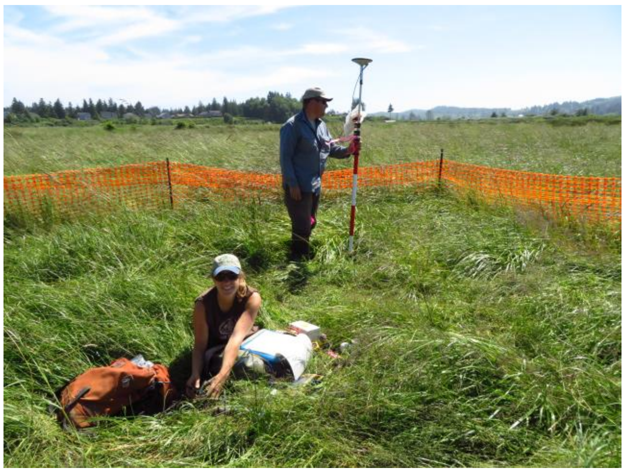
Baseline monitoring at Wallooskee-Youngs restoration site, 2015, Part 2:
Blue carbon, ecosystem drivers and biotic responses

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Note:
This document finalizes reporting on the Estuary Technical Group's 2015 monitoring and blue carbon studies at the Wallooskee-Youngs restoration site and reference sites. An earlier report (Brophy et al. 2016) reported on channel morphology monitoring and provided a progress report on the blue carbon studies.

Data availability: Data from this project are available from the lead author listed above.

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Table of contents

Project overview

This report describes results from baseline monitoring of coastal “blue carbon” (carbon stored in soils of coastal wetlands), tidal hydrology (surface water level), surface water salinity, groundwater level, groundwater salinity, and vegetation composition at the Wallooskee-Youngs restoration site (referred to in this report as the “Wallooskee site”) and reference sites in the Youngs Bay estuary. This work, conducted by the Estuary Technical Group (ETG) at the Institute for Applied Ecology, was funded through a contract between ETG and the U.S. Fish and Wildlife Service's Fish Passage Program.

During 2015, ETG, working in collaboration with a team from Oregon State University led by Dr. Rob Wheatcroft, collected carbon cores from the Wallooskee site and two nearby reference sites (Daggett Point and Cooperage Slough). The cores were analyzed by Erin Peck, graduate student at Oregon State University. The results include sediment and soil carbon accumulation rates at the reference sites, estimated carbon losses that occurred when the restoration site was diked and drained, and the potential for post-restoration carbon storage at the restoration site.

To provide information on the controlling factors (ecosystem drivers) that affect carbon sequestration and the full suite of other valued tidal wetland functions, ETG also monitored the following parameters at the Wallooskee site and reference sites: tidal hydrology (surface water level), surface water salinity, groundwater level, groundwater salinity, soils, and vegetation.

Under a separate contract in 2015 with the Lower Columbia Estuary Partnership (LCEP), ETG also collected and analyzed channel morphology data from the Wallooskee site and four reference sites (Daggett Point, Cooperage Slough, Grant Island, and Fry Island). ETG generated a high-resolution channel network map at the reference sites using automated and consistent LIDAR-derived methods. ETG also analyzed channel cross-section data provided by Statewide Land Surveying for the Wallooskee site, and conducted field measurements of channel cross-sections at two of the reference sites (Daggett Point and Cooperage Slough). The resulting channel morphology data are described in a 2016 report (Brophy et al. 2016).

This project’s combination of a blue carbon study with monitoring of the physical drivers that affect carbon accumulation rates provides interpretive power and an important addition to our understanding of Pacific Northwest tidal wetland ecosystems and their valued functions.

Key findings

Key findings for blue carbon and sediment accretion

- This study was part of a larger blue carbon project extending across multiple Oregon estuaries; the larger investigation provides context and leverages results.
- Diking of the Wallooskee restoration site is estimated to have caused the loss of 34,000 tons of organic carbon (490 tons C_{org}/ha) equivalent to 130,000 tons CO_2.
- With restoration, the Wallooskee site has the potential to sequester the same amount of carbon in the future – 34,000 tons of organic carbon -- or even more, with accelerated sea level rise.
- Accretion at the reference sites was not only keeping pace with sea level rise, but exceeding it. Sediment accretion rate averaged 2.6 mm/yr across all sites, and was similar between the restoration site and reference sites.

- Recent accretion (past 20 to 50 years) at the Wallooskee restoration site was similar to the reference sites (2.8 mm/yr), suggesting that considerable sediment has been entering the site through dike-overtopping flood events.
- Despite its relatively high recent accretion rate, the Wallooskee site has subsided over a meter compared to a nearby reference site, indicating that accretion could not compensate for soil compaction and loss of soil organic matter during the period of agricultural use.
- The long-term carbon accumulation rate (CAR) was similar between the (still diked) restoration site and the reference sites, averaging 80 and 79 g/m^2/yr respectively. CAR ranged from 45 to 210 at individual cores. This range of CAR was similar to that observed at reference sites in the Snohomish Estuary (Crooks et al. 2014) and globally (Chmura et al. 2003, IPCC 2014).
- With future sea level rise, carbon accumulation at restoration and reference sites is likely to increase substantially, as shown by a carbon core outside the dike at the restoration site (core WY04S). This core showed rapid sediment accretion (12 mm/yr) and a resulting high carbon accumulation rate of 210 g C_{org}/m^2/yr, over twice the global average rate for tidal saline wetlands of 91 g C_{org}/m^2/yr (IPCC 2014).
- Carbon densities of cores at restoration and reference sites were 0.023 and 0.026 g/cm^3, respectively, similar to regional and global figures reported by others (Crooks et al. 2014, Chmura et al. 2003).

Key findings for tidal hydrology

- Diking and drainage of the Wallooskee site were effective; daily maximum water levels inside the Wallooskee restoration site were about 1.8 m (nearly 6 ft) lower than at the reference sites.
- Local high tide datums (MHW, MHHW) were calculated for each site; they were highest upstream at Cooperage Slough and were around 10-20 cm higher than at the Astoria NOAA station.
- As is typical for the PNW, daily high tides were higher in winter when rainfall adds to tide heights. The effect was strongest at Cooperage Slough, the farthest upstream site with the greatest influence of river flows on tide heights.
- Annual percent inundation was calculated for reference sites, and ranged from less than 5% for high marsh and shrub swamp, to over 40% for low marsh.

Key findings for channel water salinity and temperature

- Salinity data showed that all of the wetlands in this study should be classified in the Estuarine System (Cowardin classification system) and the Estuarine Coastal System (CMECS classification system).
- Summer daily maximum salinities were in the mesohaline range (5-8 PSU) inside the dike at the restoration site, and at Daggett Point and Grant island reference sites.
- Despite earlier reports that Cooperage Slough is a freshwater tidal wetland, daily maximum salinities at Cooperage Slough reached the mesohaline range (> 5 PSU) in late summer/early fall, approaching the salinities at Daggett Point on Youngs Bay.
- Brackish flows occurred at all sites even in winter; daily salinities peaked in the oligohaline range during spring tide cycles in the winter.
- Daily salinity maxima occurred during the higher high tide, and the highest salinities were during spring tide cycles.
- As expected, salinity decreased upriver, but there was an exception: the station just outside the dike at the restoration site, on the Wallooskee River side (Wallooskee-Out) showed low summer

salinities similar to Cooperage Slough, far up the Youngs River. Apparently, fresh river flows from the Wallooskee River had a strong influence at the Wallooskee-Out station.
- Average daily maximum water temperatures did not differ significantly between the restoration and reference sites, nor did they generally differ among sites.
- Daily maximum temperatures generally exceeded 20°C during July and August at all sites.

Key findings for groundwater level

- Groundwater data showed that the Wallooskee site was effectively drained by the system's dikes and tide gates. During summer, groundwater at the Wallooskee site dropped below the soil surface in late April/early May and remained low all summer; and groundwater levels year-round were significantly lower than at the reference sites.
- Absolute elevations of groundwater at the Wallooskee site were higher than channel water levels during spring and early summer, showing that the site's soils retain water.
- In summer, high marsh and scrub-shrub tidal wetlands at the reference sites showed the typical "spring tide reset" groundwater pattern that we have observed at other sites: groundwater rose to the soil surface during inundating spring tides, but dropped steadily during neap tide cycles.
- Winter groundwater levels at the reference sites generally hovered near the soil surface.
- During inundating tides, groundwater peaks matched the nearby tide gauge peaks, showing that the groundwater level loggers also served as local peak tide gauges.

Key findings for groundwater salinity

- This project was the first to collect detailed time series data on groundwater salinity and groundwater level across a full year in Pacific Northwest tidal wetlands. These data are important for understanding many wetland functions, including carbon sequestration potential.
- Groundwater salinity often varied significantly from channel water salinity, particularly in terms of daily variability and seasonal patterns, illustrating the importance of monitoring groundwater salinity separately.
- Groundwater salinity was significantly lower at the Wallooskee site compared to the reference sites.
- At the Wallooskee site, groundwater salinity was slightly higher than channel water salinity in spring, but the relationship reversed in summer.
- Groundwater was generally fresher than channel water at the Wallooskee site (low oligohaline vs. low mesohaline, respectively).
- Groundwater salinity at Daggett Point (the only reference site where it was monitored) was much less variable than channel water salinity. This was especially true for high marsh and shrub swamp; low marsh groundwater salinity was more responsive to tide cycles.
- At Daggett Point, the relationship between groundwater salinity and channel water salinity was complex. In the spring, groundwater salinity was similar to channel water salinity. In summer, groundwater salinity was lower; in fall and early winter, groundwater salinity was substantially higher than channel water salinity.
- When fall rains returned, groundwater salinity at the Daggett Point reference site remained higher than channel water salinity. All three habitat classes – low marsh, high marsh, and scrub-shrub tidal swamp – had reached mesohaline salinities in the fall, and remained mesohaline until mid- to late January, two months past the transition from brackish to predominantly fresh water in the channel.

- Groundwater salinity was significantly lower during the dry season at Daggett Point high marsh and shrub tidal swamp, compared to the wet season. This relates to the long lag time between precipitation cycles and the response of soil salinity described above.
- Groundwater salinity was not monitored at Cooperage Slough, based on past reports that described the site as a freshwater tidal wetland. However, channel salinities reached the mesohaline range at Cooperage Slough, so we recommend monitoring groundwater salinity at this site in the future.
- Groundwater salinity logger data matched well with validation measurements from a YSI probe, indicating that the loggers were functioning properly in the groundwater wells.
- Stratification of salinity within the groundwater well was minor; salinity differences from top of well to bottom prior to mixing were usually <1 PSU. However, results might be different in more strongly brackish or euhaline wetlands.
- We wrapped groundwater salinity loggers in copper mesh to prevent fouling; this was effective and is recommended for future monitoring (link to photo).
- We compared point-in-time soil and groundwater salinity measurements from three monitoring methods, providing useful information for other projects. Compared to YSI grab samples and soil auger sampling, dataloggers provided much better information on daily, monthly and seasonal variation in soil salinity.

Key findings for soils

- Soils at the Wallooskee site had lower pH, lower salinity, and lower organic matter content compared to the reference sites. However, probably due to the small sample size at the Wallooskee site (n=2), these differences were not statistically significant.
- At reference sites, soil pH was inversely correlated to elevation (higher pH at lower elevations); and organic matter and carbon content were positively correlated to elevation (higher carbon content at higher elevations).
- This study provided an opportunity to compare "snapshot" data from soil probe samples to a time series of groundwater salinity data. The results show that soil salinities from auger samples were generally higher than groundwater salinities using a datalogger, particularly in late summer.
- In this project, our past projects, and related projects, a wide range of conversion factors between % organic matter (by loss on ignition) and organic carbon content have been used. We recommend investigating the reasons for these differences, since they are important to conclusions about carbon storage. Understanding possible reasons for different conversion factors will help the scientific community achieve comparability and consistency across studies.

Key findings for vegetation

- Native plant cover averaged only 4% at the Wallooskee restoration site, significantly lower than at reference sites (94%).
- Cover at the Wallooskee restoration site consisted of typical coastal pasture species (creeping bentgrass, meadow foxtail, ryegrass, bluegrasses, and tall fescue).
- The Daggett Point low marsh was dominated by Lyngbye's sedge; the high marsh was dominated by lady fern, softstem bulrush, and Pacific silverweed.
- Scrub-shrub tidal swamp at Daggett Point was dominated by Hooker's willow, lady fern, and Pacific water parsley. At Cooperage Slough, the scrub-shrub tidal swamp was much more

diverse, with a mix of shrub species (Nootka rose, Douglas spiraea, Hooker willow, black twinberry, and salmonberry) and an understory dominated by lady fern.

- Although not sampled, an interesting lady fern – Nootka rose community occupies broad areas of the Cooperage Slough site.

Study sites

We investigated carbon accumulation, tidal hydrology, channel water salinity and temperature, groundwater level, ground water salinity, soil characteristics, and vegetation at the Wallooskee site and four least-disturbed tidal wetland reference sites located in the Youngs Bay estuary (Figure 1). Site characteristics are summarized in Table 1. At the time of this study's field work, the Wallooskee site was not yet restored, so data from the Wallooskee site presented in this report represent pre-restoration (baseline) data.

The Wallooskee site, totaling 193 acres, is located at the confluence of the Wallooskee River and Youngs Bay (Figure 1). The site was used as a dairy farm for 80 years prior to 2011, when dairy operations stopped. Site modifications to support agricultural use included diking, tide gates, ditching, and tiling. Restoration work, begun in late summer 2015 and completed in 2017, included grading and excavation of channels, tide gate removal, and levee lowering and breaching, with the goal of restoring tidal influence to the site and re-establishing valued tidal wetland functions. The Cowlitz Indian Tribe sponsored the restoration of this site.

The five least-disturbed reference sites (Daggett Point; the marsh just outside the dike at the Wallooskee site; Fry Island; Grant Island; and Cooperage Slough) provided examples of pre-disturbance conditions and goals for a restoration trajectory at the Wallooskee site. Reference sites were selected to represent historical conditions that were likely present at the Wallooskee site prior to diking and conversion to agricultural use. The Daggett Point reference sites was also chosen for its proximity and similar geomorphic setting to the Wallooskee site; other reference sites were added to incorporate the full salinity gradient found in the Youngs Bay estuary, which helps us understand how physical conditions relate to carbon accumulation and other wetland functions. The similarities and differences among the reference sites will help interpret post-restoration changes at the Wallooskee site by providing a “before-after-control-impact” (BACI) statistical framework – optimal for restoration effectiveness monitoring (Stewart-Oaten 1986, 1992), and will contribute to the understanding of carbon sequestration potential in Pacific Northwest tidal wetlands.

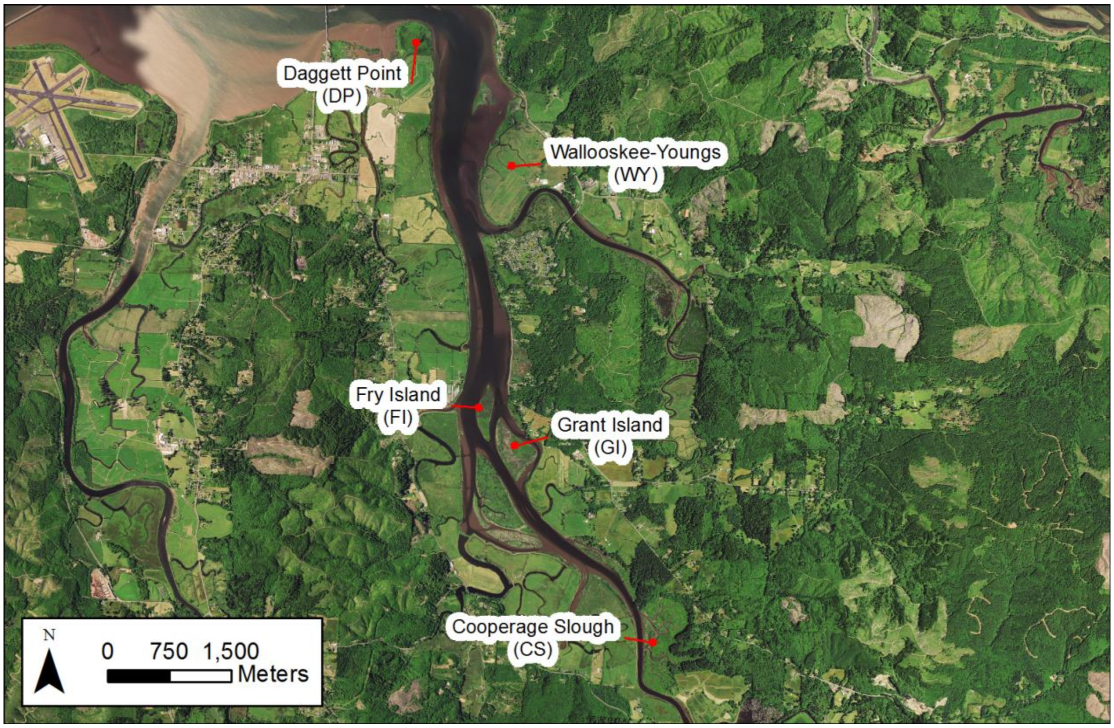

Figure 1. Overview of the study sites in the Youngs Bay estuary, Oregon. Background: NAIP 2014.

Table 1. Study site information. River miles were calculated using the Oregon DEQ RM Calculator (http://deqgisweb.deq.state.or.us/llid/llid.html).

Site	Wallooskee-Youngs (WY)	Daggett Point (DP)	Fry Island (FI)	Grant Island (GI)	Cooperage Slough (CS)
River mile	1.98	0.63	3.89	4.24	6.60
Site type	Restoration (still diked)	Reference	Reference	Reference	Reference
Historical wetland type (1850s)*	Tidal marsh; Sitka spruce or crabapple may be included on higher parts of site.	Tidal marsh; Sitka spruce or crabapple may be included on higher parts of site.	Tidal marsh; Sitka spruce or crabapple may be included on higher parts of site.	Tidal marsh; Sitka spruce or crabapple may be included on higher parts of site.	Sitka spruce swamp; combinations of willow, red alder, red cedar, hemlock; Dense understory may include salmonberry, crabapple, elderberry, gooseberry, briers, ferns, skunk cabbage, and vine maple.
Alterations, impacts	Diking, ditching, grazing, tree and shrub removal	Power transmission tower, possible dredging of large channel	Possible grazing	Enhanced and natural levee/dike, possible grazing	No known alterations
Channel condition	Ditched	Natural, meandering; one possible excavated large channel	Natural, meandering	Natural, meandering	Natural, meandering
Parameters measured**	All parameters	All parameters	Channel morphology	Tidal hydrology, surface water salinity, channel morphology	All parameters

* from Hawes et al. (2008)
** "All parameters" includes blue carbon, tidal hydrology (surface water level), surface water salinity, groundwater level, groundwater salinity, vegetation composition, and channel morphology. For results of channel morphology monitoring, see Phase 1 report (Brophy et al. 2016).

Sample design

Sample design for this project consisted of a series of monitoring stations established to represent major elevation strata at each site. The station locations were selected based on field reconnaissance and review of available data, including elevation, aerial photographs, vegetation, and soil survey maps. Sampling was distributed across the elevation and salinity gradient within each site; for reference sites, the goal was to sample at least one location representative of each major wetland class (low marsh, high marsh, and scrub-shrub tidal wetland, if present). For the restoration site, the two station locations were chosen to represent the lower and higher elevation zones within the broad pasture surface.

There were two types of monitoring stations: **channel monitoring stations** where tide gauges (water level loggers) and channel water salinity/temperature dataloggers were installed; and combined **wetland monitoring stations** where we measured groundwater level, groundwater salinity, soils and vegetation. “Blue carbon” cores (also called "carbon cores") were located directly adjacent to the combined wetland monitoring stations. By co-locating monitoring of these parameters, we could better determine the relationships between physical drivers, biotic responses (plant community composition), and carbon accumulation. The locations and elevations of the combined wetland monitoring stations are shown in Table 2; locations and elevations of channel monitoring stations are provided in Table 6.

Table 2. Locations and elevations of wetland monitoring stations at the Wallooskee site and nearby reference sites. Easting and Northing represent NAD83 UTM Zone 10 N coordinates in meters. Locations are shown in Appendix 1, Maps 1-4. See Appendix 3 for spatial reference system information. Note: WY4 was located outside of the dike and was considered a reference site throughout this study.

Site	Location	Station	Soil surface elevation (m NAVD88)	Soil surface elevation (m MHHW)	Easting	Northing
Wallooskee site	Wallooskee – inside	WY1	1.46	NA	437416	5111465
		WY3	1.89	NA	437639	5111153
Reference sites	Wallooskee – outside	WY4	1.73	-1.085	437719	5110996
	Daggett Point	DP1	1.81	-0.927	436334	5113121
		DP2	2.82	0.083	436302	5113000
		DP3	2.77	0.033	436218	5223003
	Cooperage Slough	CS2	2.52	-0.306	439455	5105495
		CS3	2.71	-0.116	439409	5106095

Sediment and "blue carbon" accumulation

Background

Tidal wetlands, including emergent marshes, forested tidal wetlands, scrub-shrub tidal wetlands, and seagrass beds, sequester carbon -- also referred to as "blue carbon" -- through burial and preservation of organic material (Mcleod et al. 2011). However, once drained and converted for human land uses such as agriculture, these areas may rapidly release carbon back into the atmosphere and lose their capacity for future carbon accumulation. Blue carbon accumulation and sequestration potential for disturbed wetlands in the Pacific Northwest is likely high following restoration (Crooks et al. 2014). Thus, information on blue carbon sequestration potential at restoration sites is required for policy makers and land-use managers to assess alternative land-use policies. Moreover, measurement of historical carbon accumulation and sequestration rates at reference sites is needed to estimate the uptake and carbon preservation potential of possible restoration sites. Finally, knowledge of historical carbon accumulation and sequestration rates in least-disturbed wetlands adds incentive for conservation of these areas.

This study sought to: 1) quantify carbon stocks and carbon accumulation rates at the Wallooskee restoration site and nearby least-disturbed reference sites (Daggett Point and Cooperage Slough);

2) estimate the carbon losses that occurred when the restoration site was diked and drained; 3) predict the post-restoration carbon accumulation capacity of the restoration site; and 4) gain some understanding of potential tidal wetland resilience to sea level rise in the study area. Briefly, soil carbon stocks were calculated by analysis of carbon content and bulk density of sediment within soil cores collected from restoration and reference sites. Blue carbon accumulation rates (CARs) were then calculated using the soil carbon density (carbon concentration) data along with sediment accumulation rates (SARs), determined by analyzing radioisotope levels (^{210}Pb and ^{137}Cs) in the cores.

Our blue carbon study at the Wallooskee site and reference sites was part of a larger investigation addressing blue carbon sequestration across a number of estuaries on the Oregon coast (Wheatcroft 2017). The larger study provides context and leverages this project to substantially advance our understanding of carbon sequestration in the Pacific Northwest. Detailed initial results from several of the sites (including the site sin this study) are provided in Peck (2017).

Methods

Field sites

Field sites are described in "Study sites" above. One carbon core was collected at the monitoring station just outside the dike at the Wallooskee site (WY04S). This was considered a reference site, as it was fully tidal, but was not included in reference site carbon core averages because it cannot be considered "least-disturbed"; its depositional environment is strongly influenced by the adjacent dike.

Field sampling

Field sampling, conducted in February 2015, was designed to meet two criteria: 1) co-locate blue carbon sampling at project monitoring stations to maximize interpretive power and maintain overall project sample design (described in "Sample design" above); 2) enable comparisons between restoration and reference sites; and 3) sample across the elevation gradient, including low marsh, high marsh and scrub-scrub-shrub tidal swamp reference samples.

Project monitoring stations and adjacent blue carbon core locations are shown in Appendix 1, Maps 1-7. Cores were either short (1.5 m in length) or long (3 m). Within the Wallooskee restoration site, two short cores were collected, one at monitoring station WY1 (lower elevation) and one at monitoring station WY3 (slightly higher). One short core was collected at monitoring station WY4, in low tidal marsh just outside the dike (Appendix 1, Map 5; Table 3). At the Daggett Point reference site, a short core was collected at monitoring station DP1 (low marsh), a long core was collected at monitoring station DP2 (high marsh), and another long core was collected at monitoring station DP3 (scrub-shrub tidal swamp). At the Cooperage Slough reference site, a long core was collected at monitoring station CS2 (high marsh), and a short core was collected at monitoring station CS3 (scrub-shrub tidal swamp).

To obtain the cores, PVC pipe 10 cm in diameter and 1.5 or 3 m in length was pounded into the ground using a sledgehammer and retrieved using a truck jack (Figure 2). The location and soil surface elevation at each core was recorded using high-precision RTK GPS/GNSS survey instruments; the spatial reference system used is described in Appendix 3. In the field, each long core was cut into two ~1.5 m sections using a pipe cutter. These sections were stored within the OSU Marine Geology Repository's refrigerated core storage facility. Reference site cores will be maintained at that facility indefinitely and are available for study by others.

Core codes (Table 3) were assigned to carbon cores; these corresponded to the adjacent wetland monitoring station codes used in subsequent sections of this report, with the addition of a letter at the end of the code indicating the core type (short or long). For example, CS02L indicates a long core at Cooperage Slough monitoring station CS2, and DP03S indicates a short core at Daggett Point monitoring station DP3.

Table 3. Locations and soil surface elevations of carbon cores at the Wallooskee site and reference sites. Easting and Northing represent UTM Zone 10 N coordinates in meters (NAD83 datum). Elevations and local mean higher high water (see **Tidal hydrology** below) are expressed in meters NAVD88 (Geoid 12A). For locations, also see Appendix 1, Maps 5-7. See Appendix 3 for spatial reference system information.

Site type	Site name	Monitoring station	Core type	Core code	Easting	Northing	Soil surface elevation (m NAVD88)	Local MHHW (m NAVD88)
Restoration	Wallooskee site	WY1	Short	WY01S	437410	5111457	1.494	2.815*
	Wallooskee site	WY3	Short	WY03S	437651	5111157	1.985	2.815*
Reference	Cooperage Slough	CS2	Long	CS02L	439454	5105485	2.582	2.826
	Cooperage Slough	CS3	Short	CS03S	439405	5106096	2.691	2.826
	Daggett Point	DP1	Long	DP01L	436337	5113121	1.808	2.737
	Daggett Point	DP2	Long	DP02L	436300	5112999	2.890	2.737
	Daggett Point	DP3	Short	DP03S	436219	5113001	2.781	2.737
	Wallooskee site	WY4	Short	WY04S	437726	5111007	1.683	2.815

* Local MHHW for the Wallooskee site refers to the gauge just outside the dike near monitoring station WY4. The interior of the Wallooskee site was diked and therefore not tidally-influenced during this study.

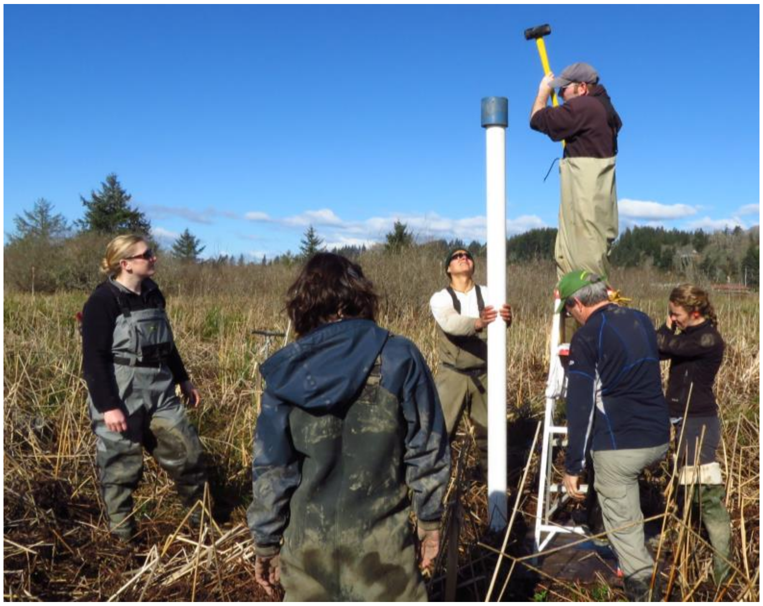

Figure 2. Driving a blue carbon core into the ground.

Laboratory analyses

Soon after collection, all core sections were scanned using a Computerized Tomography (CT) system at the Oregon State University College of Veterinary Medicine. CT imaging provides high-resolution views of sediment stratigraphy (e.g. changes in texture, presence of root mats), and CT images can be used to estimate sediment bulk density. Each core was split in half lengthwise exposing the sediment and providing a working half and an archived half. The top (~50 cm) of each working half was sectioned vertically at 2-cm increments, creating "slices" for analysis; and these "slices" (samples) were then freeze dried for 48 hr, removing all water.

Sediment bulk density

Determination of sediment bulk densities for each core was important in the calculation of carbon stocks, mass accumulation rates, and carbon accumulation rates. The gray-scale value of a CT scan is primarily related to a material's density, with lighter values more dense and darker values less dense (e.g., black indicates air). Thus, a relationship between CT-derived grey-scale value and a subset of experimentally determined dry bulk densities was derived. Sediment bulk density was experimentally determined every 2 cm for the top 50 cm of each least-disturbed core using a method described by Howard et al. (2014). Briefly, known volumes of wet sediment (0.5 -1 cm^3) were sampled using syringes and placed on pre-weighed dishes. Each sample was weighed, dried at 60 °C for 24 hr, and reweighed at room temperature.

Radionuclides

We employed two age dating techniques using the radioisotopes ^{210}Pb and ^{137}Cs, which are commonly used in tidal wetland environments. ^{210}Pb is naturally produced in the environment in two ways. As part of the ^{238}U decay series, ^{226}Ra decays within sediments to ^{222}Rn, which is released to the atmosphere as a gas. ^{222}Rn then decays to ^{210}Pb and is deposited on land by atmospheric fallout. This form of ^{210}Pb is known as unsupported or excess ^{210}Pb and often accumulates and decays in marine sediments following erosion from catchment topsoils. ^{210}Pb is also produced within sediment from the decay of ^{226}Ra. This ^{210}Pb decays to ^{214}Pb and is known as supported ^{210}Pb, because unlike excess ^{210}Pb, it is constant with depth. Assuming a constant deposition rate and a relatively unmixed profile, the activity of excess ^{210}Pb can thus be calculated from the total ^{210}Pb profile and the supported ^{210}Pb profile determined by measurement of ^{214}Pb. The slope of the activity of excess ^{210}Pb with depth ($\frac{\lambda}{\mathrm{SAR}}$) provides an estimate of sediment accumulation rate (Wheatcroft et al. 2013):

$$\mathrm{A_z} = \mathrm{A_0} e^{\left(\frac{-\lambda}{\mathrm{SAR}}\right)\mathrm{z}}$$

where A_z and A_0 is the activity of excess ^{210}Pb at a given depth (z) and at 0 cm, respectively. Λ is the ^{210}Pb decay constant (0.03101/yr) and SAR is the sediment accumulation rate (mm/yr). ^{137}Cs, primarily deposited as fallout from atmospheric nuclear weapons testing, offers an additional estimate of the sediment accumulation rate, assuming its deposition started when weapons testing commenced in 1954, peaked in 1963, and fell sharply thereafter due to the end of aboveground nuclear weapons testing.

Samples were prepared for radionuclide measurement by first removing large pieces of plant material from the dried sediment, then grinding the sample to a constant consistency using a mortar and pestle. Approximately 20 – 40 g of material were weighed into jars and the volume of the compacted and leveled sediment was recorded. Each sample was counted for ≥24 hr on Canberra gamma detectors and the activities of the radionuclides ^{210}Pb, ^{214}Pb, and ^{137}Cs were measured at their respective photopeaks (46.5, 352.0, and 661.6 keV) (Wheatcroft and Sommerfield 2005).

Mass accumulation rates (MAR; g/cm^2/yr) were calculated from the slope ($\frac{\lambda}{\mathrm{MAR}}$) of excess ^{210}Pb activities plotted against cumulative mass (m; g/cm^2), determined using the change in depth and bulk density:

$$A_z = A_0 e^{\left(\frac{-\lambda}{\mathrm{MAR}}\right)m}.$$

To help interpret results, we calculated a rough approximation of the time period represented by the sediment, mass and carbon accumulation rates. To determine the approximate time period, we divided the core depth used for the sediment accumulation rate calculation (in mm) by the sediment accumulation rate (in mm/yr) and rounded to the nearest 10 years. Sediment accumulation rates determined using ^{137}Cs represented the past 61 years, by definition (since ^{137}Cs first appeared in 1954).

Carbon content

Analyzed with sediment accumulation rates, sediment carbon densities (carbon concentrations) allow us to quantify carbon accumulation rates within the restoration site and nearby least-disturbed reference sites. Carbon densities also allow calculation of estimated carbon losses that occurred when the restoration site was diked and drained, and prediction of the post-restoration carbon accumulation capacity of these sites. Loss on ignition (LOI) was used to measure organic matter content within the sediment samples, and a relationship between organic matter and organic carbon content was additionally determined for a subset of samples using an automated elemental analyzer (CHN analyzer). The LOI technique consisted of weighing the freeze-dried sediment before and after combustion in a muffle furnace at ~550 °C for 4-8 hr (Heiri et al. 2001). The CHN method consisted of packaging ~150 mg of dried sediment into a tin capsule, followed by instrumental analysis (Howard et al. 2014). Two samples from each reference core, one with the highest measured organic content and one with the lowest, were measured. To bolster this data, samples from the larger study of which this project is a part (Peck 2017, Wheatcroft 2017) were included; these were from the Tillamook Bay and Salmon River estuaries of Oregon.

Mean carbon density within the top 50 cm for each core was calculated by averaging the product of dry bulk density and the fraction of organic carbon for each 2-cm increment. Values of carbon density for the reference sites allowed us to calculate both the mass of carbon lost after diking in the restoration site and the mass of carbon that could potentially be stored after restoration.

Carbon accumulation rates (CARs, g C/m^2/yr) were calculated in a similar manner to mass accumulation rates; however, the cumulative mass of organic carbon (m_C; g C/cm^2), calculated as the cumulative mass multiplied by the percent organic carbon, was plotted against excess ^{210}Pb activity:

$$A_z = A_0 e^{\left(\frac{-\lambda}{\mathrm{CAR}}\right)m}.$$

Descriptive statistics were calculated for the carbon cores, representing variability within each core. Due to the low number of replicate samples, statistical analysis of differences between restoration and reference cores or between habitat classes were not possible.

Results

As described above, average sediment, mass, and carbon accumulation rates for reference sites below exclude data from the carbon core outside the dike at the Wallooskee site (core WY04S), for the reasons described in "**Field sites**" above.

CT scans

The CT scans showed that the sediment cores were in good shape following collection and transportation; many biogenic and sedimentary structures typical of tidal wetland sediment were well preserved within the top 50 cm of each core (Figure 3). Some cores had roots and shoots present (Figure 3G and 3H), but in general, plant material was limited below the top few centimeters of soil. DP01 showed the most variability in density, with lighter, denser sediment deeper in the core and darker, more organic-rich sediment towards the surface (Figure 3E), probably showing the location's history of transition from more mineral-dominated mud flat conditions to the organic-rich sediment characteristic of tidal marsh.

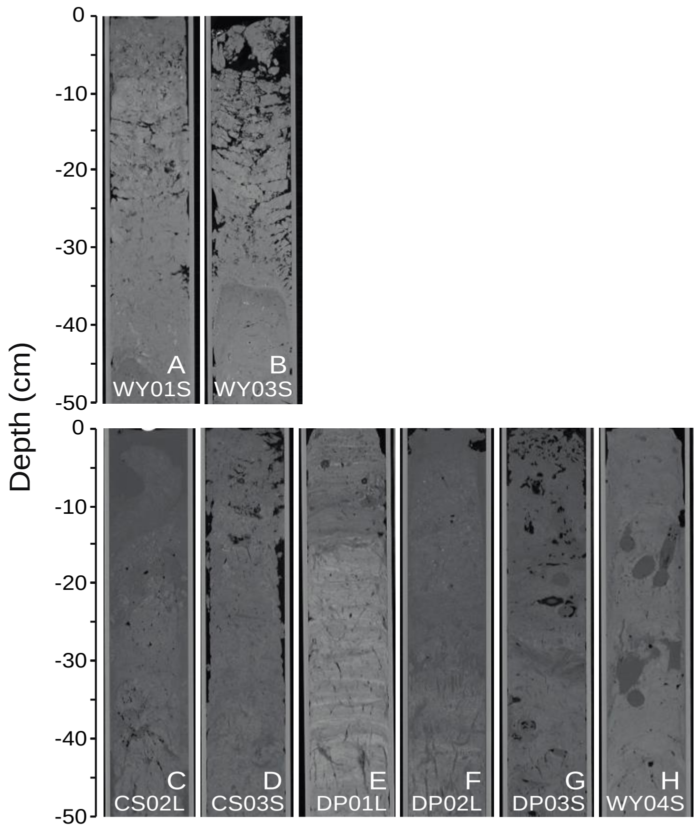

Figure 3. CT scans of the top 50 cm of each core. Lighter regions are more dense, while darker regions are less dense. Each scan is labeled with a letter A through H. Scans A – B are from cores collected from the restoration site, while scans C – H are from cores collected in reference sites.

Sediment dry bulk density was strongly correlated with CT-derived grayscale value (Figure 4). This simple linear regression ($y = 0.012x - 0.62$, $R^2 = 0.70$) was used to calculate dry bulk density from the CT scans for the cores collected at the reference sites and for WY04S.

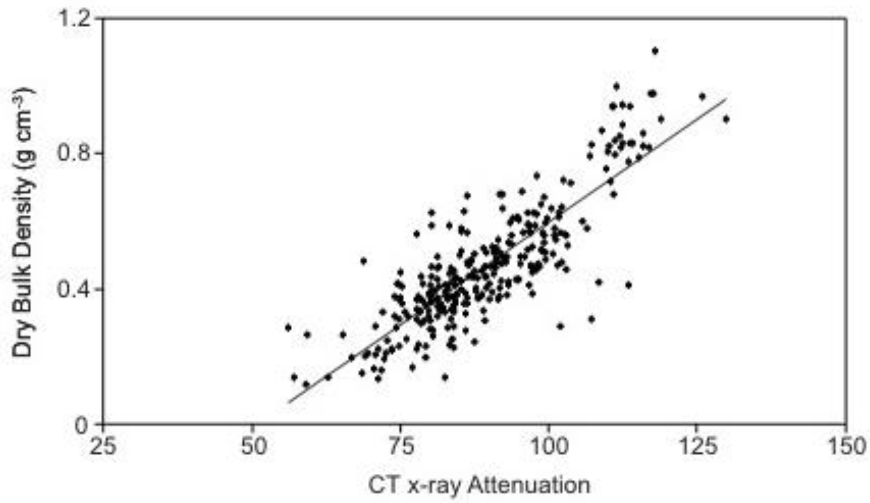

Figure 4. Relationship between dry bulk density measured directly from reference site cores and 8-bit (0 – 255) grayscale values of CT x-ray attenuation (n = 302). The linear regression has an R^2 value of 0.70, a slope of 0.012, and a y-intercept of -0.62.

