Action Effectiveness Monitoring for the Lower Columbia River Estuary Habitat Restoration Program

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Action Effectiveness Monitoring for the Lower Columbia River Estuary Habitat Restoration Program Annual Report (October 2018 to September 2019)

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Abbreviations and Acronyms

AEM	Action Effectiveness Monitoring
BPA	Bonneville Power Administration
CEERP	Columbia Estuary Ecosystem Restoration Program
CRD	Columbia River Datum
CREST	Columbia River Estuary Study Taskforce
EMP	Ecosystem Monitoring Program
ESA	Endangered Species Act
NMS	nonmetric multidimensional scaling
ORP	Oxidation-reduction potential
PIT	passive integrated transponder
RPA	Reasonable and prudent alternative
UAV	Unmanned aerial vehicles
USACE	U.S. Army Corps of Engineers

EXECUTIVE SUMMARY

The Lower Columbia Estuary Partnership manages the Action Effectiveness Monitoring (AEM) program with the goals of determining the impact of habitat restoration actions on salmon at the site and landscape scale, identifying how restoration techniques address limiting factors for juvenile salmonids, and improving restoration techniques to maximize the impact of restoration actions. To accomplish AEM program goals, the Estuary Partnership implements the Columbia Estuary Ecosystem Restoration Program (CEERP) AEM Programmatic plan (Johnson et al. 2016), employs standardized monitoring protocols, and coordinates between stakeholders to collect and share AEM data. The objectives of the AEM annual monitoring objectives were to quantify post-restoration hydrology, temperature, habitat, and vegetation within restoration sites, and determine post-restoration fish use at selected sites.

A total of twenty-nine restoration sites received AEM data collection in 2019. All monitoring was conducted following standardized protocols outlined in Roegner et al. (2009). Three restoration sites received Level 2 monitoring, and twenty-six restoration sites received Level 3 monitoring. A PIT tag array was operated at Horsetail Creek to determine type and residency time of salmonids at the site and address uncertainties related to fish passage through long culverts. Additionally, we conducted status fish sampling at North Unit Sauvie Island Phase 2 (Deep Widgeon and Millionaire Lakes) to identify fish presents five years post-restoration.

Hydrologic reconnection is intended to restore physical processes that provide site access to juvenile salmonids and restore ecological processes. Water surface elevation (WSE), water temperature, and habitat opportunity are metrics used to measure changes in these hydrologic physical processes at restoration sites. Across all restoration sites, post-restoration WSEs showed strong similarity to reference channel and marsh WSEs, which indicates recovery of lost hydrologic connectivity. Post-restoration water temperatures were also found to be similar to their reference sites. Both restored and reference marsh water temperatures were found to track slightly warmer than the main stem Columbia River temperatures. Combining WSE and water temperature provides a meaningful measure of salmonid habitat opportunity, as defined by the number of days a site has both suitable water temperatures and water levels for salmonids. Across all restoration sites, habitat opportunity was significantly increased post-restoration indicating restoration actions created useable (based on water depth and temperature) habitat for out migrating juvenile salmonids as soon as one-year post-restoration.

Hydrology and wetland elevation drive emergent wetland vegetation cover and composition. Across Level 2 restoration sites monitored in 2019, Wallacut Slough, Steamboat Slough, and North Unit Phase 2 (Widgeon), distinct high and low marsh vegetation zone development was evident based on both ground vegetation surveys and Unmanned aerial vehicle (UAV) vegetation mapping. Trends in plant community composition recovery towards reference native conditions were identified across all restoration sites 3-5 years post restoration. Wallacut and Steamboat sloughs also exhibited lags in plant community recovery in high marsh elevation zones, retaining reed canarygrass and other non-native species in these areas year 3 and 5 post restoration. The lack of high marsh plant community recovery was also echoed in the soil

conditions identified in these locations, which exhibited lower soil salinity, pH, and greater oxidation-reduction potential (ORP) levels than found at reference sites.

Five years post-restoration, we collected synoptic fish community data to determine if North Unit Phase 2 achieved the goal of fish use. Both marked and unmarked Chinook salmon were captured across the site. Additionally, the PIT array at Horsetail Creek continued to detect upstream salmonid species, including hatchery spring, fall, and summer Chinook along with hatchery Coho, steelhead, summer sockeye. 2019 detections at the site showed the fish occupied the area for a range of days to hours. These results indicate targeted salmonid use across the restoration sites, further highlighting the importance of restoring these lost marsh habitats. Status checks of fish occurrence at other Level 2 AEM sites and PIT array monitoring at Horsetail Creek will continue through 2020.

AEM data shows that restoration sites are achieving increases in connectivity and salmonid opportunity. However, plant community recovery is more variable, with lower elevation areas slowly developing native emergent vegetation and reed canarygrass dominating higher elevation wetland areas. These findings indicate that the re-establishment of natural physical processes to sites are accomplished in a relatively short period of time. However, the ecological response to physical drivers can take more time to manifest. Continued monitoring through the AEM program will elucidate and improve our understanding of the connections between physical processes, habitat responses, and the resulting benefits to juvenile salmon.

MANAGEMENT IMPLICATIONS

Action effectiveness monitoring measures changes to physical and ecological processes that influence the ability of restoration sites to support juvenile salmonids. In addition, AEM data provides project managers with vital information to determine if project design elements are meeting goals or if adaptive management is required.

At the site-scale, restoration projects are leading to the reestablishment of natural physical processes that support juvenile salmonids. Data has shown that site water levels respond immediately to hydrologic reconnection. Water temperatures at the restoration sites are generally warmer than nearby main stem waters but were generally suitable during the spring and early summer juvenile outmigration periods. The higher temperature at restoration sites can be attributed to shallower water depths, and this trend is mirrored in results seen at Ecosystem Monitoring Program (EMP) sites (Kidd et al. 2019).

As the goals of restoration activities include improving fish access to historic floodplain habitats and the quality of those habitats, we wanted to verify that fish are using restored sites. We chose to employ a “status check” of fish use at five years post-restoration. We collected fish occurrence data at four locations within North Unit Phase 2 and found juvenile salmonids at all locations. The presence of juvenile salmonid indicates that restoration benefits fish. The PIT array at Horsetail Creek continues to detect out migrating upriver juvenile salmonid species visiting the site for periods ranging from a few hours to a couple of days.

AEM research shows that restoration sites are achieving increases in hydrologic connectivity and salmonid opportunity; however, plant community recovery is more variable across sites. Given the inherent inter-annual climate variability, it is difficult to predict specific restoration outcomes on a year to year basis. However, clear trends in plant community recovery across restoration sites persist, with high marsh elevations retaining reed canarygrass and other non-native species at year 3 and 5 post restoration. The lack of high marsh plant community recovery is also echoed in the soil conditions identified in these locations, which retain lower soil salinity, pH, and greater ORP levels than found at reference sites. Additionally, areas within restoration sites that have undergone heavy construction impacts and grading have also been shown to recover on a slower timeline. Alternatively, we have observed that both soil and dominant native plant communities recover quickly (within 5 years post-restoration) in areas that are found at moderately low to mid wetland elevations. Across all these findings, wetland elevation is used as a proxy for restored wetland hydrology which, in combination with soil conditions, is the ultimate mechanism driving restoration outcomes throughout the estuary (e.g., Bledsoe and Shear 2000, Neckles et al. 2002, Davy et al. 2011, Mossman et al. 2012, Gerla et al. 2013, Kidd 2017). Through our AEM research we have found that the re-establishment of natural physical and hydrological processes to sites can be accomplished in a short period of time but understanding how these wetland sites respond ecologically will require long-term monitoring. Ultimately, this continued monitoring will elucidate long-term trends and improve our understanding of the connections between physical processes, habitat responses, and the resulting benefits to juvenile salmon.

AEMR Program Recommendations

Suggestions for Project Design

- Both restoration design and evaluation would benefit from the use of predictive modeling to determine the restoration of aquatic, marsh, and shrub-scrub plant communities. This type of modeling can be easily accomplished by incorporating anticipated restored hydrology and site elevations and comparable reference site conditions (Hickey et al. 2015). These data can also provide a platform for evaluating different restoration scenarios, such as considering different levels of hydrologic reconnection and/or marsh plain lowering and the impacts of this for multispecies and plant community habitat recovery (Hickey et al. 2015)[4].
    - Across multiple restoration projects we have seen very high and very low marsh elevations struggle to recover native plant cover within a 5-year timeline. Moving forward predictive modeling could aid in restoration design (and adaptive management efforts) to maximize the restoration of the mid to moderately low

[4] We are currently using this Ecosystem Modeling Approach (Hickey et al. 2015) at Steigerwald National Wildlife Refuge and Multnomah Channel Natural Area to evaluate and design for desired restoration outcomes.

    - marsh elevations which have been shown to recover native plant habitat and soil conditions quickly post-restoration (throughout the Estuary).
    - In addition, this will also aid project planning for determining seeding and planting zones in target high marsh areas for non-native species control and shrub-scrub development.
    - Assessing restoration success and goal-reaching post-restoration would also be easier given predictive maps and data could be compared to conditions observed post-restoration.

Suggestions for Project Monitoring

Site topography and Reference Sites

- Accessibility to ground survey technology such as RTK GPS systems has increased dramatically over the last five years and these systems allow us to easily map the overall topography of wetlands and their plant communities and channels. With this technology, we can assess the compatibility of reference and restoration wetland sites. Similar elevation gradients (and hydrology) should be sampled within reference and restoration sites for meaningful comparisons to be made post-restoration (and to aid in project design). In this report we have highlighted that the reference site elevations have generally been a poor match with each restoration site's restored elevations. Moving forward, we will aim to alter monitoring plans to sample more overlapping elevation gradients between the restoration and reference sites to correct these issues. Additionally, upon choosing reference sites to inform project design and post-restoration project success, elevations and (anticipated) hydrology should be compared to ensure that the use of reference elevation data is an appropriate proxy for hydrologic conditions.

Hydrology

- Hydrology is a critical component to all wetland restoration efforts and should be monitored for project planning, design, and to assess project success. During project design clear hypotheses should be developed to define hydrologic changes anticipated from restoration efforts. For monitoring, data loggers need to be in placed in areas that are anticipated to experience these hydrologic changes post-restoration and remain in the same location pre- and post-restoration. Given the number of issues we have experienced through the years with data loggers we recommend having at least one redundant logger be placed within the site (nearby or at the same location), that can provide additional data in case of equipment failure (which is common). Loggers need to be maintained at least every six months and we recommend all deployment and retrievals follow the new and more detailed monitoring protocols to avoid data loss (Kidd et al. 2018).

Sediment Accretion and Erosion, Channel Cross-Sections

- Understanding sediment accretion and erosion dynamics across the floodplains of newly restored wetlands is critical for tracking wetland and channel development and long-term topographic trajectories. Sediment dynamics across restoration sites can be extremely variable, making it difficult to track meaningful change without intensive and extensive monitoring efforts. We recommend shifting our current approach of sediment monitoring (one or two sediment benches placed within a site) to a more targeted application of these methods. Before restoration occurs, specific areas of interest should be selected and multiple sediment monitoring benches (a minimum of 6) should be installed along the elevation gradient and within these targeted areas. Within the sediment bench monitoring area (between the pins), we also recommend tracking dominant plant community development and soil characteristics to aid data interpretation. Channel cross-section monitoring should be similarly focused, and extreme care should be taken to resurvey the exact location of the cross-section for meaningful results to be obtained. Both channel cross-section and sediment benches need to be resurveyed using RTK GPS technology to provide topographic context and increase data usability. Updated monitoring protocols are currently in development for these methods (Kidd and Rao 2019).

Wetland Plant Community

- Native wetland plant communities provide a critical base of the salmonid food web and are essential for determining wetland restoration success (Rao et al. 2020). We have found that monitoring a randomized selection of vegetation plots each year creates a great amount of variability in the data, and makes determining what change has been caused by the restoration and what change is due to the new randomized sampling difficult. There are two approaches to addressing this issue: to (1) continue to randomize the plots annually but significantly increase the overall total number of plots surveyed, or (2) to only randomize the plots the first year of monitoring and re-visit these same plots year after year. We recommend (2)—re-visiting the same plots year after year, which provides a clear path to assessing plant community changes overtime and does not increase the overall amount of time required to conduct sampling. Additionally, as shown in this report, the collection of soil data, alongside of plant community data, can be very informative when evaluating wetland development and restoration. We recommend integrating soil data collection as an essential metric for Level 2 monitoring across sites. Further vegetation and soil monitoring recommendations are forthcoming, as we work on a comprehensive update to the *Protocols for Monitoring Habitat Restoration Projects in the Lower Columbia River and Estuary* (Roegner et al. 2009).

Utilizing UAV Technology: Site Topography, Plant Community Mapping

- The accessibility and applicability of UAV and associated sensor technology have made significant strides in the last several years. Using some of the most affordable equipment and software available we have shown that large scale site wetland plant community and topographic mapping is possible and accurate (Kidd et al. 2020). Mapping dominant native and non-native plant communities across large portions of restoration sites can aid evaluation of project success post-restoration, and guide both active restoration project design and post-restoration project adaptive management efforts. Moving forward we are working to refine our UAV monitoring methods to include tracking channel and floodplain topographic development into our analysis and reporting. We are also exploring methods of evaluating biomass and carbon stores across reference and restored wetlands using our

UAV and sensor technologies. Further UAV vegetation monitoring methods and recommendations will be included in the comprehensive update to the *Protocols for Monitoring Habitat Restoration Projects in the Lower Columbia River and Estuary* (Roegner et al. 2009).

Fish and Macroinvertebrate Monitoring

- AEMR Level 2 monitoring does not encompass comprehensive fish or macroinvertebrate monitoring as part of the standard habitat monitoring protocol. Level 2 monitoring includes limited macroinvertebrate monitoring (one or two neuston tows a year following the Level 2 monitoring schedule) and a one-time fish sampling event at year five post-restoration. Given the spatial and temporal variability of both fish and macroinvertebrate populations seen across the long-term EMP reference sites (Rao et al. 2020), we have concluded that a more comprehensive macroinvertebrate and salmonid sampling effort is required, for meaningful post-restoration food web conditions to be evaluated. Limited fish monitoring shows that juvenile salmonids are present in restoration sites after tidal reconnection, but without intensive monitoring efforts, the number of fish using the site can be difficult to ascertain. Furthermore, it is not known if the number of fish accessing a site increases as the habitat moves toward a reference state. A better understanding of how physical processes influence habitat conditions and how these resulting habitat conditions support juvenile salmonids are key to quantifying the overall impact of restoration efforts. The addition of long-term ecosystem monitoring at a select number of restoration sites would allow for these sites to be tracked alongside the EMP. The EMP sites have years of accumulated status and trends fish, macroinvertebrate, water quality, and habitat data which could be used for ongoing comparative analysis and evaluation. Selecting focal restoration sites of interest and conducting intensive fish and macroinvertebrate monitoring efforts at these sites, similar to the level of monitoring conducted across EMP sites (Rao et al. 2020), would allow for the recovery of fish use and macroinvertebrate communities to be assessed over the long-term and aid in the interpretation of how physical changes to habitat directly influence the salmonid food web.

Frequency of Monitoring

- Currently, Level 3 monitoring is conducted 1-year pre-restoration through year 5 post-restoration and Level 2 monitoring is conducted pre, 1, 3, and 5 years post restoration. Results from the last 6 years of the AEMR Level 2 and 3 monitoring indicate that restoration outcomes can be slow and variable, with sites not achieving reference level native plant community conditions by year 5 post-restoration (Johnson et al. 2018, and this report). Given these observations, we recommend that level 3 monitoring continue to occur pre through 5, 8, and 10 years post-restoration and that Level 2 monitoring should also be conducted at year 8 and year 10 post-restoration. Adding year 8 and 10 to monitoring for all level 2 and 3 metrics will aid in understanding the long-term impacts of our restoration efforts and allow for monitoring to occur over a wider spectrum of annual climate conditions. Additionally, we recommend UAV plant community mapping occur across all Level 2 and 3 sites pre-restoration, and 3, 5, 8, and 10 years post-restoration. These additional data and longer-term monitoring windows will provide greater context to assess restoration actions and outcomes and help us test ongoing hypotheses about how shifts in climate and river discharge conditions impact restoration outcomes. Synthesis reports of site conditions at year 8 and 10 post-restoration will also provide meaningful insight for ongoing adaptive management and restoration efforts.

SYNTHESIZING RESTORATION RESULTS

- The most meaningful analysis of restoration success would be one that incorporates all habitat level monitoring metrics across a site to identify recovery of salmonid habitat overtime. We are currently developing a site wide assessment of habitat opportunity that extends across the wetland's active floodplain (Johnson et al. 2018). This would incorporate floodplain topography, water surface elevation (water depth), water temperatures, and dominate plant communities to highlight salmonid habitat conditions across the active floodplain of restoration and reference sites. This active floodplain mapping approach could also be used as a tool to evaluate the impacts of climate change and shifting river discharge on wetland habitat conditions throughout the Columbia Estuary.

INTRODUCTION

Program History

The Action Effectiveness Monitoring (AEM) program is managed by the Lower Columbia Estuary Partnership (Estuary Partnership) and addresses RPA 60 of the 2008 Draft Biological Opinion (NMFS 2008). As part of the Columbia Estuary Ecosystem Restoration Program (CEERP), this program provides the Bonneville Power Administration (BPA), restoration partners (e.g., USACE and CREST), the Environmental Protection Agency, and other stakeholders with data to assess the success of restoration projects in the Lower Columbia Estuary.

In 2008, during the pilot phase of the program, the Estuary/Ocean subgroup (EOS) recommended four projects for AEM. The selected AEM sites were monitored annually until 2012 and represented different restoration activities, habitats, and geographic reaches of the river. The initial phase of AEM resulted in site scale monitoring and the standardization of data collection methods, but also highlighted the need for expanded monitoring coverage, paired restoration and reference sites, and comparable monitoring to ecosystem status and trends monitoring to evaluate reach and landscape scale ecological uplift.

To provide monitoring at all restoration sites, three monitoring levels are implemented at restoration sites as follows:

<u>Level 3</u> – includes "standard" monitoring metrics: water surface elevation, water temperature, sediment accretion, and photo points that are considered essential for evaluating the effectiveness of hydrologic reconnection restoration. This monitoring is done at all restoration sites within the CEERP. Level 3 monitoring is conducted by project sponsors.

<u>Level 2</u> – includes the Level 3 metrics and metrics that can be used to evaluate the capacity of the site to support juvenile salmon. These metrics include vegetation species and cover; macroinvertebrate (prey species) composition and abundance; and channel and wetland

elevation. This “extensive” monitoring is done at a selected number of sites chosen to cover a range of restoration actions and locations in the River and is intended to provide a means of monitoring an “extensive” area. Level 2 monitoring is conducted by the Estuary Partnership.

Level 1 – includes Level 2 and 3 metrics and more “intensive” monitoring of realized function at restoration sites, such as fish use, genetics, and diet. Since Level 1 monitoring is more expensive, it is conducted at fewer sites with the goal of relating the Level 1 results to the findings of the Level 2 and Level 3 monitoring. Level 3 monitoring is conducted by the USACE.

Program Overview

The Lower Columbia Estuary Partnership manages the Action Effectiveness Monitoring (AEM) program with the goals of determining the impact of habitat restoration actions on salmon at the site and landscape scale, identify how restoration techniques address limiting factors for juvenile salmonids, and improve restoration techniques to maximize the impact of restoration actions.

To accomplish AEM program goals, the Estuary Partnership implements the Columbia Estuary Ecosystem Restoration Program (CEERP) AEM Programmatic plan (Johnson et al. 2016), employs standardized monitoring protocols, and coordinates between stakeholders to collect and share AEM data. The objectives of the AEM annual monitoring objectives were to quantify post-restoration hydrology, temperature, habitat, and vegetation within restoration sites, and determine post-restoration fish use at selected sites.

The goals of the AEM program are to:

1. Determine the benefit of restoration actions for juvenile salmonids at the site, landscape, and ecosystem scale.
2. Improve restoration and monitoring techniques to maximize the benefits of habitat restoration projects.
3. Use the results of intensive AEM (Level 1) to focus extensive AEM efforts (Level 2 and 3) and link fish presence and habitat recovery outcomes through a lines of evidence approach.

To meet these goals, the Estuary Partnership is engaged in the following tasks:

1. Implementing AEM as outlined in the Estuary RME plan (Johnson et al. 2008), Programmatic AEM plan (Johnson et al. 2016), and following standardized monitoring protocols (e.g., Roegner et al. 2009) where applicable.
2. Developing long-term datasets for restoration projects and associated reference sites.
3. Coordinating between stakeholders to improve AEM data collection efficiency.
4. Supporting a regional cooperative effort by all agencies and organizations participating in restoration monitoring activities to create a central database to house monitoring data.
5. Capturing and disseminating data and results to facilitate improvements in regional restoration strategies.

A total of twenty-nine restoration sites received AEM data collection in 2019 (Figure 1, Table 1). The specific monitoring actions for 2019 involved quantifying water surface elevation, water temperature, habitat opportunity, and vegetation at restoration sites. Additionally, at year 5, post-restoration fish data are collected to determine the composition of the fish community. To put ecological changes at restoration sites into context, the program incorporated data from reference sites monitored in the Ecosystem Monitoring Program (EMP), which focuses on characterizing the status, trends, and juvenile salmonid usage of relatively undisturbed emergent wetlands.

All monitoring was conducted following standardized protocols outlined in Roegner et al. (2009). Three restoration sites received Level 2 monitoring, and 26 restoration sites received Level 3 monitoring. A PIT tag array was operated at Horsetail Creek to determine type and residency time of salmonids at the site and address uncertainties related to fish passage through long culverts. Additionally, we conducted status fish sampling at North Unit Sauvie Island Phase 2 (Deep Widgeon and Millionaire Lakes) to identify fish presents five years post-restoration.

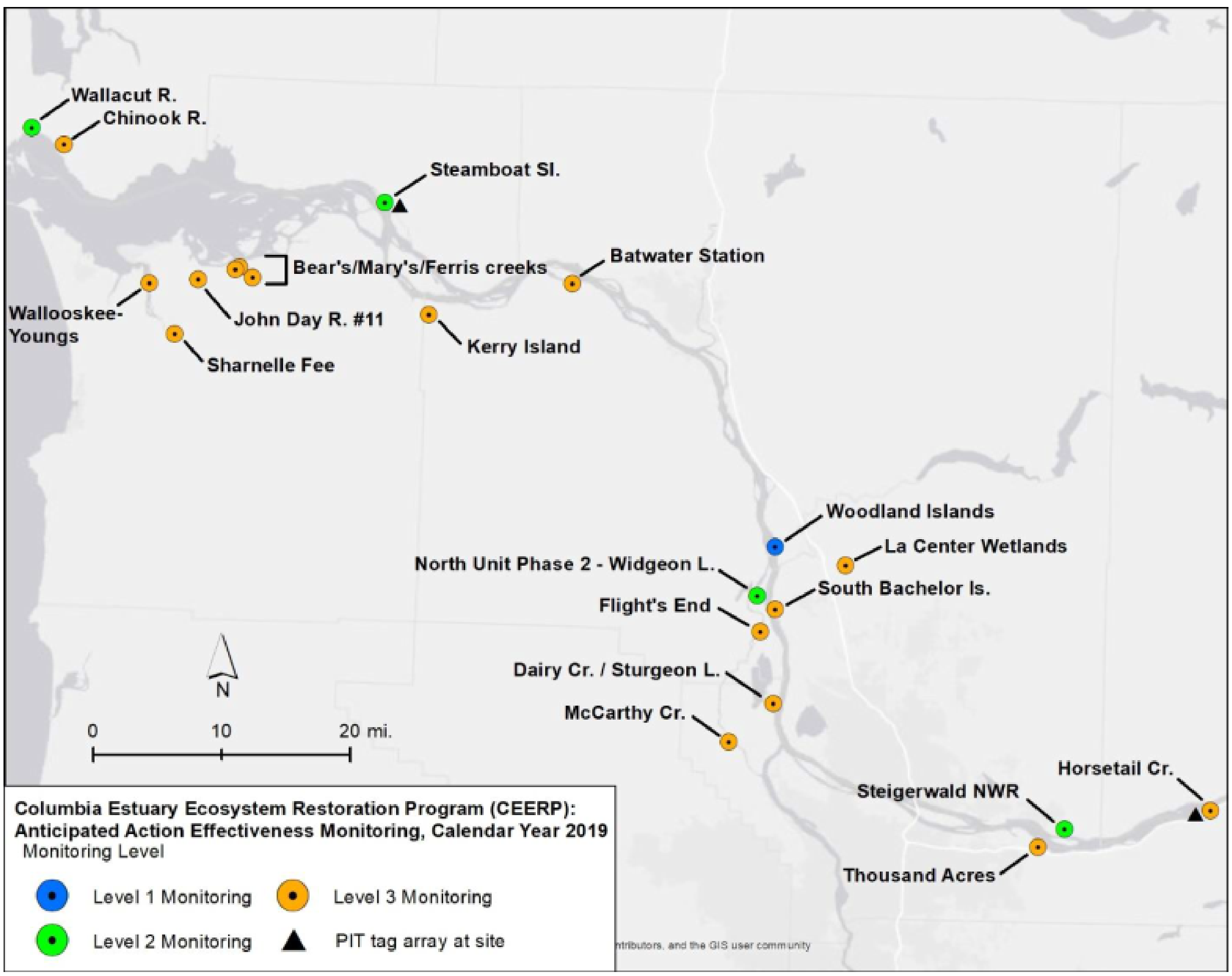

Figure 1: AEMR Level 1, 2, and 3 monitoring planned for 2019. See Table 1 for details.

Table 1. Summary of AEMR accomplished or planned from 2012 through 2022. 2019 is bolded for emphasis. Numbers in the cells show AEMR level. Orange shading signifies the year of construction. Asterisks indicate PIT tag monitoring, no direct fish capture.

Project Name	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Batwater	3	3		3	1,3	1,3	3	**3**	3		
Bear-Mary's-Ferris					3	3	3	**3**	3	3	3
Buckmire Ph 1			3	3	3				3		
Chinook River	2, 3	2, 3	2, 3	2, 3	2, 3	2, 3		**3**			
Colewort Creek	1, 3	3	1,3	3			3				
Crane-Domeyer				3	3	3	3	**3**	3	3	
Dairy Creek/Sturgeon Lk							3	**1*, 3**	1*,3	3	3
Dibblee		3	3	2,3	1,2,3	1,2,3					2,3
Elochoman Slough East				3	3	3	3	**3**	3		
Flight's End					3	2, 3	2,3	**3**	2,3	3	2,3
Government Island						3	3	**3**	3	3	3
Gnat Creek #1 and #2	3	3	3	3	3						
Horsetail		1*, 3	1*,3	1*,3	1*,3	1*,3	1*,3	**1*, 3**	1*	1*	1*
John Day River #11								**3**	3	3	3
Kandoll Farm		2, 3	2,3		2,3		2,3				
Karlson Island			3	3	1,2,3	1,2,3	3	**3**			
Kerry Island				3	3	3	3	**3**	3	3	
La Center Wetlands		3	3	2, 3	2,3	3	2,3	**3**	1,2,3		
Louisiana Swamp	3	3	3	3	3	3	3	**3**			
McCarthy Creek							3	**3**	3	3	3
Mill Road (2011)	2. 3						3				
North Unit Ph 1 Ruby	3	2, 3	2,3		2,3		2	**3**			
North Unit Ph 2 Widgeon		3	2, 3	2,3		2,3		**2,3**			
North Unit Ph 3 Jack			3	3	3	3	3	**3**	3		
Otter Point	3	3	3	3							
Sandy R Delta (dam)	1,2,3	3	3	1, 3	1,2,3						
Sharnelle Fee			3	3	3	3	3	**3**	3		
Skamakowa Creek	3	3				3					
Skipanon Slough			3		3				3		
South Bachelor Island								**3**	1,3	3	3
Steamboat Slough		2, 3		2, 3	1,2,3	1,2,3	1, 3	**2, 3**			
Steigerwald							3	**2, 3**	2.3	2,3	3
Thousand Acres			3	3	3			**3**			
Wallacut River			2, 3	2, 3	2,3	2,3	3	**2, 3**	3	2, 3	
Wallooskee-Youngs				2, 3	3	3	2,3	**3**	2, 3	3	2,3
West Sand Island							3	**3**	3	3	3
Westport Slough (USFWS)				3	3	3	3	**3**	3	3	

Project Name	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021	2022
Willow Bar				3	3	3	3	**3**	3	3	
Woodland Islands								**2,3**	2,3	2,3	

Methods

Site Selection 2019

Four restoration sites were selected for Level 2 monitoring (Table 2) in 2019 using the prioritization criteria outlined in Johnson et al (2016). Three associated reference sites were chosen to establish a before-after reference impact monitoring design, which puts pre- and post-restoration site data into ecological context (Table 2). In this report, a summary of results for level 2 monitoring metrics is provided for all sites surveyed in 2019, except for Steigerwald which is still under construction and will receive a formal post-restoration write up in 2022 (1-year post-restoration).

Table 2. Sites included in Level 2 monitoring in 2019.

RKM	Site	Project Management	Description	Construction	Pre	1 yr	3 yr	5 yr	Reference site
6	Wallacut	CLT	Tide-gate upgrade, non-native plant community treatment with herbicides	2016	2014	2017	2019	*2021*	Ilwaco Slough (RKM 6, EMP site)
57	Steamboat Slough	Army Corps	Full tidal reconnection	2014	2014	2015	2017	2019	Welch Island (RKM 53, EMP site)
142	North Unit (Phase 2)	CREST	Full tidal reconnection targeted marsh plain lowering	2014	2014	2015	2017	2019	Cunningham Lake (RKM 145, EMP site)
200	Steigerwald	LCEP	Full channel and tidal reconnection, alluvial fan restoration, and targeted marsh plain lowering	*2021*	2019	*2022*	*2024*	*2026*	Reed Island (RKM 200), and Franz Lake (RKM 221, EMP site)

Habitat Monitoring

Methods from the protocol "Lower Columbia River Estuary Habitat Action Effectiveness v1.0" were used to evaluate changes related to restoration actions and quantify ecological uplift (Roegner et al. 2009, Protocol ID: 460).

We surveyed vegetation cover and composition (Method ID: 822) to assess changes to habitat structure related to restoration actions. Vegetation cover and composition is an indicator of the production of organic matter and the detritus produced by decaying vegetation forms the base of the food web for many species in the lower Columbia River and estuary (Borde et al. 2010, Maier and Simenstad 2009). Vegetation plot elevation (Method ID: 818) was recorded to track the effectiveness of lowering marsh elevations (soil scrape down) to control invasive vegetation and promote native plant species growth. At each restoration site, two vegetation monitoring areas were established – one in an area directly impacted by restoration actions and one in an area indirectly impacted by restoration actions. Two vegetation sampling areas provide an overview of overall site condition pre- and post-restoration. Sediment Accretion (Method ID 818) was measured to determine if constructed wetlands are self-sustaining. Water Temperature (Method ID 816) was measured to determine habitat suitability for juvenile salmonids. Water Surface Elevation (Method ID 814) was measured to determine opportunity for juvenile salmonid species to access the site and determine timing and level of wetland inundation.

Soil survey - Within each quadrat, in-situ surface soil salinity, conductivity, soil placed 5 cm below the soil surface representing the midway point vertically of the sample (Bledsoe and Shear 2000, Neckles et al. 2002, Davy et al. 2011, Mossman et al. 2012, Gerla et al. 2013). All soil surveys were conducted in saturated soil conditions, timed near peak low tide (lowest tidal elevation) and surveyed in order from highest to lowest elevation. Although these soil parameters are dynamic over time depending on the precise environmental conditions present and the duration of tidal flooding, the logic in taking these in-situ samples was to capture the general gradient that existed among the different plant communities. If all samples were collected under similar conditions and at similar intervals of time, they become more comparable amongst each other. Redox potential (ORP), pH, and temperature data were collected using Extech soil probes. For detailed information about these soil parameters and tidal wetland restoration see Kidd 2017.

Fish and Macroinvertebrate Monitoring

At North Unit (Phase 2, Millionaire and Deep Widgeon) sampling was conducted to determine the fish community and whether salmon were present or absent. Fish were collected using a bag seine (BS; 37 x 2.4 m, 10 mm mesh size). All sets were deployed using a 9-ft Zodiac inflatable raft. The objective of the sampling was to determine the fish community and whether salmon were present or absent, there was no limits on the number of seine efforts at each site (Table 3). All non-salmonid fish were identified to the species level counted and released. All salmonids were measured (fork length, nearest mm), weighed (nearest g), and released. A genetic sample was taken from the caudal fin on all captured Chinook salmon at both Millionaire and Widgeon Lakes. All salmonids were checked for adipose fin clips, or other external marks, coded wire tags, and passive integrated transponder tags to distinguish between marked hatchery fish and unmarked (presumably wild) fish. A fish condition index (Fulton's) was calculated using the following equation: K = (W/L3) x 100,000.

Table 3. Total number of Bag Seine efforts done at Millionaire Lake (Millionaire 1 & Millionaire 2) and Widgeon Lake (Widgeon 1 & Widgeon 2) in April 2019.

Site	Effort #
Millionaire 1	4
Millionaire 2	3
Widgeon 1	3
Widgeon 2	2

A PIT tag detection system was installed at the confluence of Horsetail and Oneonta Creeks to monitor fish passage through a culvert located under the I-84 highway. The system consists of a Biomark FishTRACKER IS1001-MTS distributed Multiplexing Transceiver System (MTS). The MTS unit receives, records, and stores tag signals from 10 antennas, which measure approximately 6' by 6' and are mounted on the north and south sides of the 5-barrel culvert system running under the freeway. The system is powered by an 840-watt solar panel array and supported by 24-volt, 800 amp-hour battery bank backup. The unit is connected to a fiber optic wireless modem that allows for daily downloads of tag data and system voltage monitoring updates. In 2019 PIT tag was under normal operation.

In 2019, macroinvertebrate neuston tows were collected at Wallacut and North Unit Phase 2, however at the time of this reporting, these data are still under analysis and will be updated once available.

Analysis

Water-surface elevation (WSE)

WSE is the primary indicator of hydrographic conditions at a site. Continuous pre and post-restoration water level data was collected at the restoration sites and a nearby outer reference channel. The sensors collecting data 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 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

Pre- and post-restoration hydrographs for the wetland channel were created and compared to those for the outer for the outer reference channel and a nearby reference site ("a site with little or no anthropogenic influence", Borde et al., 2012). An effective restoration project would have a WSE that matches the conditions of the reference site, indicating hydrology for the site were meeting restoration principles.

Water Temperature

Water temperature data was still under analysis at the time of writing this report and will be included in the 2021 synthesis report. Monthly maximum 7-day moving average maximum (7-DMA) will be calculated for sites post-restoration to compare to an outer reference location and main stem conditions. The Columbia mainstem data collection station S8 (Washougal, EP) will be used for comparison. Previous research has shown that main stem temperatures do not vary substantially, and a single station is an adequate representation of general main stem conditions for any given time-period (Sager et al. 2014).

Vegetation

To assess species richness (defined as the total number of species) and percent cover for the herbaceous vegetation community at a given restoration site, we categorized plants species into native/non-native categories. We calculated species richness, species diversity (Equation 1), and relative cover for native and non-native plants out of the total assemblage for sampling episodes before and after restoration for seven restoration sites for which data were available. To evaluate significant ($p<0.05$) differences in plant community and soil conditions across years and sites, ANOVA and Tukey's HSD tests were performed. Data analysis was conducted using Microsoft Office Excel (2016), Exploratory (2017), and R (2020) software.
Equation 1. Shannon Diversity Index

$$H' = -\sum_{j=1}^{s} p_i \ln p_i$$

where H' = Shannon Diversity Index
p_i = importance probability in column
i= matrix elements relativized by row totals (see Greig-Smith 1983, p.163; based on Shannon and Wiener 1949).

UAV Plant Community Mapping

Quantifying the distribution and abundance of dominant plant communities over time is of fundamental importance to ecological and restoration effectiveness monitoring. Our ability to estimate plant distributions over large areas (i.e., several hectares) using traditional approaches (transect or quadrat methods) is limited because of the time and expense required. In 2019, we conducted aerial surveys using an unmanned aerial vehicle (UAV) to develop a map of the current extent (2019) of dominant native and non-native plant community distributions across the restoration sites.

Data Collection

A DJI Phantom 4 was outfitted with a Sentera Near Infrared (NIR) Camera was the UAV chosen to collect multispectral aerial images (visible or RGB, and NIR) of the restoration sites. At each site, Pix4D capture was used to create the flight polygon grid with overlaps of 80% fore-lap and 80% side-lap. The UAV was flown at 200ft above ground level (AGL), producing a high density of images (ground sampling distance (GSD) of 1.68 inches per pixel). Multispectral data was collected between 11am and 12pm to ensure consistent light conditions at all sites.
In order to geo-reference the aerial images, ground control points (GCPs) were placed at sites and surveyed. Between 5 to 10 GCPs were placed at each site, depending on the range of terrain elevations at the sites. The GCPs were 1m x 1m, black and white rectangular cardboard cut-outs, the position and elevations of which were captured using a TOPCON Real Time

Kinematic (RTK) GPS. Elevations of different vegetation communities were also collected to outline representative dominant plant communities on the site.

Figure 2: 1m x 1m rectangular ground control point (GCP)

Data Processing

Multispectral images collected by the UAV were imported into PIX4D mapper to create products that will aid in mapping vegetative communities at the site. Images from each camera were processed separately to obtain different products. RGB images were processed to obtain an Orthomosaic and a digital surface model (DSM), while NIR images were processed to determine the normalized difference vegetation indices (NDVI) of the vegetation at the site. Pix4D Mapper comparatively analyzed multiple points in the imported images to triangulate matches and create a 3D point cloud of the sites. The point cloud was then georeferenced using the collected GCP information to create an orthorectified mosaic of RGB data of the site and a corrected elevation model called a Digital Surface Model (DSM) (Figure 4). Pix4D processed NIR images also in the same manner, however, in addition to producing an Orthomosaic and a DSM, the software also produced a mosaic of the NDVI for the site (Figure 3). The NDVI is a well-established indicator for presence and condition of vegetation at a site and ranges from -1 to +1. Negative values indicate no green biomass and positive values indicate lush green biomass. Bare ground areas usually produce values of zero.

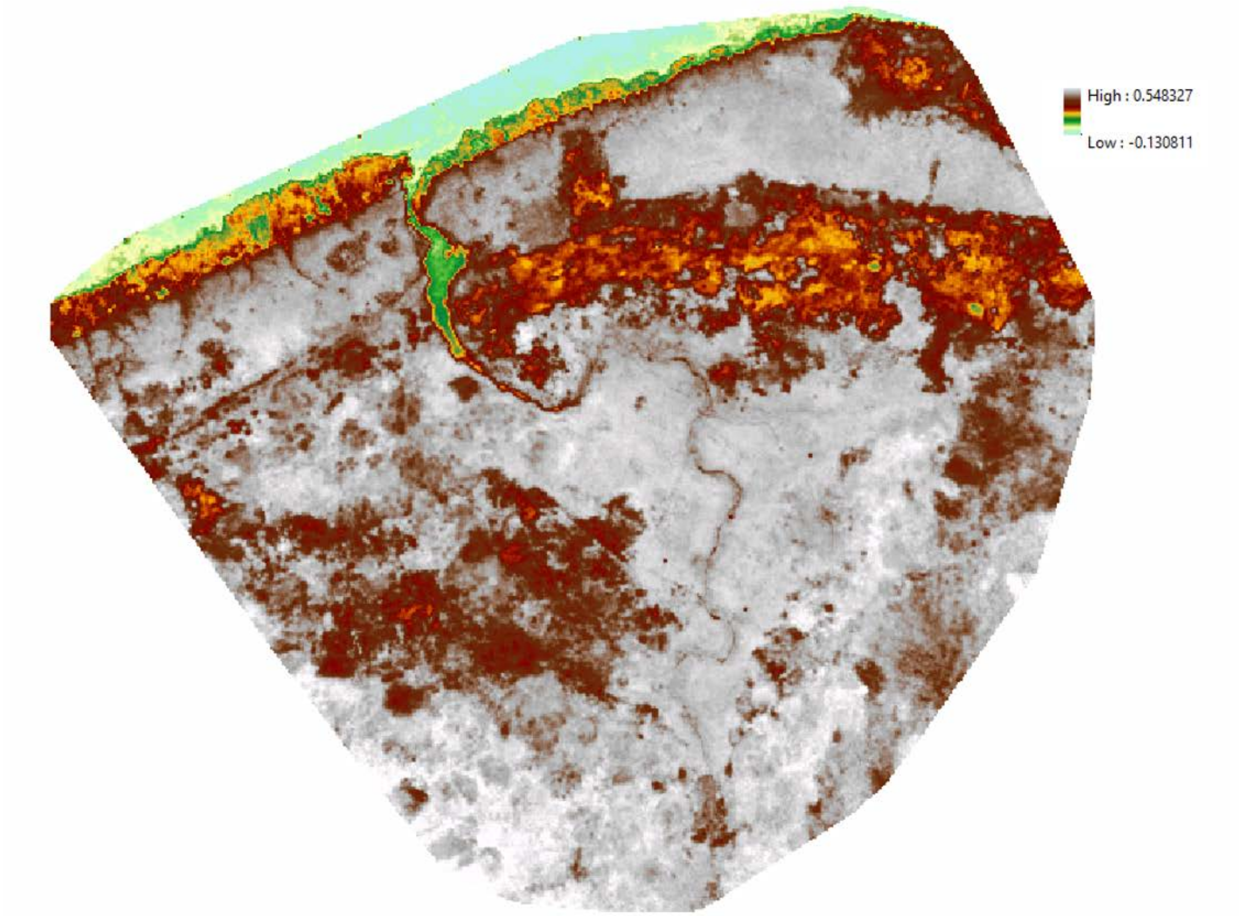

Figure 3: NDVI Mosaic for Wallacut Slough

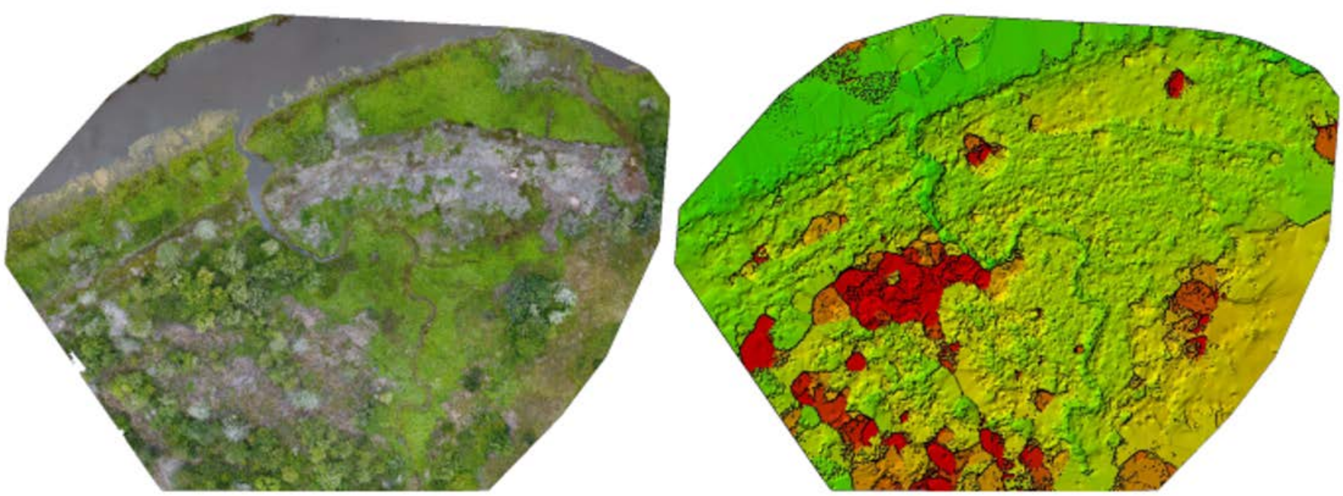

Figure 4: RGB Orthomosaic and Digital Surface model (DSM). The different colors on the DSM represent ranges of elevations present at the site, red color representing higher elevations and green representing low elevations

Data analysis

RGB and NDVI orthomosaic were combined with the DSM and ground plant community survey data in ArcGIS and R statistical software was used to model the extent of dominant native and non-native plant communities across the site. These data were evaluated for accuracy using the plant community data collected during the ground survey. The final product of this analysis is a dominant plant community map of the site in addition to estimates (in acres) of the extent of these communities.

RESULTS

2019 Water Year Overview

Habitat Restoration and Climate Variability

Long-term status and trends monitoring conducted through the Ecosystem Monitoring Program have underscored the importance and influence that shifts in annual climate and discharge conditions in the Columbia River have on tidal wetland food web dynamics and habitat conditions (Rao et al. 2020, Kidd et al. 2019). Ongoing synthesis efforts of EMP data have revealed that plant community composition of both reference and restoration sites can be heavily impacted by discharge conditions in the Columbia during the growing season, resulting in annual shifts in both reed canarygrass and native wetland plant community abundance (Rao et al. 2020, Kidd et al. 2019).

Annual climatic variations can also cause a shift in wetland and mainstem water temperatures and water biogeochemistry impacting local tidal wetland water quality conditions for salmonids. All wetland restoration sites in the estuary are impacted by these annual shifts in climatic and discharge conditions which makes simple pre-post restoration comparison difficult to interpret, especially if extreme dry or wet years fall right before or after restoration has occurred (Johnson et al. 2018). Comparing pre/post restoration success to that of a reference site tracked during the same time period can be a helpful way to account for the variability in annual conditions; however, it is critical to provide appropriate water year and climatic descriptions for any pre/post or time series analysis and comparison of habitat conditions across sites in the estuary. To aid in this, we have provided an expert from the 2020 EMP report below, which highlights these conditions experienced in 2019 through 2010. For a more detailed analysis of these data, please visit the EMP report directly (Rao et al. 2020).

Overview of 2019 and historic conditions

River flows in the Columbia and its tributaries are influenced by a combination of winter snowpack and pluvial flows driven by rainfall. High snowpack arises from cold and wet winters, while low snowpack arises from dry conditions throughout the winter, which can be either warm or cold (Figure 6). The timing of precipitation and whether it falls as snow or rain influences the timing and magnitude of the spring freshet. Typically, the freshet begins in late April/early May and persists into June. After that, the summer period tends to be dry, and river flows are low between June and October.

Compared to the previous nine years (Figure 7), discharge at Bonneville Dam during the freshet in 2019 can be characterized as dry, on the whole (Figure 5). Discharge was nearly as low as the long-term minimum until mid-March and again after the freshet subsided. The freshet itself was close to average. Thus, the water year could be described as having a lower-than-average baseline flow with an average-sized freshet. The freshet occurred in a series of peaks between April and early June.

