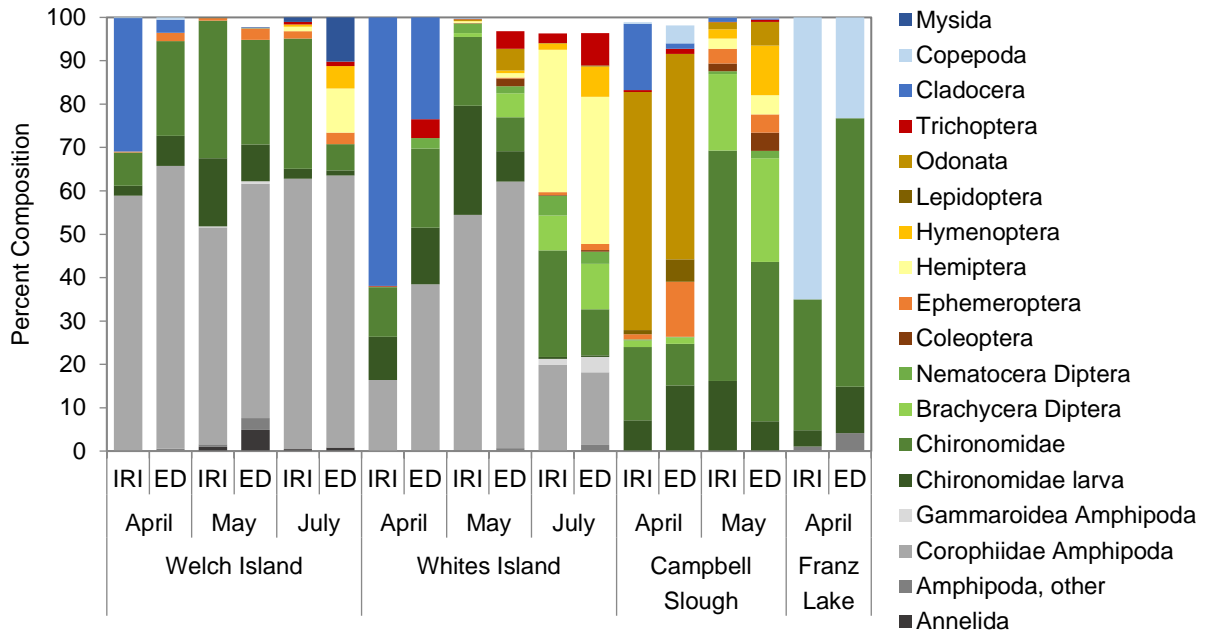
Figure 92. Comparison of the average percent composition of prey contributing to the total IRI and energy density (ED) consumed by juvenile Chinook salmon in 2016.

Mean juvenile Chinook salmon instantaneous ration (IR) and energy ration (ER) were significantly lower at Franz Lake in 2015 and 2016 compared to other sites (Welch ANOVA and Games-Howell post hoc test, p<0.001; Figure 93).
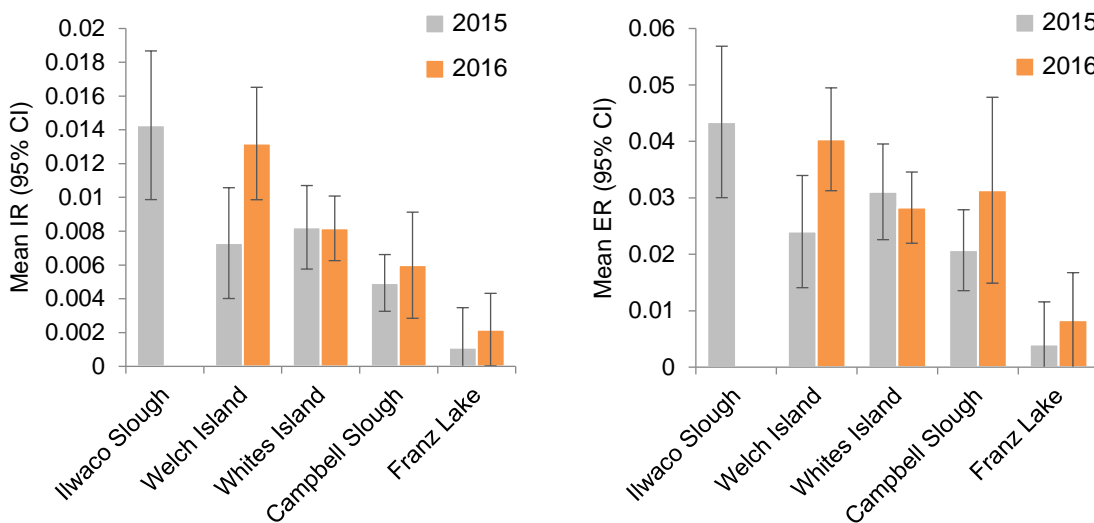
Figure 93. Feeding rates of juvenile Chinook salmon in 2015 and 2016 presented as average instantaneous ration (IR, left) and average energy ration (ER, right).

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Following methods in Fiechter et al. (2015), maintenance metabolism (J_M) was calculated for all juvenile Chinook salmon used in diet analyses from 2015 and 2016 (Figure 94). Maintenance metabolism increases with higher temperatures and with increases in fish size, such that larger fish in warmer temperatures would have higher metabolic needs. Sites generally had similar measures of maintenance metabolism within months and size classes. May temperatures at Franz Lake were on average considerably lower than at the other sites resulting in a lower maintenance metabolism for the fish sampled, all of which were within the 50-70 mm size range.
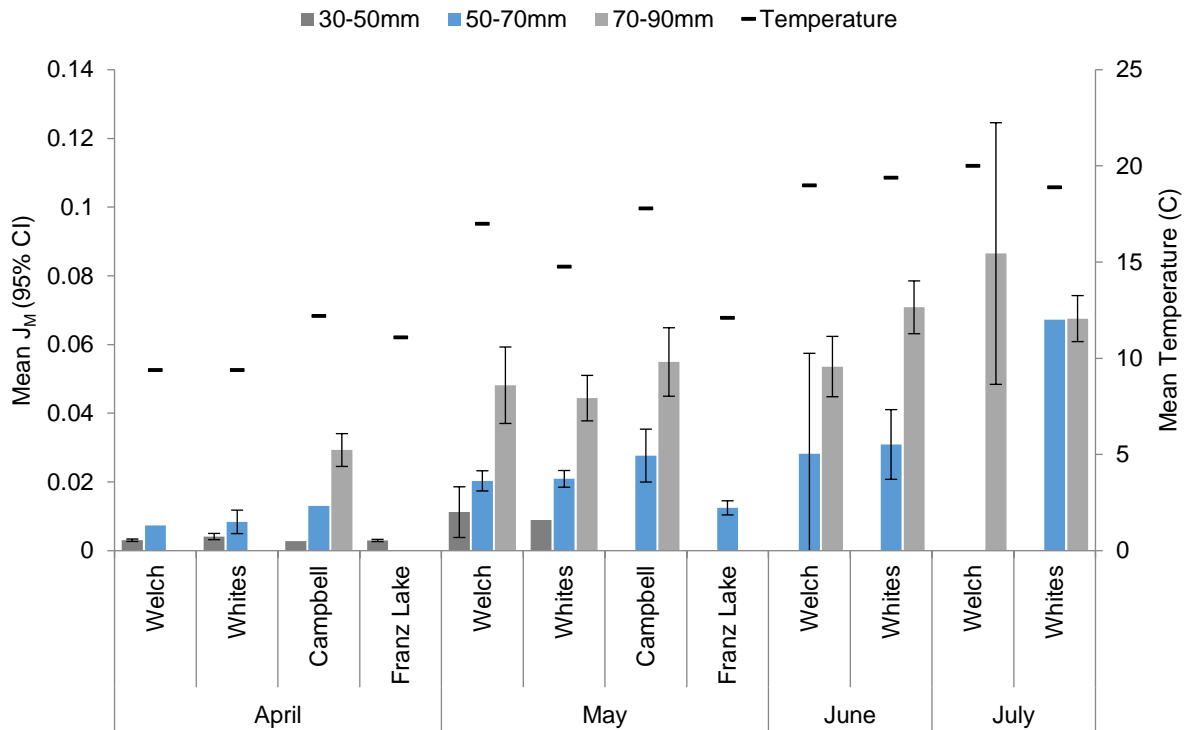
Figure 94. Average maintenance metabolism (J_M) by fish length bin, site, and month for juvenile Chinook salmon sampled in 2015 and 2016. Maintenance metabolism, which increases under unfavorable conditions, is compared to average water temperature (dashed line) at time of capture.

Combining measures of ER and J_M provides a general assessment of habitat quality and juvenile Chinook salmon growth potential at a given site and month. Figure 95 provides a graphic representation of this, plotting average maintenance metabolism against average energy ration for all fish sampled between 2008-2013, and 2015-2016, arranged by size (length) bins of fish. Samples with high energy assimilation and low metabolic costs (lower right quadrant) reflect conditions 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 (in the 30-50 mm length range) at all sites between February and April, but energy ration varied. Energy ration for small fish at Franz Lake in April was well below the 50th percentile, while those from Welch Island in February had the highest observed ratio. The small fish at Welch Island experienced on average good conditions from February to April, but in May and June had low energy ration and high maintenance metabolism, reflecting a decline in growing conditions over the season. Energy ration at Whites Island remained fairly steady from March to June for

30-50 mm fish, ranging between 0.03 and 0.05. However, as temperatures increased over that period so did the cost of metabolic upkeep indicating an added strain to the fish.

Maintenance metabolism was relatively low for medium-sized juvenile Chinook salmon (50-70 mm length) between March and May at all sites except at Campbell Slough in May. Energy ration over these months varied, and as was seen for the smaller fish, medium sized fish from Franz Lake had the lowest average energy ration. Medium sized fish from Campbell Slough in May, and Whites Island in July and August, had above average metabolic costs, but also the highest levels of energy assimilation. This demonstrates a trade-off fish may experience during summer months where temperatures increase but the more energy dense prey, such as hemipterans, hymenopterans, and other insect taxa, become available and are more often consumed.

Large sized juvenile Chinook salmon (70-90 mm length) from Campbell Slough in April had high energy assimilation and low metabolic costs, reflecting favorable growing conditions. Large fish were collected from Franz Lake in April and May. Like the other size classes, they had relatively low metabolic costs but did not consume energy dense prey. As shown for medium sized fish, large fish from Whites Island in July and August and from Welch Island in July may be experiencing a tradeoff of relatively higher temperatures (and metabolic needs) coinciding with peak emergence of energy dense prey.
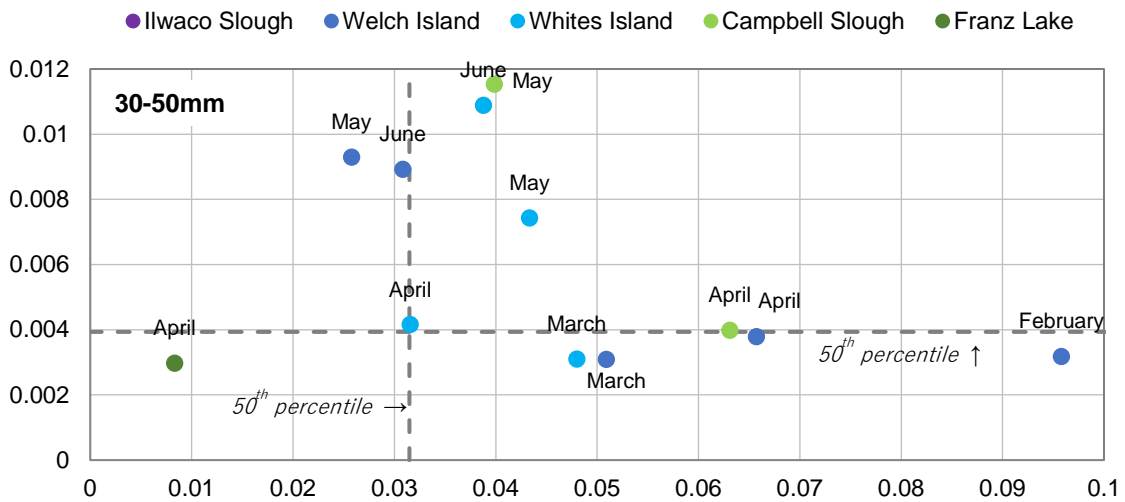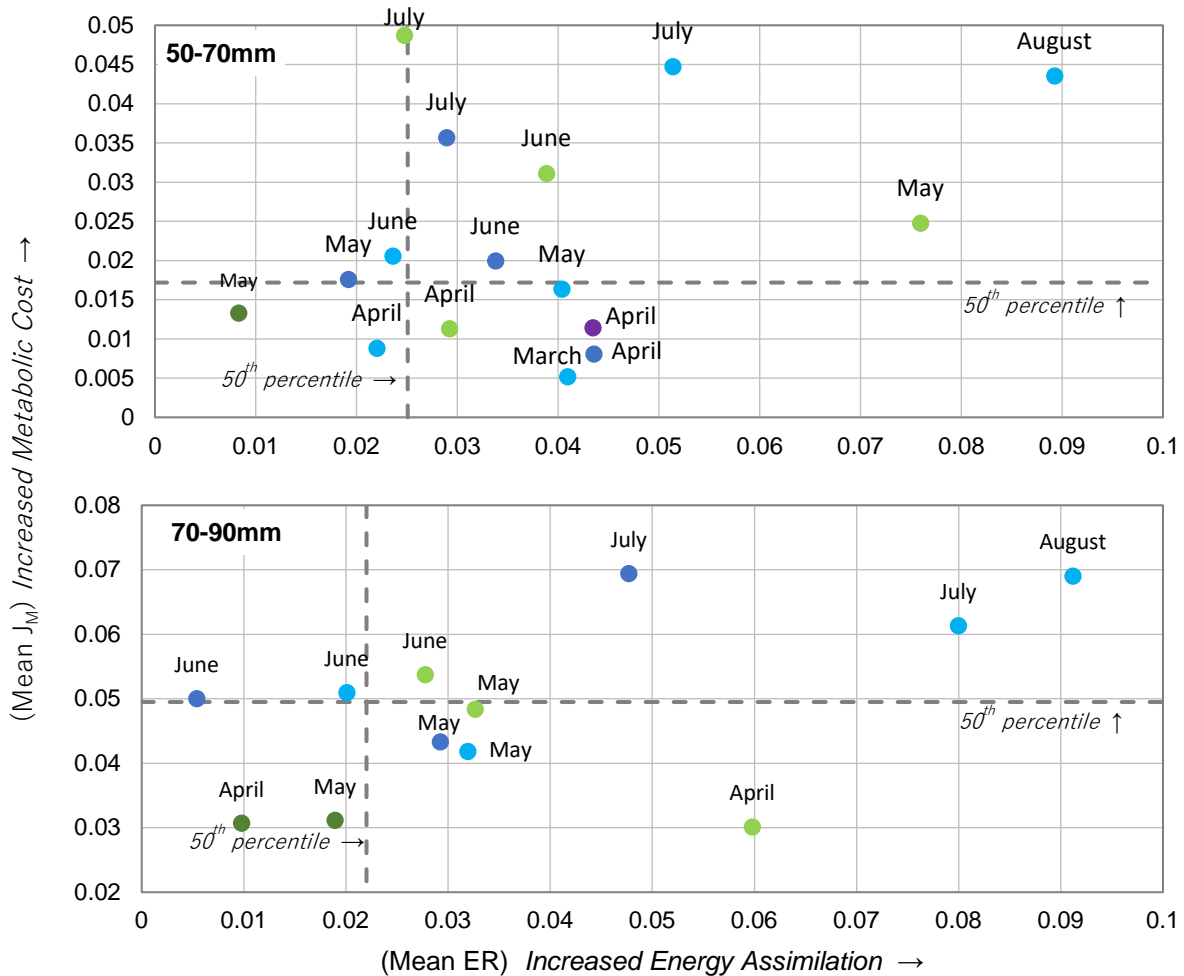
Figure 95. Quadrant charts of average maintenance metabolism (J_M) and average energy ration (ER), arranged by length bin of juvenile Chinook salmon sampled between 2008-2013, and 2015-2016. Dashed line is the 50th percentile of J_M and ER. The lower right quadrant represents conditions beneficial for juvenile Chinook growth (higher energy ration, lower metabolic needs), while the upper left quadrant represents conditions less conducive to juvenile Chinook growth (lower energy ration, higher metabolic needs).

3.6 Fish

3.6.1 Fish Community Composition

Yearly fish sampling conducted at the five trend sites (some since 2008), has included 16 different families and within those families 48 different categories that include a combination of specific species and/or unidentified species within family/genus categories. (Figure 88) (Appendix F). Threespine sticklebacks, which are present at every sample site dominate catches at Ilwaco Slough, Welch and Whites Islands. Campbell Slough and Franz lake had the highest diversity and were dominated by Cyprinids (9 species) and Fundulidae (one species, banded killifish). Ilwaco Slough, the only site with marine influence is also dominated by Embiotocidae (shiner perch) and estuarine Cottidae (Pacific staghorn sculpin). The family Salmonidae is found at each sample site and the most common species is Chinook salmon followed by chum and coho, respectively. Although seven salmonid species are represented in the study, each species does not occur at every site.

In 2017, fish communities were sampled at four trend sites (Ilwaco Slough, Welch Island, Whites Island and Campbell Slough) on a monthly basis February–June and September. No sampling occurred at the fifth trend site Franz Lake due to the lack of access caused by high Columbia River flow conditions in the spring and a forest fire in the summer. The fish community at Ilwaco Slough was comprised of three species that were equally represented (29-31%), Pacific staghorn sculpin, shiner perch and threespine stickleback, in addition Chinook salmon represented 7.5% of the total catch. The fish communities at Welch and Whites Islands were dominated by the same species in nearly equal numbers threespine stickleback (72 and 74%), Chinook salmon (15 and 11%) and banded killifish (11 and 14%). At Campbell Slough dominant fish catches were slightly more equally distributed between threespine stickleback (45%), banded killifish (23%), Chinook salmon (19%) and yellow perch (9%). Campbell Slough had the highest species diversity (13), followed by Whites (10), Welch (9) and Ilwaco (7). Campbell also had the highest catches of introduced species (Appendix F).

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 89). 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 90), where the values can vary minimally, however tend to be the highest at Campbell Slough and Franze Lake. Within site comparisons, the only standout difference was at Campbell Slough where the 2017 species richness and diversity were noticeably lower than any previous years.

In 2017, the monthly sampling at each site does not show any distinct pattern of species richness and diversity (Figure 91). The number of species each month ranges from 0-6, with the exception of Campbell Slough in June where the total number of species reached 11. Campbell Slough also had the greatest range in diversity (0.2-1.2) which reflects the higher number of species and the overall total catch of those species.

Non-native fish species do 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 92). The highest number of non-native fishes occur at Campbell Slough where the catch rates have exceeded 50% six out of the last 10 years, and range from 40-75%. 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 93). 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 2017, yellow perch comprised 98% of the total number of predatory fish captured.
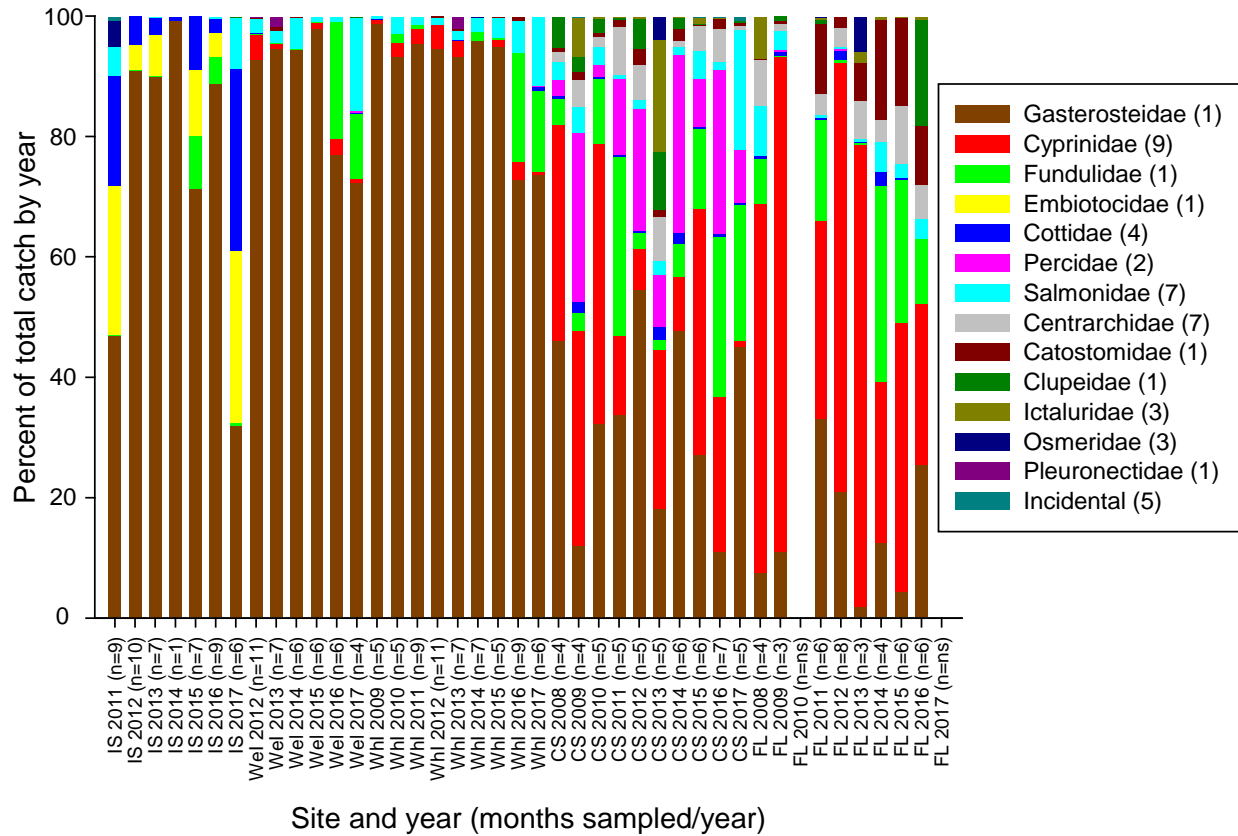
Figure 88. Fish community composition at EMP trend sites sampled from 2008-2017, presented by Family with the number of major taxonomic categories in parentheses in the legend. For each year the total number of sampling months is presented in parentheses on the X axis. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
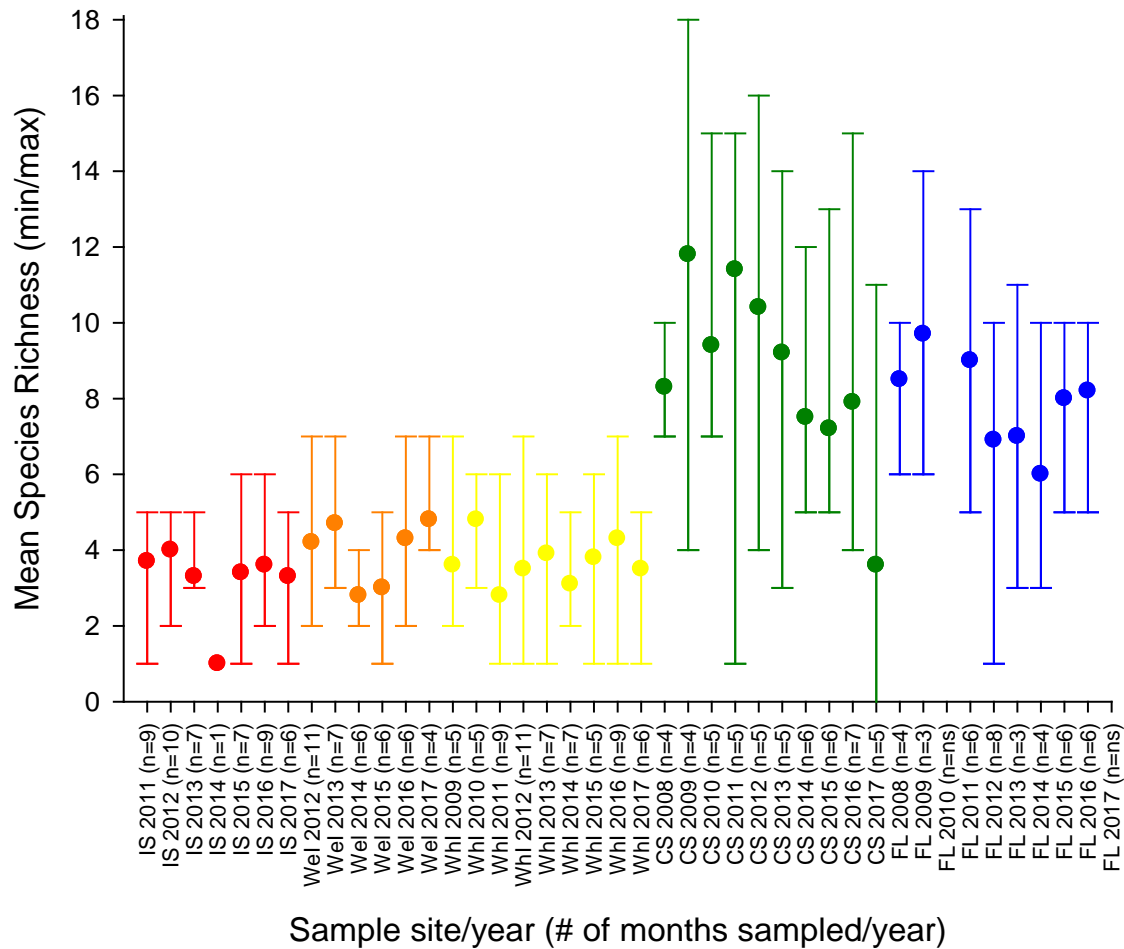
Figure 89. Mean species richness with minimum/maximum ranges for EMP trend sites sampled from 2008-2017. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
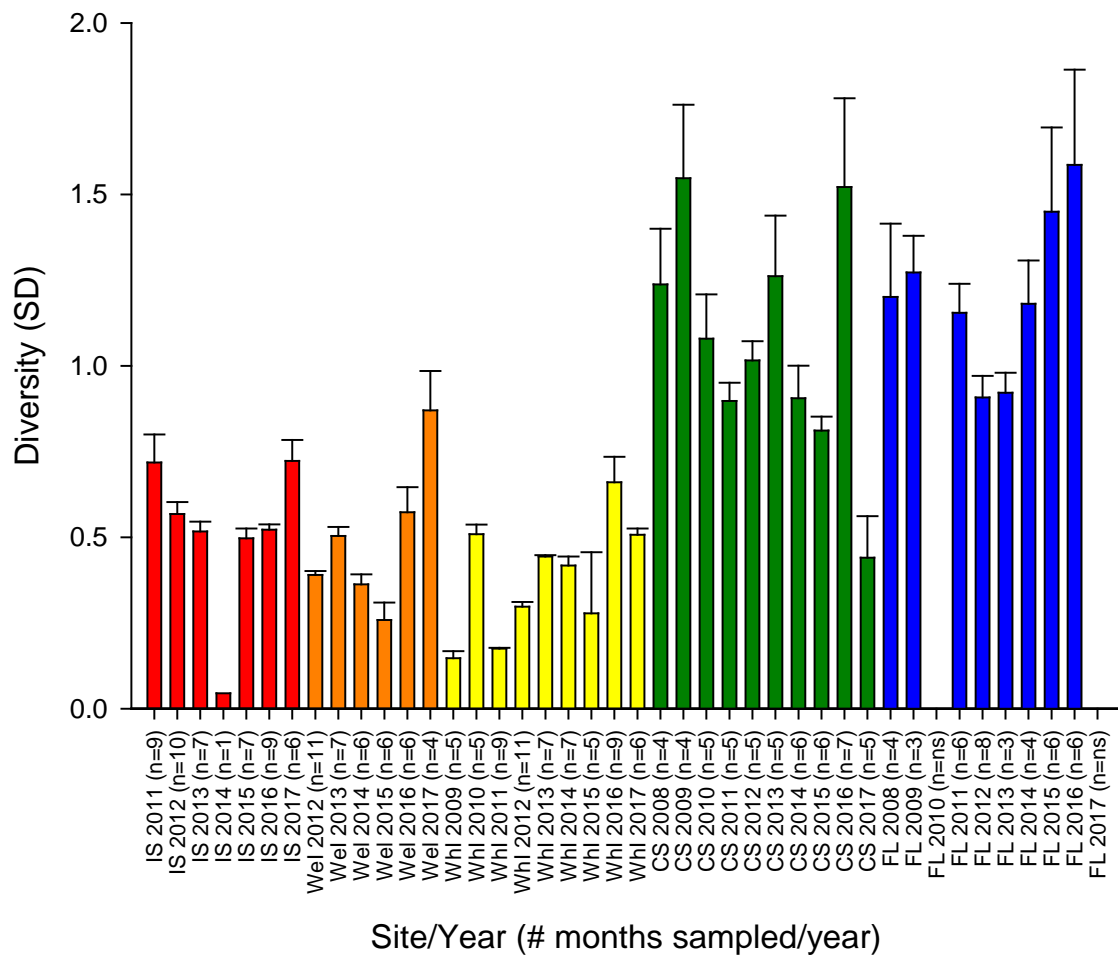
Figure 90. Mean Shannon-Weiner diversity index with standard deviation from EMP trend sites sampled from 2008-2017. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
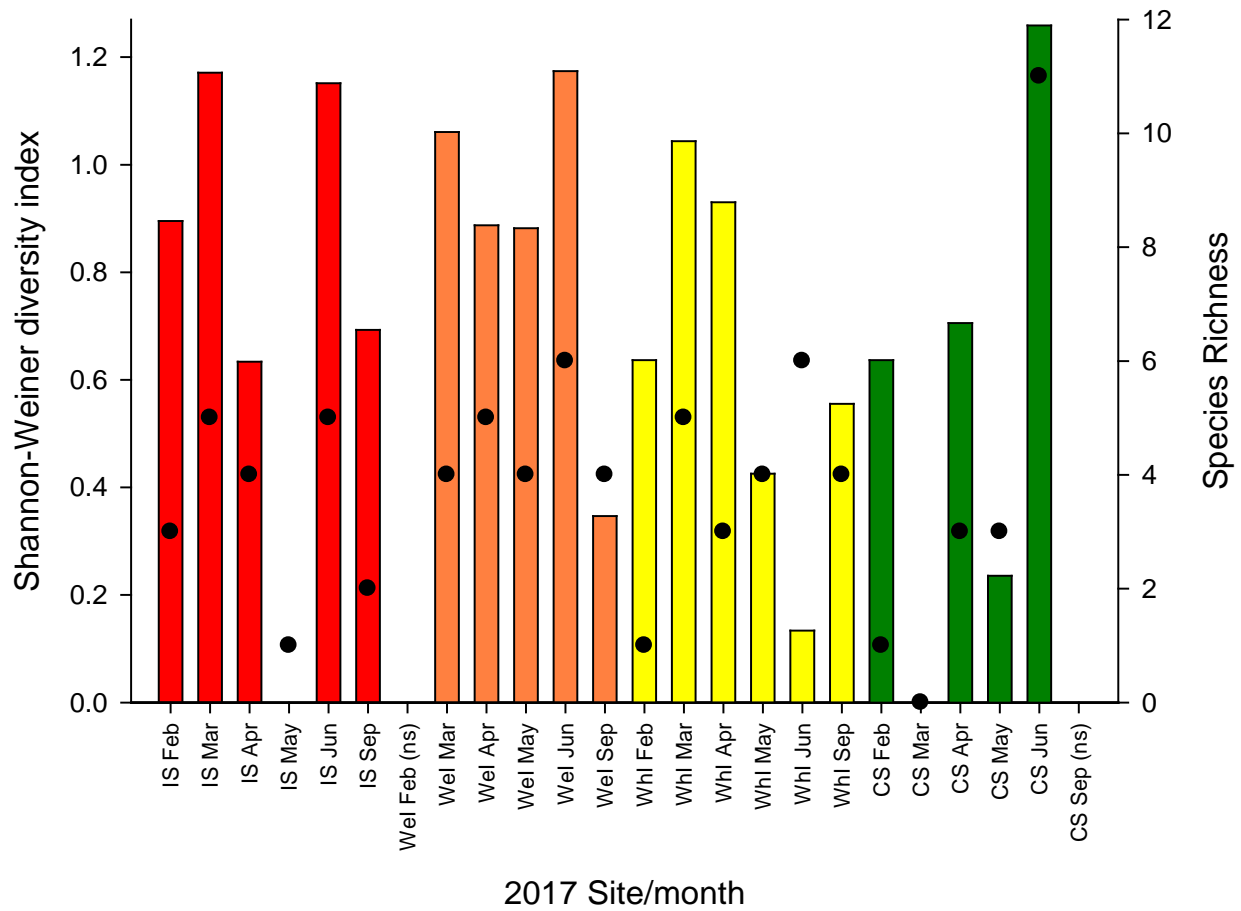
Figure 91. Shannon-Weiner diversity index (bars) and species richness (closed circles) for EMP trend sites sampled monthly in 2017. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough. No sampling was conducted at Franz Lake in 2017.
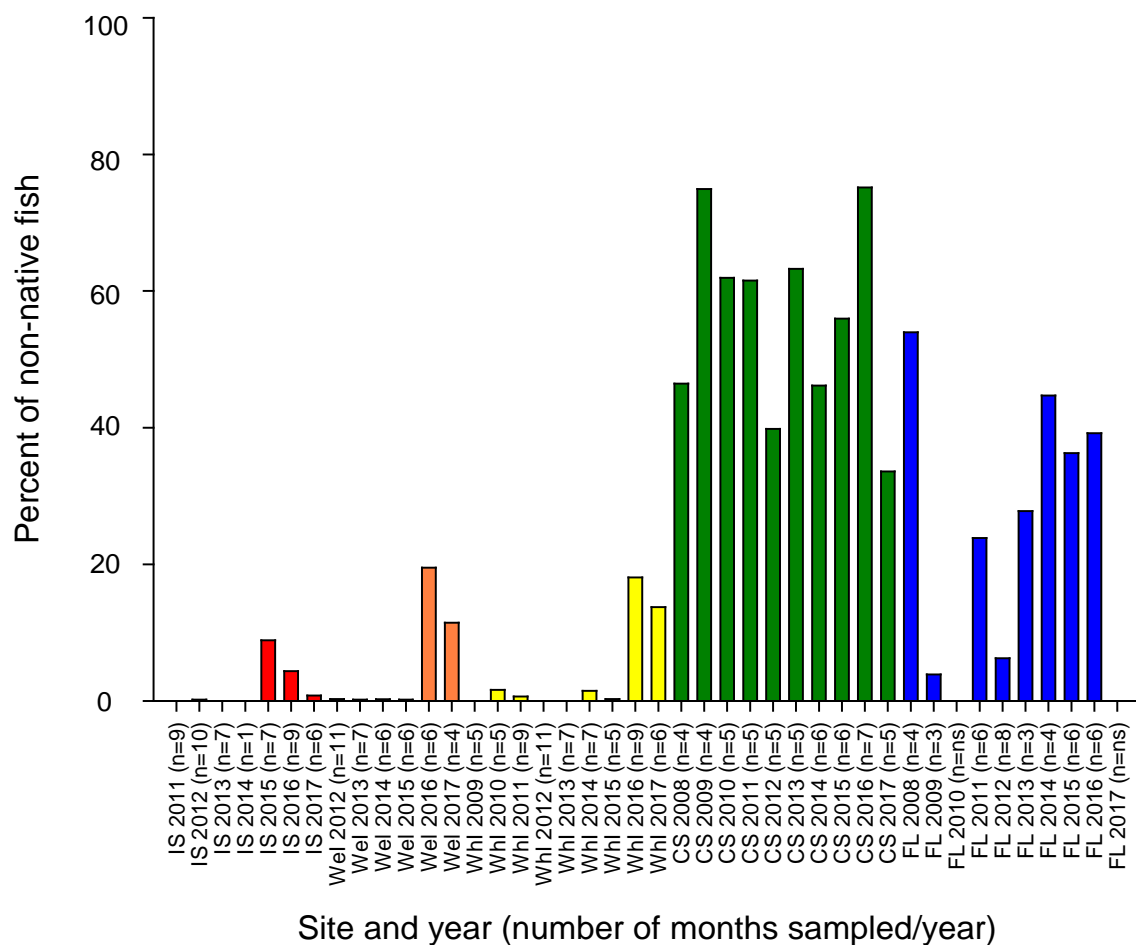
Figure 92. Percent of total fish catches per year that are non-native species for EMP trend sites sampled in 2008-2017. For each year the total number of sampling months is presented in parentheses. IS = Ilwaco Slough, WeI = Welch Island, WhI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
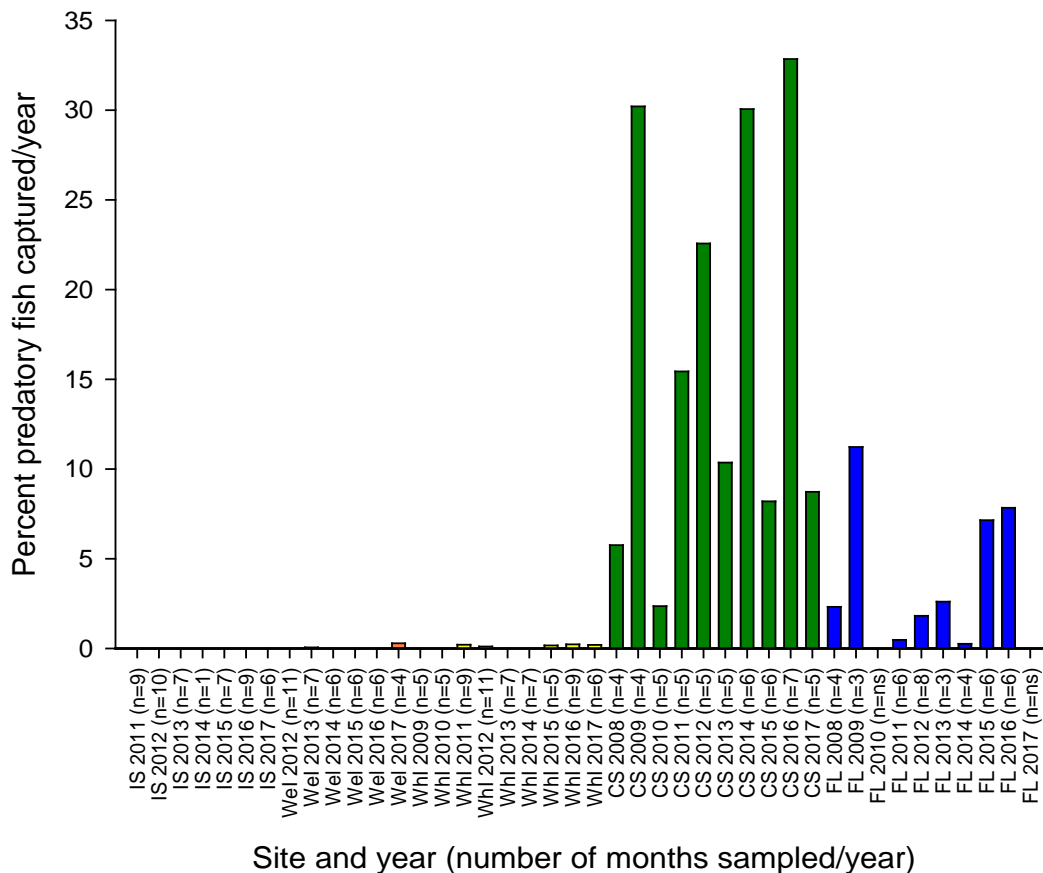
Figure 93. Total percentage of the yearly catch of fish species that have mature stages that could be predatory toward juvenile salmon. Species include small and largemouth bass, northern pikeminnow, walleye, warmouth and yellow perch. For each year the total number of sampling months is presented in 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, 2017 salmon species composition varied by site, showing distinct patterns associated with hydrogeomorphic reach (Figure 94). No sampling was conducted at Franz Lake in 2017 due to high water levels through-out the spring and a forest fire in the summer. In 2017, Chinook salmon were caught at all four sampled sites and were the dominant salmon species at Welch Island in Reach B, Whites Island in Reach C, and Campbell Slough in Reach F. At these sites, Chinook salmon comprised 90 to 100% of salmonid catches. In 2017, unmarked (presumably wild) Chinook were more abundant at Welch and Whites Island than marked hatchery Chinook, however, marked Chinook were more abundant than unmarked Chinook at Campbell Slough (Figure 95). 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 Whites Island. No trout or sockeye salmon were caught in 2017(Figure 94 and Table F.3 in Appendix F).
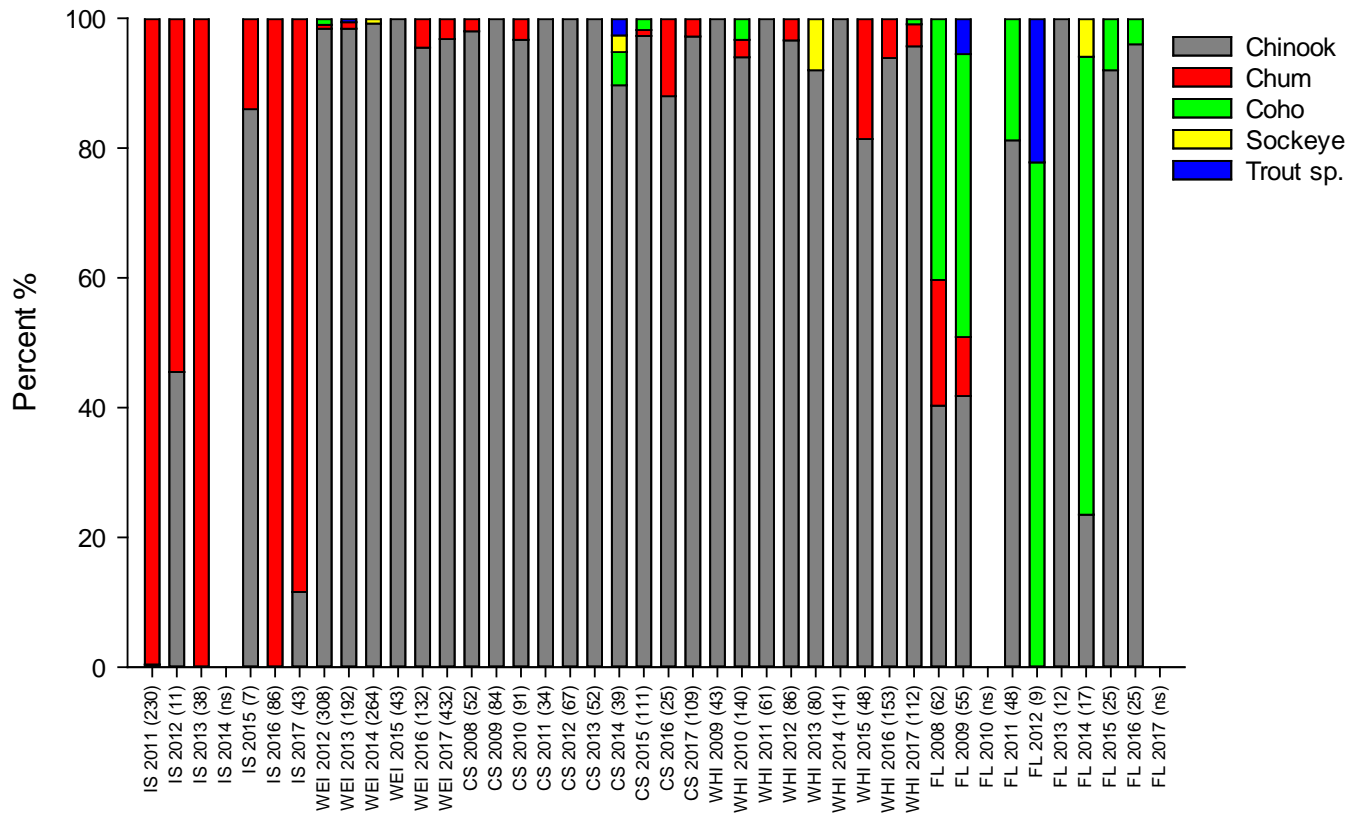
Figure 94. Percentage of salmonid species collected at EMP trends sites in 2017, as compared to percentages collected in previous sampling years. Total number of salmonids captured at a given site and year are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
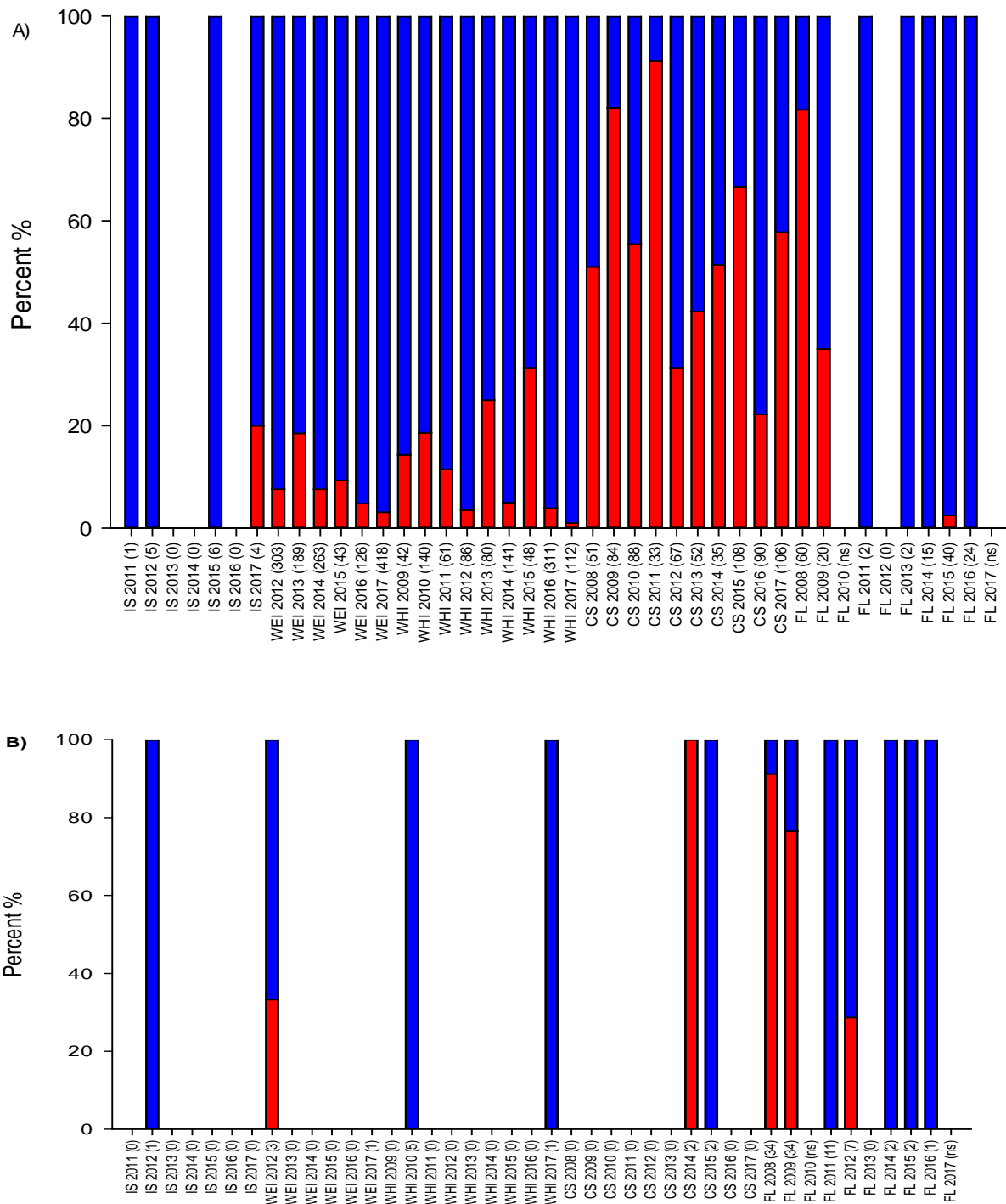
Figure 95. Percentage of marked and unmarked a) Chinook salmon and b) Coho salmon captured at the EMP sampling sites in 2017, as compared to previous sampling years. Total number of the specified salmon species captured at a given site and year are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

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3.6.2.1 *Salmon Density*

Chinook salmon

In 2017, unmarked Chinook salmon were captured at the EMP trends sites from February through June. The highest average densities of unmarked juvenile Chinook salmon were 41 and 70 per 1000 m^2 in April and May, respectively. Marked Chinook salmon were captured from March to June, with the highest average densities of 15.7 fish per 1000 m^2 in May (Figure 96). Mean Chinook salmon densities by site and year are shown in Figure 97. In 2017 the density of unmarked Chinook salmon was highest at Welch Island (73.4 fish per 1000 m^2) and Whites Island (12.8 fish per 1000 m^2) and lowest at Ilwaco Slough (1.4 fish per 1000 m^2), with densities in the 3-6 fish per 1000 m^2 range at Campbell Slough. Densities of unmarked Chinook salmon in 2017 were generally within the same ranges as previous years at all of the sites. The densities of marked Chinook salmon in 2017 were generally within the same ranges similar 2008-2015, but higher than those in 2016 (Figure 97).