Radioisotope profiles

Graphs ("profiles") of excess ^{210}Pb by depth (e.g. Figures 5 and 6) are used to determine whether the ^{210}Pb method is working as expected, and to reveal outliers or other problematic data points that should be omitted when estimating sediment and carbon accumulation rates. When the method is working well, excess ^{210}Pb should show a steady decrease with increasing depth. This study's profiles of excess ^{210}Pb generally showed this pattern (Figures 3 and 4), but there were some individual data points (core slices) that did not follow this pattern (gray points in Figures 5 and 6); these points were excluded from the calculation of sediment accumulation. For instance, any sediment slices with error ranging below the detection limit of the γ-ray spectrometers (3 Bq/kg) were excluded. In some cases, surface sediment organic content can dilute the inorganic component of the slice, causing low ^{210}Pb activities. The surface slices from cores WY03S, DP01L and DP03S showed this effect, and were therefore omitted from the analysis. Some profiles exhibited outliers, potentially due either to a change in sediment grain size or sediment movement by burrowing organisms (e.g., lower slices in core DP02L); these outliers were also excluded from the regression equation.

When the ^{137}Cs method is working well, ^{137}Cs profiles should show a "mountain-shaped" curve belowground, with a peak corresponding to the 1963 Partial Test Ban Treaty and a decrease in ^{137}Cs concentration from that point to the present. This pattern was not evident for most of the cores in this project. The ^{137}Cs profiles for all but one core showed higher-than-expected concentrations of ^{137}Cs that peaked at or near the sediment surface (data not shown), suggesting upward movement and loss of ^{137}Cs after deposition. Unlike ^{210}Pb, ^{137}Cs is relatively mobile, becoming un-adsorbed from sediment particles after deposition due to changes in sediment chemistry such as salt intrusion, which allows this "post-depositional remobilization." Cores with profiles indicating post-depositional remobilization cannot be used for the ^{137}Cs method. However, rapid accumulation of sediment can prevent post-depositional remobilization by reducing the time during which remobilization can occur.

The single core that showed the expected peak in ^{137}Cs concentration was WY04S, the core with the fastest sediment accumulation rate (which apparently prevented post-depositional remobilization of ^{137}Cs, as described above). Since two dates could be derived from the WY04S ^{137}Cs profile, the average of the dates was used to calculate the sediment accumulation rate. Because a sediment accumulation rate could be calculated for WY04S using the ^{210}Pb method, as well, these two rates were again averaged and the error was determined as the standard deviation of the values.

Sediment and mass accumulation

The sediment accumulation rates determined for the two restoration site cores (WY01S and WY03S) were 1.9 and 3.8 mm/yr respectively; these data represent approximately the last 50 years and 20 years, respectively (Table 4). Because only two cores were collected within the restoration site, we must assume that an average of the two (2.8 mm/yr) is representative. The mean rate of sediment accumulation for the reference sites was nearly the same at 2.6 mm/yr, and the reference site data represented somewhat longer time periods, 60-110 years (Table 4). SARs did not differ significantly among the reference sites (Grubb's test for statistical outliers, $p<0.05$).

Mass accumulation rates better account for sediment compaction than sediment accumulation rates since density changes with depth are accounted for in the calculation. Because of this, mass accumulation rates are often preferred measures of accumulation. The average mass accumulation rates for the restoration and reference sites were very similar, at 0.86 and 0.85 kg/m^2/yr, respectively (Table 4).

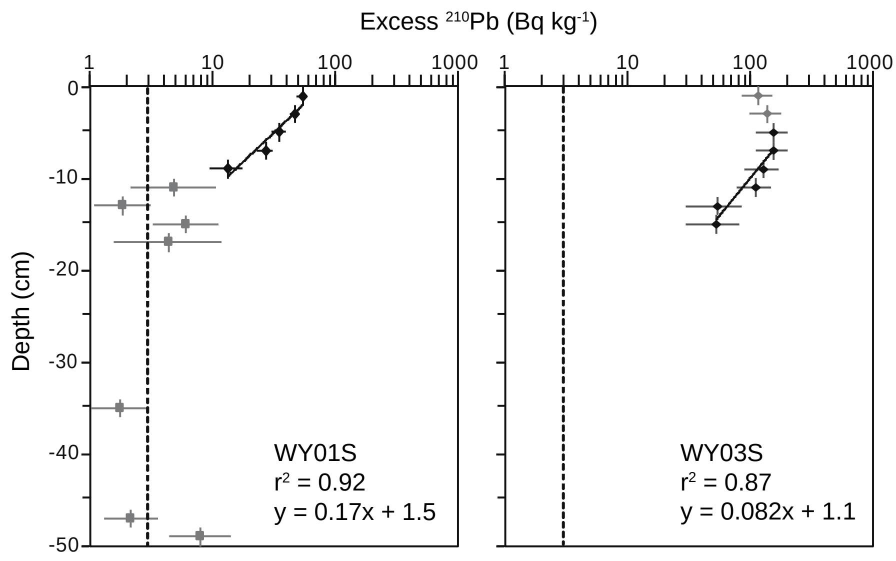

Figure 5. Depth profiles of excess ^{210}Pb measured in each restoration site core. The vertical dashed line indicates the detection limit (3 Bq/kg) of the gamma detectors. The vertical bars indicate the height of the "slice" of sediment sampled (2 cm), and the horizontal error bars represent the detector error. The gray points were not included in the regression. The black points were used to calculate sediment accumulation rates.

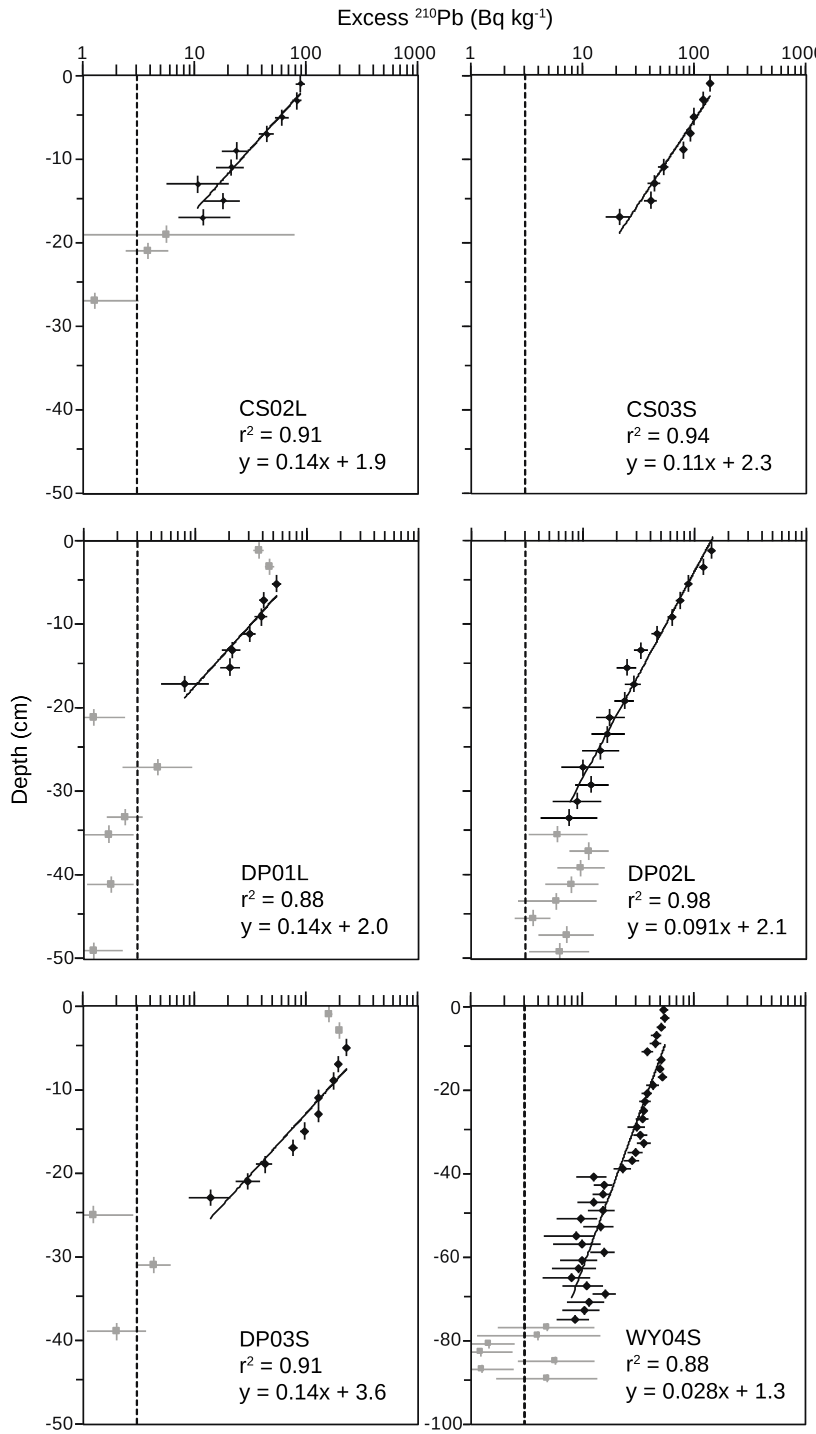

Figure 6. Depth profiles of excess ^{210}Pb activities measured in each reference site core. The vertical dashed line indicates the detection limit (3 Bq/kg) of the gamma detectors. The vertical bars indicate the height of the "slice" of sediment sampled (2 cm), and the horizontal error bars represent the detector error. The gray points were not included in the regression. The black points were used to calculate sediment accumulation rates. Note that the data are shown to a depth of 100 cm rather than 50 cm for WY04S, due to higher ^{210}Pb levels at depth.

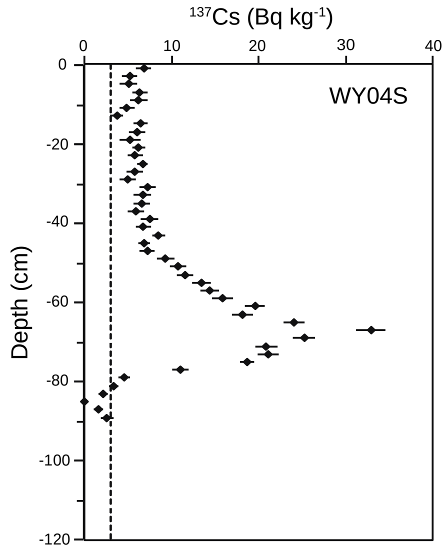

Figure 7. ^{137}Cs concentration measured in core WY04S. The vertical dashed line indicates the detection limit (3 Bq/kg) of the gamma detectors. The vertical bars indicate the height of the "slice" of sediment sampled (2 cm), and the horizontal error bars represent the detector error.

Relationship between soil organic matter and carbon content

The relationship between LOI-derived organic matter and organic carbon measured by elemental analysis (Figure 8) agreed well with equations calculated by others (Morris and Whiting 1986; Craft et al. 1991). Crooks et al. (2014), however, determined a different relationship for the Snohomish Estuary (WA). The reasons for this difference are unknown; in the meantime, it appears that region-specific relationships between organic carbon and organic matter must be determined. Given the closeness of fit, the relationship calculated here is appropriate for Northern Oregon wetlands and was thus applied to all core samples to convert organic matter to organic carbon. The relationship is quadratic because sediments high in organic matter are more subject to decomposition. Organisms that break down organic compounds preferentially take up nutrients such as phosphorus and nitrogen, thus causing the ratio of carbon to organic matter to increase with greater organic matter concentrations.

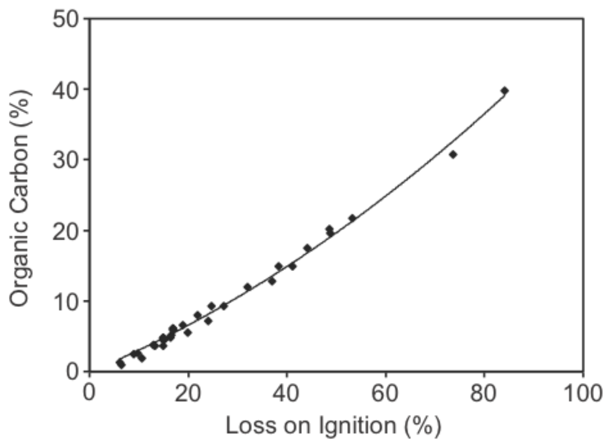

Figure 8. Relationship between organic carbon (C_{org}), measured using elemental analysis, and organic matter measured by LOI, using data from this study and other Northern Oregon sites in the Salmon River and Tillamook Bay estuaries (n = 34) (Peck 2017). The R^2 value of the regression is 0.99 and the equation is $C_{org} = 0.0021 \bullet LOI^2 + 0.29 \bullet LOI$.

Carbon profiles, densities and accumulation rates

Profiles of organic carbon showed a decreasing trend with depth (Figure 9) due to decomposition during and after burial. In general, restoration site cores (Figure 9A) had lower concentrations of organic carbon than the reference site cores (Figure 9B). DP01L and WY04S had similarly low organic carbon contents. These core locations were lowest in elevation amongst the reference sites, resulting in greater mineral sediment accumulation -- and therefore lower organic carbon density.

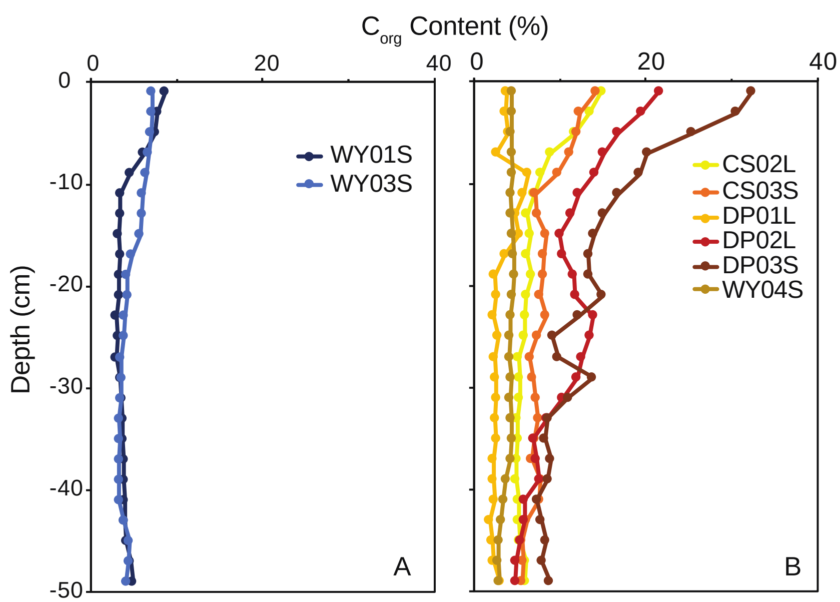

Figure 9. Organic carbon profiles for each core. Subplot A shows restoration site cores and subplot B shows reference site cores.

Table 4. Sediment, mass and carbon accumulation rates for each core location. Rates measured at WY04S were not included in the reference mean (see "**Field sites**" above). Starred (*) values were calculated using an average of ^{210}Pb method derived SAR and ^{137}Cs method derived SAR, and were not included in reference site means, as described in "**Field sites**" above.

Site type	Core Location	Sediment accumulation rate (mm/yr)	Mass accumulation rate ($kg/m^2/yr$)	Carbon accumulation rate (g $C_{org}/m^2/yr$)	Approximate time period represented (yr)*
Restoration	WY01S	1.9 ± 0.3	1.0 ± 0.1	68 ± 13	50
	WY03S	3.8 ± 0.7	0.71 ± 0.10	91 ± 19	20
	MEAN	**2.8 ± 1.3**	**0.86 ± 0.21**	**80 ± 16**	
Reference	CS02L	2.2 ± 0.3	0.71 ± 0.10	54 ± 7	80
	CS03S	2.9 ± 0.3	1.1 ± 0.1	100 ± 10	60
	DP01L	2.3 ± 0.4	0.93 ± 0.10	45 ± 6	80
	DP02L	3.4 ± 0.1	1.1 ± 0.0	130 ± 0	100
	DP03S	2.2 ± 0.2	0.42 ± 0.04	64 ± 7	110
	WY04S	12 ± 1*	6.1 ± 0.6*	210 ± 10*	70
	MEAN	**2.6 ± 0.1**	**0.85 ± 0.29**	**79 ± 36**	

* Rounded to the nearest 10 yr

Soil carbon stocks and carbon accumulation rates

Soil carbon stocks for each core location were calculated as the product of carbon density and sediment depth (Kauffman and Donato 2012; Table 5). Carbon stocks were calculated to a depth based on the portion of a nearby long core that showed no change in lithology (i.e. no major textural change to the depth in question). The CT scans of the cores showed little lithology change with the exception of DP02, which had a tsunami sand deposit at 124 cm depth. This depth was also applied to the DP03S core location, since it was collected nearby and likely exhibits the same deposit at a similar depth. The full depths of the long cores that showed no change in lithology, such as CS02L and DP01L, were used in calculations. Even though core CS03S was a short core, it was assumed that carbon stocks at that location extended to the same depth as at CS02L, since the depositional environment was similar; therefore, the depth used for CS02L was also used for the CS03S carbon stock calculation. For the Wallooskee site, the depths of the short cores were used because no long core was taken near that location. Carbon density was only available for the top 50 cm of each core (for which slices were analyzed for LOI), and the average density for the top 50 cm was assumed to be representative of the full depth for which no lithology change was observed. The CO_2 equivalent pool for each core was calculated using a conversion factor of 3.67 (the ratio of the molecular weight of CO2 to the atomic weight of carbon) (Table 5).

Carbon densities for the restoration and reference sites were similar, averaging 0.023 and 0.026 g C_{org}/cm^3, respectively (Table 5). Carbon accumulation rates were also not significantly different, averaging 80 and 79 g $C_{org}/m^2/yr$ for restoration and reference sites, respectively (Table 4).

Carbon impacts of diking and restoration

Using information collected at the reference sites, we can predict the mass of carbon lost when the Wallooskee restoration site was diked, and the mass of carbon that will be stored following restoration of the site. These masses are considered the same, since the Wallooskee site has not subsided below

Mean Tide Level, the typical lower elevation boundary for vegetated marsh (Warren Pinnacle Consulting, Inc. 2012; Thorne et al. 2015; Brophy and Ewald 2017) – in other words, as the restoration site equilibrates with sea level, it is expected to accumulate the same amount of carbon it once stored prior to diking and drainage. The high marsh location at the Daggett Point reference site is the expected wetland type and elevation for the restoration site once it equilibrates with sea level.

The area of the Wallooskee site is 70.64 hectares, and its approximate average elevation is 1.63 m NAVD88. Because elevation at the Daggett Point high marsh (core DP02L, the appropriate reference site) is 2.89 m NAVD88, the amount of subsidence at the Wallooskee site was estimated at 1.26 m, and the change in soil volume was estimated to be 890,000 m^3. This method for assessing volume change (using an average elevation change) is simplified from Crooks et al. (2014), who used a more detailed method to assess the changes in morphology of the restoration surface. However, our simple approximation is reasonable, since the Wallooskee site is relatively flat. Using the mean carbon density for the top 50 cm of DP02L (0.035 g C_{org}/cm^3), the total mass of organic carbon lost to the atmosphere after diking of the Wallooskee site was 34,000 tons C_{org}. Assuming no change in the 70.64 ha area, this mass translates to 490 tons C/ha lost. Applying a conversion factor of 3.67 to calculate the mass of CO_2 equivalents (Kauffman and Donato 2012), 130,000 tons CO_2 or 1,800 tons CO_2/ha were lost to the atmosphere due to diking and drainage of the site. As described above, these values also represent the amount of carbon and CO_2 that could be stored in the soil at the Wallooskee site following restoration.

Table 5. Mean carbon densities, soil carbon stocks and CO_2 equivalent pools for carbon cores. Carbon densities were measured within the top 50 cm of each core; core depths used to calculate soil carbon stocks are shown.

Site type	Core Location	Mean carbon density (g C_{org}/cm^3)	Core depth used to calculate carbon stock (cm)*	Soil organic carbon stock (tons/ha)	CO_2 equivalent pool (tons/ha)
Restoration	WY01S	0.026 ± 0.005	83	240	870
	WY03S	0.020 ± 0.004	87	190	700
	MEAN	**0.023 ± 0.004**			
Reference	CS02L	0.023 ± 0.003	245	620	2300
	CS03S	0.029 ± 0.006	245	780	2900
	DP01L	0.021 ± 0.004	254	590	2200
	DP02L	0.035 ± 0.009	124	480	1800
	DP03S	0.031 ± 0.011	124	420	1600
	WY04S	0.018 ± 0.002	90	180	650
	MEAN	**0.026 ± 0.007**			

* Core depth shows the depth used for the carbon stock calculation. For example, a depth of 245 cm was used for core CS02L, because no lithology change was observed to that depth.

Discussion

Sediment and carbon accumulation and loss

Historic vegetation mapping (Hawes et al. 2008) shows that prior to diking, the Wallooskee site was a tidal marsh, as was the Daggett Point reference site. Given the landscape setting of the two sites, it is likely they had similar elevations, similar vegetation, and fairly similar depositional environments. Thus, we expected sediment and carbon accumulation rates to have been similar between these two sites up until the time of diking. During the period when the Wallooskee site operated as a dairy farm (over 80 years, until 2011), we expected that sediment accumulation would have decreased due to loss of daily tidal inundation; and carbon accumulation would have decreased because the soil was drained, reducing anaerobic accumulation of deposited organic matter. In addition, previously sequestered carbon would have been lost during this period of agricultural use, due to drainage and drying of soils, which allowed oxidation and decomposition of some of the previously buried carbon.

Within the restoration site, the age dating techniques used in this study did not provide estimates of sediment accumulation rates before diking, due to poor preservation of older sediments. However, the restoration site cores did yield recent rates of sediment accumulation, mass accumulation, and carbon accumulation (past 20-50 yr), which averaged 2.8 mm/yr, 0.86 kg/m^2/yr, and 80 g C_{org}/m^2/yr respectively. Interestingly, these rates were nearly identical to the average rates at the reference sites over the past 70-110 yr (2.6 mm/yr, 0.85 kg/m^2/yr, and 79 g C_{org}/m^2/yr respectively). Thus, despite the dikes, the restoration site has not only been accumulating sediment, but its recent sediment, mass, and carbon accumulation rates appear to have been similar to the longer-term rates at the least-disturbed reference sites. Most likely, winter storms flooding the restoration site provided the sediment to maintain these relatively high recent rates of accretion. River flows during winter storms carry high sediment loads, so these flood events may have allowed the restoration site to maintain similar sediment accumulation rates despite the dikes.

Despite relatively high recent sediment and carbon accumulation rates, it is clear that diking of the Wallooskee site led to large losses of carbon to the atmosphere, because the site's elevation is about 1.26 m lower than the Daggett Point high marsh, and as described above, we expect its elevation was similar prior to diking. Subsidence of this type is commonly observed in diked tidal wetlands; it is caused by drainage of soils resulting in loss of soil organic matter through decomposition; by compaction of soils by farm machinery and livestock trampling; and by other factors (Frenkel and Morlan 1990, 1991). The Wallooskee site's elevation loss of 1.26 m probably occurred early after diking, as much as 80 years ago; and the accretion rates measured in this study are representative of only the last 20-50 years (as described above). Clearly, the site's relatively high sediment accretion rate during recent years has not compensated for that earlier subsidence, as shown by the site's net elevation loss. As the site equilibrates with sea level after restoration, carbon accumulation is likely to occur at rates similar to nearby reference sites at similar elevations, eventually leading to storage of the same amount of carbon that was lost after diking – and potentially more, as the site equilibrates to the increased sea levels predicted with climate change. However, very rapidly rising sea level could potentially "drown" the wetlands at the site if inundation is frequent and deep enough to prevent growth of sediment-trapping vegetation, preventing full restoration of the original carbon stocks.

In another monitoring project on the northern Oregon coast, we collected carbon cores and measured accretion using feldspar marker horizon plots at the Tillamook Southern Flow Corridor tidal wetland restoration site ("SFC site") and reference sites (Brophy et al. 2017) during 2014-2017. Just as observed at the Wallooskee site, the results from the Tillamook study also showed higher-than-expected sediment and carbon accumulation within the restoration site prior to dike removal (in 2014). These results were attributed to sediment inputs from river floods that overtopped the dike. Post-restoration monitoring in 2017 using the feldspar marker horizon method showed a significant increase in sediment accumulation at the SFC site (Brophy et al. 2017), indicating increased potential for carbon accumulation after restoration.

Interestingly, sediment accumulation rates in this study did not relate closely to soil surface elevations at each core. Typically, lower-elevation tidal wetlands (e.g. low marsh) have higher sediment accretion rates than higher-elevation tidal wetlands (e.g. high marsh) because more frequent inundation results in greater sediment inputs, other factors being equal (Morris 2002, Chmura et al. 2003). In this study, there was no clear relationship between the wetland surface elevation at each core site, and the sediment accumulation rate for that core. This is in contrast to our monitoring at the Tillamook sites (Brophy et al. 2017), where low marsh sediment accumulation rates averaged significantly higher than the rates at high marsh and scrub-shrub tidal swamp (Brophy et al. 2017). However, this study included only two reference cores at low marsh elevation (WY04S and DP01L), and it is possible that more extensive sampling would show more typical relationships between elevation and sediment accretion rates.

Climate change resilience

The results of this study offer hope for climate change resilience in Youngs Bay tidal wetlands. The mean accretion rate calculated from carbon cores in least-disturbed high marsh and scrub-shrub tidal swamp (2.6 mm/yr) was unexpectedly high, given an estimated sea level fall within Youngs Bay of 1.0 mm/yr over the past century (Komar et al. 2011). Elevations of least disturbed high marsh and tidal swamp in the Pacific Northwest generally equilibrate slightly higher than MHHW (Brophy et al. 2011, Janousek and Folger 2014). As relative sea level rises, multiple feedbacks between morphology and vegetation allow the tidal wetland surface to accrete at a similar pace. Briefly, as tidal inundation increases, more suspended sediment can settle onto the wetland surface. Simultaneously, marsh vegetation density increases, trapping more sediment, reducing wave energy, and increasing organic matter accumulation both above and below the wetland surface (Kirwan and Megonigal 2013). Therefore, because the mean accretion rate within the reference sites do not match sea level rise, either relative sea level rise does not control marsh accretion within Youngs Bay, or relative sea level is rising faster than previously measured. For instance, discharge from the Columbia River may increase sea level within Youngs Bay. Seasonal changes in sea surface height may have caused high accretion rates, as well. Winter in Oregon, the primary season during which sediment is delivered to the marsh surface, experiences higher sea levels than the annual average. This is due to a combination of build-up of water along the coast caused by southerly winds, increased flooding due to winter storms, and thermal expansion (Komar et al. 2011). For further discussion, see Peck (2017).

The rapid sediment accretion rate observed at WY04S (the lowest-elevation reference core) suggests that tidal wetlands in this area have considerable capacity to equilibrate with accelerated future sea level rise. Kirwan et al. (2016) note that vertical accretion within lower elevations may be the best indication of tidal wetland resistance to drowning under accelerating sea level rise. This is because high marsh equilibrates to a high elevation within the tidal frame, thereby reducing frequent flooding, but low marsh is inundated daily, simulating the stress of higher sea level. If the study sites continue accreting at a high rate under such stress, these wetlands may avoid future drowning. The rapid accretion observed at WY04S and the relatively high accretion rates at the high marsh and scrub-shrub tidal wetland cores suggest that if sediment transport regimes remain intact within the Youngs Bay watersheds, tidal wetlands in the area may be relatively resistant to drowning under future accelerated sea level rise, compared to watersheds and wetlands with lower sediment supply and more disturbed watersheds.

Carbon densities and carbon accumulation rates compared to global averages

Though the reference cores generally had higher organic carbon contents than the restoration site cores (as observed in the organic carbon profiles, Figure 9), the mean carbon densities were similar between restoration and reference cores (0.023 and 0.026 g C/cm^3 respectively); and both were similar to the global mean carbon density of 0.039 g/cm^3 for tidal saline wetlands (Chmura et al. 2003). The mean rates of carbon accumulation for restoration and reference sites (80 and 79 g $C/m^2/yr$) were very similar to the global accumulation rate for tidal marsh of 91 g $C/m^2/yr$ published by IPCC (2014).

Significance

In this study, we found that accretion rates at the reference sites were not only keeping pace with current estimates of sea level rise, but were exceeding it. Thom (1992) had similar results in the Salmon River Estuary; the two cores analyzed showed accretion rates of 3.0 mm/yr under a relative sea level rise of 1.7 mm/yr.

Rapid sediment accumulation at the low marsh just outside the Wallooskee site's dike (WY04S) suggests that wetlands within Youngs Bay may be capable of much faster accretion under the accelerated sea level rise predicted for the future. However, the landscape setting for WY04S is strongly depositional (newly accreted marsh platform outside a dike; and at the confluence of the Wallooskee and Youngs Rivers), so it may not be representative of the broader wetland landscape. Further sampling would help determine the accretion potential at wetlands in different landscape settings.

Diking of the Wallooskee site has caused an estimated loss of 34,000 tons C_{org} or 130,000 tons CO_2 that was formerly sequestered, but was lost to the atmosphere after the site was drained. The mass of carbon lost from the Wallooskee site is greater on a per-area basis (490 tons/ha) than Crooks et al. (2014) measured in the Snohomish Estuary (240 tons/ha). This is potentially due to two factors: 1) a greater difference in height between the restoration site and the reference site (more subsidence), and/or 2) higher carbon density within the high marsh reference site (DP02L), compared to reference sites in Crooks et al. (2014).

After restoration, it is predicted that this relatively large loss in carbon will be reversed, and the restoration site will once more store large amounts of carbon. In fact, restoration of the Wallooskee site offers an opportunity for new accumulation of an equal quantity of carbon to that lost – or even greater quantities, given future accelerated sea level rise. The very high carbon accumulation rate observed at WY04S (210 g $C_{org}/m^2/yr$) suggests that carbon accumulation rates may increase greatly under rapid sea level rise.

Though sea level rise to date is minor in the Pacific Northwest and human influences have been limited in comparison to East and Gulf Coast tidal wetlands, removal of dikes and wetland restoration are important because Oregon estuaries have limited opportunity for landward migration in response to sea level rise (Brophy and Ewald 2017). As sea level rises, healthy tidal wetlands respond by growing both vertically and horizontally (often landward), but on the Oregon coast, the Coast Range prevents much landward migration. Vertical growth (equilibration of the wetland surface with rising sea levels) is therefore critical to wetland survival. Reducing human pressure on wetlands through restoration will improve the chances of their survival and the maintenance of their valued ecosystem services.

Lessons learned

Data collected at the reference sites were vital to this project, as they allowed us to estimate the mass of carbon lost after diking, predict the mass of carbon that could be stored following restoration, improve our understanding of climate change resilience for Youngs Bay tidal wetlands, and compare rates of carbon accumulation at these sites to global values. Though only two cores were obtained from the restoration site, the large difference in values suggests significant heterogeneity within the site. Thus, an increased sample size in future studies is recommended. Additionally, future work should attempt to determine the cause of unexpectedly high accretion rates within the restoration site.

The methods used here allowed us to answer important questions about Youngs Bay least-disturbed and disturbed tidal wetlands. The CT scans allowed us to more easily calculate sediment bulk density and observe sedimentary and biological structures. Though the ^{210}Pb method is not commonly used in tidal wetlands that have been diked, it appeared to produce reliable sediment accretion rates and could therefore be used in future studies.

Tidal hydrology

Methods

Wetland surface elevation

Wetland surface elevation data are used to determine the tidal inundation regime (% inundation). Wetland surface elevation was calculated by averaging all ground surface elevation measurements taken at each wetland monitoring station (Table 2). These measurements were collected at the ground surface using RTK-GPS system with an occupation time of ten seconds. Differences among sites were analyzed using an ANOVA. When distributions did not meet the normality assumptions, an equivalent non-parametric test was used (either a Wilcoxon in place of a t-test or Kruskal-Wallis in place of ANOVA). All analyses were completed in R (Version 3.1.1).

Water levels

Channel water levels were measured at channel monitoring stations (Table 6) using automated water level loggers (Onset HOBO © loggers, model U20-001-01), programmed to collect data at 15 minute intervals. A logger was placed inside the dike at the Wallooskee site (Wallooskee-In), outside the dike at the Wallooskee site (Wallooskee-Out), across the Youngs River at Daggett Point (DP), upriver at Grant Island (GI), and further upriver at Cooperage Slough (CS) to sample along a salinity gradient (Table 1; Appendix 1, Maps 1-4). Water level monitoring began March 24, 2015 and continued through March 27, 2016, obtaining a year of data, at all locations except the two located at the Wallooskee site (both inside and outside of the dike). The loggers at the Wallooskee site were removed on August 21, 2015 due to the beginning of major earthmoving for restoration at that site. At all other locations (DP, GI, and CS), water level monitoring covered one wet season and one dry season during the baseline period.

Water levels were tied to an orthometric reference frame (NAVD88) using a high precision RTK GPS/GNSS system; loggers were checked for vertical movement at each maintenance interval by re-measuring the relationship between the sensor and a local benchmark with a laser level. The average vertical movement between installation and retrieval was 2.3 cm with a standard deviation of 0.96 (within the typical error of the RTK-GPS instruments), indicating negligible shifts in the water level loggers throughout the monitoring period. Raw logger data were converted from pressure values to water levels using HOBOWare Pro© software's barometric compensation assistant, using local barometric pressure data collected onsite at 15 minute intervals throughout the monitoring period. Next, data were pruned to remove reading taken when the top of the logger was submerged by less than 2 cm, or when the water temperature was below freezing (since loggers do not function well at temperatures < 0 ^{o}C).

Tidal datums were calculated for logger installations at the reference sites in R (version 3.1.1) using the "Direct Method" as described in the NOAA Computational Techniques for Tidal Datums Handbook (NOAA 2003). The NOAA Astoria tide gauge (#9439040) in Youngs Bay was the master station. The "Standard Method" could not be applied to the reference logger installations because they did not capture the full tidal range, so we used the "Direct Method" to calculate MHHW and MHW tidal datums at these locations. Tidal datums were not calculated for the logger inside the dike at the Wallooskee site (Wallooskee-In), because the site was still diked at the time of this study.

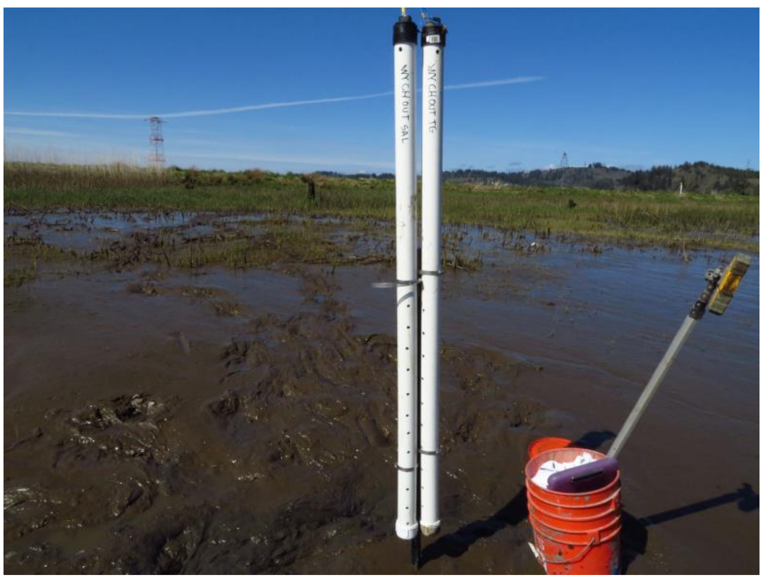

Fig. 10. Channel water level and salinity logger installation outside the dike at the Wallooskee site (Wallooskee-Out). Two stilling wells are placed side-by-side, one housing a level logger and the other housing a salinity logger. Both loggers' sensors are at the same elevation.

Fig. 11. Channel water level and salinity logger installation at Cooperage Slough. Two stilling wells are placed side-by-side, one housing a level logger and the other housing a salinity logger. Both loggers are at the same elevation.

Table 6. Locations of and dates of sampling for water level stations at the Wallooskee site and reference sites. Easting and Northing represent NAD83 UTM Zone 10 N coordinates in meters. Sensor elevations are expressed in meters NAVD88 (Geoid 12A). Plots are those that use that specific logger for percent inundation calculations. Locations are shown in Appendix 1, Maps 1-4. See Appendix 3 for spatial reference system information. Note: Wallooskee-Out was located outside of the dike and is considered a reference site.

Site	Location	Station code	Easting	Northing	Sensor elevation	Wetland monitoring stations that used this sensor for percent inundation calculations	Dates monitored
Wallooskee site	Wallooskee - In	WY In chan	437748	5111445	0.74	WY1, WY3	3/24/15 - 8/21/15
Reference sites	Wallooskee - Out	WY Out chan	437738	5110961	1.64	WY4	3/24/15 - 8/21/15
	Daggett Point	DP chan	436487	5113042	1.32	DP1, DP2, DP3	3/24/15 – 3/27/16
	Grant Island	GI chan	437634	5107819	1.36		3/24/15 – 3/27/16
	Cooperage Slough	CS chan	439441	5105550	1.36	CS2, CS3	3/24/15 – 3/27/16

Daily maximum channel water levels were extracted from the data for the five gauges over the whole monitoring period (March 2015 – March 2016). Daily maximum water levels were averaged across the baseline monitoring period at each gauge for the overlapping dates of 3/24/15 – 8/21/15. Differences in average daily maximum groundwater levels among gauges were analyzed with an ANOVA. A two-way

ANOVA was used to determine differences among gauges and wet and dry seasons (wet season being December 2015 – February 2016, dry season being July 2015 – September 2015). Average percent inundation was calculated for the eight wetland monitoring stations that combined blue carbon sampling with monitoring of groundwater level, groundwater salinity, soils, and vegetation. Average percent inundation was calculated for the available period of overlap for the restoration and reference sites (3/24/15-8/21/15), and for the entire monitoring year for the reference sites (both wet and dry seasons). The gauges used for calculation of percent inundation are found in Table 6. When distributions did not meet the normality assumptions, an equivalent non-parametric test was used (either a Wilcoxon in place of a t-test or Kruskal-Wallis in place of ANOVA). All analyses were completed in R (Version 3.1.1).