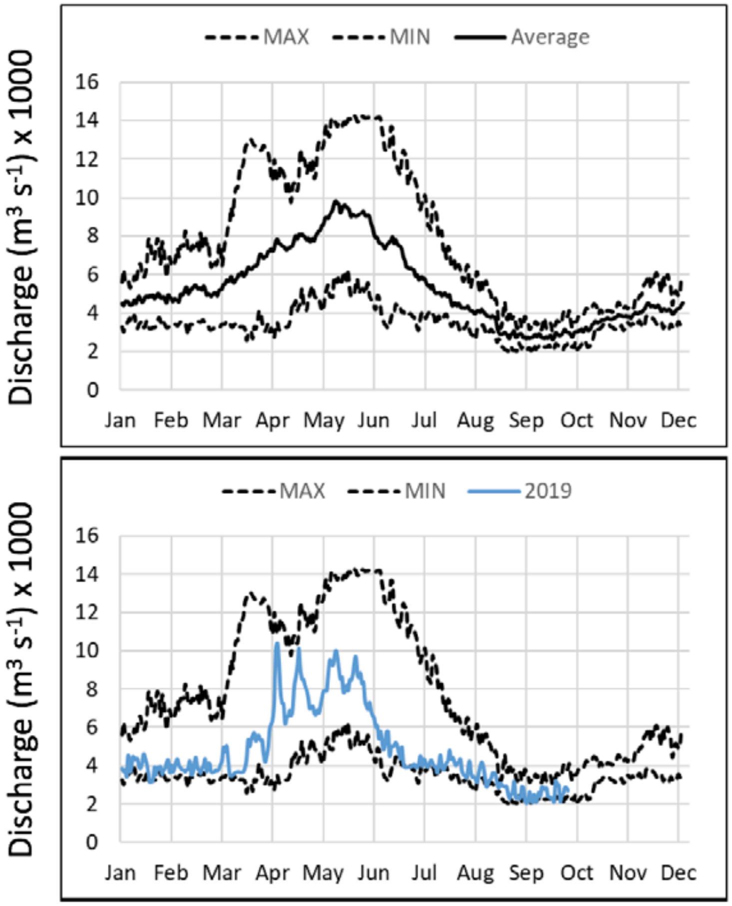

Figure 5. Top panel: Minimum, maximum, and average Columbia River discharge at Bonneville Dam between 2011 and 2019. Bottom panel: Minimum, maximum, and 2019 river discharge fluxes at Bonneville Dam.

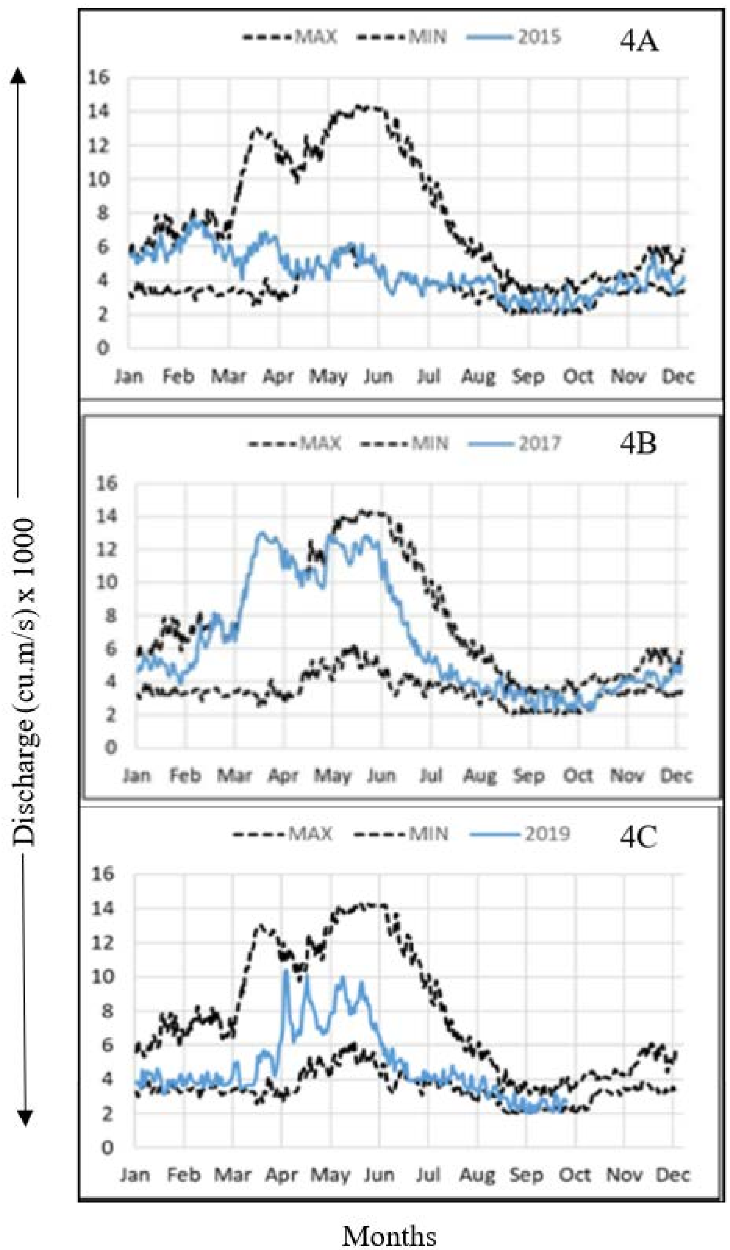

Figure 6. Comparative panels of minimum, maximum, and average river discharge at Bonneville Dam in 2015, 2017, and 2019. Panel 4A represents discharge for 2015 which consisted of warm rainy winter, low snowpack and summer drought. Panel 4B represents discharge for 2017 which consisted of high precipitation and large snowpack. Panel 4C represents discharge for 2019, described as an "dry" year.

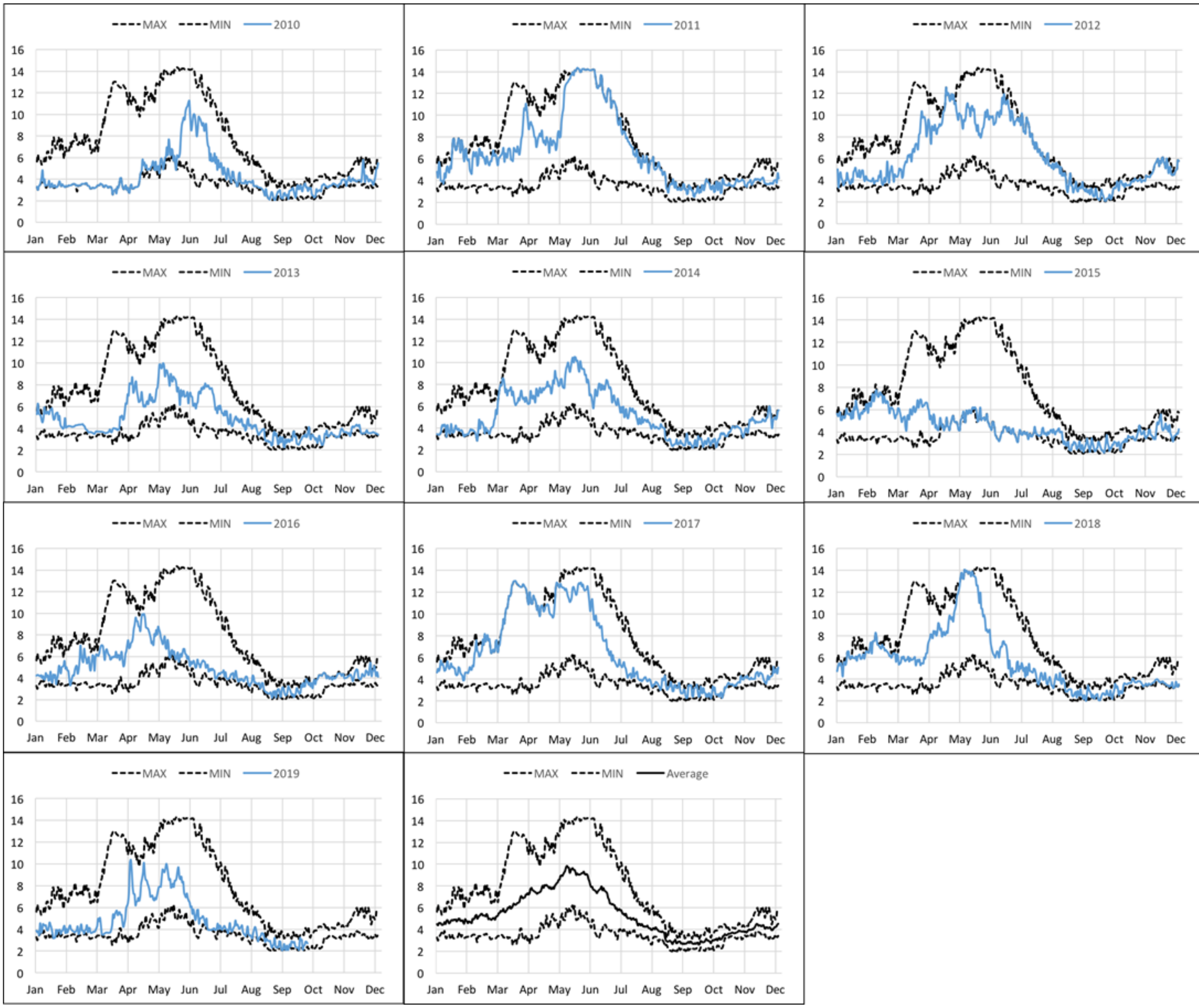

Figure 7. Daily water discharge (m^3/s) at Bonneville Dam. Panels show individual years between 2010-2019 (blue lines) and the daily max and min for all years combined. Vancouver gage web page shows recent flood stage years - https://water.weather.gov/ahps2/crests.php?wfo=pqr&gage=vapw1&crest_type=recent

Based on Figure 7 an NMDS plot of differences in river discharge and river temp between years, hydrologic conditions or cumulative discharge of the Mainstem since 2010 were classified into four categories (Table 4). The results presented in this report have compared the evolution of abiotic and biotic conditions over the monitoring years and differentiated the results between the tabulated categories. Any additional or modified freshet categories have been included in respective sub-sections.

Table 4: Classification of Monitoring years according to cumulative river discharge during the spring freshet between 2010-2019

Year	Cumulative River Discharge (m^3 x 10^{10}) for May – Aug[2]	River Temperature[1] (°C)	Classification
2019	5.9	85	**dry**
2018	7.8	79	**mid/wet**
2017	8.7	78	**wet**
2016	5.5	85	**dry**
2015	4.7	102	**very dry**
2014	7.3	86	**mid**
2013	6.7	84	**mid**
2012	9.2	59	**wet**
2011	10.4	59	**wet**
2010	6.3	47	**mid**

[1]River temperature: Number of days days that the river temp was >19 °C May –Sep
[2]Freshet: cumulative river discharge (m^3 x 10^{10}) for May – Aug. Also referred to as “Freshet condition” in this report

Wallacut Slough

Project Description and Ongoing Management Actions

Wallacut Slough is a restoration site located in Bakers Bay, near the City of Ilwaco in Washington. In 2014, Ilwaco Slough, a long-term EMP site, was chosen as a nearby reference site for ongoing monitoring and comparisons.

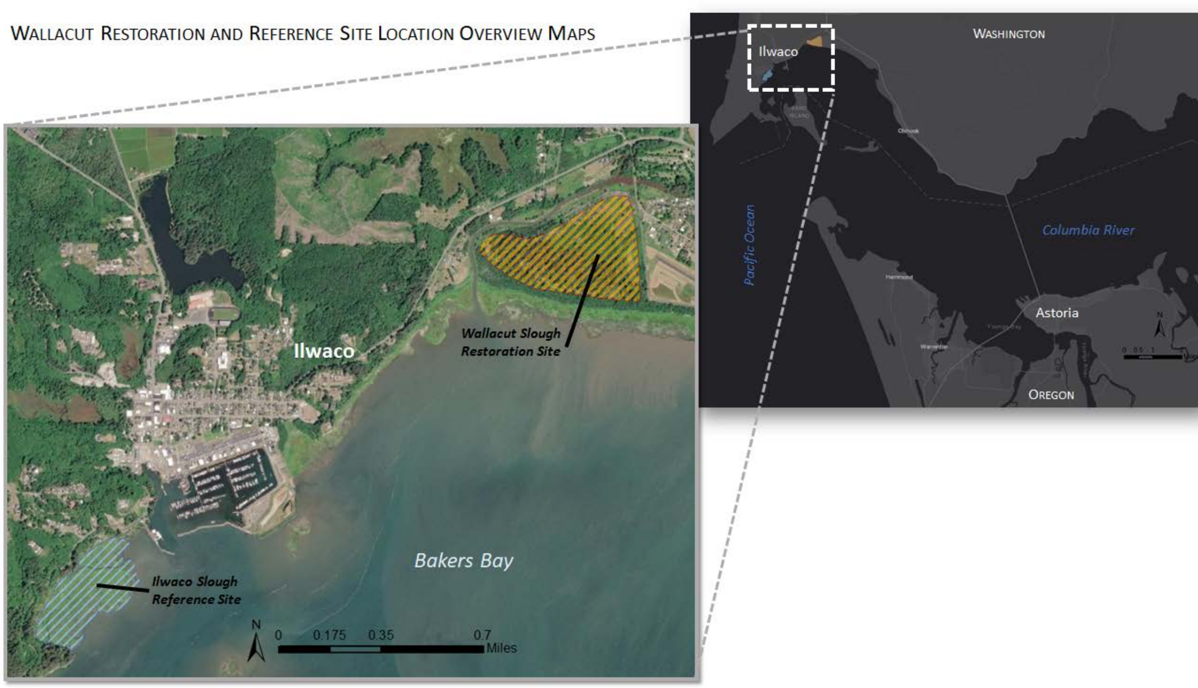

Figure 8: Overview map of Wallacut Slough Restoration Site Location and Ilwaco Slough Reference Site Location.

In 2016 tidal influence to the Wallacut Slough network was restored through the removal of barriers throughout the system (Table 2). Additional channel enhancements were conducted in areas to expand channel density and access to wetland habitat.
Project goals as defined in the SM2 (Johnson et al. 2018):

"Removing the levee and filling the borrow ditch will increase hydrologic connectivity during the tidal cycle and increase the spatial extent of inundation in the wetland. The restoration of a more natural tidal cycle will help restore ecosystem function by supporting a diverse native plant community, improving nutrient cycling, and increasing quantity and quality of off-channel habitat for aquatic species."

Two areas within the site received focal plant community monitoring, one area was located at the "Mouth" of the site near a channel re-connection and the other was located in an area in the "Upper" portion of the reconnected channel (Figure 9).

Figure 9: Map of plant community monitoring areas and the location of water surface elevation (WSE) data loggers at the Wallacut Slough Restoration Site.
During the restoration, the area near the **Mouth** of the channel was heavily impacted by grading and removal of levee materials; after restoration, this area was also targeted for non-native herbicide treatments in the spring of 2019. The area monitored in the **Upper** portion of

the restored channel received only minimal impacts during restoration and no herbicide treatments.

Plant Community Results
Sampling Overview
Wallacut Slough plant community results are reported by summarizing the results across the two major sampling areas: Wallacut Mouth and Wallacut Upper (Figure 9, Figure 11). Wallacut Mouth is located in an area that received grading and construction impacts during restoration and was targeted for non-native plant community herbicide treatments in the spring of 2019. Wallacut Upper is located just a few meters up the restored channel from the Mouth and was only minimally impacted during construction (Figure 9, Figure 11). A similar number of plots where sampled across each sampling area and these are compared to sampling conducted at the reference site, Ilwaco Slough (Table 5, Figure 8). While Ilwaco Slough is an excellent reference site in terms of proximity and general hydrology (Figure 8), located very nearby and experiencing similar tidal conditions, it is not a perfect match. The mismatch in site conditions is mainly due to Ilwaco Slough being composed of primarily low marsh elevations and plant communities and Wallacut Slough, in our monitoring areas, being composed of mid-high marsh elevations (Figure 10). During year five (2021) monitoring, we plan to seek out higher marsh elevations within Ilwaco Slough for monitoring, to improve this comparative reference site analyses. Given the differences in site elevations, extra care should be taken when interpreting comparative results.

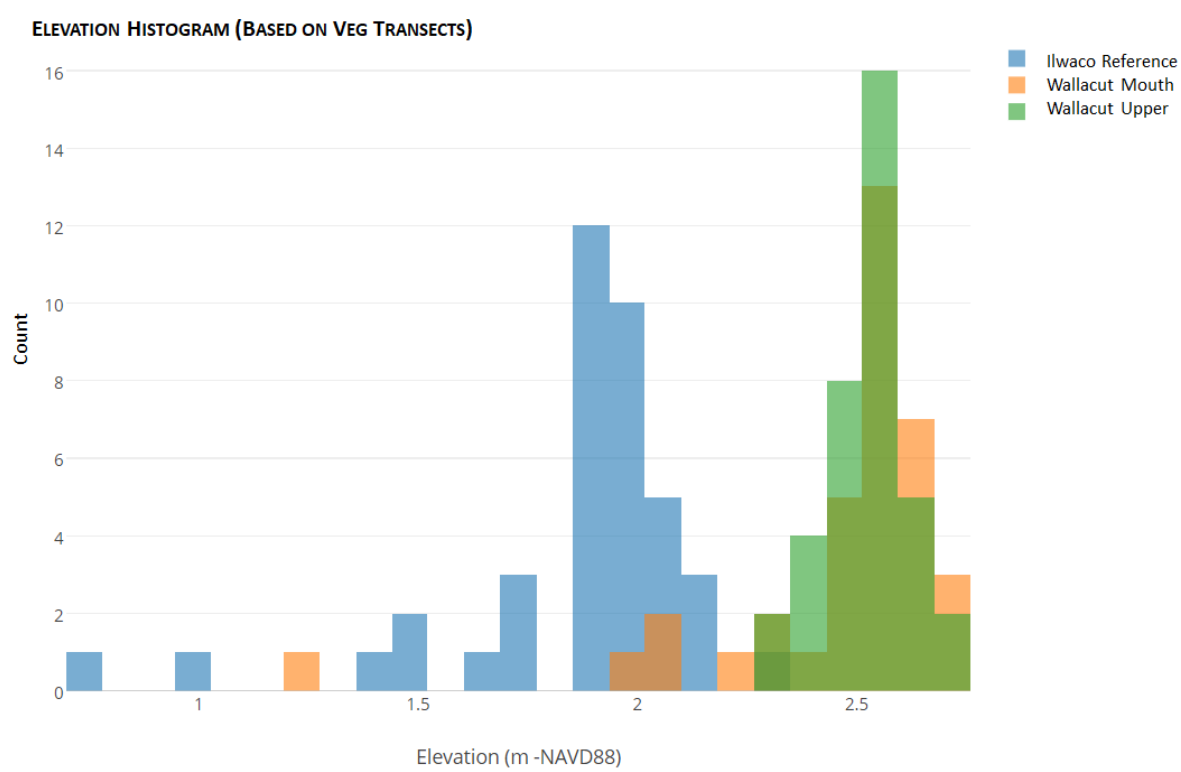

Figure 10: Wallacut Slough Restoration and Reference Site Elevation Distribution Histogram (2019).

Table 5: Wallacut Slough Restoration and Reference Site Number of Samples Plots Surveyed and Mean (±SD) Elevation, and Plant Community Relative Cover (%) Summarized by Years Post-Restoration.

Mean Relative Cover (%)	Monitoring Areas	2014		2017		2019	
		Pre		Year 1		Year 3	
		Mean	±SD	Mean	±SD	Mean	±SD
Number of sampling plots	Reference	40		40		40	
	Wallacut Mouth	36		36		36	
	Wallacut Upper	36		36		36	
Elevation, m, NAVD88	Reference	2.0	0.2	1.9	0.2	1.9	0.3
	Wallacut Mouth	2.5	0.2	2.5	0.2	2.5	0.3
	Wallacut Upper	2.5	0.3	2.5	0.2	2.5	0.1
Bareground (%)	Reference	10.7	18.8	7.0	14.0	9.2	18.2
	Wallacut Mouth	7.0	13.3	1.8	5.4	6.7	12.0
	Wallacut Upper	4.2	11.9	1.3	4.4	0.2	0.8
Standing dead (%)	Reference	0.0	0.0	5.9	11.5	3.2	5.9
	Wallacut Mouth	0.1	0.8	1.4	4.9	25.6	26.2
	Wallacut Upper	0.1	0.8	3.5	11.4	16.1	22.1
Native (%)	Reference	91	15	79	21	85	15
	Wallacut Mouth	41	28	32	32	41	31
	Wallacut Upper	44	22	69	19	83	16
Non-native (%)	Reference	9	15	21	21	15	15
	Wallacut Mouth	57	30	68	32	53	28
	Wallacut Upper	53	24	31	19	17	16

Trajectories: Native and Non-native Dominant Species

In year three post-restoration (2019) we found a mix of native and non-native plant community recovery across the sampling areas (Figure 11, Figure 12, Table 5). Sampling plots located at the Mouth were dominated by a mix of bare ground, standing dead grasses, and living non-native reed canarygrass, *Phalaris arundinacea,* and creeping bentgrass, *Agrostis stolonifera* (Figure 13, Table 6). In contrast, the plant community identified in the Upper sampling area was composed primarily of native pacific silverweed, *Argentina egedii ssp. Egedii,* and baltic Rush, *Juncus arcticus,* with very little bareground or standing dead observed (Figure 13 and Figure 14, Table 6).

Since 2014 (pre-restoration) dramatic shifts in native and non-native plant community composition have been observed in the Upper Wallacut monitoring area, with significant increases in native relative plant cover and decreases in non-native relative plant cover (Figure 12, Table 5). Very little changes were observed pre and one-year post restoration in the Mouth monitoring area. However, since 2017, one-year post-restoration, non-native plant cover, primarily *P. arundinacea,* has declined dramatically, and correspondingly both levels of standing dead vegetation and bare ground in the Mouth monitoring area have also increased since 2017

(Figure 13, Table 6 and Table 7). These shifts in plant community composition observed in the Mouth monitoring area are likely a direct result of herbicide treatments targeting *P. arundinacea* and other non-natives on the site in the spring of 2019, which took place before this monitoring occurred. Additionally, this area received greater construction impacts exposing bare mineral soil and likely causing soil compaction, further slowing the recovery of native plant communities compared to the unimpacted Upper monitoring area.

Comparisons of across monitoring areas indicate that they share similar physical characteristics both in elevation, hydrology, and soil biogeochemistry. Additionally, shifts in native and non-native species richness indicate that native species are re-populating both monitoring areas and these native plant communities will likely recover given time and with a break from herbicide applications

Figure 11: Wallacut Slough Restoration Site Plant Community Monitoring Results (2019), Native and Non-native Species Dominance Across Sampling Plots. More plant community details can be found in Table 5 and Table 6.

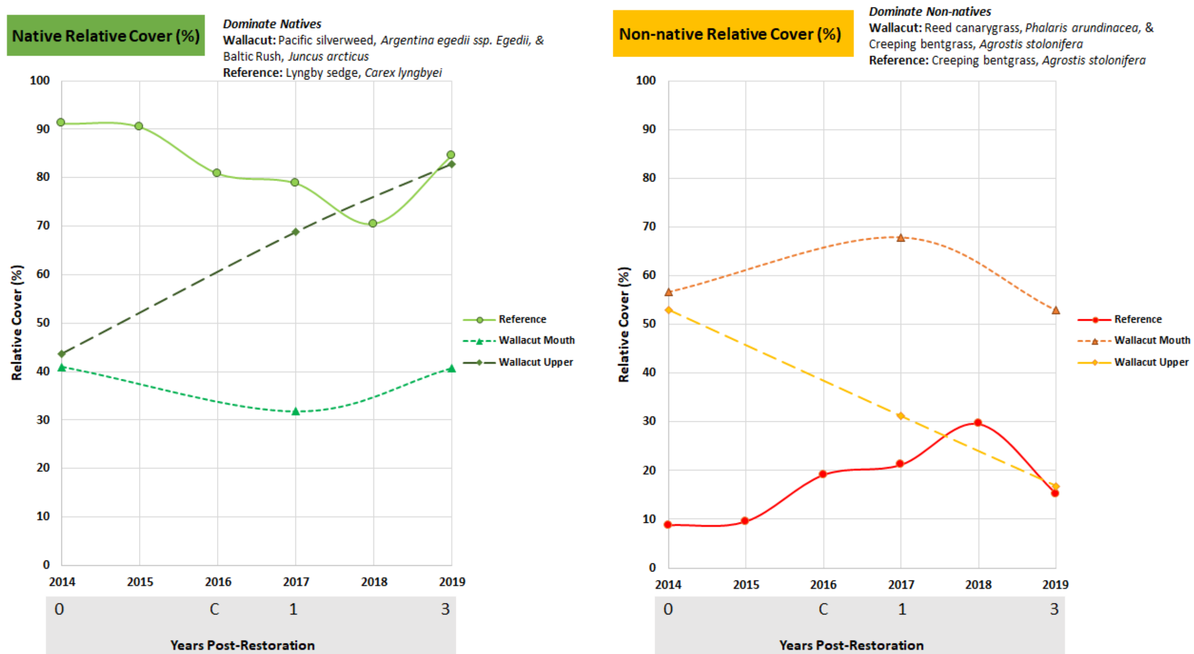

Figure 12: Wallacut Slough Restoration and Reference Site Native and Non-native Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 5 and Table 6.

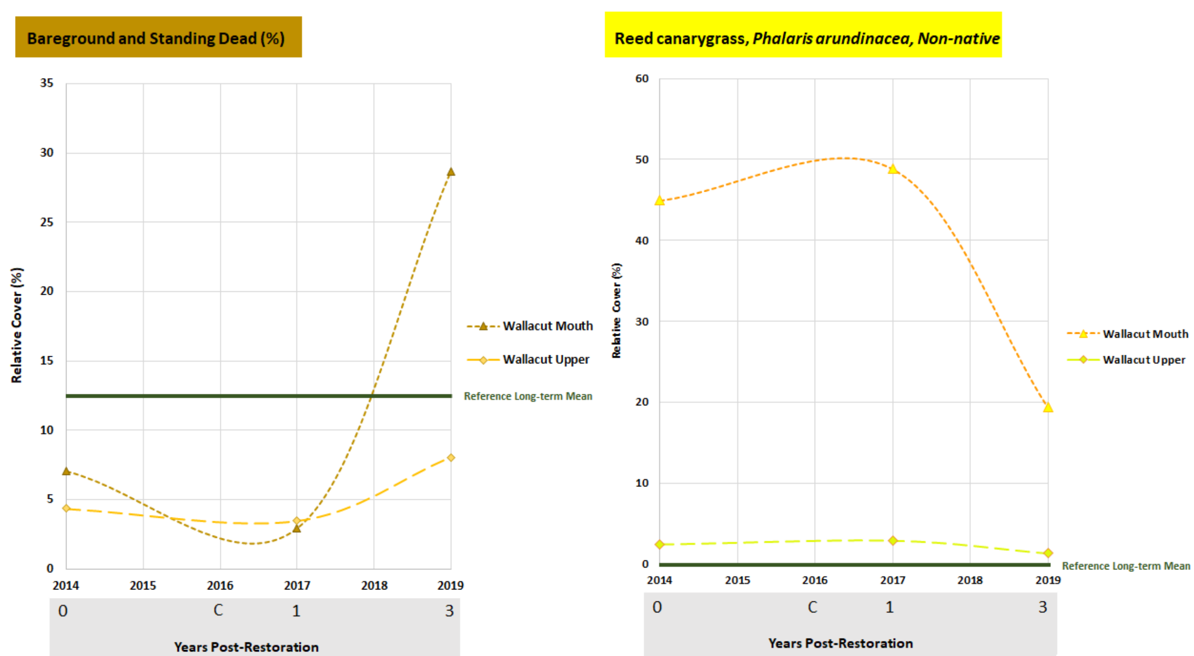

Figure 13: Wallacut Slough Restoration and Reference Site Bareground & Standing Dead, and Non-Native, Reed Canarygrass, *Phalaris arundinacea,* Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 5 and Table 6.

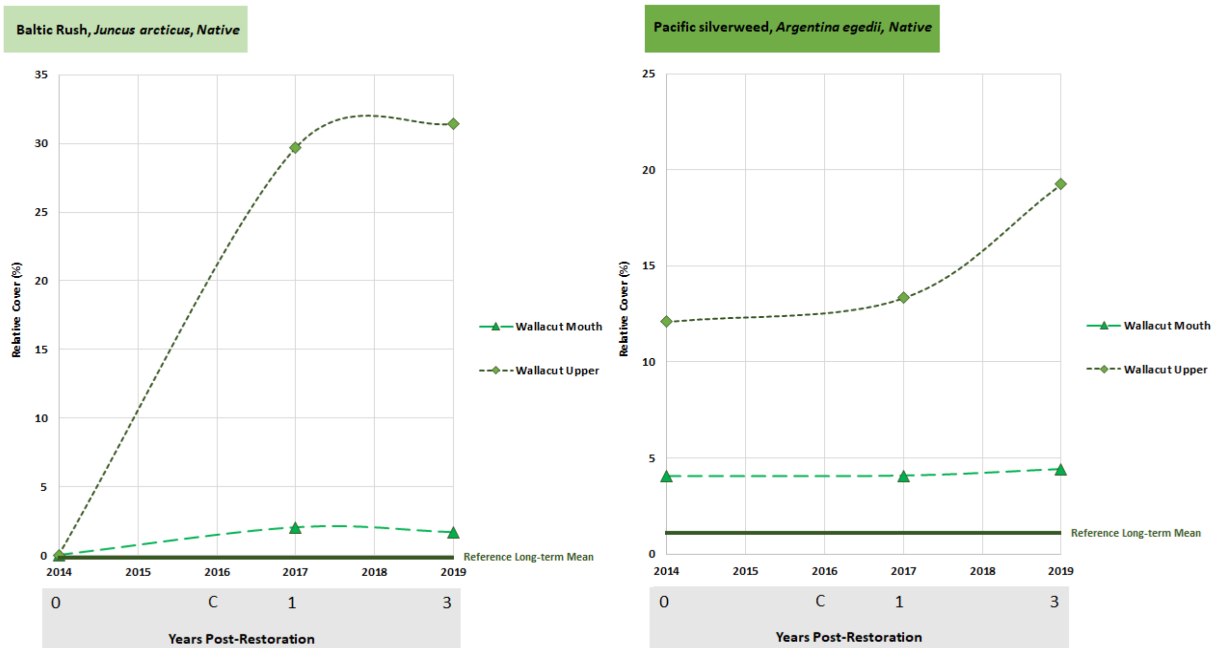

Figure 14: Wallacut Slough Restoration and Reference Site Native, Pacific silverweed, *Argentina egedii ssp. Egedii,* & Baltic Rush, *Juncus arcticus,* Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 5 and Table 6.

Table 6: Wallacut Slough Restoration and Reference Site Mean (±SD) Dominant Plant Species Relative Cover (%) Summarized by Years Post-Restoration.

Dominate Species		Monitoring Areas	2014		2017		2019	
			Pre		Year 1		Year 3	
			Mean	±SD	Mean	±SD	Mean	±SD
Non-native	Reed canarygrass, *Phalaris arundinacea*	Reference	0.0	0.0	0.0	0.0	0.0	0.0
		Wallacut Mouth	44.9	30.7	48.8	41.0	19.4	23.2
		Wallacut Upper	2.4	12.5	2.9	13.6	1.3	7.5
	Creeping bent grass, *Agrostis stolonifera*	Reference	6.0	8.0	18.4	17.8	10.8	9.5
		Wallacut Mouth	2.0	3.0	14.6	22.2	3.2	8.4
		Wallacut Upper	2.9	4.4	21.5	15.9	12.1	13.3
	Common rush, *Juncus effusus*	Reference	0.0	0.0	0.0	0.0	0.0	0.0
		Wallacut Mouth	4.1	8.5	0.4	1.8	0.0	0.0
		Wallacut Upper	34.4	22.5	0.5	3.1	0.0	0.1
Native	Pacific silverweed, *Argentina egedii ssp. Egedii*	Reference	1.8	6.2	0.6	2.6	0.8	3.3
		Wallacut Mouth	4.0	11.6	4.1	12.3	4.4	16.3
		Wallacut Upper	12.1	16.1	13.3	16.7	19.2	21.8
	Lyngby sedge, *Carex lyngbyei*	Reference	52.3	35.1	43.1	28.5	41.3	31.7
		Wallacut Mouth	2.7	14.2	3.4	16.1	2.5	10.0
		Wallacut Upper	1.1	4.5	0.6	2.1	0.0	0.1

Dominate Species		Monitoring Areas	2014		2017		2019	
			Pre		Year 1		Year 3	
			Mean	±SD	Mean	±SD	Mean	±SD
	Slough sedge, *Carex obnupta*	Reference	**0.0**	0.0	**0.0**	0.0	**0.0**	0.0
		Wallacut Mouth	**0.0**	0.0	**1.8**	6.4	**0.0**	0.0
		Wallacut Upper	**3.7**	14.7	**1.6**	6.8	**0.4**	1.7
	Baltic rush, *Juncus balticus*	Reference	**0.0**	0.0	**0.0**	0.0	**0.0**	0.1
		Wallacut Mouth	**0.0**	0.0	**2.0**	8.5	**1.7**	4.2
		Wallacut Upper	**0.0**	0.0	**29.7**	19.8	**31.4**	16.8

Trajectories: Species Richness

Over the three-year time frame, overall total species richness (TSR) has increased at the Mouth site with a mean of 7.1 TSR identified per plot in 2019 compared to 5.1 TSR in 2014 (pre-restoration) and decreased slightly in the Upper site with 6.9 TSR in 2019 compared to 7.3 TSR in 2014. Overall these TSR numbers are significantly higher than those observed at the reference site, which had 5.1 TSR in 2019 and 4.2 in 2014 (Table 5). At the Mouth, the mean native and non-native species richness has increased post-restoration, accounting for this overall increase in TSR observed in 2019 (Table 5). The Upper site has had a slight decline in non-native species richness and a greater increase in native species richness across the pre-post time frame. The Mouth and Upper sites both have experienced increased levels of native species richness in 2017, Mouth - 3.3, Upper - 4.1, and 2019, Mouth – 3.8, Upper – 4.4, compared to pre-restoration conditions (2014), Mouth – 2.9, Upper – 3.8, and similar to those observed at the reference site in 2019, 4.1. Overall, these shifts in species richness across the restoration site are to be expected and are commonly observed in restored wetlands during plant community transitions from agricultural lands to native wetland plant communities (Kidd 2017).

Table 7: Wallacut Slough Restoration and Reference Site Mean (±SD) Species Diversity Metrics Summarized by Years Post-Restoration.

Mean Diversity	Monitoring Areas	2014		2017		2019	
		Pre		Year 1		Year 3	
		Mean	±SD	Mean	±SD	Mean	±SD
Total Species Richness	**Reference**	4.2	2.9	4.8	2.5	5.1	2.8
	Wallacut Mouth	5.1	2.5	5.6	2.8	7.1	3.6
	Wallacut Upper	7.3	1.7	6.5	1.3	6.9	2.3
Non-native Species Richness	**Reference**	0.8	0.6	0.9	0.5	1.0	0.6
	Wallacut Mouth	1.8	1.1	2.0	1.0	2.8	1.6
	Wallacut Upper	2.8	1.2	2.4	0.9	2.1	1.2
Native Species Richness	**Reference**	3.5	2.5	3.9	2.3	4.1	2.6
	Wallacut Mouth	2.9	1.7	3.3	1.9	3.8	2.5
	Wallacut Upper	3.8	1.5	4.1	1.2	4.4	1.5
	Reference	0.9	0.7	1.0	0.5	1.1	0.6

Mean Diversity	Monitoring Areas	2014		2017		2019	
		Pre		Year 1		Year 3	
		Mean	±SD	Mean	±SD	Mean	±SD
Shannon Diversity Index	Wallacut Mouth	0.9	0.5	0.7	0.5	1.0	0.5
	Wallacut Upper	1.3	0.3	1.3	0.3	1.1	0.2
Evenness Index	Reference	0.6	0.4	0.6	0.3	0.7	0.3
	Wallacut Mouth	0.6	0.2	0.4	0.2	0.5	0.2
	Wallacut Upper	0.7	0.1	0.7	0.1	0.6	0.1

Drone Imagery and Plant Community Modeling Results
When combined with ground survey elevation and plant community data, aerial drone survey data can be processed to develop robust large-scale plant community maps of wetland restoration sites. In 2019, LCEP collected drone RGB and near infrared imagery of Wallacut slough near the intensively monitored ground survey locations (Figure 9). Comprehensive plant community groupings we developed from ground survey data and used the aerial imagery and sensor data to model plant community distributions across the site. These map classifications include:

- Water – the flight was conducted on 7/29/2019, at 12:05 pm which was near high-tide (gage high tide was measured as 1.5 m at 12:35 pm, Astoria, Oregon), this can be easily observed in the classification imagery, with the channel and some low elevation wetland areas showing water.
- Native wetland matrix –defined as being dominated by a mix of the following native species: Pacific silverweed, *Argentina egedii ssp. Egedii*, Baltic rush*, Juncus balticus*, Lyngby sedge, *Carex lyngbyei*, Slough sedge, *Carex obnupta* etc.
- Dead vegetation and bareground - defined as being dominated by a mix of standing dead vegetation, primarily grasses, and bareground.
- Reed canarygrass mix – defined as being dominated by non-native grasses Reed canarygrass, *Phalaris arundinacea*, and Creeping bent grass, *Agrostis stolonifera*, this mix may also contain some native species, such as those species found in the Native Wetland Matrix, in addition to native shrubs, however it is generally dominated by the non-native grasses.
- Trees and shrub-scrub – defined as being dominated by native and mature pre-existing trees and shrubs on the site. Understory of this plant community is likely a mix of native and non-native species including Reed canarygrass*, Phalaris arundinacea*, and Creeping bent grass, *Agrostis stolonifera.*

The distribution of these classifications across the survey area can be seen in Figure 15. Total acres and abundance were calculated based on map resolution and the total number of cells classified into each grouping. Within the sampling area the classification was found to have >95% accuracy when compared to known vegetation on the ground.

Utilizing the digital surface model of site elevations, the elevation ranges and overall abundance of all mapped plant communities can be seen in Figure 16. Within the surveyed area we found approximately 32% of the area (2.7 acers) was dominated by pre-existing riparian forest and shrub-scrub wetlands, 29% (2.5 acers) was composed of a native wetland plants, 19% (1.6 acers) was composed of standing dead grasses and bareground, 15% (1.2 acers) was composed of a reed canarygrass mix, and 5% (0.4 acers) was composed of open water (Figure 15). Standing dead grasses and bareground were found primarily in the lower site elevations < 3 meters. However, this also overlapped with both the elevations where the reed canarygrass mix and native wetland plant communities were identified (2.2-3.6 meters) (Figure 16, Figure 17). Based on the observed wetland plant community distributions across the elevation gradient, it is anticipated that current (2019) areas that are bareground and dead vegetation located in areas less than 3.1 meters in elevation will transition into primarily native wetland plant communities (should increase by 0.8 acres, 9%), while those located in areas 3.1 meters and above in elevation will become a mixed native and reed canarygrass dominated plant communities (increase by 0.8 acres, 10%) by year five post-restoration (Figure 18). This modeling does not account for ongoing planting efforts, which may successfully transition the higher elevation areas of the site from a non-native plant matrix to native shrub-scrub and riparian forest.

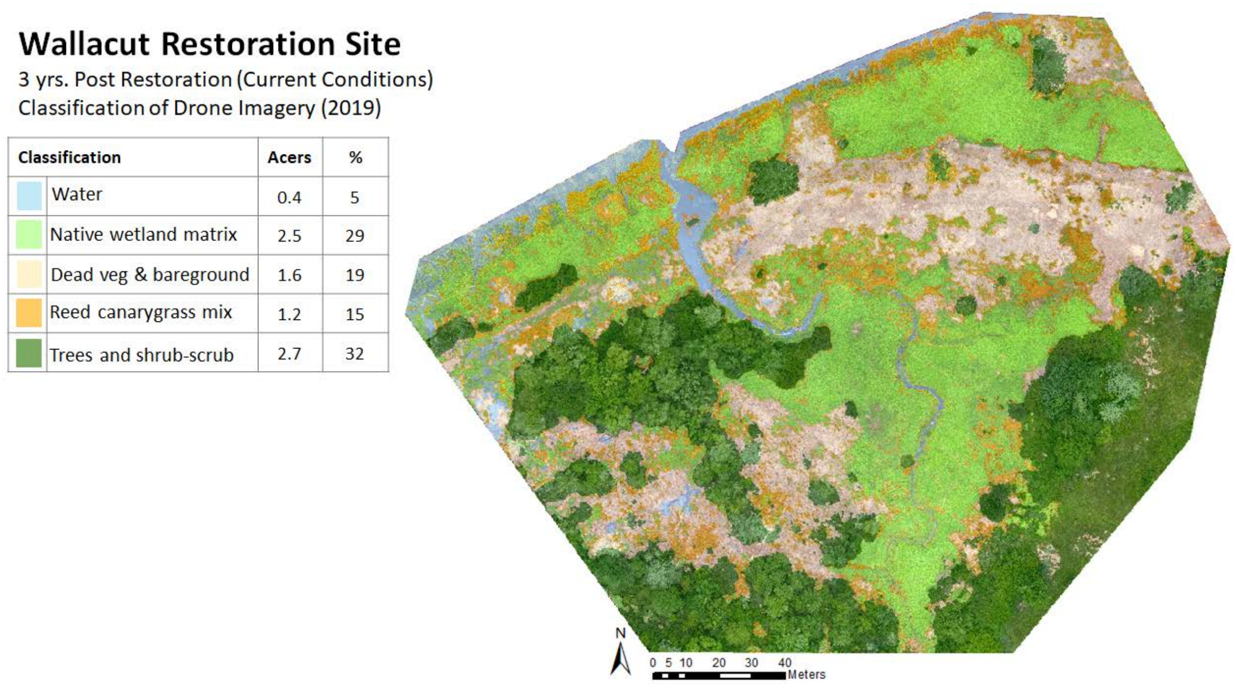

Figure 15: Wallacut Slough Restoration Site Dominate Plant Community Distributions and Abundance in 2019. These data were extracted from the modeled plant community distributions developed using 2019 drone senor data, resolution is less than 10 cm x 10 cm for each map cell.

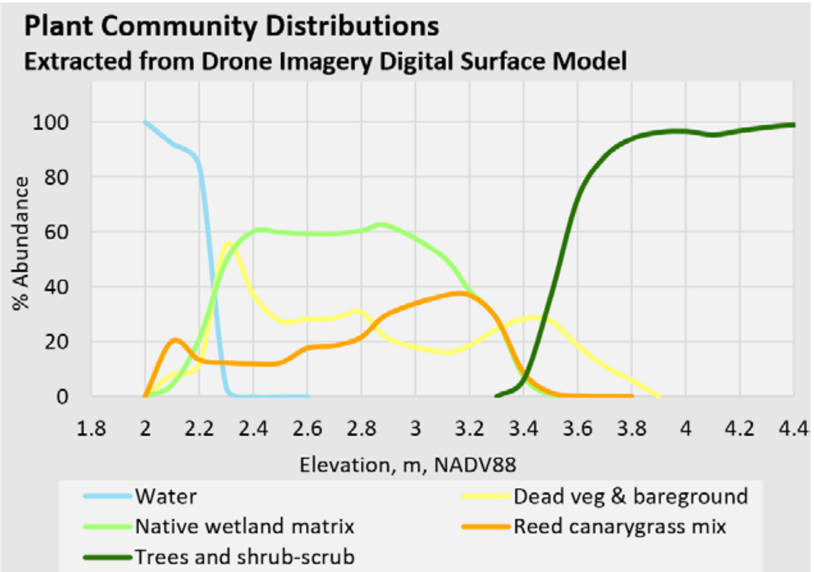

Figure 16: Wallacut Slough Restoration Site Dominate Wetland Plant Community Relative Abundance (%) Across All Classification Types Along the Elevation (m, NAV88) Gradient. These data were extracted from the modeled plant community distributions (Figure 15) and digital surface model of elevations produced from 2019 drone senor data.

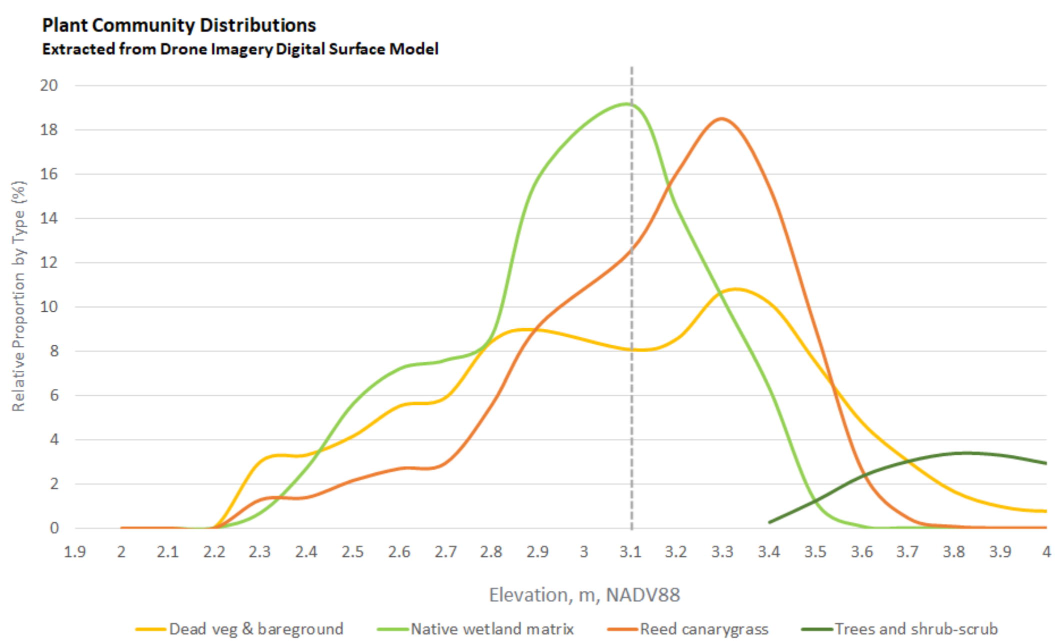

Figure 17: Wallacut Slough Restoration Site Dominate Wetland Plant Community Proportion (%) for Each Elevation (m, NAV88) Class (every 0.1 meters) (like a histogram for each community classification). These data were extracted from the modeled plant community distributions (Figure 15) and digital surface model of elevations produced from 2019 drone senor data.

