Lower Columbia River Ecosystem Monitoring Program

Annual Report for Year 12

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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 toward 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 2016, data were collected on fish, 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 the only metrics collected at Secret River (rkm 37), and Cunningham Lake (rkm 145) in 2016. 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.

A high level of connectedness exists between the conditions within the Columbia River mainstem and shallow off-channel habitats; conditions are often similar during periods of high flow, whereas they tend to diverge as flows subside. High quality conditions in both the mainstem and off-channel habitats are critical for the growth and survival of juvenile salmonids. 2016 water temperatures in the mainstem river were generally cooler than 2015, with fewer days when water temperature exceeded thresholds for salmon. At the trends sites, water temperatures were generally higher than the 2000-2014 daily averages, but lower than 2015 and elevated water temperatures occurred at Campbell Slough and Franz Lake earlier than at other sites. Larger daily fluctuations in dissolved oxygen and pH occurred at the trends sites than in the mainstem. The highest number of hours per month with dissolved oxygen levels <6 mg L^{-1} was observed at Franz Lake in July and August, while dissolved oxygen often dropped below this threshold throughout the monitoring period at Ilwaco Slough.

Hydrologic patterns, sediment accretion, and vegetation composition and cover were monitored at seven emergent wetland sites in the lower river. Cumulative inundation (i.e., sum exceedance value; SEV) varies spatially and generally increases with distance from the river mouth. However, in 2016 the SEV was similar across the four trends sites that were monitored, with the greatest inundation occurring during a series of low-level flood events in winter and spring. Sediment accretion rates in 2016 were variable in time and space (0.0 to 2.9 cm per year on average) and erosion occurred at Franz Lake. Accretion rates were most consistent over time at Welch Island (0.7 ± 0.1 cm) and most variable across years at Campbell Slough and Franz Lake. Emergent wetland vegetation cover and composition are largely driven by hydrologic patterns. Non-native reed canarygrass (*Phalaris arundinacea*) was the most dominant species of vegetation (22.7%), although the average cover of *Carex lyngyei* was only slightly less at 19.1%. An increase in cover of the native species wapato (*Sagittaria latifolia*) at Campbell Slough and Cunningham Lake and water smartweed (*Polygonum amphibium*) at Franz Lake was observed in 2016. We conducted a multi-year analysis of aboveground plant biomass collection to evaluate overall averages and variability in vegetation strata throughout the estuary. We concluded that while the sites are very productive, variability is high. The lowest variability and most consistent production occurs in communities dominated by *Carex lyngbyei*.

Diatoms typically dominated the phytoplankton community in the mainstem and in the early spring at the trends sites. Cyanobacteria blooms were observed at Campbell Slough and Franz Lake by June. Spring zooplankton communities were dominated by rotifers and the concentration of all zooplankton was

highest at Campbell Slough. Stable isotope results suggest that algae is an important carbon source for salmon prey (e.g., amphipods, cladocerans, copepods) in the summer, whereas cladocerans and copepods may consume plant detritus in the spring. In May and June, chironomids likely rely on food sources from terrestrial plants. Wild and hatchery fish have significantly different carbon and nitrogen isotopic signatures, with wild fish being lighter in ^{13}C and heavier in ^{15}N.

Benthic macroinvertebrate results show a transition in the macroinvertebrate community along the estuarine gradient, with similarities in density and biomass among the lower estuary sites (Ilwaco Slough, Welch Island, and Whites Island) and the upriver sites (Campbell Slough and Franz Lake). Total benthic invertebrate counts were highest from Ilwaco Slough and amphipods densities, in particular, were higher than at other sites. The density and biomass of invertebrates captured in neuston samples were generally higher in emergent vegetation than in open water habitats. Dipterans were most abundant in emergent vegetation at Whites Island and Campbell Slough in June and at Franz Lake in April, while amphipods were most abundant in emergent vegetation at Welch Island in June. In the lower reaches juvenile Chinook salmon diets were dominated by amphipods, whereas chironomids, odonates, and copepods were preferred prey in the upper reaches. Growing conditions decreased over the sampling season. Maintenance metabolism and energy ration data indicate that juvenile Chinook salmon likely experience a trade off in late summer when water temperatures (and thus metabolic needs) increase, but more energy dense prey (e.g., hemipterans, hymenopterans) are available and often consumed. Condition factor and stomach fullness were reduced in juvenile Chinook salmon captured at Franz Lake in 2015 and 2016, which reflected a lower feeding intensity and energy consumption at this site compared to other sites.

Genetic analysis for fish captured in 2015 showed a lower than average proportion of interior Chinook salmon stocks (e.g., Snake River fall Chinook, Upper Columbia summer/fall Chinook, and Deschutes River fall Chinook) using the trends sites. The 2015 migration period appeared to be condensed, with juvenile Chinook salmon only captured between April and June, with most stocks migrating earlier in the season than in past years, suggesting a response to unusually warm conditions. However, lipid content in juvenile salmon collected in 2015 was relatively high, suggesting energy stores were similar to those in previous years. Growth rate variability was best explained by temporal factors (day and year), as well as by river kilometer and distance from the mainstem river. We also found that variability in relative growth rate was further explained by genetic stock, contaminants, and whether a fish was hatchery produced or unmarked. Sampling in the lower reaches of two tributaries in 2015 showed that West Cascades fall Chinook were the dominant stock in the lower Grays River, with some individuals from Columbia River Rogue and Spring Creek Group fall Chinook stocks and the catch in the lower Lewis River was composed entirely of West Cascades fall Chinook. Seasonal and spatial patterns of Chinook salmon occurrence were more typical in 2016 than what was observed in 2015. Sampling across the incoming tide at three sites (Ilwaco Slough, Whites Island, and Campbell Slough) showed differences in fish community composition and density of salmon and other fish species with the tidal cycle.

The Ecosystem Monitoring Program produces 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 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.

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). 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. 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.

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 showing the greatest species diversity; although some areas are dominated by the non-native reed canarygrass (*Phalaris arundinacea*), particularly in the river-dominated upper reaches (Sagar et al. 2013). Quantifying the variability in habitat metrics allows for greater predictability of how biota respond to changing environmental conditions and improves our understanding of how the lower river functions ecologically.

Salmonids occupy the upper trophic levels in the Columbia River system and they spend portions of their life cycle in fresh water, estuarine water, and oceanic water. 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, understanding the abiotic conditions characteristic of emergent wetlands, and in the river mainstem are essential for elucidating 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), United States Geological Survey (USGS), and Oregon Health and Sciences University (OHSU).

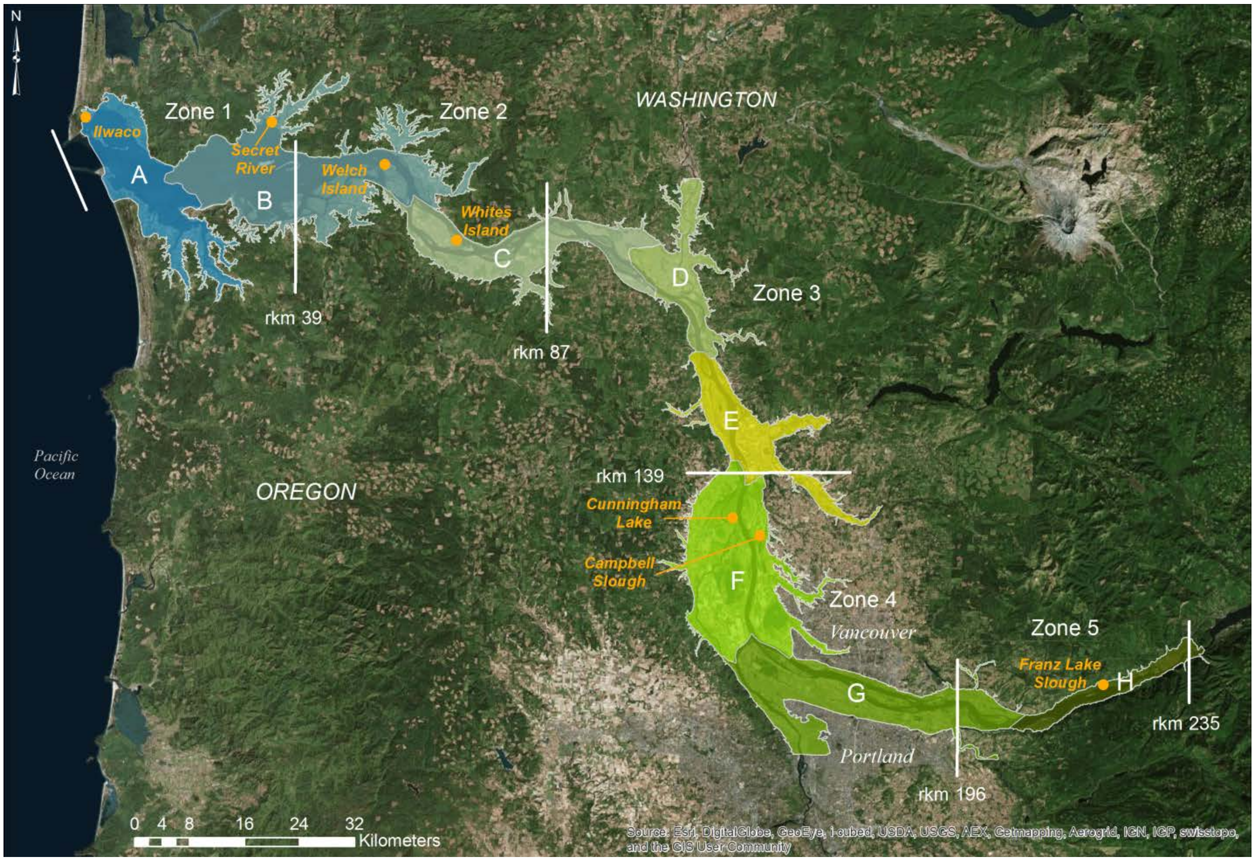
1.2 Study Area

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



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

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 2016 EMP trends sites are shown in orange.

1.3 Characterization of Emergent Wetlands in the Lower Columbia River

1.3.1 Sampling Effort, 2005-2016

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 has 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.

In 2013, the EMP sampling scheme was adjusted to no longer include data collection at status sites and monitoring efforts focused solely on the six trends sites. The six trends sites are: Ilwaco Slough (2010-2016), Secret River (2010-2016), Welch Island (2010-2016), Whites Island (2009-2016), Campbell Slough in the Ridgefield National Wildlife Refuge (2005–2016), and Franz Lake (2008-2009, 2011-2016). In 2016, an additional objective was undertaken by some EMP partners to address a question regarding juvenile salmon presence and food web resources at three trends sites (Campbell Slough, Whites Island, and Ilwaco Slough) through the incoming portion of the tidal cycle. 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 12 Contract (October 1, 2015 – September 30, 2016):

- 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 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 2016.

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

	Trend	**Campbell Slough**	CS1	2005-2016	2007-2016	2008-2016	2010-2016
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**	-			2012-2016	
H	Trend	**Franz Lake (slough)**	FLM	2008-2009, 2011-2016	2008-2009, 2011-2016	2011-2016	2011-2016
	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 – 2015.

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

1.3.2 Site Descriptions

In 2016, the EMP focused primarily on the six trends sites that were monitored over multiple years: Ilwaco Slough, Secret River, Welch Island, Whites Island, Campbell Slough, and Franz Lake Slough Habitat and hydrology data were collected at all six 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 2016 are listed in Table 2. The 2016 trends monitoring sites are described in order below, starting at the mouth of the Columbia 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, southeast 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 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*; Figure 2a). 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.

Secret River. The Secret River marsh, located in Reach B, EM Zone 1 in Grays Bay at the mouth of Secret River at rkm 37, is an extensive marsh owned by the Columbia Land Trust. The site was monitored as part of the Reference Site Study in 2008 (Borde et al. 2011). Although the marsh was present on the historical maps from the late 1880's, the marsh edge has receded approximately 400 m since then. The cause of this erosion is unknown. The marsh grades from *C. lyngbyei* and soft stem bulrush (*Schoenoplectus tabernaemontani*) in the low and mid marsh to a diverse mix of species in the upper marsh. The primary tidal channel is a low grade channel with low banks near the mouth, becoming steeper as it cuts through the higher marsh and then in to the tidal swamp above the marsh. Many smaller tidal channels also cut through the marsh plain. The marsh and the channel have large wood scattered throughout, with an accumulation at the high tide margin. Secret River was selected as a long-term monitoring site in 2012 and was sampled for all EMP metrics that year and in 2013. From 2014 through 2016, only habitat and hydrology data were collected at this site.

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.

Franz Lake. The long-term monitoring site located in Reach H, EM Zone 5, 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 2016.

Site Name	Latitude	Longitude
Ilwaco Slough	46°18.035'N	124° 2.784'W
Secret River	45° 9.561'N	122° 20.408'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) Secret River - 2016

c) Welch Island - 2016

d) Whites Island - 2016

e) Cunningham Lake - 2016

f) Campbell Slough - 2016

g) Franz Lake Slough – 2016

Figure 2. Ecosystem Monitoring sites sampled in 2016: (a) Ilwaco Slough; (b) Secret River; (c) Welch Island; (d) Whites Island; (e) Cunningham Lake; (f) Campbell Slough; (g) Franz Lake.

1.3.3 Water Year

The 2016 water year was characterized by near average water levels for most of the water year, with higher than average late winter and spring water levels, and slightly lower than average summer water levels as indicated by measurements just below Bonneville Dam (Figure 3). Peak water levels occurred in March rather than in the late spring and early summer. Winter high water was even more pronounced in December in the lower part of the estuary as runoff from the west side of the Cascade Range contributed to increased flows in the tributaries (see Section 3.1 for more information). Hydrographs of all measured years at the trends sites, including 2016, are provided in Appendix C.

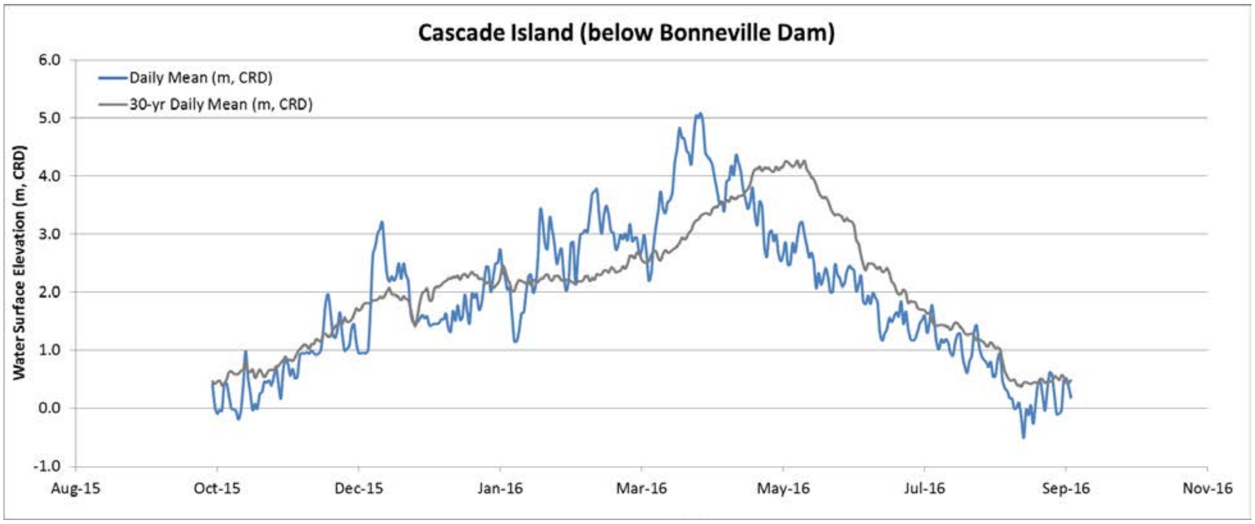


Figure 3. Water surface elevation at Cascade Island, just below Bonneville Dam (rkm 233), from October 2015 to October 2016 compared to the 29-year daily mean water surface elevation (Data from USGS National Water Information System at: http://waterdata.usgs.gov/nwis/).

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, indications 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).

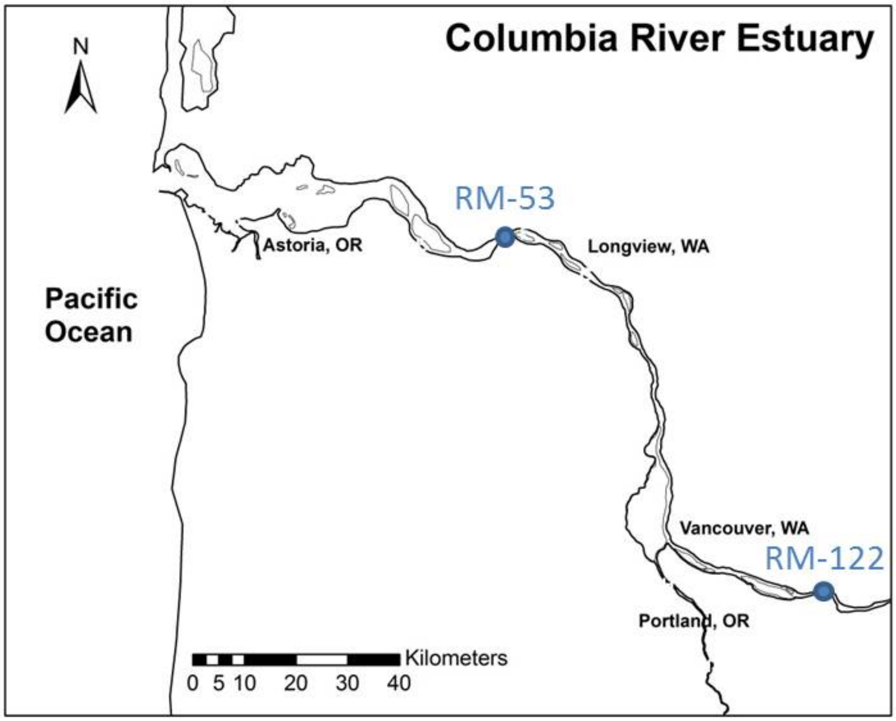


Figure 4. Station locations for the two in situ water quality monitoring platforms in the mainstem Columbia River that support the Ecosystem Monitoring Program. RM-53 (river mile 53) is Beaver Army Terminal, while RM-122 (river mile 122) is located in Camas, WA.

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.

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 2016, water quality was continuously monitored at four of the trends sites, Franz Lake, Campbell Slough, Whites Island, and Ilwaco Slough (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 at trends sites in 2016.

Site name*	USGS site number	USGS site name*	Reach	Latitude	Longitude	Monitor deployment date	Monitor retrieval date
Franz Lake	453604122060000	Franz Lake Slough Entrance, Columbia River, WA	H	45° 36' 04"	-122° 06’ 00”	3/31/16	8/30/16
Campbell Slough	454705122451400	Ridgefield NWR, Campbell Slough, Roth Unit, WA	F	45° 47’ 05”	-122° 45’ 15”	3/31/16	8/30/16
Whites Island	460939123201600	Birnie Slough, White’s Island, Columbia River, WA	C	46° 09’ 39”	-123° 20’ 16”	4/7/16	9/2/16
Ilwaco Slough			A	46° 18’ 19”	-124° 02’ 06”	3/29/16	9/2/16
Welch Island	461518123285700	Unnamed Slough, Welch Island, Columbia River, OR	B	46° 15' 18.4”	-123° 28' 56.8"	4/7/16	9/2/16

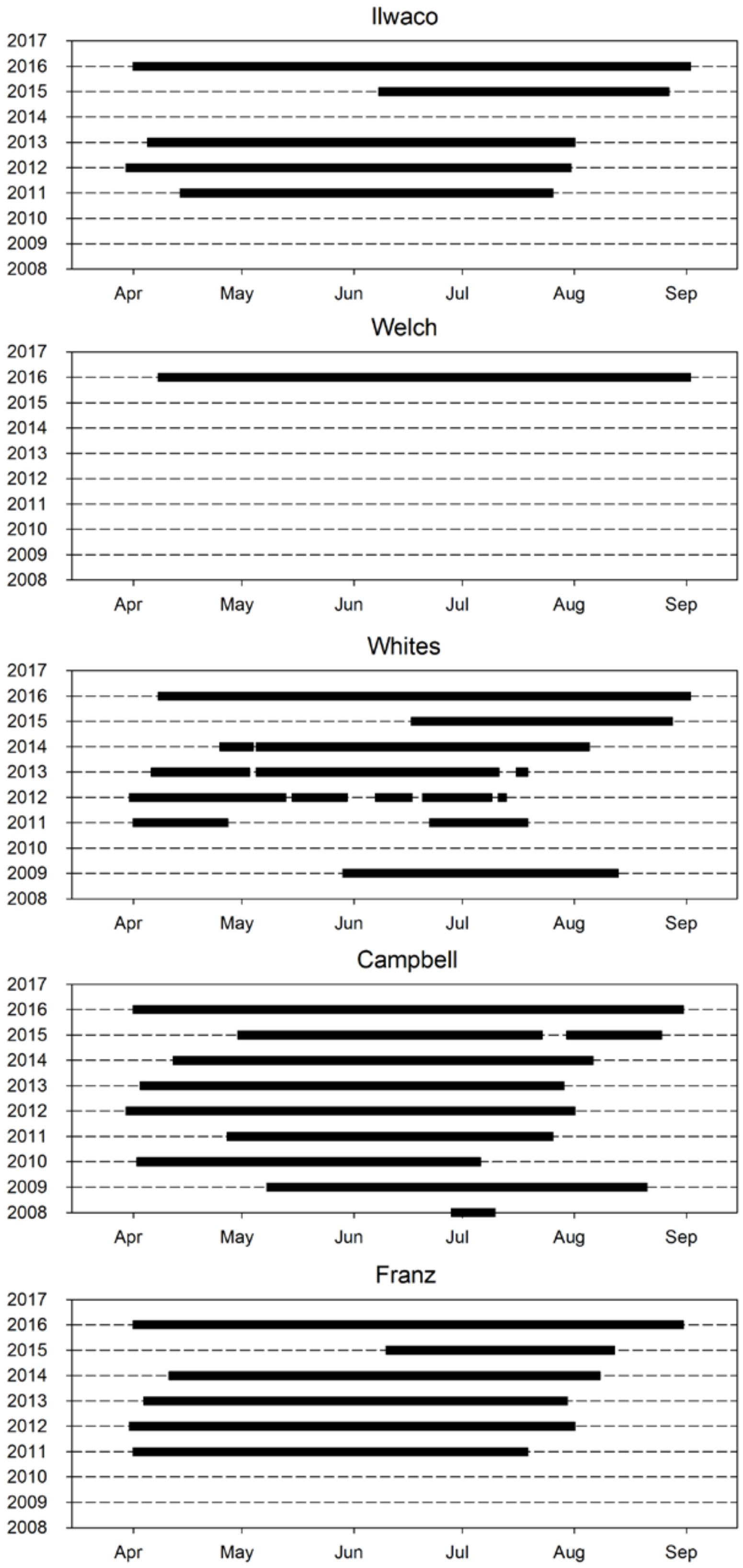
*Site names used in this report differ from official USGS site names to be consistent with site names used by other EMP partners.

Figure 5. Images showing deployment of water quality monitors (YSI sondes) at study sites.

The water quality monitors were YSI models 6600EDS and 6920V2, equipped with water temperature, specific conductance, pH, and dissolved oxygen probes. In addition, a 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 2016, the monitors were deployed from early April through early September. 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


Ilwaco

Welch

Whites

Campbell

Franz

Figure 6. Time periods corresponding to sensor deployments at trends sites (2011-2016).

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 was 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 2016, PNNL collected field data on vegetation and habitat conditions at the seven trends sites (Figure 1). Monitoring dates are provided in Table 8 and detailed maps of the 2016 monitoring sites are presented in Appendix A.

Table 8. Site location and sampling dates for each site sampled in 2016. 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	8/7/16
Secret River (low marsh)	SRM-L	37	Trend	8/6/16
Secret River (high marsh)	SRM-H	37	Trend	8/8/16
Welch Island 2	WI2	53	Trend	8/5/16
Whites Island	WHC	72	Trend	8/4/16
Cunningham Lake	CLM	145	Trend	8/3/16
Campbell Slough	CS1	149	Trend	8/2/16
Franz Lake	FLM	221	Trend	8/1/16

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 2016, we focused our efforts 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. 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 trend 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 and Secret River sites in 2012. Occasionally sensor failure or loss occurred; however, the sensors have been downloaded and redeployed every year since the initial deployment for 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

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). Six new sets of stakes were deployed in 2015; five were at a new elevation within the site and one at Secret River was installed to replace a set that had become unstable. All previously installed sediment accretion stakes at the trends sites were measured in 2016. 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).

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 sensor malfunctioned in April 2015 and was not redeployed at the site in 2015. Therefore salinity data were not collected in 2016 (Method ID 816).

2.3.2.4 *Vegetation Species Assemblage*

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

Along each transect, vegetative percent cover was evaluated at 2 – 10 m intervals. This interval and the transect lengths were based on the marsh size and/or the homogeneity of vegetation. At each interval on the transect tape, a 1 m^2 quadrat was placed on the substrate and percent cover was estimated by observers in 5% increments. If two observers were collecting data, they worked together initially to ensure their observations were “calibrated.” Species were recorded by four letter codes (1st two letters of 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 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 *Vegetation Community Mapping*

Using Trimble GeoXT and GeoXH handheld global positioning system (GPS) units, a representative portion of each site (using reasonable natural boundaries) was mapped and major vegetation communities were delineated within the site. Additionally, features of importance to the field survey (e.g., transect start/end points, depth sensor location, and photo points) were also mapped. All data were input to a GIS (ArcGIS 10; ESRI, Redlands CA), and maps of each site showing major communities and features were created (Appendix A). Four trends sites (Ilwaco Slough, Cunningham Lake, Campbell Slough, and Franz Lake) and portions of two additional sites (Welch and Whites Islands) where changes were observed were re-mapped in 2015.

2.3.2.6 *Elevation*

In previous years, elevation was measured at all trends sites, corresponding to each of the following metrics: vegetation quadrats, the water level sensor, sediment accretion stakes, vegetation community boundaries, and in the channels. In 2015, elevation was re-measured at Welch Island, Cunningham Lake, Campbell Slough, and Franz Lake. Elevations from previous years were used at Ilwaco Slough, Secret River, and Whites Island. Elevation was surveyed using a Trimble real time kinematic (RTK) GPS with survey-grade accuracy and an auto-level. All surveying was referenced to the NAVD88 vertical datum; 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:

$$\mathrm{SEV} = \sum^{n} (h_{\mathrm{elev}})$$

$i=1$

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 an 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 Similarity Analysis at Trends Sites*

Similarity analyses, using the Bray-Curtis similarity coefficient (S') as a measure of distance between years (described in Clarke and Warwick 2001), were performed on percent cover data from the trends sites by using Primer™. Percent cover data were arc-sin, square-root transformed, but were not standardized, prior to analyses.

2.3.3.3 *Floristic Quality Assessment*

A Floristic Quality Assessment (FQA) was conducted for each trend site based on the methods described in Rocchio and Crawford (2013). The FQA provides a means of estimating the ecological quality of a site and is based on a coefficient of conservatism value (C), which has been previously developed for most native species found at the trend sites, and native species richness. The C values range from 0-10 and represent the collective opinion of botanical and ecological experts from the region regarding a native species relative conservatism. The C values are defined by Rocchio and Crawford (2013) as follows:

0 – 3	Species that readily occur and persist in ecosystems that have been affected by human stressors. They can be found in a wide range of ecosystem conditions where ecological processes, function, composition, and structure range from being intact to severely modified by human stressors. If non-native species are used in the calculation of FQA metrics then they are assigned a value of 0.
4 – 6	Species that readily occur and persist in ecosystems where ecological processes, functions, composition, and/or structure have been moderately degraded/modified by human stressors. These species are often matrix-forming or dominant species.
7 – 8	Species that are mostly restricted to intact ecosystems but can persist where ecological processes, functions, composition, and/or structure are slightly degraded/modified by human stressors. These species are good indicators of intact ecosystems.
9 – 10	Species that are almost always restricted to intact ecosystems where ecological processes, functions, composition, and structure have not been (or only minimally) degraded/modified by human stressors; excellent indicators of intact ecosystems.

The Floristic Quality Index (FQI) is calculated as follows:

$$\text{FQI} = \bar{C} * \sqrt{N}$$

where $\bar{C}$ = average C values and N = native species richness. The Adjusted FQI is similar but eliminates the sensitivity of the FQI to species richness and incorporates the effect of non-native species (Miller and Wardrop 2006) using the following equation:

$$\text{Adjusted FQI} = \left(\frac{\bar{C}}{10} * \frac{\sqrt{N}}{\sqrt{S}}\right) \times 100$$

where $\bar{C}$ = average C values, N = native species richness, and S = native + non-native species richness. The Western Washington Floristic Quality Assessment Index Calculator (June 2013 version; downloaded from http://www.dnr.wa.gov/NHP-FQA) was used to calculate the FQI and other associated vegetative assessment metrics.

2.3.3.4 *Vegetation Community Change Analysis*

A change analysis was conducted comparing the GPS mapping results from the earliest available mapping and the most recent mapping conducted at each trend site (except Secret River, which was only mapped once due to the large size of the site and little observed change). The comparison was conducted in ArcGIS (version 10.4; ESRI, Redlands CA) using the following years and sites:

Ilwaco Slough	2012, 2015
Welch Island	2012, 2015
Whites Island	2009, 2015
Cunningham Lake	2006, 2015
Campbell Slough	2005, 2015
Franz Lake	2008, 2012, 2015

Three years were compared at Franz Lake because of different mapping areas being covered in each of the three years and due to the extensive changes that occurred at the site over the time period. In GIS, comparable mapped areas were intersected and the areas for each community type were calculated. The area of change for each vegetation community was then calculated and summarized in a matrix and overlaid in maps.

2.4 Food Web

2.4.1 Primary Productivity

2.4.1.1 *Emergent Wetland Vegetation*

Field Methods

From summer 2011 to winter 2017 above ground biomass was sampled to estimate the primary productivity at the six trends sites. Samples were collected in the summer during July or August during peak biomass and again in 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. Starting in 2012, the 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. Beginning in 2015, each sample was sorted in the field to separate the primary strata species from other species and to distinguish live from dead. The biomass was placed in a uniquely numbered bag and held in a cooler until samples were transported to the laboratory. Dominant vegetation species were recorded in field notebooks 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.

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

		2011-12		2012-13			2013-14		2015-16		2016-17		
Site	**Strata**	**S**	**W**	**S**	**F**	**W**	**S**	**W**	**S**	**W**	**S**	**W**	**Total**
BBM	CALY	3	4	6		6	4	4			6	6	39
BBM	CALY/AGSP	4	3	4		4	6	6			6	6	39
BBM	SAV	4	4	6		6	6						26
SRM	HM			5		5	9	9			9	9	46
SRM	LM			5		5	9	9			9	9	46
SRM	SAV			6		6	6						18
WI2	HM			5		9	9	9			12	12	56
WI2	LM			4									4
WI2	SAV			4		4	6						14
WHC	CALY		1	3		3	3	3	3	3	3	3	25
WHC	PHAR/HM	6	4	5		5	6	6	9	9	9	9	68
WHC	SALA/LM	2	3	3		3	6	6	6	6	6	6	47
WHC	SAV	8	8	6		6	6						34
CLM	ELPA/SALA								6	6	6		18
CLM	PHAR								7	7	7		21
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	29
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	731

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. 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 four trends sites in the Lower Columbia River in 2016.

Site	Reach	Date	Nutrients	Zooplankton	Phytoplankton
ILWACO SLOUGH	A	3/29/16	X	X	X
	A	4/28/16	X	X	X
	A	6/13/16	X	X	X
	A	7/8/16	X	X	X
	A	8/9/16	X	X	X
WELCH ISLAND	B	4/7/16	X	X	X
	B	5/18/16	X	X	X
	B	6/9/16	X	X	X
	B	7/6/16	X	X	X
	B	8/1/16	X	X	X
	B	9/2/16	X	X	X
WHITES ISLAND	C	4/7/16	X	X	X
	C	5/9/16	X	X	X
	C	6/7/16	X	X	X
	C	7/6/16	X	X	X
	C	8/1/16	X	X	X
	C	9/2/16	X	X	X
CAMPBELL SLOUGH	F	3/31/16	X	X	X
	F	5/3/16	X	X	X
	F	6/6/16	X	X	X
	F	6/30/16	X	X	X
	F	8/3/16	X	X	X
	F	8/30/16	X	X	X
FRANZ LAKE SLOUGH	H	3/31/16	X	X	X
	H	5/3/16	X	X	X
	H	6/2/16	X	X	X
	H	6/30/16	X	X	X
	H	8/3/16	X	X	X
	H	8/30/16	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 ($\text{NMDS}_{\text{total}}$) and (ii) only major diatom taxa ($\text{NMDS}_{\text{diatom}}$). Abundances for 25 major phytoplankton taxa ($\text{NMDS}_{\text{total}}$) and 10 major diatom taxa ($\text{NMDS}_{\text{diatom}}$) 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.

Tidal sampling

In an effort to determine whether there are differences in phytoplankton biomass and nutrients during different stages of the tide, hourly tidal sampling was conducted at Campbell Slough (Reach F) and Whites Island (Reach C) over 10 h and 8 h periods, respectively in May and June 2016. Water samples for dissolved nutrients, chlorophyll *a*, and phytoplankton species composition and abundance were collected during daylight as described above for routine sampling at the shallow water trends sites.

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 other 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 2016

Table 12. Vegetation types collected for analysis of stable isotopes of carbon and nitrogen.

SITE	DATE	VEG TYPE 1	VEG TYPE 2	VEG TYPE 3	VEG TYPE 4	VEG TYPE 5
Ilwaco Slough	3/29/16	*Carex lyngbyei*	*Ulva sp.*	*misc. seaweed*		
Campbell Slough	3/31/16	*Phalaris arundinaceae*	*misc. veg. 1*			
Franz Lake Slough	3/31/16	*Phalaris arundinaceae*	*Polyganum amphibium*	*Eleocharis cf. palustris*		
Whites Island	4/7/16	*Phalaris arundinaceae*	*cf. Salix sp.*	*misc. veg. 1 (nearshore)*	*Sagittaria latifolia*	
Welch Island	4/7/16	*Carex lyngbyei*	*Lysichiton americanus*	*Oenanthe sarmentosa*	*Equisetum sp.*	*misc. veg. 1*
Ilwaco Slough	4/28/16	*Carex lyngbyei*	*Ulva sp.*	*Ulva sp.*		
Campbell Slough	5/3/16	*Phalaris arundinaceae*	*misc. veg. 1*	*misc. veg. 2*	*thistle*	
Franz Lake Slough	5/3/16	*Phalaris arundinaceae*	*Polyganum amphibium*	*Salix sp.*		
Campbell Slough	6/2/16	*Sagittaria latifolia*	*Phalaris arundinaceae*	*Carex lyngbyei*	*Eleocharis cf. palustris*	

Franz Lake Slough	6/2/16	*Phalaris arundinaceae*	*Polyganum amphibium*	*Salix sp.*		
Whites Island	6/7/16	*Carex lygbyei*	*Phalaris arundinaceae*	*Sagittaria latifolia*	*Eleocharis cf. palustris*	*cf. Polyganum amphibium*
Welch Island	6/9/16	*Carex lygbyei*	*Eleocharis cf. palustris*	*Equisetum sp.*	*misc. veg. 1*	*misc. veg. 2*
Ilwaco Slough	6/13/16	*Carex lygbyei*	*sea lettuce*			
Campbell Slough	6/30/16	*Sagittaria latifolia*	*Phalaris arundinaceae*	*Eleocharis cf. palustris*		
Franz Lake Slough	6/2/16	*Phalaris arundinaceae*	*Polyganum amphibium*	*Salix sp.*		
Whites Island	6/7/16	*Carex lygbyei*	*Phalaris arundinaceae*	*Sagittaria latifolia*	*Eleocharis cf. palustris*	*cf. Polyganum amphibium*
Welch Island	6/9/16	*Carex lygbyei*	*Eleocharis cf. palustris*	*Equisetum sp.*	*misc. veg. 1*	*misc. veg. 2*
Ilwaco Slough	6/13/16	*Carex lygbyei*	*sea lettuce*			
Campbell Slough	6/30/16	*Sagittaria latifolia*	*Phalaris arundinaceae*	*Eleocharis cf. palustris*		
Franz Lake Slough	6/30/16	*Sagittaria latifolia*	*Eleocharis cf. palustris*	*Polyganum amphibium*		
Whites Island	7/6/16	*Eleocharis cf. palustris*	*Sagittaria latifolia*	*Phalaris arundinaceae*	*Carex lyngbyei*	
Welch Island	7/6/16	*Phalaris arundinaceae*	*Equisetum sp.*	*yellow iris*	*misc. veg. 1*	*misc. veg. 2*
Ilwaco Slough	7/8/16	*Carex lygbyei*	*Ulva sp.*	*Ulva sp.*		

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 (Table 12). 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 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. 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). Two OW and two EV samples were collected at each site per month; although, occasionally one or three tows were performed in each habitat type depending on field conditions (Table 13). Samples were preserved in 10% formalin until delivered to the laboratory for processing. The monitoring protocol can be found on monitoringmethods.org (Method ID 1622).

Table 13. The number of invertebrate tow samples (OW and EV) collected at each site per sampling event, 2008-2013 and 2015-2016.

		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	0	0	0	0	0	0	0	0	0	0	0	0	3	0	0	6	6	0	15
	May	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	6	0	12
	June	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4	4
2009	May	0	0	0	3	0	0	4	0	4	0	0	0	5	0	0	0	4	0	20
	June	0	0	0	0	0	0	0	0	0	0	0	0	4	0	0	0	0	0	4
2010	April	0	0	0	0	4	4	4	4	0	0	0	0	4	0	0	0	0	0	20
	May	0	0	0	0	4	4	4	4	0	0	0	0	4	0	0	0	0	0	20
	June	0	0	0	0	4	4	4	4	0	0	0	0	4	0	0	0	0	0	20
	July	0	0	0	0	4	0	4	4	0	0	0	0	2	0	0	0	0	0	14
2011	April	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
	May	8	0	0	0	0	0	10	0	0	4	4	4	4	0	0	0	2	0	36
	June	4	0	0	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	8
2012	February	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
	March	0	0	2	0	0	0	2	0	0	0	0	0	0	3	0	0	0	0	7
	April	0	4	5	0	0	0	6	0	0	0	0	0	0	4	2	0	0	0	21
	May	0	1	4	0	0	0	4	0	0	0	0	0	4	4	4	0	0	0	21
	June	0	6	4	0	0	0	4	0	0	0	0	0	4	2	4	0	0	0	24
2013	March	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
	May	0	4	4	0	0	0	4	0	0	0	0	0	4	0	0	0	0	0	16
	June	0	4	4	0	0	0	3	0	0	0	0	0	4	0	0	0	0	0	15
	July	0	0	4	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	10
2015	April	5	0	0	0	0	0	0	0	0	0	0	0	6	0	0	0	6	0	17
	May	0	0	2	0	0	0	4	0	0	0	0	0	2	0	0	0	5	0	13
	June	0	0	6	0	0	0	4	0	0	0	0	0	0	0	0	0	0	0	10
2016	February	0	0	2	0	0	0	6	0	0	0	0	0	0	0	0	0	2	0	10
	March	0	0	0	0	0	0	2	0	0	0	0	0	0	0	0	0	0	0	2
	April	0	0	2	0	0	0	4	0	0	0	0	0	6	0	0	0	4	0	16
	May	0	0	4	0	0	0	4	0	0	0	0	0	4	0	0	0	0	0	12
	June	0	0	6	0	0	0	4	0	0	0	0	0	6	0	0	0	0	0	16
	July	0	0	4	0	0	0	6	0	0	0	0	0	0	0	0	0	0	0	10
	August	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
	September	0	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
Total Tow Samples		19	23	65	3	16	12	97	16	4	4	4	4	76	13	10	6	35	4	411

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. If fish were not processed immediately, they were kept on ice until later in the same day when stomachs were extracted. Whole stomach samples were preserved in 10% formalin until delivered to the laboratory for processing. The total number of diet samples collected at the EMP sites since 2008 is provided in Table 14.

2.5.2.2 *Laboratory Methods*

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

Table 14. The number of Chinook salmon diet samples collected at each site per sampling event, 2008-2013, 2015-2016.

		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	0	0	0	0	0	0	0	0	0	0	0	0	6	0	0	13	15	9	0	43
	May	0	0	0	0	0	0	0	0	0	0	0	0	19	0	0	0	7	0	0	26
	June	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	13	13
2009	May	0	0	0	9	0	0	10	0	6	0	0	0	10	0	0	0	8	0	0	43
	June	0	0	0	10	0	0	0	0	0	0	0	0	9	0	0	0	0	0	0	19
2010	April	0	0	0	0	10	19	16	6	0	0	0	0	12	0	0	0	0	0	0	63
	May	0	0	0	0	17	15	14	14	0	0	0	0	24	0	0	0	0	0	0	84
	June	0	0	0	0	9	8	18	11	0	0	0	0	18	0	0	0	0	0	0	64
	July	0	0	0	0	10	0	19	11	0	0	0	0	15	0	0	0	0	0	0	55
	August	0	0	0	0	8	0	13	0	0	0	0	0	0	0	0	0	0	0	0	21
2011	May	0	0	0	0	0	0	10	0	0	10	13	10	22	0	0	0	0	0	0	65
	June	0	0	0	0	0	0	25	0	0	0	0	0	0	0	0	0	0	0	0	25
	July	0	0	0	0	0	0	2	0	0	2	0	0	0	0	0	0	0	0	0	4
2012	February	0	15	16	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	31
	March	0	0	14	0	0	0	13	0	0	0	0	0	0	13	0	0	0	0	0	40
	April	0	15	14	0	0	0	10	0	0	0	0	0	0	7	15	0	0	0	0	61
	May	0	0	30	0	0	0	11	0	0	0	0	0	18	15	18	0	0	0	0	92
	June	0	14	15	0	0	0	15	0	0	0	0	0	15	15	36	0	0	0	0	110
2013	March	0	0	9	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	9
	May	0	12	30	0	0	0	15	0	0	0	0	0	34	0	0	0	0	0	0	91
	June	0	1	23	0	0	0	13	0	0	0	0	0	9	0	0	0	0	0	0	46
	July	0	2	25	0	0	0	0	0	0	0	0	0	1	0	0	0	0	0	0	28
2015	April	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	6
	May	0	0	15	0	0	0	15	0	0	0	0	0	15	0	0	0	4	0	0	49
	June	0	0	7	0	0	0	13	0	0	0	0	0	0	0	0	0	0	0	0	20
2016	April	0	0	13	0	0	0	13	0	0	0	0	7	0	0	0	0	12	0	0	45
	May	0	0	15	0	0	0	19	0	0	0	0	13	0	0	0	0	0	0	0	47
	July	0	0	3	0	0	0	8	0	0	0	0	0	0	0	0	0	0	0	0	11
Total Tow Samples		6	59	229	19	54	42	272	42	6	12	13	30	227	50	69	13	46	9	13	1211

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:

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

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

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

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

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

where j_m is the mass specific maintenance cost at 0' C (0.003), d is the temperature coefficient for biomass assimilation (0.068), T is the temperature at 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 2016, NOAA Fisheries monitored habitat use by juvenile Chinook salmon and other fishes at five trends sites, Franz Lake in Reach H (sampled in 2008 – 2015), Campbell Slough in Reach F (sampled from 2007-2015), Whites Island site in Reach C (sampled from 2009-2015), Welch Island in Reach B (sampled from 2012-2015), and Ilwaco Slough in Reach A (sampled from 2011-2015), in order to examine year-to-year trends in fish habitat use in the lower river. Coordinates of the sampling sites are shown in Table 15.

Fish were collected from January through September 2016 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. All captured fish were identified to the species level and counted. Salmonid species (up to 30 specimens) were measured (fork length in mm) and weighed (g) and checked for adipose fin clips and coded wire tags to distinguish between marked hatchery fish and unmarked (presumably wild) fish. 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 2016 are shown in Table 15. The monitoring protocol can be found on monitoringmethods.org (Method ID 826). In addition to our standard sampling at the trend sites, at Ilwaco Slough, Whites Island, and Campbell Slough, additional beach seine sets were taken in May and June to examine changes in the fish community over the tidal cycle (see Section 2.6.2.5).

Table 15. Location of EMP sampling sites in 2016 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	3	3	2	3	3	3	3	3	3	26
Welch Island (Reach B) 46.255011° N, 123.480398° W	NS[1]	NS[1]	3	2	1	3	3	3	3	18
White Island (Reach C) 46.159350° N, 123.340133° W	3	3	3	1	1	3	3	3	3	23
Campbell Slough (Reach F) 45.783867° N, 22.754850° W	3	3	3	3	1	3	3	1	1	21
Franz Lake (Reach H) 45.600583° N, 122.103067W	3	3	1	2	NS[3]	2	1[2]	3	3	18
Total	12	12	12	11	6	14	13	13	13	106

[1] Not sampled due to permit issues
[2] Not sampled due to high water temperatures
[3] Not fishable due to high water levels

When juvenile Chinook salmon were captured, up to 30 individuals were collected for necropsy at each field site during each sampling effort. Salmon fork length were measured (to the nearest mm) and weighed (to the nearest 0.1 g), then euthanized by anesthesia with a lethal dose of MS-222. For each juvenile Chinook salmon, the following samples were collected: stomach contents for taxonomic analysis of prey; whole bodies (minus stomach contents) for measurement of lipids and persistent organic pollutants (POPs), including polycyclic aromatic hydrocarbons (PAHs), dichlorodiphenyltrichloroethanes (DDTs), polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and various organochlorine pesticides; fin clips for genetic stock identification; otoliths for aging and growth rate determination; and, when sufficient fish were available, bile for measurement of metabolites of polycyclic aromatic hydrocarbons (PAHs) and stomach contents for measurement POPs, including PCBs, PBDEs, and DDTs as well as other and various organochlorine pesticides as well as PAHs.

Samples for chemical analyses were frozen and stored at -80°C until lab analyses were performed. 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.

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). Juvenile Chinook salmon stock composition was estimated by using a regional microsatellite DNA data set (Seeb et al. 2007) that includes baseline data for spawning populations from throughout the Columbia River basin (described in Teel et al. 2009). 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 Chinook; West Cascades fall Chinook; West Cascades Spring Chinook; Middle and Upper Columbia Spring Chinook; Spring Creek Group fall Chinook; Snake River Fall Chinook; Snake River Spring Chinook; Upper Columbia River Summer/Fall Chinook; and Upper Willamette River Spring Chinook (Seeb et al. 2007; Teel et al. 2009). West Cascades and Spring Creek Group Chinook are Lower Columbia River stocks. The monitoring protocol can be found on monitoringmethods.org (Method ID 948).

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.

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:

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

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

2.6.2.3 *Otoliths (Growth Rates)*

Otoliths were extracted from juvenile Chinook salmon collected at EMP status and trends sampling sites (including toxic contaminant sampling sites; Lower Columbia Estuary Partnership 2007), as well as Action Effectiveness Monitoring sites from May to June in 2005 and 2007-2012 (n = 28 sites). Otolith data collected from action effectiveness monitoring sites and the toxic contaminant study in addition to EMP status and trends sites to allow for the most comprehensive analysis possible. Otoliths from fish ranging in fork length from 37-111 mm (mean = 67 mm, SD = 13 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).

We used a generalized linear modeling (GLM) approach to investigate the extent to which variability in somatic growth rate (dependent variable) was explained by a suite of independent variables (Table 16). Preliminary analyses indicated a non-linear relationship between growth rate and Julian Day and therefore Julian Day^2 was also included in our analyses. In addition, fork length was included in our analyses so as to account for the linear relationship we observed between growth rate and fish size (Figure 7). Water temperature was not included in this analysis because of insufficient data (only 64 of the 106 sampling events). Because of the unbalanced design of this study we ran GLMs on each of five datasets that differed in terms of the number of variables and estimates of growth rate. The Baseline dataset included all of our estimates of somatic growth rate (n=615) and 6 variables; year, Julian day, Julian day^2, off-channel distance, river kilometer, and fork length; Table 16). Each of the remaining four datasets (Stock and Hatchery/unmarked, Prey, Predator and Conspecifics, and Toxins) included these six variables, 2-4 additional variables, and a subset of the 615 estimates of growth rate. For example, the Stock and Hatchery/unmarked dataset included the variables stock and hatchery/unmarked, and had 531 estimates of growth rate. Of the 84 estimates of growth rate that were excluded from the Stock and Hatchery/unmarked dataset, 77 were removed because a stock was not assigned (i.e., genetic assignment probability was < 0.8), six others were removed because a stock had a small sample size (n<10), and one more was removed because information was not recorded as to whether it was hatchery or unmarked fish.

Using the Baseline dataset, we ran all possible GLM model combinations. All model parameters were estimated by maximizing the likelihood function. To compare models we calculated four values for each model; Akaike's Information Criterion (AIC), delta AIC, relative likelihood, and AIC weight. Smaller AIC values indicate "better" models and when comparing two models we calculated the difference in AIC values (delta AIC) (Akaike 1973, Burnham and Anderson 2002). A delta AIC of less than 2 indicates little difference between competing models; a delta AIC of 2–10 indicates moderate support for a difference between the models, and a delta AIC of greater than 10 indicates strong support (Burnham and Anderson 2002). Relative likelihood represents the likelihood of a model given the data, whereas AIC weight is the discrete probability of each model (Burnham and Anderson 2002). The best model from the Baseline dataset was defined as having a delta AIC of 0.00, and this model was used as the starting model (referred to as baseline model) for the other four datasets. This allowed us to investigate if the addition of, for example, prey richness and density (in the Prey dataset), explained more variability in somatic growth relative to the baseline model.

Table 16. List of independent variables in each of 5 datasets that were analyzed using a generalized linear modeling approach: The Baseline dataset consisted of 615 otolith-derived estimates of somatic growth and 5 independent variables. The other 4 datasets (Stock and Hatchery/unmarked, Prey, Predator and Conspecifics, and Toxins) included the same 5 independent variables, an additional 2-4 variables, and a subset of the 615 estimates of somatic growth. Percent (%) events refers to the percentage of the 106 sampling events for which data of a given independent variable were collected.

Independent variables	% events	Baseline	Stock & Hatchery/unmarked	Prey	Predator & Conspecifics	Toxins
Collection time						
1. Year	100	X	X	X	X	X
2. Julian day	100	X	X	X	X	X
Collection site						
3. Off-channel distance	100	X	X	X	X	X
4. River kilometer	100	X	X	X	X	X
Fish & invertebrates						
5. Prey richness	75			X		
6. Prey density	75			X		
7. Predator density	79				X	
8. Chinook salmon density	79				X	
9. Salmonid density (excluding Chinook)	79				X	
10. Non-salmonid density	79				X	
Chinook salmon characteristics						
11. Stock	100		X			
12. Hatchery or unmarked	100		X			
13. Fork length	100	X	X	X	X	X
14. Organochlorine pesticides*	85					X
15. Industrial contaminants*	85					X
Number of estimates of somatic growth rate		615	531	472	489	365

* Composites of individuals were used, pooled by genetic stock group.

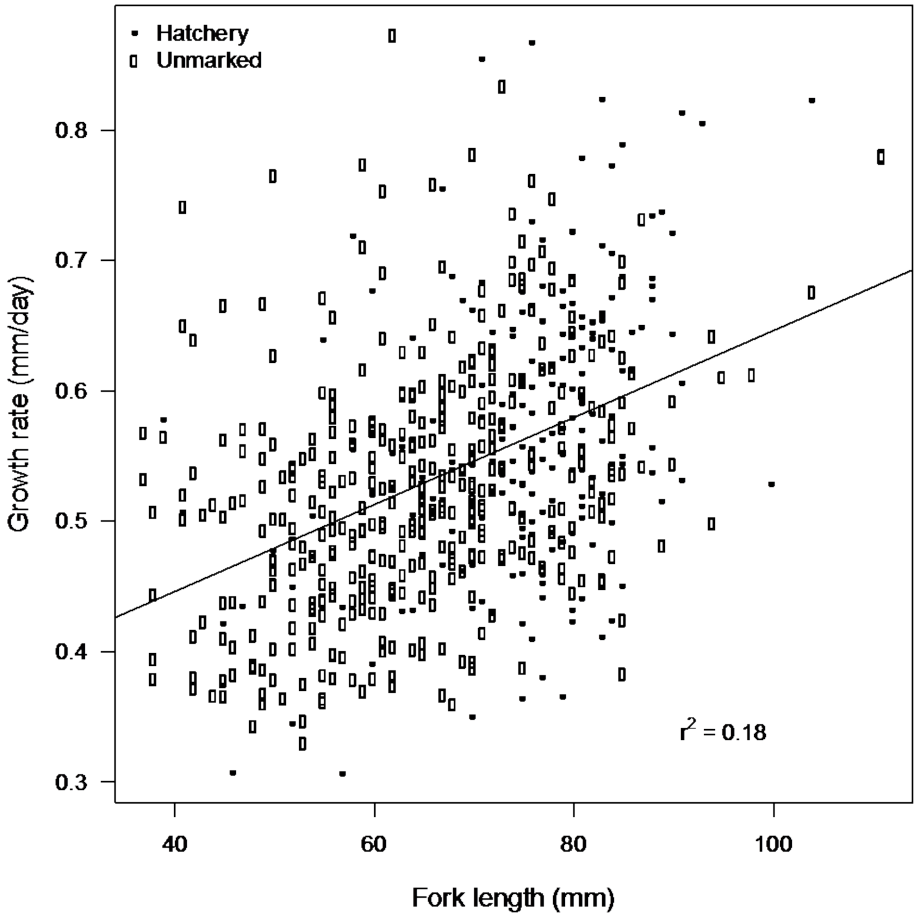


Figure 7. Significant linear regression ($p<0.01$) between otolith-derived estimates of growth rate (mm/day) and fork length (mm) at capture. Hatchery and unmarked fish are denoted by the black and white circles symbols, respectively.

2.6.2.4 *Chemical Contaminants in Chinook salmon*

Persistent Organic Pollutants in Bodies

Composite body samples (with stomach contents removed) were extracted with dichloromethane using an accelerated solvent extractor. The sample extracts were cleaned up using size exclusion liquid chromatography and analyzed by gas chromatography/mass spectrometry (GC/MS) for PCB congeners; PBDE congeners; organochlorine (OC) pesticides including DDTs, hexachlorocyclohexanes (HCHs), chlordanes, aldrin, dieldrin, mirex, and endosulfan I; and low (2-3 ring) and high (4-6 ring) molecular weight aromatic hydrocarbons as described by Sloan et al. (2006, 2014). Summed PCBs were determined by adding the concentrations of 45 congeners (PCBs 17, 18, 28, 31, 33, 44, 49, 52, 66, 70, 74, 82, 87, 95, 99, 101/90, 105, 110, 118, 128, 138/163/164, 149, 151, 153/132, 156, 158, 170/190, 171, 177, 180, 183, 187, 191, 194, 195, 199, 205, 206, 208, 209). Summed DDT levels (∑DDTs) were calculated by summing the concentrations of *p,p'*-DDT, *p,p'*-DDE, *p,p'*-DDD, *o,p'*-DDD, *o,p'*-DDE and *o,p'*-DDT. Summed chlordanes (∑CHLDs) were determined by adding the concentrations of heptachlor, heptachlor epoxide, g-chlordane, a-chlordane, oxychlordane, *cis*-nonachlor, *trans*-nonachlor and nonachlor III. Summed hexachlorocyclohexanes (∑HCHs) were calculated by adding the concentrations of a-HCH, b-HCH, and lindane (g-HCH). Summed low molecular weight aromatic hydrocarbons (∑LAHs) were determined by adding the concentrations of biphenyl, naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, 2,6-dimethylnapthalene, acenaphthene, fluorene, phenanthrene; 1-methylphenanthrene, and anthracene. Summed high molecular weight aromatic hydrocarbons (∑HAHs) were calculated by adding the concentrations of fluoranthene, pyrene, benz[*a*]anthracene, chrysene + triphenylene, benzo[*a*]pyrene, benzo[*e*]pyrene, perylene, dibenz[*a,h*]anthracene + dibenz[*a,c*]anthracene, benzo[*b*]fluoranthene, benzo[*j*]fluoranthene + benzo[*k*]fluoranthene, indeno[1,2,3-*cd*]pyrene, and benzo[*ghi*]perylene. Summed total aromatic hydrocarbons (∑TAHs) were calculated by adding the concentrations of ∑HAHs and ∑LAHs.

To adjust for the influence of lipid on toxicity, we normalized whole body contaminant concentrations for lipid, and relied primarily on lipid-normalized data to evaluate potential health effects of toxicants on juvenile salmon. Wet weight data are also presented to facilitate comparison with other studies, and to evaluate risks to predators who consume salmon that have accumulated toxicants. The monitoring protocol can be found on monitoringmethods.org (Method ID 950).

2.6.2.5 *Changes in fish community composition and salmon occurrence with the tidal cycle*

In 2016, we investigated the influence of the tidal cycle on fish assemblage, species composition, and abundance. Sampling was conducted at three of the trend sites (Ilwaco Slough, Whites Island, and Campbell Slough) which differ in tidal height and saltwater intrusion. The tidal height at Ilwaco Slough is approx. 2-2.5 m and it is highly influenced by saltwater intrusion, whereas the tidal height at Whites Island is approx. 2-2.5 m, but the site is not influenced by saltwater intrusion. Campbell Slough has a tidal height of approx.1 m, and it is not influenced by saltwater intrusion.

Tidal sampling was conducted in May and June 2016 during daylight flood tide in conjunction with regular annual EMP sampling. Fish were collected using a Puget Sound beach seine (PSBS; 37 x 2.4 m, 10 mm mesh size) deployed on foot or using a 5 m Boston Whaler, depending on the water level. The tidal sampling was conducted at a location near the center of the area along the channel edge typically sampled for the EMP. Sets were performed hourly from low tide (low tide but still ebbing or slack) to high tide (incoming flood tide), to a depth just before the high marsh was submerged (at high tide, the sampling areas at Ilwaco Slough and Whites Island are submerged and cannot be reliably sampled without compromising the efficiency of the sampling gear). For each sampling event, the time of sampling, weather, water depth, and water temperature were recorded. All captured fish were identified to species and counted, with the exception of some juvenile fish, which could only be categorized to the genus level. Catch per unit effort (CPUE) was calculated as described in Roegner et al. (2009), reported in number of fish per 1000 m^2.

Calculations

Hourly sampling points where the water was ebbing or slack, are categorized as time 0 hr; subsequent hourly sampling events where the site was flooding are categorized as time 1-4 hr. Catch per unit effort (CPUE) was calculated as described in Roegner et al. (2009), reported in number of fish per 1000 m^2.

2.6.2.6 *PIT Tag Array*

A passive integrated transponder (PIT) tag detection system was installed at Campbell Slough in June 2011, 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.

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 (i.e., daily volume fluxes) at Beaver Army Terminal (BAT; RM 53; Figure 8, Figure 9) and Bonneville Dam (Figure 10) were higher in 2016 compared to 2015, but similar to 2013 and 2014. Discharge was higher at Beaver Army Terminal (RM-53) compared to Bonneville Dam prior to the freshet in early May, similar to previous years. After the freshet, discharge at the two sites was similar, unlike in 2015, where discharge remained higher at RM-53 than at Bonneville Dam through the summer (Figure 9).

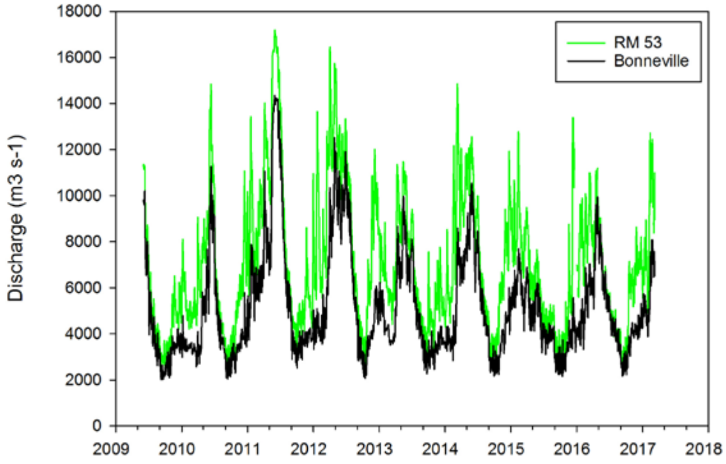


Figure 8. Daily river discharge (in m^3 s^{-1}) at Beaver Army Terminal (BAT, River Mile 53) and at Bonneville Dam from 2009 to 2016.

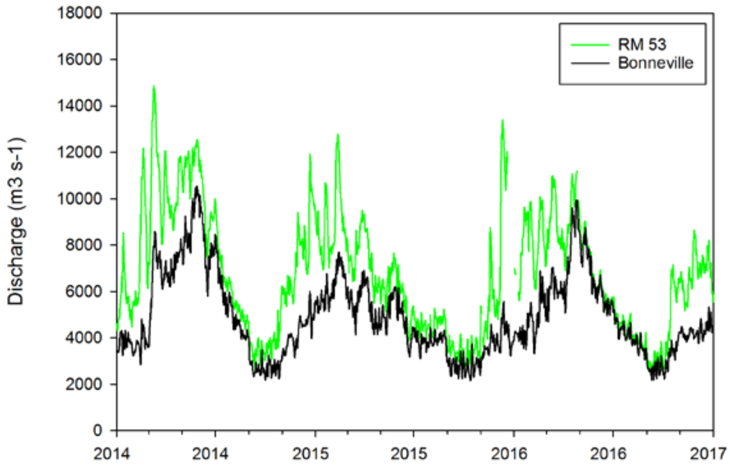


Figure 9. Daily average river discharge (m^3 s^{-1}) at Beaver Army Terminal (River Mile 53) and at Bonneville Dam in 2016.

To place flows into context, discharge volume fluxes from 2016 are shown in Figure 10. Prior to the freshet in late April/early May, discharge at Bonneville Dam was higher than the 10-year average. Following the freshet, discharge was lower than the long-term average until late summer/early autumn.

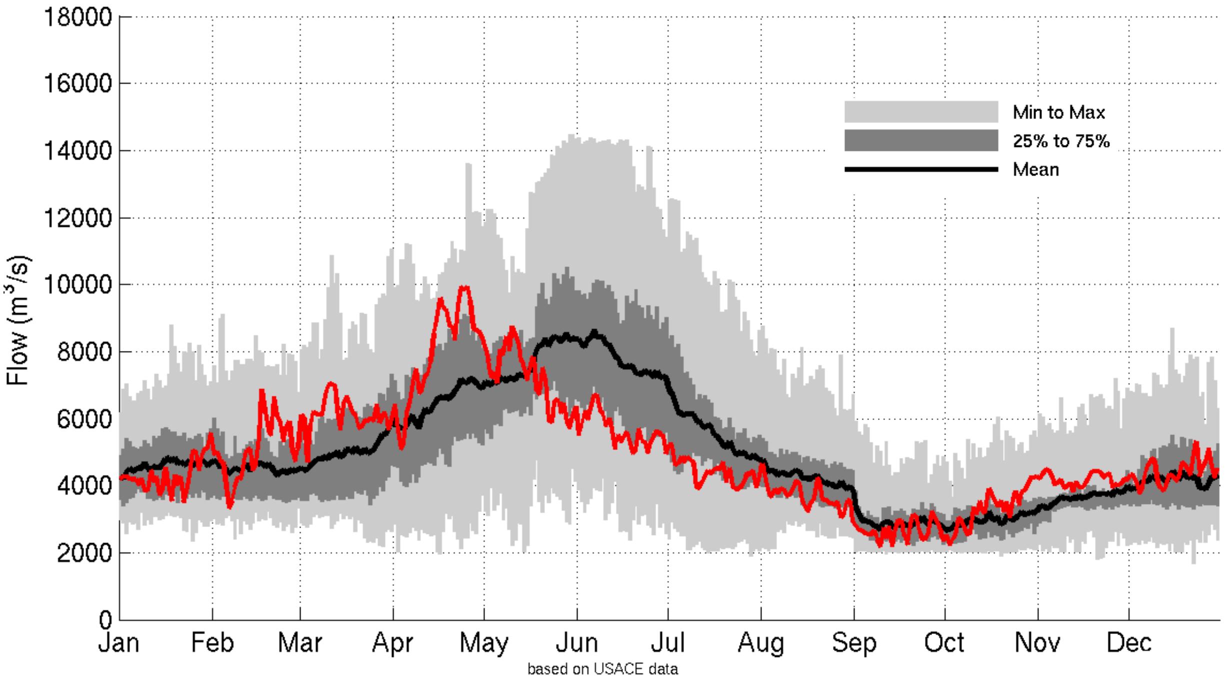
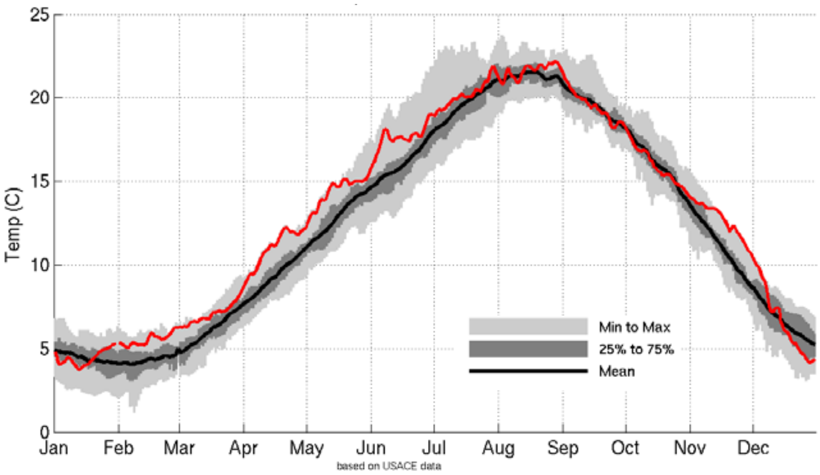


Figure 10. Columbia River flow at Bonneville Dam. Red line shows values from 2016; dark grey shows the 25-75th percentiles, and the light grey shows the minimum and maximum values for the 10-year data set.

3.1.1.2 *Water temperature at Beaver Army Terminal (RM-53) and at Camas (RM-122)*

Temperatures in the mainstem Columbia River in 2016 were warmer than the long-term average until the end of June (Figure 11). Values during late April through June reached similar levels to 2015, represented as the maximum values in the plot in Figure 11. Following the freshet, however, average daily temperatures were similar to the long-term average.



based on USACE data

Figure 11. Columbia River temperatures at Bonneville Dam in 2016 (red line) compared to the long-term average (black line), 25-75th percentiles (dark grey), and range (minimum and maximum; light grey).

There were fewer days where temperatures exceeded both 19°C and 21°C thresholds in 2016 compared to 2015 at both River Miles 53 and 122 (Figure 12). In 2016, there were just over 80 days where temperatures exceed 19°C, similar to 2009, 2013, and 2014. There were fewer days with very high temperatures (>21°C) in 2016 compared to both 2013 and 2015 (just under 40 days in 2016 compared with >50 days and >60 days in 2013 and 2015, respectively.

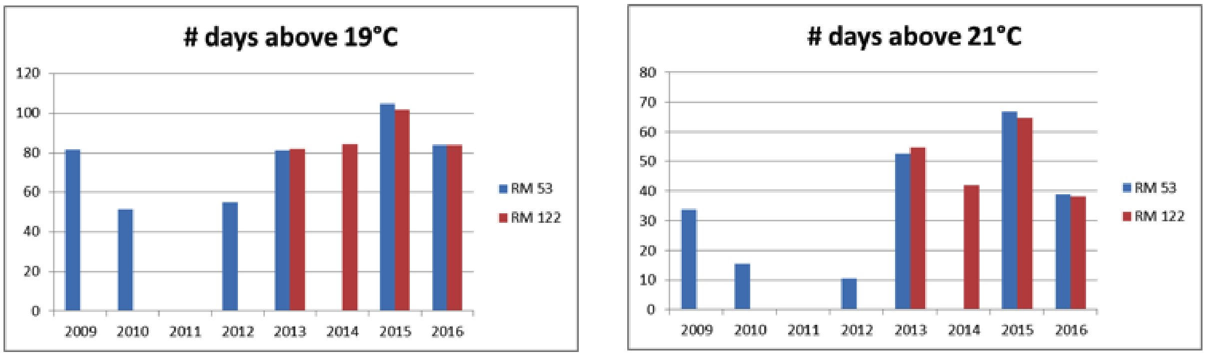


Figure 12. Number of days with temperatures above 19°C (left) or 21°C (right) at River Mile 53 and Camas, WA (River Mile 122).

3.1.1.3 *Water quality parameters in the mainstem*

Temperatures in the mainstem Columbia River peaked in late summer/early autumn when both turbidity and colored dissolved organic matter were low (Figure 13). Although nitrate reached a minimum in early September, it was never depleted. Oxygen concentrations were always sufficient to maintain percent saturation relative to the atmosphere greater than 90%, and chlorophyll concentrations did not exceed 10 μg L^{-1}. Thus, aside from high summer temperatures, water quality was good in the mainstem throughout 2016.

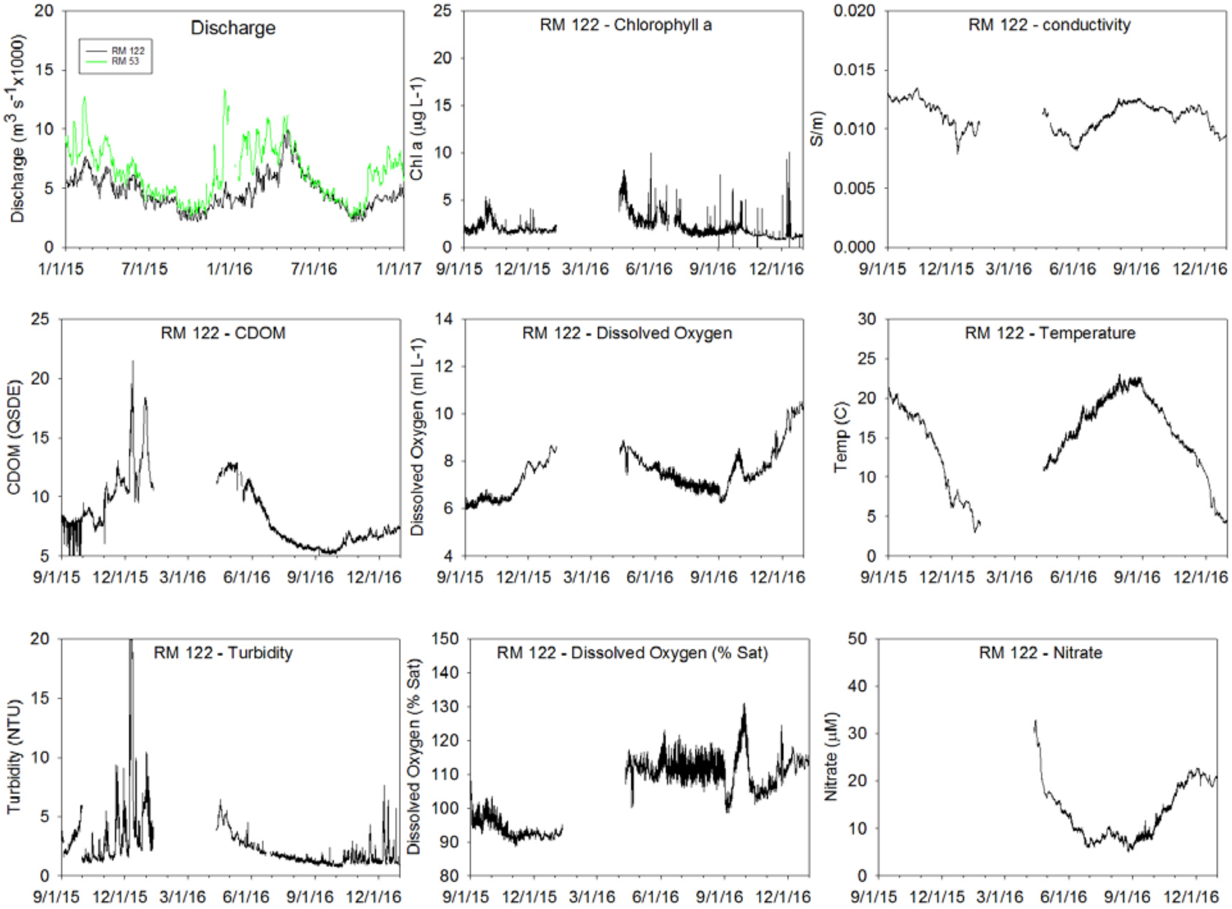


Figure 13. Water quality parameters in the mainstem Columbia River at Camas, WA (RM-122) derived from YSI sonde measurements.

3.1.2 Discrete samples from the mainsteam

3.1.2.1 *Dissolved nutrients*

Nutrient concentrations (nitrate, ammonium, and phosphate) were determined at the Port of Camas, adjacent to the LOBO mooring (Figure 14). Nitrate concentrations were highest in winter and declined following the peak in early April. In contrast, concentrations of ammonium and phosphate remained relatively constant throughout 2016. The phosphate data show a small peak at the end of May 2016, coinciding with the timing of peak river discharge during the freshet.

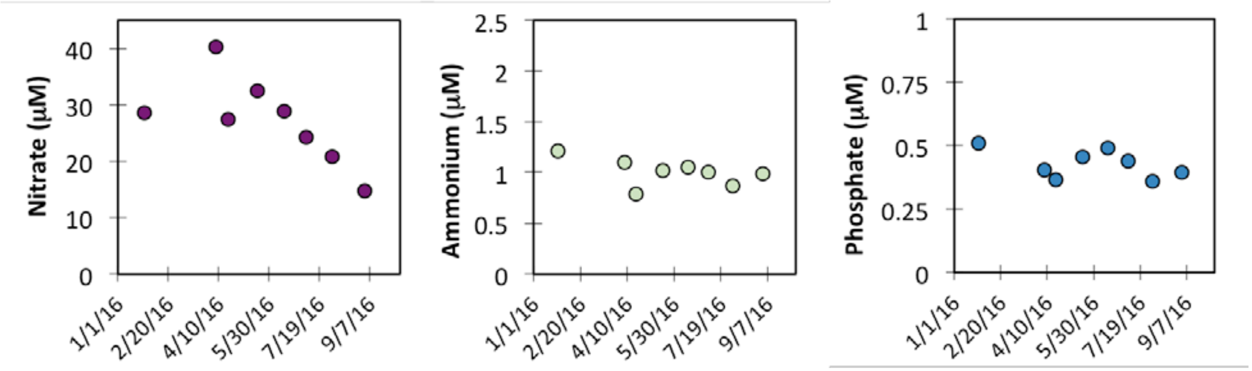


Figure 14. Discrete samples for dissolved nutrients (nitrate, ammonium, and phosphate) collected at the Camas site (River Mile 122) in 2016.

Similar to the Camas site (River Mile 122), nitrate concentrations at Beaver Army Terminal (RM-53) were higher during the winter months and declined after early April (Figure 15). Ammonium concentrations ranged from 0.8 to 2.9 μM at BAT, demonstrating higher variability compared to Camas. Similar to the nitrate concentrations—but different from observations at Camas—phosphate declined from winter to spring to summer.

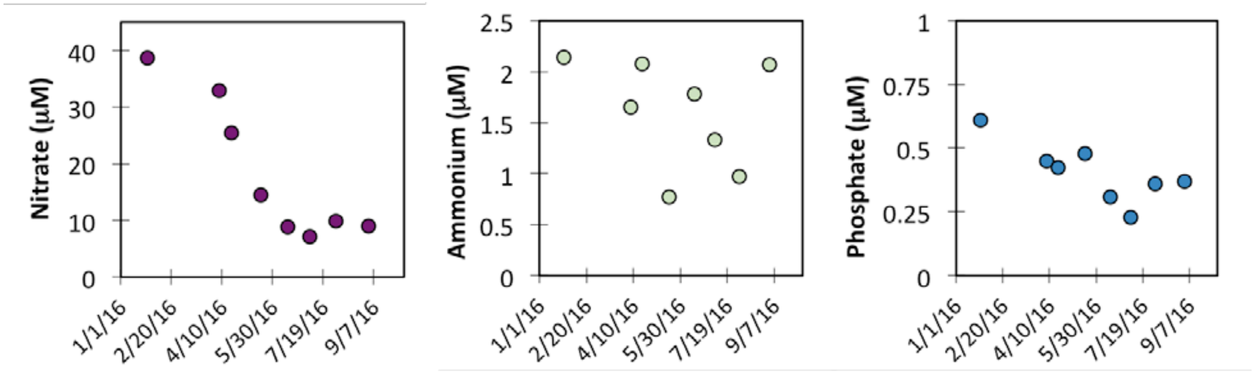


Figure 15. Discrete samples for dissolved nutrients (nitrate, ammonium, and phosphate) collected at Beaver Army Terminal (BAT) in 2016.

3.1.2.2 *Particulate nutrients (phosphorus, nitrogen)*

Based on analysis of discrete samples, there was higher phosphorus concentrations in particulate matter at Camas compared to Beaver Army Terminal, but higher particulate nitrogen at Beaver Army Terminal compared to Camas (Figure 16). This likely reflects the fact that the Willamette River tends to have higher dissolved nitrogen loads that the Columbia, which is manifest in higher nitrogen content in organic matter. There was not much seasonal variability in particulate phosphorus at Beaver Army Terminal or Camas. Particulate nitrogen, however, higher at BAT on two occasions, one corresponding approximately to the timing of the freshet, and the other in early September (Figure 16). There was little seasonal variability in particulate nitrogen at Camas.

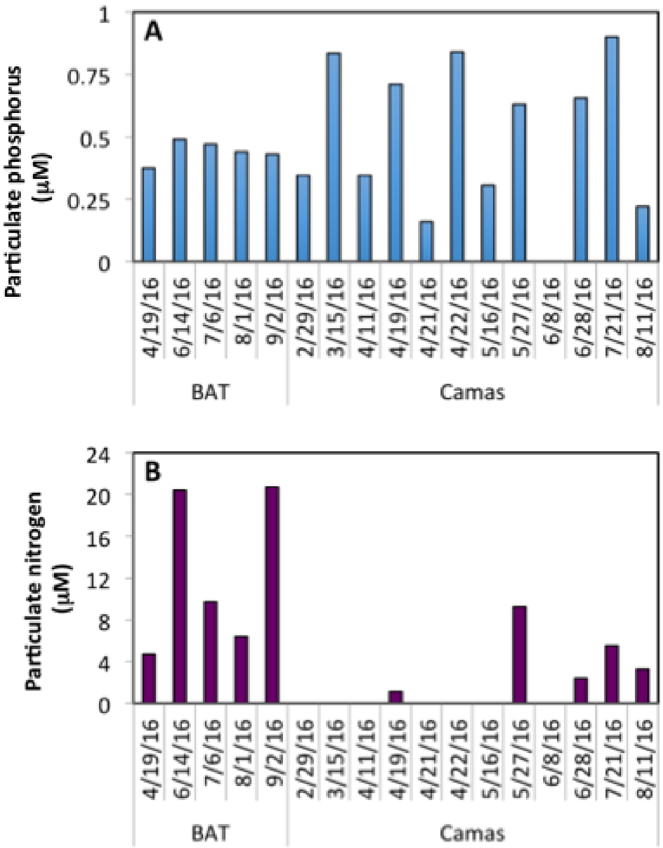


Figure 16. Concentrations of particulate phosphorus (A) and nitrogen (B) at Beaver Army Terminal (BAT) and at Camas in 2016.

3.1.2.3 *Phytoplankton in the mainstem*

Concentrations of the pigment, chlorophyll, which approximates total phytoplankton biomass, were similar at Camas and Beaver Army Terminal (Figure 17). There was a peak in chlorophyll in April and May, which was more distinct at Camas compared to Beaver Army Terminal. None of the samples had chlorophyll concentrations higher than 15 µg L^{-1}, the threshold set by the Environmental Protection Agency and Oregon Department of Environmental Quality.

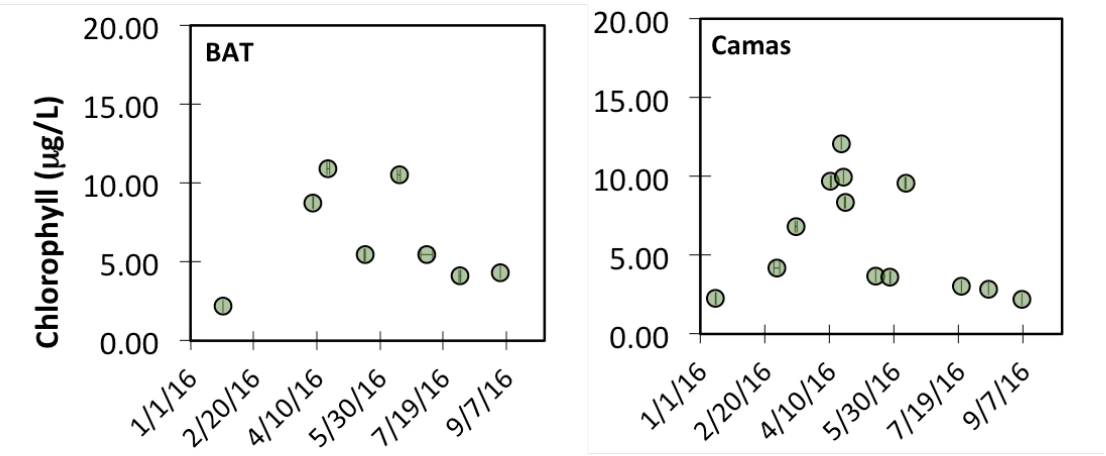


Figure 17. Chlorophyll concentrations (in µg L^{-1}) were very similar at the two mainstem sites (BAT = Beaver Army Terminal, River Mile 53) and Camas (Port of Camas, WA, River Mile 122). Error bars (± one standard deviation) are shown but fall within the markers, making them difficult to discern.

The phytoplankton species composition at Beaver Army Terminal (BAT) was dominated by diatoms (Class Bacillariophyceae) during April, May, and June 2016 (Figure 18). Chlorophytes (green algae) and cryptophytes also accounted for significant proportions of total phytoplankton at BAT. Data for Camas are not included here, but preliminary results show that the phytoplankton composition is similar to that at BAT (data analysis in progress).

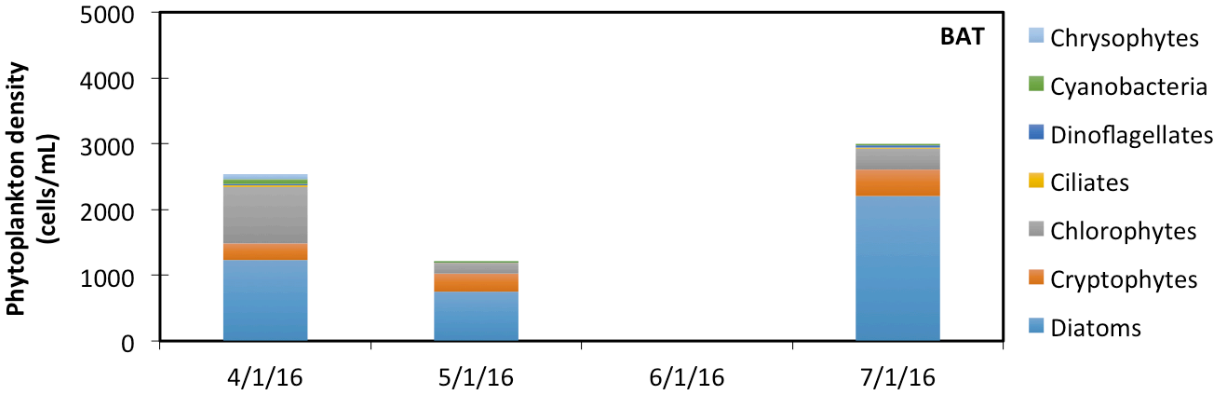


Figure 18. Composition of phytoplankton assemblages determined from grab samples at Beaver Army Terminal in April, May, and June 2016. Ciliates are included in the counts since they are in the same size range as phytoplankton; however, they are not phytoplankton but heterotrophic protists.

3.2 Abiotic Site Conditions

3.2.1 Continuous Water Quality

3.2.1.1 *Temperature at trends sites*

High temperature thresholds were exceeded earlier in the spring season at the shallow water trends sites in 2016 relative to the Columbia River mainstem. Temperatures at all four sites (Ilwaco Slough, Whites Island, Campbell Slough, and Franz Lake Slough) were higher than the 2000-2014 daily averages (Figure

19, Figure 20); the difference was smallest at Whites Island. Temperatures exceeded 20-21°C as early as April in Campbell Slough, which showed sporadic high values followed and regular high temperatures by June. Franz Lake Slough had high temperatures by June, while it was not until mid-July that temperatures exceeded 20°C at Ilwaco Slough and Whites Island.

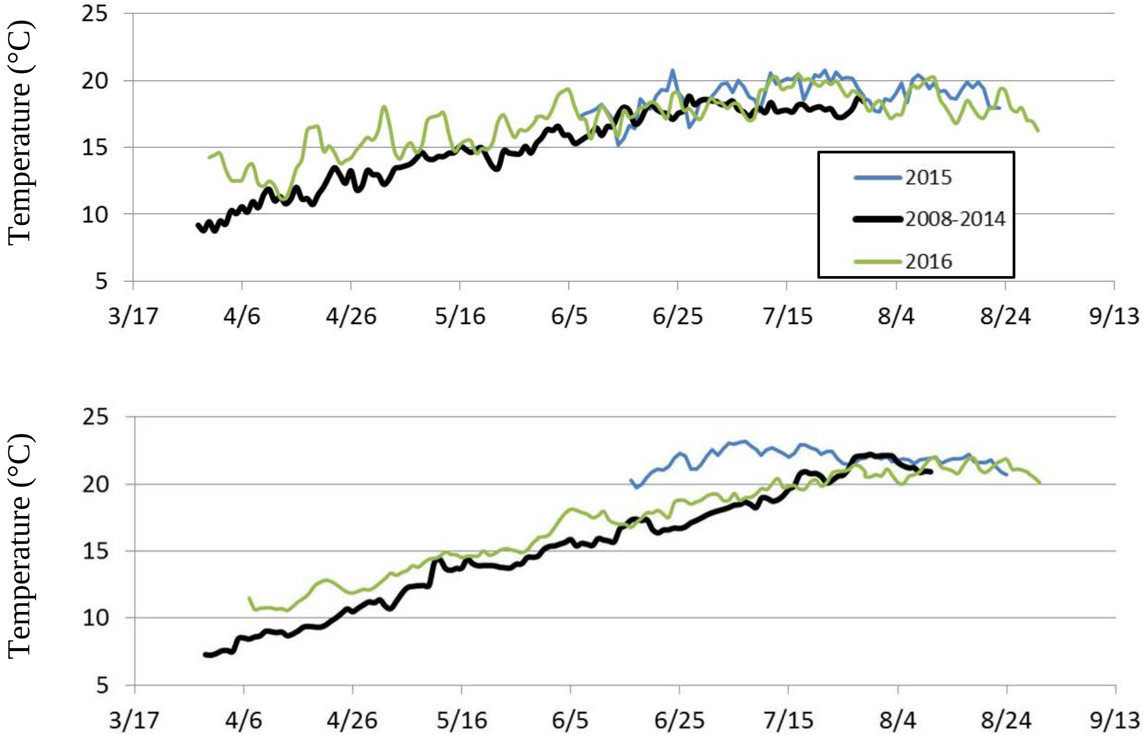


Figure 19. Daily average temperatures (°C) at Ilwaco Slough (upper) and at Whites Island (lower) determined for three time periods: 2012-2014 (black line), 2015 (blue line), and 2016 (green line).