Coho salmon

In 2017, only one unmarked coho salmon was collected at Whites Island in March (Figure 95). Coho salmon densities by site and year are listed in Appendix F. Coho salmon have been captured only sporadically at Ilwaco Slough, Campbell Slough, Welch Island and Whites Island, so their absence in 2017 was not unusual compared to previous years. No sampling was conducted at Franz Lake in 2017 due to high water, 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. Marked coho salmon, which were common at Franz Lake in 2008 and 2009, have not been observed since 2012.

Chum salmon

In 2017, chum salmon were found at the trends sites in March, April and May with the highest average density in March (15 fish per 1000 m^2; Figure 96). Chum salmon were present at Ilwaco Slough, Welch Island, Whites Island and Campbell Slough in 2017 (Figure 94). Since the beginning of this long-term study, 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 2017, as in 2016, Sockeye salmon and trout were not caught. Historic densities for sockeye salmon and trout for at all sampling have been relative low (Appendix F, Table F.4).
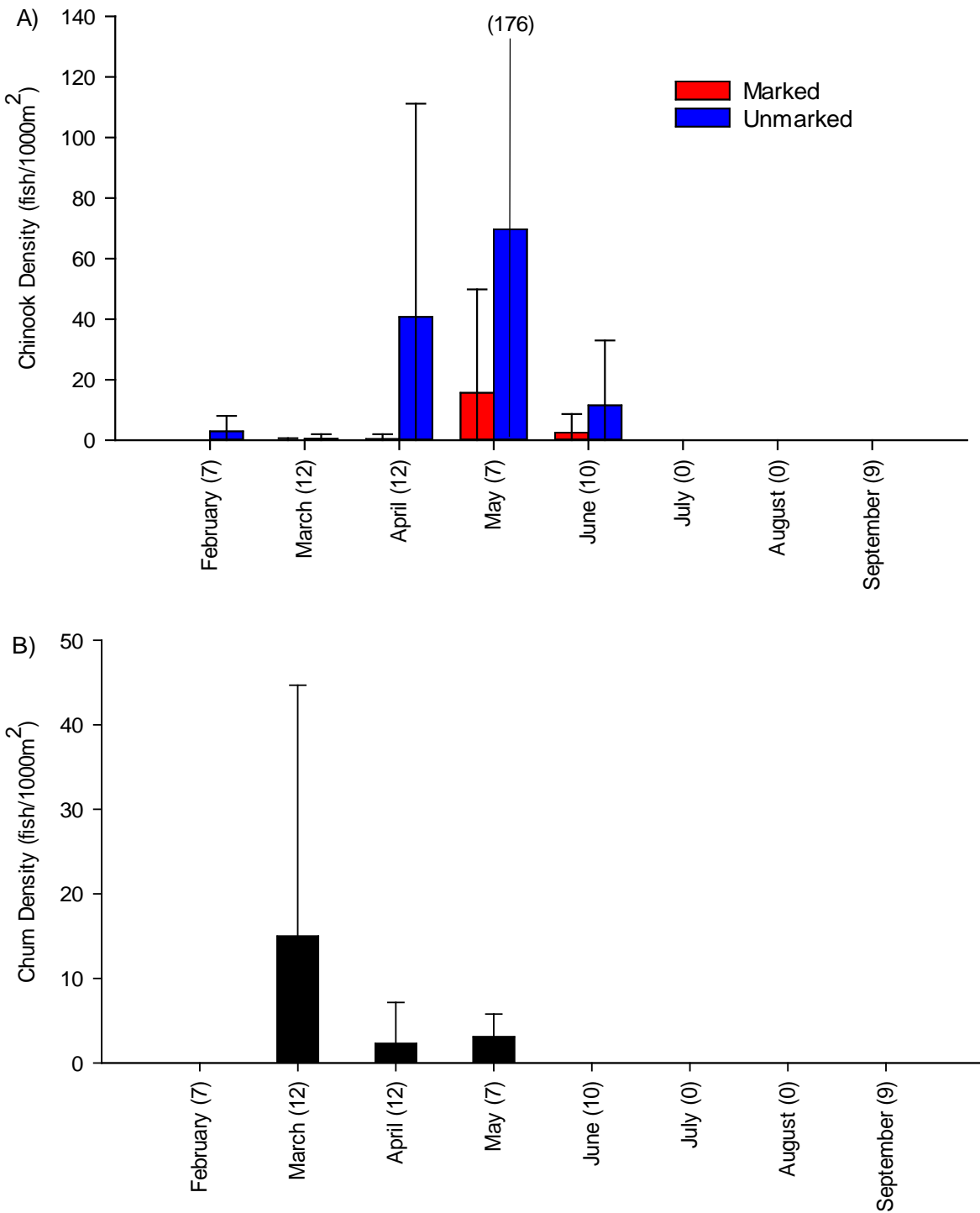
Figure 96. Mean (SE) densities (fish per 1000 m^2) of marked (red bars) and unmarked (blue bars) juvenile a) Chinook salmon, b) chum salmon by month during the 2017 sampling year (all sites combined). Total number of sampling efforts per month are presented in parentheses. May 2017 was truncated for 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 2017 therefore no monthly density for coho salmon is shown.
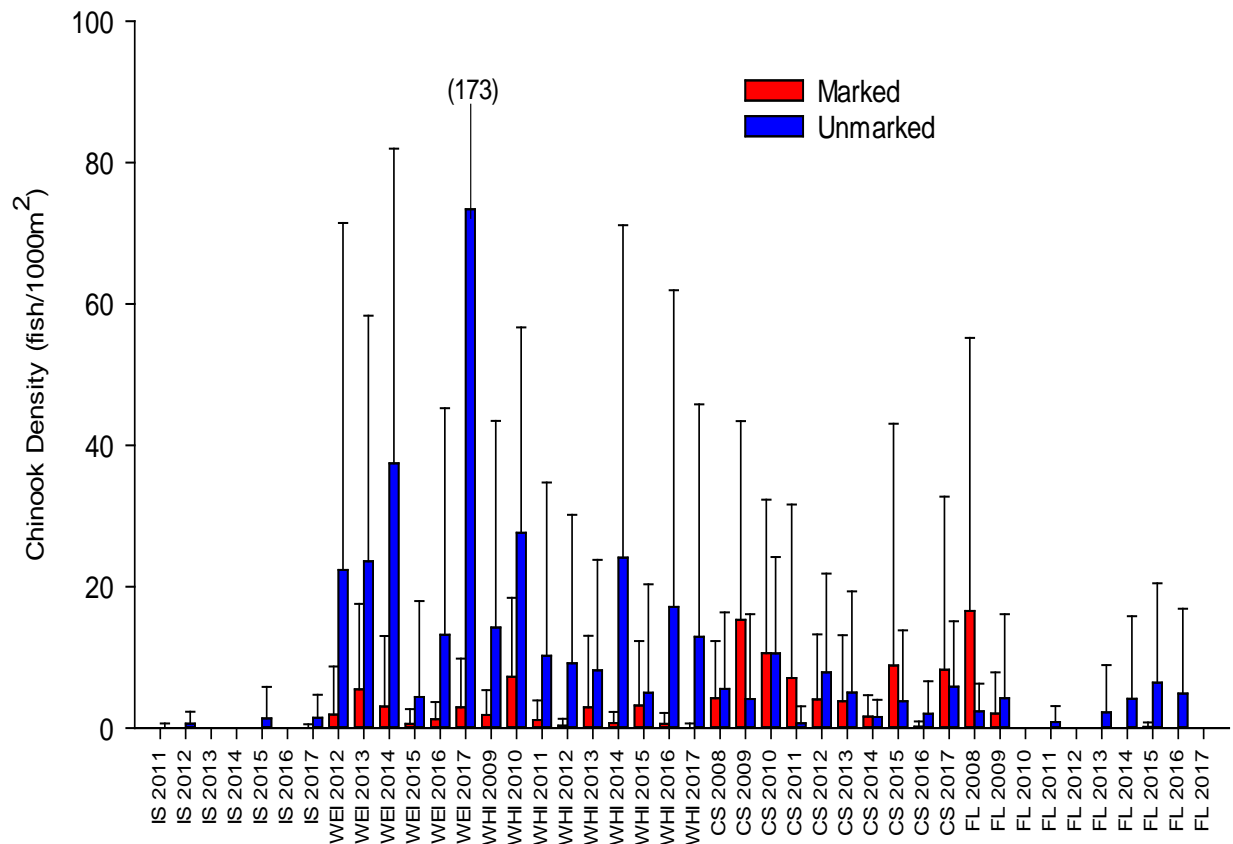
Figure 97. Marked (red bars) and unmarked (blue bars) juvenile Chinook salmon densities (fish per 1000 m²) by site and year. Welch 2017 was truncated for ease of viewing and the number in parentheses is the upper limit of the error (SE). IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.3 Salmon Metrics

3.6.3.1 *Genetic Stock Identification*

In 2016, genetics data were collected from Chinook salmon at Welch Island, Whites Island, Campbell Slough, and Franz Lake. In 2017, genetics data were collected from Chinook salmon at Ilwaco Slough, Welch Island, Whites Island, and Campbell Slough. We were unable to sample Franz Lake in 2017, thus, there are no genetics data to report for that site. 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, 85% of genetic samples assigned at 0.90 or greater.

Among unmarked fish in 2016, West Cascades fall Chinook were the most abundant stock at sites in Reaches B and C: Welch Island and Whites Island. This trend had persisted since 2010 when Reach C was first sampled as a status and trend site. Similar to previous years, the unmarked stock composition at Campbell Slough was more diverse than the lower river sites. Unmarked fish at Franz Lake were

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exclusively from the Upper Columbia River summer/fall stock in 2016. This signifies a reduction in diversity as Snake River fall and Spring Creek Group stocks are typically found at Franz Lake as well Figure 98).

Stocks of marked fish in 2016 followed annual trends of West Cascade fall stock dominating the lower reaches at Welch and Whites Islands and Spring Creek Group fall stock dominating the upper reaches at Campbell Slough and Franz Lake (Figure 98).

Among unmarked fish in 2017, West Cascades fall Chinook were the most abundant stock at sites in Reaches A, B, and C: Ilwaco Slough, Welch Island, and Whites Island. Similar to previous years, the unmarked stock composition at Campbell Slough was more diverse than the lower river sites, yet, in 2017 Spring Creek Group fall Chinook were present at a lesser percentage than in previous years (Figure 98). Interior stocks such as Snake River fall Chinook and Upper Columbia summer/fall Chinook were captured at all sites sampled in 2017 except Ilwaco Slough. Overall the stocks of unmarked Chinook present at each of the sites were generally similar over the sampling years.

Similar to previous years, the population of marked fish in 2017 was dominated by West Cascade fall stock in the lower river at Welch and Whites Islands, and by Spring Creek fall stock in the upper river at Campbell Slough (Figure 98). No marked Chinook Salmon were captured at Ilwaco Slough, nor has a marked Chinook been captured at this site for the duration of the study.

The seasonal distribution of stocks in 2016 reveals that West Cascade fall Chinook are present throughout the lower estuary during spring and summer. Interior stocks occurred earlier at upper reaches and were not present in lower reaches until June (Figure 99).

The seasonal distribution of stocks in 2017 reveals that West Cascade fall Chinook are present throughout the estuary during late winter, spring, and early summer, and that interior stocks are present later in the season. This trend echoes the seasonal pattern of proportional stock abundances observed in previous years (Figure 99). One notable difference in 2017 was the lack of Spring Creek Group stock during our April and May sampling at lower reaches. A possible explanation could be the timing of our sampling relative to large hatchery releases of the stock. In April our sampling was concurrent with a release from Spring Creek Hatchery (rkm 269), and in May our sampling occurred several days before a large release from Spring Creek Hatchery.
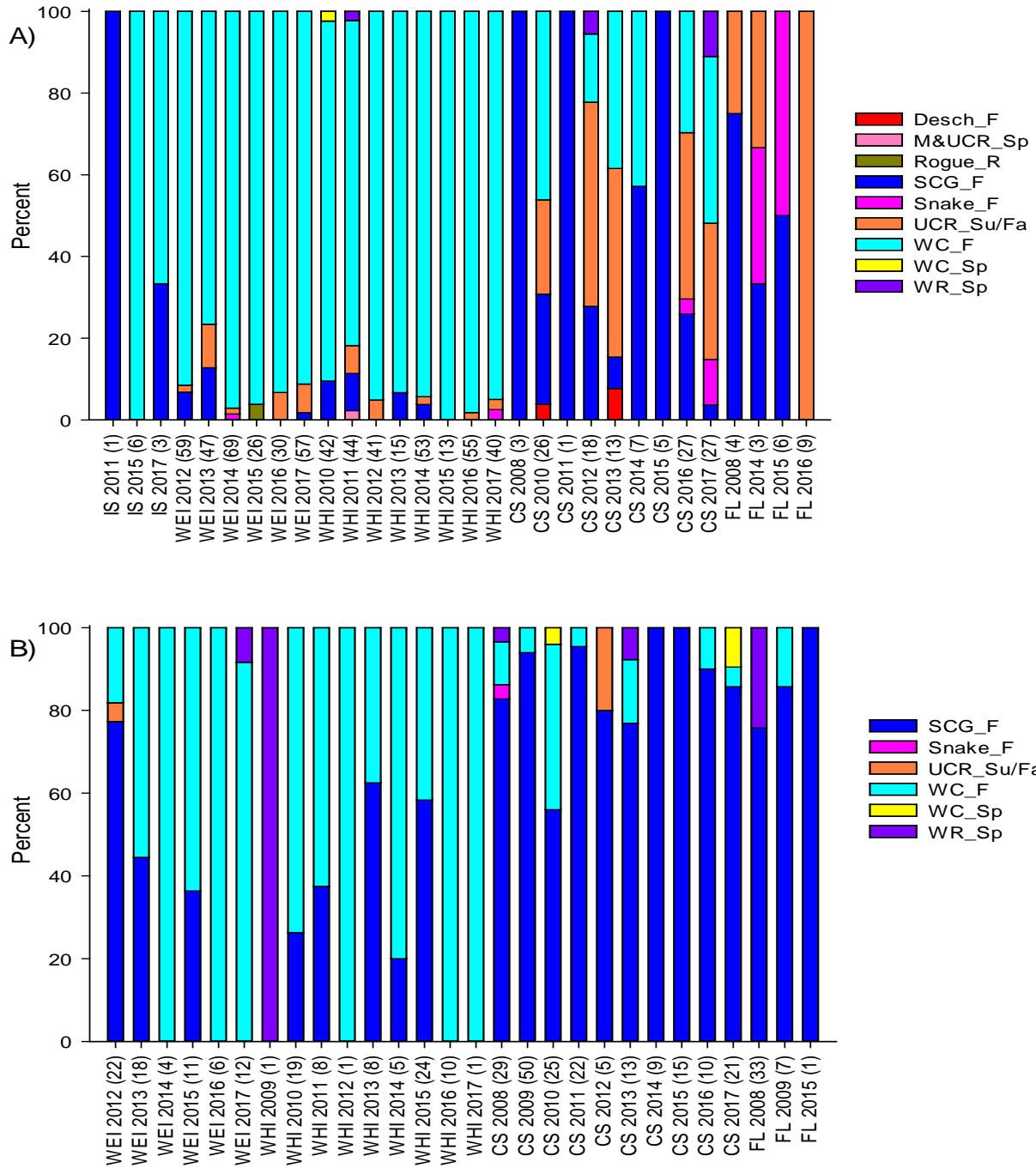
Figure 98. Genetic stock composition of A) unmarked and B) marked Chinook Salmon at trend sites from 2008–2017. Genetic sample sizes for each site is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake. Chinook salmon stocks: Desch_F = 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.
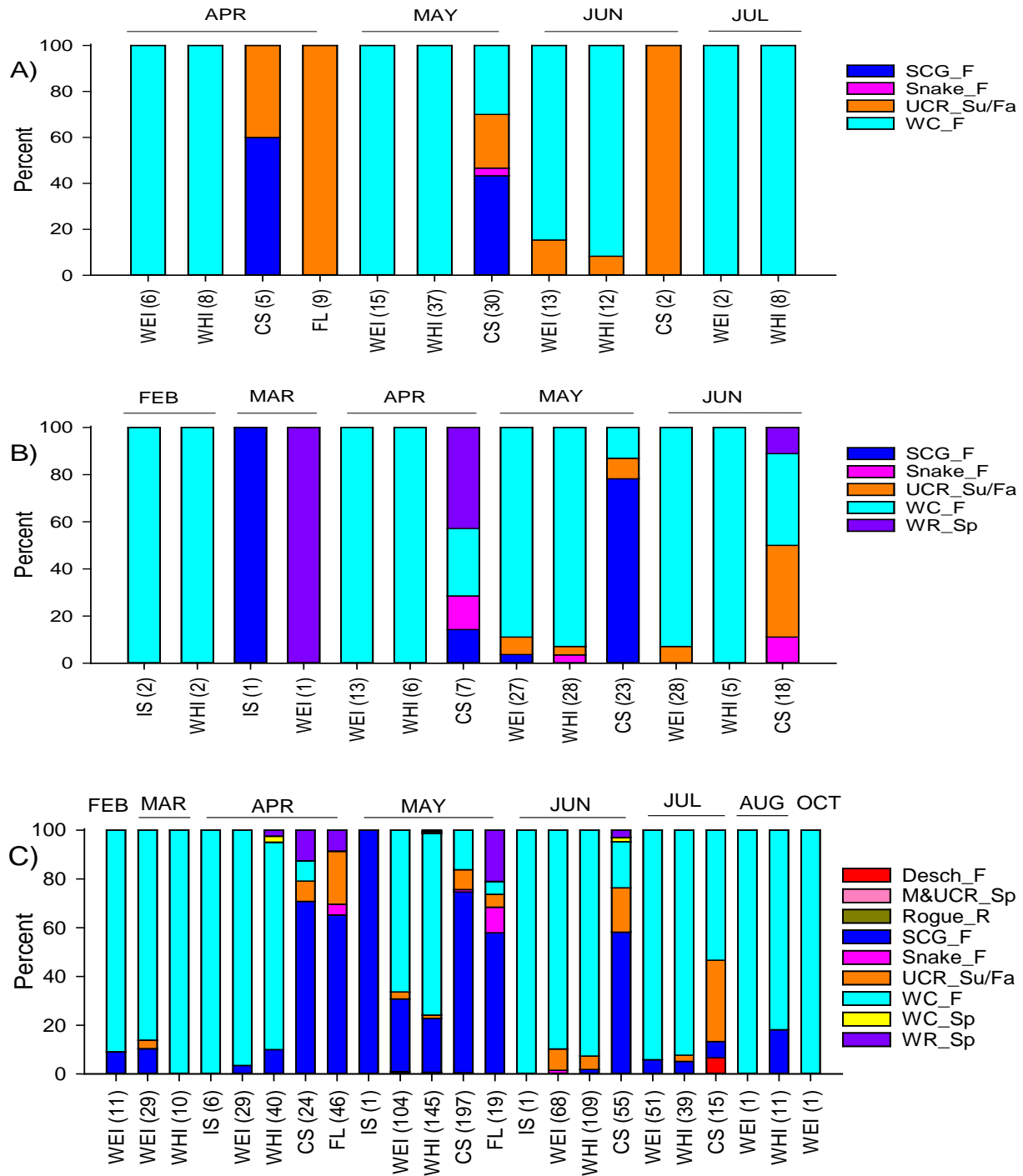
Figure 99. Seasonal percent stock composition per site for Chinook Salmon collected in A) 2016, B) 2017, and B) 2008–2016. Plots include both unmarked and marked Chinook Salmon. Genetic sample sizes for each site is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake. Chinook salmon stocks: Desch_F = Deschutes River fall, M&UCR_Sp = mid and upper Columbia River spring, Rogue_R = Rogue River, SCG_F = Spring Creek Group fall, Snake_F = Snake River fall, 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.

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3.6.3.2 *Salmon Size and Condition*

Chinook salmon

Length, weight, and condition factor. Chinook salmon were caught at all sampled locations in 2017. In 2017 the average length, weight and condition factor for unmarked Chinook captured at the following sampled sites were: Ilwaco Slough (± SD) 41.6 ± 3.05 mm; 0.51 ± 0.17 g; and 0.67 ± 0.09, Welch Island 50.12 ± 14.79 mm; 1.71 ± 2.09 g; and 0.94 ± 0.21, White Island 53.72 ± 12.15 mm; 1.77 ± 1.42 g; and 0.98 ± 0.19, and Campbell Slough 68.13 ± 8.47 mm; 3.75 ± 1.51 g; and 1.08 ± 0.05, respectively (Figure 100). The length, weight, and condition of unmarked Chinook salmon in 2017 showed similar patterns, with the largest fish typically captured at Campbell Slough (Figure 100). Within sites, there was some variation among years, though no clear increasing or decreasing trends. Unmarked Chinook sampled at Whites Island in 2017 appeared to have a similarly low average weight as seen in 2016. However, condition appears to follow the overall trend on Whites Island over the past 5 years. Ilwaco slough showed a decrease in size and condition. However this is likely due to a low sample size of only 3 unmarked Chinook in 2017.   Overall, unmarked Chinook size and condition in 2017 was very similar to Chinook sampled in 2016.

In 2017, marked Chinook salmon were caught at all four sampled locations. Only Campbell slough and Welch Island experienced significant enough catches to examine marked Chinook size and condition. In 2017 the average length, weight and condition factor for marked Chinook captured at the following sampled sites were: Welch Island (± SD) 80.27 ± 8.52 mm; 6.29 ± 2.06 g; and 1.12 ± 0.11, and Campbell Slough 87.09 ± 6.71 mm; 7.5 ± 1.85 g; and 1.11 ± 0.06, respectively (Figure 101). Campbell Slough and Welch Island show little variation in length, weight, and condition across sampled years (Figure 101). Similar to unmarked Chinook sampled in 2017, marked Chinook size and condition showed similar patterns to fish sampled in 2016.

Life History. At the trend sites in 2017, the majority of unmarked Chinook salmon were fry, 57.4%; 42.3% were fingerlings, and 0.3% were yearlings (Figure 102). At Welch Island and Ilwaco Slough, fry dominated catches, making up 78.9% and 75% of unmarked Chinook salmon, respectively. At Campbell Slough fingerlings predominated, comprising 77.8% of the catch. Whites Island showed a more even distribution of fry and fingerling with 44.1% and 55.9% of unmarked Chinook salmon, respectively. In comparison to previous years, the percentage of fry at all of the trend sites was lower than in 2016. A total of 76 (94.7%) marked Chinook salmon caught at the trends sites in 2017 were fingerlings, and 5.3% were yearlings (Figure 102). 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.

Other salmon species
A total of 58 chum salmon were captured and measured in 2017, 37 at Ilwaco Slough, 14 at Welch Island, 4 at Whites Island, and 3 at Campbell Slough. All chum were caught between March and May. Only 51 of the 58 chum captured in 2017 were used due to the difficulties of measuring weight (g) during adverse field conditions. In 2017 the average length, weight, and condition factor of chum salmon captured at the following sampled sites were: Ilwaco Slough (± SD) 42.16 ± 1.68 mm; 0.52 ± 0.07 g; and 0.69 ± 0.08, Welch Island 40.66 ± 2.17 mm; 0.54 ± 0.17 g; and 0.80 ± 0.19, White Island 39.50 ± 0.71 mm; 0.45 ± 0.07 g; and 0.73 ± 0.08, and Campbell Slough 61.66 ± 21.93 mm; 2.8 ± 3.2 g; and 0.87 ± 0.11, respectively (Figure 103). The chum salmon collected in 2017 were comparable in size to those that have been collected in previous years, not especially large or small. Similarly, the mean 2017 value for condition factor (0.71) was intermediate, between a high of 1.10 in 2008 and a low of 0.58 in 2013. 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 Franz Lake (not sampled in 2017) and Campbell Slough, the two sites farthest upstream. 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 2017, an unmarked coho found at Whites Island in March. Only four coho have been sampled from Whites Island since 2009. 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. Overall size and condition of unmarked and marked coho sampled at Franz Lake are shown below in Figures 104 and 105.

Sockeye salmon and trout were not caught at any of the trends sites in 2017. Sockeye salmon were last sampled at Welch Island in 2014 and trout were last caught at Welch Island in 2013.
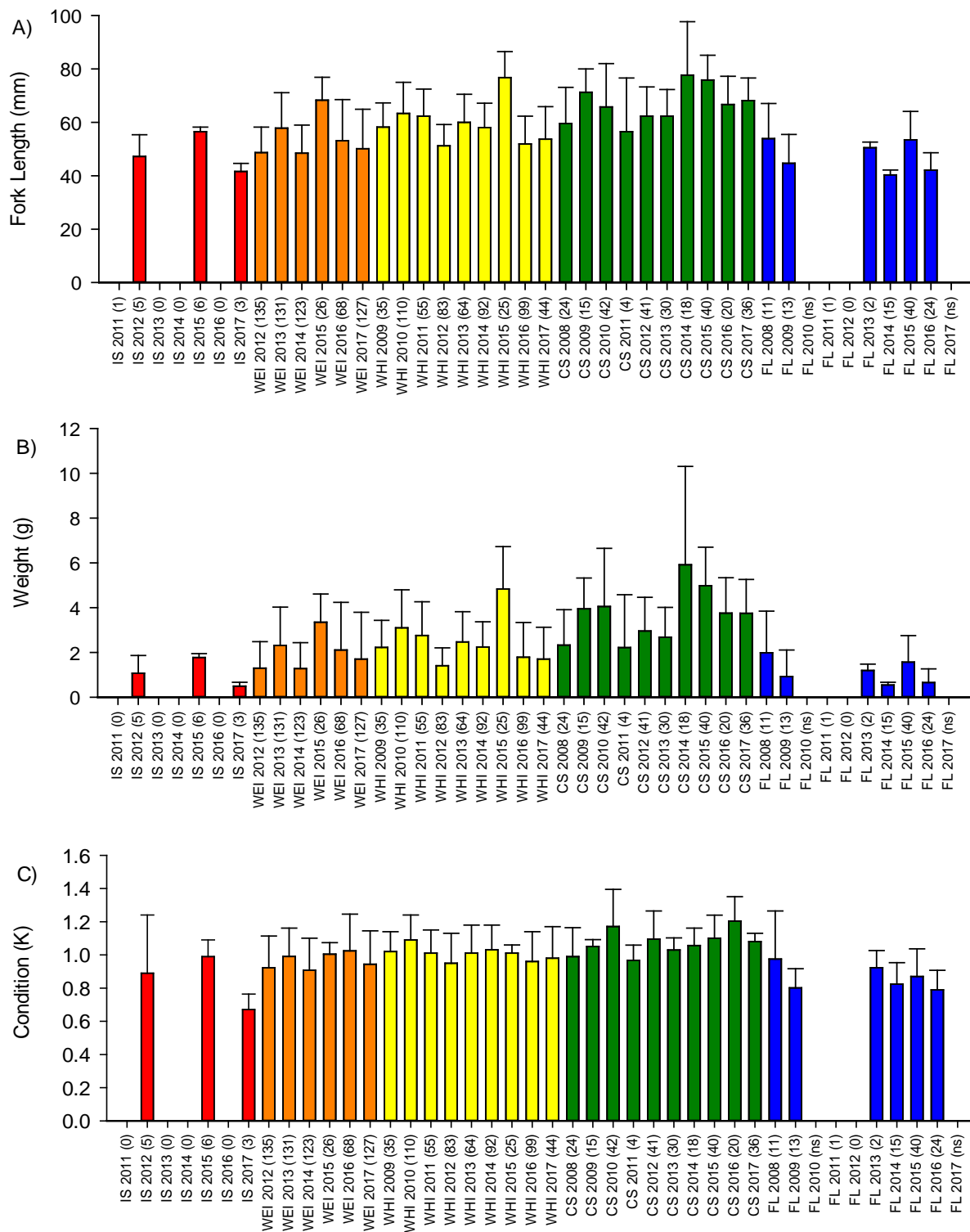
Figure 100. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of unmarked juvenile Chinook salmon at trends sites in 2017 as compared to previous years. Total number of Chinook salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
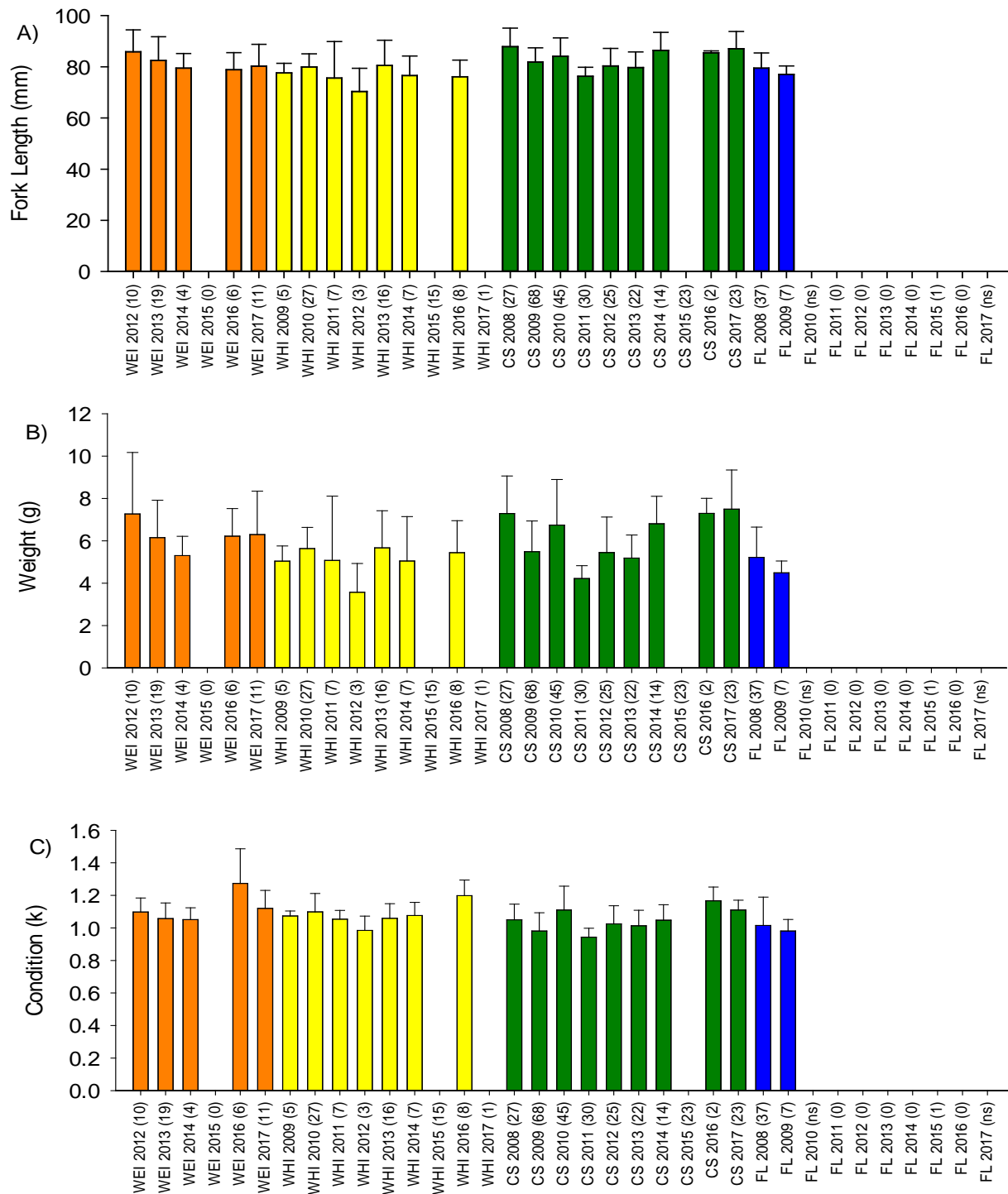
Figure 101. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of marked Chinook salmon at trends sites in 2017 compared to previous sampling years. Total number of Chinook salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
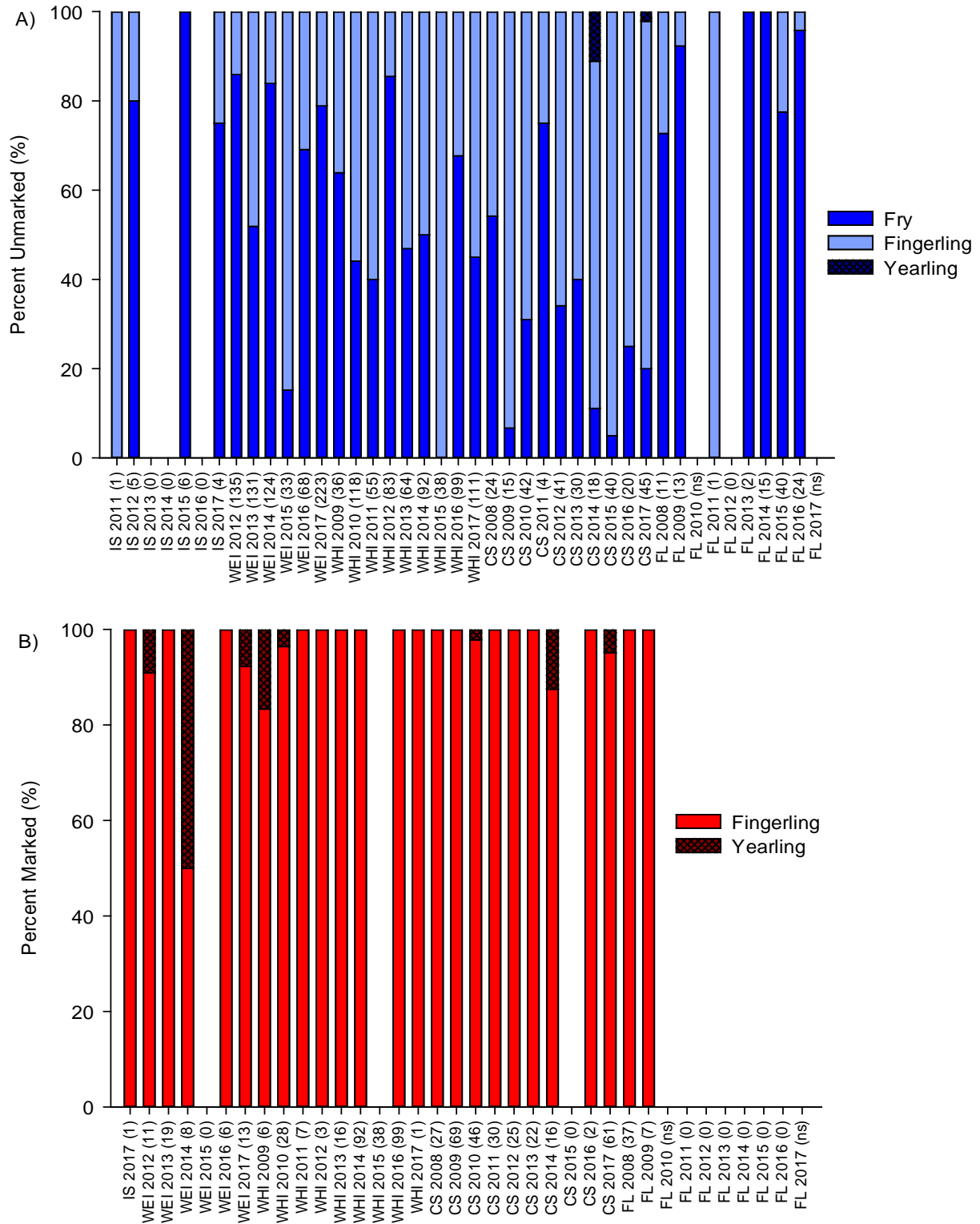
Figure 102. Percentages of life history types of a) unmarked and b) marked juvenile Chinook salmon captured at trends sites in 2017 and in previous sampling years. Total numbers of Chinook salmon captured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

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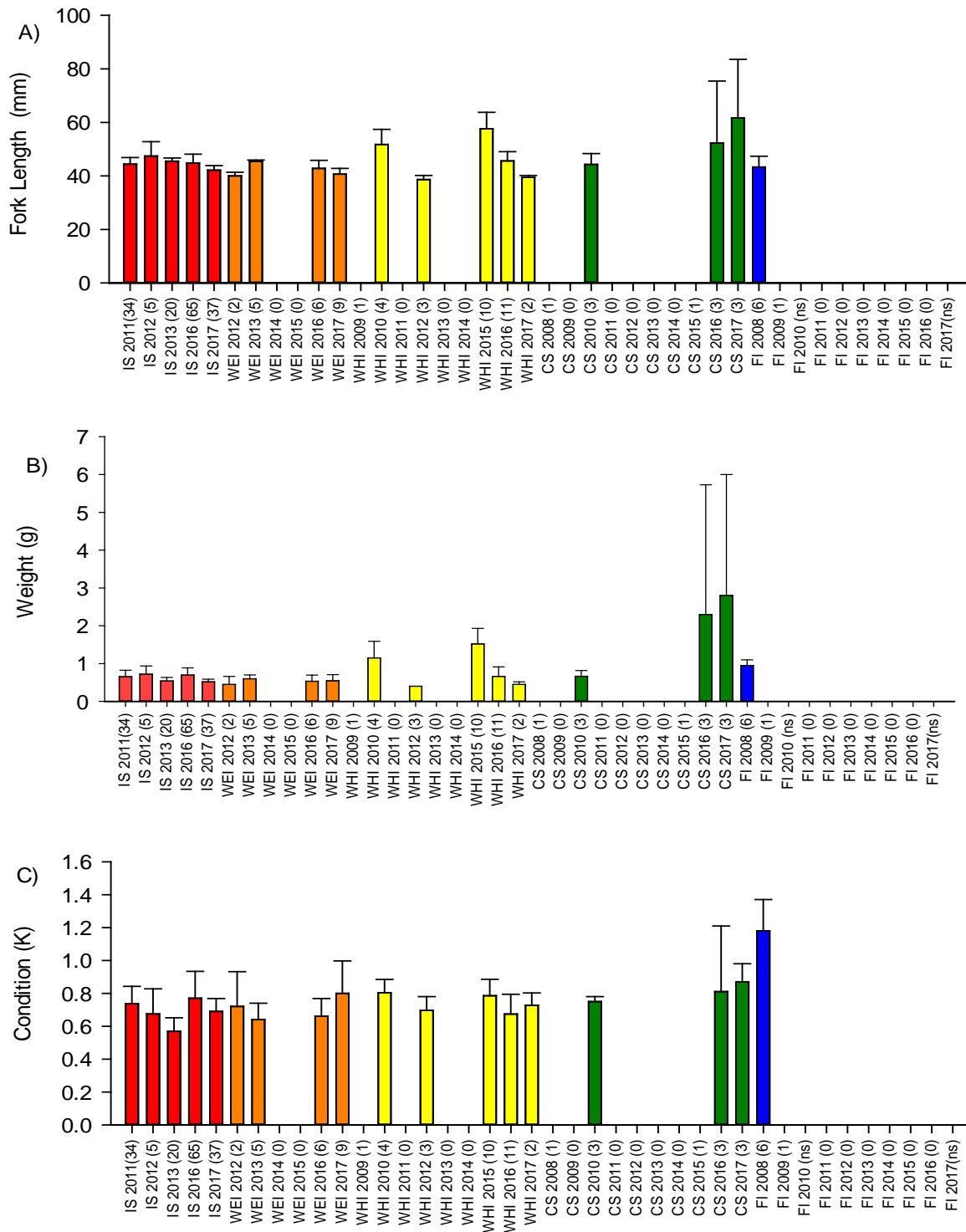
Figure 103. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of chum salmon at trends sites in 2017 compared to previous sampling years. Total number of chum salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
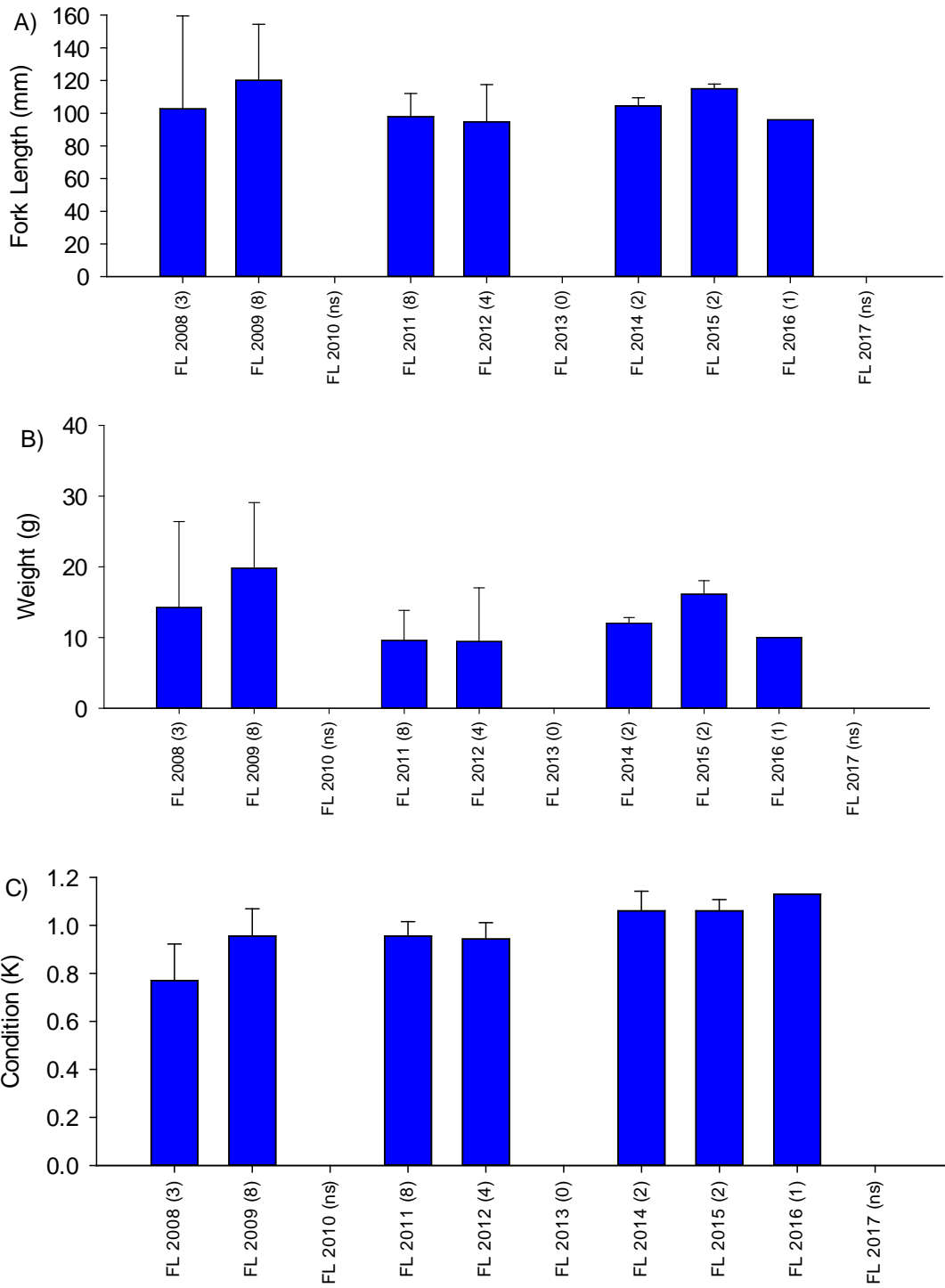
Figure 104. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of unmarked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses. No sampling was conducted on Franz Lake in 2017 due to high water levels.