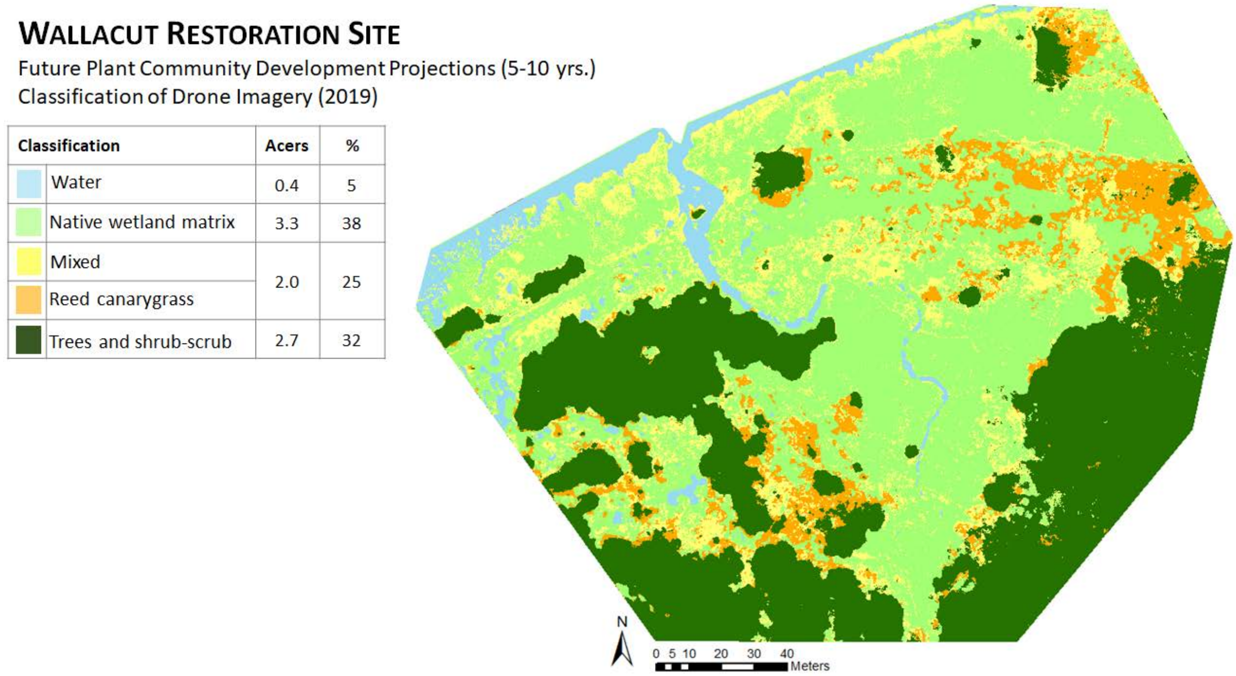

Figure 18: Wallacut Slough Map of Future Plant Community Development Predictions for 5-10 years Post-Restoration. Predictions based on existing plant community distributions and elevations observed in 2019 (Figure 15 & Figure 16). It is anticipated that current (2019) areas that are bareground and dead vegetation will transition into Native wetland plant communities (increase by 0.8 acres, 9%), Mixed Native and Non-native plant communities and Reed canarygrass dominated plant communities (increase by 0.8 acres, 10%).

Soil Monitoring Results

Re-introducing flooding to previously drained agricultural soils has significant impacts on soil characteristics and creates the biogeochemical template needed for wetland plant community and habitat restoration. Additionally, restoration actions such as scrapping down the topsoil to created lower elevations and channels can alter soil composition, revealing mineral soils and causing compaction. These restoration actions and soil manipulations can further impact plant community recovery. Monitoring soil conditions pre- and post-restoration can provide insight into the mechanisms causing restoration success and failure and LCEP has recently begun consistently monitoring soil conditions as part of their Level 2 AEMR. Due to the recent adoption of these monitoring methods we only have year three post-restoration data (2019) for Wallacut's soil conditions. However, while these soil data are limited, they still provide valuable insight into the site conditions and how they compare across monitoring areas and to the reference site.

Soil salinity can be an important indicator of wetland restoration success along the oligohaline (>0.5 ppt) to saltwater (35 ppt) marsh gradient, where saline soil conditions are restored through the re-introduction of tidal saltwater exposure. Wallacut and Ilwaco Slough are located within the hydrogeomorphic reach A, within the estuary saltwater intrusion zone (Simenstad et al. 2011). Restoration of soil salinity levels can be a critical part of native wetland plant community recovery, with the common invasive reed canarygrass, *Phalaris arundinacea,* found to be intolerant of soil salinities > 3 ppt (Kidd 2017).

The year three post-restoration mean (saturated) soil salinity was found to be significantly higher at the reference site, 5.9 ppt, compared to both the Mouth, 2.2 ppt, and Upper, 1.1 ppt, monitoring areas (Figure 19, Table 8). The soil salinity levels of the reference site, Ilwaco Slough, are characteristic of mesohaline (5.0-18.0 ppt) tidal wetlands, while at year three post-tidal restoration the Wallacut Slough monitoring areas are more characterized by oligohaline soil conditions (0.5-5.0 ppt). Comparatively, soil pH levels followed a similar trend, the highest soil pH levels found at the reference site at 6.6, compared to slightly lower pH levels identified at the Mouth, 6.2, and Upper, 5.9, monitoring areas (Figure 20, Table 8). Elevation differences across the reference and restored monitoring areas may influence these soil salinity and pH observations, the reference site being composed of primarily low marsh while the restored wetland is mid-high marsh in elevation (Table 8). Kidd (2017) found that the recovery of wetland soil salinities and pH levels were significantly influenced by wetland elevation, with low marsh zones (exposed to more tidal flooding) recovering reference salinity and pH levels more readily than high marsh zones (Kidd 2017).

Wetland elevation can also influence soil ORP conditions, higher marsh zones, exposed to less tidal flooding have higher greater oxygen levels than low marsh zones (Seybold et al. 2002, Kidd 2017). Correspondingly, both Wallacut monitoring areas also reported greater soil ORP conditions (more oxygen) than the reference site (Table 8). The mean ORP at the Mouth monitoring area being 252 mV, the Upper monitoring area being 275 mV, and the reference site being 111 mV (Figure 21).

Reed canarygrass (*Phalaris arundinacea)* was only found growing as a dominant species above 2.5 meters in elevation and in soil with ORP > 200 mV, and primarily in the more disturbed soils of the Mouth monitoring area (Figure 22). Otherwise, no clear zonation of plant communities or soil conditions were identified within the restoration site indicating wetland soil biogeochemistry conditions are still developing (Kidd 2017). Further soil monitoring as time passes should provide insight into site recovery and plant community development. The Mouth monitoring area may suffer from getter soil compaction, lower organic matter, and nutrient content from construction impacts, resulting in slower wetland plant community recovery compared to the Upper monitoring area (Wisheu and Keddy 1991, Roman and Burdick 2012, Spencer and Harvey 2012).

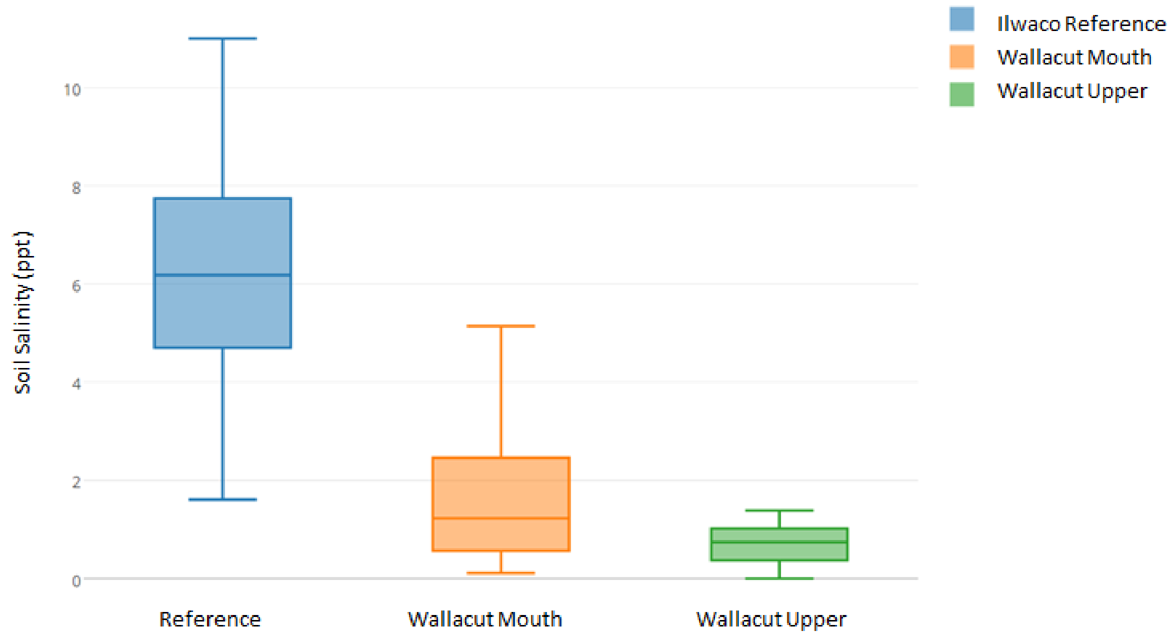

Figure 19: Boxplot of Wallacut Slough Restoration and Reference Sites (2019) soil salinity (ppt) conditions (low tide). A summary of all soil data can be found in Table 8.

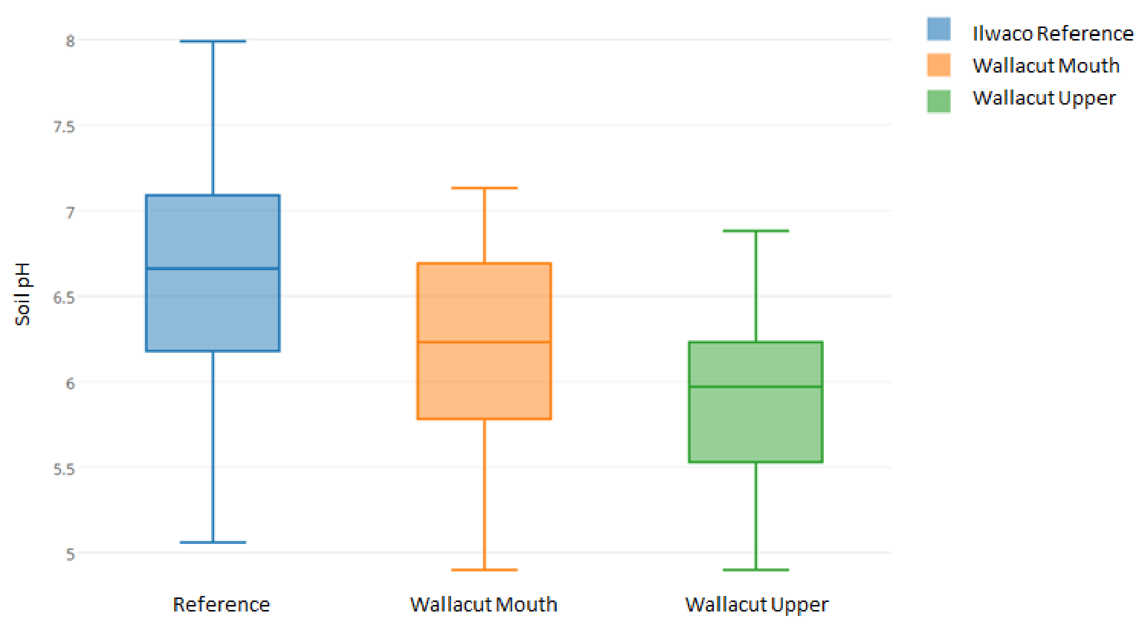

Figure 20: Boxplot of Wallacut Slough Restoration and Reference Sites (2019) soil pH conditions (low tide). A summary of all soil data can be found in Table 8.

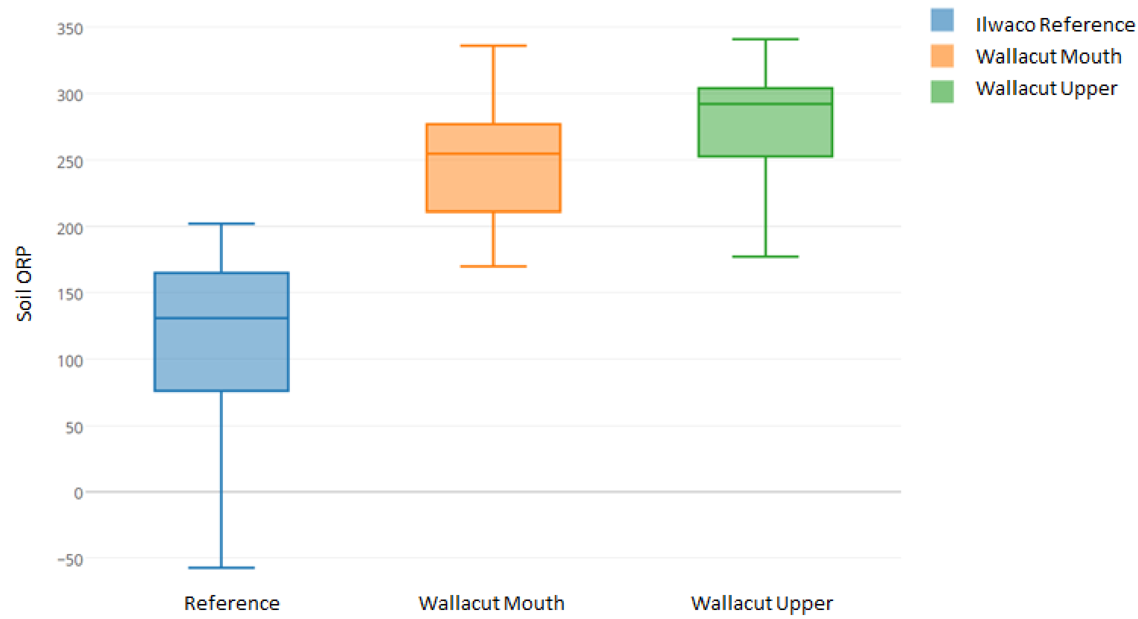

Figure 21: Boxplot of Wallacut Slough Restoration and Reference Sites (2019) soil ORP conditions (low tide). A summary of all soil data can be found in Table 8.

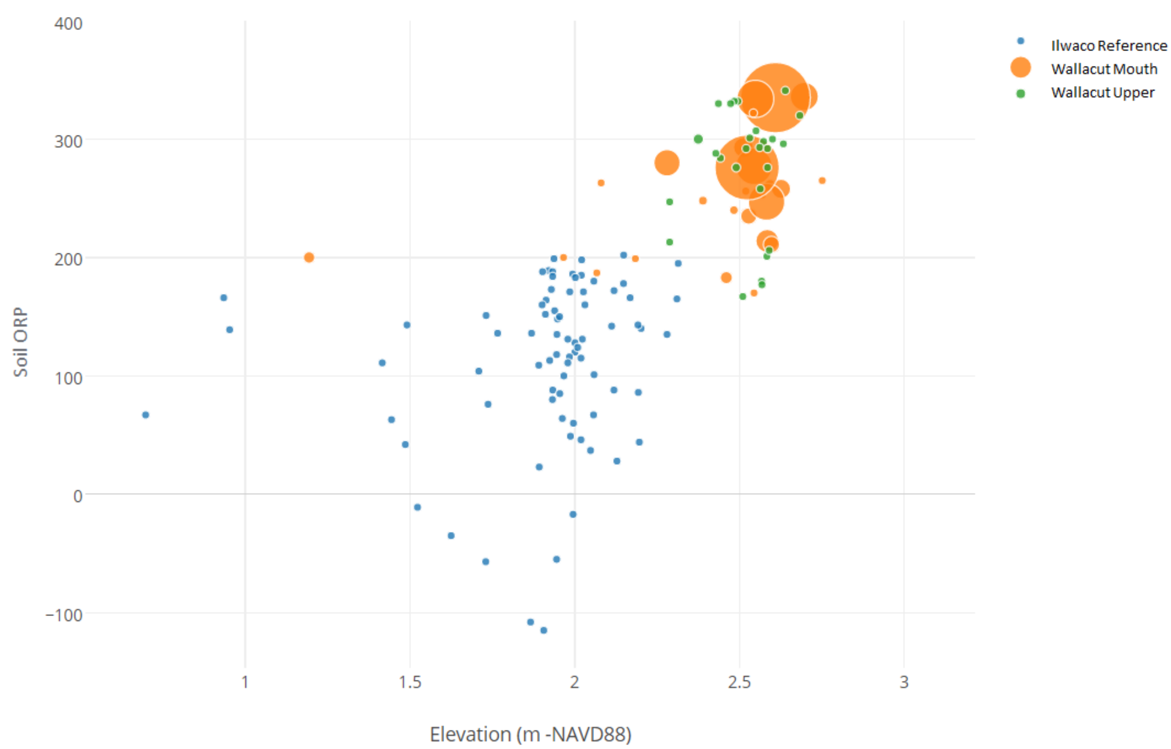

Figure 22: Wetland Soil ORP vs Elevation (m-NAVD88) of all sample plots, with graduated symbols indicating the relative cover (%) of Reed canarygrass, *Phalaris arundinacea,* for Wallacut Slough Restoration and Reference Sites (2019). For more information on plant

community abundance by monitoring area see Table 6 and a summary of all soil data can be found in Table 8.

Table 8: Wallacut Slough Restoration and Reference Site Mean (±SD) Soil Conditions, 5 years post-tidal reconnection (2019).

Soil Conditions	Reference		Wallacut Mouth		Wallacut Upper	
	Mean	±SD	Mean	±SD	Mean	±SD
Sample number	77		26		29	
Elevation (m - NAVD88)	1.9	0.2	2.5	0.2	2.5	0.2
Salinity (ppt)	5.9	2.4	2.2	3.2	1.1	1.1
pH	6.6	0.7	6.2	0.6	5.9	0.5
ORP	111	71	252	48	275	52

Water Surface Elevation & Water Temperature Results

Water Surface Elevation (WSE) for Wallacut slough and the adjacent Wallacut River is available from 2014 to 2019. Occasionally, sensor failure occurred. Due to sensor errors, pre-restoration data is unreliable and hence has not been included in this report. Sensor failure occurred at Ilwaco Slough in 2019, hence, year three post restoration hydrological comparisons have been made only between the slough and the adjacent river. Summary statistics of hydrological patterns for the slough, the adjacent river and Ilwaco reference are presented in Table 9 for construction and post-restoration period (2016 – 2019). Sensor Elevations (m, NAVD88) have also been included for each site.

Table 9: Hydrologic Summary Statistics for Wallacut Slough, adjacent Wallacut River and nearby Ilwaco Reference site for year 1 to year 5 post-restoration. All metrics are in meters, relative to the North American Vertical Datum of 1988 (NAVD88). MWL = mean water level; MHHW= mean higher high water, Tidal Range is Mean Tidal Range. Full comparative hydrographs are in Appendix B: Site Hydrographs.

Restoration Year	Monitoring Location	Sensor Elevation	MWL	MHHW	Tidal Range	Max WSE	Date of Max WSE	Period of Record	Days
2019 (Year 3)	**Wallacut Slough**	1.7	2.0	2.5	0.6	3.2	Jan 22	Jan-Dec	365
	Wallacut River	0.6	1.4	2.4	1.8	3.3	Jan 22	Jan-Dec	357
	Reference	0.7	NA	NA	NA	NA	NA	NA	NA
2018 (Year 2)	**Wallacut Slough**	1.7	2.0	2.5	0.6	3.2	Dec 20	Jan-Dec	350
	Wallacut River	0.6	1.5	2.5	1.7	3.3	Dec 20	Jan-Dec	326

Restoration Year	Monitoring Location	Sensor Elevation	MWL	MHHW	Tidal Range	Max WSE	Date of Max WSE	Period of Record	Days
	Reference	0.7	1.5	2.4	1.4	3.1	Mar 02	Jan-Nov	311
2017 (Year 1)	**Wallacut Slough**	1.7	2.0	2.6	0.7	3.5	Feb 09	Jan-Feb, Apr-Dec	249
	Wallacut River	0.6	0.9	1.9	1.7	2.5	Dec 02	May-Dec	211
	Reference	0.7	1.5	2.4	1.5	3.3	Feb 09	Jan-Feb, Aug-Dec	216
2016 (Constructed)	**Wallacut Slough**	1.7	2.0	2.6	0.8	3.1	Dec 14	Nov-Dec	45
	Wallacut River	NA	NA	NA	NA	NA	NA	NA	NA
	Reference	0.7	1.4	2.4	1.5	3.2	Oct 15	Aug-Dec	147

NA= Not available

The hydrology of Wallacut slough at year three of post-restoration has been statistically compared to the adjacent Wallacut river only. In 2019, a stronger linear relationship was observed between the MHHWs of the two sites ($R2= 0.79$, $p=0.0001$) than the MWLs ($R2= 0.68$, $p=0.015$). This may be due to the difference in elevations of sensor deployment. The influence of elevation differences is also observed in the hydrographs (Wallacut Slough) of the site, where even though the crests of the graphs overlap, there still is a slight mismatch in the range of variations. The linear relationships between the metrics suggest that the hydrology at Wallacut Slough successfully emulates that of the adjacent river.

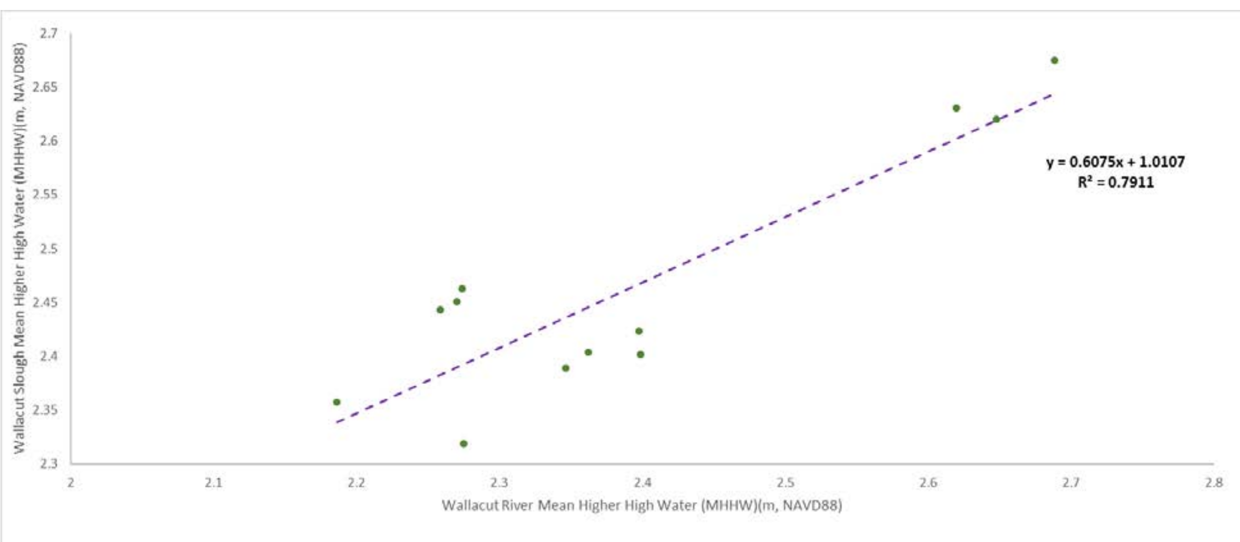

Figure 23: Scatterplot showing a linear relationship between MHHWs of Wallacut Slough and Wallacut River for 2019 ($R2= 0.79$, $p=0.0001$).

The influence of yearly freshet patterns is not observed at Ilwaco Slough or Wallacut Slough. Ilwaco is tidally dominated, with a tidal range of 1.4 m to 1.5 m throughout the monitoring period (since 2011) (Rao et.al 2020). In order to compare the hydrological patterns of the slough and river with Ilwaco, September to November months of the post-restoration period between 2017 to 2018 were selected, as these months have equal datapoints across all three sites and across years. A multiple regression model showed that a significant linear relationship between the MHHWs of Ilwaco slough and Wallacut Slough during post-restoration (adjusted R2= 0.829, p=0.015), however, there were no significant relationships found between the hydrological patterns of the slough with the adjacent river for years 2017 and 2018. This is likely be a relic of data quality issues that were not captured in post-processing.

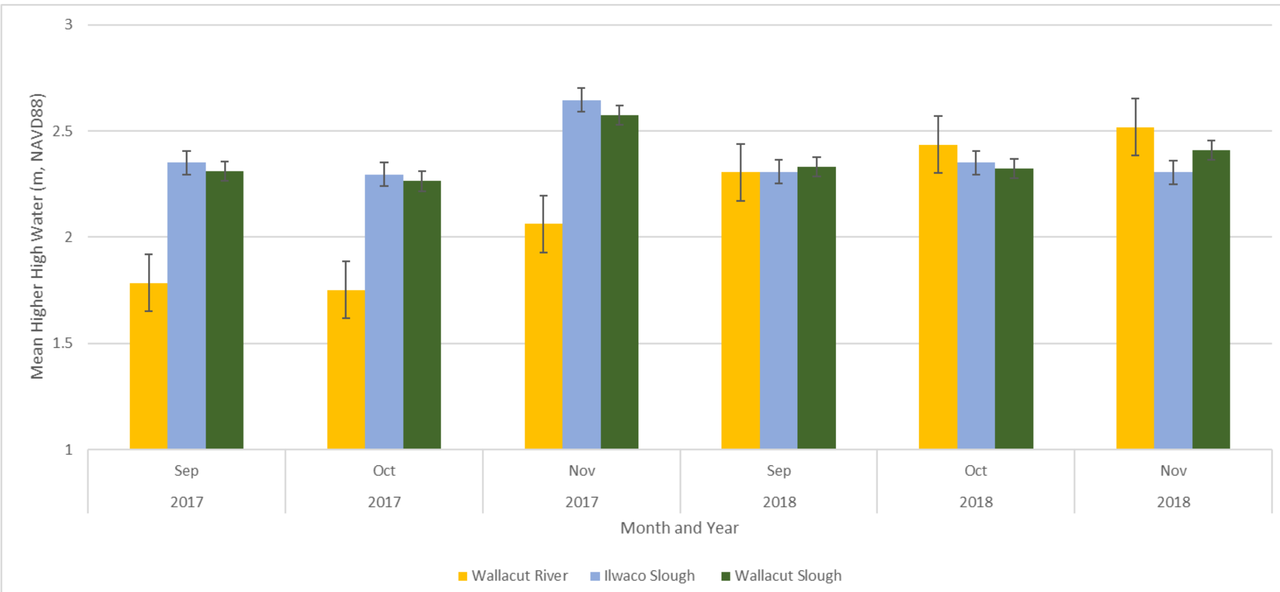

Figure 24: Bar graph representing the evolution of Wallacut slough and River hydrology in comparison to the hydrology of Ilwaco Slough (Reference) for years one (2017) and two post-restoration (2018). MHHWs are used instead of MWL to avoid interference from differences in sensor elevations.

Water temperature data were still under analysis at the time of writing this report and will be reported out on during the 2020 synthesis report.

Sediment Accretion and Erosion Monitoring

Sediment accretion is an important indicator of restoration progress. Two pairs of PVC Stakes were placed one meter apart and driven into the sediment and leveled. The distance from the plane at the top of the stakes to the sediment surface is measured as accurately as possible every 10 cm along the one-meter distance. The stakes were installed in 2016 and were measured annually. Accretion or erosion rates were calculated as differences between annual averages of 11 measurements along the 1m distance between PVC stakes. Accretion and

Erosion rates were then compared to Ilwaco Slough. In 2019, accretion and erosion rates ranged from -1 to 3cm/year since construction (Table 10).

Table 10: Sediment Accretion and Erosion annual rates for Wallacut Slough and Ilwaco Slough reference site.

Site Bench	Elevation (m, NAVD88)	Annual Summary	Pre-Restoration	Post-Restoration		
			2016	2017	2018	2019
			Construction	Year 1	Year 2	Year 3
Wallacut SET-1	NA	Rate	-2.7	0.4	3.2	1.4
		±SD	2.8	2.9	2.8	2.2
Wallacut SET-2	NA	Rate	1.6	0.9	1.7	-1.1
		±SD	2.5	1.3	1.5	2.0
Reference - BBM-1	2.61	Rate	0.0	0.4	0.9	-0.3
		±SD	0.8	0.8	0.7	0.7
Reference - BBM-2	2.49	l Rate	0.3	-2.5	1.1	-0.5
		±SD	2.4	2.3	2.1	2.6

Elevations of the sediment stakes were unavailable during the time of writing this report, hence, deeper connections between marsh elevations and sediment rates could not be made. However, a general pattern of sediment movement has been observed at the stakes. In 2019, two different observations were made at the two locations. SET-1 has shown accretion since construction in 2016, while SET-2 showed signs of accretion at one-year post restoration. In 2019, SET-2 showed signs of erosion, emulating the trend observed at Ilwaco Slough. However, there was also high variability observed in the measurements at restoration and reference sites.

Channel cross-section data were still under analysis at the time of writing this report and will be reported out on during the 2020 synthesis report.

Steamboat Slough

Project Description & Ongoing Management Actions

Steamboat Slough is a restoration site located, near the City of Skamokawa in Washington. In 2013, Welch Island, a long-term EMP site, was chosen as a nearby reference site for ongoing monitoring and comparisons (Figure 25).

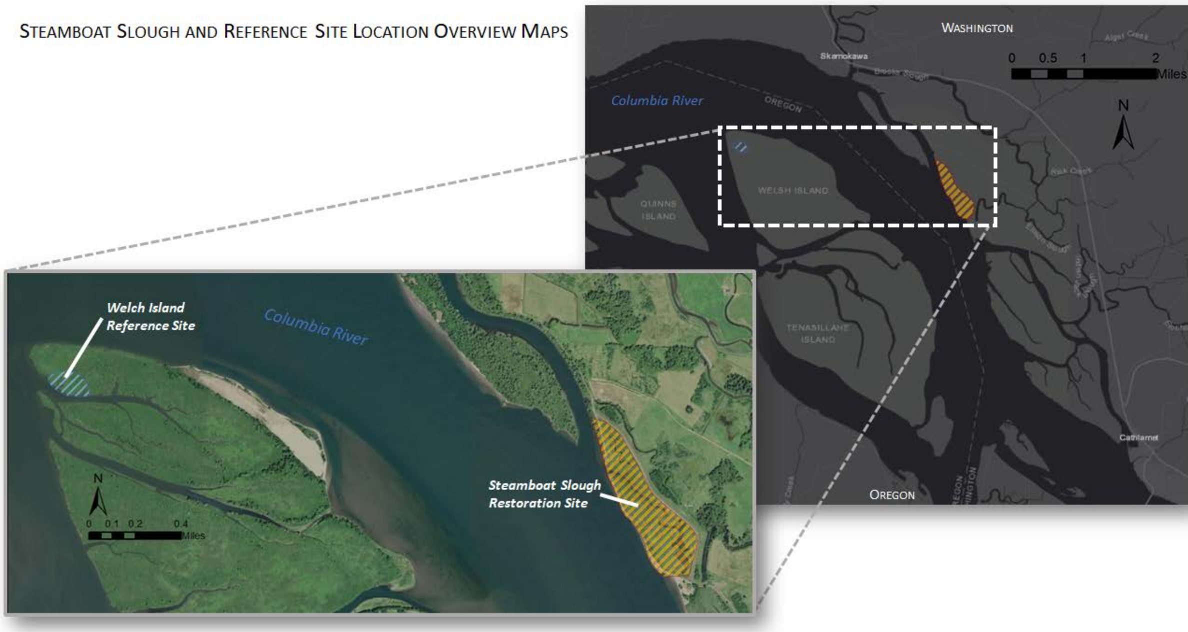

Figure 25: Overview map of Steamboat Slough Restoration Site Location and Welch Island Reference Site Location.

In 2014, tidal influence to the Steamboat Slough was restored through the removal of levee barriers throughout the system and the development of a channel network (Table 2). Channel cutting and marsh lowering enhancements were conducted in areas to expand channel density and access to wetland habitat. The United States Army Corps of Engineers (USACE) were the primary project sponsor for Steamboat Slough and were responsible for most Level 1 monitoring efforts including fish use and nutrient flux studies which can be found in the PNNL and NMFS 2020 AEM report. These studies have highlighted the use of these newly created tidal habitats by endangered salmonids and the importance of these wetlands in contributing nutrients, and macroinvertebrates to the mainstem (PNNL and NMFS 2020).

LCEP's monitoring and research efforts have focused on the Level 2 vegetation monitoring (this report and Schwartz et al. 2017) and through our EMP program have conducted a detailed and ongoing plant biomass and macro-detritus study which has been most recently reported in 2020 (Rao et al. 2020). These complementary research efforts have highlighted the overall success of the Steamboat restoration project over the last 5 years since restoration has occurred.

Steamboat Slough site has two vegetation monitoring areas which were sampled pre-restoration, one-year post-restoration, three, and five years post-restoration. Vegetation monitoring at Steamboat Slough West was established to capture changes directly related to the lowering of the marsh elevation and unrestricted connection to the Columbia River. Vegetation monitoring at Steamboat Slough East was established to track indirect changes to established wetland within the restoration site following tidal reconnection (Figure 26, Figure 27).

STEAMBOAT SLOUGH AND REFERENCE SITE, INTENSIVE MONITORING AREAS HIGHLIGHTED

Figure 26: Map Identifying Intensive Vegetation Monitoring Areas of Steamboat Slough Restoration Site Location and Welch Island Reference Site Location.

Plant Community Results
Sampling Overview
Steamboat Slough plant community results are reported by summarizing the results across the two major sampling areas: Steamboat West and Steamboat East (Figure 26, Figure 27). Steamboat East is in what was a pre-existing wetland area within the center of the site and was only minimally impacted during construction (Figure 26, Figure 27). Steamboat West was located more closely to the mouth of the channel network and was not a wetland area pre-restoration (Figure 26, Figure 27). A similar number of plots where sampled across each sampling area and these are compared to sampling conducted at the reference site, Welch Island (Table 11-Table 12). While Welch Island is an excellent reference site in terms of proximity and general hydrology (Figure 26), located very nearby and experiencing similar tidal conditions, it is not a perfect match. The mismatch in site conditions is mainly due to Welch Slough being composed of primarily mid to high marsh elevations and plant communities and Steamboat Slough, in our monitoring areas, being composed of mid-low marsh elevations (Figure 28). During future monitoring, we plan to seek out lower marsh elevations within Welch Island for monitoring, to improve this comparative reference site analyses. Given the differences in site elevations, extra care should be taken when interpreting comparative results.

Steamboat Slough Restoration Site

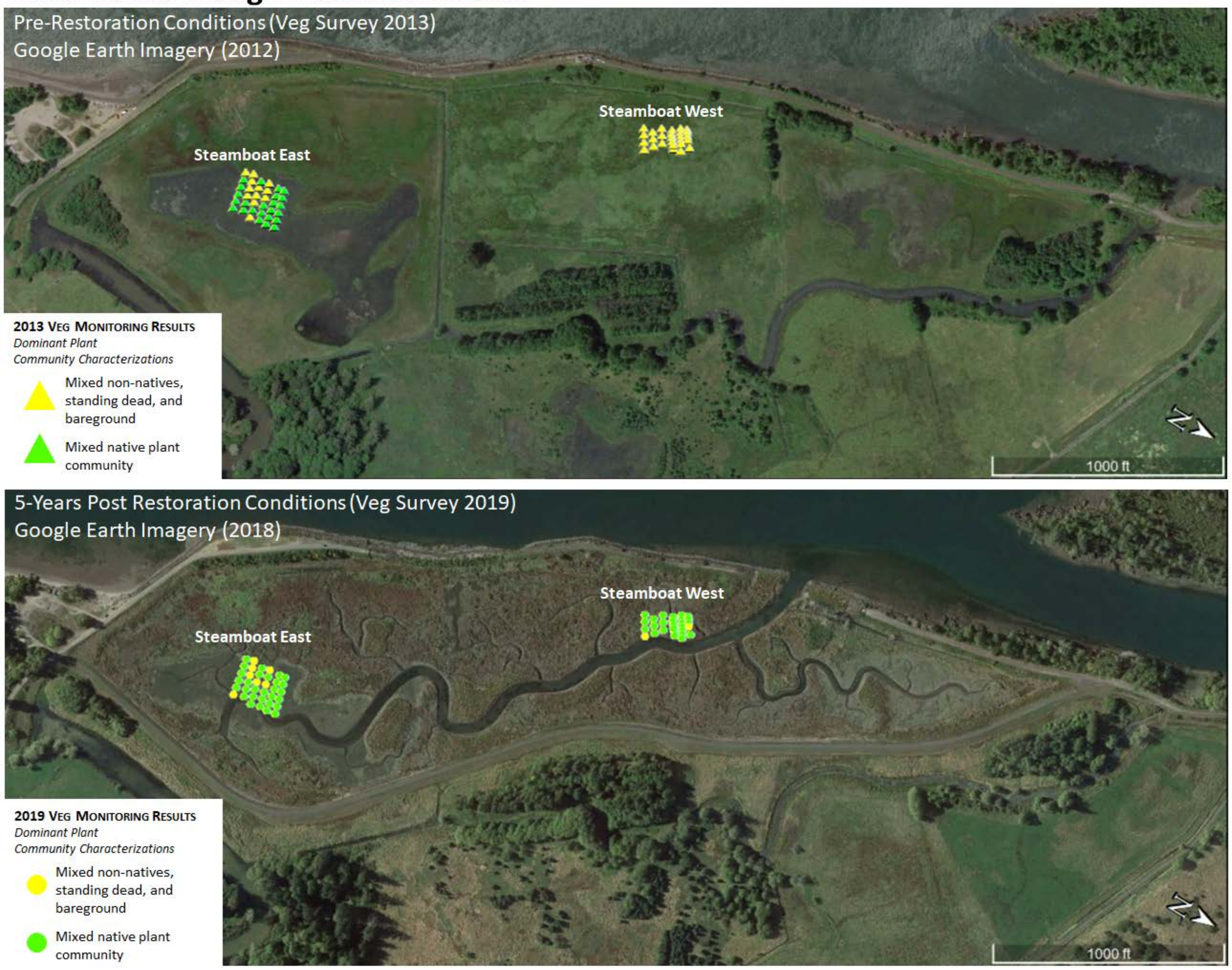

Figure 27: Steamboat Slough Restoration Site Plant Community Monitoring Results Pre (2013) and 5 years-post restoration (2019), Native and Non-native Species Dominance Across Sampling Plots. Google Earth aerial imagery pre and post restoration provides context of restoration actions and landscape change observed across the site. More plant community details can be found in Table 11-Table 12.

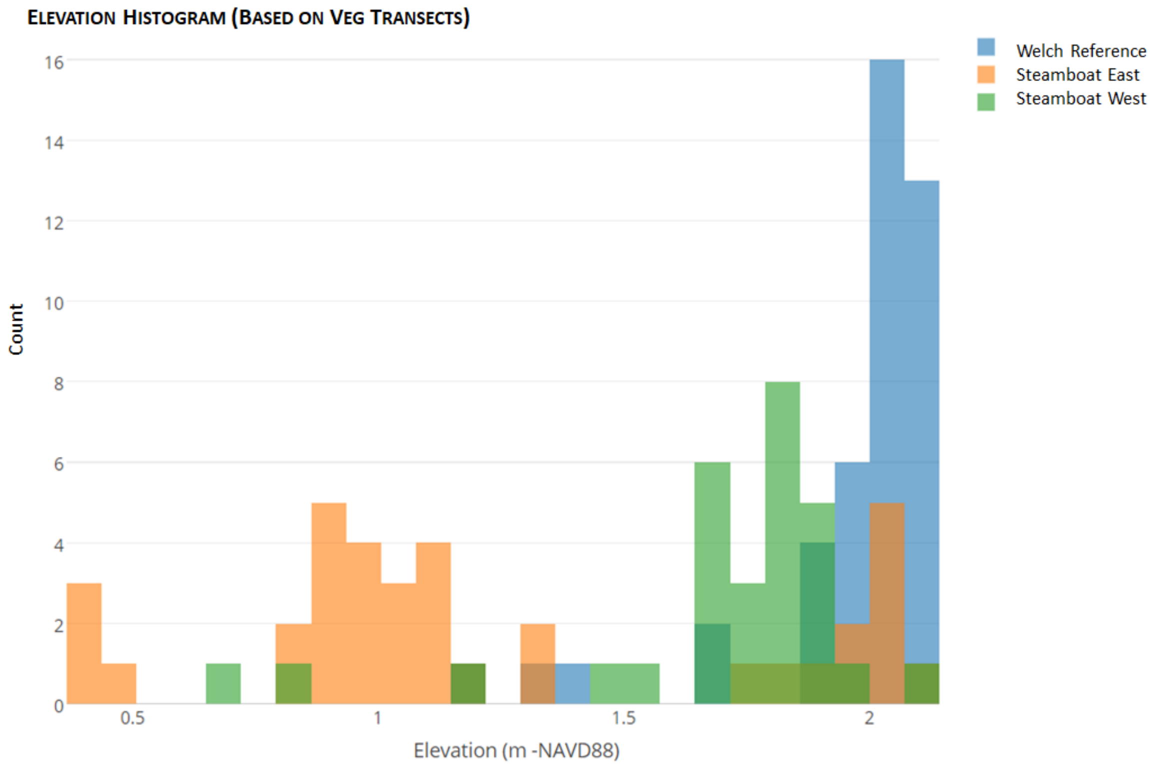

Figure 28: Steamboat Slough Restoration and Reference Site Elevation Distribution Histogram (2019).

Trajectories: Native and Non-native Dominant Species

In year five post-restoration (2019) we found successful native plant community recovery across the sampling areas (Figure 27, Figure 29, Table 11). In 2013, sampling plots located at the East sampling area were dominated by a mix of native and non-native field and wetland grasses such as non-native reed canarygrass, *Phalaris arundinacea*, field fescue, *Festuca arvernensis*, creeping bentgrass, *Agrostis stolonifera*, and native Rough bentgrass, *Agrostis scabra*, in addition to bareground/open water (Figure 27, Table 11, Table 12. In 2019, 5-years post restoration the East sampling area is now dominated by mudflat and native aquatic species Canada waterweed, Elodea canadensis, and Coontail, *Ceratophyllum demersum,* in the lower elevations (Figure 30, Table 12 and native, Wapato, *Sagittaria latifolia,* and non-native reed canarygrass, *Phalaris arundinacea,* in the higher elevation areas (Figure 31, Table 12). Overall native plant cover in the Eastern sampling area has shifted from 36% in 2013 to 50% in 2019, non-native cover has stayed relatively stable (Figure 29), and bareground has increased slightly (Figure 30). This monitoring area, as depicted in the elevation histogram (Figure 28), is composed of both some high and some very low (<1.5 meters) marsh plots, these low marsh areas will likely remain a combination of mudflat and aquatic species into the foreseeable future, while the high marsh areas will likely retain some non-native reed canarygrass, *Phalaris arundinacea*, abundance similar to those levels seen at the reference site (Figure 31, Table 12).

In contrast, sampling plots located at the West sampling area were dominated by primarily non-native field grasses such as field fescue, *Festuca arvernensis*, before restoration and after marsh lowering are now dominated by native, Wapato, *Sagittaria latifolia,* and Nodding beggars-ticks, *Bidens cernua,* among other native species, showing a stark shift from non-native dominance to native 5 years since restoration occurred (Figure 27, Figure 31). Overall native plant cover in the Western sampling area has shifted from 4.6% in 2013 to 72% in 2019, non-native cover has correspondingly declined from 95% to 22% during this time (Figure 29, Table 12), and bareground has remained low (Figure 30). This monitoring area, as depicted in the elevation histogram (Figure 28), is composed of primarily of mid-marsh elevation plots, mid-marsh elevations are known to be some of the most species rich and native dominated areas of tidal wetlands in the Columbia Estuary (Kidd 2017, Kidd et al. 2019) and this area will likely continue to become more diverse and similar to the reference marsh mid-marsh elevation areas as time goes on.

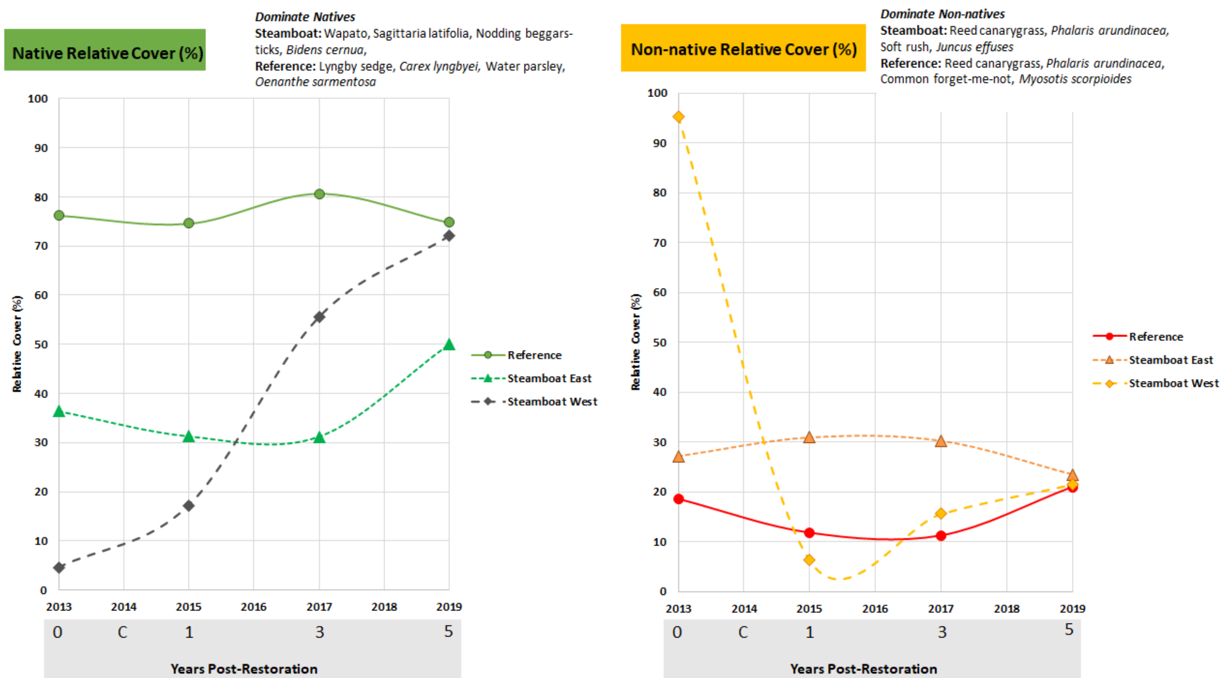

Figure 29: Steamboat Slough Restoration and Reference Site Native and Non-native Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 11 and Table 12.