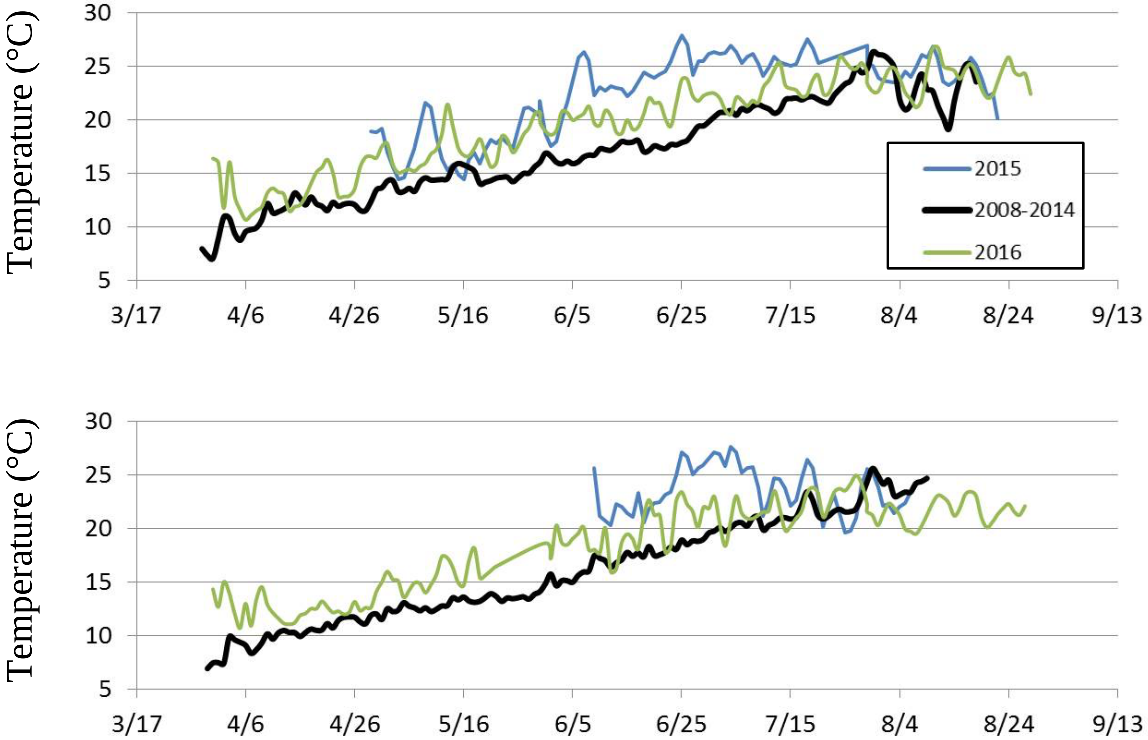


Figure 20. Daily average temperatures (°C) at Campbell Slough (upper) and at Franz Lake Slough (lower) determined for three time periods: 2012-2014 (black line), 2015 (blue line), and 2016 (green line).

3.2.1.2 *Ilwaco Slough*

Elevated instantaneous temperatures were observed at Ilwaco Slough (Reach A) as early as late March (Figure 21); however, the daily averages (reported above) were between 10-15°C. Tidal effects are clearly seen in the variation in sensor depth (sensor location was fixed) at Ilwaco Slough (Figure 21b). pH values declined to less than 6.5 by early May and never exceeded 8.5 pH units. The minimum values for instantaneous measurements of percent saturation of dissolved oxygen relative to the atmosphere fell below 50% in the spring, and declined to ~5-10% by June. 6.5 mg L^{-1} is the level set for high water quality in estuarine and coastal waters for the State of Oregon by the Environmental Protection Agency and Department of Environmental Quality (see document # 340-041-0001, “Water quality standards: beneficial uses, policies, and criteria for Oregon”). Roegner et al. (2011) use a cut-off of 6 mg L^{-1} to indicate the level below which mild biological stress would be felt by salmonids, which is consistent with the assessment by EPA/DEQ.

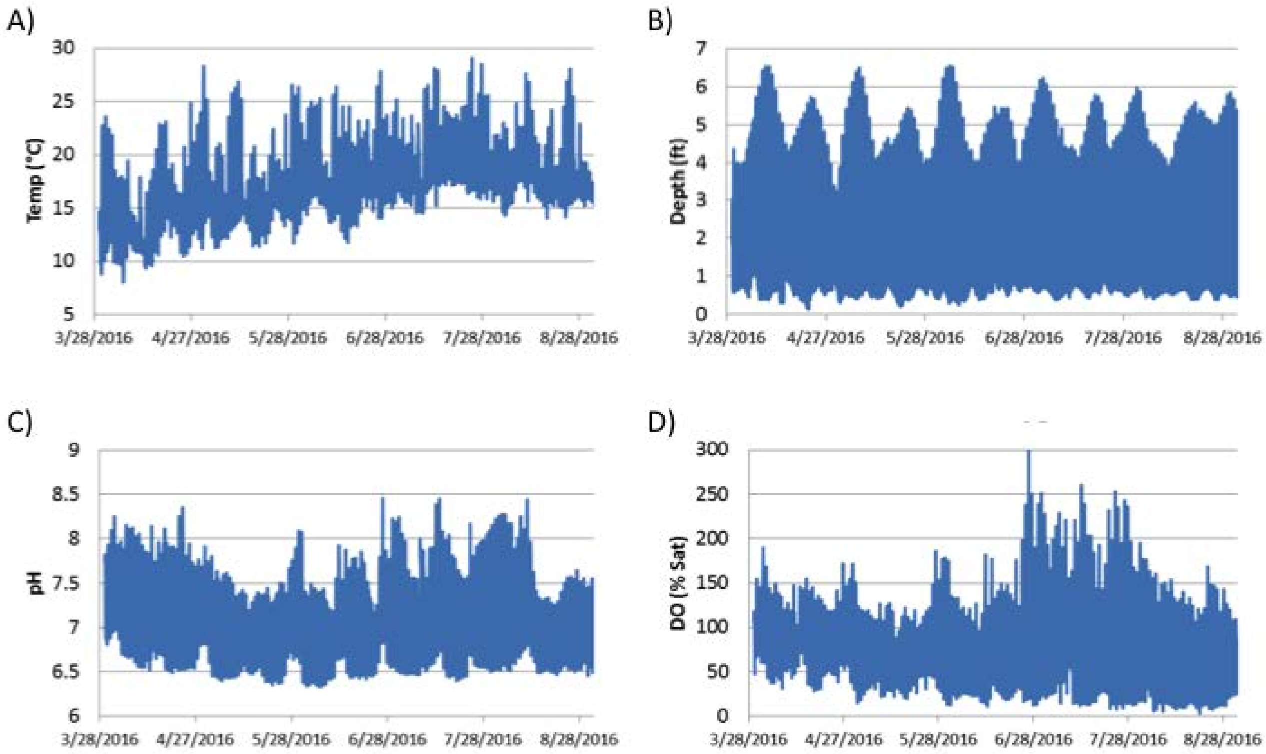


Figure 21. Hourly data from a YSI water quality sonde deployed at Ilwaco Slough (Reach A). (A) Temperature, (B) Depth of the water above the sonde, which was in a fixed position, (C) pH, (D) percent saturation of dissolved oxygen.

Unlike the other trend sites, Ilwaco Slough receives substantial inputs of seawater during tidal exchange. The seasonal change in salinity is shown in Figure 22. Salinity increased markedly between the spring (late March 2016) and late August, which typically occurs as river discharge decreases and the marine influence on the estuary increases (Chawla et al. 2008).

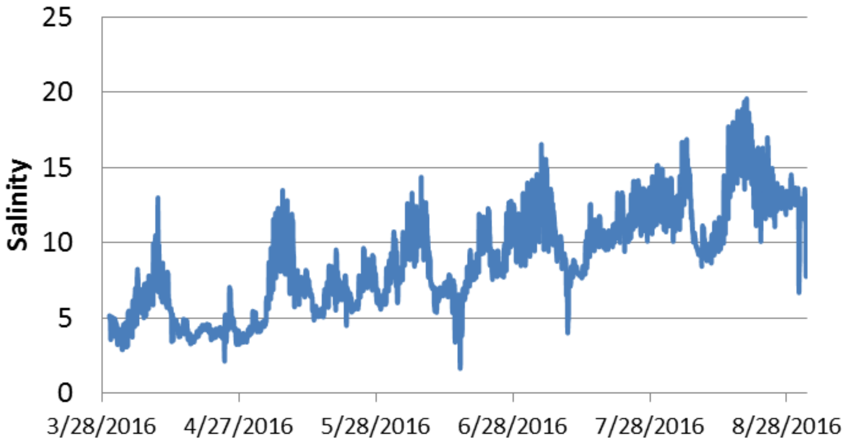


Figure 22. Hourly salinity data from a YSI water quality sonde deployed at Ilwaco Slough (Reach A). Salinity is determined from conductivity measurements.

3.2.1.3 *Welch Island*

Water temperature at Welch Island (Reach B) was higher than the 19°C threshold by mid-May (Figure 23). Temperatures frequently exceeded the 21°C threshold during late spring and during the summer. Two of the other parameters, pH and percent saturation of dissolved oxygen, also showed seasonal differences, with greater ranges in both occurring during the summer (Figure 23). The percent saturation of dissolved oxygen, which typically increases during the growing season when primary productivity is high, reached very high levels in July and August. Conductivity reached a minimum in early June and increased through the summer.

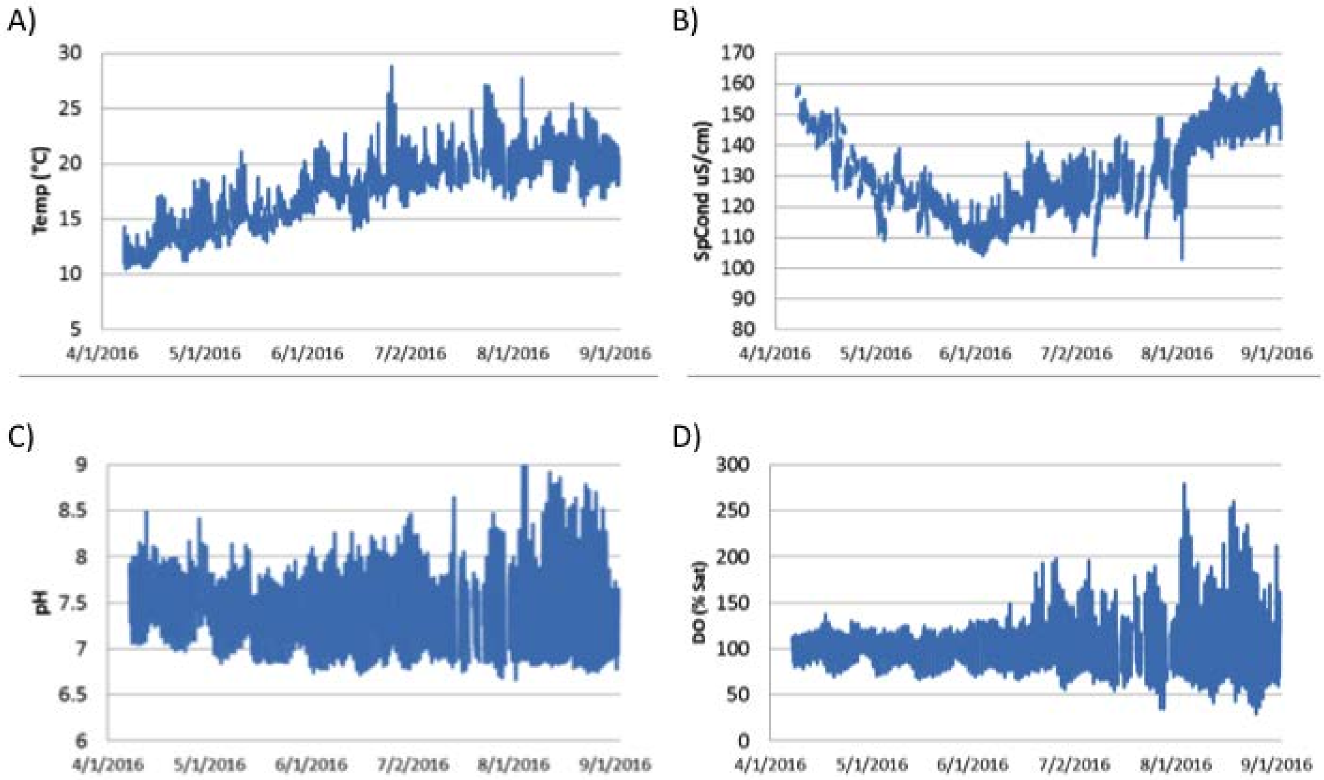


Figure 23. Hourly data from a YSI water quality sonde deployed at Welch Island (Reach B). (A) Temperature, (B) conductivity, (C) pH, (D) percent saturation of dissolved oxygen relative to the atmosphere.

3.2.1.4 *Whites Island*

Water temperatures at Whites Island exceeded 19°C by June (Figure 24). Dissolved oxygen levels never fell below 50% at Whites Island, with highs exceeding 150%. Water pH fell outside of water quality standard thresholds set by the Washington Department of Ecology (6.5–8.5) during brief events in July and August (Figure 24).

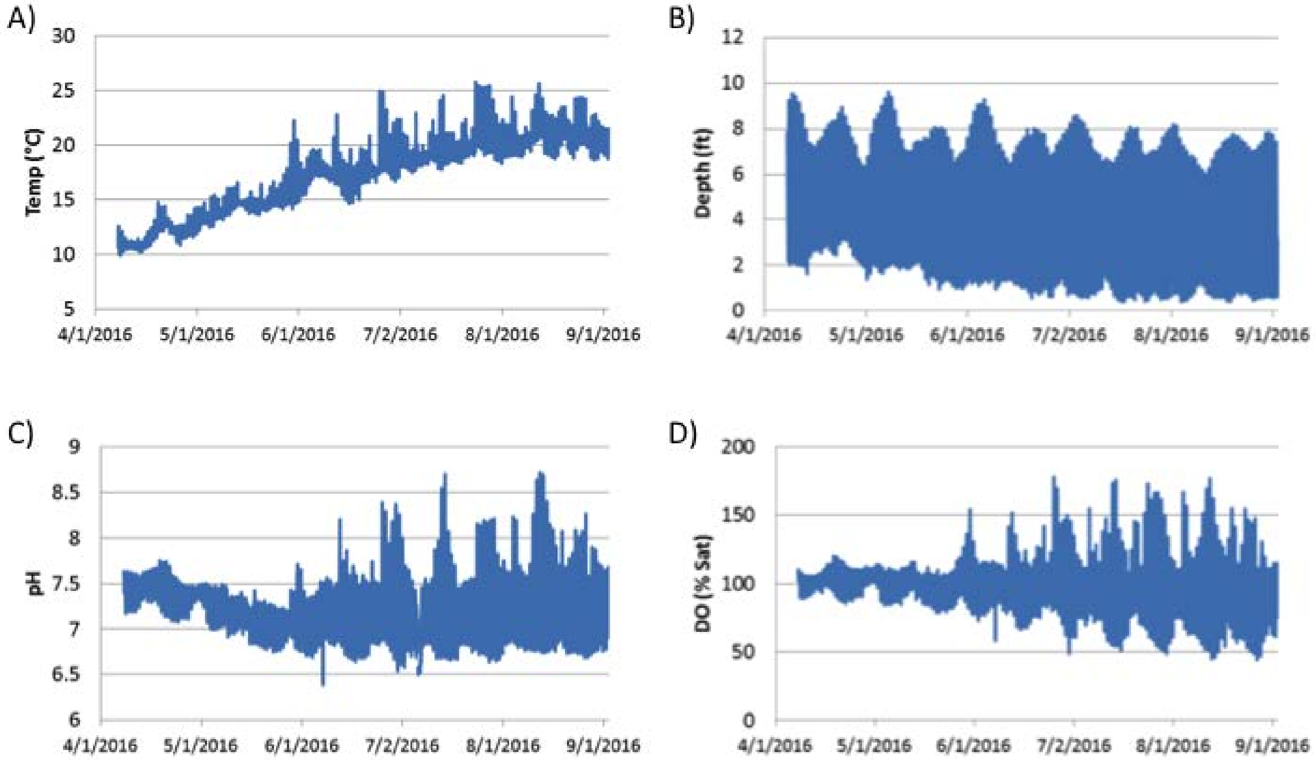


Figure 24. Hourly data from a YSI water quality sonde deployed at Whites Island (Reach C). (A) Temperature, (B) sensor depth, (C) pH, (D) percent saturation of dissolved oxygen.

3.2.1.5 *Campbell Slough*

Temperatures in Campbell Slough occasionally exceeded significant thresholds (19 °C, 21°C) by late April and frequently exceeded these thresholds by June (Figure 25). Episodes of low dissolved oxygen were noted by the end of June. Water pH exceeded thresholds set by the Washington Department of Ecology (6.5–8.5) during events in June and during all of July and August.

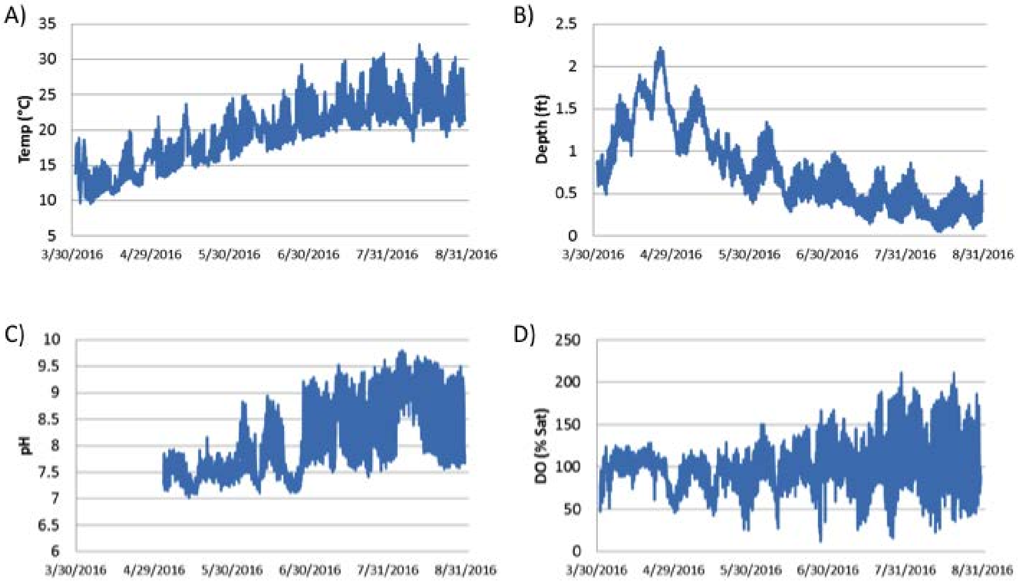


Figure 25. Hourly data from a YSI water quality sonde deployed at Campbell Slough (Reach F). (A) Temperature, (B) sensor depth, (C) pH, (D) percent saturation of dissolved oxygen.

The data show that there were three phytoplankton bloom events in Campbell Slough, one in June, one in July, and one in August which likely included cyanobacteria, based on the similar patterns in chlorophyll and phycocyanin (Figure 26).

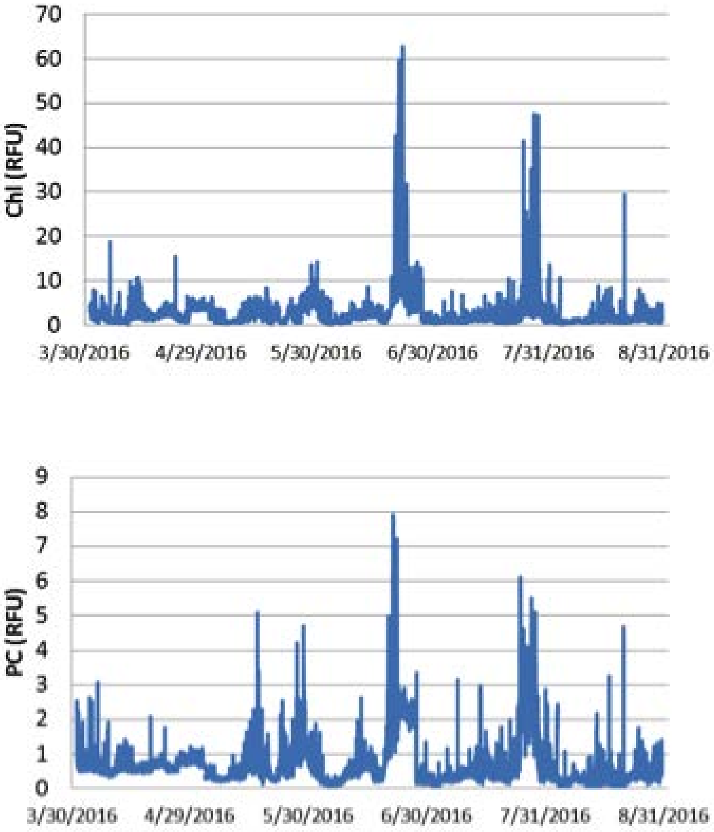


Figure 26. Pigment sensors deployed at Campbell Slough in 2016. Top: chlorophyll a; bottom: phycocyanin. Phycocyanin indicates the presence of cyanobacteria, while chlorophyll a is present in all phytoplankton. Peaks in phycocyanin were observed in May, June, July, and August 2016.

3.2.1.6 *Franz Lake Slough*

At Franz Lake Slough, there was high variability in both pH and dissolved oxygen (percent saturation relative to the atmosphere) by June when temperatures exceeded 20°C, which continued through the summer (Figure 27).

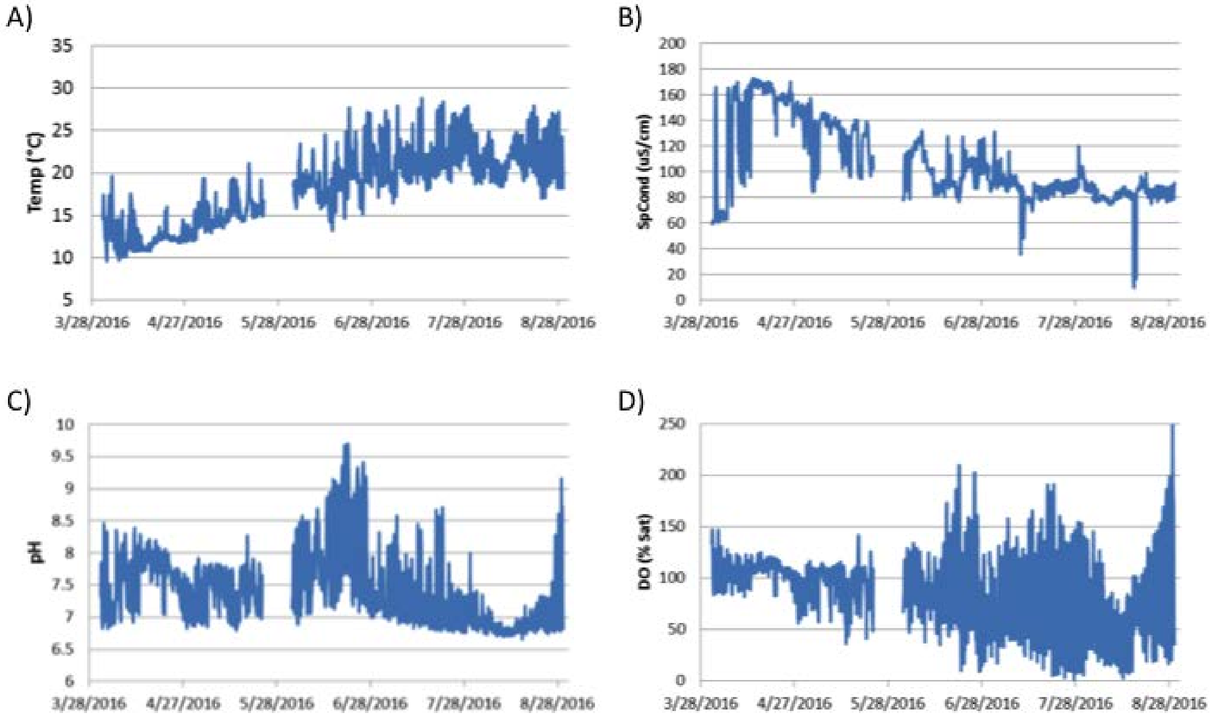


Figure 27. Hourly data from a YSI water quality sonde deployed at Franz Lake Slough (Reach H). (A) Temperature, (B) conductivity, (C) pH, (D) percent saturation of dissolved oxygen.

In contrast to Campbell Slough, there were only one or two phytoplankton bloom events in Franz Lake in 2016; there was a similar coincidence in the pattern of chlorophyll and phycocyanin (Figure 28), suggesting that cyanobacteria contributed to the blooms.

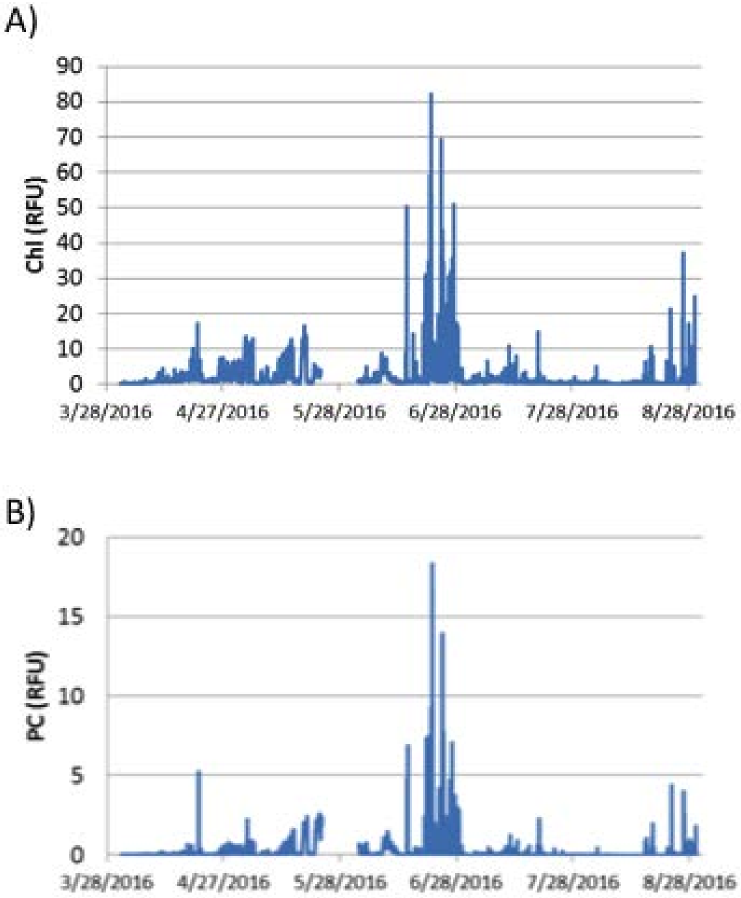


Figure 28. Pigment sensors deployed at Franz Lake Slough in 2016. Top: chlorophyll a (Chl a); bottom: phycocyanin (PC). RFU = relative fluorescence units. Phycocyanin indicates the presence of cyanobacteria, while chlorophyll a is present in all phytoplankton. Peaks in phycocyanin were observed in May, June, July, and August 2016.

3.2.2 Dissolved oxygen at trends sites

There is a diel fluctuation of dissolved oxygen due to the processes of photosynthesis and respiration. If water residence time is long or if respiration rates are high due to high inputs of organic matter, low dissolved oxygen can result. This poses a problem for any organism that undergoes aerobic respiration. A threshold of 6 mg L^{-1} has been set by the Washington Department of Ecology to indicate whether an aquatic habitat is suitable for salmonid habitat. In 2016, the number of hours in a given month with dissolved oxygen values below this threshold is indicative of potentially stressful conditions for aquatic organisms. For each of the sites with dissolved oxygen levels below this threshold, there was an increase from spring to summer (April to August; Figure 29). The highest number of hours in a month with low dissolved oxygen was observed at Franz Lake Slough in July and August; however, there were always hours in the month with low dissolved oxygen at Ilwaco Slough (Figure 30). Whites Island and Welch Island had very few hours with low dissolved oxygen, similar to the mainstem sites (data not shown).

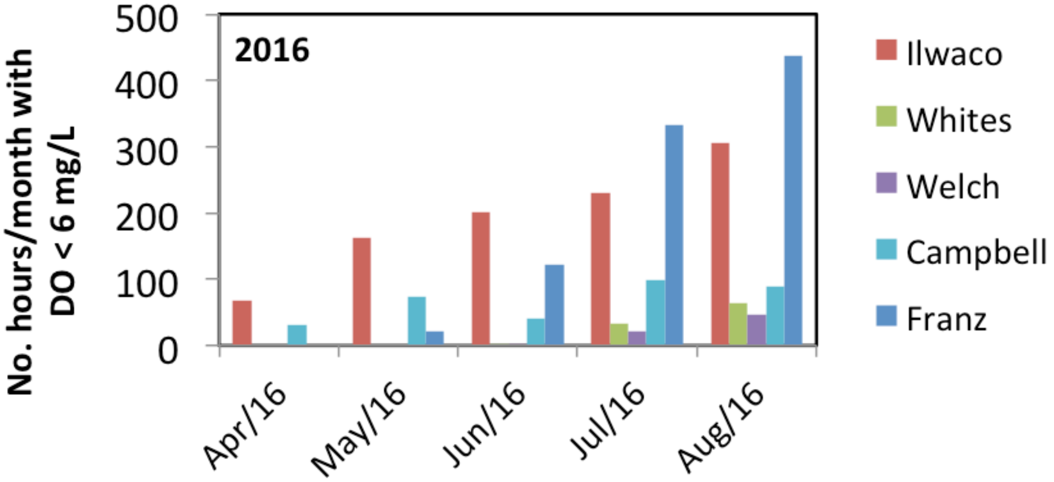
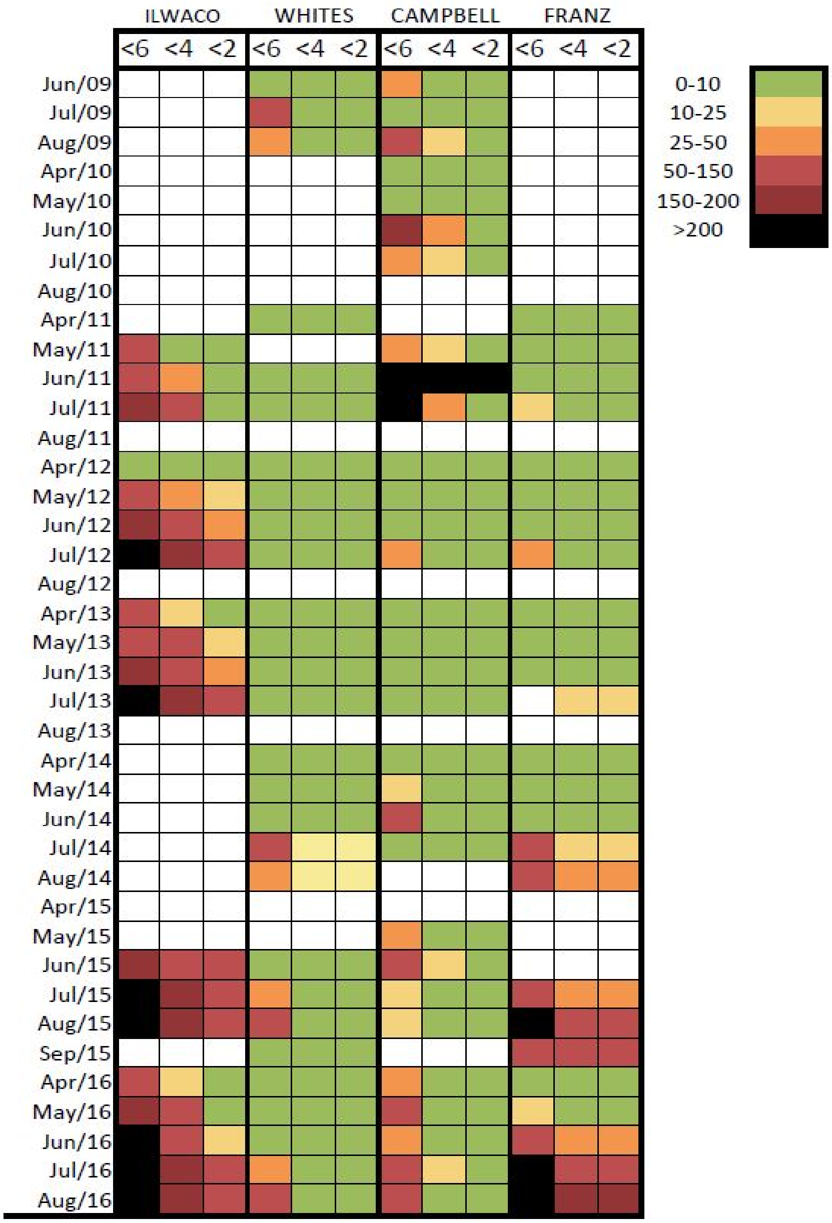


Figure 29. Number of hours per month with dissolved oxygen levels below a threshold of 6 mg L^{-1} at each of the trends sites in 2016.



	ILWACO			WHITES			CAMPBELL			FRANZ		
	<6	<4	<2	<6	<4	<2	<6	<4	<2	<6	<4	<2
Jun/09	⬜	⬜	⬜	🟩	🟩	🟩	🟧	🟩	🟩	⬜	⬜	⬜
Jul/09	⬜	⬜	⬜	🟥	🟩	🟩	🟩	🟩	🟩	⬜	⬜	⬜
Aug/09	⬜	⬜	⬜	🟧	🟩	🟩	🟥	🟨	🟩	⬜	⬜	⬜
Apr/10	⬜	⬜	⬜	⬜	⬜	⬜	🟩	🟩	🟩	⬜	⬜	⬜
May/10	⬜	⬜	⬜	⬜	⬜	⬜	🟩	🟩	🟩	⬜	⬜	⬜
Jun/10	⬜	⬜	⬜	⬜	⬜	⬜	🟫	🟧	🟩	⬜	⬜	⬜
Jul/10	⬜	⬜	⬜	⬜	⬜	⬜	🟧	🟨	🟩	⬜	⬜	⬜
Aug/10	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜
Apr/11	⬜	⬜	⬜	🟩	🟩	🟩	⬜	⬜	⬜	🟩	🟩	🟩
May/11	🟥	🟩	🟩	⬜	⬜	⬜	🟧	🟨	🟩	🟩	🟩	🟩
Jun/11	🟥	🟧	🟩	🟩	🟩	🟩	⬛	⬛	⬛	🟩	🟩	🟩
Jul/11	🟫	🟥	🟩	🟩	🟩	🟩	⬛	🟧	🟩	🟨	🟩	🟩
Aug/11	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜
Apr/12	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
May/12	🟥	🟧	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
Jun/12	🟫	🟥	🟧	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
Jul/12	⬛	🟫	🟥	🟩	🟩	🟩	🟧	🟩	🟩	🟧	🟩	🟩
Aug/12	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜
Apr/13	🟥	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
May/13	🟥	🟥	🟨	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
Jun/13	🟫	🟥	🟧	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
Jul/13	⬛	🟫	🟥	🟩	🟩	🟩	🟩	🟩	🟩	⬜	🟨	🟨
Aug/13	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜
Apr/14	⬜	⬜	⬜	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩	🟩
May/14	⬜	⬜	⬜	🟩	🟩	🟩	🟨	🟩	🟩	🟩	🟩	🟩
Jun/14	⬜	⬜	⬜	🟩	🟩	🟩	🟥	🟩	🟩	🟩	🟩	🟩
Jul/14	⬜	⬜	⬜	🟥	🟨	🟨	🟩	🟩	🟩	🟥	🟨	🟨
Aug/14	⬜	⬜	⬜	🟧	🟨	🟨	⬜	⬜	⬜	🟥	🟧	🟧
Apr/15	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜	⬜
May/15	⬜	⬜	⬜	⬜	⬜	⬜	🟧	🟩	🟩	⬜	⬜	⬜
Jun/15	🟫	🟥	🟥	🟩	🟩	🟩	🟥	🟨	🟩	⬜	⬜	⬜
Jul/15	⬛	🟫	🟥	🟧	🟩	🟩	🟨	🟩	🟩	🟥	🟧	🟧
Aug/15	⬛	🟫	🟥	🟥	🟩	🟩	🟨	🟩	🟩	⬛	🟥	🟥
Sep/15	⬜	⬜	⬜	🟩	🟩	🟩	⬜	⬜	⬜	🟥	🟥	🟥
Apr/16	🟥	🟨	🟩	🟩	🟩	🟩	🟧	🟩	🟩	🟩	🟩	🟩
May/16	🟫	🟥	🟩	🟩	🟩	🟩	🟥	🟩	🟩	🟨	🟩	🟩
Jun/16	⬛	🟥	🟨	🟩	🟩	🟩	🟧	🟩	🟩	🟥	🟧	🟧
Jul/16	⬛	🟫	🟥	🟧	🟩	🟩	🟥	🟨	🟩	⬛	🟥	🟥
Aug/16	⬛	🟫	🟥	🟥	🟩	🟩	🟥	🟩	🟩	⬛	🟫	🟫

0-10	🟩
10-25	🟨
25-50	🟧
50-150	🟥
150-200	🟫
>200	⬛

Figure 30. Number of hours per month below three thresholds for dissolved oxygen (<6 mg L^{-1}, <4 mg L^{-1}, and <2 mg L^{-1}) at four trends sites (Ilwaco Slough, Whites Island, Campbell Slough, and Franz Lake Slough). White color indicates that no data are available; legend shows values associated with colors in the table.

3.2.3 Nutrients

3.2.3.1 *Dissolved inorganic nutrients (nitrate, phosphate)*

Nitrate concentrations were highest in the early spring, and declined at all sites thereafter (Figure 31), which is consistent with patterns observed in the mainstem. However, nitrate concentrations were depleted at Franz Lake Slough by mid-late June. With the exception of Welch Island, nitrate concentrations were low throughout the system by June. In contrast to nitrate, phosphate concentrations were highest throughout the season at Franz Lake Slough and at Ilwaco Slough. Ammonium concentrations were notably higher at Ilwaco Slough compared to the other trends sites, which is indicative of stronger organic matter remineralization at that site.

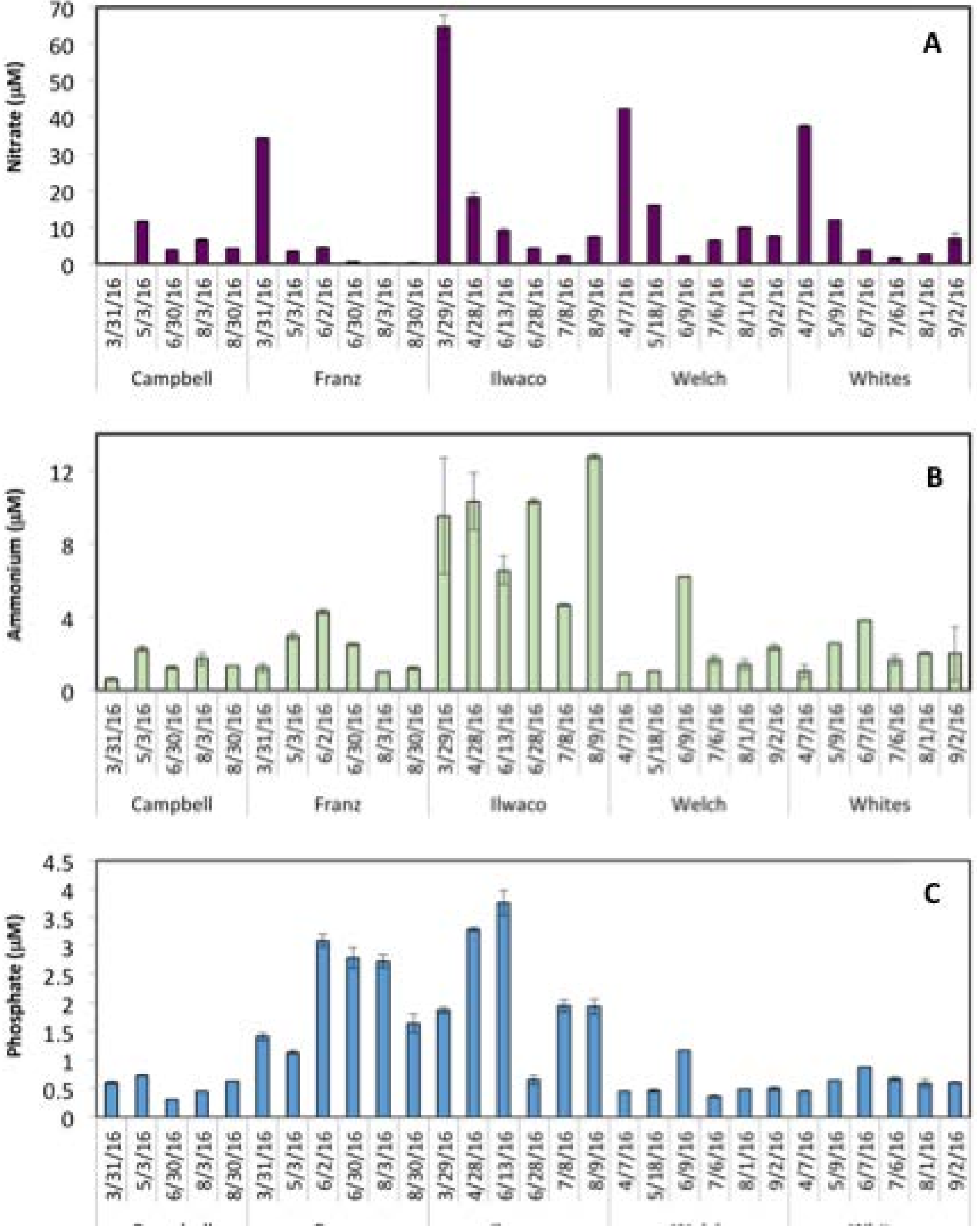


Figure 31. Time series of nutrient concentrations (nitrate, ammonium, phosphate) at each of the trends sites (Campbell Slough, Franz Lake Slough, Ilwaco Slough, Welch Island, and Whites Island) in 2016.

3.2.3.2 *Particulate nitrogen and phosphorus*

Phosphorus associated with particulate matter was highest at Franz Lake Slough and Ilwaco Slough, similar to the dissolved phosphorus concentrations (Figure 32). Particulate phosphorus concentrations were similar at Welch Island, Whites Island, and Campbell Slough.

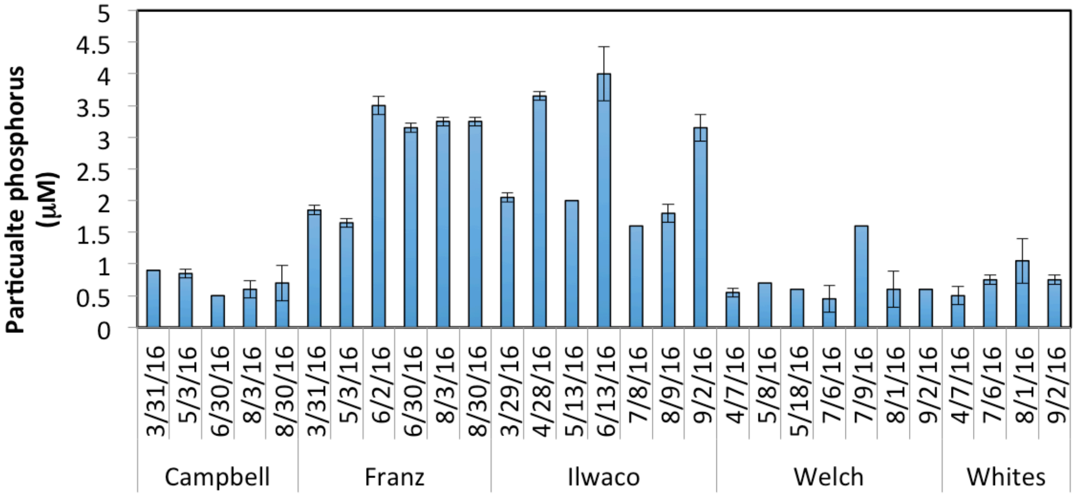


Figure 32. Particulate phosphorus concentrations at the trends sites (Campbell Slough, Franz Lake Slough, Ilwaco Slough, Welch Island, and Whites Island) in 2016.

Particulate nitrogen concentrations generally declined from spring to summer (Figure 33), with the highest value observed at Ilwaco Slough at the end of March. There was less variability in particulate nitrogen at Campbell Sough compared to the other sites, and unlike the other sites, concentrations at Franz Lake Slough did not decrease over the season, but varied from month to month.

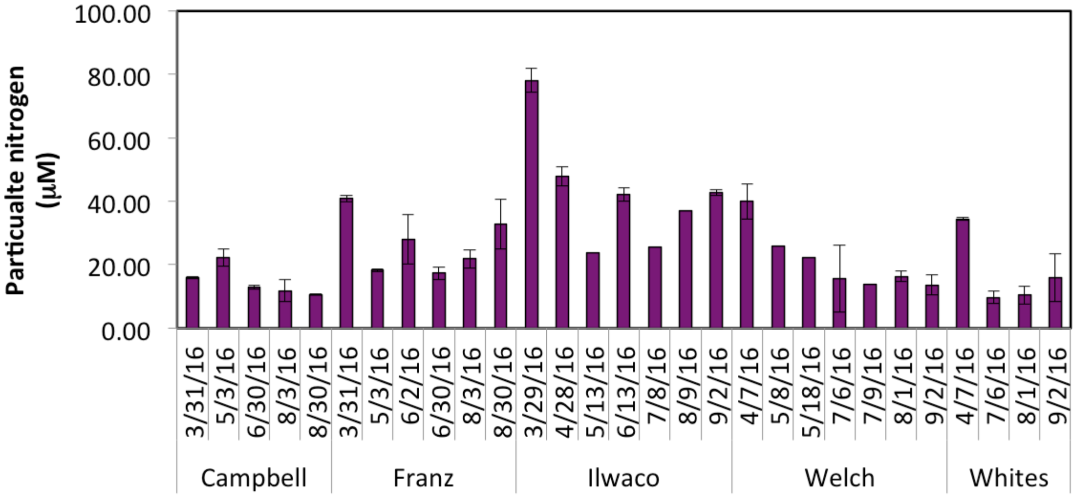


Figure 33. Particulate nitrogen concentrations at the trends sites (Campbell Slough, Franz Lake Slough, Ilwaco Slough, Welch Island, and Whites Island) in 2016.

3.3 Habitat Structure

3.3.1 Hydrology

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 Whites Island to 1.7 m, CRD at Campbell Slough (Table 26; 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 trend 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 17).

- The Secret River site, at rkm 37, is also affected by winter storm events and minimally by the spring freshet. In 2015 and 2016 the mean higher high water (MHHW) was highest at this site (Table 17). The mean water level and the average tidal range at this site were greater than at the Ilwaco Slough site in part due to the lower elevation of the tidal channel where the sensor is located. The low-elevation marsh at the site is exposed only during low tide and conversely, the high-elevation marsh is inundated only during high tide each day.

- 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 17) 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; see Figure 3), the site was inundated above the marsh surface most of the winter and spring through June in 2016.

Table 17. 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
Secret	37	1.51	0.48	2.67	2.19	3.86	12/10/2015	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 34). 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. Inundation at the Secret River low marsh site had the highest inundation frequency of all the sites monitored in 2016 due to its position at the lower end of the tidal-wetland elevation range in the lower river. 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.

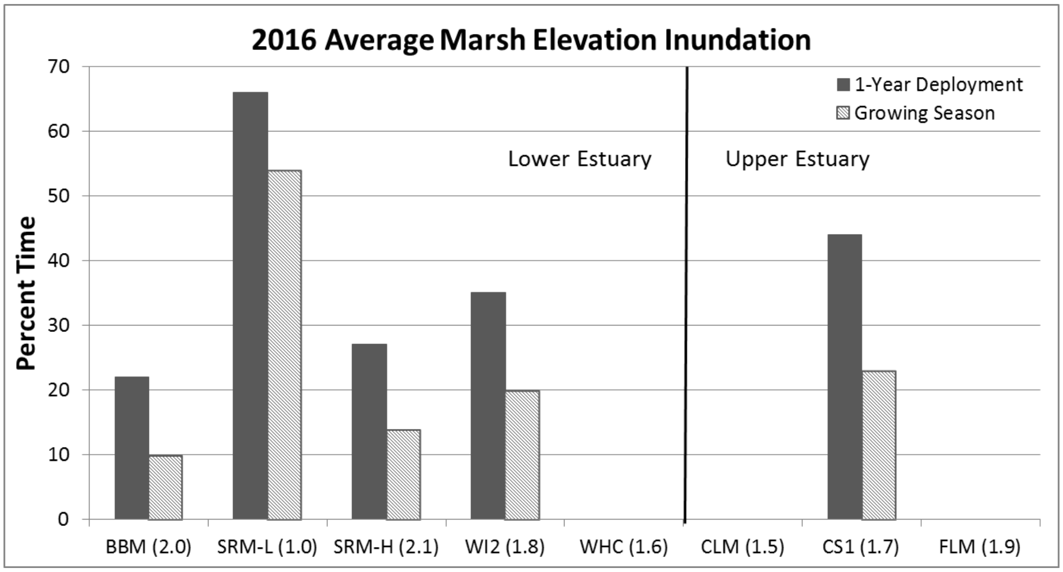


Figure 34. Inundation frequency at four of the trend sites in 2016; one-year deployment is from July 2015 to July 2016 and the growing season is from April -October. Site codes are defined in Table 8. Sites are ordered from left to right starting at the river mouth. Average site elevations are given 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 trend sites measured (Figure 35). Most of the inundation in 2016 occurred in a prolonged series of low-level flood events over the winter and spring.

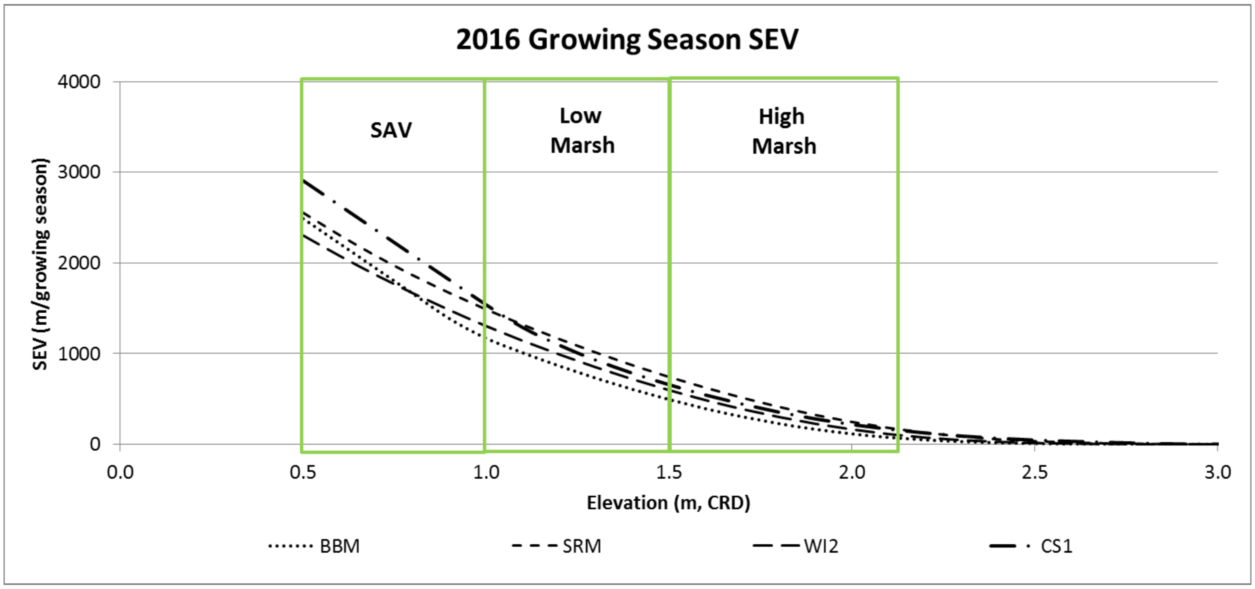


Figure 35. 2016 growing season sum exceedance values (SEVs) for the trend monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the lower river; the vertical lines represent the approximate boundaries between vegetation communities at the trend sites (SAV = submerged aquatic vegetation).

Inter-annual variation in inundation patterns is much greater at the upper estuary sites (Figure 36), where seasonal flooding can result in months of inundation during high-water years. At the lower, tidally dominated sites, inundation occurs frequently, but for a short duration of a few hours. 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 similar to 2010 and 2013, somewhat “average” water years. Inundation was similar to other years at the lower estuary sites.

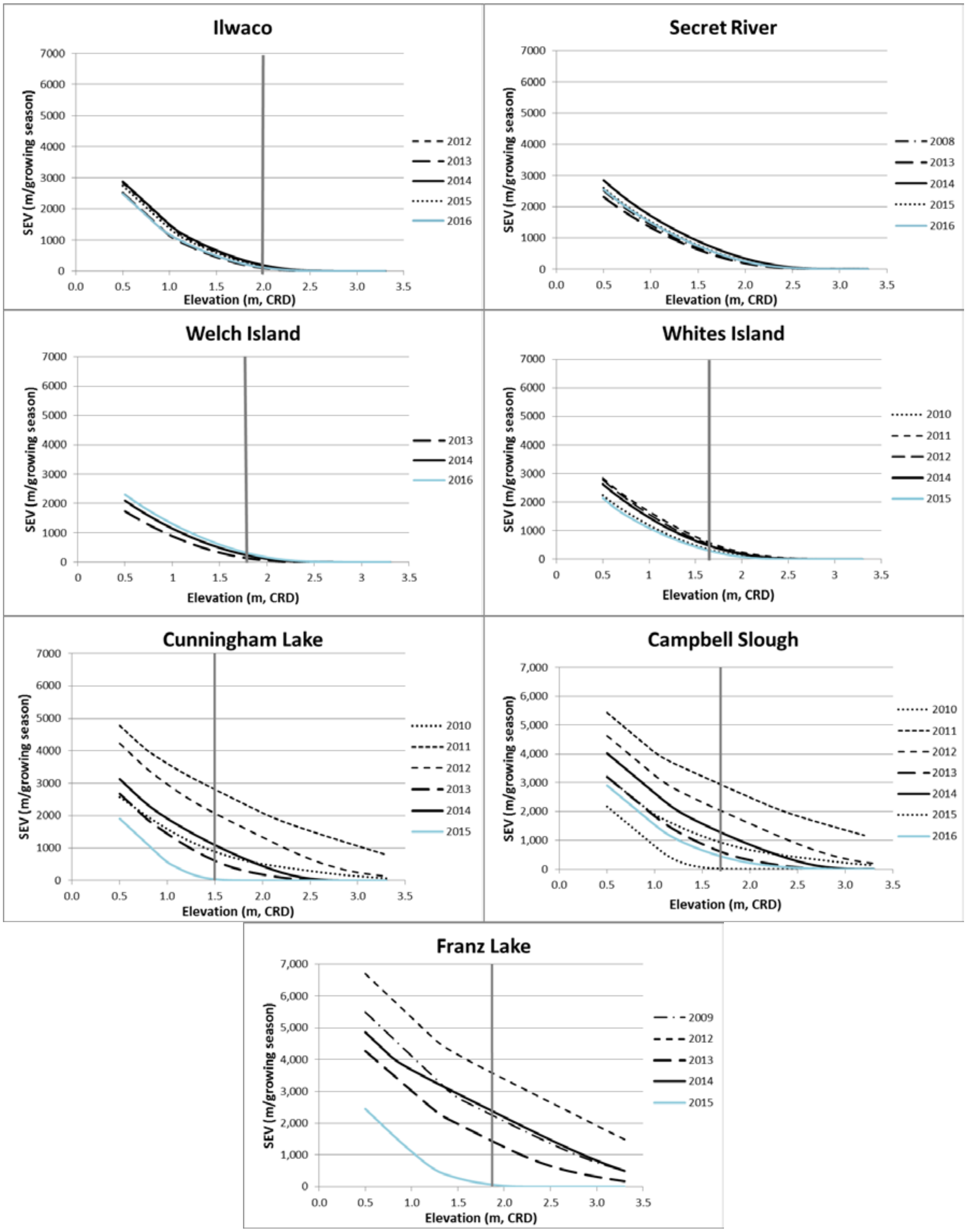


Figure 36. Annual growing season sum exceedance values (SEVs) for the trend monitoring sites based on hydrology data collected on site. Plotted lines represent the calculated SEVs for a given year at the elevations typically found at wetland sites within the lower river; the blue lines indicate the most recent year available. The vertical line represents the average elevation at each site, with the elevation of the Secret River low marsh represented by an additional dashed line. Sites are ordered with the site nearest the Columbia River mouth in the upper left panel.

3.3.2 Sediment Accretion Rates

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 18). 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 confirm hypotheses that increased accretion occurs at lower elevations (CLM; CS1) and with proximity to channels (WI2).

Average sediment accretion at the trend sites ranges from 0.3 cm to 2.2 cm per year except at the Secret River low marsh site where erosion was consistently measured every year, averaging -1.6 cm per year, until 2016 when accretion of 0.9 cm was measured (Table 18). 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 18. Sediment accretion rates at the trend sites between 2008 and 2015. WI2SRM-C is a set of stakes on the channel bank at the Secret River high marsh site and WHC-M and WHC-H represent mid-elevation and high-elevation marsh locations, respectively.

Site Code:	**BBM**	**BBM-2**	**SRM-L**	**SRM-BANK**	**SRM-H**	**WI2**	**WI2-2**	**WHC-M**	**WHC-H**
Elevation (m, CRD):	**1.81**	**1.76**	**1.06**	**2.16**	**2.09**	**1.66**	**1.69**	**1.35**	**1.88**
Year				**Annual Rate (cm)**					
08-09	ND[1]	ND	ND	0.2	ND	ND	ND	ND	-1.2
09-10	ND	ND	ND	2.8	ND	ND	ND	ND	1.0
10-11	1.7	ND	ND	0.9	ND	ND	ND	ND	0.1
11-12	0.1	ND	-2	ND	ND	ND	ND	ND	0.9
12-13	0.6	ND	-1.7	1.4	1.1	0.8	ND	1.2	0.2
13-14	0.3	ND	-1.6	1.0	0.6	0.6	ND	2.3	0.8
14-15	1.0	ND	-0.9	0.7	ND	0.7	ND	2.7	0.0
15-16	0.0	0.3	0.9	0.9	0.3	ND	0.97	2.6	ND
Average	**0.6**		**-1.1**	**1.1**	**0.7**	**0.7**		**2.2**	**0.3**
Std Dev	0.6		1.2	0.8	0.4	0.1		0.7	0.8

Site Code:	**CLM**	**CLM-2**	**CS1**	**CS1-2**	**FLM**	**FLM-2**
Elevation (m, CRD):	**1.54**	**1.26**	**1.49**	**1.91**	**1.87**	**2.13**
Year			**Annual Rate (cm)**			
08-09	ND	ND	ND	ND	0.5	ND
09-10	1.9	ND	0.4	ND	ND	ND
10-11	1.6	ND	1.7	ND	3.0	ND
11-12	1.4	ND	0.9	ND	-0.4	ND
12-13	1.3	ND	0.2	ND	3.0	ND
13-14	0.5	ND	1.5	ND	0.7	ND
14-15	-0.5	ND	-2.4	ND	1.2	ND
15-16	0.9	2.9	1.4	0.8	-0.6	-2.3
Average	1.0		0.5		1.0	
Std Dev	0.8		1.4		1.5	

ND No data.

3.3.3 Vegetation Species Assemblage

A summary of the number and aerial cover of native and non-native species at the trend sites in 2016 is provided in Table 19. The highest number of species occurred at Whites Island, however the highest number of native species at were highest at the Secret River and Welch Island sites. The lowest number of species occurred at Ilwaco Slough and at Cunningham Lake; however there were more non-native species at Cunningham Lake. Ilwaco Slough had the highest proportion of native species (88 percent) and Campbell Slough had the highest proportion of non-native species (37 percent). Welch Island had the highest proportion of native species cover (85 percent), while Whites Island had the highest cover comprised of non-native species (58 percent). Cover was greater than 100 percent at three sites in 2016 (Table 19) and similar to 2015, the highest cover was measured at Welch Island. The lowest total cover was measured at the Secret River low marsh, however the lowest high marsh cover was at Ilwaco Slough.

Table 19. Species richness and areal cover of native and non-native species at the 2016 monitoring sites.

Site	Rkm	Species Richness (native)	Species Richness (non-native)	Species Richness (all)	Percent Cover[1] (native)	Percent Cover[1] (non-native)	Percent Cover (all)
Ilwaco Slough	6	14	2	16	64.2	19.4	83.6
Secret River - High	37	31	10	41	79.1	36.5	115.6
Secret River - Low	37	22	3	25	63.4	13.0	76.4
Welch Island	53	31	10	41	110.0	20.0	130.0
Whites Island	72	29	13	42	44.3	60.8	105.1
Campbell Slough	145	22	13	35	57.6	41.2	98.8
Cunningham Lake	149	15	6	21	50.7	48.2	98.9
Franz Lake	221	22	7	29	70.3	17.8	88.1

[1] Cover values include only live herbaceous vegetation and woody species that are not solely overhead; overhanging tree cover is not included. Cover values are not relative but absolute and therefore can exceed 100% where there is more than one vertical layer in the plant community.

The aerial cover of the six most common species found at the trend sites are summarized in Table 20. During the monitoring period prior to 2016 the cover of *Carex lyngbyei* and *Phalaris arundinacea* were very similar. In 2016 the cover of *P. arundinacea* was higher than *C. lyngbyei*, due to an increase in *P. arundinacea* cover and a decrease in *C. lyngbyei* cover. Overall, there was an increase in *Sagittaria latifolia* and *Polygonum amphibium* in 2016 compared to the average of previous years.

Table 20. Common vegetation species found at the seven trend sites sampled for habitat structure. Average percent cover estuary-wide was calculated by taking the average of all years (through 2015) at each site then averaging all sites. The 2016 data was averaged by site then all sites averaged.

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 ladysthumb, 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)

Reed canarygrass (*P. arundinacea*) is present at six of the seven trend sites in the lower river (Table 21), with the extent of coverage varying depending on location and annual environmental conditions. The lowest cover was observed at the Secret River low marsh site where 5.3 percent was observed in 2008 and not since then. At Welch Island cover has remained less than 10 percent since 2012. The highest coverage has consistently been observed at Whites Island, where cover has been greater than 40 percent since the site was first monitored in 2009. Moderate cover between 20 and 35 percent was observed at Secret River since 2012; an increase from the 10 percent cover observed in 2008. The upper estuary sites have had variable cover over the monitoring period. *P. arundinacea* cover at the Cunningham Lake and Campbell Slough sites ranged from 15 to 57 percent, with lowest cover observed during years of high inundation and episodes of disturbance from cows. The average *P. arundinacea* cover in 2016 at Campbell Slough was the highest it had been since 2010. Franz Lake had consistently moderate cover of 33 to 34 percent in 2008 and 2009 followed by a decrease, starting in 2011 when high inundation favored the growth of a competitive native species, *Polygonum amphibium*, which has persisted until 2016 (Figure 37).

Table 21. Average percent cover of *Phalaris arundinacea* at the trend sites between 2005 and 2016.

		Average Percent Cover *Phalaris arundinacea*											
Site	**Rkm**	**2005**	**2006**	**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**
Ilwaco Slough	6	ND	ND	ND	ND	ND	ND	ND	0.0	0.0	0.0	0.0	0.0
Secret River–Low	37	ND	ND	ND	5.3	ND	ND	ND	0.0	0.0	0.0	0.0	0.0
Secret River–High	37	ND	ND	ND	10.4	ND	ND	ND	19.8	35.5	24.3	26.3	22.1
Welch Island	53	ND	ND	ND	ND	ND	ND	ND	5.9	9.8	8.3	8.3	9.5
Whites Island	72	ND	ND	ND	ND	43.0	47.8	56.8	42.0	56.5	48.0	53.9	49.7
Cunningham Lake	145	41.7	16.4	36.1	32.8	38.5	57.3	15.6	22.5	39.2	24.3*	52.0	47.4
Campbell Slough	149	35.6	30.7	18.4	28.9	37.9	41.5	33.6	15.2	33.1	26.6	29.5	36.0
Franz Lake	221	ND	ND	ND	33.0	34.3	ND	26.5	5.8	13.8	8.8	15.5	17.0

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

Annual vegetation cover for the dominant species is depicted in Figure 37 (cover for all species observed in 2016 is provided in Appendix D). Cover at the lower estuary sites is generally stable between years, however, some variability is evident. At Ilwaco Slough a decrease in cover has occurred in 2015 and 2016, primarily due to a 20 percent reduction in the cover of *C. lyngbyei*. Additionally, cover was lower in 2016 at the Secret River low marsh site than in previous years, with the greatest single species reduction occurring in soft stem bulrush (*Schoenoplectus tabernaemontani*). High marsh cover at the Secret River, Welch Island, and Whites Island sites is very high (>100 percent), except at Whites Island in 2011 and 2012. Of these three high marsh sites, Welch Island has had consistently the highest species richness (>40 species) since 2012, though Whites Island has also had high richness the past four years. In

2016, at the Secret River high marsh site species richness increased to over 40 with a concomitant increase in the cover of non-dominant species (Figure 37). The cover of water parsley (*Oenanthe sarmentosa*) has decreased over time at the site, while several other native species have increased including nodding beggars-ticks (*Bidens cernua*) and Douglas aster (*Symphyotrichum subspicatum*). Of concern is the introduction of non-native species in the past couple years at Secret River, including birdsfoot trefoil (*Lotus corniculatus*) and wart-removing herb (*Murdannia keisak*).

Variability in the cover of the dominant species is more evident at upper estuary sites over the monitoring period. However, cover at the upper estuary sites was high and very similar between 2015 and 2016. Wapato (*Sagittaria latifolia*) had the highest cover measured during the monitoring period at the Cunningham Lake and Campbell Slough sites in 2015 and 2016. At Franz Lake, a shift in vegetation dominance from *P. arundinacea* to water smartweed (*Polygonum amphibium*) occurred in 2012 continued through 2016 when cover reached a high of 45 percent. At the Cunningham Lake and Franz Lake sites, there is a positive relationship between the number of species and the aerial cover at the site ($R^2 = 0.69$ and 0.71, respectively).

Submerged aquatic vegetation (SAV) species occur at the lowest elevations of the sites, in the channels and in ponded depressions in the emergent vegetation. Vegetative cover data for the SAVs that occur in the marsh are reported with the emergent cover for all of the sites in Appendix D (Table D-1). At the Secret River low marsh site, waterweed (*Elodea* spp.) accounted for 25 percent of the cover in 2016, occurring throughout the low marsh in small depressions that hold water at low tide (Figure 37). At all other sites, SAV species account for less than five percent of the cover in the emergent marsh area.

Cover data for SAV species in the tidal channels are provided for the six trend sites in Appendix D (Table D-2). Prior to 2015, horned pondweed (*Zannichellia palustris*) occurred at Ilwaco Slough in the tidal channel, however in 2015 the SAV in the channel was replaced by *Zostera japonica*, a marine species that requires higher salinity than the pondweed. This species persisted in 2016 however the cover was reduced compared to 2015. Above the brackish zone SAV species are more prevalent and are dominated by the native species *Elodea* spp. and *Potamogeton* spp. At Secret River the percent cover of SAVs was 26 percent, which is less than half that observed in the previous three years. In contrast, SAV cover at Whites Island has been increasing every year since 2013 to 45 percent cover in 2016. One species in particular, the native *Potomogeton zosteriformis*, was not observed prior to 2015 and increased to 7 percent in 2016. SAV cover at Campbell Slough, nearly non-existent in 2013 and 2014, increased to 40 percent in 2015 and 32 percent in 2016. The primary cover was milfoil (*Myriophyllum* spp.) and *Elodea* spp. The channel at Franz Lake has consistently had very little SAV.

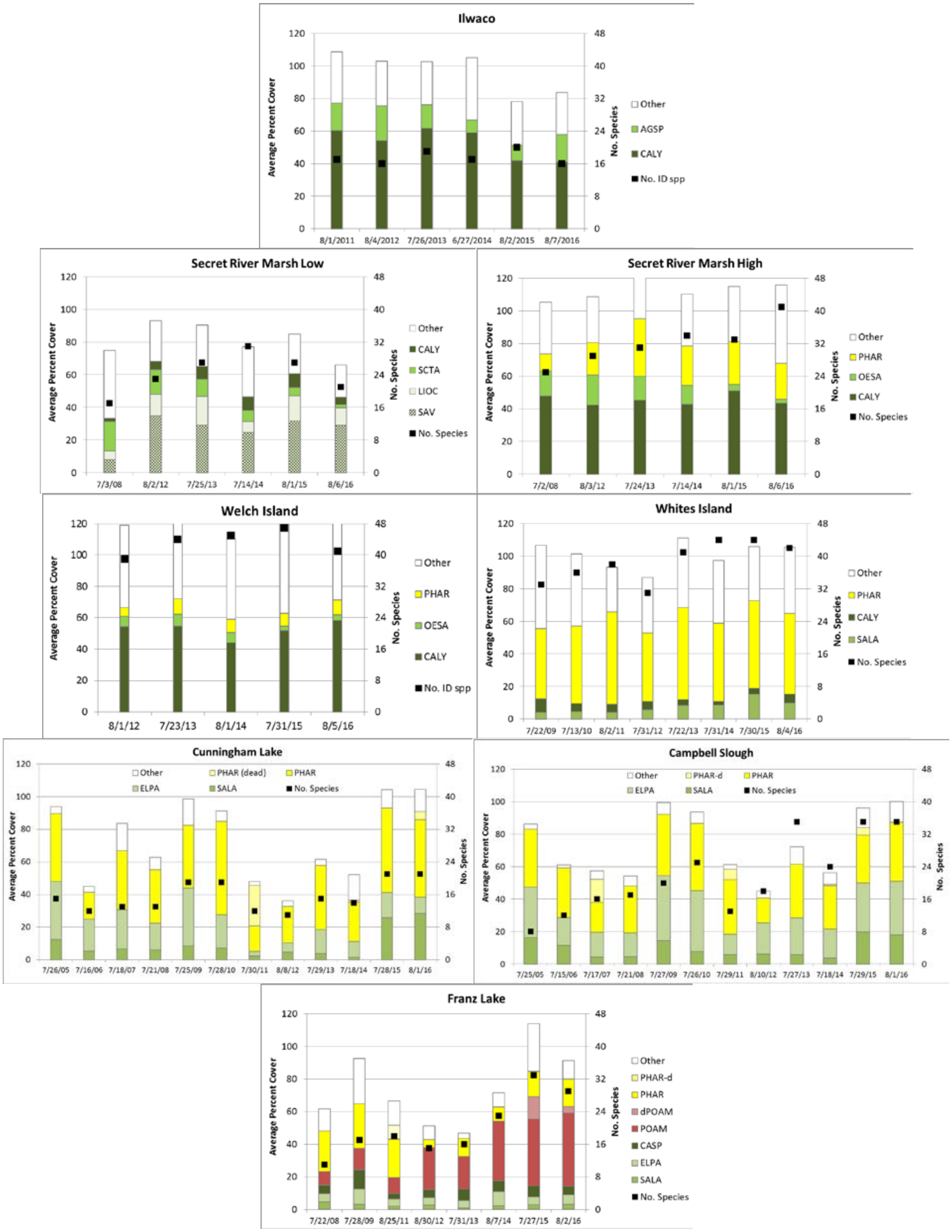


Figure 37. Average percent cover and number of identified species at the trend sites for all years monitored. Sites are presented in the order in which they occur in the River, starting near the mouth. Species codes are defined in Appendix D.

A weighted similarity analysis of vegetation cover between years at each site was conducted to evaluate change over time and differences between sites. In general, the similarity between years at the trend sites was the greatest at the lower estuary sites (Figure 38 and Table 22) with average similarity between 76 and 83 for the four marsh sites below rkm 72. The lowest average similarity was at the three upper estuary sites, all with 69 percent average similarity. Average similarity between years significantly decreases moving up river (Figure 39; regression $p < 0.001$). As the span between years increases, the pairwise similarity for a given site decreases. Thus, for those trend sites observed over a greater number of years, the average similarity decreased significantly with an increasing number of years between observations (Figure 39; regression $p = 0.001$). Similarity was slightly higher for the comparison between the 10 and 11 years at Cunningham Lake and Campbell Slough compared to the pairwise comparison of 9 years.

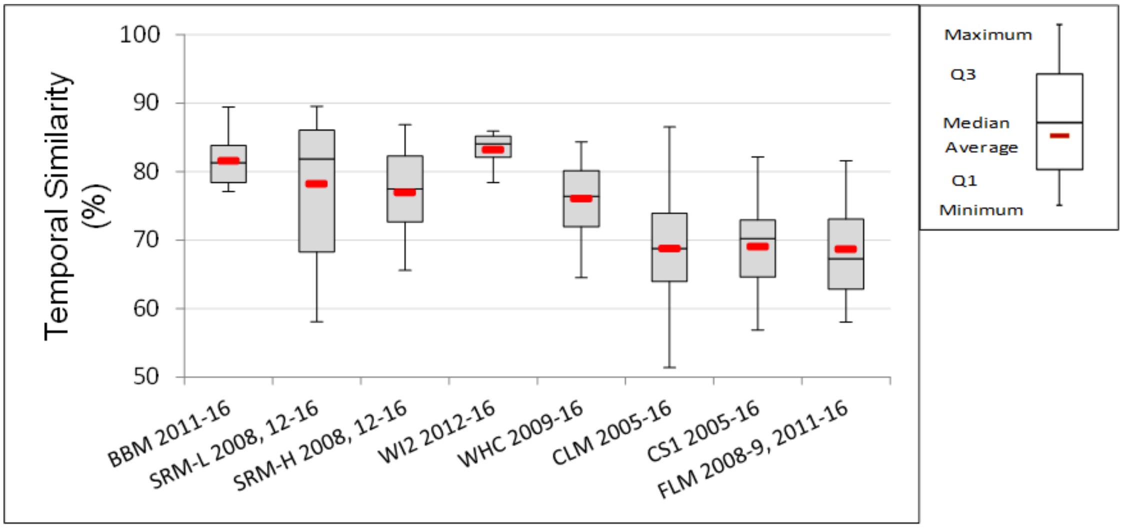


Figure 38. Box plot of the Bray-Curtis similarity calculated between years at the trend sites. Site codes are as follows: BBM = Ilwaco Slough, SRM-L = Secret River low marsh, SRM-H = Secret River high marsh, WI2 = Welch Island, WHC = Whites Island, CLM = Cunningham Lake, CS1 = Campbell Slough, FLM = Franz Lake. Sites are ordered from the mouth to the upper estuary.

Table 22. Descriptive statistics of the percent site similarity of vegetative cover at a site over time. Site codes are as follows: BBM = Ilwaco Slough, SRM-L = Secret River low marsh, SRM-H = Secret River high marsh, WI2 = Welch Island, WHC = Whites Island, CLM = Cunningham Lake, CS1 = Campbell Slough, FLM = Franz Lake. Sites are ordered from the mouth to the upper estuary.

Site	Number of Pairs	Mean	StDev	Minimum	Q1	Median	Q3	Maximum
BBM	15	81.6	3.8	77.0	78.4	81.3	83.8	89.4
SRM-L	15	78.2	10.3	58.1	68.2	81.9	86.0	89.5
SRM-H	15	76.9	6.2	65.6	72.6	77.4	82.3	86.8
WI2	10	83.2	2.7	78.4	82.1	84.0	85.1	85.9
WHC	28	76.1	5.3	64.5	71.9	76.4	80.1	84.3
CLM	66	68.7	7.9	51.4	63.9	68.8	73.9	86.5
CS1	66	69.0	5.8	56.9	64.6	70.2	72.9	82.1
FLM	28	68.7	6.8	58.0	62.8	67.3	73.0	81.6

[1] The number of comparisons (*n*) is based on the number of years a site was monitored; for example, CLM was monitored for 11 years and 55 year-to-year comparisons could be made.

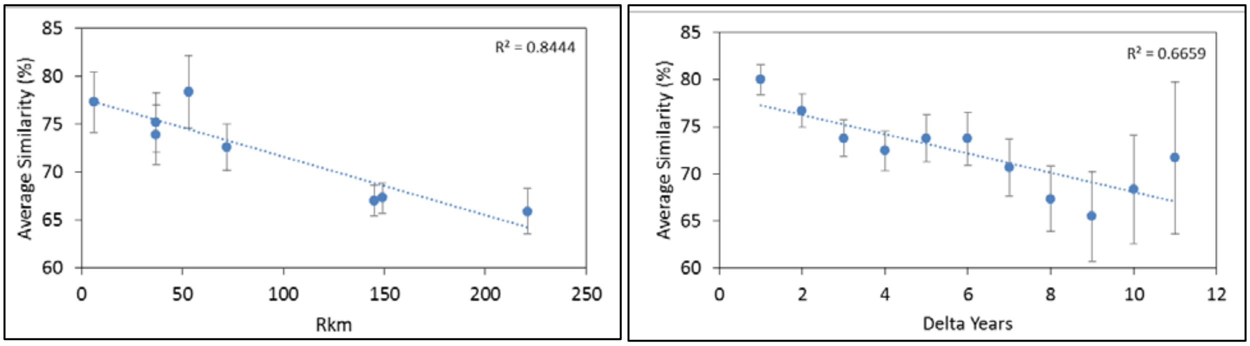


Figure 39. Average similarity of sites using the least square mean model for individual sites with the effect of the difference in years removed (left plot), and the average similarity between years with the effect of the different sites removed (right plot).

3.3.3.1 *Floristic Quality*

The metrics calculated as part of the Floristic Quality Assessment are summarized in Table 23. The mean Coefficient of Conservatism (mean C) values for the long term monitoring sites range from a low of 2.4 at Campbell Slough to a high of 4.8 at Ilwaco Slough. The percent of species with C values that are considered intolerant of human stressors (C >=7) was lowest at Whites Island and highest at Ilwaco Slough. Conversely, the percent of species with the most tolerance for human stressors (C <=3) was highest at Campbell Slough and lowest at the Secret River low marsh site. The FQI had a narrow range between 19.0 and 21.8 at the lower estuary sites and between 14.6 and 16.7 at the upper estuary sites. The Adjusted FQI had higher scores and a broader range as would be expected since non-native species are part of the calculation. Using this metric Ilwaco Slough has a much higher score than the others, followed by the Secret River low marsh site and Welch Island and Campbell Slough has the lowest score. The wetland indicator values (0 = facultative; -3 = wet facultative; and -5 = obligate) show that the native species tend to be more wetland obligate species than the non-natives. The percent of species that are hydrophytes is lowest at the Campbell Slough and Franz Lake sites, but as expected over 65 percent at all other sites.

Table 23. Floristic quality metrics for the long-term monitoring sites. Sites are ordered starting at the Columbia River mouth; site codes are provided in Table 8.

Site	BBM	SRM-L	SRM-H	WI2	WHC	CLM	CS1	FLM
Mean C (all species)	4.8	3.8	3.2	3.4	3.1	3.1	2.4	3.0
% intolerant (C value >= 7)	13	4	7	5	2	5	5	6
% tolerant (C value =< 3)	25	24	49	40	47	45	51	48
FQI (all species)	19.1	19.0	20.2	21.8	20.0	14.6	14.8	16.7
Adjusted FQI	46.5	36.7	33.9	36.3	33.4	32.5	29.1	32.1
Wet Indicator (all)	-4.5	-4.8	-3.4	-4.1	-4.1	-4.3	-3.5	-3.3
Wet Indicator (native)	-4.5	-4.8	-3.8	-4.3	-4.5	-4.7	-4.4	-3.9
% hydrophyte	75	76	66	76	74	68	62	55

3.3.3.2 *Vegetation Community Change Analysis*

Spatial analysis of vegetation communities was compared for two time periods at six of the trend sites. The results and the maps are provided in Appendix A. In summary, all sites exhibited some level of change in the boundaries and extent of the vegetation communities as summarized below.

Carex lyngbyei communities declined over the 4 year analysis period at Ilwaco Slough; 58 percent of the community comprised primarily of *C. lyngbyei* remained, while 42 percent changed to a community mixed with *Agrostis* spp. (23 percent), bare ground/pans (12 percent), or *Typha angustafolia* (7 percent). Likewise, the mixed community of *C. lyngbyei* and *Agrostis* spp that was present in 2011 also declined by 40 percent, primarily (36 percent) converting to a community of grasses with very little *C. lyngbyei*. The bare pans, or depressions, at the site more than doubled in area, whereas the prominent channel vegetation present in 2011, *Zannichellia palustris*, was not present in 2015.

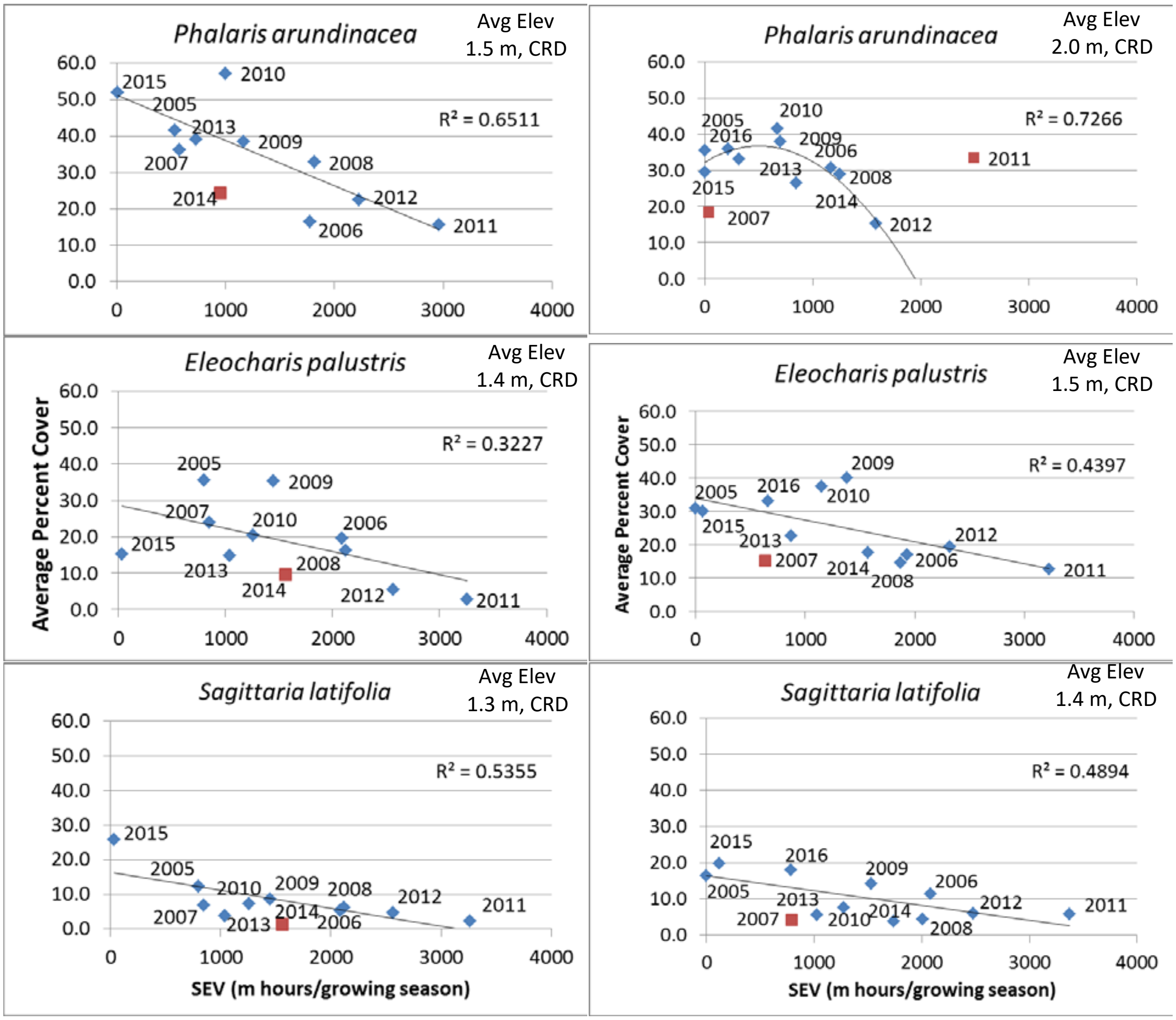
At Welch Island in 2015 we focused on re-mapping the *P. arundinacea* patches that were originally mapped in 2012 to determine if this invasive species was spreading into the native sedge dominated *(C. lyngbyei*) community. The overall coverage of *P. arundinacea* was very similar between years; however 38 percent of the area changed with some patches reduced in size while new patches developed.

Most of Whites Island is comprised of a high marsh vegetation community with a heterogeneous mix of many species and little species differentiation, making a change analysis difficult. One area of the site, however, is predominantly *P. arundinacea* and *C. lyngbyei*. We re-mapped this area in 2015 to determine whether changes in the boundary between the two species had occurred. The size of the *C. lyngbyei* patch was 40 percent smaller than in 2009 and 23 percent had shifted to *P. arundinacea* while the rest had changed to lower marsh species.

Vegetation community zonation is much more distinct at the two up-river sites located near the confluence of the Willamette River in Zone 4. Vegetation at these sites grades from *Sagittaria latifolia* at the lowest marsh elevation to *Eleocharis palustris* and then to high marsh typically dominated by *P. arundinacea*. While the boundaries of the communities are somewhat static, our change analysis at Cunningham Lake and Campbell Slough indicates that the communities often overlap at the boundaries and form mixed communities that change over time. Community change was greater at Cunningham Lake than at Campbell Slough, with the percent of the herbaceous communities that changed between 42 and 65 percent of the original area, compared to 12 to 20 percent at Campbell Slough. The lowest elevation marsh community at Cunningham Lake changed the most, with 53 percent of the area becoming unvegetated and 13 percent shifting to a slightly higher elevation community.

