Ni-les'tun Tidal Wetland Restoration Effectiveness Monitoring: *Year 2 Post-restoration (2013)*

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This study was a joint effort of the Estuary Technical Group of the Institute for Applied Ecology and the Confederated Tribes of Siletz Indians.

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EXECUTIVE SUMMARY

Purpose: This report describes the results of effectiveness monitoring at the Ni-les'tun tidal wetland restoration site, Bandon National Wildlife Refuge, Coquille River estuary, Oregon. The monitoring described in this report was conducted during 2013, which was the 2nd year after the site's dikes and tide gates were removed, restoring tidal flows to the site. Effectiveness monitoring was designed to determine whether the project is meeting its goals, and to provide information to help guide other restoration projects. The results and "lessons learned" through monitoring at this landmark project are already helping to advance restoration science at many projects in Oregon, the Pacific Northwest, and beyond.

Who did the work: This study was a collaborative, multi-disciplinary effort of the Estuary Technical Group of the Institute for Applied Ecology and the Confederated Tribes of Siletz Indians. Laura Brophy led the monitoring of tidal hydrology, plant communities, groundwater, soils, and water temperature and salinity. Stan van de Wetering led the monitoring of fish use, macroinvertebrates, and fish use of wood habitats. All team members collaborated on channel morphology monitoring, and on analyses of linkages between physical and biological characteristics at the site.

Approach and presentation: To determine project effectiveness, we used a "before-after-control-impact" (BACI) approach, comparing the 2013 data to baseline (pre-restoration) data collected in 2010-2011 (or earlier) at Ni-les'tun and a local reference site. Ni-les'tun is a large site, and understanding patterns required sampling many locations, which generated a high volume of data. The main body of this report provides summaries, representative results, and interpretation. Further results are provided in the appendices, and additional data are available from the lead authors. Throughout this report, we focus on Year 2 post-restoration monitoring results, highlighting key comparisons to pre-restoration conditions. Further details on pre-restoration conditions are contained in the baseline monitoring report (Brophy and van de Wetering 2012).

Summary of results: Post-restoration monitoring in 2013 revealed many dramatic physical and biological changes since restoration, and a restoration trajectory that is moving towards conditions at the local reference site as well as a broader set of reference sites in Oregon. Some physical and biological conditions changed rapidly, while others appear to be changing more slowly – results which were expected and which are typical of restoration sites in general. Key findings are listed below.

Key findings

To jump to figures or tables illustrating key findings, click on the hyperlink (underlined text). Use the back arrow to return to the key findings section.

Tidal hydrology

1. The tidal inundation regime at Ni-les'tun was successfully restored to closely match the adjacent river and the Bandon Marsh Unit reference site. Average daily high tides inside the restoration site were within 9 cm (3.5 in) of those in the mainstem river, showing that the site has free tidal exchange.
2. Strong post-restoration increases in channel water salinity and soil salinity across all parts of the restoration site (from the Coquille River to North Bank Road) provide clear evidence that brackish tidal flows were quickly returned to the entire site.
3. After restoration, groundwater regimes showed strong tidal influence, indicating tidal flows were affecting belowground processes as well as surface inundation.
4. Post-restoration changes in channel morphology showed that restored tidal flows are influencing channel width, depth, and substrate configuration.
5. The Ni-les'tun project restored highly prioritized Sitka spruce tidal swamp as well as tidal marsh, as shown by tidal inundation patterns in the forested areas of Ni-les'tun.

Channel morphology

6. Channels across Ni-les'tun deepened and strong head-cutting occurred in lower channels as the channels equilibrated with the restored tidal action.
7. Fine sediment was present in all excavated channels at depths ranging from 5 to 23 cm. Fine sediment is important for fish prey production, and it was absent from excavated channels prior to tidal flow restoration. Fine sediment depths were greater in non-excavated channels.
8. Longitudinal gradients differed between the reference and restoration site, but the restoration site is expected to change in the direction of the reference site.

Vegetation

9. Plant communities on the restoration site changed substantially since baseline; species that could not tolerate the restored tidal inundation and brackish salinity decreased in cover and/or condition.
10. Analyzed across the entire site, the composition of plant communities at the restoration site appears to be converging with the reference site.
11. Cover of non-native species at Ni-les'tun declined significantly after restoration; total plant cover also declined, due to increased bare ground where species that could not acclimate to restored tidal inundation and salinity died back. Cover of bare ground is expected to be temporary as brackish-tolerant native tidal wetland species re-colonize the site.

Soils

12. Soil salinity increased significantly at Ni-les'tun following restoration, and was similar to the reference site.
13. Soil carbon content increased significantly after restoration, although it was still lower than that of the reference site.

Groundwater

14. The Ni-les'tun pasture was a seasonal wetland prior to restoration, with soils that dried during summer. After restoration, the entire site was a tidal wetland year-round, with groundwater that fluctuated in response to tide levels and precipitation.
15. After restoration, the groundwater regime in middle to high elevations at Ni-les'tun followed the "spring tide reset" pattern typical of natural tidal marsh sites.

16. Using the 22 groundwater wells as "peak tide gauges", we were able to see variations in tidal inundation patterns across these large restoration and reference sites such as delays in tide peaks at wells far from the river, and variability in maximum tide heights. This information will be used to improve goal-setting and interpretation of monitoring results at other projects.

Channel water temperature and salinity

17. Restoration led to a significant increase in channel water salinity at Ni-les'tun during the spring and summer months. Post-restoration salinity in summer was somewhat lower at Ni-les'tun compared to the reference site, probably due to the restoration site's freshwater inflows and its location further upstream.
18. During spring and summer post-restoration, the daily salinity regime was very dynamic in tidal channels with freshwater flow (Fahys Creek and Redd Creek). This expanded daily range of salinities may provide osmotic regulation opportunities for juvenile salmonids and other anadromous fish during critical spring and summer periods.
19. Salinities in blind channels at Ni-les'tun slightly exceeded salinities at the reference site during summer, despite their location further upstream.
20. Restoration was associated with significantly lower water temperature at Ni-les'tun compared to the reference site in the upper portions of the channels. In other areas, water temperatures were similar to the reference site.

Wood structures and channel morphology

21. Of the 193 wood structures placed in restored marsh channels, some were lost due to bank erosion. Channel reaches with lower wood density had less bank erosion and less loss of wood structures.
22. Channel reaches with wood structures showed more scour and fill of the substrate and bank, compared to channel reaches without wood structures. These processes helped create varied habitats, including low tide refugia scour pools and sediment bars.
23. Lower channel reaches (near channel mouths) showed more channel complexity in wood reaches compared to upstream reaches. This may have been due to greater tidal forcing in the downstream channels, or to the fact that the downstream channels were not excavated during restoration, so they had more fine sediment.

Salmonid habitat opportunity and suitability

24. Restoration led to dramatic increases in habitat opportunity (access) for migrating and non-migrating juvenile salmonids. Fish access to channels at Ni-les'tun was greatly enhanced by restoration, and was greater at Ni-les'tun than at the reference site.
25. Prior to restoration, salmonid rearing conditions (water temperature and salinity) were impaired by the site's dikes and tide gates. By Year 2 after restoration, temperature and salinity were close to reference conditions, resulting in increased duration of temperatures that met the Oregon Administrative Rules' salmonid rearing criteria (<18° C) in most locations.
26. New salinity regimes created new rearing opportunities for specific fish species and age classes during the key period of summer low flow.

Salmonid habitat capacity (prey resources)

27. Macroinvertebrates colonized the newly excavated channels at Ni-les'tun. The restored benthic macroinvertebrate communities showed abundance, diversity and community structure similar to the reference site.
28. Macroinvertebrates also colonized the non-excavated, pre-existing channels at Ni-les'tun.
29. The macroinvertebrate taxa that dominated at Ni-les'tun at Year 2 after restoration (primarily *Corophium* and polychaetes) are important prey for salmonids and Pacific staghorn sculpin, suggesting that the restoration provided enhanced salmonid foraging opportunities

30. At year two, the restored marsh benthic macroinvertebrate communities were more diverse than those observed in the mainstem Coquille River habitats.

Salmonid habitat use

31. Restoration resulted in significant increases in use of key rearing habitats by age 0 chinook, age 1 coho, staghorn sculpin, and other fish species and age classes.
32. Rates of fish use of restored habitats increased both within and across seasons for age 0 chinook, age 1 coho, staghorn sculpin, and other fish species and age classes.
33. Tidal migration increased after restoration for age 0 chinook and age 1 coho.
34. About 300 age-0 coho reared in an oligohaline intertidal beaver dam pool on the restoration site. Intertidal beaver dam pool habitat is a key habitat that likely occurred at a greater rate prior to European settlement. Intertidal beaver dams and pools were also present at the Bandon Marsh Unit reference site.
35. Based on peak month catch data, we estimate the Ni-les'tun restoration site produced 6022 Chinook smolts during 2013. If ocean survival is assumed to be 1.5%, a logical conclusion would be that the Ni-les'tun restoration resulted in 90 additional adult Chinook spawners.

Ecological linkages and other results

36. Percent inundation was significantly correlated with physical and biological site characteristics such as soil salinity and plant species richness. These relationships indicate that Ni-les'tun is developing in response to the restoration of tidal flows, evidence of effective restoration.
37. Many other species besides the target species, such as surf smelt, anchovy, crangon shrimp, and larval Dungeness crab and bay pipefish, were observed using the restoration site; these were absent prior to restoration.
38. Effectiveness monitoring requires an understanding of natural changes over time at local sites. Before-After-Control-Impact (BACI) sample design and analysis is a framework that improves understanding of these changes and the effects of restoration. BACI allowed us to detect significant changes associated with restoration that would otherwise have been masked by year-to-year variability.

Lessons learned from monitoring at Ni-les'tun

In addition to a summary of results and interpretation of the project's effectiveness based on those results, the following "lessons learned" can help guide other monitoring and restoration projects.

1. Channel excavation practices affect the physical evolution of a restoring tidal wetland. Tidal channels at Ni-les'tun were excavated relative to the wetland surface, rather than relative to a fixed datum (geodetic or tidal). The wetland surface method is less expensive and less time-consuming, but led to some unnatural gradients. Channels will adjust over time as they equilibrate with tidal forces, but this adjustment will take longer than if channels had been excavated relative to a fixed datum. Therefore, we recommend excavation of channels relative to a fixed datum.
2. Appropriate on-site monumentation and high-accuracy elevation surveys are necessary to carefully track channel evolution. Channel morphology monitoring methods at Ni-les'tun included RTK-GPS "as-built" survey at baseline, and a combination of RTK-GPS survey and rapid cross-sections (survey chain and rod) during post-restoration. This combination of methods allowed collection of a much larger dataset than would otherwise have been possible, but created some challenges in data alignment and interpretation. We recommend a consistent, combined protocol be applied during baseline and post-restoration periods, using RTK-GPS (to

tie channel measurements to the geodetic datum), laser level (to measure channel depths at intervals), and permanently monumented channel cross-section endpoints (see next bullet point). This protocol maximizes efficiency and information return.

3. Permanent monuments established at channel cross-section endpoints during baseline monitoring would enhance tracking of post-restoration changes in channel morphology. Due to cost, logistics, and monitoring methodologies, we did not establish permanent monuments at this site, but they are recommended for other projects.
4. Because of the high level of variability within a single tide cycle, across monthly tide cycles, and across seasons, automated dataloggers are needed to understand salinity and temperature patterns at tidal wetland restoration sites.
5. When deploying and operating salinity/temperature dataloggers, careful placement and field validation are critical to interpretation of results. Salinity logger placement in tidal channels should be low enough to maximize immersion, but high enough to avoid sediment deposition that could clog logger orifices and distort conductivity readings. Field validation is important for detection of sediment deposition and other problems, and a systematic protocol for inspection of installations is needed.
6. Shorter data collection intervals (15 min instead of the 30 min interval we used) make field validation of salinity and temperature data much easier, but these create much larger datasets.
7. Salinity/temperature datalogging during limited, key time periods (e.g. the driest part of the summer and critical periods for fish use) may reduce costs while still allowing understanding of important patterns.
8. Collection of mainstem river bank fish migration data greatly improved our understanding of factors that drove individual sub-basin fish migration patterns.
9. Effectiveness monitoring requires an understanding of natural changes over time at local sites. Before-After-Control-Impact (BACI) sample design and analysis improves understanding of these changes and the effects of restoration. BACI allowed us to detect significant changes associated with restoration that would otherwise have been masked by year-to-year variability.
10. The broad effectiveness monitoring program at Ni-les'tun produced results that are proving extremely important for guidance at other tidal wetland conservation and restoration projects in Oregon and beyond. A few examples include:
    a. Peak tides measured at the upslope edges of Ni-les'tun and the reference site were the most important data supporting and validating the State of Oregon's updated estuarine habitat mapping methods (Lanier *et al*. 2014).
    b. Groundwater wells serving as "peak tide gauges" showed delays in tide peaks across the large restoration site, and lower peaks at the farthest wells during summer. These differences are expected at large sites, but this project is the first in Oregon to provide quantitative data on this topic. This information will be used to set appropriate performance targets for other projects such as the Southern Flow Corridor project in the Tillamook estuary (http://tillamookoregonsolutions.com).
    c. Lessons learned in channel morphology methods, such as use of permanent monuments at channel cross-section endpoints, are being applied at other monitoring sites in Oregon, including the Southern Flow Corridor project in the Tillamook estuary (http://tillamookoregonsolutions.com) and the Waite Ranch project in the Siuslaw estuary (http://www.siuslaw.org/waite-ranch-tidal-wetland-restoration-project/).
11. The relationship between wood diameter and wood buoyancy during high water events is critical for successful placement. Based on observations at Ni-les'tun, we suggest using logs with a maximum diameter of 0.6 m and with 4.6 – 6.0 m of stem that can be pushed into

channel bank top soils. Calculation of soil holding strength is also recommended, and could result in different diameter/length recommendations.

12. In areas where wood was placed at higher densities, tidal flows forced around the structures caused bank scour and eventual loss of the anchoring mechanism. Therefore, we suggest using less complex wood structures to help form low tide refugia (scour pools), rather than complex log jams like those used in freshwater stream restorations.
13. Prior to restoration, mosquito production was not known or suspected to be an issue and therefore, monitoring of mosquitoes was not included in our effectiveness monitoring program. To help control mosquitoes and eliminate mosquito-breeding habitat at Ni-les'tun, USFWS plans to excavate 40,000 additional linear feet of first- and second-order tidal channels during the summer of 2014. This additional channel restoration will further enhance tidal wetland functions at Ni-les'tun, particularly salmonid habitat availability. Other USFWS activities associated with mosquito control at Ni-les'tun, including mosquito monitoring, are described at http://www.fws.gov/oregoncoast/bandonmarsh/Mosquito.html.

REPORT ORGANIZATION: RESTORATION AND MONITORING OBJECTIVES

This report is organized by the “big picture” **restoration and monitoring objectives** listed below. Restoration objectives are those listed in the OWEB restoration grant proposal (DU 2009). Each monitoring objective contains several specific **monitoring questions**, which were answered by measuring **monitoring parameters**. This report contains results, interpretation and comparison to other projects.

Restoration Objective 1: Restoration of coastal tidally influenced wetlands through hydrological reconnection

Monitoring Objective 1: Measure restoration of tidal hydrology, tidal wetland vegetation, and the physical attributes that control tidal wetland functions across the 418-acre marsh.

Monitoring Questions:

Q1a) Was tidal hydrology successfully restored?

Parameters: Tidal hydrology (inundation frequency and duration) at restored and reference sites; elevation of wetland surface and instrumentation; tidal channel morphology (cross-sections, longitudinal sections, length); groundwater regime; channel water and soil salinity.

Q1b) Are tidal wetlands developing, with physical and biological characteristics trending towards least-disturbed reference conditions?

Parameters: Wetland plant community composition and extent; soil characteristics (stored organic carbon, salinity, pH); groundwater levels; surface water salinity and temperature.

Restoration Objective 2: Restoration of coastal and marine habitat to recover listed and at-risk species, particularly estuary dependent and anadromous fishes

Monitoring Objective 2: Measure habitat recovery and habitat use by estuary-dependent and anadromous fishes, including at-risk and and endangered species.

Monitoring Questions:

Q2a) Did restoration result in increased salmonid habitat opportunity (availability)?

Parameters: Duration and frequency of salmonid habitat availability (using channel morphology measurements and tidal elevations); surface water salinity and temperature; locations, quantities, and descriptions of large wood habitat restored.

Q2b) Did restoration result in increased salmonid habitat capacity?

Parameters: Benthic macroinvertebrate abundance and community structure within the largest of the three restored basins (Fahys Creek).

Q2c) Did restoration result in increased salmonid habitat use?

Parameters: Residency patterns (occupancy rates, catch per unit effort [CPUE], and CPUE marsh to reference ratios), tidal migration patterns, and salmonid use of large wood habitats.

Restoration Objective 3: Improve coastal resiliency to storms, flooding and climate change

Monitoring Objective 3: Measure extent of resiliency to storm-related flooding and climate change.

Monitoring Questions:

Q3a) Did restoration improve the site’s capacity to moderate storm-related flooding?

Parameters: Restored channel morphology, tidal hydrology, and inundation regime.

Q3b) Do post-restoration site conditions show potential for improved resilience to climate change?

Parameters: Plant community composition and extent; soil characteristics (% organic matter, pH, and salinity); groundwater levels.

PROJECT TIMELINE

The timeline for the Ni-les'tun tidal wetland restoration project extended across several years. Major tidal wetland restoration and monitoring activities are listed in Table 1. Information on the timing of other activities on the Refuge is available from Bandon Marsh National Wildlife Refuge.

Table 1. Dates of major tidal wetland restoration and monitoring activities at the Ni-les'tun site.

Year	**Restoration activities**	**Monitoring activities[2]**
2003[1]	• None	• Emergent wetland plant communities • Forested wetland plant communities • Soils
2005[1]	• None	• Low tide fish density • Juvenile salmonid tidal migration
2007[1]	• None	• Benthic macroinvertebrates
2009	• Removal of livestock • Excavation of a few restored tidal channels	• None
2010	• Excavation of most restored tidal channels • Ditch filling (major ditches) • Ditch disking (minor ditches)	• Tidal hydrology • Channel morphology • Emergent wetland plant communities • Groundwater (emergent wetlands) • Soils • Low tide fish density • Juvenile salmonid tidal migration • Benthic macroinvertebrates
2011	• Excavation of the last few restored tidal channels • Filling of lower Fahys Creek ditch • Completion of E and W protection dikes • Dike removal (final removal: 8/18/11) • Tide gate removal (final removal: 8/18/11)	• Tidal hydrology • Groundwater (emergent wetlands) • Forested wetland plant communities • Groundwater (forested wetlands) • Surface water temperature and salinity
2013	• Pilot tests of methods for restoring tidal channels to small pools (mosquito breeding sites)	• Tidal hydrology • Channel morphology • Emergent wetland plant communities • Forested wetland plant communities • Groundwater(emergent & forested wetlands) • Soils • Surface water temperature and salinity • Low tide fish density • Juvenile salmonid tidal migration • In-stream habitat • Wood and non-wood habitat use by fish • Benthic macroinvertebrates

[1] 2003, 2005 and 2007 monitoring activities were supported by non-OWEB funding.

[2] Only monitoring activities by our team are listed here. Several other groups are conducting research and monitoring at Ni-les'tun; further information is available from Bandon Marsh NWR.

METHODS OVERVIEW

As described above, this report is organized by monitoring objectives; methods are described briefly under each objective, and summarized in Table B1 (Appendix B). To provide context, sampling locations are described below. Methods were designed for comparability with other projects, and the methods meet regional and national standards for science-based effectiveness monitoring of tidal wetland restoration projects (Rice *et al.* 2005, Roegner *et al*. 2008, Thayer *et al*. 2005, Simenstad *et al*. 1991). Further information on methods is available in the baseline report (Brophy and van de Wetering 2012) and from the authors.

Sampling locations

Sampling was conducted at two sites within the Bandon Marsh NWR: the Ni-les'tun restoration site and the nearby Bandon Marsh Unit, a least-disturbed reference site (Appendix A, Figure A1). Sampling was stratified and distributed across all tidal wetland elevation zones and all sub-basins, including Fahys, NoName, and Redd Creek sub-basins at Ni-les'tun, and the Shipwreck and Unknown sub-basins at the Bandon Marsh Unit reference site (Appendix A, Figure A1). Sampling of vegetation, soils, and groundwater was conducted within 22 study transects strategically placed to sample major plant communities and the associated physical and biotic conditions (Appendix A, Figures A2 and A3); transects spanned the elevation range present within the site (Appendix B, Table B2). Within each transect, sampling of vegetation was randomized; groundwater was measured in a central shallow observation well 1.2m (4ft) deep, and soil samples were bulked across the entire transect. Tide gauges were deployed in the mainstem Coquille River near the mouth of Fahys Creek, inside the site in lower Fahys Creek, and in a more central location in the site in Channel 7 (Appendix A, Figure A2). To characterize salinity and water temperature across the large study area, 17 salinity/temperature loggers were deployed: one in the Coquille River opposite the mouth of Fahys Creek, 11 in tidal channels at the restoration site, and 5 in tidal channels at the reference site (Appendix A, Figures A2 and A3). Channel cross-sections were measured at 311 locations on the restoration site and 11 locations at the reference site (Appendix A, Figures A4 and A6), and flowpath elevation was determined at 335 additional locations on both sites (Appendix A, Figures A5 and A7). These channel morphology data were compared to the pre-restoration "as-built" survey (Appendix A, Figure A8). Sampling of fish and macroinvertebrates was distributed across sub-basins and elevation zones (Appendix A, Figures A9 and A10).

Baseline monitoring was conducted primarily in 2010, but was supplemented by data collected in 2003-2007 under other funding sources. This repeated sampling provided valuable perspective on site dynamics and change. Monitoring parameters in 2003-2007 included vegetation, soils, benthic macroinvertebrates, low tide salmonid density and distribution, and salmonid migration. Details of specific transects sampled in 2003 *versus* 2010 can be found in the baseline monitoring report (Brophy and van de Wetering 2012).

Post-restoration sampling was conducted at the Ni-les'tun restoration site and at the Bandon Marsh Unit reference site nearby during 2013. Transect locations at the reference site were selected based on pre-disturbance conditions at Ni-les'tun (high marsh). Pre-disturbance elevation data are not available for Ni-les'tun (nor for our coastal wetlands in general). However, based on historic vegetation and relative elevations of the restoration and reference sites, it is likely that about 30 cm (1 ft) of subsidence has occurred across much of Ni-les'tun (Brophy and van de Wetering 2012). Subsidence may have been greater on the site's lowest portions (near the mouth of Fahys Creek).

METHODS AND RESULTS BY MONITORING OBJECTIVE

This section presents methods and results organized by monitoring objective and metrics. Throughout this report, we focus on Year 2 (2013) post-restoration monitoring results, highlighting key comparisons to pre-restoration conditions. Further details on pre-restoration conditions are contained in the baseline monitoring report (Brophy and van de Wetering 2012).

The term “restoration” can have several meanings. It is often used to refer to the on-the-ground actions taken to reverse impacts due to human activities. “Restoration” can also refer to the ecological process of recovery that follows those actions. In this report, even though we recognize that recovery will take many years, we use the term “post-restoration” to refer to the period after final removal of the dikes and tide gates at Ni-les’tun in fall 2011.

1. Tidal wetland restoration

Monitoring Objective 1: Measure tidal wetland restoration

In this objective, we measured the restoration of tidal hydrology, tidal wetland vegetation, and the physical attributes that control tidal wetland functions across the 418-acre marsh.

1a. Tidal hydrology

Monitoring Question 1a: Was tidal hydrology successfully restored?

Metrics: Tidal hydrology (inundation frequency and duration) at restored and reference sites; elevation of wetland surface and instrumentation; tidal channel morphology (cross-sections, longitudinal sections, length); groundwater regime; channel water and soil salinity. *(Rationale: Tidal hydrology is a major controlling factor for biological and physical characteristics and processes in tidal wetlands. Elevation measurements allow linkage of tide heights to physical and biological site characteristics; tidal channel morphology strongly affects water movement across a large tidal wetland. Channel morphology data will also be used to quantify salmonid habitat availability. Groundwater regime, channel water salinity, and soil salinity provide corroboration of tidal influence.)*

Methods

Tidal water levels were measured using automated water level loggers (Onset HOBO® loggers, model U20-001-01) programmed to collect pressure data at 15min intervals. During baseline monitoring (2010) and Year 2 post-restoration monitoring (2013), water level loggers were located in the lowest part of the Fahys Creek channel (southwest portion of the site, Lower Fahys TG2), and in the nearby mainstem Coquille River (CoquilleR TG2) (Appendix A, Figure A2). During 2013, an additional water level logger was installed further inside the restoration site (NL Ch7 TG) as a backup, since drift logs threatened the Lower Fahys installation. This interior gauge also allowed us to detect differences in tidal hydrology patterns within the restoration site.

Water level data were collected from April 11, 2011 to November 17, 2011 during the pre-restoration period, and from May 9, 2012 to September 26, 2013 during post-restoration. The raw data (pressure values) were converted to water levels using HOBOWare Pro® software; the conversion included adjustment for barometric pressure (using local barometric pressure data) with HOBOWare Pro® software’s barometric compensation assistant. To extend our ability to compare pre- and post-restoration tidal hydrology, we also obtained water level data for 2009 (both inside and outside the restoration site) from Ducks Unlimited (Randy Van Hoy, personal communication). The 2009 data were collected using Global® water level loggers (model WL-16), which automatically compensate for barometric pressure variations.

Tidal datums were calculated from the water level data using the direct method with the Charleston NOAA gauge as the master station (NOAA/NOS 2003). Percent inundation was calculated for all sample transects, and for representative channel locations to determine fish access (habitat availability) (Appendix A, Figure A11). Data from the Coquille River gauge (CoquilleR TG2) were used for percent inundation and fish access calculations, since the gauges inside the restoration site were out of water during low tides (and since tidal datums were very similar for all three tide gauges). Daily tide height maxima were extracted from the data for the maximum period of overlap between pre- and post-restoration data (1/1/09-9/26/09 and 1/1/13-9/26/13). Within the period of overlap, Before-After-Control-Impact (BACI) analysis (2 way analysis of variance, with site and year as independent variables) was used to determine whether restoration significantly affected maximum tide height. Analysis of variance (ANOVA) was conducted in R using the general linear models ("glm") package; significant differences between means were determined using the "lsmeans" package within general linear models, and displayed with the compact letter display of pairwise comparisons ("cld") routine.

Groundwater wells provided additional data on water levels during tidal inundation events (see **Groundwater** below). The loggers in these wells measured groundwater levels, but also measured surface inundation levels during inundation events. This use of the groundwater logger data gave us a high-resolution spatial array of 22 "peak tide gauges" – 17 at the restoration site and 5 at the reference site (Appendix A, Figures A2 and A3).

Results and discussion

Tidal inundation was successfully restored at the Ni-les'tun restoration site (Figures 1 and 2). BACI analysis showed a significant effect of restoration on mean daily maximum water level, which was 1.29 m (4.2 ft) during 2009 (nontidal, pre-restoration) and rose to 2.09 m mean daily maximum tide (6.9 ft) during Year 2 post-restoration (Figure 3; Appendix B, Table B4). Although means comparison (Appendix B, Table B5) showed that post-restoration mean daily tidal maximum was slightly lower at Ni-les'tun compared to the reference site (2.09 m *versus* 2.15 m), this 6 cm difference is not ecologically significant, and may represent normal spatial variation in tidal amplitude within the estuary. Average daily higher high tides (MHHW) for the two internal tide gauges were within 9 cm (3.5 in) of mean higher high water at the mainstem river gauge, showing that the site has free tidal exchange (Appendix A, Figure A12).

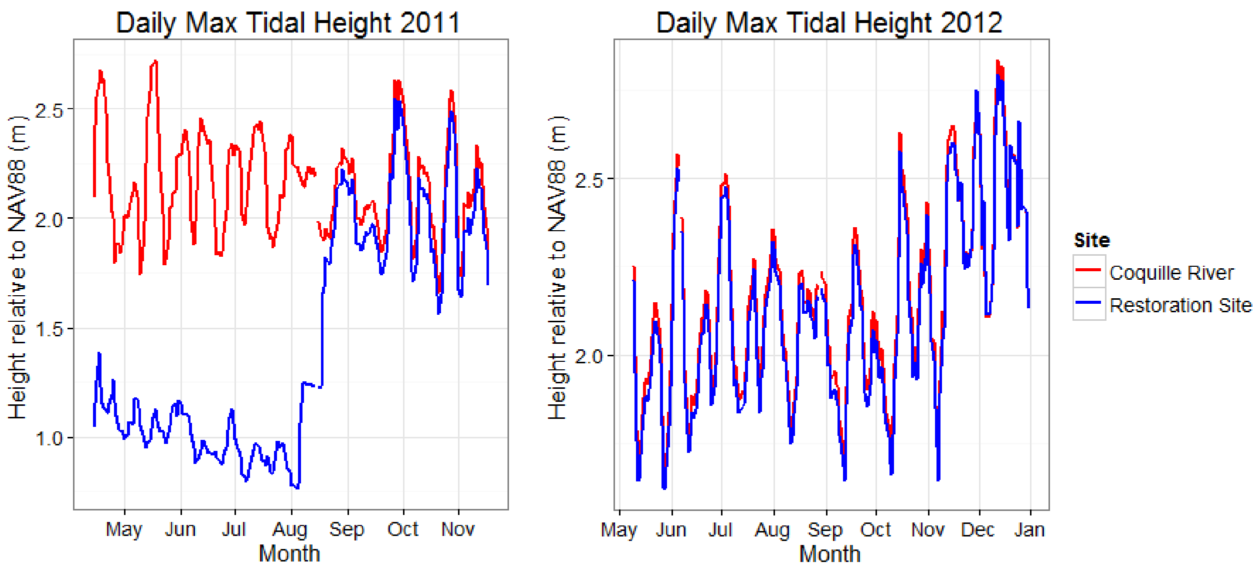

Figure 1. Pre-restoration and post-restoration daily maximum tide heights for the Ni-les'tun restoration site, compared to the adjacent Coquille River. Restoration occurred on August 17, 2011. Data are from tide gauges at Lower Fahys TG2 and CoquilleR TG2 (Appendix A, Figure A2)

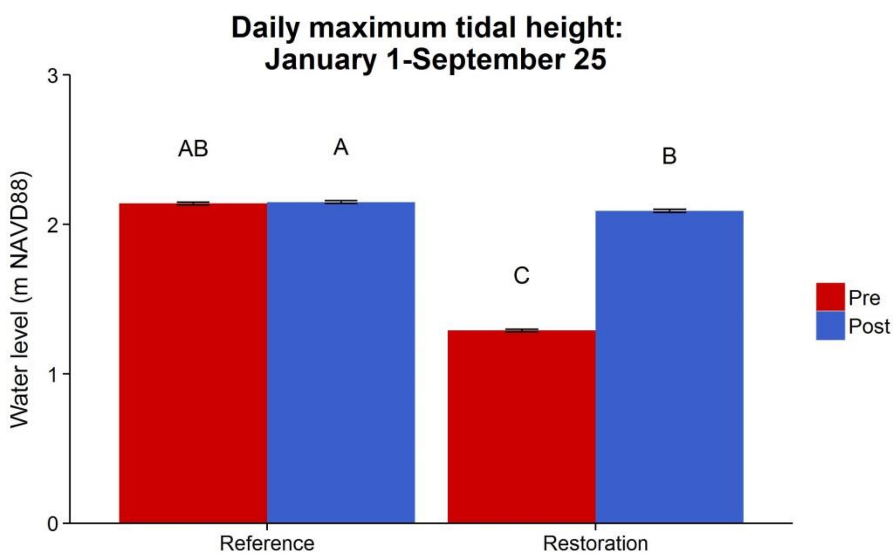

Figure 2. Daily maximum water level (pre-restoration) and daily maximum tidal height (post-restoration) during January through September at the restoration and reference sites. Pre-restoration data are from 2009, post-restoration data from 2013. Bars with a letter in common are not significantly different.