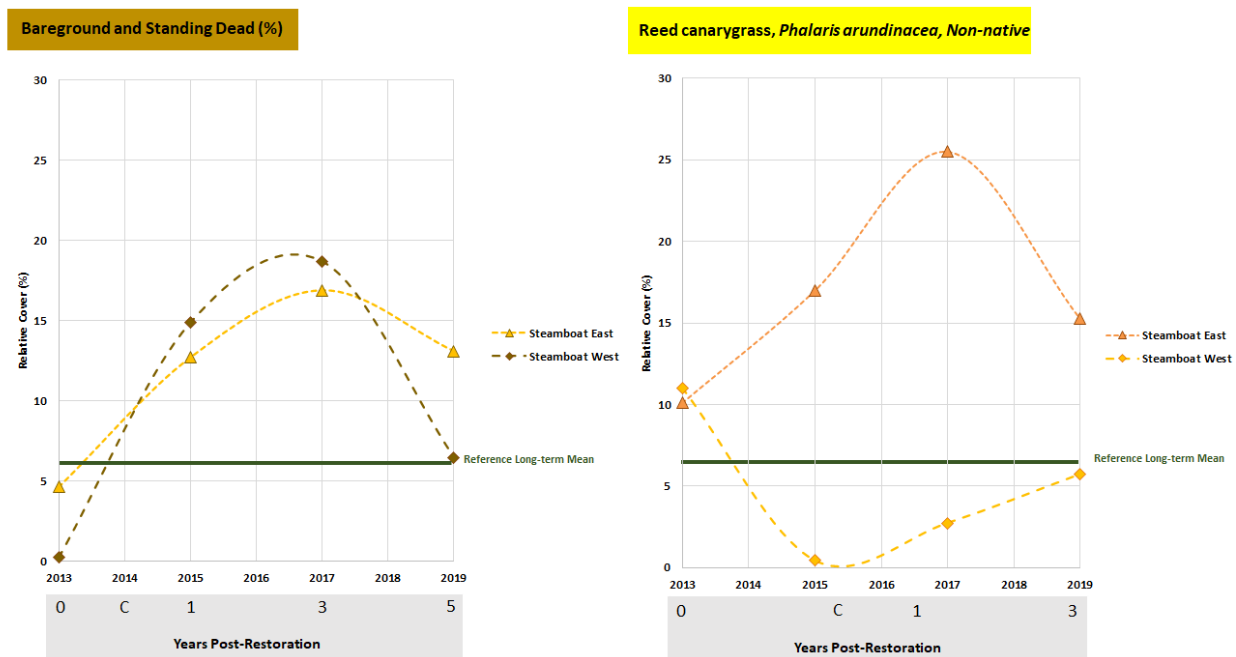

Figure 30: Steamboat Slough Restoration and Reference Site Bareground & Standing Dead, and Non-Native, Reed Canarygrass, *Phalaris arundinacea,* Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 11 and Table 12.

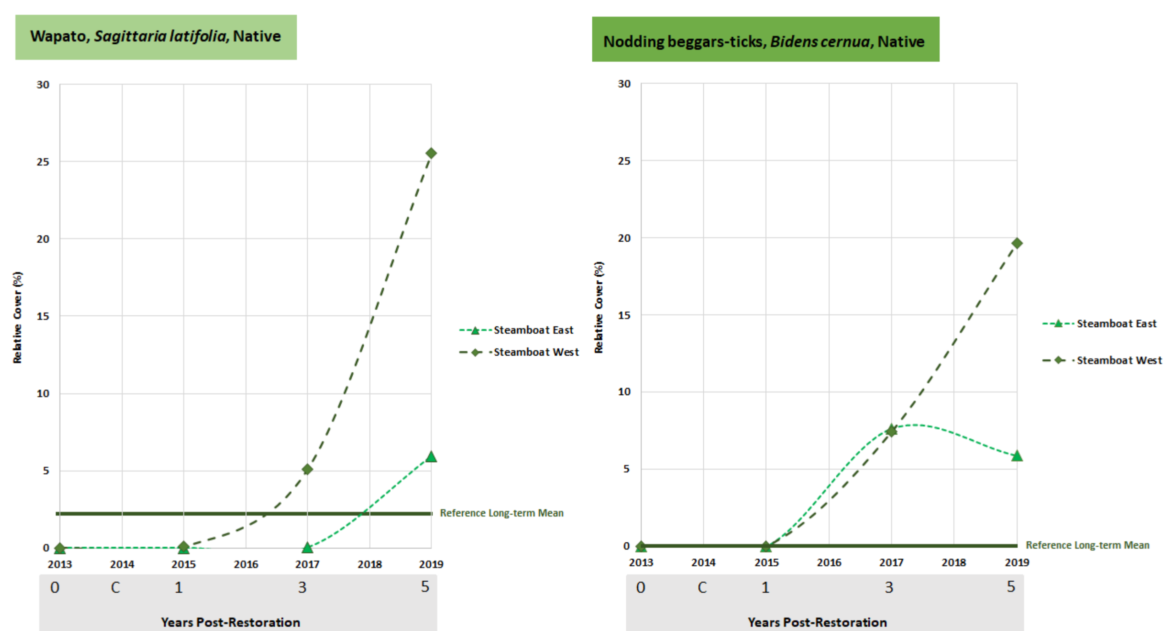

Figure 31: Steamboat Slough Restoration and Reference Site Native, Wapato, *Sagittaria latifolia, &* Nodding beggars-ticks, *Bidens cernua,* Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 11 and Table 12.

Table 11: Steamboat Slough Restoration and Reference Site Number of Samples Plots Surveyed and Mean (±SD) Elevation, and Plant Community Relative Cover (%) Summarized by Years Post-Restoration.

Mean Relative Cover (%)	Monitoring Areas	2013		2015		2017		2019	
		Pre		Year 1		Year 3		Year 5	
		Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
Number of sampling plots	Reference	46		47		47		44	
	Steamboat East	36		36		36		36	
	Steamboat West	36		35		36		29	
Elevation, m, NAVD88	Reference	1.9	0.4	1.9	0.4	1.9	0.9	2.0	0.2
	Steamboat East	1.4	0.4	1.4	0.5	1.5	0.5	1.2	0.5
	Steamboat West	1.9	0.1	1.2	1.0	1.5	0.8	1.7	0.3
Bareground (%)	Reference	1.3	4.9	10.9	18.9	2.9	8.1	9.0	20.1
	Steamboat East	2.2	8.4	12.5	17.1	15.1	19.6	13.1	18.4
	Steamboat West	0.2	0.9	13.8	23.2	17.4	24.7	6.5	16.2
Standing dead (%)	Reference	0.1	0.5	0.1	0.7	0.2	1.4	0.0	0.0
	Steamboat East	2.5	5.8	0.2	0.6	1.7	9.9	0.0	0.0
	Steamboat West	0.0	0.0	1.1	6.1	1.3	3.4	0.0	0.0
Native (%)	Reference	76.2	25.9	74.6	26.4	80.6	20.2	74.8	23.9
	Steamboat East	36.4	32.8	31.3	29.3	31.2	29.3	50.0	31.0
	Steamboat West	4.6	16.9	17.2	21.1	55.5	35.1	72.0	18.7
Non-native (%)	Reference	18.5	25.1	11.7	17.2	11.2	19.6	20.9	21.7
	Steamboat East	27.2	35.4	31.0	32.6	30.3	38.6	23.5	31.5
	Steamboat West	95.4	16.9	6.3	11.4	15.6	19.3	21.5	16.5

Table 12: Steamboat Slough Restoration and Reference Site Mean (±SD) Dominant Plant Species Relative Cover (%) Summarized by Years Post-Restoration.

	Dominate Species	Monitoring Areas	2013		2015		2017		2019	
			Pre		Year 1		Year 3		Year 5	
			Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
Non-native	Reed canarygrass, *Phalaris arundinacea*	Reference	8.1	24.1	5.5	15.7	6.5	17.8	5.9	18.7
		Steamboat East	10.1	23.5	17.0	27.3	25.5	38.5	15.2	29.7
		Steamboat West	11.0	30.3	0.4	2.5	2.7	14.2	5.7	15.5
	Common rush, *Juncus effusus*	Reference	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
		Steamboat East	0.1	0.7	0.0	0.0	1.3	5.2	2.5	13.6
		Steamboat West	2.0	4.6	0.1	0.7	5.8	11.3	8.9	12.0
	Common forget-me-not, *Myosotis scorpioides*	Reference	5.9	11.3	2.0	6.0	0.0	0.0	9.2	11.4
		Steamboat East	0.0	0.0	0.0	0.0	0.0	0.2	0.3	2.0
		Steamboat West	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

Dominate Species		Monitoring Areas	2013 Pre Mean	2013 Pre ±SD	2015 Year 1 Mean	2015 Year 1 ±SD	2017 Year 3 Mean	2017 Year 3 ±SD	2019 Year 5 Mean	2019 Year 5 ±SD
Native	Nodding beggars-ticks, *Bidens cernua*	Reference	0.1	0.6	3.9	5.1	0.4	0.9	1.5	3.3
		Steamboat East	0.0	0.0	0.0	0.0	7.6	17.5	5.9	12.5
		Steamboat West	0.0	0.0	0.0	0.0	7.4	10.1	19.6	13.7
	Lyngby sedge, *Carex lyngbyei*	Reference	39.9	26.8	35.5	24.6	48.3	25.9	27.1	17.8
		Steamboat East	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1
		Steamboat West	0.0	0.0	0.0	0.0	0.0	0.0	1.2	4.7
	Canada waterweed, *Elodea canadensis*	Reference	0.1	0.7	1.6	6.7	1.1	5.7	0.1	0.3
		Steamboat East	3.4	13.9	2.7	8.4	14.0	21.6	21.9	25.5
		Steamboat West	0.0	0.0	0.1	0.3	0.4	1.3	0.7	3.7
	Wapato, *Sagittaria latifolia*	Reference	3.4	5.0	6.3	8.2	3.2	5.5	3.6	4.6
		Steamboat East	0.0	0.0	0.0	0.0	0.0	0.2	6.0	15.3
		Steamboat West	0.0	0.0	0.1	0.2	5.1	7.9	25.5	15.9

Table 13: Steamboat Slough Restoration and Reference Site Mean (±SD) Species Diversity Metrics Summarized by Years Post-Restoration.

Mean Diversity	Monitoring Areas	2013 Pre Mean	2013 Pre ±SD	2015 Year 1 Mean	2015 Year 1 ±SD	2017 Year 3 Mean	2017 Year 3 ±SD	2019 Year 5 Mean	2019 Year 5 ±SD
Total Species Richness	Reference	11.2	3.4	11.8	3.6	10.3	3.3	11.8	4.6
	Steamboat East	5.4	2.8	5.7	2.1	6.3	3.0	5.1	2.1
	Steamboat West	4.3	1.3	5.7	4.0	9.3	5.3	7.7	2.4
Non-native Species Richness	Reference	1.8	1.2	2.1	1.2	1.6	1.0	2.5	1.4
	Steamboat East	2.6	2.8	1.9	1.3	1.7	1.3	1.4	0.9
	Steamboat West	3.9	1.2	1.6	1.3	2.1	1.5	2.1	0.9
Native Species Richness	Reference	8.9	2.9	9.7	3.3	8.0	2.8	9.3	3.8
	Steamboat East	2.7	1.6	3.8	1.4	3.9	2.1	3.6	1.7
	Steamboat West	0.3	0.6	4.1	3.1	6.9	4.2	5.6	2.1
Shannon Diversity Index	Reference	1.5	0.5	1.5	0.5	1.3	0.4	1.7	0.5
	Steamboat East	0.9	0.6	0.9	0.5	0.7	0.5	0.8	0.4
	Steamboat West	0.8	0.4	0.6	0.5	1.3	0.7	1.5	0.4
Evenness	Reference	0.6	0.2	0.6	0.1	0.6	0.1	0.7	0.1
	Steamboat East	0.5	0.3	0.5	0.2	0.4	0.2	0.5	0.2
	Steamboat West	0.6	0.2	0.4	0.2	0.6	0.2	0.8	0.1

Trajectories: Species Richness

Over the five-year time frame, overall mean total species richness (TSR) has remained relatively stable in the East sampling with a mean of 5.1 identified per plot in 2019 compared to 5.4 in

2013 (pre-restoration) and has increased in the West sampling area with 7.7 in 2019 compared to 4.3 in 2013 (Table 13). Overall, these TSR numbers are lower than those observed at the reference site, which had 11.2 TSR in 2013 and 11.8 TSR in 2019 (Table 13). In both sampling areas, the mean non-native species richness has declined post-restoration, from 2.6 in 2013 to 1.4 in 2019 for the East monitoring area and from 3.9 in 2013 to 2.1 in 2019 for the West monitoring area (Table 13). These mean non-native species richness values are similar to what has been found at the reference site, which has varied from 1.8 to 2.5 between 2013 and 2019 (Table 13). Overall mean native species richness has increased slightly in the Eastern monitoring area shifting from 2.7 in 2013 to 3.6 in 2019 and has increased significantly in the Western monitoring area from 0.3 in 2013 to 5.6 in 2019 (Table 13). Both monitoring areas still remain significantly lower in mean native species richness than the reference site, which has ranged from 8.9 in 2013 to 9.3 in 2019 (Table 13).

Overall, these shifts in species richness across the restoration site overtime correspond to the changes also seen across the vegetation communities and are similarly impacted by the differences in wetland elevations restored. High marsh areas, with reed canarygrass, *Phalaris arundinacea*, cover will likely remain relatively low in native species richness due to the resistance of these zones to change (Kidd 2017), correspondingly very low marsh-mudflat zones dominated by native aquatics are also not expected to accumulate further native species richness, given these habitats tend to accumulate less species than well drained marsh zones. The Eastern monitoring area is composed of both these very high and very low marsh elevations, which will result in a slower accumulation of native species overtime compared to the West monitoring area. The West monitoring area is primarily composed of mid-high marsh elevation and should continue to accumulate native species richness similar to those observed at the reference site.

Soil Monitoring Results

Re-introducing flooding to previously drained agricultural soils has significant impacts on soil characteristics and creates the biogeochemical template needed for wetland plant community and habitat restoration. Additionally, restoration actions such as scrapping down the topsoil to created lower elevations and channels can alter soil composition, revealing mineral soils and causing compaction. These restoration actions and soil manipulations can further impact plant community recovery. Monitoring soil conditions pre- and post-restoration can provide insight into the mechanisms causing restoration success and failure and LCEP has recently begun consistently monitoring soil conditions as part of their Level 2 AEMR. Due to the recent adoption of these monitoring methods we only have limited data for Steamboat's soil conditions, and these data were collected at slightly different locations (but within the same general monitoring areas) than the vegetation data reported above (all soil data is reported in Kidd et al. 2019 and Rao et al. 2020). However, while these soil data are limited, they still provide valuable insight into the site conditions and how they compare across monitoring areas and to the reference site.

In year 5 post-restoration (saturated) soil salinity was found to be similar across the reference site (0.21 ppt) and Steamboat monitoring areas Steamboat East (0.17 ppt) or West (0.15 ppt). Soil salinity can be an important indicator of wetland restoration success along the oligohaline (>0.5 ppt) to saltwater (35 ppt) marsh gradient, where saline soil conditions are restored through the re-introduction of tidal saltwater exposure. The soil salinity levels of Steamboat slough and its reference site Welch Island are clearly those of freshwater tidal wetlands (<0.5 ppt). All these wetland areas are located within the hydrogeomorphic reach B of the estuary, and right on the edge of the estuary saltwater intrusion zone, however their soil salinity levels indicates that salinity exposure is minimal at their locations in the river (Simenstad et al. 2011). Soil pH levels were generally found to be neutral and most similar between the reference site (6.9) and the Steamboat East monitoring area (7.0), while pH levels at the West monitoring area were on average lower (6.5), and overall, more variable (Figure 33, Table 14). Soil oxygenation can influence soil pH in wetlands, with both very low and high oxygen levels resulting in lower pH levels (Seybold et al. 2002). Correspondingly, the West monitoring area also reported greater soil ORP conditions (more oxygen) than both the East and Reference site monitoring areas (Figure 34, Table 14). The mean ORP at the West monitoring area being 177 mV, the East monitoring area being 80 mV, and the reference site being 146 mV (Table 14). Generally, the East monitoring area had a wider distribution of soil sampling locations across the elevation gradient, lending to more low marsh sampling where soil ORPs are typically lower due to greater soil moisture saturation (Figure 35).

Plant communities across the reference and restoration sites were clearly segregated along elevation and soil ORP gradients, with reed canarygrass, *Phalaris arundinacea* and lyngby sedge, *Carex lyngbyei,* growing primarily above 1.75 meters in elevation and > 150 mV soil ORP, and wapato*, Sagittaria latifolia,* and spikerush, *Eleocharis palustris* primarily growing below 1.75 meters and <150 mV soil ORP (Figure 36). Lyngby sedge, *Carex lyngbyei, and* wapato*, Sagittaria latifolia,* where also found growing in the high and low marsh zones spanning the elevation and ORP gradient, which indicates the typical pattern of grown found in these species (*Sagittaria latifolia* in the low marsh and *Carex lyngbyei* in the high marsh) may be a result of competition with other high and low marsh plant species rather than a result of suitable soil conditions alone (Kidd 2017). Overall, wetland plant community segregation is clear along both the elevation and soil ORP gradients found at the reference and restoration sites. The identification of these soil gradients indicates that wetland soil biogeochemistry conditions have been recovered at the restoration site and are positively influencing wetland plant community development (Figure 36).

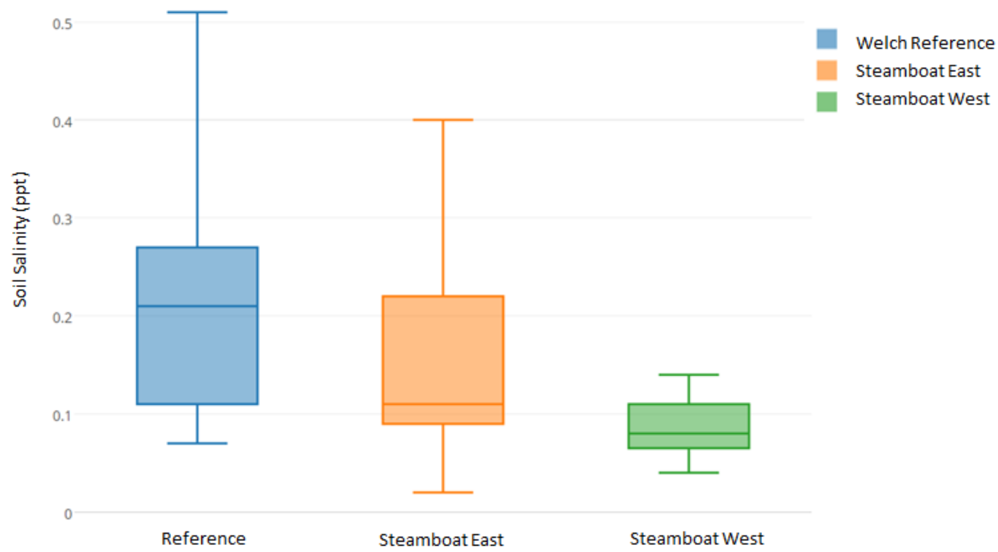

Figure 32: Boxplot of Steamboat Slough Restoration and Reference Site (2019) soil salinity (ppt) conditions (low tide). A summary of all soil data can be found in Table 14.

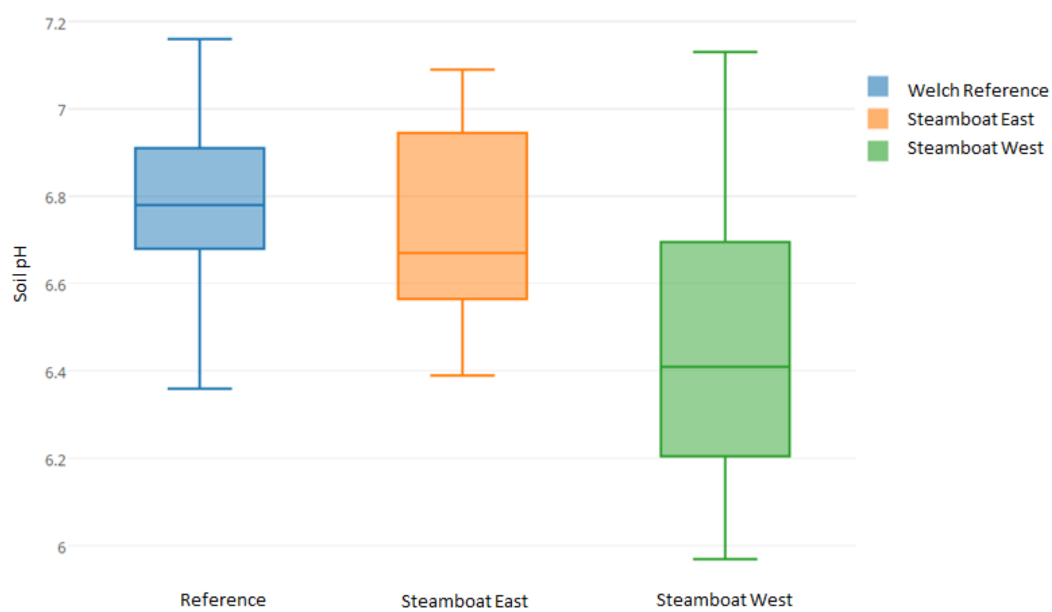

Figure 33: Boxplot of Steamboat Slough Restoration and Reference Site (2019) soil pH conditions (low tide). A summary of all soil data can be found in Table 14.

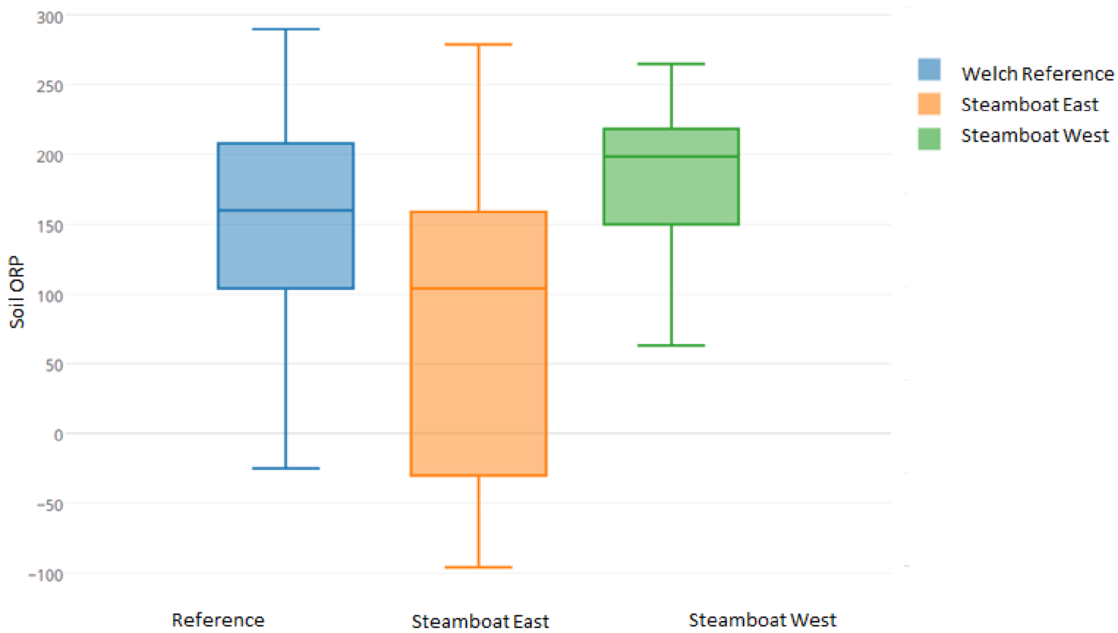

Figure 34: Boxplot of Steamboat Slough Restoration and Reference Site (2019) soil ORP conditions (low tide). A summary of all soil data can be found in Table 14.

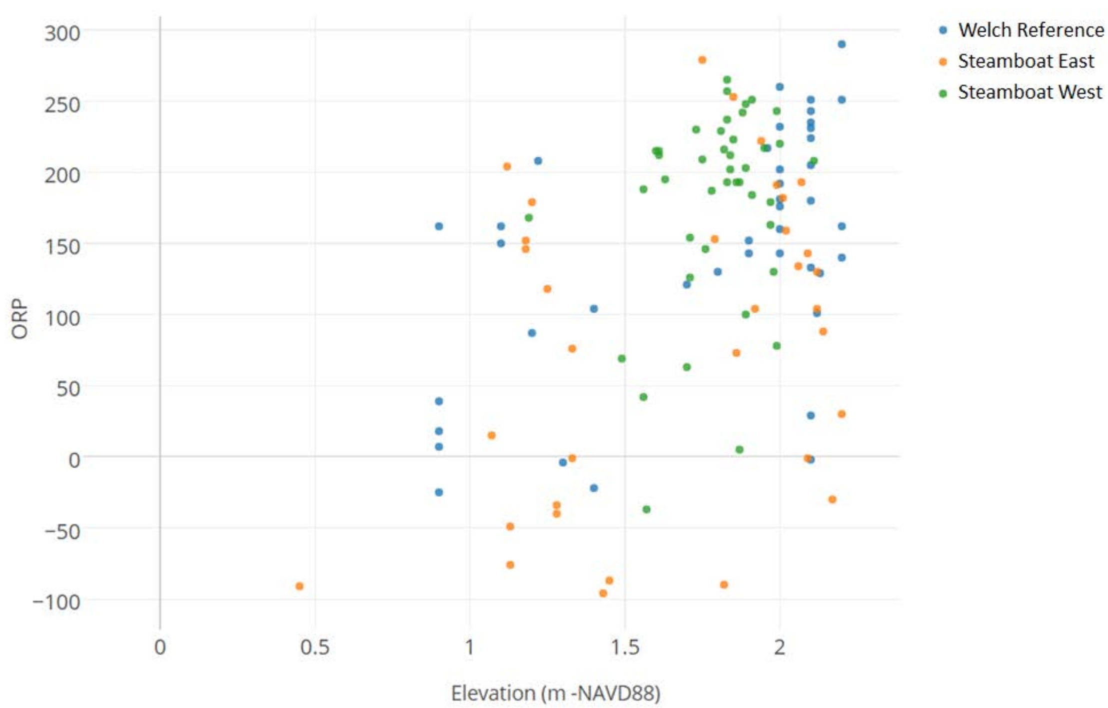

Figure 35: Wetland Soil ORP (mV) vs Elevation (m-NAVD88) across sample plots at Steamboat Slough Restoration and Reference Sites (2019).

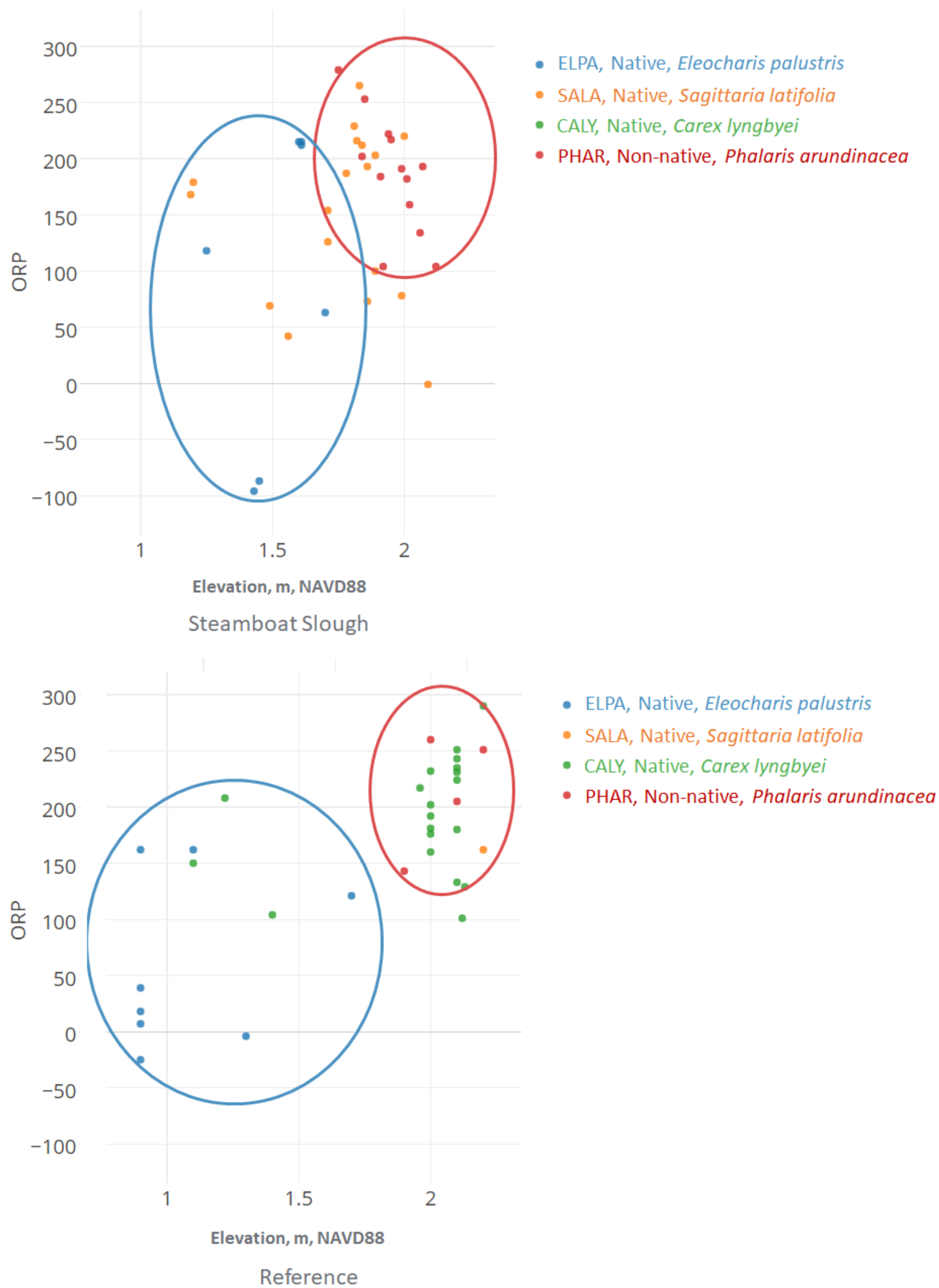

Figure 36: Wetland Soil ORP (mV) vs Elevation (m-NAVD88) of dominant plant community sample plots at Steamboat Slough Restoration and Reference Sites (2019). Native species include: ELPA, Spikerush, *Eleocharis palustris,* SALA, Wapato*, Sagittaria latifolia,* CALY, Lyngby sedge, *Carex lyngbyei,* Non-native species: PHAR, Reed canarygrass, *Phalaris arundinacea.* For more information on plant community abundance by monitoring area see Table 12 and a summary of all soil data can be found in Table 14.

Table 14: Steamboat Slough Restoration and Reference Site Mean (±SD) Soil Conditions (2019).

Soil Conditions	Reference		Steamboat East		Steamboat West	
	Mean	±SD	Mean	±SD	Mean	±SD
Sample number	41		34		44	
Elevation (m - NAVD88)	1.77	0.46	1.64	0.45	1.73	0.36
Salinity (ppt)	0.21	0.12	0.17	0.16	0.15	0.16
pH	6.9	0.4	7.0	0.6	6.5	0.4
ORP (mV)	146	85	80	110	177	69

Water Surface Elevation & Water Temperature Results

In August 2019, a water level sensor was installed in the channel close to Steamboat East vegetation monitoring grid to report out on the evolution of the hydrology (reported as water surface elevation – WSE) of the site after tidal reconnection (Figure 25). Changes in hydrology for Steamboat Slough for 1 year and 3-year post-restoration can be found in the PNNL and NMFS 2020 AEM report. Table 15 summarizes the WSE metrics of Steamboat for year five post-restoration and compares it to the hydrology of Welch Island. Sensor elevations at each site are also provided.

Table 15: Hydrologic Summary Statistics for Steamboat Slough (adjacent to East monitoring area) and Welch Island (Reference) site for 2019 – 5 year post-restoration. All metrics are in meters, relative to the North American Vertical Datum of 1988 (NAVD88). MWL = mean water level; MHHW = mean higher high water, Tidal Range is Mean Tidal Range.. Full comparative hydrographs are in Appendix B: Site Hydrographs.

Restoration Year	Monitoring Locations	Sensor Elevation	MWL	MHHW	Tidal Range	Max WSE	Date of Max WSE	Period of Record	Days
2019 (Year 5)	**Steamboat Slough**	0.6	1.6	2.7	2.0	3.3	Dec 12	Aug-Dec	152
	Reference	0.6	1.7	2.8	2.1	3.3	Dec 12	Sept-Dec	113

Welch Island is a predominantly tidal driven site with annual maximum WSE levels coinciding with king tides during winters. The marsh elevation at Welch island was 0.92 m in 2019 (Rao et.al 2020). From Table 15 it is observed that at year five post-restoration, the hydrological patterns observed at Steamboat Slough was identical to that of Welch island. When mean water levels across the two sites were compared from September 2019 to December 2019, a statistically significant linear relationship was obtained (R^2= 0.98), suggesting that channel reconnection at Steamboat slough has positively influenced the hydrology at steamboat, making it identical to that of Welch Island. The marsh elevation at Steamboat East in 2019 was 1.2 m (Table 11).

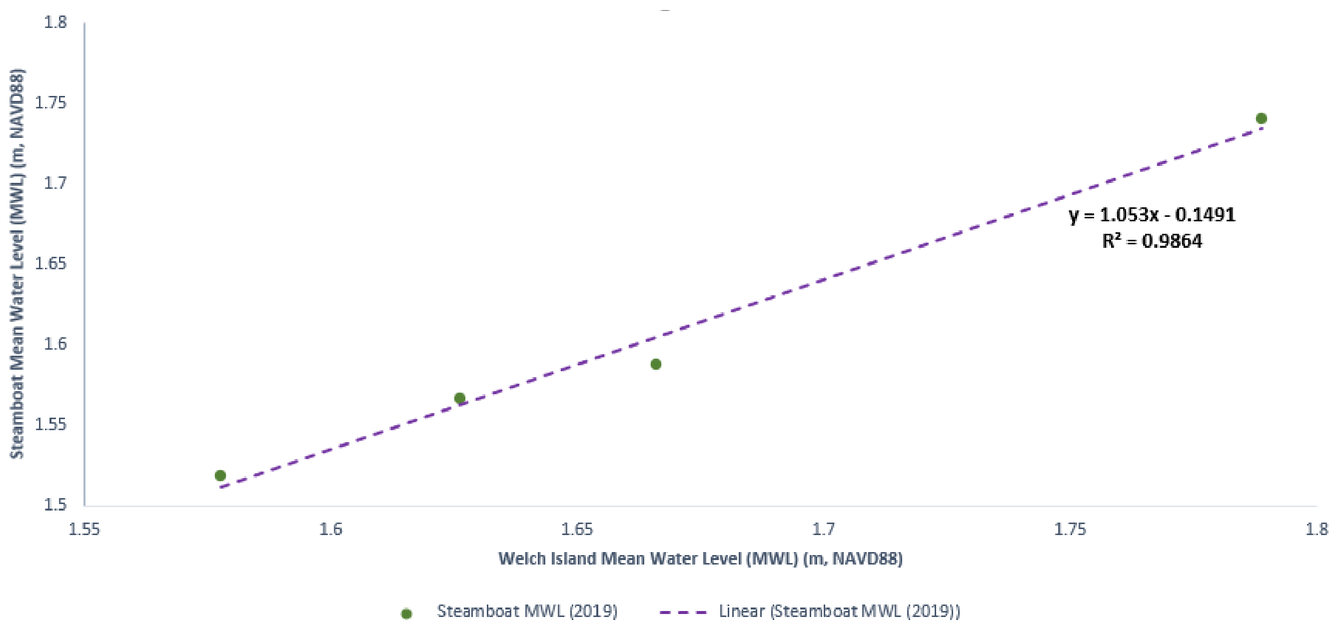

Figure 37: Scatterplot showing a linear relationship between mean water levels at Steamboat Slough and the adjacent Welch Island Reference site for September 2019 to December 2019. MWL = Mean Water Level (m, NAVD88)

In summary, these hydrology data indicate that the site has been successfully reconnected to the mainstem Columbia river and is experiencing similar flooding to its reference site Welch Island (Figure 37). It should also be noted, however, that while we observed similar hydrologic patterns between the two sites, the dataset available is small, hence, this pattern cannot be generalized over different seasons and weather patterns. Water temperature data and further analysis of hydrology data will be available in the 2020 synthesis report.

Sediment Accretion and Erosion Monitoring & Channel Cross-Sections

Accretion and Erosion Monitoring at Steamboat Slough is part of Level 1 monitoring conducted by US Army Corps hence is unavailable in this report. Please refer to PNNL and NMFS 2020 AEM reports for further information on these sections.

Sauvie Island Phase 2 (Deep Widgeon and Millionaire)

Project Description & Ongoing Management Actions

North Unit Phase 2 - Millionaire and Deep Widgeon Lakes is a restoration site located in the northern portion of Sauvie Island, Oregon. In 2014, Cunningham Lake, a long-term EMP site, was chosen as a nearby reference site for ongoing monitoring and comparisons (Figure 38). In 2014, water-control structures were removed from both Millionaire and Deep Widgeon Lakes returning full hydrologic access to the site. In strategic locations, marsh plain surfaces were scraped down to lower elevations, allowing a larger portion of the wetlands to be inundated at deeper depths for longer periods of time, thereby benefiting native plant species. Material removed to create wetland areas were placed adjacent to wetlands to create riparian berms, higher elevation zones, intendent to support native shrub-scrub plant communities.

Removal of structures reestablished upriver and local juvenile salmonid access to over 292 acres of historical habitats (Johnson et al. 2018).

Plant Community Results

Sampling Overview

Millionaire and Deep Widgeon North monitoring areas were placed in locations that received elevation manipulations (scrape down and berm placement) and Millionaire and Deep Widgeon South monitoring areas were located in nearby locations that were not impacted by ground disturbing construction (Figure 40, Figure 41). The scrape down areas can clearly be seen in the aerial images (Figure 40, Figure 41) when comparing pre and post conditions across both sites. The shift in both northern monitoring area elevation gradients are also evident in the elevation distribution histogram of pre and post-restoration conditions, where scrape down and berm placement resulted in both much higher and much lower elevations than pre-restoration conditions (Figure 39). In the elevation histogram you can also see an increase in the overall number of plots monitored at the references site, however the overall elevation gradient monitored is unchanged (Table 16).

A similar number of plots where sampled across each sampling area and these are compared to sampling conducted at the reference site, Cunningham Lake (Table 16). Cunningham Lake is an excellent reference site both in terms of proximity and general hydrology (Figure 38), located very nearby and experiencing similar seasonal flooding and tidal conditions. The only issues we identified with the reference site, in terms of comparability with the restoration site is that no sampling occurs above 3 meters in elevation (NAVD88) at Cunningham lake and the restoration site has restored berms that go up to 4 meters in elevation. However, we know from long-term monitoring data at Cunningham lake that the emergent wetland monitoring stops at 3 meters in elevation because this is where the shrub-scrub zone becomes very dense (Borde et al. 2012). Specifically, mature native pacific willow, *Salix lucida*, is found growing at 2.8 meters and greater in elevation at the reference site and as the elevation increases shifts into native Oregon ash, *Fraxinus latifolia* swamp (Borde et al. 2012). Shrub-scrub and riparian forest restoration continue to be a challenge across all restoration projects in the estuary because without ongoing maintenance of planting areas, these elevations are easily overwhelmed by reed canarygrass, *Phalaris arundinacea.* An additional challenge to restoration present at North Unit Phase 2 is ongoing heavy cattle grazing, which was observed across all monitoring areas in 2019.

NORTH UNIT PHASE 2 AND REFERENCE SITE LOCATION OVERVIEW MAPS

Figure 38: Overview map of North Unit Phase 2 Restoration Site Location (monitoring areas) and Cunningham Lake Reference Site Location (monitoring area).

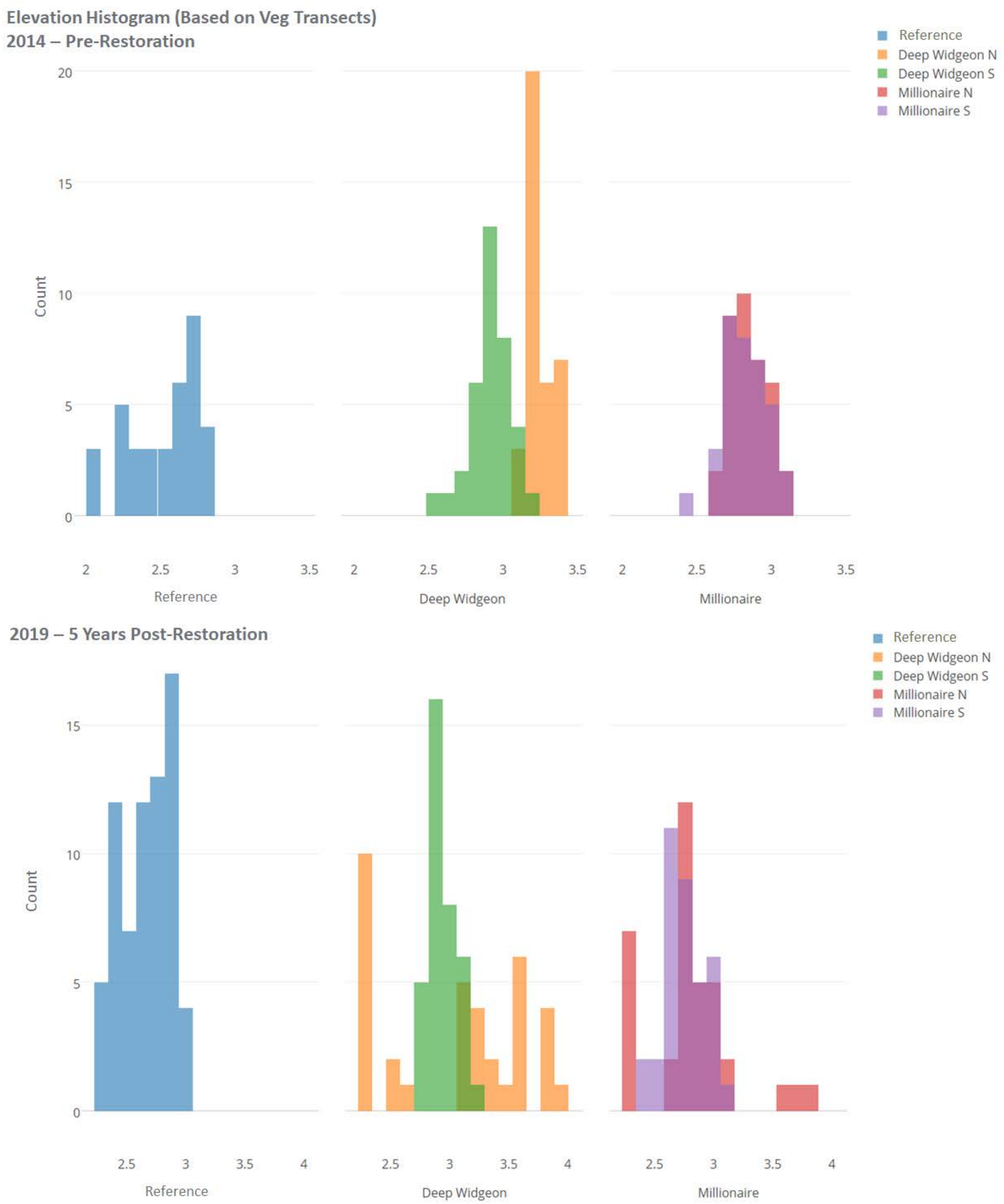

Figure 39: North Unit Phase 2 Restoration and Reference Site Elevation Distribution Histograms, Pre (2014) and 5 Years Post Restoration (2019).

Trajectories: Native and Non-native Dominant Species

In year five post-restoration (2019) both Deep Widgeon and Millionaire monitoring areas have recovered native plant communities dominated by Wapato, *Sagittaria latifolia,* and Common spikerush, *Eleocharis palustris* were the wetlands fall between 2.4-2.8 meters in elevation (Figure 40-Figure 42, Table 16-Table 17). Areas that were scraped down below 2.4 meters or shifted (berm creation) up above 2.8 meters have not recovered native plant communities (Figure 40-Figure 42, Table 16-Table 17). Bareground persists below 2.4 meters within the Millionaire and Deep Widgeon North soil scrap down areas (Figure 42-Figure 46). Millionaire and Deep Widgeon North, as depicted in the elevation histogram (Figure 39), are composed of both some high and some very low (<2.4 meters) marsh plots, these low marsh areas will likely remain a combination of mudflat and aquatic species into the foreseeable future, while the high marsh areas will likely retain non-native reed canarygrass, *Phalaris arundinacea*, abundance (Figure 45-Figure 46). While initial planting of native shrub-scrub species may have occurred across the restored high marsh zones in the restoration site, they were not observed during the 2019 monitoring.

Millionaire South and Deep Widgeon South showed the greatest overall similarities to the reference site in terms of plant community composition. This is likely due to having a similar elevation gradient to the reference site, and therefore having similar opportunities for native plant community establishment (Figure 42). The abundance of non-native Spotted ladysthumb, *Polygonum persicaria*, across the mid-low marsh zones in Millionaire North and South are likely due to grazing impacts in these areas (Table 17). *Polygonum persicaria* is grazing resistant and is commonly found growing in heavily disturbed and/or grazed wetlands (Kidd et al. 2015).

Overall, across all monitoring areas, trends in bareground, standing dead, native, and non-native plant cover follow those observed at the reference site. Specifically, similar patterns were observed across all sites in 2015 (year 1) and 2017 (year 3), which were both were extreme water years (Figure 5-Figure 7), with the very low water conditions (low river discharge and wetland water levels) in 2015 resulting in greater native and non-native cover (Figure 43-Figure 46), with lower amounts of bareground. Correspondingly, the very high water conditions of 2017 resulting in lower overall native and non-native cover and greater amounts of open water and bareground (Figure 43-Figure 46). Through our long-term Ecosystem Monitoring Program, we have identified that annual water year conditions and Columbia River discharge levels can dramatically impact plant community composition in upper river wetland sites such as those found within Sauvie Island up through Franz Lake (Kidd et al. 2019).