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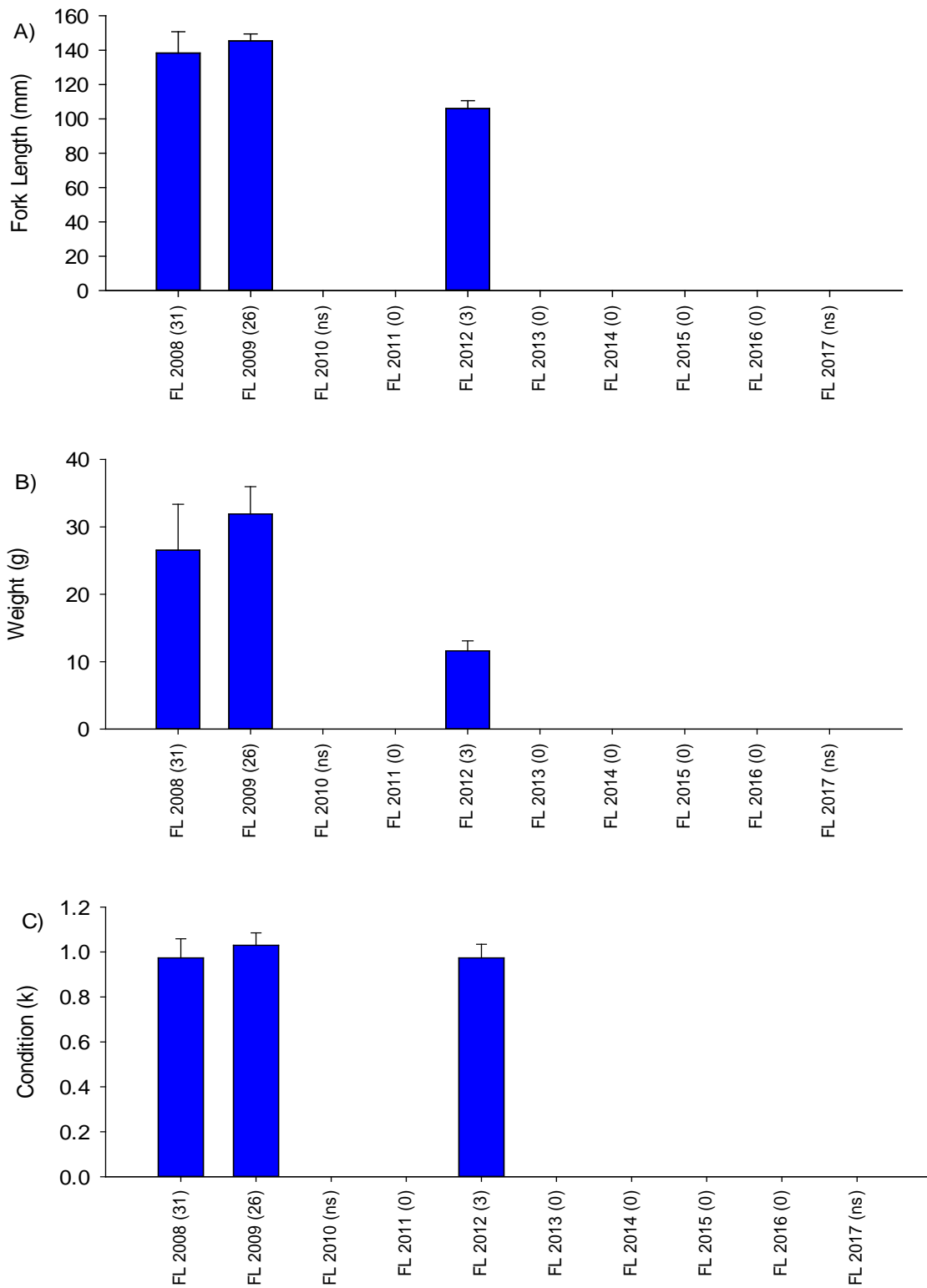
Figure 105. Mean (SD) a) length (mm), b) weight (g), and c) condition factor of marked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses. No sampling was conducted on Franz Lake in 2017 due to high water levels.

3.6.3.3 *Somatic Growth Analyses*
Processing of otoliths from fish collected in 2017 was not completed by the time of publication.

3.6.3.4 *Lipid Content of Juvenile Chinook Salmon*
Lipid data for 2017 are presented with data from 2008-2015. Samples from 2014 were compromised and are not presented. Samples from 2016 have not been processed. Lipid data in 2017 were collected at Welch Island, Whites Island, and Campbell Slough. Average gravimetric lipid percentages for these sites in 2017 fell within established ranges from 2008-2015 (Figure 106). Percent gravimetric lipid content at Campbell Slough was higher than previous years except 2015. Likewise, average percentages of triglycerides at Welch Island, Whites Island and Campbell Slough in 2017 also fell within established ranges from 2008-2015 (Figure 106). The percent triglycerides at Campbell Slough in 2017 was the highest level reported at the site, edging out levels observed in 2010.

Seasonally, both lipid content and triglyceride levels tend to increase in late winter and early spring and reach a peak by late spring or summer. From 2008-2015 lipid content peaked in April and triglyceride levels peaked in July. In 2017, both lipid content and triglycerides peaked in May (Figure 107).
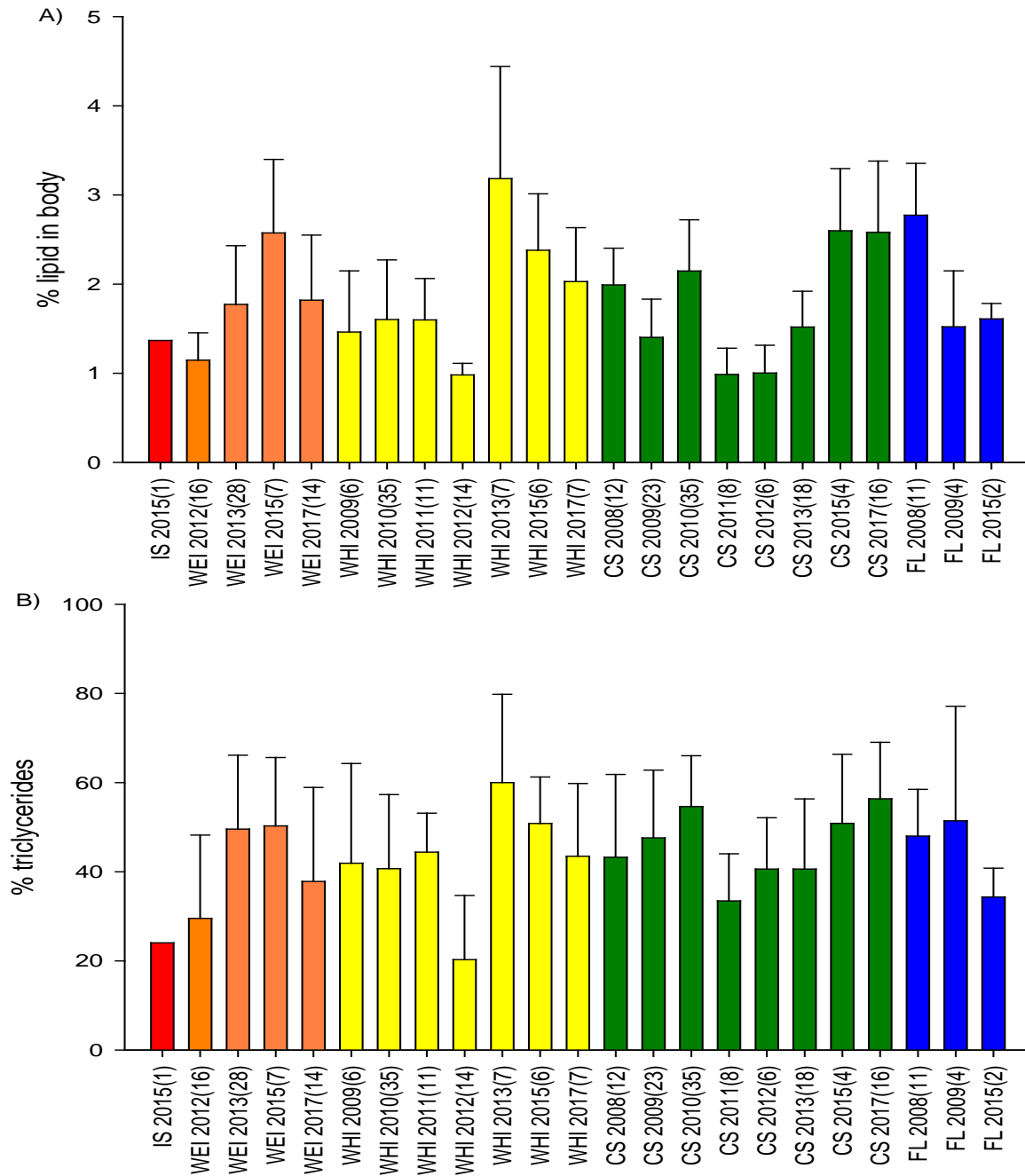
Figure 106. Mean (SD) A) body lipid content (%) and B) % of total lipids that were triglycerides in Chinook salmon collected from trend sites in 2017 compared to previous sampling years. The number of composite samples analyzed is indicated in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.
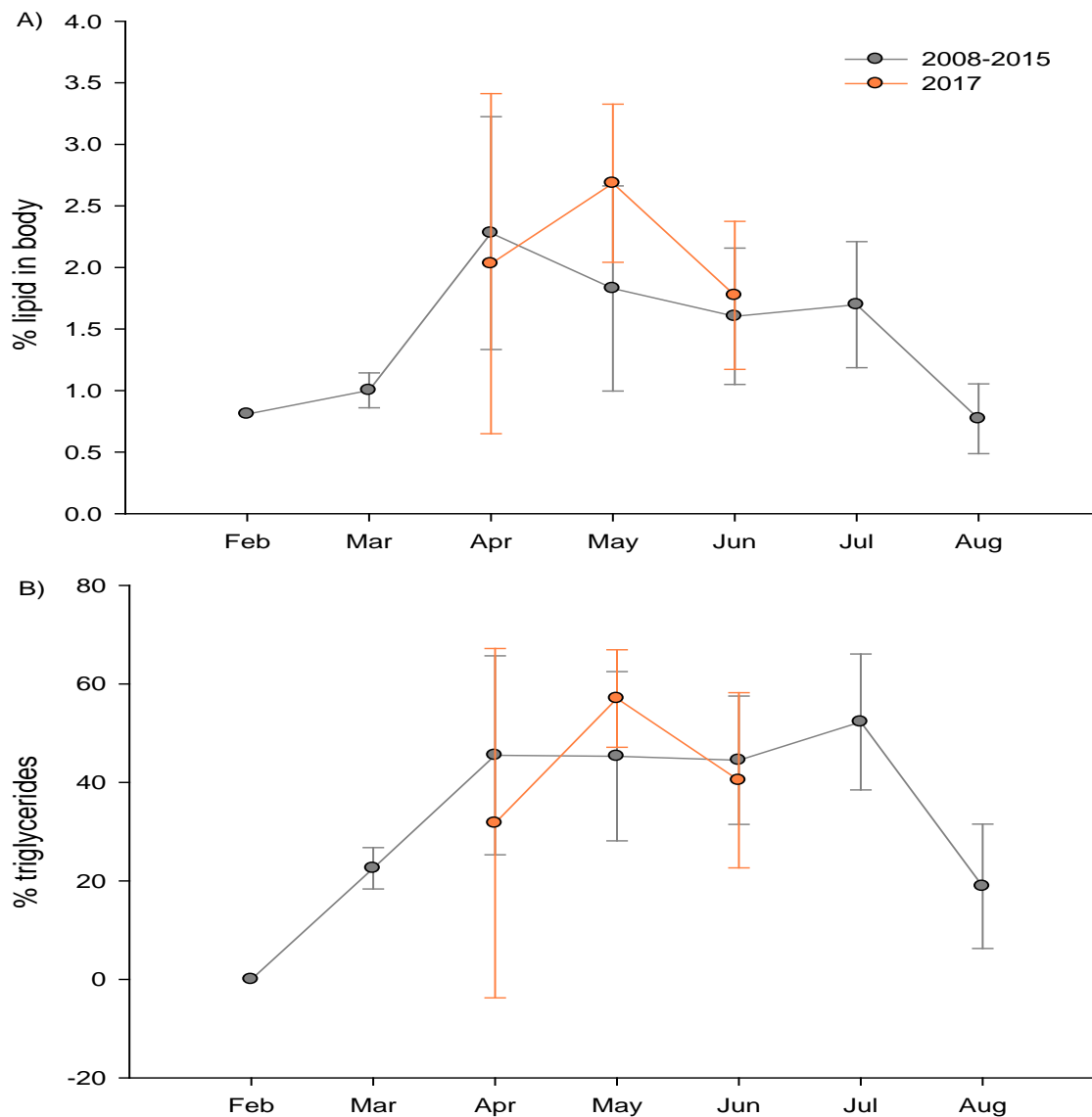
Figure 107. Seasonal trend of mean (SD) A) body lipid content (%) and B) % of total lipids that were triglycerides in Chinook salmon collected from trend sites in 2017 compared to previous sampling years.

3.6.4 PIT-Tag Array Monitoring of Juvenile Salmon Residence

The PIT detection array at Campbell Slough was non-operational in 2017. Repeated efforts to reset the equipment were unsuccessful. In the fall of 2017, we removed both antennas and discovered that the structural integrity of one of the antennas had failed. This allowed water to enter the antenna rendering it inoperable. The other antenna was also inoperable due to a severed power and communications cable. We plan to revamp the Campbell Slough PIT array in 2018. Six repurposed antennas measuring 4' x 10' will be installed vertically allowing for better detection coverage of the slough. A repurposed elevated platform to house the electronics and solar panels also will be constructed.

The Horsetail Creek PIT detection array was operational from February 25–November 30, 2017. Although not all 10 antennas were operating, we had coverage of two antennas on the downstream side and three antennas on the upstream side of the culvert.

Twenty-nine individual fish were detected from May 7–November 26. Thirty-eight percent of fish detected were juvenile fall Chinook. The second most prevalent category was juvenile steelhead at 24%. Juvenile spring Chinook and adult coho salmon represented 14 and 7 % of detections, respectively. One northern pikeminnow and four unknown (no tag data in the regional database; www.ptagis.org) fish were also detected. Detection numbers and residence times are listed in Table 30. Residence times provided are a measure of elapsed time from first to last overall detection. We have not yet analyzed the data to determine whether a fish passed through the culvert and how much time was spent on the upstream side in the restoration area. Most fall Chinook originated from hatcheries in the Bonneville Pool, however, one was Snake River stock. The majority of steelhead and spring Chinook were also from Snake River populations.

Efforts are underway to repair and replace damaged antennas at Horsetail Creek in order to achieve greater detection efficiency in 2018.

Table 30. Number and residence time (max and median) of fish detected at Horsetail Creek PIT array in 2017. 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.

	N	Residence time	
		Max	Median
Fall Chinook	11 (1)	2.5 h	33 m
Spring Chinook	4 (1)	1 h	12 m
Steelhead	7 (4)	24.5 d	47 m
Coho (adult)	2	20.5 h	11 h
Northern Pikeminnow	1	21 d	
Unknown	4	38 m	14 m

4 Status and Trends Discussion

4.1 Mainstem Conditions

River discharge fluxes in the Columbia River were higher than normal in 2017. There was an early and sustained freshet that was similar in magnitude and timing to observations from 2011 and 2012. Early high flows were likely driven by high precipitation in February and March; February was the wettest on record and March had twice the normal rainfall (NOAA National Weather Service). High flows resulted in high water elevations, making Franz Lake Slough inaccessible until May.

Although 2017 river discharge fluxes were high relative to other years during winter, spring, and autumn, they were similar to the average for the late spring and summer. The data 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). While there was a similar number of days where water temperatures exceeded 19 ºC in 2017 compared to other recent years (2013, 2014, 2016), the number of days with temperatures exceeding high thresholds (21, 22, or 23 ºC) was greater in 2017 than in any other year except 2015. Climate change, generally manifest 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 temperature, which is thought to occur due to reduced food availability (Burke et al., 2013, Daly et al., 2013).

By comparison to other years, 2017 was average in terms of source contributions to total Columbia River determined at river mile 53 (Beaver Army Terminal). Similar to other years, there was a relatively large fraction of total Columbia River flow from the Willamette and tributaries in early spring, which 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).

Dissolved nitrate concentrations exceeded 50–60 μM in March–April and declined rapidly during the month of April when phytoplankton growth was strong. 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), 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 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 observed between July and September when salinity increased; low dissolved oxygen saturation was also observed at Campbell Slough and Franz Lake Slough. The low values in the two upstream sites were due to intense biological activity rather than to inputs from water masses with low dissolved oxygen. At both Campbell and Franz, dissolved oxygen concentrations fluctuated widely, with values as low as 40% of that expected from equilibrium with the atmosphere to >120%. Photosynthesis within algal blooms leads to high values and respiration of organic matter accounts for low values.

pH values were in the target range for good water quality at Ilwaco, Welch, and Whites Island throughout the sampling season in 2017; in contrast, high values were observed at Campbell Slough and Franz Lake Slough during periods associated with strong growth of phytoplankton, particularly cyanobacteria. At Franz Lake Slough, pH dropped to ~5 during the spring. These values were outside of the range set by the Washington Department of Ecology (7–8.5) for good water quality. 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, where values as high as 18 µM were measured; however, pH at Ilwaco did not approach 9, so the observed concentrations were not harmful. Ammonium concentrations were generally <2 µM at the other trends sites.

Water temperatures were warmest at Campbell Slough, exceeding 25ºC in August. In 2017, a 20ºC threshold was exceeded at Welch Island and Whites Island from mid-July through August, while at Campbell Slough and Franz Lake Slough temperatures exceed 20ºC 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.

Nutrient concentration at the trends sites sometimes exceeded recommended benchmarks for good water quality. For example, nitrate concentrations exceeded the upper recommended concentration (28.5 µM) in the early spring at Whites Island, Welch Island, and Ilwaco. At the other extreme, nitrate was almost completely drawn down at Franz Lake Slough between March and June, coincident with strong phytoplankton growth. Following this period, there was a summer increase in phosphate at Franz Lake Slough that was not observed at the other trends sites; concentrations increased from < 1µM to almost 3 µM between late June and the end of July. This coincided with a steady decrease in dissolved oxygen saturation (<80% of atmospheric equilibrium), and presumably contributed to the large chlorophyll peak in August at Franz. Campbell Slough did not show a similar late summer peak in phytoplankton biomass.

By examining relationships between key nutrients that fuel growth of primary producers (i.e., N and P), we can infer whether nutrient limitation of growth is occurring at the five trends sites. At Ilwaco, N:P ratios in the dissolved and particulate phases were generally close-to or less-than 16:1, suggesting that phytoplankton growth was not limited by N or P; after March, ratios declined to ~10, suggesting a tendency toward nitrogen limitation. The other sites tended toward phosphorus limitation, particularly during the period of strong spring growth (March); Welch Island and Whites Island in particular demonstrated clear P-limitation after mid-March, where N:P ratios increasing to >20 by the end of the summer. Thus for each of these two sites, there was a general pattern of P-limited spring growth, nutrient replete late-spring/early summer growth, and mildly P-limited autumn growth. In contrast, although early spring growth was P-limited at Campbell Slough and Franz Lake Slough, by April, N:P ratios were <10 (for both dissolved and particulate fractions), indicating a tendency toward nitrogen limitation of growth. The higher-than-expected nitrogen content of particulate matter relative to ratios of dissolved nitrogen to

phosphorus in late summer at Franz Lake Slough likely reflects the fixation of N_2 by cyanobacteria, which were abundant at this site during the summer.

An investigation of patterns in net ecosystem metabolism (NEM), which indicates whether a system is considered to be net autotrophic (i.e., dominated by organic matter production through photosynthesis) or net heterotrophic (i.e., dominated by organic matter respiration), showed that—despite large hourly fluctuations—both the site with the greatest exchange with the mainstem (Whites Island) and the site with the least exchange (Campbell Slough) were dominated by organic matter production (i.e., positive NEM) over the last few years. Positive NEM generally equates with good water quality (Caffrey, 2004), particularly with respect to dissolved oxygen content; indeed, dissolved oxygen saturation exceeded 100% for nearly the entire year in the mainstem, with greater day-to-day variability observed during the summer months when river discharge fluxes were smaller. This is likely influenced by the shallow nature of the sites, which facilitates air-water exchange. This same characteristic, however, means that heat exchange is also rapid, and the shallow sites warm substantially during the summer months.

4.3   Habitat Structure

4.3.1 Sediment and Hydrology

Sedimentation and hydrology data are currently under analysis and not available at the time of this report.

4.3.2 Vegetation Community Condition and Dynamics

In 2017 we saw a negligible decline in average total vegetation cover from 2016, however this was driven by large declines of 30-39% at two sites, Campbell Slough and Cunningham Lake (Figure 39). This is drop in total cover at these sites, Campbell Slough and Cunningham Lake, is possibly explained by the occurrence of heavy grazing at both sites in 2017 and the timing of the freshet resulting in more bare ground at these upriver sites than seen in previous years. The other four sites all increased in cover by 10-30% (Ilwaco Slough, Welch Island, Franz Lake) or held steady (Whites Island). Three sites (Ilwaco Slough, Welch Island, Franz Lake) all reached their highest recorded cover values in 2017. Ilwaco Slough fully recovered its cover values after having declined substantially in 2015-2016. Franz Lake continued its steady annual increase in cover since 2012. At Whites Island, the total cover held steady, but the relative proportions of native and non-native shifted, with a 6% decline in non-native cover. The average proportion of vegetation cover that was native across all sites increased slightly in 2017. Native cover increased or held steady at all sites except Ilwaco where the proportion of native vegetation decreased slightly compared to non-native cover.

Total species richness at all sites declined in 2017 from its high in 2015-2016. Only Whites Island increased in number of species. There was a greater reduction in the number of non-native species than in the number of native species, with particularly large reductions in the number of non-native species at Campbell Slough and Cunningham Lake.

The reasons for the changes seen this year are not entirely clear as yet. We are still awaiting the hydrological and salinity data so have not yet been able to evaluate water level dynamics for 2017.

4.4 Food Web

4.4.1 Primary Production

4.4.1.1 *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. In addition, we sample the standing dead biomass the following winter (February) to determine the amount of die-back. Interestingly, the proportion of live material in the winter samples was greater in the lower part of the estuary where the growing season starts much earlier. As long as live and dead material is separated in the winter samples then this early spring growth can be removed from the analysis; however, if separation is not done then sampling should be conducted earlier to avoid the confounding effects of new spring growth. Live material in the winter at Cunningham Lake and Campbell Slough were 33 and 9 percent, respectively. The reason for the difference is not known, however, it may be the higher elevation at Campbell Slough results in earlier vegetative growth than at Cunningham Lake.

Overall, 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 marshes is not uncommon (Appendix E), consistently occurring in the strata dominated by *C. lyngbyei*. In 2017, the highest average summer biomass was observed at Whites Island *P. arundinacea*/HM strata, with nearly 1,200 g/m^2, however, the multi-year analysis of the summer biomass revealed high variability between years for these strata, with 2017 biomass being lower than 2016. Across sites, the high marsh and low marsh averages from summer of 2017 were similar to those from 2011, the initial monitoring year. The upper estuary sites were generally more variable, perhaps due to the effects of inundation, though data were not sufficient for linear regression analysis of this relationship.

Specific conditions at the sites can explain the variation observed in the summer biomass. The greatest high marsh variability, averaged across years, was observed at Whites Island in the *P.arundinacea*/HM and *C. lyngbyei* strata and at Franz Lake site in the *P. amphibium* stratum. High biomass production occurred at these sites in 2017 (with the exception of Franz Lake, which was not sampled in 2017), as well as low biomass production in 2012, presumably due to high water conditions. In addition, variability in *P. amphibium* biomass was likely caused by differentiation in the size of the plants, with the larger plants having higher individual biomass but more sparse distribution. The *P. amphibium* morphology at Franz Lake has changed from large (>2m tall) plants well adapted to floating in 2 m or more of water (such as was present at the site in the 2011-2014 growing seasons) to smaller plants (about 1 m tall) growing at a higher density. This growth habit is very productive and results in high biomass. The biomass in the low marsh strata at Campbell Slough was lowest in the high water year of 2011 and in 2013, following the low water years (the site was not sampled in 2012 due to cows entering the site). Conversely, the low marsh at the site had very high biomass in 2015 and 2016, both low inundation years. In the *P. arundinacea* stratum at Campbell Slough, biomass was fairly low in all years, presumably due to grazing effects.

The marshes in the lower river contribute a large amount of organic material to the ecosystem annually. However, the amount relative to the energy needs of the food web is unknown. Additionally, the contribution is variable by strata, year, and location in the river, making estimates of food web effects difficult to discern. Overall, the strata 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 lower estuary. No other native sedge species dominate in the marshes of the upper estuary due to the highly competitive *P. arundinacea*. Based on the results from the analysis of the proportion of plant material breakdown, one could speculate that even though cover and biomass production may be variable between years, the overall amount of native sedge that breaks down in a year would be higher than that contributed by *P. arundinacea*. If organic material from marsh plants is indeed a limiting factor for the detrital based food web in the lower river, then restoration of additional marsh area dominated by native sedges could improve those conditions.

One of the advantages of long-term monitoring is the ability to develop predictive relationships. One such recently developed model shows a strong linear relationship between summer biomass production and the amount of organic matter that is contributed to the ecosystem, within vegetation strata. This means that with varying levels of confidence, we can estimate the amount of organic matter a plant community produces based on the amount of summer biomass produced.

4.4.1.2 *Pelagic*

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, Tausz et al., in prep.). 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, with the exception of Franz Lake Slough, where a large peak in chlorophyll was observed later in the summer. This peak occurred after dissolved phosphorus concentrations increased, which occurred after the depletion of nitrate from the water column. 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 were composed of cyanobacteria, since peaks in chlorophyll coincided with peaks in phycocyanin, a pigment found mainly in cyanobacteria. 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.

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; 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. The data thus far suggest that juvenile salmonid growth is supported by amphipods, chironomids, and other crustacean prey, which is consistent with observations derived from stomach analysis (see discussion below).

4.5   Macroinvertebrates

Macroinvertebrate data collected in 2017 are still under analysis and unavailable at the time of the writing of this report. The following food web discussion is adapted from the 2016 report.

Consistent with previous sampling, the benthos from all sites were dominated by nematode and oligochaete worms (Hanson et al. 2016). Benthic densities and biomass tended to be similar among sites from the lower estuary (Ilwaco Slough, Whites Island, and Welch Island) as well as among the upper estuary (Campbell Slough and Franz Lake), though average densities were consistently greatest from Ilwaco Slough. In a comparison to previous years, benthic densities appear to be relatively stable within a site and month.

Similar to previous reporting of neuston sampling from the EMP sites (Hanson et al. 2016), macroinvertebrate density and biomass in 2016 tended to be greater in the emergent vegetation than in open water. Overall, average density was nearly six times greater in the emergent vegetation than in the open water. This difference in occurrence by habitat was lower than previously reported from the EMP sites. In previous years, average density overall was more than 20 times greater in the emergent vegetation than in the open water (Hanson et al. 2016). Overall biomass in 2016, however, was nearly 70 times greater in the emergent vegetation than in the open water.

Average dipteran density was greatest in June with similar counts reported from both Whites Island and Campbell Slough. Average dipteran density was also relatively high in April at Franz Lake; however, subsequent neuston sampling was not done at Franz Lake so little can be surmised about seasonal patterns at this site. While most insect species in the temperate zone become active during spring or summer, we often observe more than one peak in activity per year resulting from a succession of generations (Wolda 1988). An early peak in average density and biomass was observed at Whites Island in February for all macroinvertebrate taxa, as well as specifically for dipterans and amphipods.

Average densities of all macroinvertebrate taxa collected by neuston tows in 2015 and 2016 were significantly greater than previous years in both the emergent vegetation and open water habitats. The increase in open water densities in the last two study years was particularly apparent and was primarily due to greater counts of copepods and cladocerans. A number of spatial and temporal factors, such as land use changes or climatic variations (rainfall, temperature) have been shown to influence inter-annual

occurrence of biological communities (Nava et al. 2015). In the Columbia River, average daily river discharge in 2014 and 2015 were relatively low, however summer flows were similar to the long-term average (Hanson et al. 2016). Daily mainstem temperatures in 2015 were higher than the long-term average throughout the spring and summer until approximately September (Hanson et al. 2016). The repercussions of such changes in mainstem conditions on biological communities are unknown, though changing conditions are likely to affect the many macroinvertebrate taxa differently. We also note that a different laboratory (University of Washington) began analyzing the neuston samples in 2015. While efforts were made to ensure sample collection and processing protocol was consistent, greater densities in 2015 and 2016 compared to previous years may have been due to this.

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). This study showed amphipods contributed most to juvenile Chinook salmon diets from Welch Island, ranging between approximately 50 and 60 percent of the total IRI. Amphipods also contributed to Whites Island diets, but were absent in those from Campbell Slough and accounted for only one percent of the IRI from Franz Lake. Dipterans were consumed at all sites, with contributions ranging between 10 (Welch Island, April) and nearly 90 (Campbell Slough, May) percent of the total IRI. This shift from diets dominated by amphipods and dipterans at the downriver sites to primarily dipterans and other insects at the upriver sites has been consistently shown at the trend sites over the study years. Diets in April 2016 were distinguished from other months and previous study years by the presence of cladocerans at Welch Island and Whites Island, odonatans at Campbell Slough, and copepods at Franz Lake. Cladocerans and copepods were both numerically abundant in these diets, but contributed significantly less to the gravimetric composition. Consequently, the amount of energy attained from these taxa by juvenile Chinook salmon was much less relative to their percent of the IRI.

In 2015 and 2016, Franz Lake was the most different in feeding quality compared to other sites. Juvenile Chinook salmon stomachs from Franz Lake were less full, reflecting a lower feeding intensity and energy consumption compared to other sites. We also noted that condition factor was lowest in juvenile Chinook salmon captured at Franz Lake (see Section 3.6.3.2). It was the only site where copepods were consumed in significant quantities—up to 65 percent of the IRI at Franz Lake in April, and never more than one percent at other sites. While differences in consumption and feeding intensity may indicate site differences in prey supply and availability, dipteran densities at Franz Lake in April were high relative to other sites, and comparable to peak dipteran densities in June at Whites Island and Campbell Slough. Also, while copepods were very abundant in April 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. Sampling at the trend sites has shown Franz Lake to have a consistently more diverse fish community than the other sites (Hanson et al. 2016). The presence of competitors may cause fish to consume lower quantities of energetically advantageous prey taxa and is one possible explanation for the unique Chinook feeding patterns at this site.

Conditions affecting the growth potential of juvenile Chinook salmon vary over both spatial and temporal scales in the estuary. Estuarine habitat opportunity (e.g. temperature, water depth, and salinity) interacts with habitat capacity (e.g. 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. Very few fish experienced low energy assimilation and high metabolic costs at 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. While only sampled 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 (such as in Figure 95) may be a useful tool to allow us to evaluate habitat quality across various time scales. For example, it could tell us how a 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 the long-term averages from the trend sites to provide an evaluation of the new habitat relative to other areas in the estuary.

4.6 Fish

In 2017, four of five trend sites—Ilwaco Slough, Welch Island, Whites Island, and Campbell Slough— were sampled monthly from February through June, and September. We were unable to sample the fifth site, Franz Lake, due to elevated water levels in the late-winter and spring and due to a forest fire which closed the Columbia River and access roads in September. The river flows in 2017 were some of the highest since 2011 and were in stark contrast to the previous two years. The higher flows helped maintain water temperatures below average for most of the migration season, resulting in decent conditions for out-migrating juvenile salmon.

Some patterns of fish community composition remained unchanged from previous years while other metrics varied. 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 upriver at Campbell Slough 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 (and Franz Lake) were much greater than the lower river sites. In 2017 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 sample collection concluded in June, whereas in previous years sample collection typically extended into 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 measure of species richness and diversity at Campbell Slough in 2017 was the highest observed for the year (Figure 91) and likely signals an increasing trend for the summer months. The increased species diversity in the upper reaches of the estuary is primarily driven by non-native species. 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 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 salmonid species composition vary by year and more strongly by site/hydrogeomorphic reach. Chum salmon dominate catches at Ilwaco Slough in Reach A, 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, but further examination and modeling of local currents and tidal forcings would be needed before drawing any linkages. Coho salmon have frequently been observed at Franz Lake, but in the recent low flow years of 2015-2016, they have been less abundant (Figure 94). We were unable to sample during 2017 because of high flows, so it is unclear if the recent decrease in coho at Franz Lake is flow related. 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 (Figures 95 and 102).

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, especially at Welch and Whites Islands. Spring Creek group stock tend to increase in proportion during May, concurrent with large hatchery releases, and interior stocks tend to show up in greater numbers during the summer.

The temporal distributions of Chinook and chum salmon indicate the separation between the timing of estuary use. Chum salmon densities peaked in March, whereas Chinook salmon densities increased in April, peaked in May, and then started to decline. This pattern of estuary use is similar to patterns of abundance found by Roegner et al. (2012). Densities of unmarked and marked Chinook were consistent with previous years, yet tended to be toward the higher end of the range. This is especially evident at Welch Island where the 2017 density was nearly twice the previous high density measured in 2014 (Figure 97). Greater densities were likely influenced by the higher flows in 2017. However, we did not see the same increase in density at Whites Island, which may indicate that differences in gear efficiency under higher flow condition at the two sites might impact catches. The site at Welch Island is a broad gradually sloping beach, whereas Whites Island has steep marsh edges, a narrow channel and greater current than Welch Island. During a low flow year like 2015, densities of unmarked Chinook were lower, yet still within the range of variance. This relative consistency in densities across years demonstrates the importance of tidal wetlands to juvenile Chinook salmon. Chinook are seeking out 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) 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 all trend sites are consistent over the years with Chinook salmon at Campbell Slough having a slightly higher value and Chinook at Franz Lake and Ilwaco Slough having slightly lesser values. Measures of percent lipid content are variable over time across sites. In 2017 lipid content analysis was available for three sites: Welch Island, Whites Island, and Campbell Slough. Campbell Slough showed the highest percentage of lipids (2.58%), while values for Welch (1.82%) and

Whites (2.03%) Islands were also on the higher end of ranges within each site. These values of lipid content for juvenile Chinook within the Columbia River estuary 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) 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). 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 salmon move from colder tributary waters to warmer estuarine habitats with large capacities of prey production. However, factors such as the transition to brackish environments and maintaining position in an increasingly tidally driven habitat may help explain the paradoxical pattern (Chittaro et al. 2018). Additionally, 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 result in increased competition for food, underscoring the importance of tidal habitat restoration to the recovery of salmon stocks.

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 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. 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 and Steelhead 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 (Figure 1). 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, were rarely observed. However, coho salmon, which also has a yearling life history strategy, were frequently caught in Reach H closest to Bonneville Dam. Chum salmon, which have a subyearling life history, were the second most frequent species observed. Chum were seen at all sites, and their use of tidal wetlands peaked in April and was limited temporally from March-May. Chinook salmon, which have both yearling and subyearling life histories, were the predominant species observed in tidal wetlands. Subyearling Chinook salmon, in particular, represented 90% of the total salmonid catch. In contrast to chum, subyearling Chinook salmon demonstrate the protracted use of tidal wetland as evidenced by the presence of fry (<60 mm fork length [FL]) and fingerlings (60–115 mm) from February–June. Peak density of Chinook salmon occurred in May at all sites (Figure 96). These results support the findings of other studies of tidal wetlands and shallow-water habitat in the lower Columbia River (Bottom et al. 2011b, Roegner et al. 2012, Sather et al. 2016, Teel et al. 2014). However, newly emerging evidence suggests that the timing of sampling that occurs in tidal wetlands excludes yearling life histories, implying that yearlings may be under-represented by traditional sampling methods (McNatt et al. In Prep).
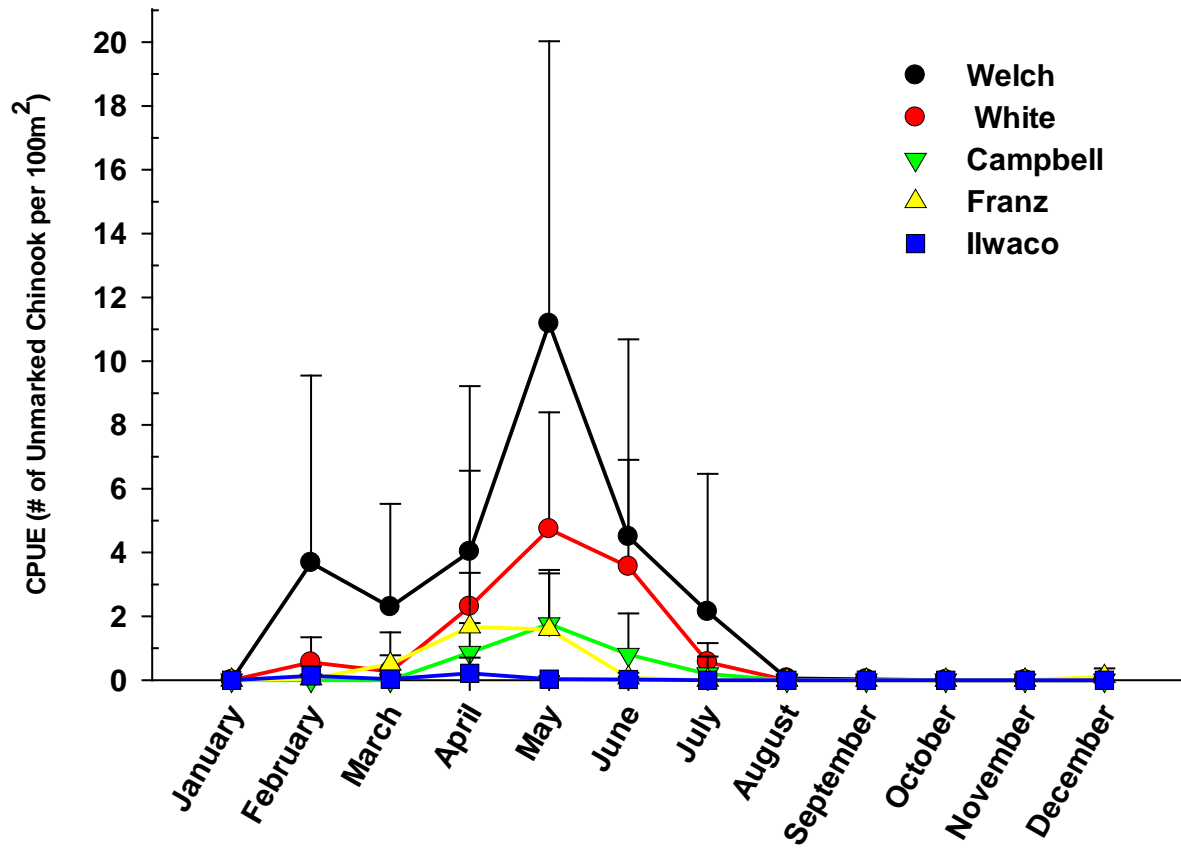
Figure 96. Monthly average density (CPUE/100m^2) of unmarked Chinook salmon at trend sites, 2008-2017.