Tides were almost completely excluded from Ni-les'tun prior to restoration, but after restoration, tidal inundation extended across the entire site, with percent inundation ranging from 1% to 31% depending on the surface elevation of the wetland (Figure 3). Before restoration, the two lowest transects on Ni-les'tun (NL T2 and NL T18) had a muted tidal regime, due to limited tidal exchange through the fish-friendly tide gate. During this period, these two transects were inundated less than 5% of the time. After restoration, these transects were inundated about 30% of the time. Percent of time inundated for the reference transects was similar in 2010 and 2013.

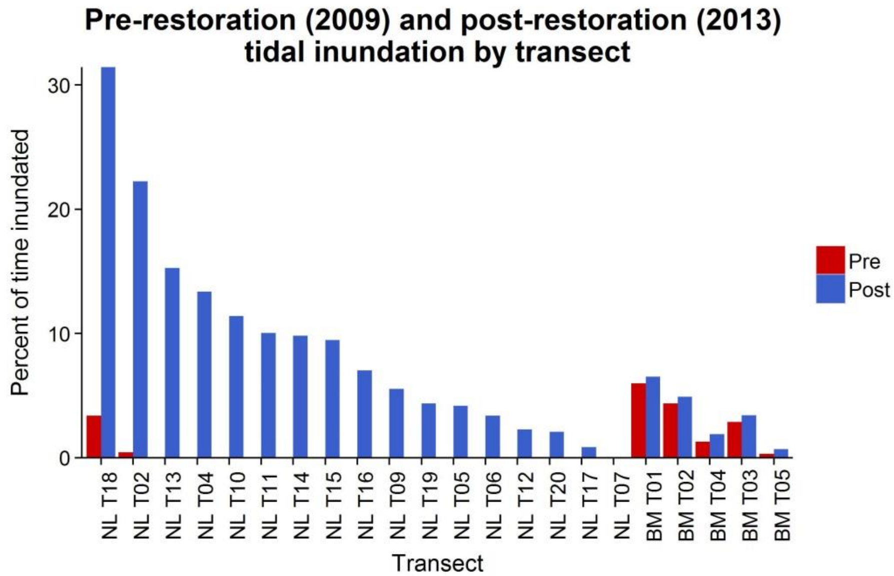

Figure 3. Percent inundation before (2009) and after restoration (2013) for sample transects at Ni-les'tun (NL) and the Bandon Marsh Unit reference site (BM). Transects are ordered by ascending elevation within each site, with NL T18 and BM T01 having the lowest elevation, NL T07 and BM T05 the highest.

Tidal inundation occurred year-round at the restoration site, but percent inundation was higher in winter than in summer (Figure 4). This pattern is typical for Pacific Northwest tidal wetlands (Seliskar and Gallagher 1983, Brophy *et al.* 2011).

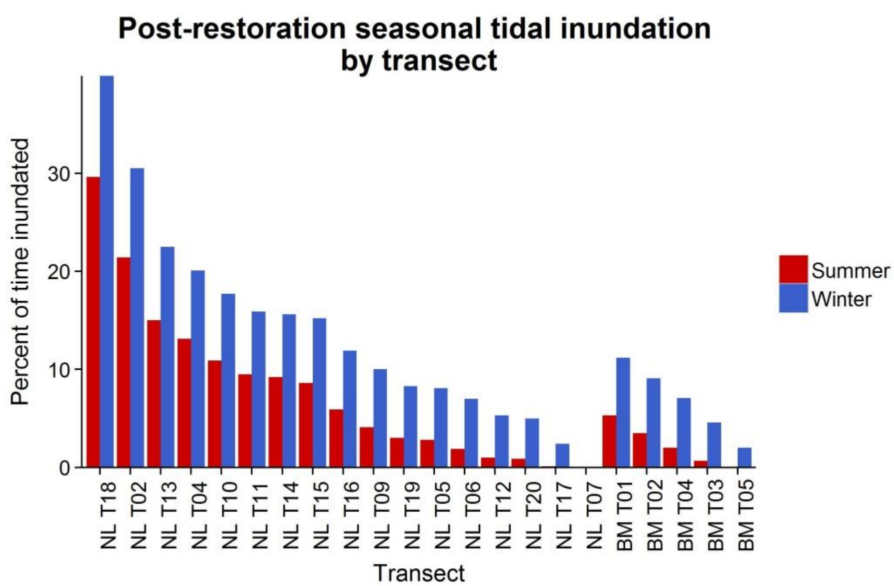

Figure 4. Percent inundation by season after restoration for sample transects at Ni-les'tun restoration site (NL) and Bandon Marsh Unit reference site (BM). Transects are ordered by elevation, as in Figure 3.

Data from groundwater wells (functioning as high spatial resolution tide gauges) showed that the tides propagated freely throughout the site; peak tide heights during monthly spring tides matched closely throughout the site, including emergent and forested transects far from the river (NL T10, NL T20), and high-elevation transects (NL T5) (Appendix A, Figure A15). Data from groundwater wells also showed that tide heights on the restoration site closely matched tide heights on the reference site, for areas at about the same elevation (Appendix A, Figure A16).

The Ni-les'tun site is large enough to have a substantial delay in tide peaks across the site. During summer, tides peaked 45 to 90 min later at the most distant transect (NL T20) compared to NL T18 (closest to the river) (Appendix A, Figure A17). The delay in winter was only half of the summer delay (not illustrated).

Wintertime high tides on the restoration and reference sites peaked around 2.8 m NAVD88 (70 cm above MHHW) (Appendix A, Figures A15, A16), showing that tidal wetlands occur far above MHHW at this site. These peaks are at approximately the 50% exceedance level calculated in NOAA's Extreme Water Levels analysis for the Charleston tide station (http://www.tidesandcurrents.noaa.gov/est/stickdiagram.shtml?stnid=9432780). The 50% exceedance elevation is currently being used as the basis for new mapping of the upslope boundary of tidal wetlands in Oregon (Lanier *et al*., 2014); the data from this project support that mapping method.

During this early post-restoration period, channels at Ni-les'tun drained slightly more slowly on ebb tides compared to the mainstem river (Appendix A, Figure A18). This is not unexpected for a large site, and may be due to the early stage of development of the tidal channel system. As the site's channel system develops, and with the construction of additional tidal channels at Ni-les'tun in 2014 (USFWS 2014b), the ebb tide pattern may more closely match the mainstem river. Channel equilibration with tidal forcing is underway, evidenced by several large headcuts in the lower Fahys channel (for example, where the channel crosses the mudflats outside the restoration area; Appendix C, Photos C1 and C2).

The tide gauge data and groundwater well data above provide strong evidence that tidal inundation has been effectively restored at Ni-les'tun. Additional evidence comes from the increases in soil salinity and channel water salinity in all parts of the site, including areas far from the Coquille River (see **Soils** and **Channel water salinity and temperature** below) and from the changes in channel morphology associated with the restored tidal flows (see **Channel morphology** below).

Elevation of wetland surface and instrumentation

Methods

Elevations of instrumentation, and the elevation of the wetland surface at sample locations, were measured in spring 2012 using high-accuracy RTK-GPS and total station equipment by Ducks Unlimited surveyor Pat Schulte (Appendix B, Tables B2 and B3). These elevations were used along with water level logger data to interpret post-restoration monitoring results, by relating biological and physical conditions to the local tidal inundation regime. The Oregon LIDAR Consortium's 2009 LIDAR DEM (Watershed Sciences 2009) provides a broad overview of elevations and was used for initial sample design (Brophy and van de Wetering 2012), but our research has shown that the LIDAR DEM is inaccurate in areas of dense vegetation (Ewald 2013), so we did not use LIDAR data to interpret our monitoring results.

Results and discussion

Elevations are referenced to the geodetic datum (NAVD88), unless otherwise stated.

Elevations of instruments and the wetland surface at sample locations are shown in Appendix B, Tables B2 and B3. These elevations were used along with water level logger data to interpret post-restoration monitoring results.

The 2008 LIDAR digital elevation model (DEM) (Watershed Sciences 2008) showed that elevations at Ni-les'tun generally ranged from 1.8 m to 2.3 m (6 to 7.5 ft) (Appendix A, Figure A13), with lower ground near the mouth of Fahys Creek, and higher ground along the river bank and in the northwest portion of the site. We resampled the LIDAR DEM because our investigations showed substantial inaccuracy (upwards bias) in the DEM due to vegetation interference (Ewald 2013). The resampled LIDAR DEM (Appendix A, Figure A14) can be used to more accurately visualize the extent of tidal influence.

Our 2010 survey showed that sample transects at Ni-les'tun ranged in elevation from 1.5 m to 2.5 m (4.9 ft to 7.6 ft), with an average of 2.1 m (Appendix B, Table B2), and spot checks showed that these elevations did not change substantially in 2013. The highest portions of the natural levee and man-made dikes (the latter removed during restoration) exceeded 3 m. During 2013, Mean Higher High Water (MHHW) averaged 2.1 m at the three tide gauges installed for this project (Appendix A, Figure A12). Brophy *et al*. (2011) measured the elevation of low and high marsh at comparable sites on the Oregon coast and found that low marsh occurred slightly below MHHW, and high marsh occurred near or just above MHHW. This was also true at the Bandon Marsh Unit reference site; low marsh at the site was generally found just below MHHW, and high marsh was found just above MHHW (see **Vegetation** below). Thus, the restoration site is expected to initially restore to a mix of low marsh and high marsh, depending on position in the elevation gradient.

The historic wetland type at Ni-les'tun was "seasonally wet prairie" subject to tidal flooding (Appendix A, Figure A24; Benner 1992) – what we currently call "high marsh." Therefore, the high marsh at the Bandon Marsh Unit – which occurs at about 2.1 to 2.4 m – is an appropriate reference area for the pasture. Based on their 2010 elevations, the lower transects at Ni-les'tun (e.g. NL T18, NL T13, NL T14) have probably undergone subsidence (elevation loss) of 0.3 to 0.6 m (1 to 2 ft). Subsidence is common at diked tidal wetlands in Oregon; it is caused by organic matter oxidation, buoyancy loss, and compaction associated with drainage, grazing, and other land use activities (Frenkel and Morlan 1991). In 2013, in the lower parts of the site, low marsh vegetation was developing (see **Vegetation** below), but over time, accretion may lead to re-establishment of high marsh (Frenkel and Morlan 1991, Thom and Borde 2002). Dynamic vegetation and soil conditions at the reference site suggest that accretion may be fairly rapid in this part of the Coquille River estuary (Brophy 2005; also see **Emergent wetland plant communities** and **Soils** below).

After restoration, the wetland surface at Ni-les'tun had many areas that held water after monthly high tides (e.g. Appendix C, Photo C6). These created breeding sites for the salt marsh mosquito (*Aedes dorsalis*). The following information is provided at the USFWS website (http://www.fws.gov/oregoncoast/bandonmarsh/Mosquito.html):

> *In 2011, Bandon Marsh National Wildlife Refuge completed the restoration of 420 acres of tidal marsh - the largest ever in Oregon. The restored Ni-les'tun tidal marsh is succeeding in increasing use by wildlife, including migratory waterfowl and shorebirds, along with native fish species, including the threatened coho salmon. However, an unanticipated by-product of the restoration*

was a large population increase of the salt marsh mosquito (Aedes dorsalis) in the summer of 2013. Shallow pools resulting from the marsh restoration, which were inadvertently created as filled ditches subsided or in ruts left by equipment, provided new breeding habitat for these mosquitoes. Though mosquitoes were present in the area prior to marsh restoration, they had much less available habitat. No other salt marsh restoration effort in Oregon had experienced this issue before.

During 2013, USFWS took several measures to control mosquitoes, and efforts are ongoing in 2014, with excavation of 40,000 additional linear feet of first- and second-order tidal channels planned for 2014. These new channels were not included in our monitoring effort, because they did not yet exist in 2013 when our monitoring occurred. Besides helping with mosquito control, the additional channels will further enhance tidal wetland functions at Ni-les'tun, particularly salmonid habitat availability. Other USFWS activities associated with mosquito control at Ni-les'tun, including mosquito monitoring, are described at http://www.fws.gov/oregoncoast/bandonmarsh/Mosquito.html.

Channel morphology

Methods

Channel types and monitoring approach

The Ni-les'tun project involved a major channel restoration effort; five miles of channels were constructed in 2009-2010. Our monitoring was designed to track changes to those excavated channels, with the goals of evaluating the performance of the restoration and informing inform future channel design efforts (see **Baseline "as-built" survey** below). During 2013 monitoring, we went beyond this initial scope, adding monitoring of the larger, pre-existing, non-excavated channels in the lower reaches of each of the three major tidal channels (Fahys, NoName, and Redd). These pre-existing channels were distinctly different from the excavated channels because they contained large amounts of soft, fine sediment which had accumulated during the pre-restoration period due to the flow barriers present on the site (dikes and tide gates). By contrast, excavated channels had channel bottom substrates were scraped by the excavator, and therefore lacked fine sediment prior to restoration of tidal exchange. We added monitoring of the pre-existing, non-excavated channels in part because we wanted to determine whether large wood placement affected these soft-sediment-filled channels differently from the hard-surfaced excavated channels.

Baseline "as-built" survey

As described in the baseline report (Brophy and van de Wetering 2012), baseline monitoring of channels consisted of an extensive RTK-GPS "as-built" survey of excavated channels, conducted by Ducks Unlimited surveyor Pat Schulte and field assistants. (Earlier ditches at the site had been filled or disked, as described in USFWS 2014b). Since it was an "as-built" survey, the baseline survey did not include the non-excavated, pre-existing channels; also, the water in these lower channels was too deep and the substrate too soft for foot access by the survey crew. The extent of the baseline survey is illustrated in Appendix A, Figure A8; pre-existing unsurveyed channels are also shown. The baseline survey provided an excellent basis for determining post-restoration change in excavated channel morphology.

Field methods

During Year 2 post-restoration (2013), channel morphology was monitored using cross-sectional profile measurements in 25 reaches of primary and secondary tidal channels at the restoration site; total length monitored was 2078 m (Appendix A, Figure A4). Eleven of the reaches were “wood reaches” which contained large wood placements (Appendix A, Figures A33 and A34) and 14 reaches were “non-wood reaches”, which did not have large wood placements. Within all monitored reaches, cross-sections were measured approximately every two channel widths, resulting in 311 cross-sections. Of these cross-sections, 130 were in the wood reaches and 181 were in non-wood reaches; 234 of the cross-sections (18 monitoring reaches) were in excavated channels, and 77 cross-sections (7 reaches) were in pre-existing, non-excavated channels. Cross-sections at the restoration site were measured by stretching a survey chain from bank to bank across the channel, and measuring the depth of the channel below the survey chain at intervals of 10% of channel width (Appendix C, Photo C1). A subset of cross-sections were tied to the geodetic reference frame (NAVD88) by surveying the starting point of the cross-section using RTK-GPS. At the reference site, 10 cross-sections were surveyed using RTK-GPS, which provided high-resolution horizontal and vertical positions (Appendix A, Figure A6).

Fine sediment was measured at each cross-section on the restoration site by pushing a marked fir dowel (3.8 cm diameter) into the sediment until firm channel bottom was reached. Three measurements were taken in each half of the cross-section, and all six measurements were averaged to obtain a single fine sediment depth value per cross-section. Fine sediment was not measured at the reference site.

Channel flow path elevation (longitudinal profile) was measured at 421 locations (330 restoration, 91 reference) using high-accuracy RTK-GPS survey equipment (Appendix A, Figures A5 and A7; Appendix C, Photo C2). Surveyed channels were selected to represent conditions across the two largest sub-basins (Fahys and and NoName) and to fill in gaps between cross-sectional profiles. At the reference site, the main northern channel (Shipwreck Channel) was surveyed from its mouth to the upper section where channel depth was about 0.5 m. To provide additional reference data, and under separate funding, we surveyed flow path elevations in the main tidal channel at Cox Island Preserve in the Siuslaw River estuary. The Cox Island channel was surveyed from its mouth to the uppermost (headwaters) channels. In addition, longitudinal profiles of the flowpaths were generated for NoName Lower and NoName Mid reaches using the flowpath data from the cross-sections described above.

Data analysis

We determined changes in channel depth for excavated channels by comparing the pre-restoration (2010) RTK-GPS “as-built” survey to the Year 2 post-restoration (2013) channel cross-sectional data. A Triangulated Irregular Network (TIN) was constructed within a GIS environment and a virtual cross-section was extracted from the pre-restoration TIN at the location of each post-restoration cross-section. Channel width could not be directly compared because of differences between the pre-restoration and post-restoration survey methodologies and pre-restoration TIN spatial registration. However, qualitative observations were made throughout the project on channel width changes.

Channel depth was extracted from both the pre-restoration cross-section and post-restoration cross-section at 10% channel width intervals and the difference calculated. Pre-restoration *versus* post-restoration channel depth was analyzed using BACI (ANOVA) with wood placement (wood *versus* no wood) and channel location (surveyed reach) as fixed effects. Results should not be extrapolated to other areas, since surveyed reaches were not randomly selected (Bennington and Thayne 1994).

Analysis of fine sediment depth used only post-restoration data, since the as-built excavated channels had no fine sediment. Comparisons of pre-restoration *versus* post-restoration fine sediment thickness were made using BACI (ANOVA) with wood placement (wood *versus* no wood) and channel location (surveyed reach) as fixed effects, as described above for channel depth analysis. ANOVA was conducted in R using the general linear models ("glm") package; significant differences between means were determined using the "lsmeans" package within general linear models, and displayed with the compact letter display of pairwise comparisons ("cld") routine.

Post-restoration (2013) longitudinal channel profiles were derived from the cross-sectional data and compared to the pre-restoration as-built RTK-GPS survey. Statistics were not calculated due to the small number of channels available for comparison and their varied settings and characteristics.

Results and discussion

Comparison of pre-restoration and post-restoration morphology of the excavated channels at Ni-les'tun showed that most channels did not change dramatically during this early two year post-restoration period. However, some changes were evident: channels had deepened slightly, and soft sediment substrates were developing. In addition, the lower reaches of each main tidal channel (pre-existing, non-excavated reaches) showed evidence of downcutting as the channels began to equilibrate with the restored tidal forcing. These changes are moving the channel system towards equilibrium with the tidal forces present at the site, and towards the reference site's channel system characteristics. Changes in channel cross-sectional area, depth, and soft sediment distribution have been observed in other tidal wetland restoration projects, particularly those with excavated channels and subsided wetland surfaces (Teal and Weishar 2005, Johnson *et al*. 2012).

Channel depth

On average, the excavated channels surveyed at Ni-les'tun deepened slightly (Figure 5), though the amount of deepening varied by channel ($p = 0.005$) (Appendix B, Tables B5 and B6). The channel that deepened the most was NoName Mid/Upr, which deepened by about 30 cm. The increase in depth in other channels ranged from 12 to 24 cm. At this early stage of the restoration trajectory, wood placement showed no significant effect on change in channel depth at a reach scale ($p=0.24$, Appendix B, Table B6). On the scale of an individual cross-section, changes are already observable, but statistically untestable by our methodology. For more information on changes at the individual cross-section scale, see **Channel cross-sections** and **Salmonid habitat availability**: **Wood structures and channel morphology** below. Wood addition may have a larger effect on channel depth over time, as the channel system develops.

It is important to note that the non-excavated channels appeared to have deepened much more than the excavated channels, but we could not describe those changes quantitatively because the non-excavated channels were not surveyed prior to restoration due to baseline monitoring goals, methods and logistics (see **Methods** above).

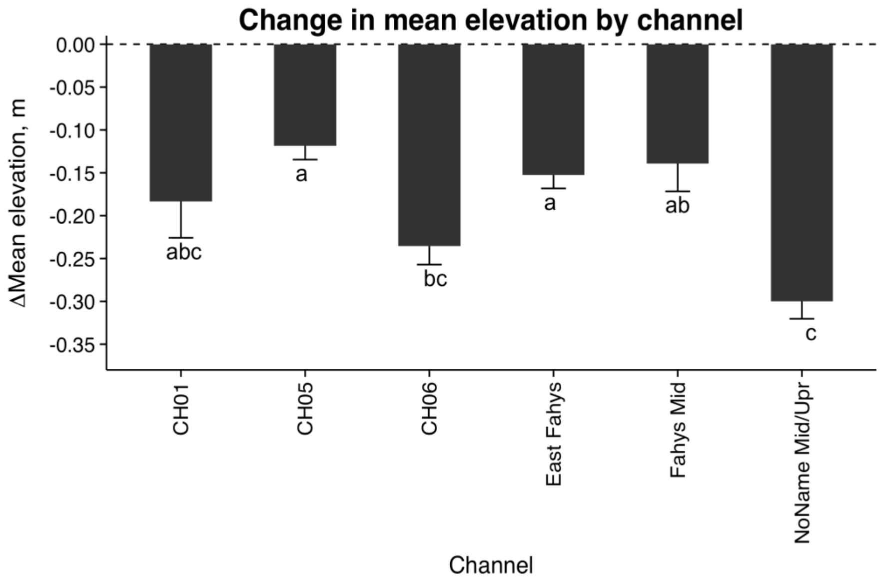

Figure 5. Change in mean channel depth (flow path elevation) between 2009 and 2013 for surveyed channel reaches at the Ni-les'tun restoration site. Bars with a letter in common are not significantly different ($p > 0.05$).

Fine sediment depth

Fine sediment is important for production of benthic macroinvertebrates and therefore important for fish foraging opportunities. As described in Methods above, fine sediment was absent from the excavated channels prior to restoration of tidal flows (due to scraping by the excavator), but non-excavated channels had abundant, deep fine sediment. In Year 2 post-restoration (2013), fine sediment was still present in the non-excavated channels, but was also present in the surveyed channels (Figure 6). Excavated channels had less sediment than pre-existing channels ($p<0.0001$) (Appendix B, Tables B7 and B8). Average fine sediment depths ranged from 5 to 23 cm in excavated channels, and from 38 to 60 cm in non-excavated channels. Although there was not a significant effect of wood placement on fine sediment depth across all channels (Appendix B, Table B8), we did observe major sediment redistribution in non-excavated pre-existing lower channels (see **Channel cross-sections** and **Salmonid habitat availability**: **Wood structures and channel morphology** below). Wood is likely to have a larger effect on fine sediment distribution in future years as the channel system develops.

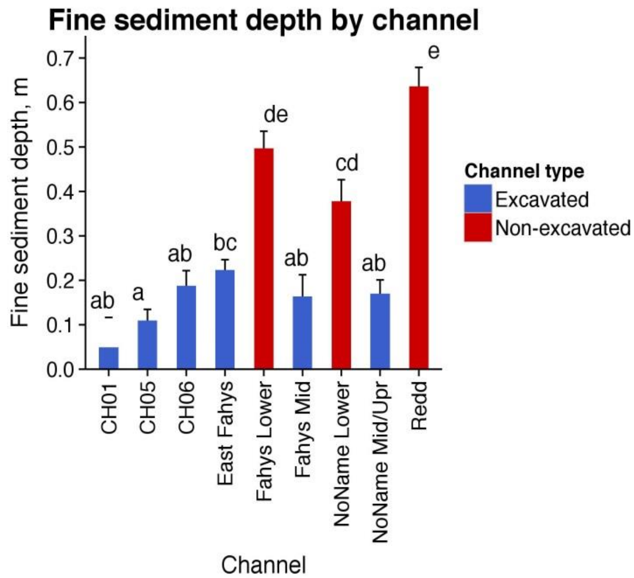

Figure 6. Fine sediment depth by channel at the Ni-les'tun restoration site in 2013. Bars with a letter in common are not significantly different (p > 0.05.

Channel cross-sections

We compared pre-restoration and post-restoration channel cross-sections for the excavated channels at Ni-les'tun. Changes in cross-sections varied in magnitude and direction. Many channels were still generally similar to their excavated dimensions, but where changes had occurred, they fell into several patterns:

1. Deepening, with fines accumulating
2. Deepening, without much accumulation of fines (i.e., degradation or scouring)
3. Broadening, becoming more shallow, and accumulating fines
4. In-filling with fines (i.e., aggradation), little depth change

Pattern 1 was the most common, but many channels showed several of these patterns along their lengths. Pattern 3 (broadening and becoming more shallow) occurred only in the lower reaches of a few channels, where channel banks were slumping. Very few examples of in-filling channels were found, suggesting that the channel system was not over-excavated. Typical examples of these changes are illustrated in Appendix A, Figure A22.

We compared cross-sectional profiles in 2013 for wood and non-wood reaches of the non-excavated, pre-existing channels at Ni-les'tun (lower Fahys, lower NoName, and lower Redd). As described above, these channels had abundant soft, fine sediment prior to restoration due to the exclusion of tidal flows, and the soft sediment was not disturbed during restoration. Presence of large wood affected channel profiles; non-wood reaches showed simple profiles with little variation, but wood reaches showed more variable and complex profiles. (Wood effects on channel morphology are discussed in detail in **Wood structures and channel morphology** below; see Figures 23 and 24.) It seems likely that the abundant soft sediment within the pre-existing, non-excavated channels could more easily be sculpted by the turbulent flows around wood structures, compared to the harder surfaces of the excavated channels. In addition, the pre-existing, non-excavated channel reaches were located near the mouth of each main tidal channel, where post-restoration tidal forcing and hydraulic head were greatest. Since these results relate mainly to salmonid habitat availability, they are discussed further in **Salmonid habitat availability**: **Wood structures and channel morphology** below.

Widths (and width:depth ratios) could not be compared between pre- and post-restoration due to differing field methods, but Ni-les'tun cross-sections were generally similar in shape to the larger channels at the reference site (Appendix A, Figures A22 and A23). However, smaller reference channels (in the middle to upper reaches of blind channel systems) had distinctly deep, narrow cross-sections (Appendix A, Figure A23, cross-sections SHP-07 to SHP-10). Few of these smaller channels were excavated at Ni-les'tun, but over time, they will develop. As they develop, they may show cross-sectional profiles similar to the reference site. Future monitoring of developing channels is recommended to track these changes. At a minimum, we recommend monitoring randomly selected reaches of the first- and second-order channels which USFWS plans to excavate in 2014 (USFWS 2014b).

Longitudinal profiles

We compared longitudinal gradients at the Bandon Marsh Unit and Cox Island reference sites with the gradient for the lower to middle reaches of one major tidal channel at Ni-les'tun (Figure 7). Comparisons were made relative to a tidal datum (MLLW) rather than the geodetic datum, since tidal channels evolve in response to tidal forces. Elevations across the longitudinal profile at Ni-les'tun and reference sites were similar (0 m to 1.2 m relative to MLLW within the lower to middle reaches), but the channels differed in shape. The surveyed channel at the reference site (Shipwreck channel) was concave, with a fairly smooth gradient and a rapid elevation gain in the upper reaches (Figure 7A). The channel had a small area of step-pool structure in the middle channel reaches (around 500 m upstream from channel mouth). The reasons for this step-pool structure are unknown but may relate to buried large wood. The Cox Island longitudinal profile was similar to the Bandon Marsh Unit profile, proportional to its longer length (it was surveyed all the way to headwaters) (Figure 7B), but was exceptionally smooth in slope. The Cox Island channel's smoothness was visually confirmed during field work and was not an artifact of sample density. The longitudinal profile of the lower and middle reaches of NoName channel at Ni-les'tun had a more convex surface, and slope was more variable (Figure 7C).

These three channels drain similar sized wetland areas at similar elevations in similar estuarine landscape settings, so their tidal channel morphology should be roughly comparable, but comparisons need to account for the fact that the portion of the channel surveyed differed between the sites. The Ni-les'tun channel was surveyed only from its mouth to its middle reaches; its headwaters (once they form) are likely to be another 800 m upslope.

The convex shape of the longitudinal profile at NoName was due to the channel construction methods used. Channels were excavated relative to the wetland surface elevation rather than relative to a geodetic or tidal datum (Appendix A, Figure A19). Since the wetland surface elevation did not have a consistent slope across the site, the result was a channel bottom that also lacked a consistent slope. We expect the Ni-les'tun channels to approach the more consistent gradients of reference system channels over time. This process could have been accelerated by excavating channels to a geodetic datum such as NAVD88 rather than excavating relative to wetland surface, but that method would have been more expensive than the method used at Ni-les'tun.

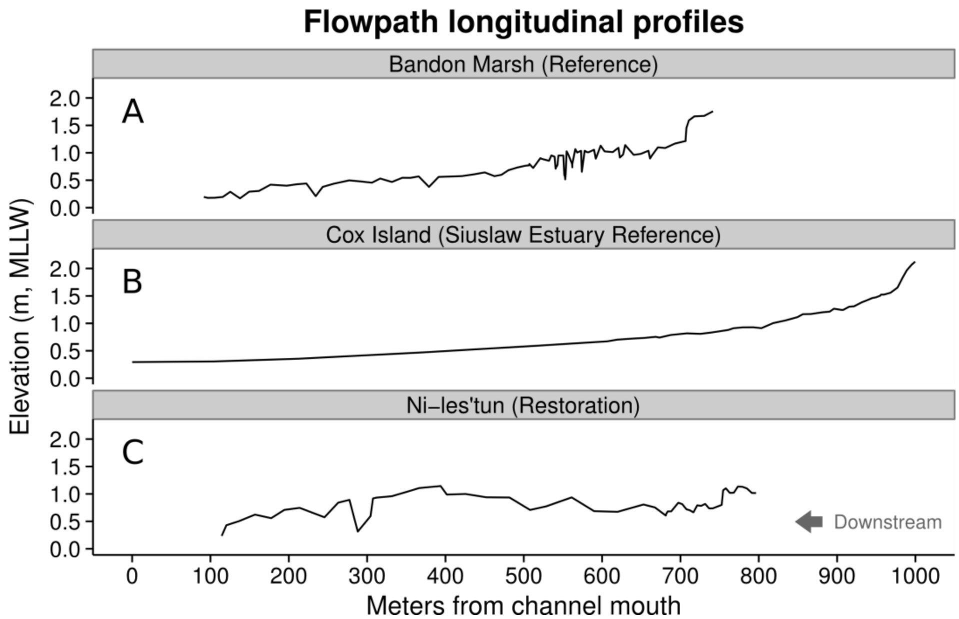

Figure 7. Flowpath longitudinal profiles for major tidal channels at two reference sites (A, B) and the NoName Lower to NoName Mid channel at the Ni-les'tun restoration site (C). Cox Island data is from Ewald, Brown and Brophy (2014). Locations of flowpath measurements are shown in Appendix A, Figures A5 and A7.

For channels with pre-restoration (as-built) RTK-GPS survey data (i.e., excavated channels), pre-restoration longitudinal profiles were plotted against post-restoration profiles and combined with fine sediment data to illustrate the magnitude of changes (Appendix A, Figure A20). Changes were generally small, confirming the overall observations of limited depth change in excavated channels (Figure 5). Fine sediment distribution along the length of each channel was variable, and as described above, fine sediment depth in excavated channels was not significantly associated with wood placement at this early stage of restoration (Appendix B, Table B8).