North Unit Phase 2: Millionaire Lake Restoration Site

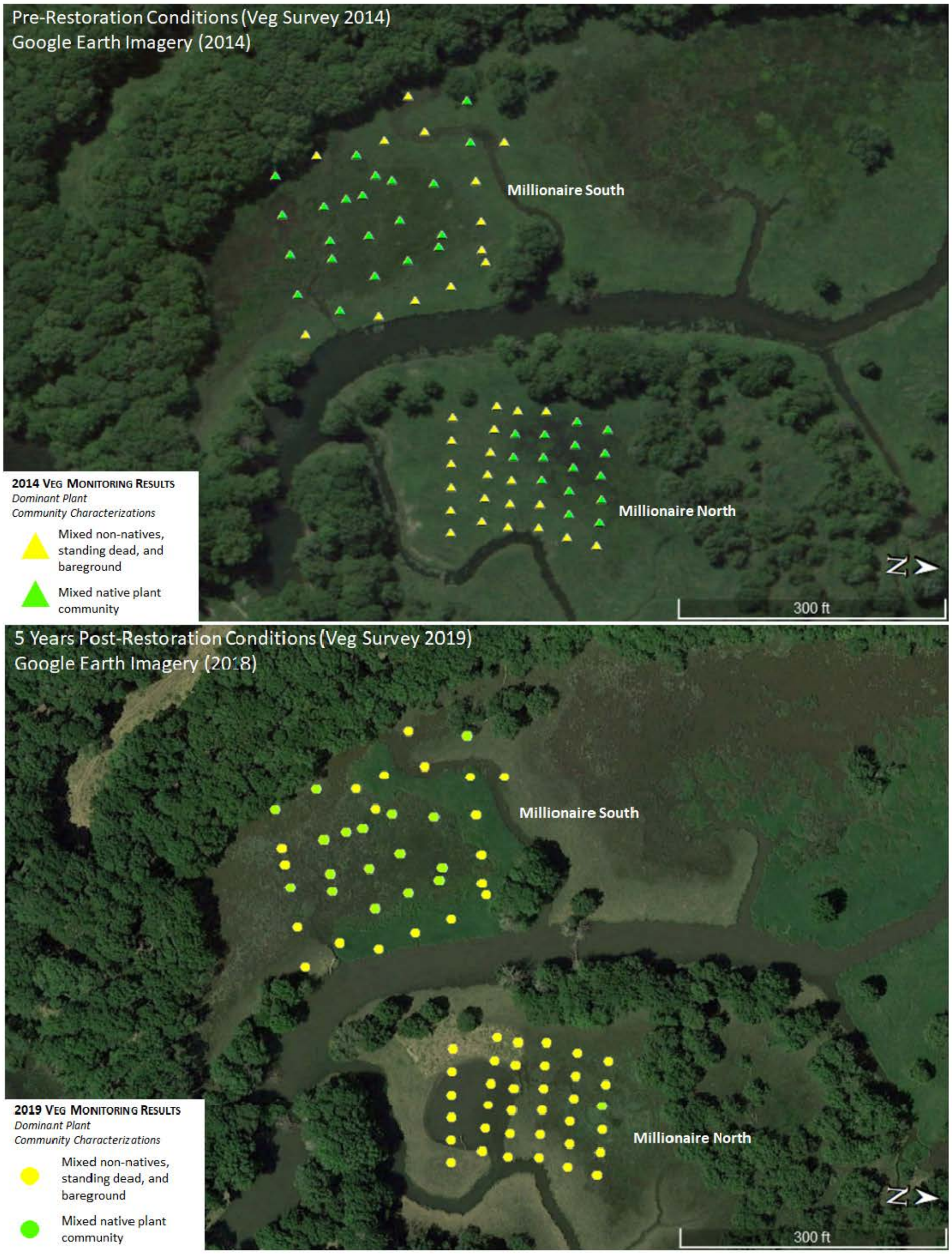

Figure 40: Millionaire Lake Restoration Site Plant Community Monitoring Results Pre (2014) and 5 years-post restoration (2019), Native and Non-native Species Dominance Across Sampling Plots. Google Earth aerial imagery pre and post restoration provides context of restoration actions and landscape change observed across the site. More plant community details can be found in

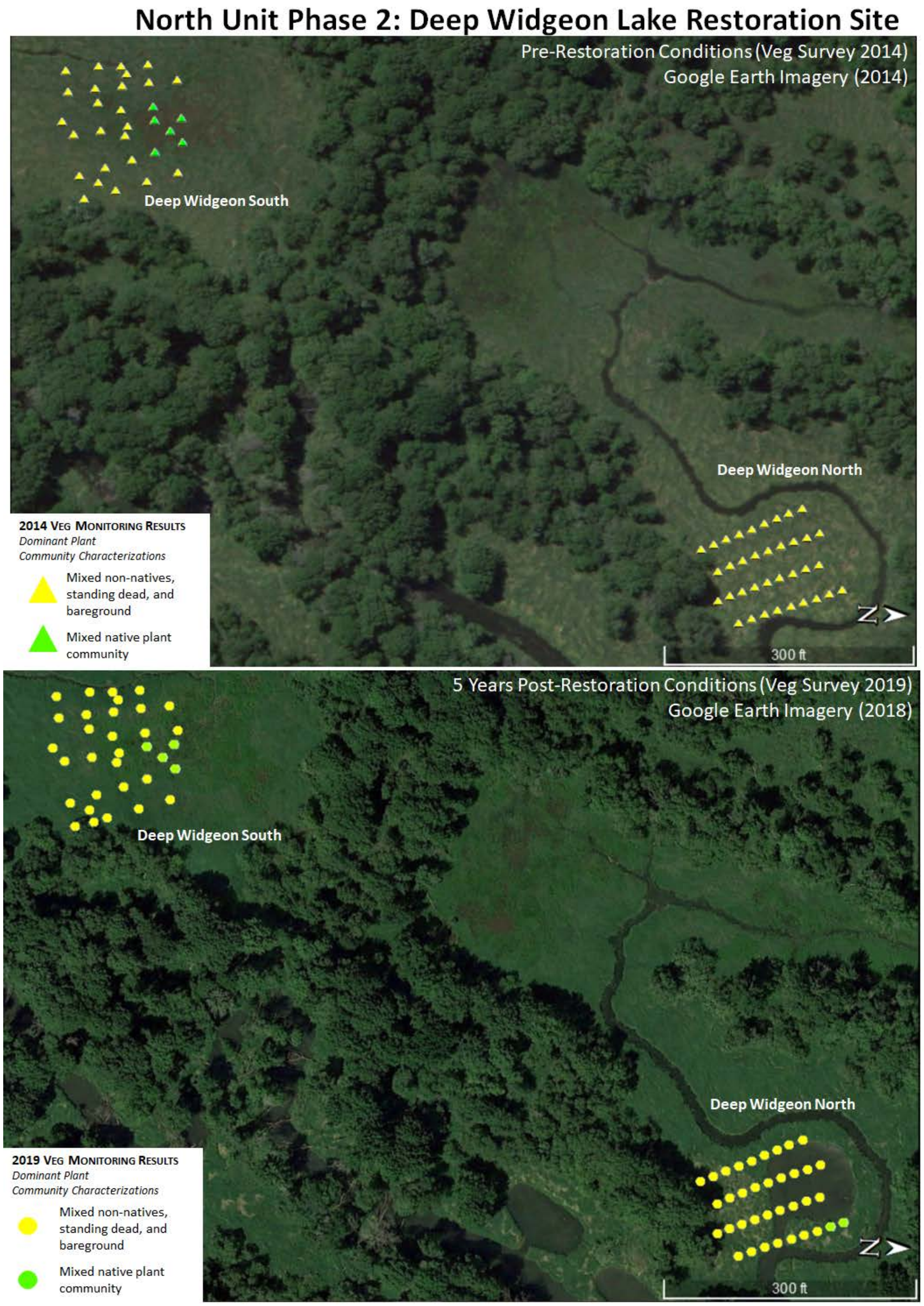

Figure 41: Deep Widgeon Lake Restoration Site Plant Community Monitoring Results Pre (2014) and 5 years-post restoration (2019), Native and Non-native Species Dominance Across Sampling Plots. Google Earth aerial imagery pre and post restoration provides context of restoration actions and landscape change observed across the site. More plant community details can be found in

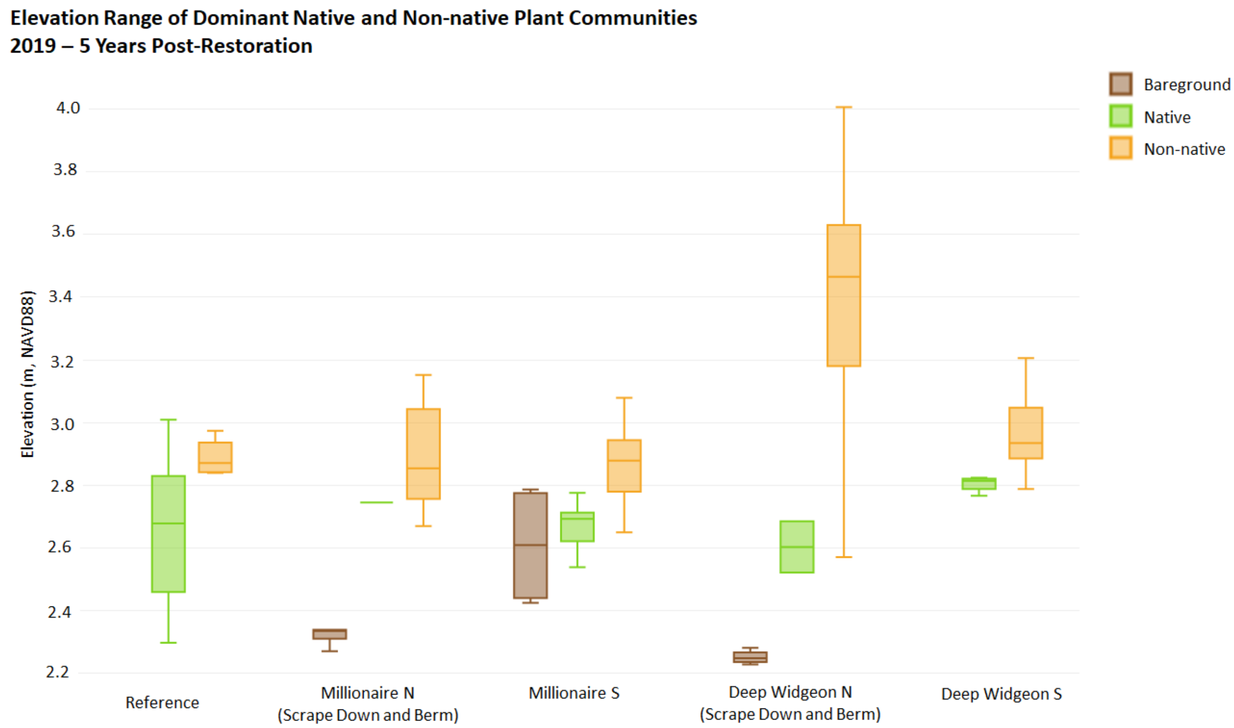

Figure 42. North Unit Phase 2: Elevation Ranges (m, NAVD88) of Dominant Plant Communities 5-years Post-Restoration (2019). More plant community details can be found in Table 16-Table 18.

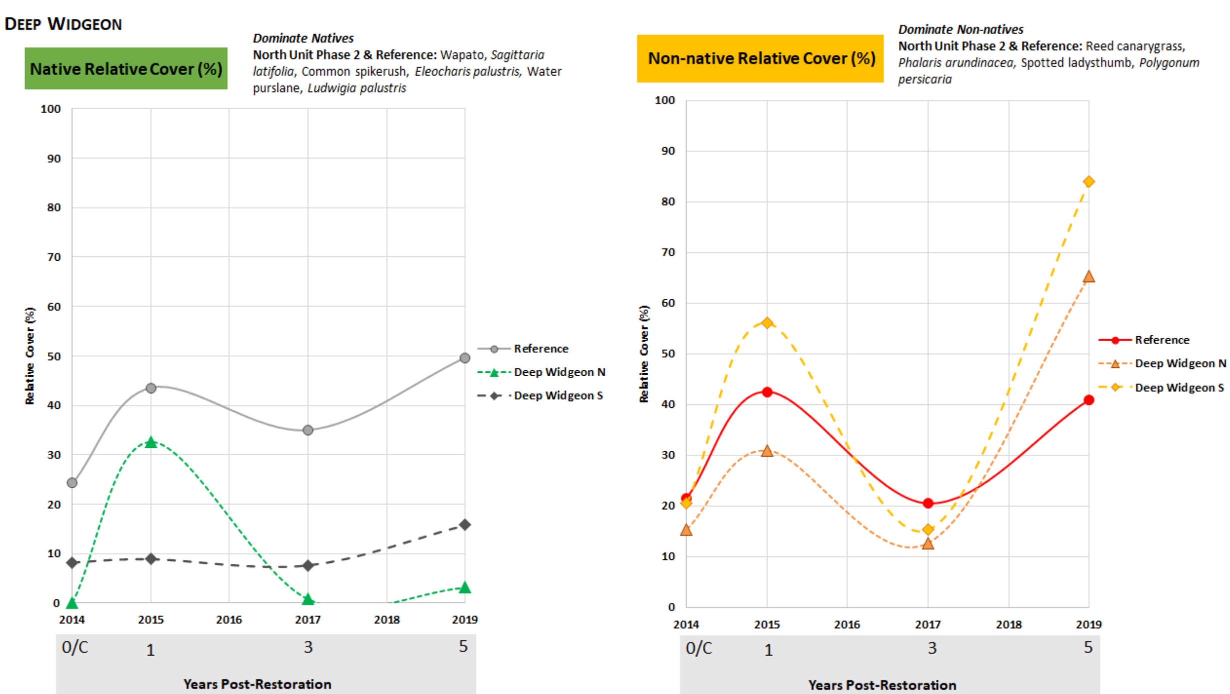

Figure 43: North Unit Phase 2: Deep Widgeon Native and Non-native Relative Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 16-Table 18.

Millionaire

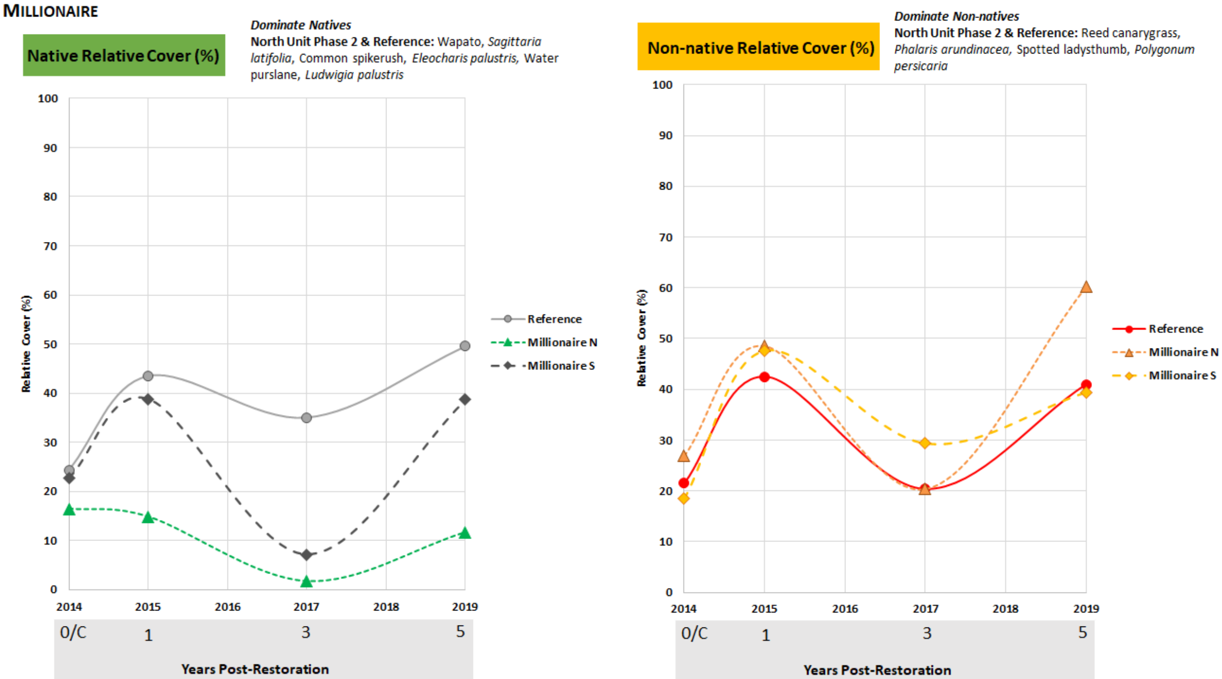

Figure 44: North Unit Phase 2: Millionaire Native and Non-native Relative Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 16-Table 18.

Deep Widgeon

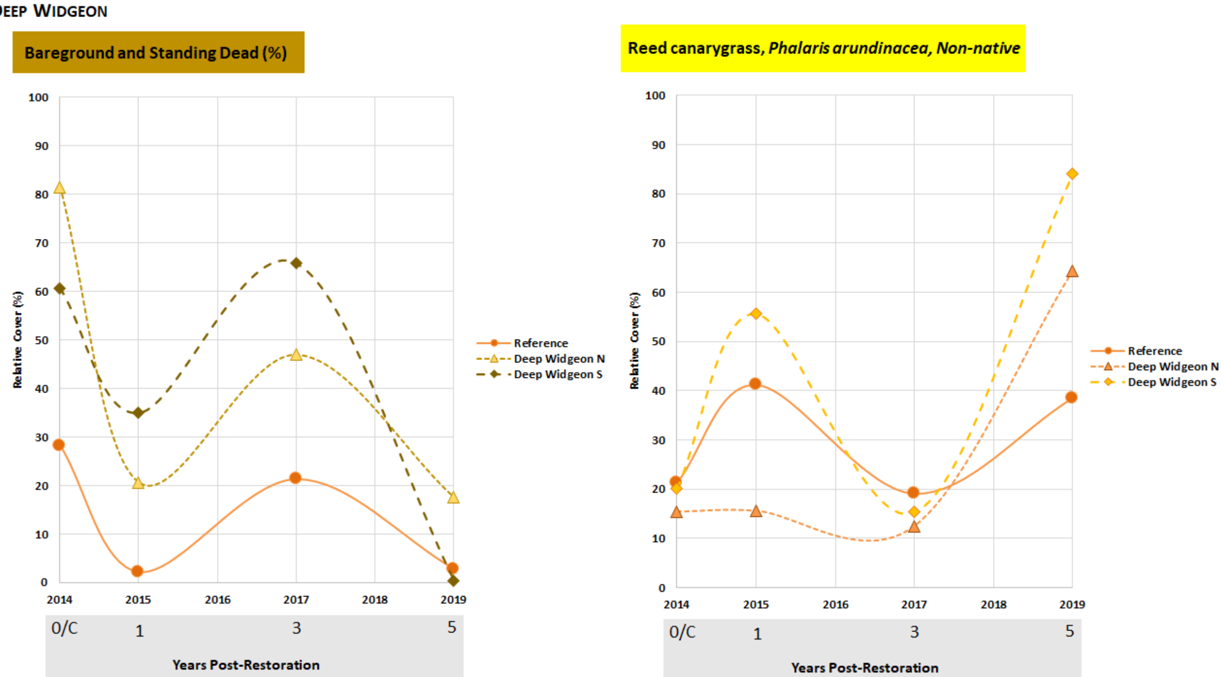

Figure 45. North Unit Phase 2: Deep Widgeon Restoration and Reference Site Bareground & Standing Dead, and Non-Native, Reed Canarygrass, *Phalaris arundinacea,* Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 16-Table 18.

Millionaire

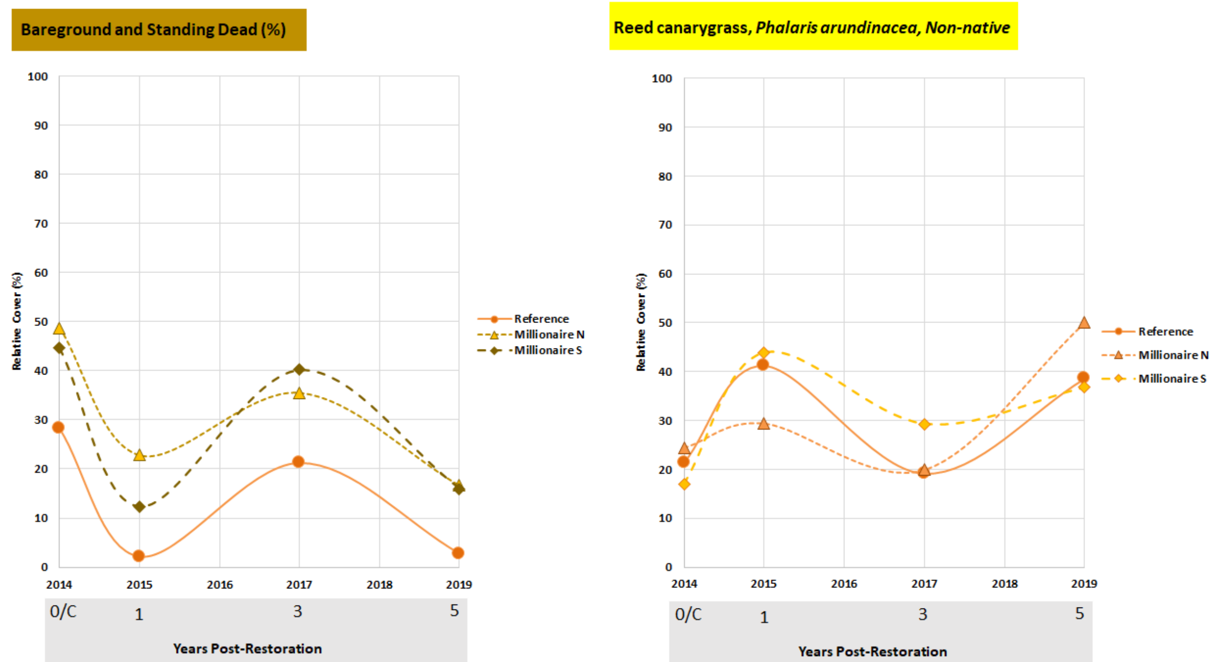

Figure 46. North Unit Phase 2: Millionaire Restoration and Reference Site Bareground & Standing Dead, and Non-Native, Reed Canarygrass, *Phalaris arundinacea,* Plant Community Abundance (% cover) Overtime. C stands for the construction year. More plant community details can be found in Table 16-Table 18.

Deep Widgeon

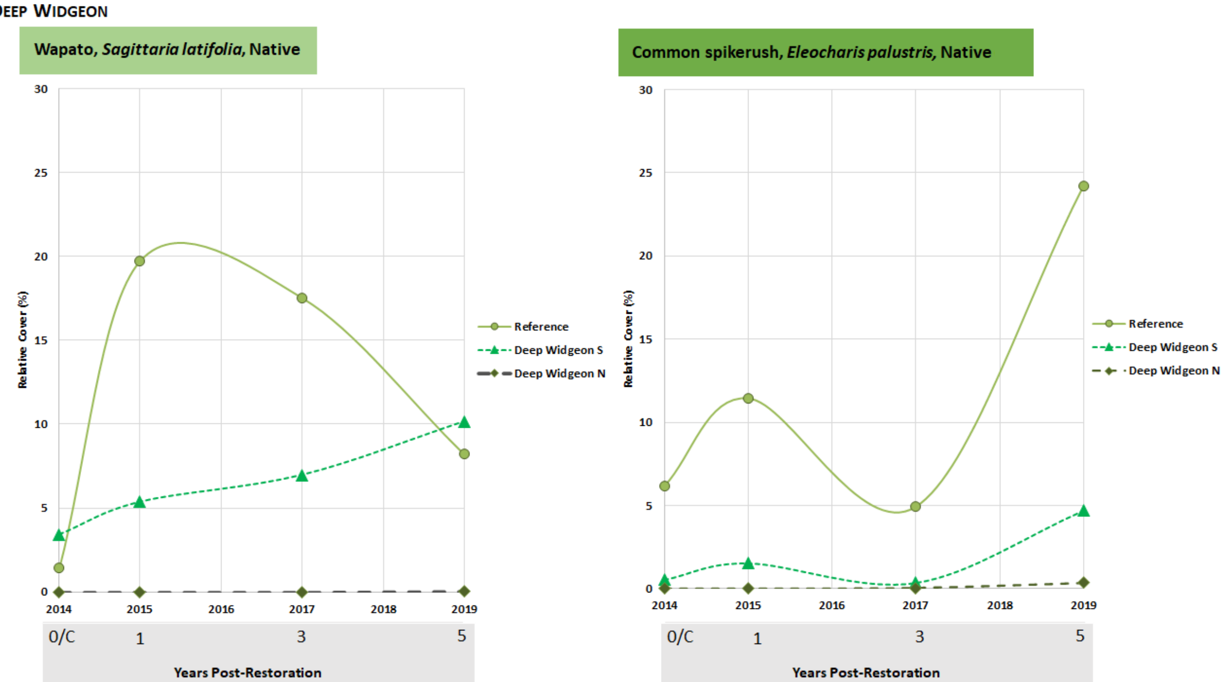

Figure 47. North Unit Phase 2: Deep Widgeon Restoration and Reference Site Native, Wapato, *Sagittaria latifolia, &* Common spikerush, *Eleocharis palustris,* Plant Community Abundance (%

cover) C stands for the construction year. More plant community details can be found in Table 16-Table 18.

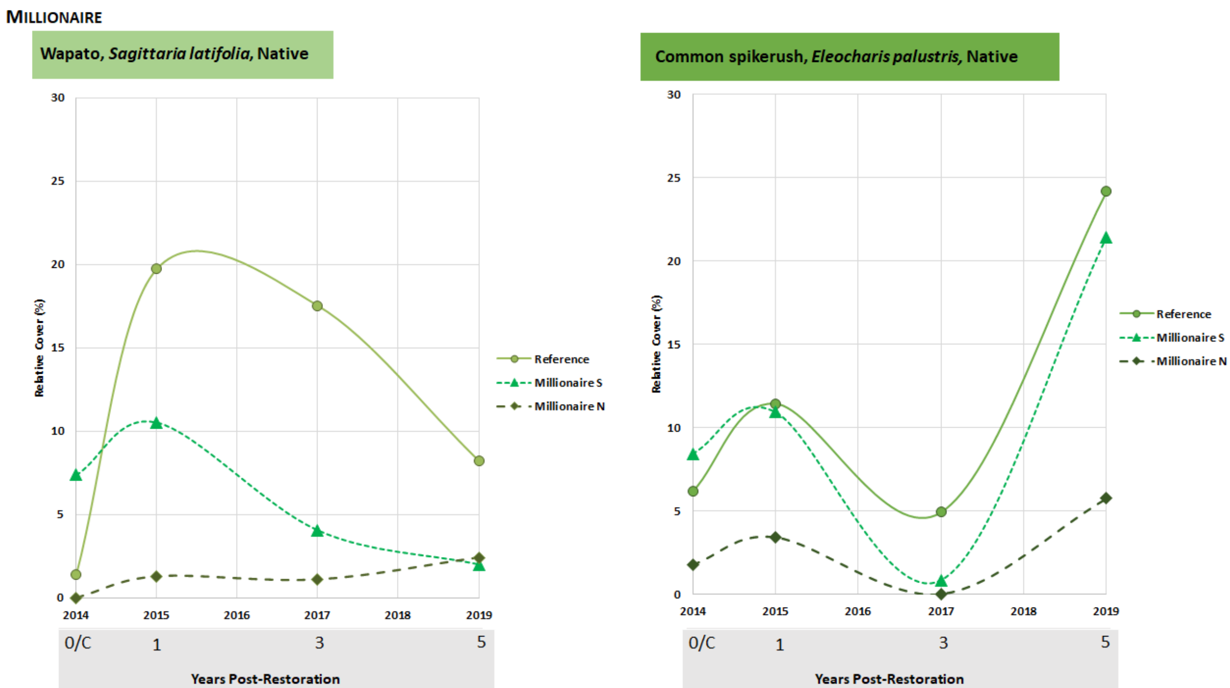

Figure 48. North Unit Phase 2: Millionaire Restoration and Reference Site Native, Wapato, *Sagittaria latifolia, &* Common spikerush, *Eleocharis palustris,* Plant Community Abundance (% cover) C stands for the construction year. More plant community details can be found in Table 16-Table 18.

Table 16. North Unit Phase 2 Restoration and Reference Site Number of Samples Plots Surveyed and Mean (±SD) Elevation, and Plant Community Relative Cover (%) Summarized by Years Post-Restoration.

Mean Relative Cover (%)	Monitoring Areas	2014		2015		2017		2019	
		Pre		Year 1		Year 3		Year 5	
		Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
Number of sampling plots	Reference	36		69		69		70	
	Deep Widgeon N	36		36		36		36	
	Deep Widgeon S	36		36		36		37	
	Millionaire N	36		36		36		36	
	Millionaire S	36		36		36		36	
Elevation, m, NAVD88	Reference	2.5	0.2	2.8	0.2	2.8	0.2	2.7	0.2
	Deep Widgeon N	3.2	0.1	3.2	0.6	2.7	0.6	3.1	0.6
	Deep Widgeon S	2.9	0.1	3.0	0.1	2.6	0.1	2.9	0.1
	Millionaire N	2.8	0.1	2.9	0.4	2.5	0.4	2.8	0.4

Mean Relative Cover (%)	Monitoring Areas	2014		2015		2017		2019	
		Pre		Year 1		Year 3		Year 5	
		Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
	Millionaire S	2.8	0.2	2.9	0.2	2.4	0.2	2.8	0.2
Bareground (%)	Reference	24.2	21.6	1.8	5.6	5.6	14.6	2.8	7.7
	Deep Widgeon N	0.0	0.2	13.9	19.8	32.0	43.3	17.5	21.4
	Deep Widgeon S	42.2	12.5	7.7	16.0	36.8	48.2	0.4	2.5
	Millionaire N	37.2	18.6	17.6	21.6	0.5	1.2	16.5	19.5
	Millionaire S	40.0	18.1	11.6	17.4	30.9	38.4	15.7	24.5
Standing dead (%)	Reference	4.1	7.1	0.4	2.1	15.7	19.3	0.0	0.0
	Deep Widgeon N	81.4	16.2	6.8	17.0	14.9	24.8	0.1	0.3
	Deep Widgeon S	18.4	20.0	27.2	19.4	28.9	30.7	0.0	0.0
	Millionaire N	11.5	14.0	5.2	13.6	35.0	37.8	0.2	0.8
	Millionaire S	4.7	9.1	0.6	2.8	9.4	20.3	0.2	0.9
Native (%)	Reference	24.3	19.3	43.5	29.1	35.0	23.8	49.6	33.5
	Deep Widgeon N	0.1	0.4	32.6	21.1	0.8	1.6	3.2	11.0
	Deep Widgeon S	8.2	13.3	8.9	14.1	7.6	17.5	15.9	27.6
	Millionaire N	16.5	19.6	14.8	16.5	1.8	3.9	11.7	11.4
	Millionaire S	22.7	19.5	38.8	31.7	7.1	11.3	38.6	31.1
Non-native (%)	Reference	21.6	28.9	42.5	36.2	20.4	21.2	40.9	36.7
	Deep Widgeon N	15.4	15.5	30.9	26.9	12.7	16.3	65.4	45.3
	Deep Widgeon S	20.5	16.0	56.0	21.1	15.3	22.2	84.0	27.8
	Millionaire N	26.9	22.9	48.5	33.3	20.3	27.6	60.2	36.5
	Millionaire S	18.5	25.4	47.6	41.3	29.4	37.6	39.4	38.4

Table 17. North Unit Phase 2 Restoration and Reference Site Mean (±SD) Dominant Plant Species Relative Cover (%) Summarized by Years Post-Restoration.

Dominate Species		Monitoring Areas	2014		2015		2017		2019	
			Pre		Year 1		Year 3		Year 5	
			Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
Non-native	Reed canarygrass, *Phalaris arundinacea*	Reference	21	29	41	37	19	22	39	37
		Deep Widgeon N	15	15	16	26	12	16	64	46
		Deep Widgeon S	20	16	56	22	15	22	84	28
		Millionaire N	24	24	29	34	20	28	50	41
		Millionaire S	17	25	44	39	29	38	37	38
	Spotted ladysthumb, *Polygonum persicaria*	Reference	0	0	1	2	1	3	2	3
		Deep Widgeon N	0	0	0	0	0	0	0	0
		Deep Widgeon S	0	0	0	0	0	0	0	0
		Millionaire N	2	4	0	0	0	0	9	16
		Millionaire S	0	1	0	1	0	0	1	4

Dominate Species		Monitoring Areas	2014		2015		2017		2019	
			Pre		Year 1		Year 3		Year 5	
			Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
Native	Common spikerush, *Eleocharis palustris*	Reference	6	12	11	17	5	7	24	27
		Deep Widgeon N	0	0	0	0	0	0	0	2
		Deep Widgeon S	1	2	2	5	0	2	5	12
		Millionaire N	2	4	3	6	0	0	6	10
		Millionaire S	8	12	11	12	1	2	21	27
	Water purslane, *Ludwigia palustris*	Reference	2	5	2	5	3	6	2	4
		Deep Widgeon N	0	0	1	2	0	1	1	2
		Deep Widgeon S	3	9	1	2	0	2	0	0
		Millionaire N	12	14	2	4	0	0	0	1
		Millionaire S	0	1	8	12	0	0	6	9
	Wapato, *Sagittaria latifolia*	Reference	1	3	20	17	18	18	8	8
		Deep Widgeon N	0	0	0	0	0	0	0	0
		Deep Widgeon S	3	6	5	8	7	17	10	17
		Millionaire N	0	0	1	4	1	3	2	3
		Millionaire S	7	9	11	11	4	9	2	4

Trajectories: Species Richness

Over the five-year time frame, overall mean total species richness (TSR) has increased across the Reference site and the Millionaire North and South monitoring areas; with mean the reference site hosting a mean TSR of 3.8 in 2014 and 5.2 in 2019, Millionaire North had a mean TSR of 2.8 in 2014 (pre) and 4.1 in 2019 (5 yr post), and Millionaire South had a mean TSR of 3.8 in 2014 and 5.2 in 2019 (Table 18). Across the Deep Widgeon monitoring areas TSR has remained below the reference site levels and relatively unchanged, with Deep Widgeon North having a TSR of 1.3 in 2014 and 1.7 TSR in 2019, and Deep Widgeon South having a TSR of 2.2 in 2014 and 1.8 in 2019 (Table 18). Correspondingly, Deep Widgeon monitoring areas have experienced very little change in both mean native and non-native species richness pre and post restoration (Table 18). While, Millionaire North and South have experienced an increase in native species richness similar to what has been observed across the reference site (Table 18). Overall, plant community Shannon Diversity and Evenness has declined post-restoration across the North Unit Phase 2 monitoring areas, remaining below reference level, except for Millionaire South which holds the most similar conditions to the reference site (Table 18).

Overall, these shifts in species richness over time across the restoration site correspond to the changes also seen across the vegetation communities and are similarly impacted by the differences in wetland elevations restored (Figure 39). High marsh areas, with reed canarygrass, *Phalaris arundinacea*, cover such as those seen across the Deep Widgeon monitoring areas, will likely remain relatively low in native species richness due to the resistance of these zones to change (Kidd 2017), correspondingly very low marsh-mudflat zones dominated by native aquatics are also not expected to accumulate further native species richness, given these habitats tend to accumulate less species than well drained marsh zones. The Northern monitoring areas, particularly Deep Widgeon North, are composed of both these very high and very low marsh elevations, which will result in a slower accumulation of native species overtime compared to the South monitoring areas (Figure 39). Millionaire South monitoring area is primarily composed of low to mid-high marsh elevation and should continue to accumulate native species richness similar to those observed at the reference site. Continued heavy grazing may have long-term impacts to native plant communities across all sites.

Table 18. North Unit Phase 2 Restoration and Reference Site Mean (±SD) Species Diversity Metrics Summarized by Years Post-Restoration.

Mean Relative Cover (%)	Monitoring Areas	2014		2015		2017		2019	
		Pre		Year 1		Year 3		Year 5	
		Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
Total Species Richness	Reference	3.8	1.8	4.0	2.2	5.7	2.3	5.2	2.5
	Deep Widgeon N	1.3	0.7	4.0	2.7	2.6	1.6	1.7	2.5
	Deep Widgeon S	2.2	1.6	1.9	1.4	1.1	1.1	1.8	1.0
	Millionaire N	2.8	1.5	4.3	2.2	1.8	1.0	4.1	2.5
	Millionaire S	3.8	1.5	5.5	2.3	2.2	1.7	5.4	2.8
Native Species Richness	Reference	2.8	1.5	3.0	2.0	4.0	1.7	3.7	1.9
	Deep Widgeon N	0.1	0.4	2.3	1.6	0.8	1.4	0.5	1.2
	Deep Widgeon S	1.1	1.5	0.8	1.2	0.4	0.7	0.8	0.9
	Millionaire N	1.4	1.4	2.4	1.8	0.8	0.9	2.3	1.9
	Millionaire S	2.7	2.0	3.9	2.7	1.3	1.6	3.8	2.1
Non-native Species Richness	Reference	0.9	0.6	1.0	0.7	1.1	0.7	1.4	0.8
	Deep Widgeon N	1.0	0.0	1.6	1.4	0.8	0.7	1.1	1.0
	Deep Widgeon S	1.1	0.4	1.1	0.2	0.6	0.5	1.0	0.0
	Millionaire N	1.3	0.5	1.8	1.2	0.9	0.5	1.5	1.0
	Millionaire S	1.1	0.9	1.6	1.2	0.8	0.6	1.4	1.1
Shannon Diversity Index	Reference	0.64	0.29	0.78	0.44	0.80	0.39	0.85	0.45
	Deep Widgeon N	0.31	0.15	0.94	0.46	0.32	0.26	0.13	0.34
	Deep Widgeon S	0.57	0.26	0.45	0.28	0.25	0.26	0.27	0.37
	Millionaire N	0.67	0.27	0.72	0.38	0.23	0.13	0.55	0.41
	Millionaire S	0.66	0.24	0.90	0.46	0.28	0.25	0.75	0.45
Evenness	Reference	0.53	0.24	0.59	0.22	0.45	0.19	0.52	0.21
	Deep Widgeon N	0.12	0.24	0.49	0.36	0.31	0.22	0.07	0.17
	Deep Widgeon S	0.34	0.38	0.29	0.37	0.32	0.34	0.28	0.33
	Millionaire N	0.55	0.38	0.51	0.20	0.13	0.16	0.34	0.26
	Millionaire S	0.49	0.20	0.52	0.22	0.31	0.34	0.43	0.24

Drone Imagery and Plant Community Modeling Results
Multiple drone flights have been conducted across North Unit Phase 2, both in 2017 and in 2019. At the time of this report these data are still under analysis and a detailed trajectory report of these data will be provided in the 2021 AEMR synthesis report.

Soil Monitoring Results
Re-introducing flooding to previously drained agricultural soils has significant impacts on soil characteristics and creates the biogeochemical template needed for wetland plant community and habitat restoration. Additionally, restoration actions such as scrapping down the topsoil to created lower elevations and channels can alter soil composition, revealing mineral soils and causing compaction. These restoration actions and soil manipulations can further impact plant community recovery. Monitoring soil conditions pre- and post-restoration can provide insight into the mechanisms causing restoration success and failure and LCEP has recently begun consistently monitoring soil conditions as part of their Level 2 AEMR. Due to the recent adoption of these monitoring methods we only have limited data for North Unit Phase 2 soil conditions: no soil data was collected at Deep Widgeon North and no soil salinity data was collected due to equipment issues during surveying. However, while these soil data are limited, they still provide valuable insight into the site conditions and how they compare across monitoring areas and to the reference site.

Wetland elevations from across the soil monitoring locations echo what was identified in the vegetation surveys with all monitoring areas across the North Unit Phase 2 restoration site being slightly higher in elevation than the reference site (Figure 39, Figure 49). Within the restoration site, the Millionaire South monitoring area had both the greatest similarity of plant community distributions and elevation ranges to the reference site (Figure 50), while sampling areas within Millionaire North and Deep Widgeon South were higher in elevation and correspondingly hosted more non-native and non-native mixed plant communities.

In year 5 post-restoration (saturated) soil pH levels were generally found to be neutral across all sites, however the restoration sites hosted slightly more acidic soil conditions (5.9-6.2) compared to the reference site (6.5) (Figure 51, Table 19). Overall, soil pH levels were consistent with non-native, reed canarygrass, *Phalaris arundinacea, and* non-native mix areas hosting lower soil pH levels (<6.5) across both the restoration and reference sites. Native, spikerush, *Eleocharis palustris,* showed greater variability and lower soil pH levels across the restoration sites (5.8) compared to the reference site (6.6) (Figure 52, Table 20). Native, wapato, *Sagittaria latifolia* plant communities showed a similar trend, with soil pH levels being lower on average at the reference (7.2) site compared to the restoration sites (6.6) (Figure 52, Table 20).

Soil oxygenation can influence soil pH in wetlands, with both very low and high oxygen levels resulting in lower pH levels (Seybold et al. 2002). Correspondingly, the Deep Widgeon South and Millionaire North also reported greater soil ORP conditions (more oxygen) overall (Figure 53) and in the Native, spikerush, *Eleocharis palustris*, monitoring areas compared to both the

Millionaire South and the Reference site (Figure 54, Table 20). The mean ORP at Deep Widgeon South was 312 mV, Millionaire North 188 mV, Millionaire South 209 mV, and the reference site 201 mV (Table 19Table 14). The greatest variability in soil ORP conditions was observed across the Millionaire North sampling area, which is composed of both high and very low elevations (Figure 50), resulting in both high and very low soil ORP conditions, as soil ORP conditions tends to follow the elevation gradient, with greater ORP levels observed in the high marsh and lower ORP level in low marsh zones due to greater soil moisture saturation (Figure 54, Table 20).

Plant communities across the reference and restoration sites were clearly segregate along elevation and soil ORP gradients, with reed canarygrass, *Phalaris arundinacea* growing primarily above 2.7 meters in elevation and > 150 mV soil ORP, and wapato*, Sagittaria latifolia,* and spikerush, *Eleocharis palustris* primarily growing below 2.7 meters and <150 mV soil ORP (Figure 54, Figure 55, Table 20). Reed canarygrass, *Phalaris arundinacea, and* wapato*, Sagittaria latifolia,* where also found growing in the mid-high marsh zone together spanning the elevation and ORP gradient, which indicates the typical pattern of growth found in these species (*Sagittaria latifolia* in the low marsh) may be a result of competition with other high marsh plant species rather (like reed canarygrass) than a result of suitable soil conditions alone (Kidd 2017). Indicating that in the absence of reed canarygrass and other competitors, wapato would grow across the low and mid-high marsh zones. Soil conditions across all sites echo the differences in wetland soil saturation along the elevation gradient and can inform adaptive management actions. Overall, wetland plant community zonation is clear along both the elevation and soil ORP gradients found at the reference and restoration sites (Figure 54, Figure 55, Table 20). The identification of these soil gradients indicates that wetland soil biogeochemistry conditions are recovering at the restoration site and are positively influencing wetland plant community development (Figure 54, Figure 55, Table 20).

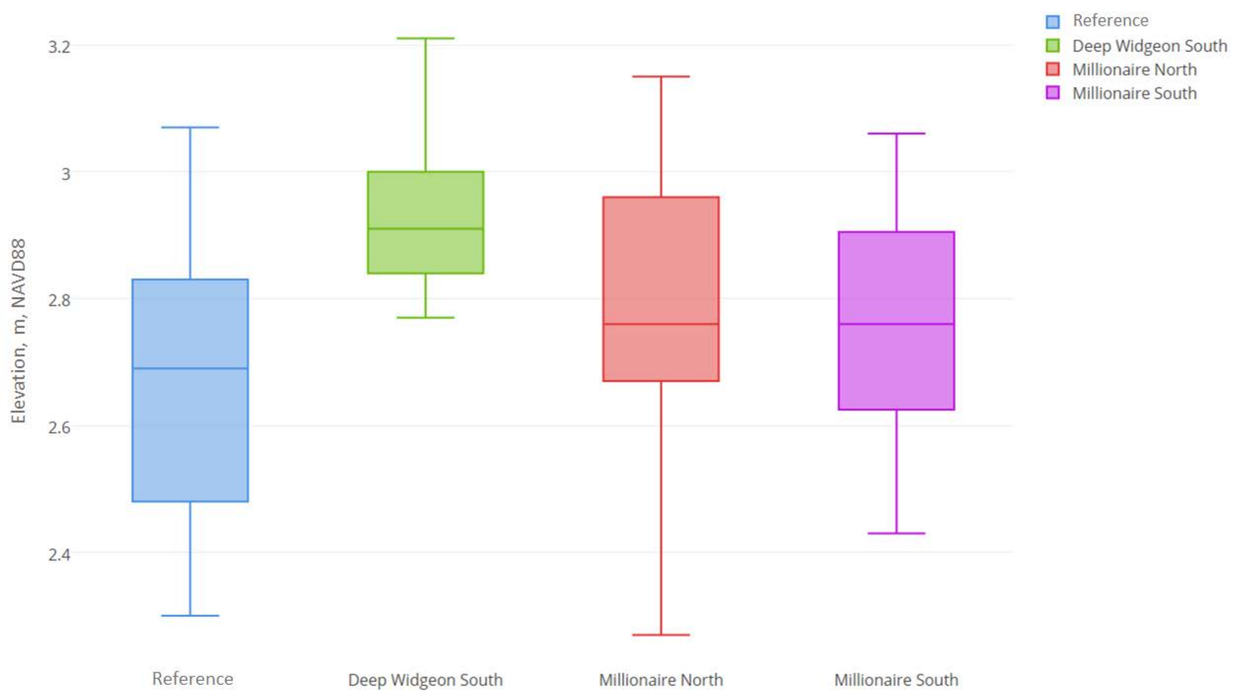

Figure 49. North Unit Phase 2 and Reference Site Elevation Ranges (m, NAVD88) of Soil Sampling Locations 5-years Post-Restoration (2019). A summary of all soil data can be found in Table 19.