Results and discussion

Wetland surface elevation and subsidence

Across all sample plots, the surface elevation at the Wallooskee site was significantly lower than at the reference sites (1.75 m NAVD88 and 2.25 m NAVD88, respectively; $p < 0.0001$; Figure 12). Surface elevations were also significantly different among sites (Table 7, Figure 12), with Wallooskee-In and Wallooskee-Out having similar elevations to each other, but lower surface elevations than Daggett Point and Cooperage Slough. Based on the elevation of the Daggett Point high marsh (DP2), we estimate about 1.26 m of subsidence at the Wallooskee site. This amount of subsidence is comparable to other diked sites in the Pacific Northwest; for example, the Southern Flow Corridor site in the Tillamook Bay estuary averaged about 62 cm of subsidence (Brown et al. 2016), and Waite Ranch in the Siuslaw River estuary has subsided about 1 to 1.5 m (Brophy et al. 2015). Wetland subsidence occurs after the conversion of tidal wetlands to agricultural uses; it is caused by oxidation and compaction of soils following the removal of tidal influence and drainage of soils via diking, ditching, and installation of tide gates (Turner 2004, Frenkel and Morlan 1991).

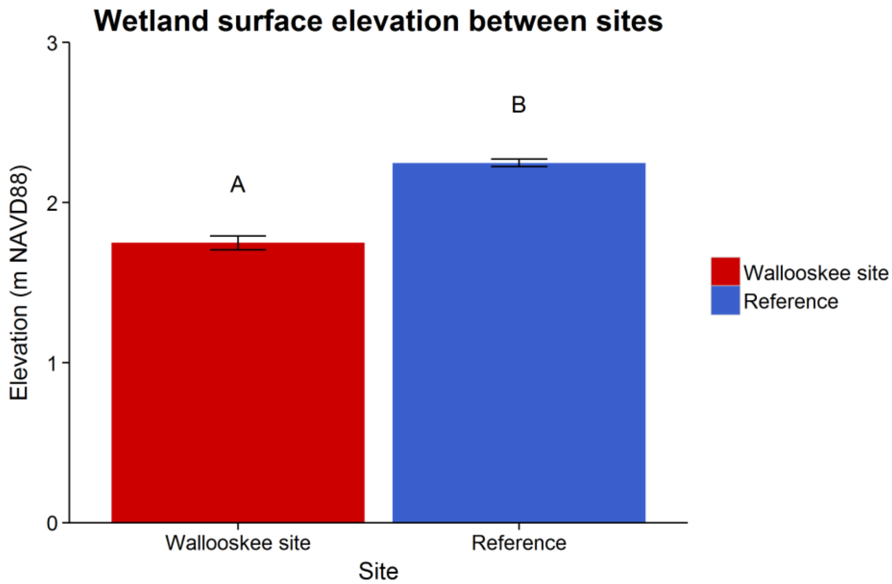

Figure 12. Wetland surface elevation by site for the Wallooskee site and nearby reference sites, averaged across all habitat classes. Error bars show one standard error; columns with no letters in common are significantly different (Wilcoxon test, $p < 0.05$).

Table 7. Average elevation (across habitat classes) and sample size (number of elevation measurements) for locations at the Wallooskee site and reference sites. Location elevations are expressed in meters NAVD88 (Geoid 12A). Note: Wallooskee-Out was located outside of the dike and is considered a reference site.

Site	Location	Elevation (m NAVD88)	Elevation (m MHHW)	Sample size
Wallooskee site	Wallooskee – In	1.75	NA	26
Reference sites	Wallooskee - Out	1.75	-1.07	11
	Daggett Point	2.50	-0.24	58
	Cooperage Slough	2.50	-0.33	31

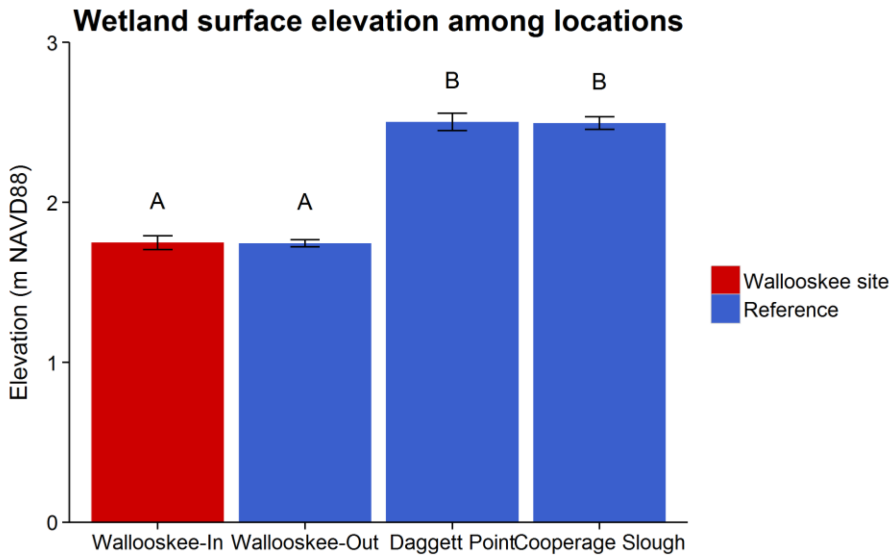

Figure 13. Wetland surface elevation by site for the Wallooskee site and reference sites, averaged across all habitat classes. Error bars show one standard error; columns with no letters in common are significantly different (Kruskal-Wallis, $p < 0.05$). Note that Wallooskee-Out was located outside of the dike and is considered a reference site.

Following restoration of tidal inundation at the Wallooskee site, we expect the tidal wetland surface elevation to increase as sediment is accreted, and eventually to approach the elevation of the Daggett Point reference site. Tidal wetland surface equilibration with sea level is a function of both organic matter accumulation and mineral sediment accretion (Cahoon et al. 2006), and repeated post-restoration monitoring of the wetland surface elevation and sediment accretion at the site will be important to track the site's trajectory towards reference conditions.

Water level

Tidal datums calculated from our water level loggers at reference sites are presented in Figure 14, along with data from the NOAA tide station at Astoria. High water tidal datums (MHHW and MHW) were similar among all sites. We could not calculate MTL, MLW, and MLLW for any of our reference loggers because the installations were located in tidal channels that empty at low tide.

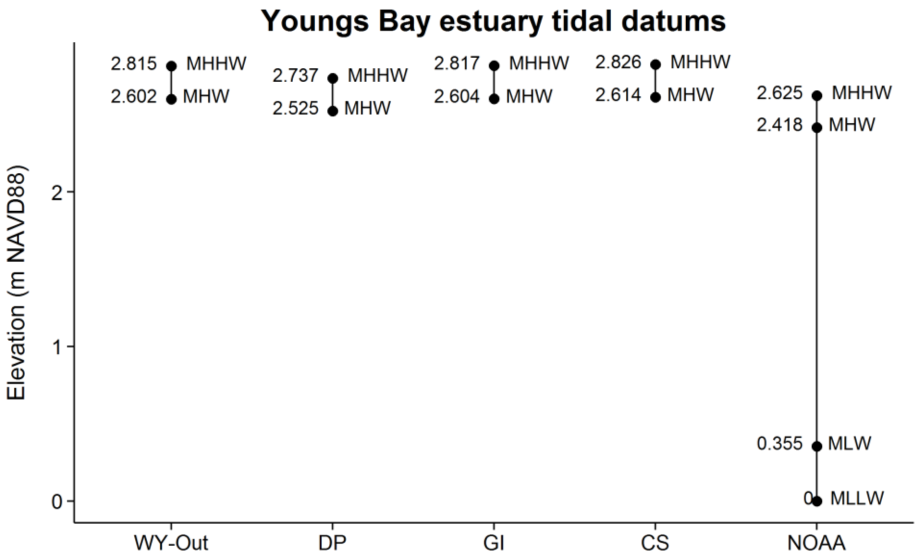

Figure 14. Tidal datums calculated from ETG water level logging data from March 2015 through March 2016 near the Wallooskee site. NOAA tidal datums presented in this figure (far right column) were published for the 1983-2001 tidal epoch at the Astoria tidal station. All elevations are expressed relative to NAVD88 (Geoid 12 A).

Daily maximum channel water levels were significantly lower at the Wallooskee site than the reference sites ($p < 0.001$; Table 8; Figures 15 and 16) for the dates of March 24, 2015 – August 21, 2015. The average daily maximum channel water level was 1.12 m NAVD88 at the Wallooskee site, and 2.92 m NAVD88 at the reference sites. The channel at the Wallooskee site did not dry out during the summer of 2015, and had muted tidal influence throughout the summer, probably due to a leaky tide gate downstream of the logger. Daily maximum channel water level was significantly different among logger locations ($p < 0.001$); WY-In had the lowest daily maximum channel water level, and all of the other logger locations were similar (Table 8, Figures 17 and 18).

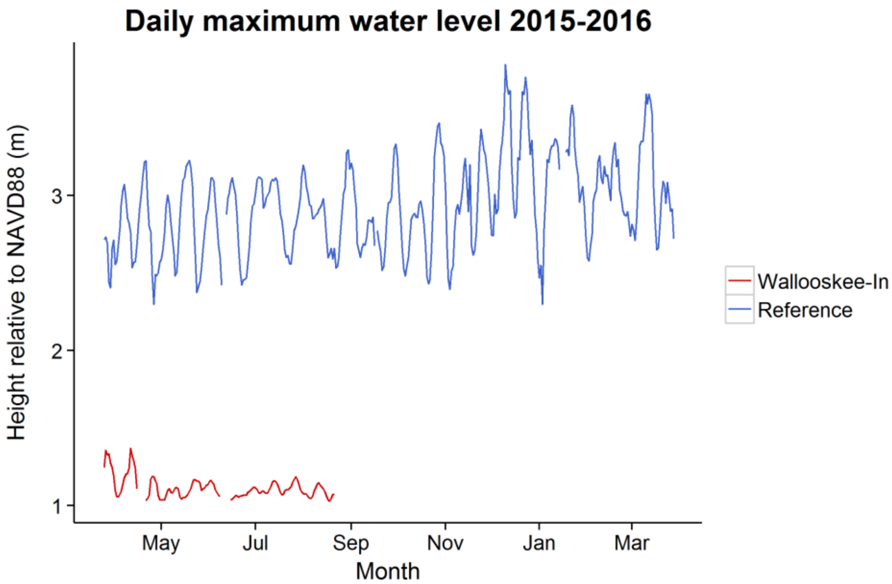

Figure 15. Daily maximum channel water level relative to NAVD88 across the logger stations at the Wallooskee and reference sites March 2015 – March 2016.

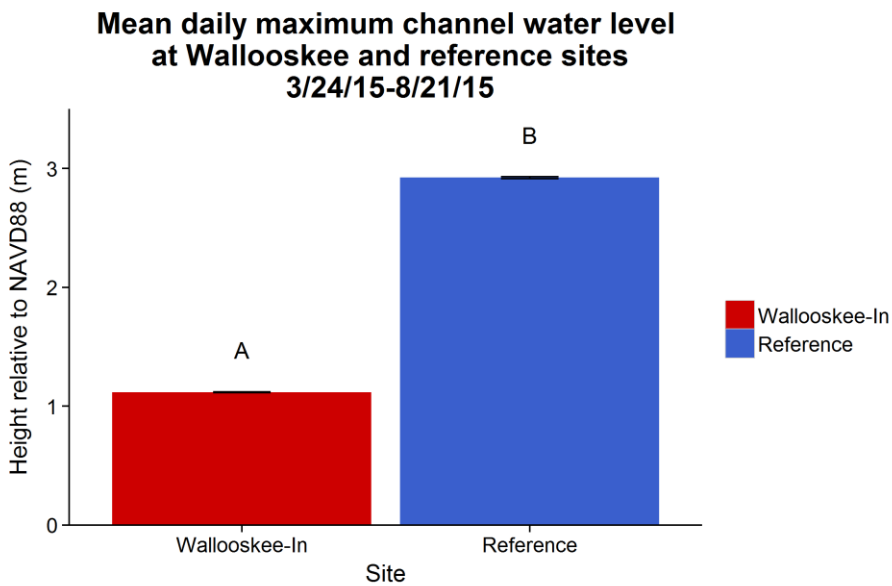

Figure 16. Mean daily maximum channel water levels for the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (Wilcoxon test, $p < 0.05$).

Table 8. Mean and standard error of daily maximum channel water levels at each logger at the Wallooskee site and reference sites, 3/24/15-8/21/15). Note WY-Outside was located outside the dike and is considered a reference site.

Site	Location of logger	Mean daily maximum channel water level in meters relative to NAVD88 (standard error)	Mean daily maximum channel water level in meters relative to MHHW
Wallooskee site	Wallooskee - In	1.12 (0.00)	NA
Reference sites	Wallooskee - Out	2.84 (0.01)	0.03
	Daggett Point	2.89 (0.02)	0.52
	Grant Island	2.96 (0.02)	0.14
	Cooperage Slough	2.96 (0.02)	0.13

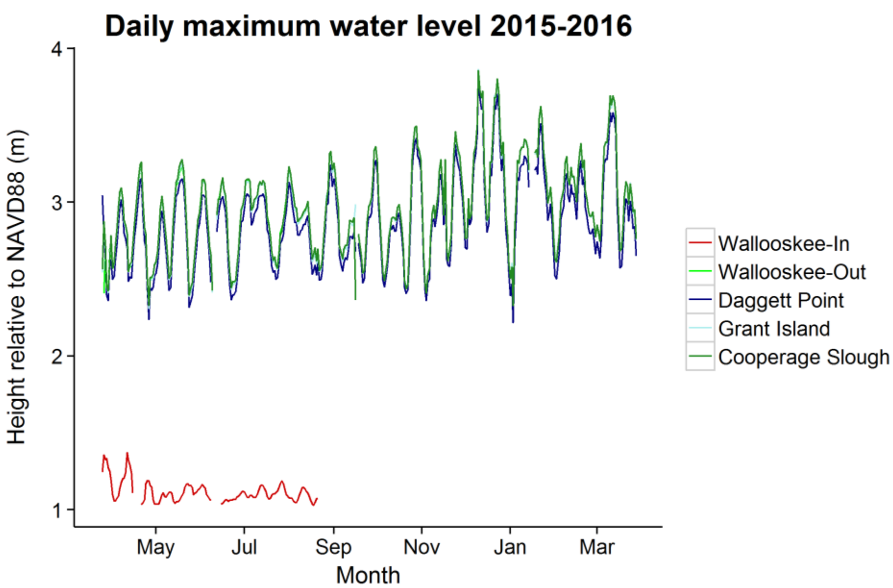

Figure 17. Daily maximum channel water level relative to NAVD88 at each logger station, March 2015 – March 2016.

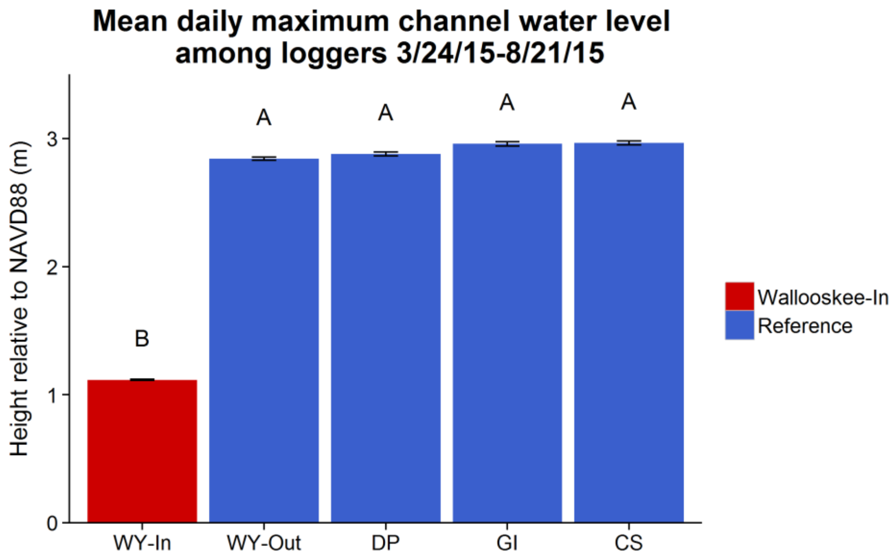

Figure 18. Mean daily maximum channel water levels among loggers at the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Mean daily maximum channel water levels were significantly higher during the wet season compared to the dry season; there was no significant difference among reference loggers within a season, but a consistent trend upwards is visible as you progress upstream from Daggett Point to Cooperage Slough (Tables 9 and 10, Figure 19). Higher winter tides are typical of PNW estuaries with a strong fluvial component to the inundation regime (Brophy et al. 2014, Brown et al. 2016), and of our sites, Cooperage Slough has the highest fluvial influence due to its landscape setting (more confined valley).

Table 9. Summary of two-way ANOVA results for daily maximum channel water level in wet and dry seasons among reference sites. Bold text indicates significant differences ($p < 0.05$).

Factor	**p-value**
Site	0.07
Season	**< 0.001**
Site*Season	0.94

Table 10. Mean and standard error of daily maximum channel water levels at each logger at reference sites, for the dry season (July 2015 – September 2015) and wet season (December 2015 – February 2016).

Site	Location	Season	Mean daily maximum channel water level in meters relative to NAVD88 (standard error)	Mean daily maximum channel water level in meters relative to MHHW
Reference	Daggett Point	dry	2.82 (0.02)	0.01
		wet	3.07 (0.02)	0.26
	Grant Island	dry	2.90 (0.02)	0.08
		wet	3.17 (0.03)	0.35
	Cooperage Slough	dry	2.91 (0.03)	0.08
		wet	3.17 (0.03)	0.34

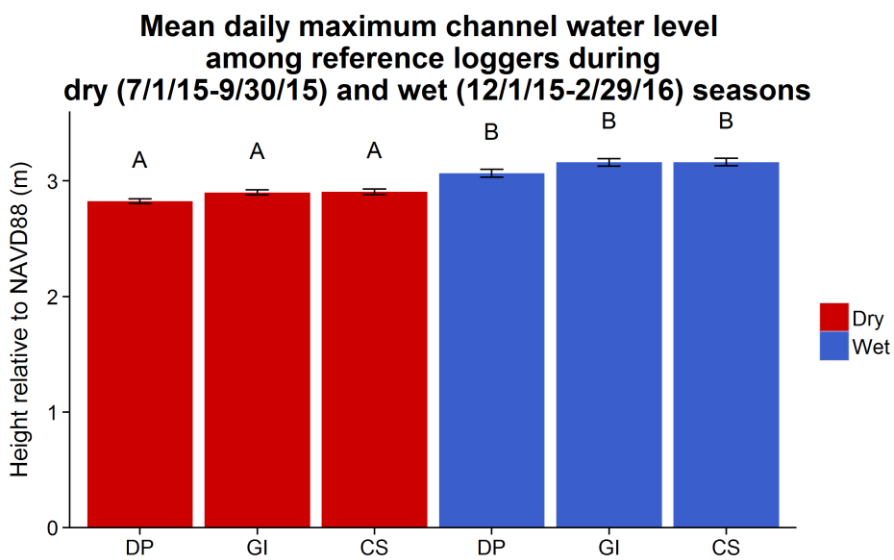

Figure 19. Mean daily maximum groundwater levels among reference stations during the dry (July – September 2015) and wet seasons (December 2015 – February 2016). Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Percent inundation (the percent of time that the ground surface was inundated) was calculated for each wetland monitoring station, habitat, and season. As would be expected, percent inundation decreased with increasing elevation (Figure 20). Percent inundation was also lowest in the diked site (which did not inundate at all during the March-August monitoring period), followed by emergent marsh, and then scrub-shrub tidal swamp (Figure 21). Percent inundation at the reference stations was higher in the wet season (winter), and lower in the dry season (summer) (Figures 22 and 23), an expected pattern for PNW tidal wetlands (Seliskar and Gallagher 1983, Brophy et al. 2011, Brophy et al. 2014, Brown et al. 2016). Seasonal change in inundation could not be determined at the restoration site due to the limited monitoring duration.

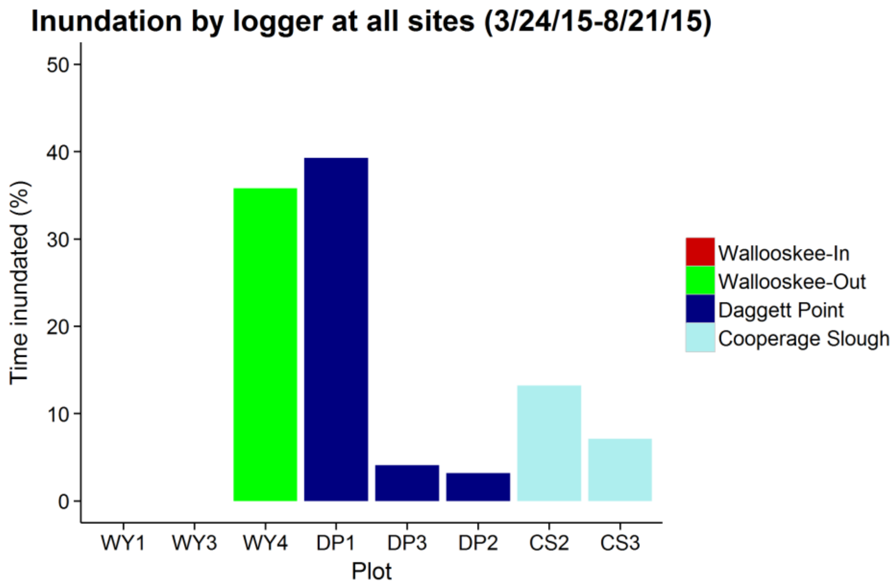

Figure 20. Average percent inundation at each wetland monitoring station at the Wallooskee site and reference sites between March 24, 2015 and August 21, 2015. Stations within a site are ordered by ascending elevation from left to right within each site, with WY1, WY4, DP1, and CS2 having the lowest elevation within their sites, and WY3, WY4, DP2, and CS3 the highest. (Note that WY4 is considered a reference site since it is outside the dike.)

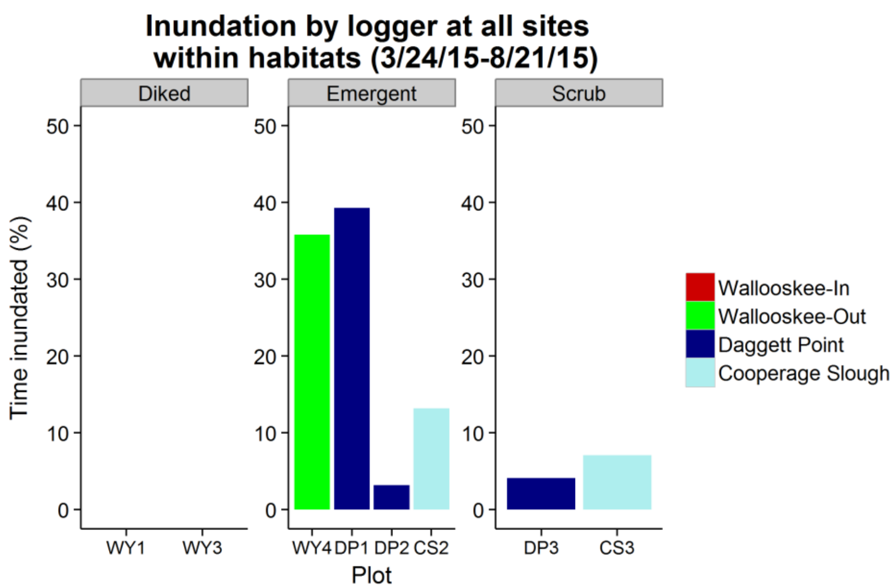

Figure 21. Average percent inundation for wetland monitoring stations at the Wallooskee site and reference sites, by habitat type, between March 24, 2015 and August 21, 2015. Stations within a habitat type are ordered by ascending elevation from left to right within each location, with WY1, WY4, DP3 having the lowest elevation within their habitat type, and WY3, CS2, and CS3 the highest.

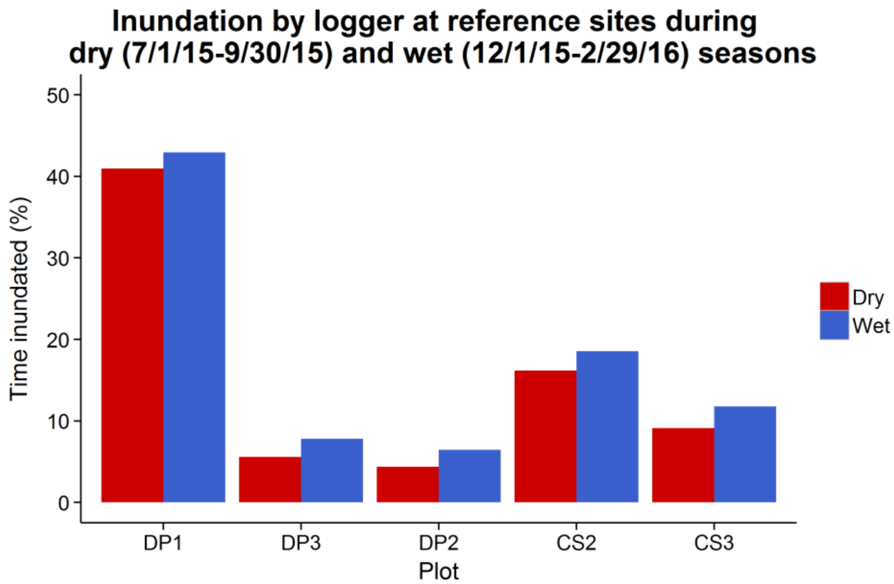

Figure 22. Average percent inundation at each wetland monitoring station at the reference sites during the dry (7/1/15-9/30/15) and wet (12/1/15-2/29/16) seasons. Stations within a site are ordered by ascending elevation from left to right within each location, with DP1 and CS2 having the lowest elevation within their sites, and DP2 and CS3 the highest.

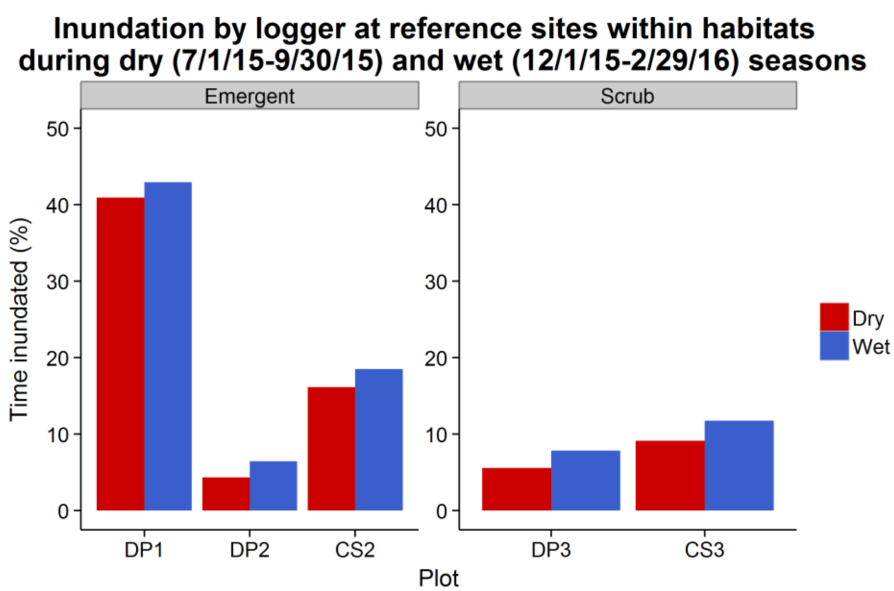

Figure 23. Average percent inundation at each wetland monitoring station at the reference sites in each habitat type during dry (7/1/15-9/30/15) and wet (12/1/15-2/29/15) seasons.

After project implementation, full tidal influence will be restored across the majority of the Wallooskee site, with percent inundations expected to be similar to the reference sites. We have previously documented rapid recovery of tidal hydrology (inundation frequency and depth) with dike removal, although drainage from restoration sites can be delayed on ebb tides compared to reference sites, likely due to channel systems in transition (Brophy et al. 2014).

Channel water salinity and temperature

Methods

Channel water salinity and temperature were measured using Odyssey conductivity-temperature data-loggers programmed to collect data at 15 minute intervals. Salinity/temperature loggers were placed adjacent to water level loggers at each location; loggers were installed in stilling wells (Figures 10 and 11). To sample along the salinity gradient in the Youngs Bay estuary, these dual water level/salinity-temperature logger installations were placed inside the dike at the Wallooskee site (Wallooskee-In), outside the dike at the Wallooskee site (Wallooskee-Out), across the estuary at Daggett Point (DP), upriver at Grant Island (GI), and further upriver at Cooperage Slough (CS) (Table 1; Appendix 1, Maps 1-4). Channel water salinity and temperature monitoring began March 24, 2015 and continued through March 27, 2016, obtaining a year of data, at all locations except the two located at the Wallooskee site (both inside and outside of the dike). Those at the Wallooskee site were pulled August 21, 2015 when channel excavation began at that site.

At each datalogger download, a YSI salinity probe (Model 30) was used to measure salinities in the water column within the salinity stilling well. To determine whether stratification was occurring within the wells, YSI measurements were taken at the top of the water column in the well before removing the logger; at the bottom after removing the logger; and after mixing the water column.

Raw logger data were converted from conductivity values to salinity values using a standard formula (Fofonoff and Millard 1983) plus logger-specific calibration data. Loggers were calibrated before and after each deployment period using a multi-point conductivity and temperature calibration procedure, and the resulting calibration formula was deployment-specific. Salinity logger elevations were the same as the adjacent channel water level loggers (Table 6). Using data from the channel water level loggers, data were “trimmed” to remove data collected when conductivity-temperature loggers were out of the water. Therefore, all data presented in this report were collected when the datalogger was immersed.

Daily maximum channel salinity and temperatures were extracted from the data for the five gauges over the whole monitoring period (March 2015 – March 2016). Daily maximum channel salinity and temperatures were averaged across the baseline monitoring period at each gauge for the overlapping dates of 3/24/15 – 8/21/15. Differences in average daily maximum channel salinity and temperature between sites were analyzed with a t-test. Differences among gauges for that time period were analyzed with an ANOVA. A two-way ANOVA was used to determine differences among reference gauges and wet and dry seasons (wet season being December 2015 – February 2016, dry season being July 2015 – September 2015). When distributions did not meet the normality assumptions, an equivalent non-parametric test was used (either a Wilcoxon in place of a t-test or a Kruskal-Wallis in place of ANOVA). All analyses were completed in R (Version 3.1.1).

Results and discussion

Channel water salinity

Channel water salinity logger data matched well with validation measurements from the YSI probe, indicating the loggers were functioning properly in the stilling wells. Stratification of salinity within the stilling wells was minor; salinity differences from top of well to bottom prior to mixing were usually <1 PSU. However, results might be different in more strongly brackish or euhaline wetlands.

We analyzed average daily maximum salinity, a metric that helps identify biologically important differences among locations and time periods. (Due to strong salinity fluctuation across tide cycles, average salinities have a high level of variability that obscures those biologically important differences.)

Year-round average daily maximum salinity for all sites was in the oligohaline range (around 3 PSU) and did not differ significantly between reference and restoration sites ($p=0.41$; Figures 24 and 25). The salinity observed at the Wallooskee site was probably due to a leaky tide gate. The site showed a muted tidal pattern typical of diked sites where freshwater flows back up behind tide gates at high tide (Figure 15; Appendix 2, Figures A1 and A4), and channel water salinities gradually increased throughout the dry season.

As expected, daily maximum salinities differed significantly among stations ($p < 0.001$), generally decreasing upstream. Cooperage Slough had the lowest salinity (2.36 PSU), while Daggett Point had the highest (3.56 PSU), followed by Grant Island, Wallooskee-In, and then Wallooskee-Out (Table 11, Figures 26 and 27). The relatively low salinity at Wallooskee-Out, despite its relatively downstream setting, was likely due to freshwater influence from the Wallooskee River.

At all sites, salinity increased from May through October (Figure 26), a pattern typically seen in PNW tidal wetlands, where marine influence increases as the dry season progresses, due to reduced precipitation and low river flows (Brophy et al. 2014, Brown et al*.* 2016).

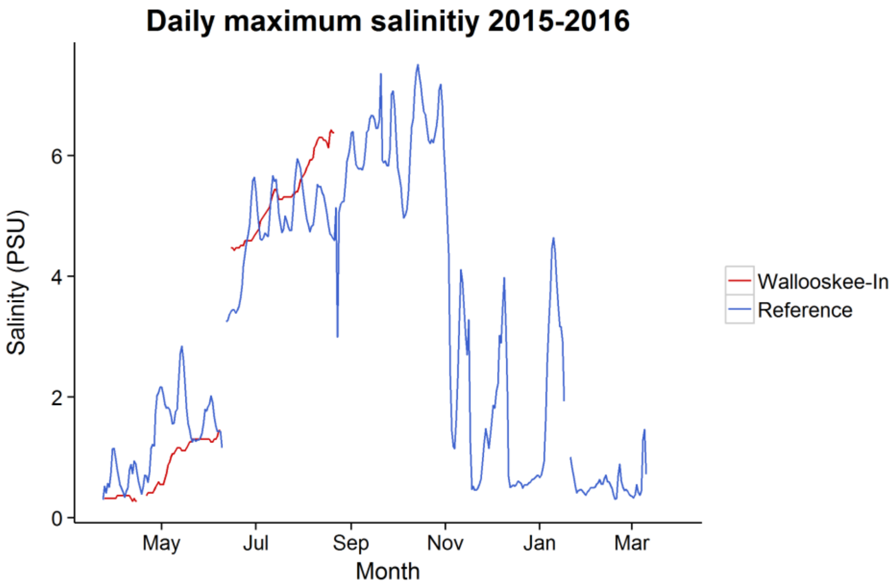

Figure 24. Daily maximum channel salinity for the Wallooskee-In logger, compared to the average daily maximum across all reference stations, March 2015 – March 2016.

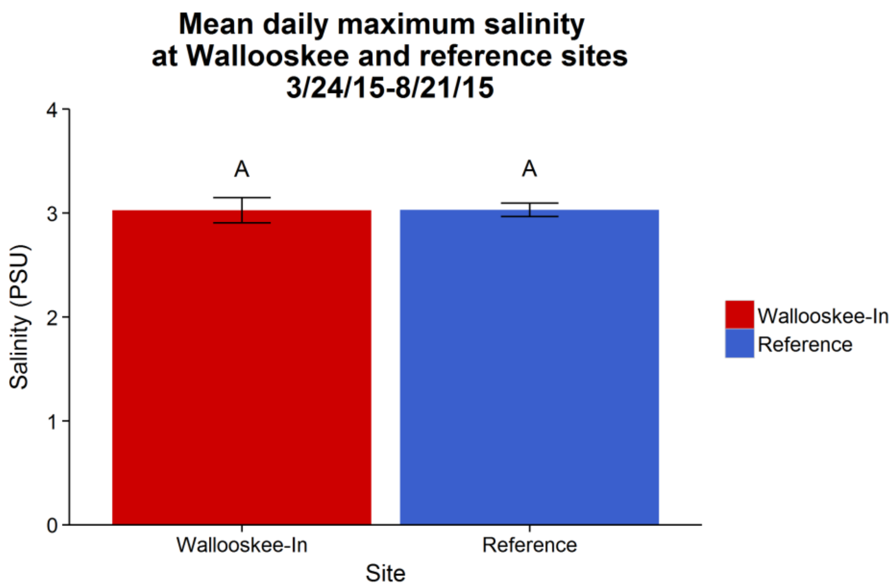

Figure 25. Mean daily maximum salinity for the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (Wilcoxon test, $p < 0.05$).

Table 11. Mean and standard error of daily maximum channel salinity at each logger at the Wallooskee site and reference sites, 3/24/15-8/21/15). Note that WY-Outside was located outside the dike and is considered a reference site.

Site	Location of logger	Mean daily maximum channel salinity in PSU (standard error)
Wallooskee site	Wallooskee - In	3.03 (0.12)
Reference sites	Wallooskee - Out	2.46 (0.07)
	Daggett Point	3.56 (0.15)
	Grant Island	3.42 (0.14)
	Cooperage Slough	2.36 (0.09)

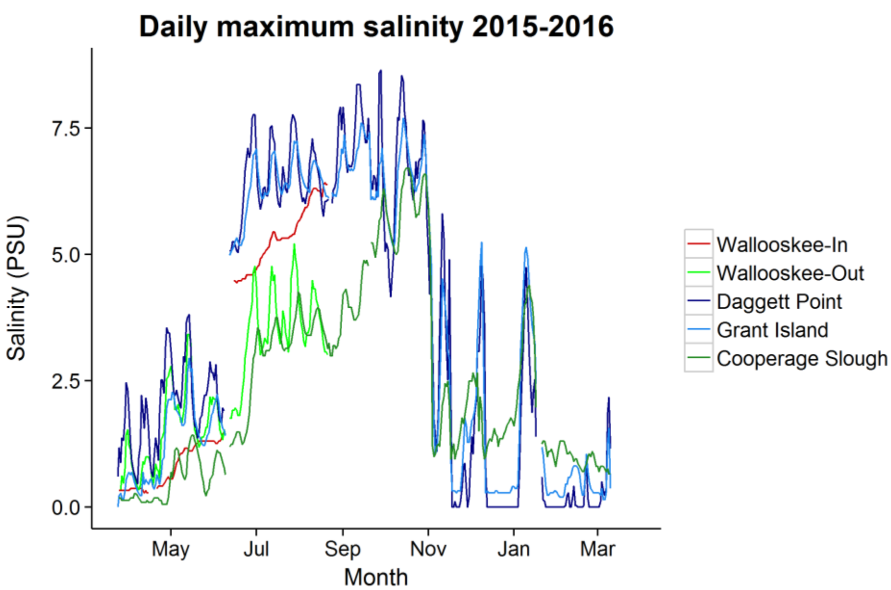

Figure 26. Daily maximum channel salinity at each logger station, March 2015 – March 2016. Note: Wallooskee loggers were removed in August 2015, when onsite restoration began (including channel excavation). Time series discontinuity in late June was due to instrument recalibration coinciding with a period of rapidly increasing salinity.