NoName Mid/Upr (bottom panel, Appendix A, Figure A20) was unique among the surveyed reaches. This channel had a "reverse gradient" – that is, the channel bottom elevation was lower at the upstream end of the survey than at the downstream end (by about 25-50 cm). This is probably due to the method of channel construction, as described above and illustrated in Appendix A, Figure A19. The reverse gradient of the wetland surface (as shown by the top of bank gradient) is due to higher ground on the natural levee near the river and lower ground in the interior of the site. This pattern is common on the Oregon coast (Brophy 2007a; Brophy *et al*. 2011). Because the channel was excavated relative to the wetland surface, the channel bottom also had a reverse gradient. Field observations confirmed that this channel reach holds some water at low tide rather than draining completely - the consequence of the reverse gradient. As for the other channels, we expect this channel to eventually equilibrate with tidal forcing and develop a longitudinal gradient in the normal direction, but equilibration will take longer than if it had been constructed relative to a geodetic or tidal datum.

The pre-existing, non-excavated channel reaches at Ni-les'tun showed convex longitudinal profiles and thick sediments (Appendix A, Figure A21). These pre-existing channel reaches were located near the mouth of each major tidal channel. As described above, lower reaches at Ni-les'tun show the most dynamic conditions (including headcutting and channel deepening), yet pre-restoration (as-built) RTK-GPS survey data were not available for these areas due to prohibitive depths and soft substrates prior to restoration (see Methods above). Therefore, pre- *versus* post-restoration change could not be determined for these reaches. However, elevation and sediment thickness decrease towards the downstream end of the surveyed reaches (Appendix A, Figure A21), suggesting that scouring may be occurring.

The most dramatic changes in channels were headcuts which initiated adjacent to the restoration actions (tide gate and dike removal) and are proceeding upstream as the system equilibrates to tidal forcing. As described above, we were unable to quantify these changes due to lack of baseline (as-built) survey data. However, the visual change is striking. One highly visible headcut is on lower Fahys where the channel crosses the mudflats (Appendix C, Photos C3 and C4). These headcuts could be tracked in the future as they proceed upstream to areas that were surveyed, using boat-based or alternative survey methods. Large changes to channel cross-section were also observed near dike breaches and tide gate removals in a restoration project in the Columbia River estuary (Johnson *et al*. 2012).

Lessons learned about channel morphology monitoring

Channel morphology methods included RTK-GPS "as-built" survey at baseline, and a combination of RTK-GPS survey and rapid cross-sections (survey chain and rod) during post-restoration. This combination of methods allowed collection of a much larger dataset than would otherwise have been possible, but created some challenges in data alignment and interpretation.

Due to cost, site logistics, and monitoring methodologies, we did not establish permanent monuments at channel cross-section endpoints during baseline monitoring. Such monuments would be installed at a distance back from the bank edge to prevent loss due to channel widening. Although permanently monumented cross-sections are more costly than the methods used in this study, they would enhance tracking of post-restoration changes in channel morphology at both the reach and cross-section scale. Changes in the width, lateral migration, and depth are easily identified using this design. A combination of monumentation of a subset of channel cross-sections, along with rapid chain-and-rod cross-sections between the monuments, might be a good compromise to optimize the number of measurements to capture reach level changes and to describe cross-section-level changes. Monumentation should be planned to avoid conflict with machinery operations during restoration, and carefully reviewed with the engineering team. For more information please contact the authors.

1b. Physical and biological conditions at Ni-les'tun

Monitoring Question 1b: Are tidal wetlands developing physical and biological characteristics trending towards reference conditions?

Metrics: Wetland plant community composition and extent; soil characteristics (stored organic carbon, salinity, pH); groundwater levels; surface water salinity and temperature. *(Rationale: Soil characteristics, groundwater levels and surface water characteristics are controlling factors in tidal wetland plant community development and many other wetland functions. Note: channel morphology is also a key physical characteristic; it is addressed under Question 1a above.)*

Tidal hydrology

This parameter is discussed under Monitoring Question 1a above.

Emergent wetland plant communities

Methods

As described above, sampling at Bandon Marsh NWR was stratified and distributed across all tidal wetland elevation zones and all sub-basins. In 2013, data on emergent wetland plant community composition was collected within the study transects established in 2010 (Appendix A, Figures A2 and A3). Five of these transects (NL T2, NL T4, NL T5, BM T1, and BM T2) were also monitored during the early baseline period in 2003. Transects were 100 m long, and stratified to sample major elevation zones, sub-basins, and major vegetation zones. Visual estimates of percent cover by species were made within 15 randomly placed 1-sq m quadrats along each transect. Quadrats were placed 1 m off the transect's central axis (left or right side randomly determined), at random distances from the transect end post (but at least 3 m apart and 3 m from the transect end post). Visual cover estimates followed the Oregon Department of State Land's Routine Monitoring Protocol (Oregon DSL 2009).

For emergent wetlands, comparisons of pre-restoration *versus* post-restoration plant community metrics (percent cover of natives and non-native species, total % cover, and species richness) were made using BACI (ANOVA) with site (restoration site *versus* reference site) and year (2010 *versus* 2013) as categorical independent variables. ANOVA was conducted in R using the general linear models ("glm") package; significant differences between means were determined using the "lsmeans" package within general linear models, and displayed with the compact letter display of pairwise comparisons ("cld") routine. BACI ANOVA could not be performed for forested wetlands, because there was only one forested wetland transect at the reference site. A multivariate technique, non-metric multidimensional scaling (NMDS), was used to summarize differences in plant composition among transects pre- and post-restoration. The analysis was run in the R package 'vegan' on percent cover data after averaging all plots within a transect.

Results

Emergent wetlands: Changes across all transects

Plant communities across the restoration site were markedly different between 2010 and 2013; species that could not tolerate the restored tidal inundation and brackish salinity decreased in cover or showed poor condition (Appendix C, Photos C5 and C6). The degree to which species intolerant of inundation and salinity decreased depended on elevation and distance from the Coquille River (the source of brackish water). The species that decreased included some native species, such as Pacific silverweed (*Potentilla anserina*), and some non-native species such as birdsfoot trefoil (*Lotus corniculatus*).

Changes in plant community metrics averaged across all transects at Ni-les'tun and the Bandon Marsh Unit reference site are shown in Figure 8 and Tables B9 to B12 in Appendix B. Non-native species cover at Ni-les'tun declined from 58.8% in 2010 to 39.9% in 2013. This decline was largely due to dieback of pasture species intolerant of the restored inundation and brackish salinity, especially tall fescue (*Schedonorus arundinaceus*) and birdsfoot trefoil. As the pasture species died back, species richness per transect decreased significantly at Ni-les'tun, dropping from 5.0 in 2010 to 3.0 in 2013 ($p < 0.03$). Total plant cover was significantly lower at Ni-les'tun in 2013 (87.1%) compared to 2010 (115.8%) ($p > 0.05$); this was due to bare ground created when some species died back. We expect the increase in bare ground to be temporary, since inundation, soils and salinities in 2013 were conducive to development of native tidal marsh vegetation (see **Soils** and **Channel water temperature and salinity** below). The decrease in species richness may persist for many years, since tidal marsh restoration sites in Oregon are often dominated by a small number of highly competitive species (particularly the native species Lyngbye's sedge, *Carex lyngbyei*) (Frenkel and Morlan 1990; Brophy 2007b, 2010; Dionne *et al*. 2012).

As was true during the baseline monitoring in 2010, cover of native species in 2013 was higher at the reference site (92.3%) compared to Ni-les'tun (47.1%) (p = 0.03). Cover of native species had been slightly higher at Ni-les'tun before restoration (56.6% in 2010), but the decline in 2013 was not statistically significant. As the site's vegetation adjusts to the restored inundation and brackish salinity in future years, we expect a decrease in cover of native species which are intolerant of those conditions, such as slough sedge (*Carex obnupta*), but we expect this decline to be offset by increased cover of native tidal marsh species adapted to those conditions. On the other hand, cover of non-native creeping bentgrass (*Agrostis stolonifera*) increased at Ni-les'tun in 2013 (16.6%) compared to 2010 (11.3%). Creeping bentgrass is a common dominant in undiked high marsh (including transect BM T1 at the Bandon Marsh Unit reference site - Appendix B, Table B14), and it is also a common colonizer of restored high marsh sites (Brophy 2007b, Brophy *et al*. 2011, Cornu and Sadro 2002). Therefore, creeping bentgrass is likely to be dominant in at least parts of Ni-les'tun for many years.

Overall, NMDS analysis suggested that the post-restoration composition of plant communities at Ni-les'tun was converging with that of the reference site (Appendix A, Figure A25). Monitoring in 2015 will provide information on whether this trend continues, but further monitoring beyond 2015 is recommended, since important plant community changes extend across many years after restoration (Cornu and Sadro 2002).

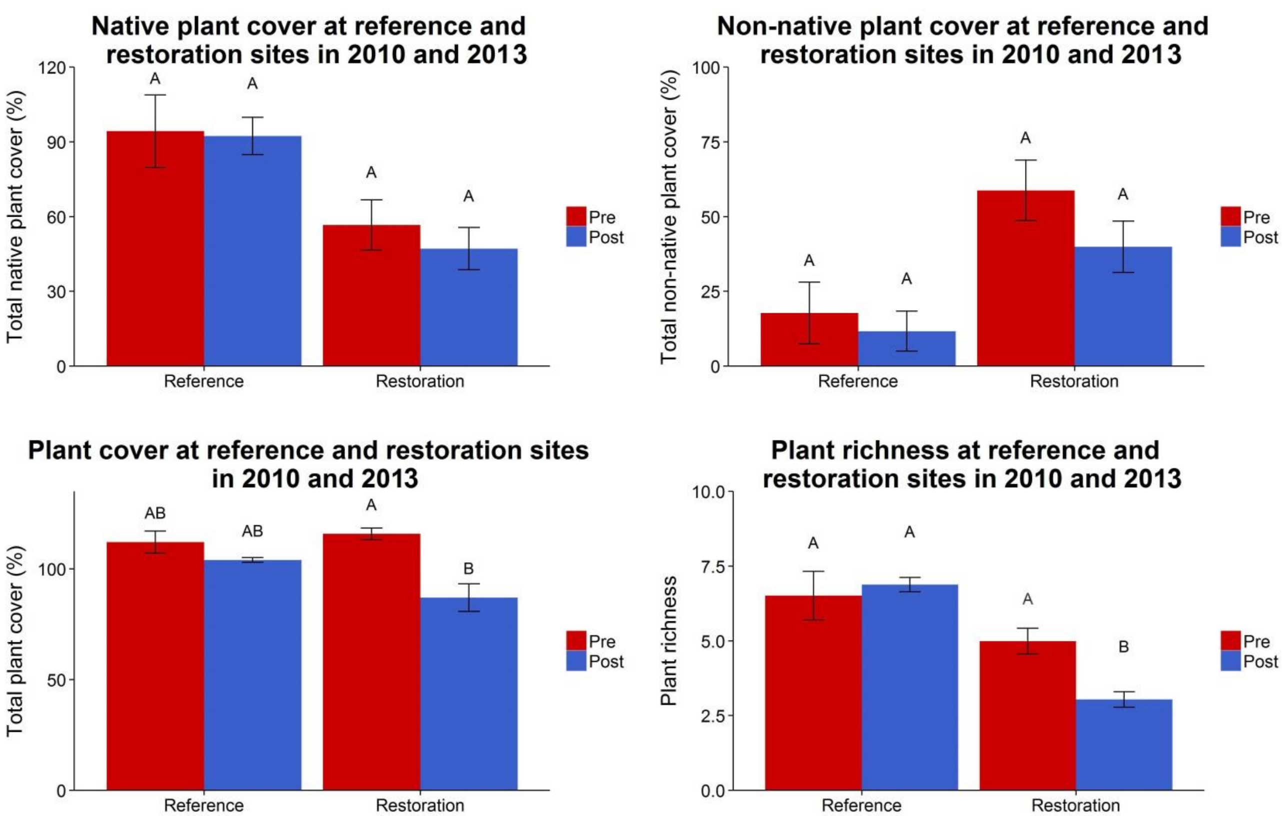

Figure 8. Pre-restoration (2010) *versus* post-restoration (2013) comparisons for plant community metrics at Ni-les'tun restoration site and Bandon Marsh Unit reference site. Bars with a letter in common are not significantly different (p > 0.05).

Emergent wetlands: Changes by transect

As described in the baseline report (Brophy and van de Wetering 2012) and the early baseline report (Brophy 2005), muted tidal influence just prior to dike removal had already led to major vegetation change in the lowest parts of Ni-les'tun (lower Fahys sub-basin)before restoration. Vegetation in this area (NL T18, NL T2) still changed between 2010 and 2013, but the change was not as dramatic as it might have been if tides had been fully excluded prior to restoration. For example, the lowest transect at Ni-les'tun in 2003 (NL T1) was dominated by Pacific silverweed and tall fescue (Brophy 2005). NL T1 was lost due to disturbance during early construction, but a new transect (NL T18) was established in the same general area. During 2010 baseline monitoring, NL T18 was dominated by the native tidal wetland species saltgrass (*Distichlis spicata*, 79.0% cover), a clear indication of occasional brackish inflows and muted tidal influence in this area prior to restoration. By 2013, saltgrass was the sole dominant (97.4% cover).

In some interior areas of the restoration site, even though percent cover by species did not change much since pre-restoration monitoring, the condition of the vegetation definitely changed. Senescence (die-back) of species intolerant of the restored salinity and inundation was apparent in these areas, showing the influence of the restored tides. Cover of these species is likely to be greatly reduced by the next monitoring event in 2015. For example, at NL T4, slough sedge – a salt-intolerant wetland species - still had average cover of 67.6% in 2013, but 39.5% of the slough sedge cover was senescent (browning) and only 28.1% was healthy. Similarly, tall fescue was still dominant across much of the restoration site in 2013 (Appendix B, Table B14), but its condition was very poor in most areas and its clumps were breaking down, being gradually covered by salt-tolerant wetland colonizers like saltbush (*Atriplex patula*). Tall fescue is not tolerant of extremely wet conditions; its wetland indicator status for the Oregon coast is FAC, indicating it is equally likely to be found in uplands and wetlands (Lichvar *et al*. 2014).

Forested wetland plant communities

Methods

The same field methods were used to monitor forested wetland vegetation during pre-restoration (2010) and post-restoration (2013) periods. Field measurements were made within four permanent transects, three at the Ni-les'tun restoration site (NL T6, NL T7 and NL T20) and one at the Bandon Marsh Unit reference site (BM T5). Plots were randomly located along study transects; plots were 9.1 m (30 ft) wide (4.6 m on each side of the transect) and the same length as the transect. Transect length varied depending on vegetation density: BM T5 and NL T6 were 53.0 m (174 ft); NL T7 was 68.6 m (225 ft); and NL T20 was 56.4 m (185 ft) long.

Sample unit size and vegetation measurements varied by stratum (herbaceous, shrub or tree). Sample units were nested within the overall plot following methods described in Peet *et al*. (1998). For shrubs, stems of each species were counted within 4.6 m by 4.6 m (15 by 15ft) plots placed on a randomly selected side of the transect at random distances from the starting point. Only stems branching below knee height were counted. Trees were counted within the entire plot (9.1 m = 30ft wide; length = transect length) except at BM T5, where exceptionally high tree density required a smaller plot size. At BM T5, trees were counted within the same plots as shrubs, but tree plots were extended to 9.1 m (30 ft) from the transect. At all transects, the diameter of each tree was measured at breast height (DBH). Herbaceous vegetation in forested wetlands was measured using visual estimates of percent cover within 1-sq m plots. Herbaceous vegetation plots were placed 1 m off the transect just inside the near and far boundaries of each shrub plot.

Transect NL T7, north of North Bank Road, was not tidally influenced either before or after restoration, largely due to its elevation (2.9 m = 9.5 ft NAVD88). Because this transect was not tidally-influenced and therefore unlikely to have been affected by the tidal wetland restoration actions, it was excluded from BACI analyses designed to determine the effect of restoration.

Because there was no replication at the reference site (only one location there was suitable for a reference forested tidal wetland transect), BACI analysis was not possible for forested wetland plant community metrics. Instead, we compared pre- and post-restoration data where possible using t-tests.

Results

NL T6, the lowest forested wetland transect, was very wet but entirely fresh prior to restoration. At this transect, density and basal area of Sitka spruce (*Picea sitchensis*) and tree-sized individuals of Pacific crabapple (*Malus fusca*) increased from 2011 to 2013 (Appendix B, Tables B16 and B17). Both of these species are tolerant of brackish salinity (Christy and Brophy 2007), so their increase may be due to the restoration of brackish tidal flows. The post-restoration increase in soil salinity (from 0.4 PSU in 2010 to 15.3 PSU in 2013) also supports this hypothesis (see **Soils** below). Changes in shrub density and herbaceous cover, however, did not show a clear pattern. Of the two shrub species present in the transect that are intolerant of brackish salinity, salmonberry (*Rubus spectabilis*) increased after restoration, while red elderberry (*Sambucus racemosa*) decreased (Appendix B, Table B17). Shrub-sized individuals of Pacific crabapple also decreased, but that could have been due to their growth into tree-sized specimens. For herbaceous species, a former dominant intolerant of brackish water (skunk cabbage, *Lysichiton americanum*) decreased greatly between 2011 (28% cover) and 2013 (1% cover), while slough sedge -- also intolerant of brackish water – increased to 96% of herbaceous cover (Appendix B, Table B18). It seems likely that the plant community at NL T6 has not yet equilibrated with the increased salinity, and further changes will probably occur over the next few years.

At NL T20, the highest of the forested tidal transects (2.3 m = 7.6 ft NAVD88), no major changes in tree density, tree basal area, shrub density, or herbaceous cover were observed between 2011 and 2013. This aligns with other monitoring data: although this transect is tidally inundated, tidal inundation occurs infrequently (only on spring tide cycles in winter), and soil salinity at the transect did not change from 2011 to 2013 (see **Soils** below). These minor changes in controlling factors were apparently too small in magnitude to create shifts in vegetation, at least at this stage of restoration.

At the reference site (BM T5), density increased for nearly all tree species between 2011 and 2013, and basal area of Sitka spruce increased. This aligns with our 2011 field observations of the abundance of small trees at this transect, and its ecotonal position (on the transition zone between tidal marsh and upland forest). Shrub density and herbaceous cover at this transect remained similar between 2011 and 2013.

Plant community mapping

Methods

We mapped wetland vegetation using aerial photography and field ground-truthing. High-resolution digital aerial photographs (15 cm pixel size) of Bandon National Wildlife Refuge were flown on May 20, 2013 and orthorectified by Eagle Imaging of Corvallis, Oregon. Images were taken at low tide from a vertical angle, using onboard GPS for automated georeferencing. We traversed the project sites on foot to correlate field vegetation with patterns in the aerial photographs. Map units were delineated in the field on enlarged printouts of the aerials. Digital vegetation maps were created in ArcGIS 9.3 by georeferencing the field maps and tracing the map unit boundaries into the GIS at a scale of 1:2000; the polygon size threshold was about 0.1 ha (0.25A). Vegetation maps were saved as shapefiles (NL_vegmap_2013.shp and BM_vegmap_2013.shp).

Following the National Vegetation Classification Standard (The Nature Conservancy 1994), we used a two-level hierarchical vegetation classification scheme. Plant associations represented fine gradations of dominant species; as in 2003 and 2010 monitoring, these were finely divided to reflect small differences in community composition. Alliances, the coarser level, were described by a single major dominant species that characterized a larger area. This two-level classification allows flexibility in tracking future vegetation change.

We also characterized plant communities as native-dominated or non-native-dominated, based on the alliance level classification. Native-species alliances such as Baltic rush and slough sedge were considered native-dominated, and non-native alliances such as tall fescue were considered non-native-dominated. The percent cover of native species *versus* non-native species varied within these alliances.

Results and discussion – Ni-les'tun restoration site

In 2013, a total of 86.2 ha (213.0 ac) was occupied by native-dominated vegetation alliances (Appendix B, Table B20). Non-native alliances occupied 103.0 ha (254.5 ac). The percentage of the mapped area occupied by native *versus* non-native-dominated alliances was very similar in 2013 (46% native, 54% non-native) and 2010 (47% native, 53% non-native). However, within this overall ratio, some alliances increased or decreased considerably. The native-dominated Baltic rush alliance increased strongly, from 4.5 ha in 2010 to 27.0 ha in 2013 (Appendix B, Table B21). Baltic rush is a tidal wetland species that was also a strong component of pre-restoration pasture communities at Ni-les'tun, allowing a more rapid response to the restored inundation regime and brackish salinity. Slough sedge cover increased slightly in 2013, but as described in **Changes by transect** above, its condition was very poor in most of the areas where it was technically dominant (based on percent cover), and its cover will probably decline sharply in the next few years. No other native alliances increased substantially during this early post-restoration period, but it is likely that some native species will increase during the next few years. For example, widespread colonization of restored tidal marsh by native Lyngbye's sedge has been documented starting 5 to 10 years after restoration at several Oregon sites (Frenkel and Morlan 1990; Brophy 2007b, 2010) and continuing for decades (Dionne *et al*. 2012).

Two native alliances decreased in area or condition at Ni-les'tun, as plant communities adjusted to the changes in salinity and inundation. Pacific silverweed decreased strongly, from 30.7 ha (75.9 ac) to 3.7 ha (9.1 ac) (Appendix B, Table B21). Although this species is tolerant of wetland conditions and brackish salinities, the individual plants may not have been adapted to the new environment after restoration. As described above, the slough sedge alliance increased slightly in area but decreased in condition in 2013.

The most widespread non-native alliance at Ni-les'tun – tall fescue – decreased from 95.1 ha (235.0 ac) in 2010 to 69.0 ha (170.5 ac) in 2013 (Appendix B, Table B21), and this species' condition was poor (senescent) across much of the remaining 69 ha. The areas formerly occupied the tall fescue alliance were mostly dominated by Baltic rush communities in 2013. The non-native creeping bentgrass alliance increased from only 0.33 ha in 2010 to 31.4 ha in 2013; this species colonized most of the areas that had been occupied by Pacific silverweed in 2010. Creeping bentgrass species is likely to remain dominant and even increase over the next few years, as it is a rapid colonizer of restoration sites and a common non-native component of otherwise least-disturbed high marsh in Oregon (Cornu and Sadro 2002, Adamus 2005, Brophy *et al*. 2011).

The geographic pattern of vegetation changes at Ni-les'tun during 2010 to 2013 reflected elevation and salinity gradients, as well as site history. As described above, the lowest portion of Ni-les'tun (near the mouth of Fahys) was already subject to muted tidal influence prior to restoration. Shifts in dominants in this area were relatively minor, probably because the 2010 dominants were already adapted to some tidal influence. Greater shifts occurred further north as former freshwater wetland communities died back and were replaced by salt-tolerant species and those able to withstand the frequent tidal inundation.

In the northwest portion of the pasture, communities that had been dominated by a mix of invasive reed canarygrass and native slough sedge changed greatly; the reed canarygrass was almost completely dead by 2013. This is probably due to reed canarygrass intolerance of brackish salinities in tidal waters flooding the site; reed canarygrass declined sharply after restoration at other brackish sites (Brophy 2007b, 2010). Across much of the site, where the non-native pasture grass tall fescue was generally dominant prior to restoration, the degree of change depended on elevation, with low areas showing decreasing cover and/or deteriorating condition of tall fescue. Many of these areas were still dominated by tall fescue in 2013, but the cover of this species is likely to decline rapidly over the next few years as the deteriorating plants die. As described above, some areas dominated by slough sedge also remained technically dominated by this species in 2013, but its condition was poor due to the restoration of brackish tidal flows.

Changing vegetation was also very noticeable in the forested wetlands in the northwest part of Ni-les'tun. Here, the Sitka spruce and willows that had been dominant in 2010 were dying back along the east side of the forest, particularly closest to Fahys Creek. Slough sedge become dominant in these areas, and black twinberry (*Lonicera involucrata*), a species that is tolerant of brackish water, was increasing in this area and appeared healthy.

The plant community shifts observed at Ni-les'tun during 2010-2013 may be similar to shifts that could occur with sea level rise. As tidal waters inundate areas that were formerly nontidal wetlands, plant communities and other organisms may respond by "retreating" or "migrating" to higher ground or areas farther from the tidal water bodies – if such areas are available (Harley *et al*. 2006). During 2010-2013 we saw this type of "retreat" of the slough sedge alliance from the area near NL T4 into the former Sitka spruce forest to the west, and similar "retreat" of tall fescue pasture grasses from large areas of the site.

Results and discussion – Bandon Marsh Unit reference site

Plant communities changed very little from 2010 to 2013 at the Bandon Marsh Unit reference site. The area of native-dominated communities was similar between the two years (80.8 ha in 2010 *versus* 81.4 ha in 2013), as was the area of non-native-dominated communities (8.9 ha in 2010 *versus* 9.5 ha in 2013) (Appendix B, Table B24). There was a small increase in the area of tufted hairgrass dominated communities along the landward edge of the site (Appendix A, Figure A29); percent cover of tufted hairgrass also increased slightly at transect BM T1 (from 18.6% in 2010 to 22.7% in 2013) and considerably at BM T2 (from 25.1% in 2010 to 40.3% in 2013) (Appendix B, Table B14). Changes between 2003 and 2010 suggested that this site may still be developing towards high marsh with ongoing accretion; it is a relatively young tidal marsh, with much of its area having accreted since the 1800s (Brophy and van de Wetering 2012). If so, the increased area and cover of tufted hairgrass may reflect this "maturation" process.

Soils

Methods

Soil samples were collected from the surface rooting zone (0-12 inches) in all vegetation transects in 2010 and 2013, using a Dutch auger and pooling 10 to 20 random subsample locations along each transect. Samples were collected during the dry season (August). Subsamples were bulked in the field, then delivered to the Oregon State University Central Analytical Laboratory for analysis. At the lab, large roots were removed, samples were dried and homogenized, and a subsample was removed for analysis. Electrical conductivity and pH of the soil solution were measured using an electrical conductivity meter and a reference electrode with a pH meter, respectively. Percent organic matter was determined by loss on ignition (Craft *et al.* 1991) by burning in a kiln at approximately 450°C for eight hours. Particle size analysis was conducted by the quick hydrometer method, after repeated treatment with hydrogen peroxide to remove organic material (Dane and Topp 2002). Soil salinity was derived from electrical conductivity using a standard formula (Fofonoff and Millard 1983). We calculated percent soil carbon from percent organic matter using a conversion specific to high organic soils (0.68 x %OM) (Kasozi *et al*. 2009).

Statistical analysis of soil data from emergent wetland transects was conducted separately from the forested wetlands, because we expected emergent wetlands to respond much more strongly to restoration. Forested wetlands, unlike the emergent wetlands, had never been cleared, ditched or drained, and they were located on higher ground where tidal inundation occurred less often. (In fact, one of the three forested transects, NL T7, did not experience tidal inundation at all during the monitoring period.)

For emergent wetlands, comparisons of pre-restoration *versus* post-restoration soil characteristics (soil salinity, % OM, % C, and pH) were made using BACI (ANOVA) with site (restoration site *versus* reference site) and year (2010 *versus* 2013) as categorical independent variables. ANOVA was conducted in R using the "general linear models" (**glm**) package; significant differences between means were determined using the "**lsmeans**" package within general linear models, and displayed with the "compact letter display of pairwise comparisons" (**cld**) routine. BACI ANOVA could not be performed for forested wetland soils, because there was only one forested wetland transect at the reference site. Therefore, we used t-tests to compare pre-restoration and post-restoration soil characteristics at the Ni-les'tun forested transects (NL T6, NL T7 and NL T20).

Results

The primary change in Ni-les'tun emergent wetland soils after restoration was a large and significant increase in soil salinity (p = 0.04) (Figure 9; Appendix B, Table B26). Average soil salinity in Ni-les'tun's emergent wetlands was 19.7 PSU in 2013, compared to 3.7 PSU in 2010 (Appendix B, Table B27). Using BACI analysis, this increase could not be statistically attributed to restoration; salinity also increased at the reference site (32.3 PSU in 2013 *versus* 15.7 PSU in 2010) (Appendix B, Tables B26 and B27). Lower rainfall during spring and summer 2013 compared to the same period in 2010 (http://www.ncdc.noaa.gov/temp-and-precip/ranks.php) probably explained the reference site's higher salinity in 2013; most likely, the year-to-year variability in precipitation prevented statistical separation of the restoration effect from the weather effect. Channel water salinity showed similar year-to-year variability, with higher salinity at the reference site in 2013 compared to 2010 (see **Channel water salinity and temperature** below.)

Soil salinity at three Ni-les'tun transects (NL T2, NL T13, and NL T18) was higher than marine salinity (32-35 PSU) after restoration (Appendix B, Table B21). NL T18 and NL T2 are located in the lower Fahys sub-basin, close to the river, but NL T13 is located farther from the river. NL T13 (34.4 PSU in 2013) is located in a blind channel system lacking freshwater inflow; post-restoration channel water salinity was also higher here than in other parts of the site (see **Channel water salinity and temperature** below). High salinity at NL T13 could be due to evaporative concentration of salts when inundating brackish tidal waters remain on the surface during hot summer days. Such evaporative concentration may be more likely in a system with relatively few tidal channels compared to reference conditions, since drainage is slower. Although several miles of tidal channels were constructed at Ni-les'tun, the site's channel density is still only a fraction of the channel density at a typical least-disturbed high marsh reference site (So *et al*. 2009).

Average soil carbon content was higher in emergent wetlands at Ni-les'tun after restoration (9.4% in 2013, compared to 6.6% in 2010) (Appendix B, Table B20), but the restoration effect (site*year interaction) was significant only at the 10% level. Early increases in surface soil organic carbon content could be related to dieback of pasture grasses; further monitoring will reveal whether the effect continues. MacClellan (2012) reported that soil carbon content of restored tidal wetlands of varying ages did not differ significantly from soil carbon content of least-disturbed reference sites, while unrestored sites had significantly lower soil carbon content. However, factors other than restoration status may relate to MacClellan's findings. For example, restored sites may have higher soil carbon compared to unrestored sites simply because wetter sites are more likely to be restored (due to lower agricultural productivity and breakdown of dike/tide gate systems).

Of the forested transects, only the lowest-elevation transect (NL T6) showed greatly increased soil salinity after restoration (15.3 PSU in 2013, compared to 0.4 PSU in 2010) (Appendix B, Table B21). The higher-elevation forested transects (NL T7, NL T20) did not show increases in salinity, nor did any of the forested wetland soils show significant changes in pH, % OM, or % C after restoration (p = 0.77). These results matched expectations, because the forested wetlands were much less strongly affected by the restoration actions compared to the emergent wetlands. The forested areas had never been cleared, ditched or and drained, and they are located at high elevations, where tidal inundation occurs infrequently (Appendix A, Figure A31).

Soil characteristics at the reference site did not change significantly between 2010 and 2013 (Figure 9; Appendix B, Table B22).