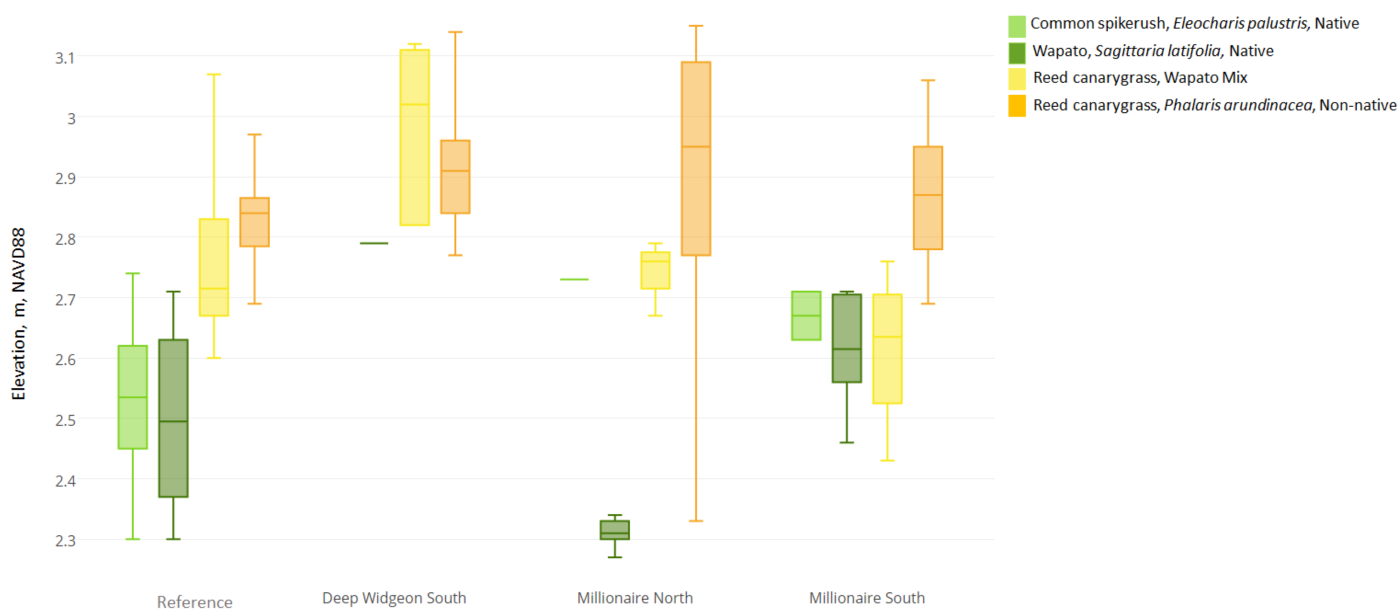

Figure 50: North Unit Phase 2 and Reference Site Elevation Ranges (m, NAVD88) Across the Dominant Native and Non-native Plant Communities, 5-years Post-Restoration (2019). A summary of all soil data can be found in Table 19 and Table 20.

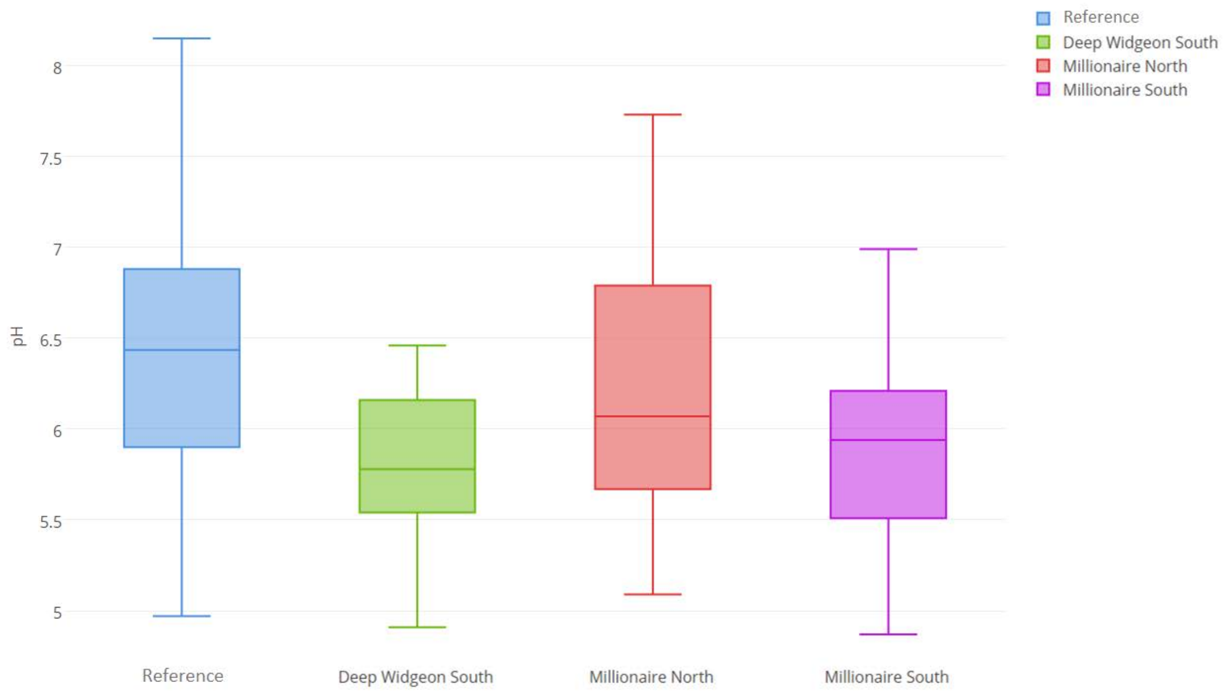

Figure 51: Boxplot of North Unit Phase 2 Restoration and Reference Sites (2019) soil pH conditions (low tide). A summary of all soil data can be found in Table 19.

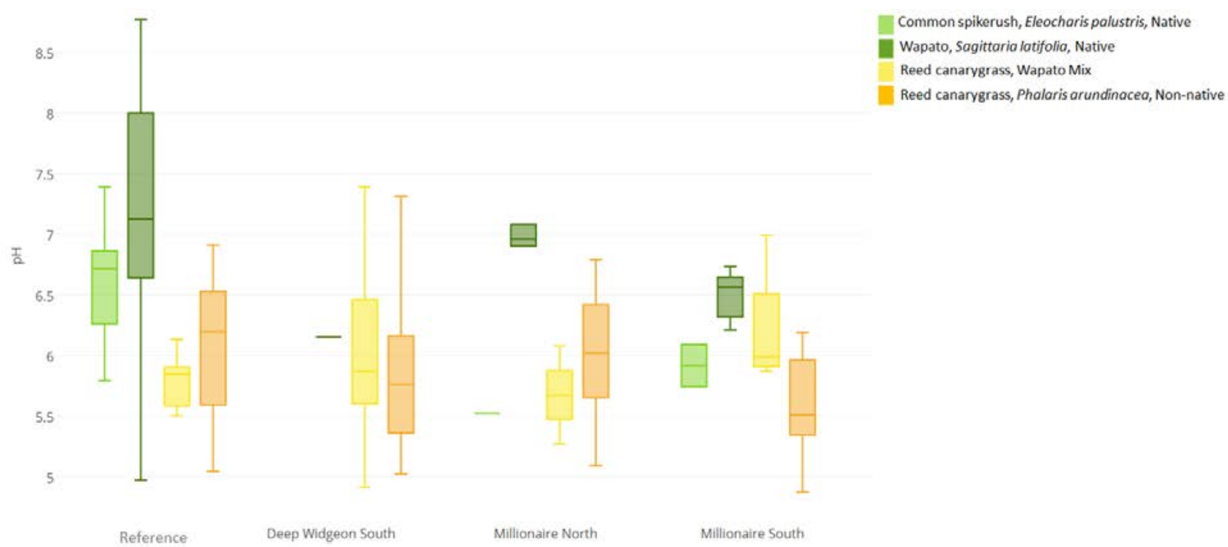

Figure 52: Boxplot of North Unit Phase 2 Restoration and Reference Sites (2019) soil pH conditions (low tide) Across the Dominant Native and Non-native Plant Communities. A summary of all soil data can be found in Table 19 and Table 20.

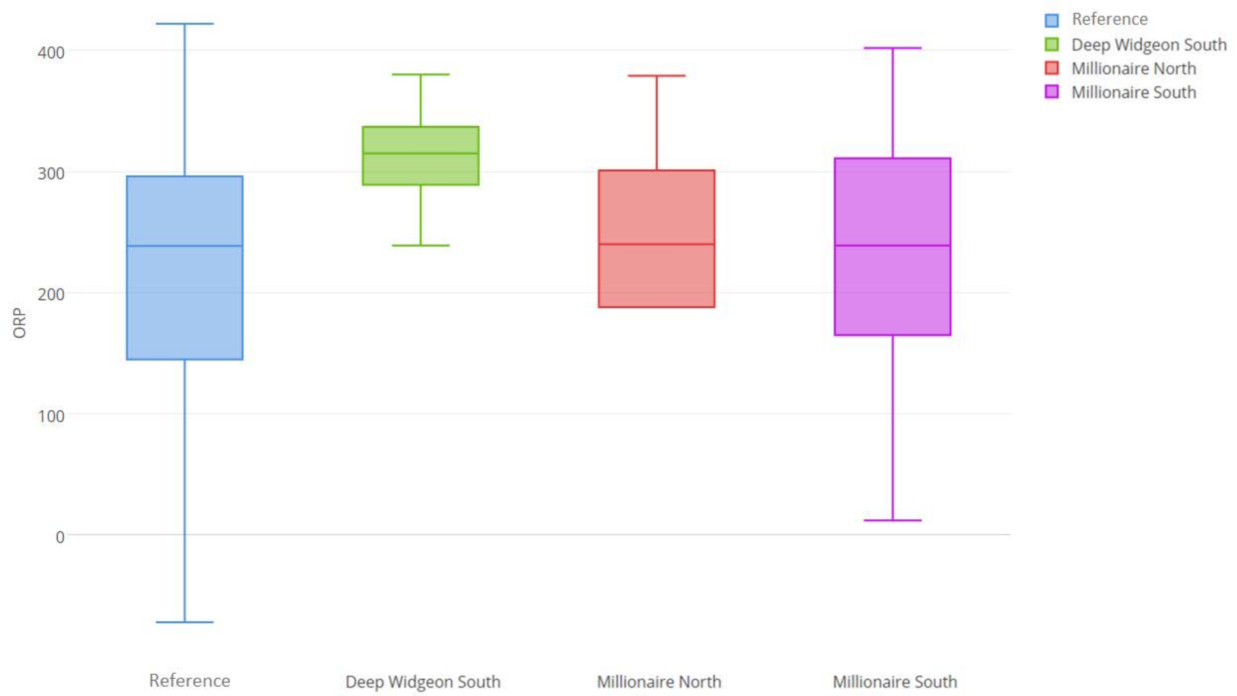

Figure 53. Boxplot of North Unit Phase 2 Restoration and Reference Sites (2019) soil ORP (mV) conditions (low tide). A summary of all soil data can be found in Table 19.

Figure 54. Boxplot of North Unit Phase 2 Restoration and Reference Sites (2019) soil ORP (mV) Conditions (low tide) Across the Dominant Native and Non-native Plant Communities. A summary of all soil data can be found in Table 19 and Table 20.

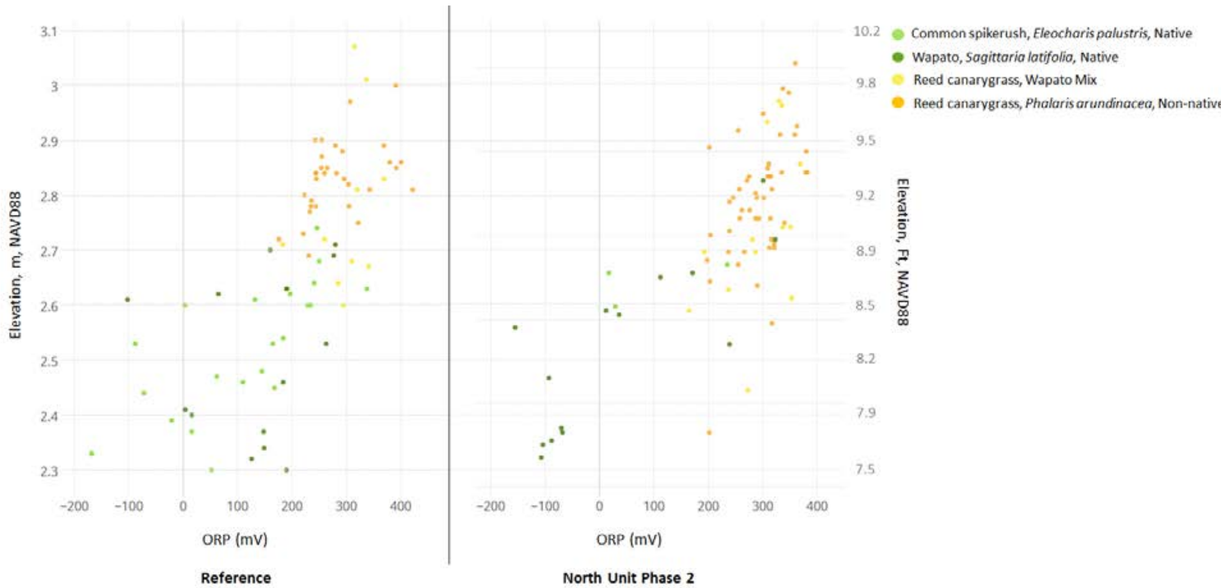

Figure 55. Wetland Soil ORP (mV) vs Elevation (m-NAVD88) across sample plots at North Unit Phase 2 Restoration and Reference Sites (2019). Dominant native and non-native plant communities highlighted, detailed information in Table 20.

Table 19. North Unit Phase 2 and Reference Site Mean (±SD) Soil Conditions (2019).

Soil Conditions	Reference		Deep Widgeon South		Millionaire North		Millionaire South	
	Mean	±SD	Mean	±SD	Mean	±SD	Mean	±SD
Sample number	62		29		30		29	
Elevation (m -NAVD88)	2.7	0.2	2.9	0.13	2.8	0.4	2.8	0.16
pH	6.5	0.8	5.9	0.6	6.2	0.7	5.9	0.6
ORP (mV)	201	133	312	36	188	156	209	142

Table 20. North Unit Phase 2 and Reference Site Mean (±SD) Soil Conditions (2019) Across the Dominant Native and Non-native Plant Communities.

Soil Conditions Across Dominant Plant Species			Elevation, m, NAVD88		pH		ORP, mV	
Species	Site	Count	Mean	±SD	Mean	±SD	Mean	±SD
Common spikerush, *Eleocharis palustris,* Native	North Unit Phase 2	3	2.7	0.1	5.8	0.3	94	123
	Reference	15	2.5	0.1	6.6	0.4	119	130
Wapato, *Sagittaria latifolia,* Native	North Unit Phase 2	14	2.5	0.2	6.6	0.5	36	164
	Reference	9	2.5	0.2	7.2	1.0	139	111
Reed canarygrass and Wapato, Mix	North Unit Phase 2	13	2.8	0.2	6.0	0.7	294	63
	Reference	10	2.8	0.2	5.8	0.3	302	52
Reed canarygrass, *Phalaris arundinacea,* Non-native	North Unit Phase 2	52	2.9	0.2	5.8	0.5	296	51
	Reference	23	2.8	0.1	6.1	0.5	286	62

Water Surface Elevation & Water Temperature Results

Water Surface Elevation (WSE) data for North Unit Phase 2 sites and their adjacent outer channel is available from 2013 – 2018. The hydrologic patterns from the restoration sites and the adjacent outer channel are compared to Cunningham Lake, a long-term Ecosystem Monitoring Program (EMP) site, which has been used as a reference site for this restoration project. Occasionally, sensor failure occurred. Summary statistics of hydrological patterns for the lakes, the adjacent outer channel and Cunningham lake reference are presented in Table 21 for pre-restoration, construction and post-restoration periods (2013 – 2018). Sensor Elevations (m, NAVD88) have also been included for each site.

The hydrology of the reference site, Cunningham lake, is predominantly influenced by the spring freshet. The annual tidal range varies between 0.4 m to 0.6 m showing very little tidal influence. The sensor at Cunningham Lake is in the very upper reach of the channel and is therefore elevated above the lowest water levels. The average marsh elevation at Cunningham Lake is 2.7 m.

Table 21: Hydrologic Summary Statistics for Millionaire and Deep Widgeon lakes, adjacent outer channel and Cunningham lake Reference site for pre-restoration, construction and year 1 to year 4 post-restoration. All metrics are in meters, relative to the North American Vertical Datum of 1988 (NAVD88), Tidal Range is Mean Tidal Range. MWL = mean water level; MHHW= mean higher high water. Full comparative hydrographs are in Appendix B: Site Hydrographs.

Restoration Year	Monitoring Location	Sensor Elevation	MWL	MHHW	Tidal Range	Max WSE	Date of Max WSE	Period of Record	Days
2018 (Year 4)	Millionaire	2.1	3.0	3.3	0.5	5.6	May 17	Jan-Dec	365
	Deep Widgeon	1.4	2.9	3.2	0.6	5.5	May 17	Jan-Dec	365
	Outer channel	1.6	2.9	3.2	0.7	5.5	May 17	Jan-Nov	330
	Reference	2.1	3.3	3.5	0.5	7.0	May 17	Jan-Dec	365
2017 (Year 3)	Millionaire	2.1	3.5	3.7	0.5	6.1	Mar 30	Jan-Dec	365
	Deep Widgeon	1.4	3.3	3.7	0.6	6.0	Mar 30	Jan-Dec	365
	Outer channel	1.6	3.3	3.6	0.6	6.0	Mar 30	Jan-Dec	365
	Reference	2.1	2.7	3.0	0.6	4.2	Dec 30	Jan, Aug-Dec	193
2016 (Year 2)	Millionaire	2.1	3.1	3.5	0.6	4.7	Jan 22	Jan-Dec	366
	Deep Widgeon	1.4	3.1	3.4	0.6	4.4	Mar 12	Jan-Dec	366
	Outer channel	1.6	3.2	2.9	0.6	4.4	Mar 12	Jan-Dec	365
	Reference	2.1	2.7	2.9	0.4	3.6	Nov 26	Aug-Dec	152
2015 (Year 1)	Millionaire	2.1	2.9	3.3	0.7	5.6	Dec 10	Jan-Dec	365
	Deep Widgeon	1.8	2.4	2.7	0.5	5.3	Dec 10	Jan-Dec	365
	Outer channel	1.6	2.5	2.9	0.7	4.7	Dec 10	Jan-Dec	365
	Reference	2.1	2.8	3.1	0.5	4.0	May 28	Jan-Jul	209
2014 Constructed	Millionaire	2.1	3.1	3.3	0.5	4.9	Mar 10	Jan-Dec	348
	Deep Widgeon	1.8	3.2	3.4	0.4	5.0	Mar 10	Jan-Dec	350

Restoration Year	Monitoring Location	Sensor Elevation	MWL	MHHW	Tidal Range	Max WSE	Date of Max WSE	Period of Record	Days
	Outer channel	1.6	3.1	3.4	0.6	5.1	Mar 10	Jan-Dec	350
	Reference	2.1	2.9	3.2	0.4	4.7	Mar 10	Jan-Dec	365
2013 Pre-restoration	Millionaire	2.1	2.5	2.8	0.6	3.1	Dec 03	Sep-Dec	114
	Deep Widgeon	1.8	2.7	2.8	0.2	3.1	Dec 04	Sep-Dec	110
	Outer channel	1.6	2.6	3.0	0.8	3.7	Dec 02	Sep-Dec	110
	Reference	2.1	2.6	2.9	0.5	4.3	Feb 09	Jan-Dec	365

NA= Not available

From Table 21, it is observed that the hydrologic patterns in millionaire lake stabilize around year three post-restoration while the average tidal range is similar to that of Cunningham lake. Due to differences in sensor elevations at the outer reference channel and the general lack of overlap in timelines of data, we used regression models to statistically evaluate the hydrologic patterns at the sites. A multiple regression model of mean water levels for 2018 at Millionaire, the nearby outer channel and Cunningham reference shows a strong linear relationship (R^2= 0.99, p=0.000) between the three. This leads us to infer that the hydrologic patterns at millionaire are emulating those at the nearby channel and the reference site.

We also compared the hydrologic patterns of the three sites from October to December months of each monitoring year (2013-2018). These months were chosen looking at overlapping data points. Figure 56 shows the variations in water level elevations pre-restoration and the timeline of stabilization of hydrology at the restoration site. However, since 2019 data was unreliable, the pattern observed at year four post-restoration could not be shown as definitive.

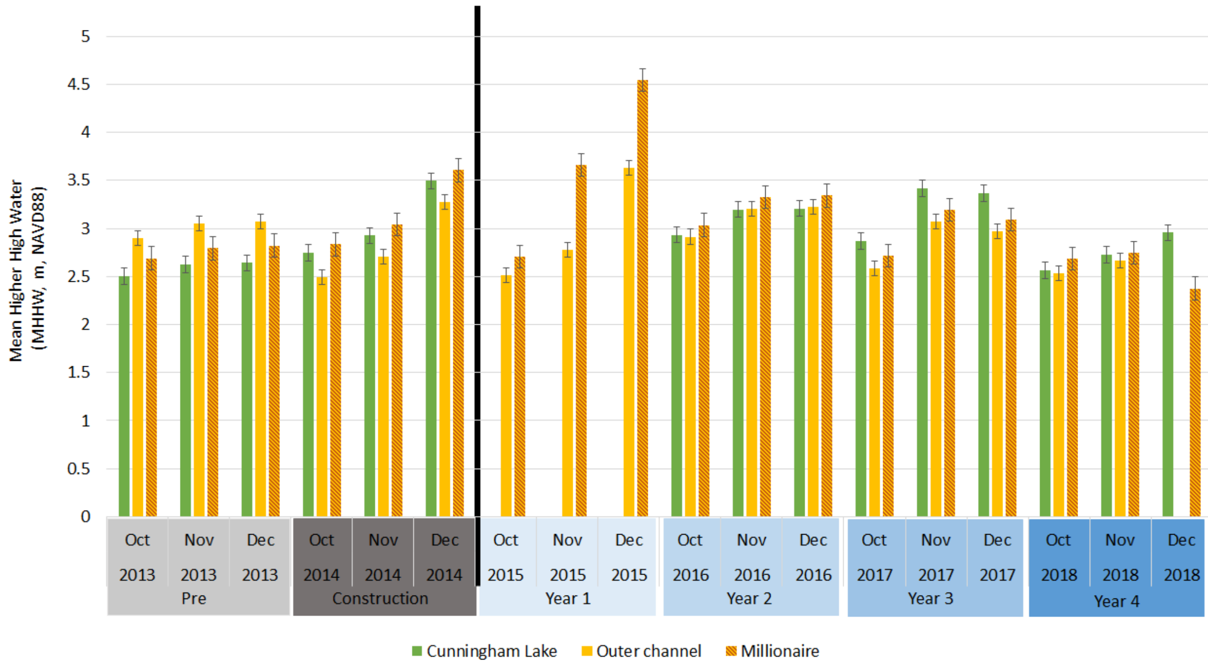

Figure 56: Bar graph representing the evolution of Millionaire and outer channel hydrology in comparison to the hydrology of Cunningham Lake for pre-restoration, construction. and years 1 to 4 post-restoration. MHHWs are used instead of MWL to avoid interference from differences in sensor elevations.

Figure 57 shows that over that over the course of post-restoration monitoring, the hydrologic pattern of Millionaire has evolved to emulate that of Cunningham Lake. However, it should also be noted that there are site-wise differences, such as differences in marsh elevations which plays an important role in inundation and plant community development.

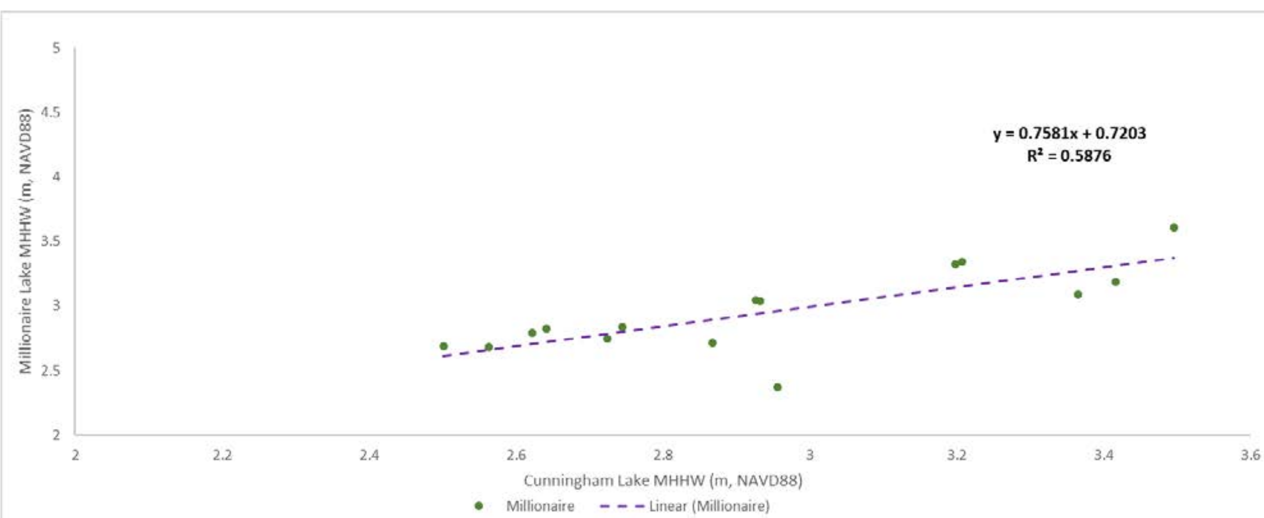

Figure 57: Scatterplot showing a positive linear relationship between MHHWs of Millionaire and Cunningham Lake for October to December months for monitoring period (2013 – 2018). MHHW= Mean Higher High Water. All elevations in m, NAVD88.

In the case of Deep Widgeon, from Table 21 it can be observed in the mean higher high water levels (MHHW) that the hydrologic stabilization after restoration is achieved around year three post-restoration. The main hydrologic drivers and annual tidal patterns at this site are similar to that of Cunningham lake due to it's location on the lower Columbia river.

Due to sensor errors in 2019, data from that year has not been included in the analysis. The sensor elevations at all three sites vary greatly, therefore, mean higher high water levels have been used for analysis than mean water levels. In 2018, a multiple regression model shows a strong linear relationship between the MHHWs of the three sites (R^2= 0.99, p=0.000), representing successful hydrologic reconnection of the site.

When MHHWs for October to December months for the monitoring period were compared across sites, it is observed that the hydrology of Deep Widgeon started stabilizing in 2016 (Figure 58). When analyzed statistically, a linear relationship was found between Deep widgeon and Cunningham Lake (R^2= 0.59, p=0.03) for the monitoring time period (Figure 59).

Figure 58: Bar graph representing the evolution of Deep Widgeon and outer channel hydrology in comparison to the hydrology of Cunningham Lake for pre-restoration, construction and years 1 to 4 post-restoration. MHHWs are used instead of MWL to avoid interference from differences in sensor elevations

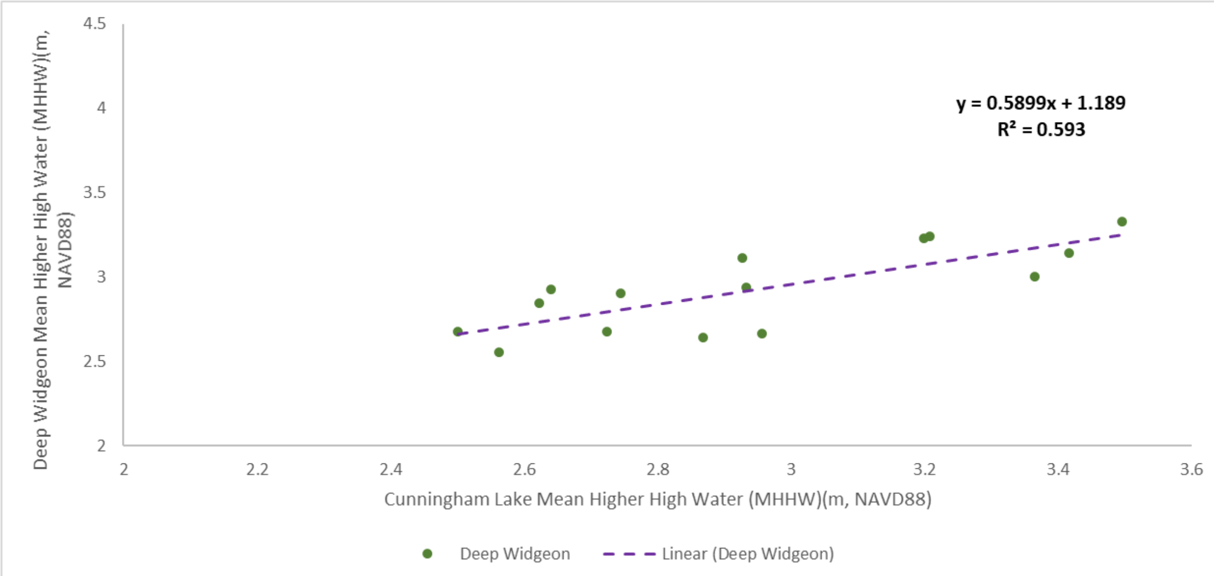

Figure 59: Scatterplot showing a positive linear relationship between MHHWs of Deep Widgeon and Cunningham Lake for October to December months for monitoring period (2013 – 2018). MHHW= Mean Higher High Water. All elevations in m, NAVD88.

Water temperature data and further analysis of hydrology data will be available in the 2020 synthesis report.

Sediment Accretion and Erosion Monitoring

North Unit Phase 2: Millionaire Lake Restoration Site

Four pairs of PVC Stakes were installed at Millionaire lake during pre-construction in 2014, however, due to cattle activity at the sites between years 1 and 3 post-restoration, 2 sed stakes were removed (Table 22). Hence, this study reports out on the remaining two pairs of sedimentation stakes at the site. It should also be noted that accretion/erosion rates at three years post was measured and calculated as differences in averages between 2015 and 2018 since no data was collected in 2016 and 2017. Accretion rates for year 5 were calculated as difference between annual averages between 2018 and 2019. These annual rates were then compared to that of Cunningham Lake to discern any observed similarities (Table 22)..

In 2019, sediment accretion and erosion rates at Millionaire lake ranged from -0.5 to 0.6 cm/year (Table 22), erosion was also observed at the reference site and the Deep Widgeon. Generally, however, very little pattern overtime was discernable across sites, this can be attributed to breaks in monitoring as well as high variability in the data (as shown by the standard deviation).

Table 22: Sediment accretion and erosion annual rates (cm/yr) for Millionaire Lake (Mill), Deep Widgeon (DW), and Cunningham Lake reference sites.

Site Bench	Elevation (m, NAVD88)	Annual Summary	Pre-Restoration	Post-Restoration		
			2014	2015	2018	2019
			Construction	Year 1	Year 3	Year 5
Mill SED 3	**3.13**	**Rate**	**ND***	**0.7**	**-0.1**	**-0.5**
		±SD	ND	2.4	0.7	0.9
Mill SED 4	**2.9**	**Rate**	**ND**	**-0.6**	**1.2**	**0.6**
		±SD	ND	3.4	1.3	1.8
DW SED-1	**3.56**	**Rate**	**ND***	**3.5**	**0.6**	**-3.2**
		±SD	ND	2.5	4.6	1.25
DW SED-2	**3.29**	**Rate**	**ND**	**1.5**	**-1.4**	**0.0**
		±SD	ND	2.3	0.6	0.8
DW SED-3	**3.17**	**Rate**	**ND**	**-1.9**	**0.0**	**-0.8**
		±SD	ND	5.7	4.2	2.1
Reference CLM-1	**3.53**	**Rate**	**-0.5**	**0.9**	**1.5**	**-1.1**
		±SD	1.0	1.3	1.5	1.2

*ND represents No Data

North Unit Phase 2: Deep Widgeon Restoration Site

Three pairs of PVC stakes were installed at Deep Widgeon in 2014, denoting areas of high, mid and low elevations (m, NAVD88). The measurements and calculations for accretion and erosion rates at Deep Widgeon also follow the same procedure as Millionaire Lake. In 2019, erosion was observed at all three sedimentation stakes. The annual erosion rate ranged from -3.2 to 0 cm/year (Table 22). When annual rates of SED-1 (high marsh elevation) was compared to that of Cunningham lake, it appears that the two sites show similar trends of sediment movement. However, this observation could not be explored deeply due to lack of data at Cunningham lake low elevation sedimentation stakes and the general high variability in the measurements collected.

Channel cross-section data were still under analysis at the time of writing this report and will be reported out on during the 2020 synthesis report.

Juvenile Salmonids

Two sites, Millionaire and Deep Widgeon Lakes were sampled in 2019 to try to identify fish communities and whether salmonids were present following previous habitat restoration effort (Figure 60-Figure 63). Widgeon Lake was sampled on April 16th and 17th and Millionaire Lake was sampled April 17th and 18th. The Deep and Widgeon Lakes area will be referred to as only Widgeon Lake in the following report for ease of interpretation and reporting. High and low water conditions can vary dramatically at these mid-river site and images of these conditions have been provided for reference, Figure 60-Figure 63, the high water conditions are characteristic of these sites in April when fish sampling occurred, and provide migrating and resident fish the greatest access to these habitats.

Figure 60. Millionaire Lake located in Cunningham slough on the Oregon side of the Columbia River. ML1 and ML2 indicate the two primary sampling location. Site sampled on April 17 - 18, 2019. Map depicts Low water levels commonly observed in summer and fall months.

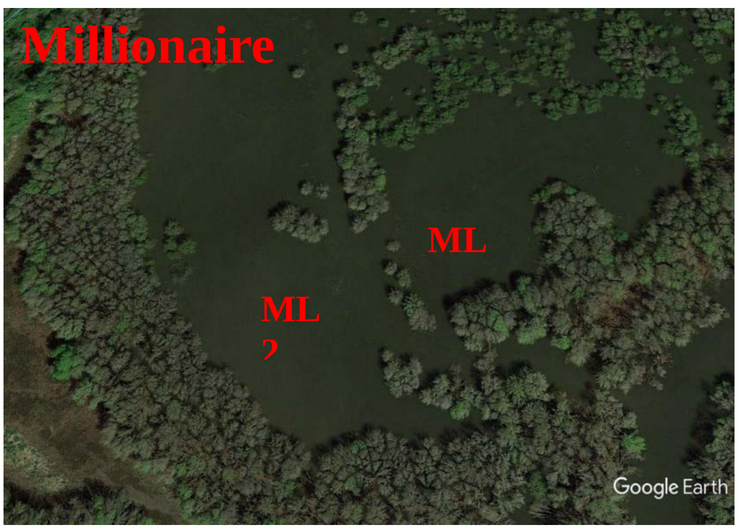

Figure 61. Millionaire Lake located in Cunningham slough on the Oregon side of the Columbia River. ML1 and ML2 indicate the two primary sampling location. Site sampled on April 17 - 18, 2019. Map depicts High water levels observed during sampling and those commonly seen during winter and spring months.

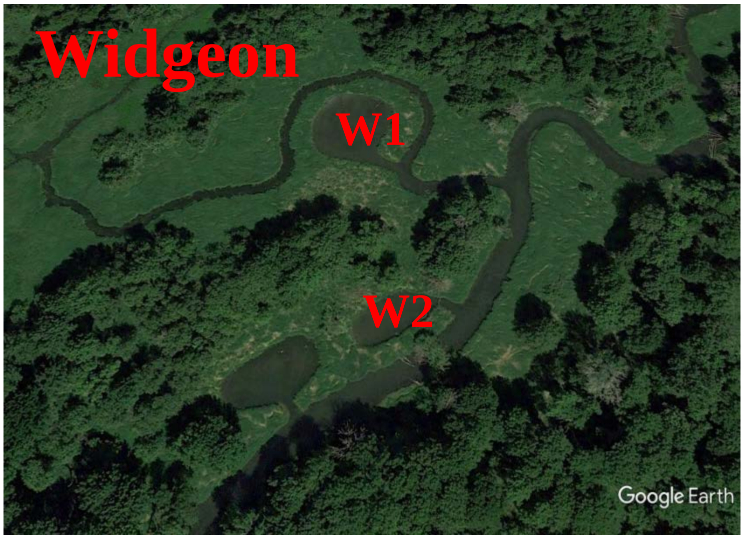

Figure 62. Widgeon Lake located in Cunningham slough on the Oregon side of the Columbia River. W1 and W2 indicate the two primary sampling location. Site sampled on April 16 - 17, 2019. Map depicts Low water levels commonly observed in summer and fall months.

Figure 63. Widgeon Lake located in Cunningham slough on the Oregon side of the Columbia River. W1 and W2 indicate the two primary sampling location. Site sampled on April 17 - 18, 2019. Map depicts High water levels commonly observed in winter and spring months.

Chinook salmon

Chinook salmon were caught at all sampling sites on Millionaire and Widgeon Lakes (Figure 64 and Figure 65, Table 23). Both marked (adipose fin clip) and unmarked (no adipose fin clip) salmon were caught at all sampled locations. There were 2 unmarked and 3 marked Chinook captured at Millionaire site 1 and 2 unmarked and 8 marked Chinook captured at Millionaire site 2. There were 3 unmarked Chinook and 7 marked Chinook captured at Widgeon site 1 and 1 unmarked Chinook and 22 marked Chinook captured at Widgeon site 2 (Figure 64 and Figure 65, Table 23). Across all four sampling locations Widgeon site 1 had the greatest overall

Chinook density (Figure 65). Despite seeing both mark and unmarked Chinook salmon at all sampling sites no other salmon species were sampled or observed (Table 23). A full description of Chinook salmon conditions for both sites can be found in Table 23 and Figure 66.

Other salmon species
No other salmon species were captured at Millionaire and Widgeon lakes in 2019.

Non-Salmon Catch Results
A total of 7 species were caught at both Millionaire Lake sites and a total of 9 different species were observed at both Widgeon Lake sites (Figure 67). There were 13 different species sampled across both lakes, with 7 species being non-native to the Columbia River. Threespine sticklebacks, redsided shiner, and peamouth were the most observed species at both lakes sampled.

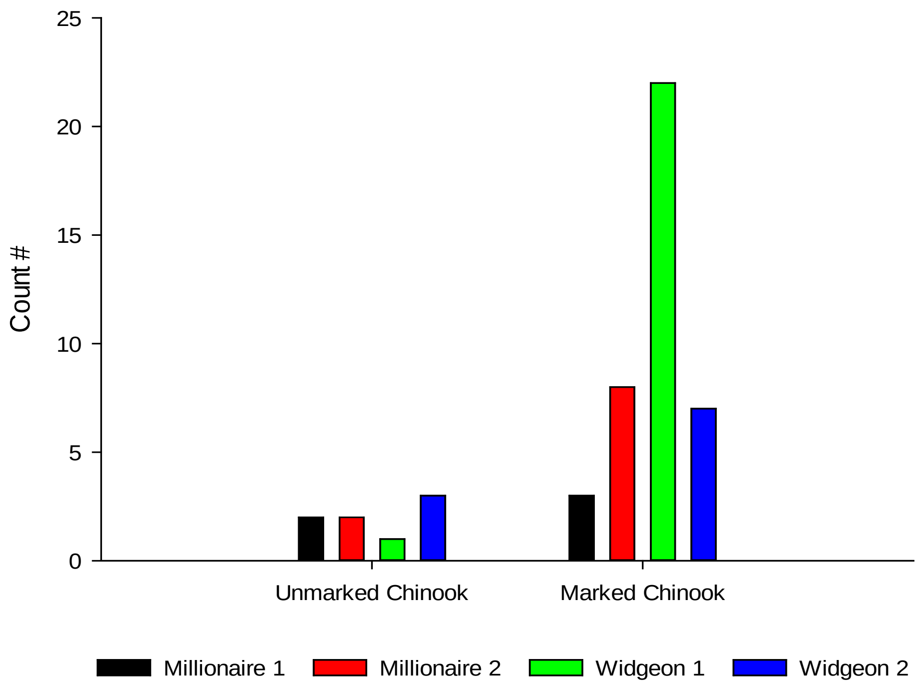

Figure 64. Total number of unmarked and marked Chinook caught in each sampling site at Millionaire Lake (Millionaire 1 & Millionaire 2) and Widgeon Lake (Widgeon 1 & Widgeon 2) in April 2019.

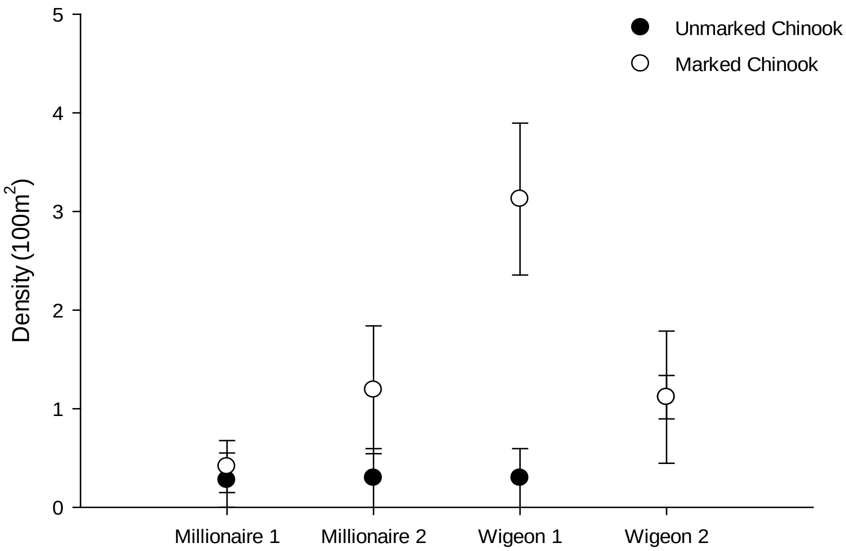

Figure 65. Unmarked and marked Chinook density (100m2) caught in each sampling site at Millionaire Lake (Millionaire 1 & Millionaire 2) and Widgeon Lake (Widgeon 1 & Widgeon 2) in April 2019. Error bars represent mean standard error (SE).

Table 23: Total number of Chinook caught, mean fork length (mm), mean weight (g), and mean Fulton's condition index (k) for Chinook salmon collected from Millionaire and Widgeon Lakes in April 2019. Numbers in parentheses represent one standard error. Yearling Chinook were included in all averages below.

	Millionaire 1		**Millionaire 2**	
Variable	**Unmarked**	**Marked**	**Unmarked**	**Marked**
Number caught	2	3	2	8
Fork Length (mm)	114 (7.01)	114 (24.64)	109 (1.01)	129 (7.06)
Weight (g)	15.95 (2.45)	17.87 (7.65)	14.3 (0.30)	22.11 (2.71)
Condition (k)	1.07 (0.03)	0.96 (0.03)	1.10 (0.01)	0.97 (0.02)

	Widgeon 1		**Widgeon 2**	
Variable	**Unmarked**	**Marked**	**Unmarked**	**Marked**
Number caught	1	22	3	7
Fork Length (mm)	65	69.63 (0.71)	104.67 (16.85)	79.43 (7.47)
Weight (g)	2.8	3.40 (0.10)	13.43 (4.96)	5.89 (1.89)
Condition (k)	1.02	1.00 (0.01)	1.02 (0.02)	1.03 (0.02)

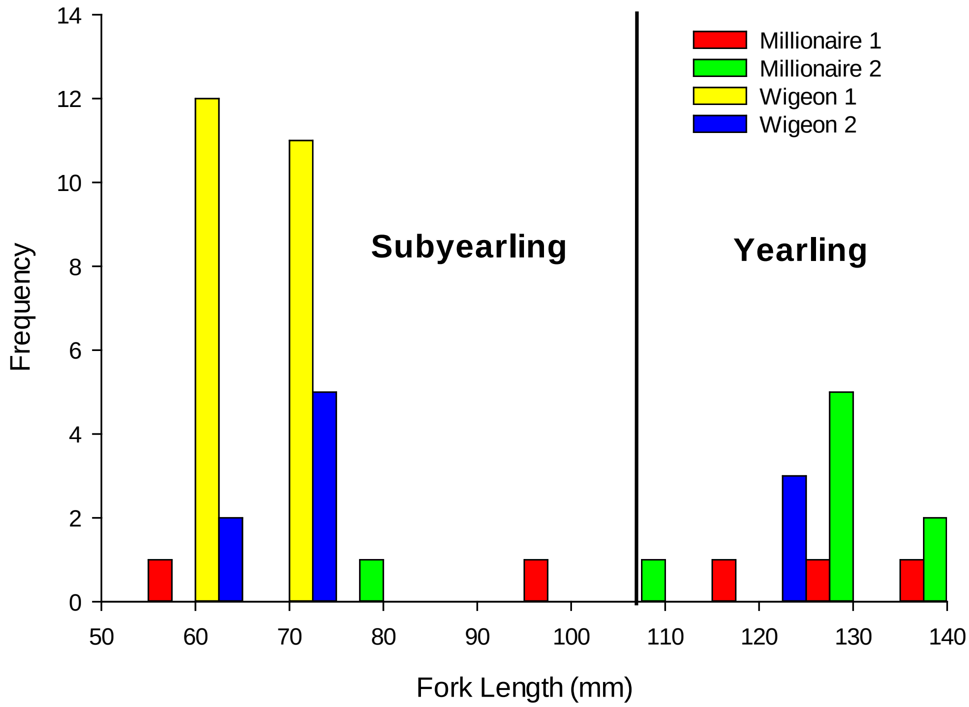

Figure 66. Length frequency of all Chinook caught at each sampling site at Millionaire Lake (Millionaire 1 & Millionaire 2) and Widgeon Lake (Widgeon 1 & Widgeon 2) in April 2019.

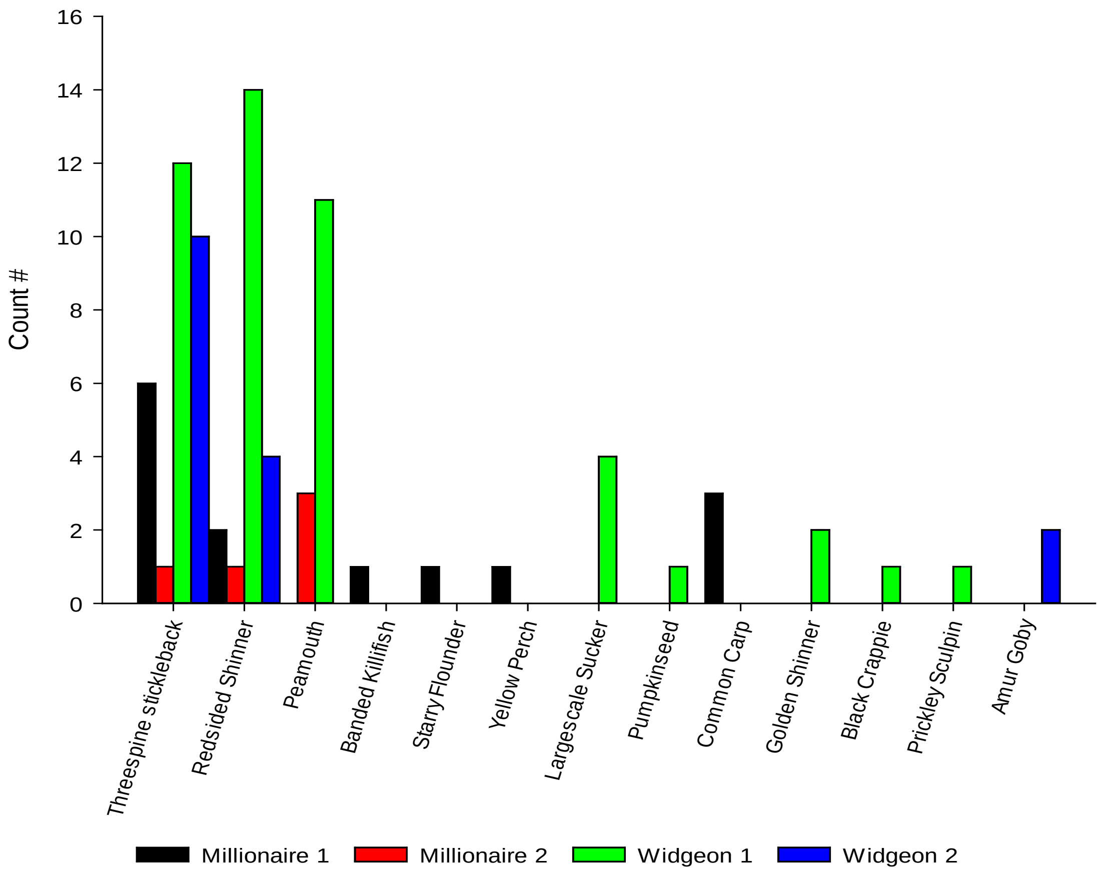

Figure 67. Total number of each fish species (non-salmon) caught at each sampling site at Millionaire Lake (Millionaire 1 & Millionaire 2) and Widgeon Lake (Widgeon 1 & Widgeon 2) in April 2019.