Hatchery releases influenced the demographics of our salmonid catches. For example, coho peak abundance at Franz Lake (Reach H) occurred in May and was driven by hatchery releases. However, a smaller peak of natural origin coho also occurred at Franz Lake in late fall–early winter, leading us to infer that the site provides important over-wintering habitat (Figure 97). Across all of the sites mean Chinook salmon fork length remained close to 40 mm from February–April and was indicative of the influx of newly emerged fry (Figure 98). However, in May, mean fork length increased by ~20 mm and was coincident with hatchery releases of fall Chinook sized 80–90 mm. Mean fork length of unmarked subyearling Chinook also increased during the April–May timeframe. Likely causes for this trend include an influx of larger fish that had reared in natal streams, and fish growing as they reside and rear in the estuary for an extended period. Increases in mean fork length may also coincide with seasonal increases of prey and water temperature.
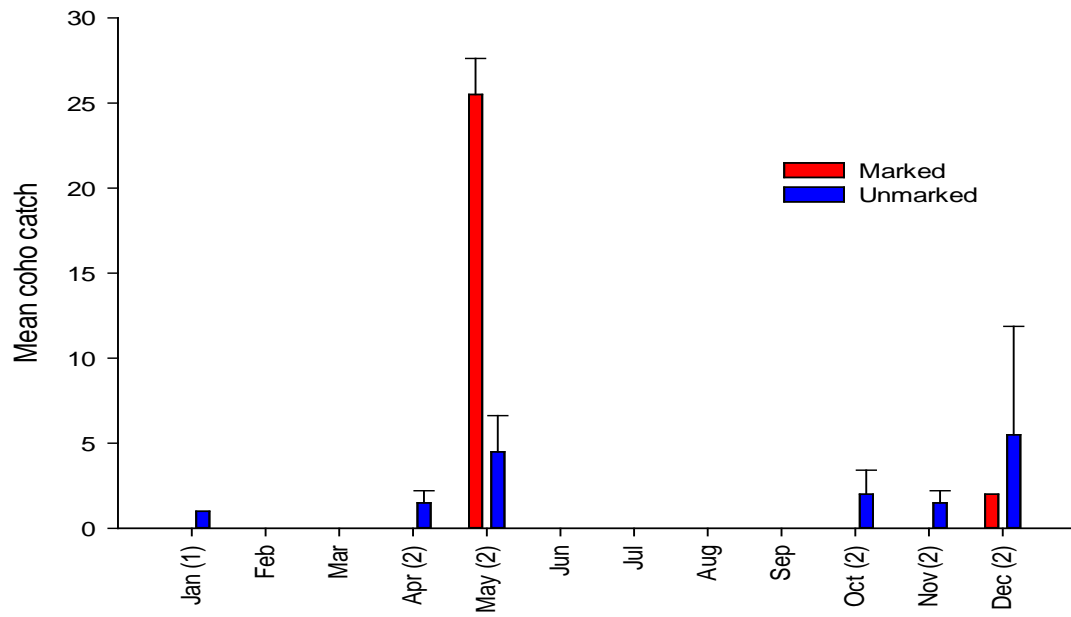
Figure 97. Mean monthly abundance of coho at Franz Lake (2008–2016).
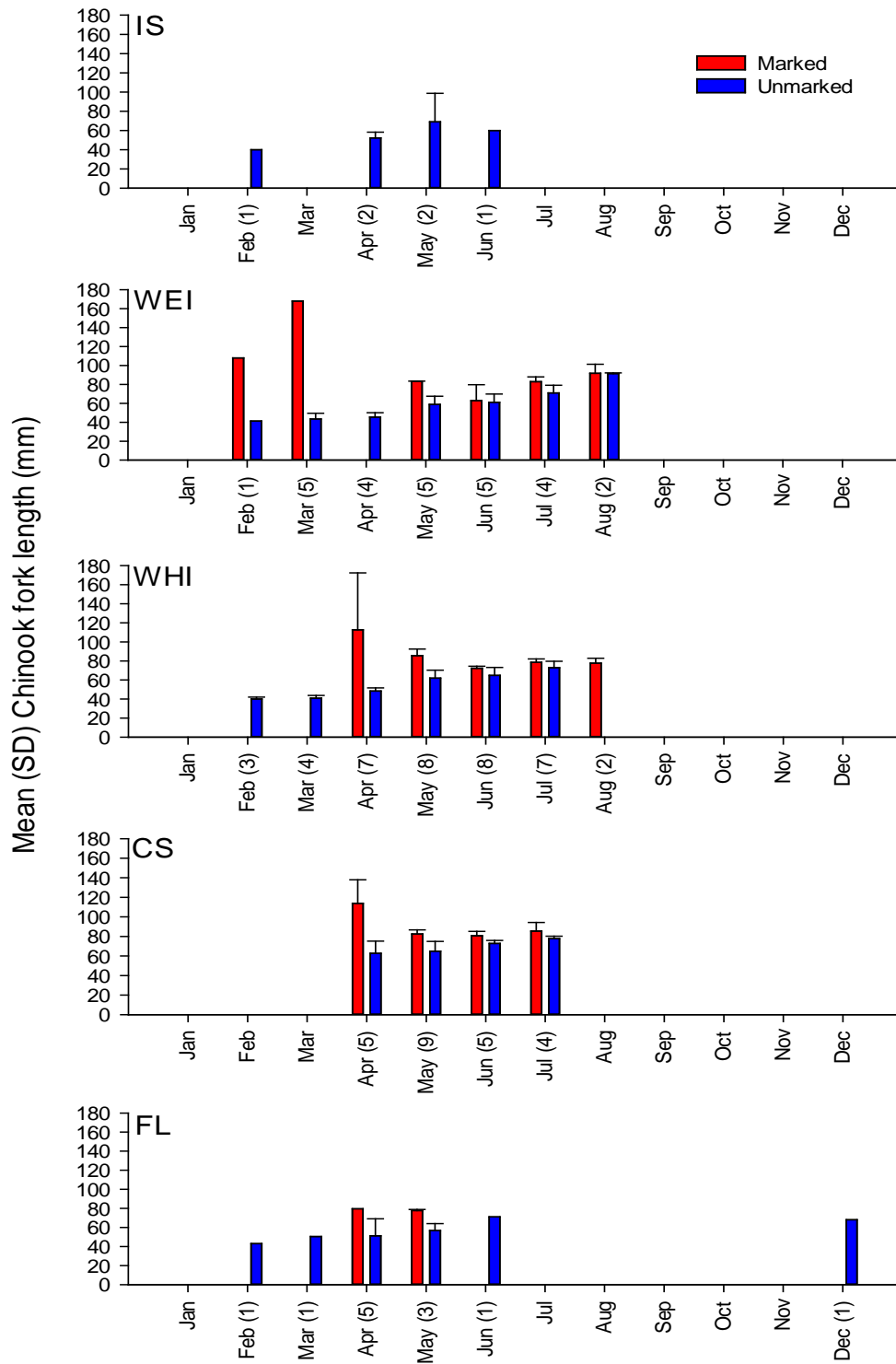
Figure 98: Mean (SD) monthly fork length of Chinook salmon at each trend site (2008-2016).

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5.2.2 Fish Condition

The condition of unmarked Chinook salmon increased over the course of the migration period, with the lowest values observed in February (mean K=0.75) and the highest values observed in July (mean K=1.23) (Figure 99). Limited data indicated that after July condition starts to decline, likely due to high water temperature. Condition factor of marked Chinook salmon was narrower than for unmarked Chinook salmon (0.94–1.22), but still trended upward during the migration season. Similar to unmarked fish, condition factor appeared to decrease in late summer (Figure 100).
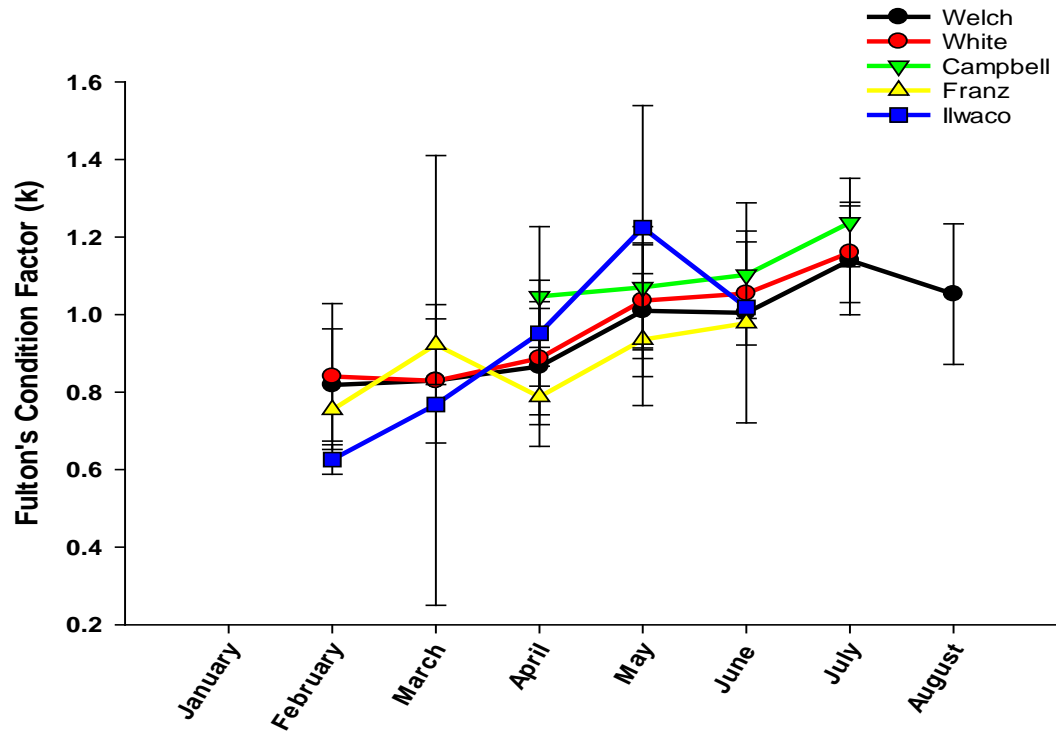
Figure 99. Monthly mean condition (K) of unmarked Chinook salmon captured at the trends sites 2008–2017.
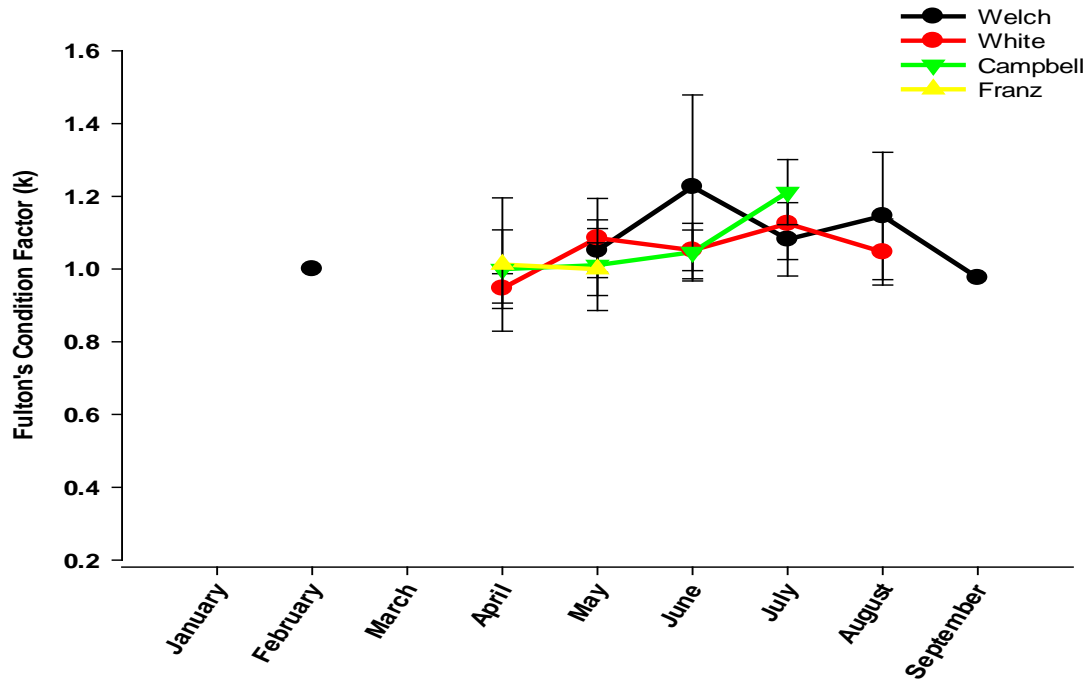
Figure 100. Monthly mean (SD) Fulton's condition factor of unmarked (top panel) and marked (bottom panel) Chinook Salmon, 2008-2017.

Somatic growth analyses from otoliths 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 in Reach B for 15 days or more.

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.

5.3   Characterization of Habitat Conditions in the lower Columbia River

River systems comprise a complex and diverse set of habitats, including deep channels and shallow, off-channel areas. The latter are often characterized by high organic matter production that is transported to the deeper main channels, where it supports the riverine food web (Jassby and Cloern, 2000, Moreira-Turcq et al. 2013). In addition, off-channel habitats can act as either sources or sinks of dissolved nutrients in a river floodplain ecosystem (Junk et al. 1989), depending on relative rates of nitrification (Strauss et al., 2004) or denitrification (Houser and Richardson 2010), for example. These processes tend to occur at higher rates in shallow floodplain areas compared to the main channel (Houser and Richardson 2010) and can result in significant retention within the floodplain (Venterink et al. 2003).

Salmonid access to tidal wetlands is influenced by hydrologic connectivity, water depth, water temperature, dissolved oxygen, and other physical conditions. Access to tidal wetlands varies spatially,

with a daily tidal connection at lower river sites, and annually and seasonally in the middle to upper river sites with mainstem river flows. At sites closer to the river mouth (Ilwaco and Welch Island), tidal influence and winter storms have a stronger influence on surface water levels 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 while at the farthest upstream trend site at Franz Lake, the tidal signal is difficult to discern from the influence of dam operations. Additionally, Welch and Whites islands in Reach B and C, respectively, are closer and more well connected to the mainstem; they have a greater similarity to river conditions compared to Campbell Slough (Reach F) and Franz Lake Slough (Reach H), which are further afar from the mainstem and comparatively more isolated once water recedes after the spring freshet (Jay et al. 2014).

5.3.1 Assessment of Connectivity among Trends Sites in the Lower Columbia

In order to characterize the importance of shallow, off-channel habitats for riverine productivity, some idea of the degree of connectivity among off-channel and mainstem habitats is needed. Connectivity, defined by Pringle (2001) as "the water-mediated transfer of matter, energy and organisms between elements of the hydrological cycle", is influenced both by the magnitude of river flow and with geomorphic features (e.g., channel width and depth, and distance from the mainstem), or hydrogeomorphology (Scheidegger 1973, Ward and Stanford 1995). The degree to which habitat characteristics differ between channel and off-channel sites depends on the relative hydrological connectivity between them, as well as site-specific landscape features (Hamilton and Lewis 1990, Thomaz et al. 2007). Evidence for connectivity can come from observed differences in water characteristics (e.g., temperature) between the site and the mainstem, expressed as Δ(property) (here, ΔT^oC). Thus, small differences denote high connectivity, and larger differences represent lower connectivity[1] (Tausz, 2015).

Although the large-scale diking of floodplains has severed historical connections between productive shallow habitats and the mainstem river, limiting direct use by fish and reducing indirect benefits to aquatic species (e.g., export of organic matter to fuel food webs) (Fresh et al. 2005), none of the trends sites within the Ecosystem Monitoring Program is completely isolated from the Columbia's mainstem; however, the rate of exchange between off-channel, emergent marshes differs among the sites, with Welch Island and Whites Island in Reach B and C, respectively, having a greater similarity to river conditions compared to Campbell Slough (Reach F) and Franz Lake Slough (Reach H), which are comparatively more isolated once water elevation recedes following the spring freshet (Jay et al. 2014). These findings came from comparisons between high-resolution measurements of temperature in the off-channel sites compared to mooring located in the mainstem Columbia (Tausz, 2015).

5.3.2 Influence of Connectivity on Habitat Capacity

Shallow floodplains typically differ from deeper channels in a river's mainstem in terms of temperature, light availability, flow velocity, and dissolved oxygen concentration (Lewis et al. 2000, Amoros and Bornette 2002), which contribute to the site's capacity (Simenstad and Cordell 2000) to support juvenile salmonids. Since off-channel habitats are often much shallower than the main channel, they tend to warm faster when air temperatures rise in the summer months. Temperature is one of the most important environmental parameters controlling aquatic community structure because of its influence on metabolic processes as well as its effect on density stratification and thus gas exchange. High temperatures can negatively affect physiological functions in vertebrates, particularly among species of fish (Coutant

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[1] Note that this definition captures the importance of the rate of exchange of materials between sites rather than focuses on isolation vs. direct connection.

1977). Salmonids are highly sensitive to temperature fluctuations since temperature influences their ability to efficiently metabolize and grow (Beechie et al. 2013). Water temperature can have a direct impact on juvenile salmon usage of shallow water habitats since suboptimal growth and increased predation risk increases for juvenile salmon reared at temperatures above 16 °C (Marine and Cech 2004). If waters in shallow, refuge habitats are too warm, juvenile salmonids may avoid those habitats altogether and instead risk predation and starvation in deeper, cooler areas (Vigg and Burley 1991, Sommer et al. 2001). Temperature is also an important variable that influences lifecycle events, or phenology, in many organisms. For example, many aquatic insects use temperature as a cue for larval emergence, which can influence food availability for insectivorous fish (Ward and Stanford 1982).

In addition to direct physiological effects, warmer temperatures typical of off-channel habitats may negatively affect habitat quality, or capacity, in indirect ways. For example, high temperatures can exacerbate nutrient-driven eutrophication through increased rates of nitrification, carried out by nitrifying bacteria, observed at higher temperatures (Strauss et al. 2004). As surface temperatures warm, water column mixing is reduced, which often leads to blooms of high-temperature and high-light adapted phytoplankton such as cyanobacteria (Paerl and Huisman 2008), particularly when nutrient loads are high (Xu et al. 2010). Thus, in addition to potentially deleterious effects on salmon metabolism from elevated temperatures, autochthonous nutrient sources may fuel more intense algal blooms, for example as repeatedly seen in Campbell Slough and Franz Lake Slough during the summer (Hanson et al. 2016, Hanson et al. 2017). Interestingly, the warmest year on record—2015—had substantial cyanobacteria blooms at the less-connected sites (Campbell Slough, Franz Lake Slough); however, the blooms were not dominated by nitrogen-fixing taxa, but instead were dominated by non-nitrogen fixing taxa such as *Merismopedia* spp. This observation is consistent with the idea that nitrogen was not limiting to the growth of cyanobacteria, which would be expected under bloom conditions with high algal biomass and strong drawdown and suggests the potential for strong nutrient remineralization at this site at high temperatures.

In productive aquatic systems, warm temperatures are sometimes associated with hypoxia, particularly in environments where water residence times are long, and biological oxygen demand is high. Hypoxic conditions have been associated with spatial and temporal avoidance of particular habitats by juvenile fish (Craig and Crowder 2005, Ludsin et al. 2009), including salmonids (Birtwell and Kruzynski, 1989). Dissolved oxygen concentrations below 2 mL/L are considered harmful; a threshold of 6 mL/L has been set for optimal performance (Washington Department of Ecology). Based on these criteria, both Ilwaco Slough and Franz Lake Slough have been shown to have frequent sub-optimal dissolved oxygen levels, as demonstrated by the number of hours of levels below thresholds of 2, 4, and 6 mL/L each month. Although fluvial phytoplankton can produce excess oxygen via photosynthesis during the day, high respiration rates can occur at night when bacterial respiration rates are high, establishing a diel pattern (Tyler et al. 2009, D'Avanzo and Kremer 1994). As phytoplankton blooms senesce, respiration by decomposing bacteria exceeds photosynthesis; in highly eutrophic systems, hypoxia can persist throughout the diel cycle for extended periods and have detrimental effects on benthic organisms and fish (Paerl et al. 1998). This phenomenon is more common in lentic environments where slowly flushed water tends to become stratified, but it has also been observed in estuaries (Paerl et al. 1998, Stanley and Nixon 1992) and in river such as the Klamath, which is connected to shallow impoundments (Otten et al. 2015), and in shallow, backwater sites in the upper Mississippi (Houser 2005), where hypoxic conditions can develop due to intense respiration of organic matter.

Juvenile salmon are also susceptible to high and low pH levels. At high pH, ammonium (NH_4^+) becomes the toxic ammonia gas (NH_3) upon acid dissociation. Among the trends sites in the lower Columbia, wide pH fluctuations have been observed at Ilwaco, Campbell, and Franz Lake Slough. The fluctuations at Ilwaco follow the intrusion of ocean water during upwelling periods in the summer when low-pH, low-oxygen water can enter Baker Bay in Reach A. At Campbell Slough and Franz Lake Slough, in contrast,

low-pH waters occur in response to changes in carbonate chemistry that accompany strong growth of algae, which draw down CO_2 and drive pH upward. At night, CO_2 is produced through respiration, reducing pH.

5.4 Characterization of Salmonid Prey Conditions in the lower Columbia River

5.4.1 Juvenile Salmon Prey and Diet

The EMP study has consistently identified two major prey items consumed by juvenile Chinook salmon collected at the emergent wetland and backwater slough trends sites. These taxa, Chironomidae 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 and 2016 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 101). 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.
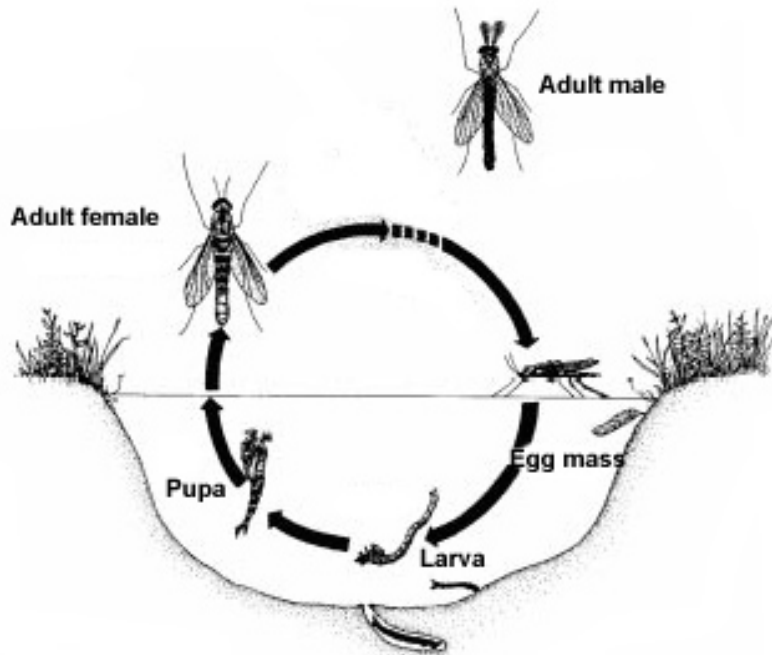
Figure 101. Life cycle of chironomid flies (image from North Carolina State University, Cooperative Extension).

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). This pattern is evident and consistent in results from the EMP study between 2008-2013, and 2015-2016. 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. While the number of juvenile Chinook diets from Ilwaco Slough in Reach A is limited to five fish collected in April 2015, they fed exclusively on amphipods. Roughly 40% of these amphipods were identified as *Americorophium*, with another 44% unidentified members of the Corophiidae family. Together, the trends sites demonstrate a shift in prey consumption along the estuarine gradient that is consistent with previous studies. 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.

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.

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 102). Farther up the estuary in Reach C, the river valley is more constricted, and areas of intermittently exposed habitat are typically limited to narrow sandy beaches (Simenstad et al. 2011).
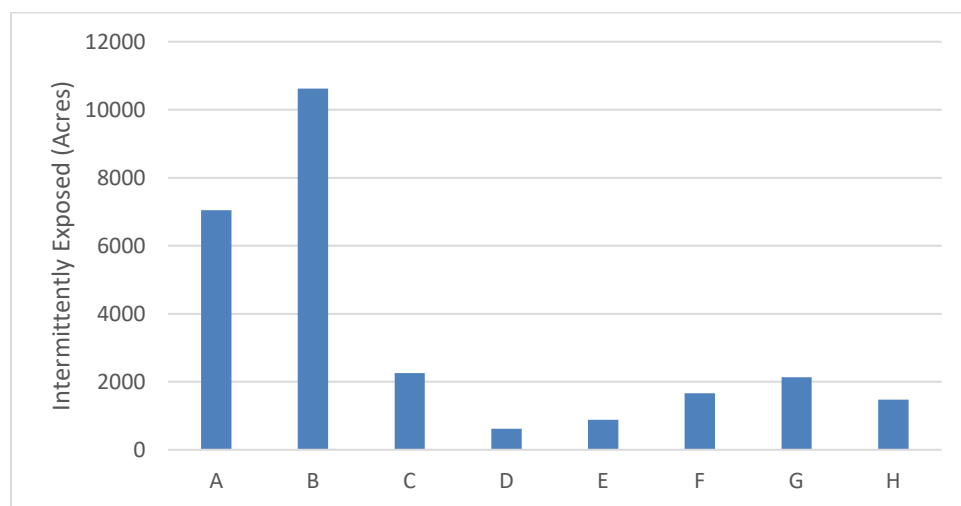
Figure 102. Area (acres) of the Intermittently Exposed Geomorphic Catena by Hydrogeomorphic Reach, as mapped in the Columbia River Estuary Ecosystem Classification.

Amphipods occurred regularly in monthly benthic core samples at Ilwaco Slough in Reach A between April and July in 2015 and 2016. 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.

We can begin to make comparisons of habitat quality across trends sites by evaluating average fish condition at each site. Energy ration is a measure of fitness that reflects both stomach fullness and energy consumed (based on the energy densities of prey taxa). Compiled over EMP study years for the months of April–June, mean energy ration from Franz Lake was lower than at the other trends sites (Figure 103). Franz Lake was the only site where copepods (a relatively energy-poor prey item) were consumed in large quantities, comprising up to 60 percent of the IRI (Index of Relative Importance) in April for the smallest size class, and never more than one percent at other sites. Elsewhere, for example in the Mokelumne River in central California, juvenile fall-run Chinook salmon have been shown to feed primarily on zooplankton and supplemented their diet with chironomids (Merz 2001). Differences in consumption and feeding intensity may indicate site differences in prey supply and availability. Dipteran densities in neuston tows at Franz Lake were variable, but at times high relative to other sites. Also, while copepods were very abundant in April 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. The EMP may benefit from a more rigorous prey sampling regime that could better identify patterns in abundance and help explain differences in habitat capacity between sites.
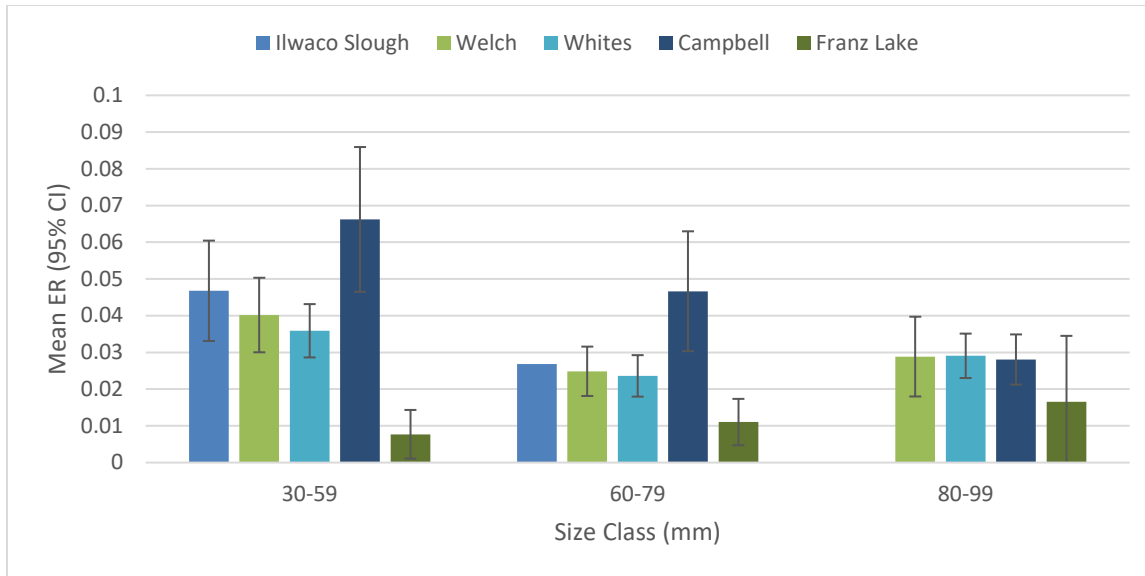
Figure 103. Mean energy ration (ER) by site and juvenile Chinook salmon size class (fork length, mm) compiled over EMP study years for months April, May, and June. Error bars represent the 95 percent confidence intervals.

In addition to differences in feeding patterns across sites, we looked at ontogenetic shifts in prey consumption based on fish length. Juvenile salmon are gape-limited predators and research from the Columbia River estuary has shown that larger juvenile Chinook salmon consume a more diverse range of prey than do smaller individuals, and consume greater proportions of relatively large benthic prey, such as *Americorophium* spp. (Lott 2004, Bottom et al. 2011). Prey selection associated with fish size was also evident from Welch Island and Whites Island. For example, in May at Welch Island, the percent IRI for amphipods ranged from 20% for fish between 30–59 mm length, to 78% for fish between 60–79 mm, and 72% for fish between 80–99 mm. This was coupled with a decline in the relative importance of dipteran in the diets of larger fish: 77% for the smallest size class, 22% for the mid-size class, and only 6% for juvenile Chinook between 80-99 mm. Larger fish collected from Whites Island also consumed more amphipods than their smaller counterparts. However, fish from Franz Lake and Campbell Slough maintained high IRI values for chironomids and other dipteran flies across size classes. We did not see evidence at trends sites in 2015 and 2016 of differences in prey diversity among size classes, according to Shannon-Weiner and Evenness Indices. Hemipterans and other non-dipteran insects were more likely to appear in the stomachs of juvenile salmon greater than 60 mm in length. The presence of hemipterans in the diets corresponded with their abundance in the environment, which tended to occur later in the summer.

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

5.5.1 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 104). Marsh plants provide more biomass and are a higher source of energy than plankton 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.
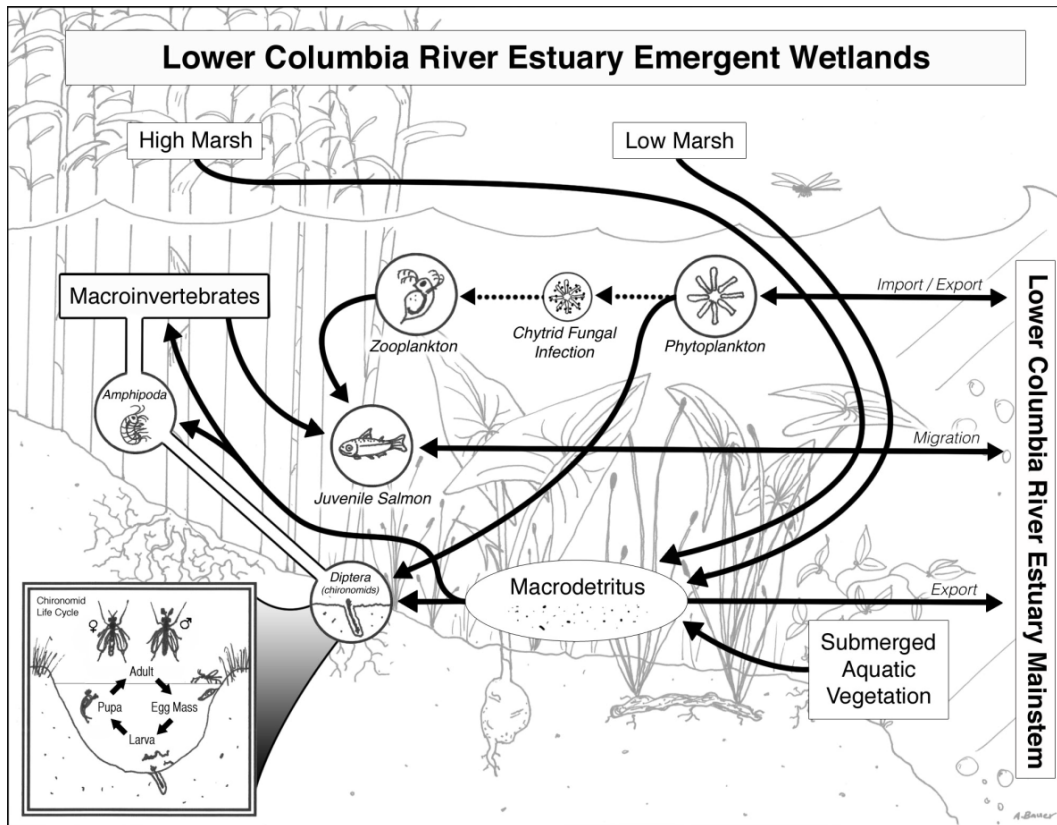
Figure 104: Conceptual model of food web interactions within Lower Columbia River emergent wetlands.

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.

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.

5.5.1.1 *Plant Assemblages in Columbia River Tidal Marshes*

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 28. 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 28. Cover (%) of native and non-native species at the 2016 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)

5.5.1.2 *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 105, 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.
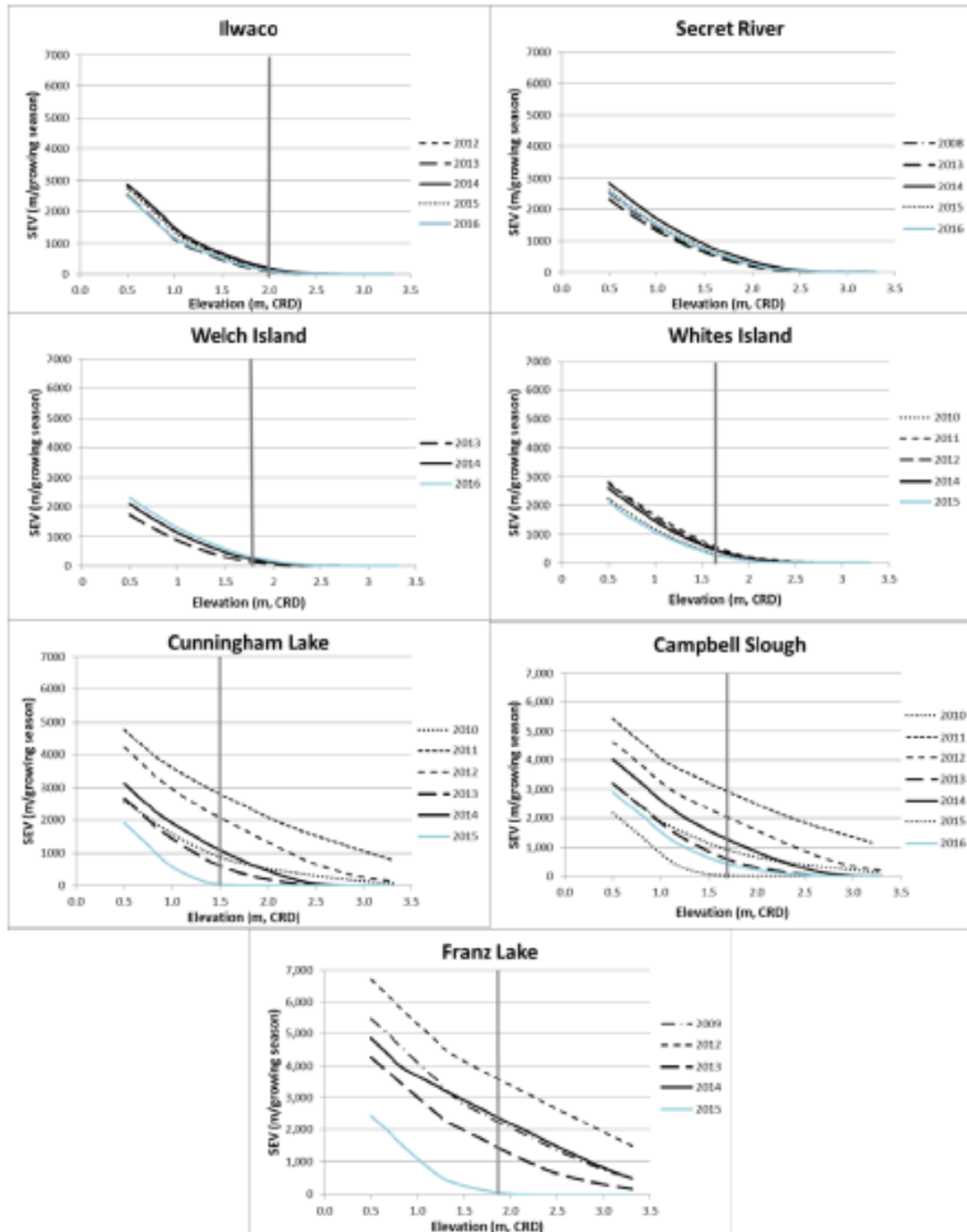
Figure 105. Cumulative inundation of wetlands during the annual growing season, as measured by sum exceedance values (SEVs). Plotted lines represent the calculated SEVs for a given year; the blue lines indicate the most recent year available. The vertical line represents the average elevation at each site. Sites are ordered with the site nearest the Columbia River mouth in the upper left panel.

5.5.1.3 *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 106). 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 107). 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.
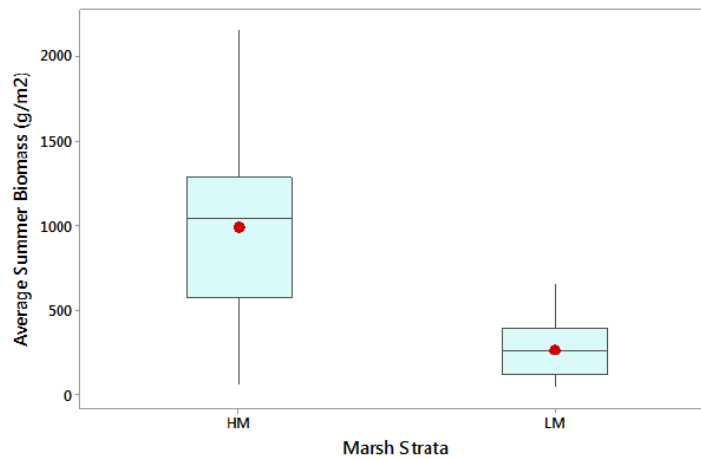
Figure 106. Overall average summer biomass (g dry weight/m^2) from the high marsh (HM) and low marsh (LM) strata.
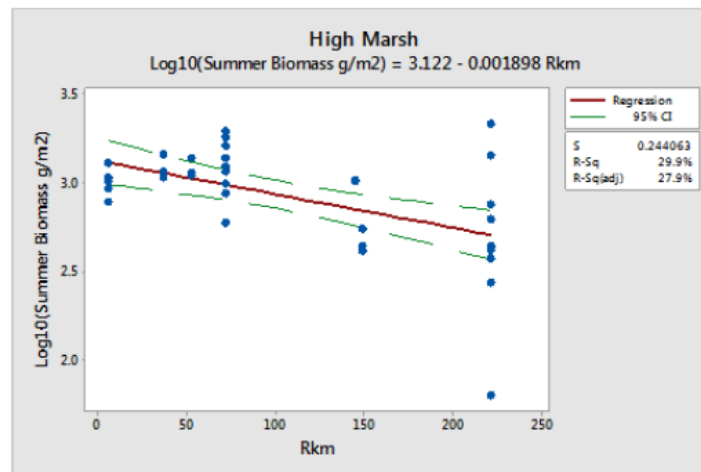
Figure 107. Average annual summer biomass (g dry weight/m^2) compared to river km for the high marsh strata. Results are transformed by Log10 for statistical analysis.

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.

5.5.1.4   *Phytoplankton Distribution*

Fluvial phytoplankton distributions in space and time 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). 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. 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).

5.6   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

The BPA now requests this section in the report to emphasize the application of results to management actions, adaptive management, and lessons learned. Please provide any ideas for the following:

1) *How could our results be used by fish and wildlife managers to inform program strategies (such as habitat restoration, predation, hatchery, or hydrosystem operation)?*
2) *How are our results applicable at the watershed, subbasin, and Columbia Basin scale?*
3) *Describe how your results will be shared with resource managers*

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 valuable for capturing the range of annual variability of environmental conditions, and the longer a monitoring program is implemented, the more descriptive the dataset becomes.

The lower river and estuary provides 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 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. In 2017, a higher than average number of days where water temperatures exceeded recommended temperature maximums in the mainstem were observed (2015 was the only other year that surpassed 2017 in this regard). Such a result is concerning since 2017 river discharge patterns were considered to be relatively high compared to other years, yet these higher flows 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 concentrations may cause hypoxic conditions that are harmful to aquatic life, as well as nutrient inputs that can trigger algae growth, including the proliferation of cyanobacteria. Monitoring the water quality in the lower river provides contextual information that identifies critical time 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 features 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 features to components of the food web to improve our ability to predict how hydrology will influence salmon production and survival. 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, noxious phytoplankton blooms can develop 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.

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. 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. Results from the EMP are presented and discussed at an annual Science Work Group meeting. In addition, EMP results will also be shared with regional partners at the Columbia River Estuary Conference in April 2018. 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. 2018) and, in 2018, contributed to the Columbia Estuary Ecosystem Restoration Program, Synthesis Memorandum (see Johnson and Fresh 2018).