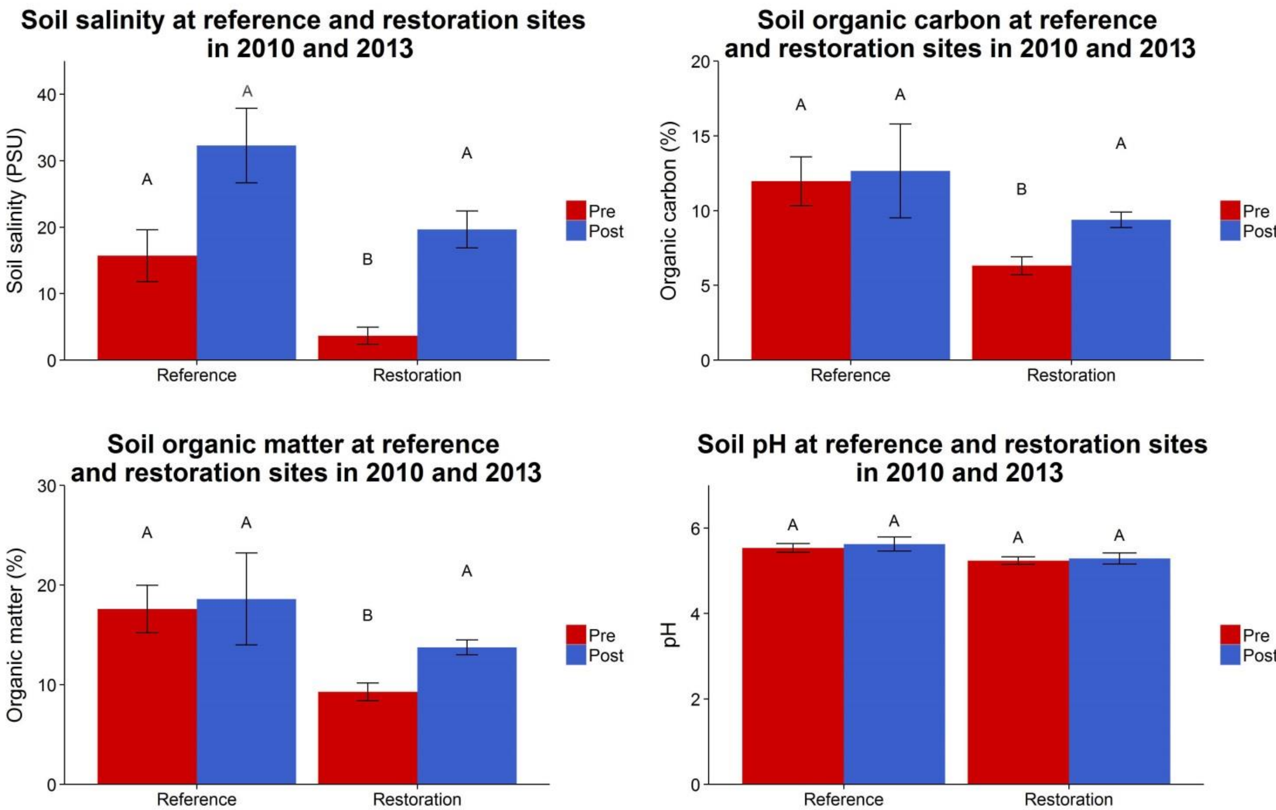

Figure 9. Pre-restoration (2010) *versus* post-restoration (2013) comparisons for soil metrics at Ni-les'tun restoration site and Bandon Marsh Unit reference site. Bars with a letter in common are not significantly different (p > 0.05).

Groundwater levels

Methods

Groundwater was monitored at each sample transect using standard shallow groundwater observation wells (Sprecher 2000); well design was slightly modified by adding tall risers to prevent overtopping of the wells by surface tidal flows (Brophy *et al.* 2011) (Appendix C, Photo C7). Monitoring in 2013 used the same observation wells installed in 2010 (17 wells at the Ni-les'tun restoration site and 5 at the Bandon Marsh Unit reference site) (Appendix A, Figures A2 and A3). Each well was approximately 1.2 m deep, so groundwater levels more than 1.2 m below the soil surface could not be tracked; deeper groundwater monitoring is not recommended for understanding wetland hydrology (Sprecher 2002). Automated water level loggers (Onset HOBO Model U20-001-01) were deployed in each well; water level was recorded at 15 min intervals from May 2010 through August 2011 (pre-restoration) and from September 2012 through September 2013 (post-restoration). Pressure data were adjusted for barometric pressure (using local barometric pressure data) and converted to water levels using HOBOWare Pro® software.

Groundwater levels were determined relative to two datums: the geodetic reference frame (NAVD88) and the soil surface. Groundwater levels relative to the soil surface were used to understand prevalence of wetland hydrology (Environmental Laboratory 1987), and to understand plant community development and soil processes. Groundwater levels relative to the soil surface were expressed as negative numbers when groundwater is below the soil, but when the soil is inundated, values were positive since the water surface was higher than the soil surface. Groundwater levels relative to NAVD88 were used to understand relationships between groundwater levels and tidal inundation patterns.

Daily maximum groundwater level was calculated both relative to NAVD88 and relative to the soil surface for the full monitoring period. We also calculated shallow groundwater duration (the greatest number of continuous hours per year that groundwater was within 30 cm of the soil surface), and compared that calculation for pre- and post-restoration periods. (Groundwater that remains within 30 cm of the soil surface for a substantial part of the growing season defines wetland hydrology [Environmental Laboratory 1987). Before-After-Control-Impact (BACI) 3-way ANOVA with site, year, and season (wet *versus* dry) as independent categorical variables was used to determine whether restoration significantly affected maximum daily groundwater level. (Season was included as an independent variable because the restoration site already met wetland hydrology criteria in winter prior to restoration.) A 2-way ANOVA (site and year as factors) was used to determine the restoration effect of restoration on continuous duration of shallow groundwater. ANOVA was conducted in R using the general linear models ("glm") package; significant differences between means were determined using the "lsmeans" package within general linear models, and displayed with the compact letter display of pairwise comparisons ("cld") routine.

Results

Average across all transects

Restoration significantly increased groundwater levels at Ni-les'tun, and increased the amplitude of tidally-related groundwater level fluctuation. Tidal groundwater regimes have returned to the entire Ni-les'tun restoration site, restoring the dynamic water table movements typical of tidal wetlands (Figure 10). The effect of restoration on groundwater levels was highly significant ($p < 0.0001$). Although the degree of tidal influence on groundwater level at Ni-les'tun was generally similar to the reference site (Figure 10), average post-restoration groundwater levels at Ni-les'tun were significantly higher than at the reference site (Figure 11), reflecting subsidence that has occurred at the site and possibly also reflecting more compact soils (Brophy and van de Wetering 2012; also see **Sampling locations** above).

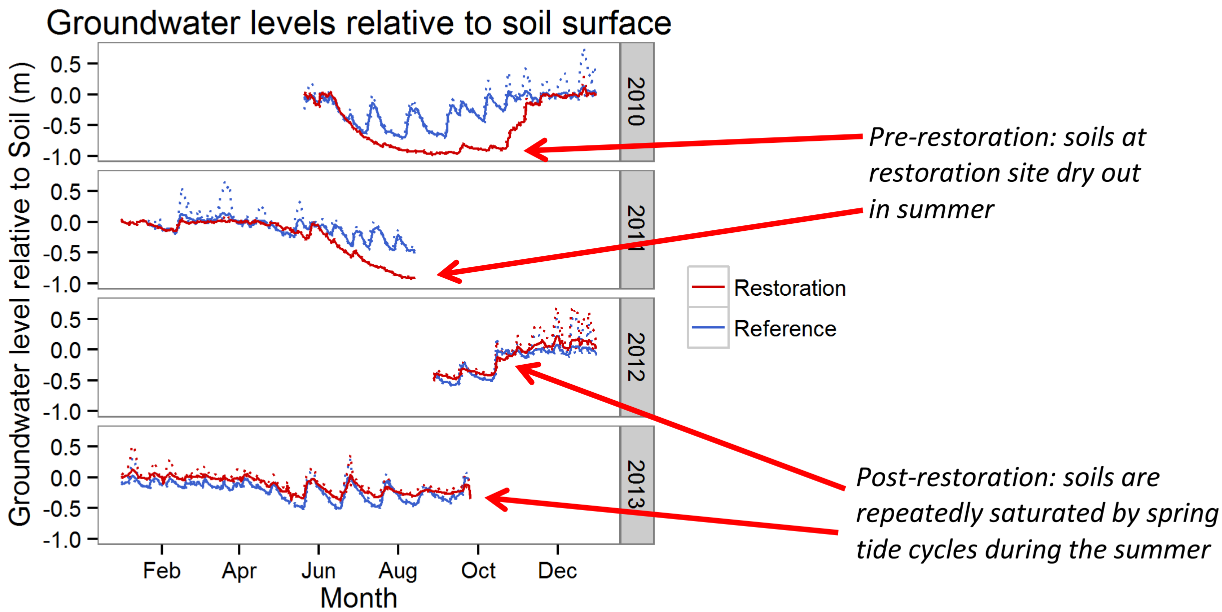

Figure 10. Average groundwater level relative to the soil surface across all transects at restoration and reference site during pre-restoration (2010-2011) and post-restoration (2012-2013). After restoration, groundwater fluctuations at the restoration closely matched patterns at the reference site, and the summer drying period was absent.

Before restoration, summer groundwater levels at Ni-les'tun averaged 70 cm below the soil surface (Figure 11), indicating that many areas did not meet wetland criteria in the summer. After restoration, average groundwater levels rose to within 30 cm of the soil surface year-round, indicating that the site had wetland hydrology year-round rather than just seasonally.

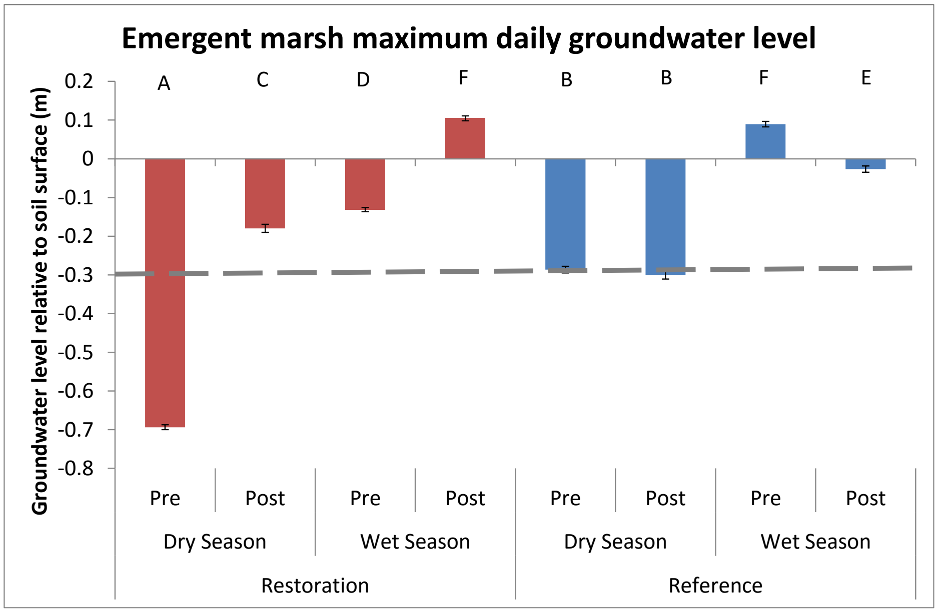

Figure 11. Mean daily maximum groundwater level relative to the soil surface for emergent marsh at Ni-les'tun restoration site and Bandon Marsh Unit reference site, dry and wet seasons, pre-restoration (2010-2011) and post-restoration (2013) periods. The dashed gray line indicates a depth of 30 cm below the soil surface, which defines wetland hydrology. Bars with a letter in common are not significantly different ($p > 0.05$).

Shallow groundwater duration did not differ significantly between the restoration site and reference site either before or after restoration. Shallow groundwater duration for restoration and reference sites averaged over 4000 hours both before and after restoration, well above the threshold of approximately 1000 hours that defines wetland habitat (Environmental Laboratory 1987).

Emergent wetlands: summer

As described in the baseline monitoring report (Brophy and van de Wetering 2012), the Ni-les'tun pasture was a seasonal wetland prior to restoration, with water tables at or near the surface across the entire pasture in winter. Because groundwater levels were already high, restoration had little effect on soil saturation in winter (although restoration did create tidal inundation peaks in winter). By contrast, restoration strongly affected soil saturation at Ni-les'tun in summer. Therefore, summer groundwater patterns were the most important to understanding the effects of restoration at this site.

Prior to restoration, soils at Ni-les'tun dried out during summer, with groundwater levels that dropped to non-wetland levels (>30 cm below the soil surface) around July 1 (2010) and remained there until the fall rains returned (Brophy and van de Wetering 2012). During the pre-restoration period, distinct differences in groundwater regime were observed across the Ni-les'tun site, as described in the baseline monitoring report (Brophy and van de Wetering 2012): most areas were seasonal wetlands, but the northwest parts of the site (e.g. NL T5) showed the influence of hillslope seepage on groundwater even in summer; and the lowest parts of the site near the Fahys tide gates showed muted tidal influence.

After restoration, the only transects at Ni-les'tun that did not have year-round daily groundwater maxima within 30 cm of the soil surface were those at the highest elevations and farthest from tidal channels

(NL T5, NL T12, and NL T17, and NL T20) – and even these showed tidal groundwater patterns very similar to the reference site. Similar tidally-influenced groundwater patterns have been observed at other least-disturbed reference sites in Oregon (Brophy 2009, Brophy *et al*. 2011).

Transects at low and middle elevations at Ni-les'tun (the vast majority of the site) had groundwater at or near the soil surface all summer, and showed inundation peaks corresponding to high tides (NL T18 and NL T13 in Figure 12 are typical examples). These patterns are similar to the low marsh groundwater patterns observed in the Siuslaw estuary (Brophy and Lemmer 2013). Higher transects (e.g. NL T19 in Figure 12) had groundwater that dropped below 30 cm during neap tide cycles, but rose to the surface during each spring tide cycle – what we refer to as a "spring tide reset" groundwater pattern (Brophy *et al*. 2011, Brophy and van de Wetering 2012). This "spring tide reset" was also the typical pattern at the reference site (Figure 13) and at other least-disturbed high marshes in Oregon (Brophy *et al*. 2011). At the highest emergent transect on the river bank at Ni-les'tun (NL T17), groundwater patterns were similar to pre-restoration patterns, with tidal influence only during spring tide cycles and deep drying in summer. The next three highest transects at Ni-les'tun (NL T5, NL T12, and NL T16) showed clear "spring tide reset" groundwater patterns similar to the reference site and NL T17, but with less drying and greater tidal influence (Appendix A, Figure A30). As the tidal channel system develops and with the addition of further channel excavation, we expect these areas to show even more "spring tide reset" events during summer.

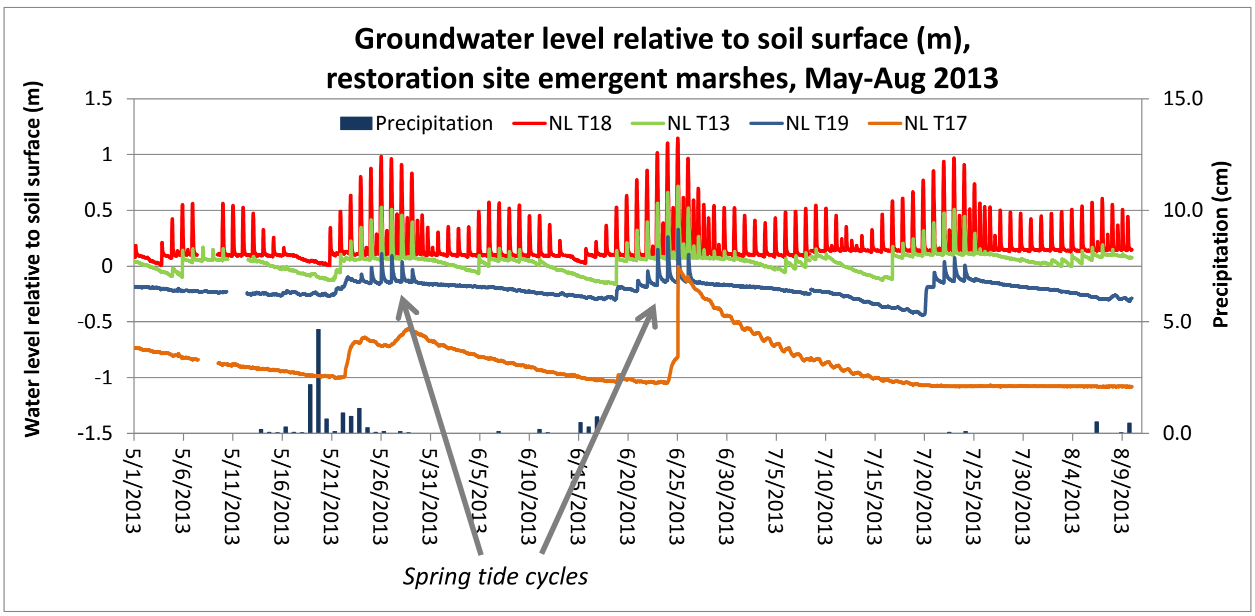

Figure 12. Spring and summer groundwater levels after restoration (relative to the soil surface) for four representative emergent marsh transects at Ni-les'tun, covering the full range of elevations across the site All show tidal influence on groundwater; NL T17 is on the high river bank and dries out during summer.

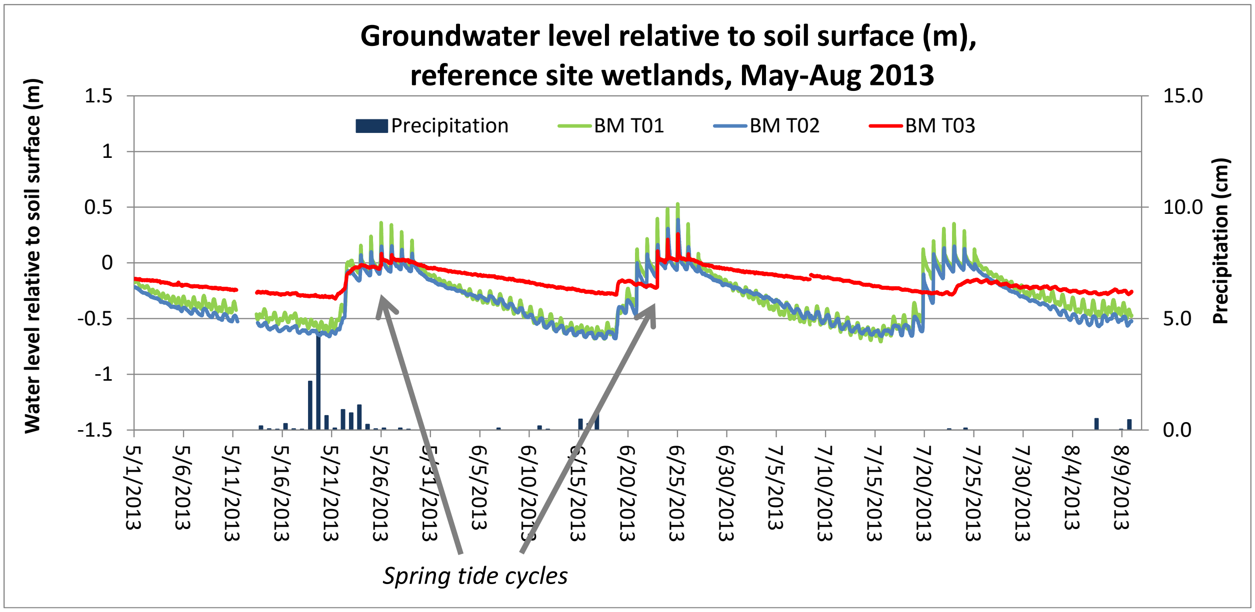

Figure 13. Spring and summer groundwater levels at emergent marsh transects, Bandon Marsh Unit reference site. BM T2 and BM T2 show the “spring tide reset” groundwater pattern typical of high tidal marsh. BM T3 shows the influence of non-channelized freshwater inflows (hillslope seepage) as well as tidal peaks during spring tide cycles.

Emergent wetlands: Winter

After restoration, wintertime groundwater levels were high throughout Ni-les’tun, generally remaining within 30 cm of the soil surface (Figure 14). This was not a change from pre-restoration conditions; as described above, Ni-les’tun was a seasonal wetland prior to restoration (i.e., it met the definition of wetland in winter months). Winter groundwater patterns were similar at Ni-les’tun (Figure 14) and the reference site (Figure 15). The Ni-les’tun tidal peaks matched closely with the peaks from our in-channel tide gauges, indicating full penetration of tides throughout the site (see **Tidal hydrology** above and Appendix A, Figure A15).

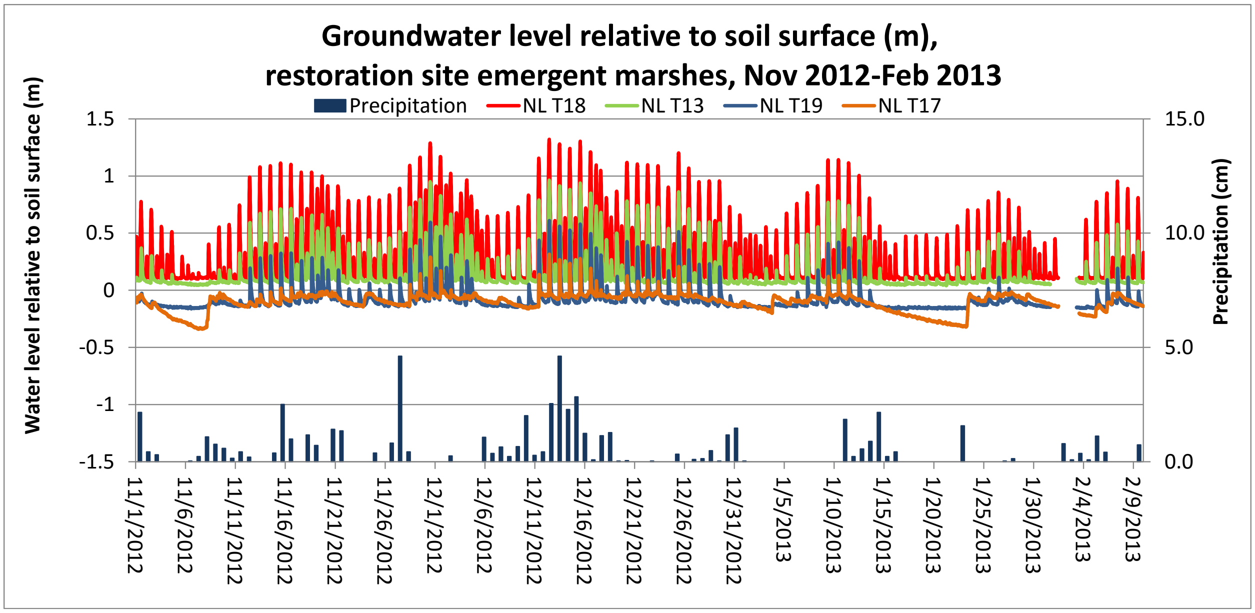

Figure 14. Winter groundwater levels after restoration (relative to the soil surface) for four representative emergent marsh transects at Ni-les'tun. These four transects cover the full range of elevations present across the restoration site (for elevations, see Appendix B, Table B2).

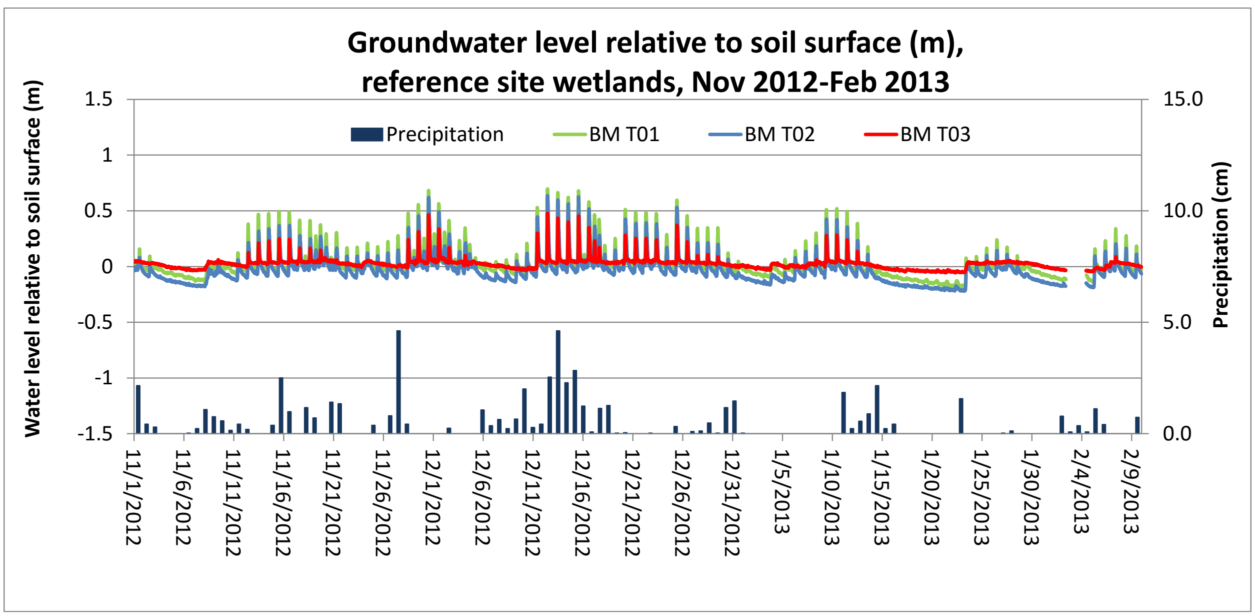

Figure 15. Winter groundwater levels at emergent marsh transects, Bandon Marsh Unit reference site.

Forested wetlands

Groundwater data showed clearly that the Ni-les'tun project restored highly-prioritized tidal swamp (forested tidal wetland) as well as tidal marsh. Forested transects NL T6 and NL T20 showed tidal groundwater patterns and tidal surface inundation year-round, especially in winter (Figure 16; Appendix A, Figures A31 and A33). NL T7 (north of North Bank Road) was not tidally influenced, but groundwater records showed evidence of "backup" of freshwater outflows by high tides following precipitation events

(Figure 16). Summer and winter groundwater patterns in Ni-les'tun's forested tidal wetlands were similar to the forested wetland at the reference site (BM T5 in Appendix A, Figures A32 and A33).

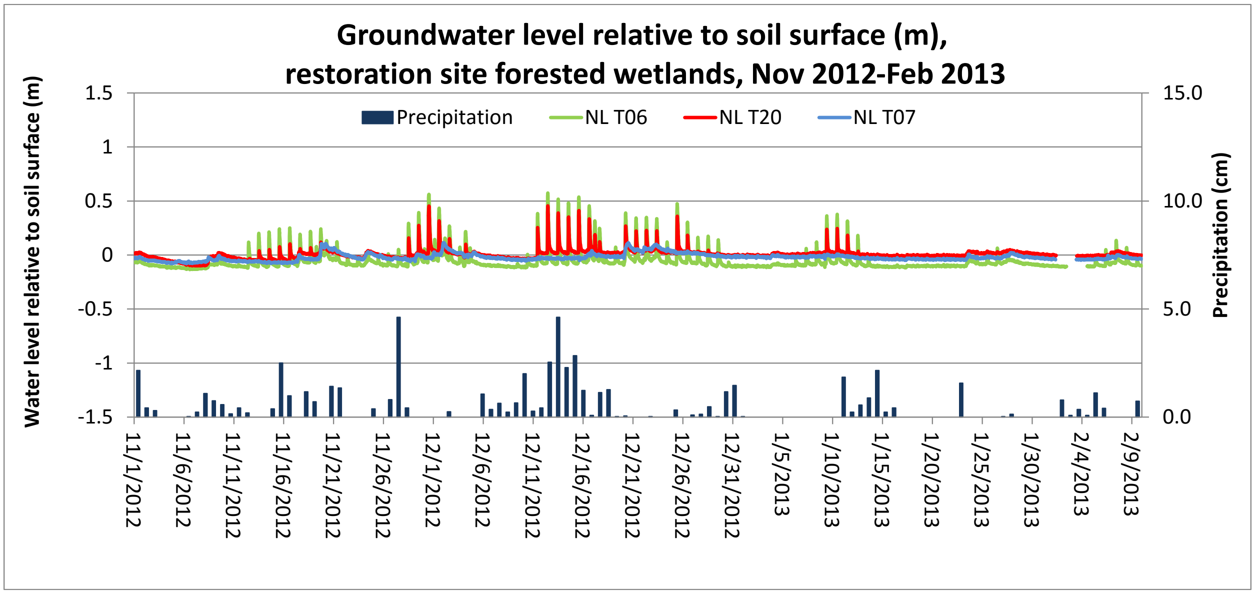

Figure 16. Winter groundwater levels after restoration (relative to the soil surface) at Ni-les'tun forested transects. NL T7 is non-tidal.

Channel water salinity and temperature

Methods

Surface water temperature and salinity were measured using Odyssey conductivity-temperature dataloggers at 17 locations: 10 on the restoration site, two in the mainstem Coquille River, and five at the reference site (Appendix A, Figures A2 and A3). Two loggers were deployed at a single site (Channel 7) in the restoration site to test installation methods; these are "CH7 8498" and "CH7 CHK 8502" in Figure A2. Conductivity and temperature were logged at 30 min intervals from May 1 through November 17, 2011 and February 2 through September 24, 2013. Loggers were installed in stilling wells for protection and to avoid fouling by debris; sensors were placed about 15 cm above the bottom of the channel to maximize the inundated period. Field validation and QA/QC were conducted as described in the project's Sampling and Analysis Plan (Brophy 2013). Data from some loggers and some time periods were lost due to storm damage and sediment deposition.

Statistical analysis was conducted separately for channel mouth *versus* upper channel locations, since we expected these areas to respond differently to the restoration action. (For example, brackish salinity was already present at channel mouth locations prior to restoration due to leaky tide gates.) Loggers included in the channel mouth category were FAHY MTH 8239, NONAM MTH 8231, REDD MID 8240, BM UNKC 8233, and SHPWRK A 8238; loggers in the upper channel category were FAHY MID 8230, FAHY RD 8241, BM UNK A 8235, BM UNK B 8232, and SHP WRK B 8229 (see Appendix A, Figures A2 and A3 for locations). Pre- *versus* post-restoration salinity were analyzed using May 1 through August 15 data, since this was a period of overlap between pre- and post-restoration datalogger records and is also the period of interest for fish use of the site (see **Salmonid habitat opportunity** below). Comparisons of pre- and post-restoration salinity were made using BACI (ANOVA) with location (restoration site *versus* reference site) and year (2011 *versus* 2013) as categorical independent variables. ANOVA was conducted in R using the general linear models ("glm") package; significant differences between means were determined using the "lsmeans" package within general linear models, and displayed with the compact letter display of pairwise comparisons ("cld") routine.

Results

This section provides an overview of salinity and temperature changes at Ni-les'tun following restoration. For detailed discussion of salinity and temperature monitoring results as they relate to fish habitat suitability, see **Salmonid habitat opportunity** below.

Salinity

Restoration led to a significant increase in channel water salinity at Ni-les'tun (Figures 17 and 18; Appendix B, Tables B23 and B24), and brackish tidal flows penetrated to the uppermost channels on the site (Figures 19, 33, and 34). The increase in salinity was greatest for the upper channels farthest from the river, because brackish salinity was already present at channel mouth locations prior to restoration due to leaky tide gates. Daily mean salinity during May through August in the middle and upper reaches of the Fahys channel averaged 0.05 PSU before restoration and 6.0 PSU after restoration; daily maximum salinity in these locations increased from 0.1 before restoration to 15.4 PSU after restoration (Appendix B, Table B26). At channel mouths, daily mean salinity did not increase significantly, but daily maximum salinity increased significantly, from 12.8 PSU in 2011 to 25.9 PSU after restoration (Figure 18; Appendix B, Table B25).