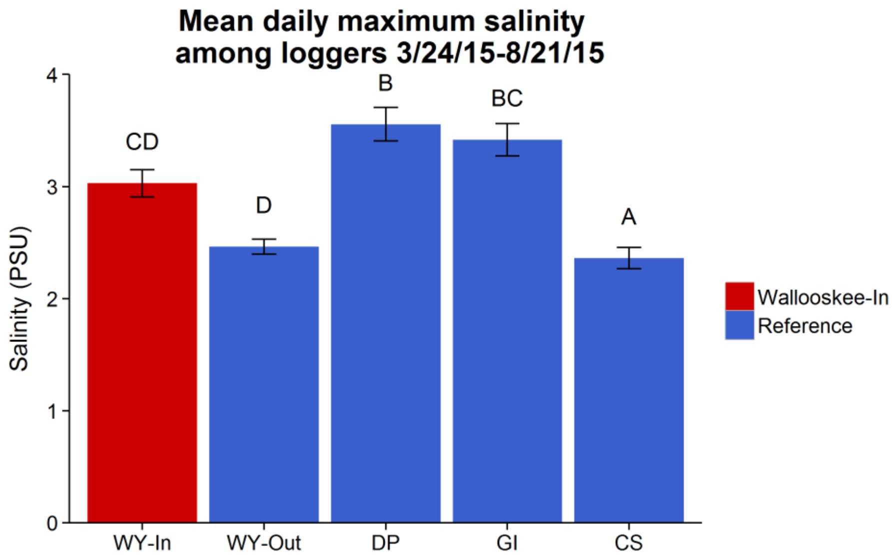

Figure 27. Mean daily maximum channel salinity compared among loggers at the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Since both the Cowardin and CMECS classification systems (Cowardin et al. 1979, FGDC 2012) use summer (low flow) salinities to classify estuarine wetlands, we also analyzed salinities separately for the dry season (July-September 2015) versus the wet season (December 2015-February 2016). As would be expected, daily maximum channel salinities were significantly higher during the dry season compared to the wet season, and there was a significant difference among reference loggers within a season (Tables 12 and 13, Figure 28). During the dry season, salinities were similar at Daggett Point and Grant Island, and significantly lower at Cooperage Slough. Dry season salinities at Daggett Point and Grant Island were in the low mesohaline range; Cooperage Slough was in the mid-oligohaline range. During the wet season, Daggett Point and Grant Island had the lowest salinities, while Cooperage Slough had slightly higher salinities. Even though there were significant differences among the logger stations, salinities during the wet season all averaged below 2 PSU.

Based on dry season salinities (Figure 26), all of the reference sites in this study are classified in the Estuarine System in Cowardin (Cowardin et al. 1979) and in the Estuarine Coastal Subsystem in CMECS (FGDC 2012). Cooperage Slough has previously been classified as a freshwater tidal wetland (LCEP 2007, The Wetlands Conservancy 2008), but the authors did not provide or cite salinity data to support that classification.

Table 12. Summary of two-way ANOVA results for daily maximum channel salinity in wet and dry seasons among reference sites. Bold text indicates significant differences ($p < 0.05$).

Factor	**p-value**
Site	**< 0.001**
Season	**< 0.001**
Site*Season	**< 0.001**

Table 13. Mean and standard error of daily maximum channel salinity at each logger at reference sites, for the dry season (July 2015 – September 2015) and wet season (December 2015 – February 2016).

Site	Location	Season	Mean daily maximum channel salinity in PSU (standard error)
Reference	Daggett Point	Dry	6.82 (0.07)
		Wet	0.80 (0.14)
	Grant Island	Dry	6.64 (0.04)
		Wet	1.23 (0.16)
	Cooperage Slough	Dry	2.82 (0.08)
		Wet	1.67 (0.10)

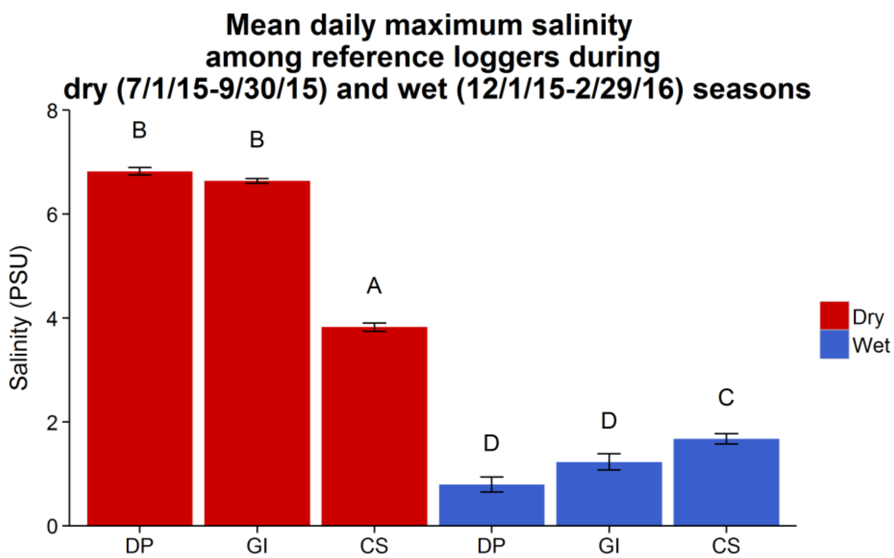

Figure 28. Mean daily maximum salinity among reference stations during the dry (July – September 2015) and wet seasons (December 2015 – February 2016). Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Channel water temperature

There were no significant differences in channel water temperatures between the Wallooskee site and reference sites ($p = 0.09$; Figures 29 and 30). Across the entire period of overlapping record (3/24/2015-8/21/2015), average temperatures at the Wallooskee site was 18.93 °C compared to 19.19 °C at the reference site. When compared among logger stations across this period of overlap, there were no significant differences in temperature ($p = 0.06$; Table 14, Figures 31 and 32). At some other restoration sites, we have observed higher water temperatures at diked sites (e.g. Brown et al. 2016), but the muted tidal influence to the site could explain the lower temperatures at the Wallooskee-In logger. As expected, channel water temperatures at all sites increased throughout the summer, and began to decrease in August (Figure 31).

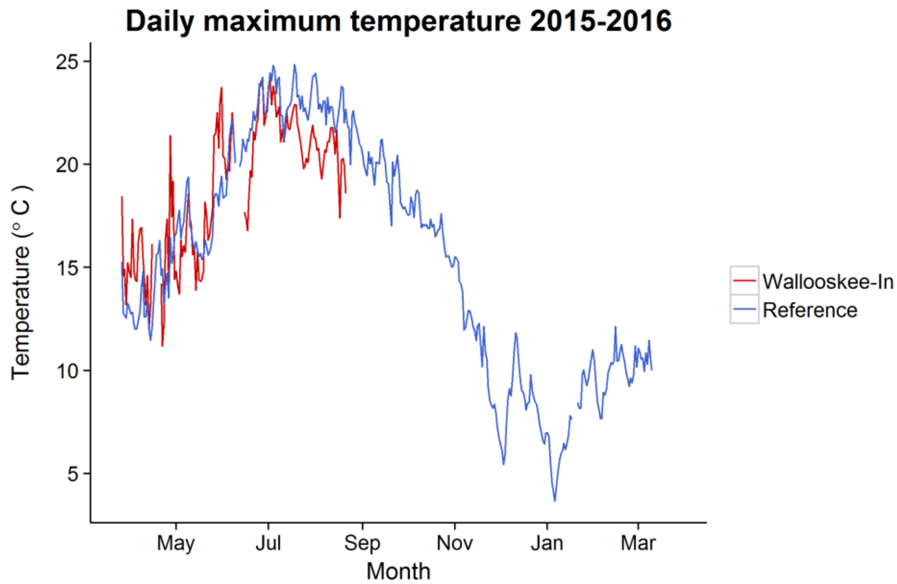

Figure 29. Daily maximum channel water temperature across the logger stations at the Wallooskee and reference sites March 2015 – March 2016.

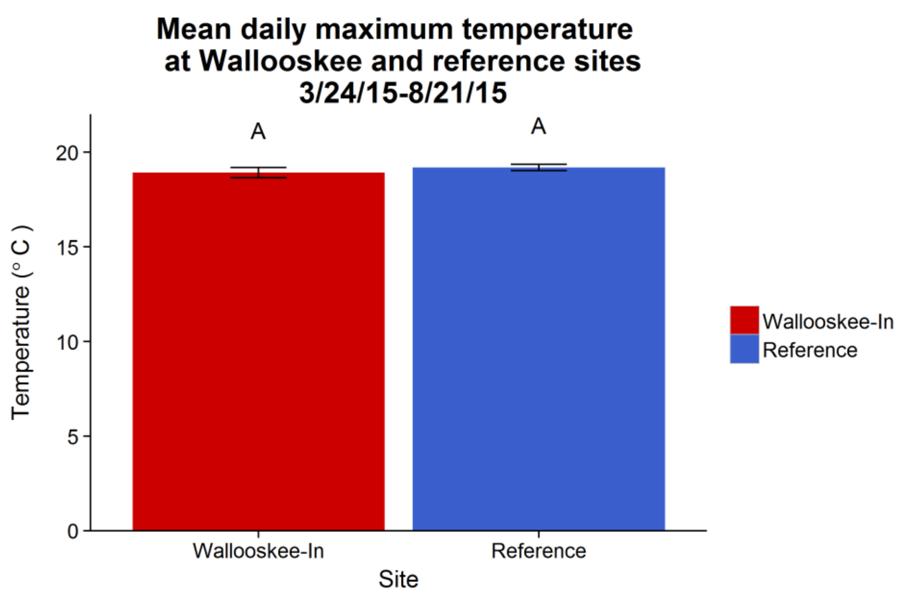

Figure 30. Mean daily maximum channel water temperature for the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (Wilcoxon test, $p < 0.05$).

Table 14. Mean and standard error of daily maximum channel water temperatures at each logger at the Wallooskee site and reference sites, 3/24/15-8/21/15). Note WY-Outside was located outside the dike and is considered a reference site.

Site	Location of logger	Mean daily maximum channel water temperature in °C (standard error)
Wallooskee site	Wallooskee – In	18.93 (0.27)
Reference sites	Wallooskee – Out	18.80 (0.31)
	Daggett Point	19.66 (0.33)
	Grant Island	19.43 (0.34)
	Cooperage Slough	18.89 (0.36)

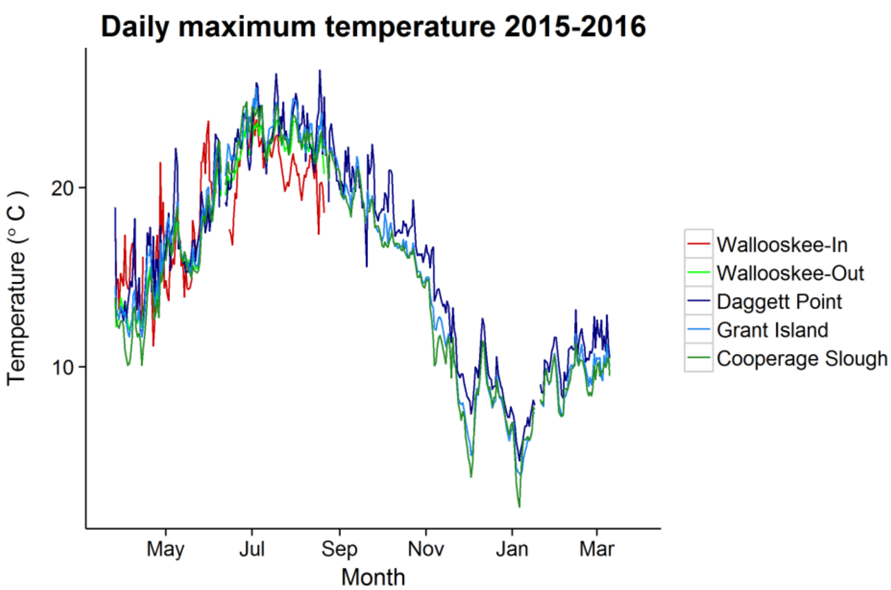

Figure 31. Daily maximum channel water temperature at each logger station, March 2015 – March 2016.

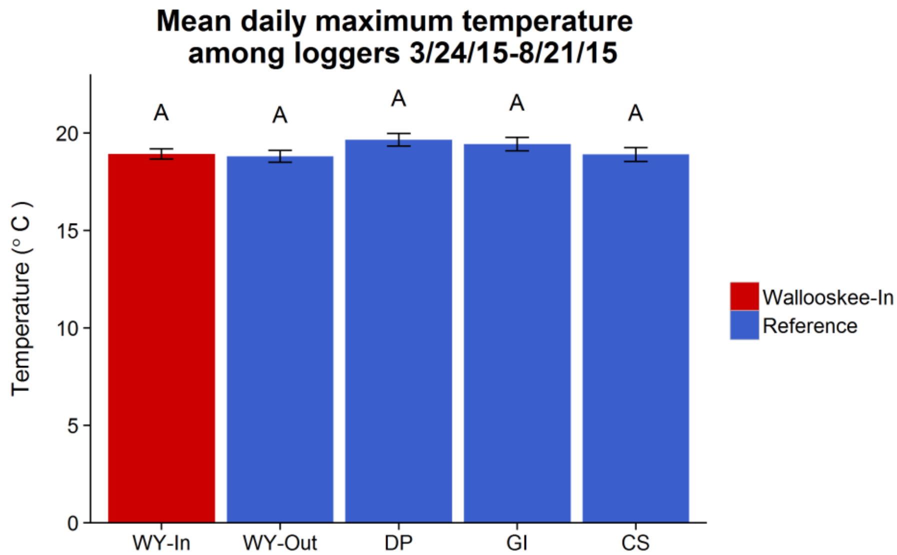

Figure 32. Mean daily maximum channel water temperature among loggers at the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

We also analyzed channel water temperatures separately for the dry season (July-September 2015) versus the wet season (December 2015-February 2016). As would be expected, channel water temperatures were significantly higher during the dry season (summer) compared to the wet season (winter), but were not significantly different among logger stations (Tables 15 and 16, Figure 33).

Table 15. Summary of two-way ANOVA results for daily maximum channel water temperature in wet and dry seasons among reference sites. Bold text indicates significant differences ($p < 0.05$).

Factor	**p-value**
Site	0.12
Season	**< 0.001**
Site*Season	0.14

Table 16. Mean and standard error of daily maximum channel water temperature at each logger at reference sites, for the dry season (July 2015 – September 2015) and wet season (December 2015 – February 2016).

Site	Location	Season	Mean daily maximum channel water temperatures in °C (standard error)
Reference	Daggett Point	dry	22.21 (0.22)
		wet	9.31 (0.20)
	Grant Island	dry	21.85 (0.22)
		wet	8.17 (0.21)
	Cooperage Slough	dry	21.60 (0.20)
		wet	8.03 (0.21)

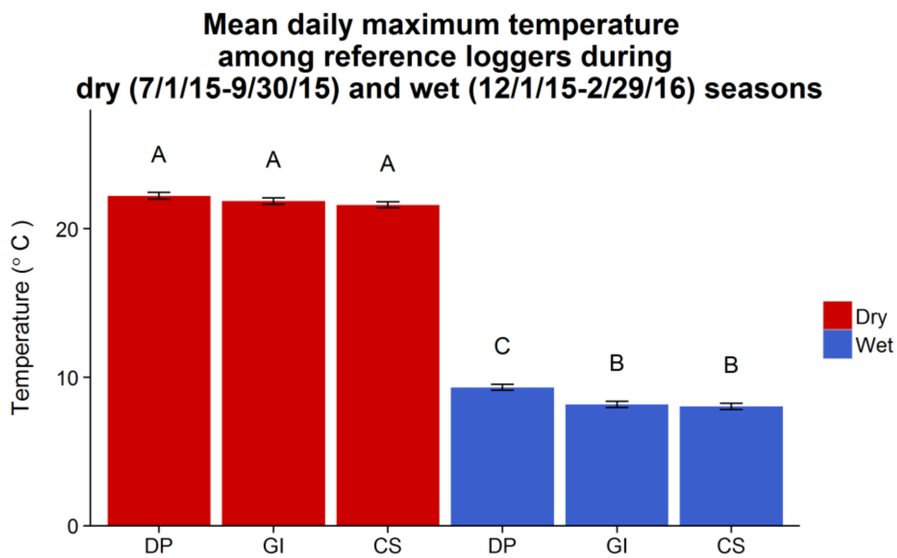

Figure 33. Mean daily maximum channel water temperatures among reference stations during the dry (July – September 2015) and wet seasons (December 2015 – February 2016). Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Groundwater level

Methods

Groundwater monitoring began in March 24, 2015 and went through August 21, 2015 at all stations at the Wallooskee site, including WY4 outside of the dike. Groundwater loggers at the Wallooskee site were pulled in August 2015 due to the beginning of major restoration activities at the site. Monitoring for all reference site groundwater level stations began March 24, 2015 and ended March 27, 2016, obtaining a year of data, except for the two stations with the lowest elevations (DP1 and WY4). During the winter wet season, groundwater was continuously at the soil surface at these two stations.

Therefore, installation of groundwater level and groundwater salinity monitoring equipment was delayed until June 2015, when groundwater levels were expected to begin to drop below the soil surface (Table 17). The groundwater level logger at CS3 was stolen sometime after the September 2015 re-deployment, therefore CS3 had a sampling period of March 24, 2015 – September 16, 2015 (Table 17).

All eight groundwater level wells were located at the wetland monitoring stations, adjacent to the blue carbon cores, groundwater salinity wells, vegetation plots, and soil samples (Appendix 1, Maps 1-4). At the restoration site, orange construction fencing was used to protect the area from disturbance during early restoration activities in spring 2015 (Figures 34 and 35).

Table 17. Locations of and dates of sampling for groundwater stations at the Wallooskee site and reference sites. Easting and Northing represent UTM Zone 10 N coordinates in meters (NAD83 datum). Locations are shown in Appendix 1, Maps A1-A4. See Appendix 3 for spatial reference system information. Note: WY4 was located outside of the dike and is considered a reference site.

Site	Location	Station code	Easting	Northing	Dates monitored
Wallooskee site	Inside dike	WY1	437416	5111465	3/24/15-8/21/15
		WY3	437639	5111154	3/24/15-8/21/15
Reference sites	Outside dike	WY4	437719	5110996	6/8/15-8/21/15
	Daggett Point	DP1	436334	5113121	6/8/15-3/27/16
		DP2	436302	5113000	3/24/15-3/27/16
		DP3	436218	5113003	3/24/15-3/27/16
	Cooperage Slough	CS2	439455	5105495	3/24/15-3/27/16
		CS3	439409	5106095	3/24/15-9/16/15

Groundwater levels were monitored using standard shallow groundwater observation wells (Sprecher 2000). Wells were approximately 1.5 m deep, therefore groundwater levels more than 1.5 m below the soil surface could not be tracked. Groundwater levels more than 1.5m below the soil surface did not occur at the reference sites during the observation period, but did occur in summer inside the Wallooskee site (stations WY1 and WY3). Groundwater levels were monitored using automated water level loggers (Onset HOBO © loggers, model U20-001-01), which were programmed to collect pressure data at 15 minute intervals. Raw logger data were converted from pressure values to water levels using HOBOWare Pro © software's barometric compensation assistant, which adjusts pressure values to water levels using local barometric pressure data collected onsite at 15 minute intervals throughout the monitoring period. Data were trimmed to remove records when the water depth was less than 2 cm above the top of the logger.

Groundwater levels were tied to both the orthometric reference frame (NAVD88) and the soil surface (Table 18); analyses were conducted on data expressed relative to the soil surface, since that is the biologically meaningful metric. Groundwater levels below the soil surface were expressed as negative numbers, and values that were positive (above the soil surface) were changed to "0", as they represented wetland surface inundation and therefore a groundwater level at the soil surface. Groundwater levels relative to NAVD88 were used for comparison with tidal hydrology.

Figure 34. Groundwater level and groundwater salinity wells inside protective construction fencing at the Wallooskee site, wetland monitoring station WY3, spring 2015.

Figure 35. Closeup view of groundwater level and groundwater salinity logger installations at wetland monitoring station WY3, Wallooskee site.

Table 18. Soil surface elevations (m) and habitat description of groundwater stations at the Wallooskee site and reference sites. The GEOID12A model was used to compute NAVD88 orthometric elevation. Note: Wallooskee site -- outside was located outside of the dike and is considered a reference site.

Site	Location	Station	Soil surface elevation (m NAVD88)	Habitat
Wallooskee site	Inside dike	WY1	1.46	diked
		WY3	1.89	diked
Reference sites	Outside dike	WY4	1.73	emergent
	Daggett Point	DP1	1.81	emergent
		DP2	2.82	emergent
		DP3	2.77	shrub
	Cooperage Slough*	CS2	2.52	emergent
		CS3	2.71	shrub

Daily maximum groundwater levels were extracted from the data for the eight groundwater wells over the whole monitoring period (March 2015 – March 2016). Daily maximum groundwater levels were averaged across the baseline monitoring period at the Wallooskee site and reference sites, as well as for individual habitats (diked, emergent, and shrub), and individual wells for the overlapping dates of 3/24/15 – 8/21/15. Differences in average daily maximum groundwater levels between the Wallooskee site and reference sites were analyzed using a t-test. Differences among habitats and among wells for all overlapping dates were tested using an ANOVA. A two-way ANOVA was used to determine differences between wells and between wet and dry seasons (wet season being December 2015 – February 2016, dry season being July 2015 – September 2015). A simple linear regression was used to relate the average daily maximum groundwater level at each station to soil surface elevation at the reference sites. The Wallooskee site only had a total of two stations, so this regression analysis was not run for the Wallooskee site. When distributions did not meet the normality assumptions, an equivalent non-parametric test was used (either a Wilcoxon in place of a t-test or Kruskal-Wallis in place of ANOVA). All analyses were completed in R (Version 3.1.1).

Results and discussion

Average daily maximum groundwater levels were significantly lower at the Wallooskee site than reference sites ($p < 0.001$; Figures 36 and 37). The average daily maximum groundwater level was 0.49 m below the soil surface at the Wallooskee site and 0.03 m below soil surface at the reference sites (Figures 36 and 37). The summer drying period began in May 2015 at the Wallooskee site and continued at least through August 2015 – typical of seasonal, non-tidal wetlands in the Pacific Northwest (Brophy et al. 2014, Brown et al. 2016). This summer pattern did not occur at the reference sites (Figure 36); instead, the high marsh and scrub-shrub tidal wetlands showed the typical "spring tide reset" pattern we have observed at other sites (Brophy 2009, Brophy et al. 2014), in which groundwater rises to the surface during spring tides, then drops gradually during neap tide cycles (Appendix 2, Figures A6 and A8). Average daily maximum groundwater levels were 0.49 m below soil surface during the monitoring period. Daily maximum groundwater levels at the reference sites were within 20 cm of the soil surface year-round (Figure 36).

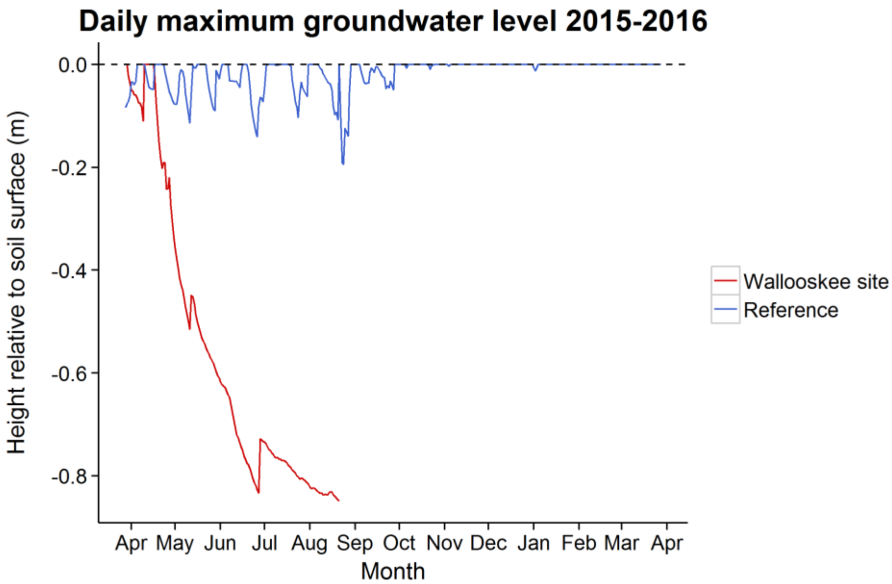

Figure 36. Daily maximum groundwater level relative to soil surface across all stations at the Wallooskee site and reference sites March 2015 – March 2016. Dashed line indicates the soil surface.

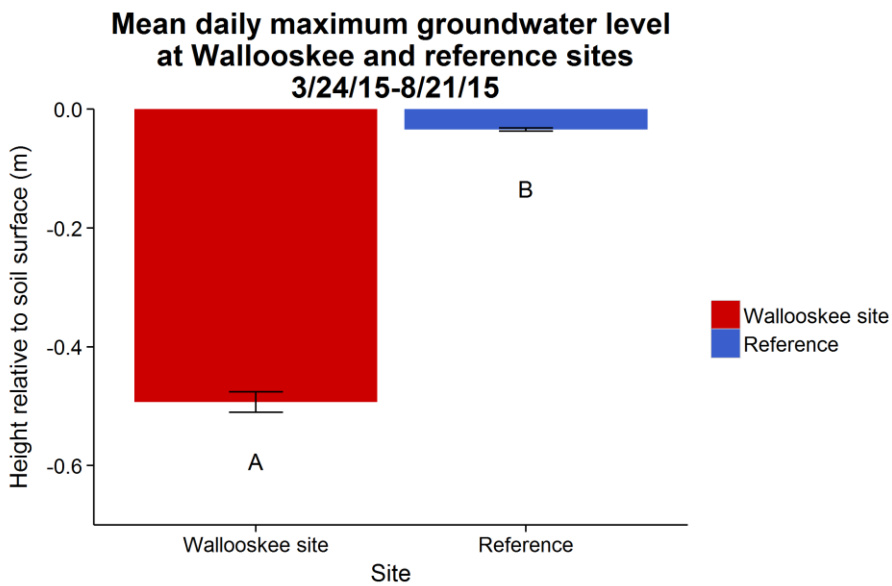

Figure 37. Mean daily maximum groundwater levels for the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (Wilcoxon test, $p < 0.05$). *Note that the bottom of the colored bar indicates the mean, since water levels are below the soil surface*.

Analysis of groundwater level by station showed significant differences among stations ($p < 0.001$). Mean daily groundwater levels were significantly lower at WY3 compared to WY1 (0.56 m and 0.39 m below soil surface, respectively), and both were significantly lower than any mean daily maximum groundwater level at the reference stations (Figure 39). Mean daily groundwater levels were at the soil surface at both WY4 and DP1 (Table 19, Figures 38 and39). CS3 had the lowest mean daily maximum groundwater level of all the reference stations (0.09 m below soil surface), likely due to CS3 having one of the higher elevations of the reference stations (Table 19, Figures 38 and 39). Mean daily maximum groundwater level at the reference stations was not significantly correlated to surface elevation ($p = 0.17$, $R^2 = 0.27$; Figure 40) when analyzed across all habitat types; but analysis of variance by habitat class showed that scrub-shrub tidal swamps had a slightly lower water table (Figure 41) compared to emergent marsh -- again, likely because of their slightly higher elevation. The regression of groundwater levels on elevation was not run for the Wallooskee site, which contained only two stations.

Table 19. Mean daily maximum groundwater levels at each station at the Wallooskee site and reference sites, 3/24/15-8/21/15. Negative numbers indicate groundwater levels below the soil surface.

Site	Location	Station	Mean daily maximum groundwater level in meters relative to soil surface (standard error)
Wallooskee site	Inside dike	WY1	-0.39 (0.03)
		WY3	-0.56 (0.02)
Reference sites	Outside dike	WY4	0.00 (0.00)
	Daggett Point	DP1	0.00 (0.00)
		DP2	-0.04 (0.00)
		DP3	-0.04 (0.03)
	Cooperage Slough	CS2	-0.01 (0.00)
		CS3	-0.09 (0.01)

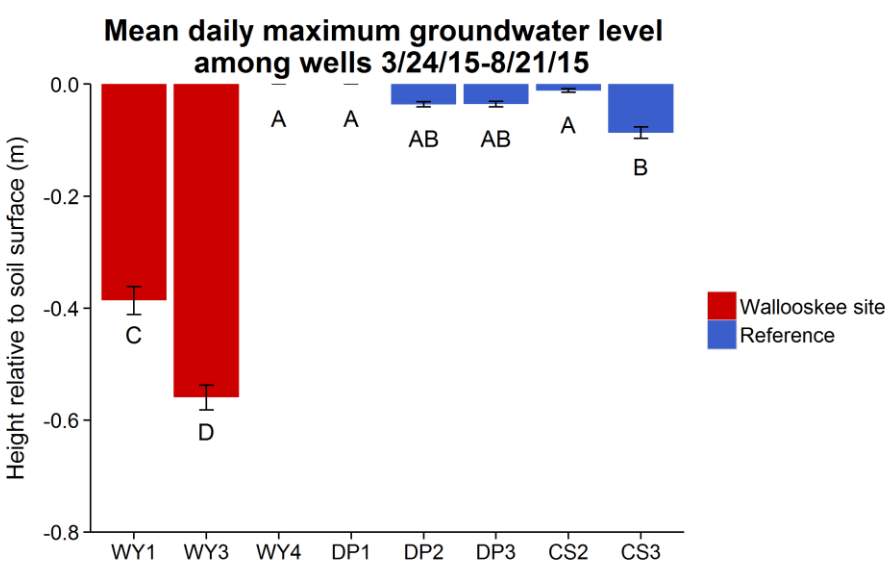

Figure 38. Mean daily maximum groundwater levels among stations at the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

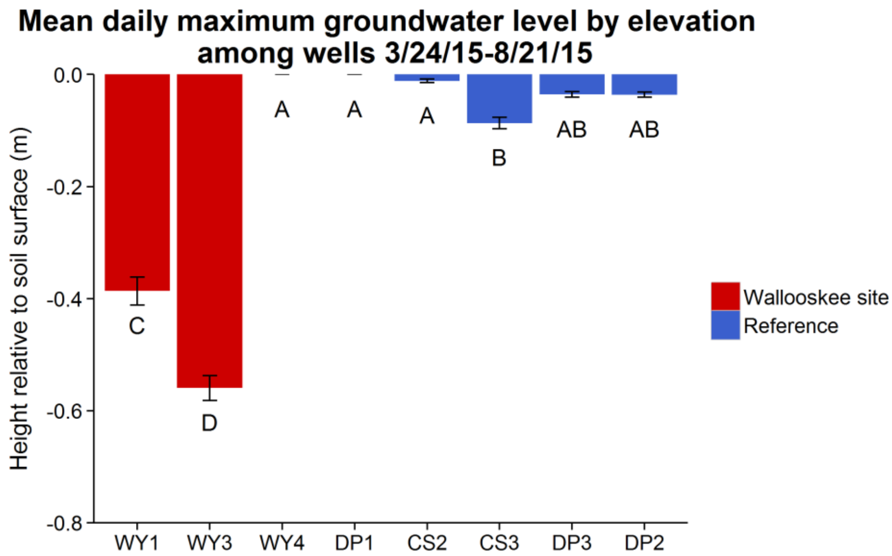

Figure 39. Mean daily maximum groundwater level among stations at the Wallooskee site and reference sites. Wells are ordered by ascending elevation from left of right, separated by inside the dike (Wallooskee site) and outside the dike (reference), with WY1 and WY4 having the lowest elevation, and WY3 and DP2 the highest.

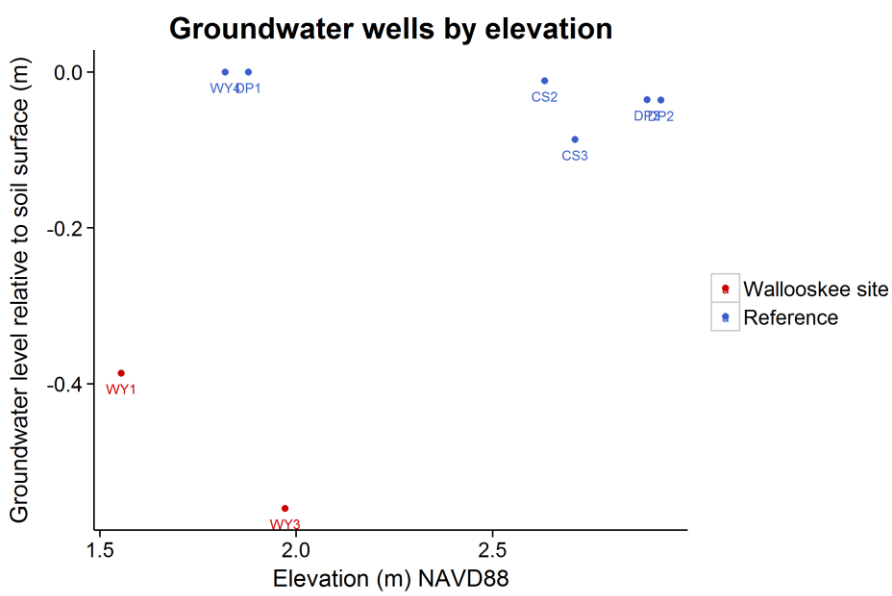

Figure 40. Mean daily maximum groundwater level at each station at the Wallooskee site and reference sites (3/24/2015-8/21/2015) along an elevation gradient.

Mean daily maximum groundwater levels significantly differed among habitat type ($p < 0.001$; Figure 41), with the diked marsh having the lowest daily maximum groundwater level (0.50 m below soil surface), followed by the two scrub-shrub tidal swamps (DP3 and CS3) (0.06 m below soil surface), and emergent marsh having the highest daily maximum groundwater level (0.02 m below soil surface). This is expected; despite the diked marsh having the lowest average elevation (1.76 m NAVD88), tide gates and flow barriers have greatly reduced tidal influence and the site dries out during the summer months (which were the months that were monitored). The shrub habitats had an average elevation of 2.80 m NAVD88, which was higher than the emergent marsh's average elevation of 2.32 m NAVD88, explaining the lower groundwater levels found in the shrub habitat. The relationship between surface elevation and groundwater level is seen in other projects along the Pacific Northwest coastline (Brophy et al. 2014, Brown et al. 2016).

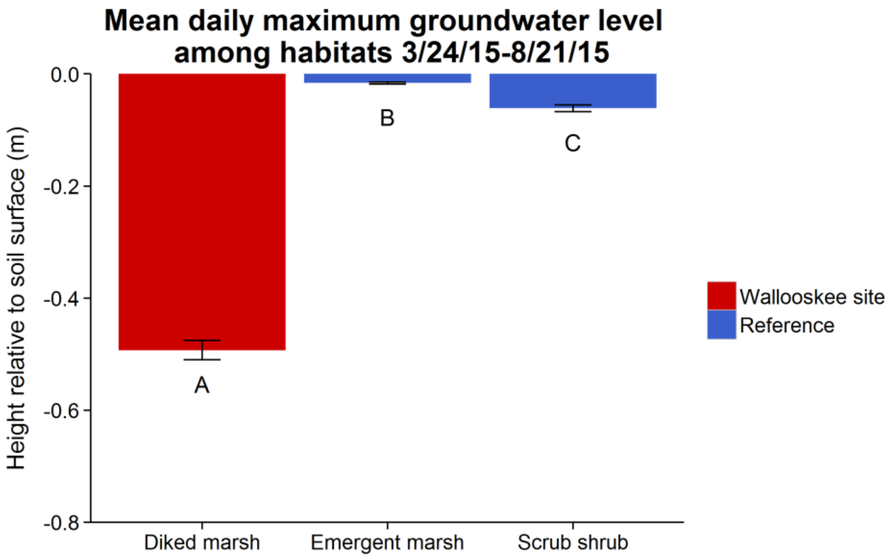

Figure 41. Mean daily maximum groundwater level among habitats at the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Mean daily maximum groundwater levels were significantly different among reference stations during March 24, 2015 – September 16, 2015 (the full period of overlap for all five reference stations due to a stolen logger) ($p < 0.001$). Groundwater levels were closest to the soil surface at DP1 and CS2, followed by DP2 and DP3, and groundwater levels were the lowest at CS3 (Table 20, Figure 42). These results are similar to those found for the date range of March 24, 2015 – August 21, 2015. Among habitats at the reference sites, with dates extending through September 16, 2015, scrub-shrub wetlands had significantly lower groundwater levels than emergent marsh ($p < 0.001$), with groundwater levels of 0.06 m and 0.02 m below soil surface for scrub shrub and emergent marsh, respectively (Figure 43).

Table 20. Mean daily maximum groundwater levels at each station at reference sites, 3/24/15-9/16/15. Negative numbers indicate groundwater levels below the soil surface.

Site	Location	Station	Mean daily maximum groundwater level in meters relative to soil surface (standard error)
Reference	Daggett Point	DP1	0.00 (0.00)
		DP2	-0.04 (0.01)
		DP3	-0.04 (0.01)
	Cooperage Slough	CS2	-0.01 (0.00)
		CS3	-0.08 (0.01)

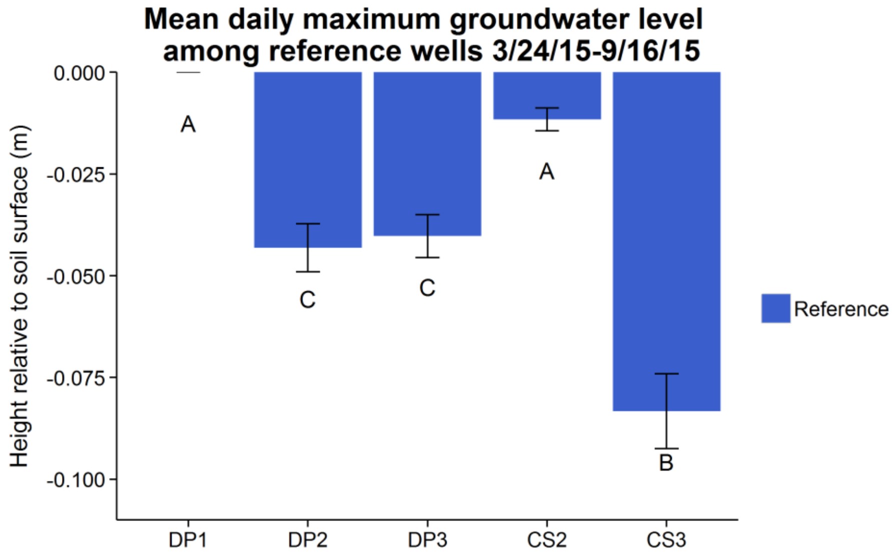

Figure 42. Mean daily maximum groundwater levels among stations at the reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

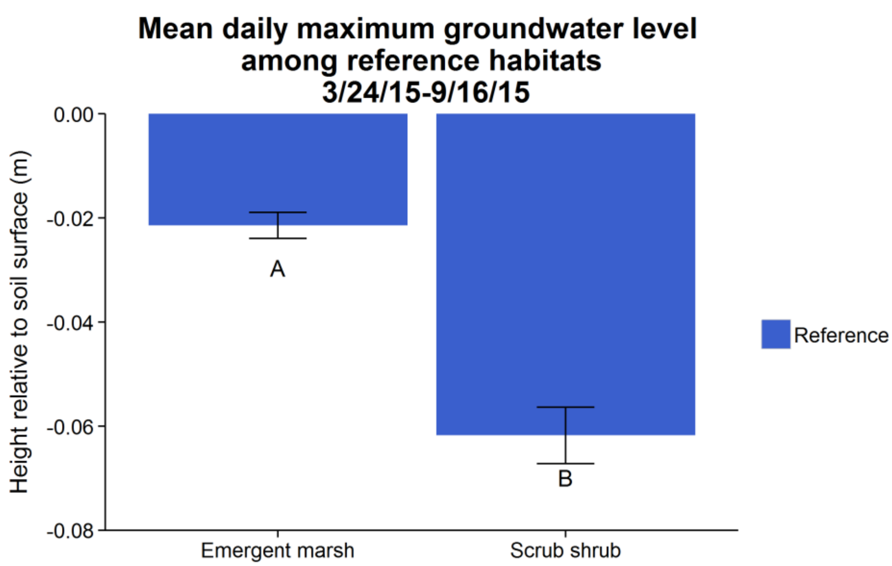

Figure 43. Mean daily maximum groundwater level among habitats at the reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Mean daily maximum groundwater levels among reference stations between the dry (7/1/15-9/30/15) and wet (12/1/15-2/29/16) seasons were significantly different ($p < 0.001$), though only DP2 and DP3 in the dry season were different when compared to all other stations and seasons (Table 21, Figure 44). DP2 and DP3 had significantly lower groundwater levels during the dry season compared to DP1 and CS2, and compared to all stations during the wet season.