Franz Lake exhibited the most obvious change of all the trend sites, with a shift in the dominant vegetation community from *P. arundinacea* in 2008 to *Polygonum amphibium* in 2012 (Figure 37). Our change analysis between those years indicated that 75 percent of the *P. arundinacea/P. amphibium* community shifted primarily *P. amphibium* by 2012. Additionally, portions of low marsh (*Eleocharis palustris*) and the willow community (*Salix* spp.) also shifted to *P. amphibium* by 2012. This vegetation community continued to persist in 2015, with only small area changes within the community.

The sites in Zone 4 are strongly influenced by riverine hydrology, which affects interannual variability, but also tides that drive vegetation community zonation. Thus, these sites were evaluated for the effects of hydrology, specifically inundation as measured by the sum exceedance value (SEV), on vegetation cover. In general, vegetation cover in each vegetation community is reduced by increased inundation (Figure 40). The *P. arundinacea* community at Campbell Slough does not follow the same linear cover decrease in response to inundation as the other strata or as the Cunningham Lake strata; instead slightly lower cover was observed at low inundation levels.



a) Cunningham Lake b) Campbell Slough

Figure 40. Annual average percent vegetation cover of the dominant species at a) Cunningham Lake and b) Campbell Slough as related to annual growing season inundation (sum exceedance values; SEV) calculated at the average elevation for each species (elevation for each strata indicated in the upper right corner of each plot). The red square dots on the plots represent years when data was not comparable for reasons described in the methods; the points were not included in the regression.

3.3.4 Channel Morphology and Inundation

Channel morphology at the trend sites exhibited low inter-annual variability in years prior to 2016; therefore, only the channel mouth cross section was surveyed this year. 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 Secret River channel had the greatest area: close to 20 m^2 for most of its length. 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 24. 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 codes are provided in Table 8.

		Physical Metrics					
Site (year)	**Cross Section**	**Thalweg Elevation (m, CRD)**	**Bank Elevation (m, CRD)**	**Channel Depth (m)**	**Cross Section Area (m^2)**	**Channel Width (m)**	**Width:Depth Ratio**
BBM (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
SRM (12)	0*	0.12	0.92	0.80	8.9	22.8	28.5
	1	0.32	1.42	1.09	19.3	22.6	20.6
	2	-0.04	2.13	2.17	22.5	14.9	6.87
	3	-0.03	1.98	2.01	20.7	15.1	7.52
WI2 (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
WHC (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
CLM (15)	1	0.82	1.26	0.44	5.5	18.3	41.6
CS1 (15)	1	0.80	1.47	0.67	11.7	23.0	34.3
FLM (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 percent. 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. At Secret River, the channel thalweg is inundated 71-99 percent of the time and the channel bank around 50 percent reducing up the channel to less than 10 percent of the time. Difference 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 differences in channel 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 25. Inundation frequency for channel thalweg (+50 cm) and channel bank (+10 cm) during peak juvenile chinook migration period (March – July). Site codes are provided in Table 8.

		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**
BBM	**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
SRM	**0**	78	nd	nd	nd	nd	77	85	83	81	59	nd	nd	nd	nd	56	63	63	64
	1	73	nd	nd	nd	nd	69	77	72	71	47	nd	nd	nd	nd	43	49	46	46
	2	99	nd	nd	nd	nd	88	97	93	90	14	nd	nd	nd	nd	14	22	17	17
	3	99	nd	nd	nd	nd	87	96	92	89	3	nd	nd	nd	nd	4	8	5	6
WI2	**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
WHC	**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
CLM	**1**	nd	nd	67	98	98	73	96	51	nd	nd	nd	64	98	98	80	97	59	nd
CS1	**1**	nd	nd	67	97	98	86	98	74	85	nd	nd	23	96	96	68	96	6	64
FLM	**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 the annual monitoring period from 2016 - 2017 and those from a multi-year analyses 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 is sampled within vegetation strata to reduce variability associated with sampling across strata. The dominant species for the strata are identified in Table 26. 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 26). When the stratum was a mix of two dominant species the results were mixed. At Ilwaco Slough (BBM) only 45 percent of the CALY/AGSP samples were comprised of these two species, however, the ELPA/SALA strata were both over 90 percent those species. The strata that were a high or low marsh mix were expectedly comprised of lower proportions of a single species, with the dominant species ranging from 50 to 70 percent of the samples (Table 26). The low marsh at Secret River is the only site that did not have any dominant species and all were species in the samples were categorized as “Other” species.

Table 26. Average proportion of dominant species within the samples of each stratum and average proportion of samples that are live and dead in summer and winter.

			Species Present in Stratum Samples							Summer		Winter	
Site	**Stratum Code**	**Stratum**	**CALY**	**CALY/ AGSP**	**ELPA/ SALA**	**PHAR**	**POAM**	**SALA**	**Other**	**Dead**	**Live**	**Dead**	**Live**
BBM	CALY	*C. lyngbyei*	0.98						0.02	0.09	0.91	0.61	0.39
BBM	CALY/ AGSP	*C. lyngbyei/ Agrostis* spp		0.45					0.55	0.18	0.82	0.77	0.23
SRM	HM	High marsh mix	0.56			0.15			0.29	0.05	0.95	0.74	0.26
SRM	LM	Low marsh mix							1.00	0.01	0.99	0.60	0.40
WI2	HM	High marsh mix	0.50			0.16			0.34	0.11	0.89	0.81	0.19
WHC	CALY	*C. lyngbyei*	0.90						0.10	0.06	0.94	0.85	0.15
WHC	PHAR/ HM	*P. arundinacea*/ High marsh mix				0.64			0.36	0.06	0.94	0.88	0.12
WHC	SALA/ LM	*S.latifolia*/ Low marsh mix						0.67	0.33	0.04	0.96	0.85	0.15
CLM	ELPA/ SALA	*E. palustris/ S.latifolia*			0.96				0.04	0.03	0.97	0.97	0.03
CLM	PHAR	*P.arundinacea*				0.92			0.08	0.10	0.90	0.91	0.09
CS1	ELPA/ SALA	*E. palustris/ S. latifolia*			0.92				0.08	0.06	0.94	1.00	0.00
CS1	PHAR	*P. arundinacea*				0.88			0.12	0.07	0.93	0.66	0.34
CS1	SALA	*S. latifolia*						1.00	0.01	0.04	0.96	1.00	0.00
FLM	PHAR/ HM	*P. arundinacea/* High marsh mix				0.70			0.30	0.12	0.88	0.96	0.04
FLM	POAM	*P. amphibium*				0.04	0.95		0.01	0.20	0.80	0.97	0.03

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 greater than 15 percent standing dead (Table 26). In the winter, samples from the upper estuary sites were generally greater than 90 percent dead, with the proportion decreasing toward the mouth. The samples from the two sites closest to the mouth were between 60 and 77 percent standing dead (Table 26).

2016-2017 Results

Annual biomass sampling results for summer 2016 and winter 2017 are summarized in Figure 41 and Table 27. The highest amount of summer biomass and the greatest variability was in the samples from the Whites Island *P. arundinacea*/high marsh stratum and the Franz Lake *P. amphibium* stratum. The contribution of organic matter was also greatest from the Whites Island *C. lyngbyei* and the *P. arundinacea*/high marsh strata (Table 27). We were not able to sample Cunningham Lake or Franz Lake in the winter due to high water therefore the measured contribution from those sites could not be calculated. However, the proportion of summer biomass contributed to the ecosystem is relatively constant for specific strata within sites or reaches of the River. In previous years the contribution from the Franz Lake *P. amphibium* stratum was approximately half the summer standing stock and therefore would have contributed roughly 1000 g/m^2.

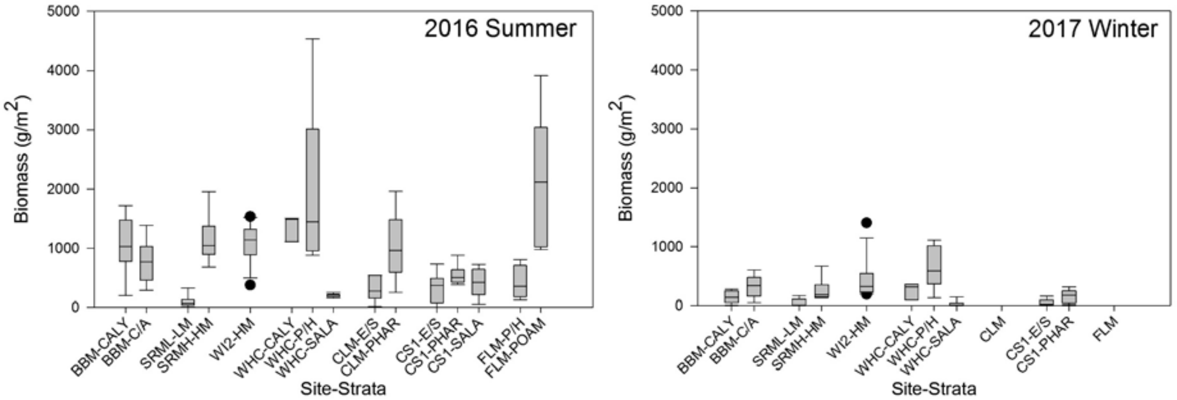


Figure 41. Aboveground biomass (dry weight g/m^2) for all sites from summer 2016 and winter 2017 sampling

Table 27. Aboveground biomass (dry weight g/m^2) for all sites from summer 2016 and winter 2017 sampling. Contribution of organic matter is calculated as the Summer standing stock minus Winter standing stock. Sites are ordered starting at the mouth and strata are ordered by elevation lowest to highest.

		Summer 2016 (live + dead standing stock)			Winter 2017 (dead standing stock)			
Site	Vegetation Strata*	n	Average Dry wt (g/m^2)	StdDev Dry wt (g/m^2)	n	Average Dry wt (g/m^2)	StdDev Dry wt (g/m^2)	Contribution Dry Wt (g/m^2/year)
BBM	CALY	6	1057.8	509.7	6	141.4	107.7	916.3
BBM	CALY/AGSP	6	771.0	376.6	6	325.1	195.4	445.9
SRM	LM	9	99.6	92.7	9	47.8	62.1	51.8
SRM	HM	9	1147.5	389.0	9	268.2	173.6	879.3
WI2	HM	12	1094.7	319.6	12	432.1	334.6	662.7
WHC	SALA	6	198.4	31.5	6	32.2	55.5	166.2
WHC	CALY	3	1366.8	220.1	3	253.0	142.7	1113.7
WHC	PHAR/HM	9	1947.0	1321.1	9	623.4	351.9	1323.6
CLM	ELPA/SALA	6	305.1	205.5	nd	nd	nd	nd
CLM	PHAR	7	1035.2	561.3	nd	nd	nd	nd
CS1	SALA	6	417.6	245.7	6	0.0	0.0	417.6
CS1	ELPA/SALA	7	350.0	246.1	6	48.1	56.4	301.9
CS1	PHAR	6	541.7	178.1	6	158.8	109.0	383.0
FLM	PHAR/HM	6	418.9	278.2	nd	nd	nd	nd
FLM	POAM	6	2154.8	1143.6	nd	nd	nd	nd

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 2016 are summarized here. The results from the general linear model (GLM) indicated that the covariate rkm (p=0.006) and the main effects of year (p=0.003) and broad strata (HM and LM; p=0.000) were significantly different (Figure 42). The year effect of 2015 after removal of all other effects was significantly greater than 2011 and 2013 (ANOVA, Tukey p=0.007; Figure 43). Regression analysis of the HM and LM summer biomass by rkm indicate that the fit for this relationship is very low (R^2 0.30 and 0.06, respectively), however, biomass in the HM strata is consistently greater in the lower estuary and more variable in the upper estuary (Figure 44).

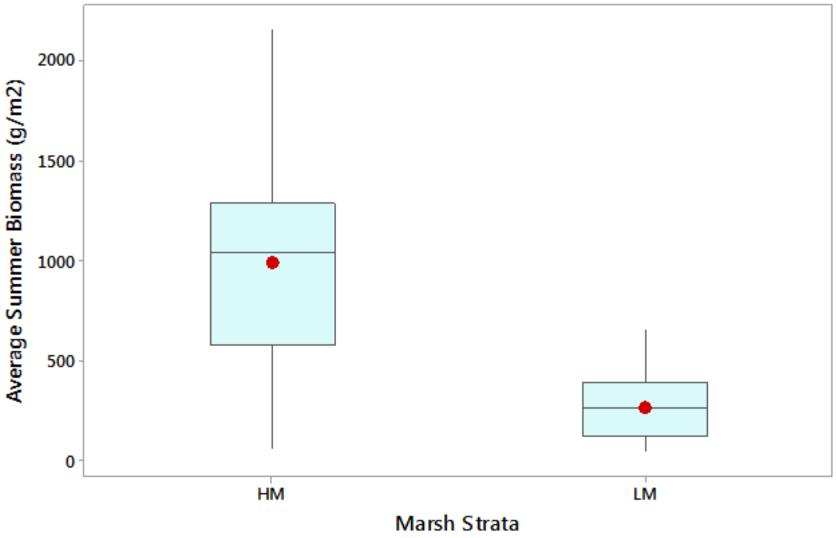


Figure 42. Average summer biomass (g dry weight/m^2) from the high marsh (HM) and low marsh (LM) strata.

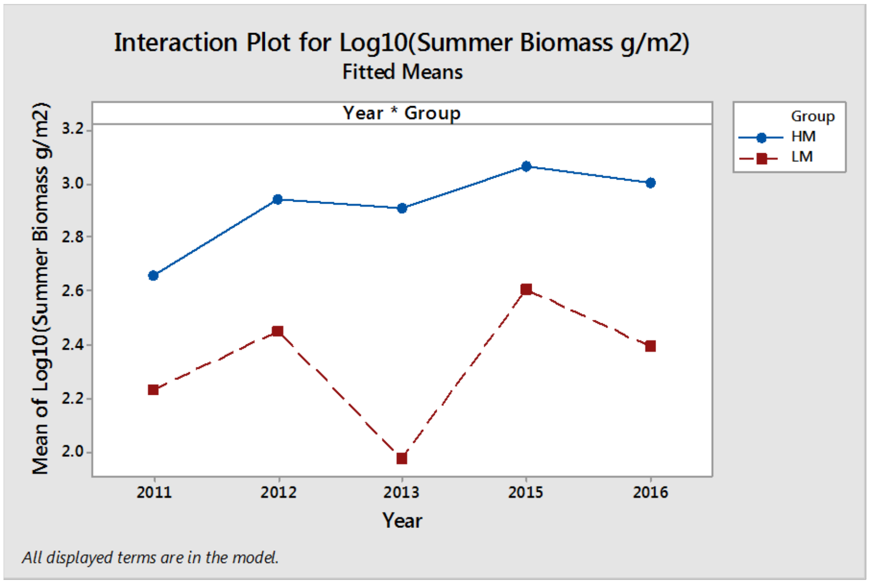


Figure 43. Average annual low marsh and high marsh summer biomass (g dry weight /m^2) by year. Results are transformed by Log10 for statistical analysis.

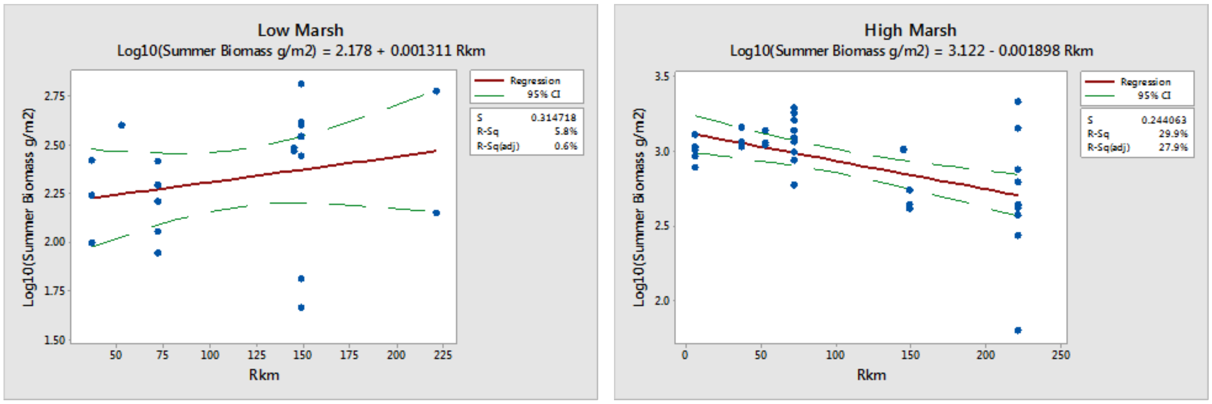


Figure 44. Average annual summer biomass (g dry weight/m^2) compared to rkm for low marsh (left) and high marsh (right) strata. Results are transformed by Log10 for statistical analysis. Note the different scales on the x- and y-axes.

A one-way ANOVA indicated no significant difference in summer biomass between the LM site specific strata (Figure 45) or averaged strata (Figure 46). Pairwise comparisons (ANOVA, Tukey) for the HM site specific strata indicated that the summer biomass from the PHAR/HM stratum at FLM was significantly less (p=0.000) than most other site specific strata. The exceptions were the PHAR strata at CLM and CS1 which were not significantly different than PHAR/HM at FLM and the PHAR/POAM stratum at FLM, which was significantly less than all other site specific strata (Figure 45). The PHAR/POAM strata only occurred in 2011 when the site was transitioning from being dominated by *P. arundinacea* (PHAR) to *P. amphibium* (POAM; Figure 37). When the high marsh strata for all sites were averaged the only strata that was significantly different was the PHAR/POAM strata (ANOVA, Tukey, p=0.000; Figure 46). Biomass production was highest and most consistent in the strata dominated by *C. lyngbyei*, averaging 1149 ±241 g dry weight/m^2 compared to the strata dominated by *P. arundinacea* which averaged 759 ± 520 g dry weight/m^2.

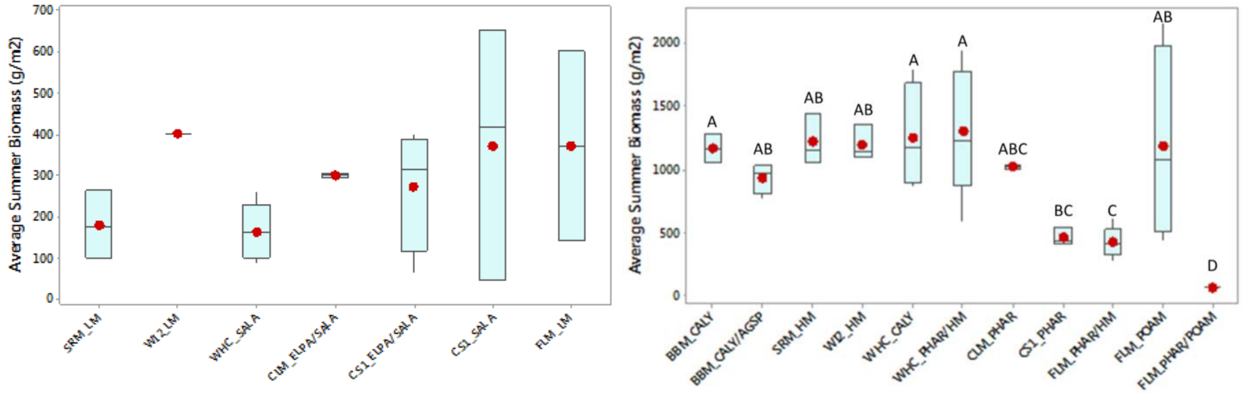


Figure 45. Average summer biomass (g dry weight/m^2) from site specific low marsh (left) and high marsh (right) strata for all years. Different letters above bars indicate means that were significantly different (p=0.000, Tukey pairwise comparison; no significance between low marsh strata). See Table 26 for definition of site and strata codes. Note different scales on the two plots.

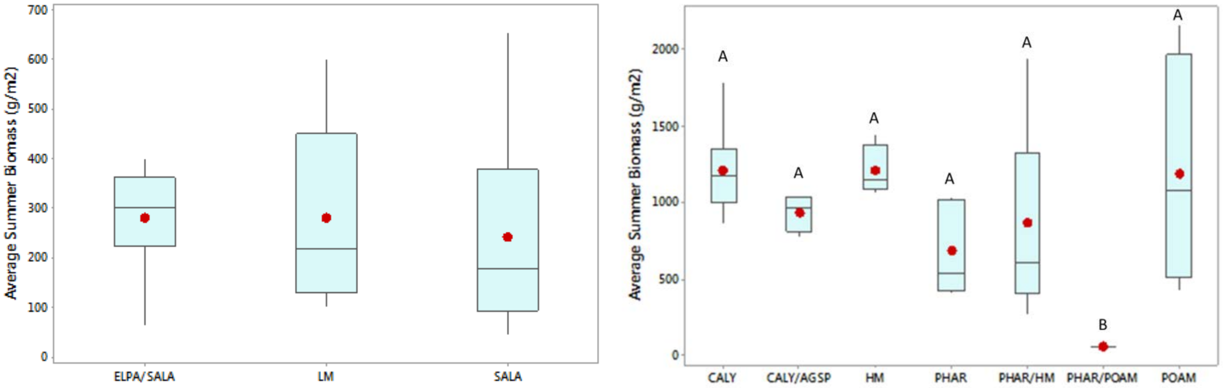


Figure 46. Average summer biomass (g dry weight/m^2) by low marsh (left) and high marsh (right) vegetation strata for all sites and all years. Different letters above bars indicate means that were significantly different (p=0.000, Tukey pairwise comparison; no significance between low marsh strata). See Table 26 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.

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 47). 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 48). 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 49). 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.

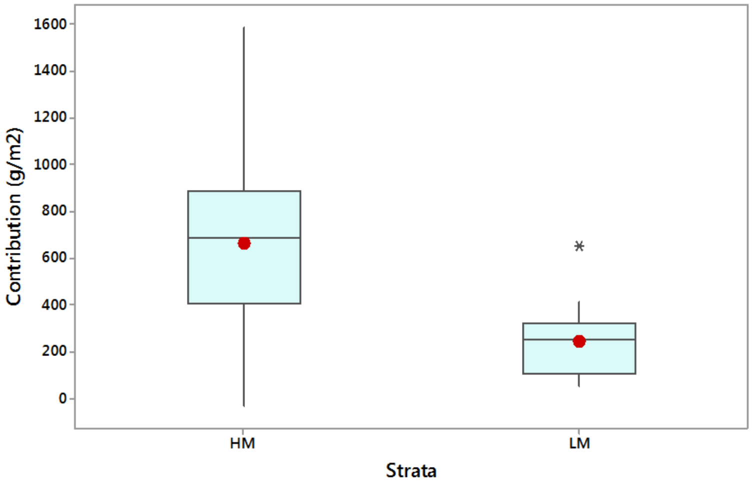


Figure 47. Average annual organic matter contribution (g dry weight/m^2) from the low marsh (LM) and high marsh (HM) strata.

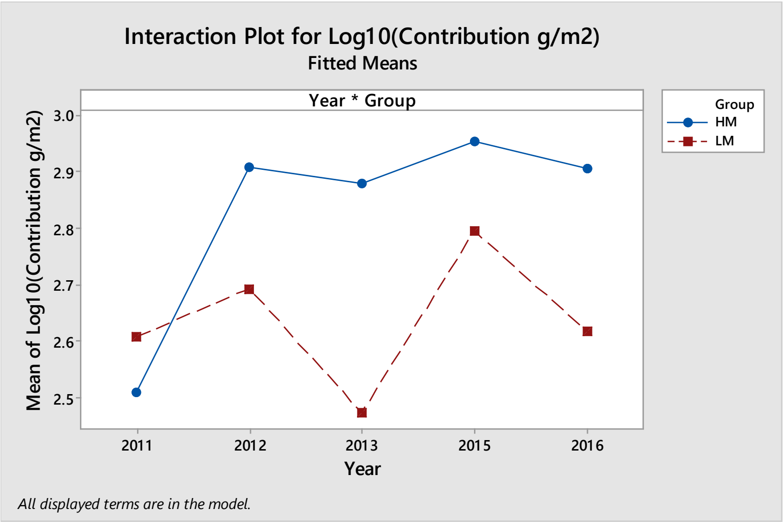


Figure 48. Average annual low marsh and high marsh organic matter contribution (g dry weight /m^2) by year. Results are transformed by Log10 for statistical analysis.

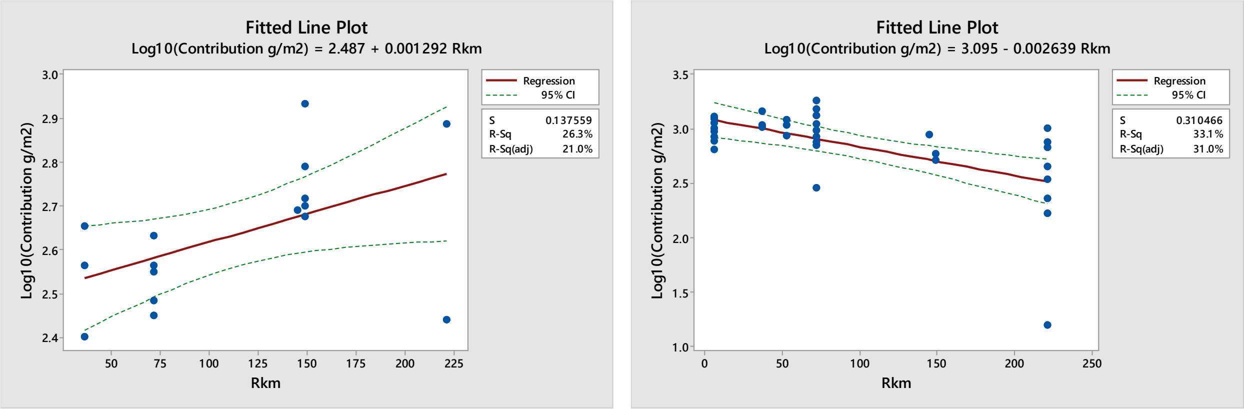


Figure 49. Average annual organic matter contribution (g dry weight/m^2) compared to rkm for low marsh (left) and high marsh (right) strata. Results are transformed by Log10 for statistical analysis. Note the different scales on the y-axes.

Pairwise comparisons (ANOVA) indicate no significant difference in the contribution between sites, the LM site specific strata (Figure 50), or LM averaged strata (Figure 51). 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 50). When the high marsh strata for all sites were averaged CALY was significantly greater than PHAR/HM (ANOVA, Tukey p=0.000; Figure 51) and the PHAR/POAM stratum was significantly lower than all other strata.

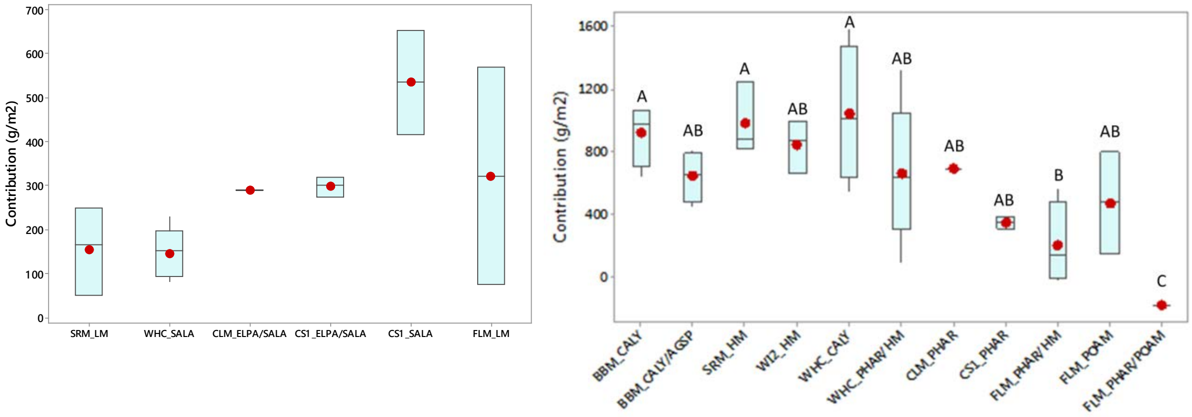


Figure 50. Average annual organic matter contribution (g dry weight/m^2) from site specific low marsh (left) and high marsh (right) strata for all years. Different letters above bars indicate means that were significantly different (p=0.000, Tukey pairwise comparison; there was no significance between low marsh strata). See Table 14 for definition of site and strata codes. Note different scales on the two plots.

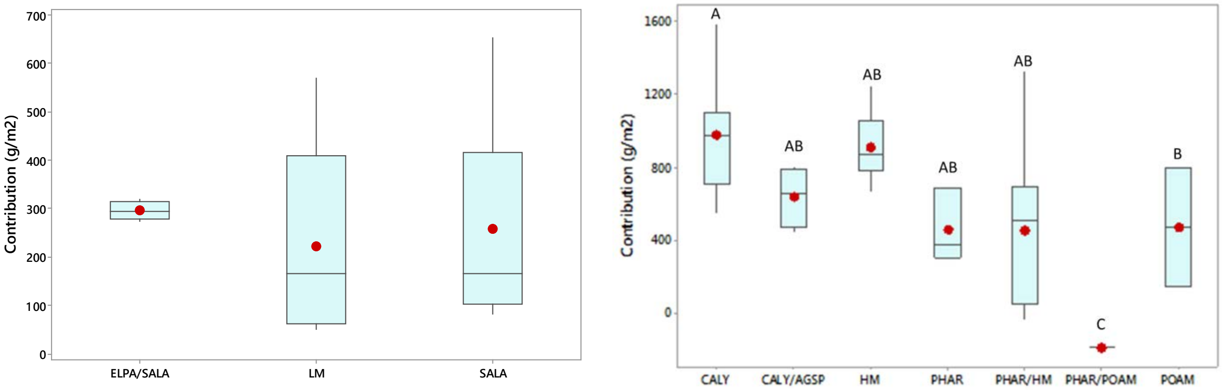


Figure 51. Average annual organic matter contribution (g dry weight/m^2) by low marsh (left) and high marsh (right) vegetation strata for all sites and all years. Different letters above bars indicate means that were significantly different (p=0.000, Tukey pairwise comparison; there was no significance between low marsh strata). See Table 14 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 52). Nearly all of the submerged aquatic vegetation and low emergent marsh biomass is broken down over the winter (Table 28). 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 (Table 27), 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 26), 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 very few other species present (<12 percent of the dry weight of the samples, Table 26), had much lower variability (0.72 ± 0.04, Table 28). The *P. amphibium* stratum also had a high proportion of the plant standing stock remaining and was variable between years.

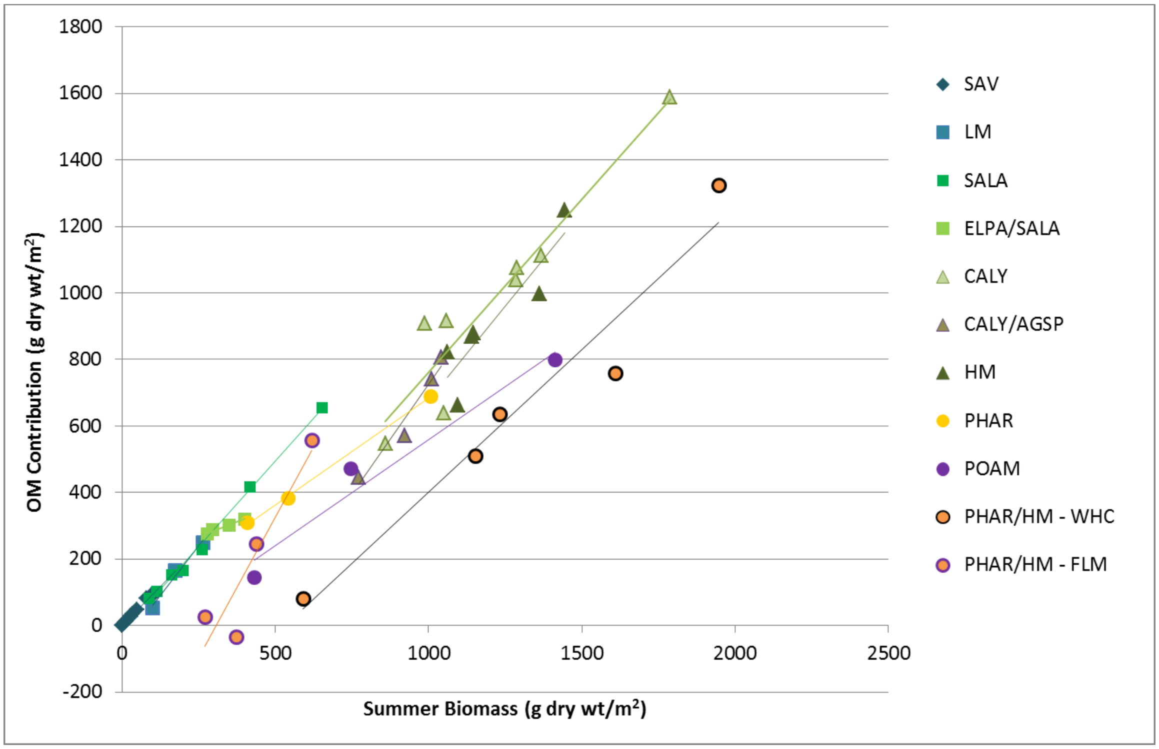


Figure 52. Linear regression between summer biomass and annual organic matter (OM) contribution for different wetland vegetation strata. Each dot represents the average amount for each strata within a year based on 3-12 replicates. Codes are defined and regression equations are provided in Table 28.

Table 28. Average proportion of summer biomass that is 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 52.

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

Vegetation Cover to Biomass Comparison

Summer biomass was compared to summer percent cover results within the vegetation strata identified for the biomass sampling plan. The data taken as a whole, all strata together, yielded a statistically significant, but moderate linear relationship between cover and biomass ($R^2 = 0.49$, p=0.000). Similarly, when the strata were divided into low marsh and high marsh strata biomass was moderately correlated with cover (HM $R^2 = 0.55$, p=0.001 and LM $R^2 = 0.45$, p = 0.000; Figure 53). When the strata were split out the biomass from the PHAR/HM stratum had a moderate and significant correlation with cover ($R^2 = 0.55$, p = 0.013). Some strata had a stronger linear relationship, in particular, SALA, CALY/AGSP, PHAR, and POAM had R^2 values greater than 0.70, though only SALA was statistically significant (p = 0.005; Table 29 and Figure 54).

Table 29. Regression results for vegetation cover and biomass (see Figure 53).

Vegetation Stratum Code	Vegetation Stratum	*n*	R^2	p-value
All Strata		50	0.49	**0.000**
All LM Strata	Low marsh	17	0.55	**0.001**
All HM Strata	High marsh	33	0.45	**0.000**
LM	Low marsh mix	3	na	na
SALA	*S. latifolia*	8	0.76	**0.005**
ELPA/ SALA	*E. palustris/ S.latifolia*	6	0.38	0.192
CALY/ AGSP	*C. lyngbyei/ Agrostis* spp	4	0.85	0.079
CALY	*C. lyngbyei*	4	na	na
HM	High marsh mix	6	na	na
PHAR/HM	*P. arundinacea*/ High marsh mix	10	0.55	**0.013**
PHAR	*P. arundinacea*	5	0.70	0.078
POAM	*P. amphibium*	4	0.76	0.125

na – strata where the vegetation cover or biomass did not meet regression criteria

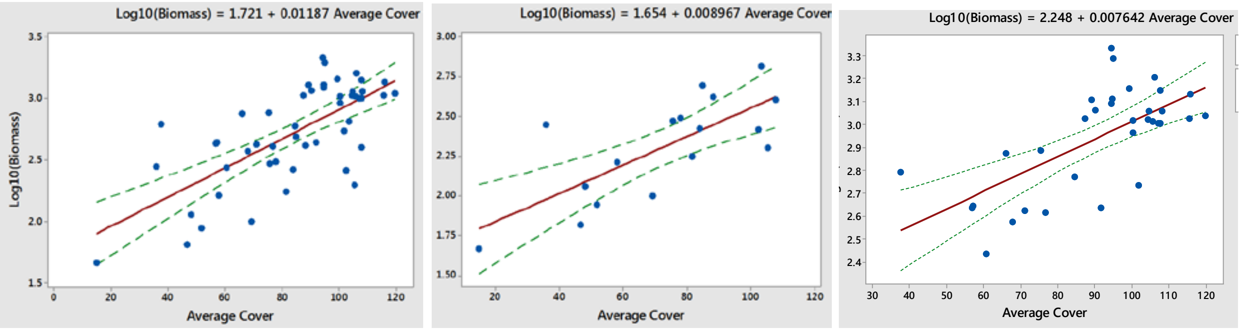


Figure 53. Regression plots comparing average vegetation cover and vegetation biomass for all strata (left), low marsh strata (middle), and high marsh strata (right). Biomass was transformed by Log10 for statistical analysis. The green dashed lines represent the 95% confidence interval.

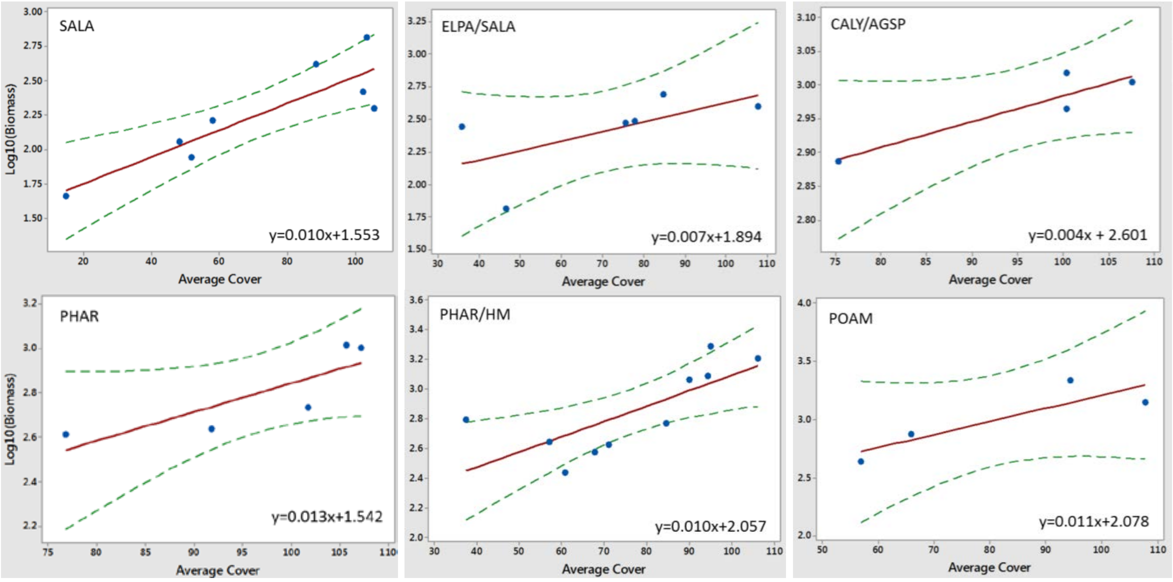


Figure 54. Regression plots comparing average vegetation cover and vegetation biomass for specific strata. Biomass was transformed by Log10 for statistical analysis. The green dashed lines represent the 95% confidence interval.

3.4.1.2 *Pelagic*

Phytoplankton biomass at each of the trends sites was estimated by the concentrations of chlorophyll *a* (chl). The highest chl concentrations were observed at Franz Lake Slough in May (38.9 mg L^{-1}) and June (46.8 mg L^{-1}) (Figure 55), which is consistent with the pigment data produced by the YSI sonde. These high values far exceeded values observed at the other sites. During the spring and early summer (March-June), chl was higher in Reaches C-H compared to Reaches A and B, whereas by August-September, chl was highest in Reaches A and B. The criterion set for high water quality for non-stratified lakes and rivers in the State of Oregon is a depth-averaged, 3-consecutive-month average chl concentration less than 15 µg L^{-1} (Environmental Protection Agency/Department of Environmental Quality). Since the high chl values observed at Franz Lake Slough only persisted for two months, water quality was not considered impaired in 2016.

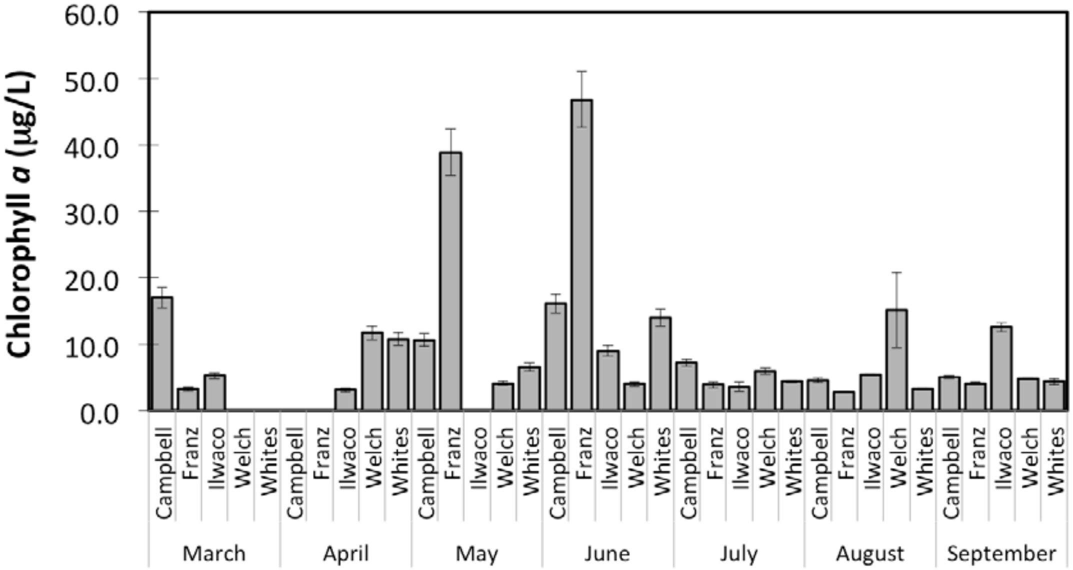


Figure 55. Chlorophyll *a* concentrations determined from grab samples at the trends sites (Campbell Slough, Franz Lake Slough, Ilwaco Slough, Welch Island, and Whites Island).

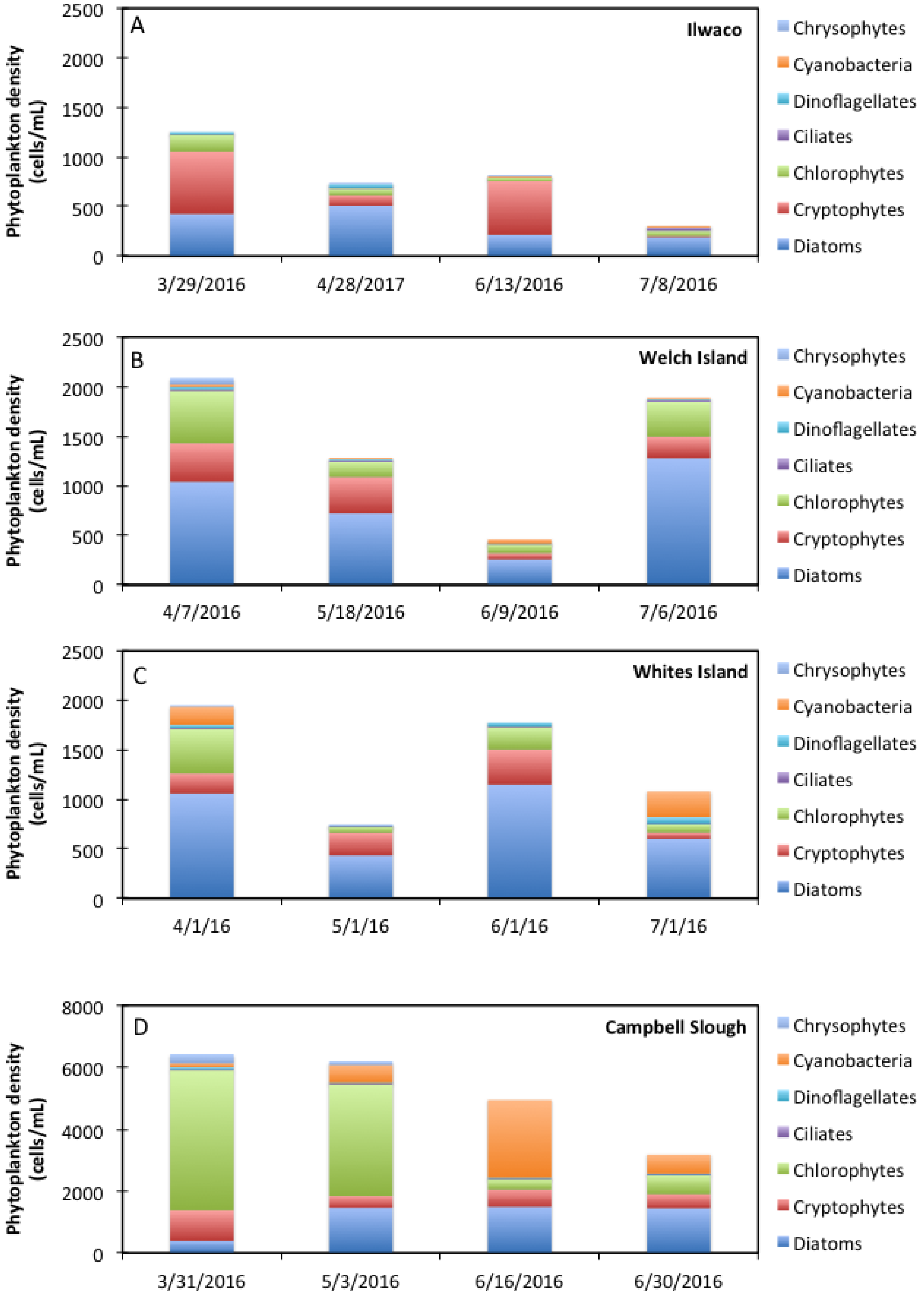
3.4.1.3 *Phytoplankton species composition*

The densities of various 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 included, since there are some species that can be photosynthetic (e.g., *Mesodinium rubrum*; Lindholm 1985 #2820; Herfort et al. 2011a, 2011b).

Among the trends sites, Ilwaco Slough had the lowest densities of phytoplankton, which were numerically dominated by cryptophytes in April and June (Figure 56A). Diatoms were numerically dominant in May and July and included mainly benthic forms (*Acnanthes* spp., *Bacillaria* spp., *Cocconeis* spp., *Nitzschia capitoradiata*, and several pennates that could not be identified using light microscopy) with the centric, *Melosira* sp., becoming more abundant in June-July.

In contrast to Ilwaco Slough, temporal variability in phytoplankton densities were larger at the other sites (Figure 56). At Welch Island, phytoplankton densities in early spring and summer were higher than the transition periods before and after the freshet (early May). At Whites Island, phytoplankton densities decreased during the freshet period, but increased again by early June. Summer values were similar to the freshet period. At both Welch and Whites Island, the phytoplankton assemblage was dominated by diatoms in the spring and summer, and while cyanobacteria did not make a large contribution to total phytoplankton densities at Welch Island, summertime increases in cyanobacteria abundance were observed at Whites Island.

The phytoplankton assemblages at Campbell Slough and Franz Lake Slough included greater proportions of flagellates, including chlorophytes, chrysophytes, and cryptophytes, compared to Welch and Whites. In addition, these sites—particularly Franz Lake Slough—had high abundances of cyanobacteria during the summer (June and July). Surprisingly, cyanobacteria were noted at Whites Island during the period associated with the spring bloom (April, pre-freshet), and their abundances were already increasing at Campbell Slough by early May (Figure 56).


A
Ilwaco
B
Welch Island
C
Whites Island
D
Campbell Slough
Phytoplankton density (cells/mL)
0
500
1000
1500
2000
2500
4000
6000
8000
3/29/2016
4/28/2017
6/13/2016
7/8/2016
4/7/2016
5/18/2016
6/9/2016
7/6/2016
4/1/16
5/1/16
6/1/16
7/1/16
3/31/2016
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Cyanobacteria
Dinoflagellates
Ciliates
Chlorophytes
Cryptophytes
Diatoms

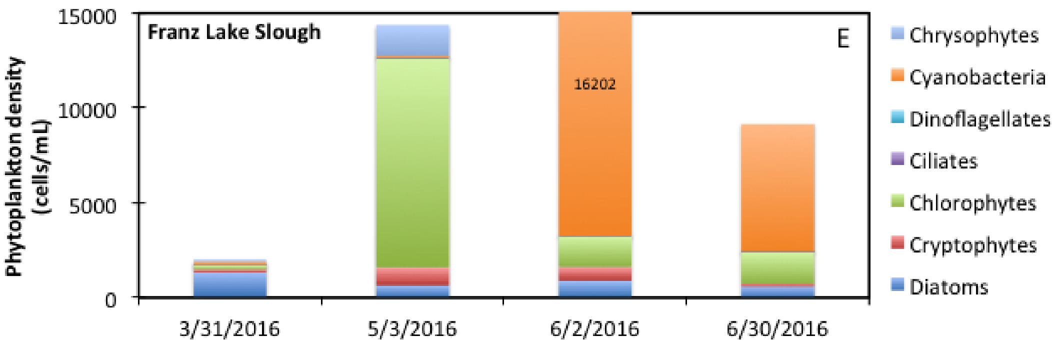


Figure 56. Densities of different phytoplankton groups (diatoms, cryptophytes, chlorophytes, ciliates, dinoflagellates, cyanobacteria, and chrysophytes) at five trends sites, (A) Ilwaco Slough, (B) Welch Island, (C) Whites Island, (D) Campbell Slough, and (E) Franz Lake Slough. Cyanobacteria were so much more abundant at Franz Lake Slough compared to any other taxonomic group at the sites that the numerical value is included (16202 cells/mL) to make visualizing inter-site comparisons easier by avoiding extension of the y-axis.

3.4.1.4 *Spatiotemporal patterns in the phytoplankton assemblage*

A dendrogram produced by comparing Bray-Curtis similarities among the phytoplankton assemblages at the trends sites shows high similarity between sites in Reaches B and C (Welch Island, Whites Island) and between sites in Reaches F and H (Campbell Slough and Franz Lake Slough). Not surprisingly, similarities were higher among samples collected close in time (Figure 57). Because of the very different environmental conditions at Ilwaco Slough that correspond with ocean influence and tidal fluctuations (high salinity, variable pH and dissolved oxygen), assemblages at that site were different from the other sites and samples obtained from different months differed from each other.

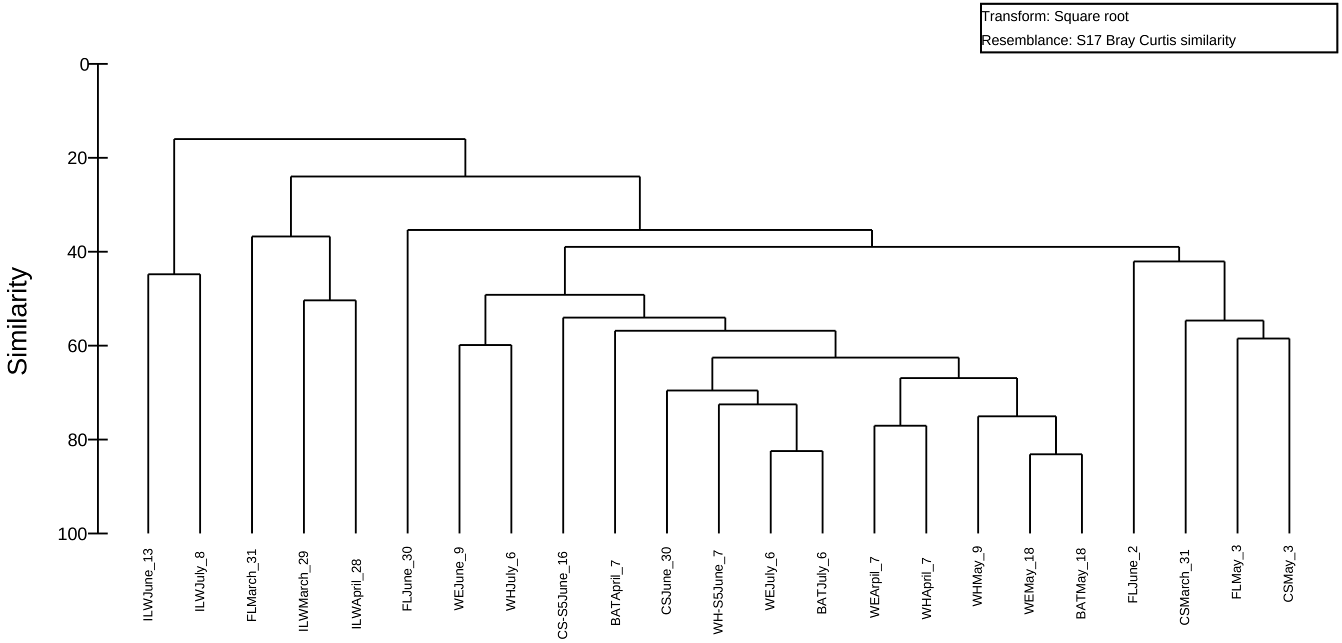


Figure 57. Bray-Curtis similarity among trends sites with respect to phytoplankton assemblages (includes all taxa). Phytoplankton abundance data were square-root transformed prior to clustering. The sites tended to cluster according to season (e.g., samples from April and May were more similar to each other). Samples from Ilwaco Slough were distinctly different from the other sites. BAT = Beaver Army Terminal (RM-53), CS = Campbell Slough, FL = Franz Lake Slough, Ilw = Ilwaco Slough, WE = Welch Island, WH = Whites Island.

After performing a Principal Components Analysis, a small number of diatom species were found to account for significant proportions of variability in the phytoplankton data. Three of the species contributing to the first three principal components (as determined by eigenvector values) were *Asterionella formosa*, *Navicula capioradiata*, 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). In contrast, *N. capitoradiata* is a benthic species that is very common at Ilwaco Slough. In MultiDimensional Scaling (MDS) plots, the different niches occupied by these three diatom species emerge. *A. formosa* tends to be more abundant and dominant in the early spring (February–April), while *S. potamos* tends to become abundant later in the season (Figure 58).

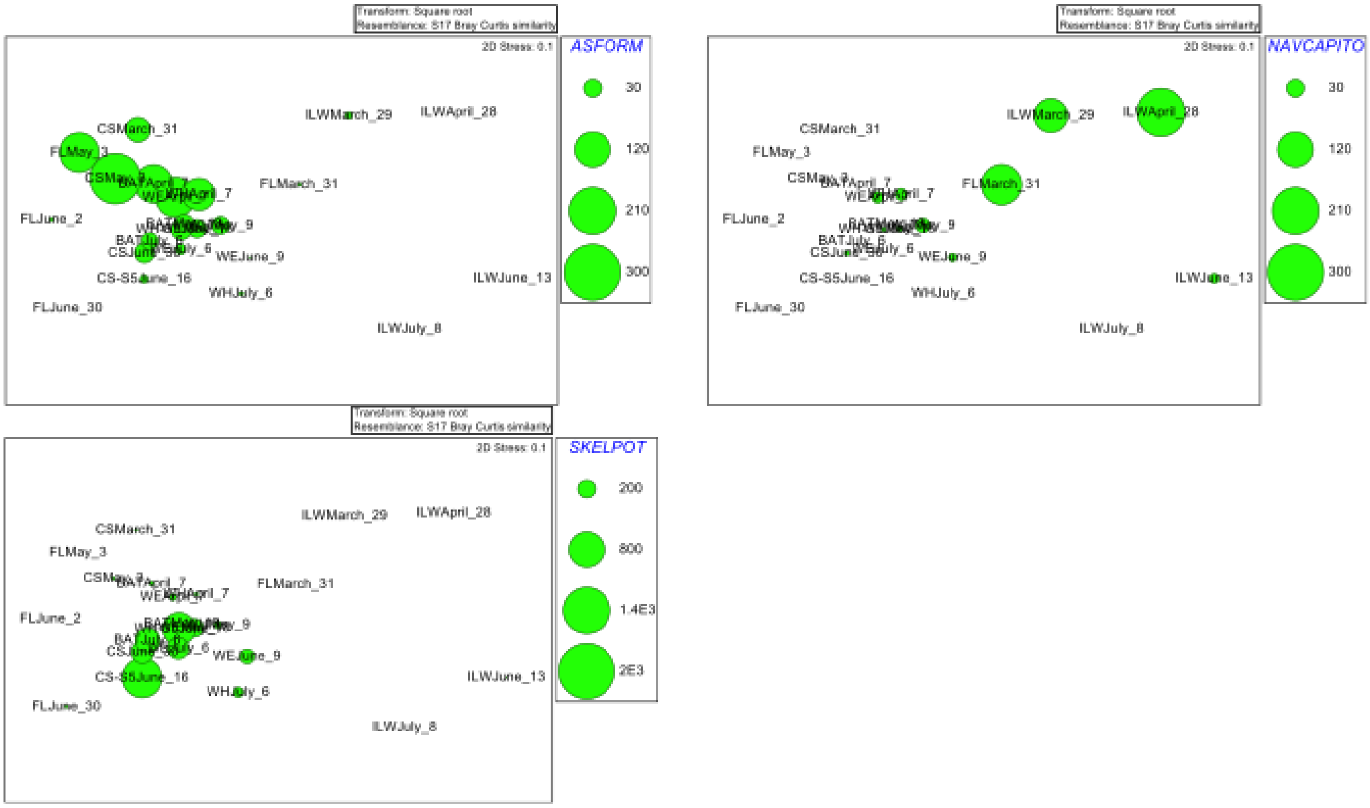


Figure 58. Multidimensional Scaling (MDS) plots showing similarities (as determined by the Bray-Curtis similarity metric) among sites for three diatom species accounting for variability in the phytoplankton data set. ASFORM = Asterionella formosa, NAVCAPITO = *Navicula capitoradiata*, and SKELPOT = *Skeletonema potamos*.

3.4.2 Spring zooplankton assemblages

As in previous years, the abundances of zooplankton were far higher at Campbell Slough than at the other trends sites (Figure 59). We noted that copepods (including adults and nauplii) were heavily parasitized at Campbell Slough in early May, 2016. There was an absence of cladocerans at Ilwaco Slough and Welch Island in late March/early April. Rotifers were abundant and diverse at all sites in the spring.

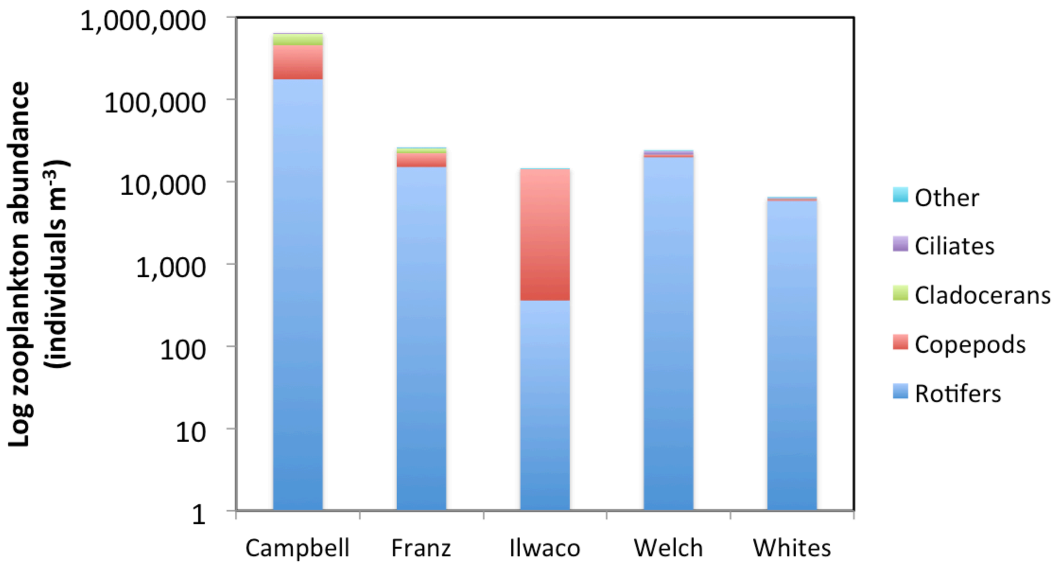


Figure 59. Abundances of zooplankton groups at the five trends sites (Campbell Slough, Franz Lake Slough, Ilwaco Slough, Welch Island, and Whites Island) in spring (late March/early April, 2016). Other included nematodes, larval chironomids, polychaetes, tardigrades, and ostracods.

3.4.3 Stable Isotope Ratios of Carbon and Nitrogen

3.4.3.1 *Wild vs. hatchery reared fish*

Stable isotope ratios of carbon ($^{13}C/^{12}C$) and nitrogen ($^{15}N/^{14}N$) were determined on fish muscle collected from multiple trends sites in May and June 2016. There were significant differences between average values of $\delta^{13}C$ and $\delta^{15}N$ between wild and hatchery reared fish, with wild fish being lighter in ^{13}C and heavier in ^{15}N ($p<0.01$; Figure 60). An analysis of C and N isotopes of hatchery food from Warm Springs, Willard, Little White Salmon, Carson, and Spring Creek National Fish Hatcheries (NFH) showed that the average $\delta^{13}C$ was -21.80 ±0.20 ‰ (i.e., heavier than the wild fish) and the average $\delta^{15}N$ was 9.15 ±0.88 ‰ (i.e., lighter than the wild fish), which could account for the difference between hatchery and wild fish.

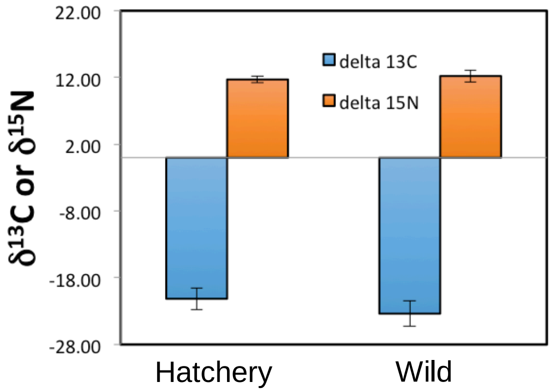


Figure 60. Stable isotope ratios of carbon ($\delta^{13}C$) and nitrogen ($\delta^{15}N$) for hatchery reared and wild fish. Differences between wild and hatchery reared fish in both $\delta^{13}C$ and $\delta^{15}N$ were significant ($p< 0.01$).

3.4.3.2 *Temporal differences in stable isotope ratios*

Fish. When all fish were included (i.e., wild and hatchery reared), the average stable isotope ratios of carbon and nitrogen in muscle from juvenile Chinook salmon differed between early May and early June. In May, average δ^{13}C was -23.15 ±2.0 ‰ (n=30), while in June it was -21.99 ±2.1 ‰ (n=39). The average δ^{15}N values for May and June were 12.24 ±1.0 ‰and 11.83 ±0.5 ‰, respectively.

Organic matter sources. There were larger seasonal differences in isotopic signatures of carbon among organic matter sources to the food web compared to nitrogen. Average isotopic signatures for the group of pooled invertebrates were -25.56 ±5.70 ‰ and 7.11 ±1.12 ‰ (for carbon and nitrogen, respectively) in May compared to 22.98 ±1.5 ‰ and 7.41 ±0.75 ‰ (for carbon and nitrogen, respectively) in June. Average carbon isotopic signatures of pooled plant groups were heavier in May (-25.87 ±5.1 ‰) compared to June (-26.74 ±4.7 ‰). Larger differences were observed in the nitrogen isotopes relative to the carbon isotopes; in May, the average δ^{15}N of the pooled plant groups was lighter (4.75 ±1.83 ‰) compared to June (6.80 ±1.03 ‰). However, the differences among the plants were driven somewhat by differences in the composition of plants rather than differences in the isotopic signatures of the same plant species. For example, if only the same plant species were compared, with the average pooled values for the four overlapping taxa (*Calex lyngbyei*, *Phalaris arundinaceae*, *Polygonum amphibium*, and *Ulva* sp.), the differences in δ^{13}C and δ^{15}N between spring and summer were smaller (compare δ^{13}C of -25.88 ±5.9 ‰ in May with -25.67 ±6.2 ‰ in June and δ^{15}N of 5.01 ±1.5 ‰ vs. 6.45 ±1.2 ‰). The average isotopic signature of carbon and nitrogen within periphyton was heavier in June compared to May (-25.27 ±4.9 ‰ in May vs. -17.90 ± 6.68 ‰ in June); however, the average δ^{15}N within periphyton was heavier in June compared to May (4.18 ± 2.0 ‰ in the spring vs. 6.68 ± 0.8 ‰ in the summer).

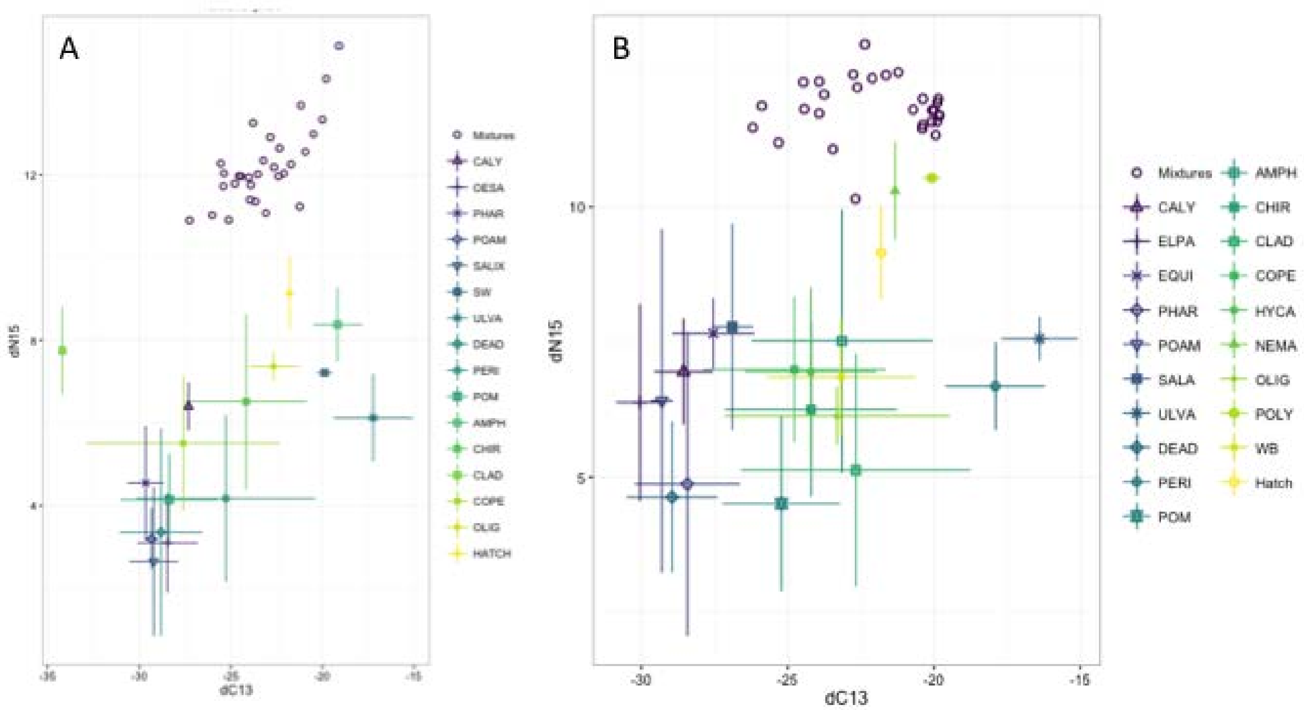


Figure 61. Two plots showing sources isotopic signatures of carbon and nitrogen in "isospace", where values corresponding to delta 15N (=dN15) are plotted against delta ^{13}C (=δ^{13}C). A) May source values, B) June source values. "Mixtures" refers to stable isotope signatures from fish muscle tissue. CALY = *Caryx lyngbyei*, ELPA = *Eleocharis* cf. *palustris*, EQUI = *Equisetum* sp., OESA = *Oenanthe sarmentosa*, PHAR = *Phalaris arundinaceae*, POAM = *Polygonum amphibium*, SALA = *Sagittaria latifolia*, SALIX = *Salix* sp., SW = miscellaneous seaweed, ULVA = *Ulva* sp., DEAD = dead vegetation (mixtures), PERI = periphyton, POM = particulate organic matter, AMPH = amphipods, CHIR = chironomids, CLAD = cladocerans, COPE = copepods, HYCA = *Hydracarina* sp., NEMA = nematodes, OLIG = oligochaetes, POLY = polychaetes, WB = water boatmen, Hatch = hatchery food. These data do not include corrections for isotope fractionation.

In general, carbon isotopes can be used to determine the source of carbon 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‰. From these typical values, it is likely that amphipods, cladocerans, copepods, oligochaetes, nematodes, and waterboatmen are consumers of algae in the summer; in the spring, cladocerans and copepods have more depleted δ^{13}C values, suggesting that other sources, including plant detritus might be important. The average δ^{13}C signature of chironomids was -24.2‰ in both May and June, which is the high end for terrestrial plants.

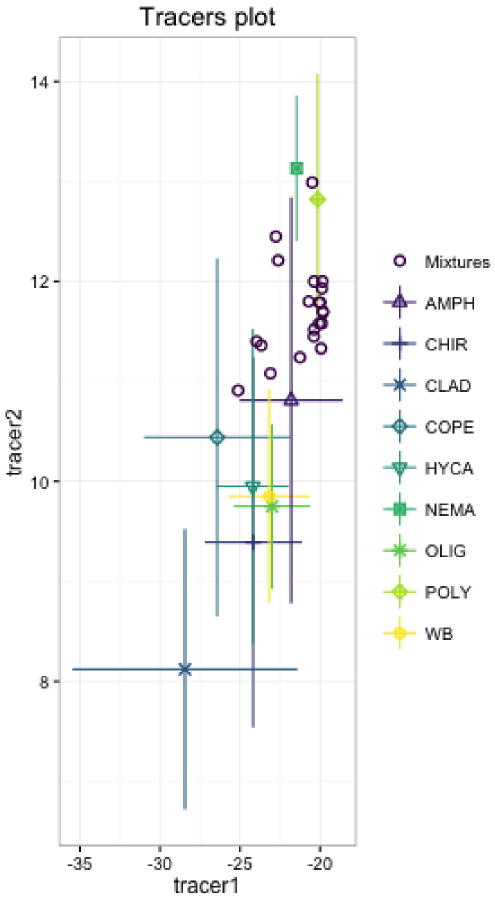


Figure 62. Isospace plot showing isotopic signatures of hatchery fish ("mixtures") and potential food sources, excluding hatchery food. Tracer 1 = $\delta^{13}C$; tracer 2 = $\delta^{15}N$). In this figure, a correction of 3‰ was applied to account for fractionation of N isotopes with increasing trophic level.

For the hatchery fish, preliminary results from the isotope mixing model suggest that there was a preference for amphipods for fish collected in May and June 2016 (Figure 63).

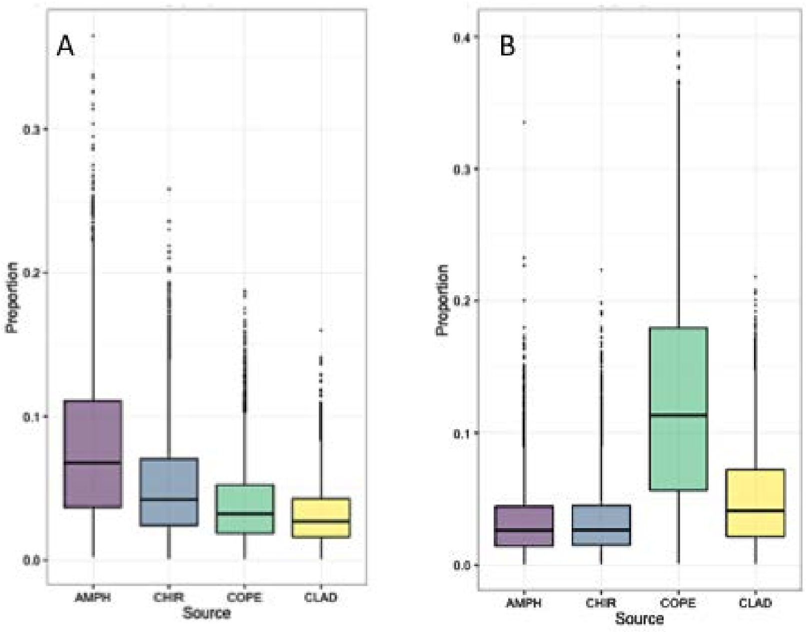
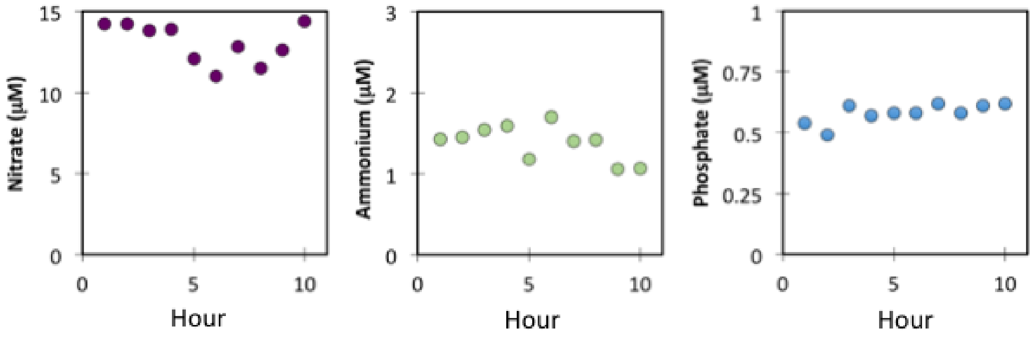


Figure 63. Preliminary results showing a comparison of the proportion of the diet of (A) hatchery reared and (B) wild juvenile salmon captured in May and June 2016 accounted for by amphipods (AMPH), larval chironomids (CHIR), copepods (COPE), and cladocerans (CLAD).

3.4.4 Tidal Sampling

To determine whether tidal flow had any effect on nutrient concentrations, samples were collected hourly at Campbell Slough and Whites Islands in early May and early June at high slack tide prior to the flood. Concentrations of nitrate, ammonium, and phosphate were determined during outwash. At Campbell Slough, nutrient concentrations varied over the tidal cycle to a greater extent in June than in May (Figure 64). In June, there was a clear decrease of nitrate over time, while at the same time phosphate increased.

A) Campbell Slough (5/8/2016)



B) Campbell Slough (6/6/2016)

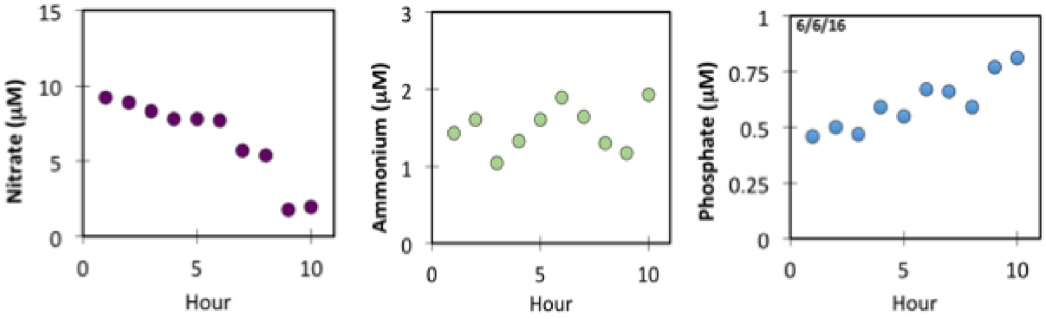


Figure 64. Concentrations of nitrate, ammonium, and phosphate during tidal outwash studies at Campbell Slough on (A) May 8, 2016 and (B) June 6, 2016. Hour 0 corresponds to the first sampling point of the tidal cycle, starting with slack high tide (i.e., prior to flood).

At Whites Island, nitrate increased and both ammonium and phosphate decreased during outwash (Figure 65). The temporal patterns were similar between the two sampling missions. While both ammonium and phosphate were present at similar concentrations in May and June, nitrate was lower in June compared to May. At the same time that nitrate decline, chlorophyll concentrations increased (Figure 66), consistent with the idea that the disappearance of nitrate occurred due to uptake by aquatic biota.

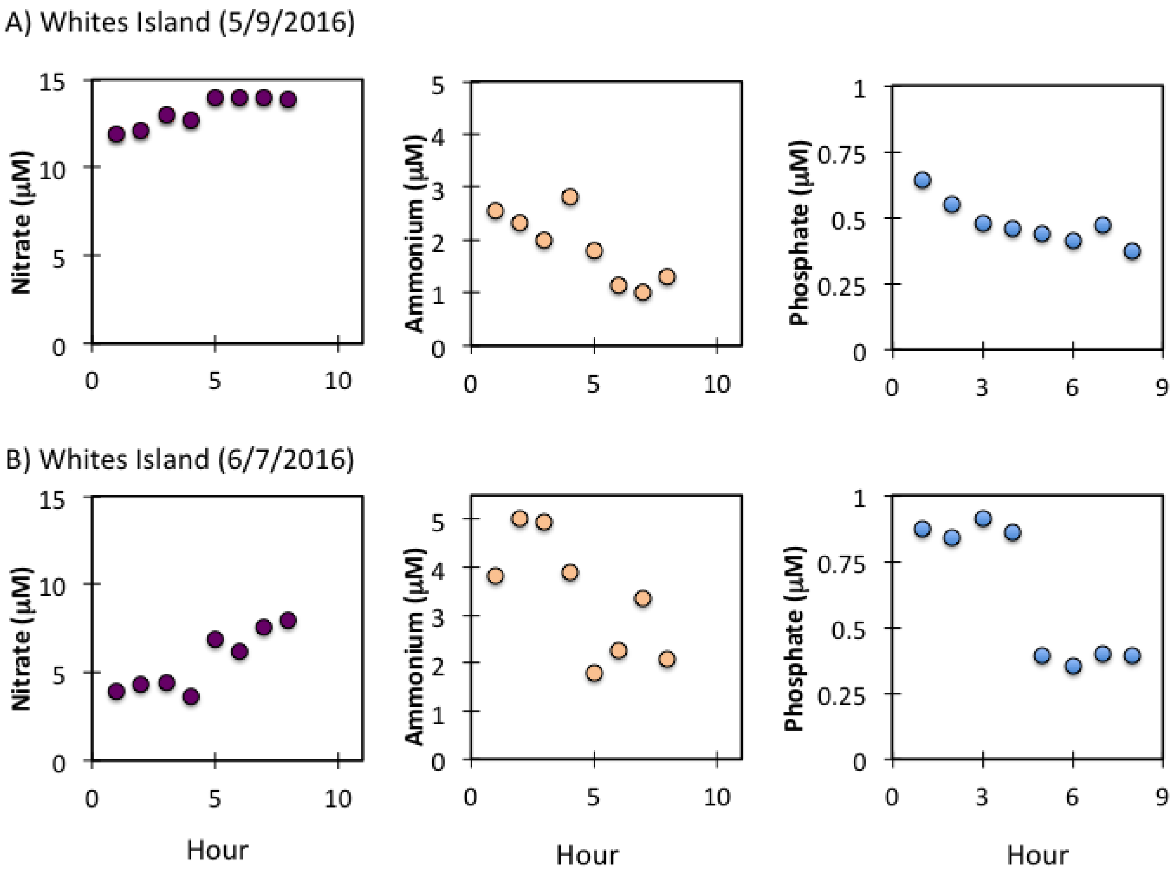


Figure 65. Concentrations of nitrate, ammonium, and phosphate during tidal outwash studies at Whites Island on (A) May 8, 2016 and (B) June 6, 2016. Hour 0 corresponds to the first sampling point of the tidal cycle, starting with slack high tide (i.e., prior to flood).

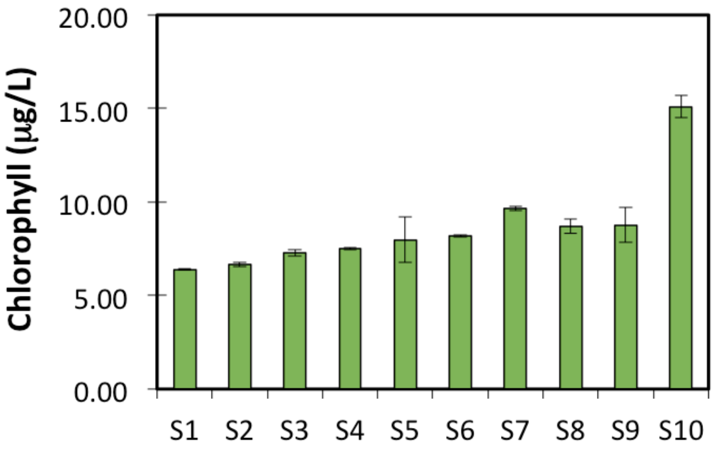


Figure 66. Chlorophyll concentrations during the 2016 tidal sampling at Whites Island in June.

3.5 Macroinvertebrates

3.5.1 Salmon Prey Availability

3.5.1.1 *Benthic*

Benthic core samples from all sites were dominated by nematode and oligochaete worms (Figure 67). 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.

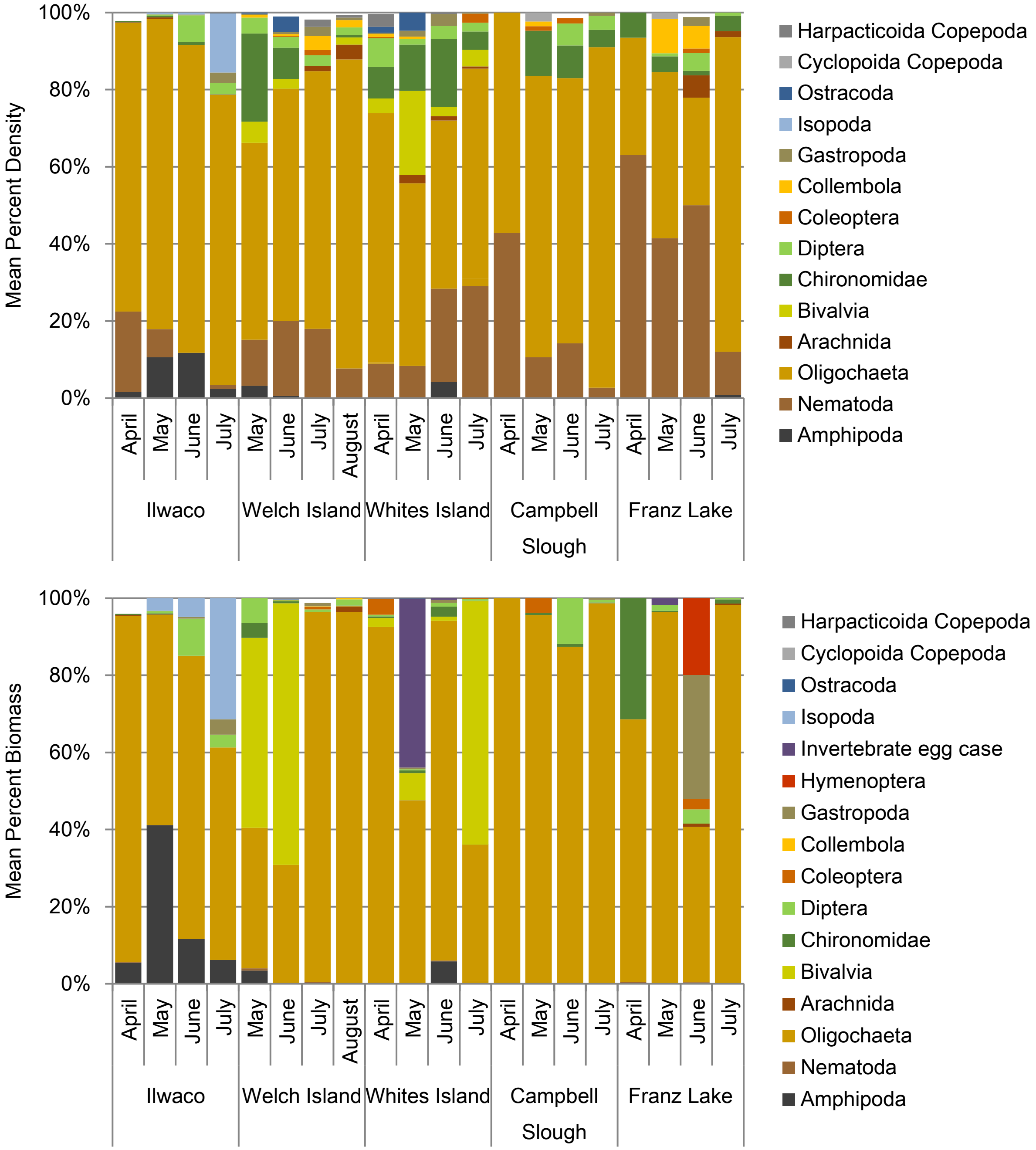


Figure 67. Composition of benthic core samples by mean percent abundance (top) and biomass (bottom) in 2016.

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 68). 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.