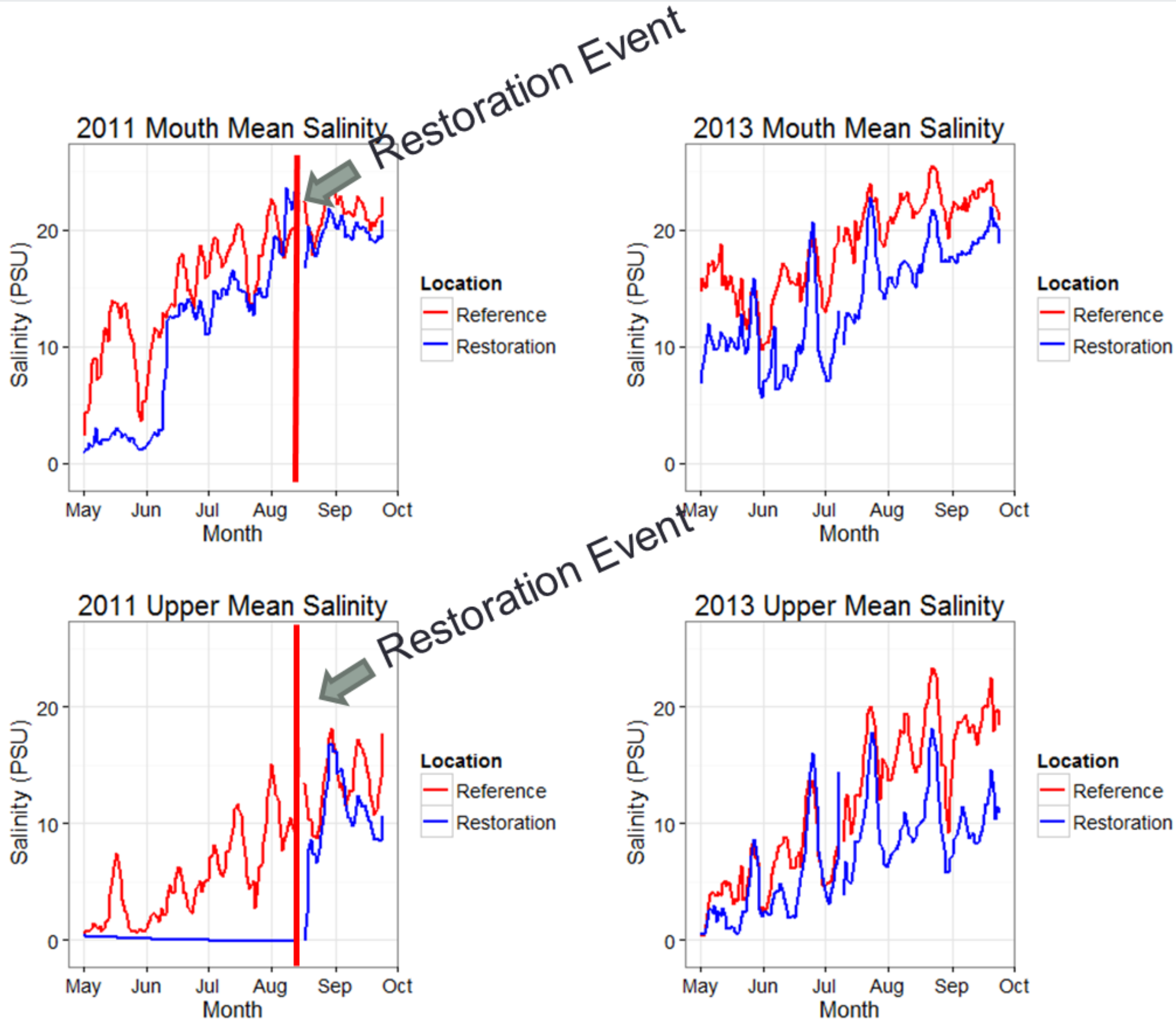

Figure 17. Average daily salinity at channel mouth and upper channel locations at Ni-les'tun restoration site and Bandon Marsh Unit reference site during pre-restoration (2011) and post-restoration (2013).

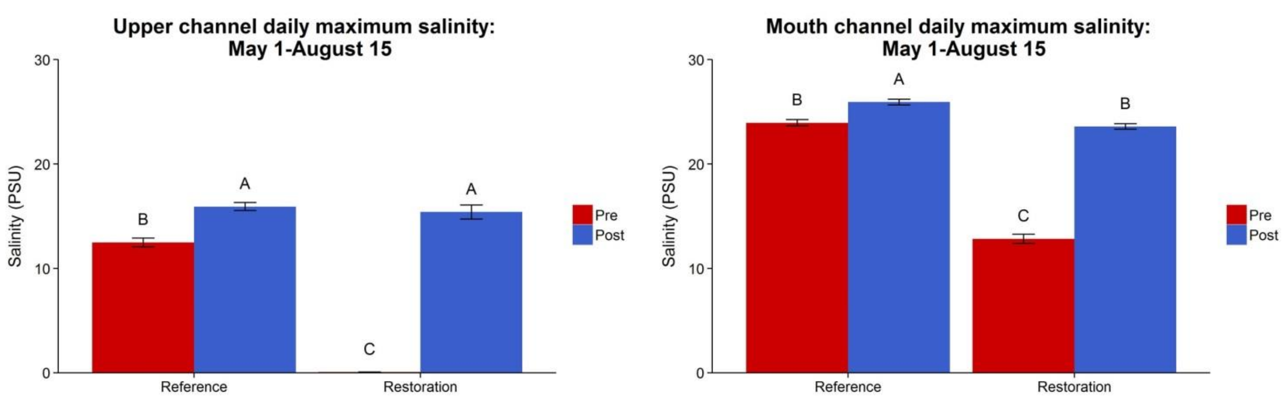

Figure 18. Daily maximum salinity at upper channel and channel mouth locations at the restoration and reference sites, pre-restoration (2011) *versus* post-restoration (2013), May 1 - August 15. Bars with a letter in common are not significantly different ($p > 0.05$). Daily mean salinity comparisons are found in Appendix A, Figure A34 and Appendix B, Tables B25 and B26.

Spring tide peak salinities were high throughout Ni-les'tun after restoration. For example, peak salinities in the Fahys channel just upstream of North Bank Road (FAHY RD 8241 logger) were generally close to marine salinity (28-30 PSU), and peak salinities in upper Channel 5 (CH5 UPR 8500 logger) exceeded 32 PSU. Peak salinities may have strong effects on plant community development, since they occur during the dry summer season when plants are most stressed. However, in general, Ni-les'tun salinities were lower than the reference site after restoration ($p = 0.0001$; Figure 18 right side, and Appendix A, Figure A34). This is probably due to the site's freshwater inflows (Fahys Creek and Redd Creek) as well as its location farther upstream in the estuary.

During spring and summer post-restoration, the daily salinity regime was very dynamic in the restoration site's tidal channels with freshwater flow (Fahys Creek and Redd Creek) (Figure 19). These channels now provide a wide range of osmotic regulation opportunities for juvenile salmonids and other anadromous fish during critical spring and summer periods. For further details, see **Salinity** in **Salmonid habitat opportunity** below.

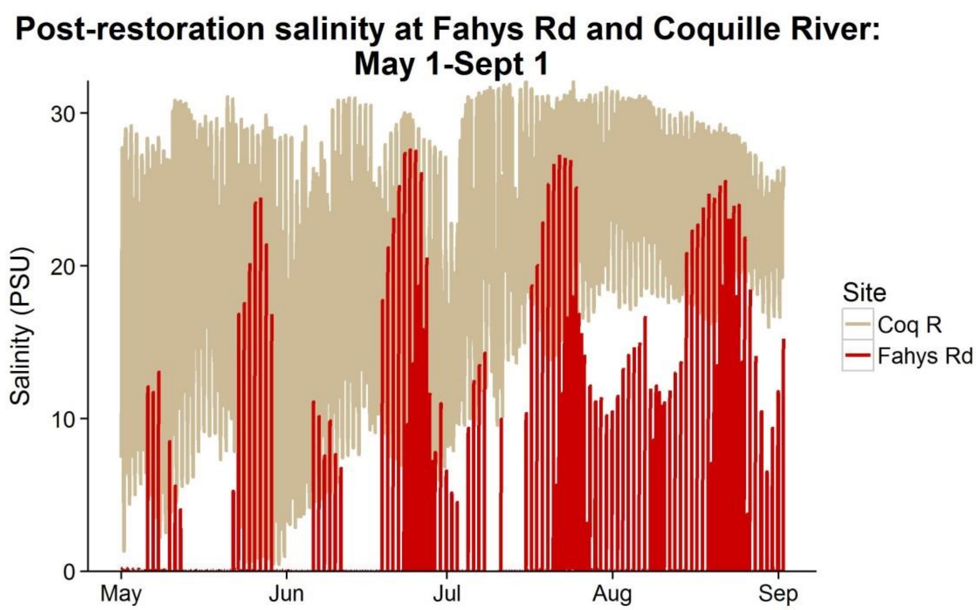

Figure 19. Channel water salinities in the logger located farthest from the river (FAHY RD 8241 logger) and in the Coquille River, May to September 2013. Colored bars indicate the full range of salinity fluctuation during each day. At the Fahys Rd logger, salinity fluctuated between zero and nearly 30 PSU during many summer days.

Vertical stratification of salinity and temperature may have been present within tidal channels at Ni-les'tun, both because of freshwater outflows and because of denser seawater moving onto the site from tidal forcing. If stratification was present, the salinity of tidal waters flooding the wetland surfaces could be lower than our observations near the bottom of tidal channels. However, post-restoration plant community composition clearly indicated that brackish water was inundating the entire Ni-les'tun site (see **Emergent wetland plant communities** above).

Temperature

BACI analysis was essential to our understanding of temperature monitoring results. BACI analysis showed that restoration was associated with significantly lower daily mean and daily maximum water temperature compared to the reference site in Ni-les'tun's upper channels (Figures 20 and 21; Appendix A, Figure A35). At channel mouth locations at Ni-les'tun, restoration was associated with lower daily maximum temperatures relative to the reference site (Appendix A, Figure A35; Appendix B, Tables B27-B30). Daily mean temperature at channel mouths after restoration was not significantly different from pre-restoration conditions or from reference (Figure 21). Temperatures were higher in 2013 than in 2011, but this was not the effect of restoration; temperatures rose at both the reference site and the restoration site, and the restoration site rose less (Appendix B, Tables B29-B30). BACI allowed us to separate year-to-year variability from restoration effects.

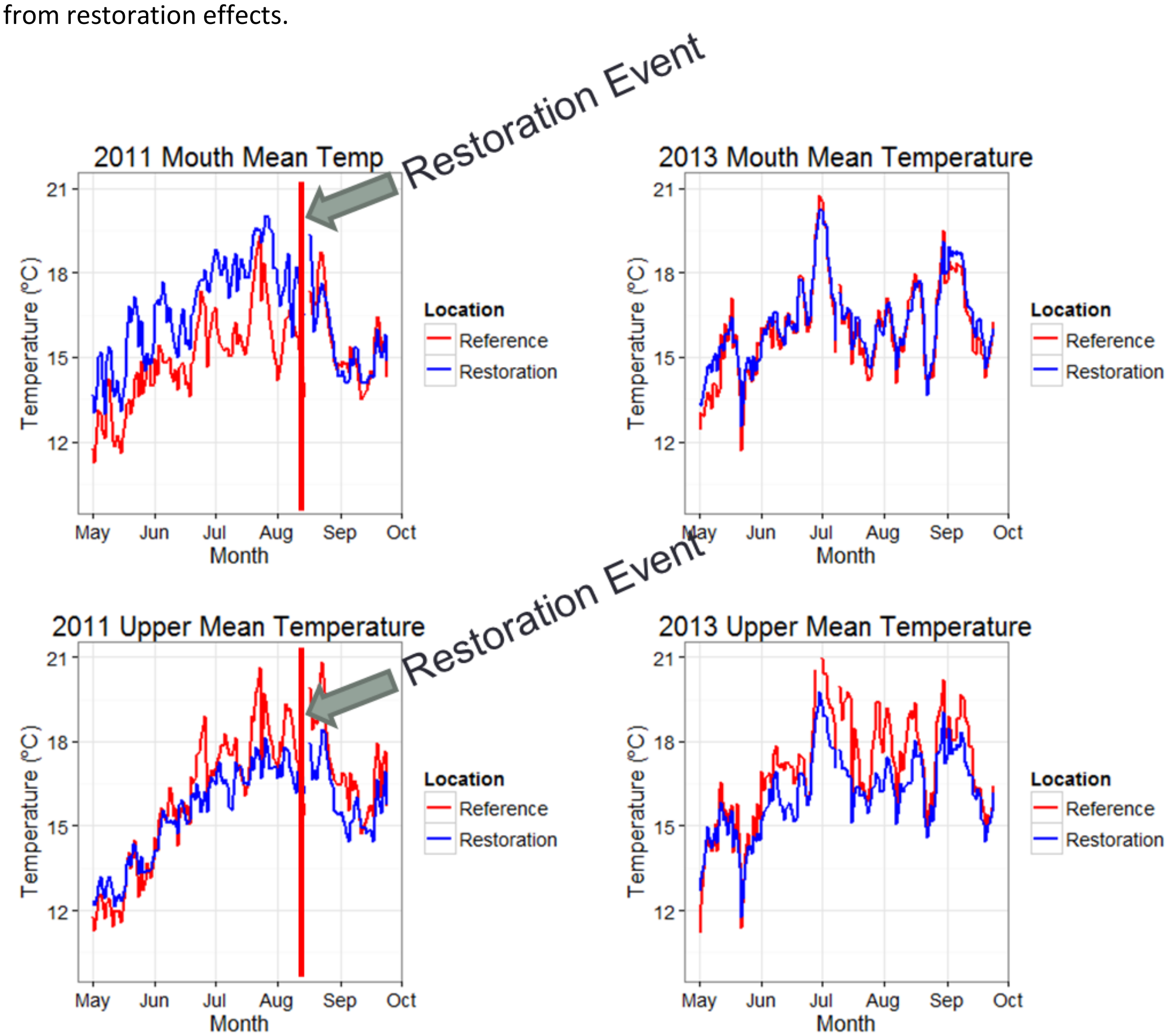

Figure 20. Daily mean water temperatures at channel mouth and upper channel locations, before and after restoration at the Ni-les'tun restoration site (red) and Bandon Marsh Unit reference site (blue).

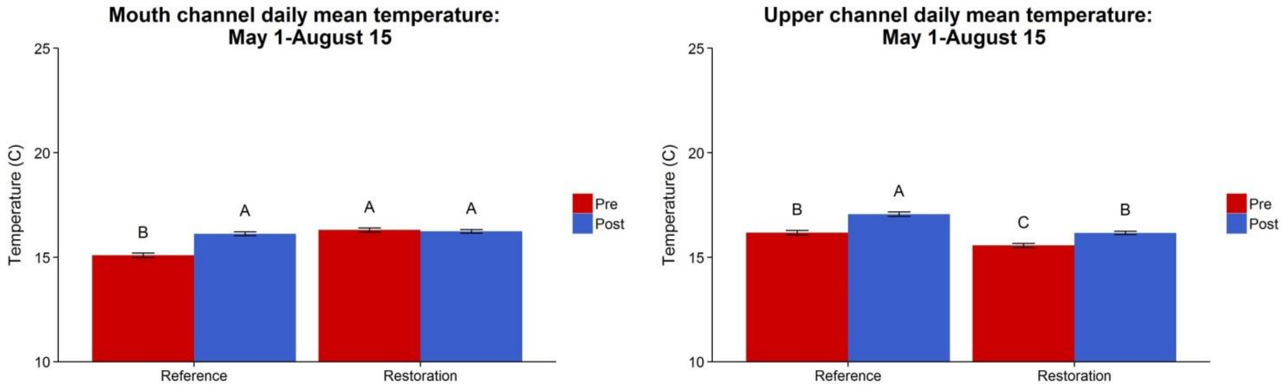

Figure 21. Daily mean temperature at upper channel and channel mouth locations at the restoration and reference sites, pre-restoration (2011) *versus* post-restoration (2013), May 1 - August 15 period. Bars with a letter in common are not significantly different ($p > 0.05$). Other comparisons are found in Appendix A, Figure A35, and Appendix B, Tables B27-B30.

Lessons learned about salinity logger deployment and operation

Placement of salinity/temperature loggers was challenging. Loggers were installed near channel bottoms so they were immersed as much as possible, but channels deepened in three locations, so those loggers were adjusted downwards during their deployment to attempt to capture more data. After this adjustment, sediment re-deposition occurred in some locations. Although we conducted field inspections of logger installations, sediment re-deposition was not always detected, and in some cases, sediment clogged the logger orifices, reducing logger accuracy. Data from these locations were omitted from analyses. We recommend frequent and careful inspections of installations and of logger positioning relative to the sediment surface. We also recommend field validation inside and outside protective housings to accurately validate field measurements.

Field validation and regular calibration of salinity/temperature loggers were conducted as described in the project Sampling and Analysis Plan (SAP) (Brophy 2013). However, field validation was challenging because of the distance and travel time between loggers and the 30 min data collection interval. (Field validation must be conducted at the time of a data collection event.) We recommend shorter data collection intervals (e.g. 15 min) to facilitate field validation at the beginning and end of every deployment period. However, these shorter intervals create much larger datasets, making data processing more time-consuming, so there are clearly trade-offs.

Linkages between percent inundation, physical and biological characteristics

In tidal wetlands, the inundation regime plays a predominant role in site structure and biological development. Therefore, we expected many physical and biological site characteristics to relate closely to elevation and inundation regime. To investigate these relationships, we conducted linear regression on percent inundation *versus* several of our monitoring metrics: soil carbon content, soil salinity, shallow groundwater duration, prevalence index, total plant cover, native plant cover, and plant species richness. Shallow groundwater duration is described in **Groundwater** above. Prevalence index is a measure of the proportion of plant cover that consists of hydrophytic (wetland-tolerant) species (U.S. Army Corps of Engineers 2008). Percent inundation calculations are described in **Tidal hydrology** above.

Soil salinity, shallow groundwater duration, prevalence index, and plant species richness were significantly correlated with percent inundation ($R2 = 0.70, 0.65$, and 0.38 respectively; $p \leq 0.01$ in all cases) (Appendix A, Figure A36; Appendix B, Table B31). The causal links for these relationships are quite clear: soil salinity is higher where brackish tidal flows inundate more frequently, and shallow groundwater duration also relates to frequency of inundation. Prevalence index is likely to be higher in wet areas that inundate frequently. Plant species richness is generally lower where growing conditions are more harsh, such as brackish and very wet areas. These close relationships suggest that biological and structural characteristics at Ni-les'tun were changing in response to the restored controlling factors such as inundation and salinity regimes -- factors known to affect vegetation composition and diversity (Watson and Byrne 2009, Janousek and Folger 2014). Such biological and structural responses to restored controlling factors provide evidence of effective restoration.

2. Salmonid habitat functions

Monitoring Objective 2: Measure habitat recovery and habitat use by at-risk and endangered species

2a. Salmonid habitat opportunity (availability)

Monitoring Question 2a: Did restoration result in increased salmonid habitat opportunity (availability)?

Metrics: Duration and frequency of salmonid habitat availability (using channel morphology measurements and tidal elevations); surface water salinity and temperature; locations, quantities, and descriptions of large wood habitat restored. *(Rationale: Habitat availability is determined by analyzing water levels [tidal exchange]; surface water salinity and temperature affect habitat use; large wood location and size have been correlated to salmonid behavior and habitat use.)*

More than 20 km (13 miles) of drainage ditches were filled during the multiyear restoration process. Of the original 20 km of ditches at Ni-les'tun, 5 km (3.1 miles) were estimated to be potentially habitable by juvenile fish during the winter and early spring seasons. During 2009-2010, the larger ditches were filled, the smaller ditches were disked to reduce their impact on site drainage, and approximately 8 km (5 miles) of restored channels were excavated (Appendix A, Figure A8).

Duration and frequency of tidal channel habitat availability

Methods

We used our tidal elevation data sets to evaluate the extent of tidal habitat available within the three restoration sub-basins (Fahys, NoName, and Redd) and the reference sub-basin (Shipwreck). Our tidal fish migration results show juvenile salmonids entered the tidal channels when water depths exceeded 0.46 m. We calculated percent inundation and the number of hours that fish could access a series of representative channel bottom locations (Appendix A, Figure A11) by adding 0.46 m to surveyed channel bottom elevations. Calculations were made for access prior to tidal flow restoration (but after channel construction) in 2009, and after restoration in 2013, for a full year period and for the month of May. The results allow comparison of juvenile salmonid tidal migration opportunities (access) before and after restoration and between the restoration and reference sites.

Results

Fish access to channels at Ni-les'tun was greatly increased by restoration. Before restoration, fish could access the middle channels of the NoName sub-basin less than 2% of the time, compared to about 27% after restoration (Figure 22; Appendix B, Table B32). Fish could access the mouth of Channel 5 (in the center of the site) less than 14% of the time prior to restoration, compared to 46% after restoration (Figure 25; Appendix B, Table B32). Based on the channel depth changes observed during 2010 to 2013 (see **Channel depth** above), it seems likely the Ni-les'tun channels will continue to deepen, so fish access will remain high or increase. USFWS also plans to excavate additional tidal channels during summer 2014 as part of the next phase of restoration (USFWS 2014b); this additional channel restoration will further enhance salmonid habitat availability.

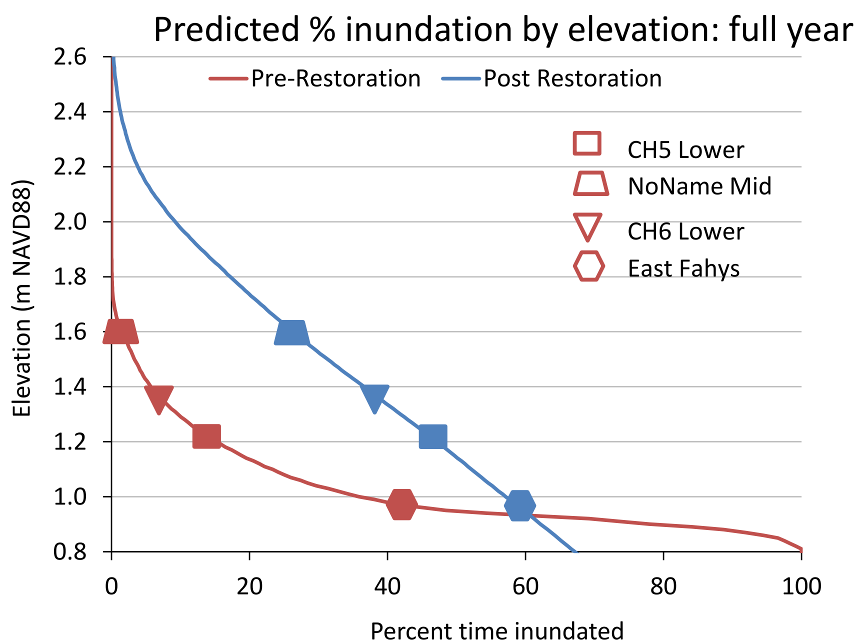

Figure 22. Frequency of fish habitat access opportunity for 4 typical channel locations at Ni-les'tun before (2009) and after restoration (2013). Curves show predicted percent inundation by elevation; markers show required water surface elevations at the 4 locations that would allows fish access (0.46m above channel bottom). For locations, see Appendix A, Figure A11; for data, see Appendix B, Table B32.

Fish access through channel mouths at Ni-les'tun was substantially greater than at the reference channel. Using the reference marsh channel (Shipwreck) as the standard, hours of inundation available for tidal migration were estimated at 2.2, 2.1 and 1.4 times that of the reference channel mouth, respectively, for Fahys Lower, NoName Lower, and Redd Lower (Figure 23). When considering the extent of tidal migration opportunities in Fahys mainstem and upstream into tributary Channel 5, inundation time decreased on an approximately linear scale (Figure 24). For NoName (Figure 25), the upper reaches had slightly longer duration of fish access due to the lower channel bottom elevations in upper *versus* middle reaches (the "reverse gradient" described in **Channel morphology: Longitudinal profiles** above).

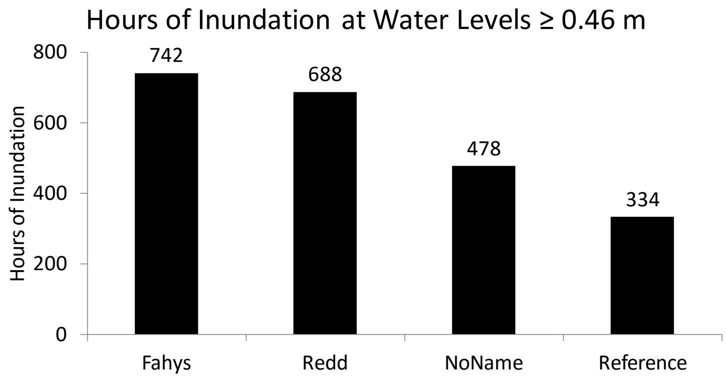

Figure 23. Hours of inundation at the mouths of the three restored and single reference sub-basins during the month of May 2013.

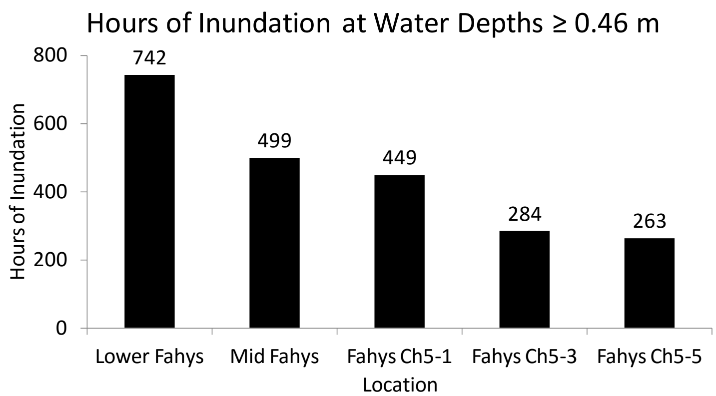

Figure 24. Hours of inundation at various locations moving upstream through Fahys sub-basin during the month of May 2013.

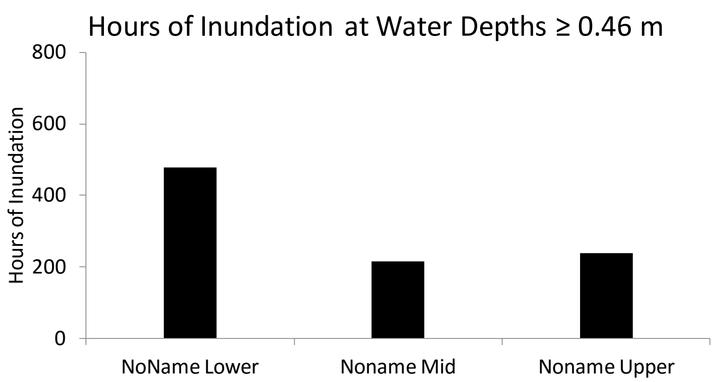

Figure 25. Hours of inundation at various locations moving upstream through NoName sub-basin during the month of May 2013.

Discussion

The history, landscape setting and hydrology of the restoration site have contributed to the morphological differences observed between the reference and restoration channels. For example, historical freshwater flows into the restoration site carried fine sediments which accumulated in ditches and settled behind tide gates. Pasture management activities included ditch cleaning and dredging, which altered channel morphology prior to the restoration effort. An unanswered question is the extent to which the freshwater perennial streams in the Fahys and Redd sub-basins have influenced channel bottom gradients. Our surveys of channel bottom elevations at Ni-les'tun suggest that the greater the wintertime freshwater stream discharge (which is related to stream catchment area), the lower the elevation of the tidal channel bottom, providing fish with more time for marsh channel access during daily tidal cycles. For example, the Fahys Creek catchment area is greater than Redd Creek's catchment, both of which are much larger than the catchments of NoName channel and the reference channel. These relationships are reflected in the relative hours of inundation at the channel mouths: the greater the catchment area, the greater the hours of inundation (Figure 23). Similar relationships have been documented in channel allometry studies in California (Coats *et al*. 1995) and Washington (Hood 2002). As the restoration marsh channels move toward equilibrium, the channel bottoms will always maintain lower elevations than those of the reference channel, as long as upstream sediment sources and sediment transport remain equal. This, in turn, suggests greater rates of juvenile salmonid accessibility in marsh channels associated with larger catchment areas (e.g., Fahys and Redd) and less accessibility (hours of inundation) in "blind" tidal channels (without connection to a freshwater stream), such as NoName and the reference channel.

Temperature

Methods

Stream temperature data summaries and analysis provided earlier in this report examined mean and maximum daily temperatures. In this section, we further that analysis by examining temperatures within categories recognized within the Oregon Adminstrative Rules. These include ≤13° C (spawning), 13.1-18° C (core cold water + rearing and migration), and >18° C (growth limiting and potential indirect mortality).

Results

As described in **Channel water salinity and temperature** above, BACI analysis and means comparison showed that spring and summer water temperatures at the restoration site and reference site were generally warmer in 2013 compared to 2011, reflecting year-to-year variability in weather. BACI analysis also showed that restoration was associated with a significant reduction in spring and summer water temperatures at Ni-les'tun compared to the reference site, for 3 of the 4 comparisons conducted (see **Channel water salinity and temperature** above). However, water temperature changes before and after restoration at Ni-les'tun were more complex than can be shown through BACI analysis. Below, we present another view using analysis based on fish habitat criteria and subdivided by month.

After restoration, Fahys Lower experienced less time at temperatures >18° C from May to August (Figure 26). The more extreme pre- *versus* post-restoration shifts in this highest temperature category occurred during late summer; the percent of time in this category dropped from 84% to 21% and 81% to 9% in July and August, respectively. In addition, the percentage of time in the ≤13° C range increased after restoration, with the exception of the month of May. Overall, the trend in Fahys Lower was toward cooler temperatures across both seasons.

After restoration, Fahys Mid experienced more time at temperatures >18° C during May and June, and less time during August (Figure 26). The most extreme shift was a 48% reduction in time spent in the <13° C category during May. Overall, the trend at Fahys Mid was toward warmer temperatures in the spring and cooler temperatures in the summer. Fahys Upper experienced more time at temperatures >18° C during June, July, and August (Figure 26) and a 35% reduction in temperatures <13° C during May. Overall the trend here was toward warmer temperatures during May through August.

A comparison of pre and post-restoration temperatures in the Coquille River show that during the post-restoration period, May was warmer and August was cooler. During post-restoration, Coquille River temperatures in the <13° C category decreased during May, June and August, but increased during July (Figure 26). Temperatures in the >18° C category increased slightly during May and June and dropped during July and August (Figure 26). These post-restoration shifts were largely due to a warmer, drier spring followed by increases in upwelling and the corresponding cooler saline waters that flood the lower estuary during the low flow period of July and August.