Table 21. Summary of two-way ANOVA results for daily maximum groundwater level in wet and dry seasons among reference sites. Bold text indicates significant differences ($p < 0.05$).

Factor	p-value
Site	**< 0.001**
Season	**< 0.001**
Site*Season	**< 0.001**

Table 22. Mean and standard error of daily maximum groundwater levels at each station at reference sites, for the dry season (July 2015 – September 2015) and wet season (December 2015 – February 2016). Negative numbers indicate groundwater levels below the soil surface.

Site	Location	Station	Season	Mean daily maximum groundwater level in meters relative to soil surface (standard error)
Reference	Daggett Point	DP1	dry	0.00 (0.00)
			wet	0.00 (0.00)
		DP2	dry	-0.06 (0.01)
			wet	0.00 (0.00)
		DP3	dry	-0.06 (0.01)
			wet	0.00 (0.00)
	Cooperage Slough	CS2	dry	-0.01 (0.00)
			wet	-0.00 (0.00)

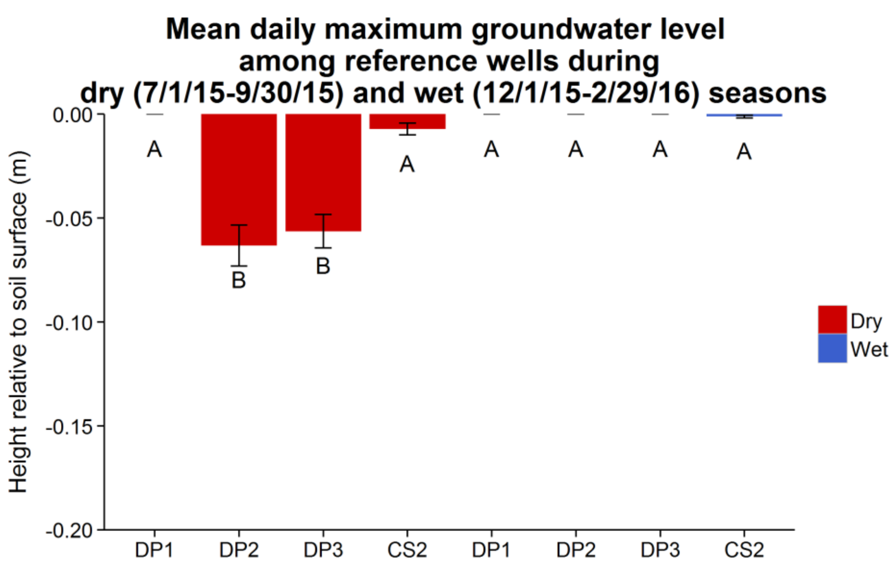

Figure 44. Mean daily maximum groundwater levels among reference stations during the dry (July – September 2015) and wet seasons (December 2015 – February 2016). Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Few other projects have monitored groundwater levels at tidal wetland restoration sites, but one study in Oregon showed that once tidal influence was restored to the restoration site, groundwater regimes began a trajectory towards those at reference conditions (Brophy et al. 2014). We expect this trajectory to be very gradual. Despite the rapid return of full tidal influence, three years after restoration at the Ni-lest'tun Unit of the Bandon Marsh National Wildlife Refuge, tidal groundwater patterns still differed from those found at reference sites (Brophy et al. 2014). Groundwater regimes may be affected by compaction, subsidence of soil, and tidal channel development, each of which are factors that may change slowly after the restoration of a site and re-establishment of tidal influence (Brophy and van de Wetering 2012, Brophy et al. 2014).

Figures 45 and 46 illustrate the dynamic groundwater patterns in summer at the reference sites; daily time series graphs show the even more dynamic daily patterns (Appendix 2, Figures A6, A8, A10). Such fluctuations in groundwater are associated with high levels of soil biotic activity and very productive ecosystems (Mitsch and Gosselink 1993). The reference site groundwater regimes contrast sharply with the much less dynamic water table at the diked Wallooskee site (Figure 36; Appendix 2, Figure A5).

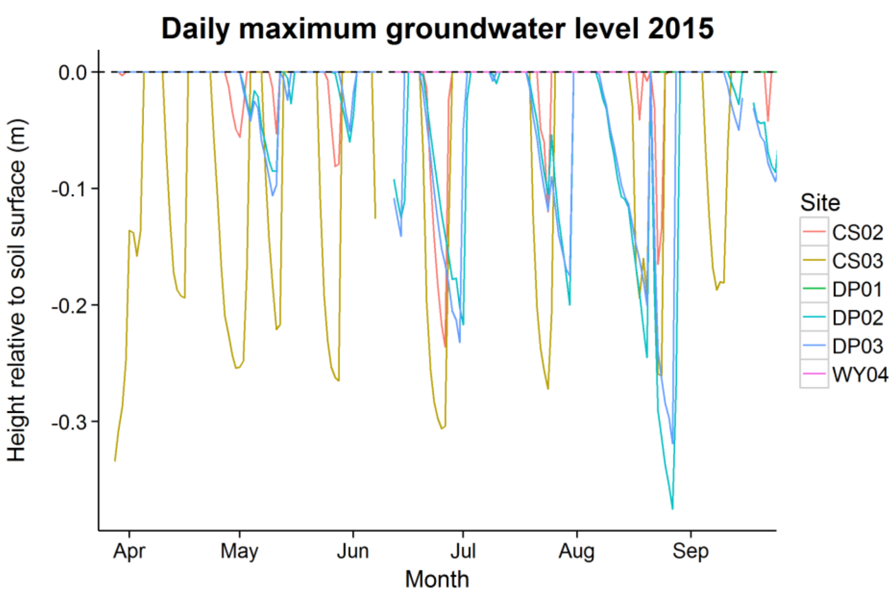

Figure 45. Daily maximum groundwater level across all stations at reference sites April 2015 – September 2015.

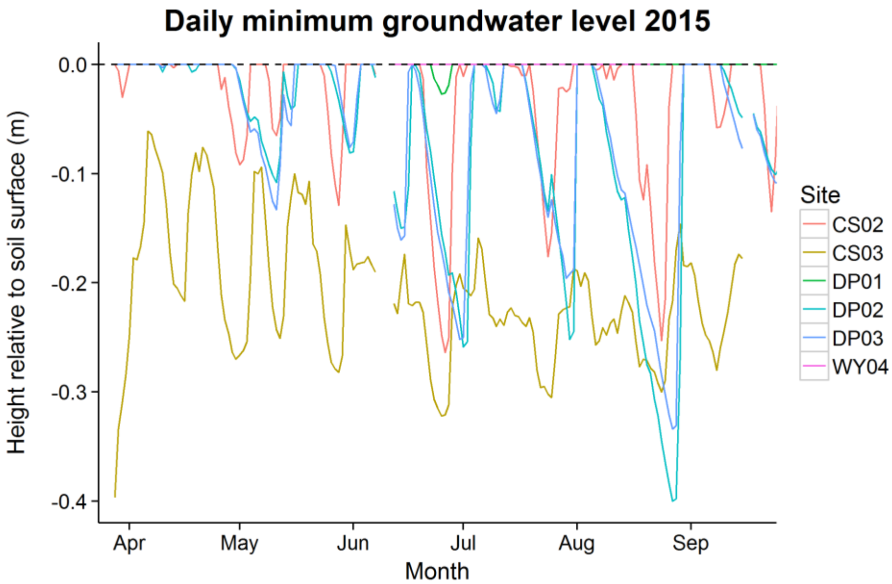

Figure 46. Daily minimum groundwater level across all stations at reference sites April 2015 – September 2015.

Groundwater salinity

Methods

Groundwater salinity monitoring began in March 24, 2015 and went through August 21, 2015 at all stations at the Wallooskee site, including WY4 outside of the dike. Wallooskee site loggers were pulled in August 2015 at the beginning of major earthmoving for restoration. Monitoring for reference stations at Daggett Point began March 24, 2015 and ended March 27, 2016, obtaining a year of data, except for the two stations with the lowest elevations (DP1 and WY4) which were installed at the same time as the groundwater level loggers in June 2015 (Table 23). All groundwater salinity wells were co-located at the wetland monitoring stations alongside the “blue carbon” cores (Appendix 1, Maps 1-4). We did not monitor groundwater salinity at Cooperage Slough, because based on existing reports (LCEP 2007, Lev et al. 2008), we expected the site to be a freshwater tidal wetland. (In retrospect, because our monitoring revealed that this is in fact an estuarine wetland with summer salinities in the mid to upper oligohaline and even extending into the low mesohaline class, we should have monitored groundwater salinity at Cooperage Slough.)

Table 23. Locations and monitoring dates for groundwater salinity stations at the Wallooskee site and reference sites. Easting and Northing represent NAD83 UTM Zone 10 N coordinates in meters. Locations are shown in Appendix 1, Maps 1-4. See Appendix 3 for spatial reference system information. Note WY.04 was located outside of the dike and is considered a reference site.

Site	Location	Station	Easting	Northing	Dates monitored
Wallooskee site	Inside dike	WY1	437416	5111465	3/24/15-8/21/15
		WY3	437639	5111154	3/24/15-8/21/15
Reference sites	Outside dike	WY4	437719	5110996	6/8/15-8/21/15
	Daggett Point	DP1	436334	5113121	6/8/15-3/27/16
		DP2	436302	5113000	3/24/15-3/27/16
		DP3	436218	5113003	3/24/15-3/27/16

Groundwater salinity was monitored in standard shallow groundwater observation wells (Sprecher 2000). Wells were approximately 1.5 m deep, therefore groundwater salinities that occurred when groundwater levels were more than 1.5 m below the soil surface were not tracked. Groundwater salinity was monitored using Odyssey conductivity-temperature data-loggers programmed to collect data at 15 minute intervals and wrapped in copper mesh to prevent fouling (Figure 47). Lab tests were run prior to deployment, and it was determined that the copper mesh did not affect the data. Groundwater salinity logger elevations were the same as the adjacent groundwater level logger elevations (Table 6).

At each datalogger download, a YSI salinity probe (Model 30) was used to measure salinities in the water column within the groundwater well. To determine whether stratification was occurring within the wells, YSI measurements were taken at the top of the water column in the well before removing the logger; at the bottom after removing the logger; and after mixing the water column.

Loggers were calibrated before and after each deployment period using a multi-point conductivity and temperature calibration procedure, and the resulting calibration formula was deployment-specific. Using data from the groundwater level loggers, data were “trimmed” when conductivity-temperature loggers were out of the water. Raw logger data were converted from conductivity values to salinity values using a standard formula (Fofonoff and Millard 1983) plus logger-specific calibration data.

For comparability with channel water salinity data, we analyzed average daily maximum groundwater salinity. Daily maximum groundwater salinities were extracted from the data for the six groundwater salinity wells over the whole monitoring period (March 2015 – March 2016). Comparisons between the restoration site and the reference sites were made for the overlapping dates of 3/24/15 – 8/21/15 (restoration site loggers were pulled in August 2015 due to commencement of site construction).

Differences in average daily maximum groundwater salinity between the Wallooskee site and reference sites were analyzed using a t-test. A two-way ANOVA was used to determine differences at the reference stations among wells and between wet and dry seasons (wet season being December 2015 – February 2016, dry season being July 2015 – September 2015). A simple linear regression was run on the average daily maximum groundwater salinity at each station by elevation at the reference sites. The Wallooskee site had only a total of two stations, so this analysis was not run for the Wallooskee site. When distributions did not meet the normality assumptions, an equivalent non-parametric test was used (either a Wilcoxon in place of a t-test or Kruskal-Wallis in place of ANOVA). All analyses were completed in R (Version 3.1.1).

Figure 47. Odyssey salinity/temperature datalogger wrapped in copper mesh to prevent fouling.

Results and discussion

Groundwater salinity logger data matched well with validation measurements from the YSI probe, suggesting the loggers were functioning properly in the groundwater wells. Stratification of salinity within the groundwater well was minor; salinity differences from top of well to bottom prior to mixing were usually <1 PSU. However, results might be different in more strongly brackish or euhaline wetlands.

Average daily maximum groundwater salinities were significantly lower at the Wallooskee site than reference sites ($p < 0.001$; Figures 48 and 49). The average daily maximum groundwater salinity was 1.71 PSU at the Wallooskee site, compared to 3.20 PSU at the reference sites. At the Wallooskee site, groundwater salinities remained under 2 PSU through August, and did not increase in summer up to the end of the monitoring period. Since tide gates block tidal influence at the Wallooskee site, the low salinities at the site were expected; however, it was interesting that the channel water was more saline, edging into the low mesohaline in late summer. This may have been due to some leakage from the site's tide gates.

Overall, groundwater salinity at the reference sites increased through November, then dropped during the rainy months of December through March (Figure 48; Appendix 2, Figures A11-A15). This pattern reflects typical surface water salinity patterns observed in Oregon tidal wetlands, as winter precipitation dilutes marine waters (e.g. Brophy et al. 2014, Brown et al. 2016). The persistence of groundwater salinity into November, despite normal precipitation in October (https://www.ncdc.noaa.gov/temp-and-precip/climatological-rankings/) probably reflected the time required to flush salinity out of the site's heavy soils.

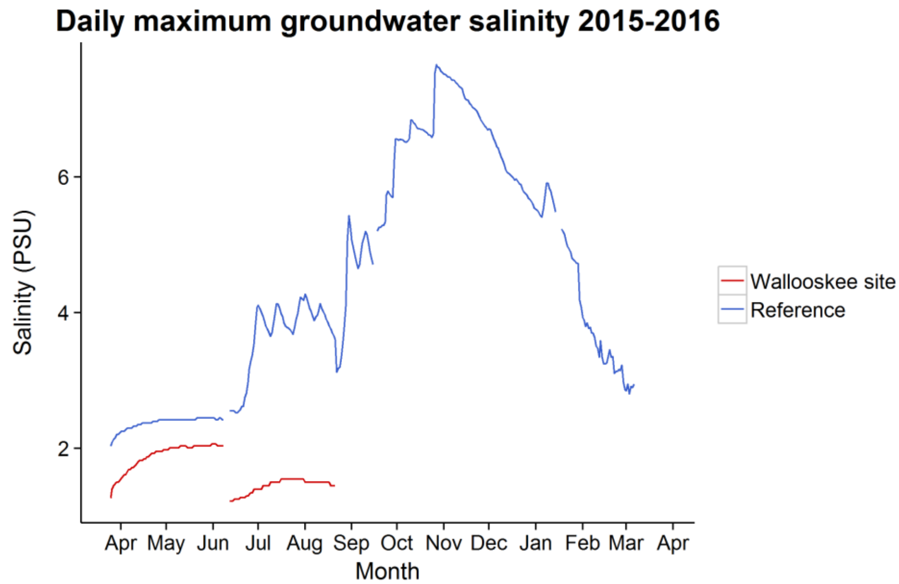

Figure 48. Daily maximum groundwater salinity across all stations at the Wallooskee site and reference sites March 2015 – March 2016.

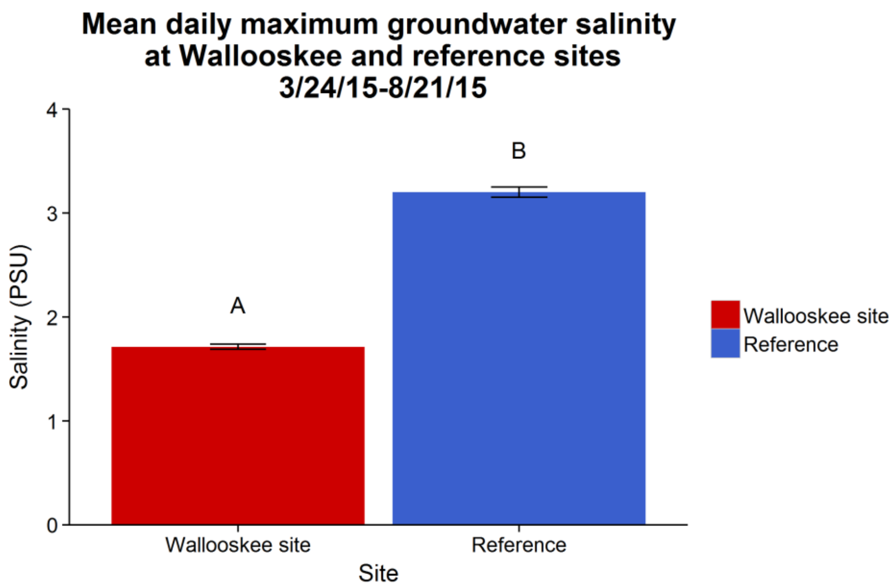

Figure 49. Mean daily maximum groundwater salinities for the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (Wilcoxon test, $p < 0.05$).

Analysis of groundwater salinity showed significant differences among stations ($p < 0.001$; Table 24, Figures 50 and 51). Groundwater salinities at the Wallooskee site (WY1 and WY3) were significantly lower than all other stations (average daily maximum 2.09 and 1.45 PSU, respectively). Salinities were significantly higher at the low marsh stations WY4 and DP1 (4.55 and 4.81 PSU, respectively) than at all other stations. DP2 had significantly higher groundwater salinity compared to DP3, which was expected as it was lower in elevation. Mean daily maximum groundwater salinities at reference stations were significantly correlated to surface elevation ($p = 0.02$, $R^2 = 0.95$; Figure 52); groundwater salinity decreased with increasing elevation. This regression was not run on the Wallooskee site, which contained only two stations.

Table 24. Mean and standard error of daily maximum groundwater salinity at each station at the Wallooskee site and reference sites, 3/24/15-8/21/15).

Site	Location	Station	Mean daily maximum groundwater salinity, PSU (standard error)
Wallooskee site	Wallooskee - inside dike	WY1	2.09 (0.04)
		WY3	1.48 (0.01)
Reference sites	Wallooskee – outside dike	WY4	4.55 (0.09)
	Daggett Point	DP1	4.81 (0.08)
		DP2	2.66 (0.02)
		DP3	2.32 (0.04)

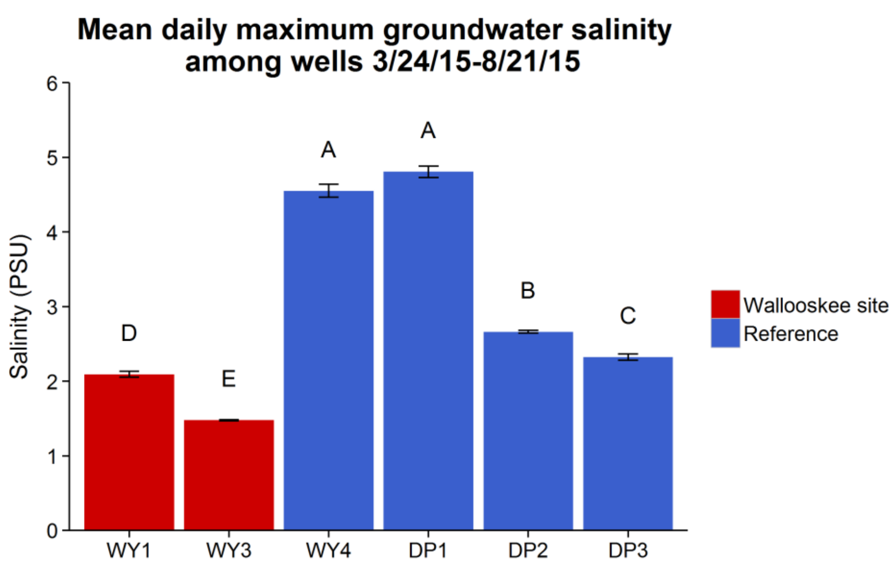

Figure 50. Mean daily maximum groundwater salinity among stations at the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

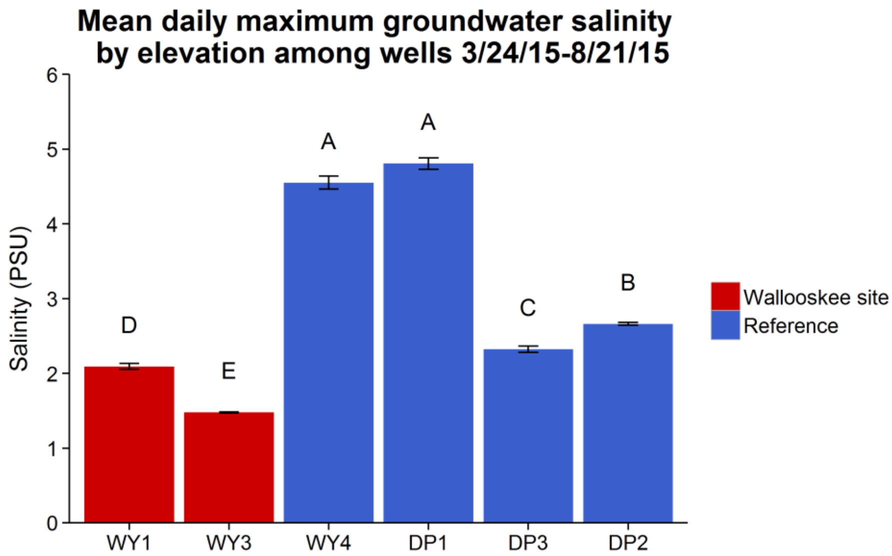

Figure 51. Mean daily maximum groundwater salinity among stations at the Wallooskee site and reference sites. Wells are ordered by ascending elevation from left of right, separated by inside the dike (Wallooskee site) and outside the dike (reference), with WY1 and WY4 having the lowest elevation, and WY3 and DP2 the highest.

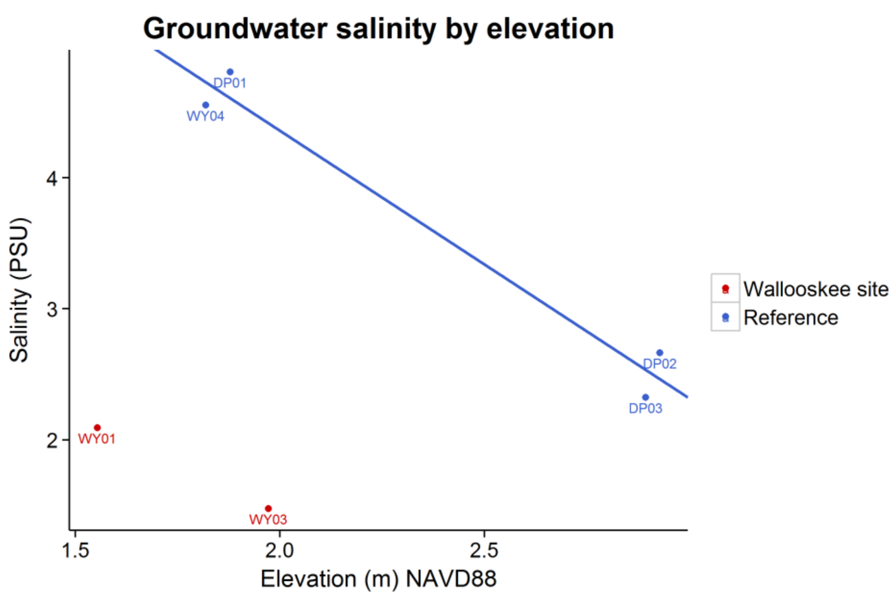

Figure 52. Mean daily maximum groundwater salinity at each station at the Wallooskee site and reference sites (3/24/2015-8/21/2015) along an elevation gradient (R^2 = 0.95, p = 0.02).

Mean daily maximum groundwater salinities among reference stations between the dry (7/1/15-9/30/15) and wet (12/1/15-2/29/16) seasons were significantly different ($p < 0.001$), with DP1 having significantly higher groundwater salinities in the wet and dry season compared to DP2 and DP3 during both seasons (Tables 25 and 26, Figure 53). Groundwater salinity was significantly lower during the dry season at the high marsh and shrub stations DP2 and DP3, compared to those same stations during the wet season. Interestingly, this was the opposite of the trend we observed in channel salinity (Figure 28).

The contrasts between groundwater salinity and channel salinity at Daggett Point relate to the long lag time between seasonal precipitation changes and groundwater salinity responses, visible in Figures A12-A15 (Appendix 2). Since groundwater salinity is rarely measured in PNW tidal wetlands, these observations could be widespread, or site-specific.

Table 25. Summary of two-way ANOVA results for daily maximum groundwater salinity in wet and dry seasons among reference sites. Bold text indicates significant differences (p < 0.05).

Factor	p-value
Site	**< 0.001**
Season	**0.05**
Site*Season	**0.03**

Table 26. Mean and standard error of daily maximum groundwater salinity at each station at Daggett Point reference site, for the dry season (July 2015 – September 2015) and wet season (December 2015 – February 2016).

Site	Location	Station	Season	Mean daily maximum groundwater salinity, PSU (standard error)
Reference	Daggett Point	DP1	dry	5.92 (0.16)
			wet	6.31 (0.27)
		DP2	dry	3.12 (0.08)
			wet	4.02 (0.07)
		DP3	dry	3.34 (0.08)
			wet	4.46 (0.05)

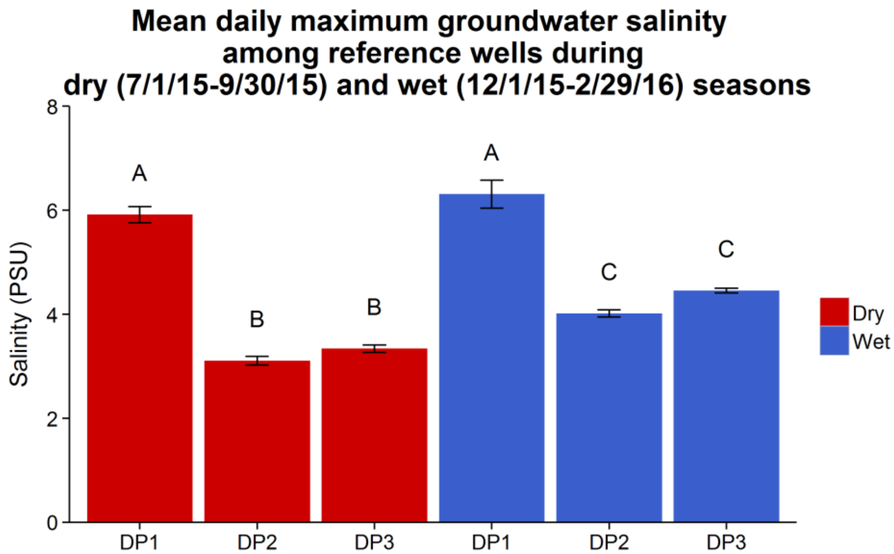

Figure 53. Mean daily maximum groundwater salinity among reference stations during the dry (July – September 2015) and wet seasons (December 2015 – February 2016). Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

Groundwater salinity regimes, though little studied, are likely closely related to groundwater level regimes, which can take years to achieve similar patterns to those at reference sites after restoration (Brophy *et al.* 2014). Factors that slow the restoration of natural patterns in groundwater level at restoration sites, such as compacted soil and low channel density, could also slow the restoration of natural groundwater salinity regimes.

As far as we are aware, no other studies have used belowground dataloggers to measure groundwater salinity across tidal cycles and seasons in tidal wetland restoration and reference sites in the Pacific Northwest. Future monitoring work will determine whether our observations are typical, and will help interpret the observations. For example, the Pacific Northwest Blue Carbon Working Group's NERRS Science Collaborative project will measure groundwater salinity and other ecosystem drivers for a full year at 32 sites from northern California to Puget Sound (Cornu 2017, Janousek et al. 2017). The Blue Carbon Working Group project will include low marsh, high marsh, and forested tidal wetland sites; its primary goal is to establish a database of carbon stocks in Pacific Northwest tidal wetlands.

Because groundwater salinity monitoring is new, we have included a variety of graphs to illustrate the relationships between groundwater salinity, channel water salinity, tide cycles, and seasons in Appendix 2.

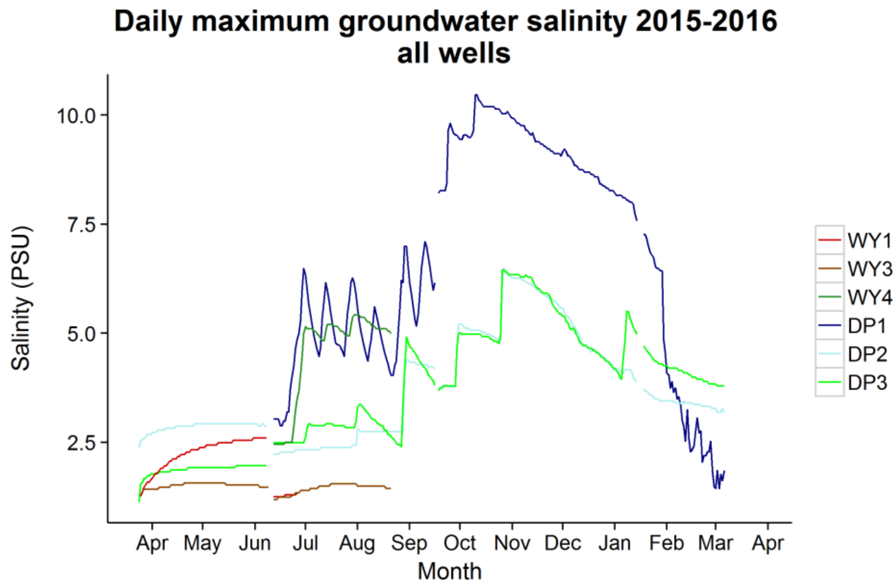

Figure 54. Daily maximum groundwater salinity at all stations at the Wallooskee site and reference sites March 2015 – March 2016.

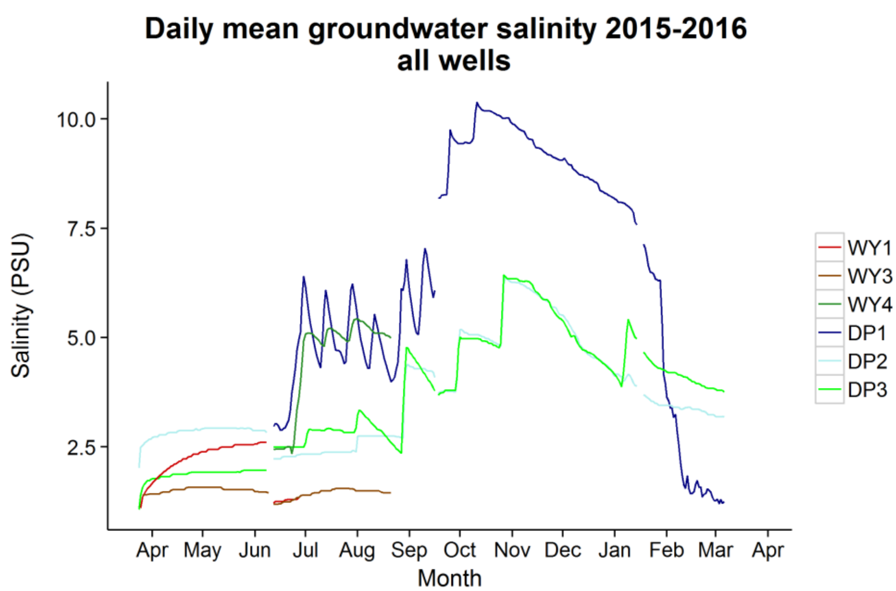

Figure 55. Daily mean groundwater salinity at all stations at the Wallooskee site and reference sites March 2015 – March 2016.

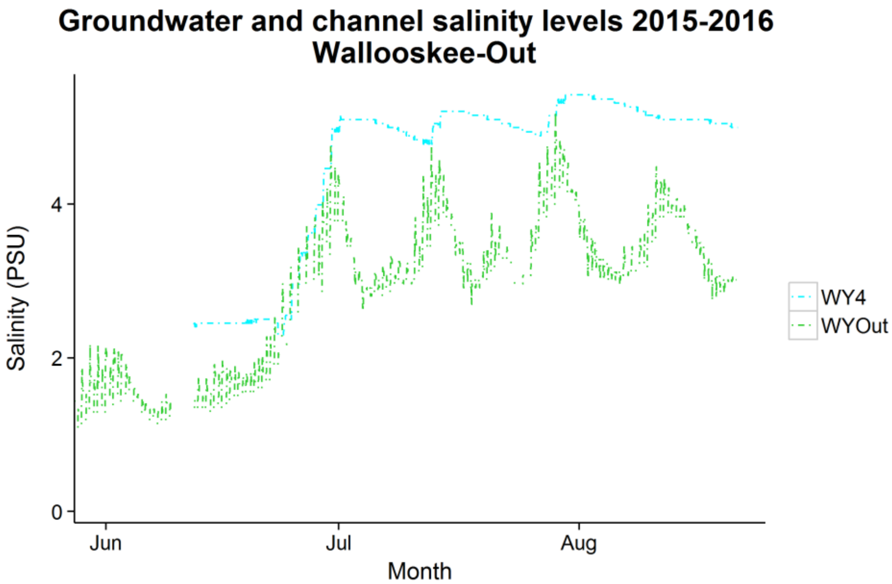

Figure 56. Groundwater salinity (blue, WY4) and channel salinity (green, WYOut) just outside of the Wallooskee site March 2015-August 2015.

Soils

Methods

Soil samples were collected from the surface rooting zone (0-20.3 cm or 0-8 in) using a Dutch auger in February 2015, March 2015, June 2015, August/September 2015 (August for the restoration site, and September for the reference sites), and March 2016. Samples were taken at each wetland monitoring station. At each plot, 6 random subsamples were pooled, bulked in the field, and then mailed to AgSource Laboratory in Umatilla, Oregon, for analysis. At the lab, large roots were removed, samples were dried and homogenized, and a subsample was removed for analysis.

The February 2015 samples were tested at the AgSource Laboratory for electrical conductivity (subsequently converted to salinity), pH, and percent organic matter (subsequently converted to carbon content). Electrical conductivity and pH were measured using a conductance meter and pH meter, respectively. Percent organic matter was determined using the loss on ignition (LOI) method (Craft et al. 1991), with ignition in a muffle furnace for two hours at 360°C. Soil salinity was calculated from electrical conductivity using a standard formula (Fofonoff and Millard 1983). Carbon content was calculated using a conversion specific to high organic soils (0.68 x % LOI) (Kasozi et al. 2009). It is important to note that this conversion factor is fairly close to the one used by Crooks et al. for high-organic soils in the Snohomish River estuary (0.55 x % LOI), but in our blue carbon study, a very different relationship between LOI and carbon content was derived ($C_{org} = 0.29\ LOI + 0.0021\ LOI^2$). These differences are important to conclusions about carbon storage and require further investigation to achieve consistency across studies.

For the April 2015, June 2015, August/September 2015, and March 2016 samples, laboratory analysis included only electrical conductivity. The goal for these sampling events was to gain knowledge of seasonal variation in soil salinity, and to provide several points of comparison to the groundwater salinity data gathered using dataloggers (see **Groundwater salinity** above). The other parameters (pH and organic matter content) were not expected to change greatly across seasons, and were therefore not tested for these later samples.

All soil metrics measured in February 2015 (pH, soil salinity, organic matter, and carbon content) were compared between the Wallooskee site and reference sites using a t-test, and among habitats using an ANOVA test. A simple linear regression was used to determine the relationship between elevation and two of the three soil metrics (pH, organic matter, and carbon content) at the Wallooskee site and reference sites for the February sample date. Analysis of salinity for the February sample date included only two reference stations (DP1 and WY4) because the sample size at stations DP2, DP3, CS2 and CS3 proved inadequate for testing conductivity. This was not considered problematic, since the dry season (August/September) salinity data were of greater interest (see next paragraph), and salinity data was obtained for all stations during the dry season sampling.

Statistical analysis of soil salinity data used only the dry season data (from August/September sample dates), because salinity during the dry season was expected to be highest and therefore variability among sites was expected to be greatest. A simple linear regression was used to determine the relationship between elevation and salinity at the reference sites. (The Wallooskee site was excluded from this analysis, because due to the site's dikes and tide gates, it was not expected to show the typical relationships between elevation and salinity found in least-disturbed wetlands.) Soil salinities were compared between the Wallooskee site and reference sites using a t-test, and among habitats using an ANOVA test. Soil salinity among months and between sites was tested using a two-way ANOVA.

For all tests, when distributions did not meet the normality assumptions, an equivalent non-parametric test was used (either a Wilcoxon in place of a t-test or Kruskal-Wallis in place of ANOVA). All analyses were completed in R (Version 3.1.1).

Many monitoring projects measure soil salinity only once a year (usually during the dry season), or at best a few times a year, using "grab sample" techniques. This study provided an opportunity to compare such “snapshot” data to a time series of groundwater salinity data. We compared the soil sample results, Odyssey salinity datalogger measurements, and YSI validation measurements to learn more about what methods might prove most useful in future projects aimed at understanding soil salinity, a very important ecosystem driver.

Results and discussion

Below, we report first on soil salinities during the late summer dry season, when the diked restoration site is expected to differ most strongly from the reference sites.

Soil salinity was significantly lower at the Wallooskee site compared to the reference sites (0.41 and 8.16 PSU, respectively) during the August/September 2015 sampling (Figure 57). Soil salinity did not differ significantly among habitat types (diked, emergent and shrub) (Figure 58). Soil salinity at the reference sites followed an expected trend of increasing over the dry season, and becoming fresher in the spring (Table 27, Figures 59 and 60). As expected, that seasonal pattern of increasing salinity during summer was not seen at the Wallooskee site, due to a lack of tidal inundation at the site (Figures 59 and 60).

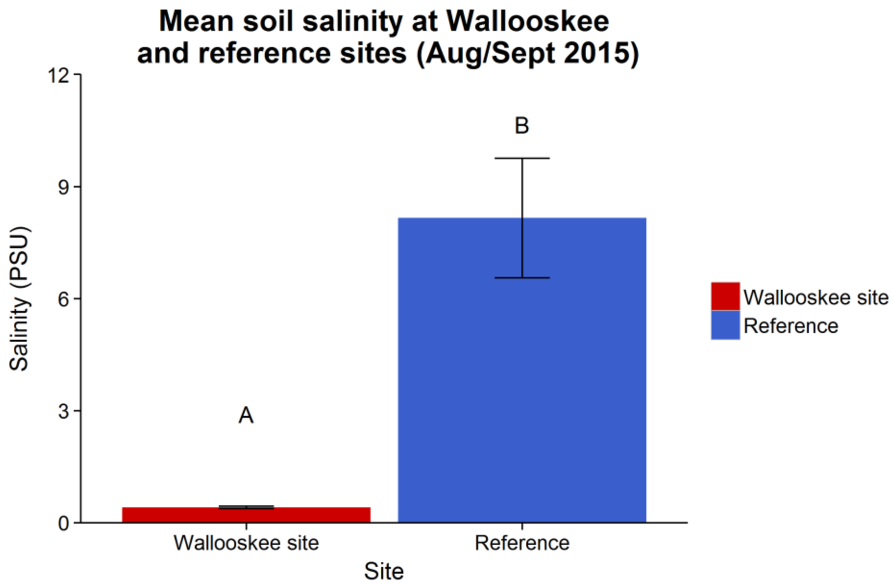

Figure 57. Mean soil salinity for the Wallooskee site and reference sites, August/September 2015. Error bars show one standard error; columns with no letters in common are significantly different (t test, $p < 0.05$).