Horsetail Creek

Juvenile Salmonids

In 2019, the Horsetail Creek PIT detection array collected data from March 20 to October 13, 2019. Although not all 10 antennas were operating, we had coverage of four antennas on both the downstream and upstream sides of the culvert. Fifteen individual fish were detected between April 26 and August 14. Forty-seven percent (N=7) of fish detected were hatchery fall Chinook, one of which originated from the Snake River. All other fall Chinook salmon originated from the middle Columbia Basin. The second most prevalent category was hatchery spring Chinook at 27% (N=4). One hatchery Coho released in the Umatilla River was also detected. Additionally, two northern pikeminnows and one unknown (no tag data in the regional database; www.ptagis.org) fish were detected. Detection numbers and residence times are listed in Table 24.

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

Table 24: Number and residence time (max and median) of fish detected at Horsetail Creek PIT array in 2018. Residence time is a measure of elapsed time from first to last overall detection, not a measure of time spent upstream of the array. Numbers in parentheses represent the number of known wild origin fish in the total.

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

DISCUSSION

Wallacut Slough

Plant Community

Over the three-year time period since restoration has occurred we have observed significant increases in native relative plant cover and decreases in non-native relative plant cover across portions of the site (Figure 12, Table 5). Additionally, native species richness has increased since restoration, reaching reference levels across all monitoring areas. Bareground, standing dead, and non-native species such as reed canarygrass, *Phalaris arundinacea,* however, continue to dominate in the areas that were more heavily impacted during construction and where ongoing non-native plant management has been conducted. While differences in plant community recovery have been found across the site, overall soil pH, salinity, and ORP conditions between these areas indicate that they share a similar level of wetland soil recovery. Based on these results, it is hypothesized that approximately 50% of the areas currently covered in bareground, standing dead, and *P. arundinacea* will transition into a native wetland plant matrix by year 5 post-restoration, if given a break from herbicide treatments (Figure 18). On-going monitoring of both plant community and soil development at Wallacut Slough will aid us in understanding how construction and herbicide treatments impact long-term wetland plant community recovery.

Water Surface Elevation & Water Temperature

Water Surface Elevation (WSE) is an important indicator of overall site hydrology and, in addition to water temperatures, is an strong predictor for habitat conditions for juvenile salmonids (Schwartz et al, 2018). While water temperature data was still under analysis at the time of writing this report, previous studies at the site show that 7-day moving average maximum (7DMAM) temperatures were similar to the nearby outer channels but were generally warmer than the mainstem.

In 2019, instead of focusing on the 2 year flood elevation, we provide a more comprehensive description of how the site hydrology has evolved since restoration by comparing metrics across outer reference sites. Wallacut slough appears to have reached complete hydrologic connectivity in 2019 with it's adjacent channel, after taking nearly two years to stabilize. From Table 9 it is seen that annual tidal conditions and water levels emulate that of Ilwaco Slough, it's reference long-term monitoring site. The hydrographs of Wallacut Slough clearly depict the variations is WSE as well as some underlying issues. While we showed strong linear relationships between hydrology of the slough and it's reference channels and EMP site, there are some limitations that have to be addressed. Due to sensor errors and failures, crucial data such as 2019 Ilwaco Slough WSE and pre-restoration metrics could not be used to determine the entire evolutionary picture of Wallacut hydrology. Another limiting factor is the large variations in sensor elevations, which causes variations in mean lower low water (MLLW) and mean water level (MWL), which are also important indicators for OBL plant communities. Based on these results, we recommend that a deeper study needs to be made into inundation patterns along with habitat opportunity in order to effectively represent successful restoration efforts. We aim to present these in the 2020 AEM Synthesis report.

Sediment Accretion and Erosion Monitoring

Studying sediment accretion at restoration sites not only gives us an indication of the stability of the site, but also provides an understanding of tidal action in marshes closer to the mouth of the river (Callaway et al. 1997). Sedimentation stakes installed at Wallacut slough showed opposite trends, with one pair of stakes showing similar patterns to Ilwaco slough. However, the lack of elevation data makes it difficult to draw any definitive conclusions. While our study did not draw conclusions for sediment accretion at Wallacut slough, there are several studies in the lower Columbia river and estuary that infer positive results of restoration on sediment accretion. A study of restoration sites in the lower Columbia river by Borde et. Al (2011) showed that restored sites had higher rates of accretion than reference sites. In terms of site proximity, Kidd (2017) studied sediment dynamics in Youngs bay to show that low-marsh zones accreted more sediment than high marsh zones, while also studying the effect of tidal range on sediment loading at restoration sites. Based on our study and others, there are several recommendations to be made to make sediment monitoring more robust and informative. We recommend adding more sedimentation stakes to be able to profile sediment movement across the entire monitoring site, instead of localized points of measurement. Another recommendation would be to collect soil conditions data as well as dominant vegetation species information. This information will help draw important conclusions and lessons about interactions between native and non-native vegetative communities and sediment loading characteristics.

Steamboat Slough

Plant Community Results

Over the five-year time-period since restoration has occurred, we have observed significant increases in native relative plant cover and decreases in non-native relative plant cover site (Figure 27). While native plant cover has increased significantly, overall native species richness remains below reference levels. Overall, the recovery in species composition and richness across the restoration site correspond to the differences in wetland elevations restored. At year five post-restoration, high marsh areas that continue to host reed canarygrass, *Phalaris arundinacea*, cover will likely remain relatively low in native species richness and cover due to the resistance of these zones to change (Kidd 2017, Schwartz et al. 2018). Correspondingly, very low marsh-mudflat zones dominated by native aquatics are also not expected to accumulate further native species richness, given that these habitats tend to accumulate less species than well drained marsh zones. In addition, wetland plant community segregation is clear along both the elevation and soil ORP gradients found at the reference and restoration sites. The identification of these soil gradients indicates that wetland soil biogeochemistry conditions have been recovered at the restoration site and are positively influencing wetland plant community development (Figure 36). Ongoing monitoring of both plant community and soil development at Steamboat slough will aid us in understanding the timeline of long-term wetland plant community recovery, especially in high and very low marsh zones which appear to have longer recovery timelines than mid-marsh habitats.

Water Surface Elevation & Water Temperature Results

Hydrology of Steamboat Slough and Welch Island in year five restoration are exact copies of one-another, showing complete hydrologic reconnection (Table 15, Steamboat Slough). However, the pattern of stabilization was not available for analysis or reporting as these data were collected as part of Level 1 monitoring by US Army Corps of Engineers. Studies in the Columbia River and other tidal marshes have indicated that hydrologic connectivity takes years to stabilize, hence a longer monitoring period is advised. Moreover, the deployed sensor as part of Level 2 monitoring is only in the east channel, hence cannot be used to draw conclusions for the inundation patterns of the west monitoring area, however given this area is upstream of the west monitoring area – they are likely very similar. Please refer to PNNL and NMFS 2020 AEM reports for additional information on hydrology of Steamboat Slough.

Sauvie Island Phase 2 (Deep Widgeon and Millionaire)

Plant Community Results

Overall, the plant community and soil data collected across North Unit Phase 2 and the reference site indicate that restoration was successful at recovering native plant communities where elevations at the restoration site are less than 2.8 meters and greater than 2.4 meters which defines the mid-low marsh zone (Figure 42). These elevations and resultant plant communities are a good match with the reference site, which experiences similar hydrology/flooding conditions. Very low-marsh, <2.4 meters, scrape down areas across North Unit Phase 2 continue to only host non-native aquatics and bareground, and in the future may only recover native plant communities in consecutive very dry years (such as 2015, Figure 5-Figure 7). Additionally, elevations >2.8 meters will continue to be dominated by reed canarygrass, *Phalaris arundinacea*, across the restoration site. These conditions are also found at the reference site, however, at the reference site reed canarygrass quickly gives way to native shrub-scrub plant communities at 3.0 meters and greater. To avoid non-native plant community dominance in the high marsh zones we recommend active shrub-scrub planting and maintenance (multiple years of maintenance) at or above 2.8 meters in elevation across the hydrologically connected wetlands on Sauvie Island.

An additional factor impacting both native plant community recovery and the abundance of bareground at North Unit Phase 2 is the presence of very heavy cattle grazing. While moderate grazing can help reduce reed canarygrass abundance, very heavy grazing is detrimental to water quality, soil, and increases both bareground and non-native species abundance in wetlands (Kidd et al. 2015). Heavy grazing also negatively impacts young shrub-scrub planting development (plantings get eaten and/or trampled) and this, in combination with a lack of planting maintenance (young plantings get overgrown quickly by reed canarygrass), has undermined the restoration of berm shrub-scrub development across the restoration site (Kidd et al. 2015). To improve restoration success and wetland conditions we recommend active grazing control and fencing repair be implemented across North Unit Phase 2 and any other restoration projects were grazing occurs.

Water Surface Elevation & Water Temperature Results

In 2019, instead of focusing on the 2 year flood elevation, we provide a more comprehensive description of how the site hydrology has evolved since restoration by comparing metrics across outer reference sites. Overall, the hydrology of North Unit phase 2 showed positive signs of reconnection with it's adjacent channel and emulates the hydrologic pattern of it's reference site: Cunningham Lake. The hydrologic trajectory for North Unit: Millionaire and Widgeon lakes has been presented for the entire monitoring period, which shows a clear disconnect in patterns between North Unit and it's outer reference channel during pre-restoration and construction. The hydrology of Millionaire and Deep Widgeon took at least till year three post-restoration to stabilize and reach complete connectivity (North Unit Phase 2: Deep Widgeon, North Unit Phase 2: Millionaire Lake). A multiple regression model fitted to these sites show a significant linear relationship between all three sites for year four restoration (Table 21, Figure 57, Figure 59). Due to sensor failures and errors, 2019 data could not be included in this report, hence, the pattern observed in year four post-restoration could not be concluded.

While water temperature data was still under analysis at the time of writing this report, previous studies at the site show that 7-day moving average maximum (7DMAM) temperatures were similar to the nearby outer channels but were generally warmer than the mainstem. This may be due to shallow water depths at the restoration sites (Kidd et. al 2019). We aim to present habitat opportunity metrics and inundation patterns in the 2020 AEM Synthesis report.

Sediment Accretion and Erosion Monitoring

Sedimentation stakes installed in North Unit Phase 2: Millionaire lake and Deep Widgeon showed initial signs of accretion, however, have since then showed gradual signs of erosion. On comparison with Cunningham Lake, no discernable pattern could be drawn. There are several theories for sediment movement at these sites. Since North Unit phase 2 is not directly connected to the mainstem like Wallacut and Steamboat, these areas may receive less sediment loading. Hence, the sedimentation stakes show signs of erosion. Although, there may be another contrasting theory: several studies have shown that sedimentation stakes show localized signs of erosion and scouring, hence may not accurately depict sediment dynamics at the sites. Given than Millionaire and Deep Widgeon each have only two or three pairs of Sedimentation stakes respectively, this alternate theory may also hold true. Hence, there is a need for more sedimentation stakes installation as well as longer monitoring periods at restoration sites to be able to draw definitive conclusions. Additionally, heavy cattle grazing can cause erosion and issues with sediment accretion and erosion monitoring accuracy. Without the removal of heavy grazing it will be difficult to determine or define patterns of sediment movement within these restoration sites.

Juvenile Salmonids

Our check-in fish monitoring efforts clearly show that juvenile salmonids are present in Millionaire and Deep Widgeon Lakes restoration sites five years after tidal reconnection has occurred. Marked and unmarked chinook salmon were caught across all sampling locations within Millionaire and Deep Widgeon Lakes (Figure 64 and Figure 65, Table 23). Despite seeing both marked and unmarked Chinook salmon at all sampling sites no other salmon species were sampled or observed (Table 23), however 13 other species of fish species caught including 7 species non-native to the Columbia River. Overall, threespine sticklebacks, redsided shiner, and peamouth were the most observed species at both lakes sampled. The fish community results, while limited, are similar to what has been observed across the reference sites intensively monitored through the EMP program (Rao et al. 2020). Chinook salmon being the most abundant salmon species identified across the reference sites, in addition to a combination of native and non-native fish frequently identified at Campbell slough and Franz lake which are the closest reference sites to Millionaire and Deep Widgeon lakes fished through the EMP program (Rao et al. 2020). Further fish monitoring would provide more context with the number and species of fish using the site over the season and how this may shift as the site ages and develops more mature habitat condition.

Horsetail Creek

Juvenile Salmonids

Data from off-channel PIT detection arrays indicate that off-channel habitat is used by a wide variety of stocks and species including Chinook and coho salmon, as well as steelhead. The extent of use varies among stock. Fall Chinook typically are the most abundant in these areas and reside longer than other stocks. However, at Horsetail Creek individual steelhead have been shown to reside for several months. One caveat to off-channel use is that northern pikeminnow, a known predator of juvenile Chinook salmon has also been detected in these habitats and tend to reside for weeks to months. Thus, extended use of these habitats could increase juvenile salmon vulnerability to predation. The ecological trade-off between predation risk and foraging opportunity in tidal wetlands, as in tributaries and the ocean, is the mechanistic driver of survival. Increases in foraging opportunities through habitat restoration and efforts to decrease predators (especially non-native predators) may help tilt the scale towards improved salmon survival.

CONCLUSION

The final goal of AEM restoration efforts is the establishment of functional wetland processes and habitat that support juvenile salmonids. Action effectiveness monitoring tracks the ecological impact of restoration work and provide valuable information to manage restoration sites adaptively. Furthermore, AEM shows that the rate at which physical processes and habitats recover after restoration activities vary, depending location in the estuary, degree of tidal reconnection, and pre-existing site conditions. For example, physical processes in a wetland like water surface elevation (duration, frequency, depth, and timing of flooding), water temperature, and overall habitat opportunity change rapidly after reconnection and become closer to conditions in reference sites. Other aspects of wetlands recover over a longer period, such as changes in the vegetation community and soil conditions. The trend for sites five years post-restoration indicates that they have slightly less native cover and a similar amount of reed canarygrass as reference sites. Limited fish monitoring shows that juvenile salmonids are present in restoration sites after tidal reconnection, but, without intensive monitoring efforts, the number of fish using the site can be difficult to ascertain. Furthermore, it is not known if the number of fish accessing a site increases as the habitat moves toward a reference state. A better understanding of how physical processes influence habitat conditions and how these resulting habitat conditions support juvenile salmonids are key to quantifying the overall impact of restoration efforts.

MANAGEMENT IMPLICATIONS

Action effectiveness monitoring measures changes to physical and ecological processes that influence the ability of restoration sites to support juvenile salmonids. In addition, AEM data provides project managers with vital information to determine if project design elements are meeting goals or if adaptive management is required.

At the site-scale, restoration projects are leading to the reestablishment of natural physical processes that support juvenile salmonids. Data has shown that site water levels respond immediately to hydrologic reconnection. Water temperatures at the restoration sites are generally warmer than nearby main stem waters but were generally suitable during the spring and early summer juvenile outmigration periods. The higher temperature at restoration sites can be attributed to shallower water depths, and this trend is mirrored in results seen at Ecosystem Monitoring Program (EMP) sites (Kidd et al. 2019).

As the goals of restoration activities include improving fish access to historic floodplain habitats and the quality of those habitats, we wanted to verify that fish are using restored sites. We chose to employ a "status check" of fish use at five years post-restoration. We collected fish occurrence data at four locations within North Unit Phase 2 and found juvenile salmonids at all locations. The presence of juvenile salmonid indicates these restoration benefit to the fish. The PIT array at Horsetail Creek continues to detect out migrating upriver juvenile salmonid species visiting the site for periods ranging from a few hours to a couple of days.

AEM research shows restoration sites are achieving increases in hydrologic connectivity and salmonid opportunity; however, plant community recovery is more variable across sites. Given the inherent inter-annual climate variability, it is difficult to predict specific restoration outcomes on a year to year basis. However, clear trends in plant community recovery across restoration sites persist, with high marsh elevations retaining reed canarygrass and other non-native species at year 3 and 5 post restoration. The lack of high marsh plant community recovery is also echoed in the soil conditions identified in these locations, which retain lower soil salinity, pH, and greater ORP levels than found at reference sites. Additionally, areas within restoration sites that have undergone heavy construction impacts and grading have also shown to recover on a slower timeline. Alternatively, we have observed that both soil and dominate native plant communities recover quickly (within 5 years post-restoration) in areas that are found at moderately low to mid wetland elevations. Across all these findings wetland elevation is used as a proxy for restored wetland hydrology which, in combination with soil conditions, is the ultimate mechanism driving restoration outcomes throughout the estuary (Bledsoe and Shear 2000, Neckles et al. 2002, Davy et al. 2011, Mossman et al. 2012, Gerla et al. 2013, Kidd 2017). Through our AEM research we have found that the re-establishment of natural physical and hydrological processes to sites can be accomplished in a short period of time but understanding how these wetland sites respond ecologically will require long-term monitoring. Ultimately, this continued monitoring will elucidate long-term trends and improve our understanding of the connections between physical processes, habitat responses, and the resulting benefits to juvenile salmon.

AEMR Program Recommendations

Suggestions for Project Design

- Both restoration design and evaluation would benefit from the use of predictive modeling to determine the restoration of aquatic, marsh, and shrub-scrub plant communities. This type of modeling can be easily accomplished by incorporating anticipated restored hydrology and site elevations and comparable reference site conditions (Hickey et al. 2015). These data can also provide a platform for evaluating different restoration scenarios, such as considering different levels of hydrologic reconnection and/or marsh plain lowering and the impacts of this for multispecies and plant community habitat recovery (Hickey et al. 2015)[5].

  - Across multiple restoration projects we have seen very high and very low marsh elevations struggle to recover native plant cover within a 5-year timeline. Moving forward predictive modeling could aid in restoration design (and adaptive management efforts) to maximize the restoration of the mid to moderately low marsh elevations which have been shown to recover native plant habitat and soil conditions quickly post-restoration (throughout the Estuary).
  - In addition, this will also aid project planning for determining seeding and planting zones in target high marsh areas for non-native species control and shrub-scrub development.
  - Assess restoration success and goal-reaching post-restoration would also be easier given predictive maps and data could be compared to conditions observed post-restoration.

Suggestions for Project Monitoring

Site topography and Reference Sites

- Accessibility to ground survey technology such as RTK GPS systems has increased dramatically over the last five years and these systems allow us to easily map the overall topography of wetlands and their plant communities and channels. With this technology, we can assess the compatibility of reference and restoration wetland sites. Similar elevation gradients (and hydrology) should be sampled within reference and restoration sites for meaningful comparisons to be made post-restoration (and to aid in project design). In this report we have highlighted that the reference site elevations have generally been a poor match with each restoration site's restored elevations, moving forward we will aim to alter monitoring plans to sample more overlapping elevation gradients between the restoration and reference sites to correct these issues. Additionally, upon choosing reference sites to inform project design and post-restoration project success elevations and (anticipated) hydrology should be compared to ensure the use of reference elevation data is an appropriate proxy for hydrologic conditions.

Hydrology

- Hydrology is a critical component to all wetland restoration efforts and should be monitored for project planning, design, and to assess project success. During project design clear hypotheses should be developed to define hydrologic changes anticipated from restoration efforts. For monitoring data loggers need to be in placed areas that are anticipated to experience these hydrologic changes post-restoration and remain in the same location pre- and post-restoration. Given the number of issues we have experienced

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[5] We are currently using this Ecosystem Modeling Approach (Hickey et al. 2015) at Steigerwald National Wildlife Refuge and Multnomah Channel Natural Area to evaluate and design for desired restoration outcomes.

through the years with data loggers we recommend having at least one redundant logger be placed within the site (nearby or at the same location), that can provide additional data in case of equipment failure (which is common). Loggers need to be maintained at least every six months and we recommend all deployment and retrievals follow the new and more detailed monitoring protocols to avoid data loss (Kidd et al. 2018).

SEDIMENT ACCRETION AND EROSION, CHANNEL CROSS-SECTIONS

- Understanding sediment accretion and erosion dynamics across the floodplain of newly restored wetlands is critical for tracking wetland and channel development and long-term topographic trajectories. Sediment dynamics across restoration sites can be extremely variable making it difficult to track meaningful change without intensive and extensive monitoring efforts. We recommend shifting our current approach of sediment monitoring (one or two sediment benches placed within a site) to a more targeted application of these methods. Before restoration occurs specific areas of interest should be selected and multiple sediment monitoring benches (minimum of 6) should be installed along the elevation gradient and within these targeted areas. Within the sediment bench monitoring area (between the pins), we also recommend tracking dominant plant community development and soil characteristics to aid data interpretation. Channel cross-section monitoring should be similarly focused, and extreme care should be taken to resurvey the exact location of the cross-section for meaningful results to be obtained. Both channel cross-section and sediment benches need to be resurveyed using RTK GPS technology to provide topographic context and increase data usability. Updated monitoring protocols are currently in development for these methods (Kidd and Rao 2019).

WETLAND PLANT COMMUNITY

- Native wetland plant communities provide a critical base of the salmonid food web and are essential for determining wetland restoration success (Rao et al. 2020). We have found monitoring a randomized selection of vegetation plots each year creates a great amount of variability in the data, and makes determining what change has been caused by the restoration and what change is due to the new randomized sampling difficult to determine. There are two approaches to addressing this issue, one would be to 1) continue to randomize the plots annually but significantly increase the overall total number of plots surveyed or 2) to only randomize the plots the first year of monitoring and re-visit these same plots year after year. We recommend (#2) re-visiting the same plots year after year, which provides a clear path to assessing plant community changes overtime and does not increase the overall amount of time required to conduct sampling. Additionally, as shown in this report, the collection of soil data, alongside of plant community data, can be very informative when evaluating wetland development and restoration. We recommend integrating soil data collection as an essential metric for Level 2 monitoring across sites. Further vegetation and soil monitoring recommendations are forthcoming, as we work on a comprehensive update to the *Protocols for Monitoring Habitat Restoration Projects in the Lower Columbia River and Estuary* (Roegner et al. 2009).

Utilizing UAV Technology: Site Topography, Plant Community Mapping

- The accessibility and applicability of Unmanned aerial vehicles (UAV) and associated sensor technology have made significant strides in the last several years. Using some of the most affordable equipment and software available we have shown that large scale site wetland plant community and topographic mapping is possible and accurate (Kidd et al. 2020). Mapping dominant native and non-native plant communities across large portions of restoration sites can aid evaluation of project success post-restoration, and guide both active restoration project design and post-restoration project adaptive management efforts. Moving forward we are working to refine our UAV monitoring methods to include tracking channel and floodplain topographic development into our analysis and reporting. We are also exploring methods of evaluating biomass and carbon stores across reference and restored wetlands using our UAV and sensor technologies. Further UAV vegetation monitoring methods and recommendations will be included in the comprehensive update to the *Protocols for Monitoring Habitat Restoration Projects in the Lower Columbia River and Estuary* (Roegner et al. 2009).

Frequency of Monitoring

- Currently, Level 3 monitoring is conducted pre- through year 5 post-restoration and Level 2 monitoring is conducted pre, 1, 3, and 5 years post restoration. Results from the last 6 years of the AEMR level 2 and 3 monitoring indicate that restoration outcomes can be slow and variable, with sites not achieving reference level native plant community conditions by year 5 post-restoration (Johnson et al. 2018, and this report). Given these observations, we recommend level 3 monitoring continue to occur pre through 5, 8, and 10 years post-restoration and that Level 2 monitoring should also be conducted at year 8 and year 10 post-restoration. Adding year 8 and 10 to monitoring for all level 2 and 3 metrics will aid in understanding the long-term impacts of our restoration efforts and allow for monitoring to occur over a wider spectrum of annual climate conditions. Additionally, we recommend UAV plant community mapping occur across all Level 2 and 3 sites pre-restoration, and 3, 5, 8, and 10 years post-restoration. These additional data and longer-term monitoring windows will provide greater context to assess restoration actions and outcomes and help us test ongoing hypotheses about how shifts in climate and river discharge conditions impact restoration outcomes. Adding synthesis reports of site conditions at year 8 and 10 post-restoration will also provide meaningful insight for ongoing adaptive management and restoration efforts.

Fish and Macroinvertebrate Monitoring

- AEMR Level 2 monitoring does not encompass comprehensive fish or macroinvertebrate monitoring as part of the standard habitat monitoring protocol. Level 2 monitoring includes limited macroinvertebrate monitoring (one or two neuston tows a year following the Level 2 monitoring schedule) and a one-time fish sampling event at year five post-restoration. Given the spatial and temporal variability of both fish and macroinvertebrate populations seen across the long-term EMP reference sites (Rao et al. 2020), we have concluded a more comprehensive macroinvertebrate and salmonid sampling effort is required, for meaningful post-restoration food web conditions to be evaluated. Limited fish monitoring shows that juvenile salmonids are present in restoration sites after tidal reconnection, but, without intensive monitoring efforts, the number of fish using the site can be difficult to ascertain. Furthermore, it is not known if the number of fish accessing a site increases as the habitat moves toward a reference state. A better understanding of how physical processes influence habitat conditions and how these resulting habitat conditions support juvenile salmonids are key to quantifying the overall impact of restoration efforts. The addition of long-term ecosystem monitoring at a select number of restoration sites would allow for these sites to be tracked alongside the Ecosystem Monitoring Program. The EMP sites have years of accumulated status and trends fish, macroinvertebrate, water quality, and habitat data which could be used for ongoing comparative analysis and evaluation. Selecting focal restoration sites of interest and conducting intensive fish and macroinvertebrate monitoring efforts at these sites, similar to the level of monitoring conducted across EMP sites (Rao et al. 2020), would allow for the recovery of fish use and macroinvertebrate communities to be assessed over the long-term and aid in the interpretation of how physical changes to habitat directly influence the salmonid food web.

Synthesizing Restoration Results

- The most meaningful analysis of restoration success would be one that incorporates all habitat level monitoring metrics across a site to identify recovery of salmonid habitat overtime. We are currently developing a site wide assessment of habitat opportunity that extends across the wetland's active floodplain (Johnson et al. 2018). This would incorporate floodplain topography, water surface elevation (water depth), water temperatures, and dominate plant communities to highlight salmonid habitat conditions across the active floodplain of restoration and reference sites. This active floodplain mapping approach could also be used as a tool to evaluate the impacts of climate change and shifting river discharge on wetland habitat conditions throughout the Columbia Estuary.

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Appendix

Appendix A: Site Sampling Reports

The summaries are presented in order starting from the mouth of the estuary to up-river. Additional background information about the sites sampled in the AEMR Program is often available in restoration project planning documents and reports, or in previous monitoring reports. To the extent possible, these are cited in the descriptions of each site.

Equipment

Equipment for each of the metrics sampled is outlined below.

- *Vegetation*: 100-m tapes for the baseline and transects, a compass for determining the baseline and transects azimuth, 1-m quadrat, data sheets, and plant books for species identification. GPS to identify location of base stakes and quadrats.
- *Sediment Accretion Rate:* 2 gray 1-inch PVC conduit pipes, at least 1.5m long, construction level, meter stick. GPS to identify location of stakes.
- *Neuston Tows:* To assess the availability of salmon prey at sites, we conducted neuston tows in both open water (OW; in the center of the channel) and emergent vegetation (EV; along the edge of the wetland channel among vegetation). Samples were preserved in 95% ethanol.
- *Photo Points:* camera, stake for including in photo, previous photos at location for reference, GPS to identify location of point.
- *Elevation*: Topcon GPS with real-time kinematic (RTK) correction. Other survey equipment in case GPS equipment is non-functional, including an auto-level, tripod, and stadia rod.

Wallacut Slough – Survey July 29, 2019, Ilwaco July 30
Welch August 1
Steamboat August 2
Cunningham August 12
Millionaire August 14 Deep August 15

Wallacut Restoration

General Site Location

The site is located near the mouth of the Wallacut River, which empties into Baker Bay, at approximately rkm 7.

Ecosystem Type

Diked, planned restoration site

Current Role of Site in the CEERP
The Wallacut site is owned by the Columbia Land Trust. The site is slated for hydrologic reconnection through the removal of three culverts, removal of a low levee, ditch filling, and tidal channel creation. In addition, invasive species removal of gorse (*Ulex europaeus* L.) has been implemented to increase native species colonization.

Dates of Sampling in 2019
29 July

Types of Sampling in 2019

- Vegetation: Herbaceous cover (2 sample areas of 36 quadrats each, 72 quadrats total)
- Insect Neuston Tows
- Photo Points: 2
    - Top of dike near the location of the lower vegetation monitoring plot
- Elevation: collected elevation at all vegetation quadrats

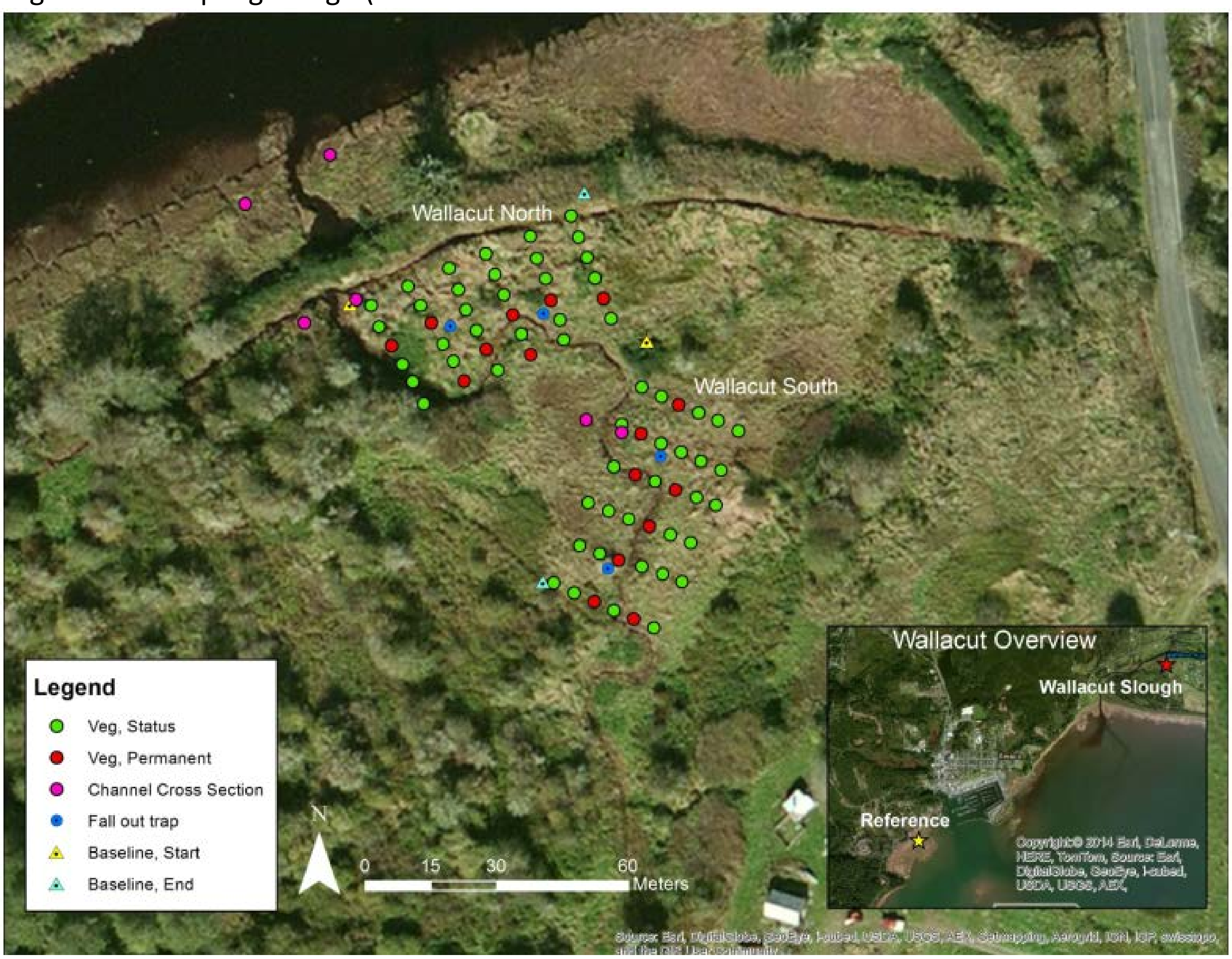
Vegetation Sampling Design (


Figure 68)

2 sampling areas were set up. New vegetation sample areas were established to capture the current condition and potential change that would occur as follows:

Mouth Veg Sample area (Wallacut North)

- Located in area near the mouth of the channel
- 60 m x 30 m, with 36 quadrat locations
- Baseline azimuth: 60° magnetic
- Transect azimuth: 105° magnetic
- Transect spacing: 10 m, random start: 5
- Quadrat spacing: 5 m, random starts: 2, 1, 2, 2, 3, 2
- 8 permanent quadrats, randomly selected, systematically to ensure coverage on all transects

Upper Veg Sample area (Wallacut South)

- Located in area that will be affected by the dike removal, but away from the channel excavation.
- 60 m x 30 m, with 36 quadrat locations
- Baseline azimuth: 185° magnetic
- Transect azimuth: 95° magnetic
- Transect spacing: 10 m, random start: 9
- Quadrat spacing: 10 m, random starts: 2, 2, 4, 2, 3, 2
- 8 permanent quadrats, randomly selected, systematically to ensure coverage on all transects

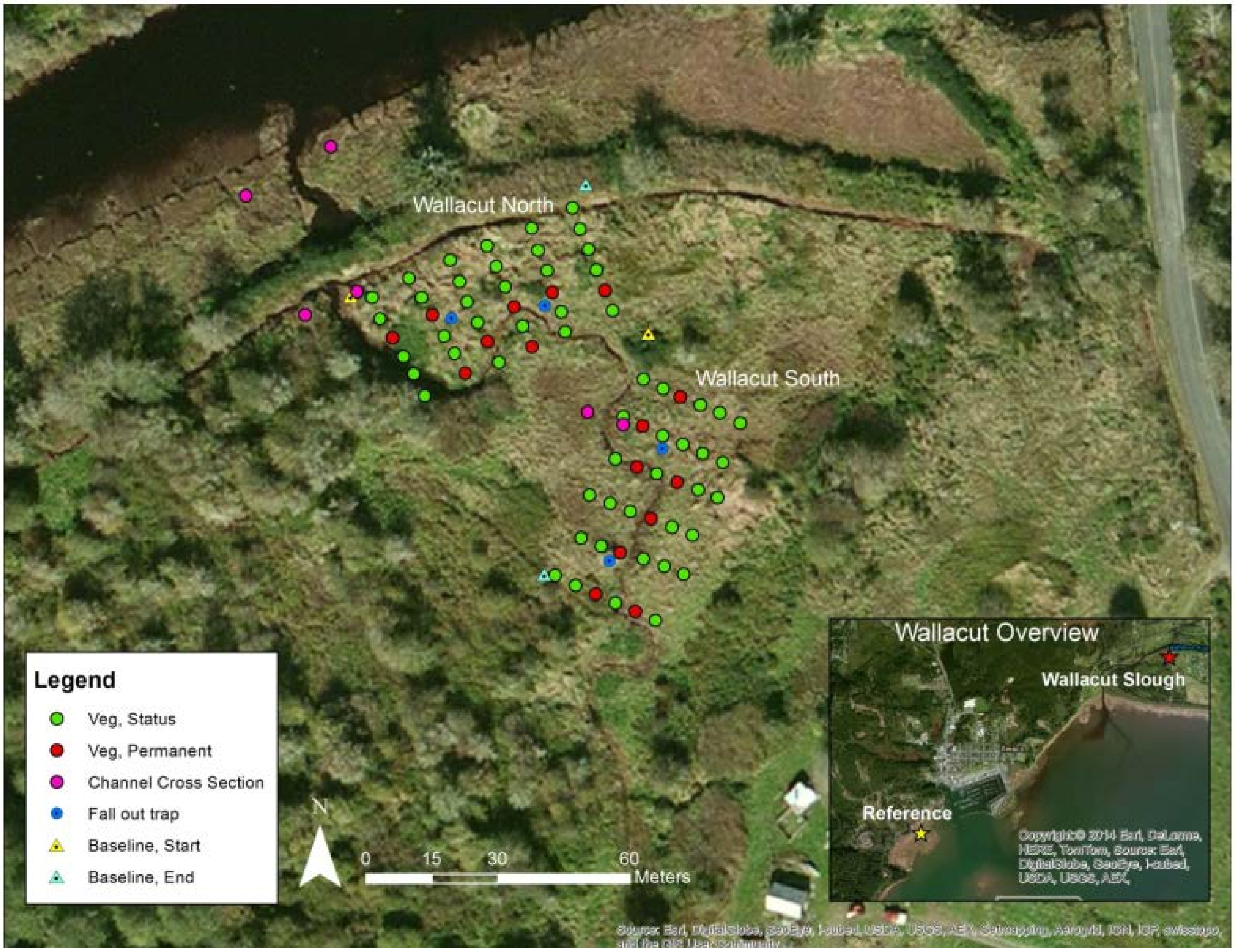


Figure 68. 2019 vegetation and macroinvertebrate sampling locations at Wallacut restoration site.

Markers Left on Site
All marking stakes are white ¾ inch PVC. We marked the following locations:

- End stakes of the baseline for the vegetation sample areas.
- Permanent quadrat stakes; 2 stakes per location in the diagonal corners (SW and NE).

Macroinvertebrate Sampling
Two macroinvertebrate neuston tows were collected in the Wallacut Slough vegetation sampling areas.

Ilwaco Reference
General Site Location
Northwest side of Baker Bay west of Ilwaco marina.

Ecosystem Type

Tidal brackish emergent wetland

Sampling History in CEERP
This long-term monitoring site has been surveyed annually since 2011 site as part of the Estuary Partnership's Ecosystem Monitoring Program.

Current Role of Site in the CEERP
Ilwaco is being sampled as a reference site for baseline monitoring for the restoration actions being conducted in 2019 at Wallacut Restoration site.

Dates of Sampling in 2019
30 July

Types of Sampling in 2019
See map below for sampling locations (Figure 69).

- Vegetation: Herbaceous cover (1 sample area of 40 quadrats)
- Insect Neuston Tows
- Photo Points:
    - 360° from 2 m east of the 0 m baseline stake
- Sediment Accretion Rate: measured one previously installed pair of stakes
- Elevation: collected elevation at all vegetation quadrats

Vegetation Sampling Design
Status Sampling. The sampling design implemented for the EMP was used for monitoring. This sampling design is similar to that used for the AEMR sampling except that the same quadrats are
sampled from year to year to evaluate trends.

Vegetation Sample Area

- Veg sample area covered the mid-marsh elevation gradient which contained primarily
- *Agrostis stolinifera* and *Carex lyngbyei*.
- 200 m x 100 m, with 40 quadrat locations
- Baseline azimuth: 240° magnetic
- Transect azimuth: 330° magnetic
- Transect spacing: 50m, random start: 16
- Quadrat spacing: 10 m, random starts: 4, 7, 2, 6
- Trends Sampling. No permanent plots were placed at this site. Future trends monitoring will
  be conducted according to the EMP sample design.

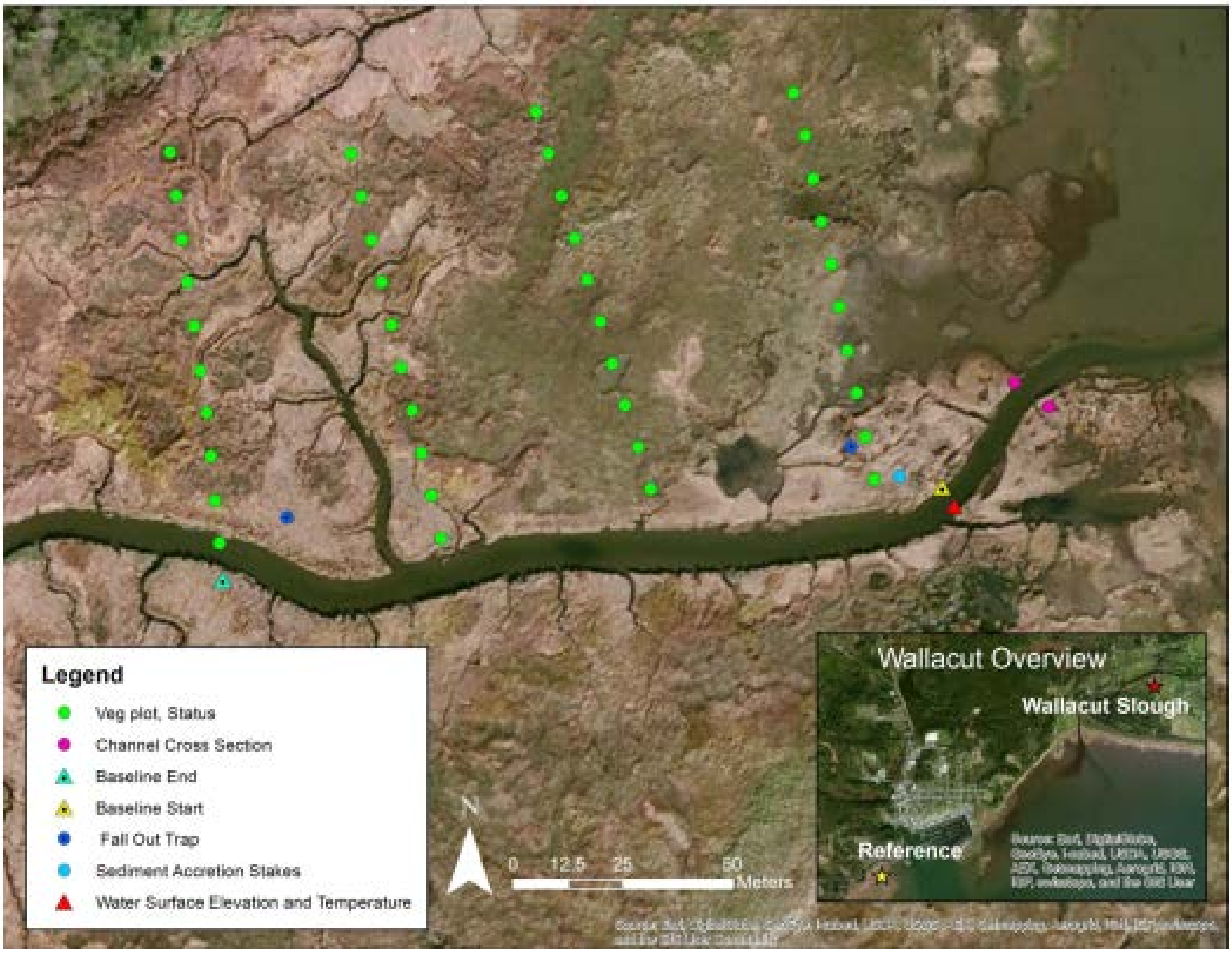


Figure 69. 2019 vegetation and macroinvertebrate sampling locations at Ilwaco marsh. (Fall out traps collected in 2014, neuston tows collected in 2019)

Markers Left on Site
All marking stakes are white ¾ inch PVC. Marks left:

- End stakes at each of the transects in the vegetation sample area.
- In addition, 2 1" gray pvc sediment accretion stakes are located on the site and a depth sensor is located inside 1 ½" PVC on a t-post in the channel.

Macroinvertebrate Sampling
Two macroinvertebrate fall out traps were placed in two separate locations within the vegetation
sampling area.

Steamboat Slough
General Site Location
Julia Butler Hanson (JBH National Wildlife Refuge)

Ecosystem Type
Formerly diked, restoration site

Dates of Sampling in 2019
2 August

Types of Sampling in 2019
See map below for sampling locations (Figure 70).

- Vegetation: Herbaceous cover (2 sample areas of 36 quadrats, 72 quadrats total)
- Insect Neuston Tows
- Elevation: collected elevation at all vegetation quadrats

Vegetation Sampling Design
East Vegetation Sample Area

- Located at east end of site in former constructed wetland low elevation area. Vegetation sample area spanned elevation gradient from lowest elevation with submerged aquatic vegetation (SAV) and bare mud through low marsh up to high elevation that was not formerly excavated.
- 70 m x 60 m, with 36 quadrat locations
- Baseline azimuth: 330° magnetic
- Transect azimuth: 240° magnetic
- Transect spacing: 12 m, random start: 10
- Quadrat spacing: 10 m, random starts: 7, 8, 1, 1, 1, 0

West Vegetation Sample Area

- Located in area that will be affected by the dike removal, near proposed site of excavated channel.
- 70 m x 60 m, with 36 quadrat locations
- Baseline azimuth: 312° magnetic
- Transect azimuth: 42° magnetic
- Transect spacing: 12 m, random start: 10
- Quadrat spacing: 10 m, random starts: 0, 7, 3, 9, 1, 5
- 8 permanent quadrats, randomly selected, systematically to ensure coverage on all Transects, Trends Sampling. Within the vegetation sample areas, we revisited trend sampling plots.