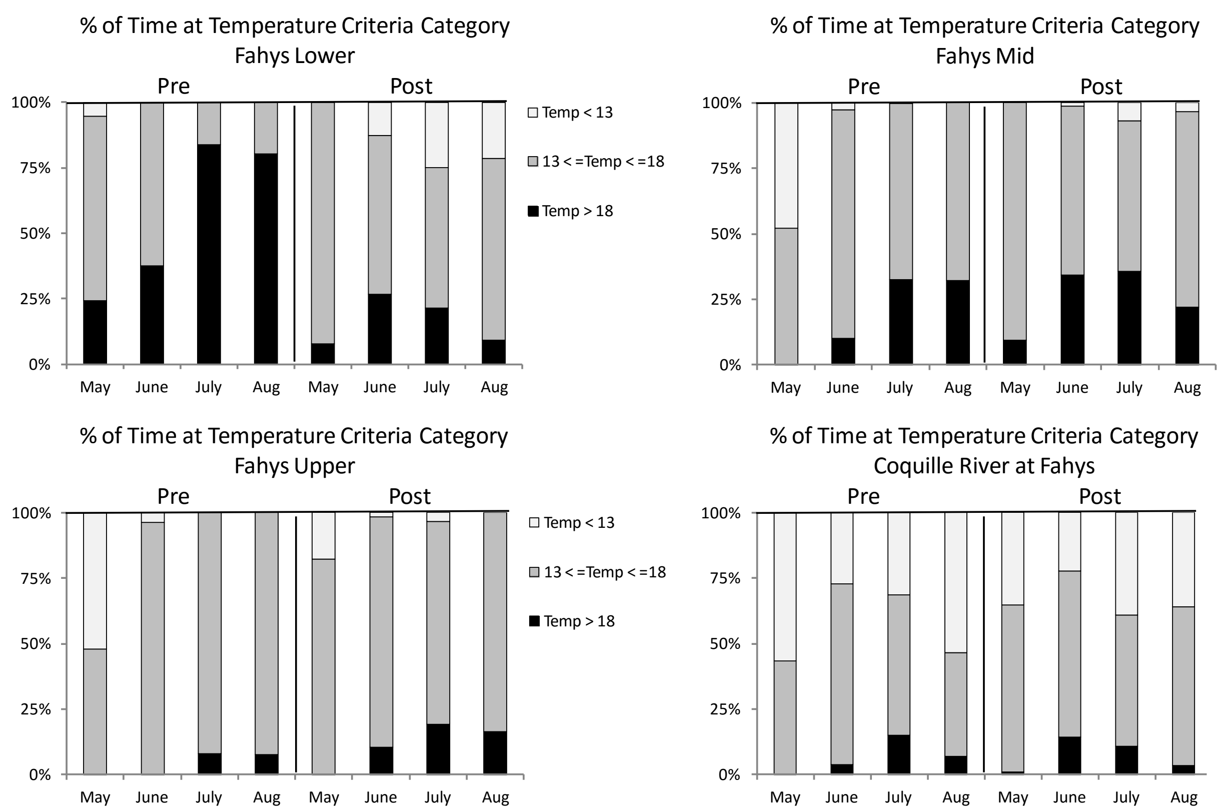

Figure 26. Percent of time that temperatures were within Oregon DEQ stream temperature criteria categories during May-August, pre-restoration (2011) and post-restoration (2013), for Fahys Lower, Mid and Upper at Ni-les'tun and the Coquille River mainstem. See Appendix A, Figure A2 for monitoring locations and Figure A4 for reach names. Labels in Figure A2 are as follows: Fahys Lower = Fahy Mth 8239; Fahys Mid = Fahy Mid 8230; Fahys Upper = Fahy Rd 8241; Coquille River at Fahys = CoqR 8234. Temperature ranges in the center of the figure apply to all panels.

We also analyzed May and August post-restoration water temperatures in the mainstem Coquille compared to the Fahys sub-basin and the reference site (Shipwreck A), to better understand habitat suitability for fish growth and rearing. During May the restored Fahys sub-basin provided temperatures very similar to the Coquille River and the reference sub-basin (Shipwreck) (Figure 27). The exception was Fahys Upper, where the overall temperature range was truncated in comparison to the mainstem river.

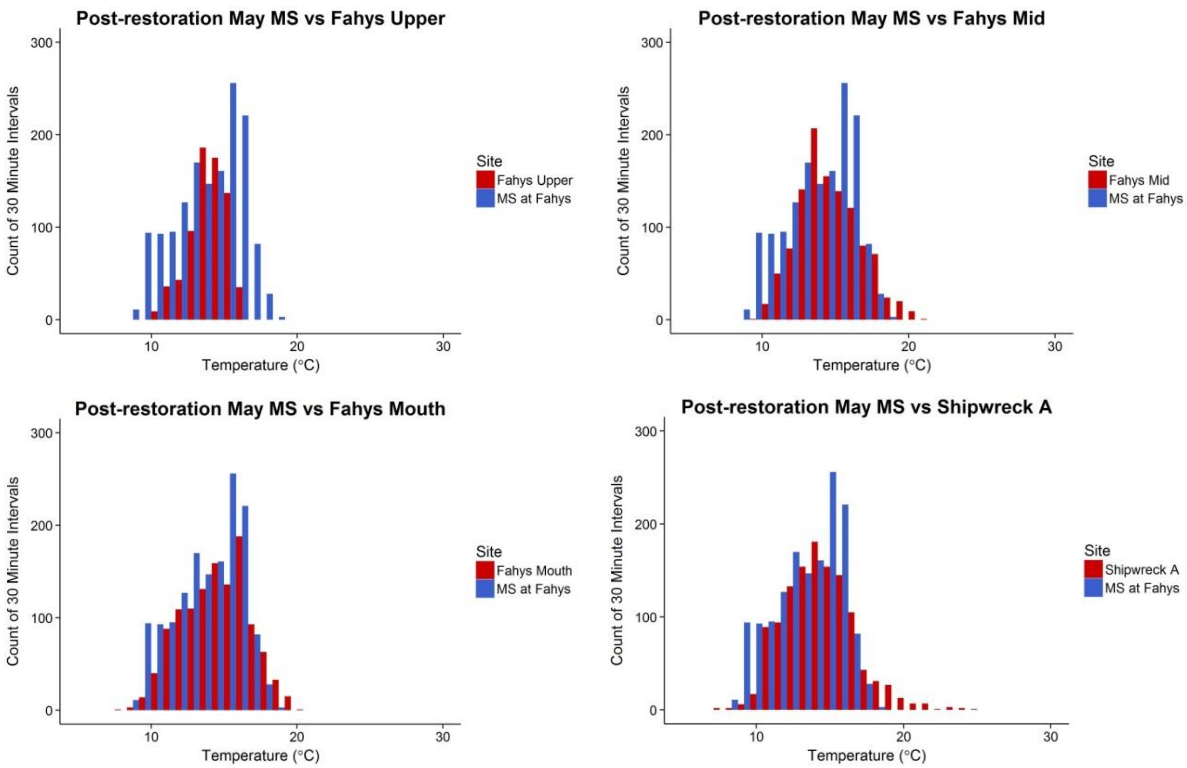

Figure 27. May temperature histograms for the post-restoration period for the mainstem Coquille ("MS at Fahys," labeled "CoqR 8234" on Figure A2 in Appendix A), three Fahys channel reaches (see Figure 26 caption for location codes), and the lower reference reach (Shipwreck A, labeled "Shipwrk A 8238" on Figure A3, Appendix A).

During August, Fahys Upper continued to be truncated relative to the river, but the temperature pattern was centered higher than the river (Figure 28). Fahys Mid developed a similar pattern to Fahys Upper. Fahys Mouth was very similar to the mainstem river, while the reference (Shipwreck A) had a wider distribution of temperatures.

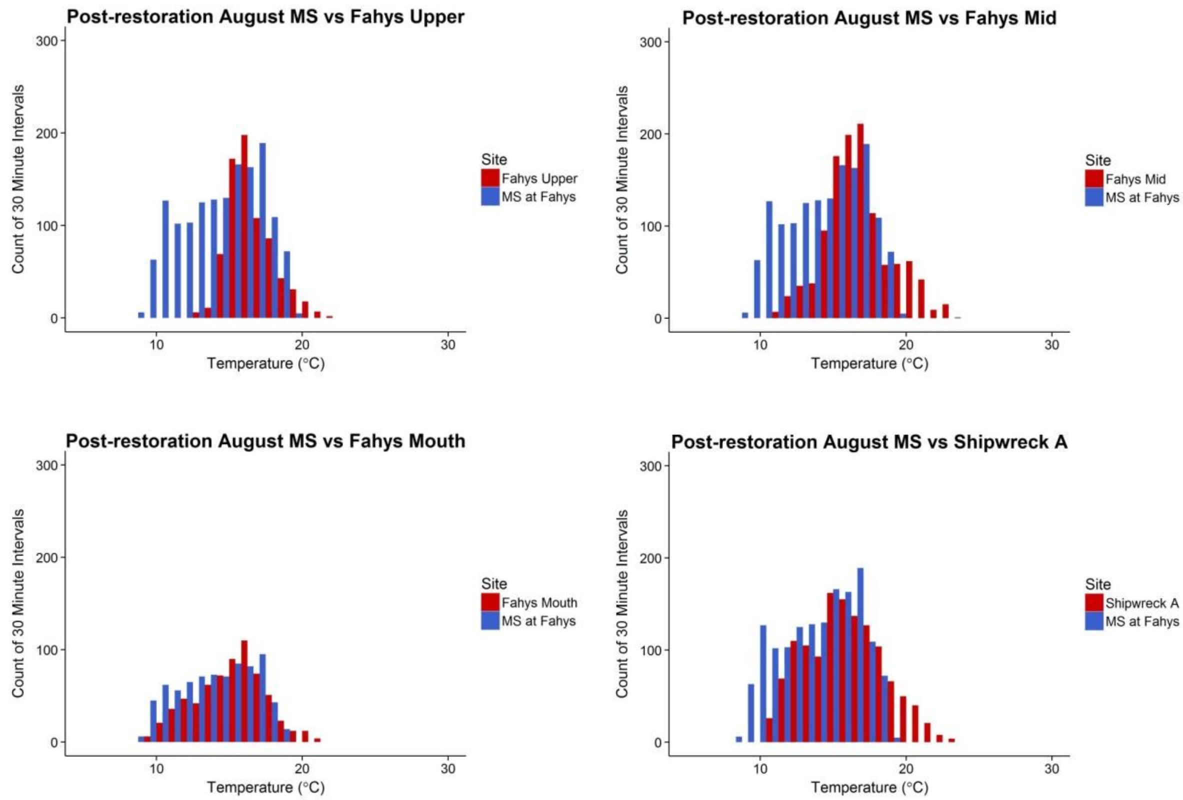

Figure 28. August temperature histograms for the post-restoration period for the mainstem Coquille ("MS"), three Fahys channel reaches, and lower reference reach (Shipwreck A).

Discussion

The removal of the dikes and tide gates at Ni-les'tun, the subsequent channel head cutting process, and the resulting changes in channel bottom elevation in Fahys Lower resulted in full tidal exchange during all river flow conditions. The reduction in >18° C temperatures during the spring and summer seasons in Fahys Lower led to improved fish rearing conditions and was in large part a result of cooler more saline tidal waters entering the channel. The reduction in >18° C temperatures during the summer months led to improved rearing conditions in Fahys Mid. Again, this was largely due to the intrusion of cooler saline water. The Fahys Upper temperatures were most heavily influenced by volume and source. Fahys Creek water provided more influence during the spring season when flows remained higher, whereas during the summer low flow period the water which moved upstream from Fahys Mid during flood tides was more influential. These shifts were somewhat similar to the inter-annual trends observed in the river.

Patterns in the May *versus* August temperature histograms (Figures 27 and 28) suggest the August temperatures in Fahys Mid and Upper reaches were less productive for juvenile salmonid growth, compared to the mainstem river. This is true of the reference sub-basin (Shipwreck A) as well.

Salinity

Methods

We summarized salinity data from restored channels in three categories: <10, 10-20, and >20 PSU to examine overall effects of restoration on channel conditions.

Results

As described in **Channel water salinity and temperature** above, BACI analysis showed significant increases in spring and summer daily maximum and mean salinity at channel mouth and upper channel locations (Figure 18; Appendix A, Figure A34). Salinities in the 10-20 PSU and >20 PSU categories increased during May, June, July and August in Fahys Upper and Fahys Mid after restoration (Figure 29). Although we don't have data for Fahys Lower during the same period, based on our 2012 data we expect that the Fahys Lower post-restoration salinity pattern would be similar to that of the mainstem river, which suggests that salinities increased significantly during May, June, July and August (Figure 29).

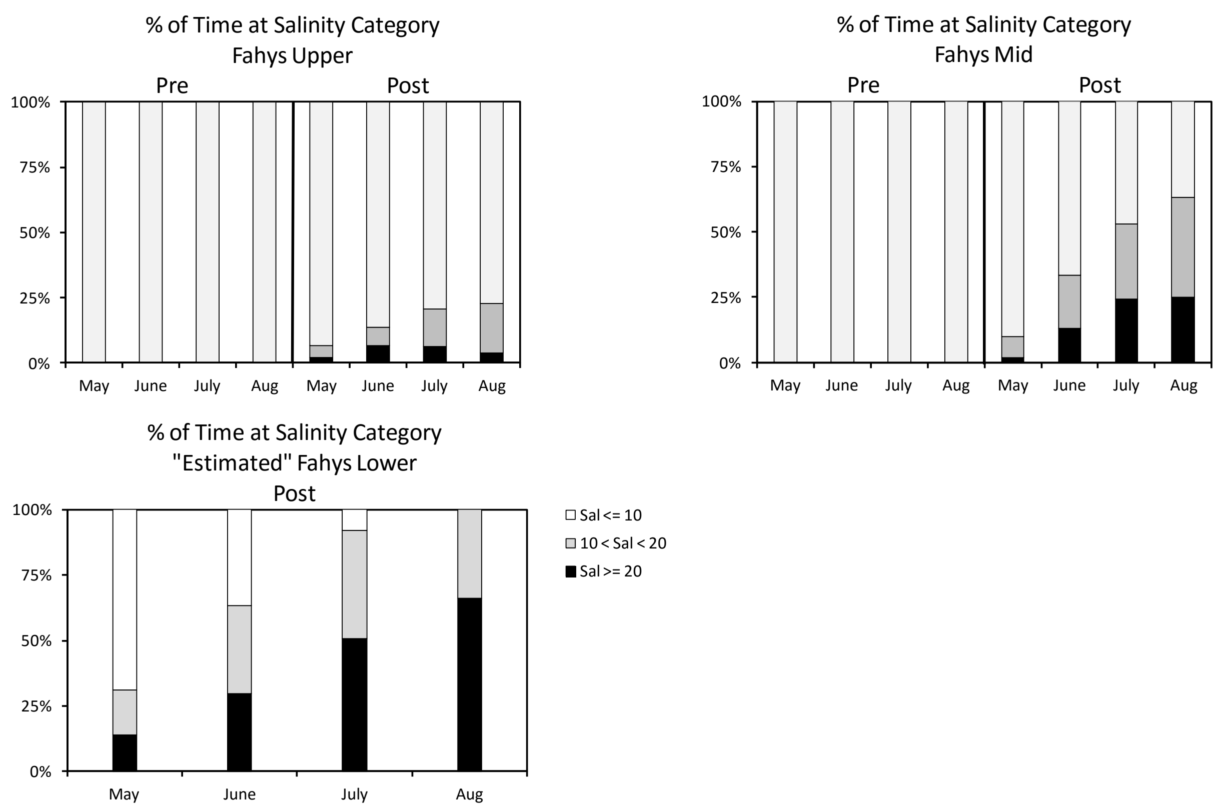

Figure 29. Percent of time in salinity categories (see lower panel) for Fahys Upper, Mid and estimated Fahys Lower (based on Mainstem Coquille River data) during May-August 2011 and 2013.

Time series records from salinity loggers (Figures 30 and 31) allow visualization of seasonal patterns that help describe how the restored site responded to increased tidal exchange. A specific example of this is the comparison of Fahys Mid or Redd Mid to channels 7, 5 Lower, 5 Upper, and NoName Mid. Fahys Mid and Redd Mid (Figure 30) have a constant supply of perennial stream water and show higher amplitude in daily salinity readings throughout the summer season. The data for the sites with less freshwater input showed a reduction in daily amplitude from the wet season to the dry season. This can be seen in NoName and reference/Shipwreck A where there is limited freshwater input (Figure 31) and in Ch 7 where there is no freshwater input (Figure 31). A more extreme example of this effect can be seen in Ch 5. The degree of daily fluctuation is truncated in Ch 5 Lower and is reduced even more in Ch 5 Upper (Figure 31).

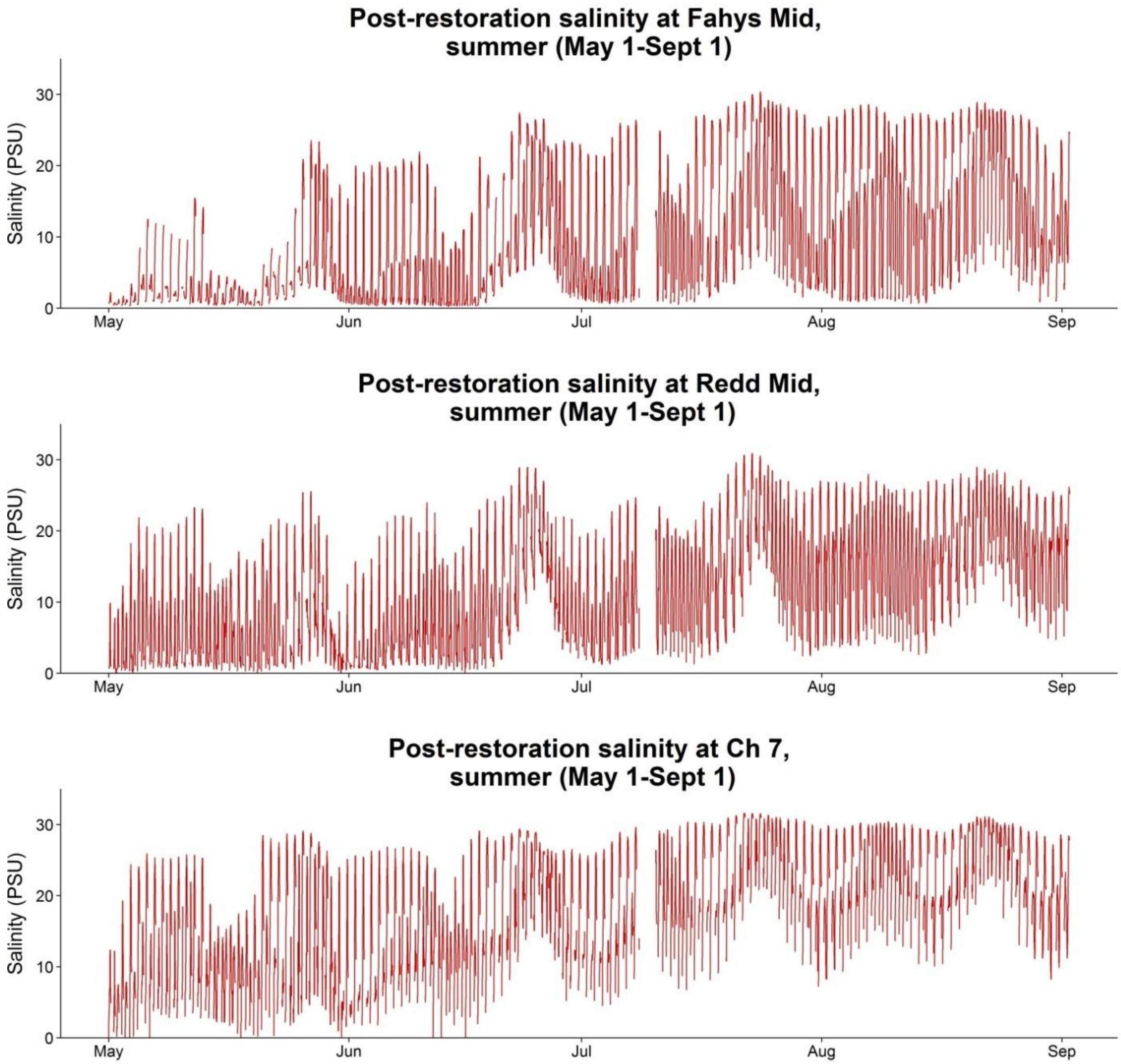

Figure 30. Fifteen minute interval salinity data for Fahys Mid, Redd Mid, and Ch 7 during May 1 through September 1, 2013. Locations are shown in Figure A2, Appendix A.

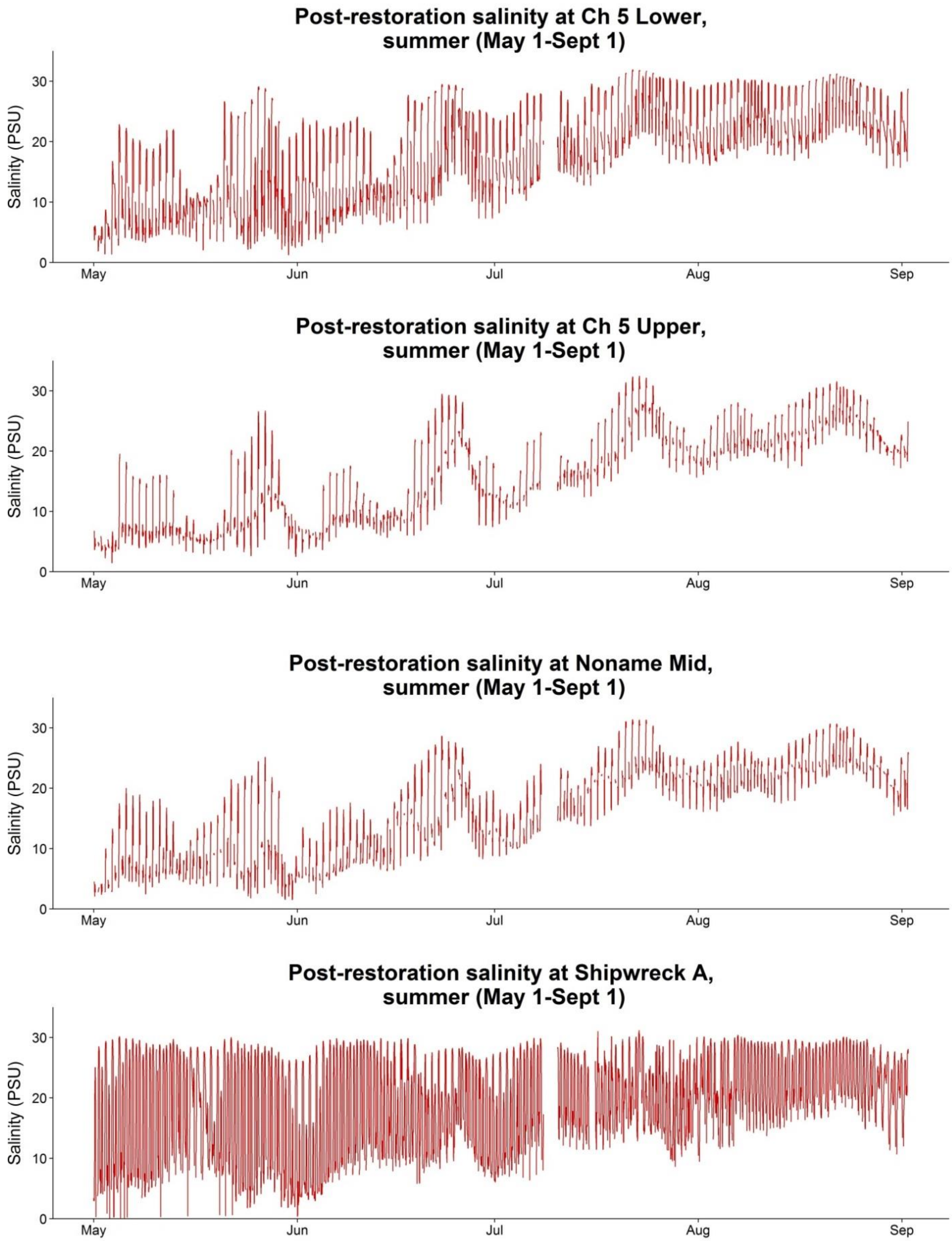

Figure 31. Fifteen minute interval salinity data for Ch 5 Lower, Ch 5 Upper, NoName Mid, and Reference Lower (Shipwreck A), May 1 - September 1, 2013. Locations are shown in Appendix A, Figs. A2 and A3.

Discussion

It is our experience that age-0 chinook that arrive in the Coquille Estuary early in the year (March – May) spend several months avoiding higher salinity habitats as they grow prior to the smolting process and eventual ocean migration (van de Wetering, 2005-2013, unpublished data). We suggest these fish will use lower estuarine habitats where salinities remain below ~20 PSU. Development of salinity tolerance is a process tied to both ontogenic (physiological) and environmental cues (McCormick 1994). Growth and development of each fish will affect the time at which they become tolerant to polyhaline (18-30 PSU) salinities (McCormick 1994). Based on our experience with age-0 chinook in the Coquille estuary, we suggest that the salinities present in all three reaches of Fahys by June of post-restoration were high enough to exclude all fish except those that had developed increased salt tolerance. For fish to use the Fahys Lower and Fahys Mid reaches after the month of June they would likely be required to have developed a tolerance to salinities greater than 20 PSU for more than 25% of the time. This is supported by the general pattern of age-0 chinook distribution described in the following chapters. Our post-restoration salinity data for a series of channels within the restoration marsh further describes the complexity of patterns that can exist and that may influence distribution of juvenile fish that have not become tolerant to polyhaline conditions. See **Salmonid habitat use** below for more discussion on sampling locations, salinity and fish use.

Wood structures and channel morphology

Methods

One hundred thirty root mass logs and 63 stem logs (Figure 32) were placed in wetland channels in the restoration site's three sub-basins. A total of 16 wood placement reaches were constructed, each approximately 100 m in length (Appendix A, Figures A37 and A38). During the post-restoration period, channel morphology was monitored in a subset of these reaches by gathering cross sectional data at regular intervals. Cross sections were monitored in 14 wood reaches and 14 adjacent non-wood reaches in the three restored sub-basins and one reference sub-basin. A total of 2078 m of longitudinal distance was surveyed at intervals of two channel widths, resulting in 311 cross sections. Cross section data were gathered by taking a depth measurement from a level line placed across the top of the channel's right and left banks. Depth measurements were taken at 10% intervals across the full channel width. Measurements were converted to line graphs to visualize variability in channel bank and channel bottom configuration, presence of low tide refugia for fish, and other elements of channel complexity.

Figure 32. Ground view of restored in-stream wood habitats.

Results

Methodological differences prevented direct, quantitative analysis of channel profile complexity between pre-restoration and post-restoration (see **Methods** above). However, visual inspection of graphed cross-sectional profiles suggests that wood placement influenced channel morphology through the erosion of bank and bed materials, leading to the creation of scour pools and fine sediment bars. The degree of channel complexity appeared to relate directly to the amount of tidal forcing at the monitored location: channel profiles were most complex in the downstream channel reaches near the three sub-basin channel mouths, and less noticeable in the upstream tributaries. Differences between the downstream channel reaches and the upstream tributaries could also have been related to the fact that the lower channels were not excavated (they were "pre-existing" channels), whereas the upstream tributary channels were excavated during the restoration process. As described in **Channel types and monitoring approach** above, the non-excavated, pre-existing channels were distinctly different from the excavated channels because they contained large amounts of soft, fine sediment which had accumulated during the pre-restoration period due to the flow barriers present on the site (dikes and tide gates). By contrast, excavated channels had channel bottom substrates that were scraped by the excavator, and therefore lacked fine sediment prior to restoration of tidal exchange.

Channel morphology was more complex in wood reaches compared to adjacent non-wood reaches. Fahys Lower showed the greatest channel complexity, with both bed scour and fill as well as widening of the channel *via* bank failure (Figure 33). Although we didn't sample any scour pools greater than 2.5 m in depth during our cross-sectional profile measurements, we observed two pools in Fahys Lower that had depths greater than 3.0 m at low tide. Redd Lower also showed bank failures and scour, but to a lesser extent than Fahys Lower (Figure 34). The channel cross section profile (Figure 34) for Redd Lower shows a pre-restoration channel width greater than that of Fahys Lower, even though the associated watershed was smaller. NoName Lower also had noticeable responses, but is not presented here, because the footprint left by the pre-restoration ditching makes graphical representation of morphological changes difficult.

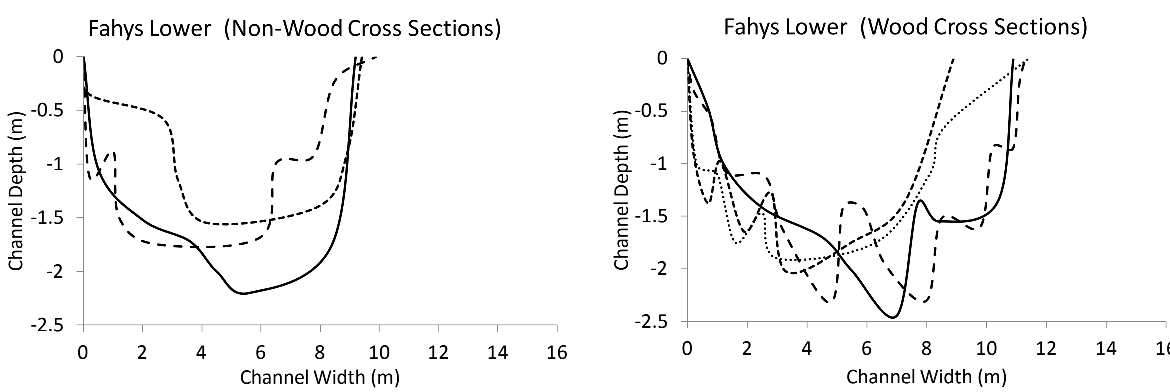

Figure 33. Lower Fahys cross sections from non-wood reaches (left) and wood reaches (right). Dotted, dashed and solid lines indicate different individual cross-sections. All data are post-restoration (2013).

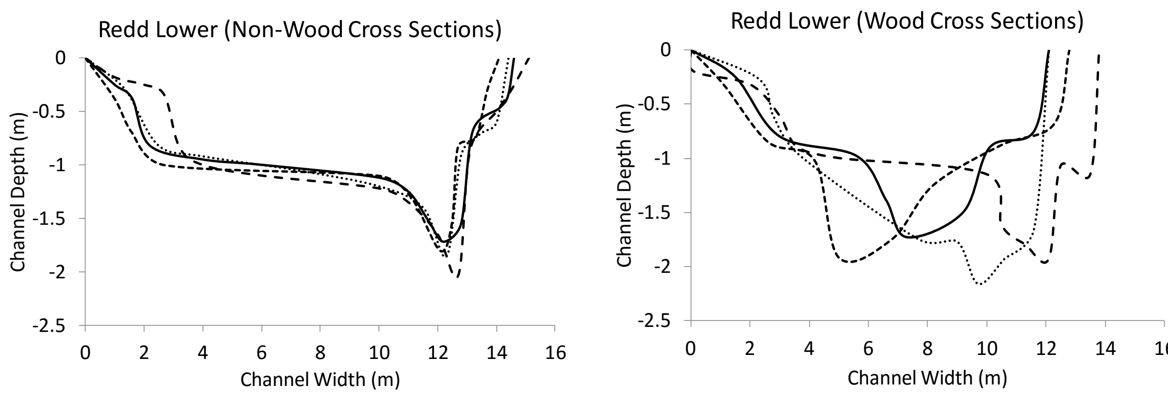

Figure 34. Lower Redd cross sections from the non-wood reaches (left) and wood reaches (right). Dotted, dashed and solid lines indicate different individual cross-sections. All data are post-restoration (2013).

Although we observed fewer changes upstream in the tributaries, channel bottom scour, fill, and bank failure remained greater in wood reaches compared to non-wood reaches (Appendix A, Figure A39). The reference site cross-sections (non-wood) showed only a small amount of profile variability (Appendix A, Figure A40).

Discussion

While channel morphology at the restoration site will most likely remain dynamic and evolving for several years to come, our data suggests that the wood structures placed at Ni-les'tun may have created more complex habitats for a host of species. Gonar *et al.* (1988) described wood distribution in Oregon estuaries during the last century. Historically, wood input likely occurred through a wide range of pathways that are currently limited due to regional land management practices. Because we are unlikely to see natural processes bring wood to our limited tidal marsh habitats in the near future, we suggest that continued wood placement in future restorations will likely benefit a range of species.

As described in Methods above, logistical challenges during the pre-restoration monitoring prevented collection of cross-section data for the non-excavated channels. We therefore lacked baseline data to compare to the 2013 post-restoration data in these channels, and could not use BACI analysis to determine whether channel morphology changes were specifically associated with wood placement. We recommend future monitoring of these same reaches using the same methods, which would allow BACI analysis and determination of the effect of wood placement on channel morphology.