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8 Appendices

Appendix A. Site Maps and Habitat Change Analysis
Site maps and habitat change analysis data collected in 2017 are still under analysis and unavailable at the time of the writing of this report. The following details are adapted from the 2016 report.
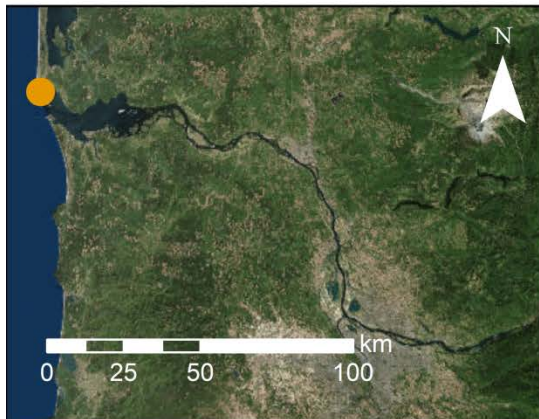
Contents:

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

km
0 25 50 100

Meters
0 50 100 200

229

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

230

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

231
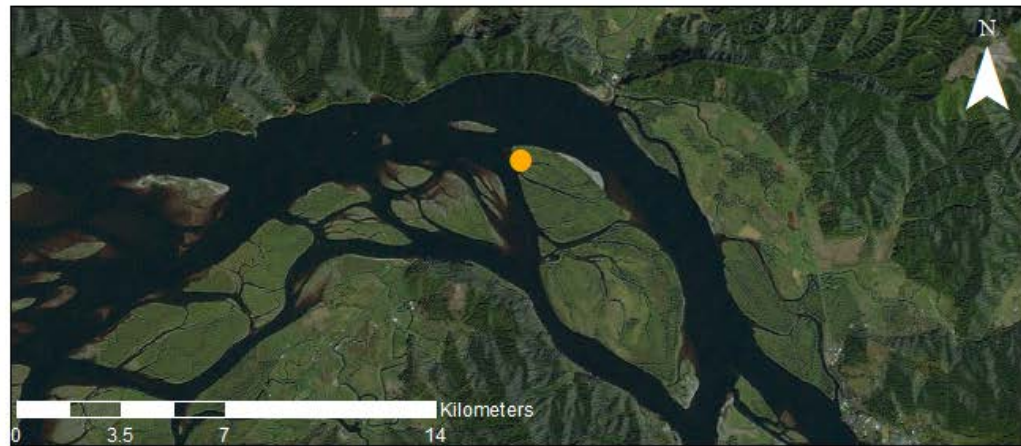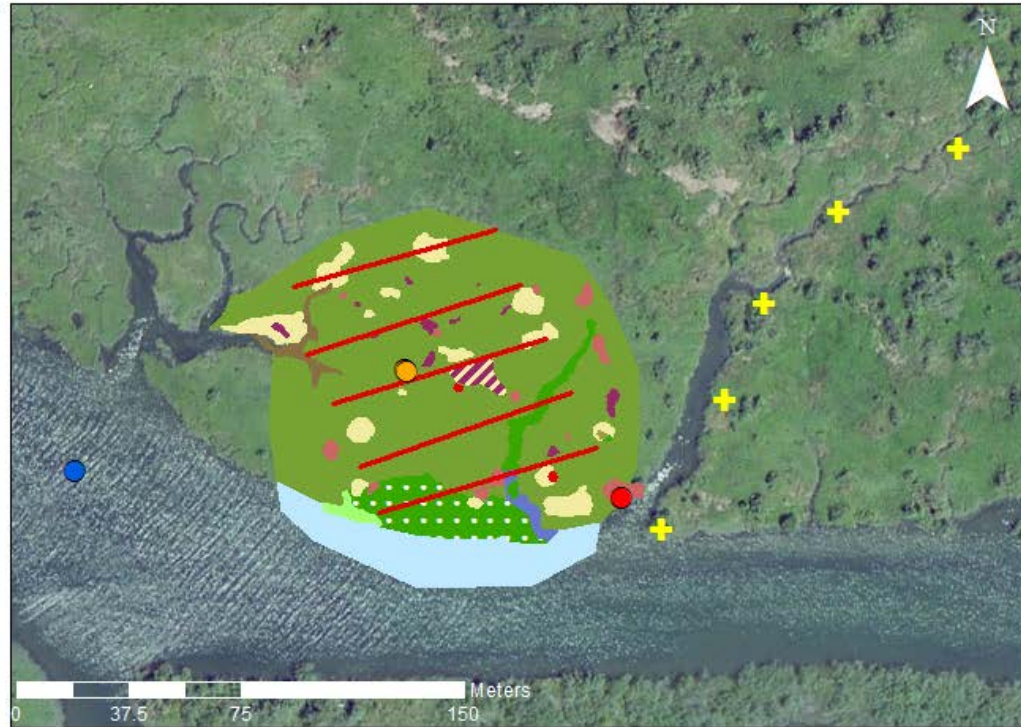
White's Island, 2015
GPS Mapping

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
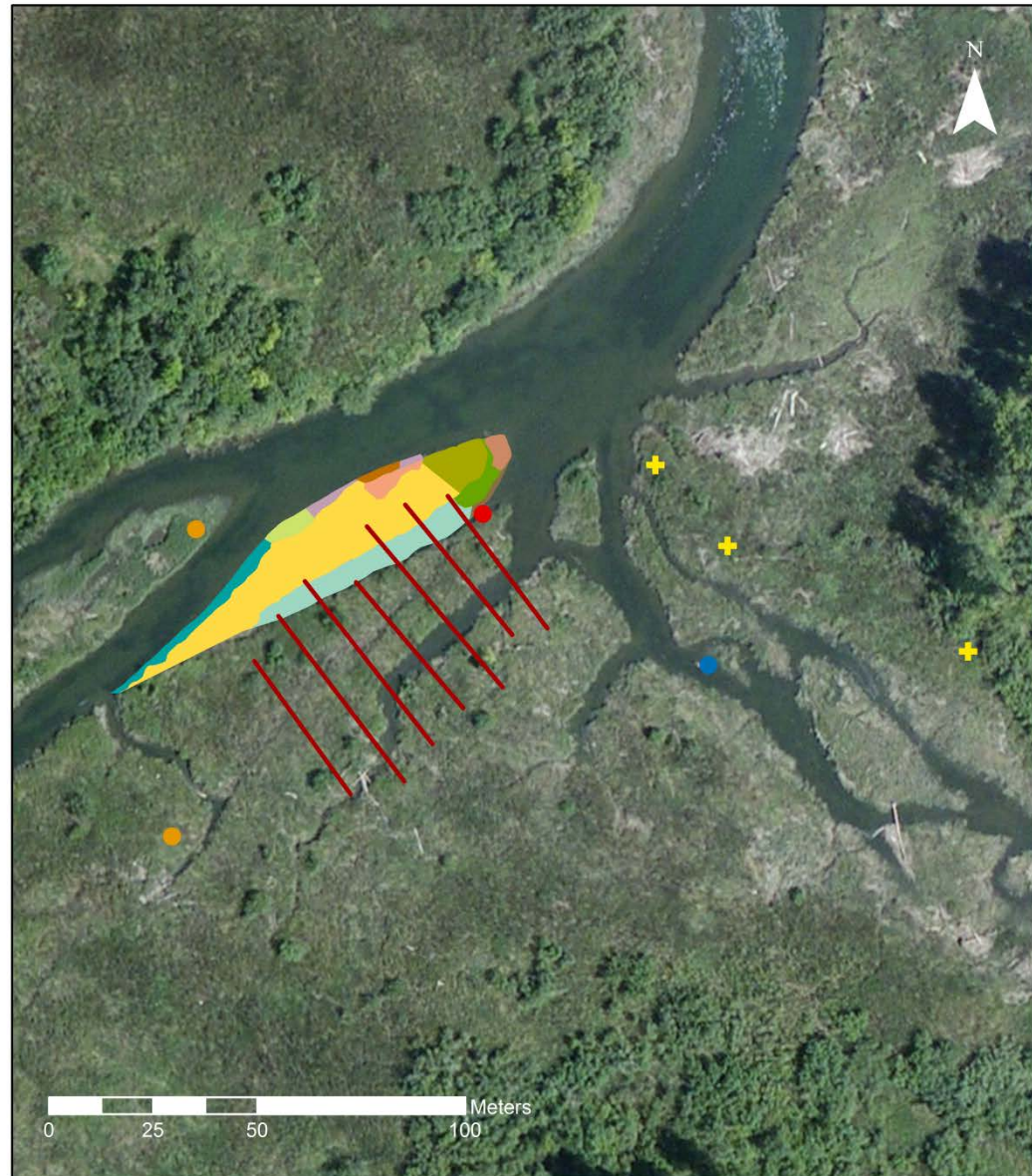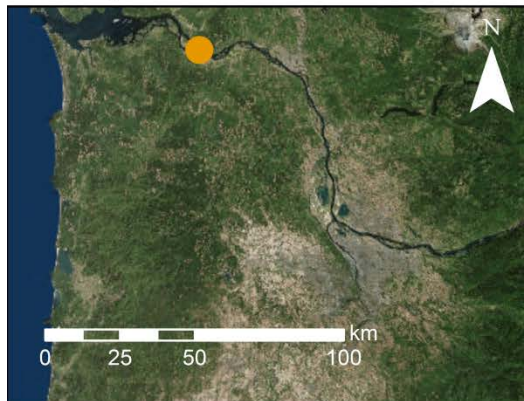
Cunningham Lake, 2015
GPS Mapping

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
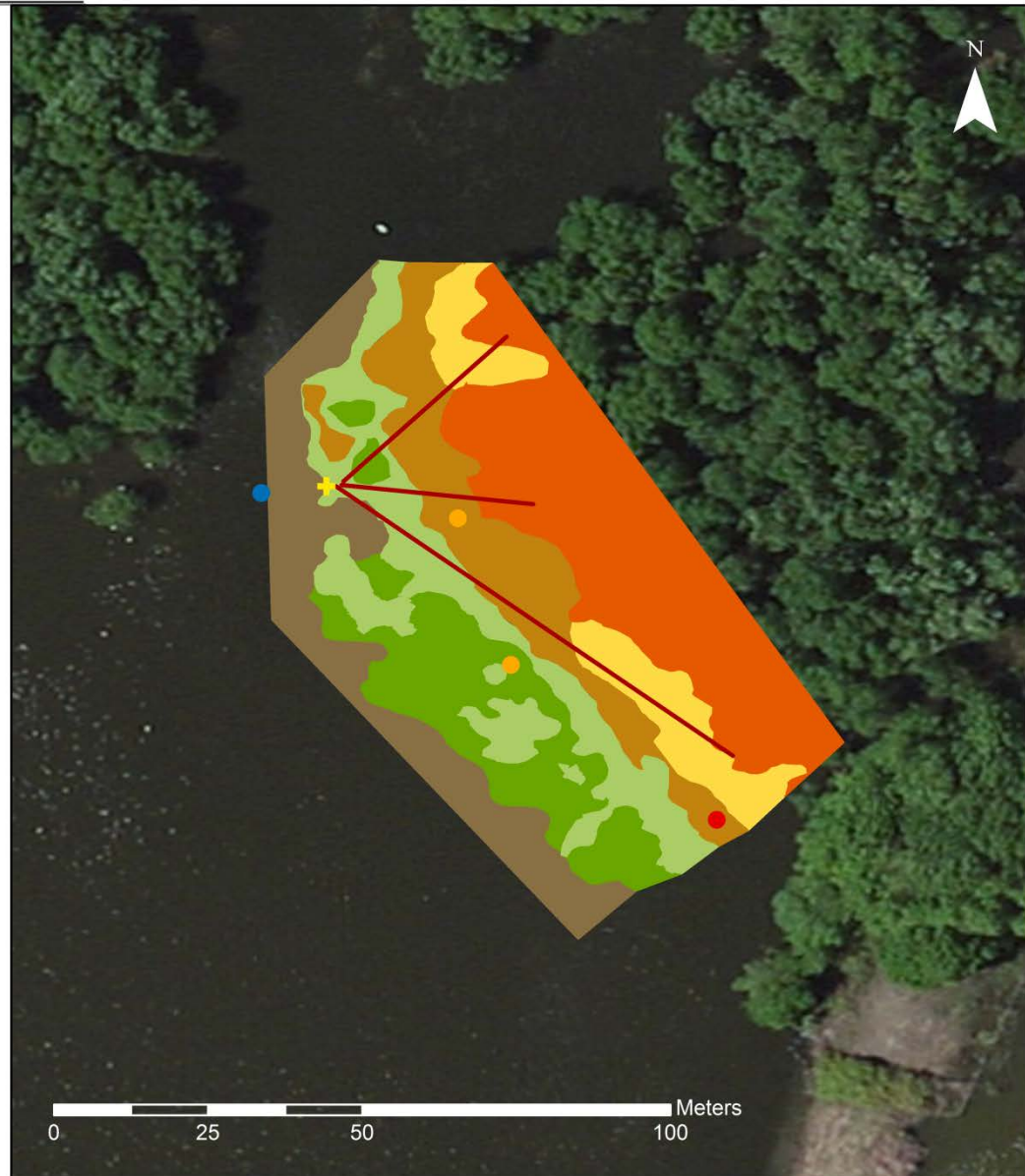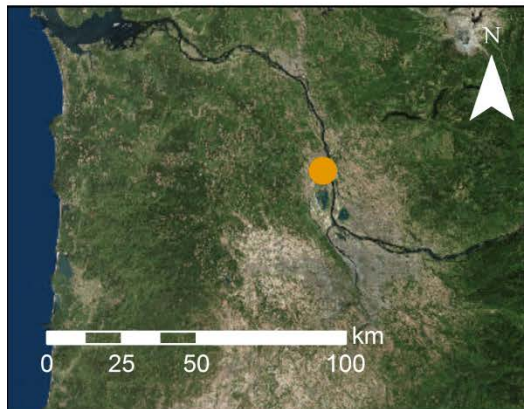
Campbell Slough, 2015
GPS Mapping

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

234
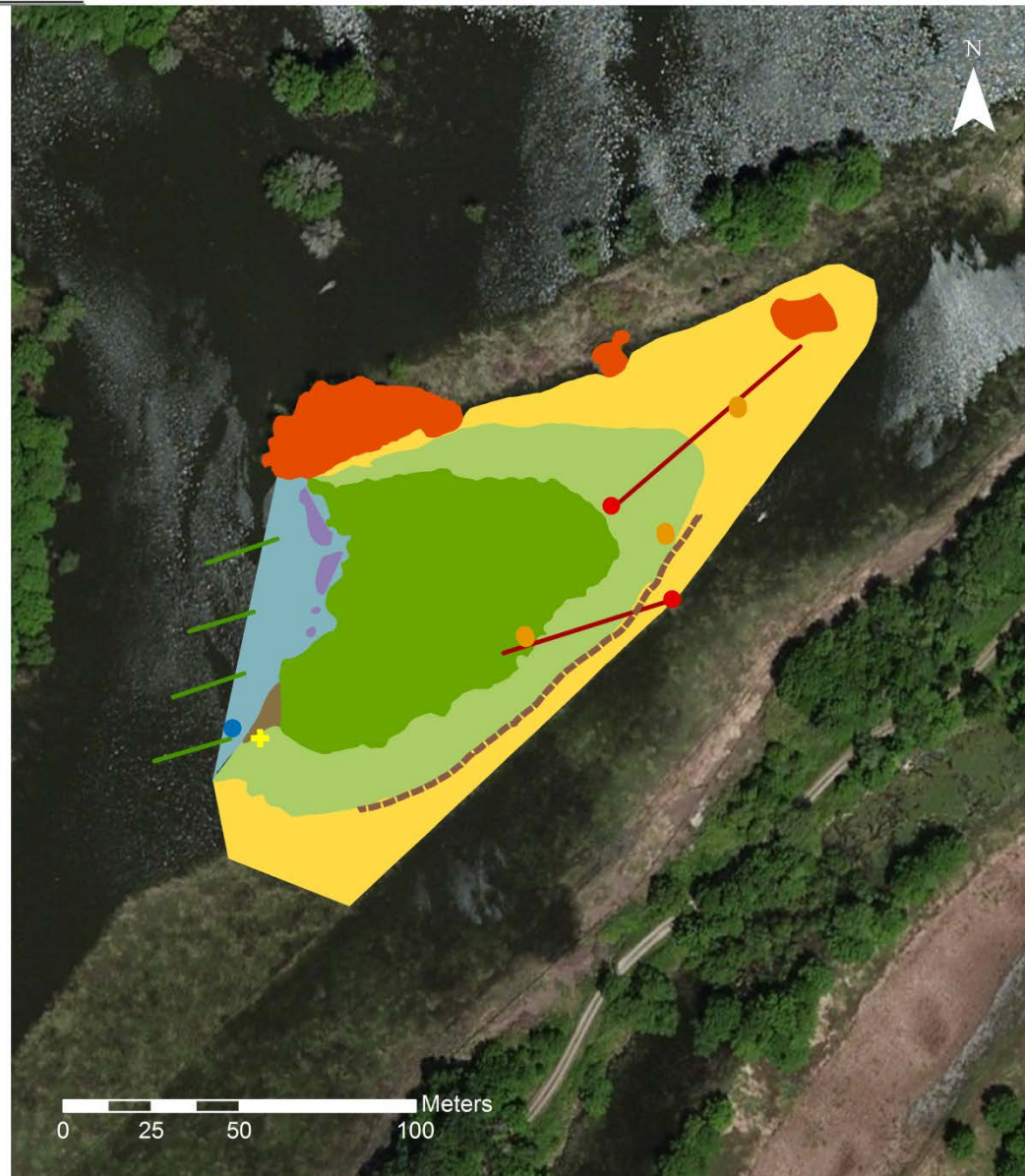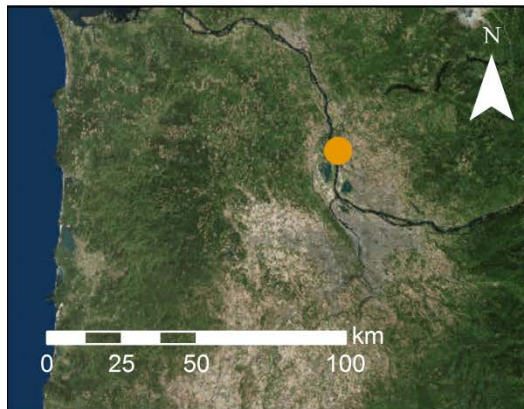
Franz Lake, 2015
GPS Mapping

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
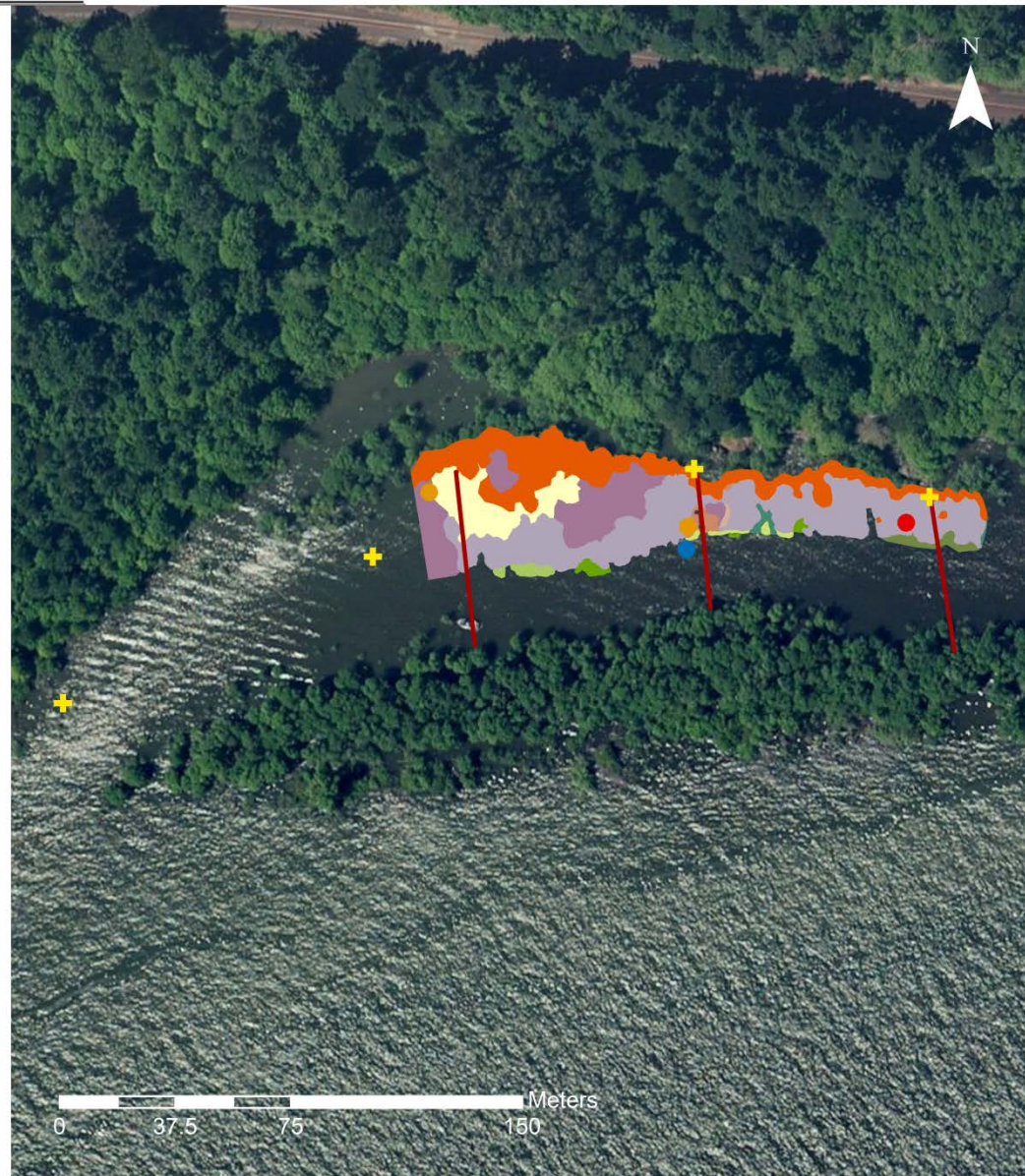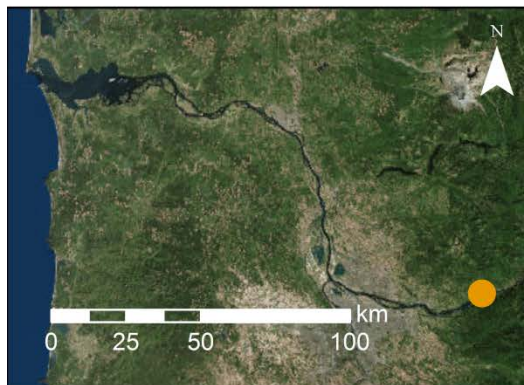
Table A1. Habitat change analysis of vegetation communities at the trend sites; comparison of overlapping areas for the earliest year mapped and the latest year mapped. All area units are square meters. Vegetation communities are ordered from the lowest elevation to the highest elevation at a site; species 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					AGSP, CALY	
Area Changed:	6416		Channel, ZAPA	Pan	CALY	AGSP, CALY	TYSP
No Change:	6895		1558	383	6455	4792	134
2015 Vegetation Community	Channel	2164	1548				
	Pan	804		241	559	4	
	Bare	127			68	58	
	CALY	3898	10	142	3045	85	
	AGSP, CALY	4048			1188	2860	
	AGSP, DECE, GLSP	1754				1754	
	TYSP	517			356	28	134

Welch Island

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

Whites Island

Area Compared:	1585	2009 Vegetation Community						
Area Changed:	729	Channel	SALA	ELPA	ELPA, SALA	ELPA, SCAM	CALY	PHAR
No Change:	855	**163**	**252**	**115**	**18**	**82**	**191**	**763**
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 labels)

Cunningham Lake

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

Campbell Slough

Area Compared:	13476	2005 Vegetation Community			
Area Changed:	2551	SALA	ELPA, 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 labels)

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
2012 Vegetation Community							
Channel	1182	1081	4	69	28		
Channel, SALA	147	112		35			
SALA	37	15	9	4	9		
ELPA	344	133	5	92	106	8	
CASP	178	6	6	39	121	6	
POAM, SALA	7				7		
POAM	2310	71		249	1645	344	
SASP, CASP	41					41	
SASP	1475				260	1180	35

Franz Lake

		2012 Vegetation Community												
Area Compared:	5203	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
2015 Vegetation Community														
SALA	41	9		11	13					8				
SALA, ELPA, CASP	79			19	27		9			24				
ELPA	25			7	6		11							
ELPA, SALA	58			45		5	8							
CASP	37				4		14			18				
PHAR, HEAU	49						18	31						
POAM	2131		8	40			69	93	25	1748	12			137
POAM, PHAR	867		6	72			47	27		152	103	135		324
POAM, SASP	10							10						
SASP	1387						18	31		170	39	8	20	1100
SASP, PHAR	522									39	43	136		304

239

Baker Bay
Vegetation Change 2011 to 2015

2011

2015

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
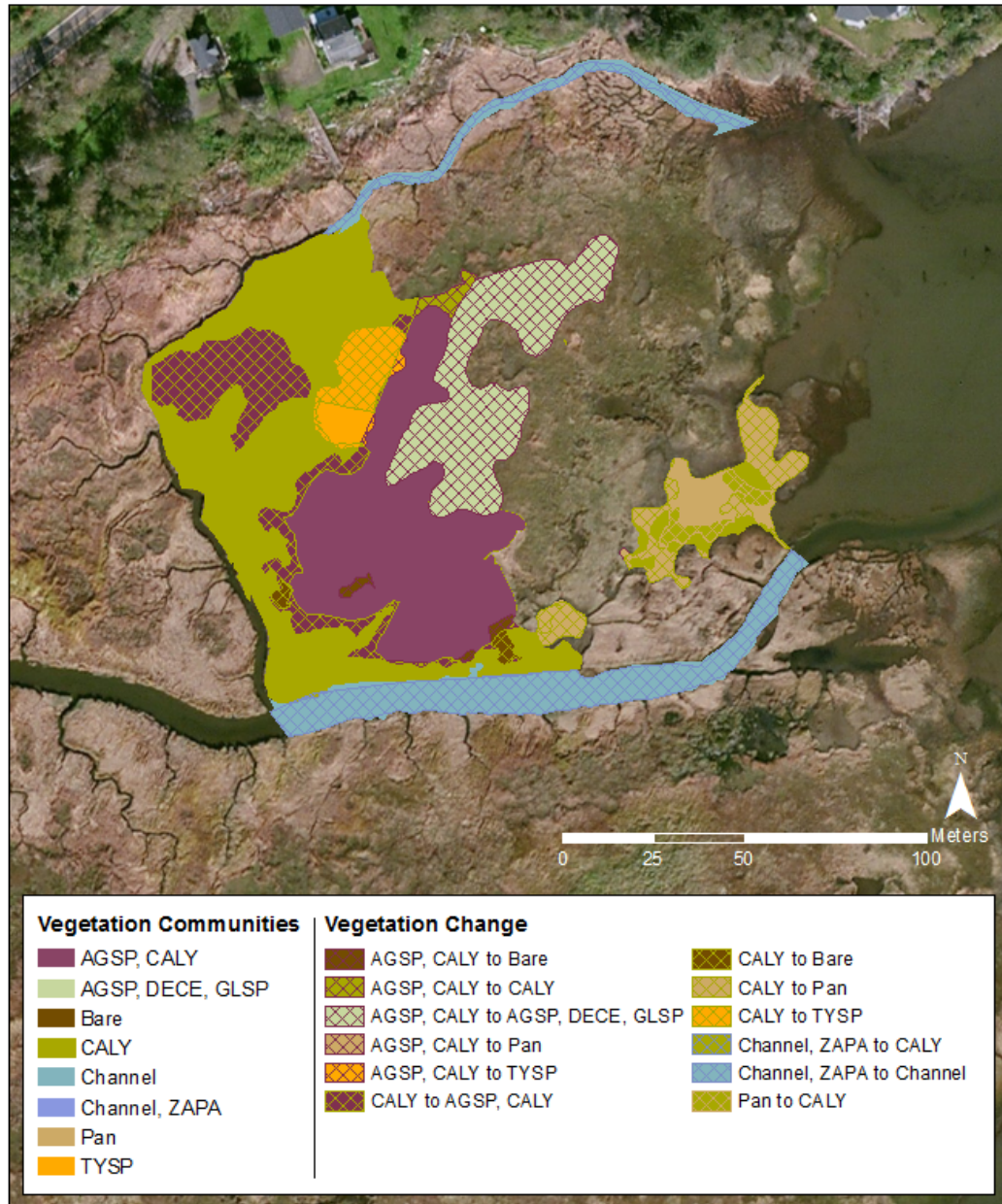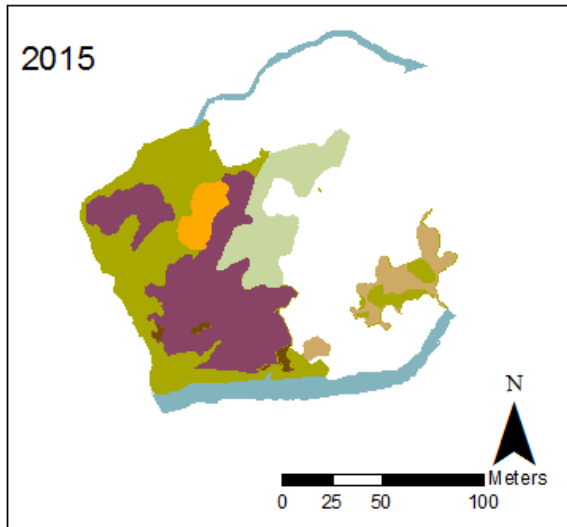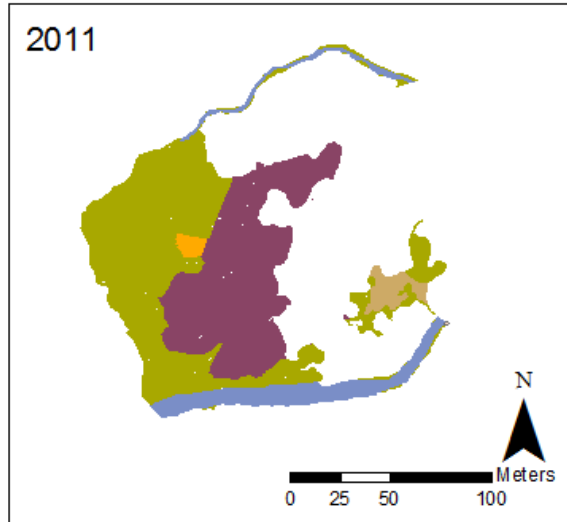
Welch Island
Vegetation Change 2012 to 2015
PHAR Specific

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

241
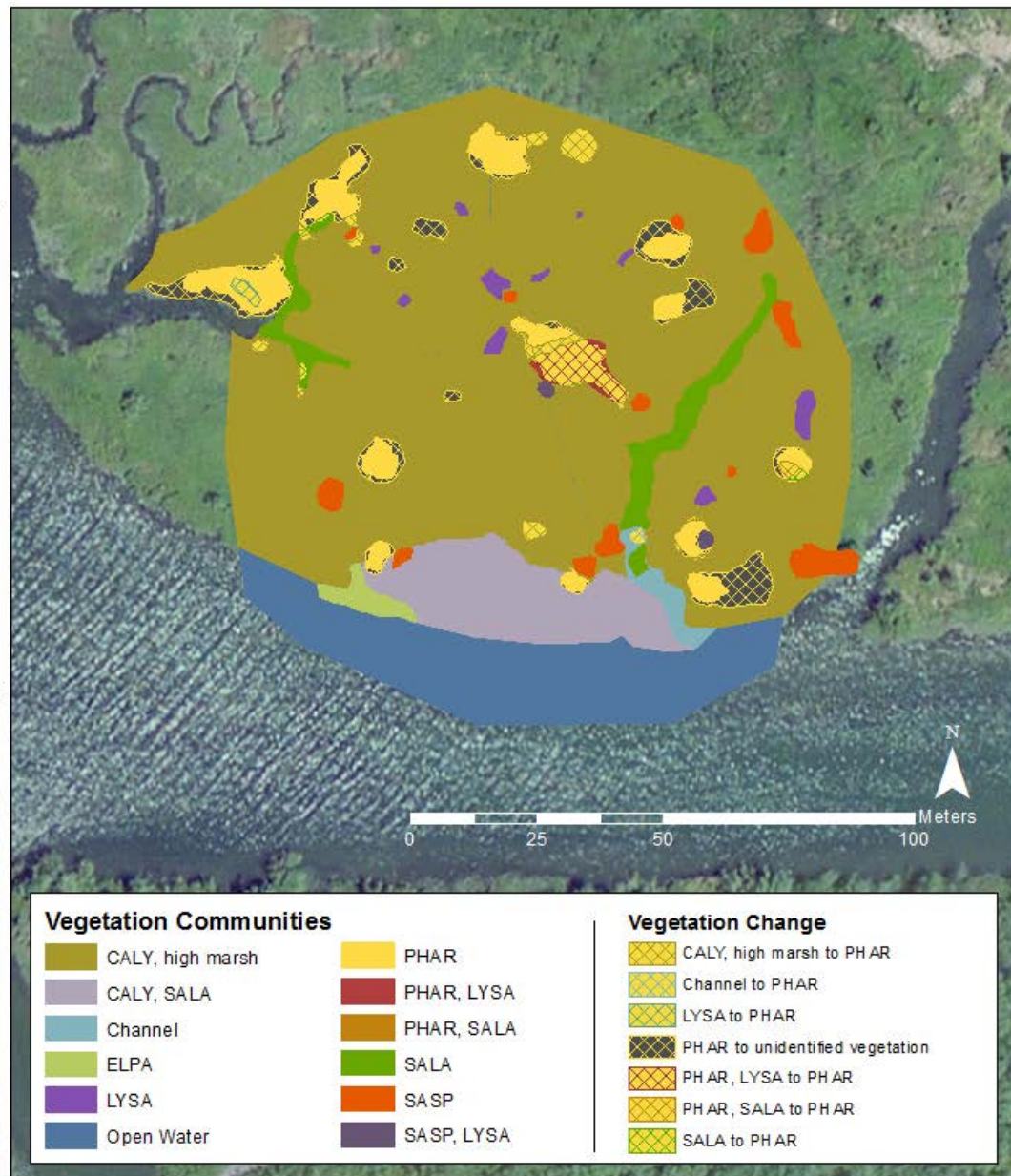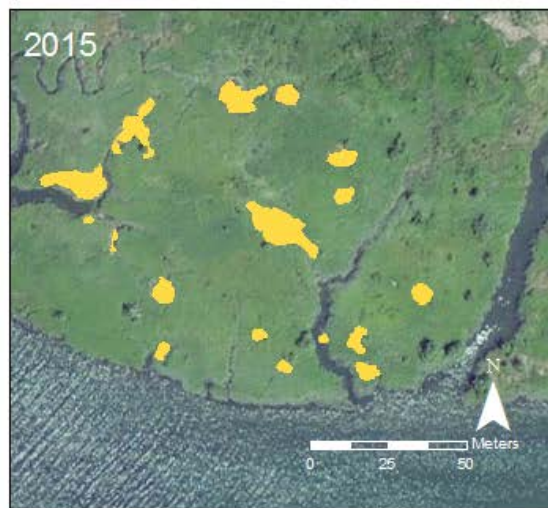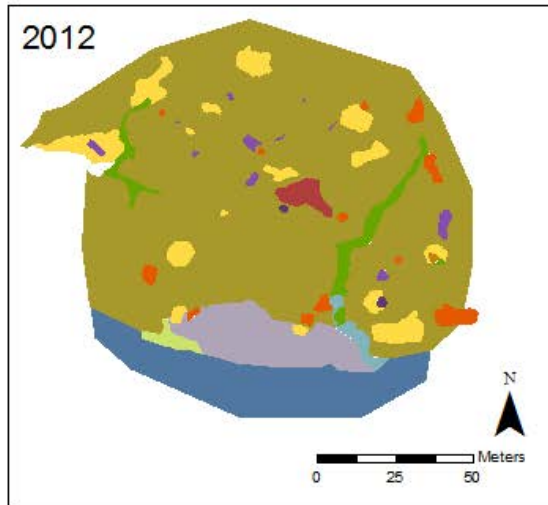
White's Island
Vegetation Change 2009 to 2015

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

242

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
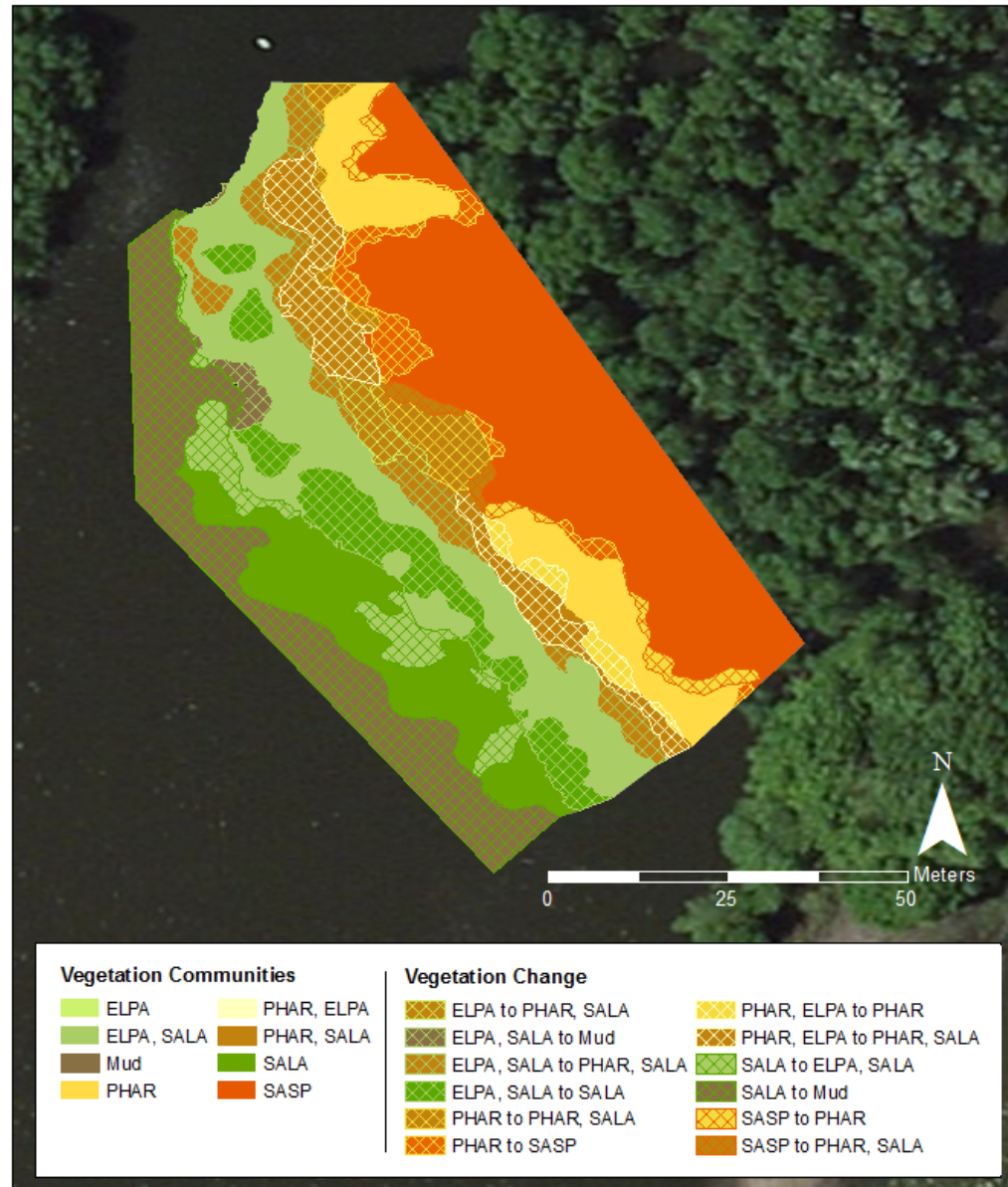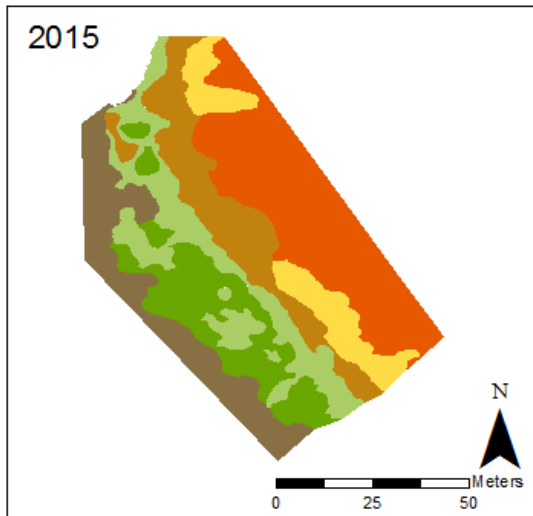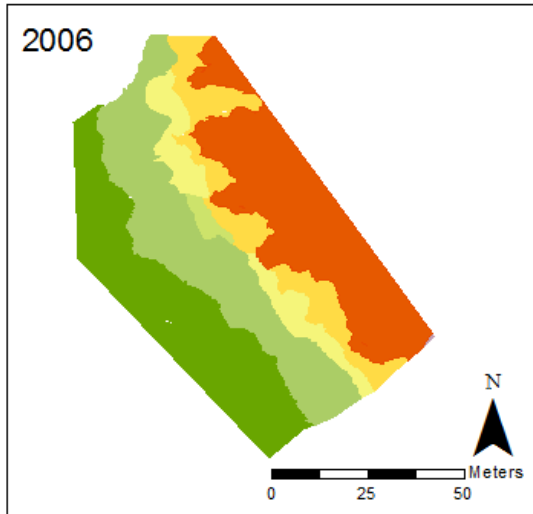
Campbell Slough
Vegetation Change 2005 to 2015

2005

N

Meters
0   25   50

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
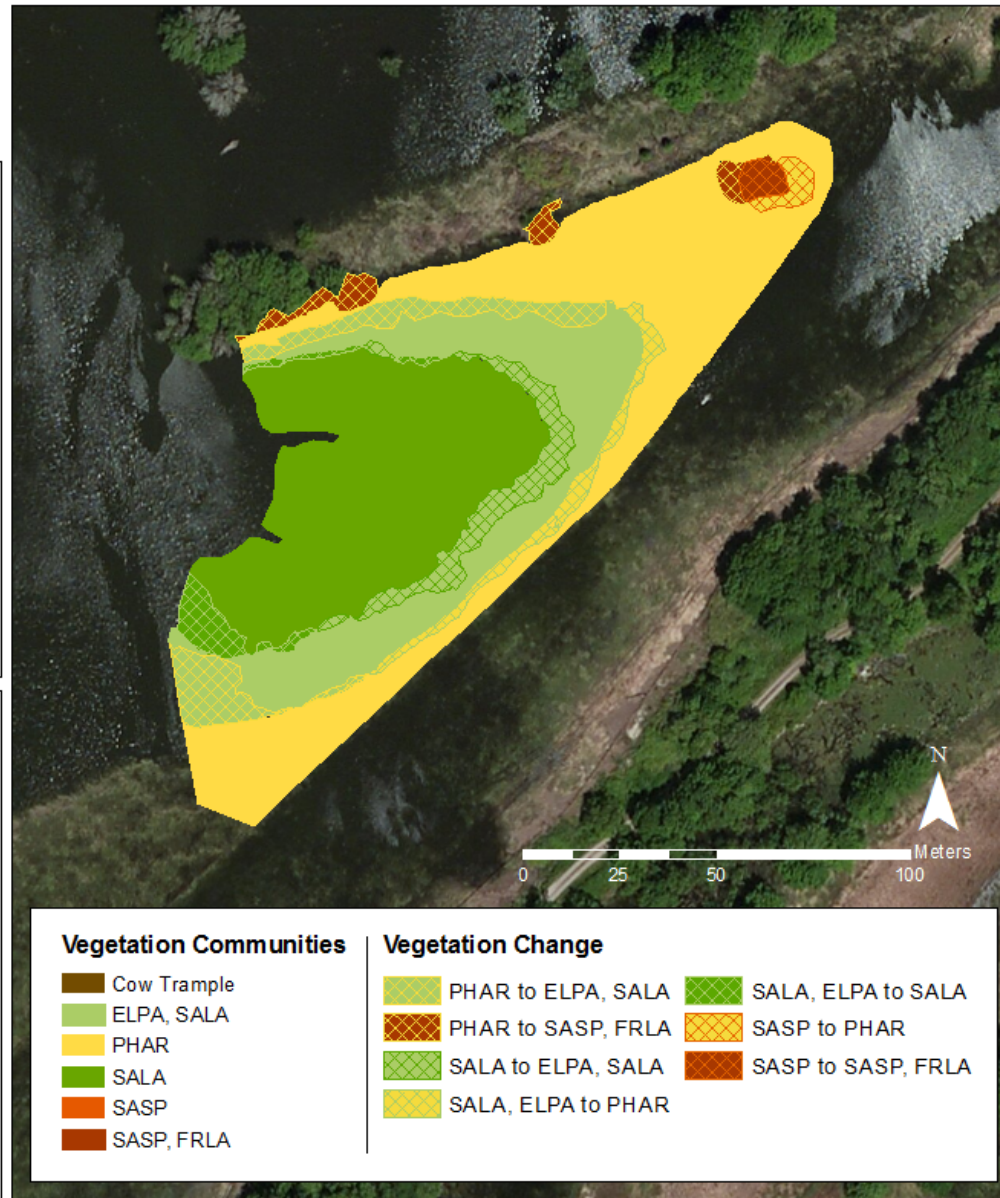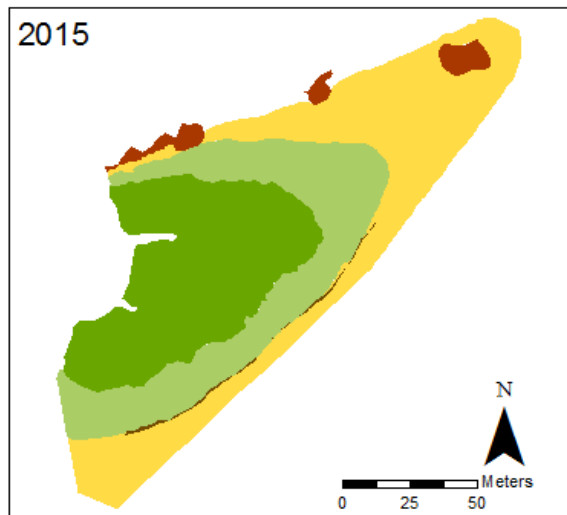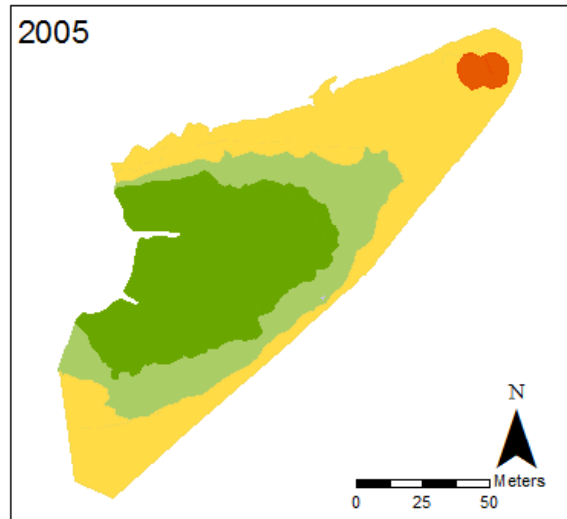
Franz Lake
Vegetation Change 2008, 2012 & 2015
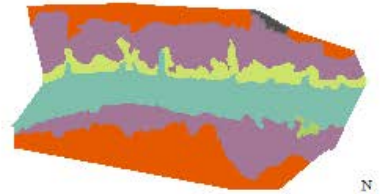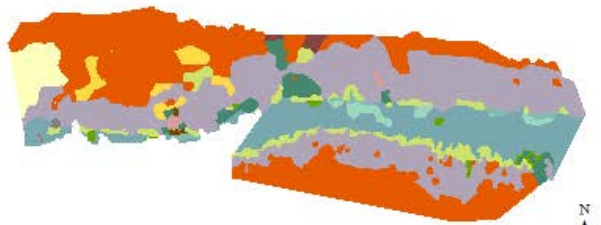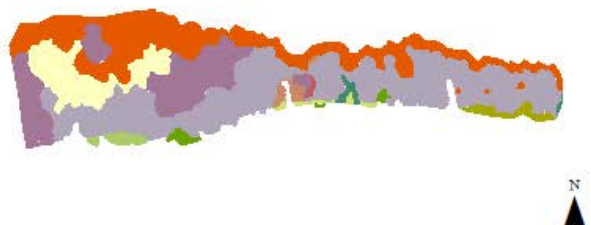
2008

2012
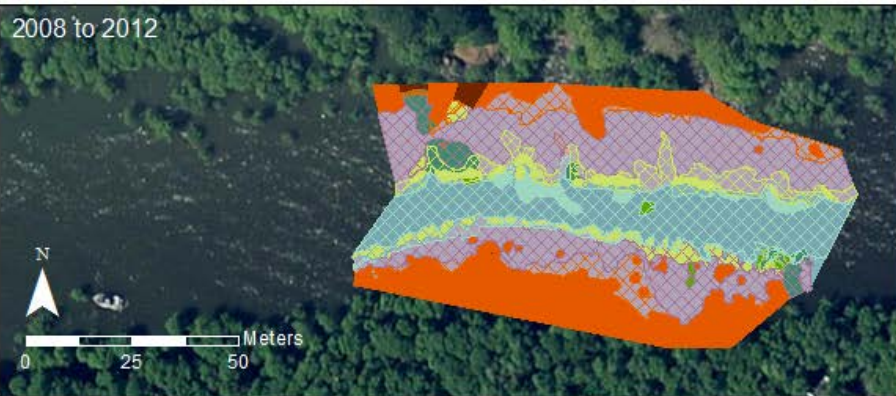
2015

2008 to 2012

2008 to 2015

2012 to 2015

245
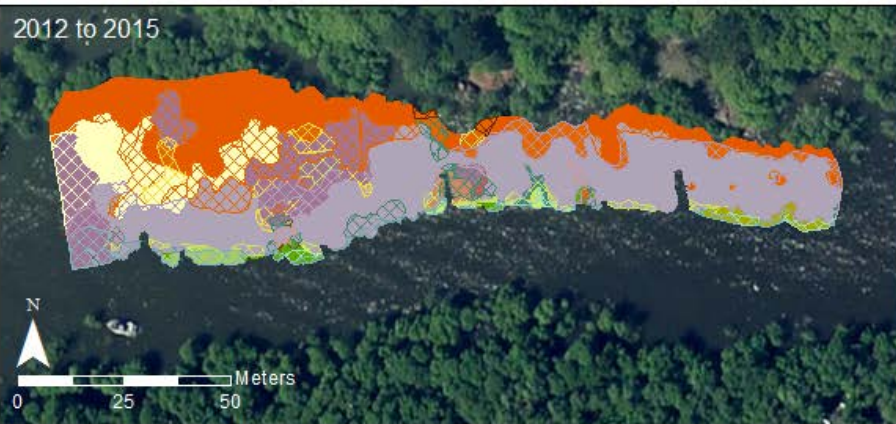
Appendix B. Annual photo points from EMP trends sites

Photo points collected in 2017 are still under analysis and unavailable at the time of the writing of this report. The following details are adapted from the 2016 report.