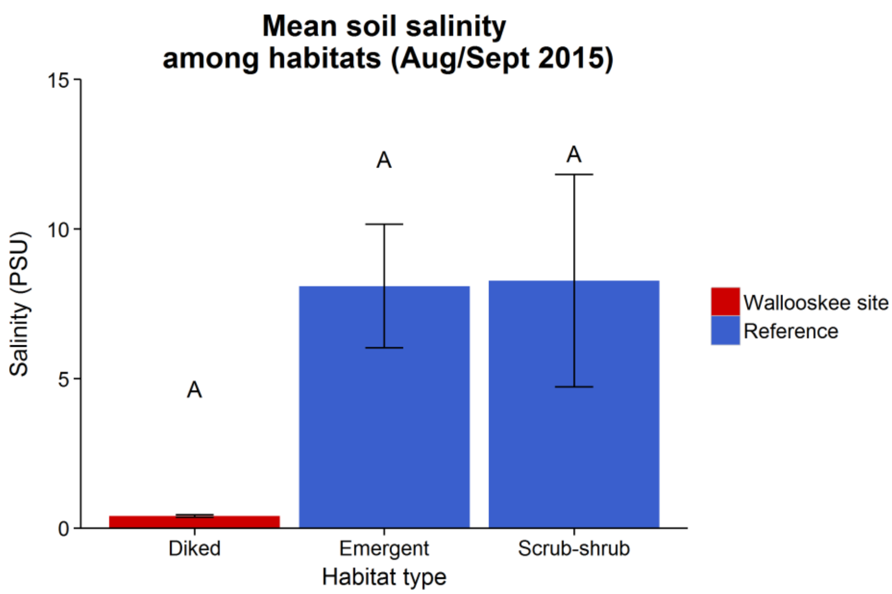

Figure 58. Soil salinity among habitat types at the Wallooskee site and reference sites, August/September 2015. Error bars show one standard error. There were no significant differences among the habitats for any parameter ($p < 0.05$).

Table 27. Summary of ANOVA results for the relationship between soil salinity across months and between sites at the Wallooskee site and reference sites, 2015. Bold text indicates significant relationship ($p < 0.05$).

Metric	p-value
Month	**< 0.001**
Site	**< 0.001**
Month x site	**0.02**

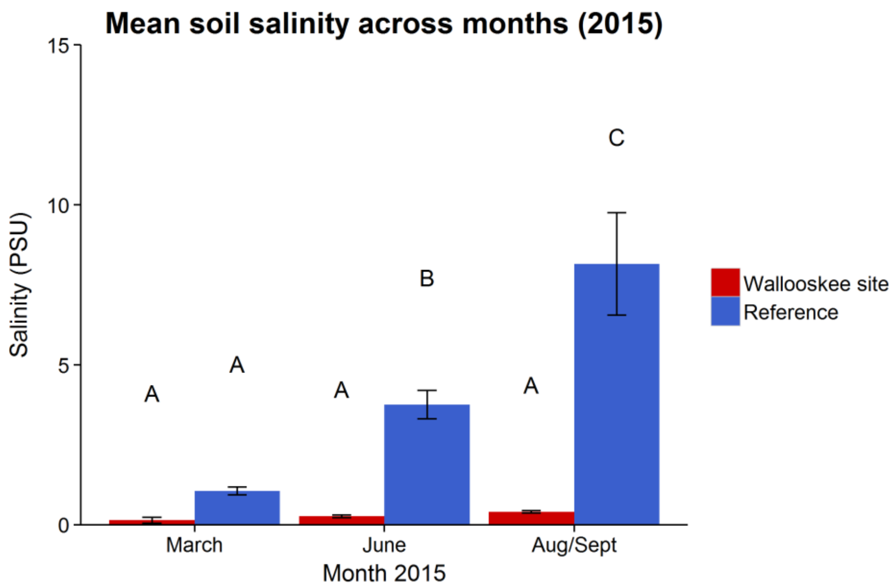

Figure 59. Soil salinity among months at the Wallooskee site and reference sites, March, June, and Aug/Sept 2015. Error bars show one standard error; columns with no letters in common are significantly different (t test, $p < 0.05$).

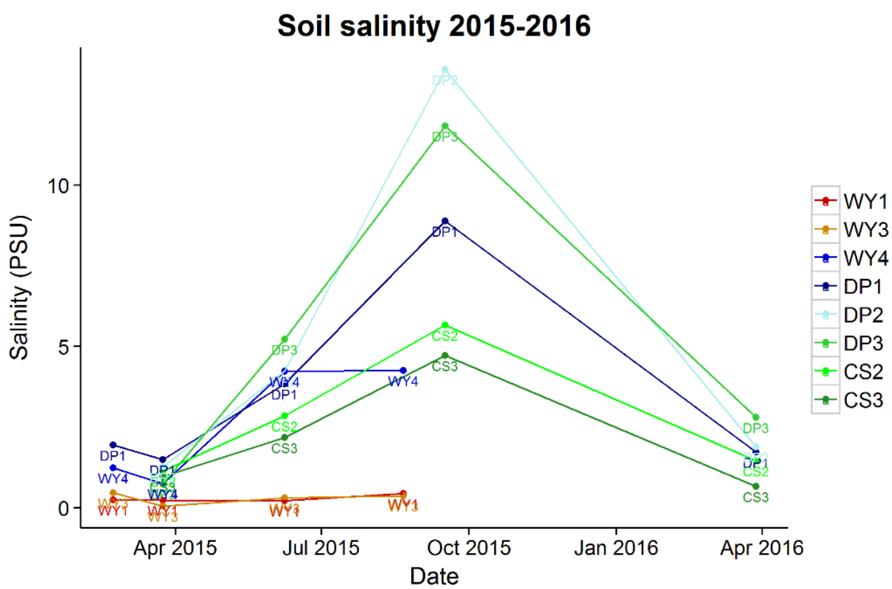

Figure 60. Soil salinity among stations at the Wallooskee site and reference sites over time, February 2015 – March 2016.

During the February sampling, soil metrics did not differ significantly between the Wallooskee site (inside the dike) and reference sites (Table 28, Figure 61), likely due to the small sample size (n=2) at the restoration site. However, trends were visible in the data, including lower pH, lower salinity, and lower organic matter and carbon content at the restoration site compared to the reference sites.

Soil metrics did not different significantly among habitat types during the February 2015 sampling (Table 29, Figure 62). Again, this result is probably attributable to the low sample number. The shrub stations (DP3 and CS3) showed higher organic matter and carbon content compared to emergent marsh; this is in general agreement with results from the carbon cores (see **Sediment and blue carbon accumulation** above).

Table 28. Soil characteristics at Wallooskee site (inside dike) versus reference sites, February 2015. None of the differences were significant.

		Mean (standard error)	Sample size	p-value
Soil pH	Wallooskee site	5.40 (0.20)	2	0.06
	reference sites	6.32 (0.19)	6	
Soil salinity (PSU)	Wallooskee site	0.35 (0.11)	2	0.15
	reference sites	1.59 (0.50)	2*	
Organic matter (%)	Wallooskee site	15.80 (0.50)	2	0.11
	reference sites	22.17 (3.25)	6	
Organic carbon (%)	Wallooskee site	10.74 (0.34)	2	0.11
	reference sites	15.07 (2.21)	6	

* Only two salinity samples were available from reference sites in February, because samples from DP2, DP3, CS2, and CS3 were too small to allow salinity measurements.

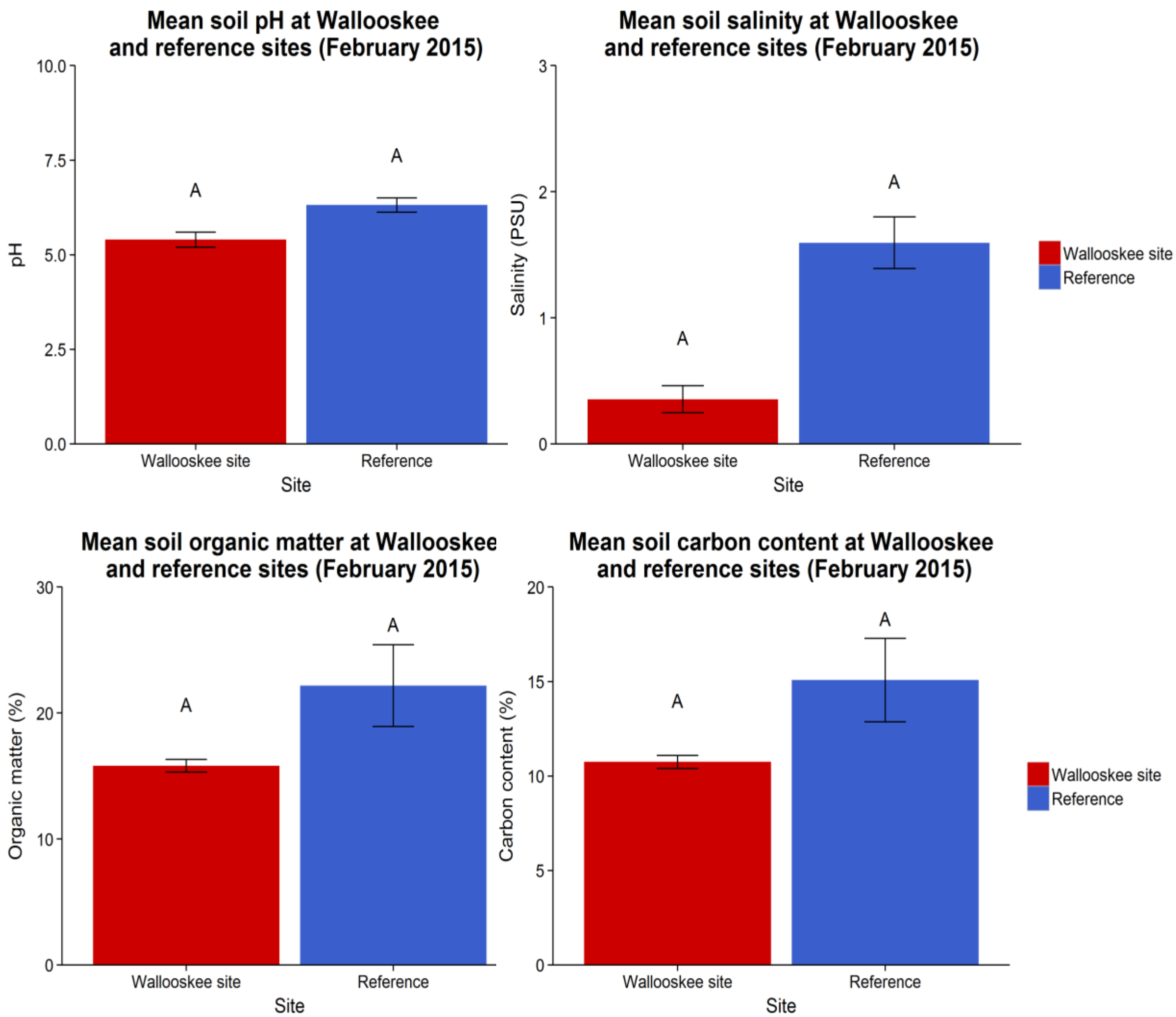

Figure 61. Mean soil pH, soil salinity, organic matter, and carbon content for the Wallooskee site and reference sites, February 2015. Error bars show one standard error; columns with no letters in common are significantly different (t test, $p < 0.05$). Note that the reference category in the salinity chart includes only two stations (DP1 and WY4) due to high organic content of soils in February 2015 at DP2, DP3, CS2 and CS3, which resulted in inadequate sample remaining for salinity analysis.

Table 29. Soil characteristics by habitat class across sites, February 2015. There were no significant differences among the habitats for any parameter.

	Habitat	Mean (standard error)	Sample size	p-value
Soil pH	diked	5.40 (0.20)	2	0.23
	emergent	6.43 (0.28)	4	
	scrub-shrub	6.10 (0.10)	2	
Soil salinity (PSU)	diked	0.35 (0.11)	2	0.08
	emergent	1.59 (0.50)	2	
	scrub-shrub	n/a*	0	
Organic matter (%)	diked	15.80 (0.50)	2	0.22
	emergent	21.03 (4.85)	4	
	scrub-shrub	24.45 (3.05)	2	
Organic carbon (%)	diked	10.74 (0.34)	2	0.22
	emergent	14.30 (3.30)	4	
	scrub-shrub	16.63 (2.07)	2	

* Salinity could not be determined for the scrub-shrub samples in February due to very high organic matter content, leading to inadequate remaining sample size for textural analysis.

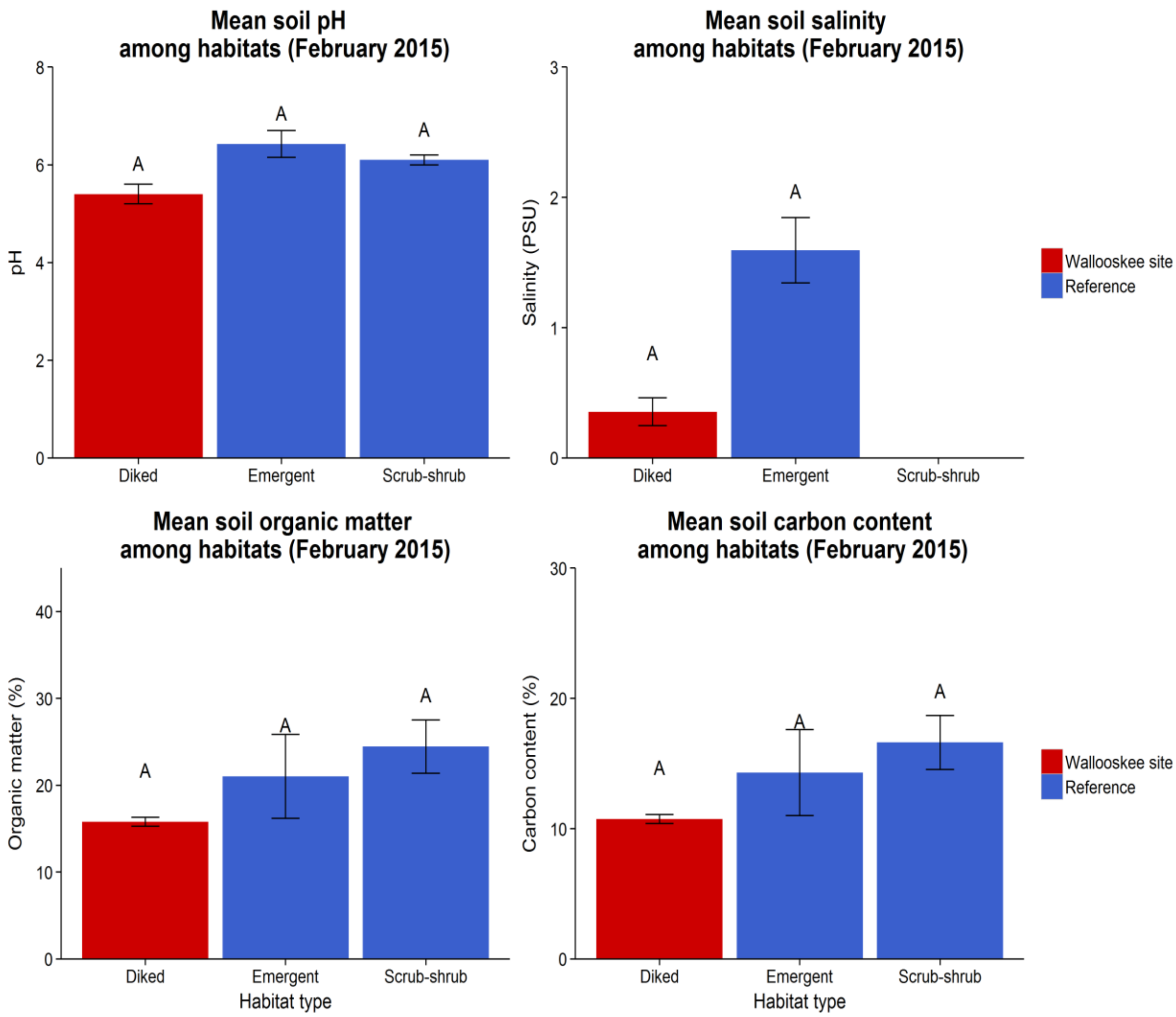

Figure 62. Soil pH, soil salinity, organic matter, and organic carbon among habitat types at the Wallooskee site and reference sites, February 2015. Error bars show one standard error; columns with no letters in common are significantly different (ANOVA test, $p < 0.05$).

We analyzed relationships between soil characteristics and elevation at the reference sites, but not at the Wallooskee site, due to the small sample size (n=2). The inverse relationship between soil pH and elevation was significant at the reference sites ($p = 0.003$, $R^2 = 0.89$), as was the direct correlation between elevation and organic matter ($p = 0.03$, $R^2 = 0.67$) (and also organic carbon [$p = 0.03$, $R^2 = 0.67$]) (Table 30, Figure 63).

Table 30. Summary of simple linear regression results for the relationship between soil metrics and elevation at reference sites, February 2015. Bold text indicates significant relationship ($p < 0.05$).

Metric	**R^2 value**	**p-value**
Soil pH	**0.89**	**0.003**
Organic matter	**0.67**	**0.028**
Carbon content	**0.67**	**0.028**

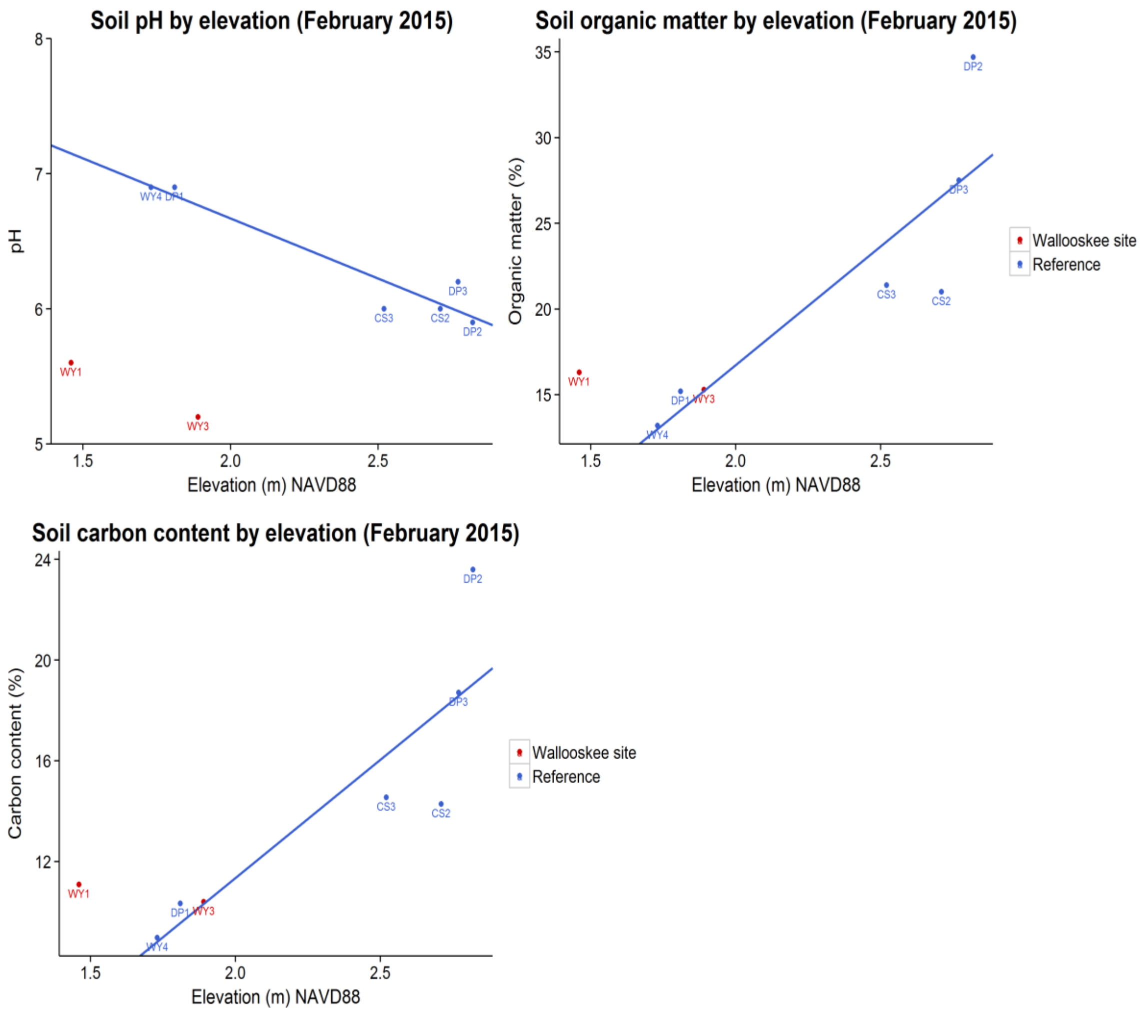

Figure 63. Soil pH, organic matter, and organic carbon along an elevation gradient for all plots at the Wallooskee site and reference sites, February 2015. Lines represent the reference site regressions reported in Table 30. Note: WY4 was located outside of the dike and is considered a reference site.

Comparison between soil/groundwater salinity monitoring methods

As described in **Methods** above, this project offered a unique and important opportunity to compare three different methods of measuring soil/groundwater salinities. Table 31 shows the relationships among soil and groundwater salinities measured using three measurement methods: soil auger samples, groundwater salinity from Odyssey dataloggers, and groundwater salinity from YSI validation measurements in groundwater wells. Briefly, we found that soil salinities from auger samples were generally higher than the groundwater salinity measurements; and that the YSI validation measurements agreed fairly closely with the Odyssey datalogger measurements. Stratification of salinity within the groundwater well was minor; salinity differences from top of well to bottom prior to mixing were usually <1 PSU. However, results might be different in more strongly brackish or euhaline wetlands.

Table 31. Field measurements of soil/groundwater salinity during 5 field visits using 3 methods: soil auger samples, groundwater salinity from Odyssey dataloggers, and groundwater salinity from YSI probe validation measurements in groundwater wells, Wallooskee site and reference sites, Mar. 2015 – Mar. 2016.

		Salinity (PSU)					
	Site	**Restoration**		**Reference**			
	Location	**Wallooskee, inside dike**		**Wallooskee, outside dike**	**Daggett Point**		
Date	**Measurement method**	**WY1**	**WY3**	**WY4**	**DP1**	**DP2**	**DP3**
3/26/2015	Soil - auger	0.23	0.05	0.75	1.49	1.26	0.75
3/26/2015	GW - Odyssey	1.11	1.38	NA	NA	2.20	1.06
3/26/2015	GW - YSI - mixed	0.40	0.80	1.00	1.60	1.60	0.80
6/9/2015	Soil - auger	0.23	0.31	4.23	3.84	4.25	5.22
6/9/2015	GW - Odyssey	1.21	1.19	2.45	2.93	2.22	2.49
6/9/2015	GW - YSI - top	1.00	0.50	1.60	2.00	1.40	1.40
6/9/2015	GW - YSI - bottom	1.20	0.70	2.10	2.50	2.20	1.60
6/9/2015	GW - YSI - mixed	1.20	0.70	2.10	2.30	2.10	1.60
8/21/2015	Soil – auger	0.44	0.37	4.26	site not visited	site not visited	site not visited
8/21/2015	GW - Odyssey	dry	1.44	5.00			
8/21/2015	GW - YSI - top	dry	1.40	3.70			
8/21/2015	GW - YSI - bottom	dry	1.30	3.90			
8/21/2015	GW - YSI - mixed	dry	1.30	3.80			
9/16/2015	Soil – auger	NA*	NA*	NA*	8.88	13.58	11.83
9/16/2015	GW - Odyssey				8.21	3.72	3.68
9/16/2015	GW - YSI - top				5.70	3.40	3.30
9/16/2015	GW - YSI - bottom				5.70	3.40	3.30
9/16/2015	GW - YSI - mixed				5.60	3.40	3.30
3/28/2016	Soil – auger				1.72	1.88	2.80
3/28/2016	GW - Odyssey				1.12	3.19	3.74
3/28/2016	GW - YSI - top				0.20	0.40	0.30
3/28/2016	GW - YSI - bottom				2.40	2.20	2.50
3/28/2016	GW - YSI - mixed				1.70	1.00	0.80

* loggers were pulled from Wallooskee site on 8/21/15

Vegetation

Methods

Vegetation was monitored in June 2015 at the Wallooskee site and reference sites (Daggett Point and Cooperage Slough). Vegetation monitoring was conducted at the combined wetland monitoring stations where we also monitored groundwater level, groundwater salinity, and soils (Appendix 1, Maps 1-4).

At each station, three 1 m^2 quadrats ("vegetation plots") were sampled. The quadrats were placed at random compass bearings and random distances (but within 5m) from the groundwater wells at the wetland monitoring station. At each vegetation plot, elevation was measured using an RTK-GPS receiver with a ten second occupation and the elevations from the three plots were averaged to obtain a station elevation (Table 2). Visual estimates of species percent cover were made within each plot, following Bonham (1989). Percent cover represented the area within the plot that was covered, in vertical projection, by the species in question. Percent cover estimates summed to 100% within a plot, including bare ground and other unvegetated surfaces.

Scientific and common names of plants in this report are based on the Oregon Flora Project's checklist (http://www.oregonflora.org/checklist.php). In the sections below, the term "dominant" was used to describe species with more than 20% cover.

Table 32. Dominant vegetation at sample stations at the Wallooskee site and nearby reference sites. WY4 was located outside of the dike and was considered a reference site throughout this study.

Site		Location	Dominant species
Wallooskee site	Wallooskee-inside	WY1	double flowered creeping buttercup, bluegrass (*Poa sp.*)
		WY3	creeping bentgrass, tall fescue, meadow foxtail
Reference sites	Wallooskee-outside	WY4	Lyngbye's sedge, soft-stem bulrush, common spikerush
	Daggett Point	DP1	Lyngbye's sedge
		DP2	lady fern, soft-stem bulrush
		DP3	coastal willow, lady fern, Pacific water parsley
	Cooperage Slough	CS2	soft-stem bulrush
		CS3	Nootka rose, Douglas' spiraea, coastal willow, lady fern

Species richness, total plant cover, native plant cover, and non-native plant cover were calculated from the raw field data. Each metric was calculated on a per plot basis, then averaged per station. Metrics were then compared between the Wallooskee site and reference sites using a t-test. When distributions did not meet the normality assumption, an equivalent non-parametric test was used (a Wilcoxon test). A simple linear regression was used to determine the relationship between elevation and species richness at the reference sites. This regression was not run for the restoration site, since relationships between vegetation and elevation are expected to be substantially different at the restoration site and it could not be analyzed separately due to small sample size. A multivariate technique, non-metric multidimensional scaling (NMDS), was used to summarize and visualize differences in plant community compositions between the Wallooskee site and reference sites. All analyses were completed in R (Version 3.1.1) using average percent cover per station as the dependent variable, except in the case of species richness.

Results and discussion

Dominant vegetation was different at each station (Table 32). Mean native plant cover was low (4.2%) at the Wallooskee site, compared to 94.3% at the reference sites, a significant difference (0 = 0.05) (Table 33, Figure 64). Mean total plant cover did not differ significantly and was near 100% for the Wallooskee site and for reference sites (Table 33, Figure 64). Mean species richness did not differ significantly between the Wallooskee site and reference sites (4.7 and 5.4 respectively) (Table 33, Figure 64). Eight plant species averaged > 2% cover at the Wallooskee site, and 12 species averaged > 2% cover at each location at the reference sites (Table 34). Many additional species were present at < 2% cover (data available upon request). A complete list of species found in sample plots is found in Table 36.

Plots inside the dike at the Wallooskee site were dominated by the non-native species creeping bentgrass, which averaged 21.7% across the sampled area. While not "dominant" based on this study's definition (>20% cover), other non-natives, such also meadow foxtail, double flowered creeping buttercup, and tall fescue were consistently present throughout the Wallooskee site (17.6%, 14.3%, and 16.7%, respectively). The non-native invasive reed canarygrass was present, but not dominant (Table 34). Together, these species comprise a typical moist pasture species mix for coastal Oregon.

The Daggett Point low marsh (DP1) was dominated by Lyngbye's sedge; the high marsh was dominated by lady fern, softstem bulrush, and Pacific silverweed. Scrub-shrub tidal swamp at Daggett Point was dominated by Hooker's willow, lady fern, and Pacific water parsley. At Cooperage Slough, the scrub-shrub tidal swamp was much more diverse, with a mix of shrub species (Nootka rose, Douglas spiraea, Hooker willow, black twinberry, and salmonberry) and an understory dominated by lady fern. Although not sampled, an interesting lady fern – Nootka rose community occupies broad areas of the Cooperage Slough site (Figure 67). We have observed lady fern as a dominant in other oligohaline to freshwater tidal wetlands (e.g. Brophy et al. 2011).

With only two sample stations at the Wallooskee site, no relationship between species richness and elevation could be calculated. The relationship between species richness and elevation was not significant at the reference sites ($p = 0.19$ including CS2, $p = 0.07$ excluding CS2; Figure 66). Station CS2, despite its higher elevation, was dominated by only one native species (76% soft-stem bulrush), and therefore uncharacteristic of the diversity typically found at in high marsh at outer coast least-disturbed sites. Least-disturbed sites in the PNW generally have strong correlations between species richness and elevation (*e.g.*, Janousek and Folger 2014, Brown et al. 2016), and the lack of significance in this study was likely due to the small sample size. When CS2 was excluded, the R^2 value was high (61%), indicating that even though the p-value was above the 5% threshold of significance ($p = 0.07$), elevation still explained 61% of species richness variability. Species richness at DP2 was the highest sampled (11.3 species on average). At this plot, there was a mix of fresh and brackish-tolerant species (Table 35, Figure 65).

Table 33. Plant community metrics at Wallooskee site versus reference sites, June 2015. Bold text indicates significant differences ($p < 0.05$).

		Mean (standard error)	p-value
Species richness per plot	Wallooskee site	4.7 (0.4)	0.68
	reference sites	5.4 (1.5)	
Total plant cover (%)	Wallooskee site	100.0 (0.0)	0.34
	reference sites	95.1 (2.5)	
Native cover (%)	Wallooskee site	4.1 (2.4)	**0.006**
	reference sites	94.3 (2.4)	
Non-native cover (%)	Wallooskee site	75.7 (13.9)	**0.05**
	reference sites	0.6 (0.5)	

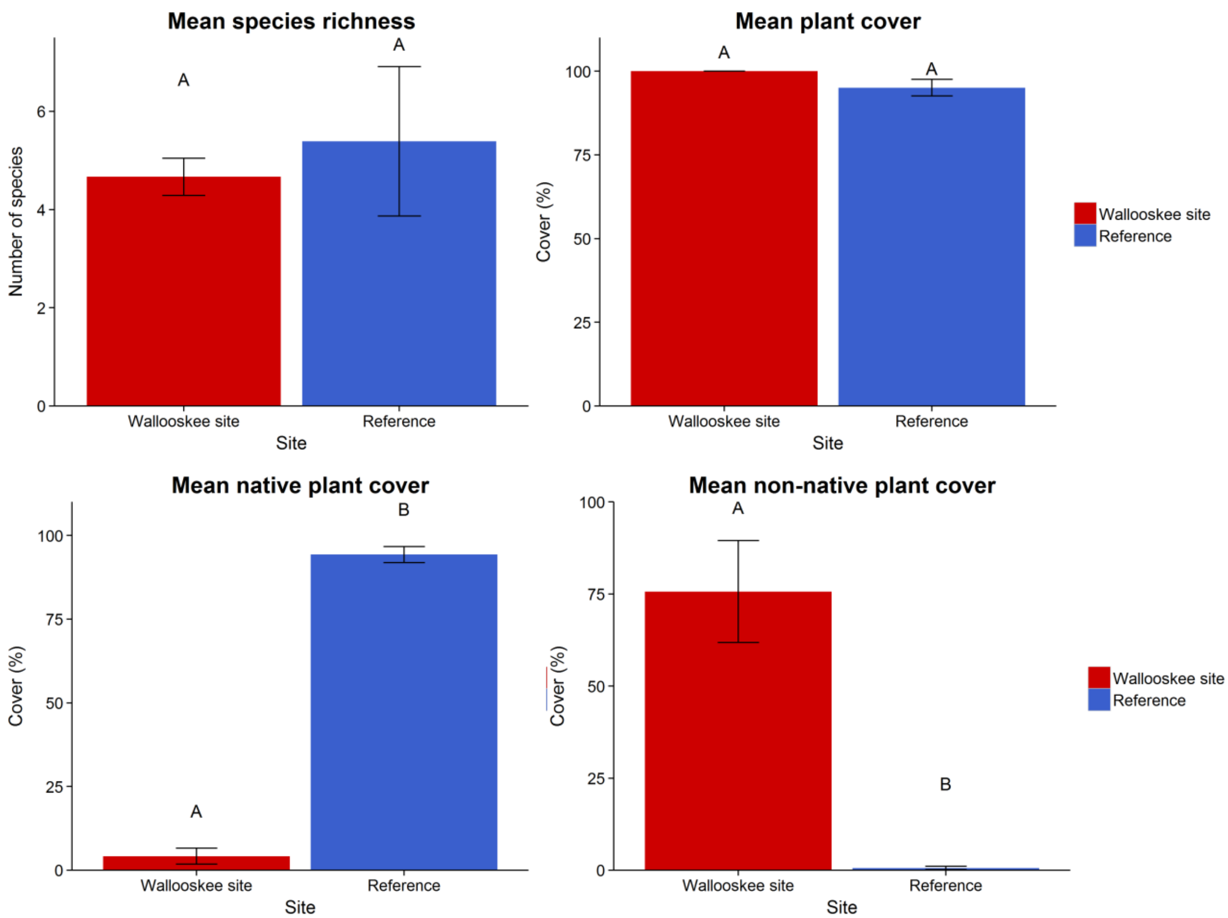

Figure 64. Mean plant species richness per plot, total cover, native cover and non-native cover for the Wallooskee site and reference sites. Error bars show one standard error; columns with no letters in common are significantly different (t test, $p < 0.05$).

Table 34. Average percent cover by plant species and by site (for species averaging over 2% cover within any site) at the Wallooskee site and reference sites, 2015. Green rows indicate native species; orange rows indicate non-native species. Note: Wallooskee site -- outside was located outside of the dike and is considered a reference site.

Plant species	Common name	Wallooskee site - inside	Wallooskee site - outside	Daggett Point	Cooperage Slough
Agrostis stolonifera	creeping bentgrass	21.7	0.0	0.0	0.0
Alopecurus geniculatus	water foxtail	4.2	0.0	0.0	0.0
Alopecurus pratensis	meadow foxtail	17.7	0.0	0.0	0.0
Athyrium filix-femina	lady fern	0.0	0.0	25.3	22.8
Carex lyngbyei	Lyngbye's sedge	0.0	41.3	30.4	0.0
Eleocharis palustris	common spikerush	0.0	36.0	0.0	0.0
Lolium sp.		8.5	0.0	0.0	0.0
Lonicera involucrata	black twinberry	0.0	0.0	0.0	4.2
Oenanthe sarmentosa	Pacific water parsley	0.0	0.0	12.0	0.0
Phalaris arundinacea	reed canarygrass	5.0	0.0	0.0	0.0
Poa sp.		11.7	0.0	0.0	0.0
Potentilla anserina	Pacific silverweed	0.0	0.0	4.7	0.0
Ranunculus repens	double flowered creeping buttercup	14.3	0.0	0.0	0.0
Rosa nutkana	Nootka rose	0.0	0.0	0.0	8.3
Rubus spectabilis	salmonberry	0.0	0.0	0.0	4.0
Salix hookeriana	coastal willow	0.0	0.0	10.0	5.0
Schedonorus arundinaceus	tall fescue	16.7	0.0	0.0	0.0
Schoenoplectus tabernaemontani	soft-stem bulrush	0.0	21.7	9.6	38.0
Spiraea douglasii	Douglas spiraea	0.0	0.0	3.9	5.8
Typha latifolia	broad-leaf cattail	0.0	0.0	3.2	0.0
	bare ground	0.0	0.0	5.6	6.7
	Total	99.8	99.0	95.8	94.8

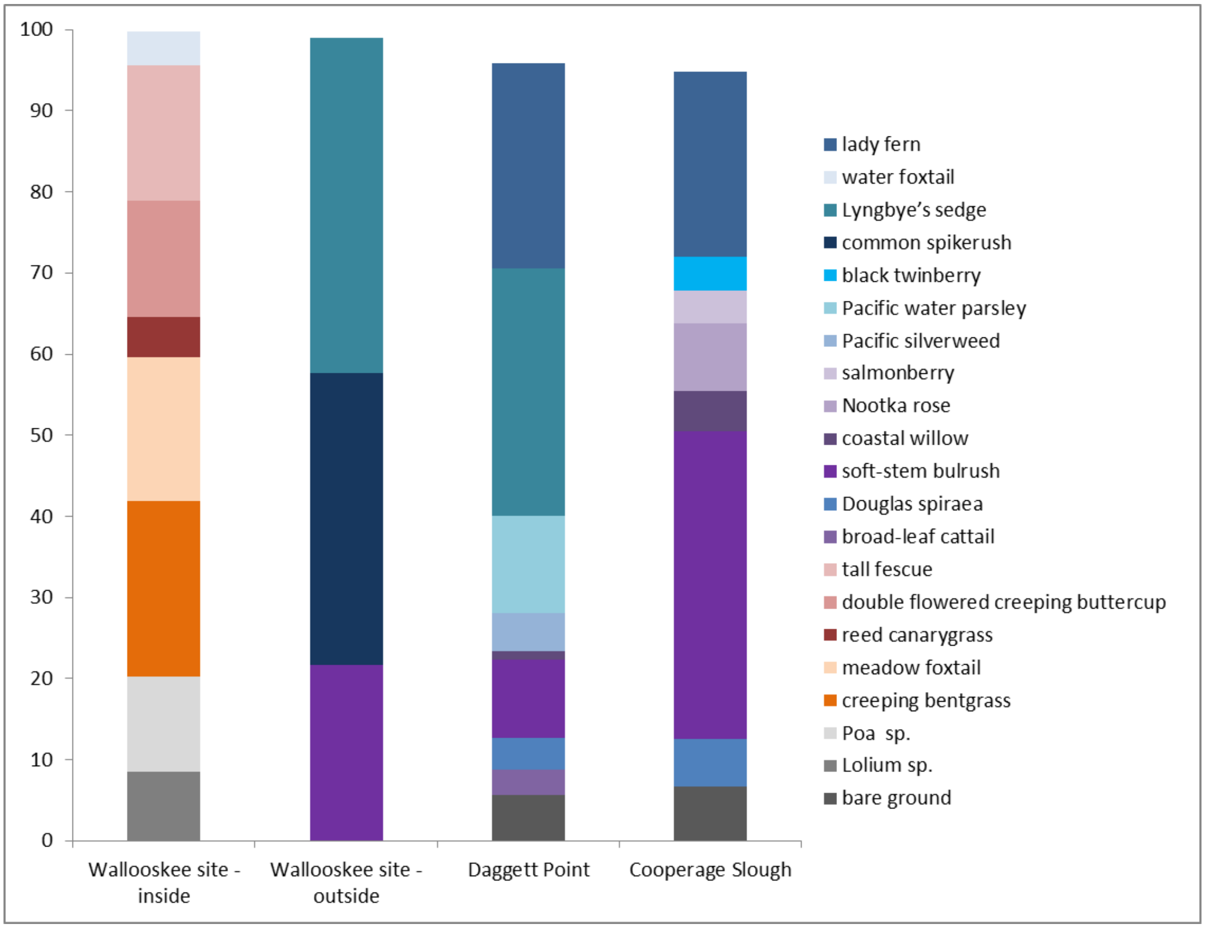

Figure 65. Average percent cover by species (for species averaging over 2% cover) at the Wallooskee site and reference sites, 2015. Blue/purple colors indicate native species, while red/orange colors indicate non-native species. Grey colors indicate unknown origins and bare ground. Note that "Wallooskee site – outside" was located outside of the dike and is considered a reference site.