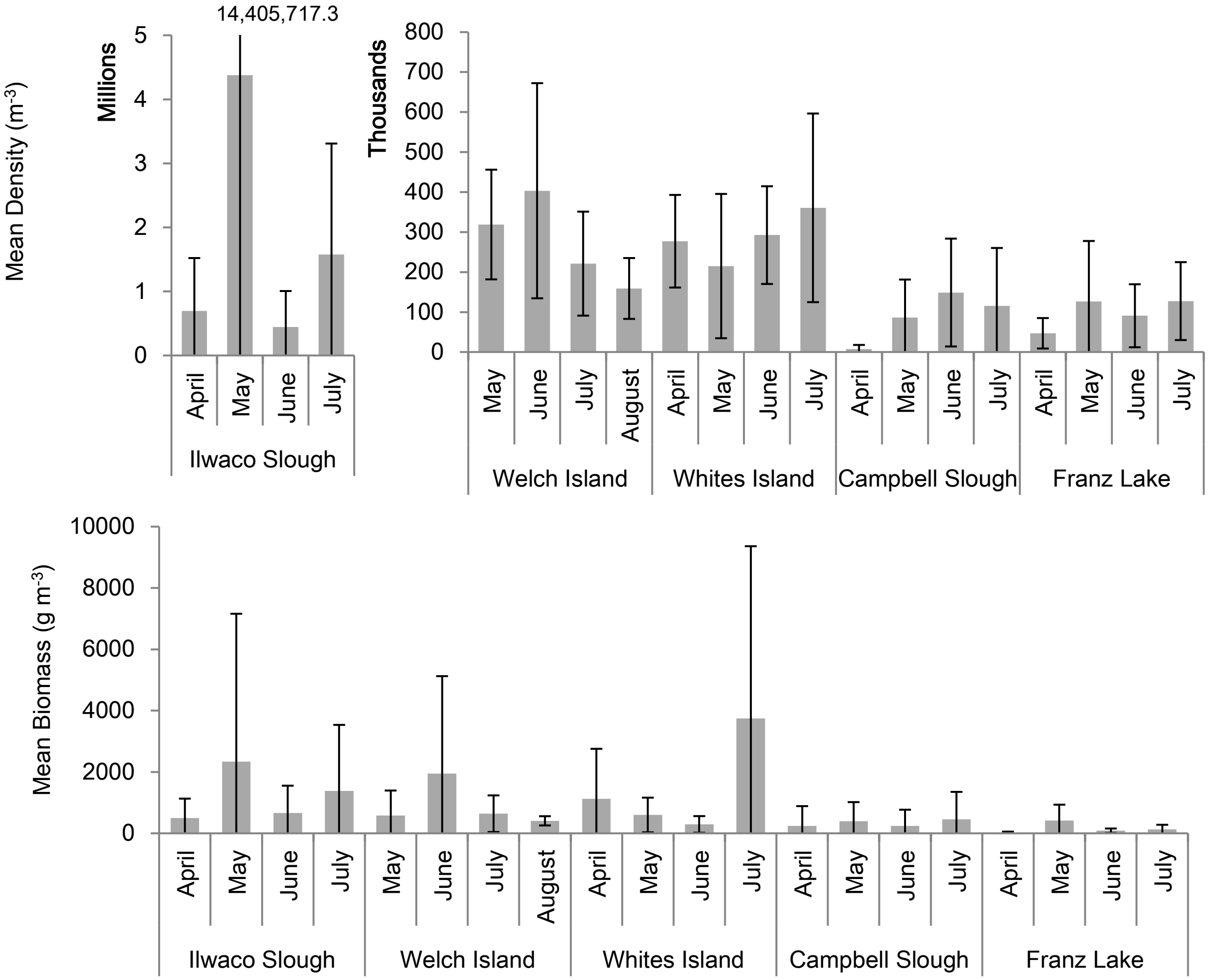


Figure 68. Average density (top) and biomass (bottom) per cubic meter of all invertebrate taxa collected by benthic core in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted. Densities from Ilwaco Slough are shown on a different scale than other sites.

Dipterans were collected from all sites in each month sampled, except Campbell Slough in April (Figure 69). 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.

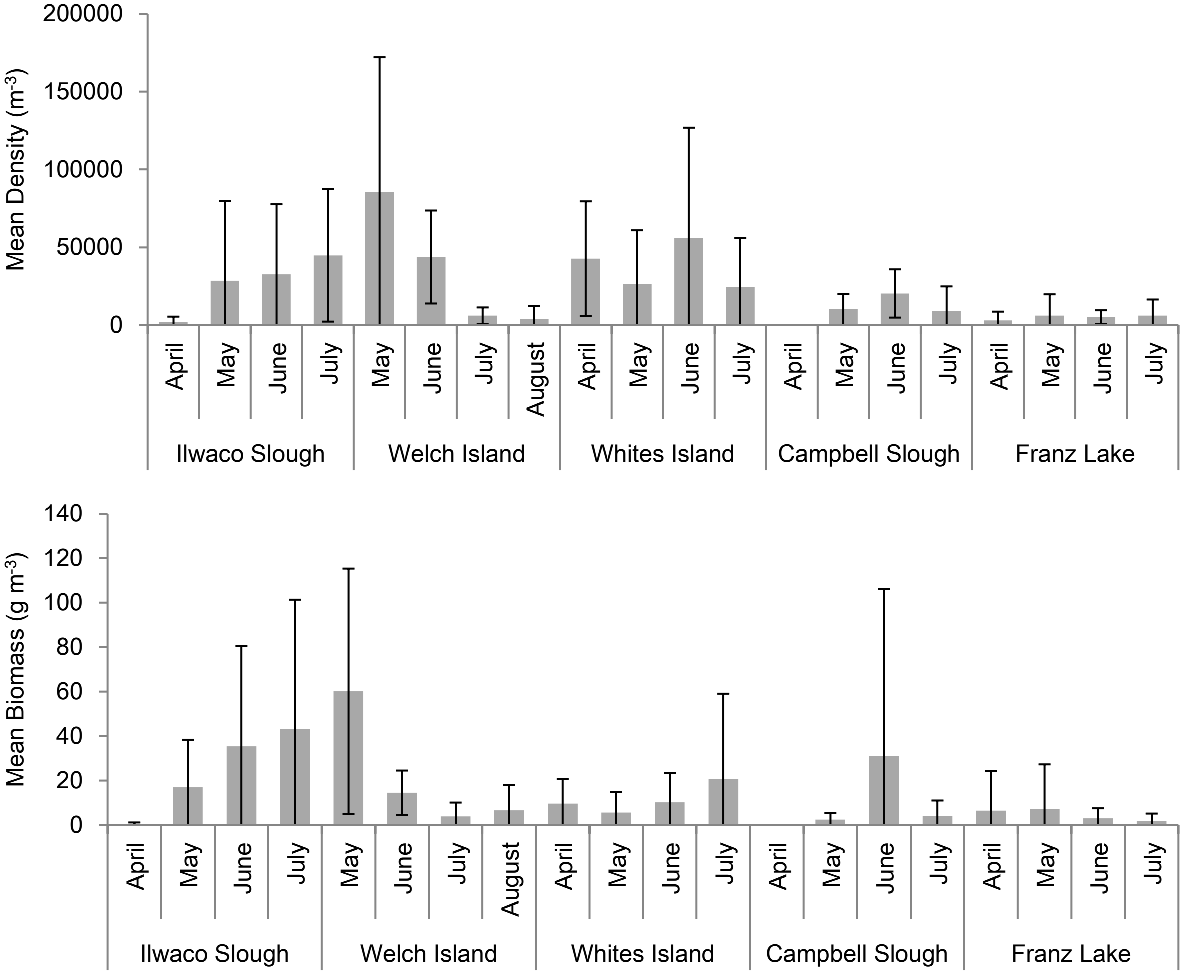


Figure 69. Average density (top) and biomass (bottom) per cubic meter of Diptera (including Chironomidae) collected by benthic core in 2016. Error bars are 95% confidence intervals (CI) around the mean.

Amphipods were abundant at Ilwaco Slough in May, though their occurrence at that time was highly variable (Figure 70). 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.

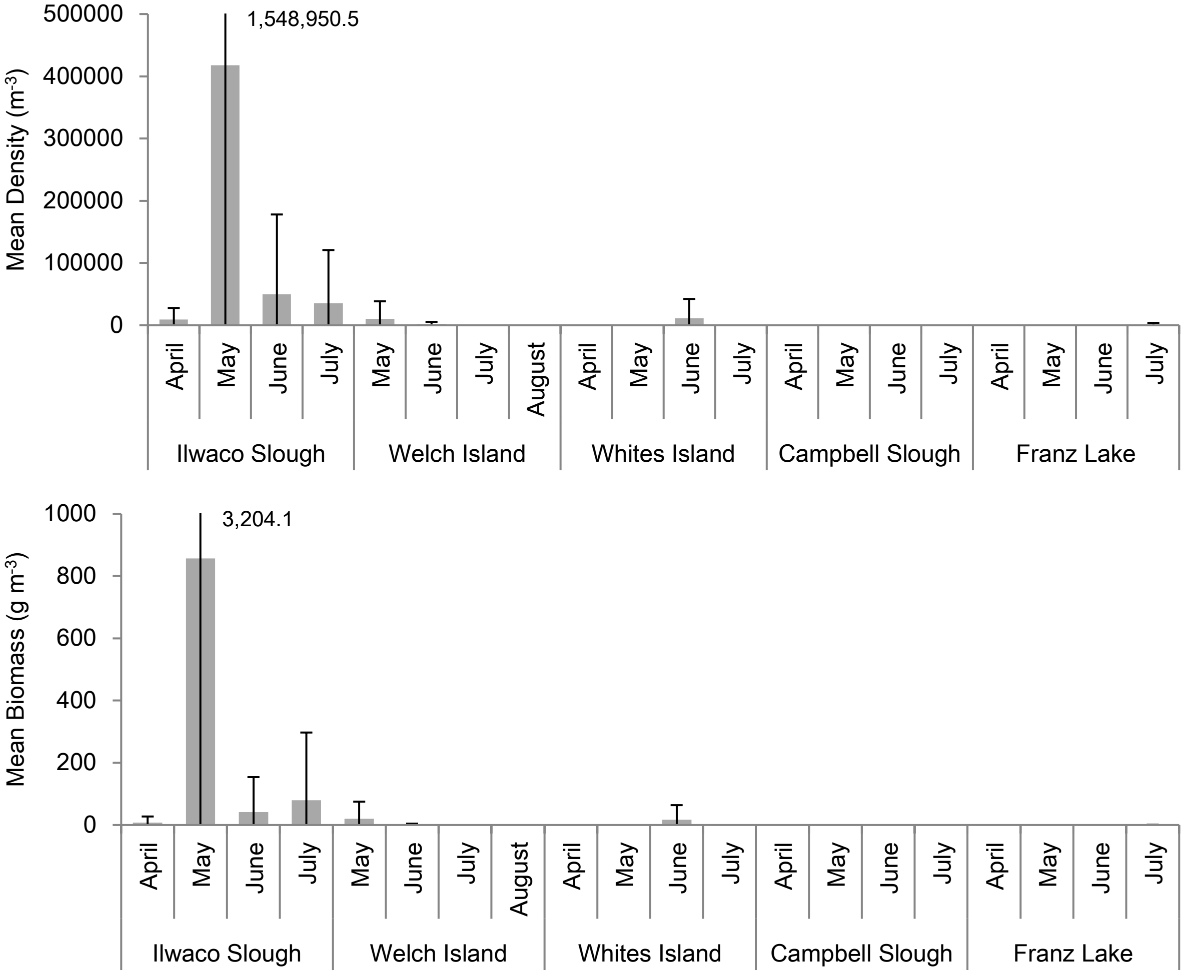


Figure 70. Average density (top) and biomass (bottom) per cubic meter of Amphipoda collected by benthic core in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted.

Average density of all taxa was compared to benthic samples from previous years (Figure 71). 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.

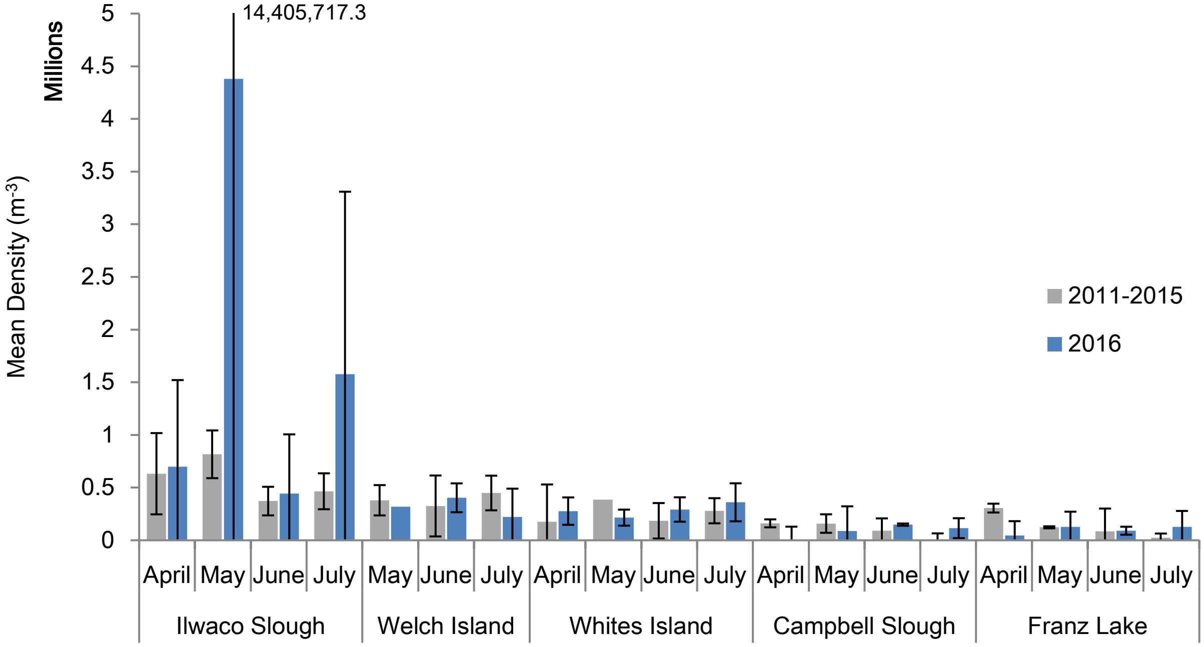


Figure 71. Comparisons between average density (count per cubic meter) summarized between 2011 and 2015 to average density in 2016 of all benthic invertebrate taxa by sample site and month. Error bars represent 95% confidence intervals. Where CI extends off the chart, the maximum range is noted.

3.5.1.2 *Neuston*

Neuston samples were composed of a diverse array of benthic/epibenthic, terrestrial riparian, and planktonic taxa (Figure 72). 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.

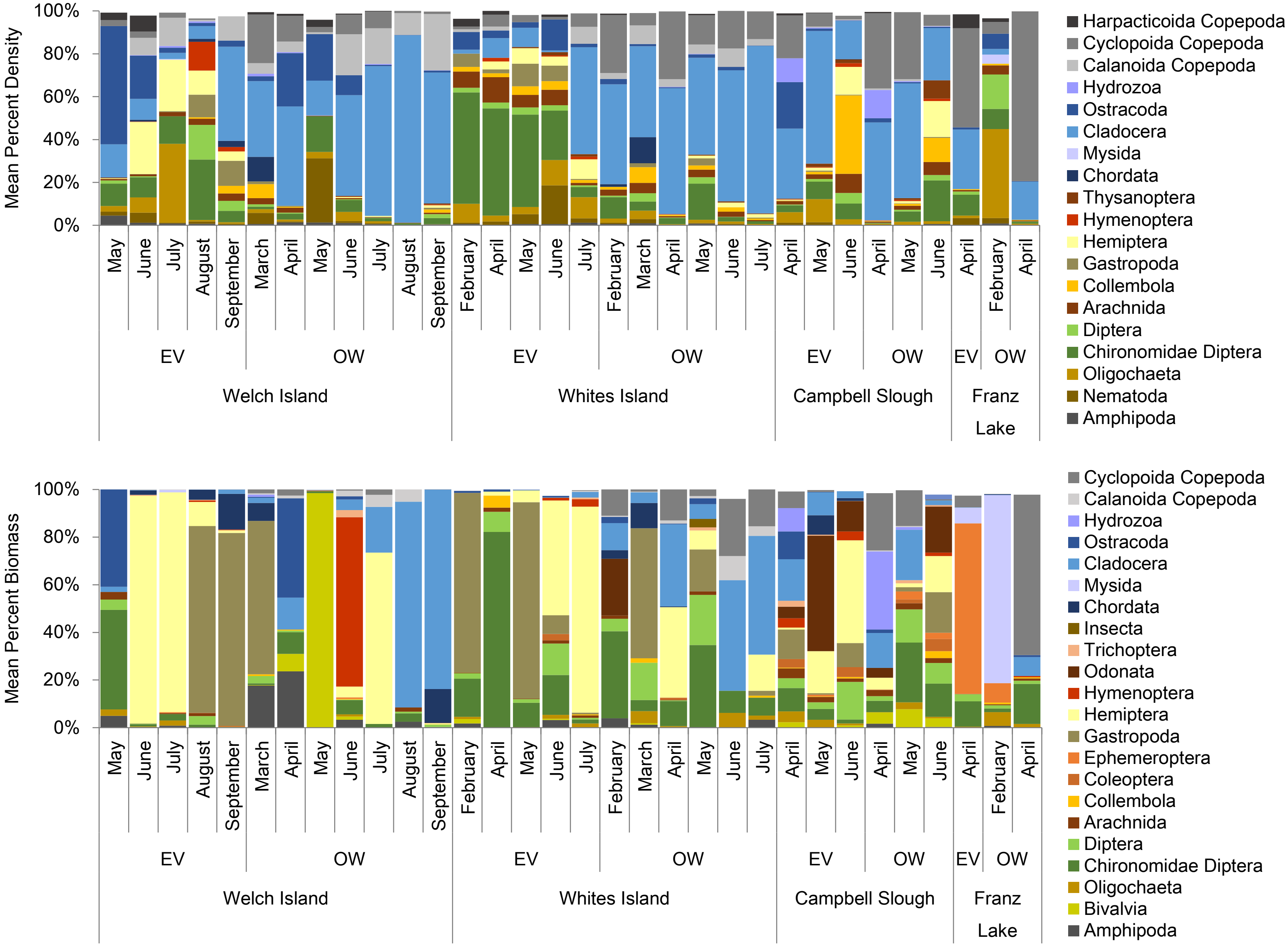


Figure 72. Composition of neuston tows in emergent vegetation (EV) and open water (OW), by mean percent abundance (top) and biomass (bottom) in 2016.

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 73). 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 73). 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.

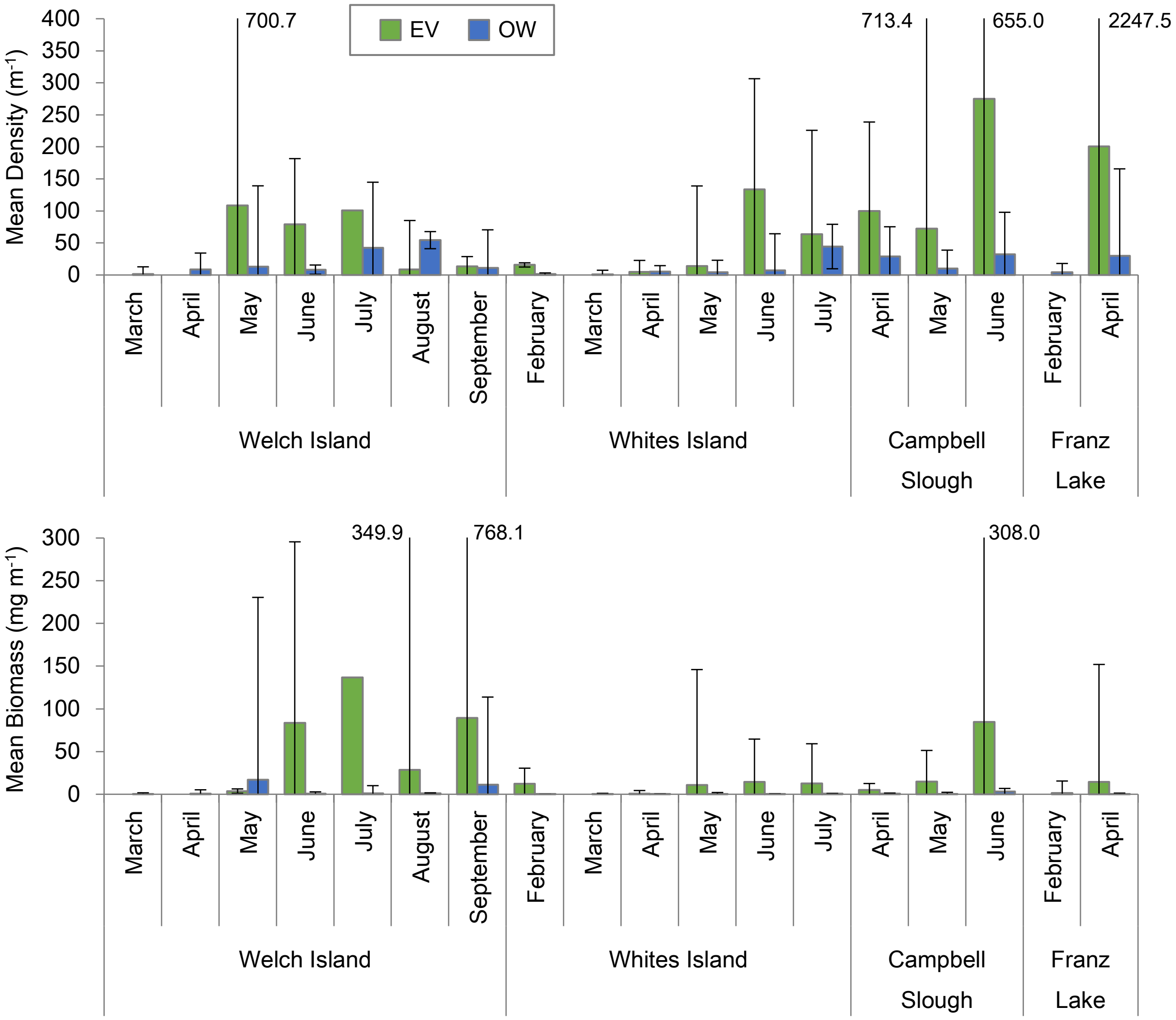


Figure 73. Average density (top) and biomass (bottom) per meter towed of all invertebrate taxa collected by neuston tow in emergent vegetation (green) and open water (blue) in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted. 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 74). 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.

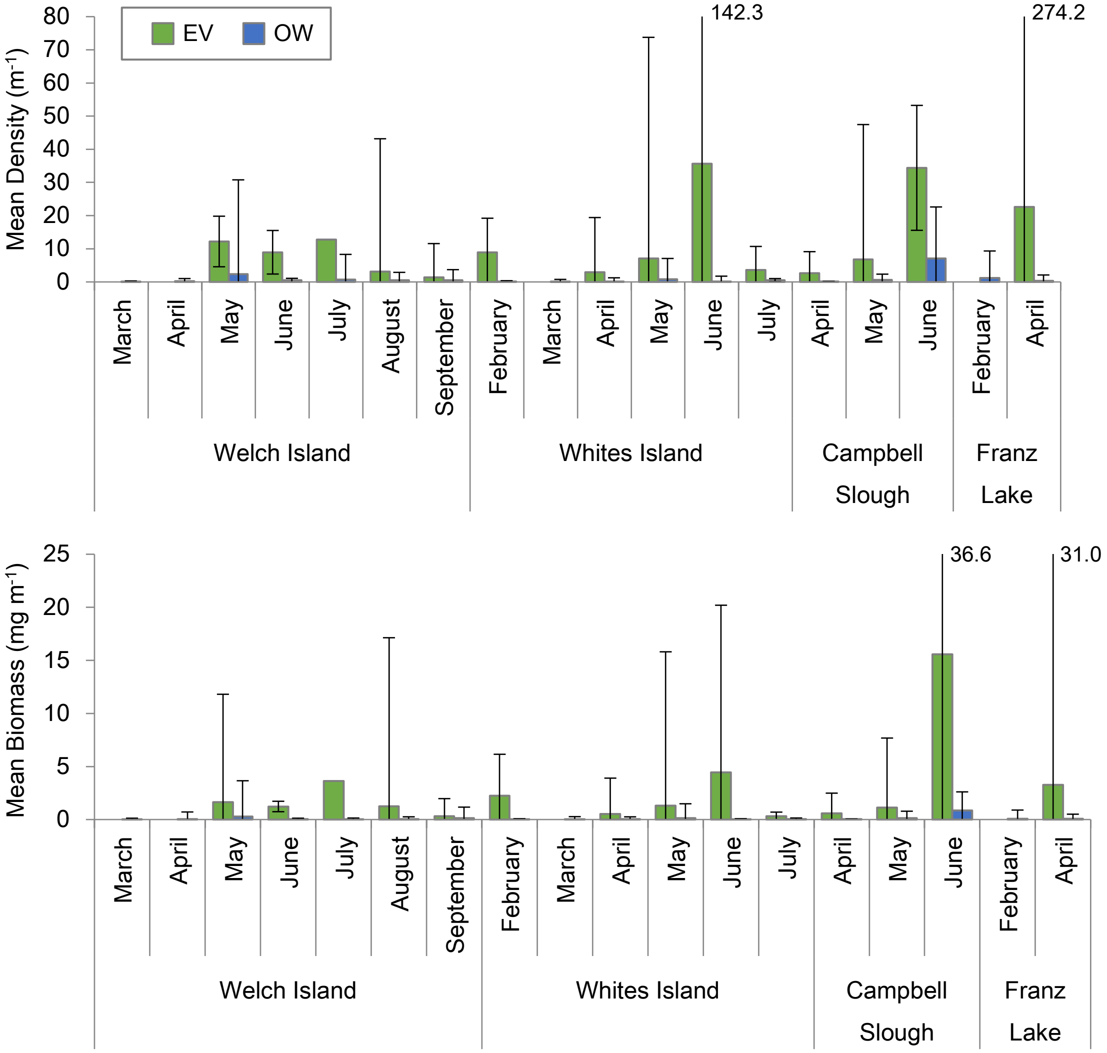


Figure 74. Average density (top) and biomass (bottom) per meter towed of Diptera (including Chironomidae) collected by neuston tow in emergent vegetation (green) and open water (blue) in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range 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 75). 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.

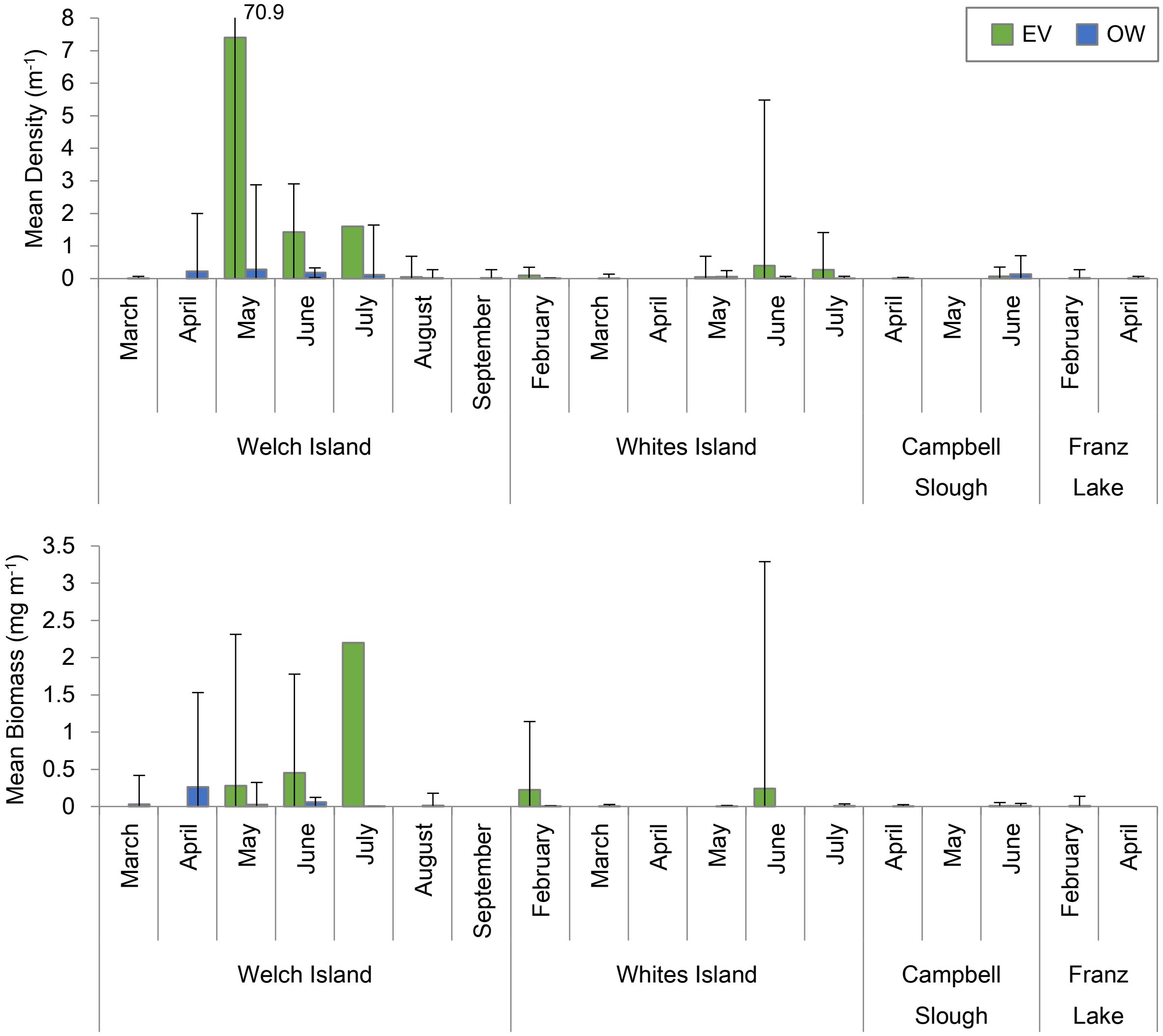


Figure 75. Average density (top) and biomass (bottom) per meter towed of Amphipoda collected by neuston tows in emergent vegetation (green) and open water (blue) in 2016. Error bars are 95% confidence intervals (CI) around the mean. Where CI extends off the chart, the maximum range is noted. Emergent 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 76). 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.

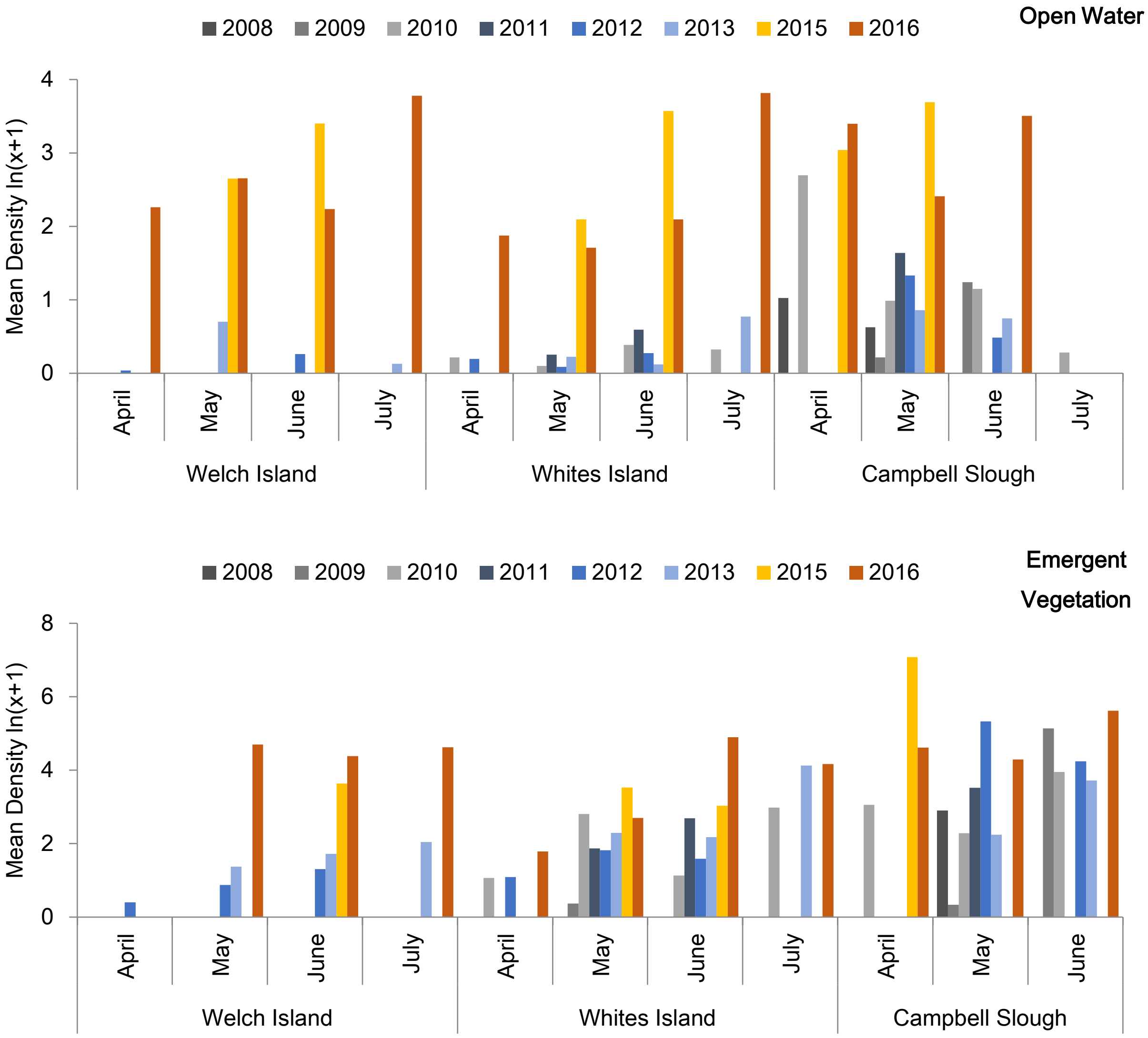


Figure 76. Comparisons across years of log-transformed average density per meter towed of all invertebrate taxa collected by neuston tow in open water (top) and emergent vegetation (bottom).

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 77). Significant differences in cladoceran densities were found between years for open water samples (p=0.001; Figure 78). 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.

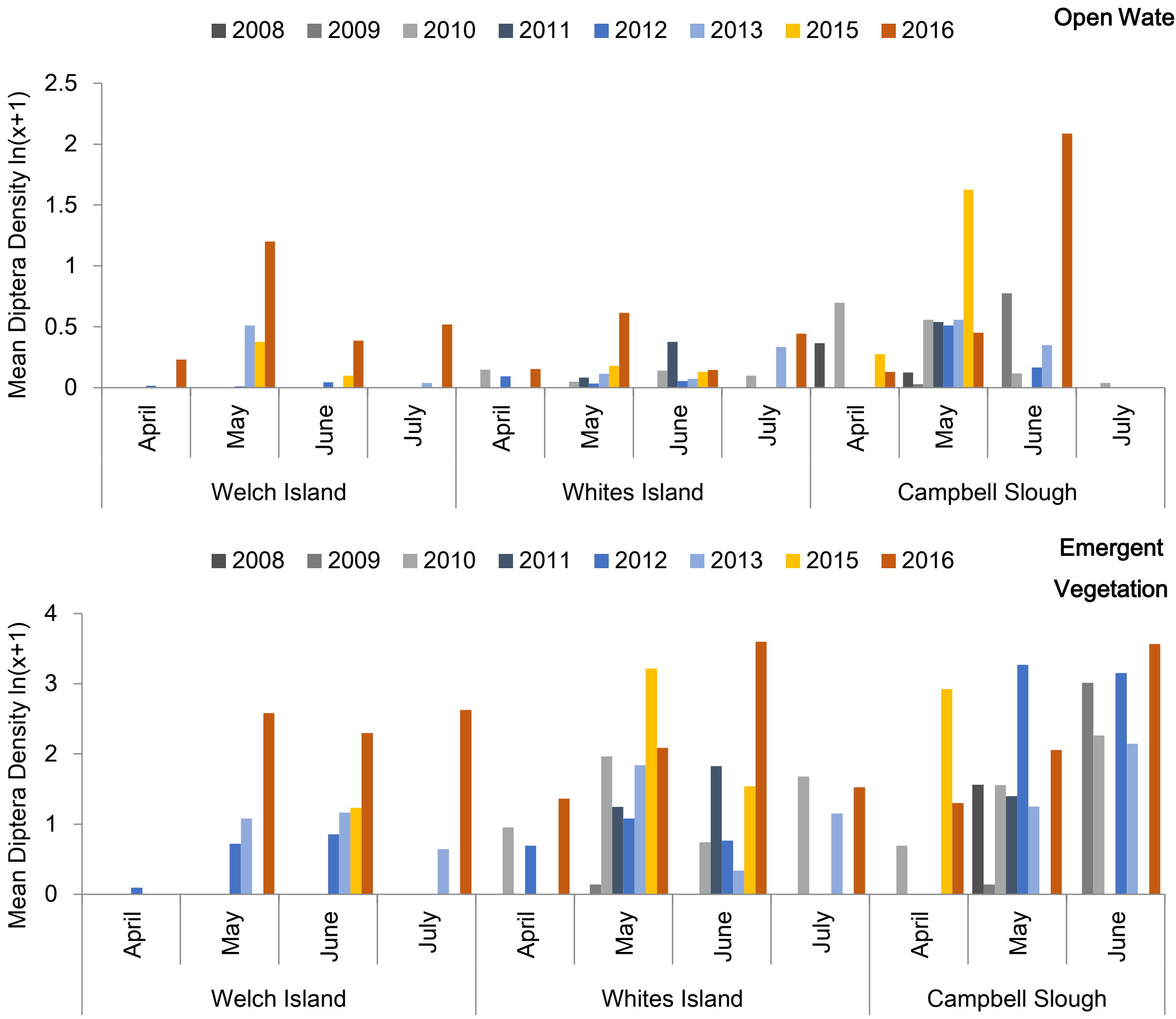


Figure 77. Comparisons across years of log-transformed average density per meter towed of all Diptera taxa collected by neuston tow in open water (top) and emergent vegetation (bottom).

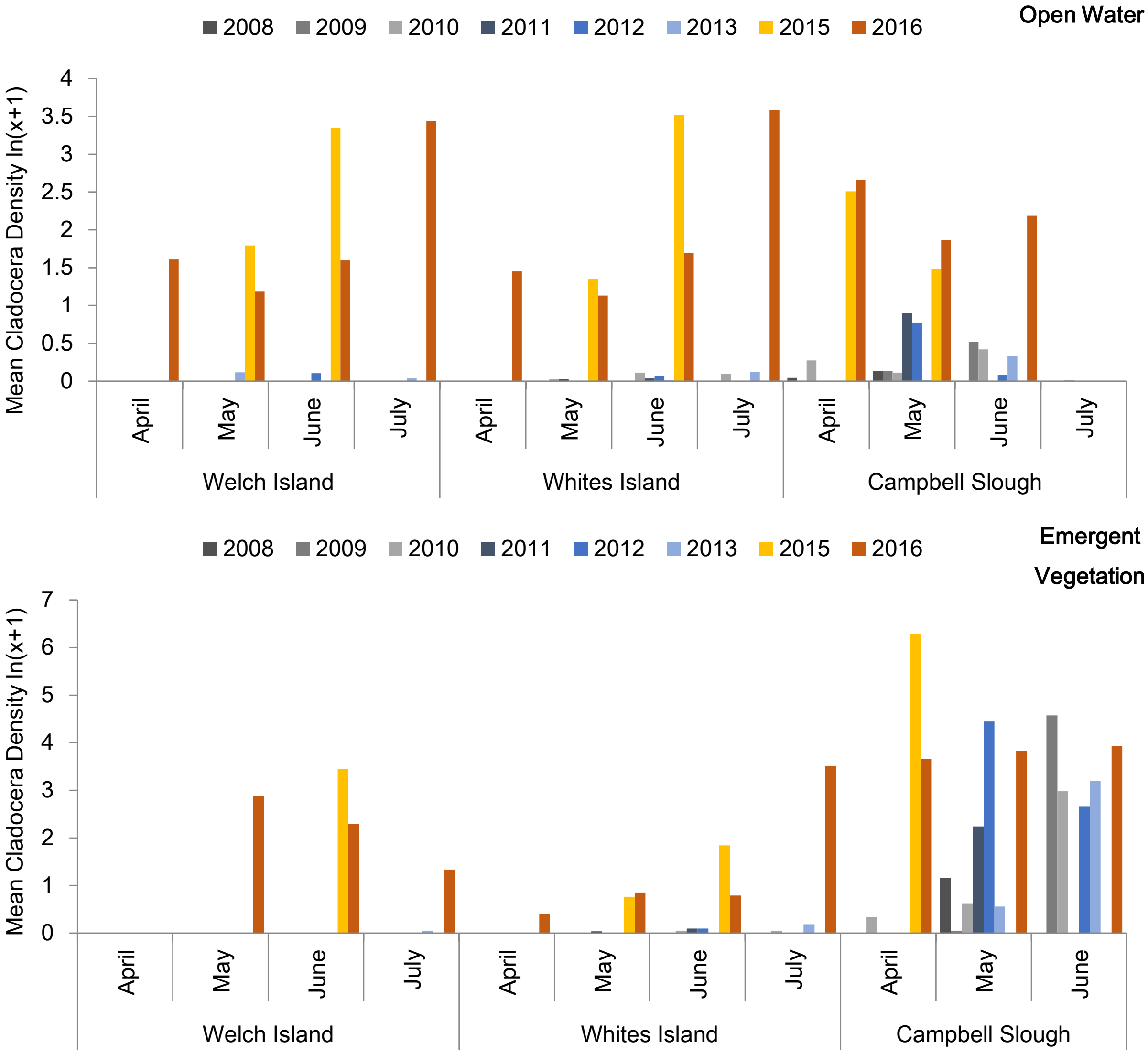


Figure 78. Comparisons across years of log-transformed average density per meter towed of all Cladocera taxa collected by neuston tow in open water (top) and emergent vegetation (bottom).

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 79).

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 80).

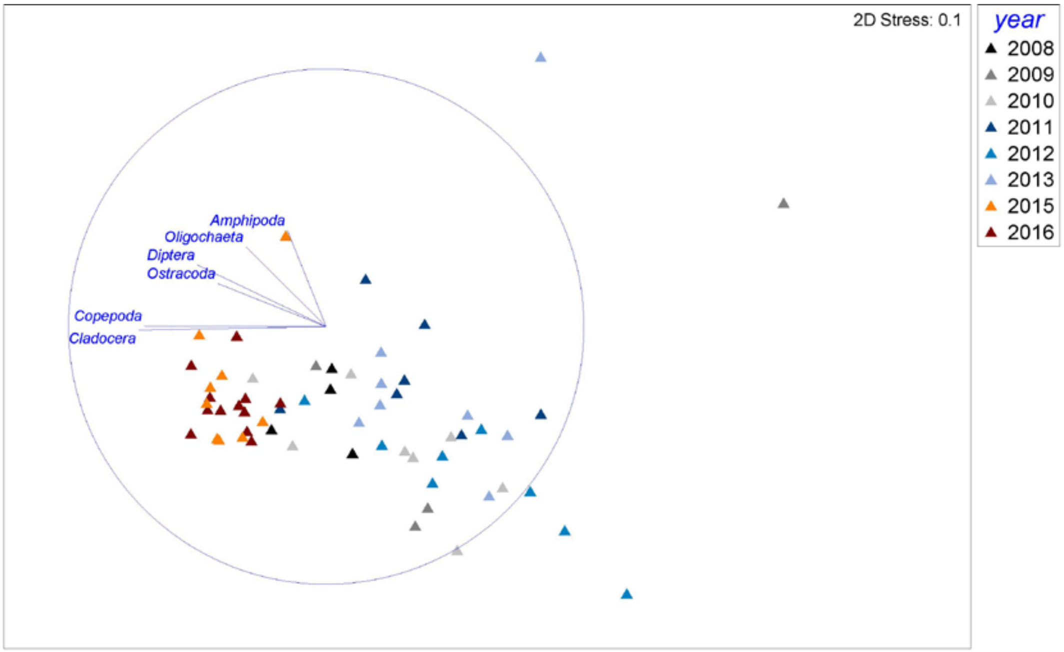


Figure 79. Nonmetric multidimensional scaling (NMDS) plot based on Bray-Curtis similarities between log transformed average monthly densities of taxa collected by open water neuston tows between 2008 and 2016. Significant correlation with variables (Pearson R > 0.4) are represented as blue vectors.

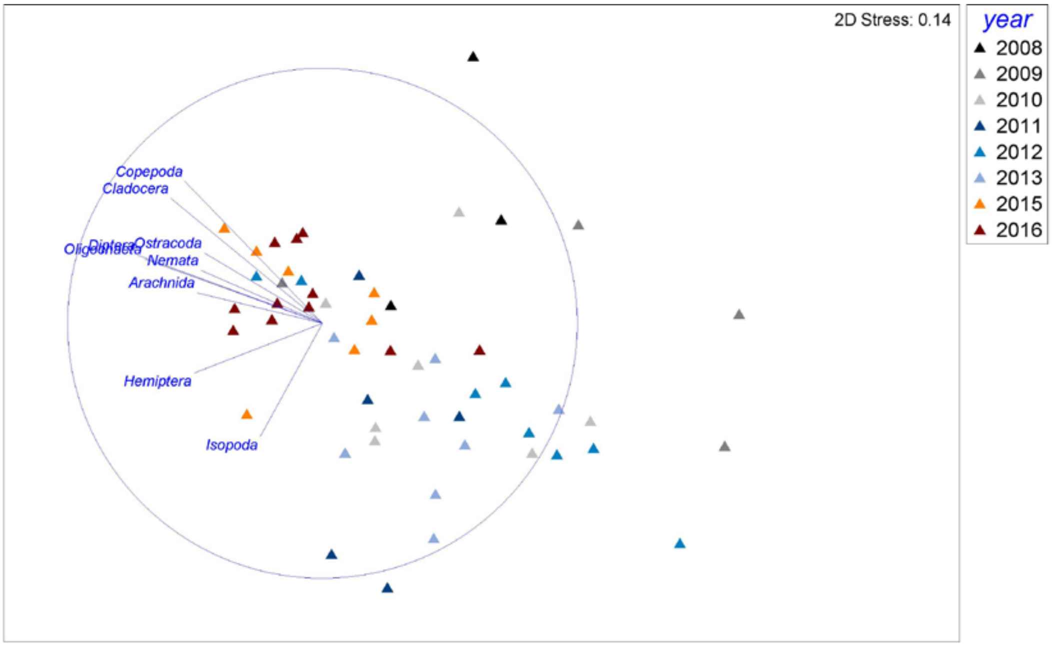


Figure 80. Nonmetric multidimensional scaling (NMDS) plot based on Bray-Curtis similarities between log transformed average monthly densities of taxa collected by emergent vegetation neuston tows between 2008 and 2016. Significant correlation with variables (Pearson R > 0.5) are represented as blue vectors.

3.5.2 Salmon Diet

Salmon diet composition, as measured by an index of relative importance (IRI), varied within site across months as well as across sites (Figure 81). 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.

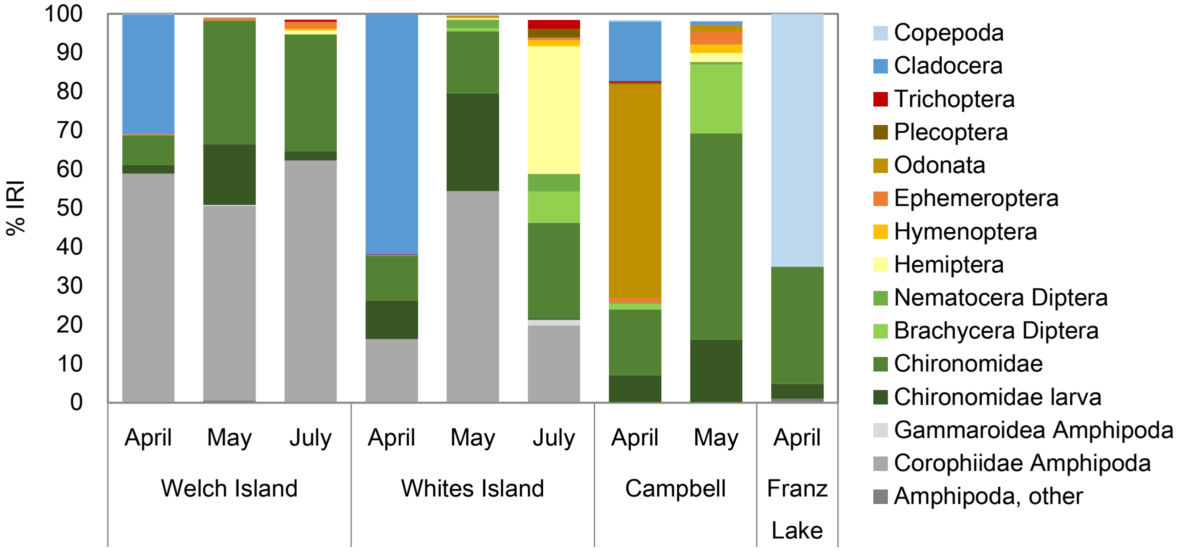


Figure 81. Contribution of prey to juvenile Chinook salmon diets by percent IRI in 2016.

Percent IRI in 2016 was compared to juvenile Chinook diets sampled in previous years (2008-2013, and 2015; Figure 82). 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.

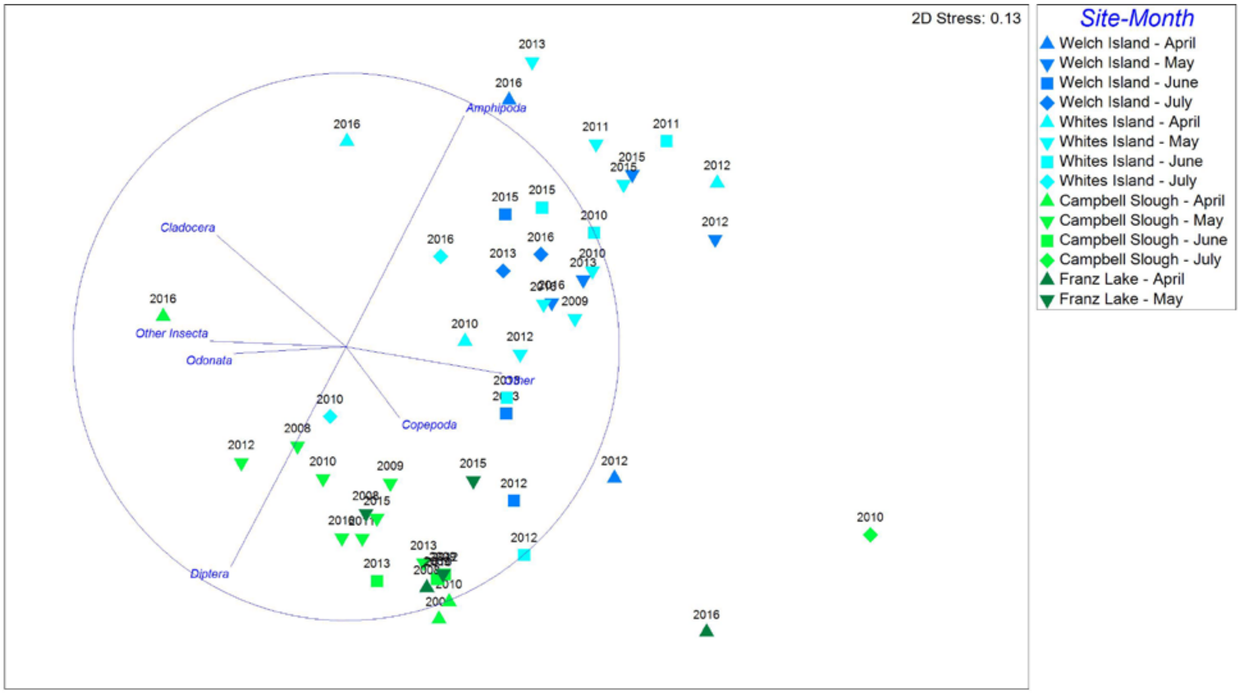


Figure 82. Two-dimensional nonmetric multidimensional scaling (NMDS) plot based on Bray-Curtis similarities between transformed percent IRI of major prey groups in diets sampled between 2008 and 2016. Significant correlation with variables (Pearson R > 0.2) are represented as blue vectors.

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 83). 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.

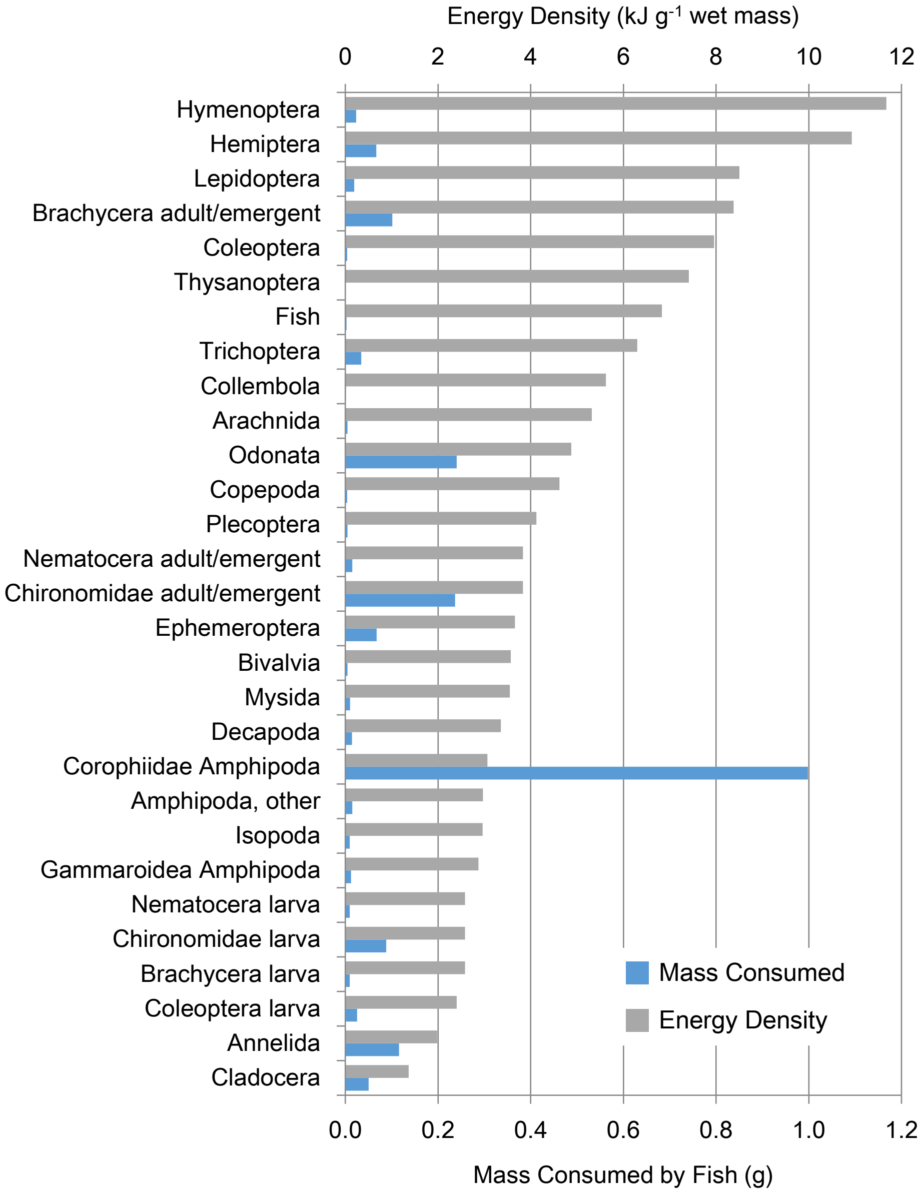


Figure 83. Energy density of individual prey taxa and total mass consumed by all juvenile Chinook salmon sampled in 2016. Energy densities were acquired from the literature and compiled in David et al. (2016).

Figure 84 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.

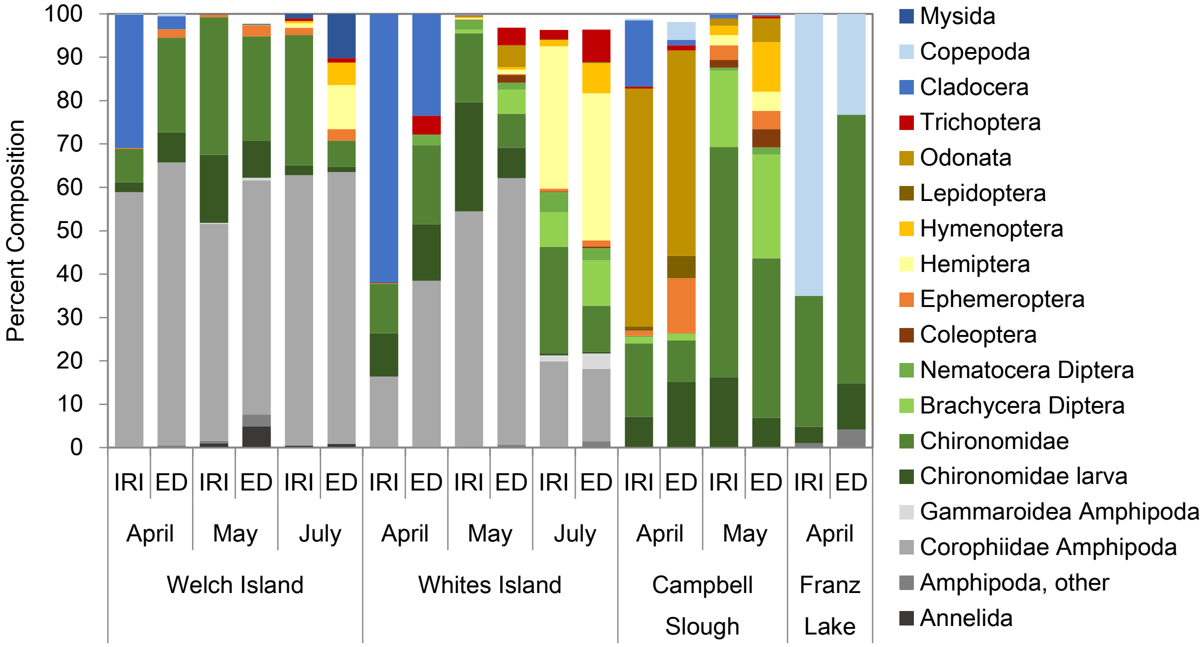


Figure 84. Comparison of the average percent composition of prey contributing to the total IRI and energy density (ED) consumed by juvenile Chinook salmon in 2016.

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 85).

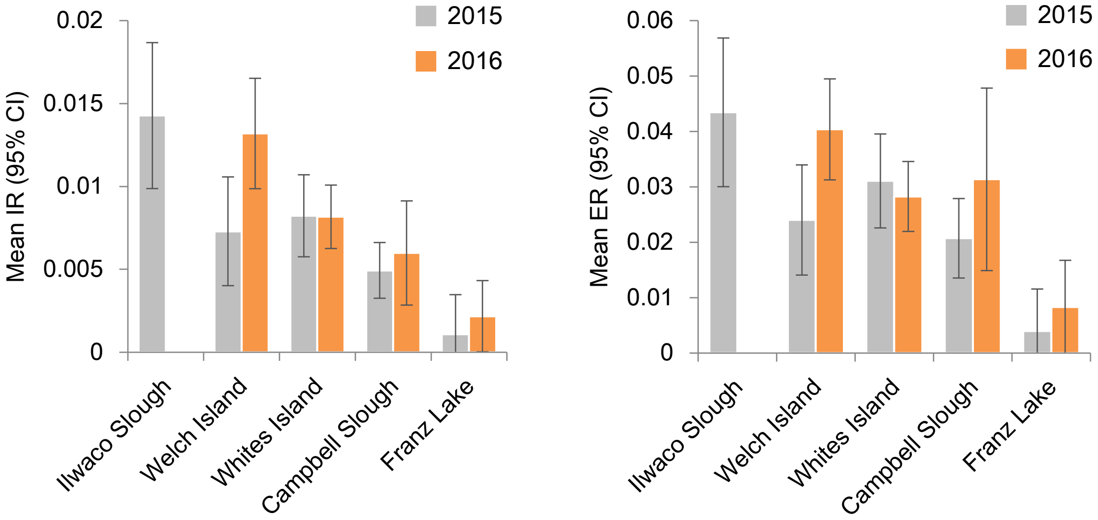


Figure 85. Feeding rates of juvenile Chinook salmon in 2015 and 2016 presented as average instantaneous ration (IR, left) and average energy ration (ER, right).

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 86). Maintenance metabolism increases with higher temperatures and with increases with 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.

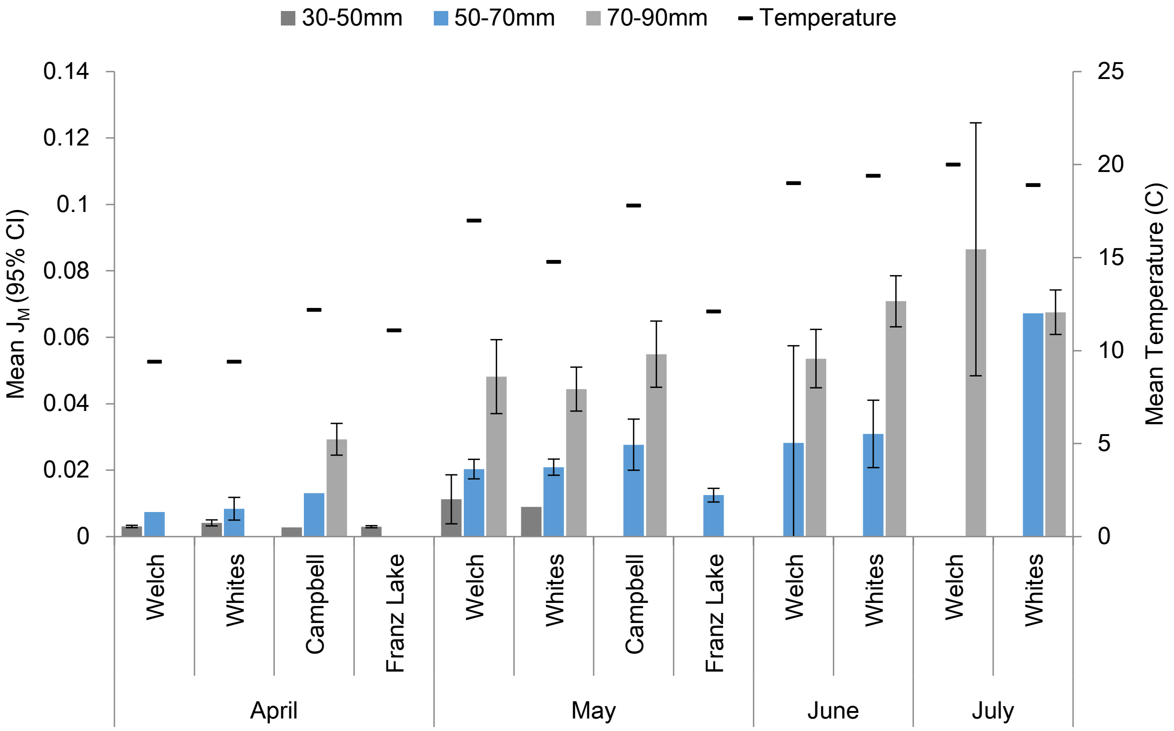


Figure 86. Average maintenance metabolism (J_M) by fish length bin, site, and month for juvenile Chinook salmon sampled in 2015 and 2016. Maintenance metabolism, which increases under unfavorable conditions, is compared to average water temperature (dashed line) at time of capture.

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 87 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.

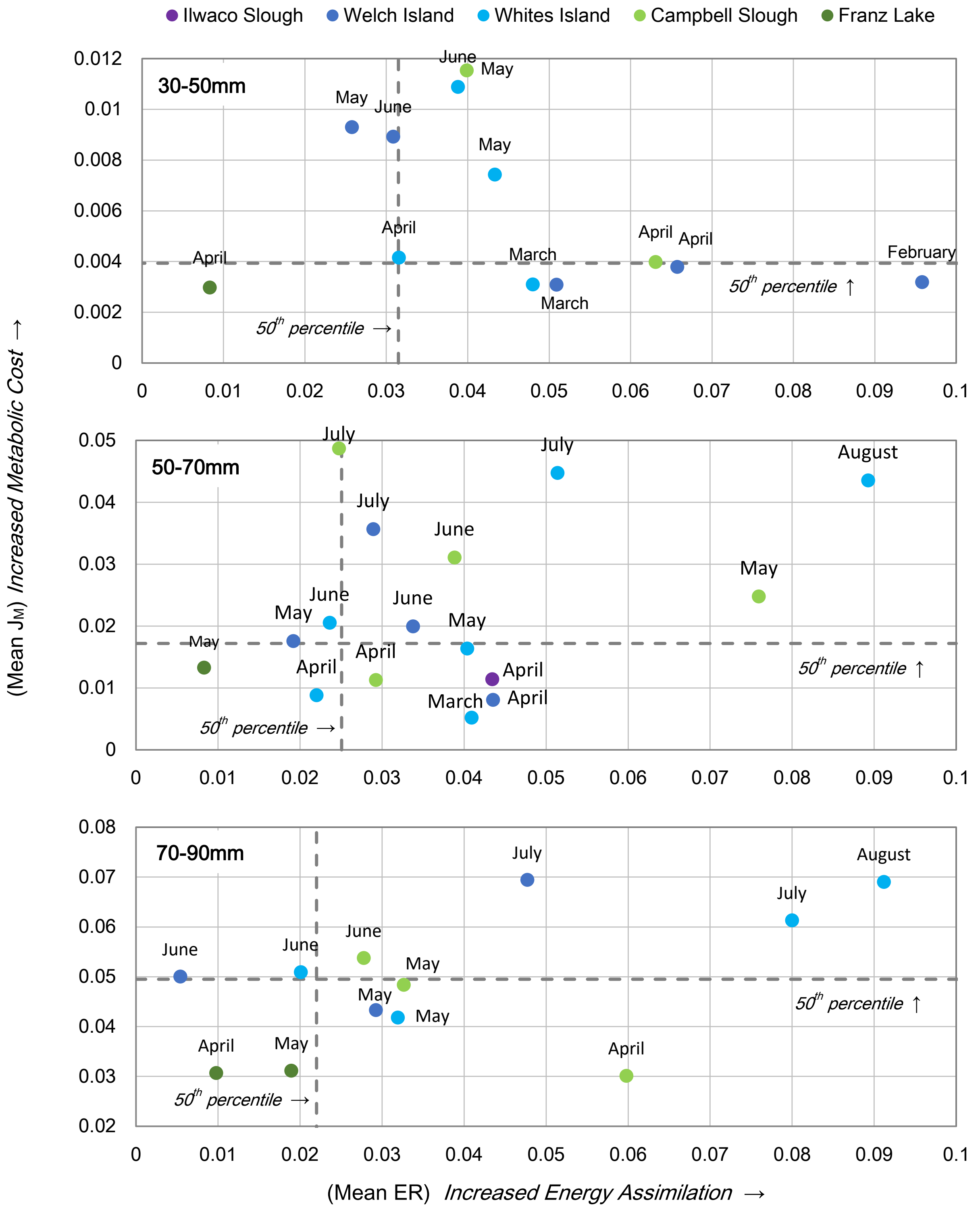


Figure 87. Quadrant charts of average maintenance metabolism (J_M) and average energy ration (ER), arranged by length bin of juvenile Chinook salmon sampled between 2008-2013, and 2015-2016. Dashed line is the 50th percentile of J_M and ER. The lower right quadrant represents conditions beneficial for 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

In 2016, fish communities at Ilwaco Slough in Reach A, Welch Island in Reach B and Whites Island in Reach C were dominated by three-spined stickleback (*Gasterosteus aculeatus*), which accounted for 89% of the total catch at Ilwaco Slough, 77% of the catch at Welch Island, and 73% of the catch at Whites Island (Figure 88). Other species present at Ilwaco Slough included banded killifish (*Fundulus diaphanous*), shiner perch (*Cymatogaster aggregata*), and staghorn sculpin (*Leptocottus armatus*), the latter two of which are saltwater species not found at other sites. Each of these species accounted for 2-4% of the total catch. Chum salmon were also present, making up 0.124% of the total catch. At Welch Island and Whites Island, species present that accounted for more than 1% of the total catch included juvenile salmonids, chiselmouth *(Acrocheilus alutaceus),* and banded killifish. Killifish were considerably more abundant at these sites than in previous years, accounting for 18-19% of the total catch. The species assemblages at Campbell Slough in Reach F and Franz Lake in Reach H were much more diverse than those at Ilwaco Slough, Welch Island, and Whites Island. Stickleback, while abundant, were less dominant than at the lower river sites, accounting for 11% of the total catch at Campbell Slough and 23% at Franz Lake (Figure 88). In addition to stickleback, other prominent species at Campbell Slough, in order of abundance, included yellow perch (*Perca flavescens*), 27% of the catch; banded killifish, 27% of the catch; carp (*Cyprinus carpio*), 15% of the catch, northern pikeminnow (*Ptychocheilus oregonensis*), 4.8% of the catch; tui chub, 3.8% of the catch, and pumpkinseed (*Lepomis gibbosus*), 3.7% of the catch. Juvenile salmonids accounted for 1.5% of the catch. At Franz Lake, a variety of species in addition to stickleback were present including, in order of abundance, American shad (*Alosa sapidissima*), 16% of the catch; chiselmouth, banded killifish 11% of the catch; pumpkinseed, 11% of the catch, largescale sucker (*Catostomus macrocheilus*), 9% of the catch, chiselmouth, 9% of the catch; northern pikeminnow (*Ptychocheilus oregonensis*), 8% of the catch; carp (*Cyprinus carpio*), 5% of the catch; juvenile salmonids, 3% of the catch; and tui chub, 3% of the catch. At all sites, fish community composition in 2016 was generally comparable to previous monitoring results from 2007-2015 (Figure 88), although the specific species present and their proportions differed somewhat from year to year.The proportions of banded killifish in the catches at Welch Island and Whites Island, for example, were noticeably higher in 2016 compared to previous years.

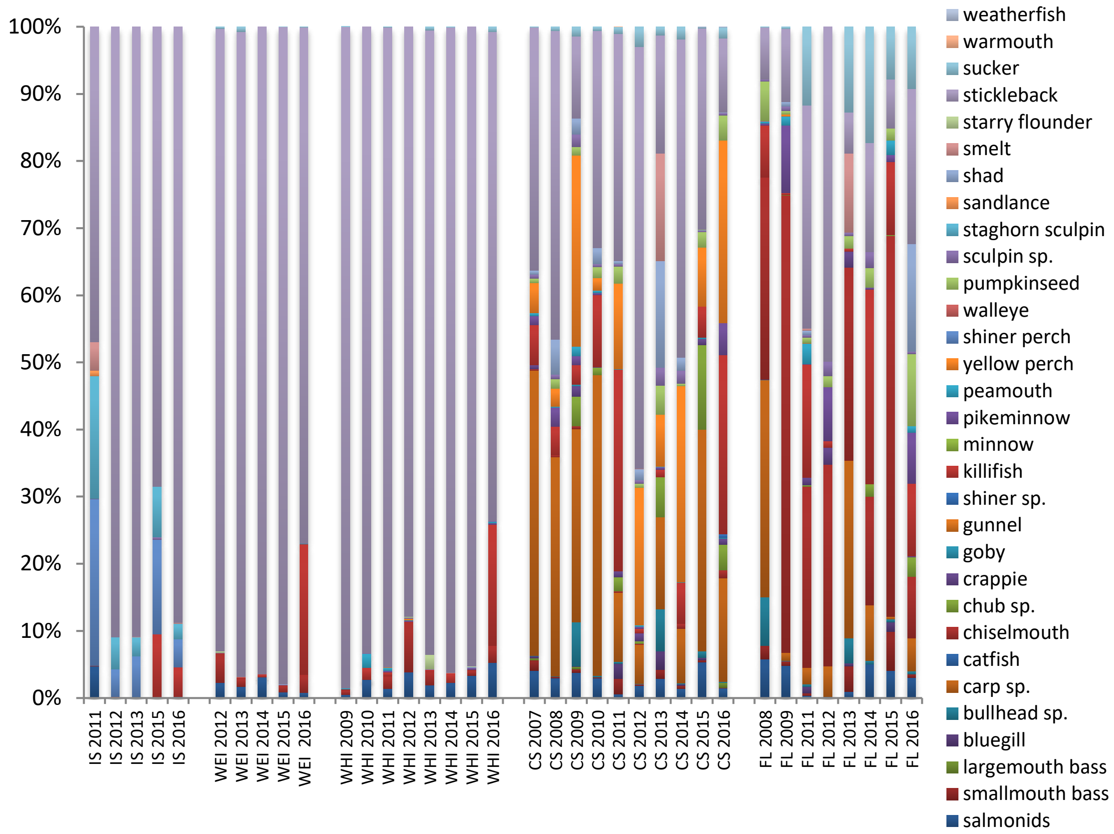


Figure 88. Fish community composition at EMP trends sites sampled 2007-2016. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

In 2016, as in previous sampling years, significant differences were found in species richness (ANOVA, $p = 0.0013$) and species diversity ($p < 0.0001$), among the trends sites (Figure 89). Mean diversity was significantly higher at Franz Lake and Campbell Slough than at Welch Island, Whites Island, or Ilwaco Slough (Tukey's Multiple Range test, $p < 0.05$). Species richness was significantly higher at Campbell Slough and Franz Lake than at Ilwaco Slough or Whites Island (Tukey's Multiple Range test, $p < 0.05$). For individual sites, there were no significant differences in species richness or species diversity by year (ANOVA, $0.0711 < p < 0.7485$)

In 2016, non-native fish species made up higher than usual percentages of the catches at Ilwaco Slough, Welch Island and Whites Island (4.4%, 19%, and 18%, respectively), because of the large number of banded killifish caught at these sites. For Ilwaco Slough, this represents a trend first observed in 2015, but for Welch Island and Whites Island, 2016 was the first year when such high percentages of killifish were observed. The percentages of non-native species in catches were substantially higher at Campbell Slough (75%) and Franz Lake (44%) than at the other sites, and were among the highest reported for these two sites over all the years they have been sampled (Figure 90). Predatory fish species known to feed on juvenile salmon, such as largemouth bass (*Micropterus salmoides*), smallmouth bass (*Micropterus dolomieu*), northern pikeminnow, and walleye (*Sander vitreus)* were absent at Ilwaco Slough and Welch Island (Figure 90) and made up only a small percentage of the catches at Whites Island (0.2%). They were somewhat more abundant at Campbell Slough and Franz Lake, where they made up 5.6% and 8.1% of the catch, respectively. At all of our sampling sites, the percentages of non-native fish species observed in 2016 were relatively high compared to percentages observed in previous years. The percentages of predatory fish were comparable to those observed in previous years at Ilwaco Slough, Welch Island, and

Whites Island, but relatively high in comparison to previous years at Campbell Slough and Franz Lake (Figure 76).

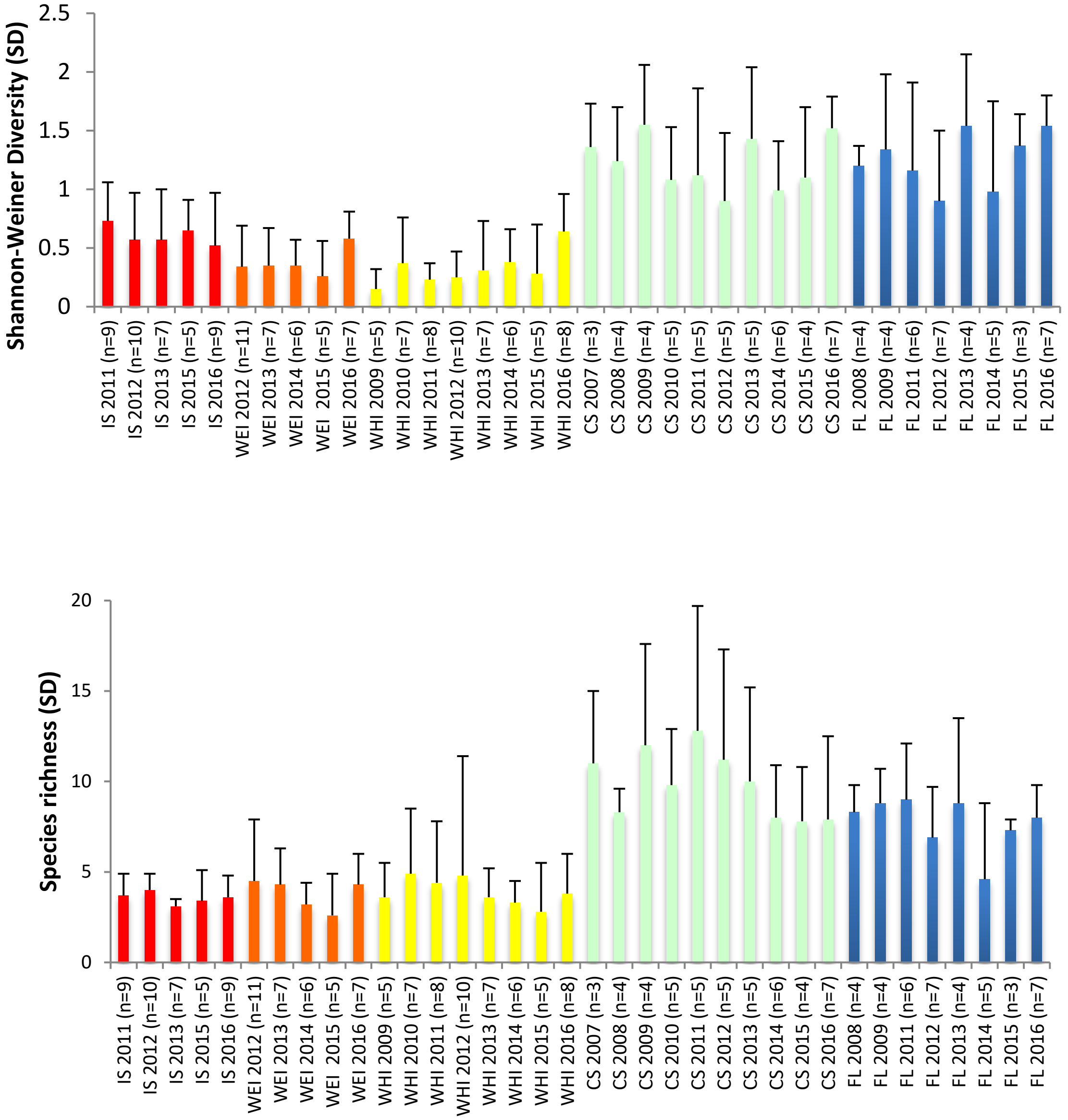


Figure 89. a) Shannon-Weiner diversity index and b) species richness (number of species) in mean (standard deviation, SD) values per sampling event (i.e., per monthly sampling event) at the EMP sampling sites in 2016 as compared to previous sampling years. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

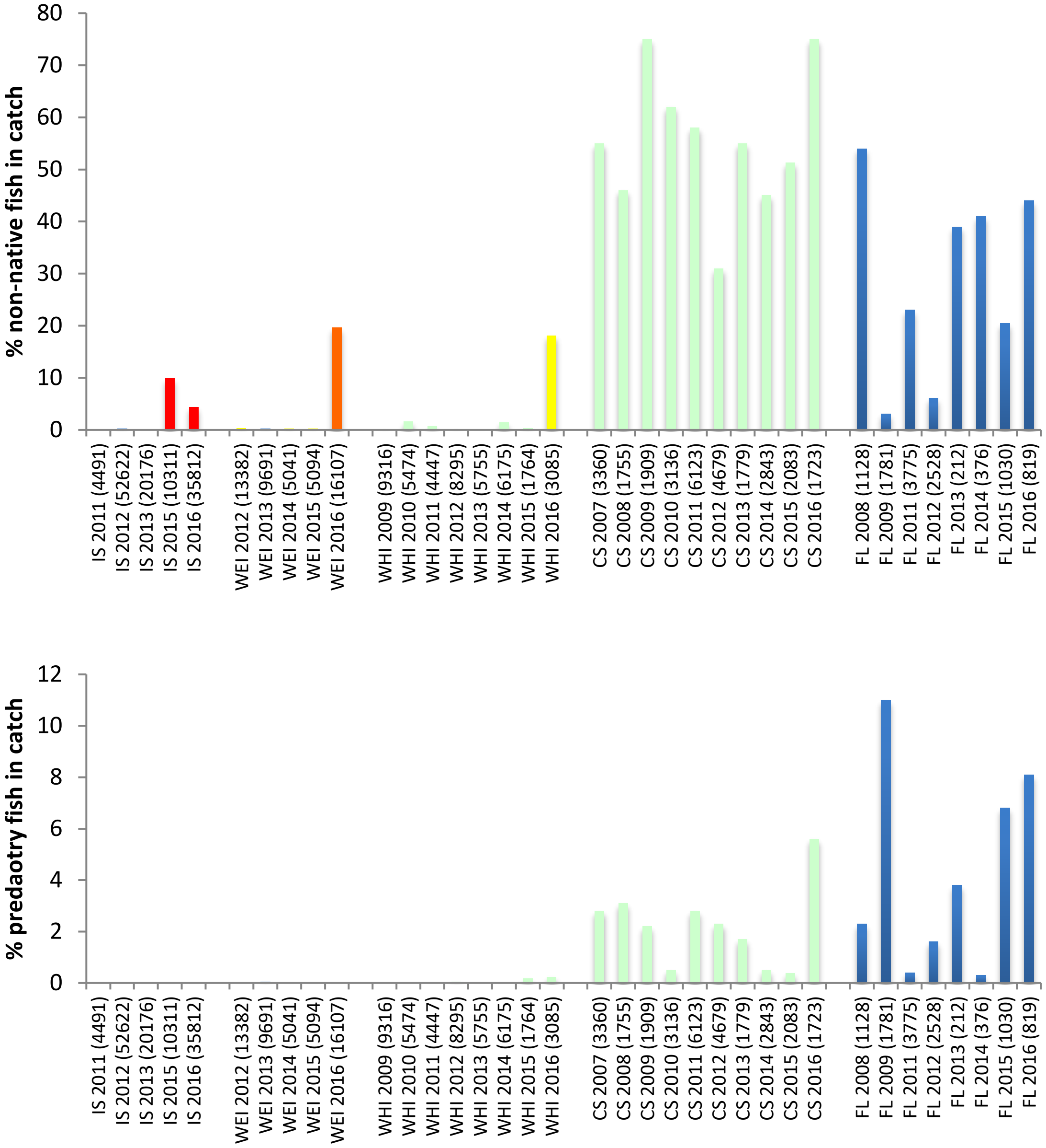


Figure 90. Percentages (based on total number of fish caught) of a) non-native fish species and b) % of fish that are recognized predators of juvenile salmon (i.e., smallmouth and largemouth bass, northern pikeminnow, walleye) in 2016 and in previous sampling years. Numbers contained in parentheses represent total fish catch at a site within a given year. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.1.1 *Salmon Species Composition*

Similar to previous sampling years, 2016 salmon species composition varied by site, showing distinct patterns associated with hydrogeomorphic reach (Figure 91). In 2016, chum salmon were the only salmon species captured at Ilwaco Slough in Reach A, while Chinook salmon were the dominant salmon species at Welch Island in Reach B, Whites Island in Reach C, and Campbell Slough in Reach F, and Franz Lake in Reach H. At these sites, Chinook salmon comprised 90% to 100% of salmonid catches. At all of the sites sampled in 2016, with the exception of Ilwaco Slough where no Chinook salmon were caught, unmarked (presumably wild) fish were more abundant than marked hatchery fish, accounting for 88-100% of the Chinook salmon collected (Figure 92). This pattern is typical for Welch and Whites Islands, and has been evident at Franz Lake since 2011. However, at Campbell Slough, the proportion of marked to unmarked Chinook salmon in 2016 was low in comparison to previous years. In addition to Chinook salmon, small numbers of chum salmon were found at Welch Island, Whites Island, and Campbell Slough. Only one unmarked coho salmon was collected at Franz Lake. No trout or sockeye salmon were caught in 2016.

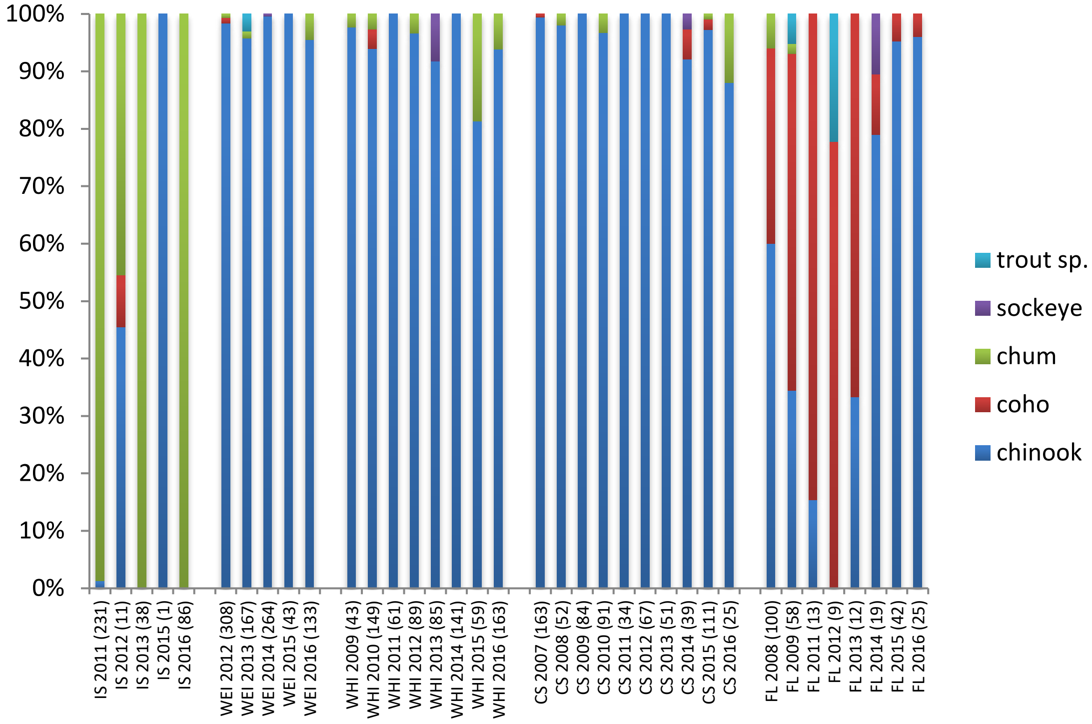


Figure 91. Percentage of salmonid species collected at EMP trends sites in 2016, as compared to percentages collected in previous sampling years. Total number of salmonids captured at a given site and year are presented in parentheses. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

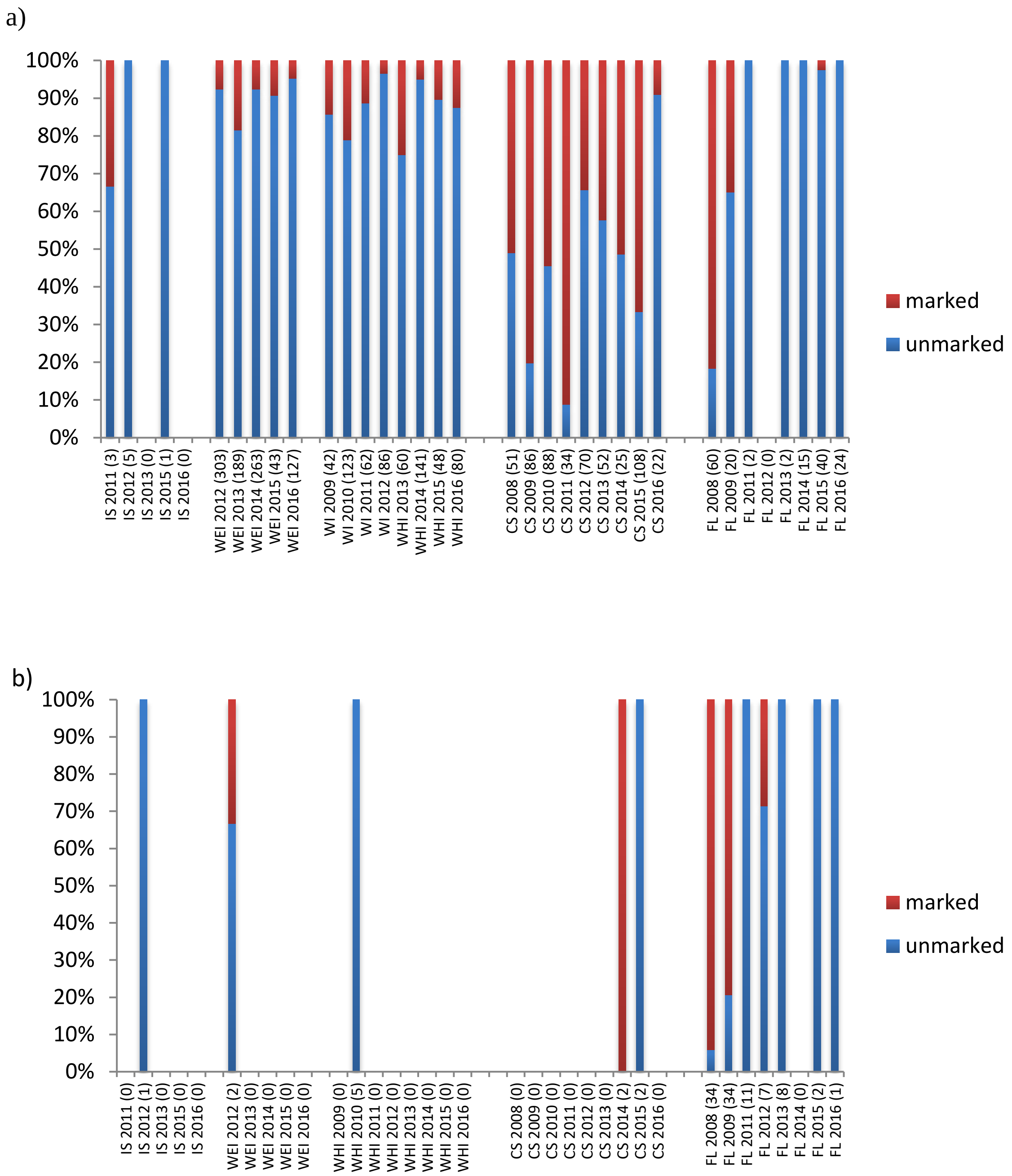


Figure 92. Percentage of marked and unmarked a) Chinook salmon and b) coho salmon captured at the EMP sampling sites in 2016, as compared to previous sampling years. Total number of the specified salmon species captured at a given site and year are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.1.2 *Salmon Density*

Chinook salmon. In 2016, unmarked Chinook salmon were captured at the EMP trends sites from February through September. The highest average densities of unmarked juvenile Chinook salmon were 43 and 70 fish per 1000 m^2 in April and May (Figure 6). Marked Chinook salmon were captured from May to August, with the highest average densities of 1.5 fish per 1000 m^2 in May and June (Figure 6). Mean Chinook salmon densities by site and year are shown in Figure 94. In 2016 the density of unmarked Chinook salmon was highest at Welch Island (28 fish per 1000 m^2) and Whites Island (30 fish per 1000 m^2) and lowest at Ilwaco Slough, where no Chinook salmon were found, with densities in the 3-6 fish per 1000 m^2 range at Campbell Slough and Franz Lake. Densities of unmarked Chinook salmon in 2016 were generally within the same ranges as previous years at all of the sites, but not especially high at any of the sites. The densities of marked Chinook salmon in 2016 were lower than the densities of unmarked Chinook salmon, and generally lower in 2016 compared to other years, with average values at all sites of less than 1 fish per1000 m^2.