31 July 2011

15 February 2012

4 August 2012

4 February 2013

26 July 2013

20 September 2013

3 February 2014

27 June 2014

Baker Bay – PP1

Secret River – PP1 [HIGH MARSH]

5 February 2010

2 August 2012

9 August 2013

1 August 2015

Secret River – PP1 [HIGH MARSH]

6 August 2016

Secret River – PP2 [LOW MARSH]

1 December 2011

2 August2012

15 July 2014

1 August 2015

Secret River – PP2 [LOW MARSH]

6 August 2016

Secret River – PP3 [CHANNEL]

1 December 2011

15 July 2014

Secret River – PP3 [CHANNEL]

1 August 2015

6 August 2016

Welch Island – PP1

1 August 2012

3 February 2013

23 July 2013

Welch Island – PP1

1 August 2014

31 July 2015

5 August 2016

Whites Island – PP1

22 July 2009

----

13 July2010

2 August 2011

----

15 February 2012

Whites Island – PP1

31 July 2012

5 February 2013

22 July 2013

Whites Island – PP1

4 February 2014

31 July 2014

30 July 2015

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

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.[8]

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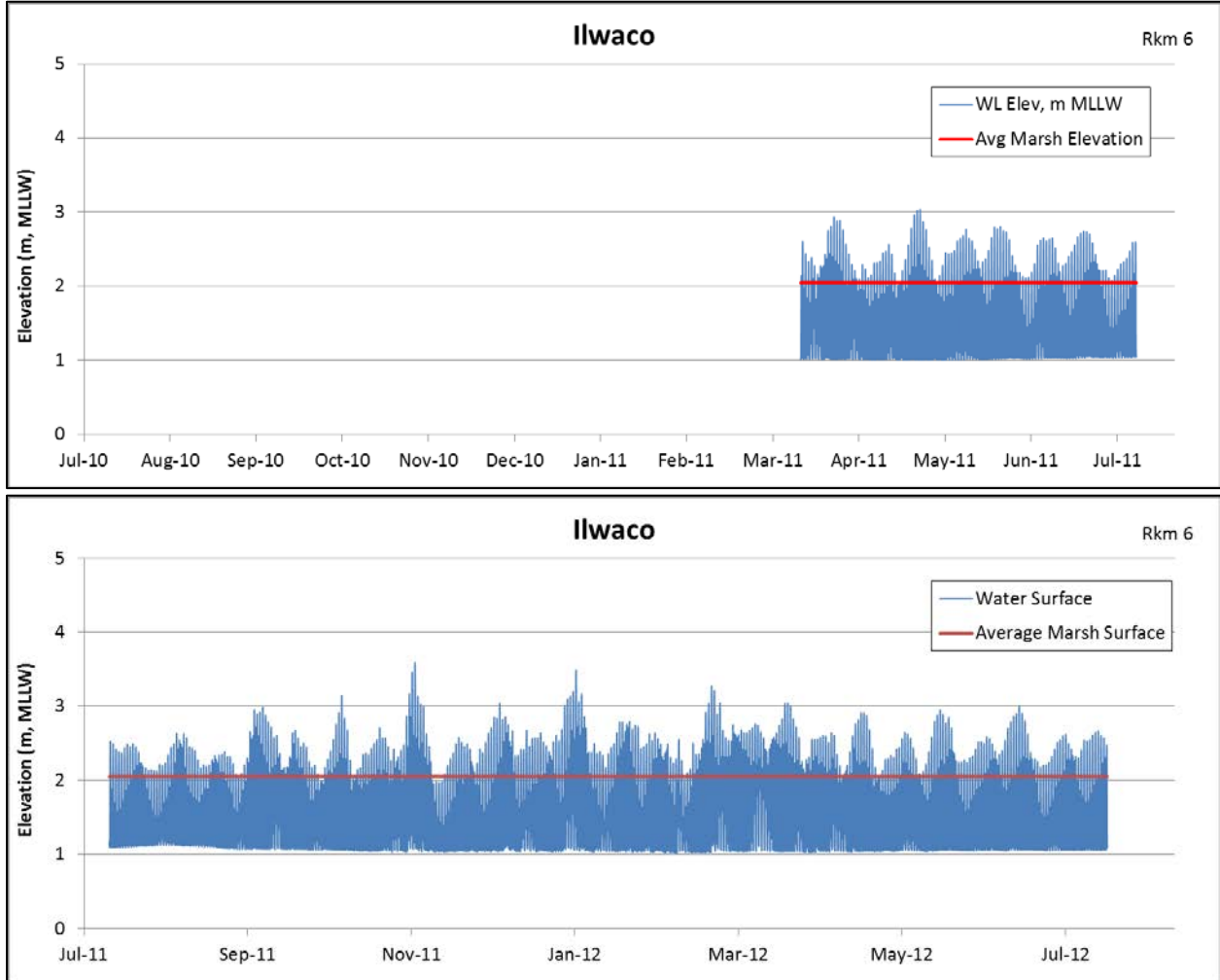
[8] Hydrology data collected in 2017 are still under analysis and unavailable at the time of the writing of this report. The following details are adapted from the 2016 report.
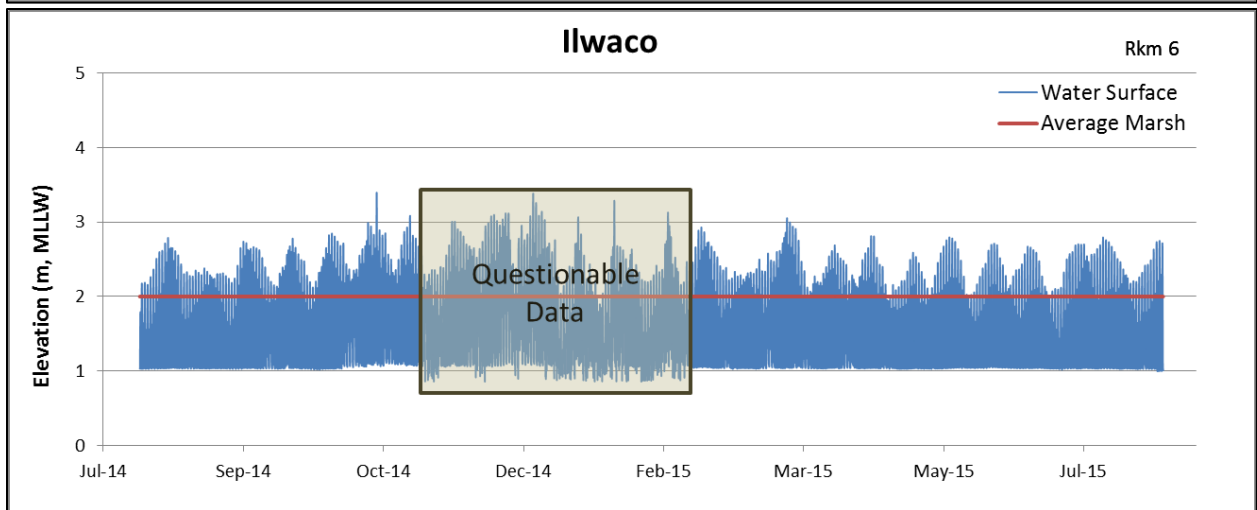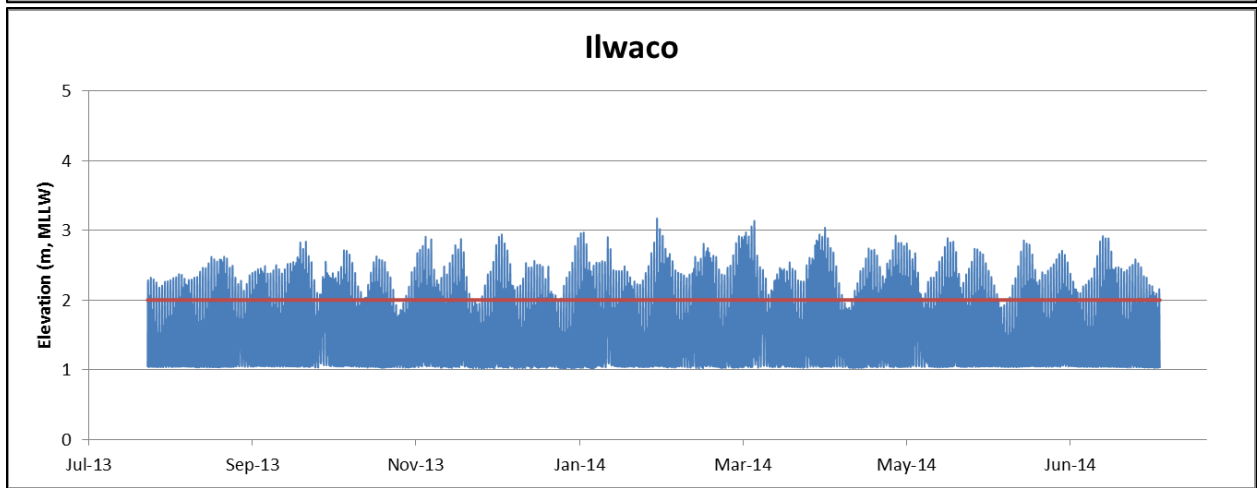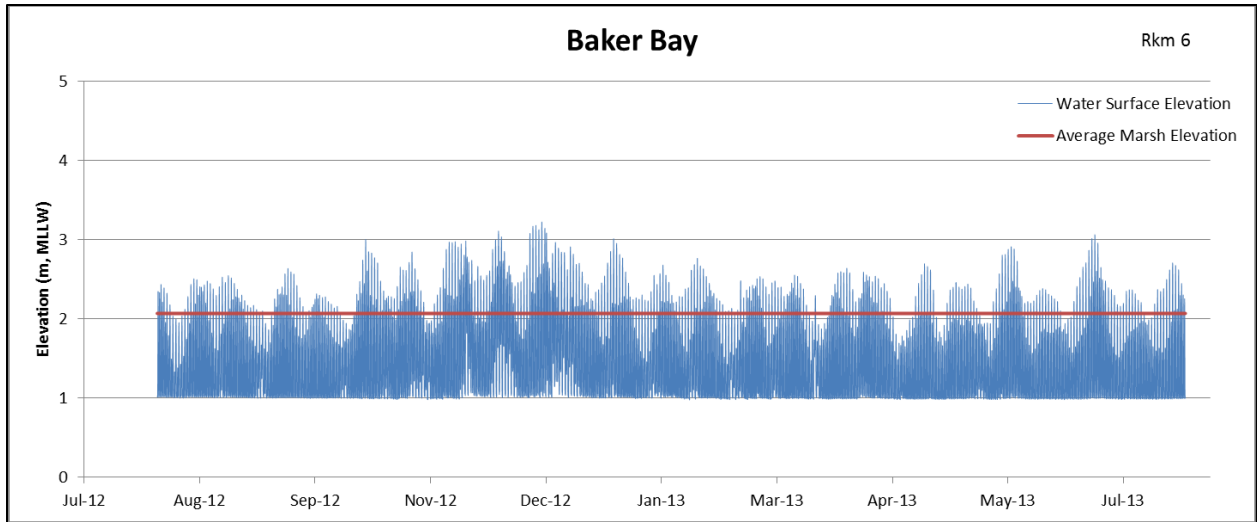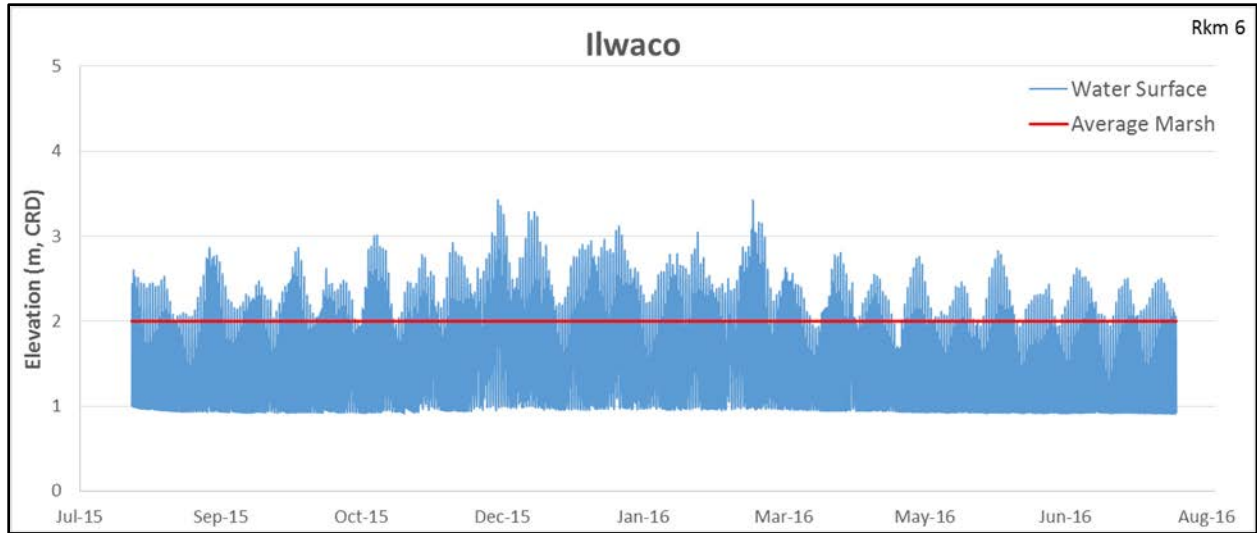
Figure C - 1. Water surface elevation data from the Ilwaco study site for the years 2011-2016. The red line represents the average elevation of the marsh sampling area. In Nov 2014-Fb 2015, the sensor appears to have become disconnected from the deployment post; measurements from this time should not be used in calculations.
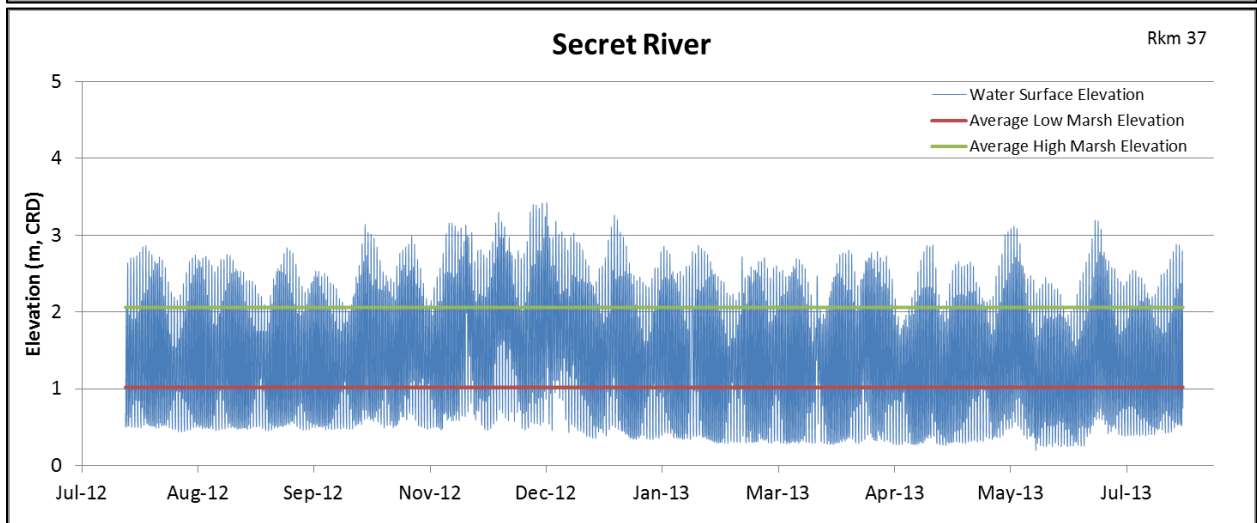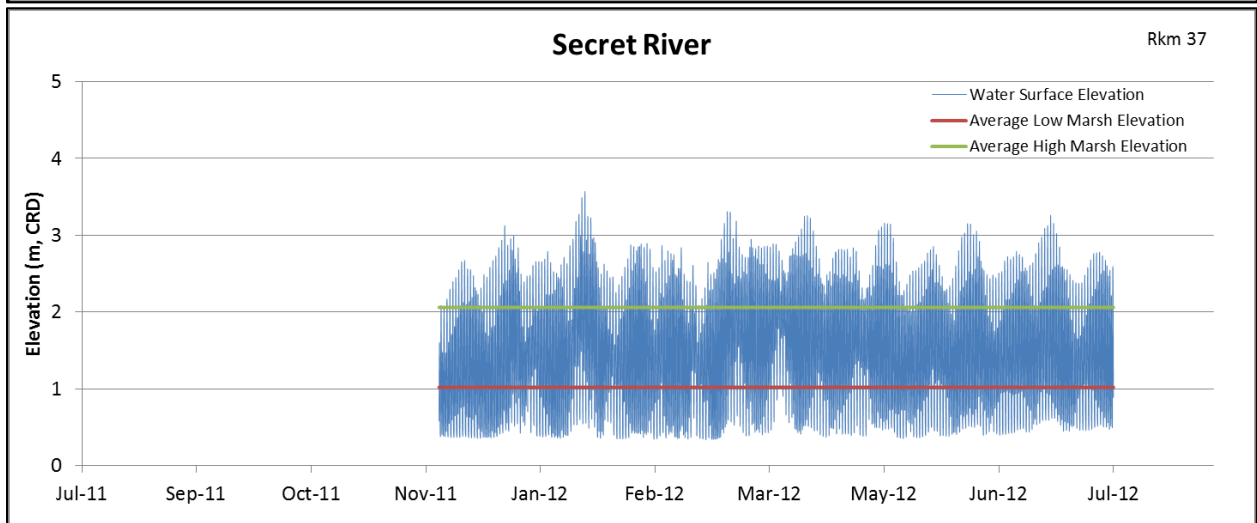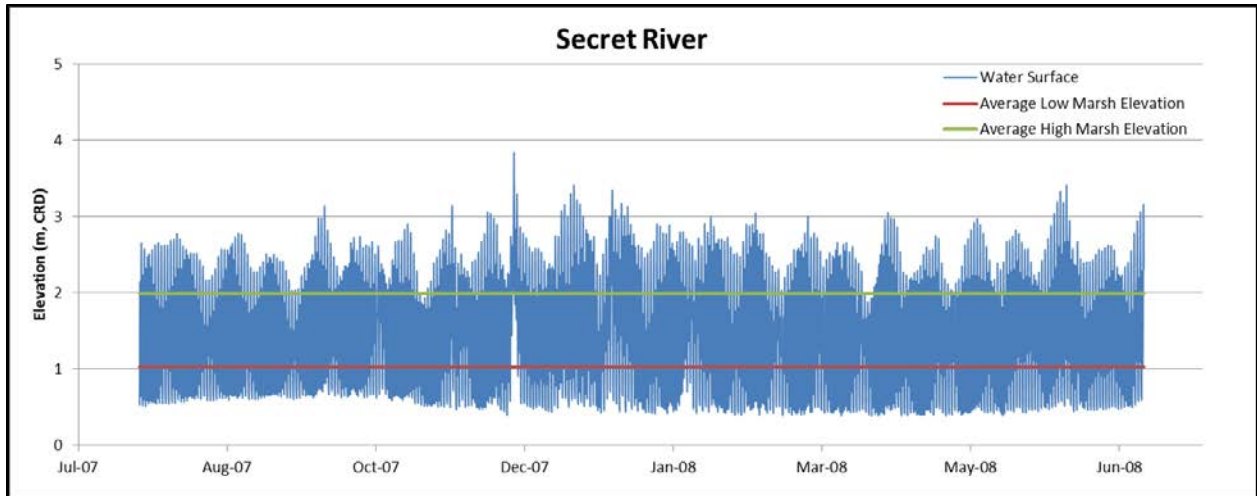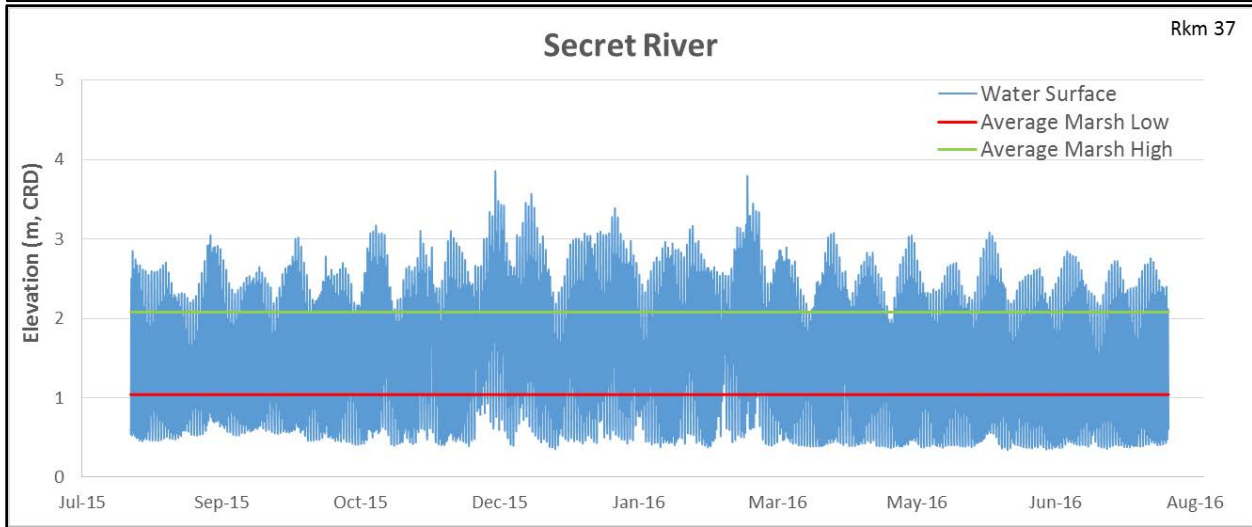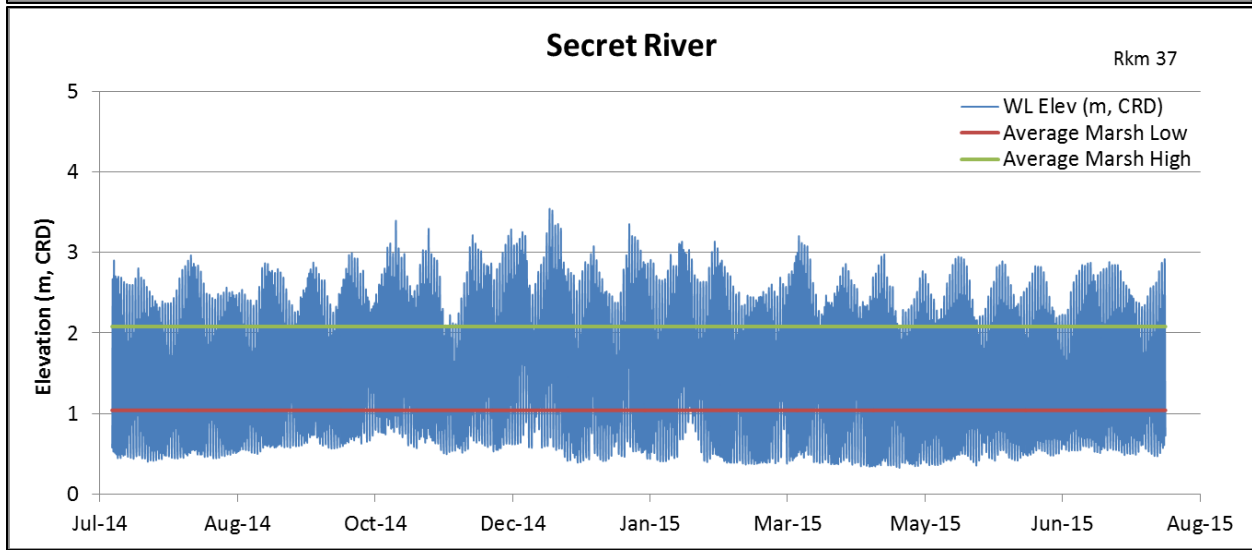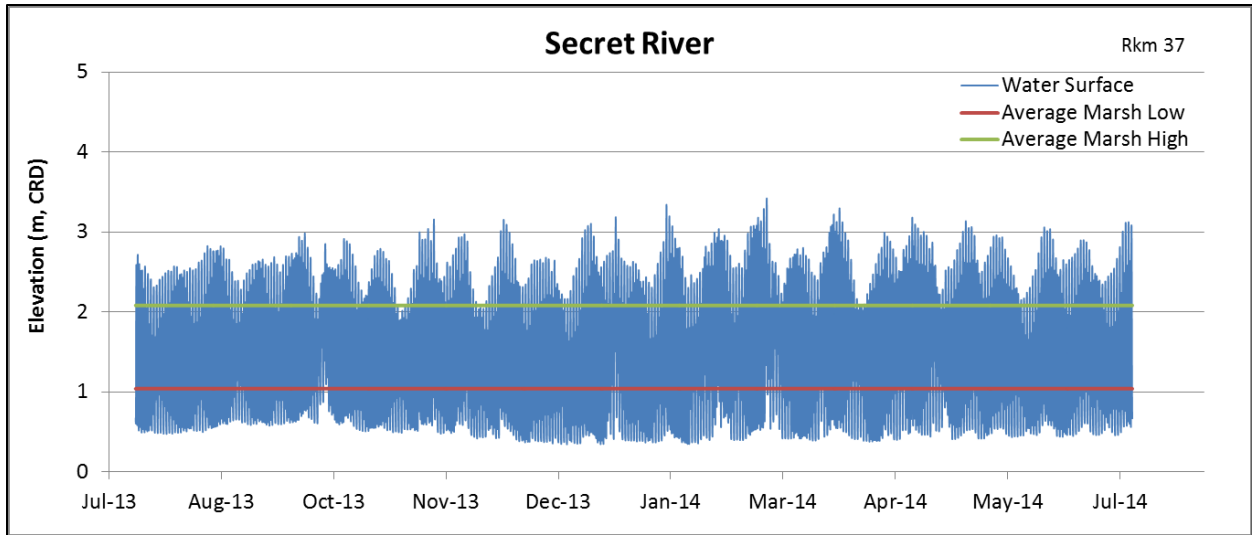
Figure C -2. Water surface elevation data from the Secret River study site for the years 2007-2008 and 2011-2016. The red line represents the average elevation of the marsh sampling area.
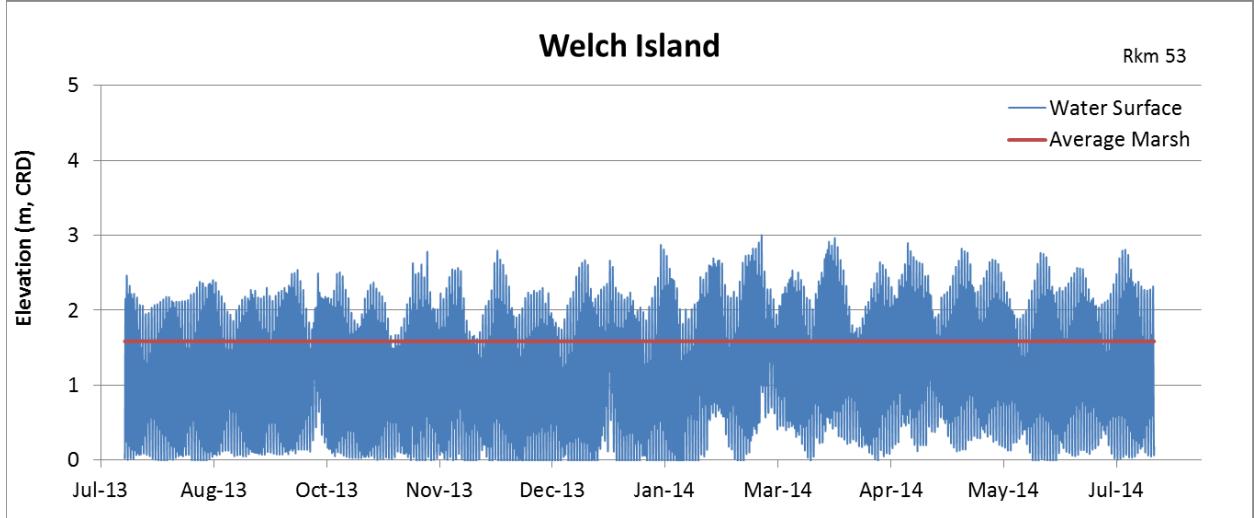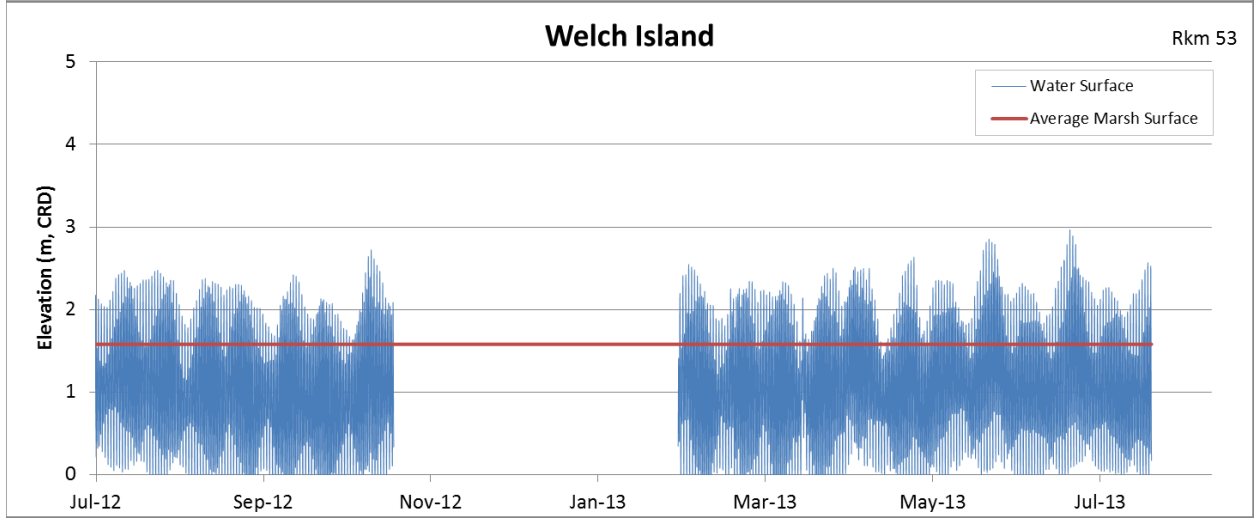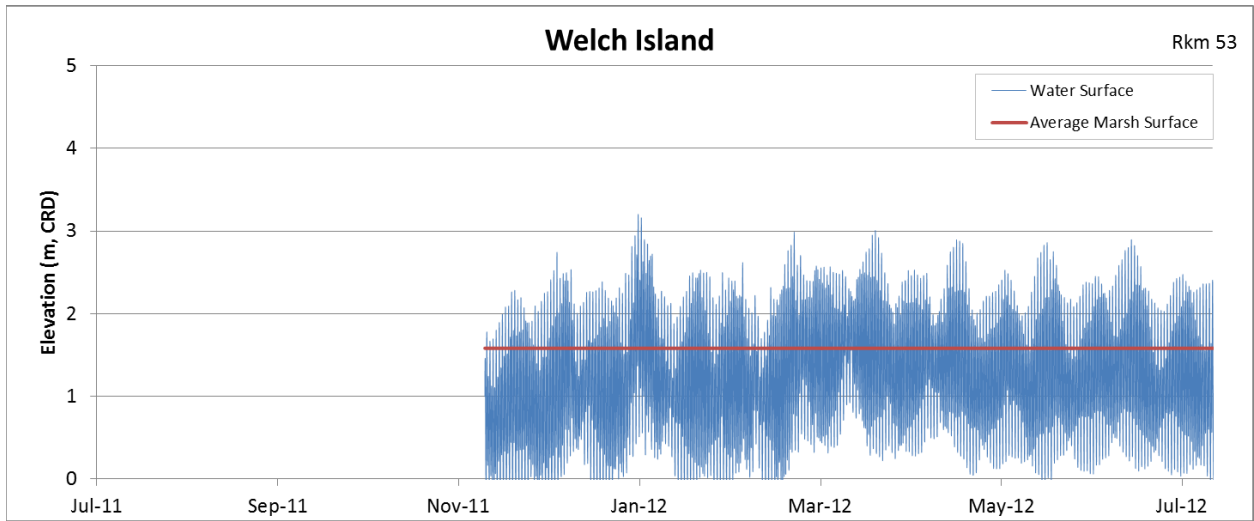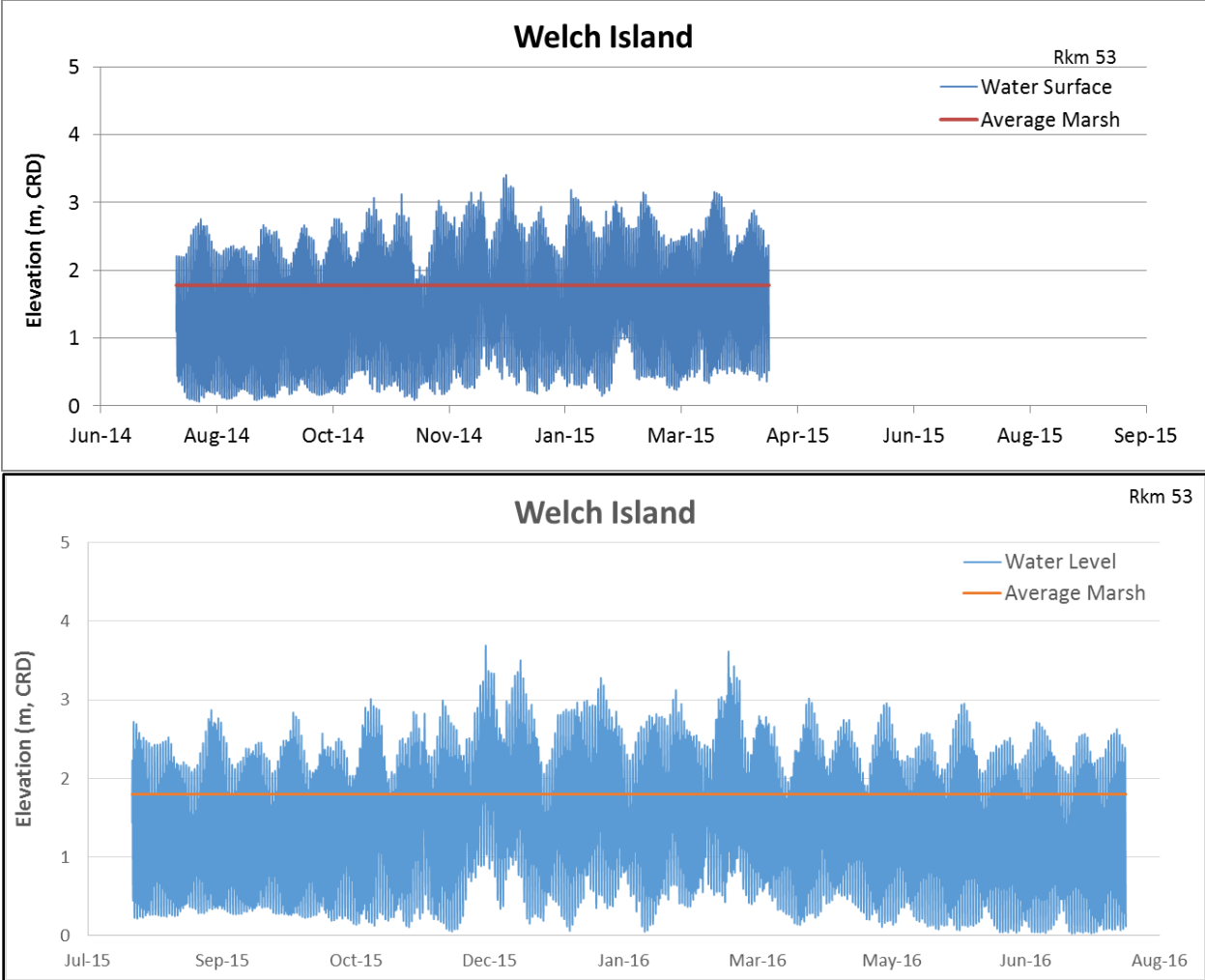
Figure C -3. Water surface elevation data from the Welch Island study site for the years 2012-2015. The red line represents the average elevation of the marsh sampling area. The sensor was displaced between early November 2012 and February 2013 therefore no data is reported for that time.
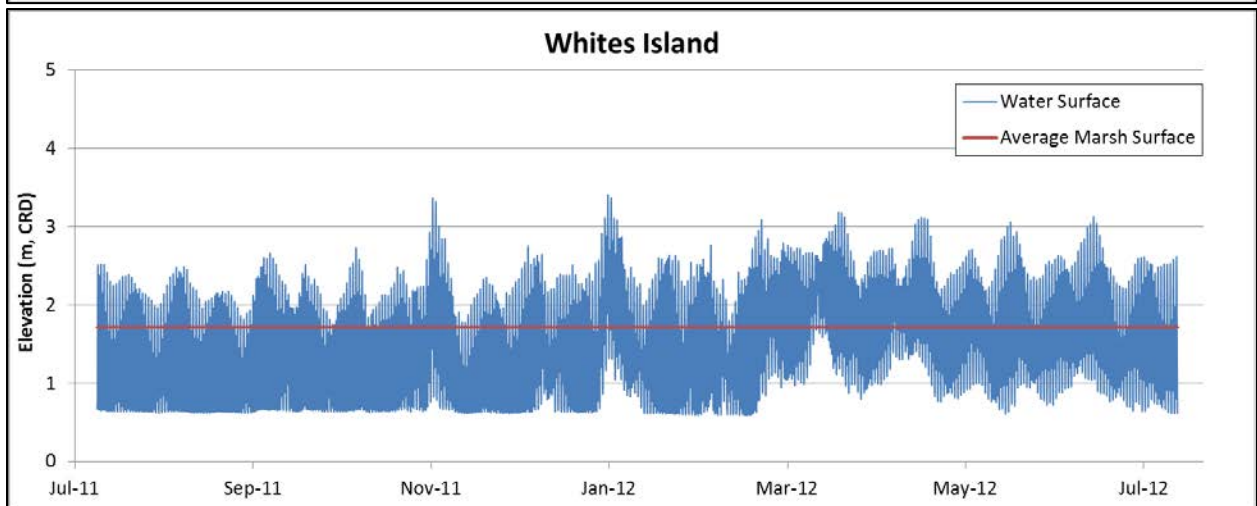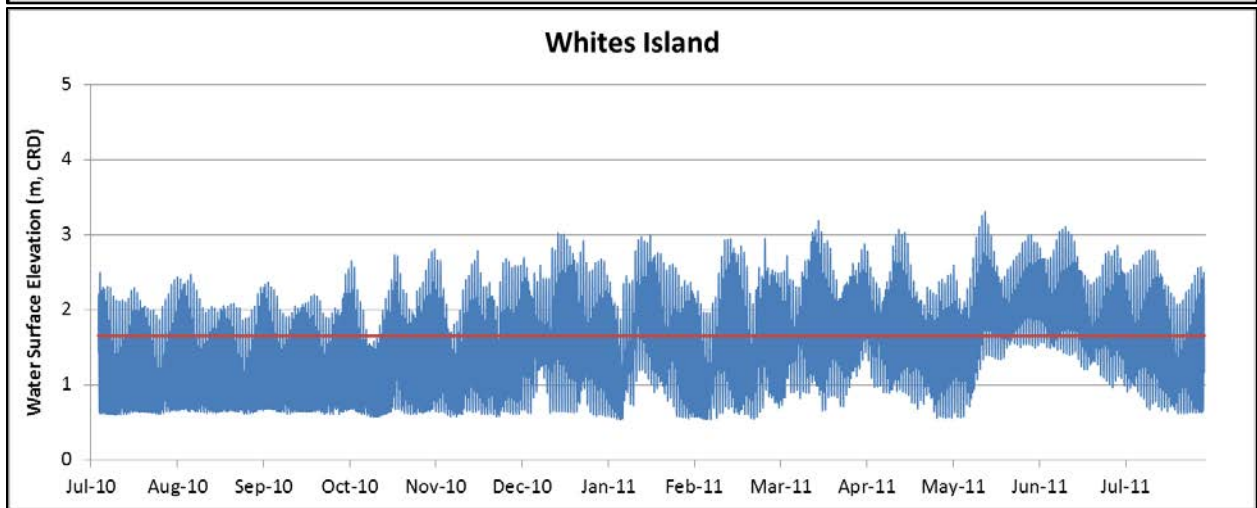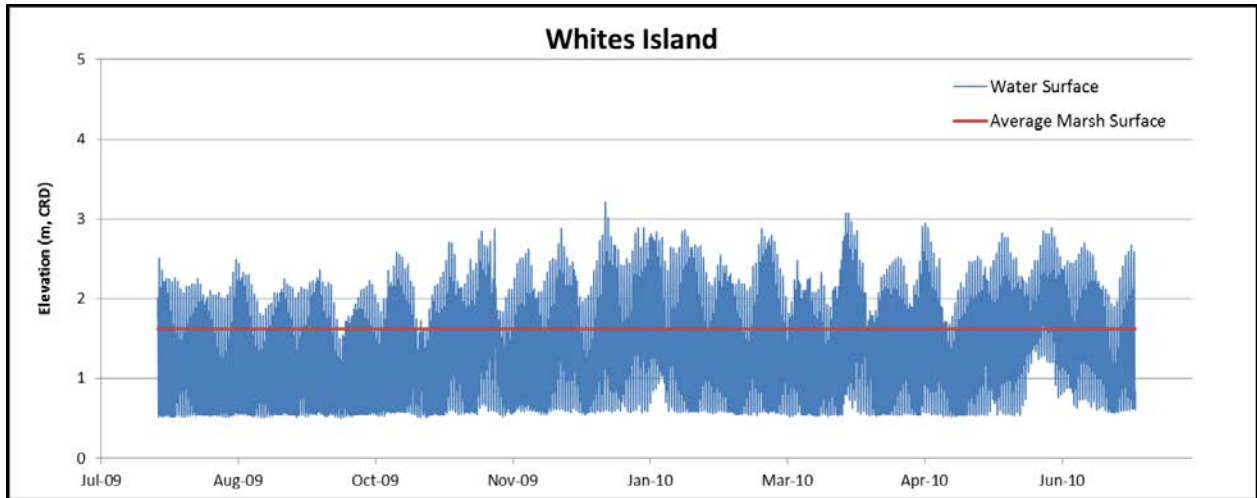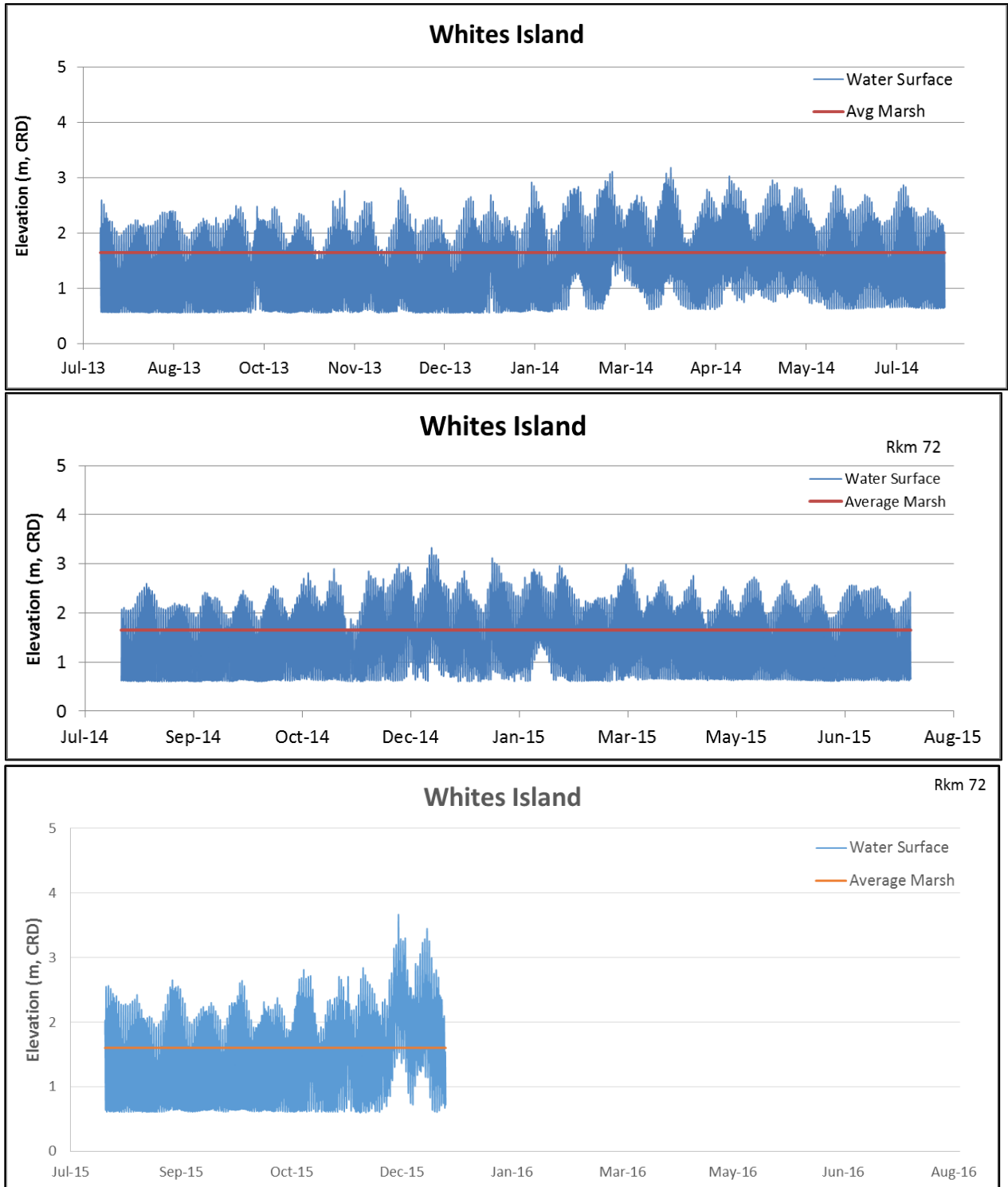
Figure C -4. Water surface elevation data from the Whites Island study site for the years 2009-2012 and 2013-2015. The red line represents the average elevation of the marsh sampling area. No data from 2013 and 2016 due to sensor failure.