Table 35. Average percent cover by species and by station (for species averaging over 2% cover at any station) at the Wallooskee site and reference sites, 2015. Green rows indicate native species; orange rows indicate non-native species. Note: Wallooskee site -- outside was located outside of the dike and is considered a reference site.

		Wallooskee site - inside		Wallooskee site - outside	Daggett Point			Cooperage Slough	
Plant species	**Common name**	**WY1**	**WY3**	**WY4**	**DP1**	**DP2**	**DP3**	**CS2**	**CS3**
Athyrium filix-femina	lady fern	0.0	0.0	0.0	0.0	29.3	46.7	10.7	35.0
Agrostis stolonifera	creeping bentgrass	0.0	43.3	0.0	0.0	0.0	0.0	0.0	0.0
Alopecurus geniculatus	water foxtail	8.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Alopecurus pratensis	meadow foxtail	11.7	23.7	0.0	0.0	0.0	0.0	0.0	0.0
Carex lyngbyei	Lyngbye's sedge	0.0	0.0	41.3	91.3	0.0	0.0	0.0	0.0
Eleocharis palustris	common spikerush	0.0	0.0	36.0	0.0	0.0	0.0	0.0	0.0
Impatiens glandulifera	policeman's helmet	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.0
Juncus effusus	soft rush	0.0	0.0	0.3	0.0	5.0	0.0	0.0	0.0
Lolium sp.		16.7	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Lonicera involucrata	black twinberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.3
Oenanthe sarmentosa	Pacific water parsley	0.0	0.0	0.0	0.0	4.3	31.7	0.0	0.0
Phalaris arundinacea	reed canarygrass	10.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Poa sp.		23.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Potentilla anserina	Pacific silverweed	0.0	0.0	0.0	0.0	14.0	0.0	0.0	0.0
Ranunculus repens	double flowered creeping buttercup	28.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0
Rosa nutkana	Nootka rose	0.0	0.0	0.0	0.0	0.0	0.0	0.0	16.7
Rubus spectabilis	Salmonberry	0.0	0.0	0.0	0.0	0.0	0.0	0.0	8.0
Salix hookeriana	Hooker's willow	0.0	0.0	0.0	0.0	0.0	30.0	0.0	10.0
Schedonorus arundinaceus	tall fescue	1.7	31.7	0.0	0.0	0.0	0.0	0.0	0.0
Schoenoplectus tabernaemontani	soft-stem bulrush	0.0	0.0	21.7	0.3	27.5	1.0	76.0	0.0
Scirpus microcarpus	small-fruited bulrush	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.7
Spiraea douglasii	Douglas spiraea	0.0	0.0	0.0	0.0	0.0	11.7	0.0	11.7
Typha latifolia	broad-leaf cattail	0.0	0.0	0.0	3.3	2.8	3.5	0.0	0.0
Vicia nigricans	giant vetch	0.0	0.0	0.0	0.0	0.0	0.0	0.0	3.3
	bare ground	0.0	0.0	0.0	5.0	11.6	0.0	13.3	0.0

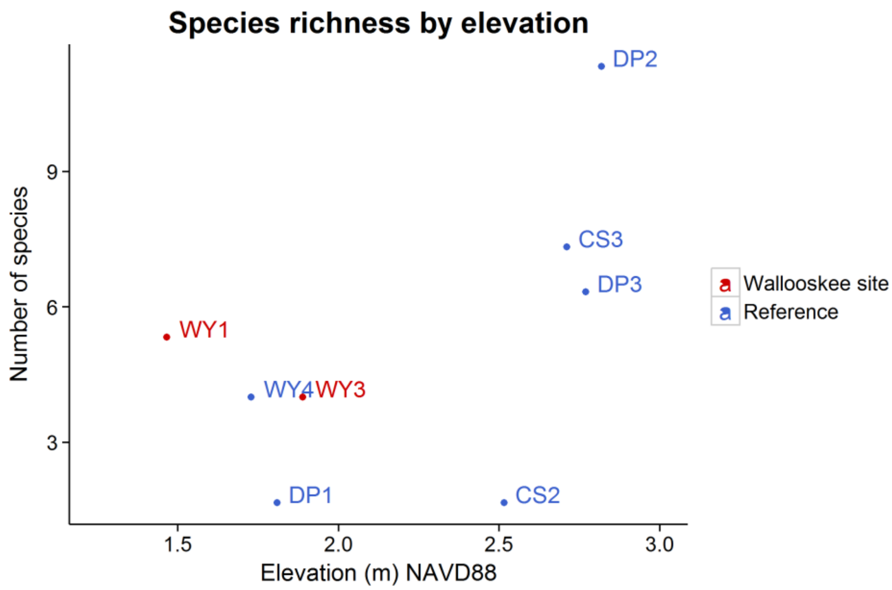

Figure 66. Species richness along an elevation gradient for all plots at the Wallooskee site and reference sites. Note WY4 was located outside of the dike and is considered a reference site.

The NMDS showed that vegetation sampling locations at the Wallooskee site grouped together, as did vegetation plots at the reference sites, though the sample size at the Wallooskee site did not allow for a stress value (Figure 66). Over time we expect the vegetation sample locations at the Wallooskee sites to cluster more closely with those at the reference sites in the NMDS analysis, as species composition converges between the Wallooskee site and reference sites. At other restoration projects during initial stages of restoration, non-native species often die back, leading to increased bare ground (Brophy et al. 2014, Brown and Brophy 2015), but over several years, post-restoration monitoring is expected to should show native species returning to the site.

Figure 67. Lady fern – Nootka rose plant community at Cooperage Slough.

Table 36. List of species found in sample quadrats at the Wallooskee site and nearby reference sites, and their common names, 2015. Green rows and "N" indicate native species; orange rows and "NN" indicate non-native species.

Scientific name	**Common name**	**Origin**
Agrostis stolonifera	creeping bentgrass	NN
Alisma triviale	northern water plantain	N
Alopecurus geniculatus	water foxtail	N
Alopecurus pratensis	meadow foxtail	NN
Athyrium filix-femina	lady fern	N
Carex densa	dense sedge	N
Carex lyngbyei	Lyngbye's sedge	N
Carex obnupta	slough sedge	N
Convolvulus sp.	bindweed	
Eleocharis palustris	common spikerush	N
Epilobium ciliatum	purple leaved willowherb	N
Galium aparine	stickywilly	N
Galium trifidum	small bedstraw	N
Holcus lanatus	velvetgrass	NN
Impatiens glandulifera	policeman's helmet	NN
Iris pseudacorus	yellow flag	NN
Juncus effusus	soft rush	NN
Lathyrus sp.	wild pea	
Lolium sp.	ryegrass	
Lonicera involucrata	black twinberry	N
Lotus corniculatus	birdsfoot trefoil	NN
Oenanthe sarmentosa	Pacific water parsley	N
Phalaris arundinacea	reed canarygrass	NN
Poa sp.		
Potentilla anserina	common silverweed	N
Ranunculus repens	double flowered creeping buttercup	NN
Rosa nutkana	Nootka rose	N
Rubus spectabilis	salmonberry	N
Rubus ursinus	Pacific blackberry	N
Salix hookeriana	coastal willow	N
Schedonorus arundinaceus	tall fescue	NN
Schoenoplectus tabernaemontani	soft-stem bulrush	N
Scirpus microcarpus	small-fruited bulrush	N
Spiraea douglasii	Douglas spiraea	N
Symphyotrichum subspicatum	Douglas' aster	N
Typha latifolia	broad-leaf cattail	N
Veronica serpyllifolia	thyme leaved speedwell	Unk
Vicia nigricans	giant vetch	N

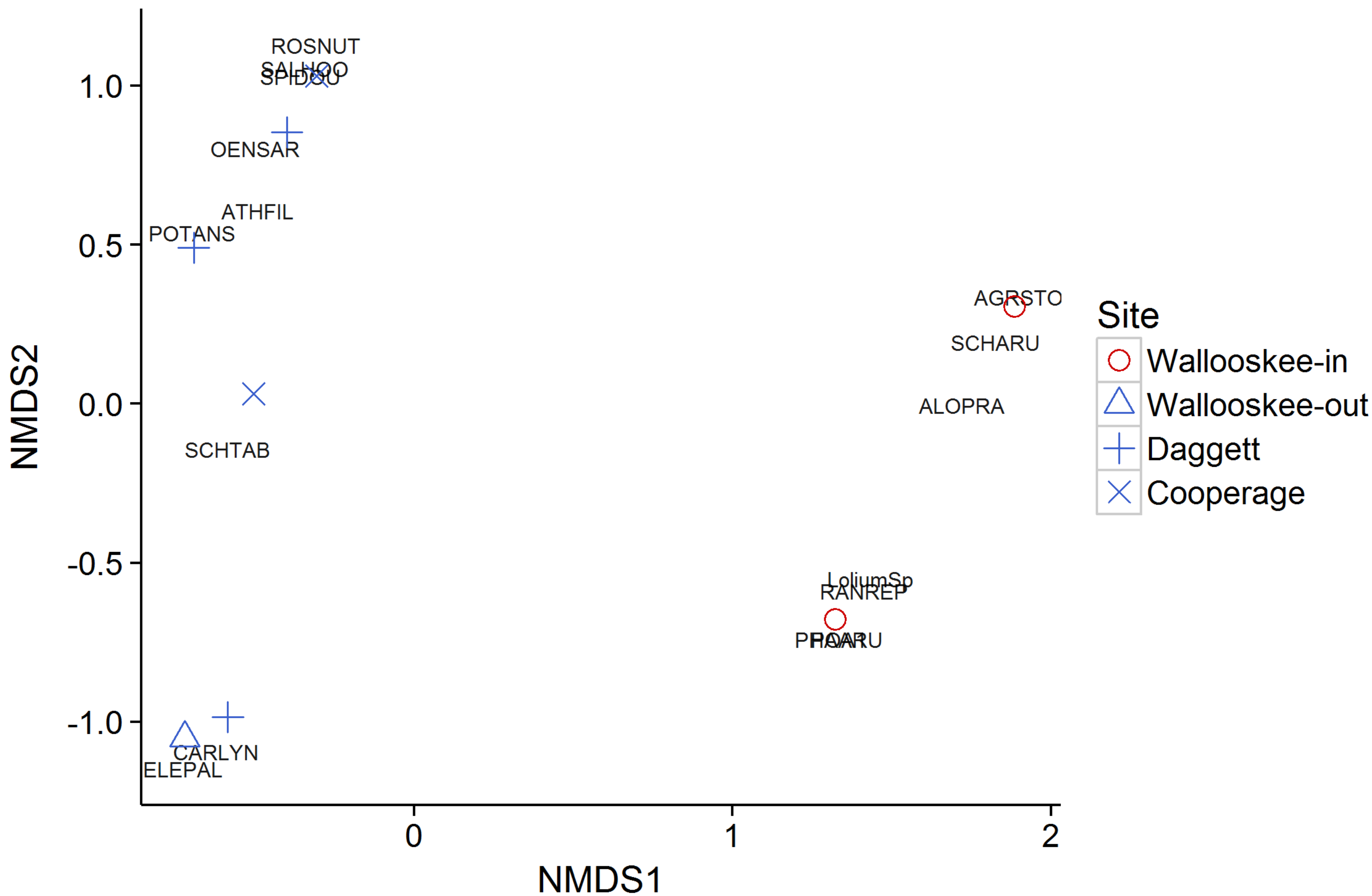

Figure 68. Non-metric multidimensional scaling (NMDS) plot for the Wallooskee and reference plant plots. Red dots indicate Wallooskee plots and blue dots indicate reference plots. Each dot represents a single plot. Dots closer together are more compositionally similar. The centroids of plant species used in the analysis are indicated by six letter species codes on the plot. Only species that had an average cover of greater than 10% were included in the analysis. Note: Wallooskee -- out was located outside of the dike and is considered a reference site.

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Appendix 1. Maps

Map 1. ETG channel and wetland monitoring stations at Wallooskee-Youngs restoration site, 2015-2016

Map 2. ETG channel and wetland monitoring stations at Daggett Point reference site, 2015-2016

Map 3. ETG channel monitoring station at the Grant Island reference site, 2015-2016

Map 4. ETG channel and wetland monitoring stations at the Grant Island reference site, 2015-2016

Map 5. Blue carbon cores at the Wallooskee-Youngs restoration site, 2015. Cores were adjacent to wetland monitoring stations shown in Maps 1-4.

Map 6. Blue carbon cores at the Daggett Point reference site, 2015

Map 7. Blue carbon cores at the Cooperage Slough reference site, 2015

Appendix 2. Additional figures

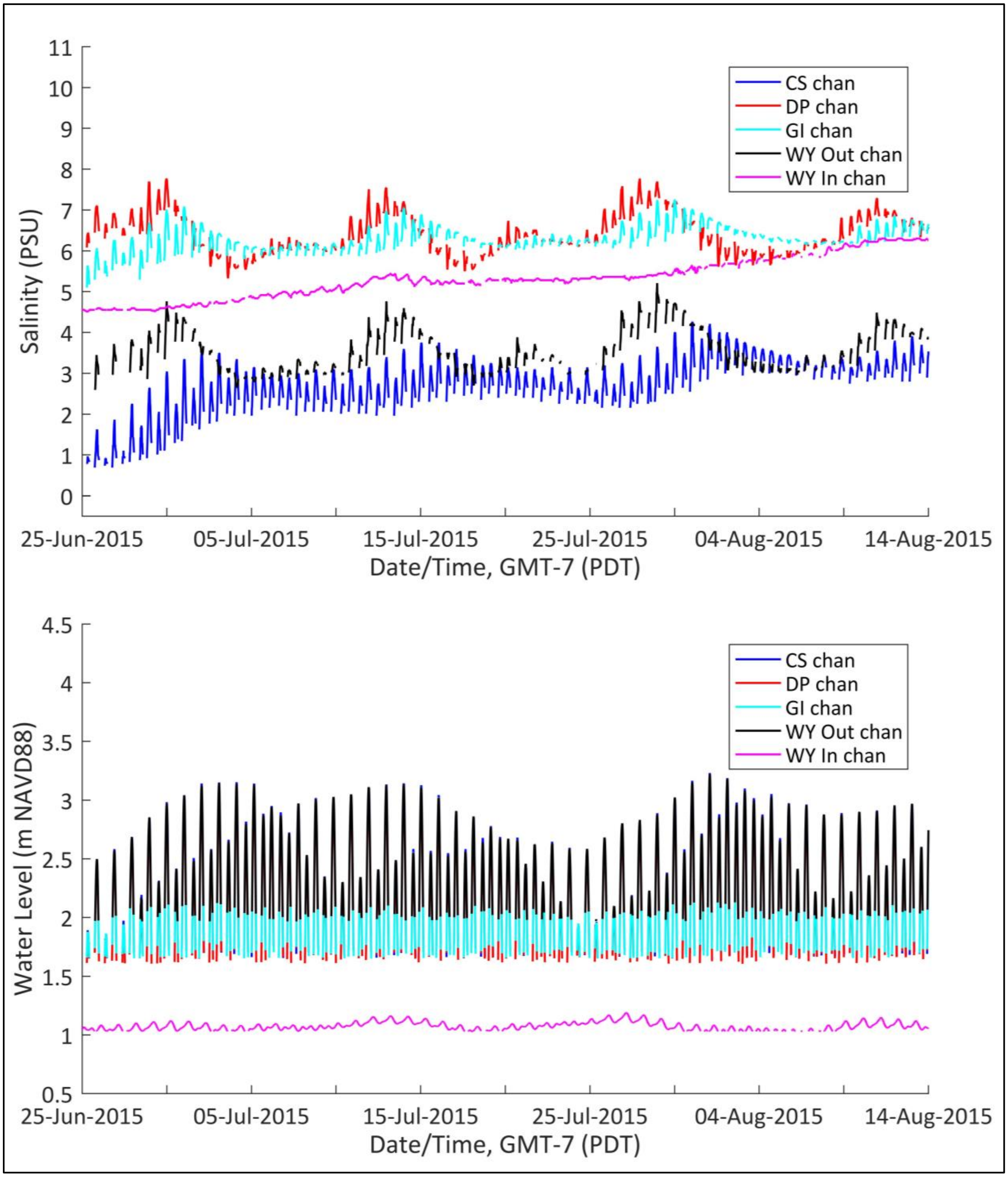

Figure A1. Salinity and tide heights at all channel monitoring stations, summer 2015. For station codes, see Table 6. Tide heights (bottom graph) were similar for all reference stations (CS, DP, GI and WY Out), so lines overlap. The flat baseline for the reference site loggers in the bottom graph indicates the sensor elevation, which was near mean tide level – therefore, loggers only measured high tides.

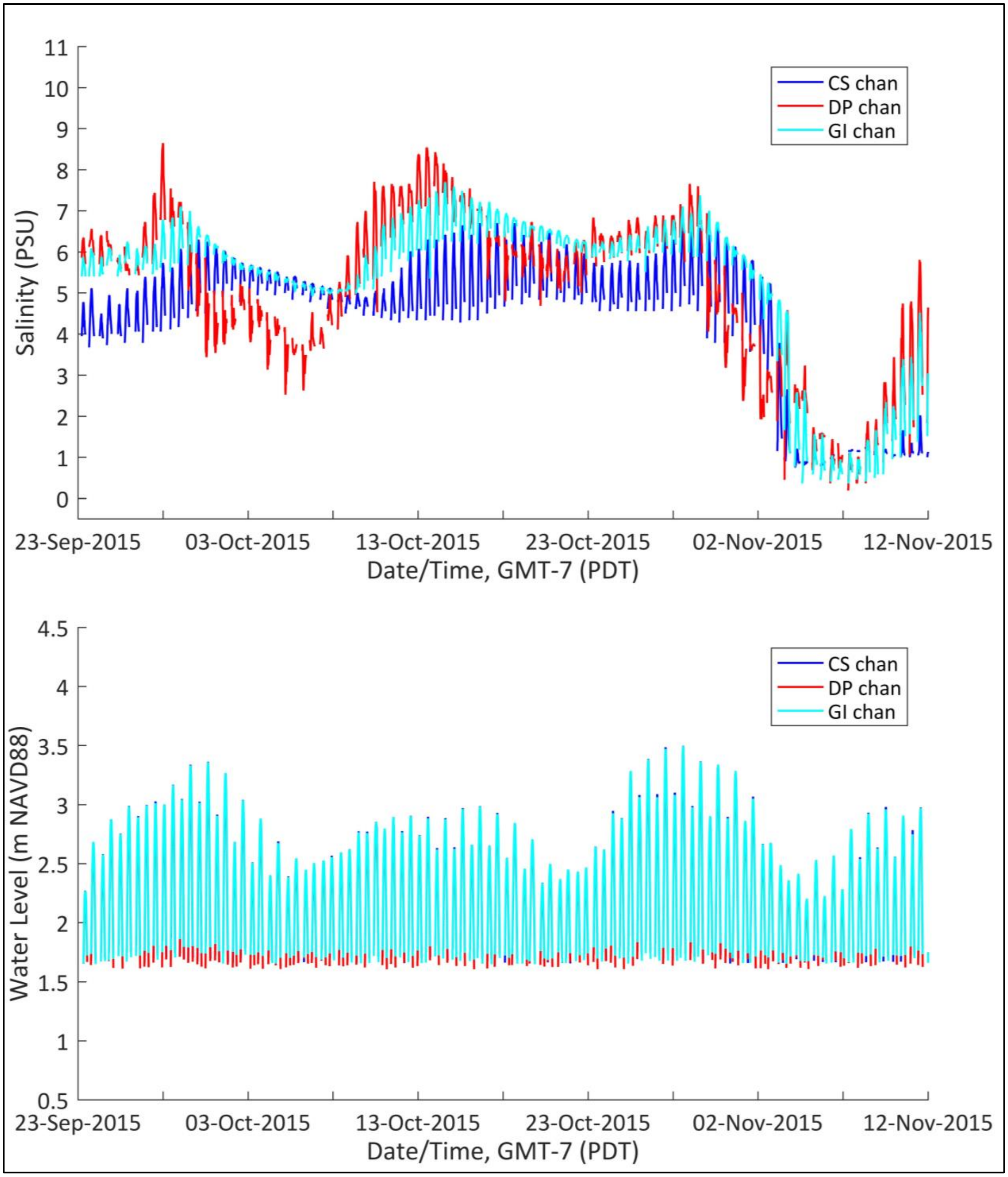

Figure A2. Salinity and tide heights at reference site channel monitoring stations, fall 2015. (WY In and WY Out loggers were removed in August 2015 when site construction began.) For station codes, see Table 6. Tide heights (bottom graph) were similar for all reference stations (CS, DP, GI and WY Out), so lines overlap. The flat baseline in the bottom graph indicates the sensor elevation, which was near mean tide level – therefore, loggers only measured high tides.

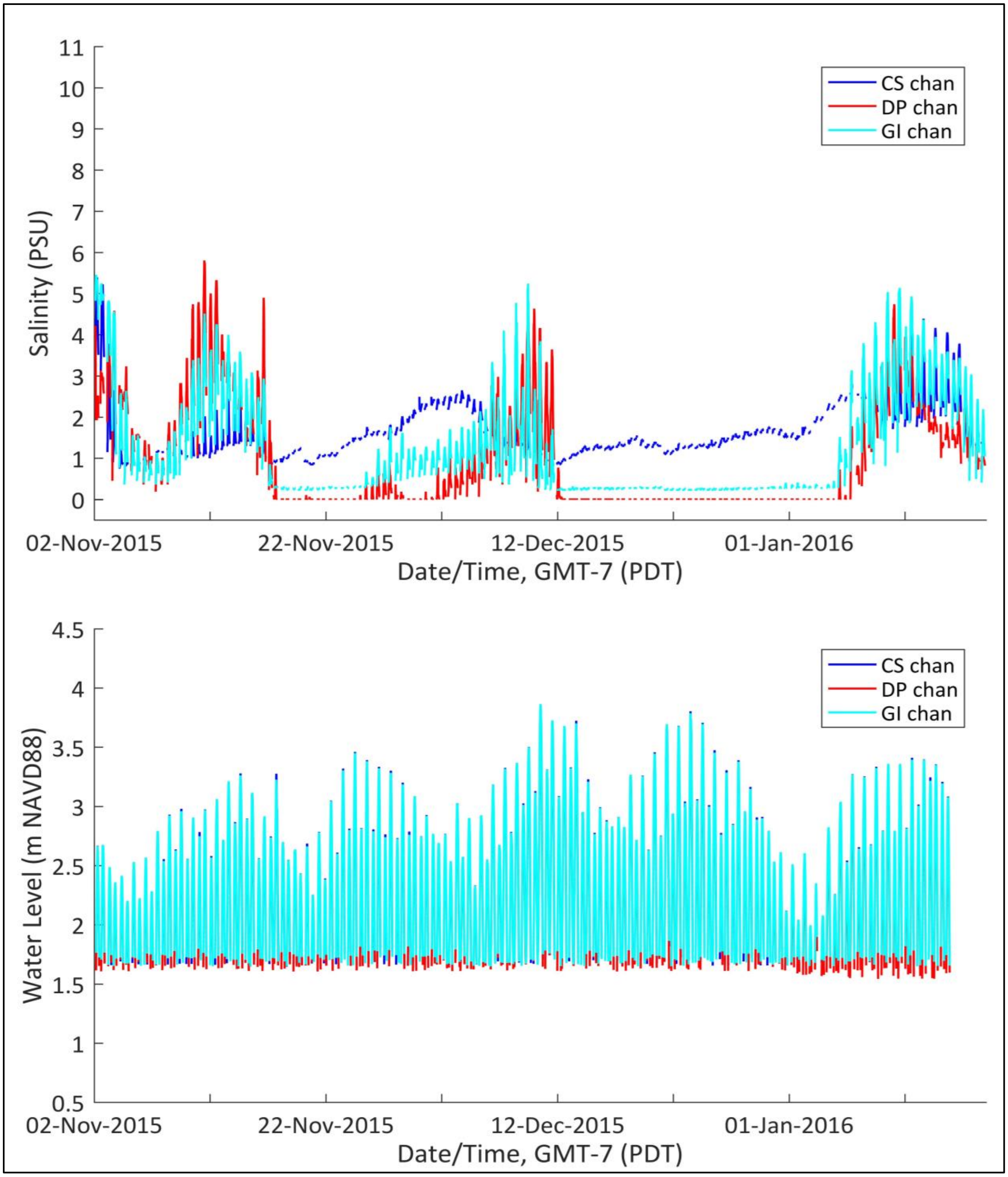

Figure A3. Salinity and tide heights at all channel monitoring stations, winter 2015-2016. (WY In and WY Out loggers were removed in August 2015 when site construction began.) For station codes, see Table 6. Tide heights (bottom graph) were similar for all reference stations (CS, DP, GI and WY Out), so lines overlap. The flat baseline in the bottom graph indicates the sensor elevation, which was near mean tide level – therefore, loggers only measured high tides.

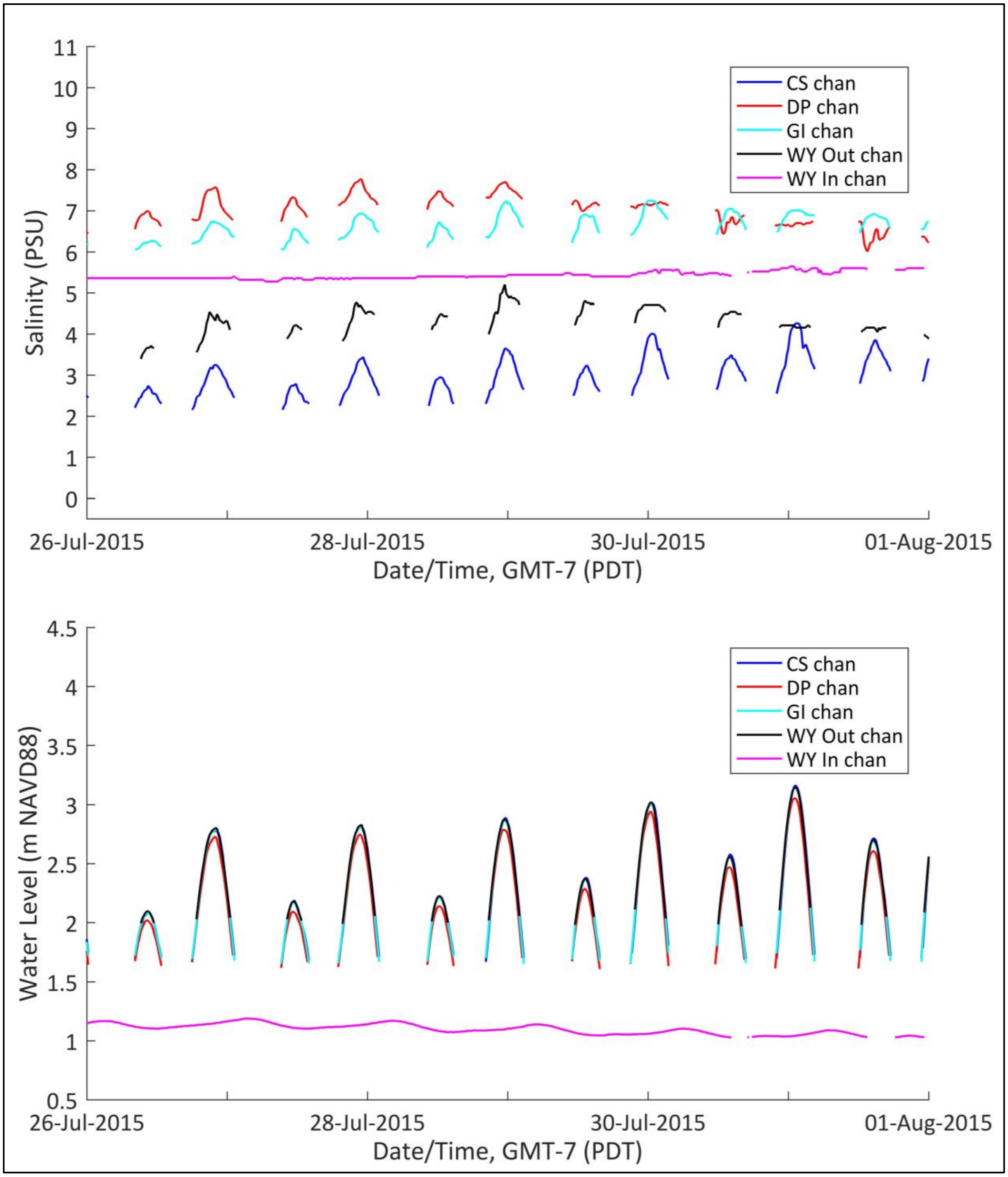

Figure A4. Salinity and tide heights at all channel monitoring stations across a few tide cycles in summer 2015, showing relationships between salinity and tide peaks. Legend in top graph also applies to bottom graph. For station codes, see Table 6. Note: Loggers measured only high tides, due to their sensor heights near mean tide level in channels which often emptied at low tide.

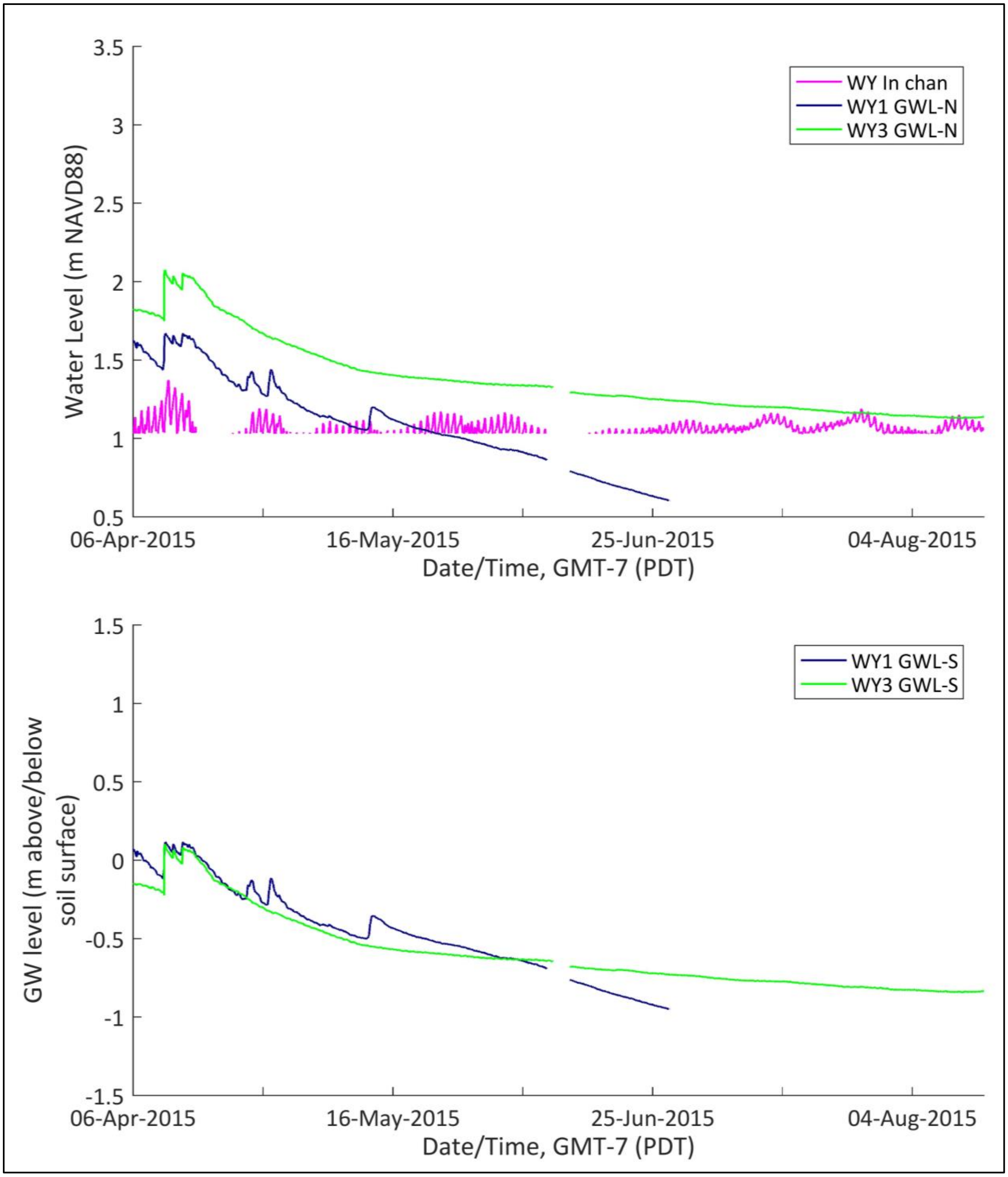

Figure A5. Channel water level versus groundwater level at the Wallooskee site for the full period of record (loggers were removed in August 2015 due to site construction). "WY In chan" is the channel water level station inside the dike. WY1 GWL-N and WY1 GWL-S are the same groundwater station; WY3 GWL-N and WY3 GWL-S are also the same station. The upper graph shows water levels on the NAVD88 elevation datum; the lower graph shows groundwater levels relative to the soil surface.

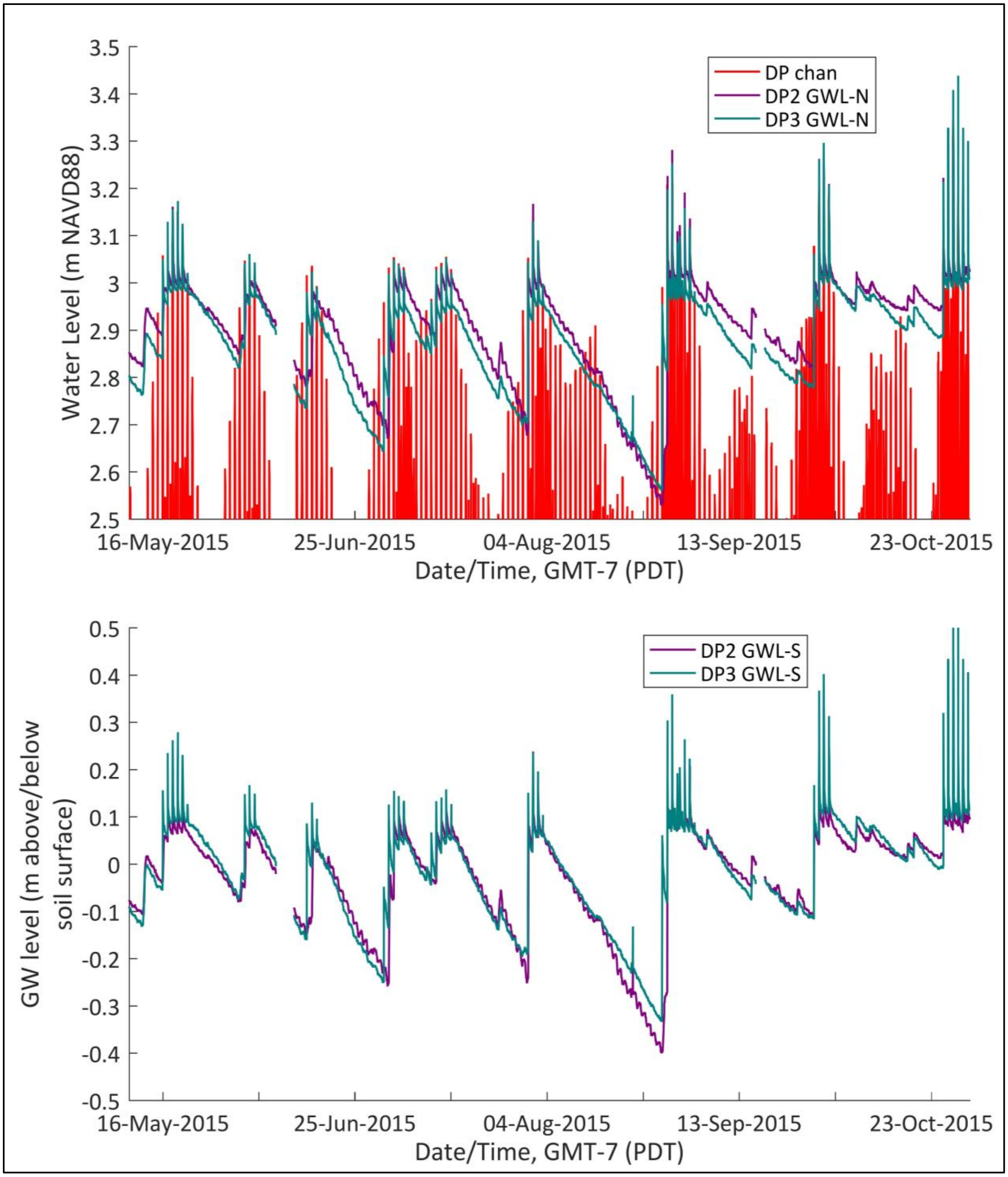

Figure A6. Channel water level versus groundwater level in high marsh and scrub-shrub tidal swamp at the Daggett Point reference site, summer 2015. "DP chan" is the channel water level station; DP2 GWL-N is the high marsh groundwater station; and DP3 GWL-N is the shrub swamp groundwater station. The upper graph shows water levels on the NAVD88 elevation datum; the lower graph shows groundwater levels relative to the soil surface. Groundwater levels for station DP1 (low marsh) are not shown; they remained at or above the soil surface throughout the monitoring period, with peaks that matched the tide peaks.