Coho salmon. In 2016, one marked coho salmon was collected in June (density of 0.03 fish per 1000 m^2), and one unmarked coho salmon in January (density of 0.35 fish per 1000 m^2; Figure 93). Coho salmon densities by site and year are shown in Figure 94. Only two coho salmon were caught in 2016, one at Campbell Slough in Reach F and one at Franz Lake in Reach H (0.2 fish per 1000 m^2 at both sites). Coho salmon have been captured only sporadically at Ilwaco Slough, Welch Island and Whites Island, so their absence in 2016 was not unusual compared to previous years. At Campbell Slough, 2016 represents the second year that coho salmon were captured since systematic sampling for salmon density began in 2008. At Franz Lake, the only site where coho salmon are consistently collected, coho salmon density was at its lowest reported level in 2016 and has shown a consistent decline since 2011. Marked coho salmon, which were common at the site in 2008 and 2009, have not been observed since 2012.

Chum salmon. In 2016, chum salmon were found at the trends sites in March and April, with the highest average density in April (19 fish per 1000 m^2; Figure 93). Chum salmon were present at Ilwaco Slough, Welch Island, Whites Island and Campbell Slough in 2016 (Figure 95). 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 2016, as in 2015, sockeye salmon and trout were not caught, thus densities at all sites in 2016 remained low relative to those between 2008 and 2014 (Figure 95).

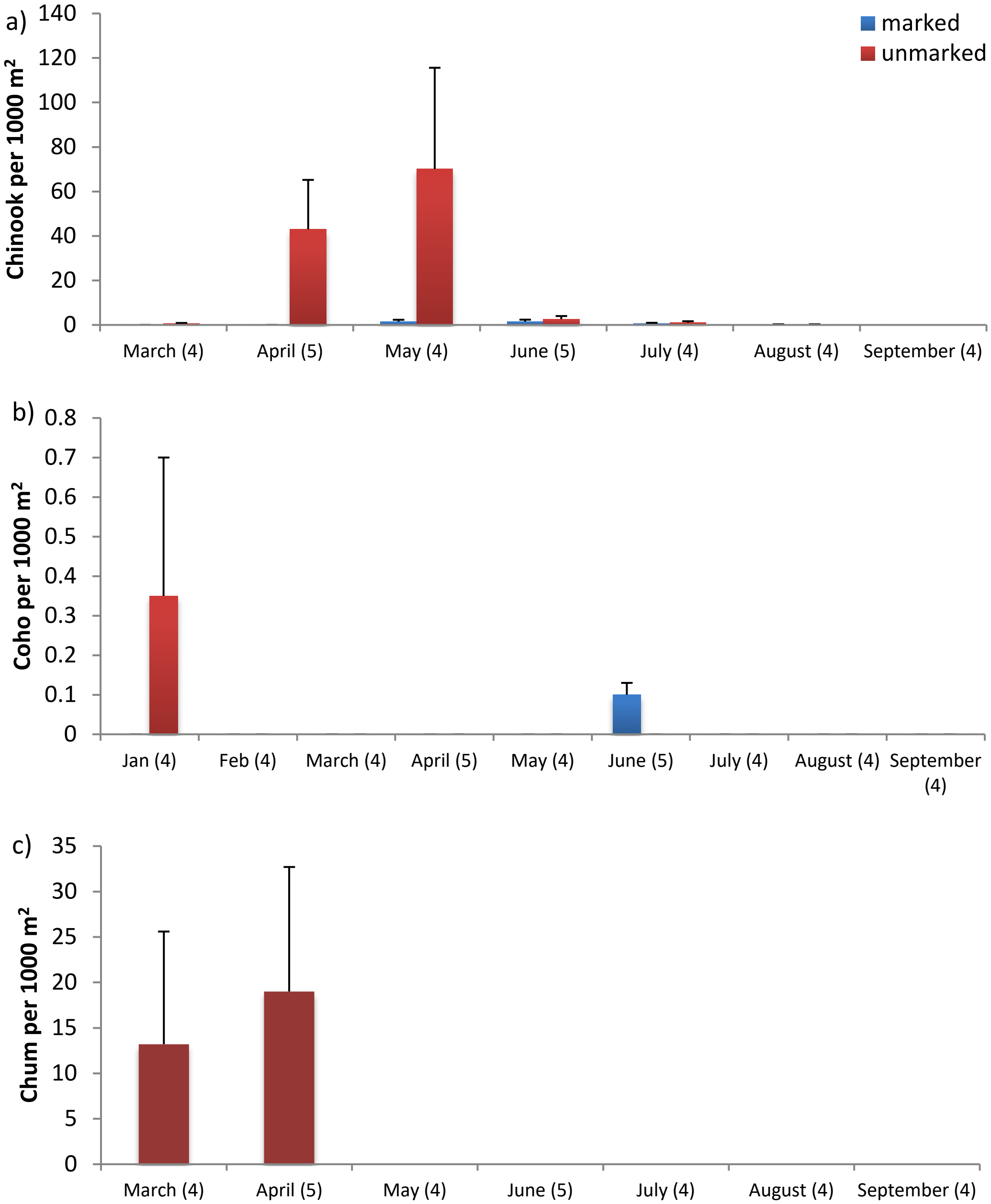


Figure 93. Mean (SE) densities (fish per 1000 m^2) of marked (blue bars) and unmarked (red bars) juvenile a) Chinook salmon, b) coho salmon, and c) chum salmon by month during the 2016 sampling year (all sites combined).

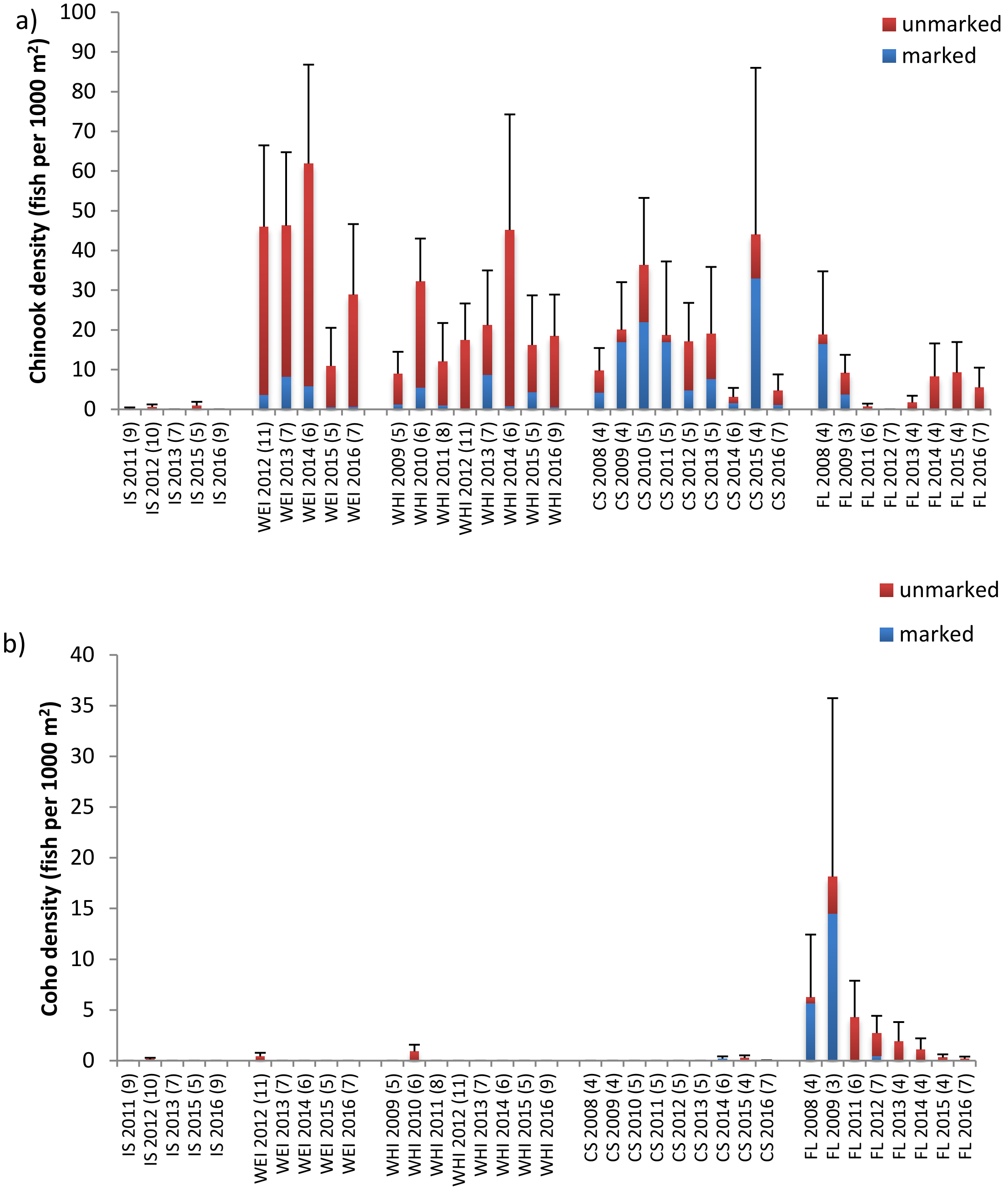


Figure 94. Marked (blue bars) and unmarked (red bars) juvenile a) Chinook salmon and b) coho salmon densities (fish per 1000 m^2) by site and year. Total number of salmonids captured per year at a site is presented in parentheses. IS = Ilwaco Slough, WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

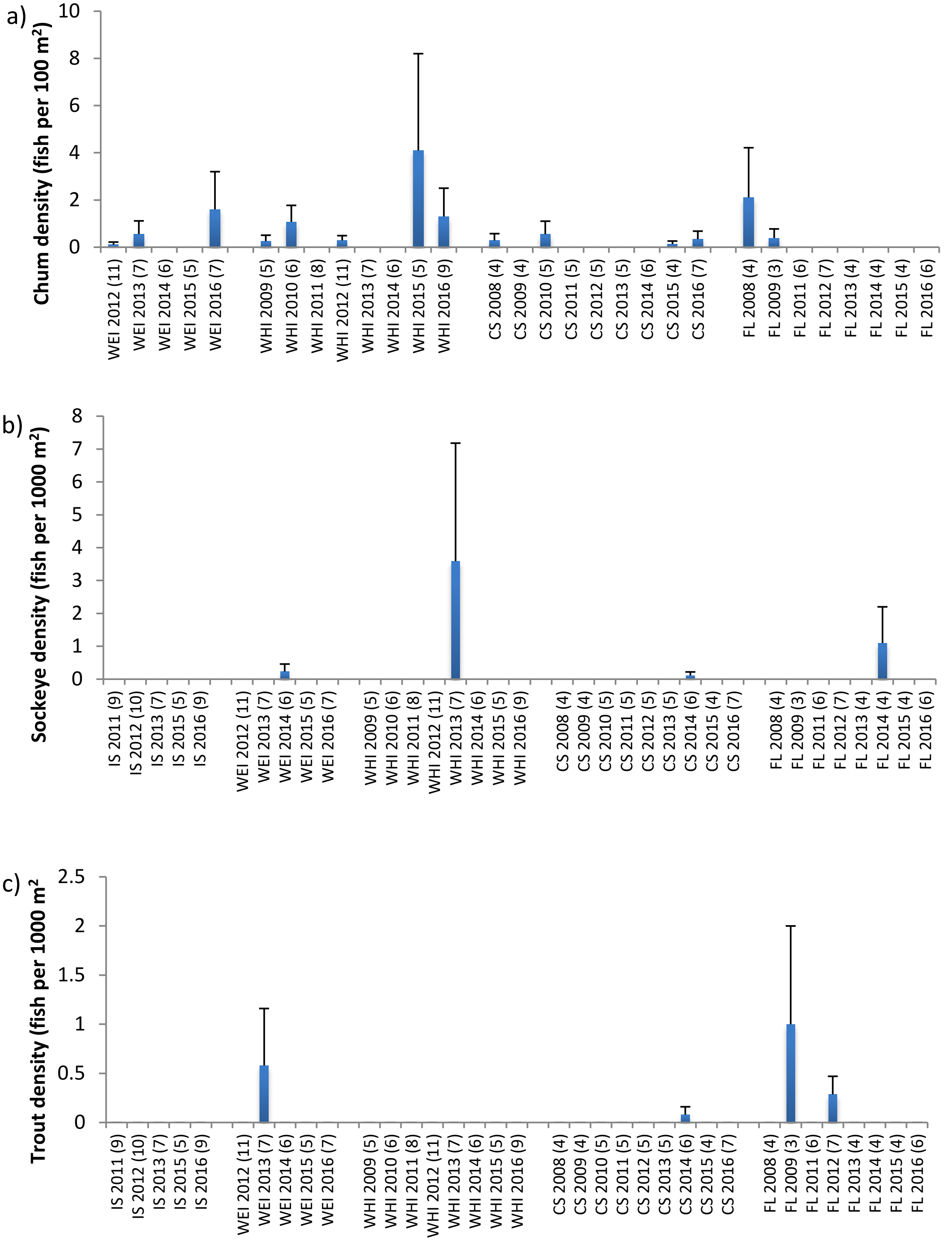


Figure 95. Juvenile a) chum salmon, b) sockeye salmon, and c) trout densities (fish per 1000 m^2) by year captured at trends sites. Total number of salmonids captured per year at a site is presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.2 Juvenile Chinook Salmon Stocks in Columbia River Tributaries

In 2015, we conducted a pilot study to monitor the occurrence of juvenile salmon in the lowest reaches of two tributaries of the Columbia River, the Grays River and the Lewis River (see Hanson et al. 2016 for map of tributary sampling locations). These sites were sampled by beach seine in April, May, and June. Our survey showed that no marked hatchery salmon were present at these areas, and the unmarked Chinook salmon present were predominantly fry. Based on these results, we hypothesized that most of the Chinook salmon using the tributary sites were of local origin. The genetics data on these fish, which are now available (Figure 96), are consistent with this hypothesis.

At the Grays River, West Cascades fall Chinook were the most common stock, with some individuals from the Columbia River Rogue and Spring Creek Group fall Chinook stocks also present. This pattern was fairly similar to that observed at the Welch Island and Whites Island sampling sites. In the lower Lewis River, the only stock observed was West Cascades fall Chinook. This is quite different from the stock composition of fish observed at Campbell Slough, the closest of the trends sites, where Spring Creek Group fall Chinook were most common.

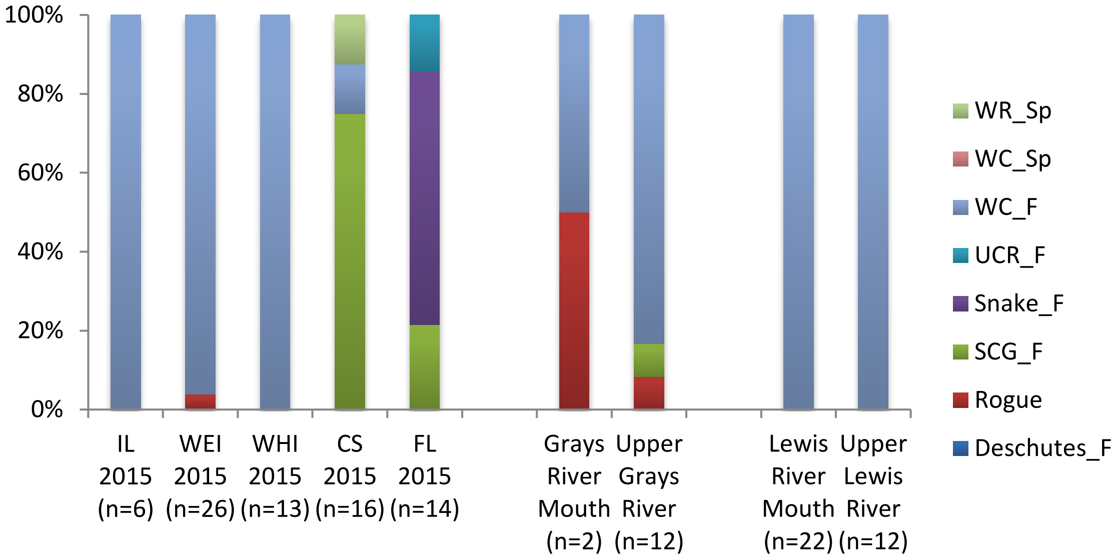


Figure 96. Genetic stock composition of juvenile Chinook salmon from the trend sites in 2015, as compared to the stock composition of juvenile Chinook salmon sampled from tributary sites in the Grays and Lewis River. Chinook salmon stocks: WR_Sp =Willamette River Spring, WC_Sp = West Cascade Spring, WC_F = West Cascade Fall, UCR_F = Upper Columbia River Fall, Snake_F = Snake River Fall, SCG_F = Spring Creek Group Fall, Rogue = Rogue River, Deschutes_F = Deschutes River Fall.

3.6.3 Salmon Metrics

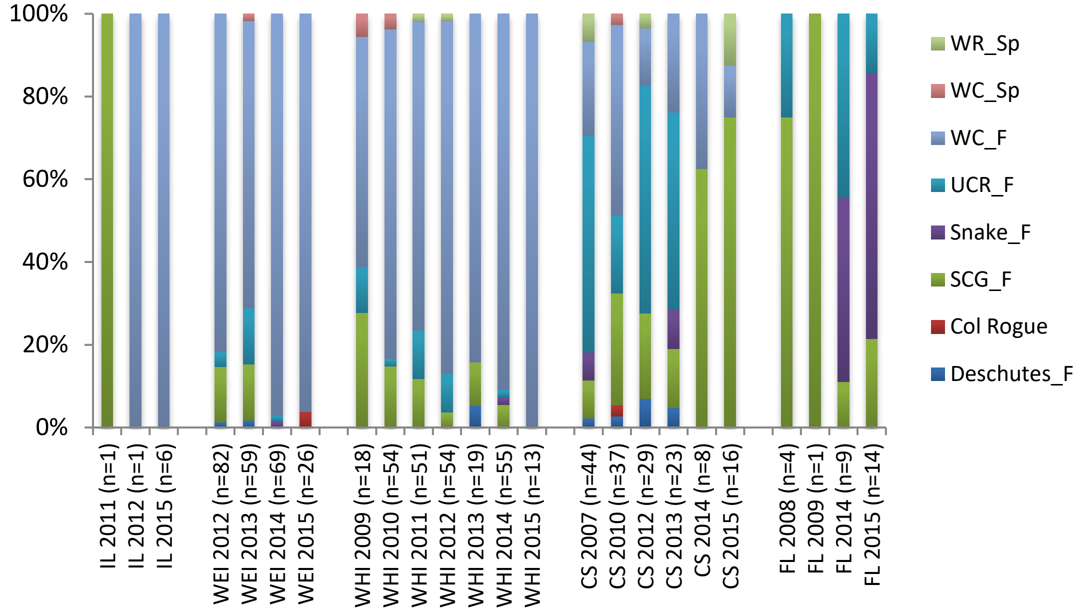
3.6.3.1 *Genetic Stock Identification*

In this report we present the Chinook salmon genetic stock data collected in 2015. Genetic samples collected in 2016 are currently undergoing analysis. In 2015, genetics data were collected from Chinook salmon at Welch Island, Whites Island, Campbell Slough, and Franz Lake. Too few Chinook salmon were collected at Ilwaco Slough to allow for genetic stock identification. Among unmarked fish, West Cascades fall Chinook were the most abundant stock at Welch Island and Whites Island, and Spring Creek Group Fall Chinook were most prominent at Campbell Slough (Figure 97). Interior stocks such as Snake River fall Chinook and Upper Columbia summer/fall Chinook were also captured at the trends sites. The stocks present at each of the sites were generally similar over the sampling years, although the interior stocks were somewhat less abundant at the Welch Island, Whites Island, and Campbell Slough sites in 2015. In 2015, as in 2014 Snake River fall Chinook and Upper Columbia River summer/fall Chinook were the predominant stocks at Franz Lake.

In 2015, marked Chinook salmon were collected and analyzed for genetic stock at Welch Island, Whites Island, and Campbell Slough. All marked fish collected in 2015 were either Spring Creek Group fall and West Cascades fall Chinook salmon (West Cascades fall Chinook were most common at Welch Island and Whites Island, while Spring Creek Group fall Chinook were most common at Campbell Slough; Figure 97). As with unmarked fish, the stock composition of marked fish at the trends sites did not vary greatly from year to year, although in 2015 the number of fish from interior stocks was especially low.

Water temperatures were unusually warm in 2015, and the seasonal distribution of Chinook salmon stocks showed some associated patterns (Figure 98). Early season (February and March) migrants, which in previous years were primarily West Cascades and Spring Creek Group fall Chinook, were entirely absent. Similarly, later season (June to August) migrants, which in past years were primarily Snake River Fall, and Deschutes Fall Chinook, were rare or absent. The migration period appeared to be much shorter, with fish observed only from April through June, and the timing of migration appeared earlier for some stocks. For example, Snake River fall Chinook and Upper Columbia summer/fall Chinook, which are typically most common in June and July, were found only in April and May. West Cascades fall Chinook salmon, which had previously been found from February through August, were found only from April through June. Spring Creek Group fall Chinook, usually found from February through June with the highest proportion of fish in May, were found predominantly in April.

a)



b)

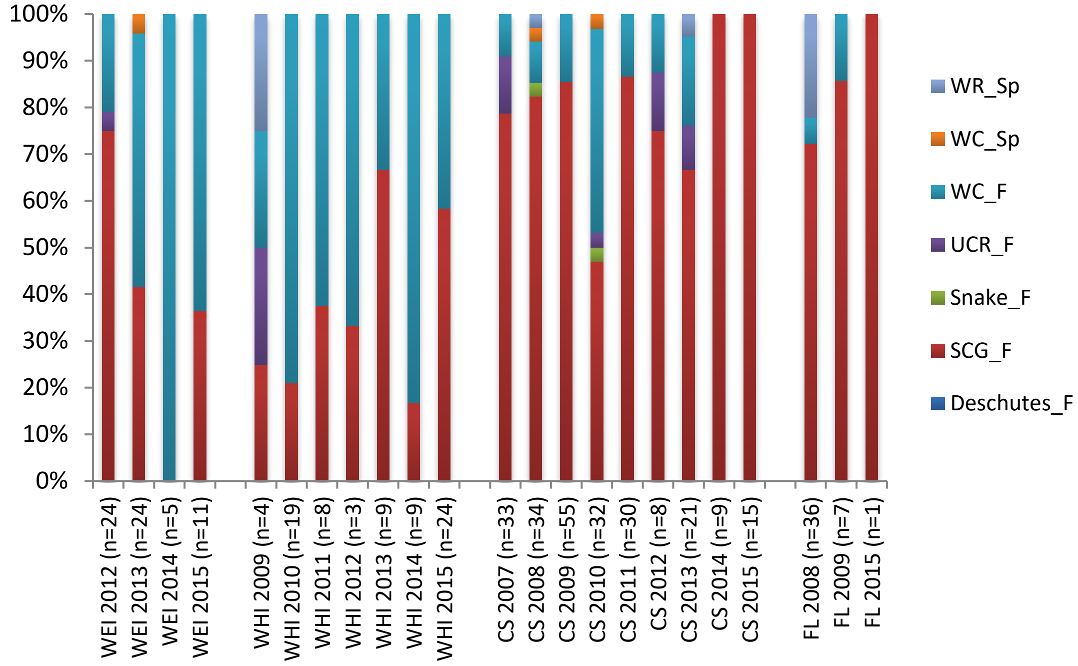


Figure 97. Genetic stock composition of a) unmarked and b) marked Chinook salmon at the trends sites from 2007-2015. Sample sizes for each site are presented in parentheses. Ilwaco Slough is not shown, as no new data were collected from this site in 2015 for temporal comparison. WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough. Chinook salmon stocks: WR_Sp =Willamette River Spring, WC_Sp = West Cascade Spring, WC_F = West Cascade Fall, UCR_F = Upper Columbia River Fall, Snake_F = Snake River Fall, SCG_F = Spring Creek Group Fall, Rogue = Rogue River, Deschutes_F = Deschutes River Fall.

Figure 98. Proportions of fish per sampling month for Chinook salmon stocks collected a) 2015 compared to b) previous years. Figures show combined data from Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake, and include only unmarked Chinook salmon. Chinook salmon stocks: WR_Sp =Willamette River Spring, WC_Sp = West Cascade Spring, WC_F = West Cascade Fall, UCR_F = Upper Columbia River Fall, Snake_F = Snake River Fall, SCG_F = Spring Creek Group Fall, Rogue = Rogue River, Deschutes_F = Deschutes River Fall.

3.6.3.2 *Salmon Size and Condition*

Chinook salmon

Length, weight, and condition factor. No Chinook salmon were caught at Ilwaco Slough in 2016. At the other sites, compared to previous sampling years, the length, weight and condition of unmarked Chinook salmon in 2016 showed similar patterns, with the largest fish typically captured at Campbell Slough (Figure 99). Within sites, there was some variation among years, though no clear increasing or decreasing trends. Significant differences in length among years were observed at Welch Island ($p < 0.0001$), Whites Island ($p < 0.0001$), Campbell Slough ($p < 0.0001$), and Franz Lake ($p < 0.0001$). Significant differences in weight among years were also observed at Welch Island ($p < 0.0001$), Whites Island ($p < 0.0001$) and Campbell Slough ($p < 0.0001$), and Franz Lake ($p = 0.0013$). At Whites Island and Franz Lake, 2016 fish were relatively smaller compared to other sampling years, but at other sites length and weight were not especially high or low (Tukeys multiple range test, $p < 0.05$). Differences in condition factor were observed among years for Welch Island ($p < 0.0001$), Whites Island ($p < 0.0001$) and Campbell Slough ($p < 0.0001$), with relatively high values at Campbell Slough and Welch Island, but relatively low values at Whites Island. At Franz Lake ($p = 0.1574$) condition factor did not differ significantly among years.

In 2016, marked Chinook salmon were caught at Welch Island, Whites Island, and Campbell Slough. At Welch Island, length and weight did not differ significantly across years, but condition factor was different across years ($p = 0.0011$) and was significantly higher in 2016 than in other years (Figure 100). Significant differences in length among years were observed at Whites Island ($p = 0.0221$), where fish were significantly smaller in 2016 than in other years. Weight showed a similar pattern, but was not significantly different among years ($p = 0.06$). At both Welch Island ($p = 0.0011$) and Whites Island ($p < 0.0001$), condition factor differed significantly among years and was significantly higher in 2016 than in other years. At Campbell Slough, the fish collected in 2016 did not differ significantly in length, weight, or condition factor than those caught in previous years. However, there were significant differences in length, weight, and condition factor among fish caught from 2008 to 2015. For example, length, weight, and condition factor were all significantly higher in fish caught in 2014 than in fish caught in 2011.

Size class distribution. At the trends sites in 2016, the majority of unmarked Chinook salmon were fry (69%), 31% were fingerlings, and none were yearlings (Figure 101). At Welch Island and Whites Island, fry predominated, making up 71% and 70% of unmarked Chinook salmon, respectively. At Campbell Slough, fingerlings predominated, comprising 75% of the catch. At Franz Lake, 96% of the 24 unmarked Chinook that were caught in 2016 were fry. In comparison to previous years, the percentage of fry at all of the trend sites was higher than in 2015. At Welch Island and Franz Lake, the 2016 proportions of fry were significantly higher than the overall proportion for all sites combined (Heterogeneity G-test, $p < 0.05$). At Whites Island and Campbell Slough, the 2016 proportions of fry were not significantly higher or lower than the overall proportion of fry for all sites combined (Heterogeneity G-test, $p > 0.05$).

Of the 45 marked Chinook salmon caught at the trends sites in 2016, 100% were fingerlings; no yearlings were found (Figure 101). In comparison to previous sampling years, the proportion of yearlings encountered did not differ significantly from previous years at Welch Island, Whites Island, Campbell Slough, or Franz Lake (Heterogeneity G-test, $p > 0.05$). No marked Chinook salmon were caught at Ilwaco Slough.

Other salmon species

A total of 84 chum salmon were captured and measured in 2016, 65 at Ilwaco Slough, six at Welch Island, six at Whites Island, and three at Campbell Slough, all caught between February and April. The average length, weight, and condition factor of these fish (± SD) were 45 ± 5 mm; 0.7 ± 0.6 g; and 0.75 ± 0.17, respectively (Figure 102). The chum salmon collected in 2016 were comparable in size to those that have been collected in previous years, not especially large or small. Similarly, the mean 2016 value for condition factor (0.75) 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 and Campbell Slough, the two sites farthest upstream (Figure 13). 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). Significant differences in length among years were observed at Welch Island ($p < 0.0336$) and Whites Island ($p < 0.0001$). At Welch Island, chum salmon collected in 2016 were of intermediate size, while at Whites Island, they were relatively small in comparison with other years, though larger than those collected in 2012. Significant differences in weight among years were observed at Ilwaco Slough ($p = 0.0053$) and Whites Island ($p < 0.0001$). At Whites Island fish size was relatively low in 2016 in comparison to other sampling years, while at Ilwaco Slough, it was relatively high. Differences in condition factor were also observed among years for Ilwaco Slough ($p < 0.0001$) and Whites Island ($p < 0.0459$), with relatively high values at Ilwaco Slough, but relatively low values at Whites Island. No significant differences in length, weight, or condition factor by year were found for chum collected at Campbell Slough, but some unusually large chum salmon were collected at this site in 2015 and 2016, one in 2015 was 62 mm and one in 2016 was 79 mm. The typical size range for chum in this part of the river is 40-60 mm.

Only one coho salmon was caught in 2016, an unmarked coho found at Franz Lake in January. Franz Lake is the only site where coho salmon have been caught consistently enough to compare size measurements by sampling year, and even at this site, only unmarked coho salmon were caught in all sampling years including 2016. Mean length, weight, and condition factor are shown for unmarked coho salmon from Franz Lake in (Figure 103). Mean length (± SD) varied from 82 ± 11 mm in 2013 to 120 ± 34 mm in 2009. Coho salmon collected in 2016 were of intermediate size (96 mm), but length did not differ significantly among sampling years ($p = 0.2490$). Weight, however, differed among years ($p = 0.0109$), with the highest value in 2009 (19.8 ± 9.3 g) and the lowest in 2013 (5.1 ± 2.5 g). Again, the weight of the one fish collected in 2016 was intermediate compared to other years (10 g). Condition factor was not different among sampling years ($p = 0.1688$), but the lowest value was observed in 2008 (0.77 ± 0.15) and the highest in 2016 (1.13). Condition factor in 2014, 2015, and 2016 were the highest values observed over the sampling period.

Sockeye salmon and trout were not caught at any of the trends sites in 2016.

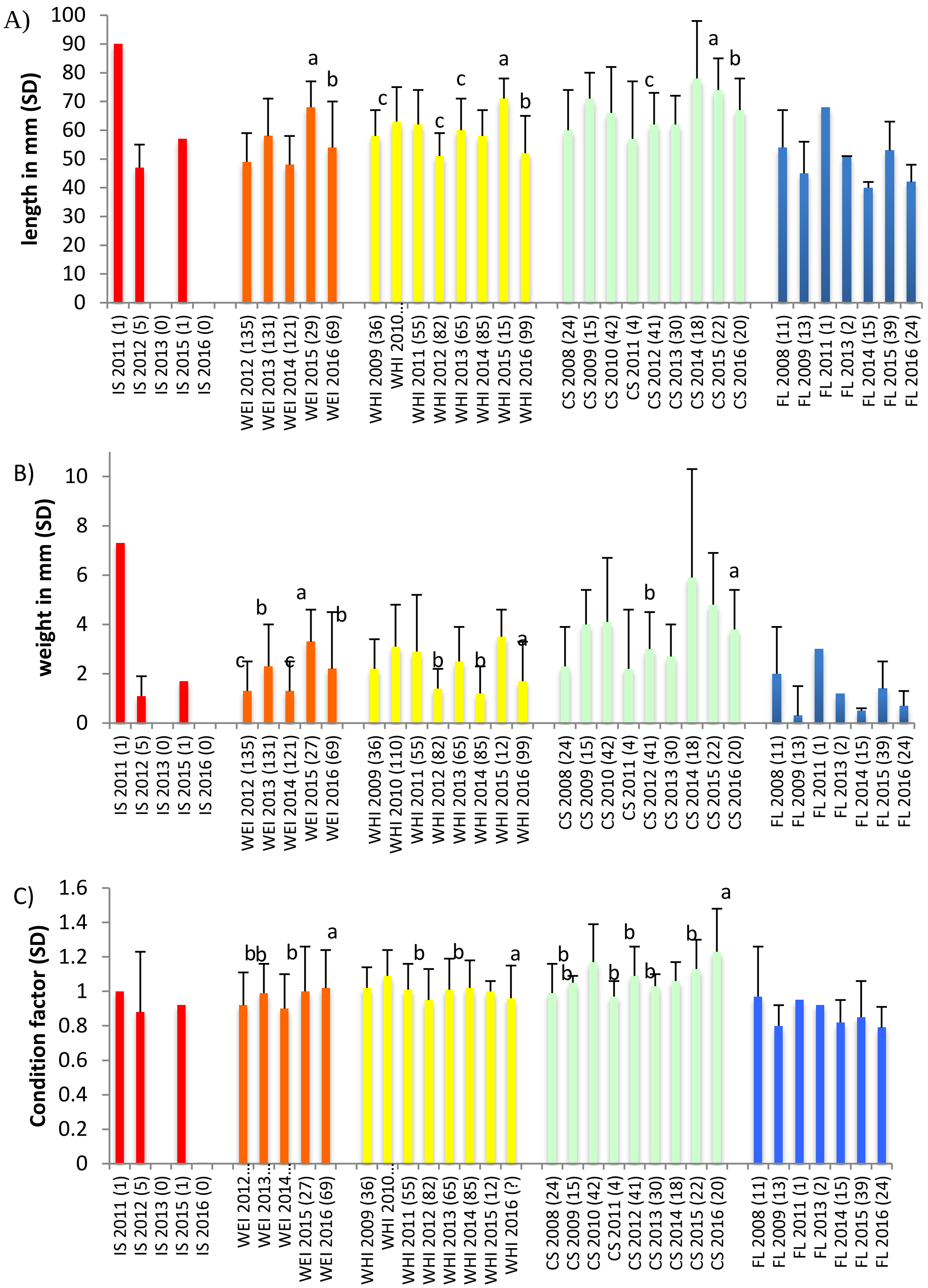


Figure 99. Mean (SD) a) length (mm), b) weight (g), and c) condition factor (± SD) of unmarked juvenile Chinook salmon at trends sites in 2016 as compared to previous years. Within the sites, values with different letter superscripts are significantly different (Tukey's multiple range test, $p < 0.05$). Total number of Chinook salmon captured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

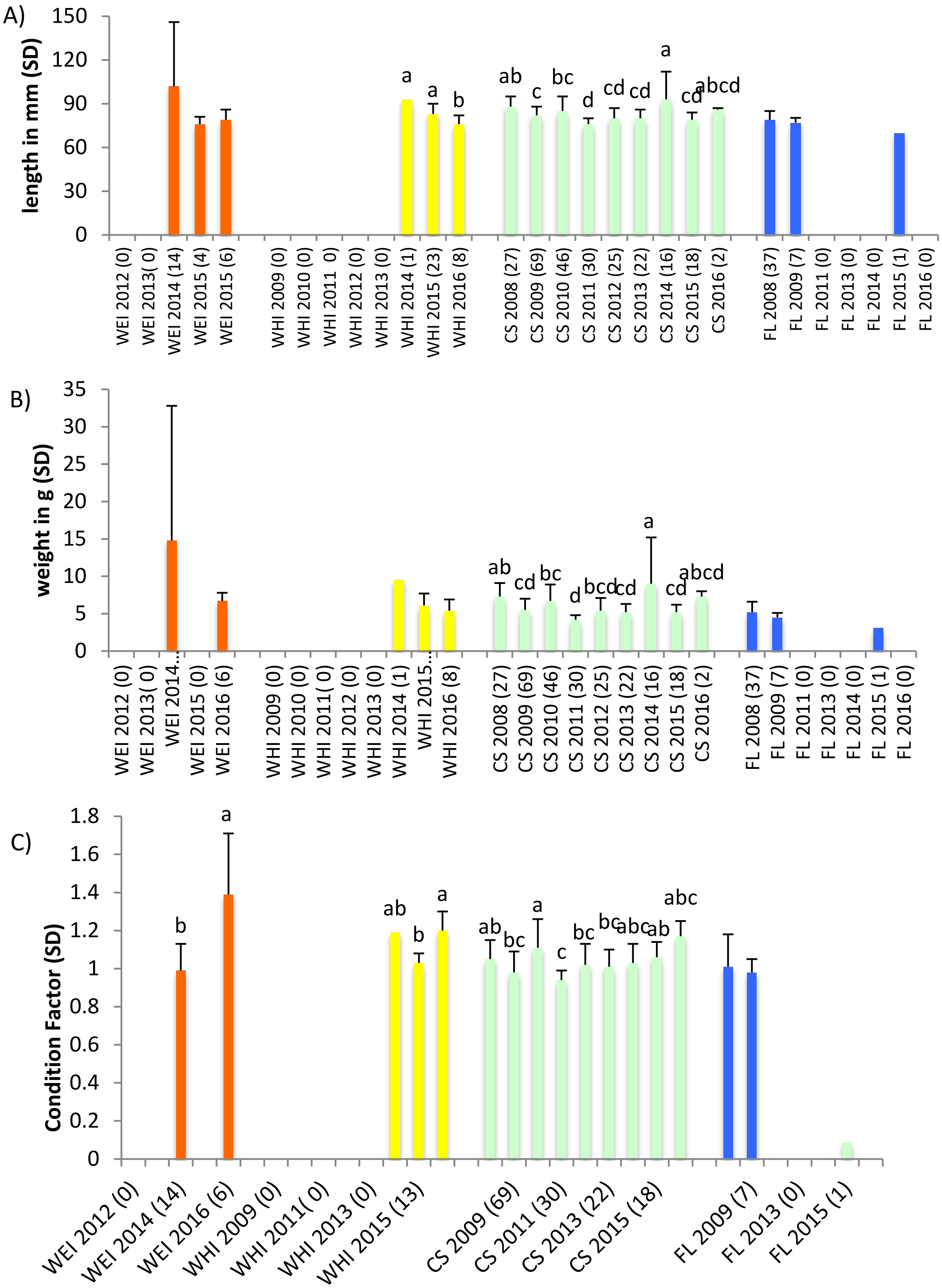


Figure 100. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of marked Chinook salmon at trends sites in 2016 compared to previous sampling years. Total number of Chinook salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

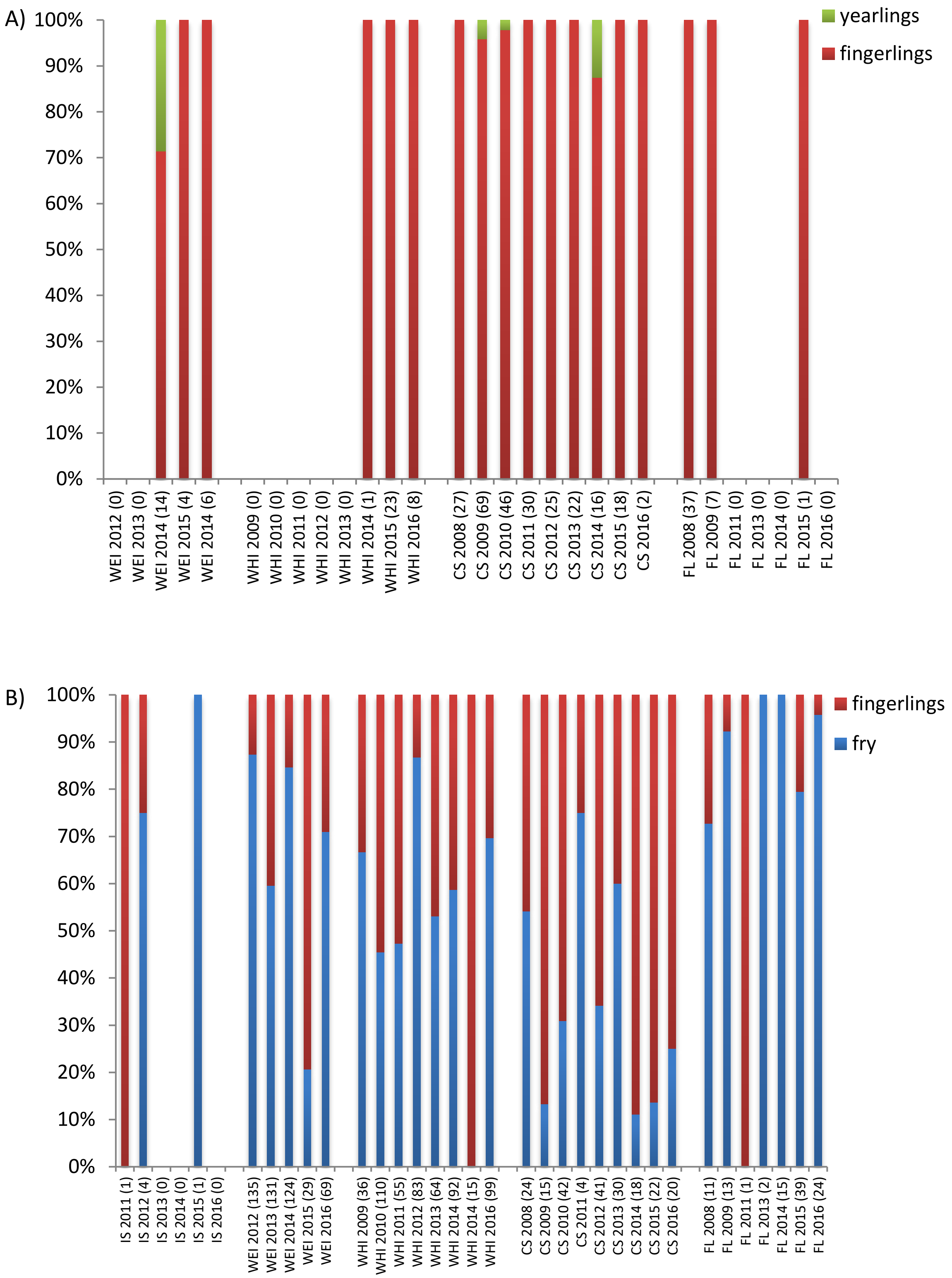


Figure 101. Size class distribution of a) marked and b) unmarked juvenile Chinook salmon captured at trends sites in 2016 and in previous sampling years. Total numbers of Chinook salmon captured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

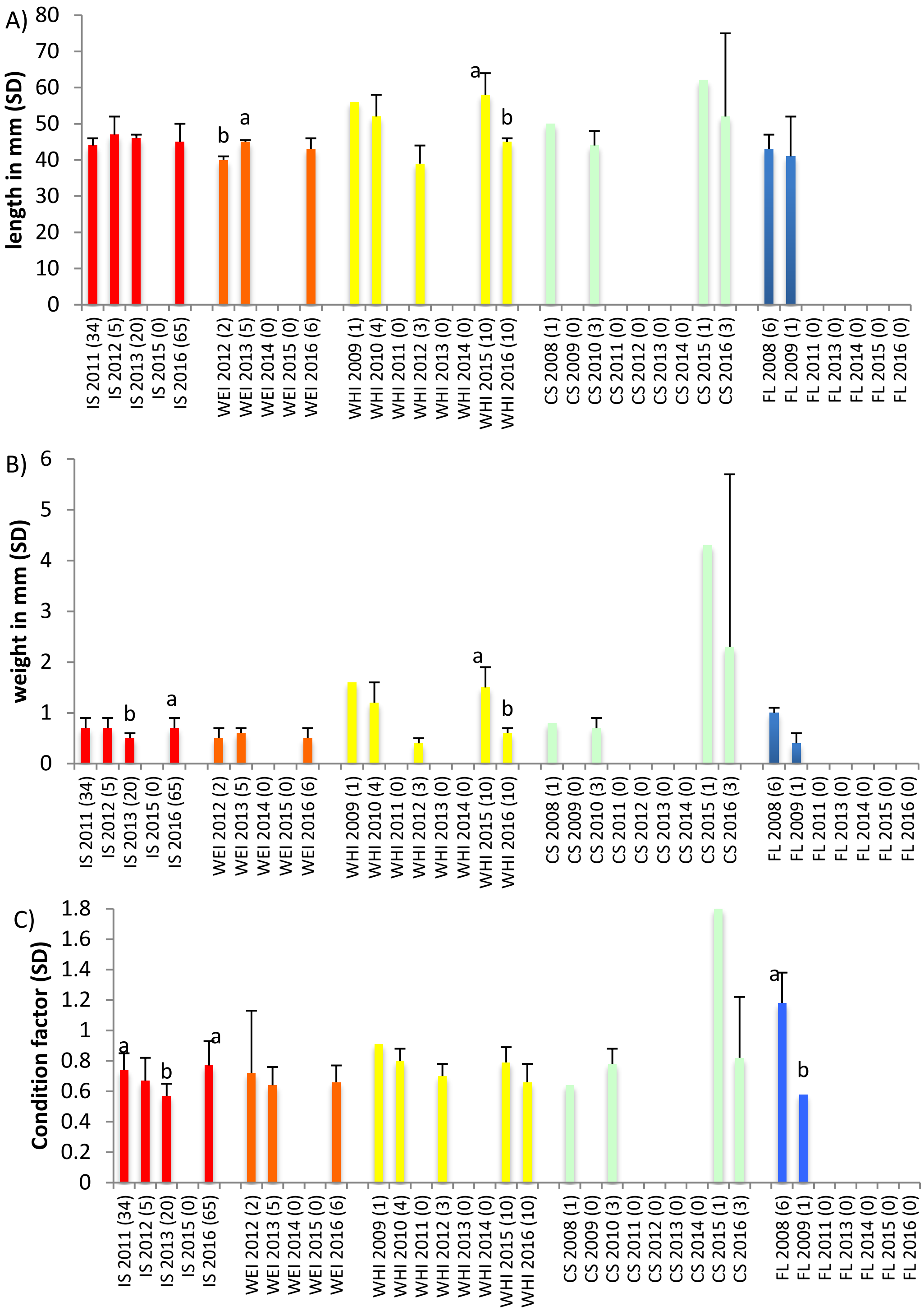


Figure 102. Mean (SD) a) length (mm), b) weight (g) and c) condition factor of chum salmon at trends sites in 2016 compared to previous sampling years. Total number of chum salmon weighed and/or measured per year at a site are presented in parentheses. IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

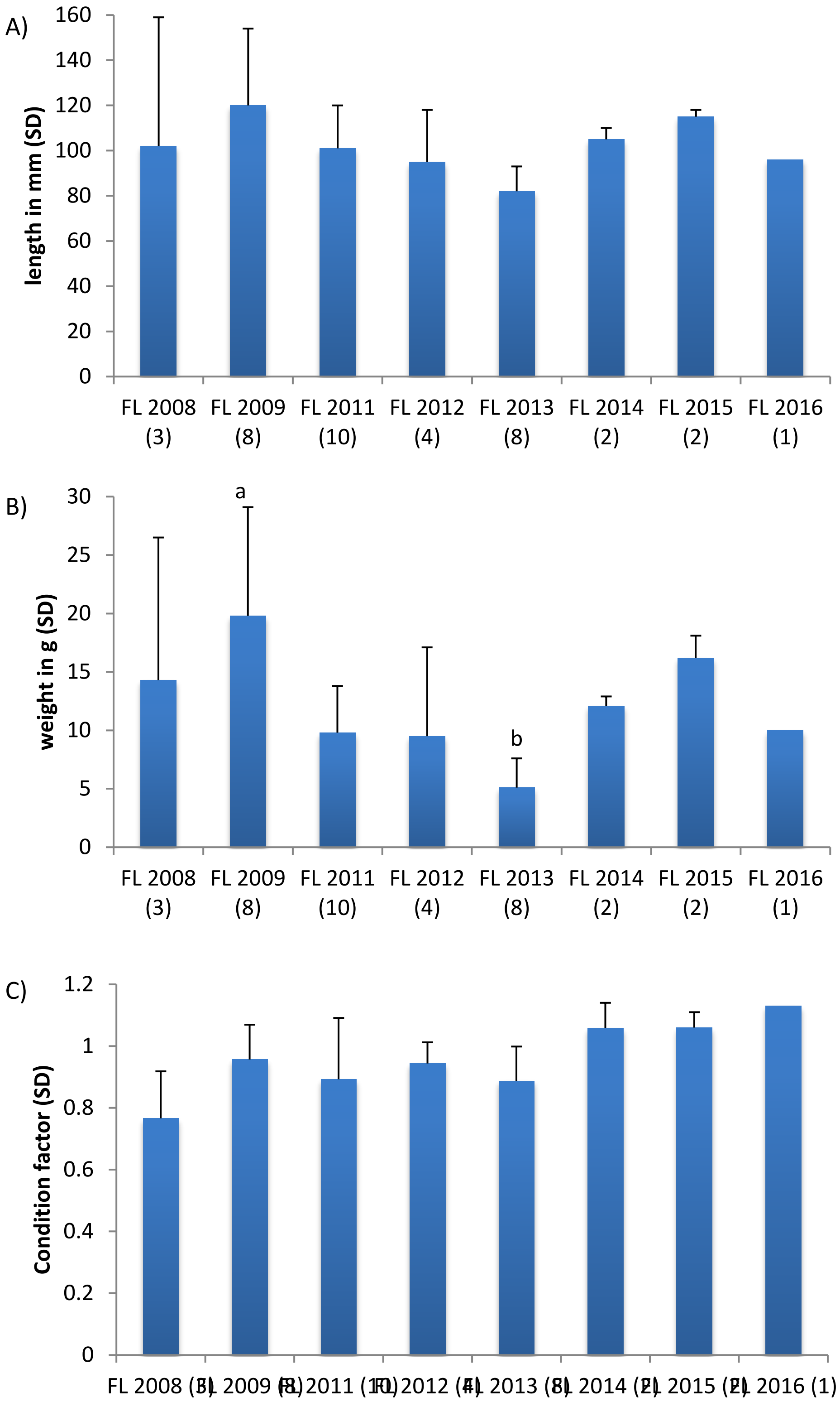


Figure 103. Mean a) length (mm), b) weight (g), and c) condition factor of unmarked coho salmon at Franz Lake by sampling year. Total number of coho salmon captured at Franz Lake per year are presented in parentheses.

3.6.3.3 *Somatic Growth Analyses*

A total of 615 otoliths were analyzed from fish ranging in fork length from 37-111 mm (mean of 67 mm and standard deviation of 13 mm). Otolith derived back-calculated estimates of somatic growth rate ranged from 0.31 to 0.87 mm/day with an average of 0.54 mm/day. Of these otoliths, 399 individuals (65%) were classified as unmarked, 215 as marked, and unrecorded for one individual. Across all years, otolith sample sizes were greatest at Campbell Slough and Whites Island, which accounted for 14% and 10%, respectively, of all 615 estimates of growth rate, while at the reach level, sample abundance was greatest in Reaches F, G, and H, which comprised 31%, 22%, and 21%, respectively, of all estimates of growth rate. Lastly, at the stock level, West Cascades fall and Spring Creek fall stocks were the most numerous, and accounted for 33% and 29%, respectively, of the 615 otoliths.

Our GLM approach to understand what factors explain variability in somatic growth rate indicated the importance of when fish were collected and secondarily by where they were collected. Generally, somatic growth rate indicated an increasing trend with river kilometer and a decreasing trend, of approximately 6%, over the eight years of this study (Figure 104). Specifically, our GLM analysis of the Baseline dataset indicated that two models were indistinguishable because they had a delta AIC ≤ 2.0 (Table 30). In these two top models, variability in somatic growth rate was best explained by fork length and year, Julian day, Julian day^2, and river kilometer, and to a lesser degree by off-channel distance. The best model (i.e., delta AIC of 0.0 in Table 30) showed a significant negative relationship between growth rate and Julian day and all years (except 2007), and a significant positive relationship was observed between growth rate and fork length, Julian day^2, and river kilometer.

When we used the best model from the Baseline dataset (i.e., growth rate ~ year + Julian day + Julian day^2 + fork length + river kilometer; hereafter referred to as the baseline model) to investigate the relative importance of prey richness and density (Prey dataset), predator and conspecific density (Predator and conspecific dataset), genetic stock and hatchery/unmarked (Stock and hatchery/unmarked dataset), and organochlorines pesticides and industrial contaminants (Toxins dataset) we found improvement in the amount of variability explained in the Stock and Toxins datasets only (Table 30). Specifically, the best model in the Stock dataset included an interaction between stock and hatchery/unmarked and it had a delta AIC value that was 19.6 lower than the baseline model, suggesting that including both of these variables substantially improved explanatory power (Figure 105). The best model (i.e., delta AIC of 0.0 in the Stock dataset of Table 30) showed a significant negative relationship between growth rate and Julian day, and several years (2008, 2010, 2011, and 2013), and a significant positive relationship was observed between growth rate and fork length, Julian day^2, and river kilometer.

Analysis of the Toxins dataset indicated three models that were indistinguishable (Table 30). In fact, one of these three models was the baseline model, which suggested that toxins provided a weak improvement to explaining variability in growth rate. The best model (i.e., delta AIC of 0.0) included the variable of industrial contaminants, and this model had an AIC that was only 1.16 lower than the baseline model. The best model showed a significant negative relationship between growth rate and Julian day, all years (except 2007), a marginally significant negative relationship between growth rate and industrial contaminants (Figure 105), and a significant positive relationship was observed between growth rate and fork length, Julian day^2, and river kilometer.

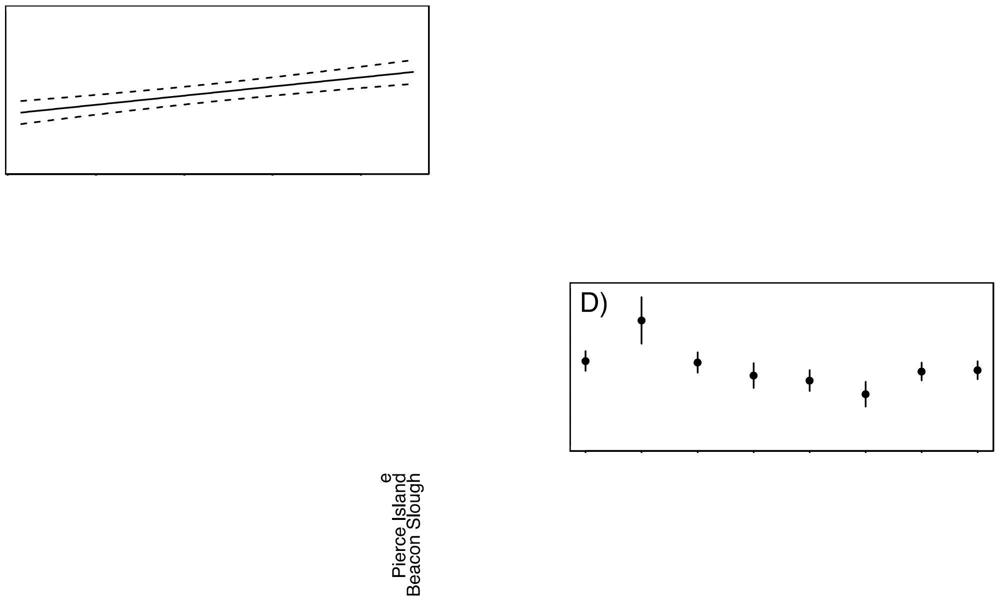
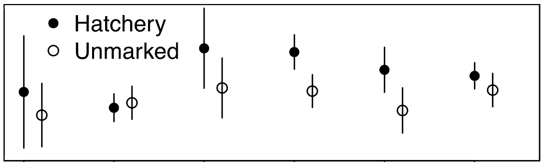


Figure 104. Model fits of growth rate (mm/day) with respect to A) river kilometer, B) reach, C) site, and D) year. The dashed lines in plot A and whiskers in plots B, C, and D represent 95% confidence limits. Reach A (plot B) is only represented by fish collected from one site, Point Adams, and 2007 (plot D) is only represented by fish from one site, Campbell Slough (Table 1). A general trend was evident of increasing growth rate (by approximately 13%) with increasing distance from the river mouth; however, that pattern masked considerable variation within reaches (panels A-C). A 6% decline in growth rate was observed over the course of the study (panel D).

Table 30. Results of the generalized linear modeling approach that assessed what independent variables explained variability in somatic growth rate. Five overlapping datasets were analyzed separately (see Table 3); *Baseline, Stock and Hatchery/unmarked, Prey, Predators and Conspecifics*, and *Toxins*. Organochlorine pesticides and industrial contaminants are labelled as OrgPest and IndCon, respectively. Relative likelihood (Rel) is the likelihood of a model given the data, and AIC weight (Wt) is the discrete probability of each model. Only models that are indistinguishable (i.e., D AIC of ≤2.0) are displayed.

Data	**Model**	**AIC**	**D AIC**	**Rel**	**Wt**
Baseline	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM	-1265.52	0.00*	1.00	0.51
Baseline	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + Off-channel	-1264.11	1.41*	0.49	0.25
Stock	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + Stock x Hatchery/Unmarked	-1101.69	0.00*	1.00	0.98
Stock	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + Stock + Hatchery/Unmarked	-1092.81	8.88	0.01	0.01
Stock	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + Stock	-1090.84	10.8	0.00	0.00
Stock	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM	-1082.06	19.6	0.00	0.00
Stock	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + Hatchery/Unmarked	-1081.61	20.0	0.00	0.00
Stock	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + Off-channel	-1080.27	21.4	0.00	0.00
Toxins	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + IndCon	-750.54	0.00*	1.00	0.35
Toxins	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + OrgPest + IndCon	-750.18	0.36*	0.84	0.29
Toxins	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM	-749.38	1.16*	0.56	0.19
Toxins	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + OrgPest	-747.99	2.55	0.28	0.10
Toxins	Year + Julian Day + Julian Day^2 + Fork Length + RiverKM + Off-channel	-747.39	3.15	0.21	0.07



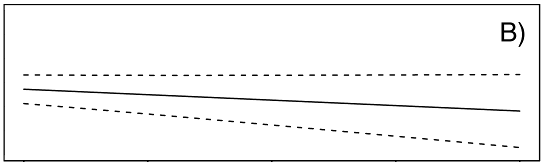


Figure 105. Model fit of growth rate (mm/day) with respect to A) genetic stock (hatchery and unmarked fish are indicated by the open and closed circles, respectively) and B) industrial contaminants (PCB's and PBDE's). The whiskers in plot A and the dashed lines in plot B represent 95% confidence limits.

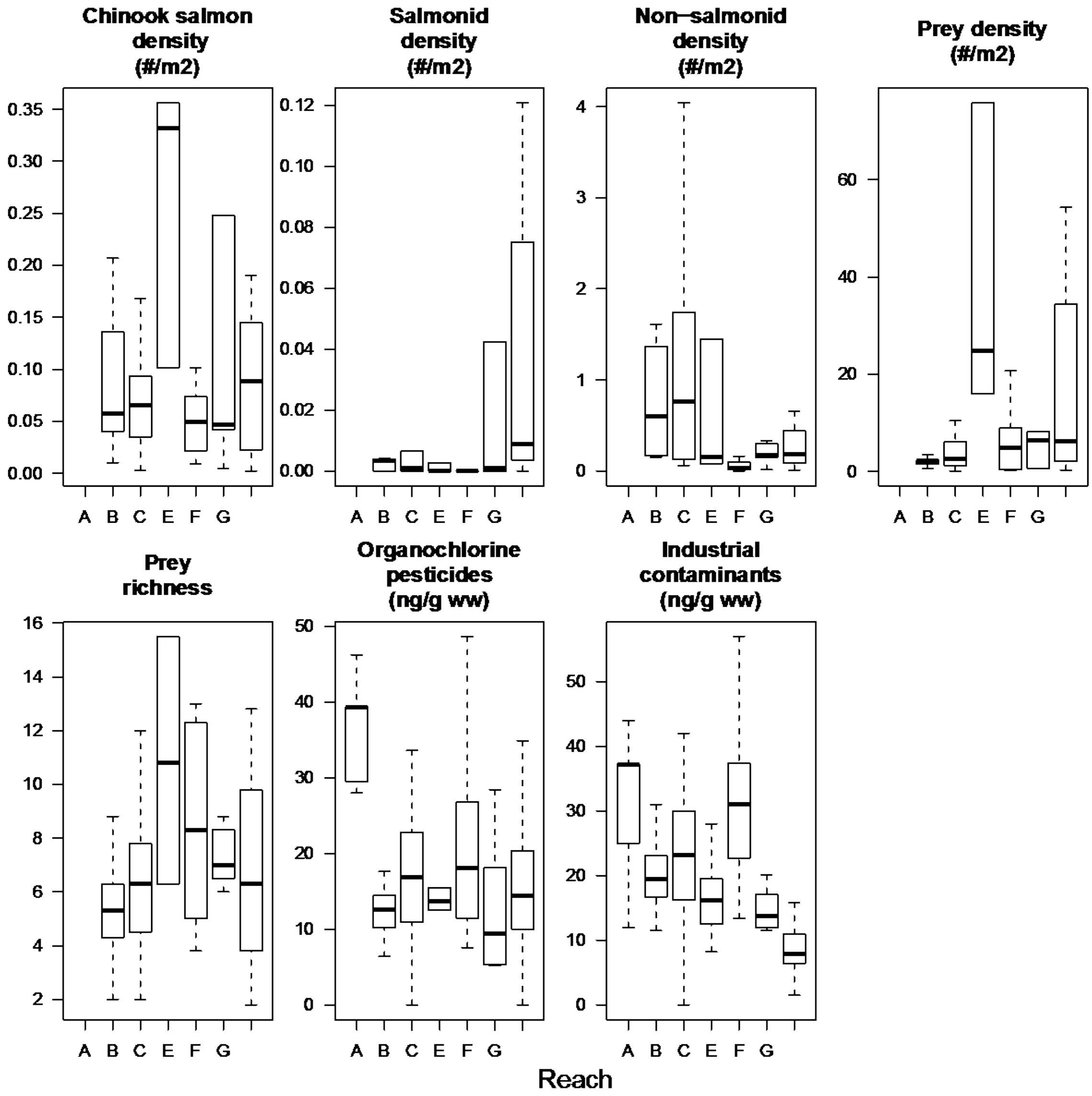


Figure 106. Boxplots of various fish, invertebrate, and contaminant data with respect to seven reaches and pooled across years. Medium, interquartile range, and extreme values are indicated as the horizontal line, box, and whiskers, respectively. The organochlorine pesticides and industrial contaminants data for Reach A were only collected from one site (Point Adams) in 2005. A boxplot of predator density was not included because this variable consisted mostly of zeros (87%).

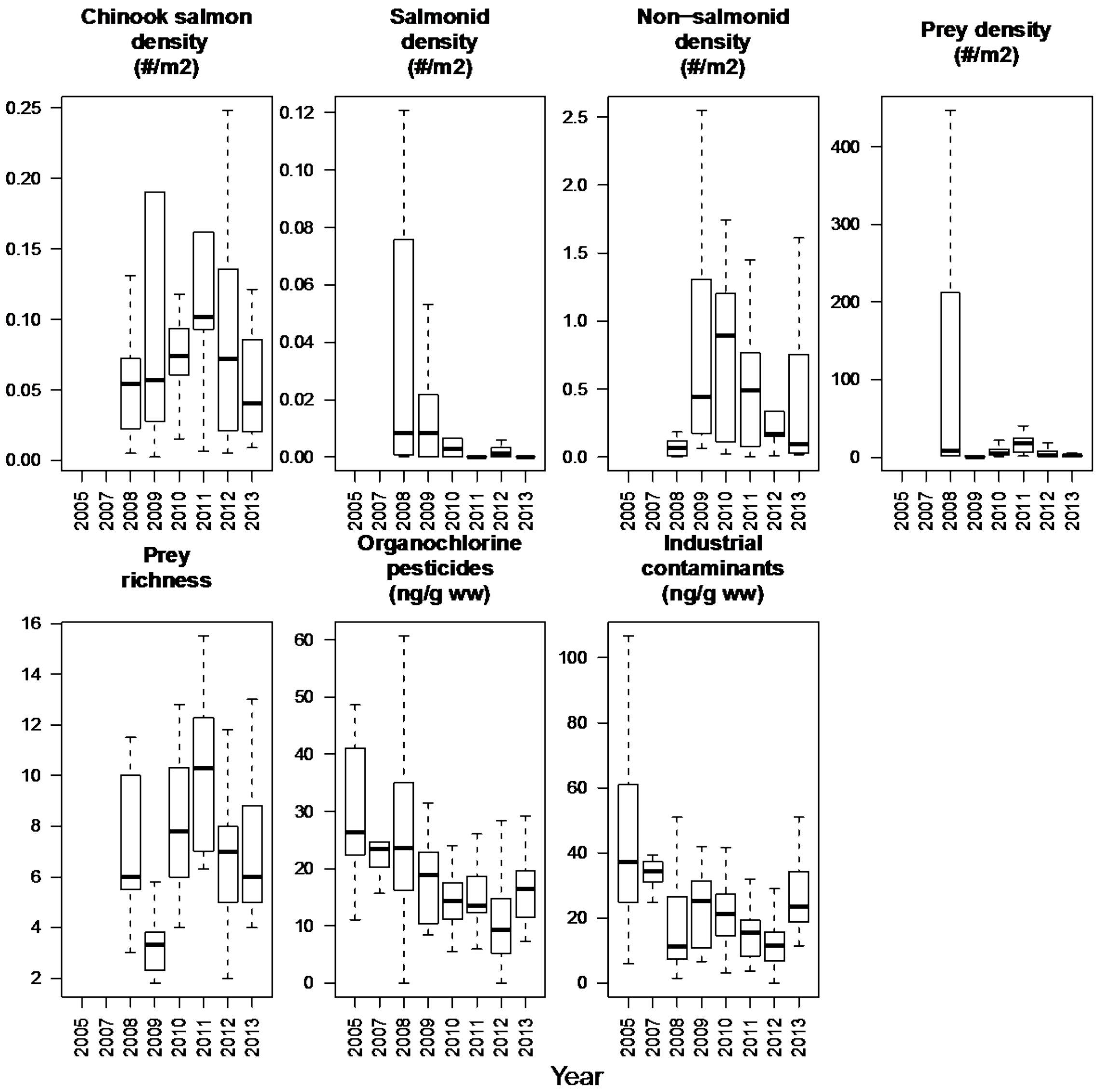


Figure 107. Boxplots of various fish, invertebrate, and contaminant data with respect to eight years and pooled across reach. Medium, interquartile range, and extreme values are indicated as the horizontal line, box, and whiskers, respectively. The organochlorine pesticides and industrial contaminants data for 2005 were only collected from one site (Point Adams). A boxplot of predator density was not included because this variable consisted mostly of zeros (87%).

3.6.3.4 *Lipid Content of Juvenile Chinook Salmon*

In this report we present data on lipid content and the proportion of lipid present as triglycerides in juvenile Chinook salmon between 2007 and 2015 (Figure 108). Because these measures did not differ significantly between marked and unmarked fish, samples from both groups of fish were pooled, increasing sample size. The 2016 samples will be analyzed when genetics data are available for the 2016 fish. Lipid samples from 2014 were compromised due to a freezer failure, so are not included in this analysis.

Currently, only one sample from Ilwaco Slough has been analyzed, so trends at this site cannot be evaluated. Significant differences in lipid content among years were observed at Welch Island ($p < 0.0001$), Whites Island ($p < 0.0001$), Campbell Slough ($p < 0.0001$), and Franz Lake ($p = 0.0049$). At Welch Island, Whites Island and Campbell Slough, the lipid content of fish collected in 2015 was relatively high in comparison to other sampling years, while at Franz Lake it was intermediate (Tukeys multiple range test, $p < 0.05$). Differences in % triglycerides were observed among years for Welch Island ($p = 0.0046$), Whites Island ($p = 0.0023$), Campbell Slough ($p < 0.0001$), but not Franz Lake ($p = 0.4524$). At all three sites where significant annual differences were detected, % triglycerides in fish collected in 2015 were relatively high compared to other sampling years.

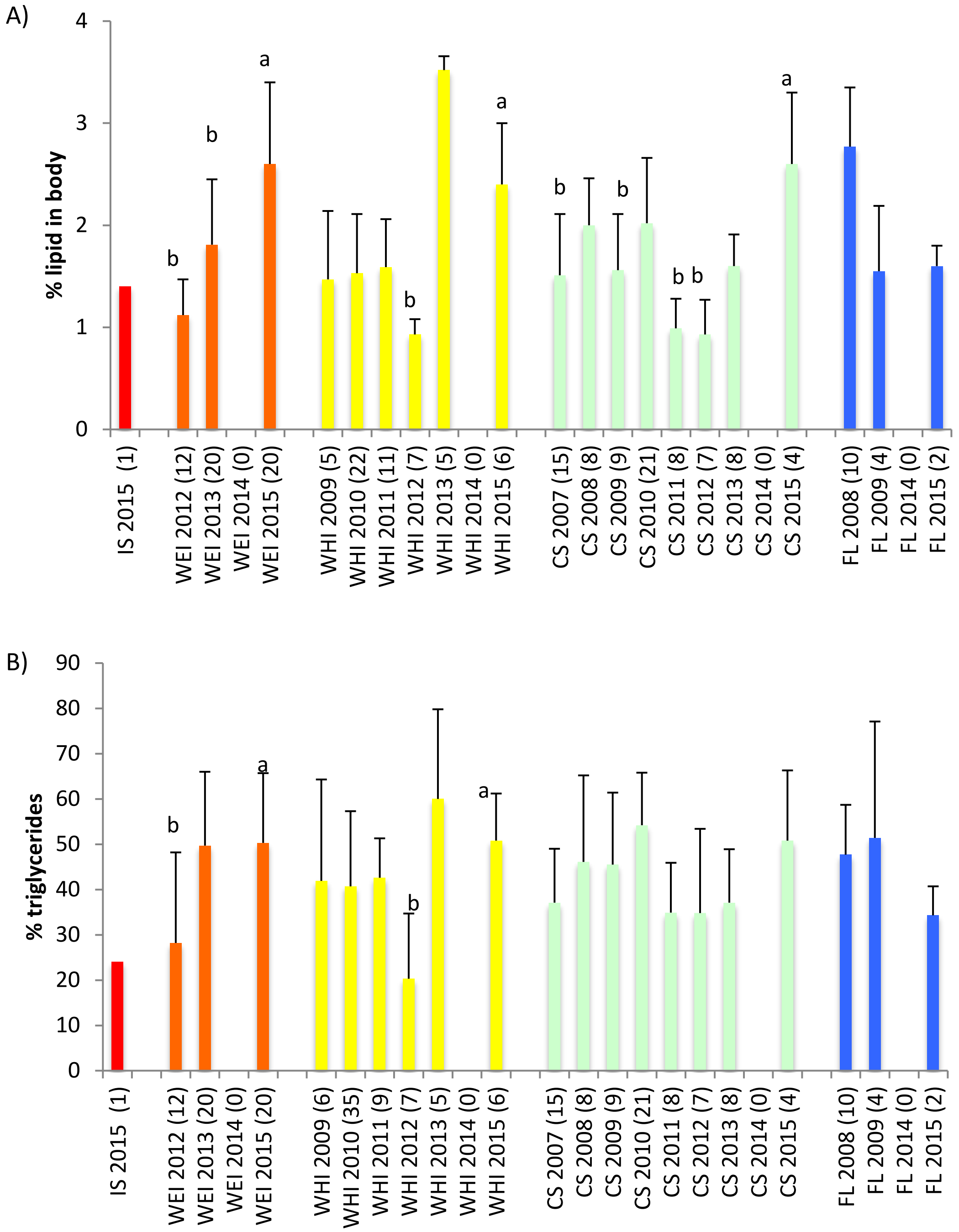


Figure 108. Mean (SD) a) body lipid content (%) and b) % of total lipids that were triglycerides in Chinook salmon collected from trend sites in 2015 compared to previous sampling years. The number of composite samples analyzed is indicated in parentheses. Letter designations indicate values within each site that are significantly different from the values measured in 2015 (ANOVA, Tukey's LSD test, $p < 0.05$). IS = Ilwaco Slough; WEI = Welch Island, WHI = Whites Island, CS = Campbell Slough, FL = Franz Lake.

3.6.3.5 *Contaminants in Juvenile Chinook Salmon*

Contaminant data for 2015 and 2016 are not yet available. The 2015 samples are currently under analysis and will be included in a later version of this report. The 2016 samples will be analyzed once genetics data are available for sample compositing.

3.6.4 Changes in fish community composition and salmon occurrence with the tidal cycle

The mean water level in the Columbia River mainstem was higher in May compared to June (Figure 3) and at all three sites the low water level was slightly higher in the May sampling period then the June sampling period (Table 31). The tidal influence may have been greater at lower river flow in June than during the higher flow in May. At Ilwaco Slough the sampling time points included from 0-3 hours for both May and June before the water level was too high to sample efficiently. At Whites Island, May sampling time points included 0-4 hours, while June sampling time points included 0-3 hours before the water level was too high to sample efficiently. At Campbell Slough, May sampling time points included 0-2 hours while June sampling time points included 0-4 hours before the water level was too high to sample efficiently.

Table 31. Depth of water (mean±stdev) at each site at each time point. n indicates total number of beach seine sets at each time point.

Sampling time (hrs after low tide)	Ilwaco Slough (m)			Whites Island (m)			Campbell Slough (m)		
	May	June	Average	May	June	Average	May	June	Average
0	0.03±0.01 (n=3)	0.15±0.01 (n=3)	0.09±0.07 (n=6)	0.33±0.24 (n=2)	0.19±0.15 (n=2)	0.25±0.19 (n=4)	1.6±0.17 (n=3)	1.5±0.0 (n=3)	1.55±0.12 (n=6)
1	0.8 (n=1)	0.91 (n=1)	0.85±0.08 (n=2)	0.15 (n=1)	0.3 (n=1)	0.23±0.11 (n=4)	2.1 (n=1)	1.5 (n=1)	1.8±0.42 (n=2)
2	0.91 (n=1)	0.91 (n=1)	0.91 (n=2)	0.5 (n=1)	0.8 (n=1)	0.63±0.18 (n=2)	2.1 (n=1)	1.5 (n=1)	1.8±0.42 (n=2)
3	1.07 (n=1)	1.12 (n=1)	1.1±0.09 (n=2)	1.2 (n=1)	1.2 (n=1)	1.2 (n=2)		1.5 (n=1)	1.5 (n=1)
4				1.2 (n=1)		1.2 (n=1)		1.8 (n=1)	1.8 (n=1)

Our sampling indicated that species composition, species richness (number of species) and fish density (CPUE) changed with tide and water depth at all three sites. Our findings at each of the sites are summarized below.

Ilwaco Slough. At low tide, at a water depth of 0.09 ± 0.07 m, the CPUE at Ilwaco Slough was 3316 fish per 1000 m^2. The catch was dominated by threespine stickleback and banded killifish, based on CPUE and percentage of the total catch (Figure 109). One hour after low tide, in conjunction with a shift in water flow to a water depth of 0.85 ± 0.08 m, we saw an increase in CPUE to 4612 fish per 1000 m^2 (Figure 2A). The CPUE and percentage of total catch increased for Pacific staghorn sculpin and shiner perch, but decreased for killifish and stickleback (Figure 109). Two hours after low tide, the water depth increased to 0.91 m, and CPUE declined to 2179 fish per 1000 m^2. Stickleback, Pacific staghorn sculpin, and shiner perch dominated the catch. The CPUE and percent composition of killifish continued to decrease during this time. Three hours after low tide, the water depth continued to increase to 1.10 ± 0.09m. CPUE continued to decrease (649 fish per 1000 m^2) and while the site was still dominated by same three species of fish (stickleback, Pacific staghorn sculpin and shiner perch), the percentage of killifish in the catch was very low (Figure 109). Overall, six species of fish were caught at Ilwaco Slough, but did not include juvenile salmon. CPUE was the highest at low tide and decreased with rising water.

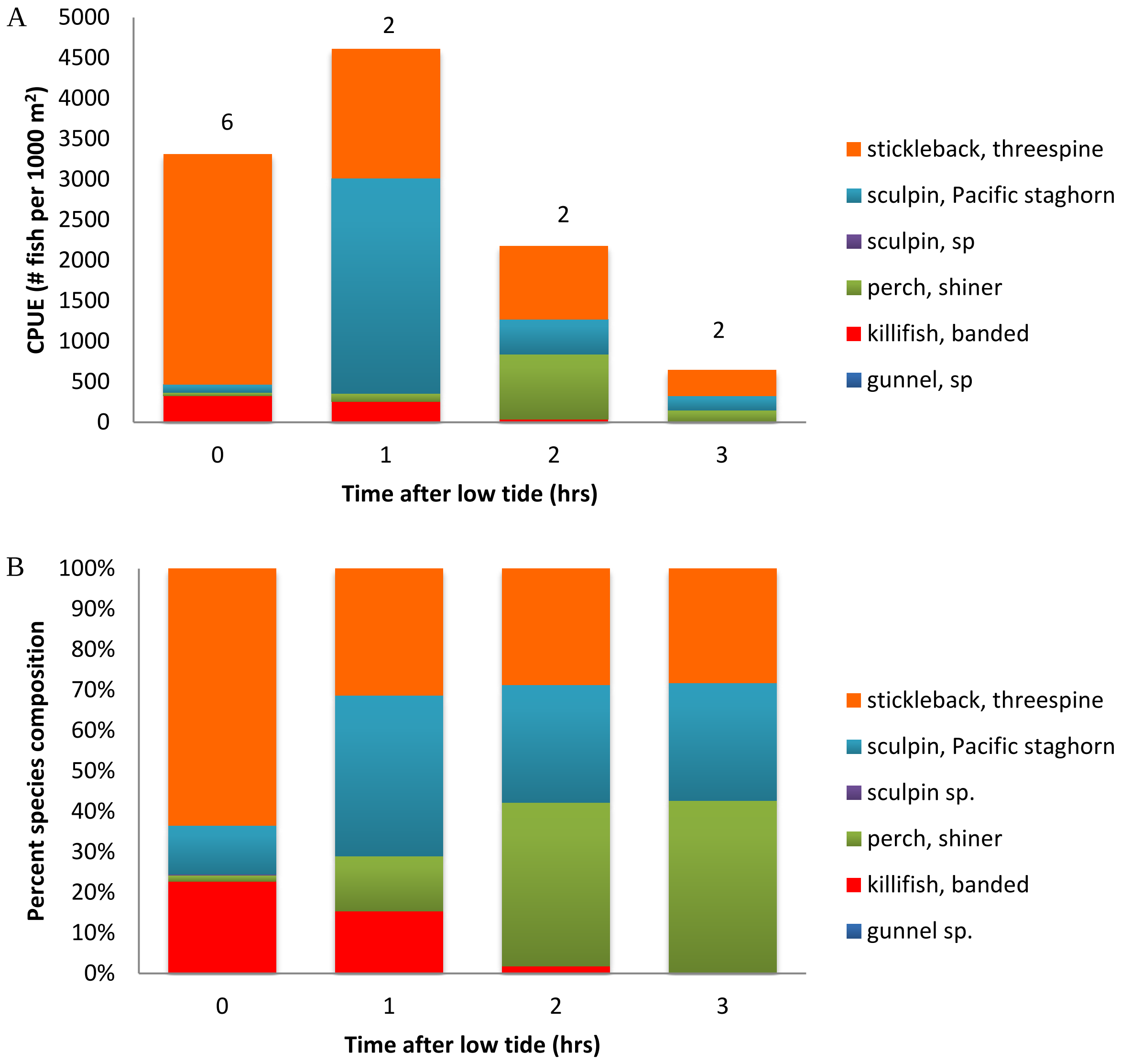


Figure 109. A) Catch per unit effort (CPUE) and B) percent species composition at Ilwaco Slough at hourly intervals during the incoming tide. Values are based on combined data from both May and June sampling periods. The numbers above the CPUE values represent the total number of sampling events at each time point.

Whites Island. At low tide/slack water, at a water depth of 0.26 ± 0.19 m, CPUE was low (342 fish per 1000 m^2), and banded killifish, threespine stickleback, and Chinook salmon dominated the catch (Figure 110). One hour later, the water depth increased to 0.23 ± 0.11 m, and CPUE decreased to 185 fish per 1000 m^2. Killifish, stickleback and Chinook salmon were again the dominant species. An increase in CPUE of Chinook salmon was observed while the CPUE of stickleback decreased. These patterns are also evident from percent composition calculations (Figure 110). Yellow perch and sculpin species were also caught in small numbers at low tide and one hour after low tide, but were absent at later sampling time points. Two hours after low tide, the water depth increased to 0.63 ± 0.18 m, accompanied by a huge increase in CPUE (3417 fish per 1000 m^2). This was mostly due to an increase in stickleback CPUE, which is also reflected in percentage of species composition (Figure 110). A slight increase in Chinook CPUE was observed but the percent of Chinook salmon in the catch decreased because of the dominance of stickleback. Small numbers of peamouth were also caught at this time point. Three hours after low tide, the water depth increased to 1.20 m. A decrease in CPUE was observed (1309 fish per 1000 m^2), although the catch was still dominated by same three species (killifish, Chinook salmon and stickleback). Chinook salmon CPUE increased, but the CPUE of killifish and stickleback declined. The percentage of stickleback in the catch decreased, while the percentages of Chinook salmon and killifish increased. Four hours after low tide, the water depth remained at 1.20 m and the CPUE continued to decline (233 fish per 1000 m^2) and only stickleback and Chinook salmon were caught. The CPUE of stickleback decreased significantly while the CPUE of Chinook salmon remained relatively the same. Chinook salmon made up 60% of the catch, while stickleback made up the remaining 40%. Overall seven species of fish were caught at Whites Island. CPUE for all species was the highest at low tide and decreased with increasing water depth. Chinook salmon CPUE was the highest 3-4 hours after low tide, with Chinook CPUE changing quite substantially depending on the stage of the tidal cycle (Figure 111).

More unmarked Chinook salmon (n = 152) were captured at Whites Island than marked Chinook salmon (n = 6). Unmarked Chinook were smaller than marked Chinook (60 ± 9 mm compared to 75 ± 5 mm; Figure 112). At lower tide (time point 0 and 1), only smaller unmarked Chinook were captured at the site, but at higher tide (time points 2-4) larger unmarked as well as marked Chinook were captured (Figure 112).