Cunningham Lake — Rkm 145

Cunningham Lake — Rkm 145

Cunningham Lake — Rkm 145
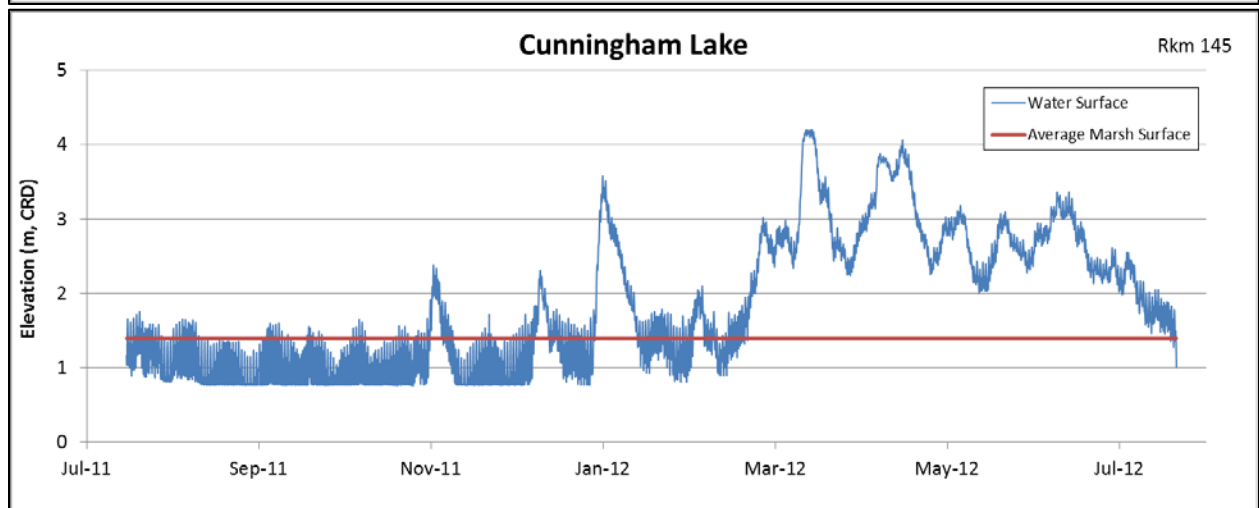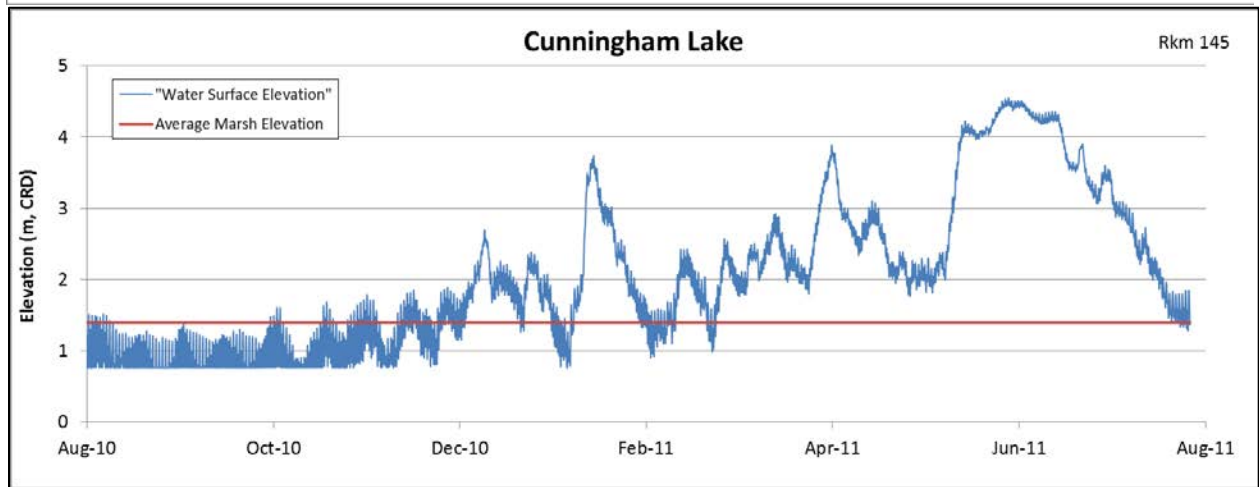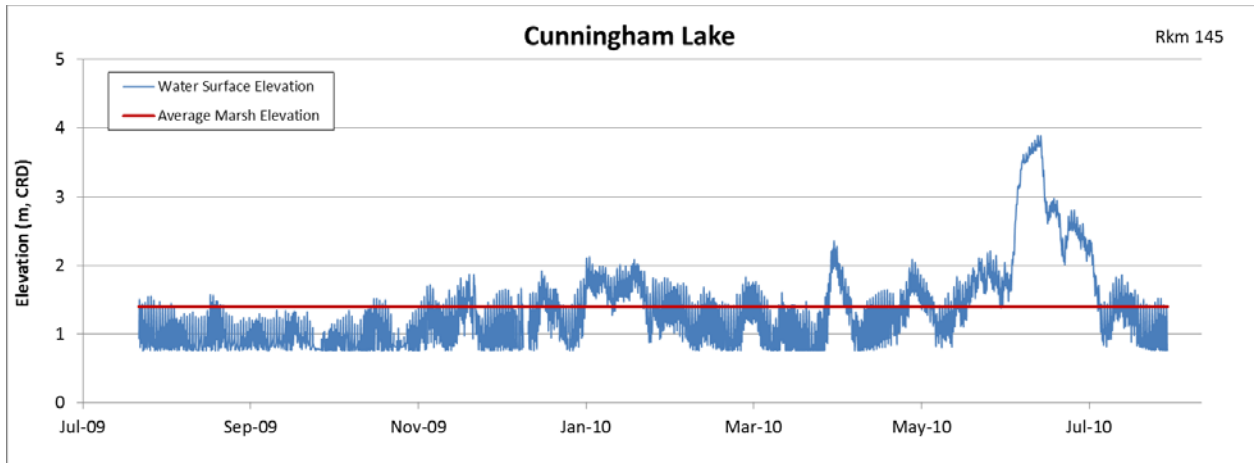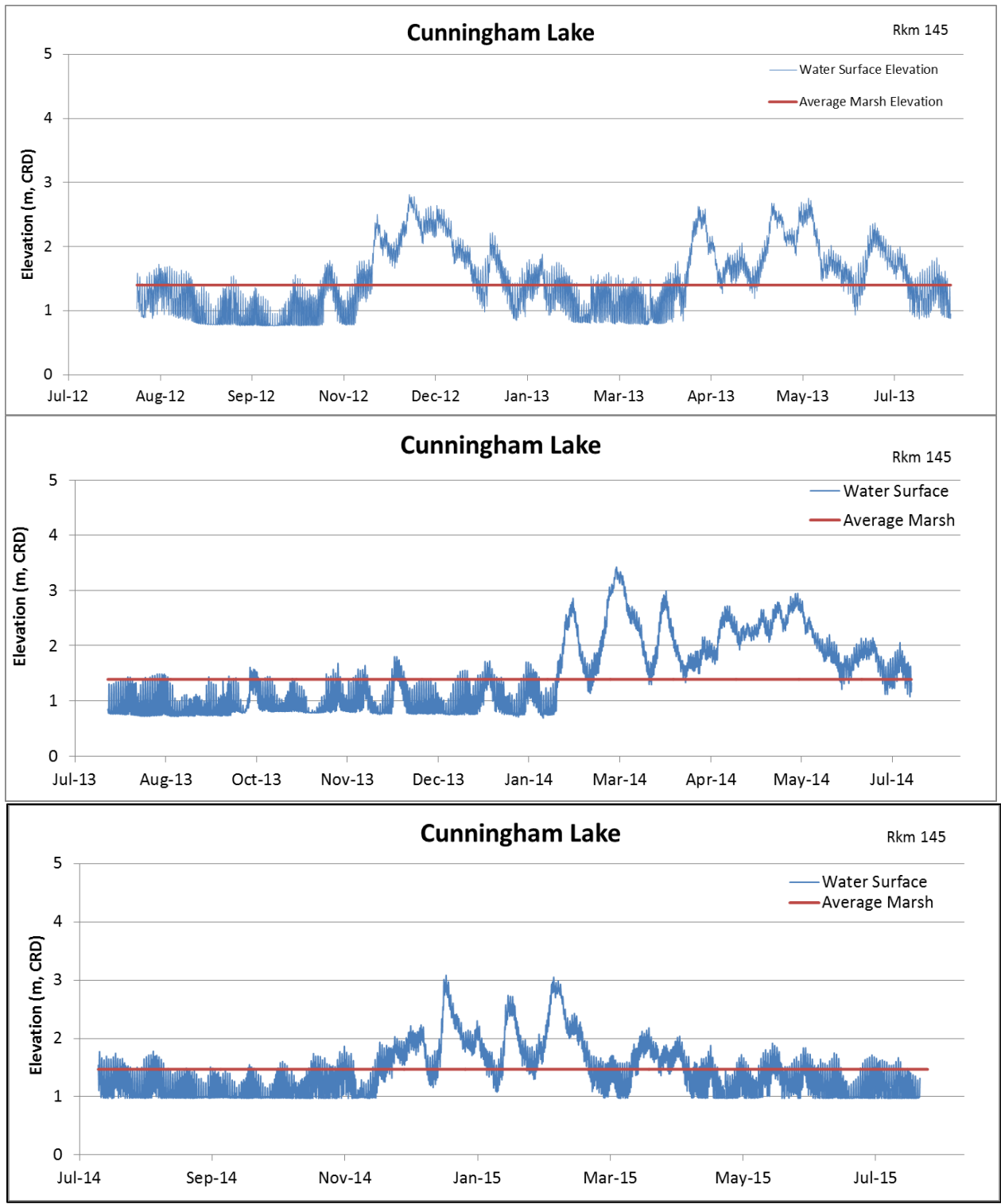
Figure C -5. Water surface elevation data from the Cunningham Lake study site for the years 2009-2015. The red line represents the average elevation of the marsh sampling area. No data was collected from 2015-2016 due to sensor failure

Campbell Slough — Rkm 149

Campbell Slough — Rkm 149

Campbell Slough — Rkm 149
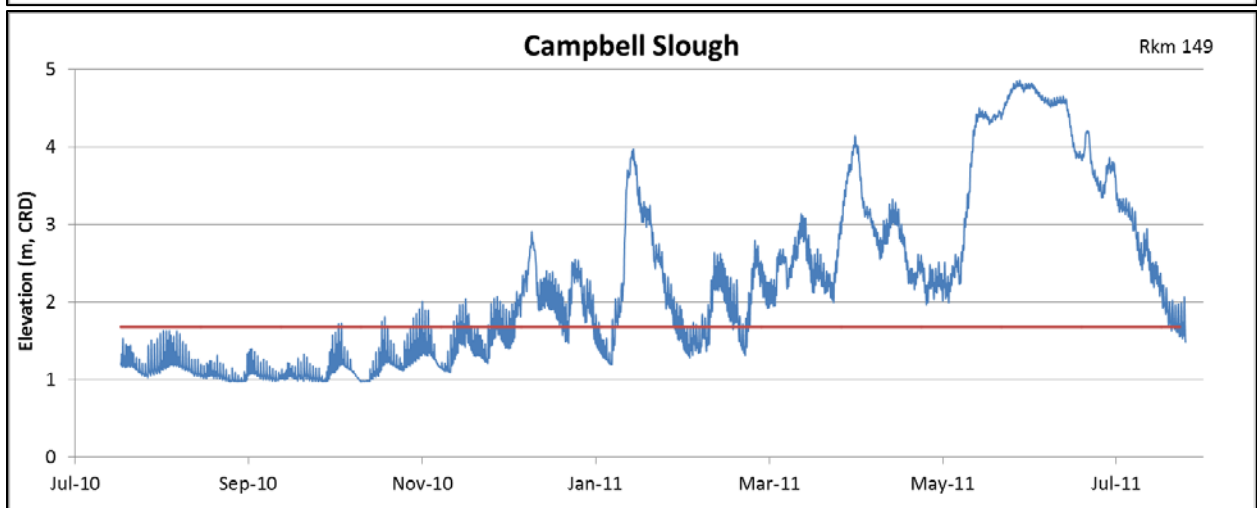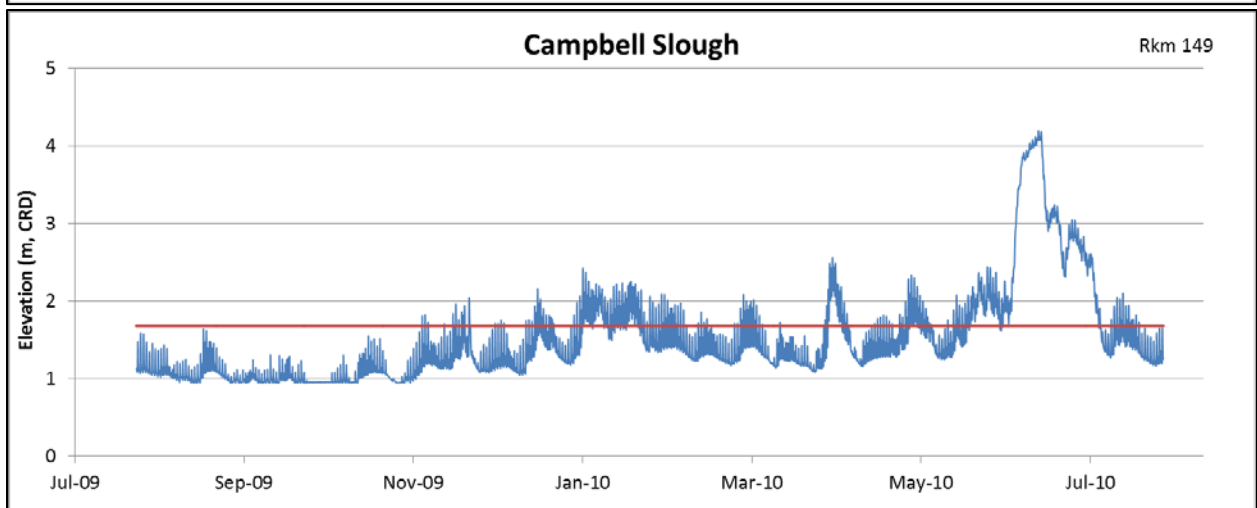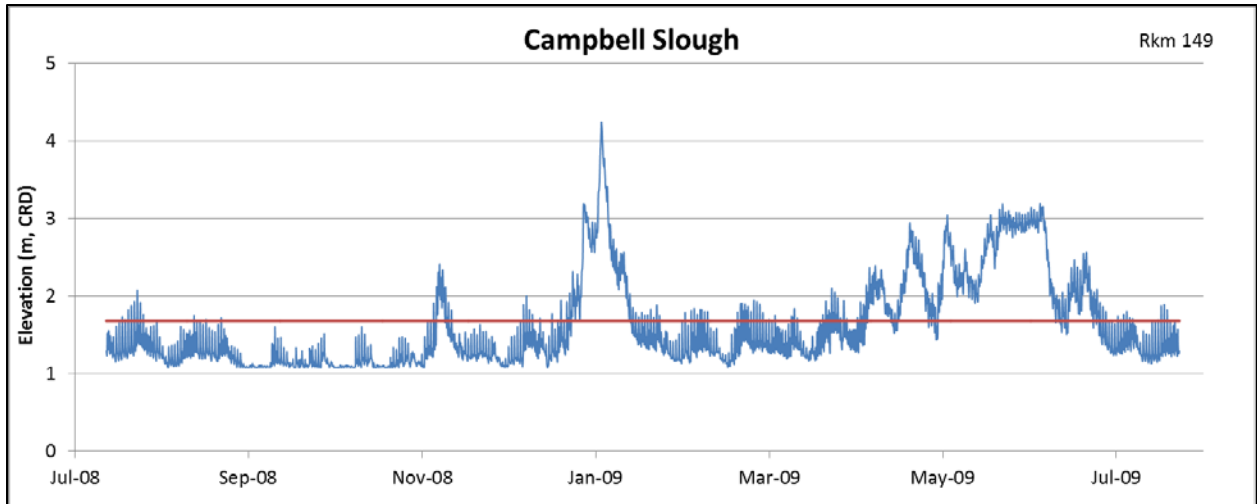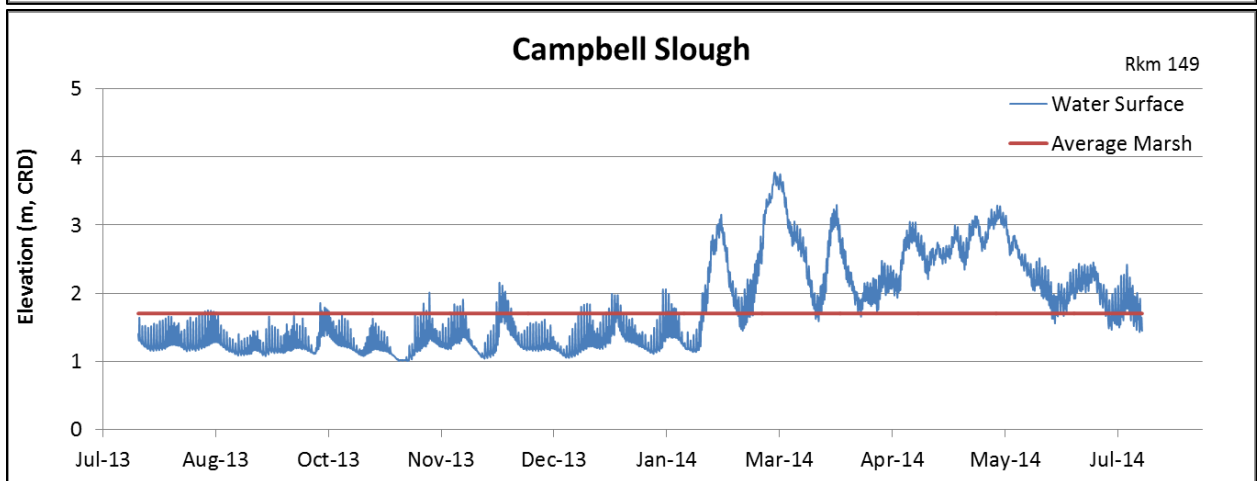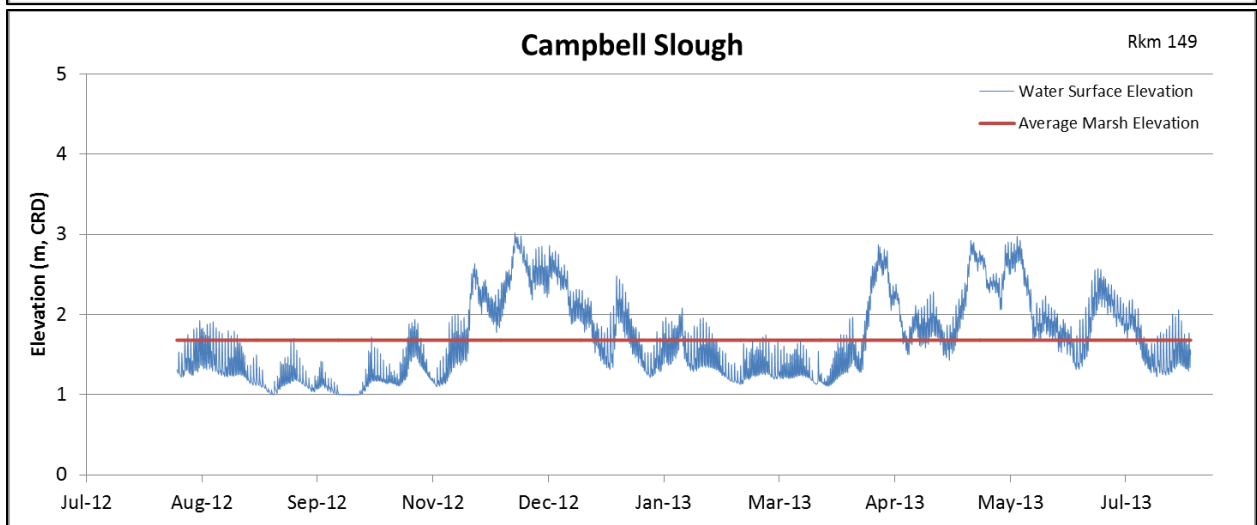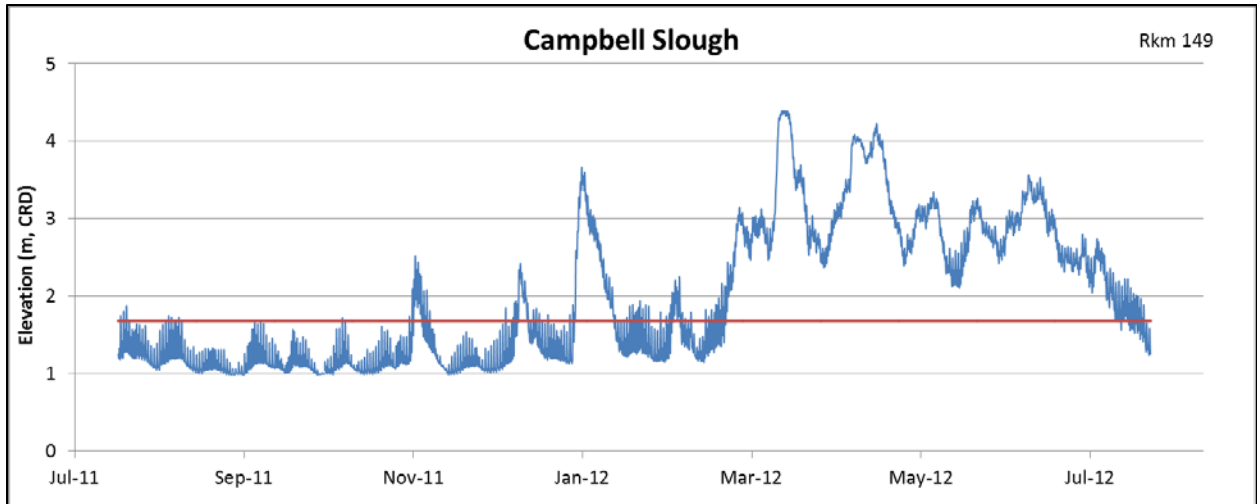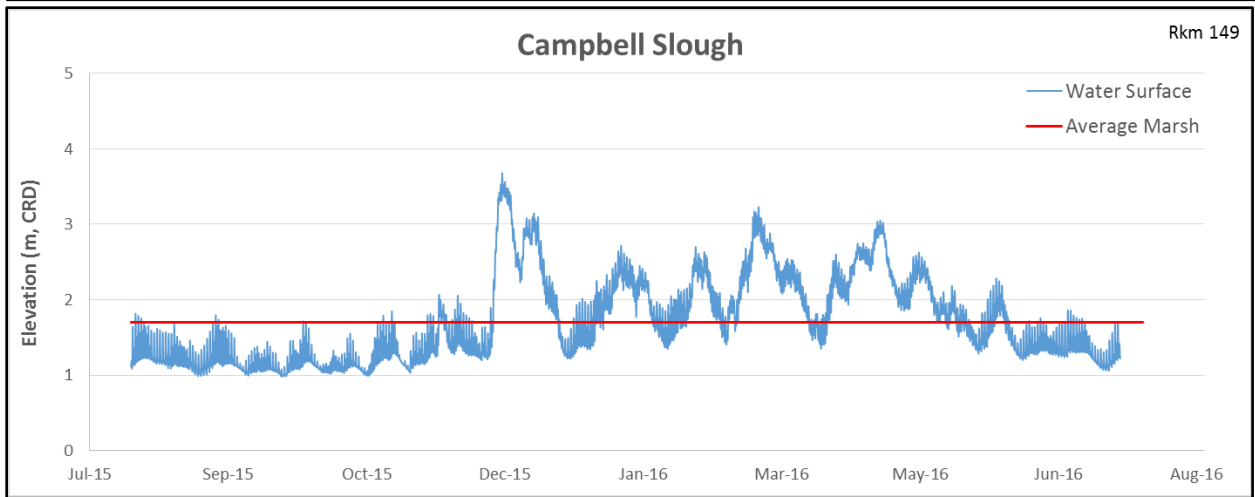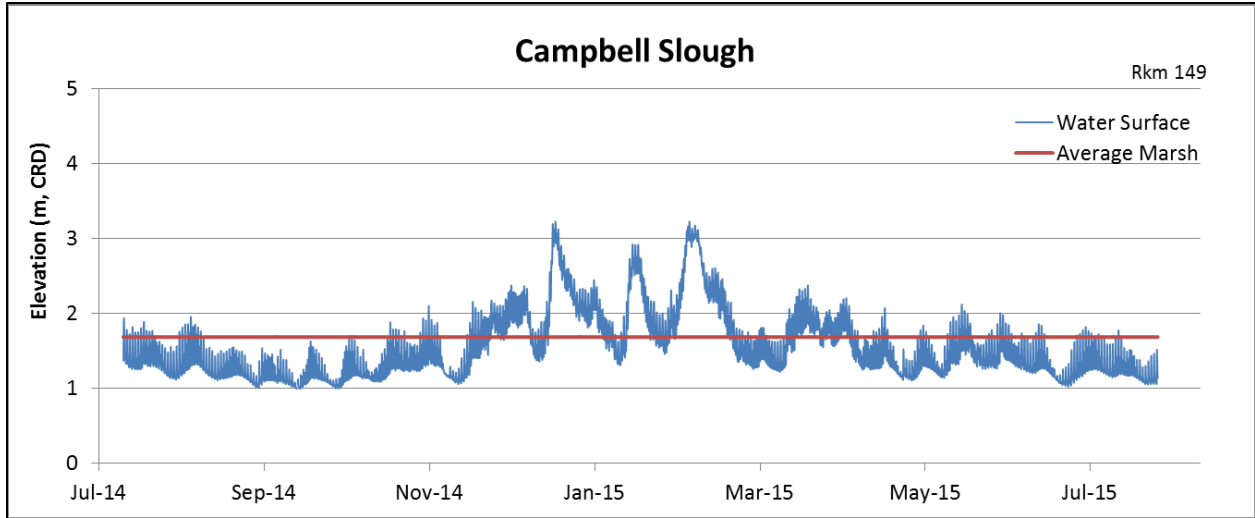
Figure C -6. Water surface elevation data from the Campbell Slough study site for the years 2008-2016. The red line represents the average elevation of the marsh sampling area.
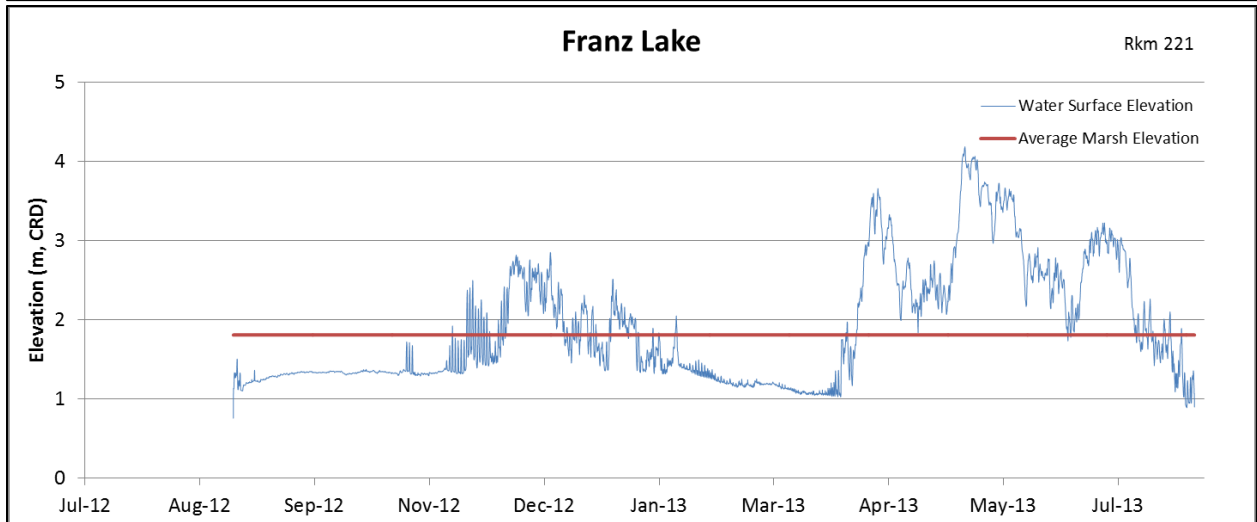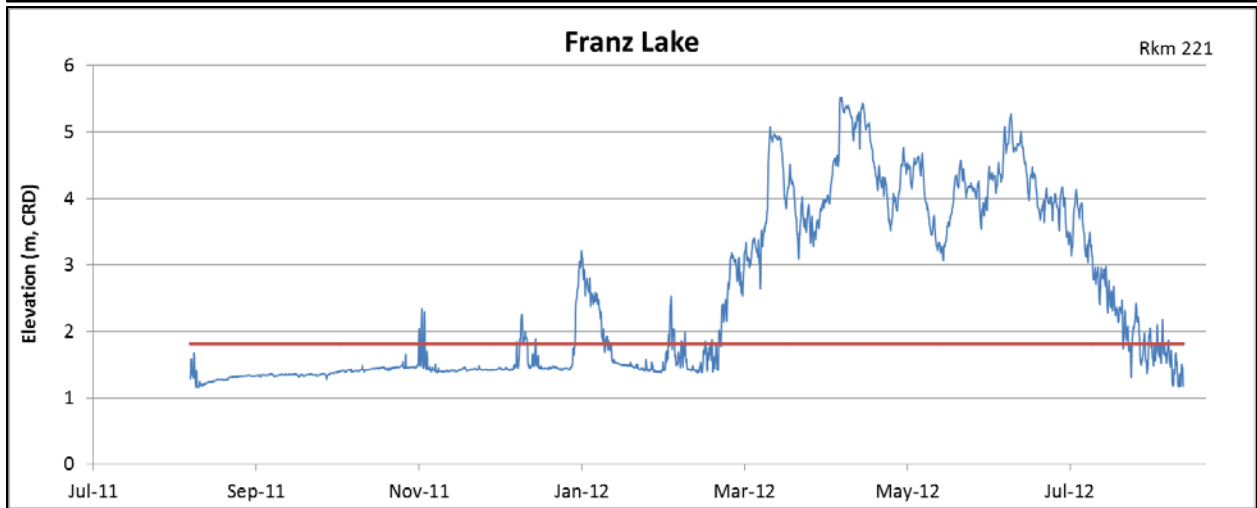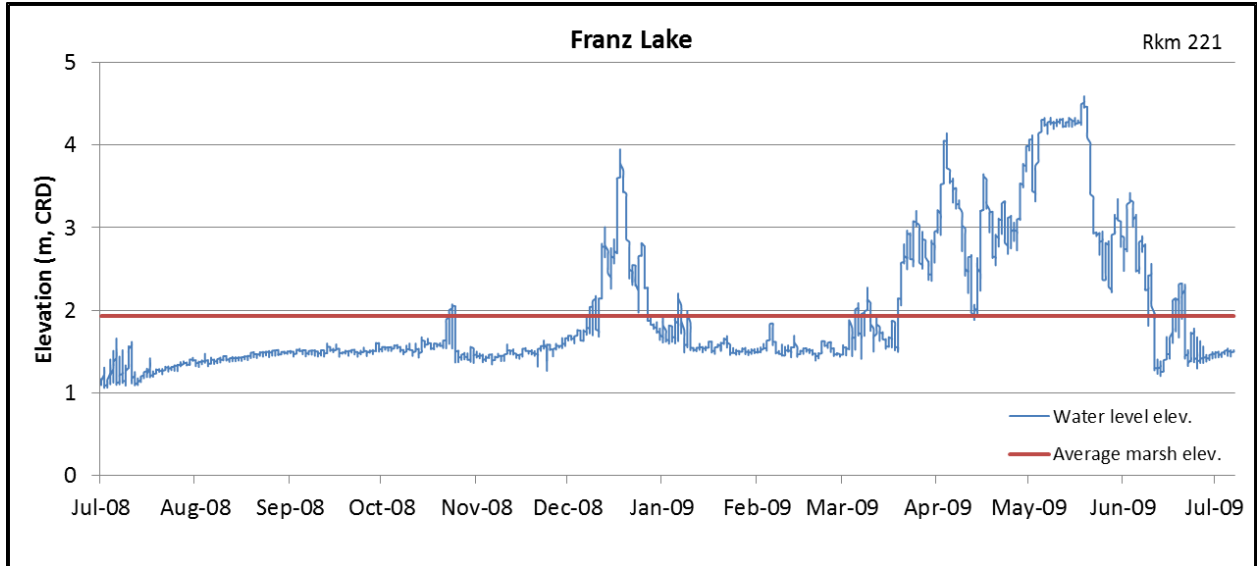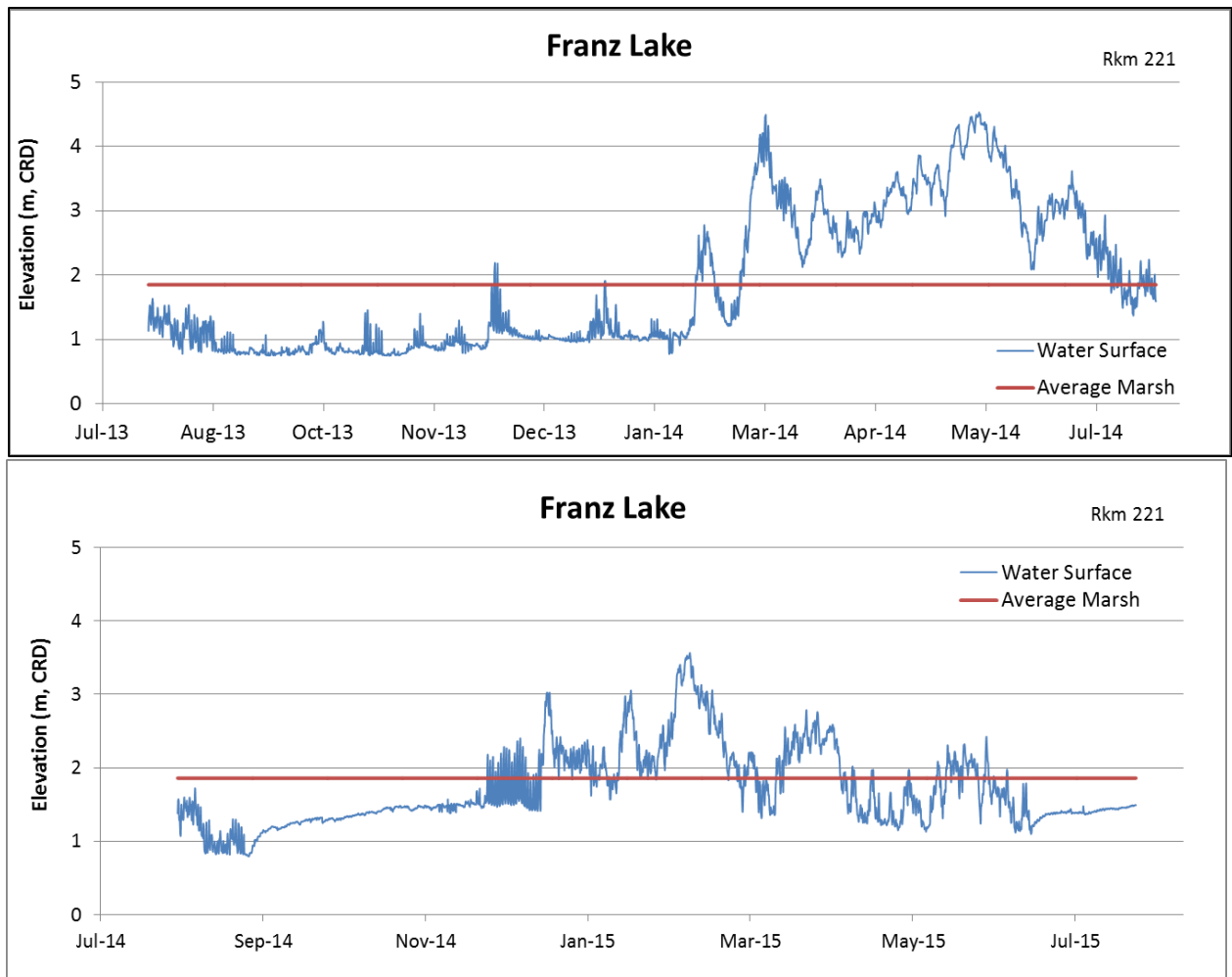
Figure C -7. Water surface elevation data from the Franz Lake study site for the years 2008-2009 and 2011-2015. The red line represents the average elevation of the marsh sampling area. Note the scale difference for the 2011-2012 plot. No data was collected for 2015-2016 due to displacement and loss of sensor.

Appendix D. Vegetation Species Cover

Table D.1. Site marsh elevation range in meters, relative to the Columbia River vertical datum CRD.

	Ilwaco Slough	Welch Island	Whites Island	Cunningham Lake	Campbell Slough	Franz Lake
Minimum	1.6	1.1	0.8	1.1	1.2	1.2
Average	2	1.8	1.7	1.5	1.7	1.9
Maximum	2.4	1.9	2.1	1.7	2.7	2.3

Table D.2. Average percent cover of each plant species at trend sites in 2017. The number in parentheses following the site name is the number of plots sampled at that site. The five-dominant species at each site are in bold red, and non-native species are shaded in peach. Species are sorted by their four-letter code (1st two letters of genus and 1st two letters of species). T = present at Trace levels.