It is worth noting that not all wood structures placed during restoration were still in place one year after restoration of tidal flows. Two factors appear to have resulted in wood structure loss. First, wood density was so high in some locations that forcing of tidal flows around the structures caused bank scour and eventual loss of the anchoring mechanism -- despite the fact that during placement, wood structures were driven 3-5 m (10-15 ft) horizontally into the channel bank. Second, there appeared to be differences in bank soil type among areas of the restoration site. In areas where sand composed a greater portion of the upper two meters of soil, wood structures were lost after top soils released the wood during high water events. The relationship between wood diameter and wood buoyancy during high water events is critical for successful placement. We calculated buoyancy but we were unable to calculate "holding" strength for the various top soils we worked with. Holding strength calculations are needed to prescribe specific log sizes appropriate for different soils, but observations at Ni-les'tun suggest that best results might be gained from using logs with a maximum diameter of 0.6 m and with 4.6 – 6.0 m of stem that can be pushed into channel bank top soils. In addition, based on fish behavior observed in the present study as well as past studies (van de Wetering 2009), we suggest using simple wood structures targeted at forming low tide refugia (scour pools) rather than complex log jams like those used in freshwater stream restorations.

2b. Salmonid habitat capacity

Monitoring Question 2b: Did restoration result in increased salmonid habitat capacity?

Metrics: Benthic macroinvertebrate abundance and community structure within the largest of the three restored basins (Fahys Creek). *(Rationale: benthic macroinvertebrates constitute a large proportion of salmonid prey; prey availability is a controlling factor in salmonid growth rate and ocean survival.)*

Macroinvertebrate abundance and community structure

Background

We conducted sampling in each of the three restored sub-basins, the single reference sub-basin, and the mainstem river where fish sampling occurred. Sampling was located at the center of the fish sampling reaches. Our initial pre-post analysis raised concerns regarding the extremely low abundances during 2010 as well as potential effects of restoration actions around the channels during the 2010 summer of pre-treatment sampling. For this reason we present analyses focused on pre-treatment data from 2007 only. The 2007 sampling did not include the reference sub-basin. Our second analysis focuses on comparisons of 2013 data only.

Methods

In September of each year, five samples were collected from each designated study reach (Appendix A, Figure A10). Only portions of the channel that remained underwater during the low tide during summer low flow conditions were sampled. A 3 x 3 adjustable grid was laid across the tidal channel, and a predetermined random grid cell was selected for sampling. Samples were taken at the center of the selected grid cell. Random selection was affected by avoidance of non-penetrable substrates such as wood and rock. Samples were standardized to target non-vegetated substrates. Benthic samples were taken using a coring device measuring 80 mm in diameter ($0.005 m^2$) and 40 mm deep, with an approximate volume of 254 cm^3. The adjustable grid was then positioned upstream of the first grid and a second sample was taken. This was repeated until five samples were collected in a given reach. Samples were stored on ice for two hours followed by a sieving process to sort macroinvertebrates from substrate and debris. Sieved samples were placed in 95% isopropyl alcohol for preservation and stored at room temperature. All invertebrates were classified to the lowest taxonomic level possible. Classification was carried out by a commercial laboratory located in Moscow, Idaho (http://www.invertebrateecology.com/).

Abundance by taxon, taxon richness (number of taxa), Shannon-Wiener Diversity Index and a Community Structure Index were calculated for all samples except the Coquille River locations. The Shannon-Wiener Diversity Index is derived from the number of taxa present and the relative abundance of those taxa (Magurran 2004). This index typically shows whether one or a few species are dominating a site – a common result when habitats are disturbed or colonization has recently occurred. The Community

Structure Index measures consistencies in macroinvertebrate communities between sites (Magurran 2004). We used a general linear mixed model to evaluate the 2013 data set (Lepš and Šmilaur 2003). For those metrics that showed significant differences among locations, pairwise means comparisons were used to determine significant differences between specific locations.

Results

2007-2013 Comparisons

Mean richness decreased in Fahys Lower and increased in Fahys Mid after restoration (Figure 35). Our results suggest richness in Fahys Upper also increased after restoration but to a lesser extent (Figure 35). The 2007 Fahys Lower taxa included five crustaceans, three polychaetes, an anemone, a round worm, a bivalve, and a snail. Those taxa from 2013 included four crustaceans (three of which were new taxa), an unknown polychaete, a roundworm, a snail, and a sponge. The 2007 Fahys Mid taxa included one crustacean, two polychaetes, an oligochaete, a roundworm, and a chironomid. Those taxa from 2013 included five new crustaceans, a polychaete, an oligochaete, a round worm, a snail, and a sponge. The 2007 Fahys Upper taxa included three crustaceans, two polychaetes, a roundworm, a chironomid, a bivalve and a spider. Those taxa from 2013 included four crustaceans (one of which was present in 2007), a polychaete, a snail and a chironomid.

Mean Shannon-Wiener Diversity Index decreased in the three Fahys locations after the restoration (Figure 35). This was mainly due to the shift away from a more evenly distributed group of species to a community dominated by *Corophium* (98%, 86% and 89% for Fahys Lower, Mid and Upper, respectively).

The mean community structure index decreased in the three Fahys locations after restoration (Figure 35). This was mainly due to a decrease in the index of diversity but was also affected by shifts in taxa after restoration. Abundance increased at all three Fahys locations (Figure 35).

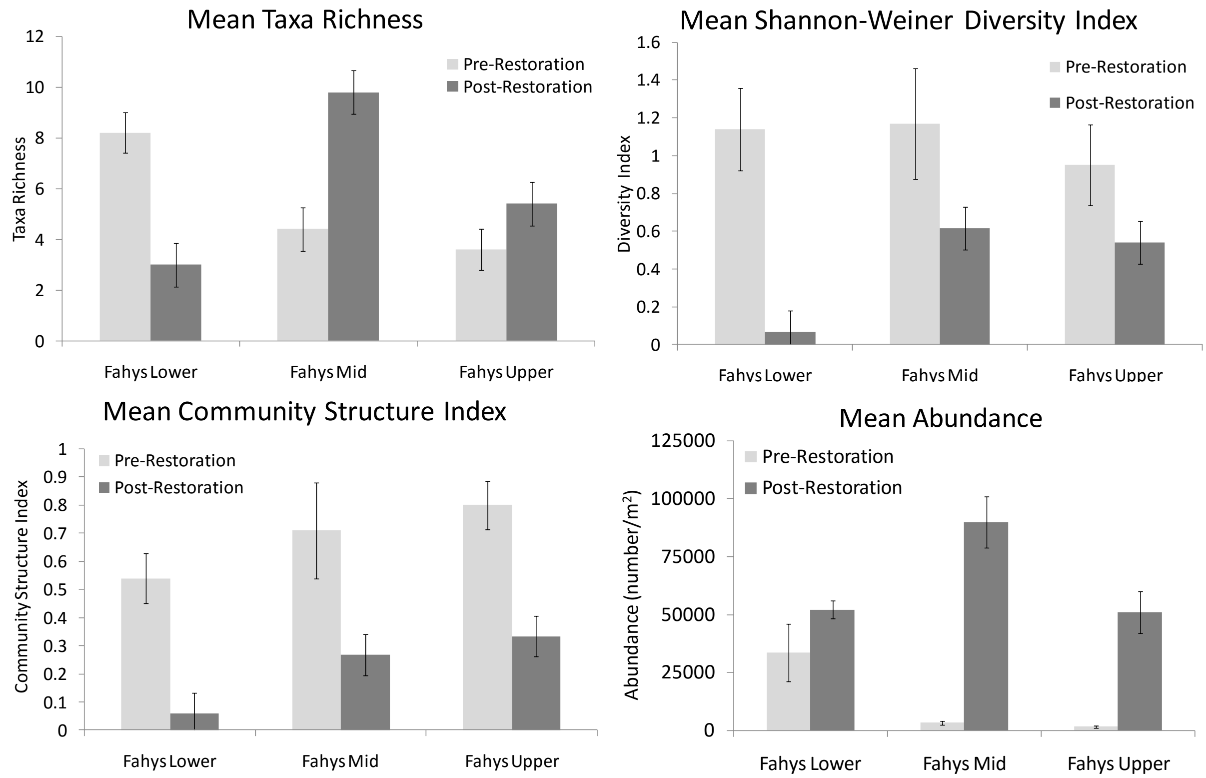

Figure 35. Pre-restoration (2007) *versus* post-restoration (2013) comparisons for macroinvertebrate community metrics at Fahys sample locations. Error bars represent one standard error of the mean (1 SEM).

We used a general linear mixed model to evaluate the 2013 multiple location data set (Tables 2 and 3). Richness, Shannon-Wiener Diversity Index, and the Community Structure Index showed a significant difference among the means ($p< 0.05$), while there wasn't a significant difference among the means for abundance. Thus, multiple comparisons were made for richness, Shannon-Wiener Diversity Index, and the Community Structure Index, but not for abundance -- although we provide abundance means and standard errors.

Table 2. Test of the global null hypothesis that the means for benthic macroinvertebrate richness, Shannon-Wiener Diversity Index, and abundance are all equal, *versus* the hypothesis that at least one mean differs from the others.

Response	F	df (numerator)	df (denominator)	Pvalue	Result
Richness	4.59	10	38	0.0003	Significant
Shannon	10.54	10	38	< 0.0001	Significant
Abundance	1.74	10	38	0.1071	Not Significant
Comm. Structure	8.56	10	38	<0.0001	Significant

Pairwise comparisons of taxa richness across locations (Table 3 and Figure 36) showed general similarity among most locations, but Fahys Mid had higher richness compared to all other locations. Within the reference site, Reference Lower was significantly different from Reference Upper (Table 3, Figure 36).

Table 3. Multiple comparisons for the mean and 1 SEM for benthic macroinvertebrate richness among locations during 2013. In the "Gp" column, rows containing the same letter are not significantly different ($p < 0.05$).

Richness

Location	Estimate	SEM	Gp
Fahys Mid	9.80	0.86	A
Reference Lower	6.80	0.86	B
Reference Mid	5.67	1.11	BC
NoName Lower	5.50	0.96	BC
NoName Upper	5.40	0.86	BC
Fahys Upper	5.40	0.86	BC
NoName Mid	4.80	0.86	BC
Redd Lower	3.80	0.86	C
Redd Mid	3.80	0.86	C
Reference Upper	3.50	1.36	C
Fahys Lower	3.00	0.86	C

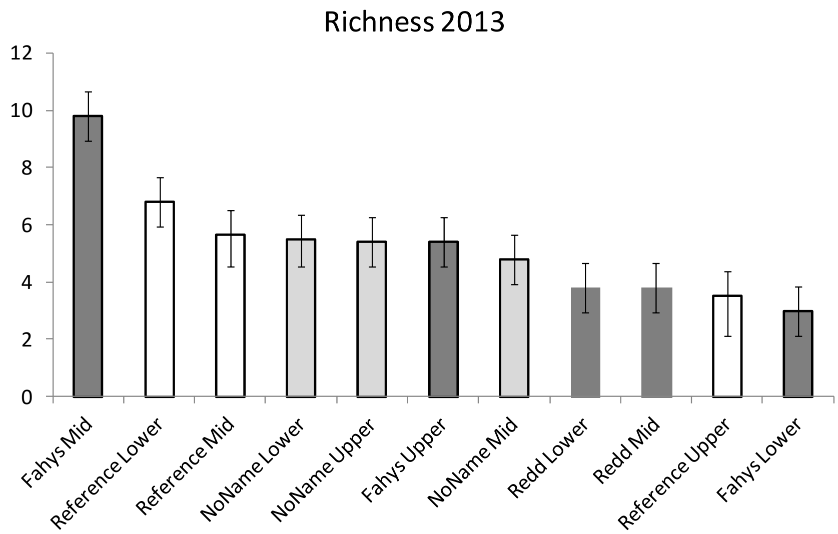

Figure 36. Mean taxa richness for benthic macroinvertebrates by location, September 2013. Error bars represent 1 standard error of the mean (1 SEM).

Pairwise comparisons of Shannon-Wiener Diversity Index across locations showed three general groups of similarity (Table 4 and Figure 37). NoName Upper and Reference Lower had significantly higher diversity than most other locations, and Redd Lower and Fahys Lower were significantly lower than most other locations. Within the reference site, Reference Lower was significantly different from Reference Mid and Upper.

Table 4. Multiple comparisons for the mean and 1 SEM for Shannon-Wiener Diversity Index for benthic macroinvertebrates among locations during 2013. Letters indicate significant differences ($p < 0.05$).

Shannon

Location	Estimate	SEM	Gp
NoName Upper	1.24	0.112	A
Reference Lower	1.09	0.112	AB
NoName Mid	0.84	0.112	BC
Reference Mid	0.71	0.145	CD
NoName Lower	0.65	0.126	CD
Fahys Mid	0.62	0.112	CD
Fahys Upper	0.54	0.112	CD
Redd Mid	0.44	0.112	D
Reference Upper	0.28	0.178	DE
Redd Lower	0.09	0.112	E
Fahys Lower	0.07	0.112	E

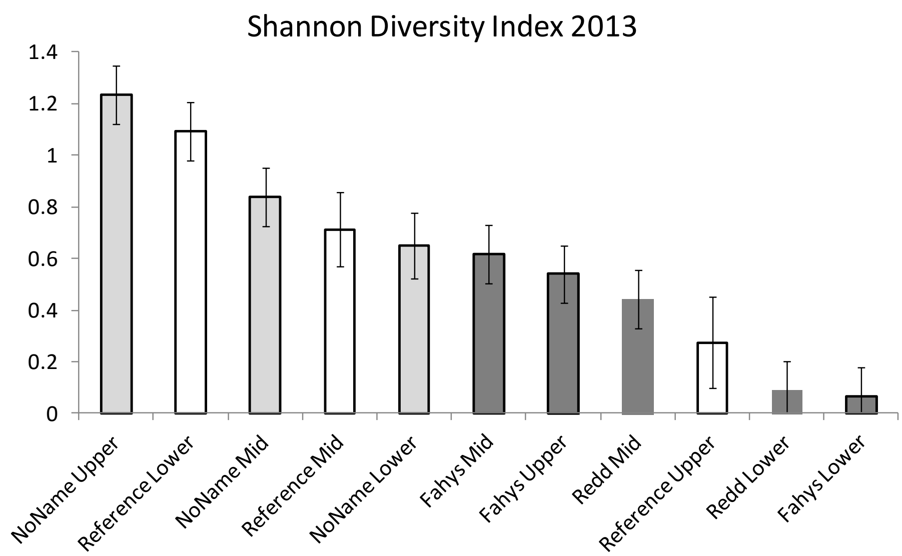

Figure 37. Mean Shannon-Wiener Diversity Index for benthic macroinvertebrate taxa by location, September 2013. Error bars represent one standard error of the mean (1 SEM).

Pairwise comparisons of Community Structure Index across locations showed broad overlap of groups, with considerable similarity among locations (Table 5 and Figure 38). NoName Upper and Reference Lower had the two highest values. Within the reference site, Reference Lower was significantly different from Reference Upper.

Table 5. Multiple comparisons for the mean and 1 SEM for community structure Index among locations during 2013. Letters indicate significant differences ($p < 0.05$).

Community Structure			
Location	Estimate	SEM	Gp
NoName Upper	0.752	0.073	A
Reference Lower	0.608	0.073	AB
NoName Mid	0.564	0.073	ABC
Reference Mid	0.39	0.094	BCD
NoName Lower	0.3775	0.081	CD
Redd Mid	0.374	0.073	CD
Fahys Upper	0.334	0.073	D
Fahys Mid	0.268	0.073	DE
Reference Upper	0.21	0.115	DE
Redd Lower	0.062	0.073	E
Fahys Lower	0.06	0.073	E

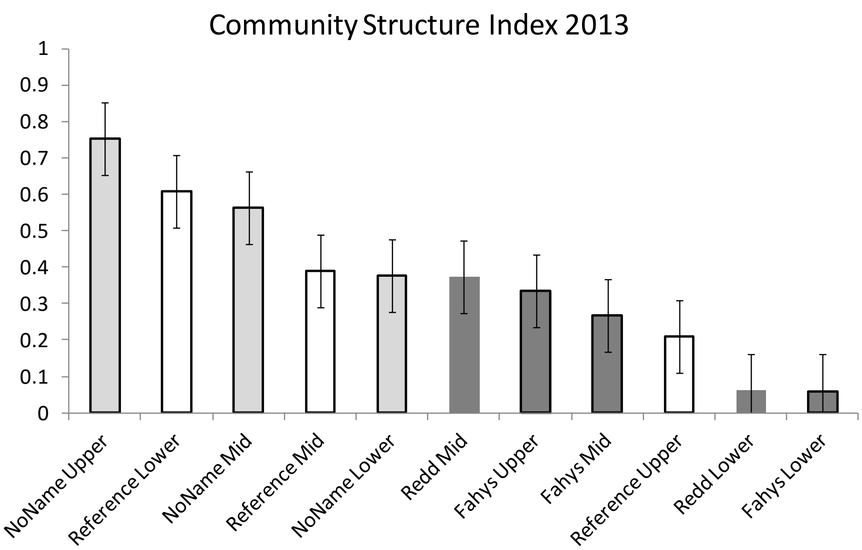

Figure 38. Mean Community Structure Index for benthic macroinvertebrates by location, September 2013. Error bars represent one standard error of the mean (1 SEM).

Although the mixed model did not show significant differences in abundance among sample locations, Reference Lower appears to have higher abundance compared to the other locations (Figure 39).

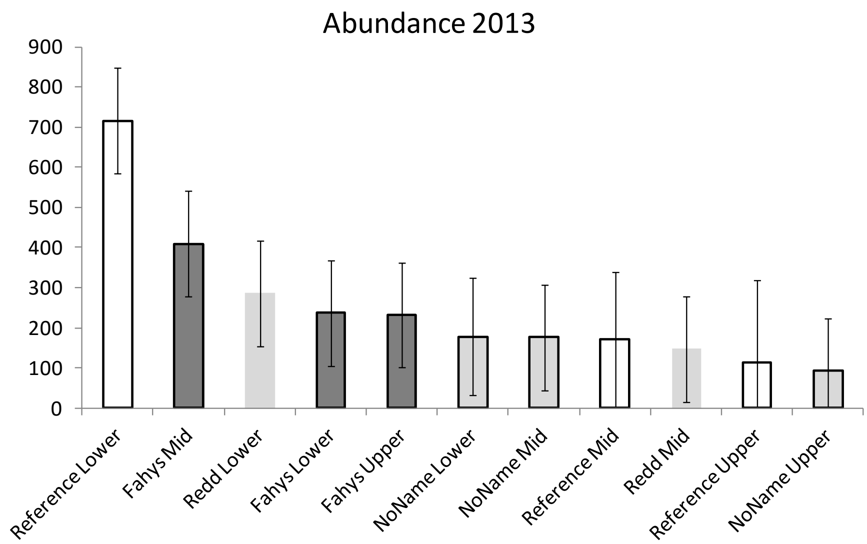

Figure 39. Mean benthic macroinvertebrate abundance per sample by location, sampled September 2013. Error bars represent one standard error of the mean (1 SEM).

Using the middle reach of each study sub-basin and the associated Coquille River bank sample locations, we compared the benthic communities available to juvenile fish in each location (Figure 40). *Corophium* was the most common and abundant taxon across all sample locations with the exception of Reference Mid (Figure 40; Appendix A, Figure A41). *Corophium* were more abundant in each river location than the paired marsh sub-basin locations, with the exception of Fahys Mid. Although we didn't include the river locations in our pairwise analysis, our results suggest all restored marsh sub-basin locations were more diverse in taxonomic composition than their adjacent river locations (Figure 40).

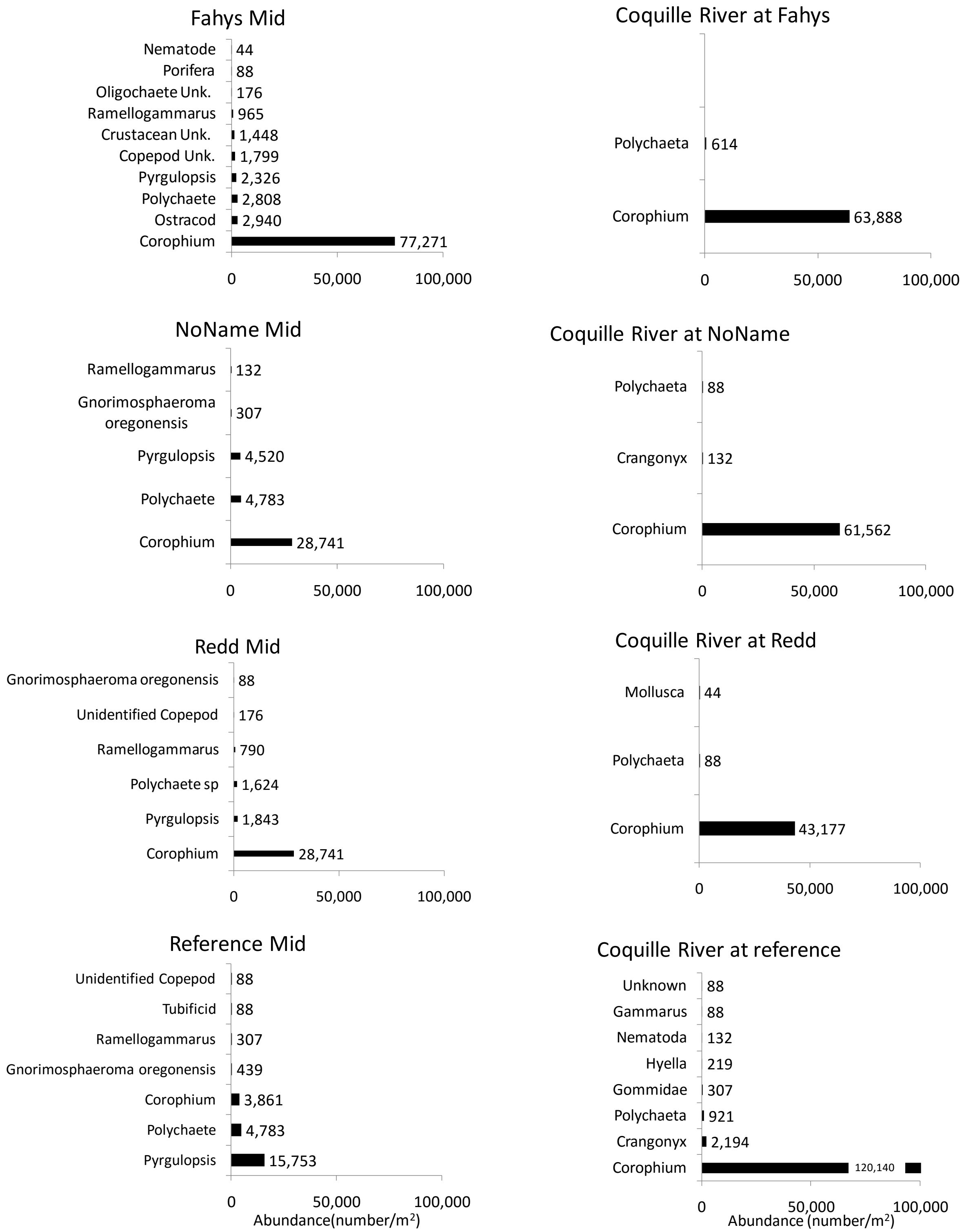

Figure 40. September 2013 benthic macroinvertebrate abundance (number/m^2)from the mid-reach sample location of each restoration sub-basin (left) and the corresponding Coquille River location near the mouth of each sub-basin (right).

Discussion

Fahys restored sub-basin (pre and post-restoration comparisons)

The benthic community of Fahys clearly changed after restoration. There was a decrease in diversity and community structure, likely due to the simplification of benthic substrates early in the restoration process and the ability of taxa such as *Corophium* to rapidly colonize. Taxa such as *Corophium* are capable of multiple generations in a single year, which can create a competitive advantage during colonization or season shifts in habitat (Desmond *et al*. 2002) and thus reduce measures of diversity and community structure by reducing evenness. The increase in Fahys Mid and Fahys Upper richness (number of taxa) was likely due to the re-introduction of tidal flows resulting in seasonally higher salinities as well as greater daily fluctuations in salinity. Increases in annual peak salinities have been shown to create habitat that is more likely to allow for a broader range of species but at less than optimal conditions (Odum 1988, Rename and Schlieper 1971, Howe *et al.* 2014). Our observations show the number of taxa increased (regardless of the shift in diversity) in the Fahys Mid and Fahys Upper reaches. The decrease in richness at Fahys Lower was likely due to extensive down cutting of the channel, which resulted in a simplification of the channel bottom substrate during early restoration. These early-restoration erosive conditions could make it more difficult for some species to colonize the substrates, and thus may influence richness more than increased annual salinities. The increase in total abundance is likely a result of the new resources made available through the physical process of restoration (digging channels, etc.), the new resources associated with tidal exchange, and the disturbance of marsh plant communities at the restoration site and their associated decomposition (Gray *et al*. 2002).

Differences among locations at the Ni-les'tun restoration site

Although there were some measurable differences among locations in richness, diversity, and community structure, those differences did not appear to relate to restoration *versus* reference site differences, but rather to channel characteristics. Locations that experienced notable scour as a result of head cutting (Fahys Lower and Redd Lower) had lower diversity and lower Community Structure Index. As channel flow paths equilibrate over time (see **Channel Morphology** above), these differences are expected to decrease. Nanami *et al*. (2005) and Degraer (2008) have shown that grain size and sediment type affect estuarine benthic macroinvertebrate community structure. Although we were unable to measure sediment type and grain size, anecdotal observations suggest there were measurable substrate differences at several sites – Fahys Mid and Upper contained more coarse grain sands; Noname Mid contained more clays while Noname Upper contained more fine organics; Redd contained small gravels and fine organics while the reference sub-basin contained a mixture of sands and silty muds. Howe *et al*. (2014) concluded that benthic/epibenthic communities responded more to small sediment-related, site-specific scales (<10 m) than to broader habitat differences, such as channel size and salinity and temperature differences. This idea is also supported by the work of Degraer *et al.* (2008) and Namani (2005). Based on our observations of channel substrates, we would anticipate stronger differences when comparing sites such as NoName Mid to Reference Lower.

Our method of sampling a centered location within a given reach could have created some biases if the sampling area (10m) was not representative of the channel system or reach as a whole. However, if this were the case one would expect to see more significant differences between locations, rather than the relative few significant differences found in our results.

Our results for the study marsh locations support the conclusions of Howe *et al*. (2014) and others: that community structure can remain similar across a range of tidal marsh channel habitat types. Considering our results for the Coquille River, we suggest that channel habitat (sediment type, grain size and rate of disturbance) is driving richness, diversity and community structure. Although we did not measure grain size for the Coquille River samples, the three locations where high winter river flows create coarse substrates had limited taxa compared to the marsh sample locations. The single Coquille River location where sediments occur in more complex layers and patches (Coquille River at Reference) had almost three times the number of taxa when compared to the other river locations. Our results suggest the simplified river bank environment provides adequate habitat for a single taxon (*Corophium*) to be successful, whereas the restored and reference marshes provide more complex habitats that support greater diversity.

Corophium, polychaetes as well as other more common and abundant taxa (*Pyrgulopsis, Gnorimosphaeroma oregonensis*, and *Ramellogammarus*; Figure 41) were present in all restoration marsh locations and were also present in the reference marsh. Others working in Eastern Pacific estuaries have shown the same suite of species we observed occurring across a range of salinity and temperature regimes (Desmond 2002; Gray 2002; Cooksey 2006; and Peterson 2010).

Corophium (Figure 41) were the key taxa that drove total abundance and Shannons Diversity Index estimates in seven of the eight restoration locations, as well as the Reference Lower sub-basin location. *Corophium* were also the dominant taxa in the river samples. Several studies have identified *Corophium* as a primary prey of age-0 chinook (McCabe Jr. *et al*. 1983; Shreffler *et al*. 1992; Miller and Simenstad 1997; Lott 2004; Gray 2005, Eaton 2010). Cooksey (2006) identified isopods (*Corophium*) and polychaetes as common prey of Pacific staghorn sculpin. Because age-0 chinook and Pacific staghorn sculpin were the two most common fishes observed during our study, the common occurrence of their preferred prey species is likely creating increases in survival and production of these fish.

Our results, specific to rate of recovery since restoration, are somewhat in contrast to those reported for the Salmon River (Gray 2005) and for the San Francisco Bay (Howe *et al*. 2014), although direct comparisons are difficult because of different study approaches and observation periods – Gray's study compared restoration of 10 – 30 years while that of Howe *et al.* compared 10 - 50 year restoration periods. As the Ni-les'tun restored channels equilibrate with tidal forces during the next decade, there will likely be significant adjustments in species composition and evenness. However, the lack of significant differences between reference and restored communities at this early stage of restoration suggests benefits to fish *via* prey resources may come relatively quickly after restoration of these types of tidal marsh channels.

Corophium

Polychaete

Pyrgulopsis

Gnorimosphaeroma Oregonensis

Ramellogammarus

Figure 41. Common benthic macroinvertebrate taxa found at the Ni-les'tun restoration site and Bandon Marsh Unit reference site

Conclusions

Our results suggest the benthic macroinvertebrate communities in the restored marshes shifted within two years post-restoration to the extent that they are difficult to differentiate from the reference marsh communities. Our results show more diverse benthic communities occurring in the restored marsh channels when compared to the mainstem river channel. We conclude that even at this very early point in the Ni-les'tun Marsh restoration process, the benthic macroinvertebrate community associated with the restored in-stream habitat resulted in dramatic increases in fish habitat capacity, and therefore likely increases in fish survival and production for the lower Coquille River system.

2c. Salmonid habitat use

Monitoring Question 2b: Did restoration result in increased salmonid habitat use?

Metrics: Residency patterns (occupancy rates, catch per unit effort [CPUE], and CPUE restoration to reference ratios), tidal migration patterns, and salmonid use of large wood habitats.

Overview

Our goal of evaluating habitat use involved two approaches. The first was to define whether and how various species age-class groups, such as age-0 Chinook, use wetland habitats as multiday residents across seasons. The second was to define patterns of daily feeding migrations into and out of the wetlands. Our approach to define residency did not involve tagging individuals to define single fish behaviors that could be used to infer population behavior patterns. Instead, we used a census of the seasonally shifting population to suggest population-based residency patterns. Our sampling was focused on the lower low tide of a daily tide cycle. Our rationale was that fish found during the lower low tide have chosen to remain in the wetland during the period when they are most susceptible to mortality, and therefore they can be considered multiday residents. Our rationale for not using mark-recapture methods was the small size of the salmonids we sampled during the spring season as well as the mortality related to repeat sampling of age-0 fish in wetland habitats. Our approach to define patterns of daily migration was to describe migration at a series of tidal channel mouth locations during the period of time when the most abundant juvenile salmonid (age-0 Chinook) was available within the broader estuary. We then used these two approaches to compare pre-restoration to post-restoration observations.