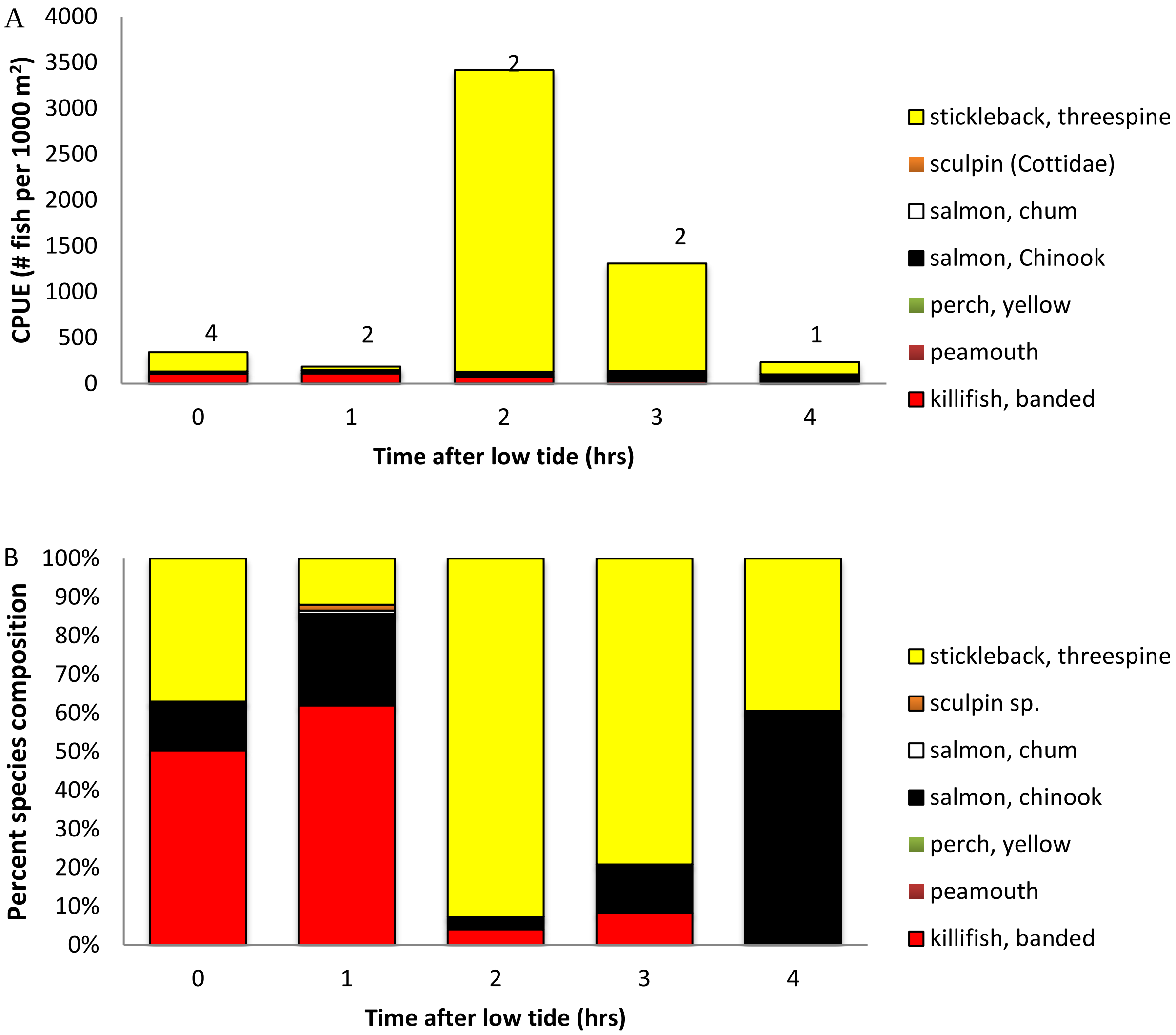


Figure 110. A) Catch per unit effort (CPUE) and B) percent species composition at Whites Island at hourly intervals during the incoming tide. Values are based on combined data from both May and June sampling periods. The numbers above the CPUE values represent total number of sampling events at each time point.

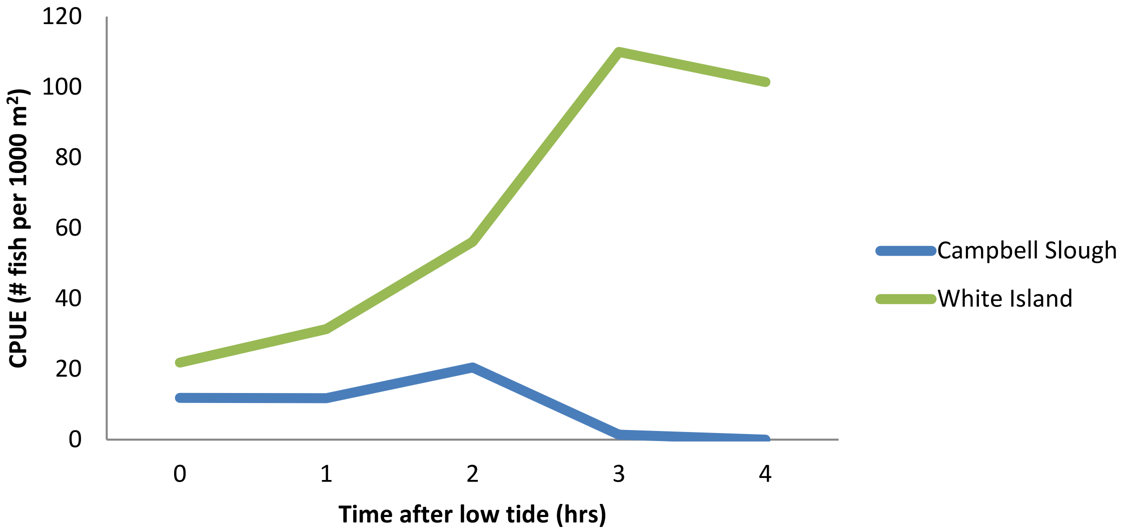


Figure 111. Mean catch per unit effort (CPUE) of Chinook salmon at Whites Island and Campbell Slough over the tidal cycle sampling period. CPUE from Ilwaco Slough is not shown because Chinook salmon were not caught at Ilwaco Slough. CPUE values are based on combined data from May and June.

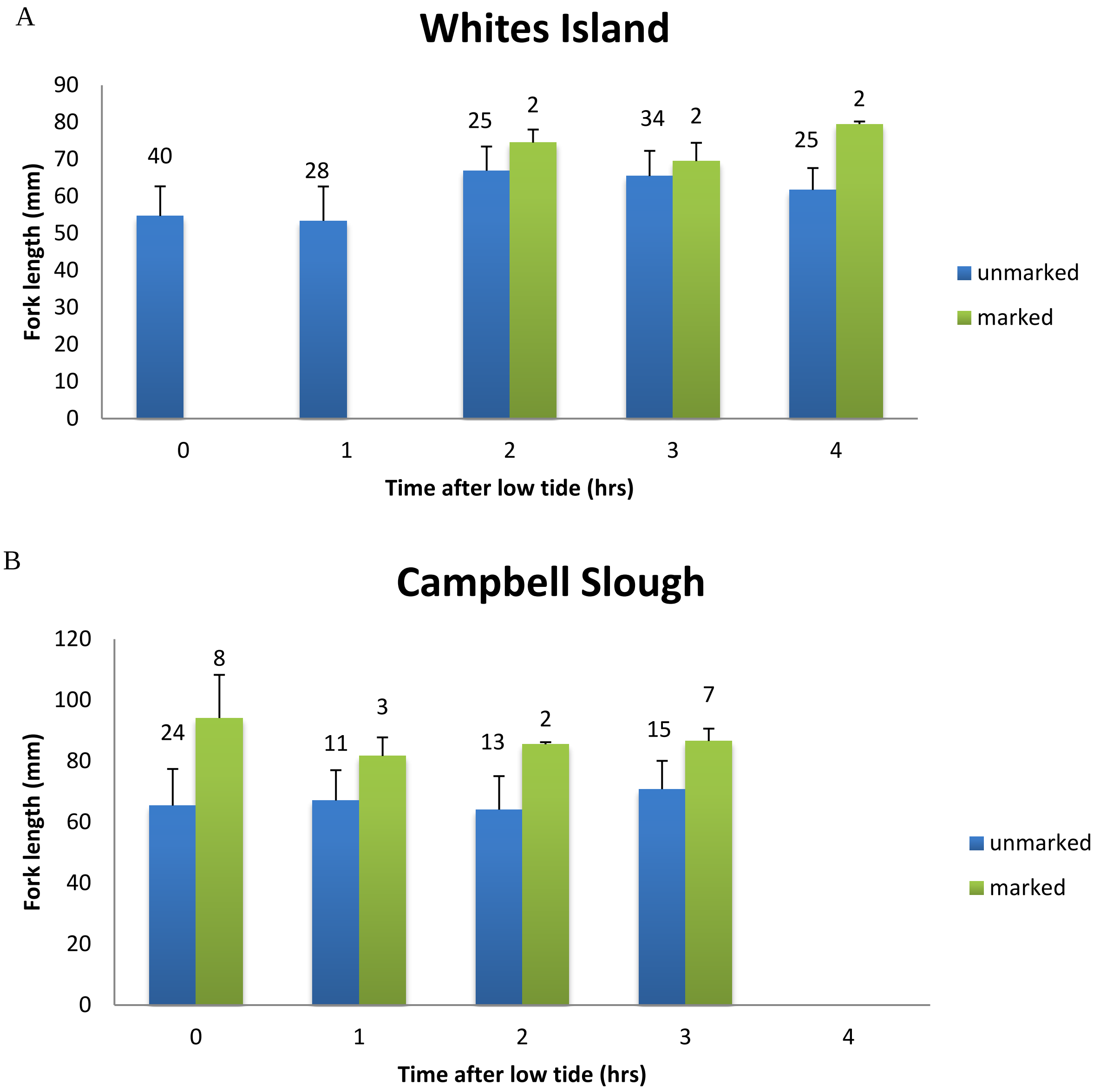


Figure 112. Mean length in mm (± SD) of Chinook salmon caught at A) Whites Island and B) Campbell Slough during the incoming tide. Values are based on combined data from both May and June sampling periods. The numbers above the bars represent the numbers of fish caught.

Campbell Slough. During low tide, at 1.55 ± 0.12 m water depth, CPUE was low (119 fish per 1000 m^2) and the catch was dominated by banded killifish, threespined stickleback, Chinook salmon, and yellow perch (Figure 113). Other species caught included goby sp., northern pikeminnow, coho salmon, sculpin sp., American shad, and largescale sucker. One hour after low tide, the water level increased to 1.80 ± 0.42 m. A slight decline in CPUE was observed (94 fish per 1000 m^2) and killifish, stickleback, Chinook salmon, and yellow perch continued to dominate the catch (Figure 113). Other species caught included goby, northern pikeminnow, and largescale sucker. Coho salmon, sculpin and shad were absent from the catch. Two hours after the low tide, the water depth remained at 1.80 ± 0.42, and a slight increase in CPUE was observed (149 fish per 1000 m^2). The catch was still dominated by killifish, stickleback, Chinook salmon, and yellow perch (Figure 113). The CPUE of other species, including chiselmouth, northern pikeminnow, sculpin, sucker, and cyprinid sp., increased slightly. Three hours after low tide at ~1.5 m depth, CPUE stayed relatively consistent (136 fish per 1000 m^2). The catch was still dominated by killifish, stickleback, Chinook salmon, and yellow perch (Figure 113). The CPUE of other species, including peamouth, northern pikeminnow, and cyprinid sp., continued to increase, while the CPUE of Chinook salmon declined. Four hours after low tide, at ~1.80 m water depth, CPUE decreased to 37 fish per 1000 m^2. Killifish, carp and pumpkinseed dominated the catch. Overall 18 species of fish were caught at the site. Chinook salmon were caught from low tide to three hours after low tide, but were absent four hours after low tide. Chinook salmon CPUE was the highest at two hours after the low tide (Figure 113, Figure 111).

More unmarked Chinook salmon (n = 63) were captured at the site compared to the marked Chinook salmon, n = 19). Both smaller unmarked Chinook salmon and larger marked Chinook salmon were caught at time points 0-3 hours (Figure 112). Overall, unmarked Chinook salmon were smaller than marked Chinook salmon captured at the site (67 ± 11mm and 91 ± 13mm, respectively).

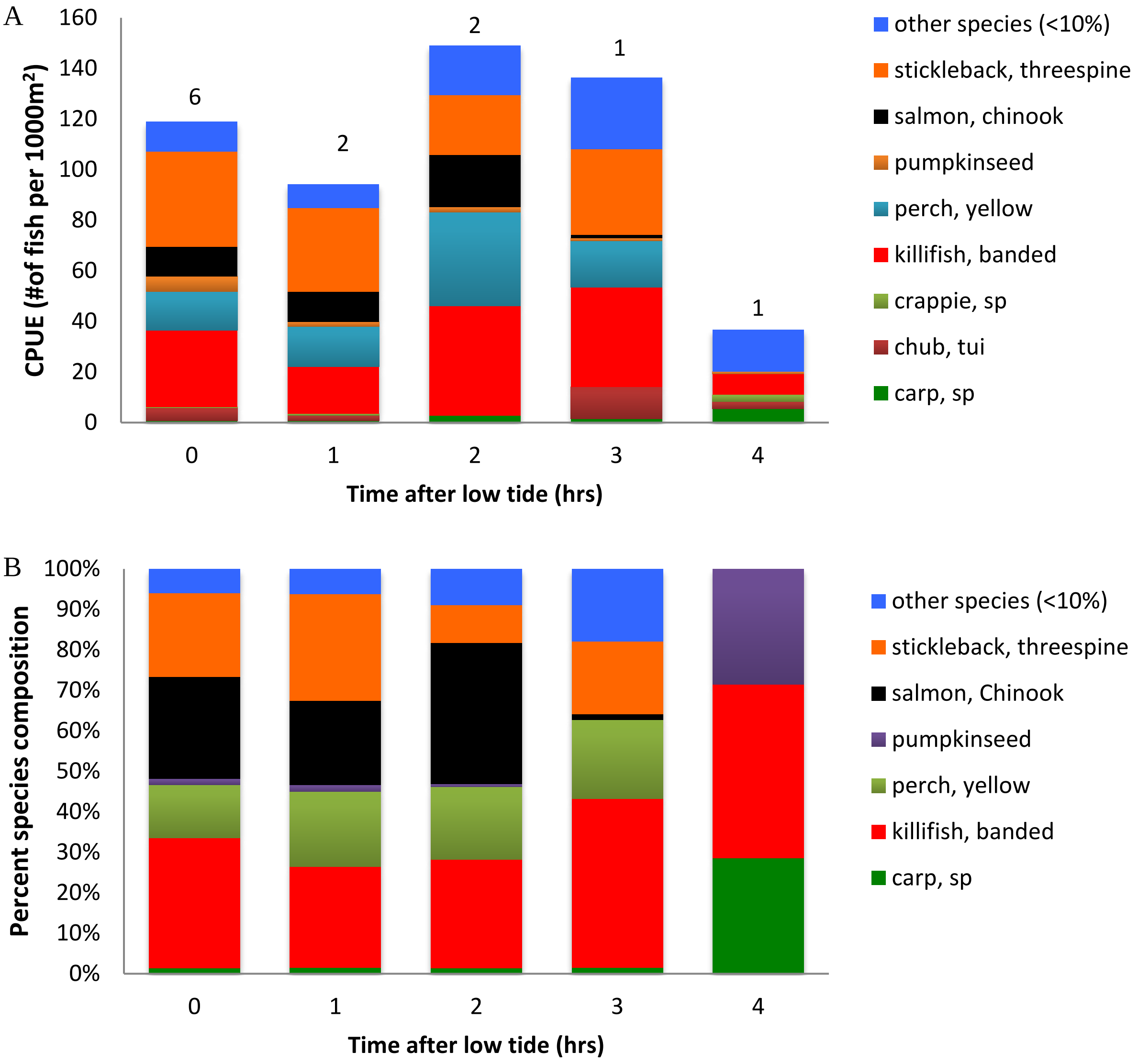


Figure 113. A) Catch per unit effort (CPUE) and B) percent species composition at Campbell Slough at hourly intervals during the incoming tide. Species that were less than 10% of the total catch at any sampling event are categorized as other species, which include: chiselmouth, goby sp., cyprinid sp., peamouth, northern pikeminnow, coho salmon, sculpin sp., shad, and sucker. Values are based on combined data from both May and June sampling periods. The numbers above the CPUE values represent total number of sampling events at each time point.

3.6.5 PIT-Tag Array Monitoring of Juvenile Salmon Residence

At the Campbell Slough PIT tag array, only two fish were detected in 2016. Both individuals were northern pikeminnow in mid-May and neither fish exhibited residency in the vicinity of the array. Salmon were not detected on the array in 2016.

4 Discussion

4.1 Mainstem Conditions

In terms of temperature and other water quality metrics, the year 2016 was similar to 2013 and 2014. Temperatures were cooler compared to 2015, but higher than the long-term average prior to the freshet. Following the freshet, temperatures were similar to previous years. During the year 2001 when river discharge was very low, daily temperatures in the mainstem were about average, with the exception of a few weeks in May where temperatures were high (Hanson et al. 2016). In contrast, temperatures in 2015 were higher than the long-term average throughout the spring and summer until approximately September.

In 2015, the total number of days where the daily average river temperature exceeded 19°C was 102 (determined from values measured at Camas, WA) or 105 days (determined from values measured at BAT), whereas in 2016 it was ~80 days at both sites. There were also similar numbers of days where temperatures in 2016 exceeded 21°C at both sites (~40 days). The number of days with average temperatures exceeding 19°C was similar to both 2013 and 2014, and the number of days exceeding 21°C was less than 2013 (but similar to 2014). Dissolved oxygen concentrations were always within the range of adequate water quality at the mainstem sites. Oxygen saturation with respect to the atmosphere tracked primary production by phytoplankton, with high values observed during the period of spring growth.

Nutrient concentrations in the mainstem showed similar patterns to previous years, with maximum values of nitrate observed in the winter. Over the course of the spring and summer, nitrate concentrations declined, presumably due to uptake and incorporation into the tissues of primary producers.

Phytoplankton biomass showed a typical temporal pattern at both Beaver Army Terminal (RM-53) and Camas, with a peak in the spring (~April) that was dominated by chain-forming diatoms. Cyanobacteria and green algae were not observed in significant abundance at the mainstem sites.

4.2 Abiotic Site Conditions

A comparison of conditions in the mainstem and shallow water trends sites shows that while conditions may be similar during periods of high flow, they tend to diverge as flows subside. For example, following the freshet and into the summer months, large daily variations in temperature, dissolved oxygen, and pH were observed at the trends sites. It is important to understand the relationship between river flow and habitat quality for juvenile salmonids, since this is likely to influence their growth and survival prior to ocean entry. 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 2015). 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).

A secondary influence of temperature on water quality and food availability is the relationship between temperature and prevalence of cyanobacteria in the phytoplankton assemblage. Cyanobacteria can become dominant when temperatures increase and water column stratification is high. We observed recurrent increases in the prevalence of cyanobacteria during the summer months in shallow water habitats, primarily in Campbell Slough and Franz Lake Slough. Surprisingly, however, cyanobacteria were noted in April at Whites Island, indicating that under certain conditions, they can increase in numbers in the absence of high temperatures. The very high abundances observed at Franz Lake Slough in the summer could reflect the increased water retention times created by beaver dams at this location. It would be helpful to better understand the relationship between beaver activity and cyanobacteria blooms to mitigate potentially negative effects of toxins associated with many bloom-forming species.

At Ilwaco Slough, in Reach A, seasonal differences in the salinity regime are also observed, which influences the distribution and species composition of marsh plants (Section 3.3.3). Over the time series of observations since June 2009, Ilwaco Slough also had the greatest number of hours of low dissolved oxygen levels of the trends sites (Figure 30). All observations after May 2012 indicate that there were more than 50 hours per month where dissolved oxygen levels were below 6 mg L^{-1}. Dissolved oxygen levels at the other sites were generally above thresholds most of the time, with some exceptions (e.g., Campbell Slough in June 2011).

4.3 Habitat Structure

4.3.1 Sediment and Hydrology

Sedimentation and hydrologic processes are the primary underlying environmental drivers dictating wetland elevation, plant species assemblages, vegetation productivity, and overall wetland condition. Understanding these processes is important for restoration planning, ensuring wetland processes are intact to keep up with sea level rise or changing fluvial inputs, and determining carbon sequestration rates as part of evolving carbon markets (Mcleod et al. 2011).

Hydrologic variability is an integral part of the tidal freshwater marsh dynamics observed in the lower river. In 2016, the hydrologic patterns were similar to those that are predicted to occur with greater frequency in future years: a series of winter and spring floods rather than a higher magnitude freshet in late spring and early summer (Hamlet et al. 2013). A flood event in December marked the peak high water for the year at the trend sites in the middle reaches of the lower river. At Ilwaco Slough water was slightly higher in March probably due to a flood event co-occurring with spring high tide. Based on water levels just below Bonneville Dam, it is likely that spring river flows elevated the water levels at Franz Lake to a greater extent in March than in December (our sensor was lost). The prolonged series of low magnitude peaks throughout the winter and spring resulted in moderate inundation levels in the upper estuary through the first three months of the growing season, but overall low inundation in the growing season as a whole. Inundation frequency in the lower estuary was less than in 2015, a markedly low water year, perhaps due to lower than average precipitation and runoff in the late spring and early summer from west side watersheds (www.usclimatedata.com). The effect of this water pattern on the vegetation is discussed below.

Sediment accretion rates are variable within the lower river and within individual sites, likely due to variation in elevation, sediment loading, and flood inundation frequency (Chmura et al. 2003) and may even be affected by the vegetation present (Marani et al. 2013). We installed sediment accretion stakes at different elevations within a site in an effort to capture this variability and generally found that accretion rates decreased with increasing elevation, however proximity to the tidal channel also seems to be an important variable. For example, a natural levee occurs along the channel at Secret River caused by increased sedimentation at the channel bank (Cazanacli and Smith 1998) and sedimentation continues to be higher there than at the marsh surface farther from the channel. Sediment accretion rates in the low marsh at Secret River have consistently been low or erosional since we started measuring in 2012. This could in part be due to decreased sediment deposition during times when wave action takes the sediment higher into the marsh at high tide (Davidson-Arnott et al. 2002). The site is likely often exposed to wave action due to the large fetch coming from the west, the prevailing wind direction in the lower estuary from March to October when vegetation is present and more likely to trap sediments (https://www.windfinder.com/windstatistics/astoria). The greatest sediment accretion rates have been measured at Whites Island in patch of *C. lyngbyei* located at a mid- to low-marsh elevation (1.3 m, CRD) very close to the primary tidal channel at the site (<10 m from marsh edge). This is a good example of conditions conducive to high accretion rates: proximity to the tidal channel, high inundation frequency (about 50 percent), and vegetation that produces high amounts of organic material and effectively traps mineral and organic material, both important sources of sediment accretion in tidal freshwater marshes (Neubauer 2008).

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

4.3.2 Vegetation Community Condition and Dynamics

An underlying premise of the EMP is that the sites included in the program are relatively undisturbed and as such provide a means of evaluating wetland process and function and to inform restoration design and monitoring. One method to evaluate the condition of lower Columbia River wetlands is to assess the floristic quality as compared to other wetlands in the region. Floristic quality assessments have been widely used nationally to evaluate and monitor natural areas and led to the development of a national database and calculator (Freyman et al. 2016). The method is based on conservation values (C-values) that are assigned to plant species. The Washington State Natural Heritage Program has developed the values of conservation for 2,721 native and non-native species in Washington and developed a floristic quality calculator (Rocchio and Crawford 2013). The mean C-values calculated for the trend site wetlands in the lower river ranged from a low of 2.4 at Campbell Slough to a high of 4.8 at Ilwaco Slough. To put these numbers in context it is helpful to know that the average native C-value for Washington State is 5.1 and is among the lowest in the country (Freyman et al. 2016). Rocchio and Crawford (2013) provide a guide to what the average scores mean relative to natural area condition in our region as follows:

5.0 = Intact
3.46 = Slightly impacted
2.16 = Highly impacted

In this context, the trend sites range from relatively intact to moderately impacted. The sites become more impacted by disturbance linearly with river kilometer, which makes sense due to the disturbance introduced by river flooding. The added disturbance presented by the cows at Campbell Slough could be the reason for this site having the lowest score. In all, this method seems useful to represent the condition of lower river wetlands and provide a means of assessing the condition of restoration sites.

4.3.2.1 *2016 Observations*

We observed some interesting patterns in vegetation species assemblage and cover in 2016. Vegetation in the channels of the trends sites varied compared to previous monitoring years. SAV cover was reduced at Secret River, whereas cover of native *Potomageton zosteraforma* increased at Whites Island and milfoil increased at Campbell Slough. Reduced SAV cover is often attributable to greater light attenuation from increased suspended sediments and/or phytoplankton in the water column (Dennison et al. 1993; Havens 2003). Perhaps changes in attenuation occurred at the sites differentially in 2016. Winter and spring storms may have increased turbidity from nearby tributaries, negatively affecting Secret River, while decreased spring freshet flooding the past two years may have reduced turbidity in the Columbia River allowing increases at Whites Island and Campbell Slough. At Franz Lake low SAV is likely attributable to beaver activity that increases water depth and herbivory (Parker et al. 2007).

Vegetation cover was reduced for the second year in a row at Ilwaco Slough. The reasons for this are not clear, however lower than normal inundation frequencies and higher than normal salinity may be causal factors. In 2016, the Secret River low marsh site had the lowest cover measured during our monitoring period. The reason for this may be the same as that which reduced the SAV cover in the channel at the site; perhaps high turbidity associated with localized spring storm events and tributary flooding. Conversely, there was high cover at the Secret River, Welch Island, and Whites Island high marshes. Biomass at Whites Island was much greater in 2016 than in previous years, indicating that conditions were favorable for vigorous growth. Similarly, cover was also high at Cunningham Lake and Campbell Sough. At the latter, however, evidence of cow grazing was apparent and likely reduced the cover of *P. arundinacea* in the high marsh. Cover in the low marsh, where the cows have less effect, was very high in 2015 and 2016 relative to previous years. *P. amphibium* continued to flourish at Franz Lake, however overall cover was lower than in 2015 primarily due to reduced cover of dead *P. amphibium* and willow (*Salix lucida*) saplings in 2016. Willow increased due to reduced beaver activity in 2015 compared to all previous years and in 2016 active willow harvesting was once again observed. One final observation at the sites was that there seemed to be a positive relationship between the number of species and the aerial cover, particularly at Cunningham Lake and Franz Lake. This could be explained by the favorable conditions at the site not only increase cover, but also allow a greater number of species to grow. A higher number of species could also contribute to greater cover. At Campbell Slough, however, some of the increase in species richness may be caused by cows bringing in new species that can grow in more disturbed conditions. Many new species were observed in the vicinity of the cow trample path.

4.3.2.2 *Vegetation Dynamics*

For many years we have documented the effect of variable inundation (SEV) on the vegetation community, particularly in the upper estuary (Sagar et al. 2013; Sagar et al. 2015). In general, increased inundation reduces vegetation cover and reduced inundation increases cover. The effect of cows at Campbell Slough has confounded the results by also acting as a driver of reduced cover. The lasting effects of this variability is difficult to document. In some cases we observed what appears to be a lag effect, whereas the effects of high water or more favorable conditions seem to continue into the following year. This should be evaluated further to improve vegetation community response predictions. Another factor that may have an effect on the vegetation in the upper estuary is the timing of inundation. Jenkins et al. (2008) found that inundation >0.85 m in spring reduced cover of *P. arundinacea*, however, inundation of 0.6-0.85 m resulted in increases at Smith Bybee Wetlands. An evaluation of SEV for the period of March to June could be evaluated to determine if there is an associated detrimental effect on the species at the trend sites.

In order to determine if there are any longer term effects occurring at the sites we conducted a change analysis based on vegetation community mapping conducted at the site. We evaluated the earliest mapped data from each site against data collected in 2015. In the lower estuary, the changes are subtle. The most notable change at Ilwaco Slough was the increased area of pans and the increase in the area of the mixed community of *C. lyngbyei* and *Agrostis* spp. Pans are a natural feature in salt marshes and are likely caused by deposition of vegetation material (Pethick 1974; Boston 1983) or by wind wave erosion if near the edge of the marsh (Perillo et al. 2003). The increase in *Agrostis* at the site is somewhat concerning since it coincides with a reduction of *C. lyngbyei*, an important native species and contributor of organic material to the system. No obvious changes occurred at Welch Island. The patches of *P. arundinacea* that occur at the site do not appear to have changed significantly in aerial coverage. At Whites Island the patch of *C. lyngbyei* does appear to have gotten smaller, reduced at the lower end by sloughing and at the upper elevation by competition from *P. arundinacea*.

In the upper estuary the changes are more evident. At Campbell Slough all the boundaries have shifted slightly and an area at the south end of the site shifted from *P. arundinacea* to low marsh after the high water in 2011 and 2012, perhaps caused by erosion in those years. Cunningham Lake had shifts in the species composition of communities and the boundaries at which they occur. At this and other sites, *Sagittaria latifolia* appears to have two growth habits: 1) growing at the lowest elevation for emergent vegetation and 2) growing at mid and high marsh elevations mixed with other marsh species. Consequently, we saw changes in both growth types at Cunningham Lake. The low elevation population shifts from almost zero with high inundation to very high cover and biomass when inundation is low during the growing season. The high elevation population shifts in the amount of cover within the low elevation *Eleocharis palustris* zone and in the high marsh *P. arundinacea* zone resulting in changes in community distribution. Also, at Cunningham Lake, the cover of *E. palustris* was reduced in 2011 and 2012 and never fully recovered. The community change analysis at Franz Lake documents the shift from a *P. arundinacea* community to that dominated by *P. amphibium*, which also occurred during the high water period of 2011 and 2012.

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 net 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 samples averaged greater than 90 percent; a confirmation that we indeed were sampling at or near the biomass peak. In addition, we sample in the standing dead the following winter (February) to determine the amount of die-back. An interesting observation regarding the winter sampling was 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 was 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 2016, the average summer biomass from

Whites Island *P. arundinacea*/HM and Franz Lake/*P. amphibium* strata was upwards of 2000 g/m^2, however, the multi-year analysis of the summer biomass revealed high variability between years for these strata. 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 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 2016, 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. Moderate inundation levels and lower than average precipitation (and possibly more sun) in the spring of 2016 may have resulted in more favorable growing conditions at Whites Island. 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 high competition afforded by *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. The analysis this year focused on the development of two such models. First, we found there to be a moderate linear relationship between cover within a vegetation strata and the biomass produced. Previous studies that have developed predictive biomass models did so using more controlled methods such as evaluating individual plant biomass to cover relationships and measuring plant morphometrics such as diameter and height (Daoust and Childers 1998; Rottgermann et al. 2000; Flombaum and Sala 2007). Our regression relationship could likely be made stronger by including measurements of plant height in addition to cover. Second, within vegetation strata, there is a strong linear relationship between summer biomass production and the amount of organic matter that is contributed to the ecosystem. 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. Put together we can begin to estimate the amount of organic matter produced by lower Columbia River marshes based on aerial cover measured within specific vegetation strata.

4.4.1.2 *Pelagic*

As in previous years, the phytoplankton assemblages differed across the trends sites, with Ilwaco Slough exhibiting most difference from the other sites due to the stronger marine influence and higher tidal range. Similar to past years, Whites Island was most comparable to the mainstem in terms of broad taxonomic representation (Tausz 2014; Tausz et al. in prep.). Given that Whites Island is situated in Reach C, designated a transport reach (Simenstad et al. 2011), this similarity is not surprising. The phytoplankton assemblage at Whites Island was dominated by diatoms, and analysis of patterns in similarity among the sites and over the dates of sampling showed that, similar to other years, the diatom, *Asterionella formosa* increased 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 2016. In each of the years between 2009 and 2016, *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); it is uncertain whether shallow water habitats with longer residence time influence rates and prevalence of parasitism upon primary producers that fuel aquatic food webs. 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.

4.5 Macroinvertebrates

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 87) 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 2016, our sampling focused on revisiting five trends sites (Ilwaco Slough, Welch Island, Whites Island, Campbell Slough, and Franz Lake) to collect additional information on temporal changes in salmon occurrence and health in these areas. In addition, we investigated the impact of the tidal cycle on fish community composition and salmon occurrence at Ilwaco Slough, Whites Island, and Campbell Slough. We also obtained genetic stock information for the Chinook salmon we sampled in a 2015 pilot study of salmon occurrence in two tributaries of the Columbia River, the Grays River and the Lewis River.

In 2015, the Pacific Northwest experienced an unusual weather year with a prolonged summer drought and high water temperatures observed throughout the region during the sampling season. In association with these conditions, we observed some unusual patterns of juvenile salmon occurrence in 2015, including unusually low juvenile Chinook salmon densities and low proportions of fry at Welch Island and Whites Island, as well as especially low densities of juvenile salmon during the summer months (Hanson et al. 2016). The genetics data for 2015, which we presented in this report, showed a lower than usual proportion of interior Chinook salmon stocks (e.g., Snake River fall Chinook, Upper Columbia summer/fall Chinook, and Deschutes River fall Chinook) and earlier downstream migration for fish from these stocks as compared to previous years, suggesting that high water temperatures may have affected their migration timing.

In 2016, temperatures were more moderate, and patterns of salmon occurrence were similar to other sampling years. In 2016, as in previous years, Chinook salmon were the dominant salmonid species at all sites except Ilwaco Slough, where no Chinook salmon were collected but chum salmon were abundant. As usual, unmarked Chinook salmon dominated catches at Welch Island, Whites Island, and Franz Lake, while higher proportions of marked hatchery fish were observed at Campbell Slough. Seasonal patterns of occurrence for Chinook salmon were more normal in 2016 than in 2015, with fish present from February through September. Also, the high proportion of fry usually found early in the sampling season at Whites Island and Welch Island were present in 2016. Typical seasonal patterns of chum salmon occurrence were observed at Ilwaco Slough, Whites Island and Campbell Slough, with fish present in March and April. Coho salmon were captured in 2016, but only in low numbers, while in both 2015 and 2016, sockeye salmon and trout were entirely absent. Although the densities of Chinook salmon were generally higher in 2016 than in 2015, they were still not especially high in comparison with other sampling years.

Patterns of fish community composition remained relatively stable, with low species diversity and richness and few non-native or predatory fish species at Ilwaco Slough, Welch Island, and Whites Island. As in previous years, we observed higher species richness and diversity, as well as higher proportions of non-native species and predatory fish species at Campbell Slough and Franz Lake. The proportion of non-native species, however, was higher than usual at Ilwaco Slough, Welch Island, and Whites Island in 2016, with a larger than usual numbers of banded killifish caught at these sites. This tendency was also observed in 2015 (Hanson et al. 2016).

In addition to monitoring Chinook salmon occurrence at the trends sites, we also monitor several indicators of salmon health, including condition factor, lipid content, growth rates, and chemical contaminants. For some of these parameters we do not yet have new data to present for fish collected in 2016. However, we did measure condition factor in 2016 and lipid content of fish collected in 2015.

In 2016, condition factor for unmarked Chinook salmon showed a pattern similar to that generally seen in previous years, with the highest condition factor values in fish from Campbell Slough and lowest values at Franz Lake (See discussion in Section 4.5 regarding feeding intensity and energy consumption in juvenile Chinook salmon at Franz Lake). Within sites, there was some variation among years at Welch Island, Whites Island, Campbell Slough, but not at Franz Lake or Ilwaco Slough. At Campbell Slough and Welch Island, condition factor was relative high in 2016 compared to previous years, but relatively low at Whites Island. Among marked Chinook salmon, condition factor has not consistently shown differences among sites, and the same was true for 2016. At Welch Island, Whites Island, and Campbell Slough (marked fish were not captured at Ilwaco Slough or Franz Lake in 2016) condition factor tended to be relatively high in 2016 as compared to other years.

In 2015, we found no evidence of reduced condition in juvenile salmon, in spite of the high summer temperatures. Condition factor was within the normal range at all of the sampling sites. Similarly, we found relatively high lipid content in juvenile salmon collected in 2015, as well as a relatively high proportion of lipids present as triglycerides, suggesting that energy stores were comparable to, or higher than those in more typical years. These findings are consistent with those of Roegner and Teel (2014), who also found little evidence for reduced condition in juvenile Chinook salmon from tidal freshwater sites in the Lower Columbia River during periods of high water temperature. Other researchers have also shown that positive growth and condition can be maintained at high water temperatures if oxygen and food supply are sufficient (Brett et al. 1982; Clarke and Shelbourn 1986; Sommer et al. 2001; Marine and Cech 2004). The low Chinook salmon densities in 2015 at our trend sites might also contribute to the maintenance of growth and condition, as there would be less competition for food. Roegner and Teel (2014) also observed that the majority of the fish they sampled during periods of high temperature had originated from genetic stocks that historically had a summer juvenile migration period (West Cascade tributaries fall and upper Columbia River summer/fall stocks; Howell et al. 1985) and suggested that these two stocks were able to maintain positive condition because they are better adapted to high summer temperatures. Based on our 2015 genetic results, we did in fact see unusually high proportions of West Cascade fall Chinook in May and June in 2015 (73% of fish in May and 100% in June in 2015, as compared to 60% and 63% in other sampling years). However this was not true for Upper Columbia summer/fall Chinook; they were rarely observed in 2015, and the few fish that were collected were found in April. Other stocks usually captured later in the summer (e.g., Snake River fall Chinook) were also absent from tidal freshwater sites by June, suggesting that extreme temperatures triggered an earlier migration time from off-channel habitats to the ocean, even for the more tolerant stocks. The lack of any clear impact on condition factor and lipid content may be in part because all juveniles migrated to the estuary as temperatures became too extreme to maintain positive condition.

Somatic Growth

Although we do not present new growth rate results for Chinook salmon from 2016 in this report, we did conduct a comprehensive analysis of all growth data collected from all sites sampled from 2005 through 2013 (see Chittaro et al. submitted) to better understand how the growth performance of juvenile Chinook salmon varied while they resided in estuarine and tidal freshwater habitats of the Columbia River. Of the suite of variables analyzed we found that variability in relative growth rate was best explained by Julian day and year, and to a lesser degree, river kilometer and off-channel distance where fish were collected.

Across the eight years of our study we observed a slight decline in performance (of approximately 6%), with fish sampled in 2010 and 2011 showing the lowest growth rate. In 2010 and 2011, we observed relatively high median densities of Chinook salmon and non-salmonids along with summer water temperatures exceeding the 19°C optimum for Chinook salmon (Brett et al. 1982). These years of elevated temperatures together with the observed higher intra- and interspecific densities could interact to negatively influence growth rate. Crozier et al. (2010), for example, found that growth correlated negatively with temperature at elevated densities, yet positively at lower densities. Maximum monthly temperatures were also relatively high in 2009 (23.5°C) and 2013 (19.4°C) but our estimates of median growth rate for these years were greater than those for 2010 and 2011. Since 2009 and 2013 had some of the lowest median densities of Chinook salmon and non-salmonids, it is possible that the impacts to performance in 2009 and 2013 were relatively smaller than those in 2010 and 2011, despite the presence of warm water.

Our analysis of the spatial variability in growth performance indicated a decreasing trend in growth rate towards the mouth of the Columbia River of approximately 13%. In particular, we observed lower growth rates for fish collected in reaches A-C. A possible explanation to this spatial pattern of growth rate is the increased metabolic cost associated with maintaining homeostasis in brackish environments (Morgan and Iwama 1991; Enberg et al. 2012) that are characteristic of Reach A and B, but not necessarily C (see Simenstad et al. 2011). It is also possible that fish in the lower reaches expend relatively more energy due to tidally influenced changes to water velocity (Reach A-C; Simenstad et al. 2011), thus resulting in less energy being available for somatic growth. For example, during higher water velocities in the Sacramento River, juvenile Chinook salmon were reported to display a more directional migration, suggesting that this was an energy saving behavior (Steel et al. 2012). Other factors and processes that could limit performance in the estuarine and tidal freshwater habitats include food availability and quality, which is linked to intra- and inter-specific density and predator density via competition and predation, respectively. Limm and Marchetti (2009) and Sommer et al. (2001) reported significant differences in growth rate of Chinook salmon among various habitats (e.g., off-channel pond, main-channel, seasonal tributaries, and floodplain) of the Sacramento River, such that higher growth rates were related to areas with greater prey consumption and presumed prey availability (Sommer et al. 2001). Work by Kaneko et al. (2015) reported a decrease in growth rate of chum salmon (*Oncorhynchus keta*) that was linked to intraspecific competition for food in an estuary following an intensive release of hatchery fish. In our study, reaches exhibiting lower growth performance had lower median prey density and richness and higher median non-salmonid density (e.g., stickleback, killifish, bass etc.) than upriver reaches, suggesting an influence of food limitation and competition on growth rate. However, results from our statistical analyses did not identify invertebrates and fish as major contributors to variability in growth rate. This is not surprising that our measure of growth rate integrates over seven days in the life of a fish, while our fish and invertebrate density and richness data are instantaneous measurements of the community, and our contaminant data represent lifetime loads. Further, growth rate is the cumulative product of interactions among many ecosystem components such as prey, predators, and habitat quality, quantity, and connectivity Searcy et al. (2007). Such interactions potentially limit the usefulness of somatic growth rate as an indicator of habitat quality. For example, Chittaro et al. (2014) reported an overlap in somatic growth rate among sites that implied comparable habitat quality. But their investigation revealed that consumption rates differed among sites, due to the influence of temperature on metabolism. In this case, similarity in growth rates occurred despite fish residing in habitats that differed in quality.

In addition to temporal and spatial factors, fish origin also appeared to influence growth rate. Growth rates of hatchery and unmarked fish showed considerable overlap similar to other measures of condition such as lipid content and condition factor. However, these rates differed among some stocks of origin. In fact, our study showed that growth rates of hatchery fish varied more among stocks than growth rates of unmarked fish, suggesting that hatchery-rearing conditions may not be not uniform among stocks. Differences in juvenile growth rate of wild Chinook salmon have previously been reported between Evolutionary Significant Units (ESU; i.e., Snake River spring/summer and Snake River fall) and were attributed to disparities in spawning season, geographic range, genetics, and life history variability (Zabel et al. 2010). However, differences in growth rate were not observed among more closely related stocks within an ESU (Zabel et al. 2010; Chittaro et al. 2014). Given that growth rates in our study overlapped regardless of whether stocks were from the same ESU or not might suggest the importance of environmental drivers such as location (i.e., river kilometer) to growth rate. Interestingly, in a similar study Goertler et al. (2016) indicated that genetic differences were not important in explaining variability in growth rate of Chinook salmon. Clearly, additional work is needed to evaluate the influence, if any, of stock on growth rate. One way to approach this question would be to conduct a common garden experiment whereby fish from different stocks are raised in common environments.

Finally, while the juvenile Chinook salmon we sampled from estuarine and tidal freshwater habitats grew at rates (0.31-0.87 mm/day) comparable to values reported in studies of similar habitats (0.2-0.7 mm/day, Healey 1991; Volk et al. 2010; Goertler et al. 2016), a comparison of the growth rates we observed in Lower Columbia River juvenile Chinook salmon with growth rates reported in other studies suggests that growth rate potential may be greater than what we observed. For example, Snake River spring/summer and fall Chinook collected upriver, by more than 450 river kilometers from our study area had growth rates substantially higher than those in our study (Connor et al. 2001; Achord et al. 2007; Zabel et al. 2010). Other studies report much higher growth rates once fish enter the ocean (MacFarlane 2010; Healey 1980; Claiborne et al. 2014). These higher growth rates reported in juvenile Chinook salmon upriver and soon after they enter the ocean suggest that it is physiologically possible for juvenile Chinook salmon to exhibit higher performance in these estuarine and tidal freshwater habitats.

Tributary Sites

In addition to the usual annual sampling efforts at our trend sites, in 2015 we sampled juvenile salmon in two tributaries of the Columbia River (lower Grays River and lower Lewis River) to collect some preliminary information on how these areas might function as habitat for threatened and endangered juvenile Columbia River salmon. We found that both of these areas were being utilized by juvenile salmon, with especially high densities of juvenile Chinook salmon at the Lewis River sites. These fish were predominantly fry and of smaller size than those found at the trend sites in the Columbia River. The genetics data on these fish, which we present in this report, confirmed our hypothesis that they were most likely locally produced and are probably part of the Lower Columbia River ESU. In the Lewis River, we found that all of the fish collected were West Cascades fall Chinook. This stock also predominated in the Grays River, though small numbers of fish from other stocks (Columbia River Rogue and Spring Creek Group Fall Chinook) were also present. Interior Columbia stocks were absent from catches at both sites. The genetic stock composition of juvenile Chinook salmon from the Grays River was similar to the stock composition at nearby sites on the Columbia, such as Welch Island. However, the stock composition was quite different in fish collected in the Lewis River than at Campbell Slough, which is the closest of the EMP trend sites.

Our findings suggest that these tributaries are used primarily by locally produced fish from the Lower Columbia ESU. These observations are consistent with those of Teel et al. (2014), who found that the tributary sites they sampled in the Lower Columbia River tended to be dominated by a single stock reflecting local production in that tributary. For example, they also found that juveniles at the tributary/Columbia River confluence sites in the Lewis River, as well as Cowlitz and Washougal Rivers and Germany Creek, were largely West Cascade fall-run fish. However, they also observed a small proportion of non-natal juveniles moved into their Lewis River Confluence site, so some utilization of this and similar sites by upriver stocks cannot be ruled out. Because of the unusual weather conditions in 2015, the generally lower proportion of interior stocks at all of our sites, and the limited sampling we conducted in our pilot study, our likelihood of observing these fish was low. Still, both our data and other data suggest that these habitats are used primarily by local stocks.

Tidal Cycle Study

In 2016, three of our trend sites (Ilwaco Slough, Whites Island, and Campbell Slough), differing in tidal influence and salinity intrusion, were sampled during May and June to assess the influence of the tidal cycle on fish species composition and density. The species composition and density at all three sites changed with tide and water depth. At Campbell Slough, the site with the least tidal influence (~1 m), salmonid species were found at the highest densities between 0-2 hours after low tide. Non-salmonid species density was inversely related to salmonid density. The tidal influence at Whites Island is approximately ~2 m, but the site is not affected by saltwater intrusion. At this site, salmonid species were caught at all phases of the tidal cycle sampled, but the highest density of salmon was observed 3-4 hours after low tide when fewer other species were present. We also found that at Whites Island, smaller unmarked Chinook salmon predominated earlier in the tidal cycle, whereas larger unmarked and marked salmon were more common later in the tidal cycle. This trend was not observed at Campbell Slough. The tidal influence at Ilwaco Slough is also ~2 m, but the site is highly influenced by saltwater intrusion. At Ilwaco Slough, changes in fish composition and density were also observed with the tidal cycle, but no Chinook salmon was observed at the site, although one Chinook salmon was caught outside the slough in a test beach seine set. This observation suggests that although Chinook salmon may be present in the area, they may not often utilize the slough for feeding and rearing.

While the data collected in our pilot study are limited, they do suggest changes in both fish community composition and density of salmon and other fishes with the tidal cycle. While the species present at the sampled sites were fairly consistent, their proportions in the catch and density varied substantially, including Chinook salmon density. At Whites Island Chinook salmon density varied from 21.9 fish per 1000 m^2 to 110 fish per 1000 m^2, while at Campbell Slough it varied from no salmon to 20.5 fish per 1000 m^2. Thus the timing of beach seine sets could influence the results of our routine EMP fish sampling. In light of these findings we are planning to incorporate additional sampling over the tidal cycle into our EMP trend site sampling in the coming year.

5 Adaptive Management & Lessons Learned

Mitigation actions, such as habitat restoration, are employed to aid salmonid population recovery (Booth et al. 2016; van Zyll de Jong and Cowx 2016), and thus knowing where juvenile salmon performance is poor could be useful for planning and prioritizing such actions. For example, our analysis of the spatial variability in growth performance may be useful for informing habitat restoration strategies as to which habitat conditions would promote salmon growth. Furthermore, our study showed that growth rates of hatchery fish varied significantly among stocks, even more than for unmarked fish. This interaction between stock of origin and hatchery with respect to growth rate suggested that hatchery-rearing conditions are not uniform among stocks, which could inform modifications to hatchery operation.

A comparison of the juvenile Chinook salmon growth rates we observed in lower Columbia River with growth rates reported in other studies upriver and soon after ocean entry suggests that growth rate potential may be much greater than what we observed in our study. For example, Snake River spring/summer and fall Chinook collected upriver (>450 rkm from our study area), had growth rates substantially higher than those in our study (Connor et al. 2001; Achord et al. 2007; Zabel et al. 2010). Other studies reported much higher growth rates once fish enter the ocean (MacFarlane 2010; Healey 1980; Claiborne et al. 2014). These higher growth rates reported in juvenile Chinook salmon upriver and at ocean entry suggest that it is physiologically possible for juvenile Chinook salmon to exhibit higher growth than what we observed in estuarine and tidal freshwater habitats.

Fish genetics data suggest that higher than normal water temperatures which occurred in 2015 affected the migration timing of juvenile salmon, particularly interior stocks that typically migrate later in the summer. This points to the importance of cold water refuge habitat, shaded refuge habitat, and flow management techniques that could ameliorate high water temperatures during the summer months and help promote salmon survival, especially for interior salmon stocks.

Collecting fish across the tidal cycle showed how the fish community changed with the incoming tide. This information could be applied to improving sampling methods and deciphering results based on the timing of collection within the tidal cycle. For example, synching collection timing with a specific tidal level could help target a particular species or stock of interest. In 2016, we ceased sampling once water levels inundated the high marsh to ensure sampling efficiency was preserved (for the type of gear used), thus, future sampling at high (slack) tide would add to our current understanding of the fish community structure under a range of tidal conditions.

Metabolic cost and energy assimilation calculated for juvenile salmon could describe habitat changes over multiple time and spatial scales, from a single out-migration season to years or decades. In addition, comparison of growing conditions among different sites to can help us understand where salmon experience relatively good or poor growing conditions. For example, sampling salmon at a recently restored site could be plotted against the long term data gleaned from the trend sites to provide an evaluation of the new habitat relative to other areas in the estuary.

Our long-term monitoring results cover a wide range of hydrologic conditions and a rich dataset on vegetation cover, organic matter production, and salmon prey which can be used to develop a predictive model of the effects of hydrosystem operation (e.g., related to the Columbia River Treaty) and climate change effects on the food web. Our research on the effect of reed canarygrass on the food web underlies the importance of determining methods to control the species and to try and reduce the likelihood of invasion in restoration projects. In addition, floristic quality assessment methods provide a means of quantifying wetland condition in reference marshes and can be used to evaluate the condition of restoration sites. Five years of plant biomass production and breakdown data indicate that the abundance of reed canarygrass in the lower Columbia River likely changed the dynamics of the food web in two thirds of the estuary where it dominates.

Knowledge of the dynamics and vulnerability of floodplain and shallow water habitat that form adverse conditions for juvenile salmon (e.g., low dissolved oxygen, high temperatures), as well as information about food availability and energy pathways through the food web can help to improve fish survival. Differences in the physical, biogeochemical, and ecological characteristics of habitats used by juvenile salmon during years characterized by different hydrologic regimes may offer insight into how environmental factors play into the survival success of juvenile salmon.

Our observations indicate that some sites with slow flushing rates during low flow periods (e.g., Campbell Slough, Franz Lake Slough) are prone to harmful algae blooms (e.g., toxin-producing cyanobacteria). High concentrations of chlorophyll indicate that these habitats may approach critical levels that exceed water quality criteria in the lower Columbia River and continued monitoring provides contextual information to pinpoint critical time periods that may be targeted for management.

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 were also shared with regional partners at the Columbia River Estuary Conference in May 2016. 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. 2017).

6 References

Achord, S., R.W. Zabel, and B.P. Sandford. 2007. Migration timing, growth, and estimated parr-to-smolt survival rates of wild Snake River spring-summer Chinook salmon from the Salmon River basin, Idaho, to the lower Snake River. Transactions of the American Fisheries Society. 136:142-154.

Akaike, H. 1973. Information theory as an extension of the maximum likelihood principle. Pages 267-281 in B. N. Petrov and F. Csaki, editors. Second international symposium on information theory. Akademiai Kiado, Budapest, Hungary

Araya, Y.N., J. Silvertown, D.J. Gowing, K.J. McConway, H.P. Linder and G. Midgley. 2010. A fundamental, eco-hydrological basis for niche segregation in plant communities. New Phytologist 189(1):1-6.

Bernard, J.M., D. Solander, and J. Květ. 1988. Production and nutrient dynamics in Carex wetlands. Aquatic Botany 30(1-2):125-147.

Biro P.A., A.E. Morton, J.R. Post, and E.A. Parkinson. 2004. Over-winter lipid depletion and mortality of age-0 rainbow trout (*Oncorhynchus mykiss*). Canadian Journal of Fisheries and Aquatic Science 61:1513-1519.

Booth, D., J. Scholz, T. Beechie, S. Ralph. 2016. Integrating limiting-factors analysis with process-based restoration to improve recovery of endangered salmonids in the Pacific Northwest, USA. Water, 8(5):174.

Borde, A.B., S.A. Zimmerman, R.M. Kaufmann, H.L. Diefenderfer, N.K. Sather, and R.M. Thom. 2011. Lower Columbia River and Estuary Restoration Reference Site Study: 2010 Final Report and Site Summaries. PNWD-4262, prepared for the Lower Columbia River Estuary Partnership by the Battelle Marine Sciences Laboratory, Sequim, Washington.

Borde A.B., V.I. Cullinan, H.L. Diefenderfer, R.M. Thom, R.M. Kaufmann, J. Sagar, and C. Corbett. 2012. Lower Columbia River and Estuary Ecosystem Restoration Program Reference Site Study: 2011 Restoration Analysis. PNNL-21433, prepared for the Lower Columbia River Estuary Partnership by Pacific Northwest National Laboratory, Marine Sciences Laboratory, Sequim, Washington.

Boston, K. 1983. The development of salt pans on tidal marshes, with particular reference to South-Eastern Australia. Journal of Biogeography 10(1):1-10.

Bottom D.L., C.A. Simenstad, J. Burke, A.M. Baptista, D.A. Jay, K.K. Jones, E. Casillas, and M. Schiewe. 2005. Salmon at River's End: The Role of the Estuary in the Decline and Recovery of Columbia River Salmon. U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Technical Memorandum NMFS-NWFSC-68, Northwest Fisheries Science Center, Seattle, Washington.

Bottom, D.L., A. Baptista, J. Burke, L. Campbell, E. Casillas, B. Craig, C. Eaton, S. Hinton, K. Jacobson, D. Jay, M.A. Lott, R. McNatt, G.C. Roegner, S. Schrode, C.A. Simenstad, S. Spilseth, V. Stamatiou, D. Teel, and J.E. Zamon. 2011. Salmon life histories, habitat, and food webs in the Columbia River estuary: final report 2002–2008. Report by the National Oceanic and Atmospheric Administration

Fisheries, Fish Ecology Division to the U.S. Army Corps of Engineers, Portland District, Contract W66QKZ20374382, Portland, Oregon. Available: http://nwfsc.noaa.gov/publications/scientificpubs.cfm.

Brett, R.J., W.C. Clarke, and J.E. Shelbourn. 1982. Experiments on thermal requirements for growth and food conversion efficiency of juvenile Chinook salmon *Oncorhynchus tshawytsha*. Canadian Technical Report of Fisheries and Aquatic Sciences. No. 1127.

Brinson, M.M., A.E. Lugo, and S. Brown. 1981. Primary productivity, decomposition and consumer activity in freshwater wetlands. Annual Review of Ecology and Systematics 12(1):123-161.

Burke, B.J., W.T. Peterson, B.R. Beckman, C.A. Morgan, E.A. Daly, M. Litz. 2013. Multivariate models of adult Pacific salmon returns. PLoS ONE 8(1):e54134.

Burnham, K.P. and D.R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. Springer-Verlag, New York.

Caffrey, J.M. 2004. Factors controlling net ecosystem metabolism in U.S. estuaries. Estuaries 27:90-101.

Cazanacli, D. and N.D. Smith. 1998. A study of morphology and texture of natural levees—Cumberland Marshes, Saskatchewan, Canada. Geomorphology 25(1): 43-55.

Chawla, A., D.A., Jay, A.M. Baptista, M. Wilkin, and C. Seaton. 2008. Seasonal Variability and Estuary-Shelf Interactions in Circulation dynamics of a river-dominated estuary. Estuaries & Coasts 31(2): 269-288.

Chittaro, P.M., R.W. Zabel, B. Beckman, D.A. Larsen, and A. Tillotson. 2015. Validation of daily increment formation in otoliths from spring Chinook salmon. Northwest Science 89:93-98.

Chmura, G.L., S.C. Anisfeld, D.R. Cahoon and J.C. Lynch. 2003. Global carbon sequestration in tidal, saline wetland soils. Global Biogeochemical Cycles 17(4):1111.

Chust, G., J.I. Allen, L. Bopp, C. Schrum, J. Holt, K. Tsarias, K. et al. 2014. Biomass changes and trophic amplification of plankton in a warmer ocean. Global Change Biology 20: 2124-2139.

Cheung, W.W.L., R.D. Brodeur, T.A. Okey, D. Pauly. 2015. Projecting future changes in distributions of pelagic fish species of Northeast Pacific shelf seas. Progress in Oceanography 130:19-31.

Claiborne, A.M., J.A. Miller, L.A. Weitkamp, D.J. Teel, and R.L. Emmett. 2014. Evidence for selective mortality in marine environments: the role of fish migration size, timing, and production type. Marine Ecology Progress Series 515:187-202.

Clarke, K.R. and R.N. Gorley. 2006. PRIMER v6: User Manual/Tutorial. PRIMER-E Plymouth.

Clarke, W.C. and J.E. Shelbourn. 1985. Growth and development of seawater adaptability by juvenile fall Chinook salmon (*Oncorhynchus tshawytscha*) in relation to temperature. Aquaculture 45:21–31.

Clarke, K.R. and R.M. Warwick. 1994. Change in marine communities: An approach to statistical analysis and interpretation. Natural Environment Research Council, London.

Connor, W.P., A.R. Marshall, T.C. Bjornn, and H.L. Burge. 2001. Growth and long-range dispersal by wild subyearling spring and summer Chinook salmon in the Snake River basin. Transactions of the American Fisheries Society 130:1070-1076.

Crozier, L.G., R.W. Zabel, E.E. Hockersmith, and S. Achord. 2010. Interacting effects of density and temperature on body size in multiple populations of Chinook salmon. Journal of Animal Ecology 79:342-349.

Daly, E.A., Auth, T.D., Brodeur, R.D., Peterson, W.T. 2013. Winter ichtyoplankton biomass as a predictor of early summer prey fields and survival of juvenile salmon in the northern California Current. Marine Ecology Progress Series 484: 203-217.

Daly, E.A. and R.D. Brodeur. 2015. Warming ocean conditions relate to increased trophic requirements of threatened and endangered salmon. PLoS ONE 10(12): e0144066. doi: 10.1371/journal.pone.0144066.

Daoust, R.J. and D.L. Childers. 1998. Quantifying aboveground biomass and estimating net aboveground primary production for wetland macrophytes using a non-destructive phenometric technique. Aquatic Botany 62(2):115-133.

David, A.T., C.A. Simenstad, J.R. Cordell, J.D. Toft, C.S. Ellings, A. Gray, H.B. Berge. 2016. Wetland loss, juvenile salmon foraging performance, and density dependence in Pacific Northwest estuaries. Estuaries and Coasts 39:767-780.

Davidson-Arnott, R.G., D. van Proosdij, J. Ollerhead, and L. Schostak. 2002. Hydrodynamics and sedimentation in salt marshes: examples from a macrotidal marsh, Bay of Fundy. Geomorphology 48(1):209-231.

Dennison, W.C., R.J. Orth, K.A. Moore, J.C. Stevenson, V. Carter, S. Kollar, P.W. Bergstrom, and R.A. Batiuk. 1993. Assessing water quality with submersed aquatic vegetation. BioScience 43(2):86-94.

Diefenderfer, H.L., A.M. Coleman, A.B. Borde, and I.A. Sinks. 2008. Hydraulic geometry and microtopography of tidal freshwater forested wetlands and implications for restoration, Columbia River, USA. Ecohydrology and Hydrobiology 8(2):339-361.

Diefenderfer, H.L. and D.R. Montgomery. 2009. Pool spacing, channel morphology, and the restoration of tidal forested wetlands of the Columbia River. U.S.A. Restoration Ecology 17:158–168.

Doney, S.C., M.H. Ruckelshaus, J.E. Duffy, J.P. Barry, F. Chan, C.A. English, H.M. Galindo, J.M. Grebmeier, A.B. Hollowed, N. Knowlton, J. Polovina, N.N. Rabalais, W.J. Sydeman, and L.D. Talley. 2012. Climate change impacts on marine ecosystems. Annual Reviews in Marine Science 4:11-37.

Elskus, A., T.K. Collier and E. Monosson. 2005. Interactions between lipids and persistent organic pollutants in fish. In: Environmental Toxicology, T.P. Mommsen and T.W. Moon (Eds), Elsevier, San Diego. pp. 119-152.

Enberg, K., C. Jørgensen, E.S. Dunlop, Ø. Varpe, D.S. Boukal, L. Baulier, S. Eliassen, and M. Heino. 2012. Fishing-induced evolution of growth: concepts, mechanisms and the empirical evidence. Marine Ecology 33:1-25.

Fiechter, J., D.D. Huff, B.T. Martin, D.W. Jackson, C.A. Edwards, K.A. Rose, E.N. Curchitser, K.S. Hedstrom, S.T. Lindley, and B.K. Wells. 2015. Environmental conditions impacting juvenile Chinook salmon growth off central California: An ecosystem model analysis. Geophysical Research Letters 42:2910-2917.

Flombaum, P. and O.E. Sala. 2007. A non-destructive and rapid method to estimate biomass and aboveground net primary production in arid environments. Journal of Arid Environments 69(2):352-358.

France, R.L. 1995. Stable isotopic survey of the role of macrophytes in the carbon flow of aquatic food webs. Vegetatio [Belgium] 124:67-72.

Fresh, K.L., E. Casillas, L.L. Johnson, and D.L. Bottom. 2005. Role of the Estuary in the Recovery of Columbia River Basin Salmon and Steelhead: An Evaluation of the Effects of Selected Factors on Salmonid Population Viability. NOAA Technical Memorandum NMFS-NWFSC-69. U.S. Department of Commerce.

Fresh, K.L., D.J. Small, H. Kim, C. Waldbillig, M. Mizell, M.I. Carr, and L. Stamatiou. 2006. Juvenile Salmon Use of Sinclair Inlet, Washington, in 2001 and 2002. Report FRT-05-06. Olympia: Washington State Department of Fish and Wildlife.

Freyman, W.A., L.A. Masters, and S. Packard. 2015. The Universal Floristic Quality Assessment (FQA) Calculator: an online tool for ecological assessment and monitoring. Methods in Ecology and Evolution 7(3): 4 pp.

Fulton, T. 1902. Rate of growth of seas fishes. Sci. Invest. Fish. Div. Scot. Rept. 20.

Goertler, P.A.L., C.A. Simenstad, D.L. Bottom, S. Hinton, and L. Stamatiou. 2016. Estuarine Habitat and Demographic Factors Affect Juvenile Chinook (*Oncorhynchus tshawytscha*) Growth Variability in a Large Freshwater Tidal Estuary. Estuaries and Coasts 39:542-559.

Gowing, D.J.G., C.S. Lawson, E.G. Youngs, K.R. Barber, and J.S. Rodwell. 2002. The water regime requirements and the response to hydrological change of grassland plant communities, Rep. BD1310, 98 pp., Cranfield Univ., Bedford, U.K.

Hamlet, A.F., M.M. Elsner, G.S. Mauger, S.Y. Lee, I. Tohver, and R.A. Norheim. 2013. An overview of the Columbia Basin Climate Change Scenarios Project: Approach, methods, and summary of key results. Atmosphere-Ocean 51(4):392-415.

Hanson, A.C., A.B. Borde, L.L. Johnson, T.D. Peterson, J.A. Needoba, S.A. Zimmerman, M. Schwartz, C.L. Wright, P.M. Chittaro, S.Y. Sol, D.J. Teel, G.M. Ylitalo, D. Lomax, and C.E. Tausz, H.L. Diefenderfer, C.A. Corbett. 2015. Lower Columbia River Ecosystem Monitoring Program Annual Report for Year 10 (October 1, 2013 to September 30, 2014). Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration. Available from the Lower Columbia Estuary Partnership, Portland, OR.

Hanson, A.C., A.B. Borde, L.L. Johnson, T.D. Peterson, J.A. Needoba, J. Cordell, M. Ramirez, S.A. Zimmerman, P.M. Chittaro, S.Y. Sol, D.J. Teel, P. Moran, G.M. Ylitalo, D. Lomax, and C.E. Tausz, M. Schwartz, H.L. Diefenderfer, C.A. Corbett. 2016. Lower Columbia River Ecosystem Monitoring Program Annual Report for Year 11 (October 1, 2014 to September 30, 2015). Prepared by the Lower

Columbia Estuary Partnership for the Bonneville Power Administration. Available from the Lower Columbia Estuary Partnership, Portland, OR.

Havens, K.E. 2003. Submerged aquatic vegetation correlations with depth and light attenuating materials in a shallow subtropical lake. Hydrobiologia 493(1):173-186.

Healey, M. 1991. Life history of Chinook salmon (Oncorhynchus tshawytscha). In Pacific salmon life histories. C. Groot and L. Margolis (Eds.). UBC Press, Vancouver. 313-393.

Healey, M.C. 1980. Utilization of the Nanaimo River Estuary by juvenile Chinook salmon, *Oncorhynchus tshawytscha* Fishery Bulletin 77:653-668.

Herfort L., T.D. Peterson, V. Campbell, S. Futrell, P. Zuber. 2011a. *Myrionecta rubra* (*Mesodinium rubrum*) bloom initiation in the Columbia River estuary. Estuarine, Coastal and Shelf Science 95:440–446.

Herfort L., T.D. Peterson, L.A. McCue, B.C. Crump, F.G. Prahl, A.M Baptista, V.C. Campbell, R. Warnick, M. Selby, G.C. Roegner, P. Zuber. 2011b. *Myrionecta rubra* population genetic diversity and its cryptophyte chloroplast specificity in recurrent red tides in the Columbia River estuary. Aquatic Microbial Ecology 62:85–97.

Howell, P., K. Jones, D. Scarnecchia, L. LaVoy, W. Knedra, and D. Orrmann. 1985. Stock assessment of Columbia River anadromous salmonids, volume I. Bonneville Power Administration, Project 83-335, Portland, Oregon.

Ibelings, B.W., A.S. Gsell, W.M. Mooij, E. van Donk, S. Van Den Wyngaert, L.N. De Senerpont Domis. 2011. Chytrid infections and diatom spring blooms: paradoxical effects of climate warming on fungal epidemics in lakes. Freshwater Biology 56(4):754-766.

Jay, D.A., K. Leffler, H.L. Diefenderfer, and A.B. Borde. 2015. Tidal-fluvial and estuarine processes in the lower Columbia River: I. along-channel water level variations, Pacific Ocean to Bonneville Dam. Estuaries and Coasts 38(2):415-433.

Jay, D.A., A.B. Borde, and H.L. Diefenderfer. 2016. Tidal-Fluvial and Estuarine Processes in the Lower Columbia River: II. Water Level Models, Floodplain Wetland Inundation, and System Zones. Estuaries and Coasts 39(5):1299-1324.

Jenkins, N.J., J.A. Yeakley, and E.M. Stewart. 2008. First-year responses to managed flooding of lower Columbia River bottomland vegetation dominated by *Phalaris arundinacea*. Wetlands 28(4):1018-1027.Kalinowski, S.T., K.R. Manlove, and M.L. Taper. 2007. ONCOR a computer program for genetic stock identification. Montana State University, Department of Ecology, Bozeman. Available: montana.edu/kalinowski/Software/ONCOR.htm.

Kaneko, N., N. Taniyama, Y. Inatani, Y. Nagano, M. Fujiwara, M. Torao, Y. Miyakoshi, and M. Shimizu. 2015. Circulating insulin-like growth factor I in juvenile chum salmon: relationship with growth rate and changes during downstream and coastal migration in northeastern Hokkaido, Japan. Fish Physiology and Biochemistry 41:991-1003.

Kentula, M.E., R.P. Brooks, S.E. Gwin, C.C. Holland, A.D. Sherman, and J.C. Sifneos. 1992. An approach to improving decision making in wetland restoration and creation. U.S. Environmental Protection Agency, Corvallis, Oregon.

Kozloff, E.N. 1996. Marine Invertebrates of the Pacific Northwest. Seattle: University of Washington Press. 511 pp.

Lassiter, R.R. and T.G. Hallam. 1990. Survival of the fattest: implications for acute effects of lipophilic chemicals on aquatic populations. Environmental Toxicology and Chemistry 9:585–595.

Liao, H., C.L. Pierce, and J.G. Larscheid. 2001. Empirical Assessment of Indices of Prey Importance in the Diets of Predacious Fish. Transactions of the American Fisheries Society 130:583-591.

Limm, M. and M. Marchetti. 2009. Juvenile Chinook salmon (*Oncorhynchus tshawytscha*) growth in off-channel and main-channel habitats on the Sacramento River, CA using otolith increment widths. Environmental Biology of Fishes 85:141-151.

Lindholm, T. 1985. Mesodinium rubrum – a unique photosynthetic ciliate. Advances in Aquatic Microbiology 3:1-48.

Lott, M.A. 2004. Habitat-specific feeding ecology of ocean-type Chinook salmon in the lower Columbia River estuary. M.Sc. Thesis. University of Washington, Seattle. 124 pp.

MacFarlane, R.B. 2010. Energy dynamics and growth of Chinook salmon (*Oncorhynchus tshawytscha*) from the Central Valley of California during the estuarine phase and first ocean year. Canadian Journal of Fisheries and Aquatic Sciences 67:1549-1565.

Maier, M.A. 2014. Ecology of diatoms and their fungal parasites in the Columbia River. Ph.D. Dissertation, Oregon Health & Science University.

Maier, M.A. and T.D. Peterson. 2017. Prevalence of chytrid parasitism among diatom opulations in the lower Columbia River (2009-2013). Freshwater Biology 62:414-428.

Maier, G.O. and C.A. Simenstad. 2009. The role of marsh-derived macrodetritus to the food webs of juvenile Chinook salmon in a large altered estuary. Estuaries and Coasts 32:984-998.

Manel, S., O.E. Gaggiotti, and R.S. Waples. 2005. Assignment methods: matching biological questions with appropriate techniques. Trends in Ecology and Evolution 20:136–142.

Marine, K.R. and J.J. Cech. 2004. Effects of high water temperature on growth, smoltification, and predator avoidance in juvenile Sacramento River Chinook salmon. North American Journal of Fisheries Management 24:198–210.

Marani, M., C. Da Lio, and A. D'Alpaos. 2013. Vegetation engineers marsh morphology through multiple competing stable states. Proceedings of the National Academy of Sciences 110(9):3259-3263.

Mason, W.R.M. 1993. Chapter 5: Key to superfamilies of Hymenoptera. In: H. Goulet and J.T. Huber (eds.). Hymenoptera of the world: an identification guide to families. Centre for Land and Biological Resources Research, publication 1894/E:65-101, Ottawa, Ontario.

Mcleod, E., G.L. Chmura, S. Bouillon, R. Salm, M. Björk, C.M. Duarte, C.E. Lovelock, W.H. Schlesinger, and B.R. Silliman. 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO2. Frontiers in Ecology and the Environment 9(10):552-560.

Merritt, R.W. and K.W. Cummins (eds). 1996. An Introduction to the Aquatic Insects of North America, 3rd ed. Dubuque (IA): Kendall/Hunt Publishing Company. 862 pp.

Morgan, J.D., and G.K. Iwama. 1991. Effects of Salinity on Growth, Metabolism, and Ion Regulation in Juvenile Rainbow and Steelhead Trout (*Oncorhynchus mykiss*) and Fall Chinook Salmon (*Oncorhynchus tshawytscha*). Canadian Journal of Fisheries and Aquatic Sciences 48:2083-2094.

Myers, J.M., C. Busack, D. Rawding, A.R. Marshall, D.J. Teel, D.M. Van Doornik, M.T. Maher. 2006. Historical population structure of Pacific salmonids in the Willamette River and lower Columbia River basins. U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-NWFSC-73, 311 p.

Nava, D., Restello, R.M., and Hepp, L.U. Intra- and inter-annual variations Chironomidae (Insecta: Diptera) communities in subtropical streams. 2015. Zoologia (Curitiba) 32(3): 207-214.

Neill, C. and J.C. Cornwell. 1992. Stable carbon, nitrogen, and sulfur isotopes in a prairie marsh food web. Wetlands 12(3):217-224.

Netboy, A. 1980. The Columbia River Salmon and Steelhead Trout, Their Fight for Survival. Seattle: University of Washington Press.

Parker, J.D., Caudill, C.C. and Hay, M.E. 2007. Beaver herbivory on aquatic plants. Oecologia 151(4):616-625.

Perillo, G.M.E. and O.O. Iribarne. 2003. Processes of tidal channel development in salt and freshwater marshes. Earth Surface Processes and Landforms 28:1473–1482.

Pethick, J. 1974. The distribution of salt pans on tidal salt marshes. Journal of Biogeography 1(1):57-62.

Rannala B. and J.L. Mountain. 1997. Detecting immigration by using multilocus genotypes. Proceedings of the National Academy of Sciences 94:9197-9201.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bulletin of the Fisheries Research Board of Canada 191:1-382.

Rocchio, F.J. and R.C. Crawford. 2013. Floristic Quality Assessment for Washington Vegetation. Natural Heritage Report 2013-03. Washington State Department of Natural Resources, Olympia, Washington.

Roegner, G.C., A.M. Baptista, D.L. Bottom, J. Burke, L.A. Campbell, C. Elliot, S.A. Hinton, D.A. Jay, M. Lott, T.A. Lundrigan, R.A. McNatt, P. Moran, C.A. Simenstad, D.J. Teel, E. Volk, J.E. Zamon, and E. Casillas. 2008. Estuarine Habitat and Juvenile Salmon Current and Historical Linkages in the Lower Columbia River and Estuary, 2002-2004. Report by National Marine Fisheries Service to the U.S. Army Corps of Engineers Portland District, Seattle, Washington, 139 p.

Roegner G.C., D.L. Bottom, A. Baptista, S. Hinton, C.A. Simenstad, E. Casillas, and K. Jones. 2004. Estuarine Habitat and Juvenile Salmon – Current and Historical Linkages in the Lower Columbia River and Estuary, 2003. Northwest Fisheries Science Center, National Marine Fisheries Service, Seattle, Washington.

Roegner, G.C., E.W. Dawley, M. Russell, A. Whiting, D.J. Teel. 2010. Juvenile salmonid use of reconnected tidal freshwater wetlands in Grays River, lower Columbia River basin. Transactions of the American Fisheries Society. 139:1211-1232.

Roegner, G.C., H.L. Diefenderfer, A.B. Borde, R.M. Thom, E.M. Dawley, A.H. Whiting, S.A. Zimmerman, and G.E. Johnson. 2009. Protocols for monitoring habitat restoration projects in the lower Columbia River and estuary. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-97, 63 p.

Roegner, G.C. and D.J. Teel. 2014. Density and condition of subyearling Chinook salmon in the Lower Columbia River and Estuary in relation to water temperature and genetic stock of origin. Transactions of the American Fisheries Society 143(5):1161-1176.

Röttgermann, M., T. Steinlein, W. Beyschlag, and H. Dietz. 2000. Linear relationships between aboveground biomass and plant cover in low open herbaceous vegetation. Journal of Vegetation Science 11:145–148.

Sagar, J.P., A.C. Hanson, A. B. Borde, L.L. Johnson, T. Peterson, K.H. Macneale, J.A. Needoba, S.A. Zimmerman, M.J. Greiner, C.L. Wright, P.M. Chittaro, O.P. Olson, S.Y. Sol, D.J. Teel, G.M. Ylitalo, D. Lomax, A. Silva and C.E. Tausz. 2014. Lower Columbia River Ecosystem Monitoring Program Annual Report for Year 9 (October 1, 2012 to September 30, 2013). Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration. Available from the Lower Columbia Estuary Partnership, Portland, OR.

Sagar, J.P., A.B. Borde, L.L. Johnson, C.A. Corbett, J.L. Morace, K.H. Macneale, W.B. Temple, J. Mason, R.M Kaufmann, V.I. Cullinan, S.A. Zimmerman, R.M. Thom, C.L. Wright, P.M. Chittaro, O.P.Olson, S.Y. Sol, D.J. Teel, G.M. Ylitalo, N.D. Jahns. 2013. Juvenile Salmon Ecology in Tidal Freshwater Wetlands of the Lower Columbia River and Estuary: Synthesis of the Ecosystem Monitoring Program, 2005–2010. Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration. Available from the Lower Columbia Estuary Partnership, Portland, OR.

Sagar, J.P., A.B. Borde, L.L. Johnson, T.D. Peterson, J.A. Needoba, K.H. Macneale, M. Schwartz, A. Silva, C.A. Corbett, A.C. Hanson, V.I. Cullinan, S.A. Zimmerman, R.M. Thom, P.M. Chittaro, O.P. Olson, S.Y. Sol, D.J. Teel, G.M. Ylitalo, M.A. Maier and C.E. Tausz. 2015. Juvenile Salmon Ecology in Tidal Freshwater Wetlands of the Lower Columbia River and Estuary: Synthesis of the Ecosystem Monitoring Program, Trends (2005–2013) and Food Web Dynamics (2011-2013). Prepared by the Lower Columbia Estuary Partnership for the Bonneville Power Administration. Available from the Lower Columbia Estuary Partnership, Portland, OR.

Sala, O.E. and A.T. Austin. 2000. Methods of estimating aboveground net primary productivity. In: Methods in Ecosystem Science (pp.31-43). Springer, New York.

Sather, N.K., E.M. Dawley, G.E. Johnson, S.A. Zimmerman, A.J. Storch, A.B. Borde, D.J. Teel, C. Mallette, J.R. Skalski, R. Farr, T.A. Jones. 2009. Ecology of Juvenile Salmon in Shallow Tidal Freshwater Habitats in the Vicinity of the Sandy River Delta, Lower Columbia River, 2008. May 2009. Prepared for Bonneville Power Administration under an agreement with the U.S. Department of Energy Contract DE-AC05-76RL01830. Pacific Northwest National Laboratory, Richland, Washington 99352.

Schwartz M.S., A.B. Borde, A. Silva, N. Elasmar, C. Kenny, and M. Vesh. 2017. Action Effectiveness Monitoring for the Lower Columbia River Estuary Habitat Restoration Program (October 2015 – September 2016). Prepared by the Lower Columbia River Estuary Partnership for the Bonneville Power Administration. Available from the Lower Columbia Estuary Partnership, Portland, OR.

Schwing, F.B., R. Mendelssohn, S. Bograd, J.E. Overland, M. Wan, S-I. Ito. 2010. Climate change, teleconnection patterns, and regional processes forcing marine populations in the Pacific. Journal of Marine Systems 70: 245-257.

Searcy, S.P., D.B. Eggleston, and J.A. Hare. 2007. Is growth a reliable indicator of habitat quality and essential fish habitat for a juvenile estuarine fish? Canadian Journal of Fisheries and Aquatic Sciences 64:681-691.

Seeb, L.W., A. Antonovich, M.A. Banks, T.D. Beacham, M.R. Bellinger, S.M. Blankenship, M.R. Campbell, N.A. Decovich, J.C. Garza, C.M. Guthrie III, T.A. Lundrigan, P. Moran, S.R. Narum, J.J. Stephenson, K.T. Supernault, D.J. Teel, W.D. Templin, J.K. Wenburg, S.F. Young, and C.T. Smith. 2007. Development of a standardized DNA database for Chinook salmon. Fisheries 32:540–552.

Shannon, C.E. and W. Weaver. 1949. The mathematical theory of communication. The University of Illinois Press, Urbana, 117 pp.

Sherwood, C.R., D.A. Jay, R.B. Harvey, P. Hamilton, and C.A. Simenstad. 1990. Historical Changes in the Columbia River Estuary. Progress in Oceanography 25:299-357.

Simenstad, C.A., J.L. Burke, J.E. O'Connor, C. Cannon, D.W. Heatwole, M.F. Ramirez, I.R. Waite, T.D. Counihan, and K.L. Jones. 2011. Columbia River Estuary Ecosystem Classification—Concept and Application: U.S. Geological Survey Open-File Report 2011-1228, 54 p.

Simenstad, C.A. and J.R. Cordell. 2000. Ecological assessment criteria for restoring anadromous salmonid habitat in Pacific Northwest estuaries. Ecol. Engineering 15:283-302.

Simenstad, C.A., K.L. Fresh, E.O. Salo. 1982. The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function. In: Kennedy, V.S. (Ed.), Estuarine Comparisons. Academic Press, New York, NY, pp. 343–364.

Simenstad, C.A., D.A Jay, and C.R. Sherwood. 1992. Impacts of watershed management on land-margin ecosystems: The Columbia River estuary. In: Watershed Management (pp. 266-306). Springer New York.

Simon, S.D., M.E. Cardona, B.W. Wilm, J.A. Miner, and D.T. Shaw. 1997. The sum exceedance value as a measure of wetland vegetation hydrologic tolerance. In: Macdonald, K.B. and F. Weinmann (eds). 1997. Wetland and Riparian Restoration: Taking a Broader View. Proceedings of Society for Ecological Restoration, 1995 International Conference, September 14-16, University of Washington, USA. Publication EPA 910-R-97-007, USEPA, Region 10, Seatte, Washington

Sloan, C.A., D.W. Brown, G.M. Ylitalo, J. Buzitis, D.P. Herman, D.G. Burrows, G.K. Yanagida, R.W. Pearce, J.L. Bolton, R.H. Boyer, M.M. Krahn. 2006. Quality assurance plan for analyses of environmental samples for polycyclic aromatic compounds, persistent organic pollutants, fatty acids, stable isotope ratios, lipid classes, and metabolites of polycyclic aromatic compounds. U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-NWFSC-77, 30 pp.

Sloan, C.A., B.F. Anulacion, K.A. Baugh, J.L. Bolton, D. Boyd, R.H. Boyer, D.G. Burrows, D.P. Herman, R.W. Pearce, and G.M. Ylitalo. 2014. Northwest Fisheries Science Center's analyses of tissue, sediment, and water samples for organic contaminants by gas chromatography/mass spectrometry and analyses of tissue for lipid classes by thin layer chromatography/flame ionization detection. U.S. Dept. of Commerce, NOAA Tech. Memo., NMFS-NWFSC-125, 61 p.

Sommer, T.R., M.L. Nobriga, W.C. Harrell, W. Batham, and W.J. Kimmerer. 2001. Floodplain rearing of juvenile Chinook salmon: evidence of enhanced growth and survival. Canadian Journal of Fish and Aquatic Science 58:325–333.

Spilseth, S.A. and C.A. Simenstad. 2011. Seasonal, Diel, and Landscape Effects on Resource Partitioning between Juvenile Chinook Salmon (*Oncorhynchus tshawytscha*) and Threespine Stickleback (*Gasterosteus aculeatus*) in the Columbia River Estuary. Estuaries and Coasts. 34:159-171.

Steel, A.E., P.T. Sandstrom, P.L. Brandes, and A.P. Klimley. 2012. Migration route selection of juvenile Chinook salmon at the Delta Cross Channel, and the role of water velocity and individual movement patterns. Environmental Biology of Fishes 96:215-224.

Tausz, C.E. 2015. Phytoplankton dynamics in off-channel habitats of the lower Columbia River estuary. M.S. Thesis, Oregon Health & Science University.

Teel, D.J., C. Baker, D.R. Kuligowski, T.A. Friesen, and B. Shields. 2009. Genetic stock composition of subyearling Chinook salmon in seasonal floodplain wetlands of the Lower Willamette River. Transactions of the American Fisheries Society 138:211-217.

Teel, D. J., D.L. Bottom, S.A. Hinton, D.R. Kuligowski, G.T. McCabe, R. McNatt, G.C. Roegner, L.A. Stamatiou, and C.A. Simenstad. 2014. Genetic identification of Chinook salmon in the Columbia River estuary: stock-specific distributions of juveniles in shallow tidal freshwater habitats. North American Journal of Fisheries Management 34:621-641.

Thorp, J.H. and A.P. Covich. 2001. Ecology and Classification of North American Freshwater Invertebrates. 2nd ed. San Diego: Academic Press. 1021 pp.

Thorpe, J. E. 1994. Salmonid fishes and the estuarine environment. Estuaries 17:76-93.

Triplehorn, C.A. and N.F. Johnson. 2005. Borror and DeLong’s Introduction to the Study of Insects, 7th ed. Belmont (CA): Brooks/Cole. 864 pp.

van Wezel A.P., D.A.M. de Vries, S. Kostense, D.T.H.M. Sijm, and Q. Opperhuizen. 1995. Intraspecies variation in lethal body burdens of narcotic compounds. Aquatic Toxicology 33:325–342.

Van Zyll De Jong, M. and I.G. Cowx. 2016. Long-term response of salmonid populations to havitat restoration in a boreal forest stream. Ecological Engineering 91:148-157.

Volk, E.C., D.L. Bottom, K.K. Jones, and C.A. Simenstad. 2010. Reconstructing Juvenile Chinook Salmon Life History in the Salmon River Estuary, Oregon, Using Otolith Microchemistry and Microstructure. Transactions of the American Fisheries Society 139:535-549.

Washington Department of Ecology (WADOE). 2001. An aquatic plant identification manual for Washington’s freshwater plants. 01-10-032. Olympia, Washington. Available online at: http://www.ecy.wa.gov/programs/wq/plants/plantid2/index.html.

Weitkamp, L.A., G. Goulette, J. Hawkes, M. O’Malley, and C. Lipsky. 2014. Juvenile salmon in estuaries: comparisons between North American Atlantic and Pacific salmon populations. Reviews in Fish Biology and Fisheries 24:713-736.

Windham, L. 2001. Comparison of biomass production and decomposition between *Phragmites australis* (common reed) and *Spartina patens* (salt hay grass) in brackish tidal marshes of New Jersey, USA. Wetlands 21(2):179-188.

Wolda, H. 1988. Insect Seasonality: Why? Annual Review of Ecology and Systematics. 19:1-18.