Code	Scientific Name	Common Name	Native	Wetland Status	Ilwaco Slough (40)	Welch Island (41)	Whites Island (44)	Cunningham Lake (69)	Campbell Slough (65)	Franz Lake (61)
AGGI	*Agrostis gigantea*	redtop; black bentgrass	No	FAC	0.0	0.6	0.5	0.0	0.0	0.0
AGSP	*Agrostis sp.*	bentgrass	Mixed	Mixed	**25.7**	0.0	0.0	0.0	0.0	0.0
ALTR	*Alisma triviale*	northern water plaintain	Yes	OBL	0.0	0.3	0.3	0.1	0.0	0.3
AMFR	*Amorpha fruticosa*	indigo bush	No	FACW	0.0	0.0	0.0	0.0	0.0	0.3
AREG	*Argentina egedii ssp. egedii*	Pacific silverweed	Yes	OBL	1.6	3.4	0.0	0.0	0.0	0.4
BESY	*Beckmannia syzigachne*	American sloughgrass	Yes	OBL	0.0	0.0	0.2	0.0	0.0	0.0
BICE	*Bidens cernua*	Nodding beggars-ticks	Yes	FACW	0.0	0.6	0.2	1.4	0.0	0.0
BIFR	*Bidens frondosa*	devil's beggartick	Yes	FACW	0.0	0.0	0.0	0.1	0.0	0.3
CAAM	*Castilleja ambigua*	paint-brush owl-clover; johnny-nip	Yes	FACW	0.7	0.0	0.0	0.0	0.0	0.0
CAHE	*Callitriche heterophylla*	Water starwort; Two-headed water starwort	Yes	OBL	0.0	0.4	0.5	0.0	0.0	0.0
CALY	*Carex lyngbyei*	Lyngby sedge	Yes	OBL	**54.1**	**70.1**	**9.0**	0.0	0.0	0.0
CAOB	*Carex obnupta*	Slough sedge	Yes	OBL	0.0	0.0	0.1	0.0	0.0	0.0

Code	Scientific Name	Common Name	Native	Wetland Status	Ilwaco Slough (40)	Welch Island (41)	Whites Island (44)	Cunningham Lake (69)	Campbell Slough (65)	Franz Lake (61)
CAPA	*Caltha palustris*	Yellow marsh marigold	Yes	OBL	0.0	4.4	0.1	0.0	0.0	0.0
CARE	*Carex retrorsa*	knotsheath sedge	Yes	OBL	0.0	0.0	0.0	0.0	0.0	0.2
CASE	*Calystegia sepium*	Hedge false bindweed	No	FAC	0.0	0.0	1.2	0.0	0.0	0.0
CASP	*Carex sp.*	Carex species	Yes	Mixed	0.0	0.0	0.0	0.0	0.0	3.2
CASP2	*Callitriche sp.*	water-starwort	Mixed	OBL	0.0	0.0	0.0	0.0	0.0	2.1
CEDE	*Ceratophyllum demersum*	Coontail	Yes	OBL	0.0	0.0	0.0	0.0	T	0.0
CIDO	*Cicuta douglasii*	Western water-hemlock	Yes	OBL	0.0	0.0	0.1	0.0	0.0	0.0
DECE	*Deschampsia cespitosa*	Tufted hairgrass	Yes	FACW	1.2	0.1	0.0	0.0	0.0	0.0
ELAC	*Eleocharis acicularis*	Needle spikerush	Yes	OBL	0.0	0.0	0.0	1.2	0.0	0.0
ELCA	*Elodea canadensis*	Canada waterweed	Yes	OBL	0.0	2.3	11.3	0.1	T	0.0
ELPA	*Eleocharis palustris*	Common spikerush	Yes	OBL	0.0	8.1	1.2	5.3	14.8	4.4
ELPAR	*Eleocharis parvula*	Dwarf spikerush	Yes	OBL	4.7	0.0	0.0	0.0	0.0	0.0
EPCI	*Epilobium ciliatum*	Willow herb	Yes	FACW	0.0	0.2	0.0	0.0	0.0	0.0
EQAR	*Equisetum arvense*	field horsetail	Yes	FAC	0.0	0.0	0.0	0.0	0.0	0.8
EQFL	*Equisetum fluviatile*	Water horsetail	Yes	OBL	0.0	0.0	0.0	0.0	T	0.0
EQPA	*Equisetum palustre*	marsh horsetail	Yes	FACW	0.0	0.0	0.0	0.1	0.0	0.9
EQSP	*Equisetum spp.*	Horsetail	Yes	Mixed	0.0	4.8	3.9	0.0	0.0	0.0
FRLA	*Fraxinus latifolia*	Oregon ash	Yes	FACW	0.0	0.0	0.0	0.0	1.6	3.0
GATR	*Galium trifidum L. spp. columbianum*	Pacific bedstraw	Yes	FACW	0.0	0.8	T	0.0	0.0	0.0
GATR3	*Galium trifidum*	small bedstraw	Yes	FACW	0.0	0.0	0.3	0.0	0.0	0.0
GLGR	*Glyceria grandis*	American mannagrass	Yes	OBL	0.0	0.0	T	0.0	0.0	0.0
GLMA	*Glaux maritima*	sea milkwort	Yes	FACW	1.1	0.0	0.0	0.0	0.0	0.0
GNUL	*Gnaphalium uliginosum*	Marsh cudweed	No	FAC	0.0	0.0	0.0	0.0	0.1	0.9
HEAU	*Helenium autumnale*	common sneezeweed	Yes	FACW	0.0	0.0	0.0	0.0	0.0	0.7
HYRA2	*Hydrocotyle ranunculoides*	Water pennywort	Yes	OBL	0.0	0.0	T	0.0	0.0	0.0
HYSC	*Hypericum scouleri*	Western St. Johns wort	Yes	FAC	0.0	0.9	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Native	Wetland Status	Ilwaco Slough (40)	Welch Island (41)	Whites Island (44)	Cunningham Lake (69)	Campbell Slough (65)	Franz Lake (61)
IMSP	*Impatiens species*	western touch-me-not, common touch-me-not, jewelweed	Yes	FACW	0.0	0.7	0.4	0.0	0.0	0.0
IRPS	*Iris pseudacorus*	Yellow iris	No	OBL	0.0	2.2	3.0	T	0.0	0.0
ISCE	*Isolepis cernua*	low bulrush	Yes	OBL	2.9	0.0	0.0	0.0	0.0	0.0
JUOX	*Juncus oxymeris*	Pointed rush	Yes	FACW	0.0	1.8	0.3	0.0	T	0.0
LAPA	*Lathyrus palustris*	Marsh peavine	Yes	OBL	0.0	0.9	0.0	0.0	0.0	0.0
LEOR	*Leersia oryzoides*	Rice cutgrass	Yes	OBL	0.0	0.5	2.4	0.1	0.0	1.2
LIAQ	*Limosella aquatica*	Water mudwort	Yes	OBL	0.0	0.0	0.1	0.0	0.0	0.0
LIDU	*Lindernia dubia*	yellow seed false pimpernel	Yes	OBL	0.0	0.0	0.0	0.0	0.1	0.0
LIOC	*Lilaeopsis occidentalis*	Western lilaeopsis	Yes	OBL	6.4	0.0	0.0	0.0	0.0	0.0
LISC	*Lilaea scilloides*	Flowering quillwort	Yes	OBL	0.0	0.0	T	0.0	0.0	0.0
LOCO	*Lotus corniculatus*	Birdsfoot trefoil	No	FAC	0.0	0.0	1.0	0.0	0.0	0.0
LUPA	*Ludwigia palustris*	False loosestrife	Yes	OBL	0.0	0.0	0.0	3.6	1.2	2.0
LYAM	*Lysichiton americanus*	Skunk cabbage	Yes	OBL	0.0	2.6	0.5	0.0	0.0	0.0
LYAM2	*Lycopus americanus*	American water horehound	Yes	OBL	0.0	0.0	0.4	0.0	0.0	0.0
LYNU	*Lysimachia nummularia L.*	Moneywort, Creeping Jenny	No	FACW	0.0	0.0	0.0	0.0	0.6	0.0
LYSA	*Lythrum salicaria*	Purple loosestrife	No	FACW	0.0	1.1	1.1	0.0	0.0	0.0
LYUN	*Lycopus uniflorus*	Northern bugleweed	Yes	OBL	0.0	0.3	0.0	0.0	0.0	0.0
MEAR	*Mentha arvensis*	wild mint	Yes	FACW	0.0	0.0	0.0	0.0	0.0	0.3
MIGU	*Mimulus guttatus*	Yellow monkeyflower	Yes	OBL	0.0	0.4	T	0.0	0.0	0.0
MUKE	*Murdannia keisak*	wart-removing herb	No	OBL	0.0	0.4	0.0	0.0	0.0	0.0
MYLA	*Myosotis laxa*	Small forget-me-not	Yes	OBL	0.0	0.0	2.0	0.0	0.0	0.0
MYSP	*Myosotis laxa, M. scorpioides*	Small forget-me-not, Common forget-me-not	Mixed	Mixed	0.0	6.2	0.4	0.0	0.0	0.0
OESA	*Oenanthe sarmentosa*	Water parsley	Yes	OBL	0.0	5.8	4.1	0.0	0.0	0.0
PHAR	*Phalaris arundinacea*	Reed canarygrass	No	FACW	0.0	8.3	43.7	22.6	23.8	12.3
PLMA	*Plantago major*	common plantain	No	FACU	0.0	0.0	0.0	0.0	T	0.0

Code	Scientific Name	Common Name	Native	Wetland Status	Ilwaco Slough (40)	Welch Island (41)	Whites Island (44)	Cunningham Lake (69)	Campbell Slough (65)	Franz Lake (61)
POAM	*Polygonum amphibium*	water ladysthumb, water smartweed	Yes	OBL	0.0	0.0	0.0	0.0	0.1	44.1
POCR	*Potamogeton crispus*	curly leaf pondweed	No	OBL	0.0	0.0	0.0	0.0	T	0.0
POHY	*Polygonum hydropiper, P. hydropiperoides*	waterpepper, mild waterpepper, swamp smartweed	Mixed	OBL	0.0	0.3	0.0	0.0	0.0	0.0
PONA	*Potamogeton natans*	Floating-leaved pondweed	Yes	OBL	0.0	0.0	0.0	T	0.9	0.0
POPE	*Polygonum persicaria*	Spotted ladysthumb	No	FACW	0.0	2.4	1.2	1.4	0.4	0.1
POPU	*Potamogeton pusillus*	Small pondweed	Yes	OBL	0.0	0.0	T	0.0	0.0	0.0
PORI	*Potamogeton richardsonii*	Richardson's pondweed	Yes	OBL	0.0	0.0	0.1	0.0	0.0	0.0
POSP	*Polygonum sp.*	Knotweed, Smartweed	Mixed	Mixed	0.0	T	0.0	0.0	0.0	0.0
POZO	*Potamogeton zosteriformis*	Eelgrass pondweed	Yes	OBL	0.0	0.0	0.1	0.0	0.0	0.0
ROPA	*Rorippa palustris*	Marsh yellow-cress	Yes	OBL	0.0	0.0	0.0	T	0.0	0.8
RUAQ	*Rumex aquaticus*	Western dock	Yes	FACW	0.0	0.0	T	0.0	0.0	0.0
RUMA	*Rumex maritimus*	Golden dock, seaside dock	Yes	FACW	0.0	0.1	0.1	0.0	0.0	0.0
SALA	*Sagittaria latifolia*	Wapato	Yes	OBL	0.0	4.8	9.1	18.2	16.0	4.6
SALU	*Salix lucida*	Pacific willow	Yes	FACW	0.0	0.0	0.0	11.0	0.0	27.4
SASE	*Salix sessilifolia Nutt.*	Columbia River willow, river willow	Yes	FACW	0.0	0.0	0.0	0.0	0.0	0.1
SASI	*Salix sitchensis*	Sitka willow	Yes	FACW	0.0	0.2	2.5	0.0	0.0	0.0
SCAM	*Schoenoplectus americanus*	American bulrush, threesquare bulrush	Yes	OBL	2.5	0.0	0.1	0.0	0.0	0.0
SCMA	*Schoenoplectus maritimus*	Seacoast bulrush	Yes	OBL	0.9	0.0	0.0	0.0	0.0	0.0
SCTA	*Schoenoplectus tabernaemontani*	Softstem bulrush, tule	Yes	OBL	0.0	0.0	0.0	0.0	0.0	0.1
SISU	*Sium suave*	Hemlock waterparsnip	Yes	OBL	0.0	2.1	0.3	0.0	0.0	0.0
SODU	*Solanum dulcamara*	Bittersweet nightshade	No	FAC	0.0	0.0	0.6	0.0	0.0	0.1

Code	Scientific Name	Common Name	Native	Wetland Status	Ilwaco Slough (40)	Welch Island (41)	Whites Island (44)	Cunningham Lake (69)	Campbell Slough (65)	Franz Lake (61)
SPAN	*Sparganium angustifolium*	Narrowleaf burreed	Yes	OBL	0.0	T	T	0.9	0.0	0.0
SYEA	*Symphyotrichum eatonii*	Eaton's aster	Yes	FAC	0.0	0.6	0.1	0.0	0.0	0.0
SYSU	*Symphyotrichum subspicatum*	Douglas aster	Yes	FACW	1.0	1.4	0.1	0.0	0.0	0.0
TRMA	*Triglochin maritima*	Seaside arrowgrass	Yes	OBL	5.7	0.0	0.0	0.0	0.0	0.0
TRWO	*Trifolium wormskioldii*	Springbank clover	Yes	FACW	0.0	0.1	0.0	0.0	0.0	0.0
TYAN	*Typha angustifolia*	Narrowleaf cattail	No	OBL	2.7	0.0	0.2	0.0	0.0	0.0
VEAM	*Veronica americana*	American speedwell	Yes	OBL	0.0	0.0	0.0	0.3	0.9	0.0
ZAPA	*Zannichellia palustris*	horned pondweed	Yes	OBL	2.6	0.0	0.0	0.0	0.0	0.0
Other Cover										
ALGAE		Algae			14.6	0.0	0.0	2.4	1.2	0.0
BG		Bare ground			12.3	6.1	16.2	25.1	48.2	28.8
d-AGSP	dead *Agrostis species*	Dead bentgrass	Mixed	Mixed	1.2	0.0	0.0	0.0	0.0	0.0
d-CALY	dead *Carex lyngbyei*	Dead Lyngby sedge	Yes	OBL	3.7	0.0	0.0	0.0	0.0	0.0
d-PHAR	dead *Phalaris arundinacea*	Dead reed canarygrass	No	FACW	0.0	0.0	0.3	17.4	1.3	0.1
d-POAM	dead *Polygonum amphibium*	Dead water ladysthumb, water smartweed	Yes	OBL	0.0	0.0	0.0	0.0	0.0	5.7
d-SALA	dead *Sagittaria latifolia*	Dead wapato	Yes	OBL	0.0	0.0	0.0	0.0	T	0.0
d-TYAN	dead *Typha angustifolia*	Dead narrowleaf cattail	No	OBL	0.3	0.0	0.1	0.0	0.0	0.0
Detritus		Detritus			9.5	13.3	18.9	4.8	9.2	3.2
DW		Drift Wrack			1.3	0.3	0.6	4.8	0.0	0.0
Grazed		Grazed			0.0	0.0	0.0	3.2	45.8	0.0
Litter		Litter			0.0	0.0	0.7	1.1	0.1	2.6
LWD		Large Woody Debris			T	0.0	1.9	0.3	0.1	1.2
MH		Mixed Herbs			0.0	0.0	T	0.6	0.2	1.5
MOSS		Moss			0.0	0.0	0.0	0.0	0.0	4.4
OW		Open Water			10.8	7.8	17.4	6.5	34.9	27.5
TRAMP		Trampled			0.0	0.0	0.0	5.8	1.2	0.0

Appendix E. Vegetation Biomass 2011-2017

Table E1. Definition of site codes and vegetation strata used for the biomass sampling design.

Site Code	Site		Strata Code	Strata
BBM	Ilwaco		CALY	*Carex lyngbyei*
SRM	Secret River		E/S	*Eleocharis palustris/Sagittaria latifolia*
WI2	Welch Island		HM	High Marsh mix
WHC	Whites Island		LM	Low Marsh mix
CLM	Cunningham Lake		LEOR	*Leerza oryzoides*
CS1	Campbell Slough		PHAR	*Phalaris arundinacea*
FLM	Franz Lake		P/H	*P. arundinacea*/High Marsh mix
			SALA	*S. latifolia*

Figure E.1. Box plots of the average aboveground vegetation biomass by site and vegetation strata. Site codes and strata codes are provided in Table E.1.
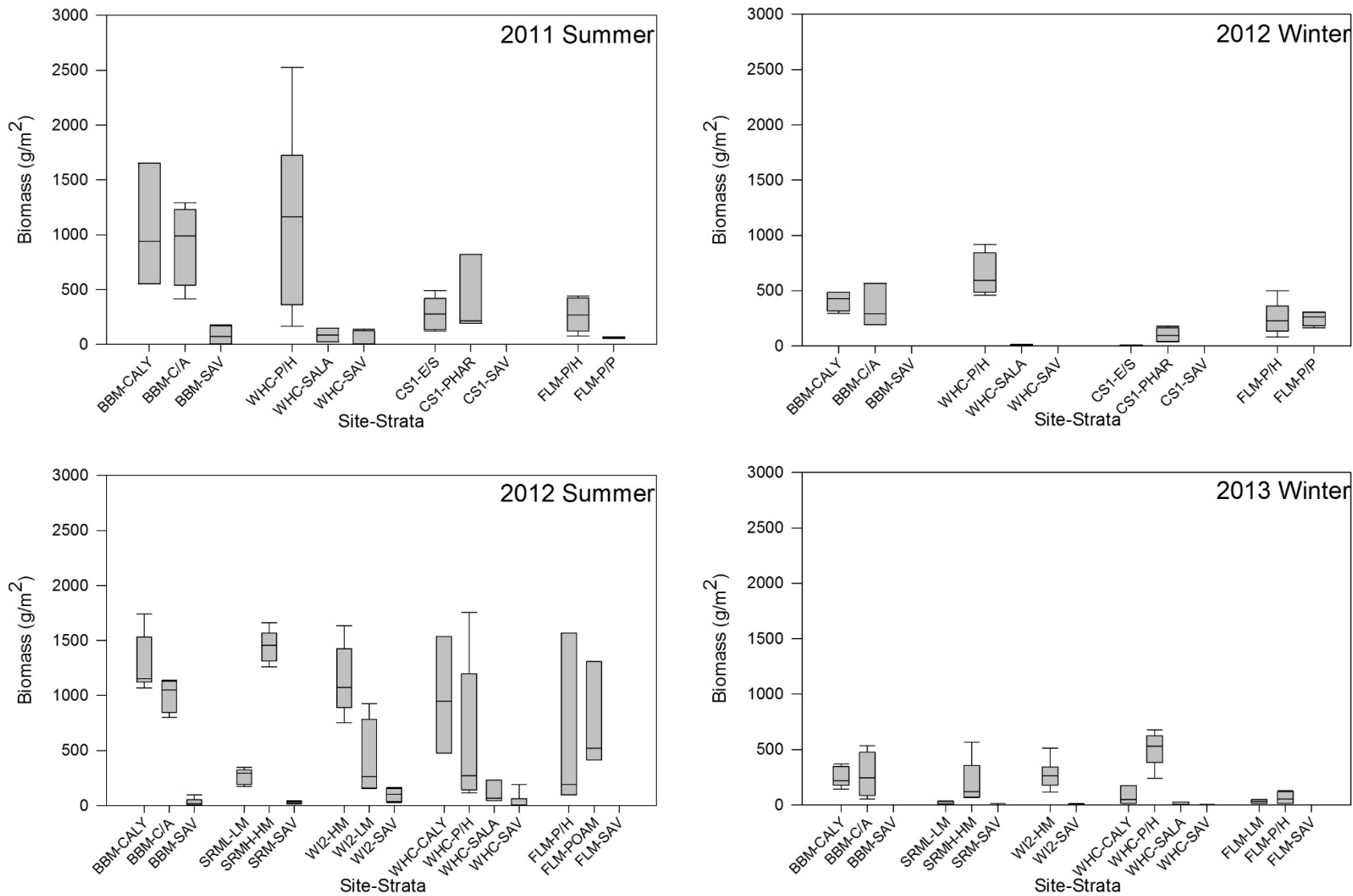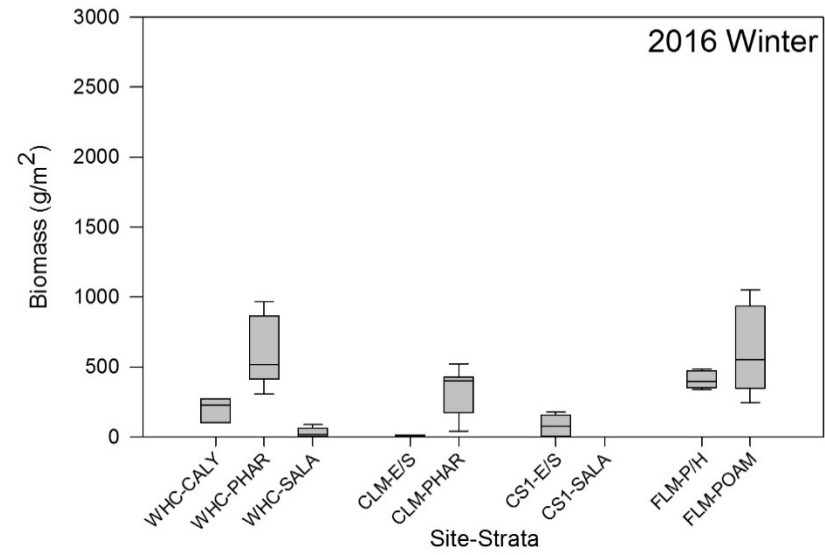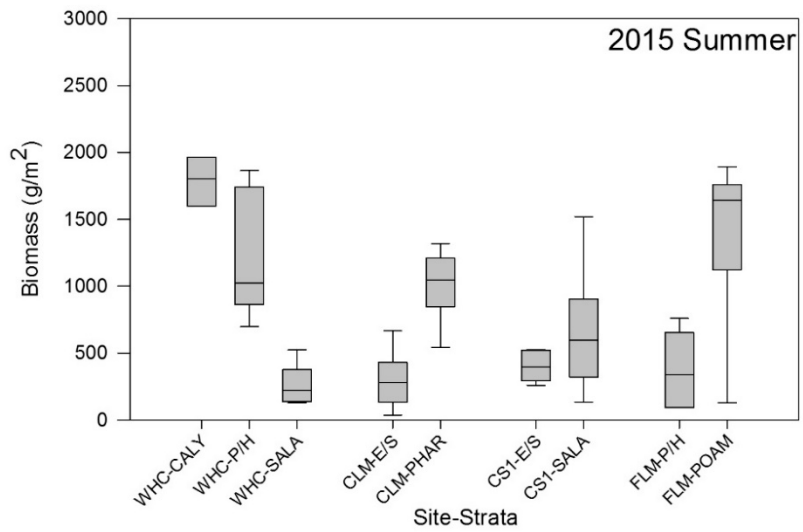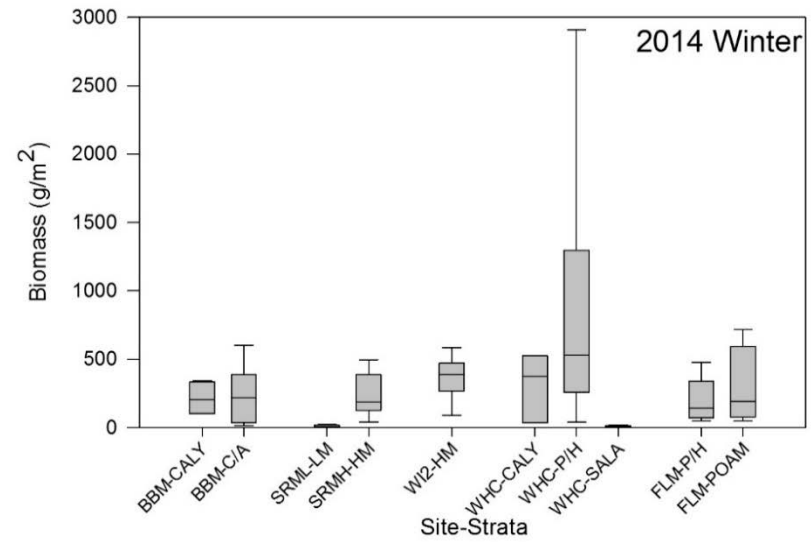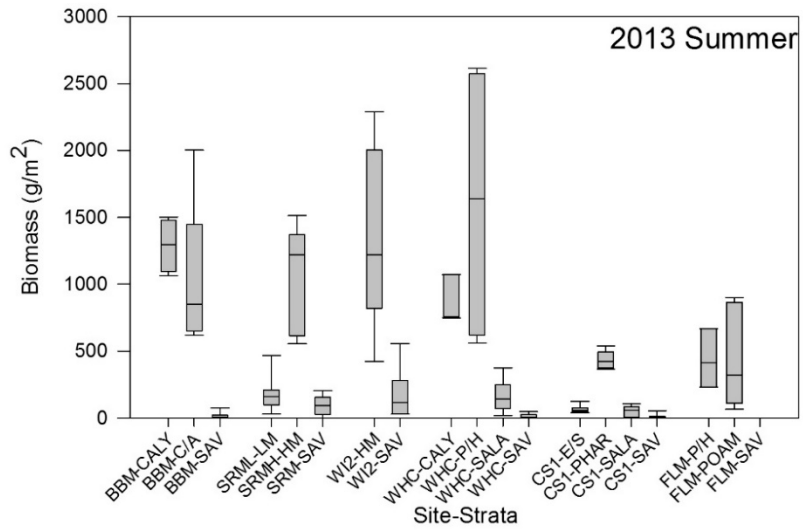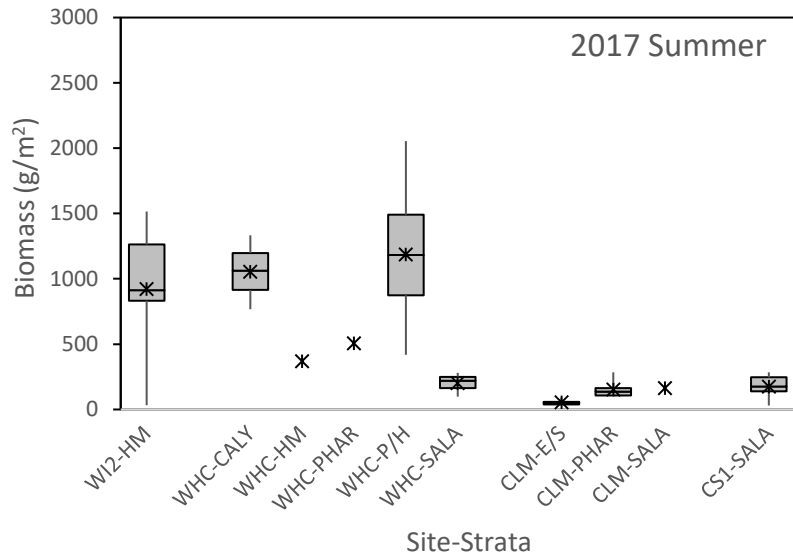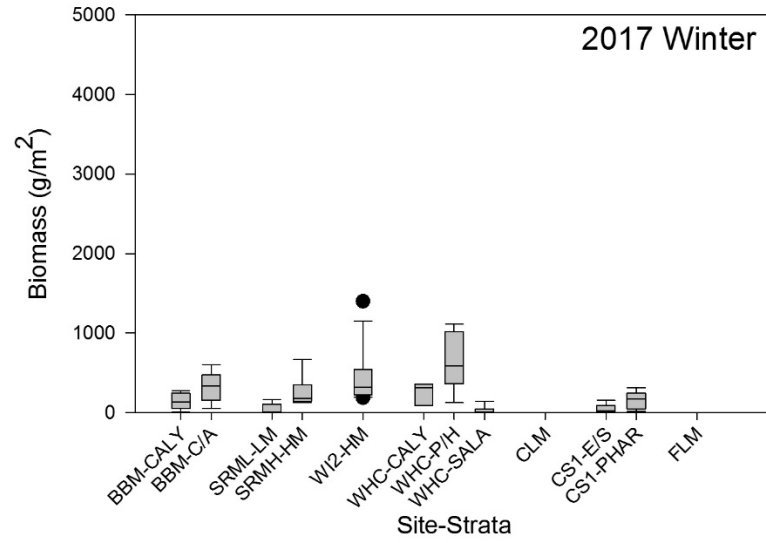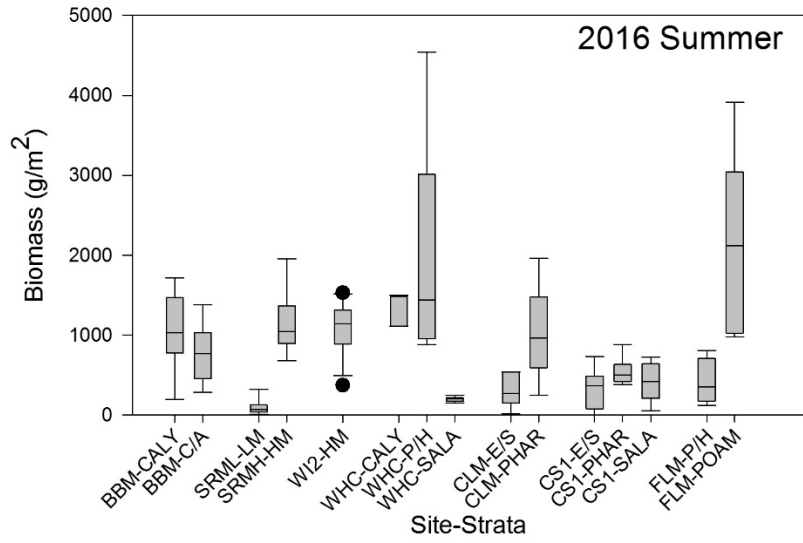
Table E2. Aboveground vegetation biomass sampling results from Summer 2011 to Summer 2017. Site codes and vegetation strata codes are defined in Table E1. Contribution is the amount of biomass, or organic material, "contributed" to the ecosystem and is calculated as the Summer biomass minus Winter biomass.

Site	Vegetation Strata*	Year	Season	n	Average Dry wt (g/m²)	StdDev Dry wt (g/m²)	Year	Season	n	Average Dry wt (g/m²)	StdDev Dry wt (g/m²)	Contribution Dry Wt (g/m²/year)
BBM	CALY	2011	Summer	3	1049.3	558.3	2012	Winter	4	409.8	91.9	639.5
BBM	CALY	2012	Summer	6	1285.4	260.6	2013	Winter	6	245.3	89.0	1040.2
BBM	CALY	2013	Summer	4	1289.9	201.2	2014	Winter	4	213.6	124.4	1076.3
BBM	CALY	2016	Summer	6	1057.8	509.7	2017	Winter	6	141.4	107.7	916.3
BBM	CALY/AGSP	2011	Summer	4	921.4	370.1	2012	Winter	3	351.3	194.3	570.1
BBM	CALY/AGSP	2012	Summer	4	1009.4	153.5	2013	Winter	4	267.1	203.5	742.4
BBM	CALY/AGSP	2013	Summer	6	1041.6	527.3	2014	Winter	6	235.7	213.4	805.9
BBM	CALY/AGSP	2016	Summer	6	771.0	376.6	2017	Winter	6	325.1	195.4	445.9
BBM	SAV	2011	Summer	4	81.8	91.3	2012	Winter	4	0.0	0.0	81.8
BBM	SAV	2012	Summer	6	28.5	38.1	2013	Winter	6	0.1	0.1	28.5
BBM	SAV	2013	Summer	6	14.5	30.1	assume 0*			0.0	na	14.5
CLM	ELPA/SALA	2015	Summer	6	295.8	212.3	2016	Winter	6	6.5	6.1	289.2
CLM	ELPA/SALA	2016	Summer	6	305.1	205.5	2017	Winter	nd	nd	nd	nd
CLM	ELPA/SALA	2017	Summer	5	54.9	21.9						
CLM	PHAR	2015	Summer	7	1007.8	265.1	2016	Winter	7	318.7	170.8	689.1
CLM	PHAR	2016	Summer	7	1035.2	561.3	2017	Winter	nd	nd	nd	nd
CLM	PHAR	2017	Summer	6	154.2	69.3						
CLM	SALA	2017	Summer	1	161.9	na						
CS1	ELPA/SALA	2011	Summer	5	277.6	150.9	2012	Winter	4	3.3	4.5	274.4
CS1	ELPA/SALA	2013	Summer	6	65.0	30.3	2014	Winter	nd	nd	nd	nd
CS1	ELPA/SALA	2015	Summer	6	399.9	115.9	2016	Winter	6	79.5	82.8	320.4
CS1	ELPA/SALA	2016	Summer	7	350.0	246.1	2017	Winter	6	48.1	56.4	301.9
CS1	PHAR	2011	Summer	3	410.1	356.0	2012	Winter	4	100.8	63.9	309.4
CS1	PHAR	2013	Summer	6	433.8	66.6	2014	Winter	nd	nd	nd	nd
CS1	PHAR	2016	Summer	6	541.7	178.1	2017	Winter	6	158.8	109.0	383.0

Site	Vegetation Strata*	Year	Season	n	Average Dry wt (g/m²)	StdDev Dry wt (g/m²)	Year	Season	n	Average Dry wt (g/m²)	StdDev Dry wt (g/m²)	Contribution Dry Wt (g/m²/year)
CS1	SALA	2013	Summer	5	45.9	46.0	2014	Winter	nd	nd	nd	nd
CS1	SALA	2015	Summer	6	654.5	469.6	2016	Winter	6	0.0	0.1	654.4
CS1	SALA	2016	Summer	6	417.6	245.7	2017	Winter	6	0.0	0.0	417.6
CS1	SALA	2017	Summer	6	176.9	92.9						
CS1	SAV	2011	Summer	8	0.4	0.8	2012	Winter	8	0.0	0.0	0.4
CS1	SAV	2013	Summer	6	9.3	22.8	assume 0*			0.0	na	9.3
FLM	LM	2011	Summer	1	141.4	na	2012	Winter	1	66.2	na	75.2
FLM	LM	2012	Summer	1	601.1	na	2013	Winter	2	30.5	23.7	570.7
FLM	PHAR/HM	2011	Summer	5	271.6	154.0	2012	Winter	7	243.9	145.6	27.7
FLM	PHAR/HM	2012	Summer	3	619.9	823.6	2013	Winter	4	62.1	56.5	557.7
FLM	PHAR/HM	2013	Summer	3	438.8	217.5	2014	Winter	5	192.5	167.8	246.3
FLM	PHAR/HM	2015	Summer	6	373.5	273.2	2016	Winter	6	405.7	60.2	-32.2
FLM	PHAR/HM	2016	Summer	6	418.9	278.2	2017	Winter	nd	nd	nd	nd
FLM	PHAR/POAM	2011	Summer	2	63.1	11.7	2012	Winter	5	247.5	64.0	-184.5
FLM	POAM	2012	Summer	3	746.7	488.4	2013	Winter	1	274.2	na	472.5
FLM	POAM	2013	Summer	6	431.2	377.2	2014	Winter	4	286.1	295.9	145.1
FLM	POAM	2015	Summer	6	1412.4	645.2	2016	Winter	6	613.7	307.5	798.7
FLM	POAM	2016	Summer	6	2154.8	1143.6	2017	Winter	nd	nd	nd	nd
FLM	SAV	2012	Summer	5	0.0	0.0	2013	Winter	6	0.0	0.0	0.0
SRM	HM	2012	Summer	5	1443.3	148.2	2013	Winter	5	194.5	209.5	1248.8
SRM	HM	2013	Summer	9	1061.7	385.8	2014	Winter	9	240.9	151.1	820.9
SRM	HM	2016	Summer	9	1147.5	389.0	2017	Winter	9	268.2	173.6	879.3
SRM	LM	2012	Summer	5	265.1	71.4	2013	Winter	5	15.2	14.9	249.9
SRM	LM	2013	Summer	9	175.0	124.2	2014	Winter	9	8.9	8.7	166.1
SRM	LM	2016	Summer	9	99.6	92.7	2017	Winter	9	47.8	62.1	51.8
SRM	SAV	2012	Summer	6	30.1	11.8	2013	Winter	6	2.4	5.1	27.7
SRM	SAV	2013	Summer	6	94.4	77.2	assume 10*			10.0	na	84.4
WHC	CALY	2012	Summer	nd	nd	nd	2012	Winter	1	18.0	na	nd

Site	Vegetation Strata*	Year	Season	n	Average Dry wt (g/m²)	StdDev Dry wt (g/m²)	Year	Season	n	Average Dry wt (g/m²)	StdDev Dry wt (g/m²)	Contribution Dry Wt (g/m²/year)
WHC	CALY	2012	Summer	3	987.8	530.0	2013	Winter	3	78.7	84.5	909.1
WHC	CALY	2013	Summer	3	860.0	183.7	2014	Winter	3	311.4	252.0	548.5
WHC	CALY	2015	Summer	3	1788.6	182.8	2016	Winter	3	199.5	86.8	1589.1
WHC	CALY	2016	Summer	3	1366.8	220.1	2017	Winter	3	253.0	142.7	1113.7
WHC	CALY	2017	Summer	3	1054.3	283.0						
WHC	HM	2017	Summer	1	367.5	na						
WHC	PHAR	2017	Summer	1	504.1	na						
WHC	PHAR/HM	2011	Summer	6	1152.3	844.4	2012	Winter	4	641.9	196.6	510.3
WHC	PHAR/HM	2012	Summer	5	590.0	682.2	2013	Winter	5	507.1	163.0	83.0
WHC	PHAR/HM	2013	Summer	6	1608.1	935.0	2014	Winter	6	849.3	1038.4	758.9
WHC	PHAR/HM	2015	Summer	9	1232.5	445.0	2016	Winter	9	596.0	239.6	636.5
WHC	PHAR/HM	2016	Summer	9	1947.0	1321.1	2017	Winter	9	623.4	351.9	1323.6
WHC	PHAR/HM	2017	Summer	9	1188.3	568.3						
WHC	SALA	2011	Summer	2	87.7	88.8	2012	Winter	3	5.6	6.4	82.1
WHC	SALA	2012	Summer	3	114.0	101.8	2013	Winter	3	10.0	14.6	103.9
WHC	SALA	2013	Summer	6	162.8	126.2	2014	Winter	6	9.5	5.4	153.3
WHC	SALA	2015	Summer	6	260.8	152.3	2016	Winter	6	31.5	33.6	229.3
WHC	SALA	2016	Summer	6	198.4	31.5	2017	Winter	6	32.2	55.5	166.2
WHC	SALA	2017	Summer	6	203.8	68.4						
WHC	SAV	2011	Summer	8	49.3	65.0	2012	Winter	8	0.4	0.5	48.9
WHC	SAV	2012	Summer	6	35.8	75.8	2013	Winter	6	0.3	0.6	35.5
WHC	SAV	2013	Summer	6	11.2	19.6	*assume 0**			0.0	na	11.2
WI2	HM	2012	Summer	5	1141.5	322.5	2013	Winter	9	272.2	122.2	869.3
WI2	HM	2013	Summer	9	1360.9	647.0	2014	Winter	9	364.6	149.8	996.3
WI2	HM	2016	Summer	12	1094.7	319.6	2017	Winter	12	432.1	334.6	662.7
WI2	HM	2017	Summer	14	922.5	455.6						
WI2	LM	2012	Summer	4	401.3	362.2	2013	Winter	nd	nd	nd	nd
WI2	SAV	2012	Summer	4	97.7	62.2	2013	Winter	4	6.2	4.2	91.5

Site	Vegetation Strata*	Year	Season	n	Average Dry wt (g/m^2)	StdDev Dry wt (g/m^2)	Year	Season	n	Average Dry wt (g/m^2)	StdDev Dry wt (g/m^2)	Contribution Dry Wt (g/m^2/year)
WI2	SAV	2013	Summer	6	173.2	197.6	*assume 10**			10.0	na	163.2

* Winter SAV was not collected in 2013; however based on observations and previous year's data we assumed that the winter biomass is 0.0 g/m^2. Sites that had >0>10 g/m^2 in previous years were assigned a value of 10 g/m^2.

na – not applicable

nd – no data

Appendix F. Fish catch summaries, 2008–2017

Table F-1. Species list of fishes captured during the EMP study, by year, 2008-2017 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
Clupeidae	American shad[i]	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
	chum salmon	x	x	x	x	x	x		x	x	x
	coho salmon	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					
	unid whitefish					x					
Osmeridae	eulachon						x	x			
	longfin smelt						x				
	unid smelt				x	x	x		x	x	
Cyprinidae	chiselmouth	x	x	x	x	x	x	x	x	x	
	golden shiner[i]				x	x					x
	northern pikeminnow[p]	x	x	x	x	x	x	x	x	x	
	peamouth	x	x	x	x	x	x	x	x	x	x
	redside shiner									x	
	speckled dace					x					
	tui chub		x	x	x	x	x	x	x	x	
	unid carp[i]	x	x	x	x	x	x	x	x	x	
	unid Cyprinidae[i]		x		x						
Catostomidae	largescale sucker	x	x	x	x	x	x	x	x	x	x
Cobitidae	oriental weatherfish[i]			x						x	
Ictaluridae	brown bullhead[i]	x	x						x		

Family	Species	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
	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
Poeciliidae	gambusia[i]			x							x
Gasterosteidae	threespine stickleback	x	x	x	x	x	x	x	x	x	x
Centrarchidae	bluegill[i]		x	x	x	x	x	x	x		
	largemouth bass[ip]		x							x	x
	pumpkinseed[i]	x	x	x	x	x	x	x	x	x	
	smallmouth bass[ip]	x	x	x	x	x	x	x	x	x	
	unid crappie[i]		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
Embiotocidae	shiner perch				x	x	x		x	x	x
Ammodytidae	sandlance				x					x	
Gobiidae	Amur goby[i]									x	x
	unid goby[i]		x						x	x	
Cottidae	mottled sculpin		x								
	Pacific staghorn sculpin				x	x	x	x	x	x	x
	Prickly sculpin										x
	unid sculpin	x	x	x	x	x	x	x	x	x	x
Pleuronectidae	starry flounder		x	x		x	x	x	x		

[i]Introduced species, [p]Predatory to salmon

Table F-2. Species list of fishes captured at the five EMP trend sites in 2017.

Family	Species_Name	Ilwaco Slough	Welch Island	White Island	Campbell Slough
Clupeidae	American shad[i]	x	x		x
Salmonidae	Chinook salmon	x	x	x	x
	chum salmon	x	x	x	x
	coho salmon			x	
	mountain whitefish			x	
Cyprinidae	golden shiner[i]				x
	peamouth		x	x	x
Catostomidae	largescale sucker				x
Fundulidae	banded killifish[i]	x	x	x	x
Poeciliidae	gambusia[i]				x
Gasterosteidae	threespine stickleback	x	x	x	x
Centrarchidae	largemouth bass[ip]		x		
	Unidentified centrarchid[i]				x
Percidae	yellow perch[ip]		x	x	x
Embiotocidae	shiner perch	x			
Gobiidae	Amur goby[i]				x
Cottidae	Pacific staghorn sculpin	x			
	Prickly sculpin		x	x	x
	unid sculpin			x	

[i]Introduced species, [p]Predatory to salmon

Table F-3. Total number of unmarked and marked Chinook salmon captured by year at each site 2008 – 2017, ns = site not sampled.

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

Table F-4. Juvenile Coho (unmarked and marked), chum salmon, sockeye salmon, and trout densities (fish per 1000 m^2) (SD), by year captured at trends sites 2008 - 2017. Total number of salmonids captured at a site is presented in parentheses under site/species/#.

Site/Species/#	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Ilwaco Slough										
Coho (unmarked) (1)	ns	ns	ns	0	0.12 (0.67)	0	0	0	0	0
Coho (marked) (0)	ns	ns	ns	0	0	0	0	0	0	0
Chum (396)	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)
Sockeye (0)	ns	ns	ns	0	0	0	0	0	0	0
Trout sp (0)	ns	ns	ns	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
Coho (marked) (1)	ns	ns	ns	ns	0.11 (0.54	0	0	0	0	0
Chum (23)	ns	ns	ns	ns	0.14 (0.71)	0.26 (1.01)	0	0	0.65 (2.68)	2.50 (5.02)
Sockeye (1)	ns	ns	ns	ns	0	0	0.12 (0.41)	0	0	0
Trout sp (1)	ns	ns	ns	ns	0	0.27 (1.05)	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)
Coho (marked) (0)	ns	0	0	0	0	0	0	0	0	0
Chum (33)	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)
Sockeye (7)	ns	0	0	0	0	1.32 (5.77)	0	0	0	0
Trout sp (0)	ns	0	0	0	0	0	0	0	0	0

Continued from Table F-4.

Site/Species/#	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Campbell Slough										
Coho (unmarked) (2)	0	0	0	0	0	0	0	0.18 (0.70)	0	0
Coho (marked) (1)	0	0	0	0	0	0	0.18 (0.51)	0	0	0
Chum (9)	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)
Sockeye (1)	0	0	0	0	0	0	0.09 (0.38)	0	0	0
Trout sp. (1)	0	0	0	0	0	0	0.78 (0.33)	0	0	0
Franz Lake										
Coho (unmarked) (31)	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
Coho (marked) (60)	5.75 (10.72)	7.92 (17.99)	ns	0	0.51 (2.17)	0	0	0	0	ns
Chum (7)	2.11 (5.23)	0.32 (1.09)	ns	0	0	0	0	0	0	ns
Sockeye (2)	0	0	ns	0	0	0	0.55 (1.56)	0	0	ns
Trout sp. (5)	0	0.95 (2.30)	ns	0	1.43 (6.54)	0	0	0	0	ns

The header column "Year" spans across the year columns (2008–2017).