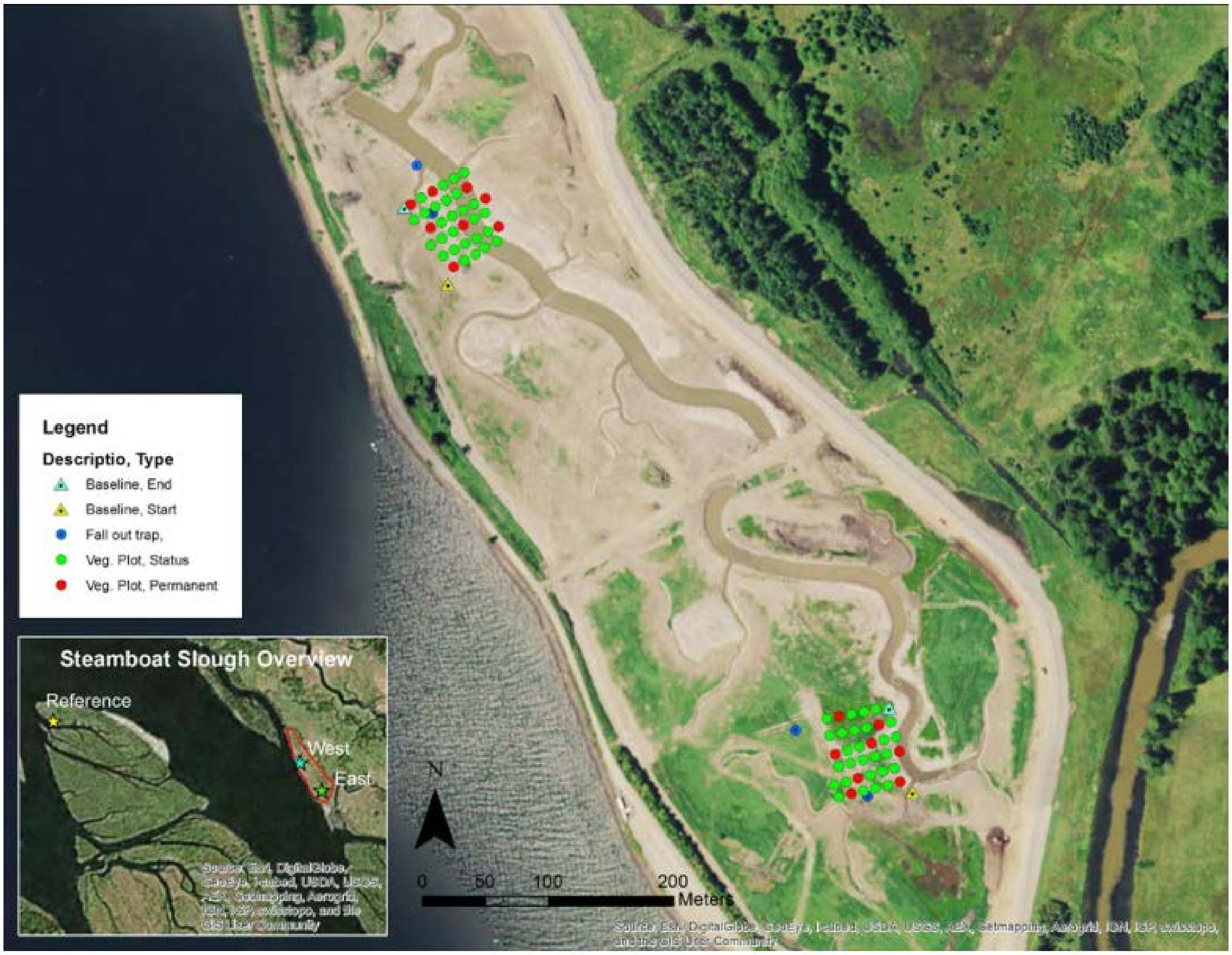


Figure 70. Vegetation and macroinvertebrate sampling locations at the Steamboat Slough restoration site. (Fall out traps collected in 2014, neuston tows collected in 2019)

Markers Left on Site
All marking stakes are white ¾ inch PVC.

Marks left: Start and End stakes at each of the transects in the vegetation sample area.

Macroinvertebrate Sampling
Two macroinvertebrate neuston tows were collected in the Steamboat vegetation sampling areas.

Welch Island
General Site Location
Welch Island is located on the northwest (downstream) corner of the island at rkm 53, which is part of
the Julia Butler Hanson Wildlife Refuge.

Ecosystem Type

Tidal emergent wetland

Sampling History in the CEERP
Two other areas of the island were monitored as part of the Reference Sites Study in 2008 and 2009
(Borde et al. 2011).

Current Role of Site in the CEERP
The area was selected as a long-term monitoring site in 2012 for the Estuary Partnership's Ecosystem
Monitoring Program.

Dates of Sampling in 2019
2 August

Types of Sampling 2019

- Vegetation: Herbaceous cover (46 quadrats)
- Insect Neuston Tows
- Sediment Accretion Rate: measured one previously installed pair of stakes
- Photo Points: one previously established point located at the 0 m end of the vegetation sample area baseline
- Elevation: collected elevation at all vegetation quadrats.
- Water surface elevation and temperature: hourly measurements collected in the channel adjacent to the vegetation sampling area; continuous collections since December 2011.

Vegetation Sampling Design (Figure 71)
Status Sampling. The same sample areas sampled for vegetation for the ecosystem monitoring program
were used for action effectiveness monitoring.

Vegetation Sample area

- Located near a tidal channel in emergent marsh vegetation.
- 100 m x 80 m, with 40 quadrat locations and 6 quadrats located in the tidal channel to the east
- of the sample area.
- Baseline azimuth: 322° magnetic
- Transect azimuth: 232° magnetic
- Transect spacing: 20m, random start: 12
- Quadrat spacing: 10 m, random starts: 6, 5, 4, 5, 0 the same quadrats are monitored each year for the trends sampling, no permanent markers were used to mark quadrat locations.

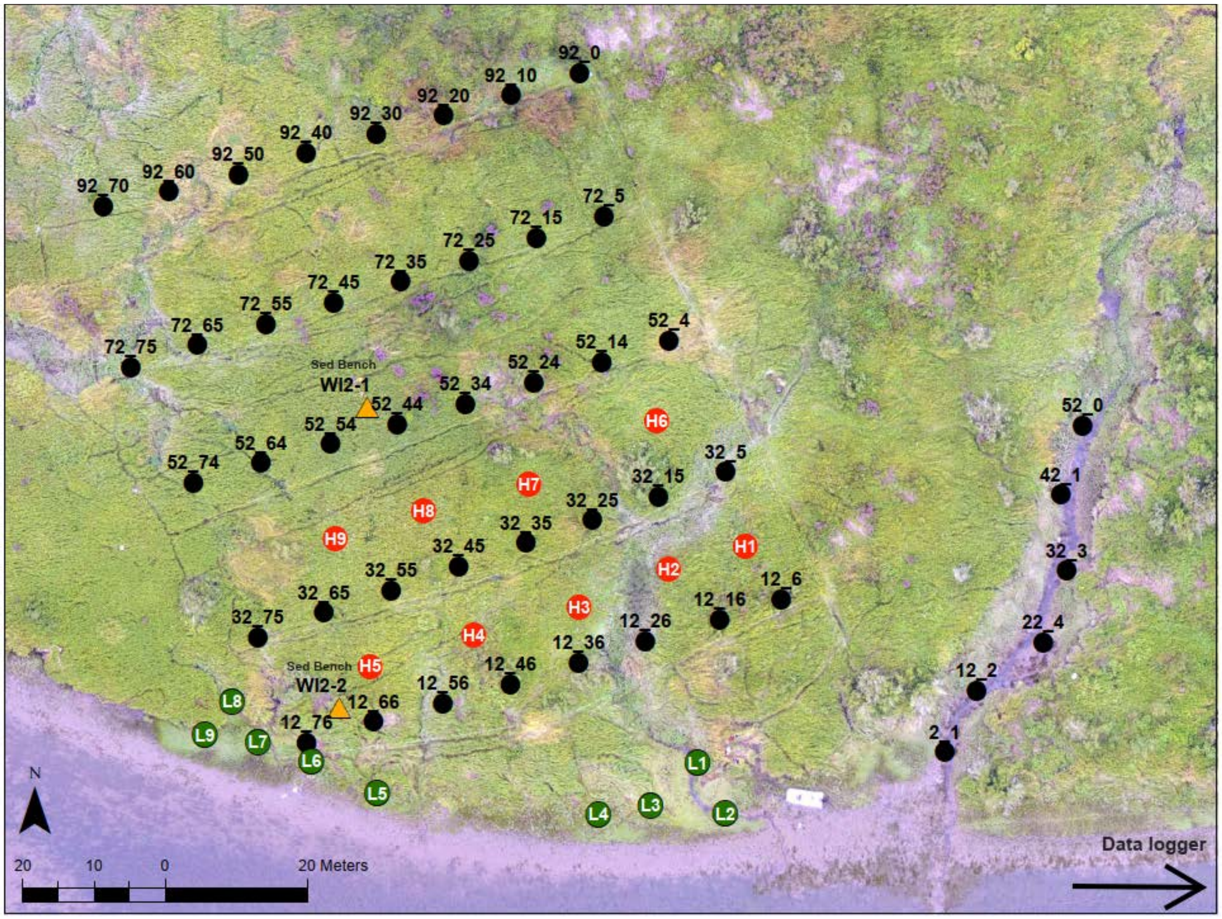


Figure 71: Vegetation sampling locations at Welch Island. (Fall out traps collected in 2013, neuston tows collected in 2019)

Markers Left on Site

All marking stakes are white 1” inch PVC. We marked the end stakes of the transects within the vegetation sample areas. One set of 2 sediment stakes are also located at the site, which are gray 1”
PVC. The depth sensor is located inside 1 ½” PVC on a t-post in the channel.

Macroinvertebrate Sampling

Macroinvertebrate fall out traps were randomly placed at locations along the edge of the vegetation
sampling area in order to avoid disturbance to the vegetation in the sampling area.

Sauvie Island North Unit Phase 2 (Deep Widgeon)

General Site Location

North End of Sauvie Island on the east side of Cunningham Slough at rkm 144.

Ecosystem Type

Formerly diked, restoration site

Dates of Sampling in 2019
15 August

Types of Sampling in 2019
See map below for sampling locations (Figure 72)
Vegetation: Herbaceous cover (2 sample areas, 72 quadrats total)

- Insect Neuston Tows
- Photo Points:
- 1 photo point at the North Veg Sample area - 360° from 2 m northeast of the 0 m baseline stake
- 1 photo points at the South Veg Sample area - 360° from 2 m south of 0 m baseline stake
- Elevation: collected elevation at all vegetation quadrats

Vegetation Sampling Design
North Veg Sample area

- 40 m x 50 m, with 36 quadrat locations
- Baseline azimuth: 229° magnetic
- Transect azimuth: 319° magnetic
- Transect spacing: 10m, random start: 4
- Quadrat spacing: 5 m, random starts: 4, 0, 4, 1
- 8 permanent quadrats, randomly selected, systematically to ensure coverage on all transects. In 2015 Permanent plot 4-15 was moved to 14-30 to capture scrape down area.

South Veg Sample area

- Veg sample area spanned the proposed elevation gradient which currently is covered by reed
- canarygrass and will be scraped down to an elevation to prevent recolonization.
- 50 m x 50 m, with 28 quadrat locations
- Baseline azimuth: 57° magnetic
- Transect azimuth: 327° magnetic
- Transect spacing: 8m, random start: 6
- Quadrat spacing:
- 4 transects with 5 quadrats at 10 m spacing
- 2 transects with 4 quadrats at 12 m spacing
- Random starts: 6, 5, 10, 3, 2, 2
- 8 permanent quadrats, randomly selected, systematically to ensure coverage on all transects

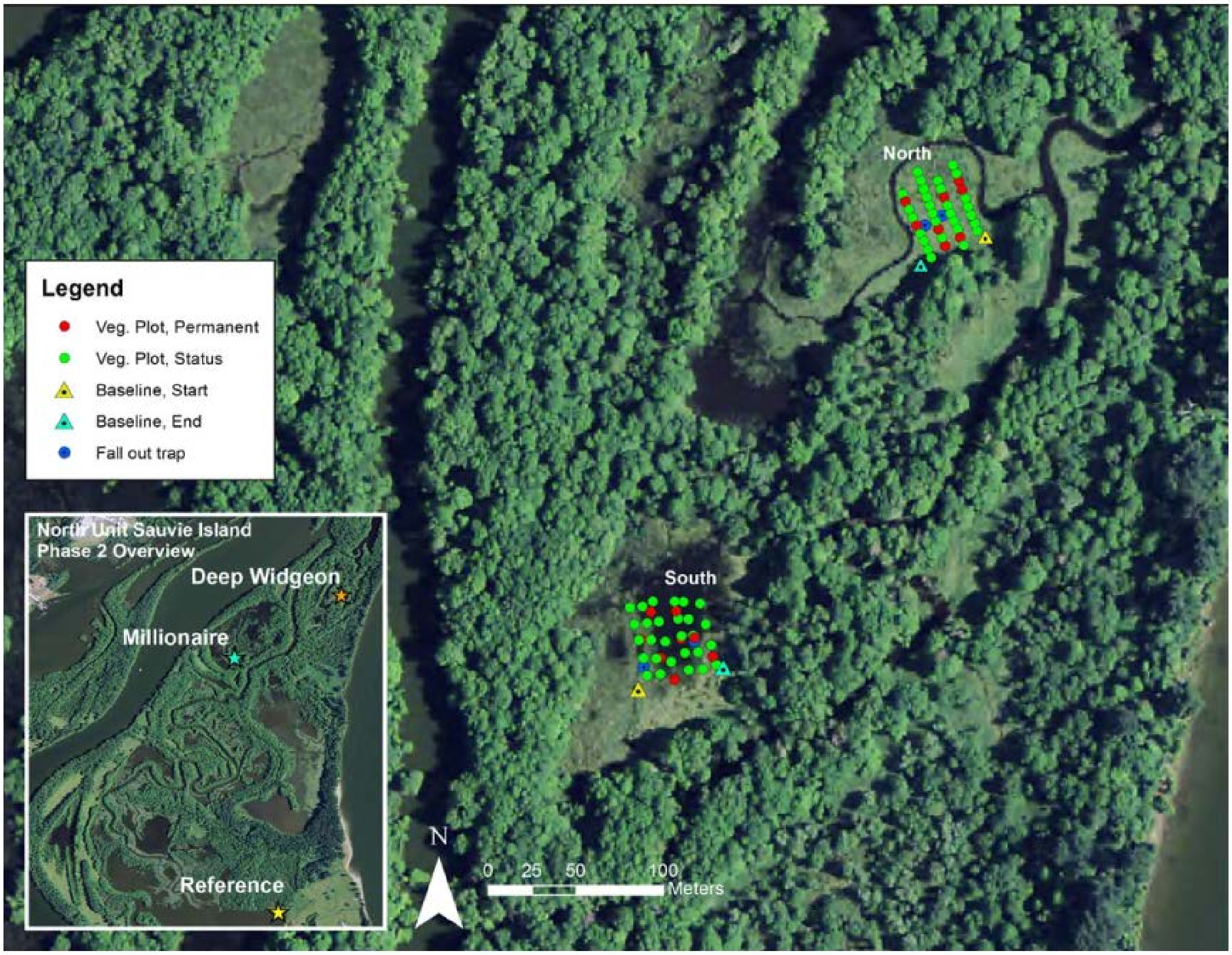


Figure 72. Vegetation and macroinvertebrate sampling locations at the North Unit Phase 2 (Deep Widgeon) restoration site. (Fall out traps collected in 2015, neuston tows collected in 2019)

Markers Left on Site
All marking stakes are white ¾ inch PVC. Marks left:
Start and End stakes of the baseline for the vegetation sample areas.
Permanent quadrat stakes; 2 stakes per location in the diagonal corners (SW and NE).

Macroinvertebrate Sampling
Two macroinvertebrate neuston tows were collected in the Deep Widgeon vegetation sampling areas.

Sauvie Island North Unit Phase 2 (Millionaire Lake)
General Site Location
North End of Sauvie Island on the west side of Cunningham Slough at rkm 144.

Ecosystem Type
Formerly diked, restoration site

Dates of Sampling in 2019
15 August

Types of Sampling in 2019

- Vegetation: Herbaceous cover (2 sample areas, 72 quadrats total)
- Insect Neuston Tows
- Photo Points:
    - 1 photo point at the North Veg Sample area - 360° from 2 m east of the 0 m baseline stake
    - 1 photo points at the South Veg Sample area - 360° from 2 m southwest of 0 m baseline stake
- Elevation: collected elevation at all vegetation quadrats

Vegetation Sampling Design (Figure 73)
North Veg Sample area

- Located at north end of the southern part of the site. Veg sample area spanned elevation
  gradient which was scraped down to an elevation to prevent recolonization of reed canarygrass.
- 60 m x 50 m, with 36 quadrat locations
- Baseline azimuth: 343° magnetic
- Transect azimuth: 253° magnetic
- Transect spacing: 10m, random start: 8
- Quadrat spacing: 8 m, random starts: 0, 5, 4, 5, 2, 0
- 8 permanent quadrats, randomly selected, systematically to ensure coverage on all
- transects

South Veg Sample area

- Located at the southern end of the southern part of the site. Veg sample area spanned elevation gradient from lowest elevation SAV and bare mud through low marsh up to an elevation dominated by reed canarygrass.
- 80 m x 70 m, with 28 quadrat locations
- Baseline azimuth: 323° magnetic
- Transect azimuth: 233° magnetic
- Transect spacing: 13m, random start: 2
- Quadrat spacing:
- 4 transects with 5 quadrats at 14 m spacing
- 2 transects with 4 quadrats at 18 m spacing
- Random starts: 5, 8, 2, 4, 5, 10
- 8 permanent quadrats, randomly selected, systematically to ensure coverage on all transects

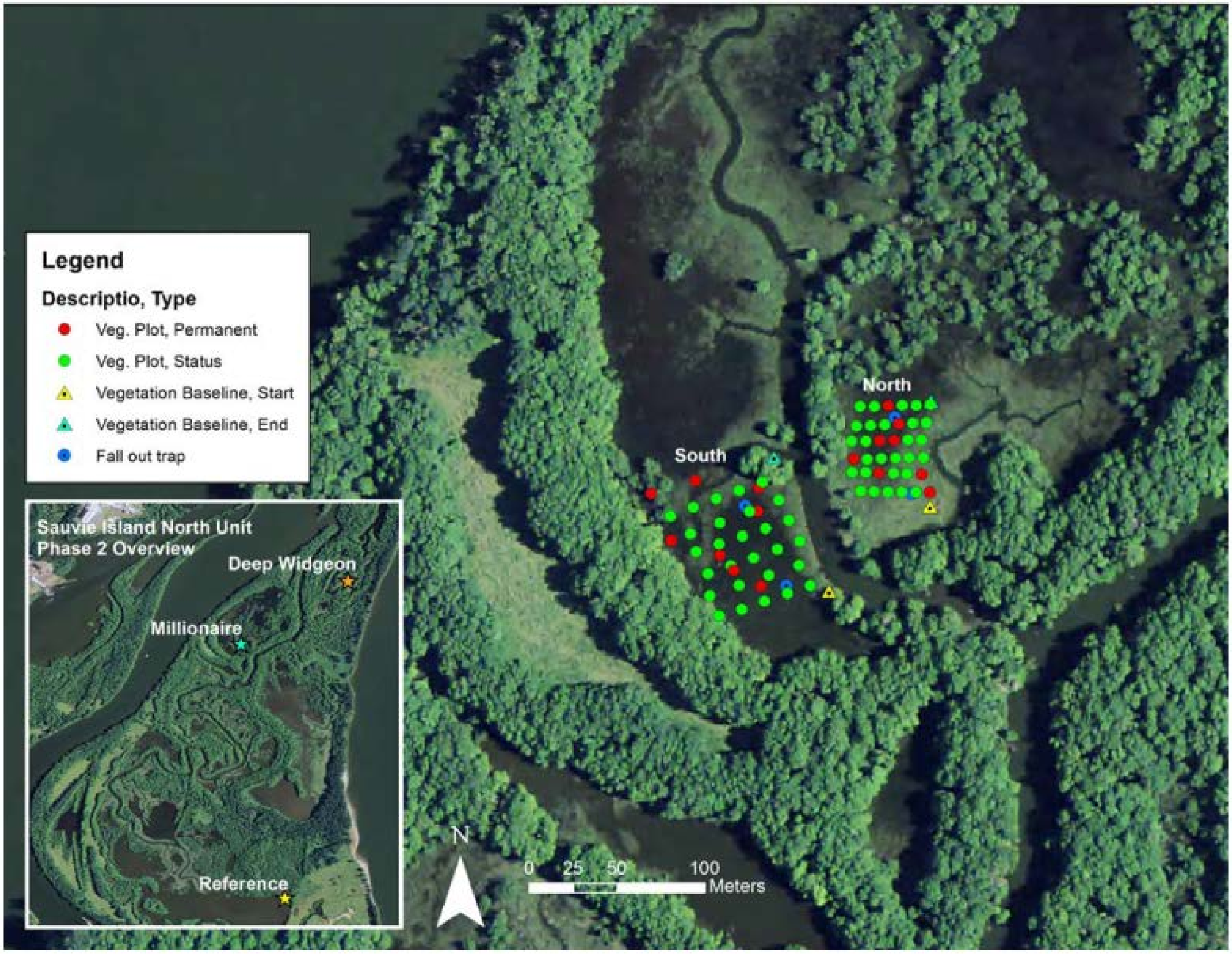


Figure 73. Vegetation and macroinvertebrate sampling locations at the North Unit Phase 2 (Millionaire Lake) restoration site. (Fall out traps collected in 2015, neuston tows collected in 2019)

Markers Left on Site

All marking stakes are white ¾ inch PVC with orange duct tape or flagging at the top were left on site from previous year's marking. Marks left:

- End stakes of the baseline for the vegetation sample areas.
- Permanent quadrat stakes; 2 stakes per location in the diagonal corners (SW and NE).

Macroinvertebrate Sampling

Two macroinvertebrate neuston tows were collected in the Millionaire vegetation sampling areas.

Sauvie Island North Unit Reference (Cunningham Lake)

General Site Location

Cunningham Lake is a floodplain lake located at rkm 145 on Sauvie Island in the Oregon DFW Wildlife Area. The mouth of the Slough is located between rkm 142 and 143 close to where Multnomah Channel meets the Columbia River. The end of Cunningham Slough is approximately 8.7 km from Multnomah Channel.
Ecosystem Type
Reference Site, Fringing Emergent Marsh at the upper extent of the extremely shallow "lake"

Dates of Sampling in 2019
12 August

Types of Sampling in 2019
See map below for sampling locations

- *Vegetation*: Herbaceous cover (70 quadrats total)
- *Insect Neuston Tows*
- *Photo Points:* 1 photo point
    - 360° panorama taken at location near south end of vegetation sample area.
- *Elevation*: collected elevation at all vegetation quadrats

Vegetation Sampling Design
Veg Sample area (Figure 74)

- Located along the fringe of the very shallow Cunningham Lake. Vegetation sample area spanned elevation gradient from unvegetated flats to the shrub/tree zone.
- 70 m x 25 m, with 36 quadrat locations
- Transect spacing: 2m, random start: 0
- Quadrat spacing: 2 m
- 8 permanent quadrats established for AEMR were monitored

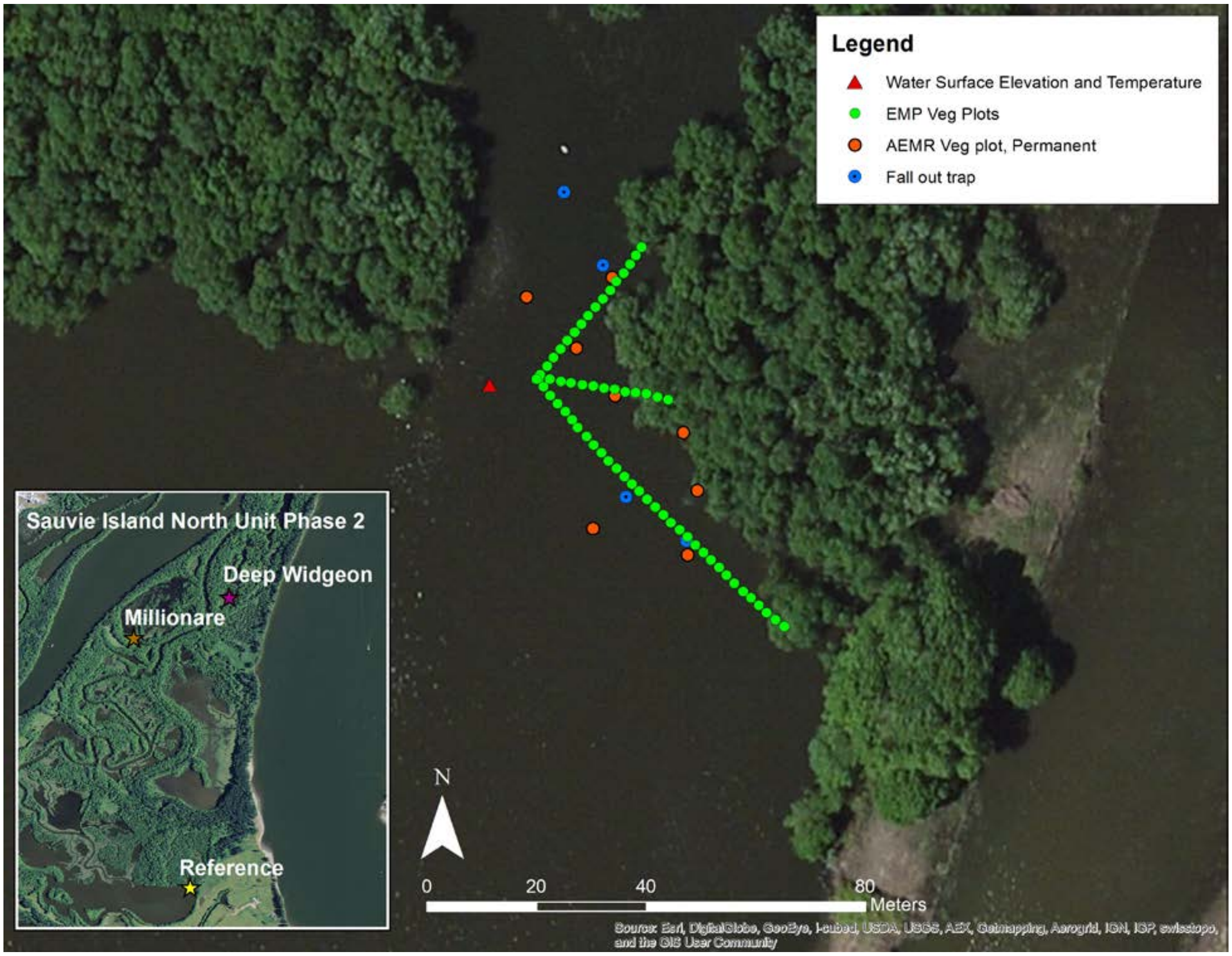


Figure 74. Vegetation and macroinvertebrate sampling locations at the Cunningham Lake reference site. (Fall out traps collected in 2015, neuston tows collected in 2019)

Markers Left on Site

All marking stakes are white ¾ inch PVC with orange duct tape or flagging at the top. We marked the following locations:

- End stakes of the baseline for the vegetation sample areas.
- Permanent quadrat stakes; 2 stakes per location in the diagonal corners (SW and NE).

In addition, 2 1” gray pvc sediment accretion stakes are located on the site and a depth sensor is located inside 1 ½” PVC on a t-post in the channel.

Macroinvertebrate Sampling

Two macroinvertebrate neuston tows were collected in the Cunningham lake vegetation sampling areas.

Appendix B: Site Hydrographs

Hydrographs are in order by site location in the River, starting at the mouth. Followed by hydrology summary statistics for each site.

Wallacut Slough

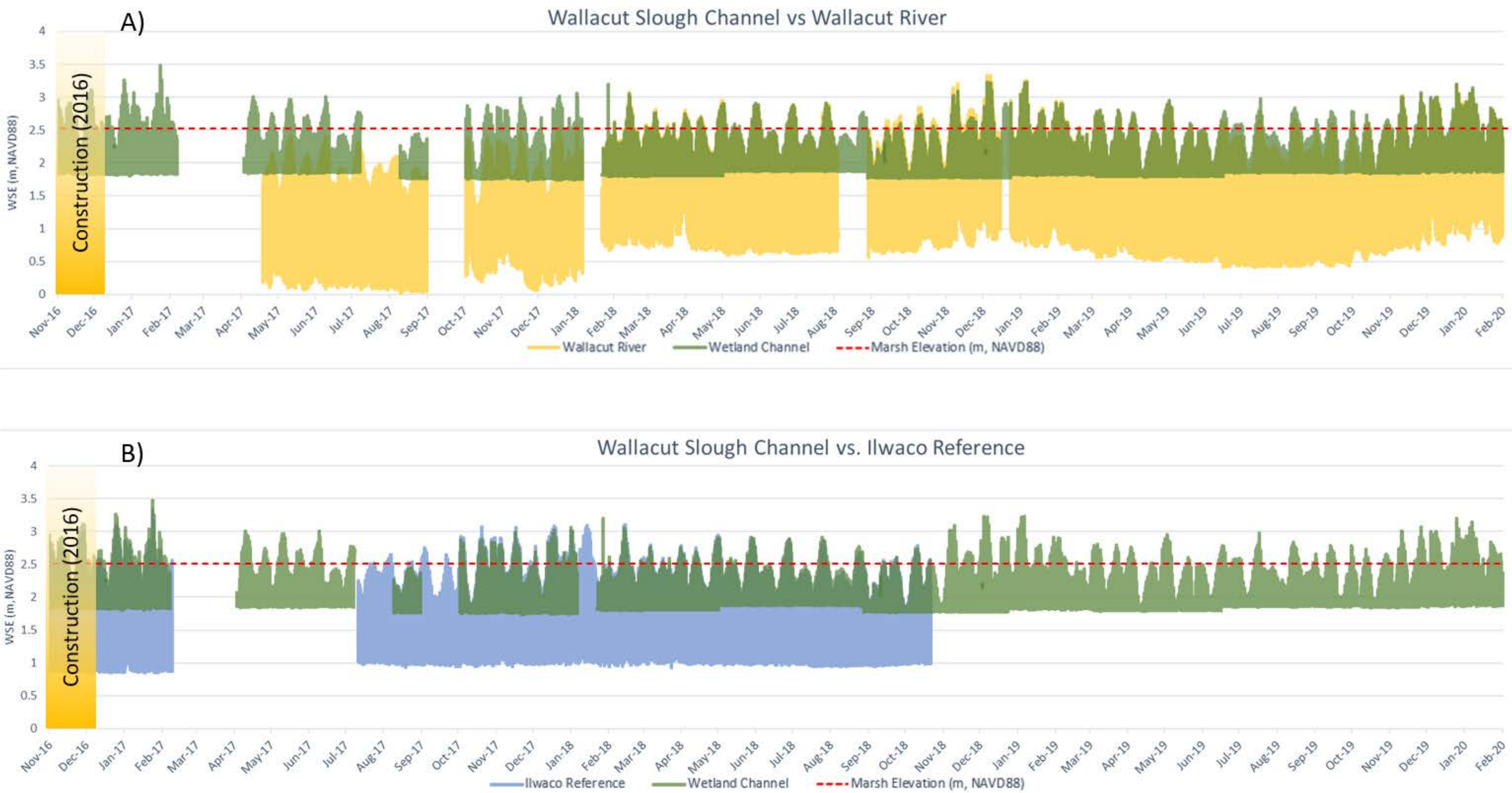


Figure B. 1: Hydrographs comparing Wallacut site hydrology during post-restoration to A) adjacent Wallacut River; B) Ilwaco Slough Reference Site

Table B. 1: Hydrologic Summary Statistics for Wallacut Slough, adjacent river and Ilwaco Slough Reference site. Statistics have been provided for post-restoration period. Pre-restoration data has been excluded due to datalogger error.

WALLACUT							
Year		2019	2018	2017	2016	2016 RESTORATION	2014
		Year 3		**Year 1**			**Pre**
Wetland Channel	Duration	Jan-Dec	Jan-Dec	Jan-Feb, Apr-Dec	Nov-Dec		May - Dec
	Days	365	350	249	45		
	MHHW	2.5	2.5	2.5	2.6		
	MWL	2.0	2.0	2.0	2.0		
	Annual Range	0.6	0.6	0.7	0.8		
	Annual Max	3.2	3.2	3.5	3.1		
Wallacut River	Duration	Jan-Dec	Jan-Dec	May-Dec	NA		
	Days	357	326	211	NA		
	MHHW	2.4	2.5	1.9	NA		
	MWL	1.4	1.5	0.9	NA		
	Annual Range	1.8	1.7	1.7	NA		
	Annual Max	3.3	5.2	2.5	NA		
Ilwaco	Duration	NA	Jan-Jul	Jan-Feb, Aug-Dec	Aug-Dec		
	Days	NA	211	216	147		
	MHHW	NA	2.5	2.4	2.4		
	MWL	NA	1.5	1.5	1.4		
	Annual Range	NA	1.5	1.5	1.5		
	Annual Max	NA	3.1	3.3	3.2		

Steamboat Slough

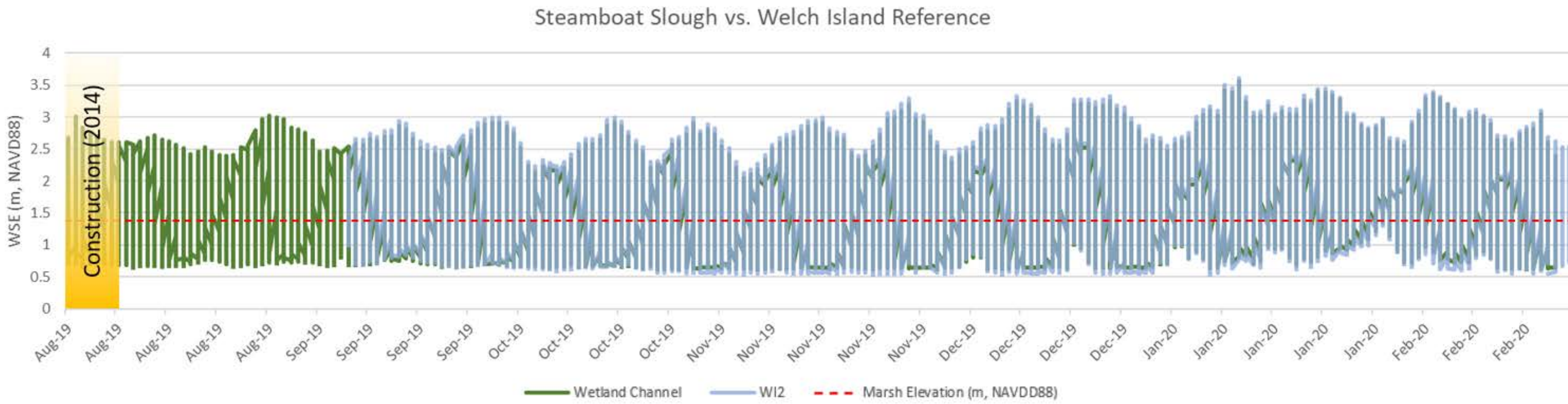


Figure B. 2: Hydrographs comparing site hydrology of Steamboat slough east channel to Welch Island reference site for 5 year post-restoration.

Table B. 2: Hydrologic Summary Statistics for Steamboat East channel and Welch Island Reference site. Statistics have been provided for only 2019. Hydrology data before year 5 post-restoration is unavailable.

STEAMBOAT		
Year		2019 **Year 5**
Wetland Channel	Duration	Aug-Dec
	Days	152
	MHHW	2.7
	MWL	1.6
	Annual Range	2.0
	Annual Max	3.3
Welch Island	Duration	Sept-Dec
	Days	113
	MHHW	2.8
	MWL	1.7
	Annual Range	2.1
	Annual Max	3.3

North Unit Phase 2: Deep Widgeon

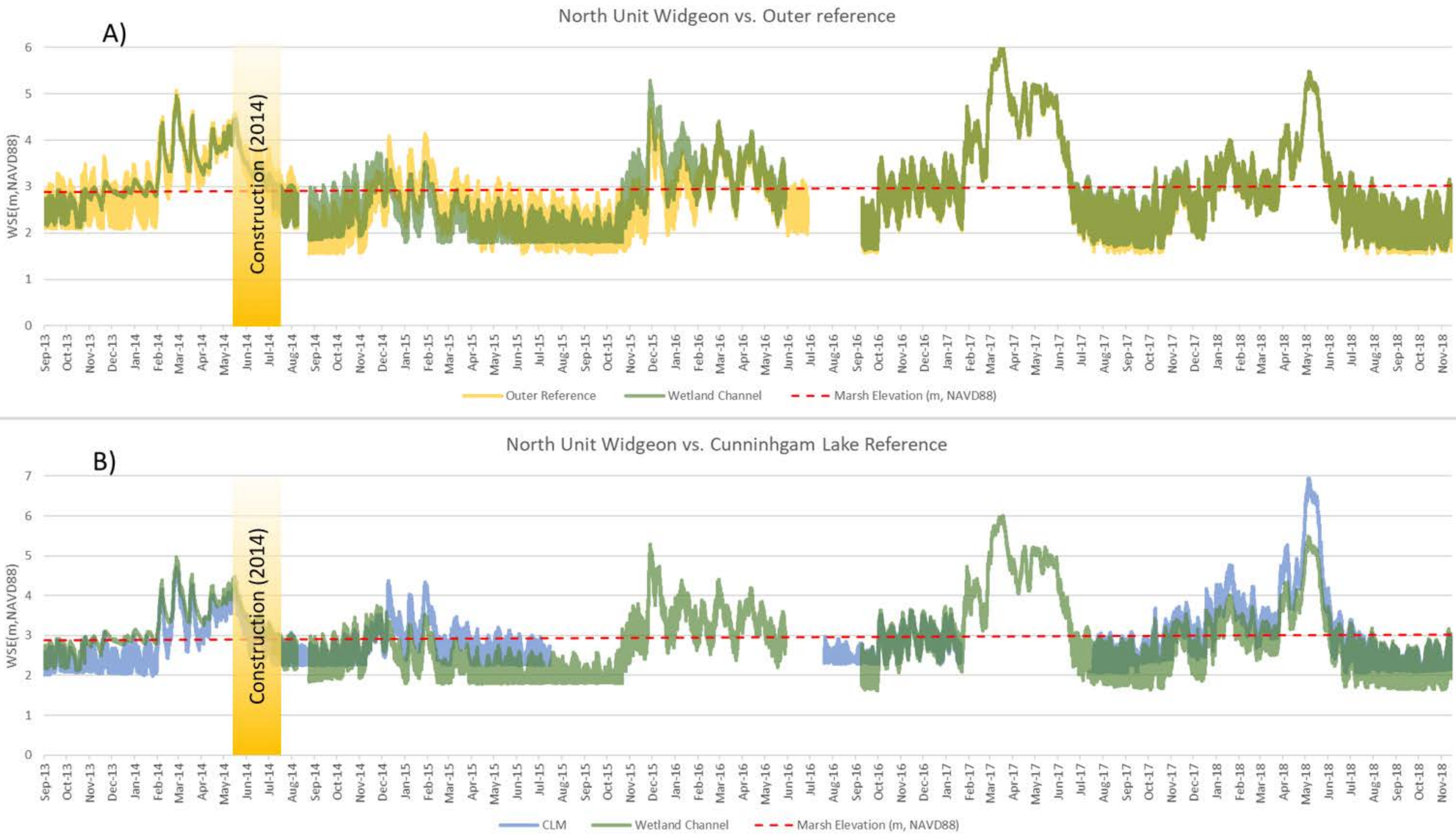


Figure B. 3: Hydrographs comparing Deep Widgeon site hydrology during pre- and post-restoration to A) adjacent outer channel; B) Cunningham Lake Reference Site

Table B. 3: Hydrologic Summary Statistics for Deep Widgeon, nearby outer channel and Cunningham Lake Reference site. Statistics have been provided for pre- and post-restoration periods. 2019 data has been excluded due to datalogger error.

North Unit Phase 2: Deep Wigeon									
Year		2019	2018	2017	2016	2015	2014 RESTORATION	2014	2013
		Year 5		**Year 3**		**Year 1**		**Pre**	**Pre**
Wetland Channel	Duration	Jan-Nov	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec		Jan-Dec	Sept-Dec
	Days		365	365	366	365		350	110
	MHHW		3.2	3.7	3.4	2.7		3.4	2.8
	MWL		2.9	3.3	3.1	2.4		3.2	2.7
	Annual Range		0.6	0.6	0.6	0.5		0.4	0.2
	Annual Max		5.5	6.0	4.4	5.3		5.0	3.1
Outer Reference	Duration		Jan-Nov	Jan-Dec	Jan-Dec	Jan-Dec		Jan-Dec	Sept-Dec
	Days		330	365	365	365		350	110
	MHHW		3.2	3.6	3.2	2.9		3.4	3.0
	MWL		2.9	3.3	2.9	2.5		3.1	2.6
	Annual Range		0.7	0.6	0.6	0.7		0.6	0.8
	Annual Max		5.5	6.0	4.4	4.7		5.1	3.7
Cunningham Lake	Duration		Jan-Dec	Jan, Aug-Dec	Aug-Dec	Jan-Jul		Jan-Dec	Jan-Dec
	Days		365	193	152	209		365	365
	MHHW		3.5	3.0	2.9	3.1		3.2	2.9
	MWL		3.3	2.7	2.7	2.8		2.9	2.6
	Annual Range		0.5	0.6	0.4	0.5		0.4	0.5
	Annual Max		7.0	4.2	3.6	4.3		4.7	4.0

North Unit Phase 2: Millionaire Lake

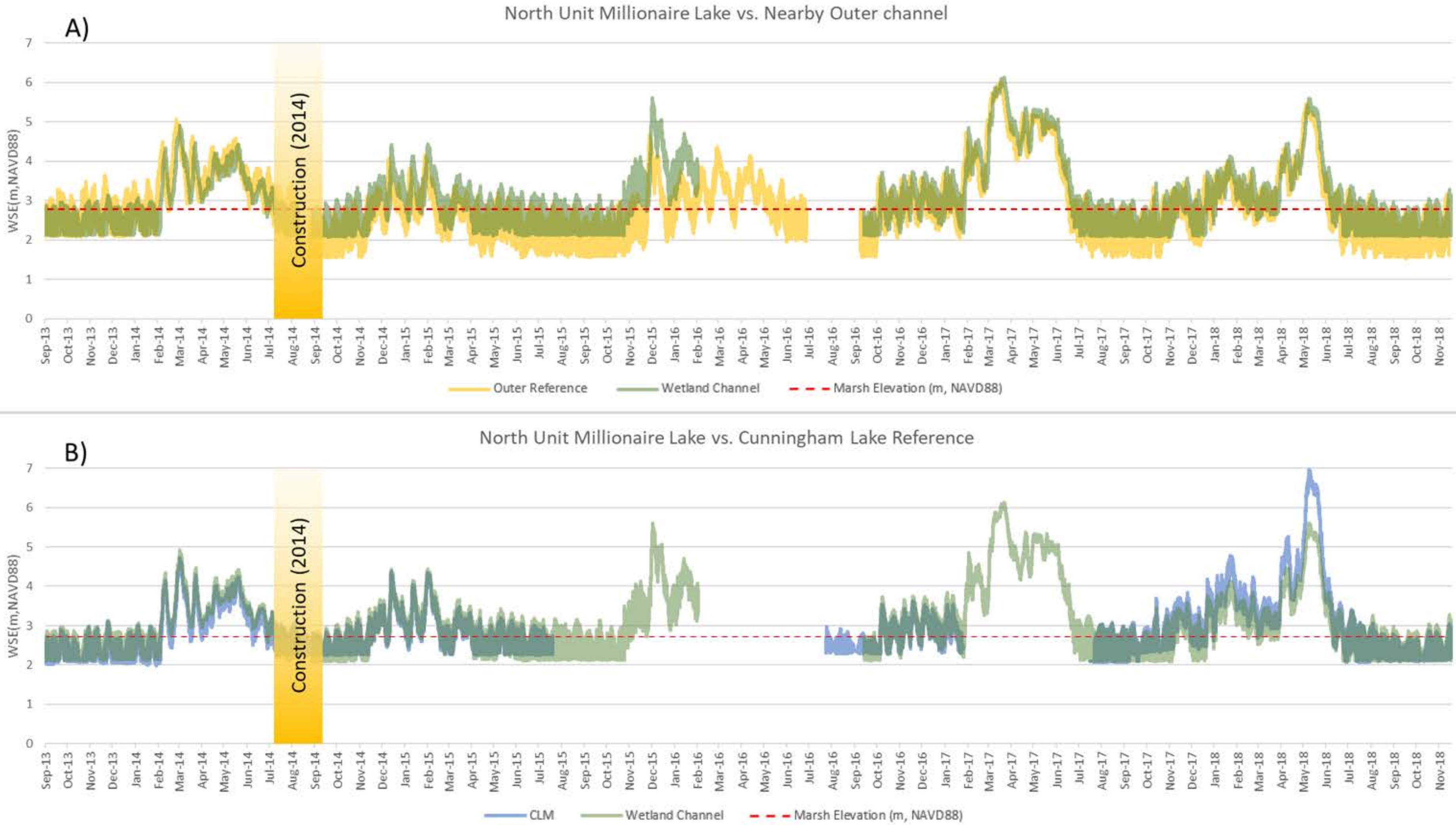


Figure B. 4: Hydrographs comparing Millionaire Lake hydrology during pre- and post-restoration to A) adjacent outer channel; B) Cunningham Lake Reference Site

Table B. 4: Hydrologic Summary Statistics for Millionaire Lake, nearby outer channel and Cunningham Lake Reference site. Statistics have been provided for pre- and post-restoration periods. 2019 data has been excluded due to datalogger error.

North Unit Phase 2: Millionaire Lake

Year		2019	2018	2017	2016	2015	2014 RESTORATION	2014	2013
		Year 5		**Year 3**		**Year 1**		**Pre**	**Pre**
Wetland Channel	Duration	Jan-Nov	Jan-Dec	Jan-Dec	Jan-Dec	Jan-Dec		Jan-Dec	Sept-Dec
	Days		365	365	366	365		348	114
	MHHW		3.3	3.7	3.5	3.3		3.3	2.8
	MWL		3.0	3.5	3.1	2.9		3.1	2.5
	Annual Range		0.5	0.5	0.6	0.7		0.5	0.6
	Annual Max		5.6	6.1	4.7	5.6		4.9	3.1
Outer Reference	Duration		Jan-Nov	Jan-Dec	Jan-Dec	Jan-Dec		Jan-Dec	Sept-Dec
	Days		330	365	365	365		350	110
	MHHW		3.2	3.6	3.2	2.88		3.4	3.0
	MWL		2.9	3.3	2.9	2.49		3.1	2.6
	Annual Range		0.7	0.6	0.6	0.7		0.6	0.8
	Annual Max		5.5	6.0	4.4	4.68		5.1	3.7
Cunningham Lake	Duration		Jan-Dec	Jan, Aug-Dec	Aug-Dec	Jan-Jul		Jan-Dec	Jan-Dec
	Days		365	193	152	209		365	365
	MHHW		3.548874	3.0	2.9	3.1		3.2	2.9
	MWL		3.261313	2.7	2.7	2.8		2.9	2.6
	Annual Range		0.516621	0.6	0.4	0.5		0.4	0.5
	Annual Max		6.954	4.2	3.6	4.3		4.7	4.0