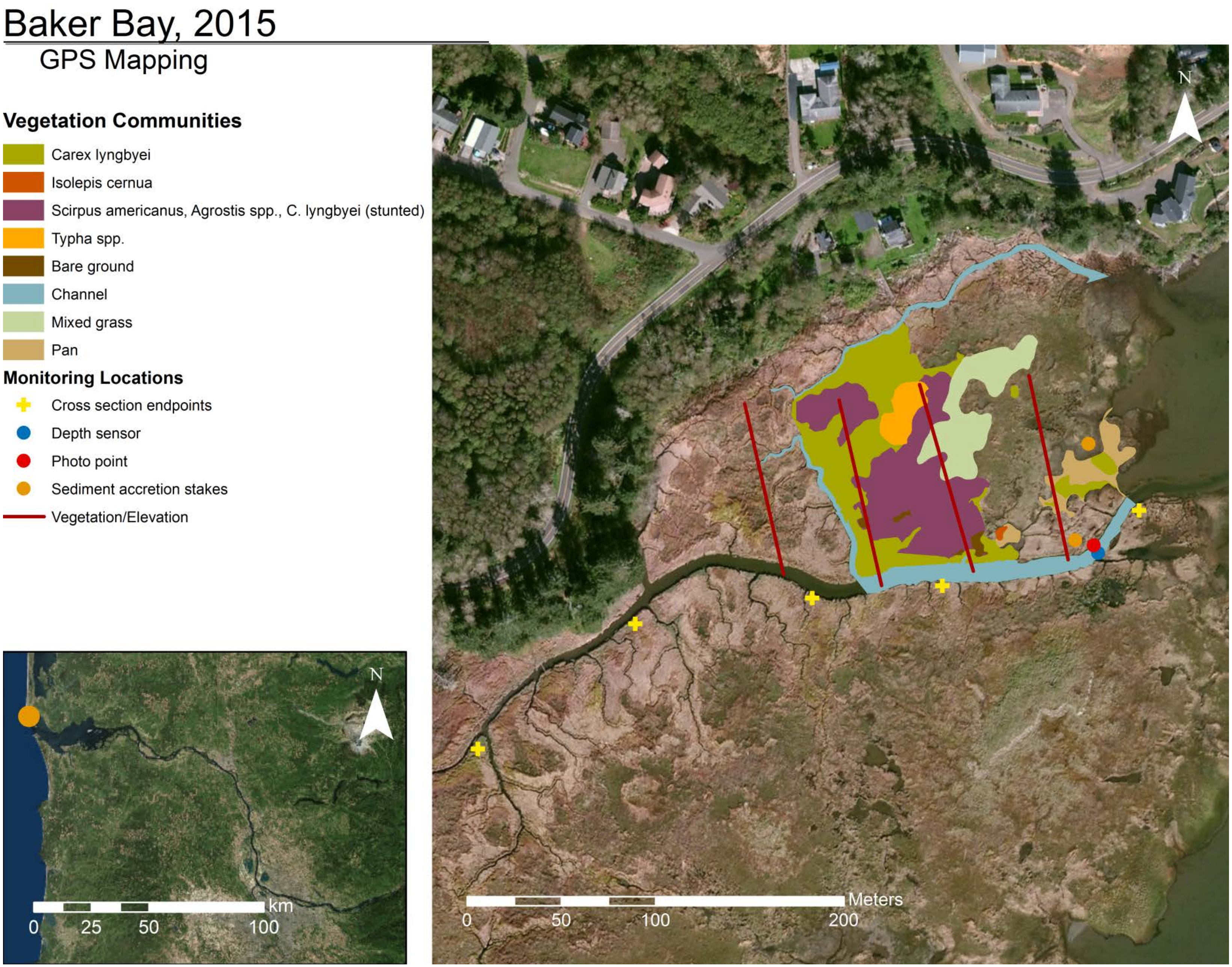
Ylitalo, G.M., G.K. Yanagida, L.C. Hufnagle Jr., M.M. Krahn. 2005. Determination of lipid classes and lipid content in tissues of aquatic organisms using a thin layer chromatography/flame ionization detection (TLC/FID) microlipid method. In Ostrander, G.K. (Ed.) Techniques in Aquatic Toxicology. CRC Press, Boca Raton, FL. Pages 227-237.

Zabel, R.W., K. Haught, and P.M. Chittaro. 2010. Variability in fish size/otolith radius relationships among populations of Chinook salmon. Environmental Biology of Fishes 89:267-278.

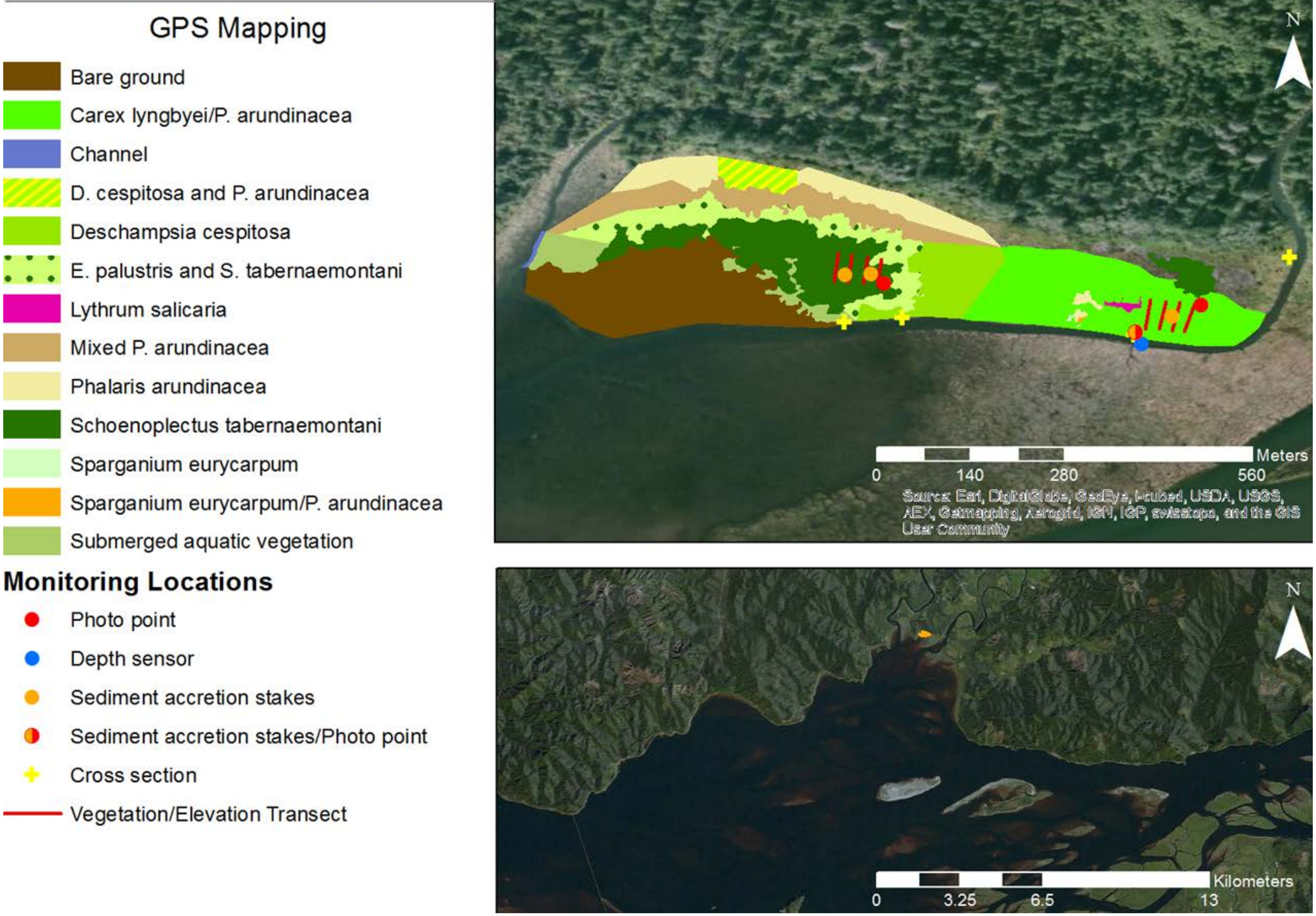
7 Appendices

Appendix A. Site Maps and Habitat Change Analysis

Contents:


Baker Bay, 2015
GPS Mapping
Vegetation Communities
Carex lyngbyei
Isolepis cernua
Scirpus americanus, Agrostis spp., C. lyngbyei (stunted)
Typha spp.
Bare ground
Channel
Mixed grass
Pan
Monitoring Locations
Cross section endpoints
Depth sensor
Photo point
Sediment accretion stakes
Vegetation/Elevation
N
Meters
0
50
100
200
N
km
0
25
50
100

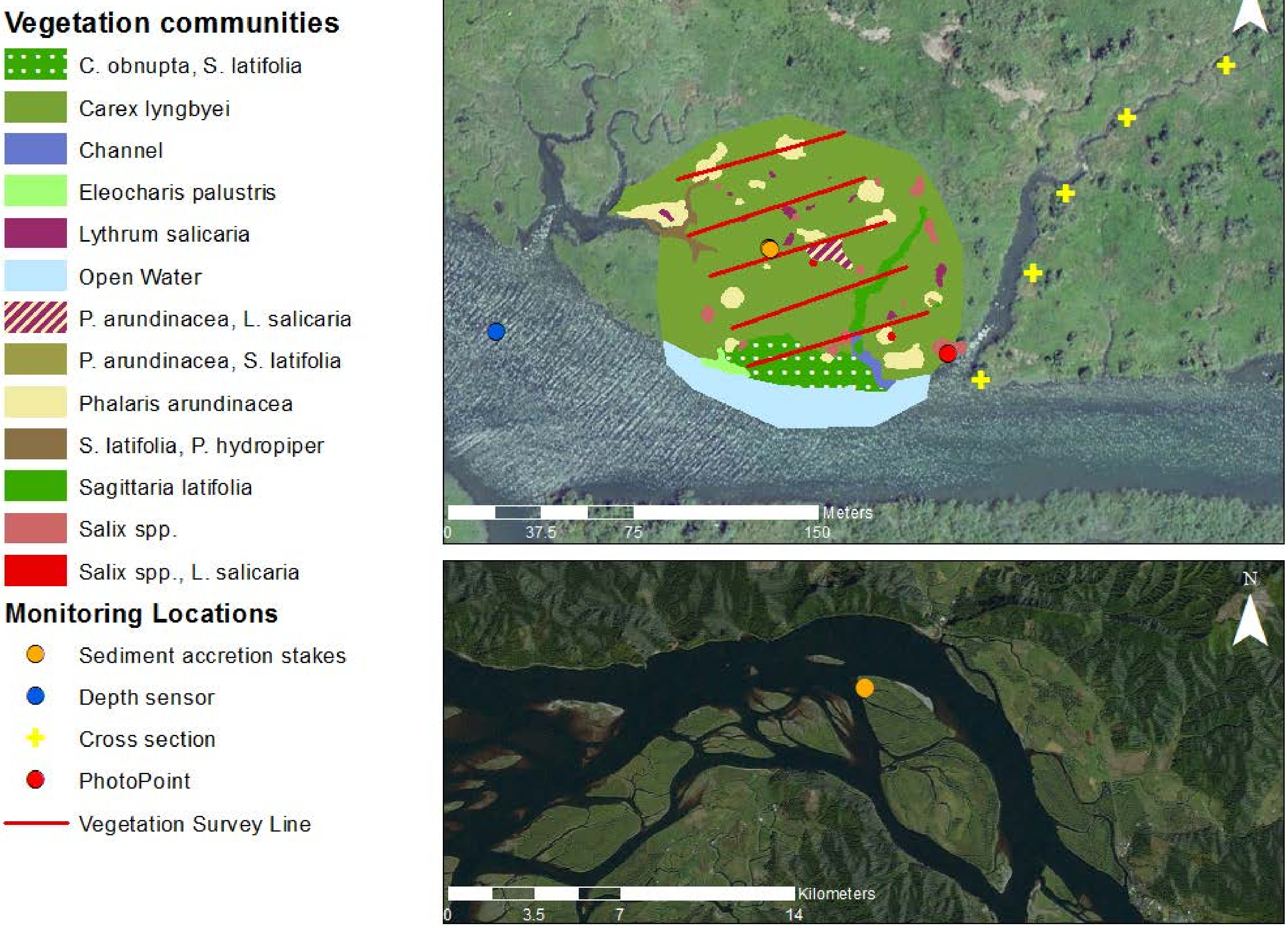
Secret River Marsh, 2013

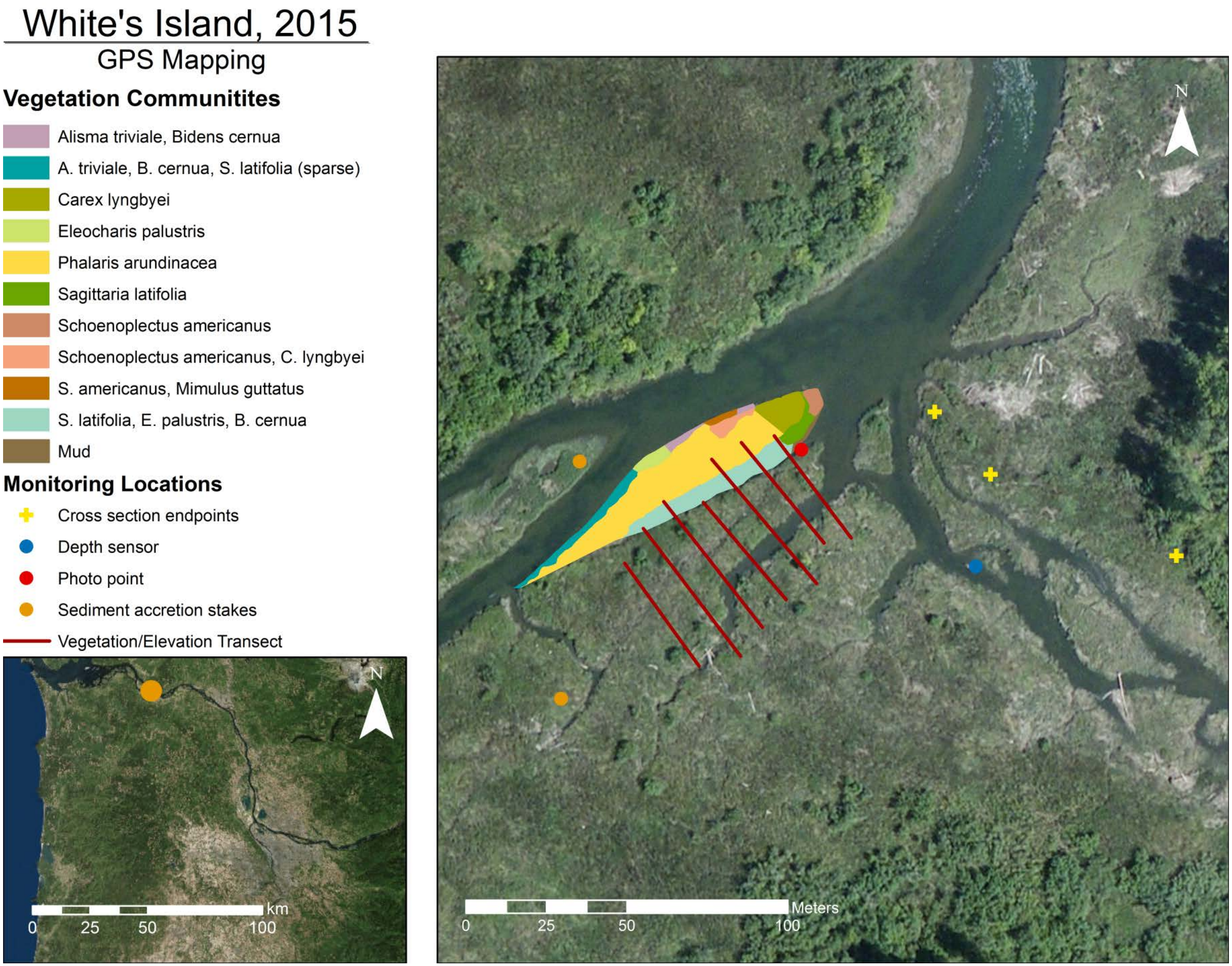


Source: Esri, DigitalGlobe, GeoEye, i-cubed, USDA, USGS, AEX, Getmapping, Aerogrid, IGN, IGP, swisstopo, and the GIS User Community

Welch Island, 2012

GPS Mapping




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

Cunningham Lake, 2015

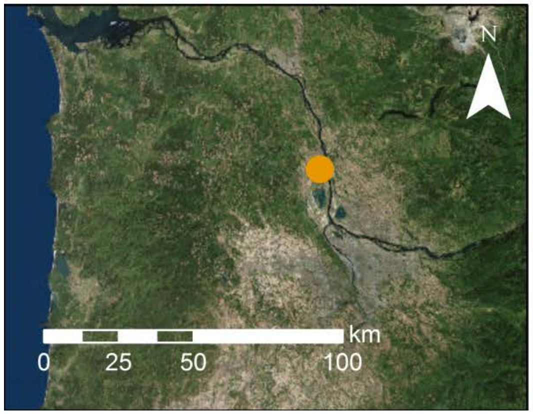
GPS Mapping

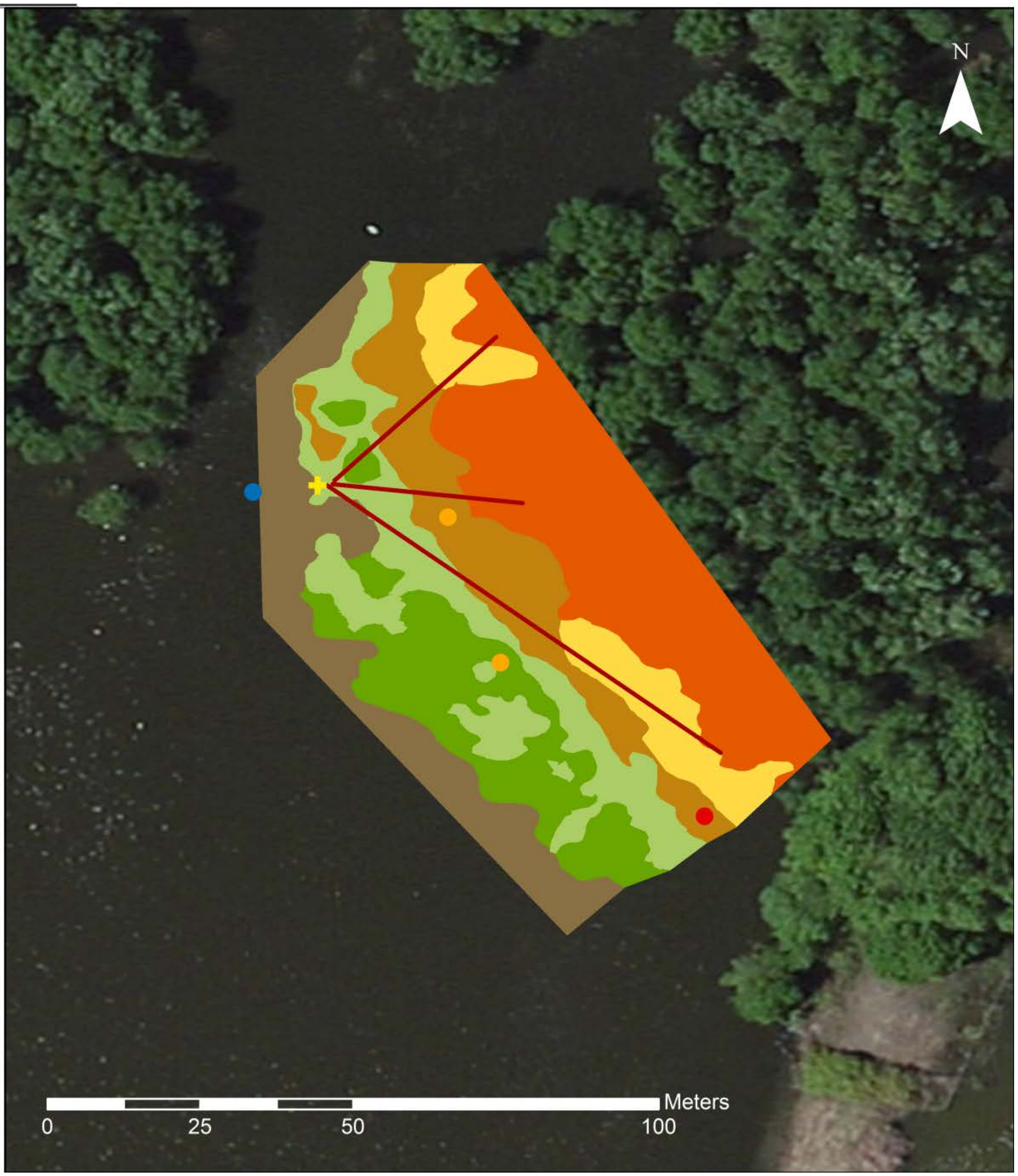
Vegetation Communitites

- 🟩 Eleocharis palustris, S. latifolia
- 🟨 Phalaris arundinacea
- 🟫 P. arundinacea, S. latifolia
- 🟩 Sagittaria latifolia
- 🟧 Salix spp.
- 🟫 Mud

Monitoring Locations

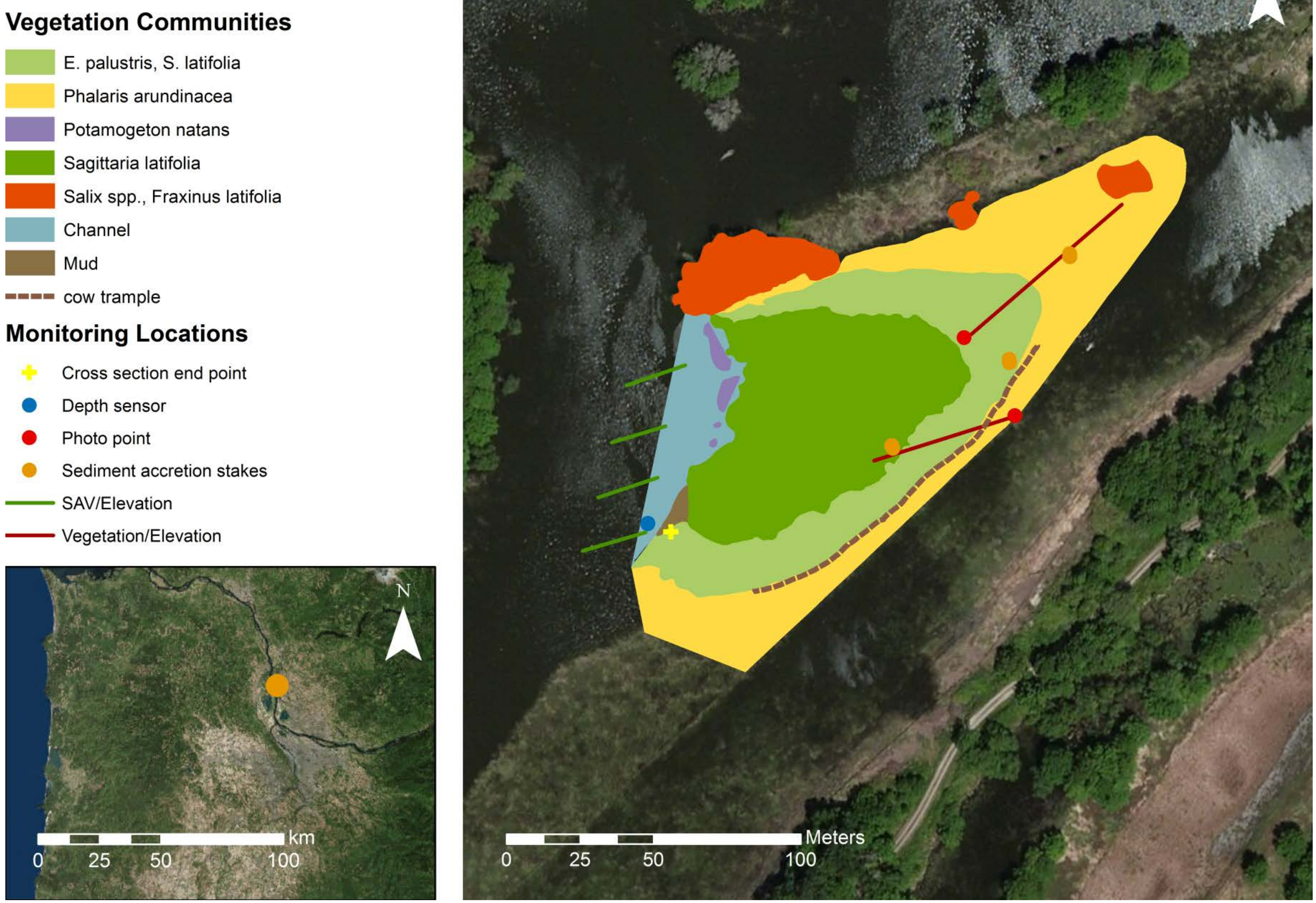
- ✚ Cross section endpoints
- ● Depth sensor
- ● Photo point
- ● Sediment accretion stakes
- — Vegetation/Elevation





Campbell Slough, 2015

GPS Mapping



Franz Lake, 2015

GPS Mapping

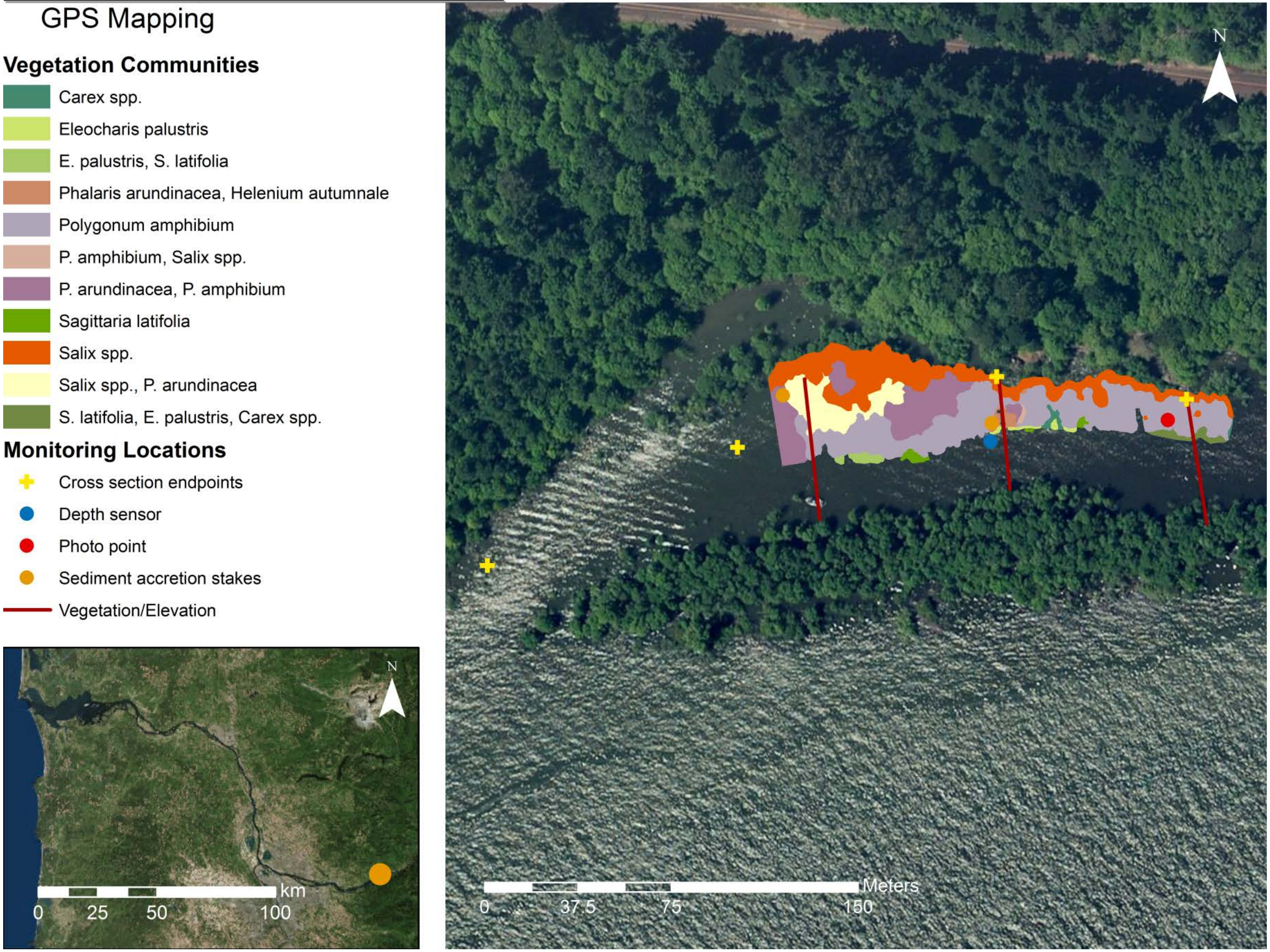


Table A1. Habitat change analysis of vegetation communities at the trend sites; comparison of overlapping areas for the earliest year mapped and the latest year mapped. All area units are square meters. Vegetation communities are ordered from the lowest elevation to the highest elevation at a site; species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

Ilwaco

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

Welch Island

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

Whites Island

			2009 Vegetation Community						
	Area Compared:	**1585**							
	Area Changed:	**729**	**Channel**	**SALA**	**ELPA**	**ELPA, SALA**	**ELPA, SCAM**	**CALY**	**PHAR**
	No Change:	**855**	**163**	**252**	**115**	**18**	**82**	**191**	**763**
2015 Vegetation Community	**Mud**	**15**	6	8					
	ALPL, BICE	**43**	31				12		
	SALA	**55**		14	31			10	
	SALA, ALPL, BICE	**107**	86				14		8
	SALA, ELPA, BICE	**297**		230	36	13			17

SCAM	**34**			34				
SCAM, MIGU	**19**	7				12		
SCAM, CALY	**39**	5					23	11
ELPA	**40**	13				26		
CALY	**114**						114	
PHAR	**823**	14		14	5	18	44	727

Cunningham Lake

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

Campbell Slough

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

Franz Lake

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

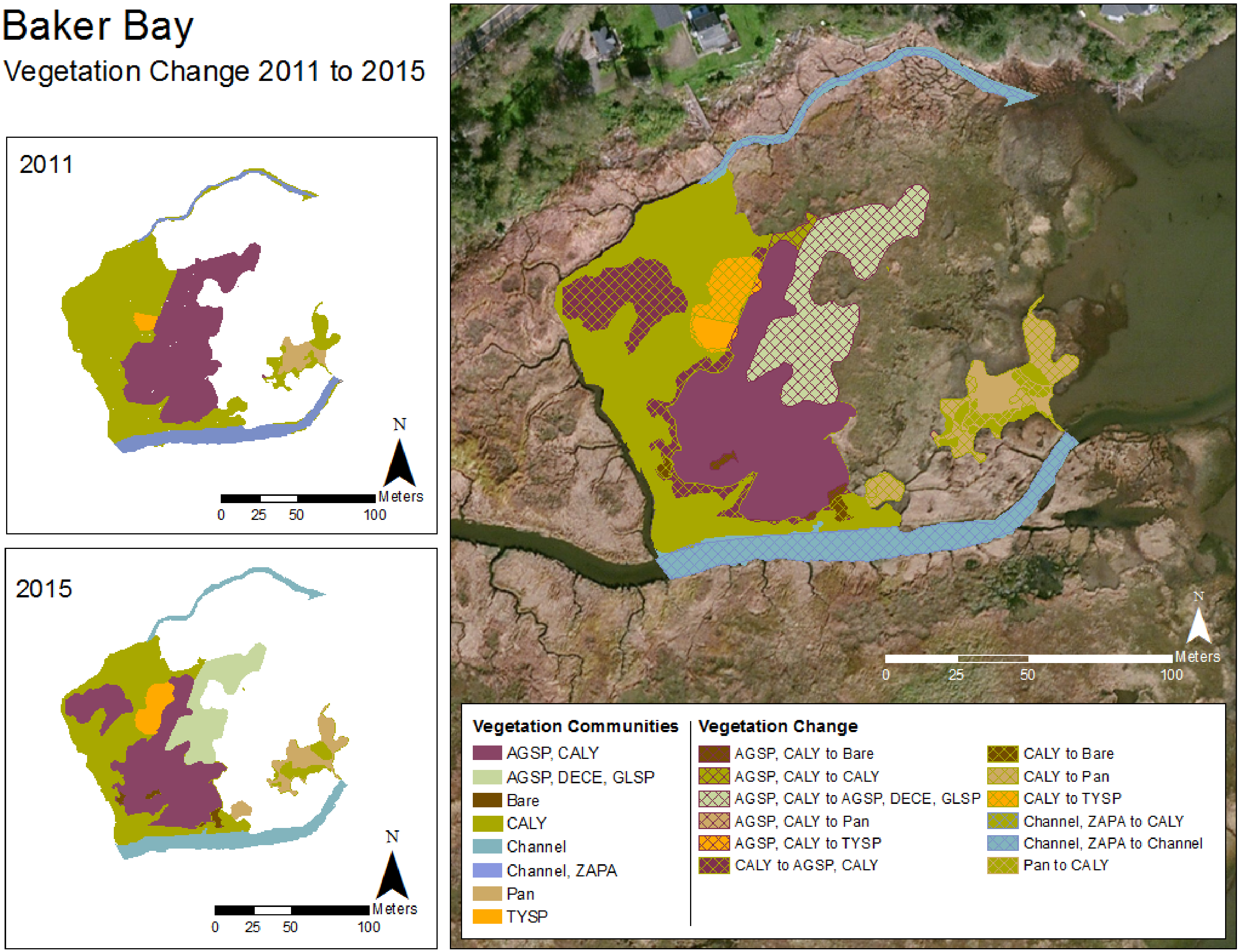
Franz Lake

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

SASP, PHAR	**522**									39	43	136		304

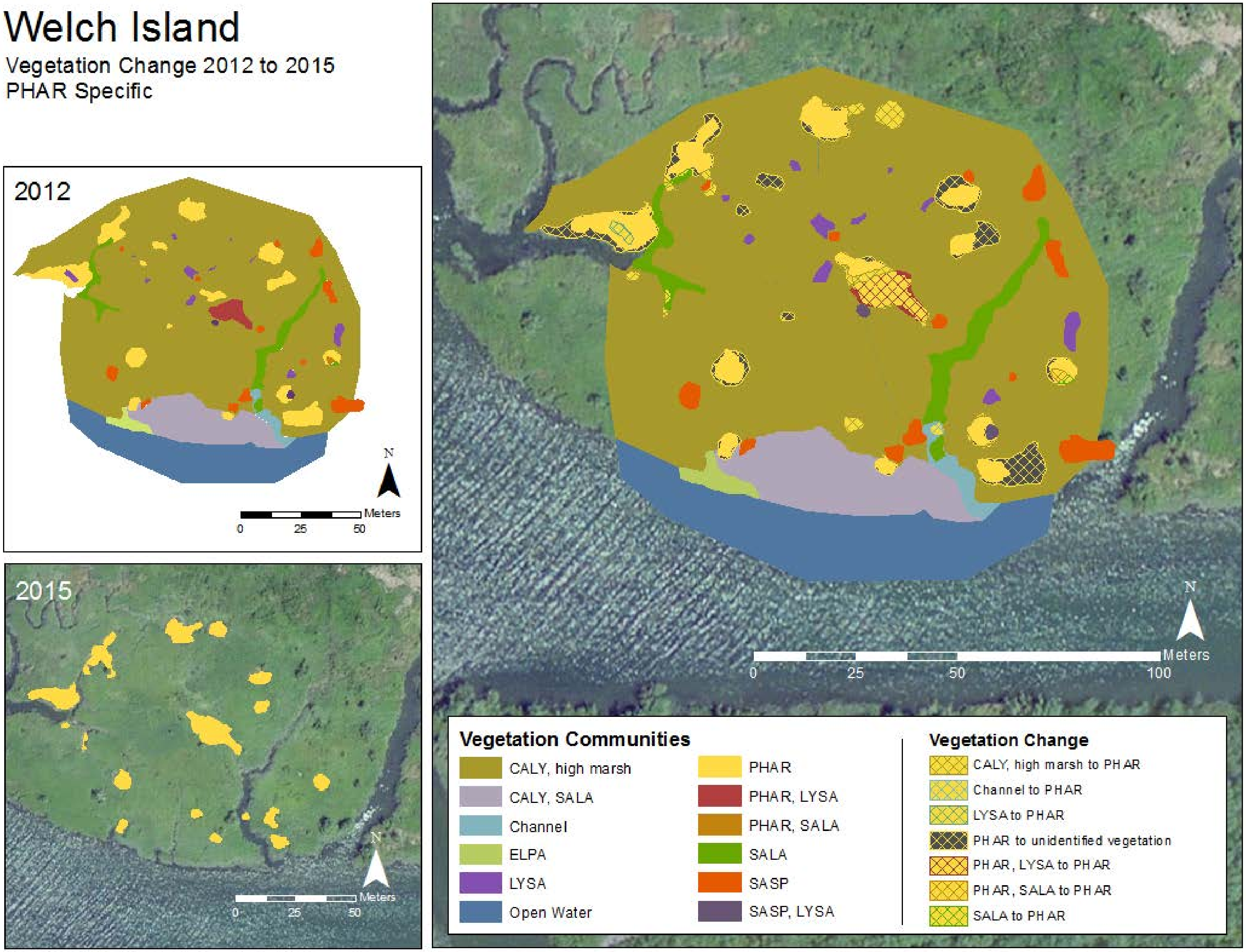
Baker Bay

Vegetation Change 2011 to 2015



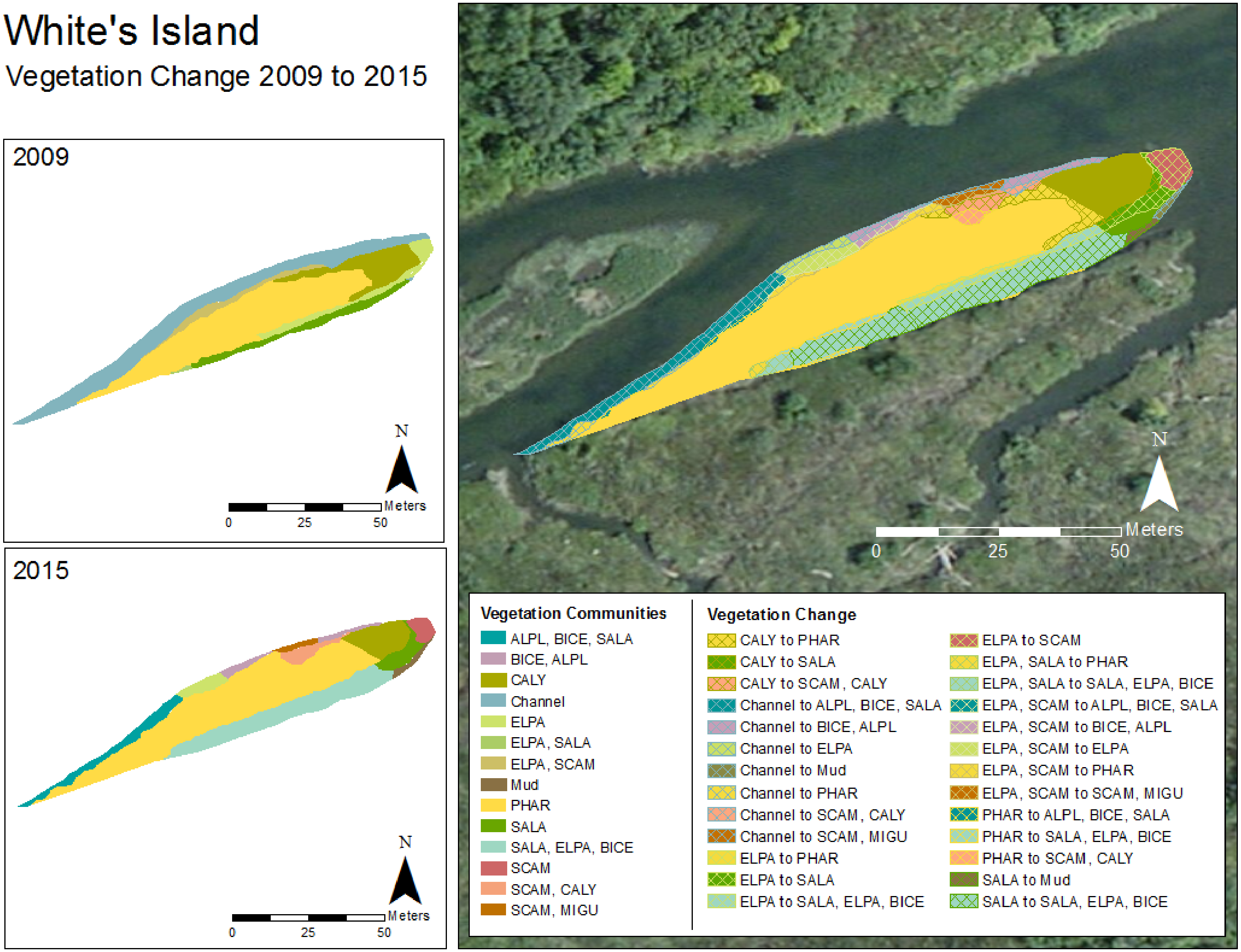
Welch Island

Vegetation Change 2012 to 2015
PHAR Specific



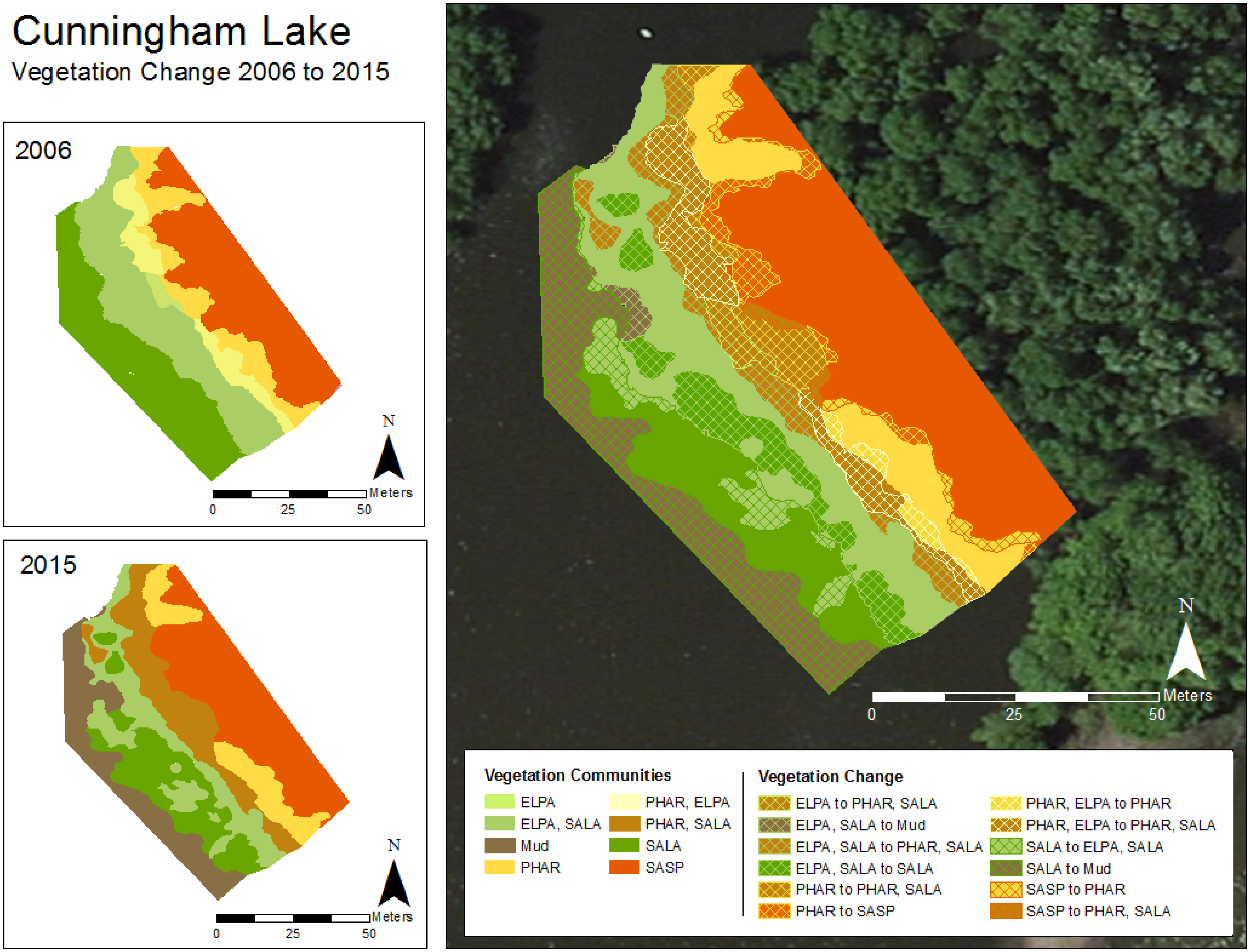
White's Island

Vegetation Change 2009 to 2015



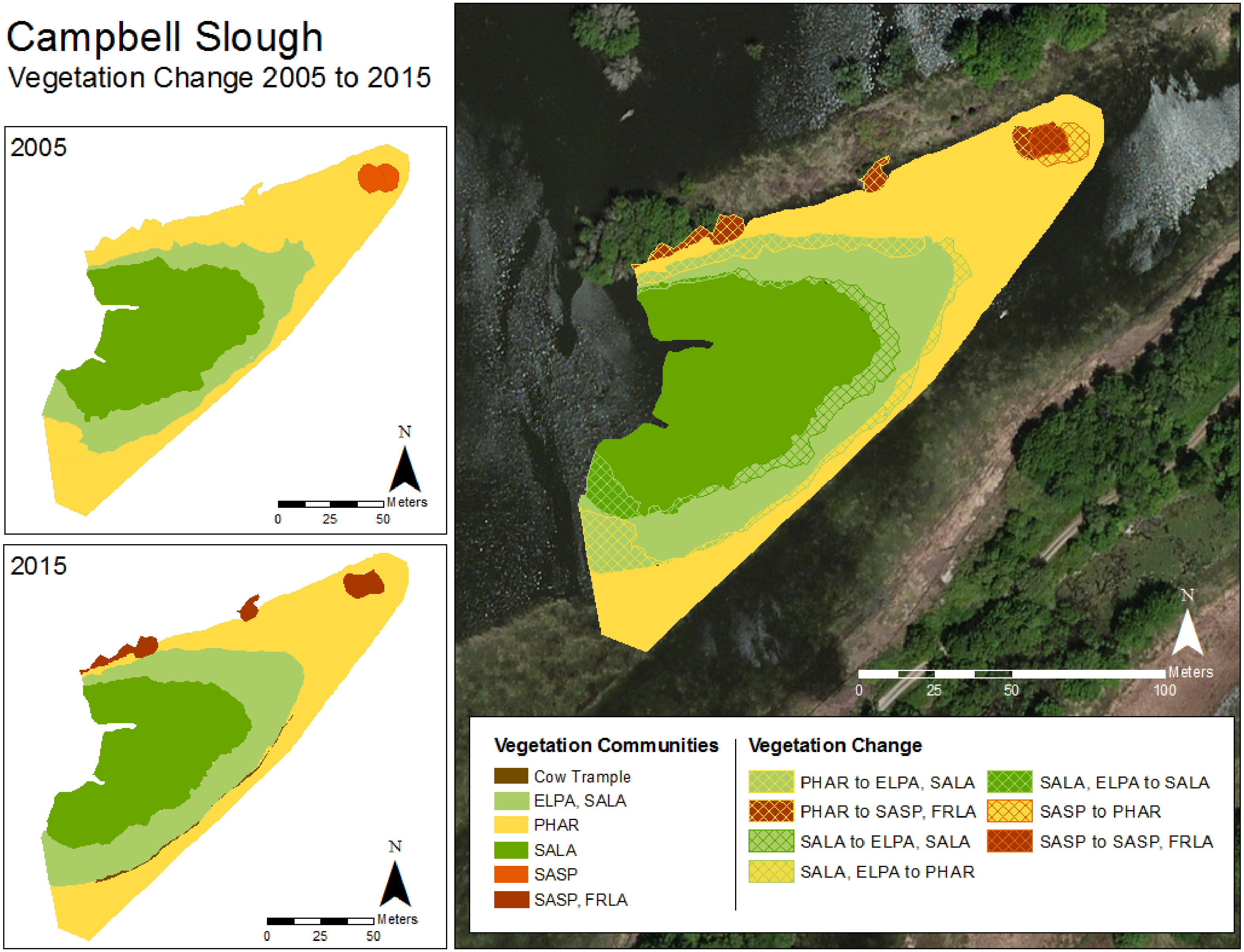
Cunningham Lake

Vegetation Change 2006 to 2015



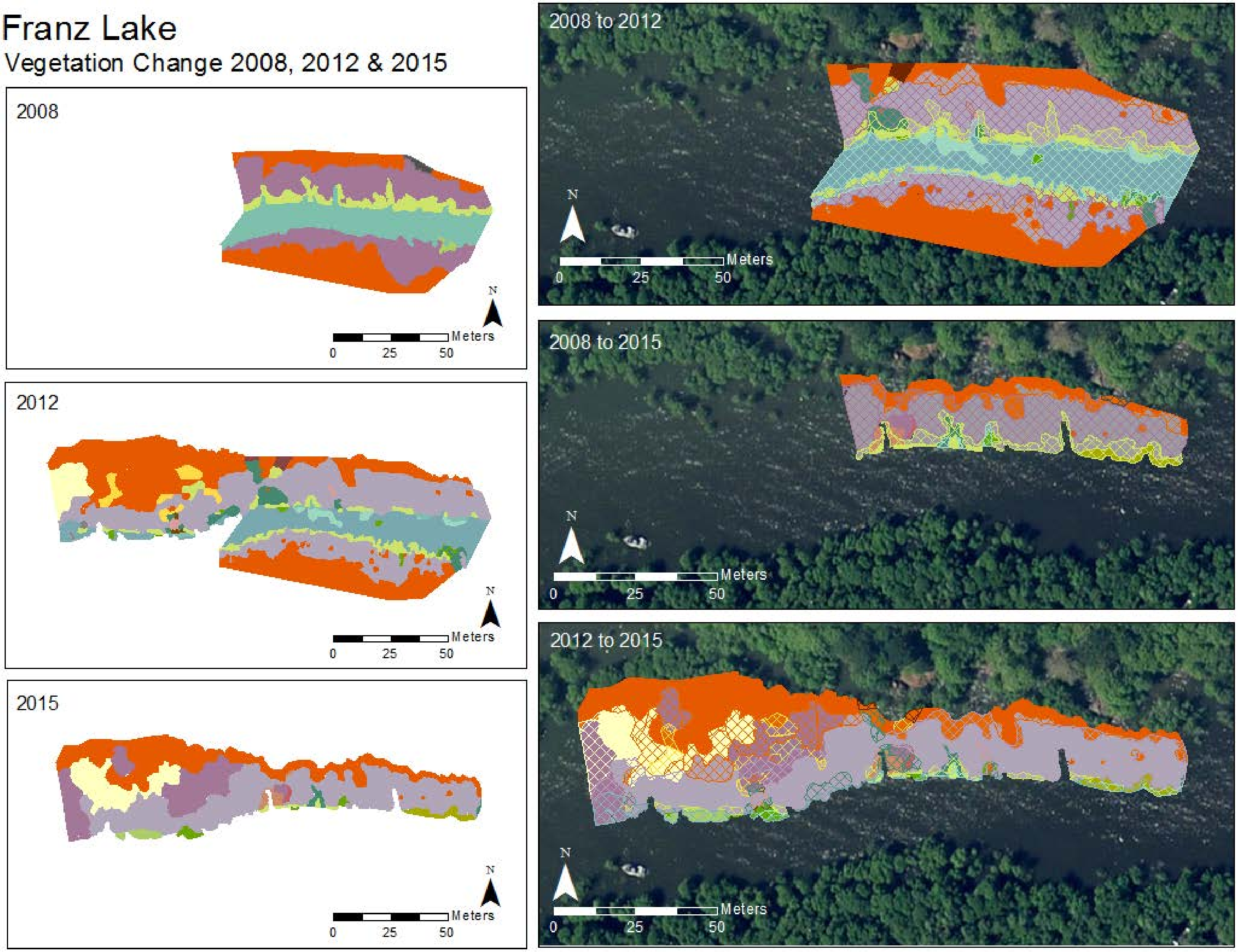
Campbell Slough

Vegetation Change 2005 to 2015



Franz Lake

Vegetation Change 2008, 2012 & 2015



Appendix B. Annual photo points from EMP trends sites

31 July 2011

15 February 2012

4 August 2012

4 February 2013

Baker Bay – PP1

26 July 2013

20 September 2013

3 February 2014

27 June 2014

Baker Bay – PP1

2 August 2015

7 August 2016

Secret River – PP1 [HIGH MARSH]

5 February 2010

2 August 2012

9 August 2013

1 August 2015

Secret River – PP1 [HIGH MARSH]

6 August 2016

Secret River – PP2 [LOW MARSH]

1 December 2011

2 August2012

15 July 2014

1 August 2015

Secret River – PP2 [LOW MARSH]

6 August 2016

Secret River – PP3 [CHANNEL]

1 December 2011

……

15 July 2014

Secret River – PP3 [CHANNEL]

1 August 2015

6 August 2016

Welch Island – PP1

1 August 2012

3 February 2013

23 July 2013

Welch Island – PP1

1 August 2014

31 July 2015

5 August 2016

Whites Island – PP1

22 July 2009

---

13 July2010

2 August 2011

---

15 February 2012

Whites Island – PP1

31 July 2012

5 February 2013

22 July 2013

Whites Island – PP1

4 February 2014

31 July 2014

30 July 2015

Whites Island – PP1

4 August 2016

Cunningham Lake – PP1

26 July 2005

18 July 2007

21 July 2008

Cunningham Lake – PP1

25 July 2009

17 May 2010

28 July 2010

Cunningham Lake – PP1

30 July 2011

8 August 2012

29 July 2013

18 July 2014

Cunningham Lake – PP1

28 July 2015

1 August 2016

Campbell Slough – PP1

29 July 2005

---

15 July 2006

---

5 September 2006

Campbell Slough – PP1

17 July 2007

---

26 July 2010

29 July 2011

Campbell Slough – PP1

15 February 2012

21 July 2012

10 August 2012

Campbell Slough – PP1

27 July 2013

18 July 2014

29 July 2015

Campbell Slough – PP1

3 August 2016

Campbell Slough – PP2

25 July 2005

----

27 July 2009

26 July 2010

29 July 2011

10 Aug 2012

Campbell Slough – PP2

27 July 2013

18 July 2014

29 July 2015

3 August 2016

Campbell Slough – PP2

22 July 2008

28 July 2009

25 August 2011

14 February 2012

21 July 2012

Campbell Slough – PP2

30 August 2012

11 October 2012

6 February 2013

31 July 2013

Campbell Slough – PP2

12 February 2014

7 August 2014

----

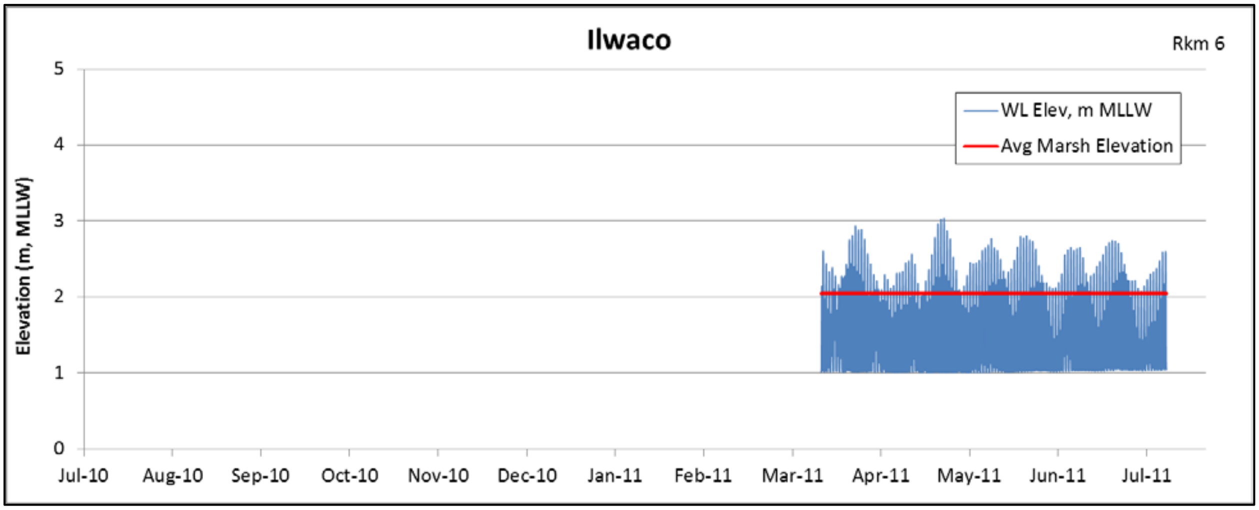
27 July 2015

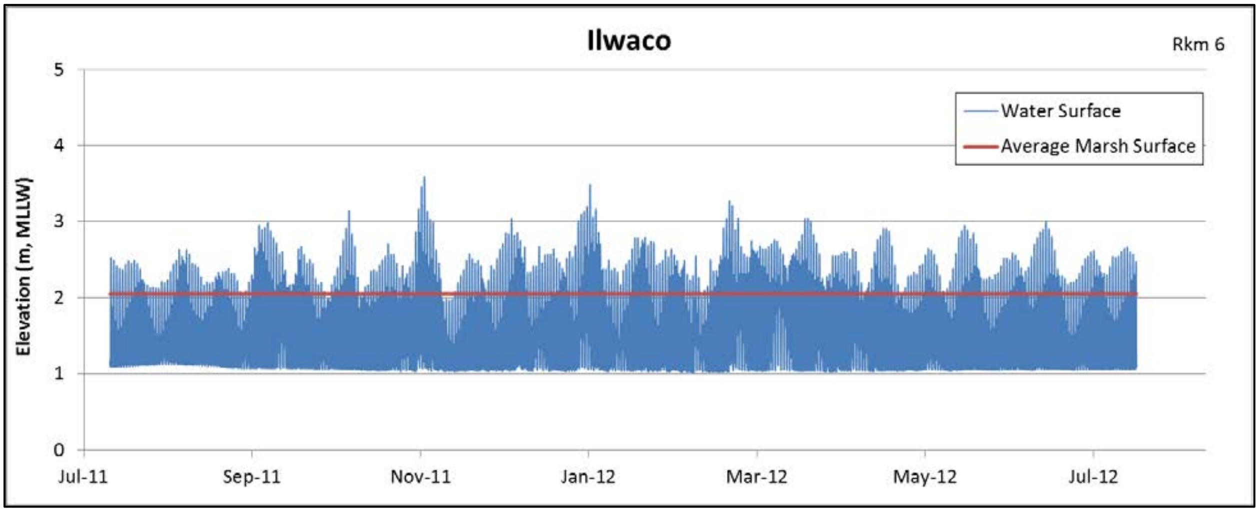
Campbell Slough – PP2

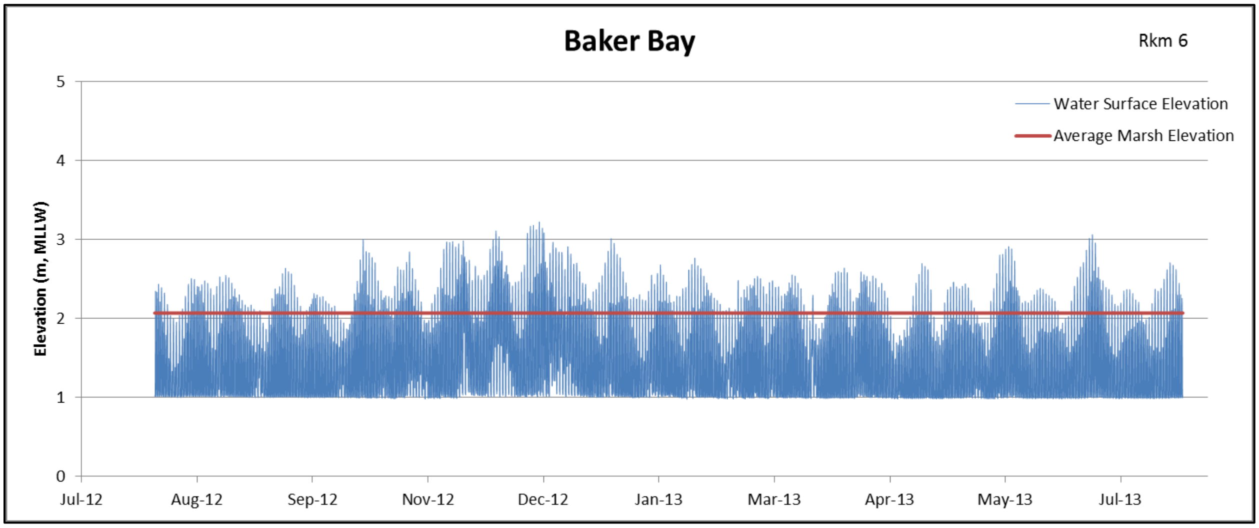
2 August 2016

Appendix C. Site Hydrographs

Hydrographs are in order by site location in the River, starting at the mouth.







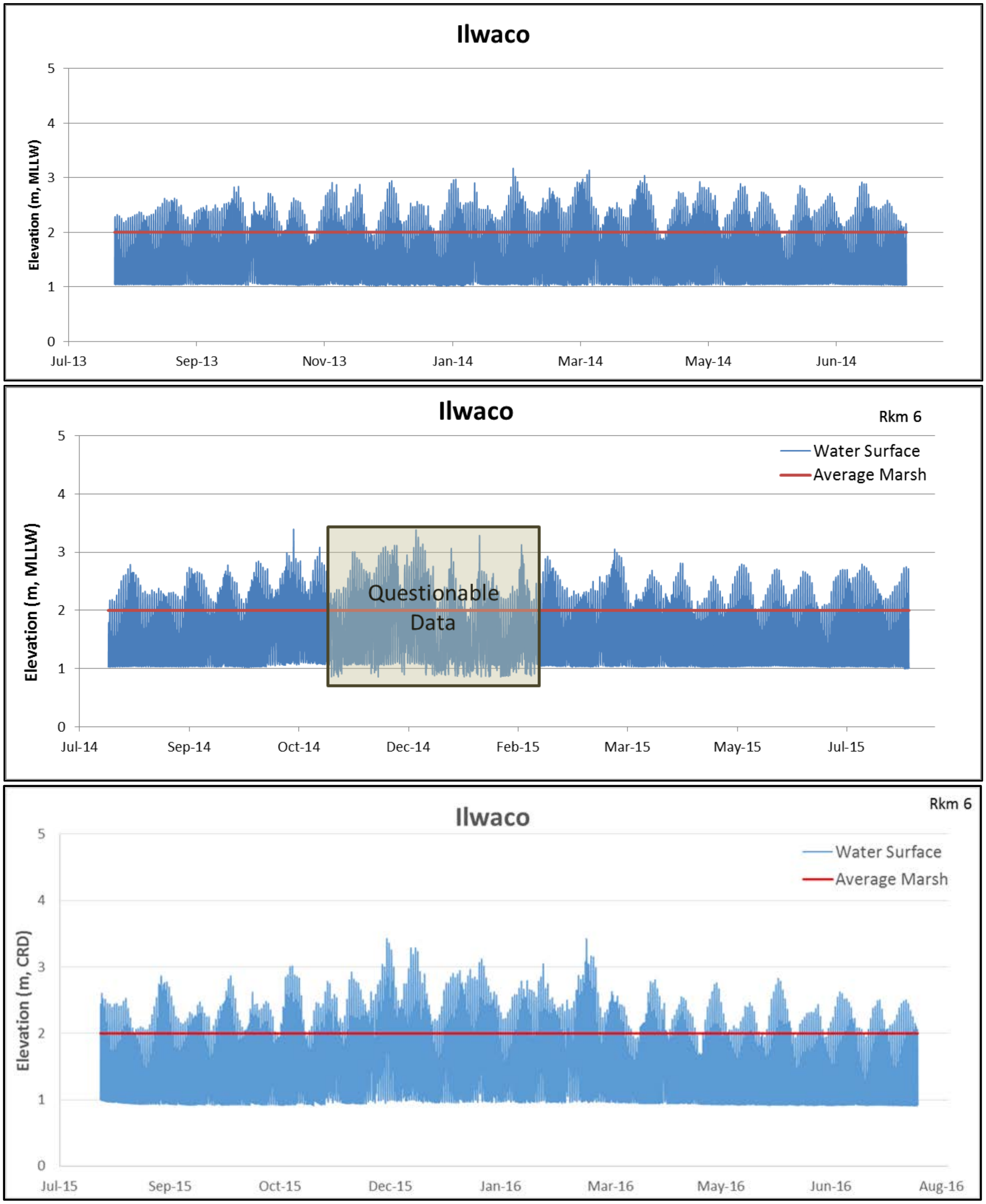
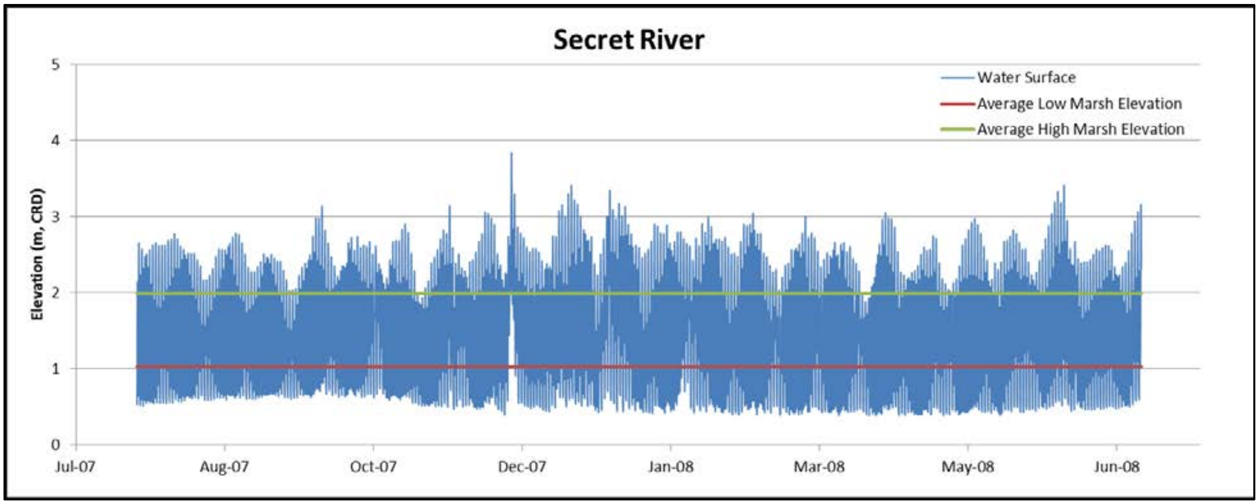
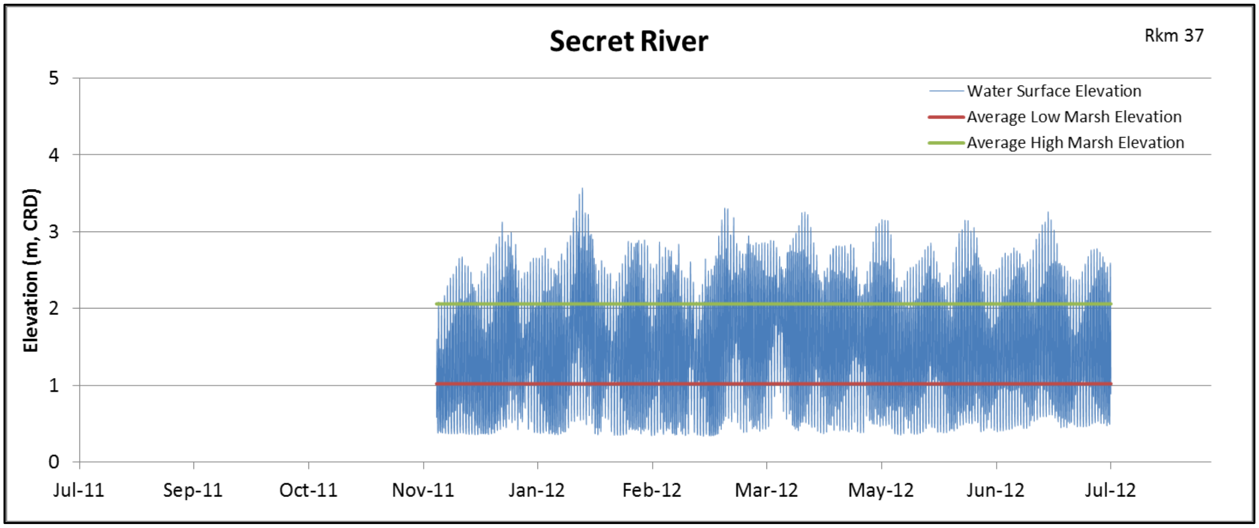
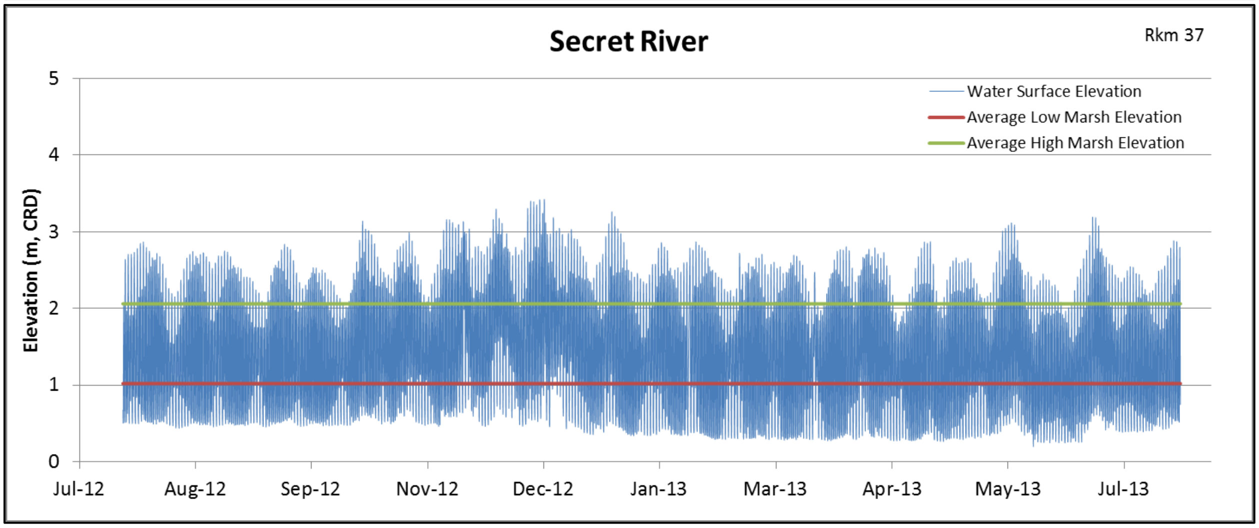


Figure A - 1. Water surface elevation data from the Ilwaco study site for the years 2011-2016. The red line represents the average elevation of the marsh sampling area. In Nov 2014-Fb 2015, the sensor appears to have become disconnected from the deployment post; measurements from this time should not be used in calculations.


Secret River
Elevation (m, CRD)
0
1
2
3
4
5
Jul-07
Aug-07
Oct-07
Dec-07
Jan-08
Mar-08
May-08
Jun-08
Water Surface
Average Low Marsh Elevation
Average High Marsh Elevation


Secret River
Rkm 37
Elevation (m, CRD)
0
1
2
3
4
5
Jul-11
Sep-11
Oct-11
Nov-11
Jan-12
Feb-12
Mar-12
May-12
Jun-12
Jul-12
Water Surface Elevation
Average Low Marsh Elevation
Average High Marsh Elevation


Secret River
Rkm 37
Elevation (m, CRD)
0
1
2
3
4
5
Jul-12
Aug-12
Sep-12
Nov-12
Dec-12
Jan-13
Mar-13
Apr-13
May-13
Jul-13
Water Surface Elevation
Average Low Marsh Elevation
Average High Marsh Elevation

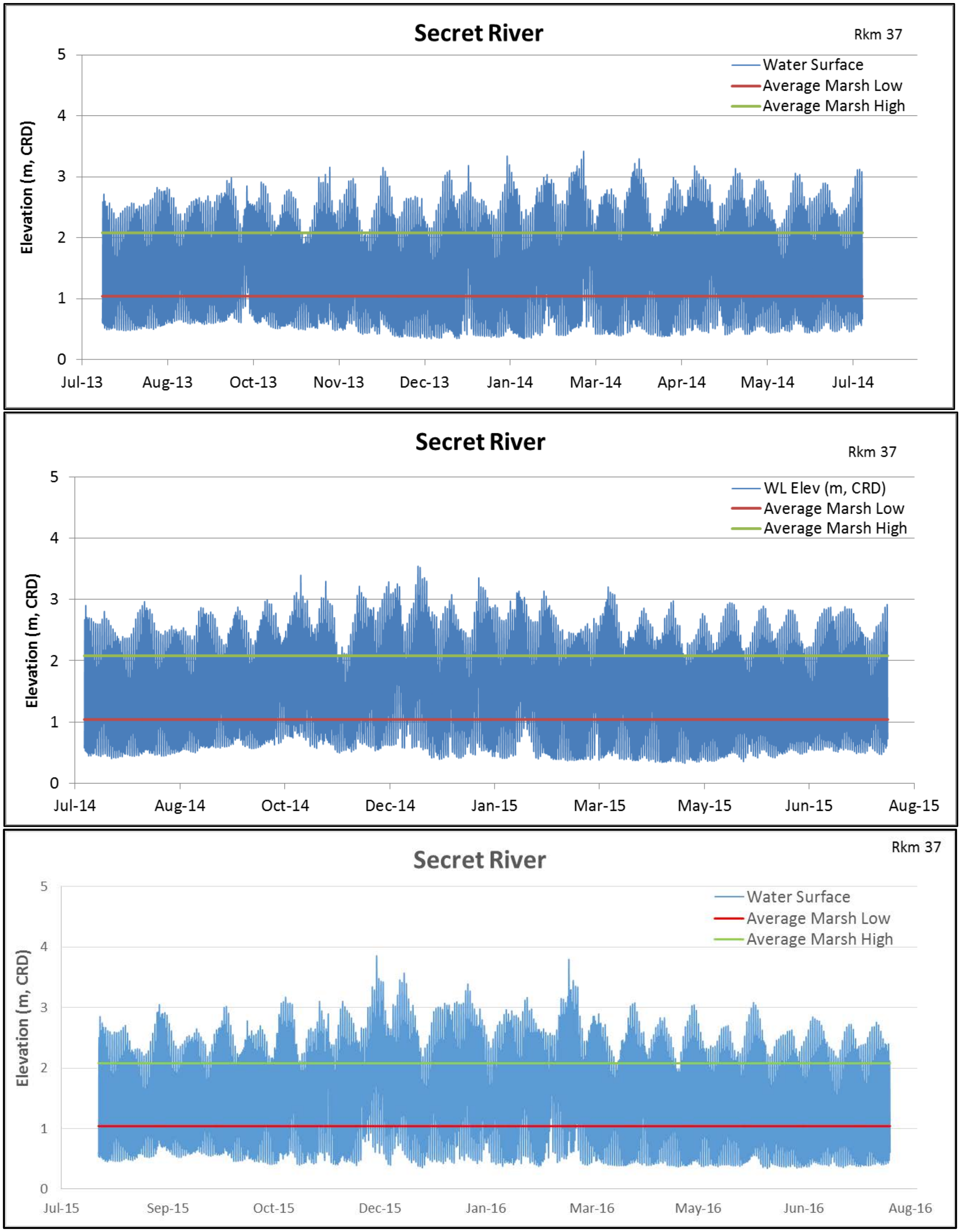
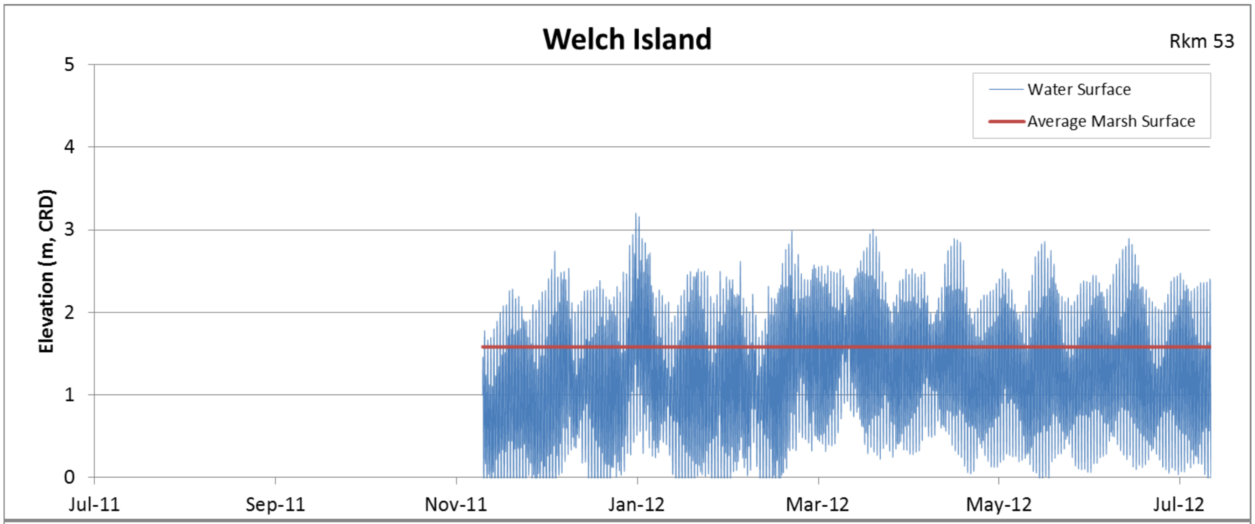
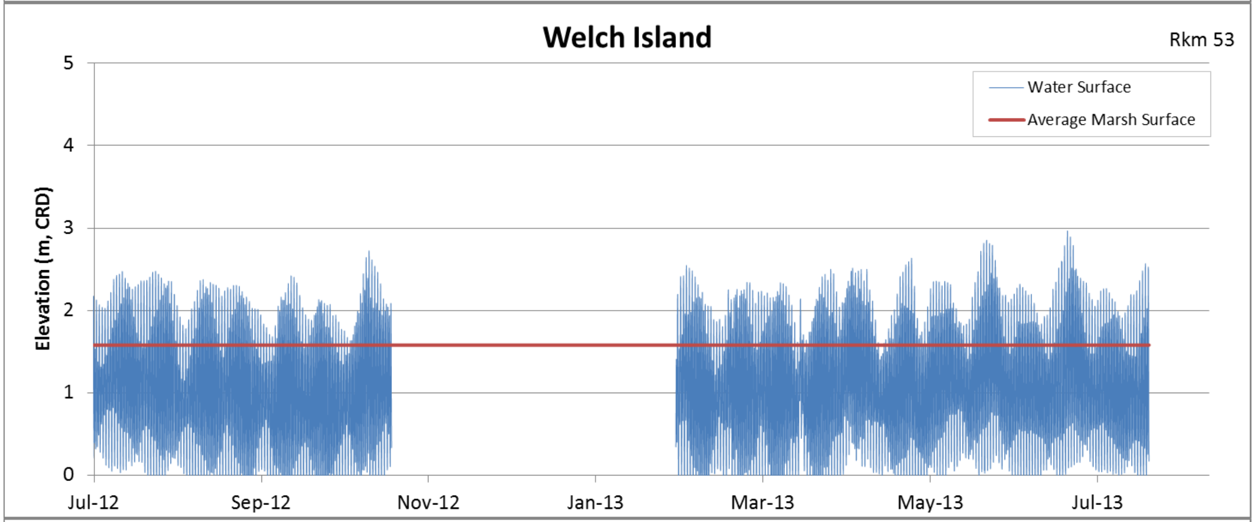
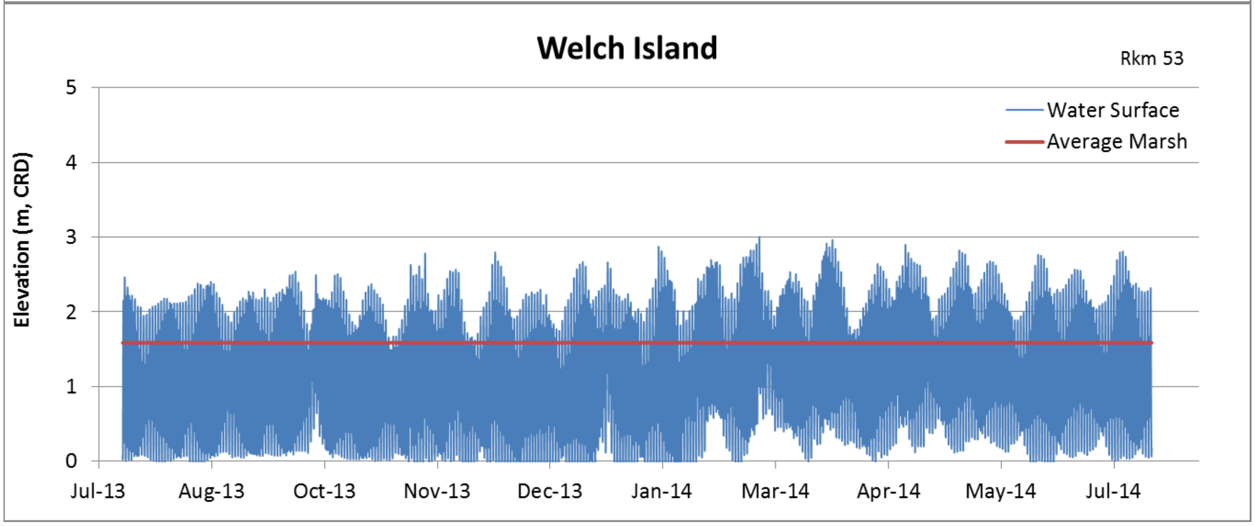


Figure A -2. Water surface elevation data from the Secret River study site for the years 2007-2008 and 2011-2016. The red line represents the average elevation of the marsh sampling area.


Welch Island
Rkm 53
Water Surface
Average Marsh Surface
Elevation (m, CRD)
0
1
2
3
4
5
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12


Welch Island
Rkm 53
Water Surface
Average Marsh Surface
Elevation (m, CRD)
0
1
2
3
4
5
Jul-12
Sep-12
Nov-12
Jan-13
Mar-13
May-13
Jul-13


Welch Island
Rkm 53
Water Surface
Average Marsh
Elevation (m, CRD)
0
1
2
3
4
5
Jul-13
Aug-13
Oct-13
Nov-13
Dec-13
Jan-14
Mar-14
Apr-14
May-14
Jul-14

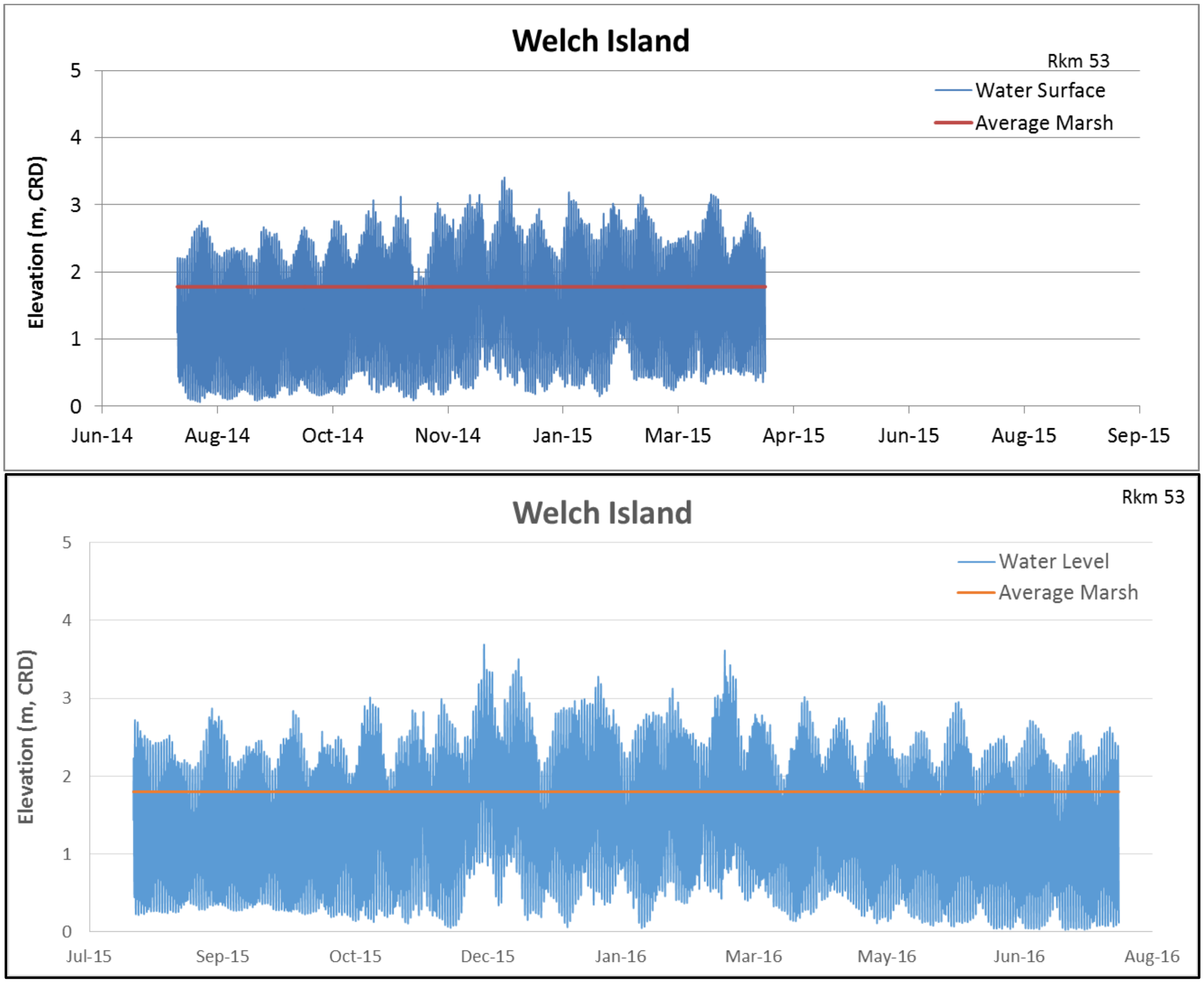
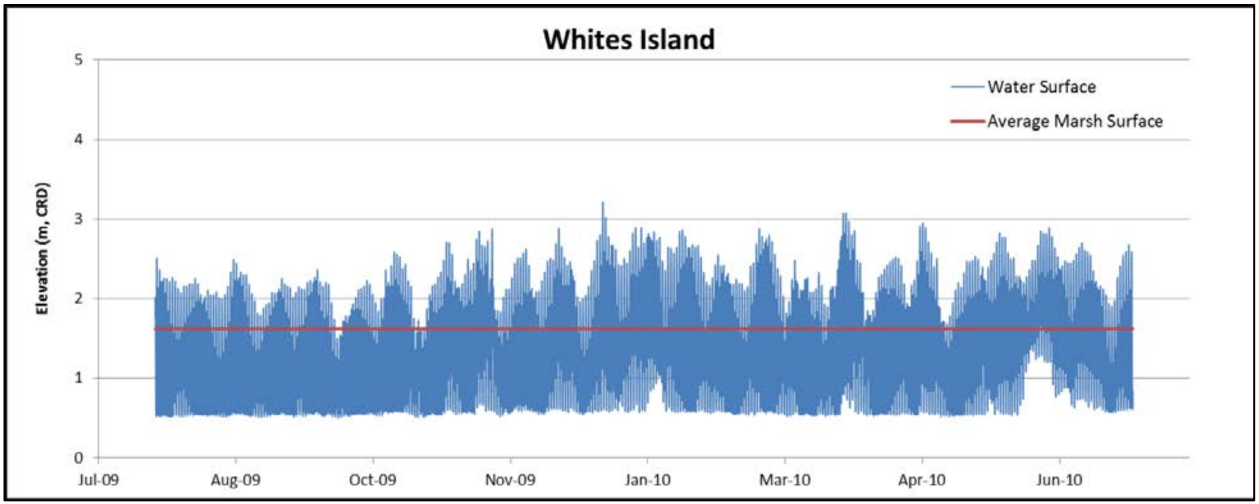
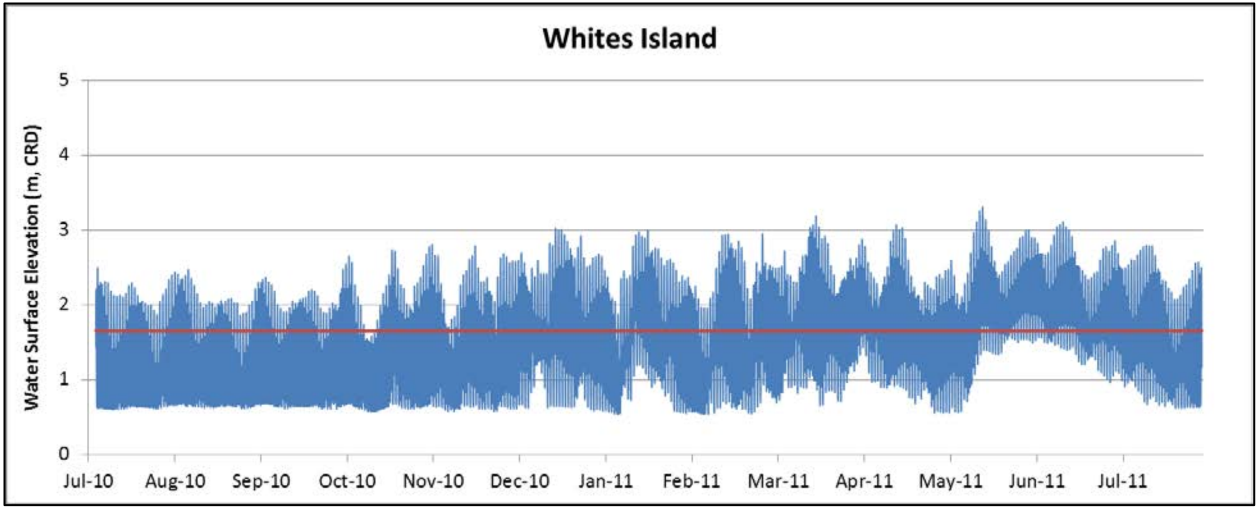
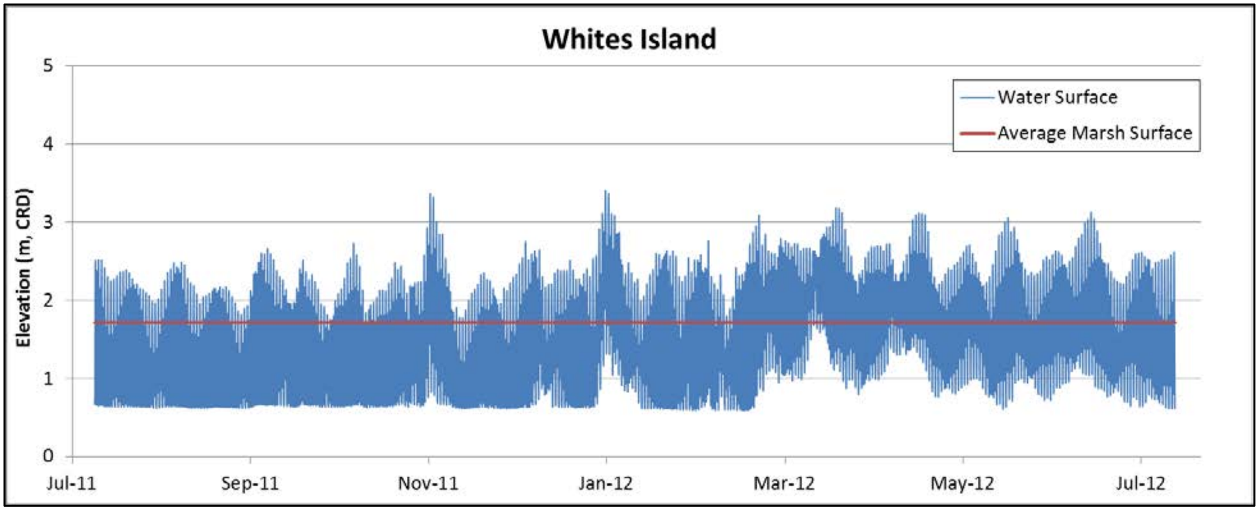


Figure A -3. Water surface elevation data from the Welch Island study site for the years 2012-2015. The red line represents the average elevation of the marsh sampling area. The sensor was displaced between early November 2012 and February 2013 therefore no data is reported for that time.