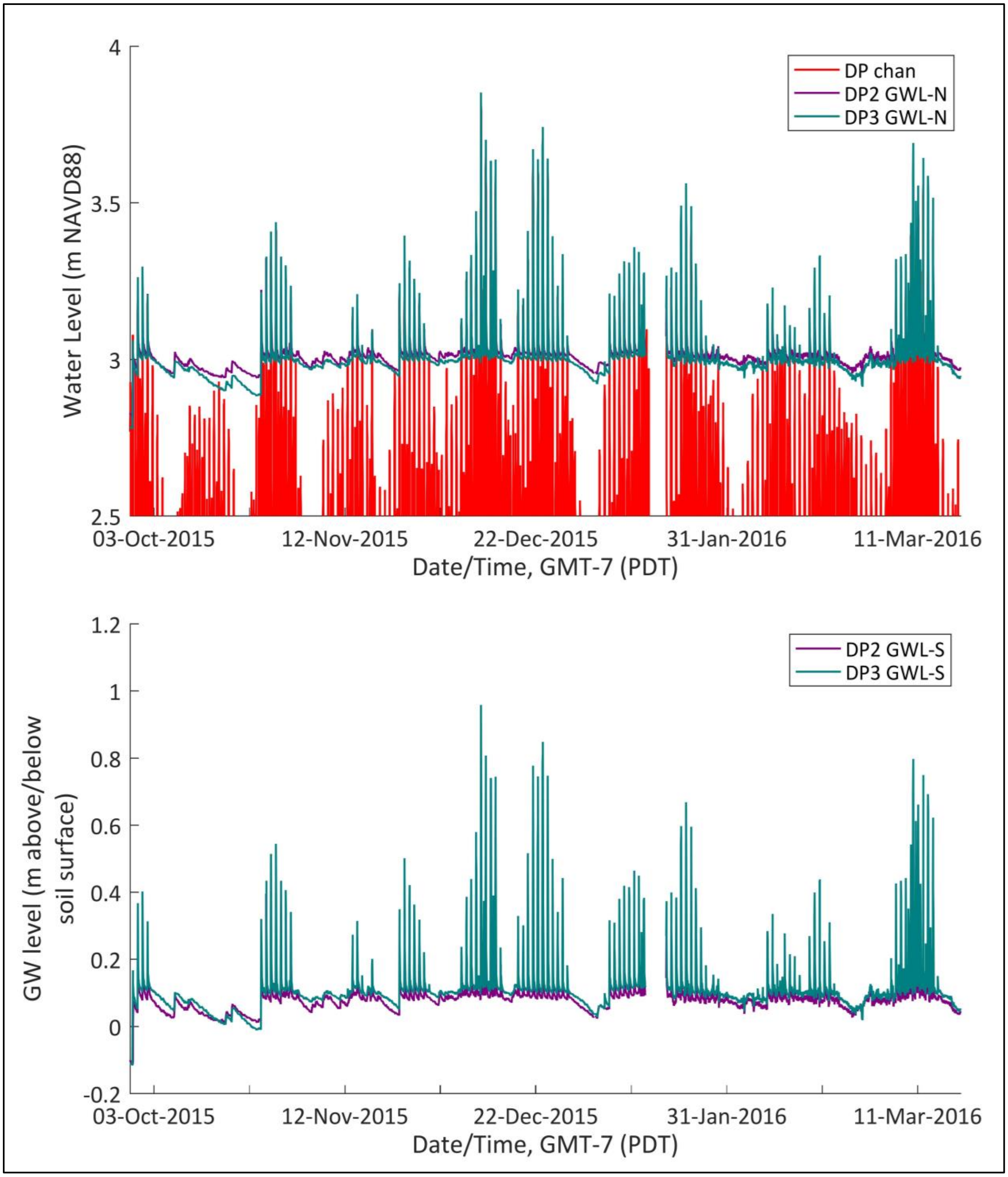

Figure A7. Channel water level versus groundwater level in high marsh and scrub-shrub tidal swamp at the Daggett Point reference site, winter 2015-2016. "DP chan" is the channel water level station; DP2 GWL-N is the high marsh groundwater station; and DP3 GWL-N is the shrub swamp groundwater station. The upper graph shows water levels on the NAVD88 elevation datum; the lower graph shows groundwater levels relative to the soil surface. Groundwater levels for station DP1 (low marsh) are not shown; they remained at or above the soil surface throughout the monitoring period, with peaks that matched the tide peaks.

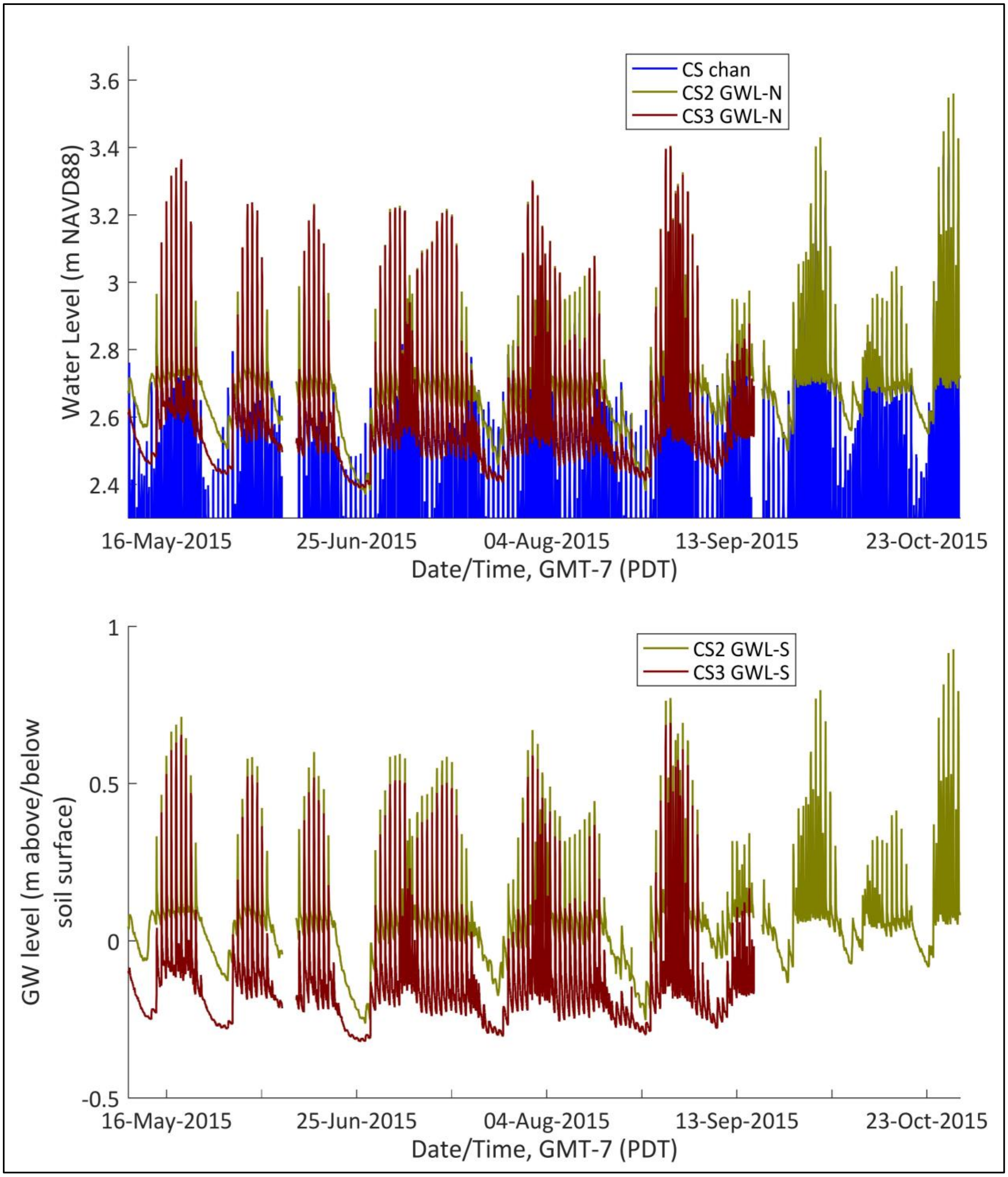

Figure A8. Channel water level versus groundwater level at the Cooperage Slough reference site, summer 2015. "CS chan" is the channel water level station; CS2 GWL-N is the high marsh groundwater station; and CS3 GWL-N is the shrub swamp groundwater station. The upper graph shows water levels on the NAVD88 elevation datum; the lower graph shows groundwater levels relative to the soil surface.

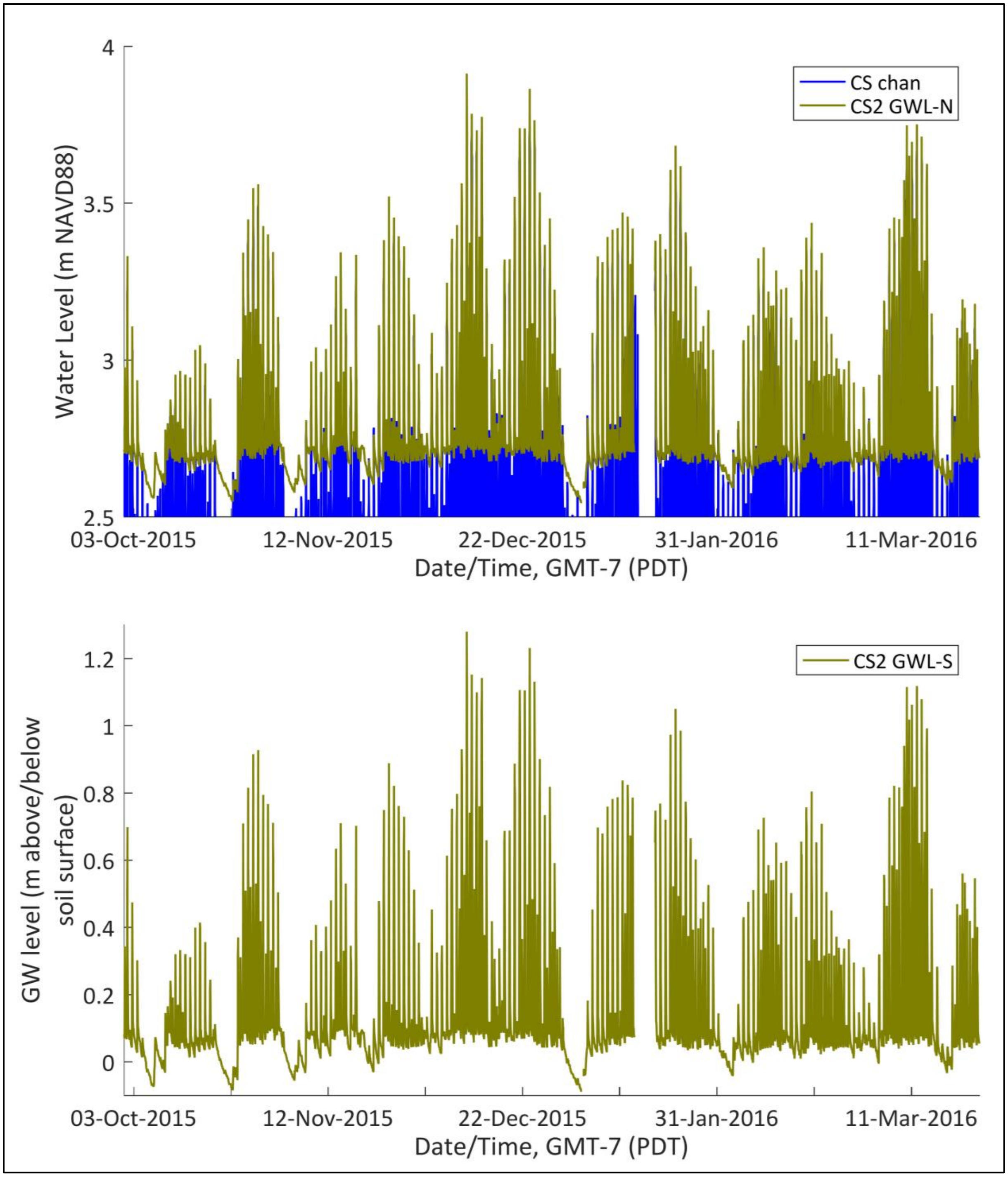

Figure A9. Channel water level versus groundwater level at the Cooperage Slough reference site in winter. "CS chan" is the channel water level station; CS2 GWL- N is the high marsh groundwater station. (No data are available from the CS3 shrub swamp groundwater well after September 2015, due to theft of the logger.) The upper graph shows water levels on the NAVD88 elevation datum; the lower graph shows groundwater levels relative to the soil surface.

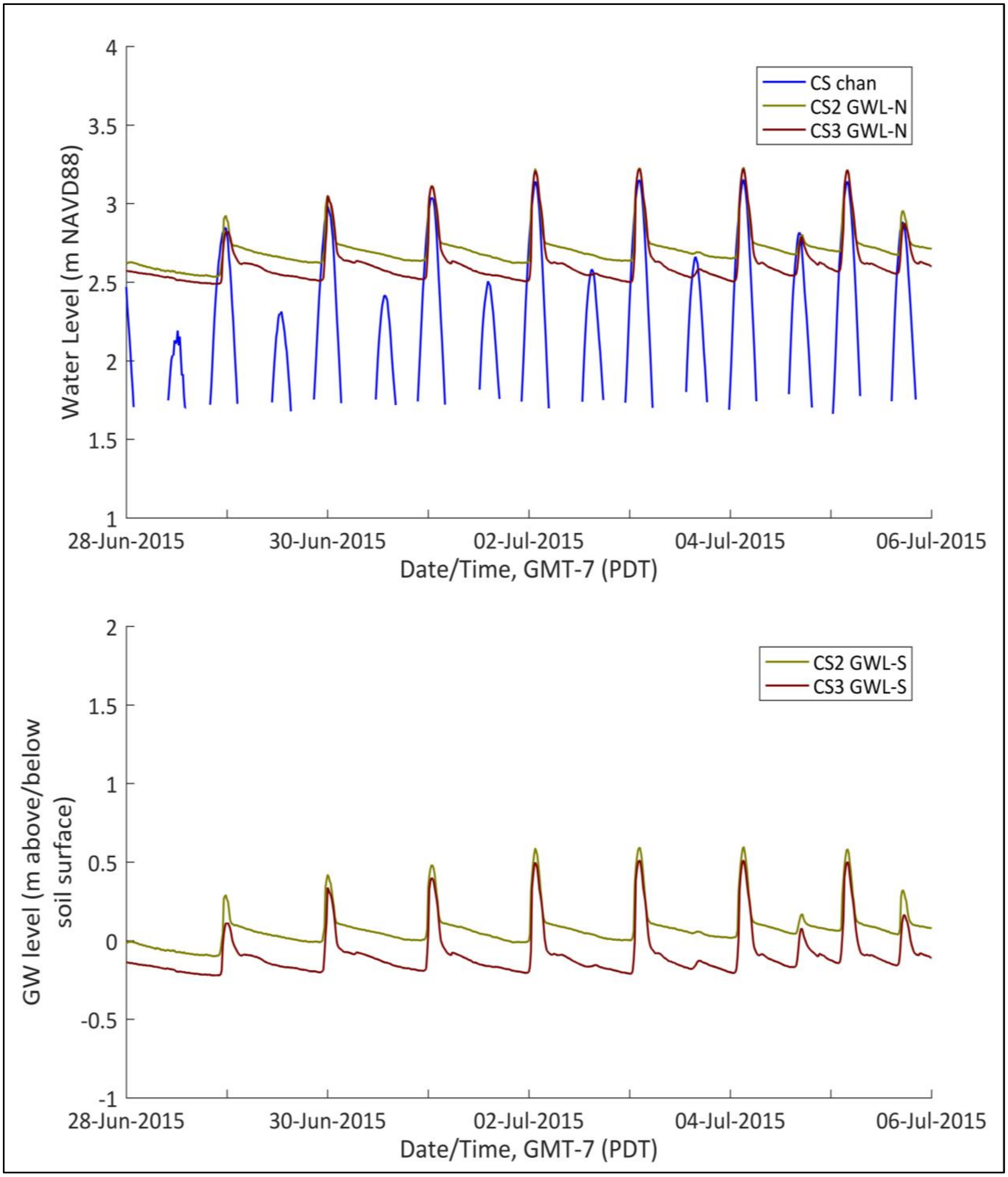

Figure A10. Channel water level versus groundwater level at the Cooperage Slough reference site for a short period in late June and early July 2015, illustrating the matching elevations of surface-inundating tide peaks and groundwater peaks. “CS chan” is the channel water level station; CS2 GWL-N and CS3 GWL-N are groundwater stations. The upper graph shows water levels on the NAVD88 elevation datum; the lower graph shows groundwater levels relative to the soil surface.

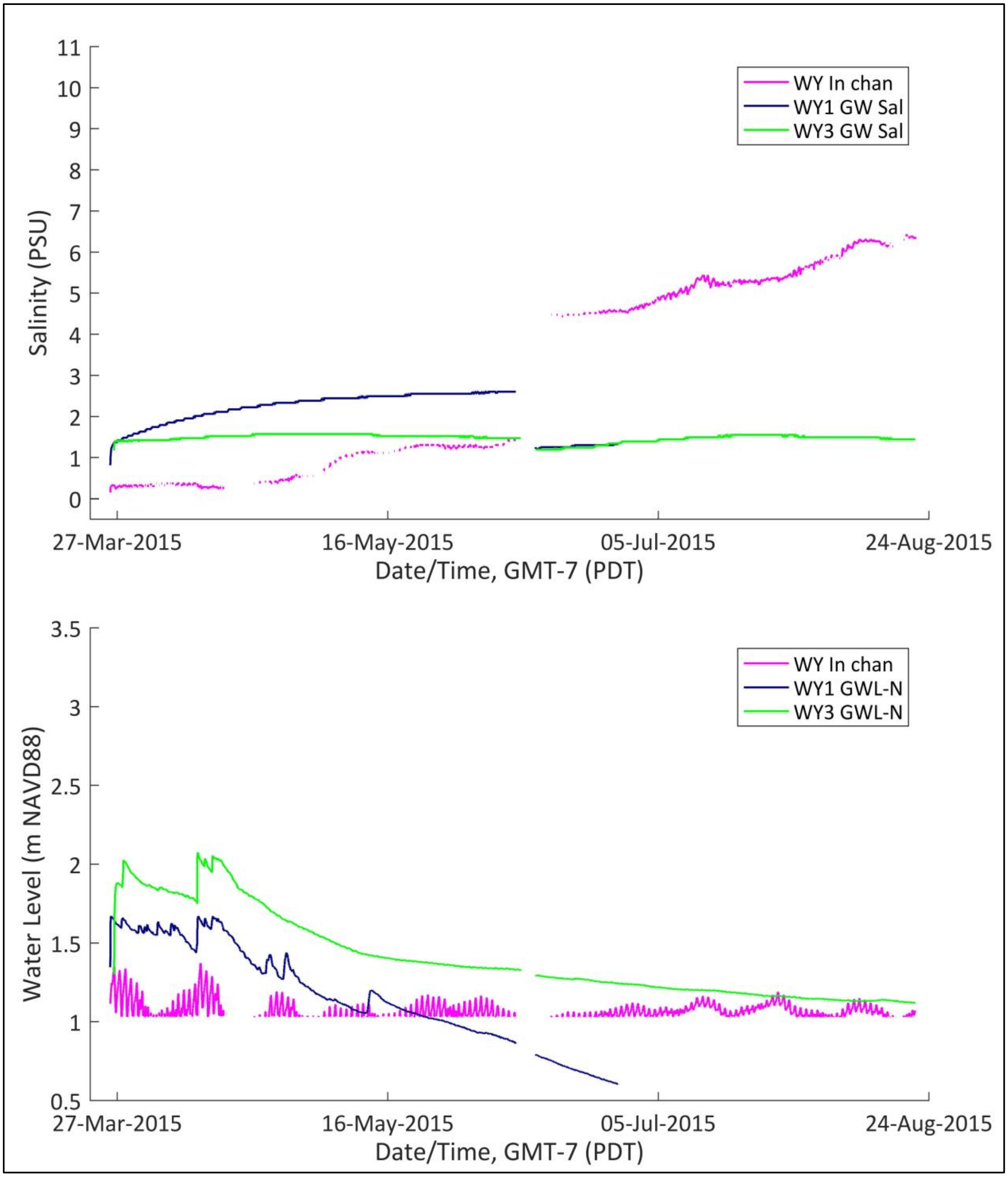

Figure A11. Salinity (top graph) and water level (bottom graph) for channel versus groundwater loggers at the Wallooskee site, for the full period of record at that site. "WY In chan" is the channel water level station inside the dike; WY1 GWL-N and WY3 GWL-N are groundwater stations.

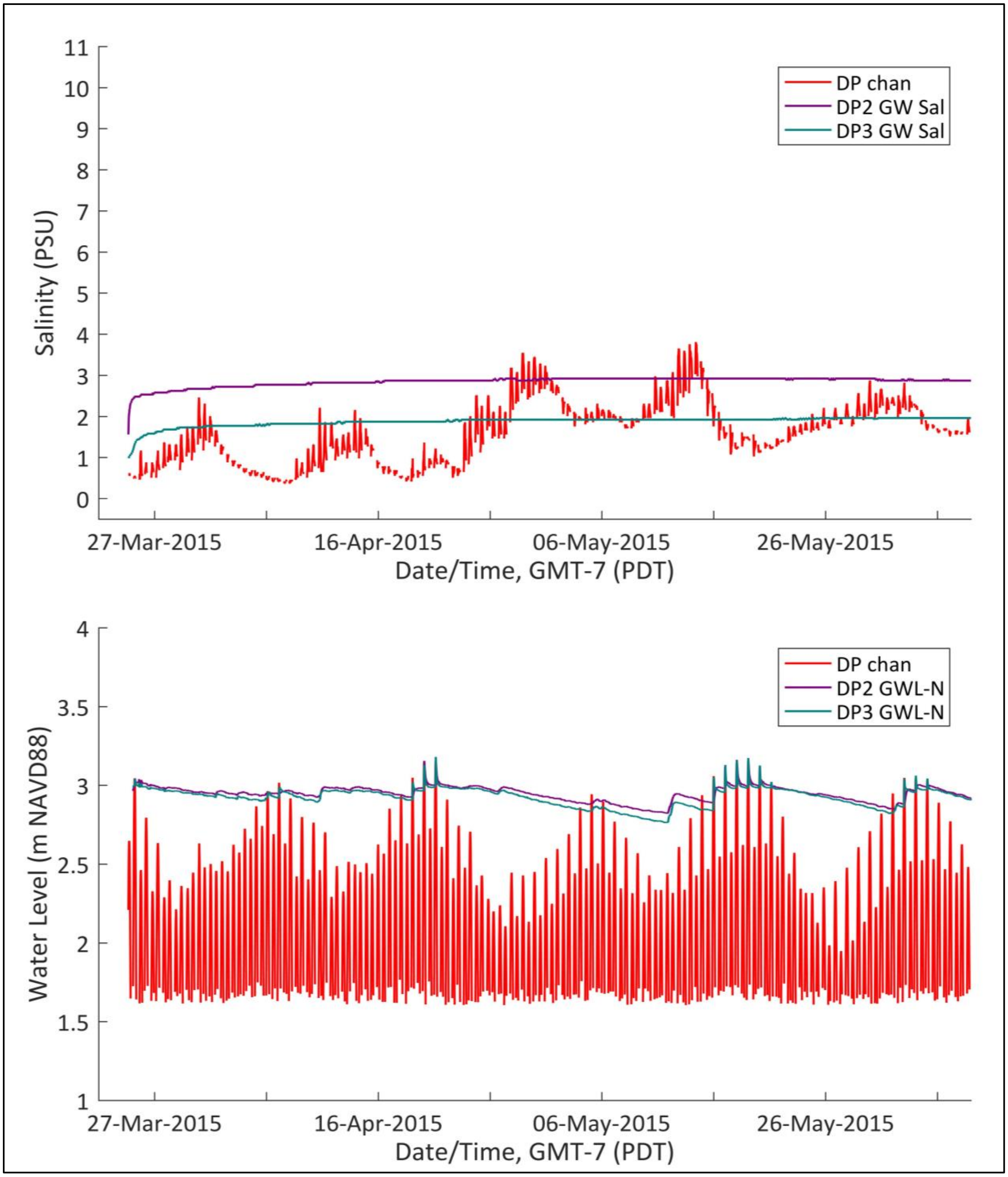

Figure A12. Salinity (top graph) and water level (bottom graph) for channel versus groundwater loggers at the Daggett Point reference site, spring 2015. “DP chan” is the channel water level station; DP1 GWL-N, DP2 GWL-N, and DP3 GWL-N are groundwater stations. The DP1 groundwater logger was not installed until June 2015.

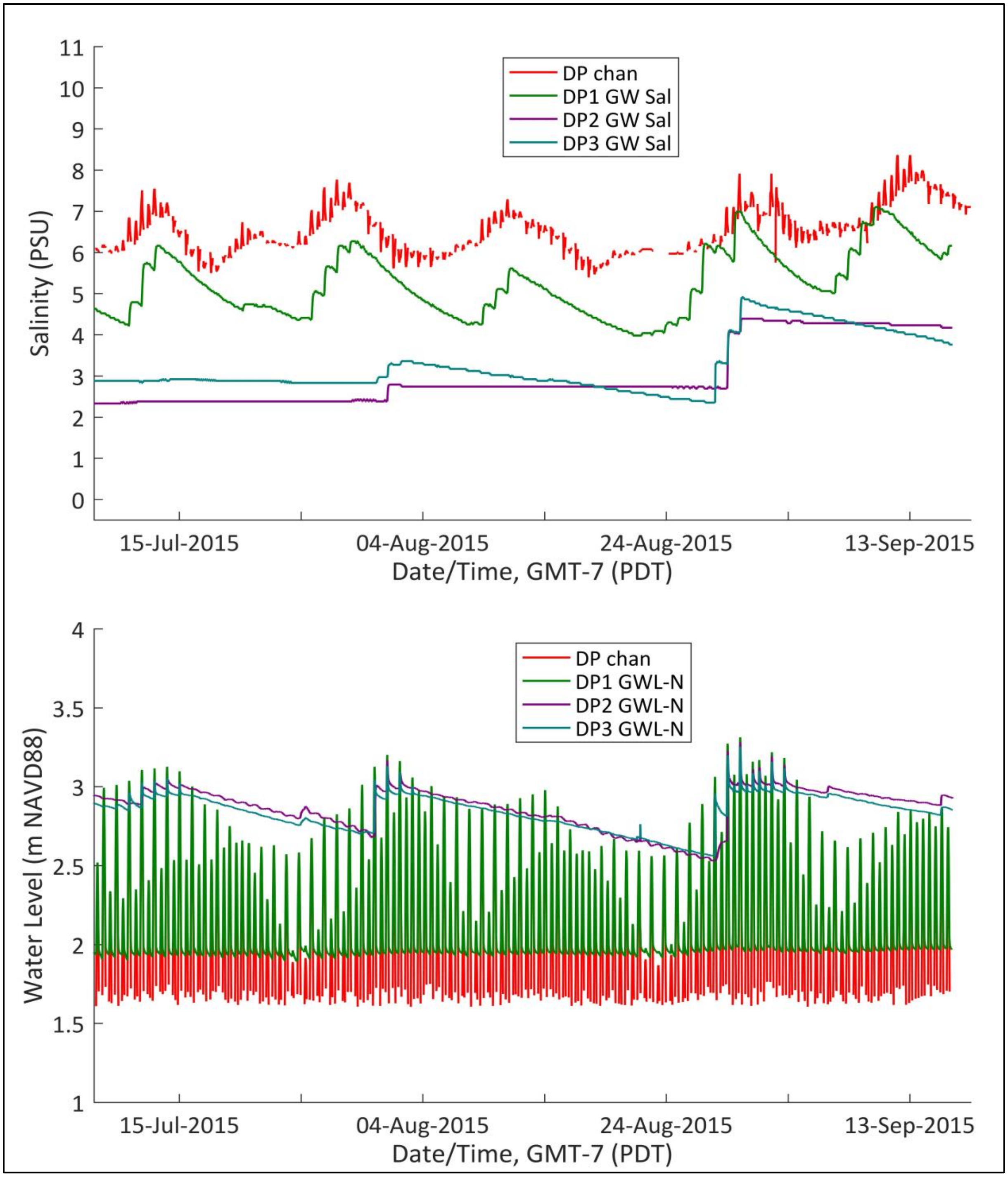

Figure A13. Salinity (top graph) and water level (bottom graph) for channel versus groundwater loggers at the Daggett Point reference site, summer 2015. "DP chan" is the channel water level station; DP1 GWL-N, DP2 GWL-N, and DP3 GWL-N are groundwater stations.

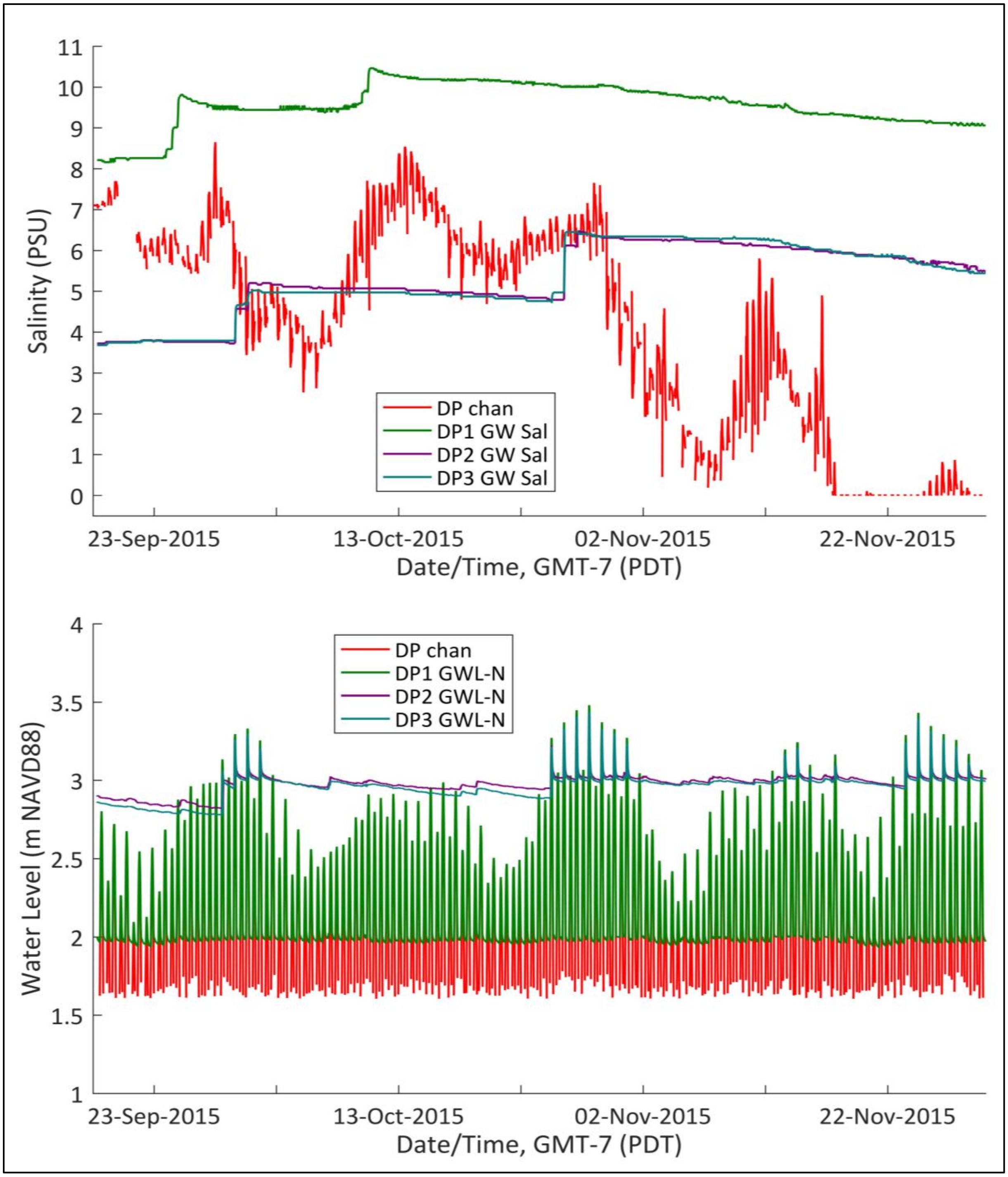

Figure A14. Salinity (top graph) and water level (bottom graph) for channel versus groundwater loggers at the Daggett Point reference site, fall 2015. "DP chan" is the channel water level station; DP1 GWL-N, DP2 GWL-N, and DP3 GWL-N are groundwater stations.

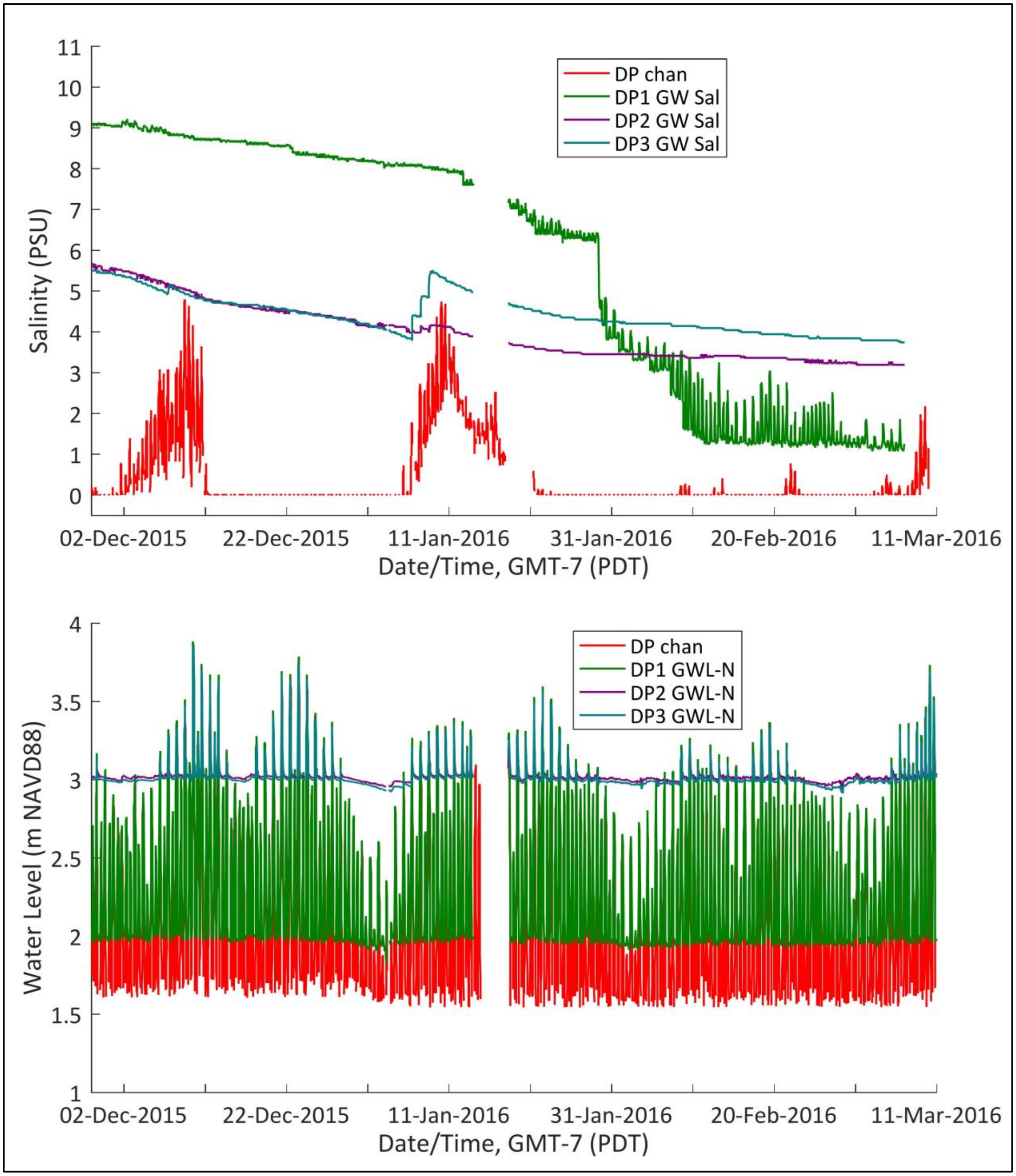

Figure A15. Salinity (top graph) and water level (bottom graph) for channel versus groundwater loggers at the Daggett Point reference site, winter 2015-2016. “DP chan” is the channel water level station; DP1 GWL-N, DP2 GWL-N, and DP3 GWL-N are groundwater stations.

Appendix 3. Spatial reference system

GPS data collected by ETG in support of monitoring at the Wallooskee site and reference sites (Daggett Point, Grant Island, and Cooperage Slough) was collected using the spatial reference system described in Table A1.

Table A1. Horizontal and vertical coordinate systems for ETG-collected GPS data

Horizontal Coordinate System	Universal Transverse Mercator (UTM) Zone 10 North
Horizontal Datum	North American Datum of 1983 (NAD83) Adjustment 2011 Epoch 2010.00
Vertical Datum	North American Vertical Datum of 1988 (NAVD88)
Geoid model	NGS Geoid 12A
Units	Meters

GPS/GNSS methods

Data was collected using a Spectra Precision ProMark 220 GNSS receiver outfitted with an Ashtech ASH111661 external GNSS antenna. The receiver collected both GPS and GLONASS L1/L2 signals at 1 Hz and received real-time kinematic corrections (RTK) from the Oregon Realtime GNSS Network (ORGN, http://theorgn.net) using a cellular data link. The receiver and antenna were mounted on an aluminum survey rod that was manually leveled or stabilized with a bipod during collection. The bottom of the survey rod was fitted with an 11 cm diameter topo shoe to prevent the survey rod from penetrating soft soil and mud. Typical occupation durations were 10 seconds for vegetation plots and general ground surface measurements. Local benchmarks, measurements of survey control, and sensor installations had a typical occupation time of 240 seconds or greater, often with multiple repeated measurements over multiple field campaigns.

Spatial data accuracy

Spatial data accuracy was calculated for each field campaign associated with this project following the National Standard for Spatial Data Accuracy (NSSDA) and repeated measurements of published NGS benchmarks near the project area. Typical absolute accuracies were 3.5 cm horizontal and 5.0 cm vertical at the 95% confidence level. Please contact the authors for more information.

Feet / meters conversion

ETG performed all analyses in meters and converted to feet when necessary for reporting. When converting to feet, we used the International Foot, which is equal to exactly 0.3048 m.