Whites Island
Elevation (m, CRD)
5
4
3
2
1
0
Jul-09
Aug-09
Oct-09
Nov-09
Jan-10
Mar-10
Apr-10
Jun-10
Water Surface
Average Marsh Surface


Whites Island
Water Surface Elevation (m, CRD)
5
4
3
2
1
0
Jul-10
Aug-10
Sep-10
Oct-10
Nov-10
Dec-10
Jan-11
Feb-11
Mar-11
Apr-11
May-11
Jun-11
Jul-11


Whites Island
Elevation (m, CRD)
5
4
3
2
1
0
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12
Water Surface
Average Marsh Surface

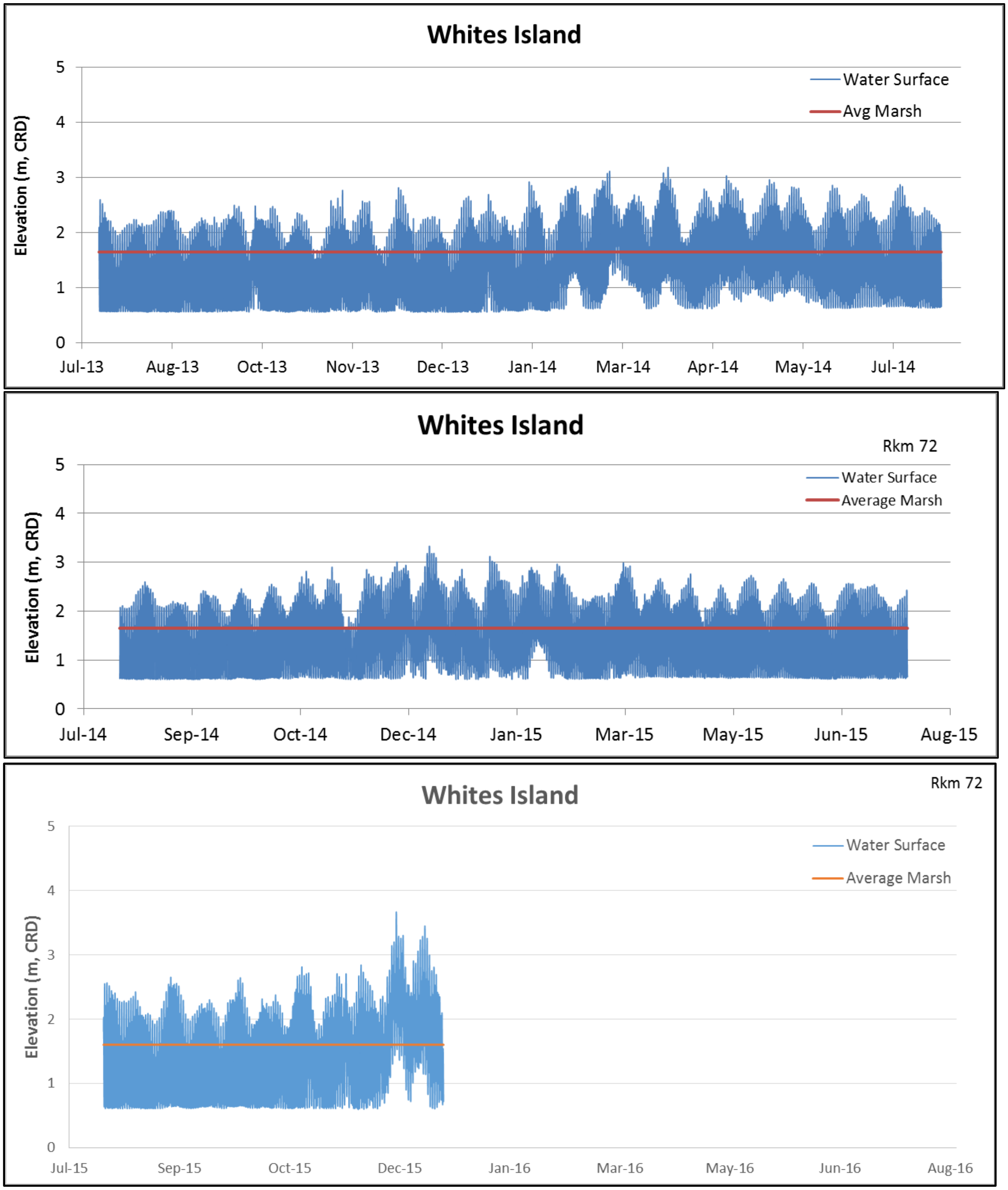
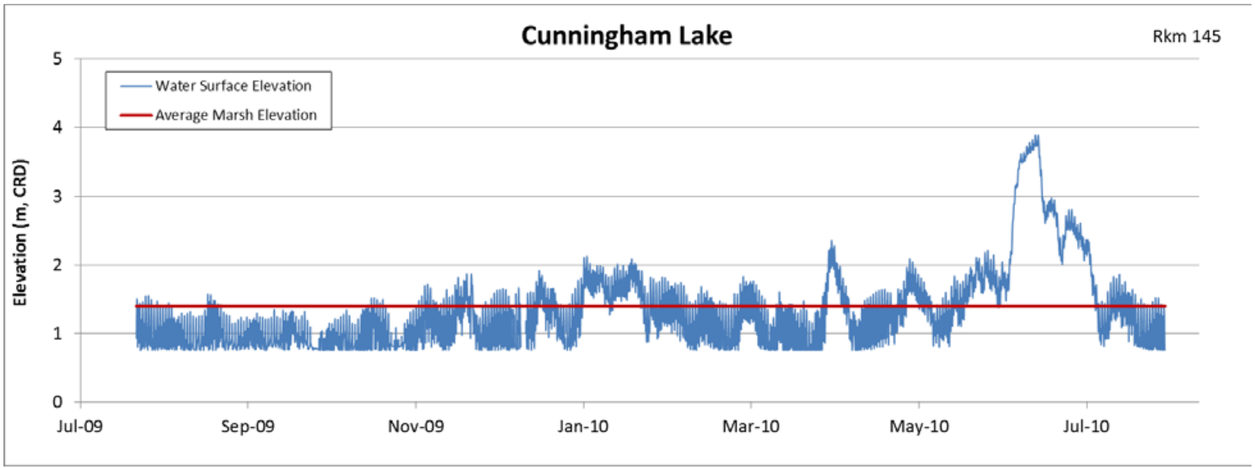
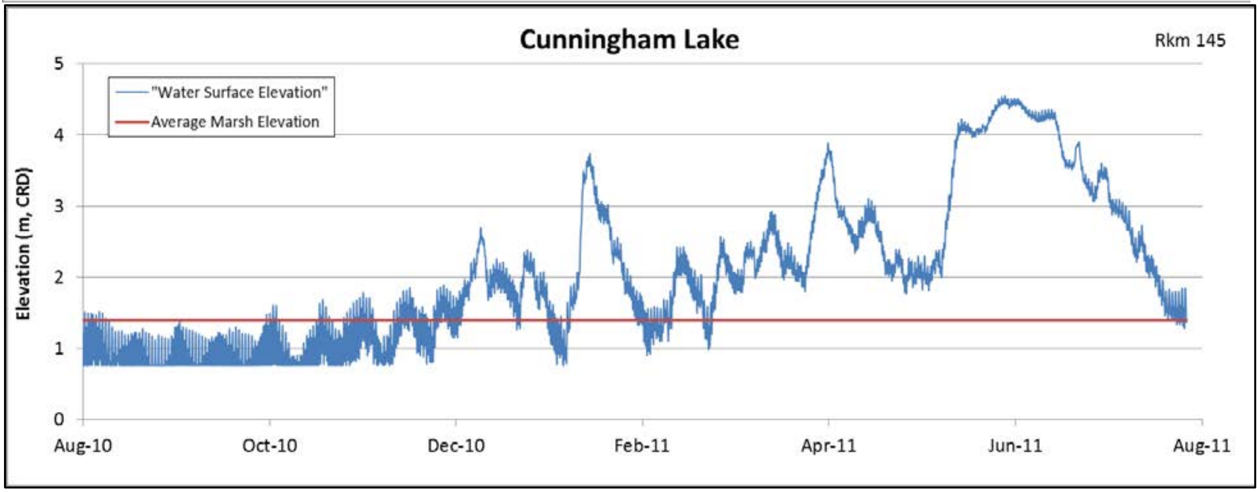
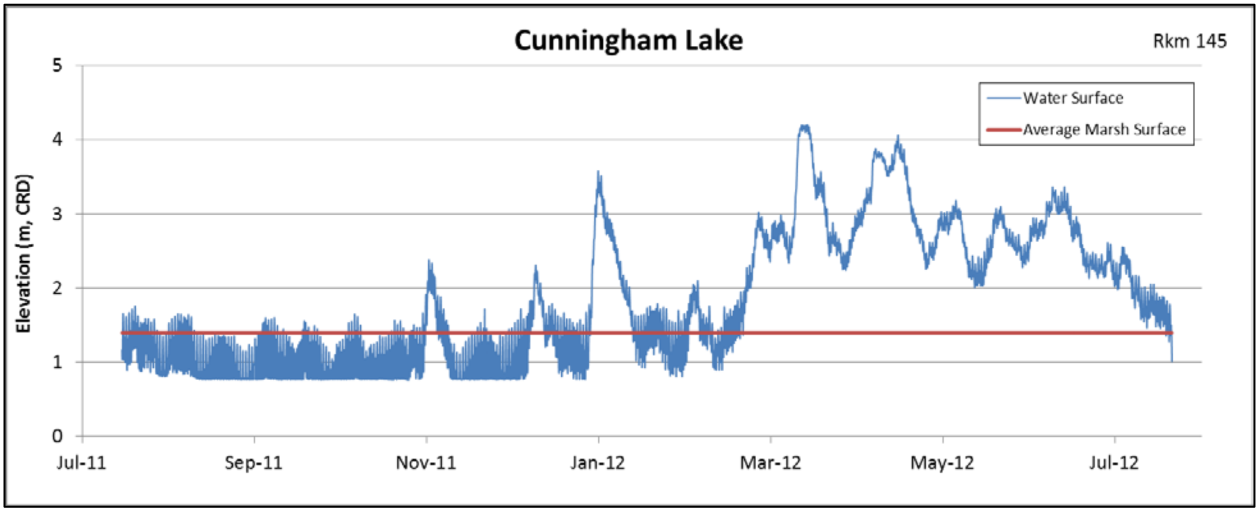


Figure A -4. Water surface elevation data from the Whites Island study site for the years 2009-2012 and 2013-2015. The red line represents the average elevation of the marsh sampling area. No data from 2013 and 2016 due to sensor failure.


Cunningham Lake
Rkm 145
Water Surface Elevation
Average Marsh Elevation
Elevation (m, CRD)
0
1
2
3
4
5
Jul-09
Sep-09
Nov-09
Jan-10
Mar-10
May-10
Jul-10


Cunningham Lake
Rkm 145
"Water Surface Elevation"
Average Marsh Elevation
Elevation (m, CRD)
0
1
2
3
4
5
Aug-10
Oct-10
Dec-10
Feb-11
Apr-11
Jun-11
Aug-11


Cunningham Lake
Rkm 145
Water Surface
Average Marsh Surface
Elevation (m, CRD)
0
1
2
3
4
5
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12

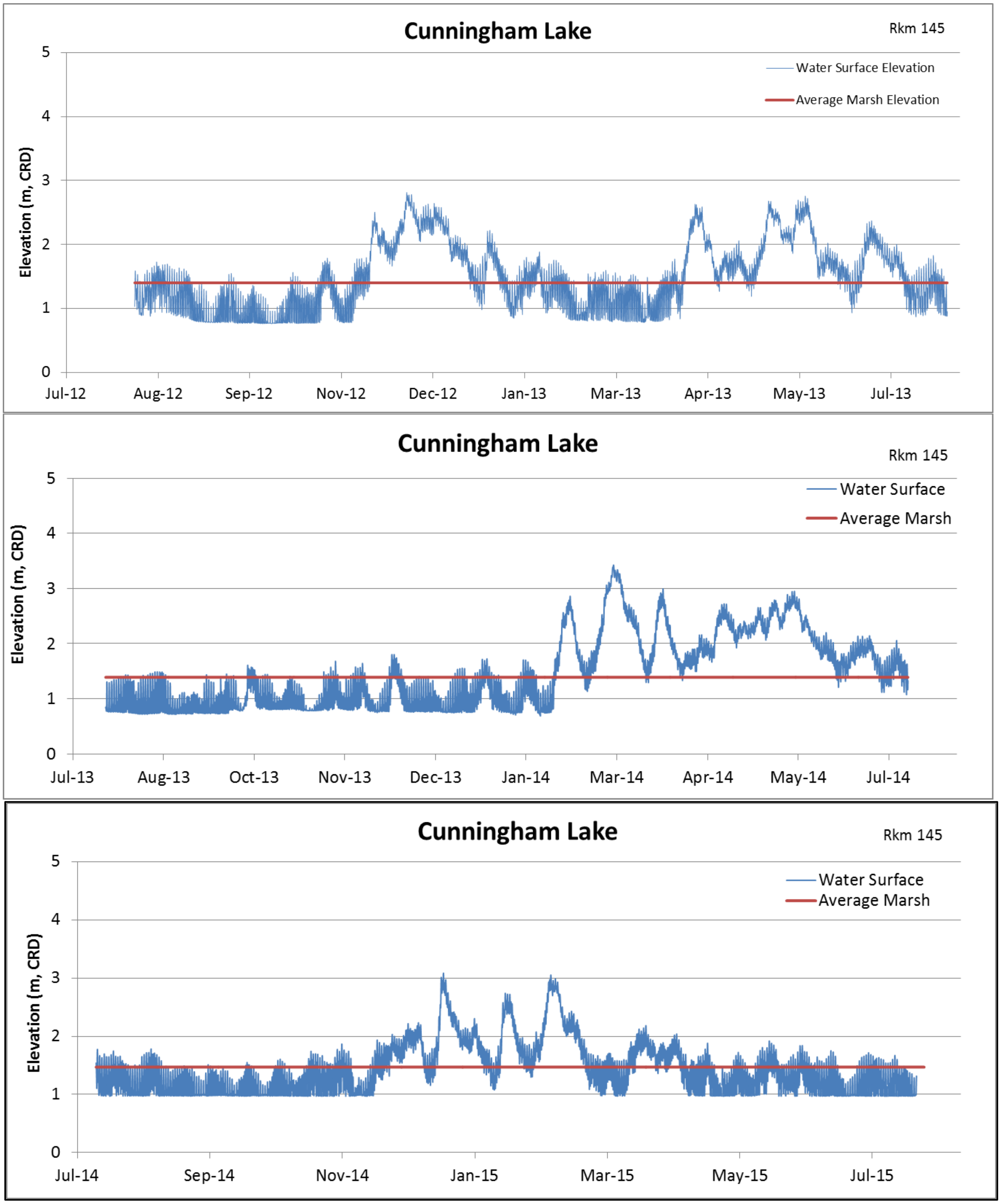
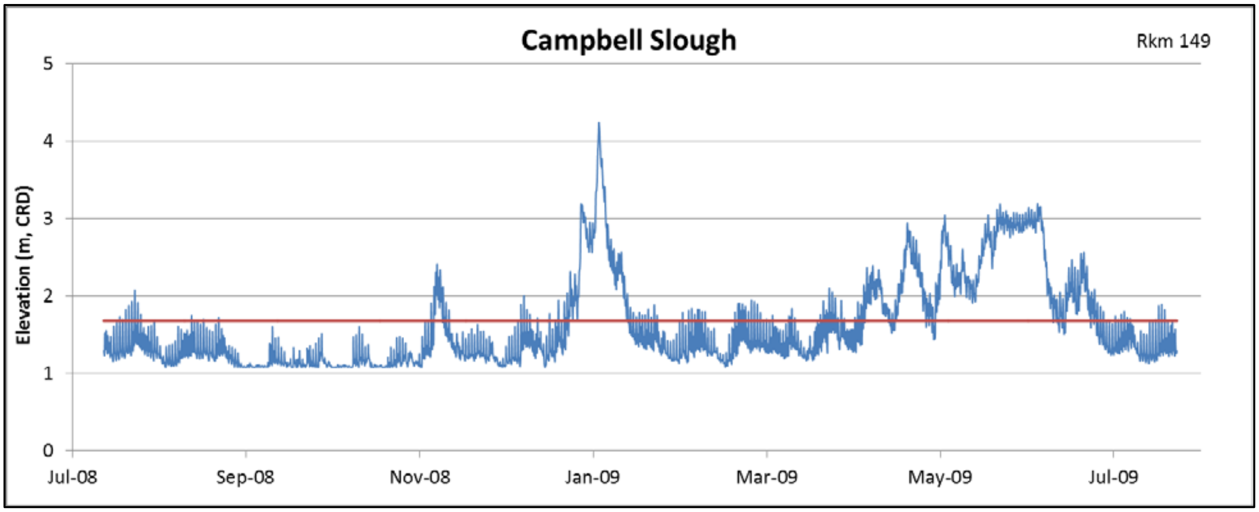
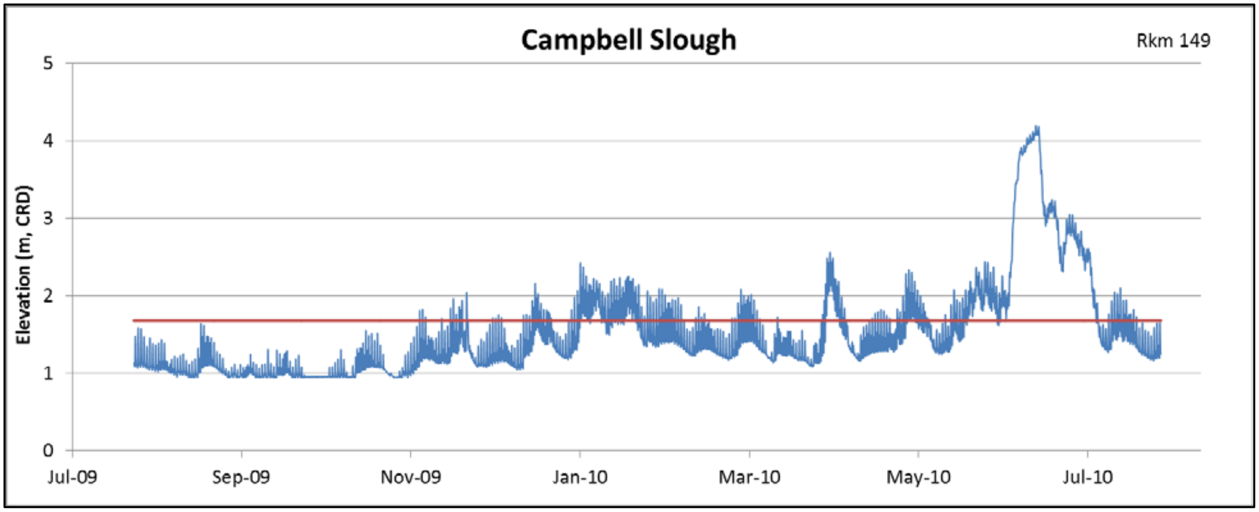
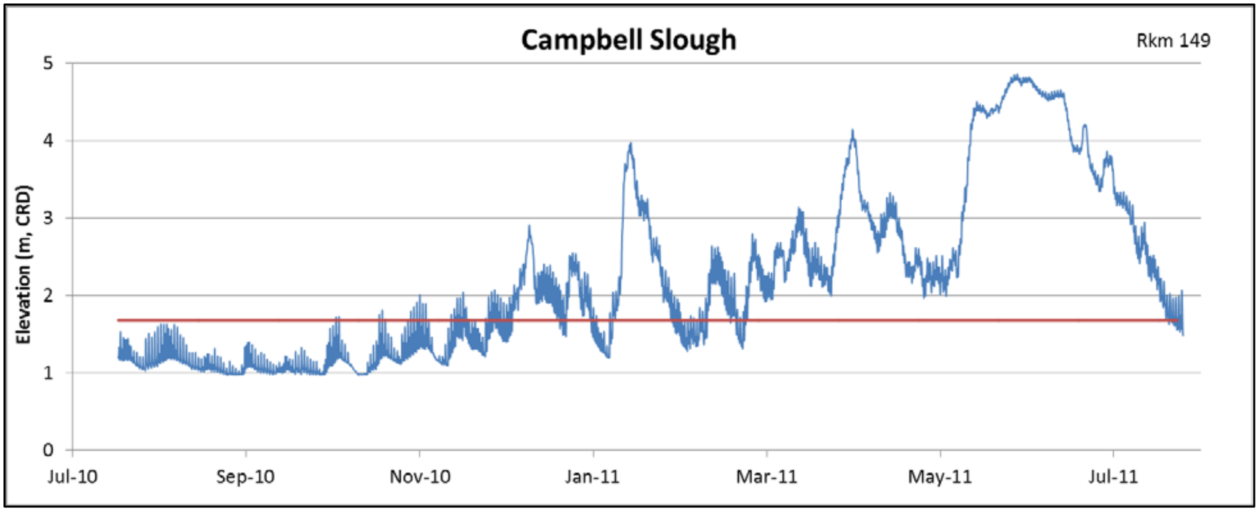
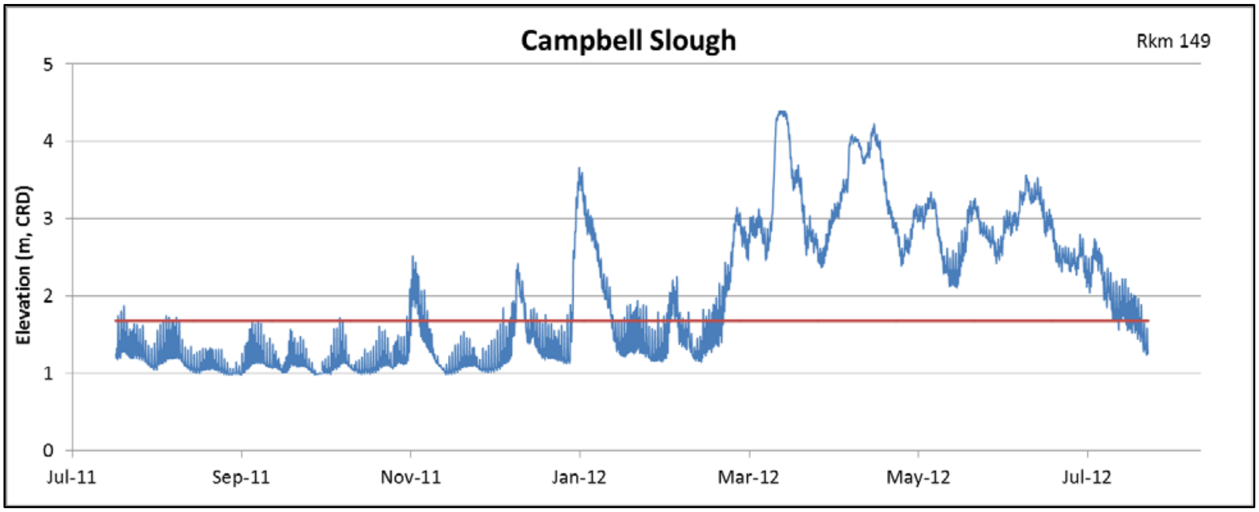


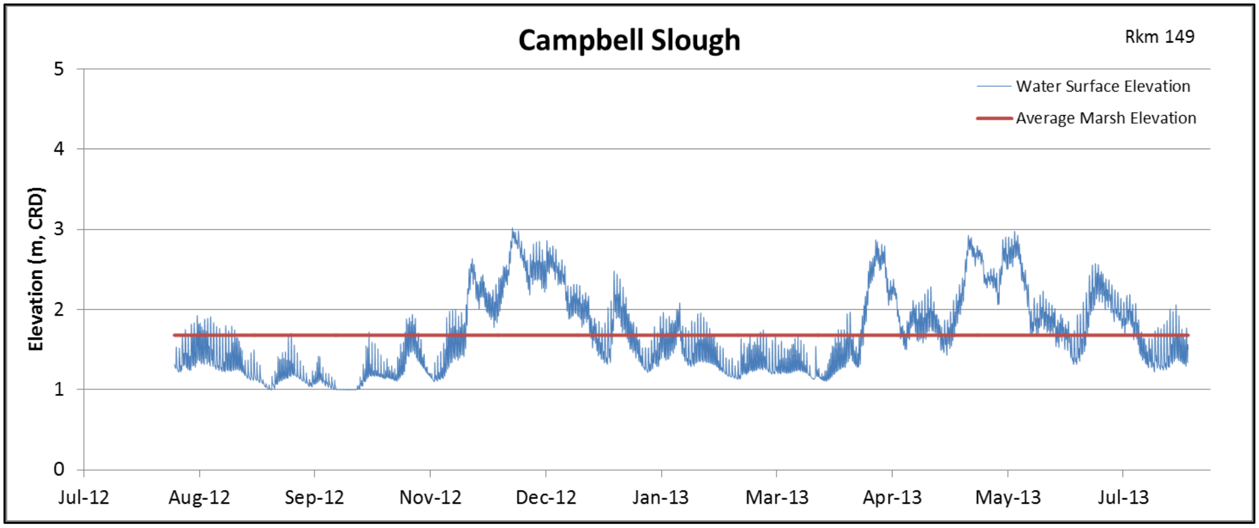
Figure A -5. Water surface elevation data from the Cunningham Lake study site for the years 2009-2015. The red line represents the average elevation of the marsh sampling area. No data was collected from 2015-2016 due to sensor failure

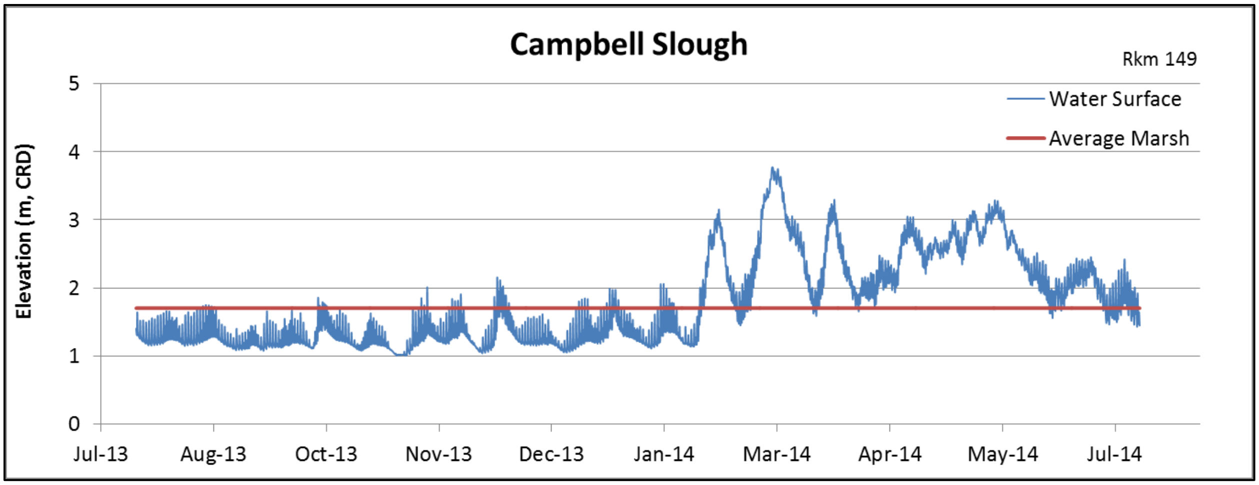

Campbell Slough
Rkm 149
Elevation (m, CRD)
0
1
2
3
4
5
Jul-08
Sep-08
Nov-08
Jan-09
Mar-09
May-09
Jul-09


Campbell Slough
Rkm 149
Elevation (m, CRD)
0
1
2
3
4
5
Jul-09
Sep-09
Nov-09
Jan-10
Mar-10
May-10
Jul-10


Campbell Slough
Rkm 149
Elevation (m, CRD)
0
1
2
3
4
5
Jul-10
Sep-10
Nov-10
Jan-11
Mar-11
May-11
Jul-11


Campbell Slough
Rkm 149
Elevation (m, CRD)
0
1
2
3
4
5
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12


Campbell Slough
Rkm 149
Water Surface Elevation
Average Marsh Elevation
Elevation (m, CRD)
0
1
2
3
4
5
Jul-12
Aug-12
Sep-12
Nov-12
Dec-12
Jan-13
Mar-13
Apr-13
May-13
Jul-13


Campbell Slough
Rkm 149
Water Surface
Average Marsh
Elevation (m, CRD)
0
1
2
3
4
5
Jul-13
Aug-13
Oct-13
Nov-13
Dec-13
Jan-14
Mar-14
Apr-14
May-14
Jul-14

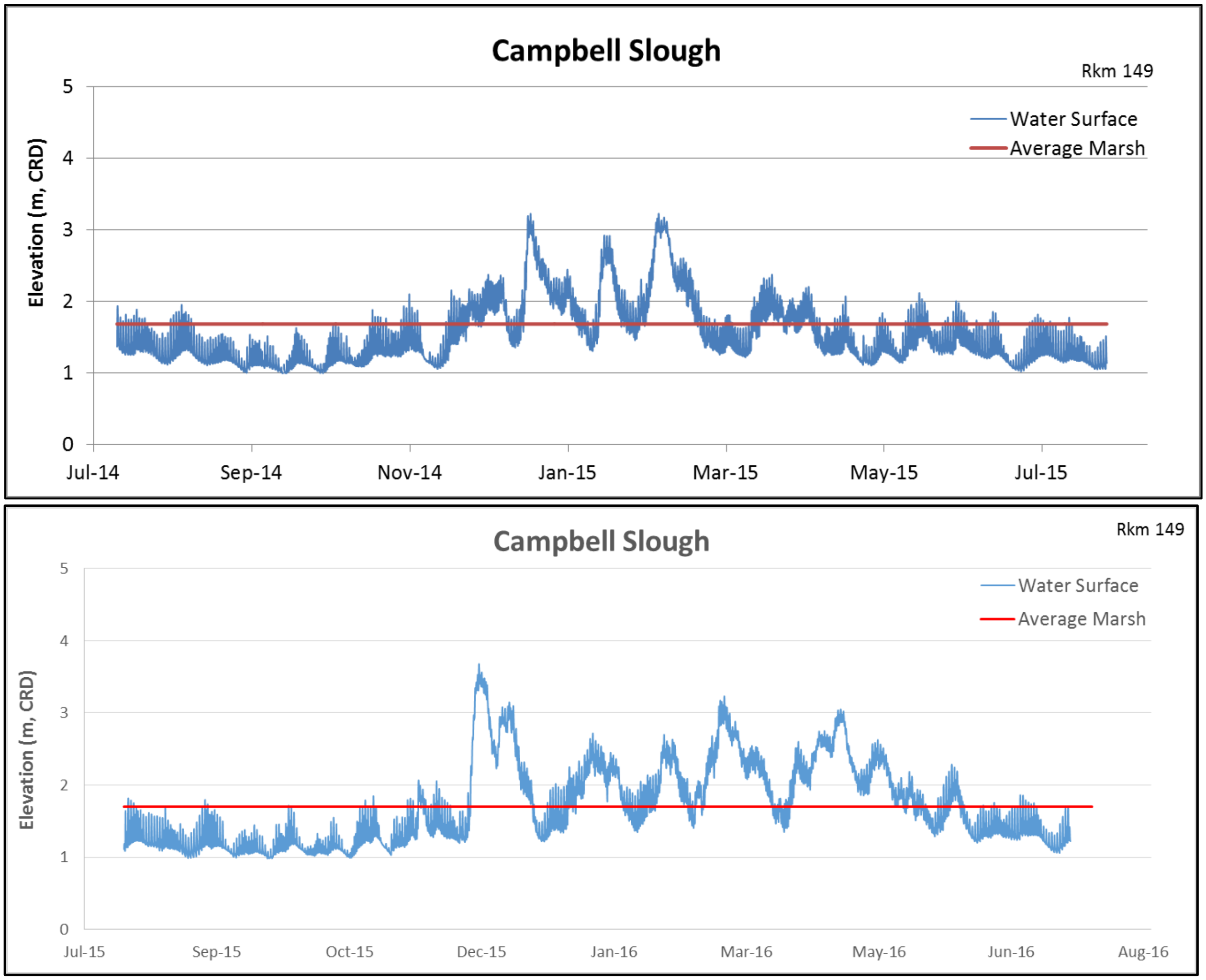
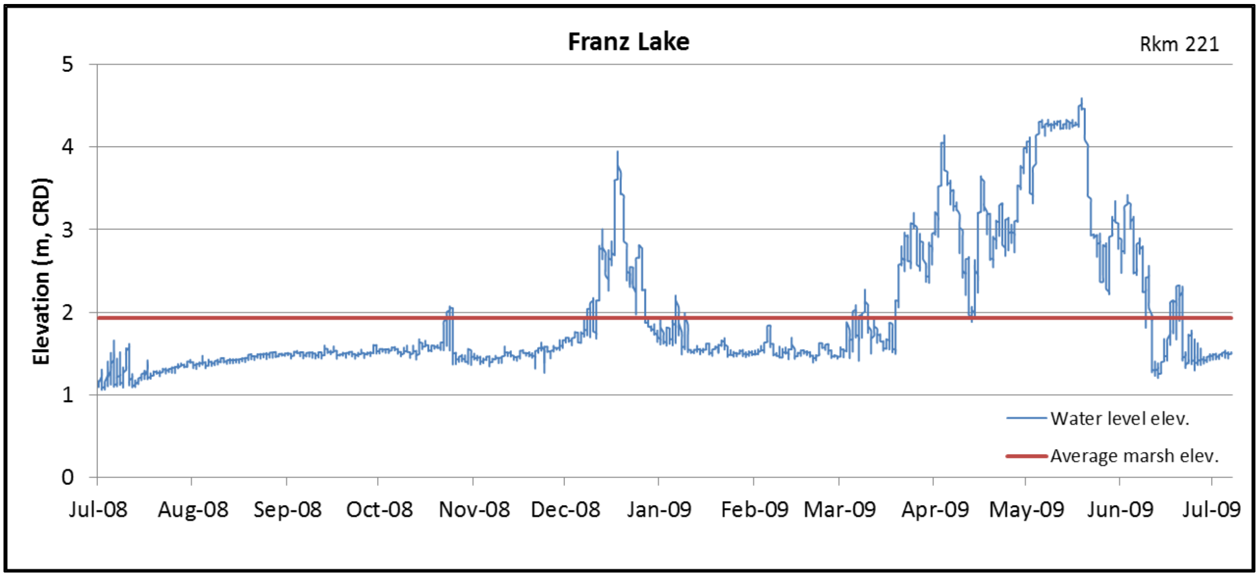
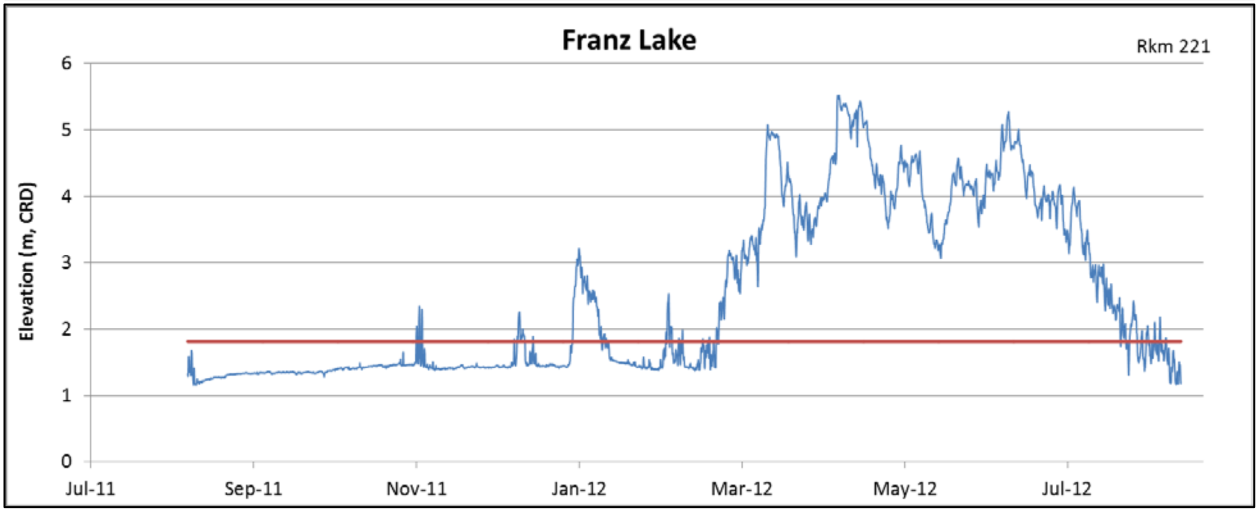
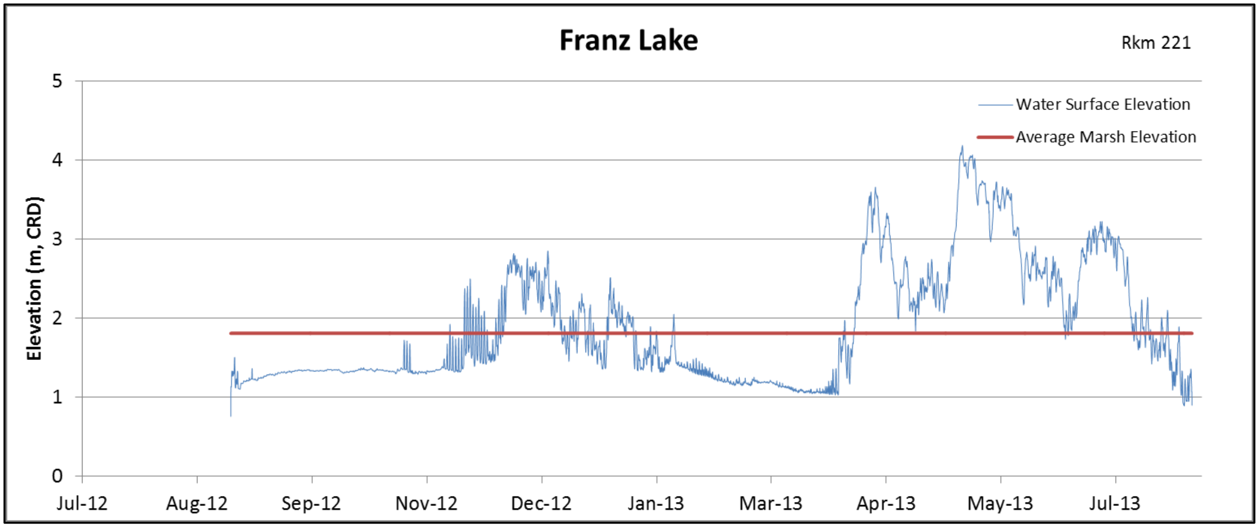


Figure A -6. Water surface elevation data from the Campbell Slough study site for the years 2008-2016. The red line represents the average elevation of the marsh sampling area.


Franz Lake
Rkm 221
Elevation (m, CRD)
5
4
3
2
1
0
Jul-08
Aug-08
Sep-08
Oct-08
Nov-08
Dec-08
Jan-09
Feb-09
Mar-09
Apr-09
May-09
Jun-09
Jul-09
Water level elev.
Average marsh elev.


Franz Lake
Rkm 221
Elevation (m, CRD)
6
5
4
3
2
1
0
Jul-11
Sep-11
Nov-11
Jan-12
Mar-12
May-12
Jul-12


Franz Lake
Rkm 221
Elevation (m, CRD)
5
4
3
2
1
0
Jul-12
Aug-12
Sep-12
Nov-12
Dec-12
Jan-13
Mar-13
Apr-13
May-13
Jul-13
Water Surface Elevation
Average Marsh Elevation

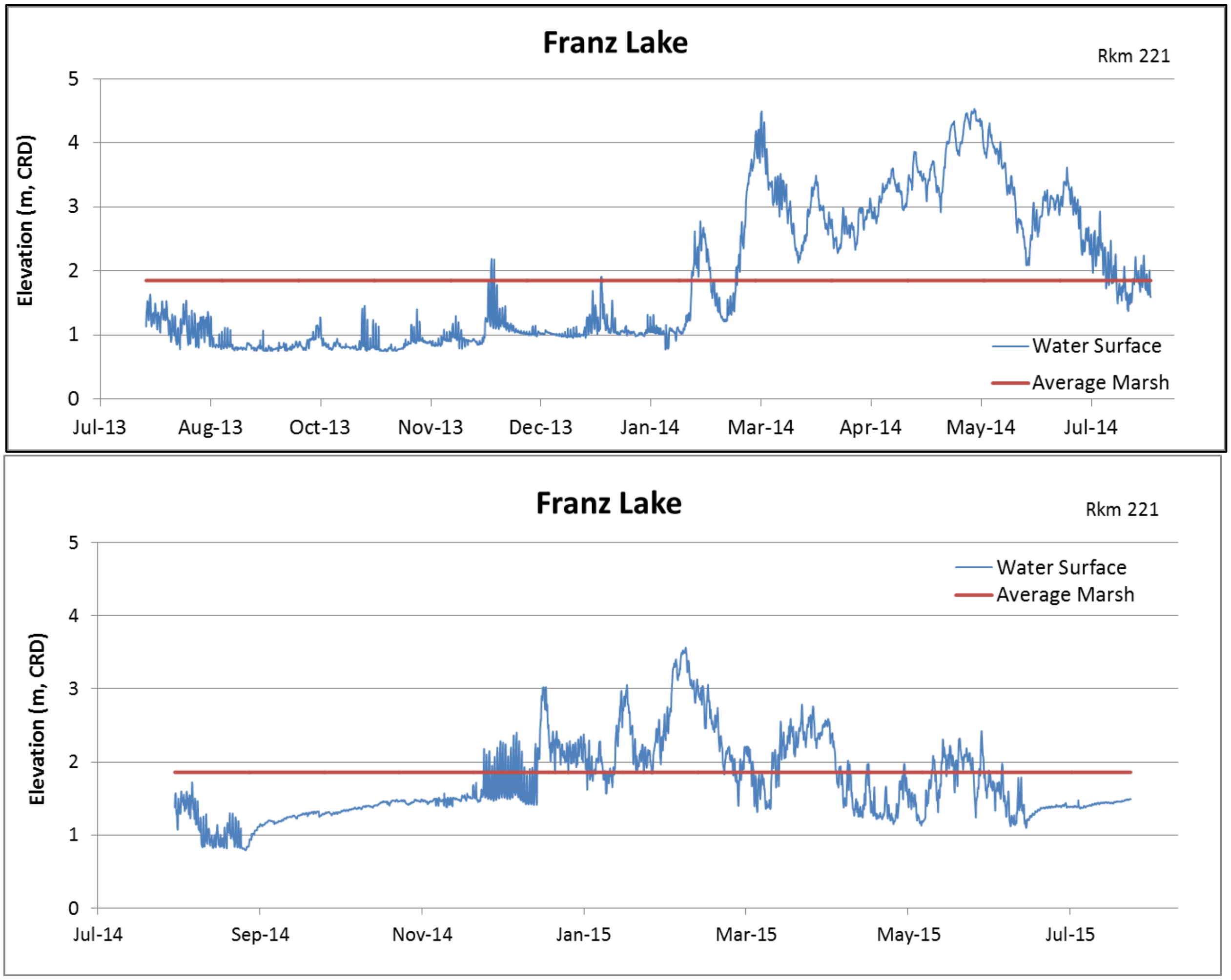


Figure A -7. Water surface elevation data from the Franz Lake study site for the years 2008-2009 and 2011-2015. The red line represents the average elevation of the marsh sampling area. Note the scale difference for the 2011-2012 plot. No data was collected for 2015-2016 due to displacement and loss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

Appendix D. Vegetation Species Cover

Table D.1. Site marsh elevation (in meters, relative to the Columbia River vertical datum CRD) and marsh vegetation species average percent cover from 2016. The three dominant cover classes are bolded in red for each site and non-native species are shaded in yellow. Overhanging tree and shrub species are not included in identification of dominant cover. Species are sorted by their four letter code (1st two letters of genus and 1st two letters of species).

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
					Elevation (m, CRD)							
				Min	1.61	1.93	0.96	1.07	0.77	1.12	1.21	1.21
				Avg	2.00	2.08	1.04	1.78	1.65	1.48	1.68	1.86
				Max	2.38	2.20	1.22	1.93	2.10	1.73	2.69	2.29
					Average Percent Cover							
ACMI	*Achillea millefolium*	common yarrow	FACU	NI	0.0	0.8	0.0	0.0	0.0	0.0	0.0	0.0
AGGI	*Agrostis gigantea*	redtop; black bentgrass	FAC	no	0.0	0.0	0.0	0.2	0.0	0.0	0.9	0.0
AGSP	*Agrostis sp.*	bentgrass	mixed	mixed	**17.1**	0.0	0.0	0.0	0.3	0.0	0.0	0.0
ALRU	*Alnus rubra*	Red alder	FAC	yes	0.0	1.0	0.0	0.0	0.0	0.0	T	0.0
ALTR	*Alisma triviale*	northern water plaintain	OBL	yes	0.0	0.5	1.1	0.7	0.8	0.0	0.0	0.0
AMFR	*Amorpha fruticosa*	indigo bush	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	T	0.3
AREG	*Argentina egedii ssp. Egedii*	Pacific silverweed	OBL	yes	1.9	3.3	0.0	4.3	0.0	0.0	0.0	0.2
BICE	*Bidens cernua*	Nodding beggars-ticks	OBL	yes	0.0	6.3	8.0	1.7	1.7	T	T	0.0
BIFR	*Bidens frondosa*	devil's beggartick	FACW	yes	0.0	0.0	0.0	0.0	0.0	T	0.0	0.3
CAAM	*Castilleja ambigua*	paint-brush owl-clover; johnny-nip	FACW	yes	0.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
CAAM2	*Carex amplifolia*	big leaf sedge	OBL	yes	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
CAAP	*Carex aperta*	Columbia sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.1
CAHE	*Callitriche heterophylla*	Water starwort; Twoheaded water starwort	OBL	yes	0.0	0.2	0.0	0.2	1.0	T	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
CAHE2	*Callitriche hermaphroditica*	northern water-starwort	OBL	yes	0.0	0.2	0.5	T	T	0.0	0.0	0.0
CALY	*Carex lyngbyei*	Lyngby sedge	OBL	yes	**40.8**	**43.5**	4.6	**58.3**	5.4	0.0	0.0	0.0
CAOL	*Cardamine oligosperma*	little western bittercress	FAC	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.5
CAPA	*Caltha palustris*	Yellow marsh marigold	OBL	yes	0.0	**6.4**	0.0	4.0	0.0	0.0	0.0	0.0
CARE	*Carex retrorsa*	knotsheath sedge	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
CASE	*Calystegia sepium*	Hedge false bindweed	FAC	no	0.0	1.5	0.0	0.0	2.4	0.0	0.0	0.0
CASP	*Carex sp.*	Carex	mixed	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.2	3.0
CAST2	*Callitriche stagnalis*	pond water-starwort	OBL	no	0.0	0.0	1.0	0.0	0.0	0.0	0.2	0.0
CEDE	*Ceratophyllum demersum*	Coontail	OBL	yes	0.0	0.0	T	0.0	0.0	0.0	0.0	0.0
CIAR	*Cirsium arvense var. horridum*	Canada thistle	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	T	0.0
DECE	*Deschampsia cespitosa*	Tufted hairgrass	FACW	yes	4.9	0.0	0.0	0.2	0.0	0.0	0.0	0.0
ELAC	*Eleocharis acicularis*	Needle spikerush	OBL	yes	0.0	0.0	0.0	0.1	0.0	0.7	0.9	0.0
ELCA	*Elodea canadensis*	Canada waterweed	OBL	yes	0.0	0.8	**24.3**	0.6	**8.0**	0.9	0.4	0.0
ELNU	*Elodea nuttallii*	Nuttall's waterweed, western waterweed	OBL	yes	0.0	0.0	0.8	0.0	0.0	0.0	0.0	0.0
ELPA	*Eleocharis palustris*	Common spikerush	OBL	yes	0.0	0.2	2.6	5.6	3.2	**9.9**	**33.1**	**5.7**
ELPAR	*Eleocharis parvula*	Dwarf spikerush	OBL	yes	1.0	0.0	0.4	0.0	0.1	0.0	0.0	0.0
EPCI	*Epilobium ciliatum*	Willow herb	FACW	yes	0.0	2.0	0.0	1.9	0.4	0.0	0.0	0.0
EQFL	*Equisetum fluviatile*	Water horsetail	OBL	yes	0.0	1.4	0.0	3.1	3.6	0.0	0.0	0.0
EQPA	*Equisetum palustre*	marsh horsetail	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.9	0.0	T
EQSP	*Equisetum spp.*	Horsetail	mixed	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	0.6
FOAN	*Fontinalis antipyretica*	common water moss	OBL	yes	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
FRLA*	*Fraxinus latifolia*	Oregon ash	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	3.0
GATR	*Galium trifidum L. spp. columbianum*	Pacific bedstraw	FACW	yes	0.0	0.0	0.0	0.0	0.1	0.0	0.0	T

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
GATR2	*Galium triflorum*	fragrant bedstraw	FACU	yes	0.0	T	0.0	0.0	0.0	0.0	0.0	0.0
GATR3	*Galium trifidum*	small bedstraw	FACW	yes	0.0	0.4	0.0	2.4	1.0	0.0	0.0	0.0
GLGR	*Glyceria grandis*	American mannagrass	OBL	yes	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0
GLMA	*Glaux maritima*	sea milkwort	OBL	yes	0.6	0.0	0.0	0.0	0.0	0.0	0.0	0.0
GLST	*Glyceria striata*	Fowl mannagrass	OBL	yes	0.0	T	0.0	0.0	0.0	0.0	0.0	0.0
GNUL	*Gnaphalium uliginosum*	Marsh cudweed	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	T	0.4
GREB	*Gratiola ebracteata*	bractless hedgehyssop	OBL	yes	0.0	0.0	0.0	T	0.0	0.0	0.7	0.0
GRNE	*Gratiola neglecta*	American Hedge-hyssop	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	1.3
HEAU	*Helenium autumnale*	common sneezeweed	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.9	1.2
HOLA	*Holcus lanatus*	Common velvetgrass	FAC	no	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0
HYSC	*Hypericum scouleri*	Western St. Johns wort	FACW	yes	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0
IMSP	*Impatiens capensis,Impatiens noli-tangere*	western touch-me-not, common touch-me-not, jewelweed	FACW	yes	0.0	0.1	0.0	4.4	1.3	0.0	0.0	0.0
IRPS	*Iris pseudacorus*	Yellow iris	OBL	no	0.0	0.0	0.0	1.5	2.5	T	T	0.0
ISCE	*Isolepis cernua*	low bulrush	OBL	yes	**5.0**	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ISSP	*Isoetes spp.*	quillwort	OBL	yes	0.0	0.0	1.4	0.0	0.0	0.0	0.0	0.0
JUAC	*Juncus acuminatus*	Tapertip rush	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	0.0
JUBU	*Juncus bufonius*	Toad rush	FACW	yes	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
JUOX	*Juncus oxymeris*	Pointed rush	FACW	yes	0.0	0.7	0.5	1.8	0.7	0.0	0.4	0.0
LAPA	*Lathyrus palustris*	Marsh peavine	OBL	yes	0.0	0.0	0.0	0.4	0.0	0.0	0.0	0.0
LEOR	*Leersia oryzoides*	Rice cutgrass	OBL	yes	0.0	0.8	0.0	0.9	2.2	5.4	0.9	3.2
LIAQ	*Limosella aquatica*	Water mudwort	OBL	yes	0.0	0.0	1.4	T	0.2	0.0	T	0.0
LIDU	*Lindernia dubia*	yellow seed false pimpernel	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.2
LIOC	*Lilaeopsis occidentalis*	Western lilaeopsis	OBL	yes	4.0	T	**10.7**	0.0	0.0	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
LISC	*Lilaea scilloides*	Flowering quillwort	OBL	yes	0.0	0.0	0.2	T	0.0	0.0	0.0	0.0
LOCO	*Lotus corniculatus*	Birdsfoot trefoil	FAC	no	0.0	3.3	0.0	0.6	1.5	0.0	1.3	0.0
LUPA	*Ludwigia palustris*	False loosestrife	OBL	yes	0.0	0.0	0.0	0.0	0.0	2.6	1.9	T
LYAM	*Lysichiton americanus*	Skunk cabbage	OBL	yes	0.0	0.5	0.0	2.9	0.8	0.0	0.0	0.0
LYNU	*Lysimachia nummularia L.*	Moneywort, Creeping Jenny	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	2.0	0.0
LYSA	*Lythrum salicaria*	Purple loosestrife	OBL	no	0.0	1.6	0.0	0.5	0.1	0.0	0.0	0.0
LYUN	*Lycopus uniflorus*	Northern bugleweed	OBL	yes	0.0	T	0.0	0.7	0.0	0.0	0.0	0.0
MEAR	*Mentha arvensis*	wild mint	FACW	yes	0.0	0.4	0.0	0.0	0.9	0.0	0.0	1.0
MESP3	*Mentha spicata*	spearmint	FACW	no	0.0	0.0	0.0	0.0	0.0	0.0	0.4	0.0
MIGU	*Mimulus guttatus*	Yellow monkeyflower	OBL	yes	0.0	0.7	T	1.7	0.2	0.0	0.0	T
MUKE	*Murdannia keisak*	wart-removing herb	OBL	no	0.0	1.3	0.0	0.4	0.0	0.0	0.0	0.0
MYLA	*Myosotis laxa*	Small forget-me-not	OBL	yes	0.0	0.0	0.0	0.0	0.5	0.0	0.0	0.0
MYSC	*Myosotis scorpioides*	Common forget-me-not	FACW	no	0.0	0.0	0.0	0.0	0.6	0.0	0.0	0.0
MYSP	*Myosotis laxa, M. scorpioides*	Small forget-me-not, Common forget-me-not	mixed	mixed	0.0	5.1	0.0	5.4	0.0	T	0.0	0.0
MYSP2	*Myriophyllum spp.*	Milfoil	OBL	mixed	0.0	0.0	2.1	0.0	0.4	0.0	0.0	0.0
OESA	*Oenanthe sarmentosa*	Water parsley	OBL	yes	0.0	2.4	0.0	3.6	1.8	0.0	0.0	0.0
PHAR	*Phalaris arundinacea*	Reed canary grass	FACW	no	0.0	**22.1**	0.0	**9.5**	**49.7**	**47.4**	**36.0**	**17.0**
PLDI	*Platanthera dilatata*	white bog orchid	FACW	yes	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
PLMA	*Plantago major*	common plantain	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.2	T
POAM	*Polygonum amphibium*	water ladysthumb, water smartweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	**44.9**
POAN2	*Poa annua*	annual bluegrass	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
POCR	*Potamogeton crispus*	Curly leaf pondweed	OBL	no	0.0	0.0	0.0	0.0	T	0.0	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
POHY	*Polygonum hydropiper, P. hydropiperoides*	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	0.0	0.9	0.0	0.5	0.6	0.2	T	0.0
PONA	*Potamogeton natans*	Floating-leaved pondweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	T	0.4	0.0
POPE	*Polygonum persicaria*	Spotted ladysthumb	FACW	no	0.0	0.5	**9.9**	1.5	0.6	0.7	0.2	T
POPU	*Potamogeton pusillus*	Small pondweed	OBL	yes	0.0	0.0	0.1	0.0	0.0	0.0	0.0	0.0
PORI	*Potamogeton richardsonii*	Richardson's pondweed	OBL	yes	0.0	0.0	1.8	0.0	0.0	0.0	0.0	0.0
POSP	*Polygonum sp.*	Knotweed, Smartweed	mixed	mixed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	T
RASC	*Ranunculus sceleratus*	Celery-leaved buttercup	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	0.0
RUAQ	*Rumex aquaticus*	Western dock	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	T	0.0
RUCR	*Rumex crispus*	Curly dock	FAC	no	0.0	0.1	0.0	0.0	0.0	0.0	0.0	0.0
RUMA	*Rumex maritimus*	Golden dock, seaside dock	FACW	yes	0.0	0.0	0.0	0.0	0.1	0.0	0.0	0.0
SALA	*Sagittaria latifolia*	Wapato	OBL	yes	0.0	2.8	0.2	**7.4**	**9.9**	**28.5**	**18.0**	3.2
SALU	*Salix lucida*	Pacific willow	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	2.8
SALU*	*Salix lucida*	Pacific willow	FACW	yes	0.0	0.0	0.0	0.0	0.0	8.1	0.3	23.8
SASI*	*Salix sitchensis*	Sitka willow	FACW	yes	0.0	0.0	0.0	0.1	0.5	0.0	0.0	0.0
SCAM	*Schoenoplectus americanus*	American bulrush, threesquare bulrush	OBL	yes	2.5	0.0	0.0	0.0	T	0.0	0.0	0.0
SCMA	*Schoenoplectus maritimus*	Seacoast bulrush	OBL	yes	0.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
SCTA	*Schoenoplectus tabernaemontani*	Softstem bulrush, tule	OBL	Yes	0.0	T	1.9	0.0	0.1	0.0	T	T
SISU	*Sium suave*	Hemlock waterparsnip	OBL	yes	0.0	0.8	0.0	1.0	0.1	0.0	0.0	0.0
SODU	*Solanum dulcamara*	Bittersweet nightshade	FAC	no	0.0	0.0	0.0	0.0	0.1	T	0.0	0.0
SPAN	*Sparganium angustifolium*	Narrowleaf burreed	OBL	yes	0.0	0.0	0.0	0.0	T	T	0.0	0.0
SPEU	*Sparganium eurycarpum*	giant burreed	OBL	yes	0.0	0.0	2.4	0.0	0.0	1.4	0.0	0.0

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret-High	Secret-Low	Welch	Whites	Cunningham	Campbell	Franz
STME2	*Stachys mexicana*	Mexican hedgenettle	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.2	0.0	T
SYEA	*Symphyotrichum eatonii*	Eaton's aster	FAC	yes	0.0	0.0	0.0	1.4	0.0	0.0	0.0	0.0
SYSU	*Symphyotrichum subspicatum*	Douglas aster	FACW	yes	0.5	3.2	0.8	0.5	0.1	0.0	0.0	0.0
TRMA	*Triglochin maritima*	Seaside arrowgrass	OBL	yes	1.7	0.0	0.0	0.0	0.0	0.0	0.0	0.0
TRWO	*Trifolium wormskioldii*	Springbank clover	FACW	yes	0.0	0.0	0.0	T	0.0	0.0	0.0	0.0
TYAN	*Typha angustifolia*	Narrowleaf cattail	OBL	no	2.3	0.0	0.0	T	2.1	0.0	0.0	0.0
VEAM	*Veronica americana*	American speedwell	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.3	T	0.0
ZAPA	*Zannichellia palustris*	horned pondweed	OBL	yes	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Other Cover												
dAGGI	*dead Agrostis gigantea*	dead redtop; black bentgrass	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
ALGAE		algae			4.5	T	24.8	0.0	0.1	0.0	0.0	0.0
BG		bare ground			22.7	13.5	25.8	7.5	7.7	11.4	12.3	16.8
dCAAM	*dead Castilleja ambigua*	dead paint-brush owl-clover; johnny-nip	FACW	yes	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
dCALY	*dead Carex lyngbyei*	dead Lyngby sedge	OBL	yes	0.0	0.0	0.0	0.9	0.0	0.0	0.0	0.0
Detritus		detritus			0.0	0.5	0.0	1.7	0.6	6.7	11.1	2.0
Drift wrack		drift wrack			1.0	0.3	0.0	0.0	0.3	0.0	0.0	0.2
dELPA	*dead Eleocharis palustris*	dead common spikerush	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.3	T
FGA		filamentous green algae			0.0	0.0	0.0	0.0	0.0	0.8	0.0	0.0
FUDI	*Fucus distichus*	Rockweed	OBL	yes	3.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Litter		litter			0.0	0.0	0.3	0.0	0.0	2.0	T	3.4
dLOCO	*dead Lotus corniculatus*	dead birdsfoot trefoil	FAC	no	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
LWD		large woody debris			0.0	0.0	0.0	0.0	1.7	0.2	0.0	1.6

Code	**Scientific Name**	**Common Name**	**Wetland Status**	**Native**	**Ilwaco**	**Secret-High**	**Secret-Low**	**Welch**	**Whites**	**Cunningham**	**Campbell**	**Franz**
dLYAM	*dead Lysichiton americanus*	dead Skunk cabbage	OBL	yes	0.0	0.0	0.0	0.1	0.0	0.0	0.0	0.0
Moss		moss			T	1.4	0.0	0.0	0.0	0.0	0.0	0.0
Open water		open water			5.3	31.0	35.8	3.1	2.4	7.8	24.1	13.4
dPHAR	*dead Phalaris arundinacea*	dead reed canary grass	FACW	no	0.0	0.0	0.0	0.0	1.2	5.1	0.7	T
dPOAM	*dead Polygonum amphibium*	dead water ladysthumb, water smartweed	OBL	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.9
dSALA	*dead Sagittaria latifolia*	dead Wapato	OBL	yes	0.0	0.0	0.0	0.0	0.0	T	1.1	T
dSALU	*dead Salix lucida*	dead Pacific willow	FACW	yes	0.0	0.0	0.0	0.0	0.0	0.0	0.2	0.0
SMG		small mixed grass			0.0	0.0	T	0.0	0.0	0.0	0.0	0.0
SMH		small mixed herbs			T	T	0.0	0.0	T	0.0	0.2	1.9
Standing dead		standing dead			0.1	0.3	0.0	1.6	0.5	0.0	0.0	0.0
dTYAN	*dead Typha angustifolia*	dead Narrowleaf cattail	OBL	no	0.1	0.0	0.0	0.0	0.0	0.0	0.0	0.0
ULVA	*Ulva lactuca*	Sea lettuce	OBL	yes	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

T = Trace

* = Overhead vegetation

Table D.2. Site channel elevation (in meters, relative to the Columbia River vertical datum CRD) and channel average percent cover from 2016. The three dominant cover classes are bolded in red for each site and non-native species are shaded in yellow. Species are sorted by their four letter code (1st two letters of genus and 1st two letters of species). Channel data was included in the marsh data for the Cunningham Lake site and no channel data was collected at Welch Island site in 2016.

Code	Scientific Name	Common Name	Wetland Status	Native	Ilwaco	Secret	Whites	Campbell	Franz
					Elevation (m, CRD)				
				Min	0.90	0.13	0.14	-0.07	0.68
				Avg	1.01	0.36	0.35	0.69	0.93
				Max	1.18	0.69	0.61	0.91	1.19
ALTR	*Alisma triviale*	northern water plaintain	OBL	yes	0.0	0.1	0.1	0.0	0.0
BICE	*Bidens cernua*	Nodding beggars-ticks	OBL	yes	0.0	0.0	1.0	0.0	0.0
CAHE	*Callitriche heterophylla*	Water starwort; Twoheaded water starwort	OBL	yes	0.0	0.0	0.9	0.0	0.0
CAHE2	*Callitriche hermaphroditica*	northern water-starwort	OBL	yes	0.0	0.1	0.1	0.0	0.0
CAST	*Carex stipata*	Sawbeak sedge	OBL	yes	0.0	0.0	0.1	0.0	0.0
CAST2	*Callitriche stagnalis*	pond water-starwort	OBL	no	0.0	0.8	0.0	0.0	0.0
CEDE	*Ceratophyllum demersum*	Coontail	OBL	yes	0.0	0.0	0.0	**1.7**	0.0
ELCA	*Elodea canadensis*	Canada waterweed	OBL	yes	0.0	**3.9**	**18.3**	**15.8**	0.0
ELNU	*Elodea nuttallii*	Nuttall's waterweed, western waterweed	OBL	yes	0.0	**10.4**	0.0	0.0	0.0
ELPA	*Eleocharis palustris*	Common spikerush	OBL	yes	0.0	0.0	4.1	0.0	0.0
LIAQ	*Limosella aquatica*	Water mudwort	OBL	yes	0.0	0.0	0.1	0.0	0.0
MYHI	*Myriophyllum hippuroides*	western milfoil	OBL	yes	0.0	0.0	0.0	**9.2**	0.0
MYSP2	*Myriophyllum spp.*	Milfoil	OBL	mixed	0.0	0.3	1.1	0.0	0.0
POCR	*Potamogeton crispus*	Curly leaf pondweed	OBL	no	0.0	5.0	0.1	4.2	0.0
POHY	*Polygonum hydropiper, P. hydropiperoides*	Waterpepper, mild waterpepper, swamp smartweed	OBL	mixed	0.0	0.0	0.1	0.0	0.0

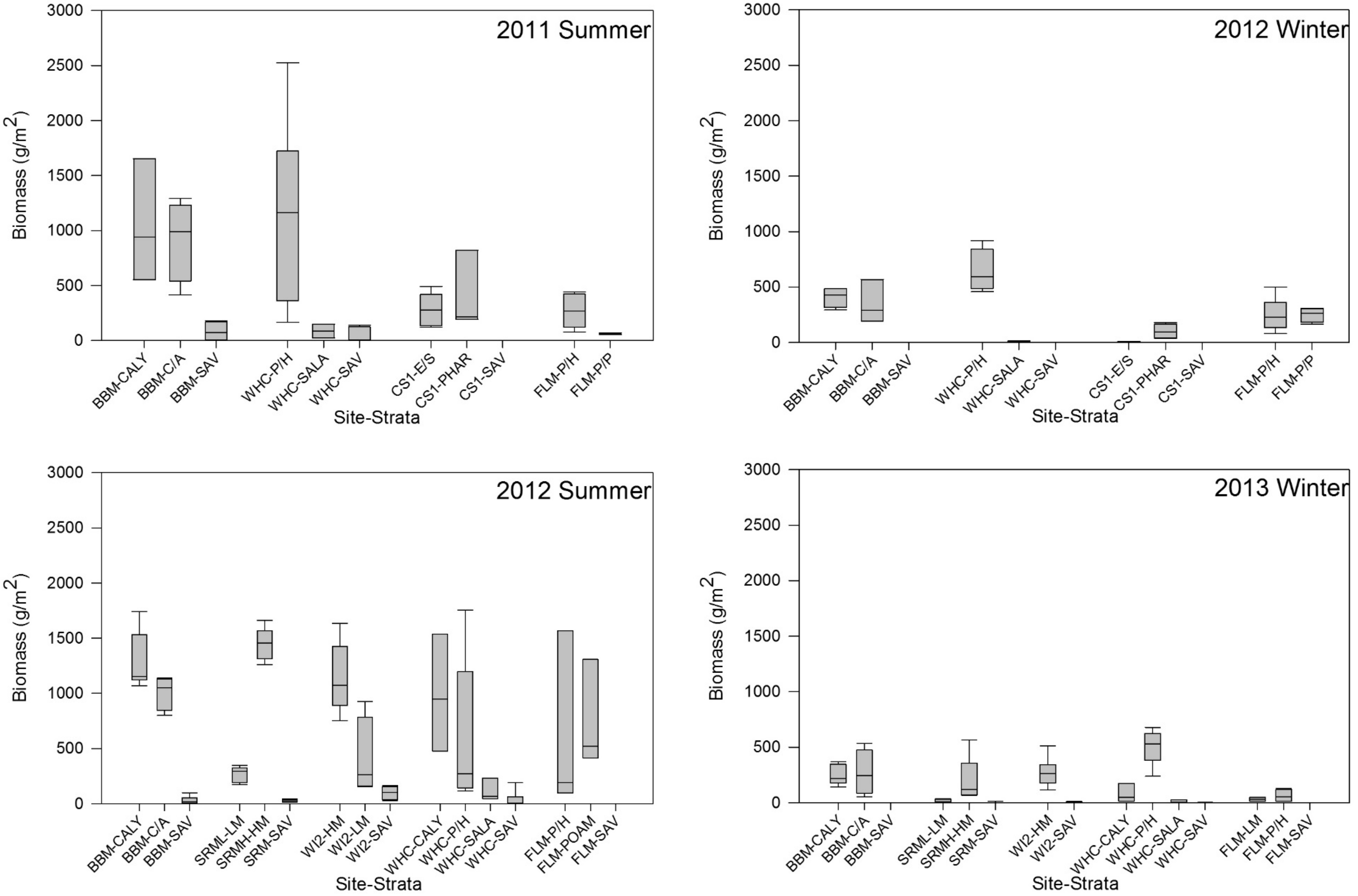
Code	**Scientific Name**	**Common Name**	**Wetland Status**	**Native**	**Ilwaco**	**Secret**	**Whites**	**Campbell**	**Franz**
POPU	*Potamogeton pusillus*	Small pondweed	OBL	yes	0.0	0.9	0.0	0.0	0.0
PORI	*Potamogeton richardsonii*	Richardson's pondweed	OBL	yes	0.0	**4.5**	**9.5**	0.0	0.0
POSP	*Polygonum sp.*	Knotweed, Smartweed	mixed	mixed	0.0	0.0	0.6	0.8	0.0
POZO	*Potamogeton zosteriformis*	Eelgrass pondweed	OBL	yes	0.0	0.0	**6.5**	0.0	0.0
SALA	*Sagittaria latifolia*	Wapato	OBL	yes	0.0	0.0	2.4	0.0	**3.8**
SCAM	*Schoenoplectus americanus*	American bulrush, threesquare bulrush	OBL	yes	0.0	0.0	0.1	0.0	0.0
ZOJA	*Zostera japonica*	Japanese eelgrass	OBL	no	**2.0**	0.0	0.0	0.0	0.0
Other Cover									
Algae		algae			0.6	0.0	0.0	0.0	0.0
BG		bare ground			98.0	74.4	65.1	0.0	15.0
Detritus		detritus			0.0	5.0	0.4	0.0	0.0
DW		drift wrack			0.0	0.0	0.0	0.0	0.3
FGA		filamentous green algae			0.0	21.9	2.5	0.0	0.0
Litter		litter			0.2	0.6	0.7	0.0	0.0
LWD		large woody debris			0.0	0.6	1.2	1.7	0.0
OW		open water			64.0	57.6	43.8	100.0	0.0
SH		shell hash			0.0	0.0	0.8	0.0	0.0
ULLA	*Ulva lactuca*	Sea lettuce	OBL	yes	0.2	0.0	0.0	0.0	0.0

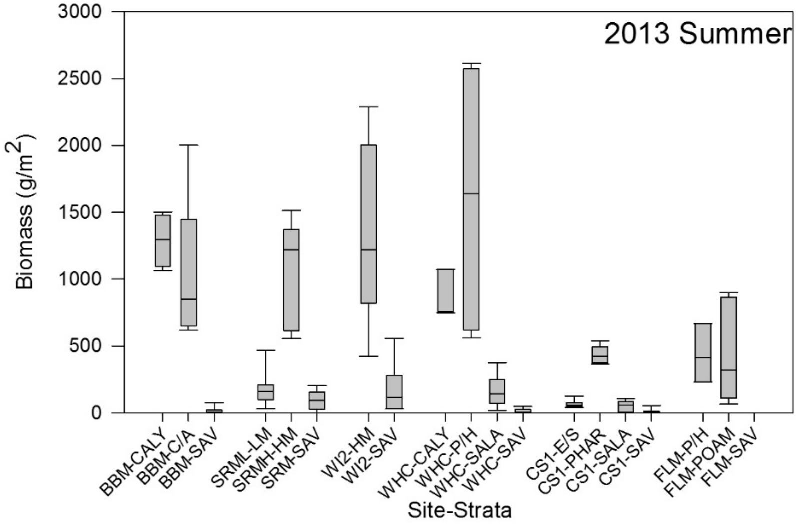
Appendix E. Vegetation Biomass 2011-2017

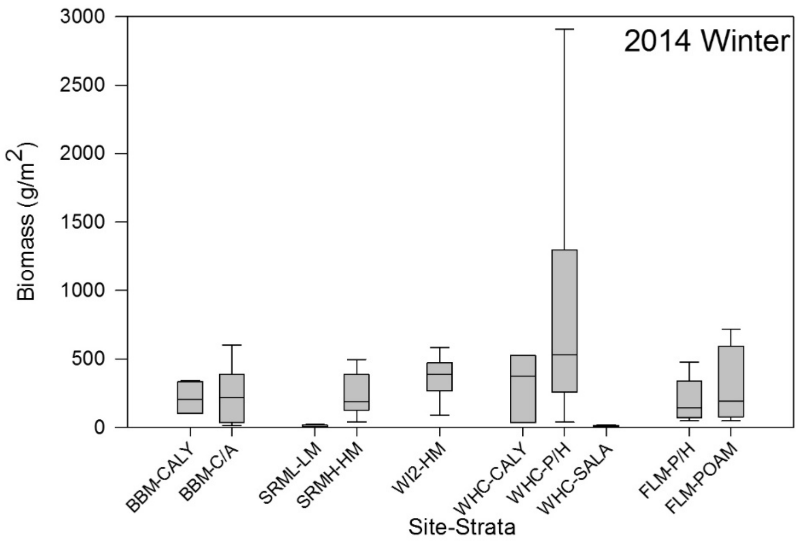
Table E1. Definition of site codes and vegetation strata used for the biomass sampling design.

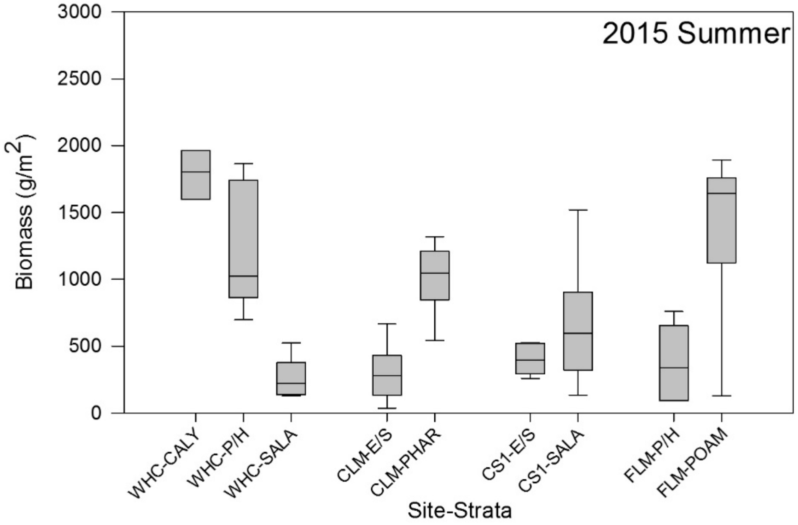
Site Code	Site		Strata Code	Strata
BBM	Ilwaco		CALY	Carex lyngbyei
SRM	Secret River		C/A	C. lyngbyei/Agrostis spp
WI2	Welch Island		E/S	Elpa palustris/Sagittaria latifolia
WHC	Whites Island		HM	High Marsh mix
CLM	Cunningham Lake		LM	Low Marsh mix
CS1	Campbell Slough		PHAR	Phalaris arundinacea
FLM	Franz Lake		POAM	Polygonum amphibium
			P/H	P. arundinacea/High Marsh mix
			P/P	P. arundinacea/P. amphibium
			SALA	S. latifolia
			SAV	Submerged Aquatic Vegetation

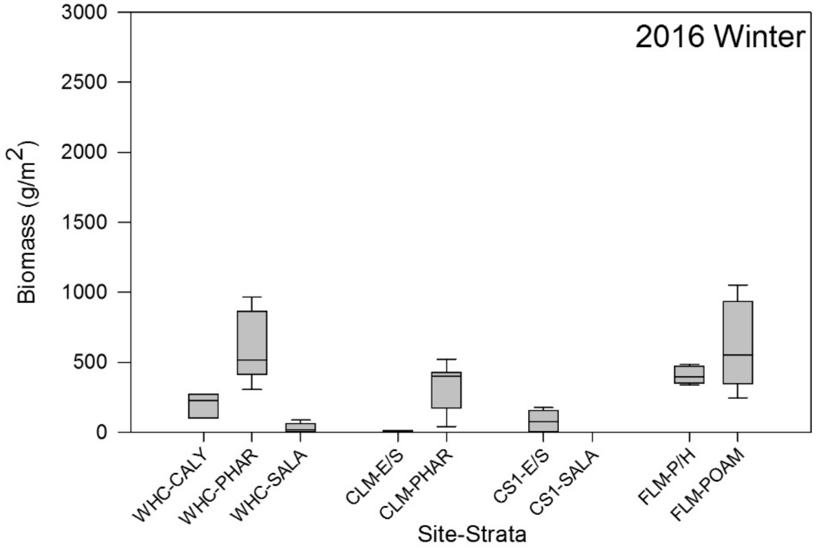
Figure E.1. Box plots of the average aboveground vegetation biomass by site and vegetation strata. Site codes and strata codes are provided in Table E.1.

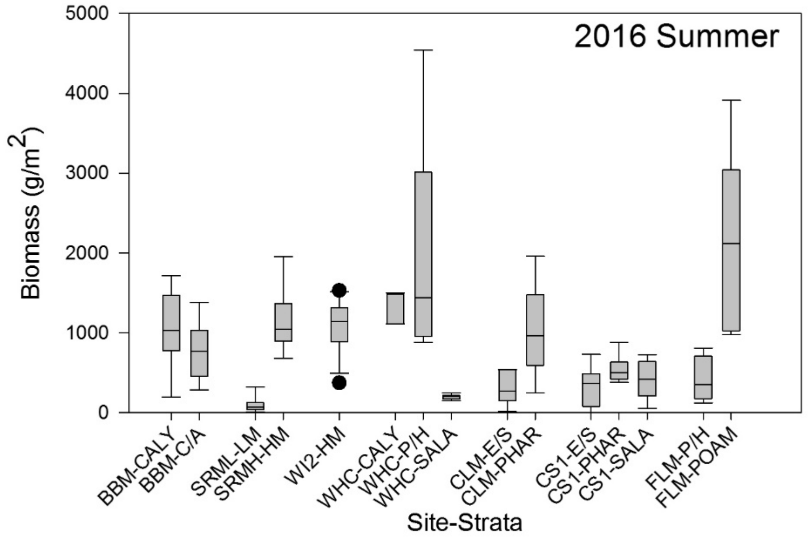



2013 Summer
Biomass (g/m^2)
3000
2500
2000
1500
1000
500
0
BBM-CALY
BBM-C/A
BBM-SAV
SRML-LM
SRMH-HM
SRM-SAV
WI2-HM
WI2-SAV
WHC-CALY
WHC-P/H
WHC-SALA
WHC-SAV
CS1-E/S
CS1-PHAR
CS1-SALA
CS1-SAV
FLM-P/H
FLM-POAM
FLM-SAV
Site-Strata


2014 Winter
Biomass (g/m^2)
3000
2500
2000
1500
1000
500
0
BBM-CALY
BBM-C/A
SRML-LM
SRMH-HM
WI2-HM
WHC-CALY
WHC-P/H
WHC-SALA
FLM-P/H
FLM-POAM
Site-Strata


2015 Summer
Biomass (g/m^2)
3000
2500
2000
1500
1000
500
0
WHC-CALY
WHC-P/H
WHC-SALA
CLM-E/S
CLM-PHAR
CS1-E/S
CS1-SALA
FLM-P/H
FLM-POAM
Site-Strata


2016 Winter
Biomass (g/m^2)
3000
2500
2000
1500
1000
500
0
WHC-CALY
WHC-PHAR
WHC-SALA
CLM-E/S
CLM-PHAR
CS1-E/S
CS1-SALA
FLM-P/H
FLM-POAM
Site-Strata


2016 Summer
Biomass (g/m^2)
5000
4000
3000
2000
1000
0
BBM-CALY
BBM-C/A
SRML-LM
SRMH-HM
WI2-HM
WHC-CALY
WHC-P/H
WHC-SALA
CLM-E/S
CLM-PHAR
CS1-E/S
CS1-PHAR
CS1-SALA
FLM-P/H
FLM-POAM
Site-Strata

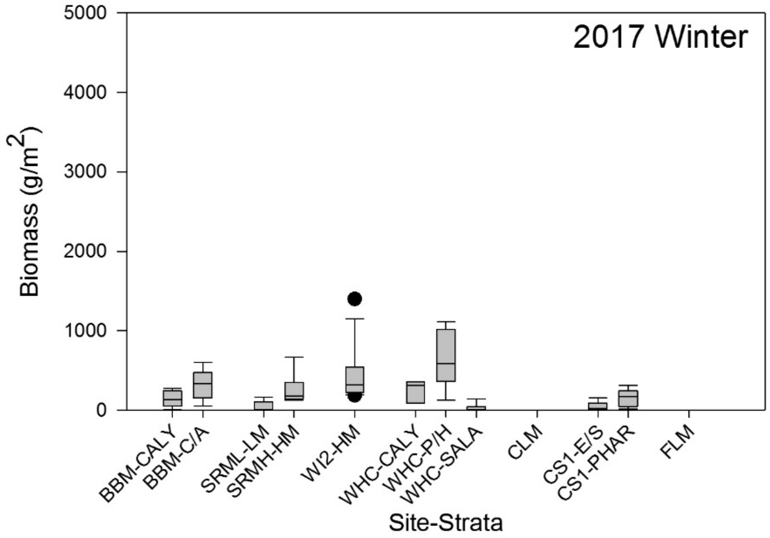

2017 Winter
Biomass (g/m^2)
5000
4000
3000
2000
1000
0
BBM-CALY
BBM-C/A
SRML-LM
SRMH-HM
WI2-HM
WHC-CALY
WHC-P/H
WHC-SALA
CLM
CS1-E/S
CS1-PHAR
FLM
Site-Strata

Table E2. Aboveground vegetation biomass sampling results from Summer 2011 to Winter 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/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)
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	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
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
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

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)
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	#DIV/0!	2012	Winter	1	66.2	na	75.2
FLM	LM	2012	Summer	1	601.1	#DIV/0!	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
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

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)
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	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	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	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
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