Methods

Wetland residency patterns

Sampling was carried out during 2005, 2010 and 2013. Sampling focused on the months of April - September when juvenile salmonids are most abundant in the Coquille Estuary (van de Wetering, unpublished data). We stratified Fahys and NoName sub-basins into three channel reaches defined by salinity, temperature and stream order (Appendix A, Figure A9). Three sample locations were then equally spaced within the full length of each sample reach. Redd was analyzed as a single stratum. Reference samples were collected at three locations in the Coquille River: downstream of the Reference (Shipwreck) sub-basin, upstream of the Fahys sub-basin, and upstream of Redd sub-basin (Appendix A, Figure A9). Three adjacent samples were collected at each river reference location. Sampling was standardized by taking all samples during the morning low slack tide, using the same net for each location, and sampling the same surface area each time a location was visited. Mainstem sample locations consisted of a seine set that measured 10 m in length and varied slightly in width, averaging five meters, with an average max depth of 1.5 m and minimum depth of 0 m. All wetland channel seine sets were 20 m in length and extended across the full width of the channel. Seine sets varied in depth from 0.1 – 0.3 m. The same sample locations were used throughout the study for comparable results.

Presence/absence across all sub-basin sample locations was used to estimate sub-basin occupancy by month. Catch per unit effort (CPUE) is an indirect measure of the abundance of a target species based on repeat sampling using standardized methods (Southerland 2000). Changes in the CPUE are inferred to signify changes to the target species' true abundance. CPUE has been used for fisheries monitoring for several years (Maundera 2006). Our seine capture data (i.e. individual sample fish counts) are hereinafter referred to as CPUE. Mean CPUE was calculated for each species for each sub-basin during its peak month of annual abundance. In addition, we calculated the ratio of the peak month mean CPUE values to the reference sample mean CPUEs for the same month. This is not a standard method, but was used to account for annual and monthly variation in the "supply" of fish that affected restored wetland use. Lastly, CPUE values for the months of May-August were used to model the effects of the restoration on various species, allowing a pre- and post-restoration statistical test of significance (BACI) by species.

Because we were enumerating animals that distribute over time and space during a broader migration, zero counts were expected. High numbers of zero count samples result in overdispersion (too many zeros in the data set). Given the tendency for overdispersion with the seasonal CPUE data, a negative-binomial hurdle model was used for analysis. The hurdle model includes a binomial model to account for the zeroes and a count model for non-zero count data (Mullahy 1986). This allows the analyst to predict the probability of an increase or decrease in sample number (fish caught) by location, as well as the probability of a non-zero value. Sample location CPUEs were aggregated to the reach level within the month and year. Reach-level CPUEs were used to obtain spatial replication for a sub-basin so that estimates of sub-basin level effects could be considered. Month effects were also evaluated in the BACI analysis model to account for within-year variation of CPUEs. Model selection was conducted with the covariates of time (pre or post), month (May-Aug), sub-basin name, and treatment (restored or reference). Data from the months of March and April were not included due to an absence of samples at some locations.

Tidal fish migration patterns (video monitoring)

Pre- and post-restoration comparisons

Because age-0 Chinook were present in our 2005 and 2010 seine samples at much greater numbers than any other salmonid species-age class in the mainstem Coquille River, they were designated as the target species for migration estimates. Pre-restoration mainstem age-0 Chinook peaked in size and number during June 2005 and 2010. Based on this pattern, June was chosen as the target sampling month with the intent to sample tidal wetland migration patterns during a period when juvenile salmonids are most likely to be in mainstem river habitats and in turn accessing tidal wetland habitats on a daily basis. Sampling occurred during the same part of the monthly tidal cycle so that fish were experiencing the same tide heights and timing of the daily cycle during the pre and post-restoration periods.

Sampling occurred at the mouths of each restoration sub-basin (Fahys, NoName, and Redd) and the mouth of Shipwreck sub-basin (reference sub-basin). Sampling during the pre-restoration period at tide gate locations involved sampling the inner end of the tide gate pipe itself. Fish migration sampling was completed using a fence of cameras (sampling transect) that bisected the inner pool. Fyke nets were used to narrow the pool width to that of the tide gate pipe. Reference sub-basin sampling occurred at the most downstream point in the channel, where the channel spills into the mainstem river (Appendix A, Figure A9).

Sampling transects consisted of a set of four cameras mounted vertically on a series of sampling poles. Camera poles were stationed in a line perpendicular to water flow. Stations were located every 0.3 m across all sample channels except Fahys, where stations were located every 0.4 m. Cameras were set at 0.2, 0.5, 0.8, and 1.1 m vertically above the channel bottom. Observations began during the lower low morning slack tide and were completed in the early evening at higher low slack tide.

Count data were developed through review of the underwater video. Visibility ranged from 0.61 to 1.8 m. All fish were enumerated within a 0.30 m field of view. Counts were recorded on a minute by minute basis, by species, and lumped into 30 minute bins for analysis. Ten percent of the video reviewed by an individual reviewer was “re-read” and validated by two additional reviewers. Counts were extrapolated by the amount of habitat not visible to the camera’s conical field of view and beyond the designated view field distance. Sub-basin camera count expansions averaged 1.5x the observed value with a range of 1.4x to 2.4x. Mainstem river camera count expansions varied ranging from 3-4x for the above Fahys location to 8-10x for the between Redd and NoName location, to 14x near the reference wetland. All salmonids observed were lumped into two categories based on size, resulting in a classification of age-0 or age-1 (smolts). Sculpin, shiner perch, and three spine stickleback were also enumerated and lumped into single classes by species rather than age-species classes. Lastly, other species such as northern anchovy *Engraulis mordax*, surf smelt *Hypomesus pretiosus*, American shad *Alosa sapidissima*, pile perch *Rhacochilus vacca*, surf perch *Embiotocidae,* bay pipefish *Syngnathus leptorhynchus*, and dungeness crab *Metacarcinus* *magister*, were observed and recorded but detailed results are not presented in this document. Expanded camera counts were summed (all "counts in" minus all "counts outs") across all cameras for a given sampling location for each 30 min period. Cumulative counts were calculated across the full tidal cycle. The peak of the cumulative estimate was used for analysis. Peak migration estimates were compared between years by sub-basin. In addition, comparisons between sub-basins and the Coquille River were made for the post-restoration period.

We used a linear model to determine the significance of the effects of year (pre-restoration *versus* post-restoration), location (restoration sub-basin *versus* reference sub-basin), and their interaction on fish abundance. Total counts for fish migrating into and out of a sub-basin were expanded, then summed to create a migration activity count. Analysis was completed for age-0 salmonids, shiner perch, three spine stickleback and sculpin.

Post-restoration targeting of age-1 coho tidal migration

During our standard May seine sampling we observed an unusually high CPUE for age-1 coho at our mainstem river sample locations in the lower portion of the river. Because of this, we carried out a single tidal migration sampling event in the mouth of Fahys and the reference sub-basin. Sampling was carried out as described in **Pre- and post-restoration comparisons** above.

Use of large wood by salmonids

The same seine sampling described above was also used to examine large wood use within the three sub-basins. Large wood structures were placed across 100 m long reaches separated by 100 m of non-wood habitat in each sub-basin (see **Wood structures and channel morphology** above). Three wood reaches were placed in Fahys, three in NoName, and two in Redd. Sample locations were stratified between wood and non-wood habitats to allow evaluation of wood as a factor in wetland residency.

Tidal migration observations (camera counts) were also used to estimate the extent of wood habitat use. Sampling transects were placed at the downstream and upstream ends of the lowest wood reach in each sub-basin. Migration patterns into and out of the wood reaches were used to evaluate the influence of wood on use of low tide refugia (wood structure scour pools).

Results

Wetland residency patterns

Age zero coho, (*Onchorhyncus kisutch),* shiner perch (*Cymatogaster aggregate*), and starry flounder (*Platichthys stellatus*) were present during different years, but their distribution was limited to one or two sample locations and their occupancy was limited to only a month or two. This resulted in variance estimates that were much greater than the means themselves, which in turn made occupancy and CPUE analysis unproductive. Cutthroat trout, *(Oncorhyncus clarki*) and steelhead trout (*Oncorhyncus mykiss*) occupancy rates were even more limited, and therefore not considered for analysis. We examined the seine sample data using occupancy rates and peak distribution densities for three dominant species (Chinook, (*Oncorhyncus tshawytscha),* Pacific staghorn sculpin, (*Leptocottus armatus*), and three spine stickleback, (*Gasterosteus aculeatus*) within each sub-basin.

In Fahys sub-basin, age-0 Chinook occupancy increased after restoration during three of the five months sampled, and decreased during two of the sampled months (Figure 42). In Redd and NoName sub-basins, age-0 Chinook occupancy increased across all months measured (Figure 42). During March and May 2013, occupancy in Redd and NoName was much greater than that in Fahys, reflecting a high rate of use by early migrant fry. During July, occupancy in Fahys increased after decreasing through the spring, reflecting use by larger age-0 Chinook that had begun using more open water habitats as they neared their ocean entry period.

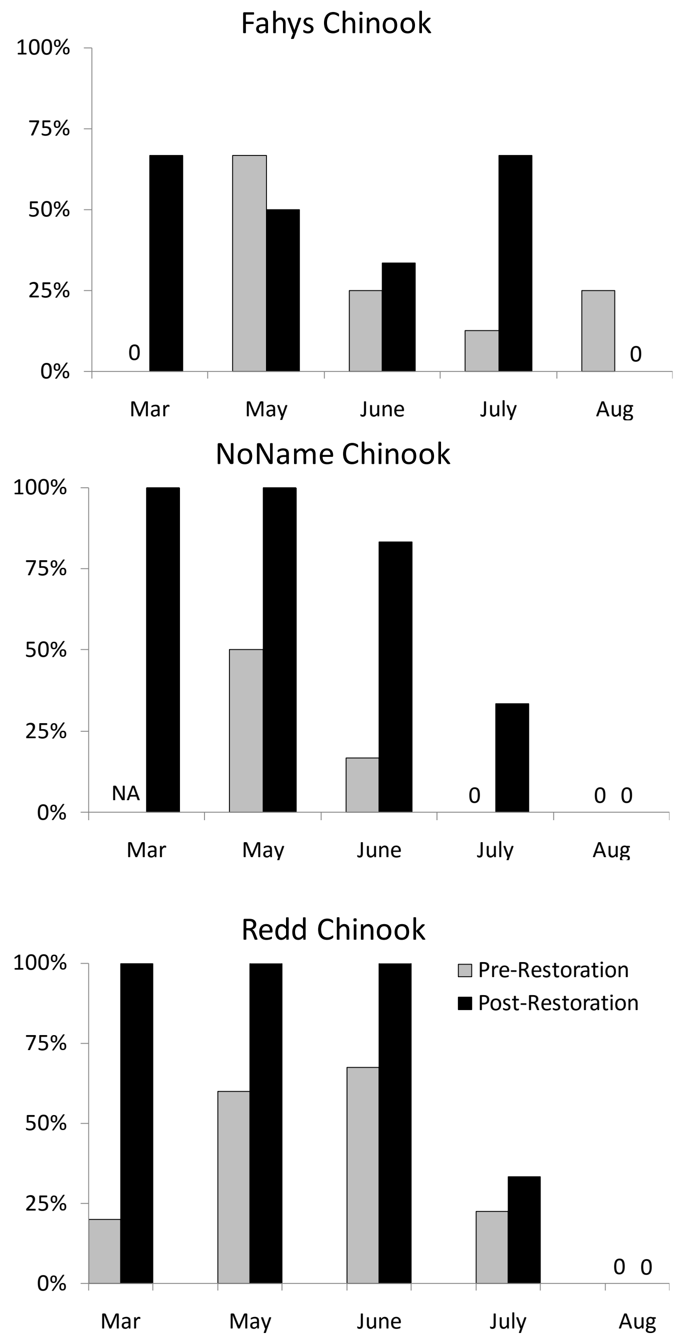

Figure 42. Age-0 Chinook monthly occupancy rates for Fahys, NoName and Redd sub-basins during pre-restoration (2005) and post-restoration (2013).

Staghorn sculpin occupancy rate in Fahys and NoName sub-basins increased after restoration during all months, while occupancy in Redd increased during all but one month (Figure 43). The highest post-restoration occupancy rates were observed in Redd early in March and May, while occupancy in Fahys was consistently high during all months. Our 2013 length data (not detailed in this report) show that fish greater than 60 mm in length were present almost exclusively during March and April, whereas smaller fish (0-20 mm) became present in larger numbers beginning in May. Fish larger than 60 mm in length appeared to be in spawning condition.

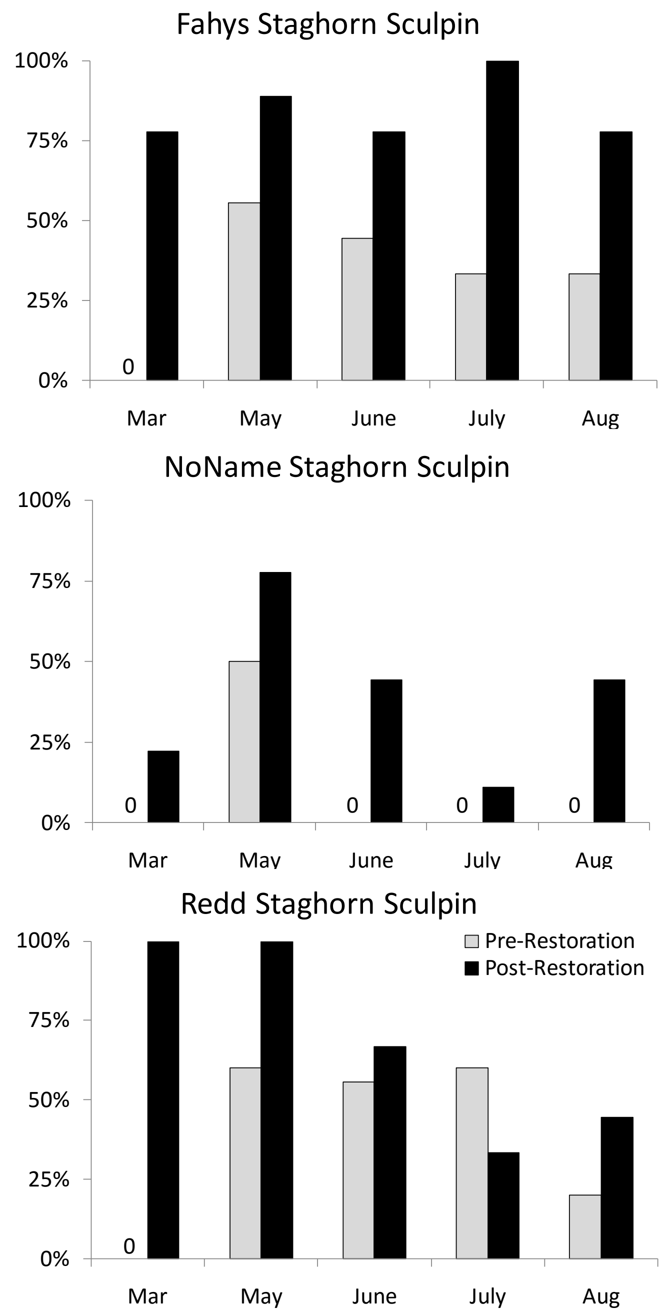

Figure 43. Staghorn sculpin monthly occupancy rates for Fahys, NoName and Redd sub-basins during pre-restoration (2005) and post-restoration (2013) sampling.

Three spine stickleback occupancy rates were estimated as the percent of sub-basin locations occupied divided by the total number of sub-basin locations available, rather than the percent of mainstem locations occupied. This was done because during several sampling months there was no mainstem reference occupancy to provide a comparison. Fahys occupancy rate dropped after restoration in all months but June (Figure 44). NoName occupancy increased during four of the five months (Figure 44). Redd occupancy rates increased during three months and decreased during two (Figure 44).

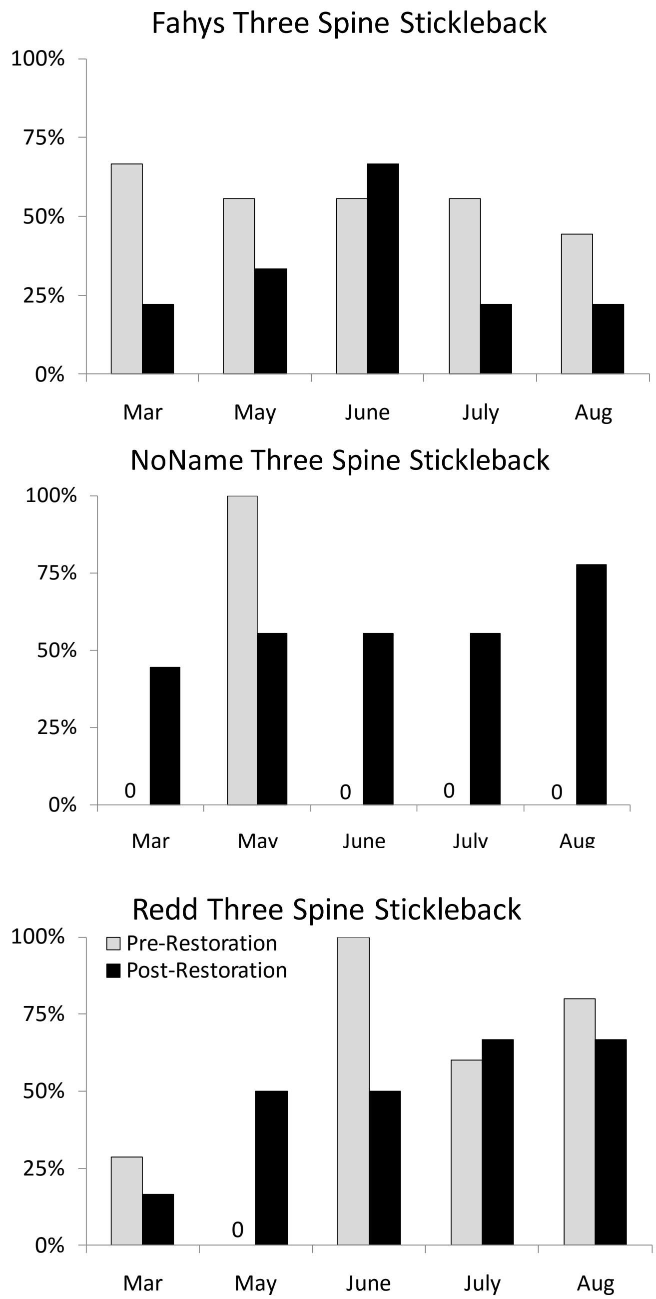

Figure 44. Three spine stickleback monthly occupancy rates for Fahys, NoName and Redd sub-basins during pre-restoration (2005) and post-restoration (2013) sampling.

Peak month mean catch per unit effort

Chinook

Mean peak CPUE values for all three restoration sub-basins were lower than reference mean CPUEs during pre-restoration sampling (2005 and 2010), and became greater than the reference during post-restoration sampling (2013) (Figure 45). During post-restoration, mean peak CPUE was highest in NoName and was associated with early migrant fry. Peak month was determined separately for each sub-basin, and reference data from that same month (averaged across Coquille River sample locations) were used for the CPUE comparisons below.

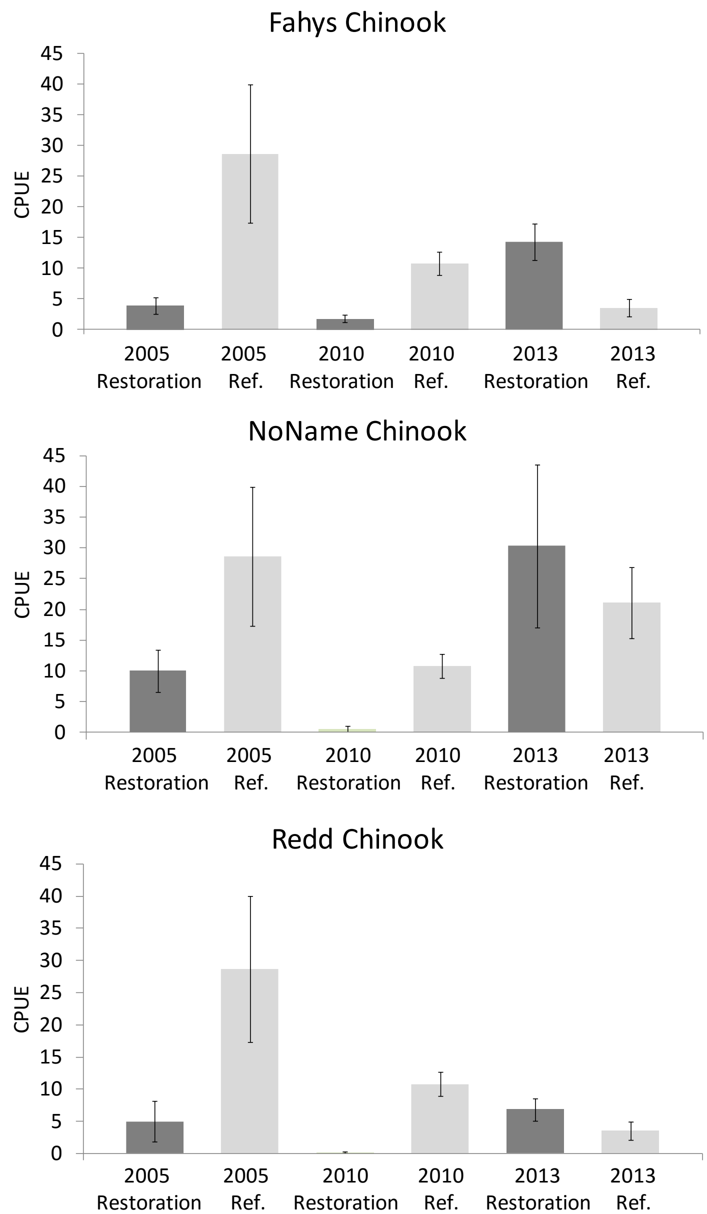

Figure 45. Age-0 Chinook peak month mean catch per unit effort (CPUE) for Fahys, NoName and Redd restoration sub-basins *versus* the reference (Coquille River) during pre-restoration (2005, 2010), and post-restoration (2013). Smaller values (NoName and Redd 2010 Restoration) were between 0 and 0.5 CPUE. Vertical lines represent one standard error of the mean (SEM).

Staghorn Sculpin

Mean peak CPUE values for all three sub-basins were lower than the reference mean CPUEs during pre-restoration sampling (2005 and 2010) and became greater than the reference during post-restoration sampling (2013) (Figure 46). Peak month was determined separately for each sub-basin, and reference data from that same month (averaged across Coquille River sample locations) were used for the CPUE comparisons below.

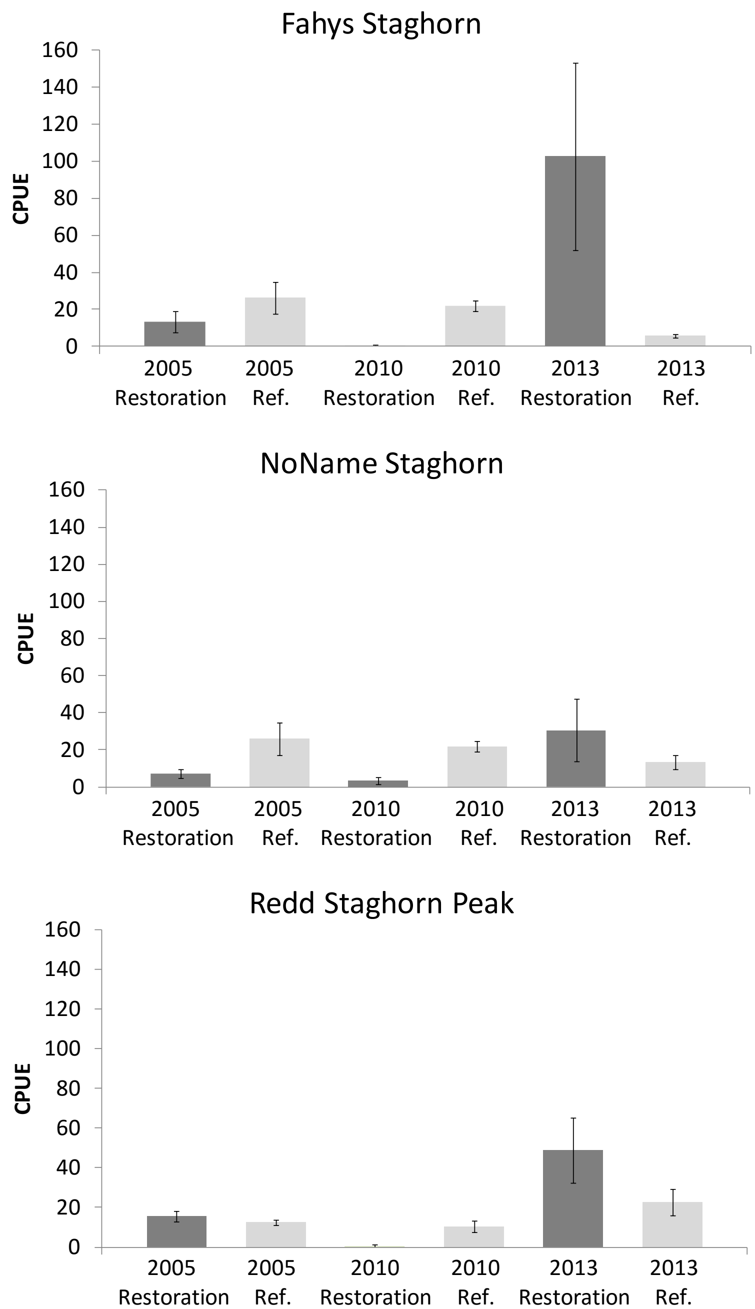

Figure 46. Staghorn sculpin peak month mean catch per unit effort (CPUE) for Fahys, NoName and Redd restoration sub-basins *versus* the reference (Coquille River) during pre-restoration (2005, 2010), and post-restoration (2013). Vertical lines represent one standard error of the mean (SEM).

Three Spine Stickleback

Almost no three spine stickleback were observed during mainstem reference sampling. Mean peak CPUE varied during the three years of observation and was also highly variable among sample locations due to the patchy nature of three spine stickleback distributions (Figure 47). These results suggest no pre- and post-treatment trend occurred. Peak month was determined separately for each sub-basin, and reference data from that same month (averaged across Coquille River sample locations) were used for the CPUE comparisons below.

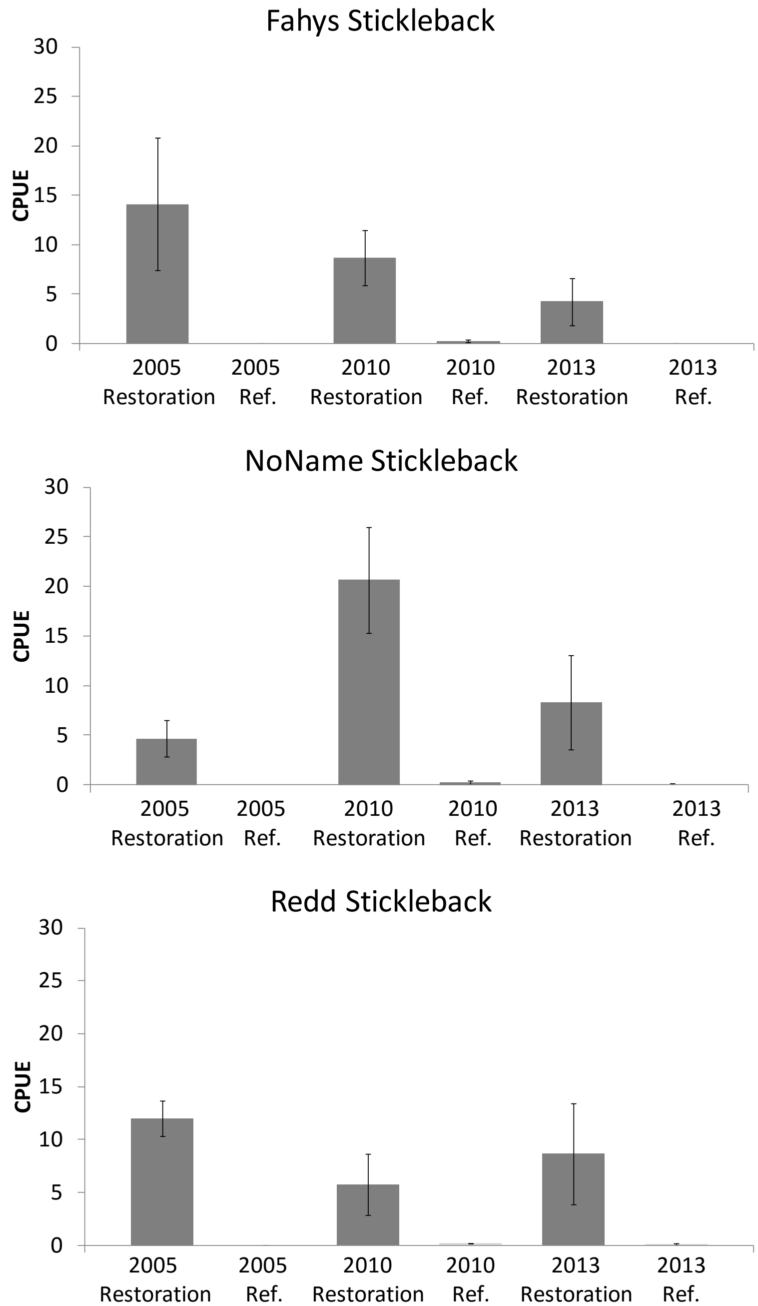

Figure 47. Three spine stickleback peak month mean catch per unit effort (CPUE) for Fahys, NoName and Redd restoration sub-basins *versus* the reference (Coquille River) during pre-restoration (2005, 2010), and post-restoration (2013). Vertical lines represent one standard error of the mean (SEM).

CPUE ratios

We examined the relative increase in fish use of the restoration site by comparing the ratio of mean peak CPUE for a given sub-basin to mean CPUE for the reference locations. Fahys age-0 Chinook ratios went from a pre-treatment low of 14% to a post-restoration high of 400%. NoName age-0 Chinook ratios went from a pre-restoration low of 4% to a post-restoration high of 144% (Figure 48). Redd ratios went from a pre-restoration low of 2% to a post-restoration high of 192% high.

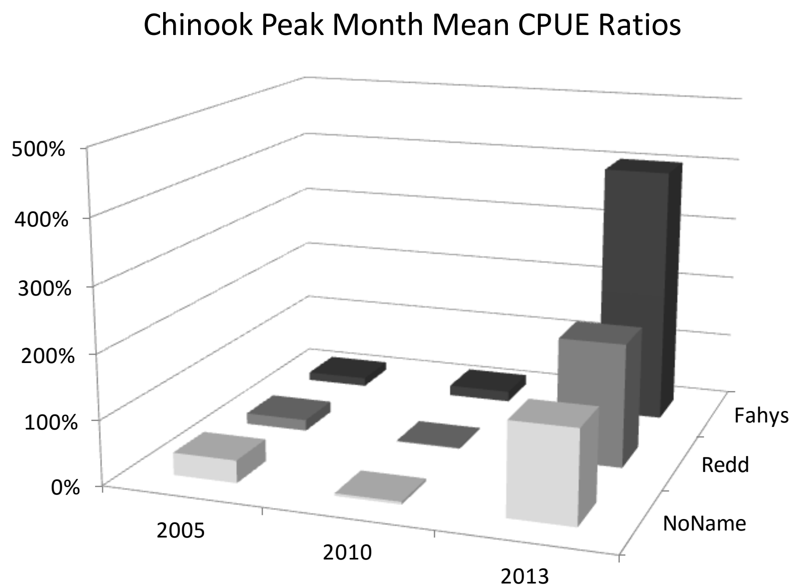

Figure 48. Age-0 Chinook peak month CPUE ratios. CPUE ratio is the ratio of mean peak CPUE for a given sub-basin to mean CPUE for reference locations.

Staghorn sculpin sub-basin ratios were 51%, 2% and 1850% for 2005, 2010, and 2013 respectively for Fahys (Figure 49). NoName ratios were 28%, 16% and 231% for 2005, 2010 and 2013 respectively.

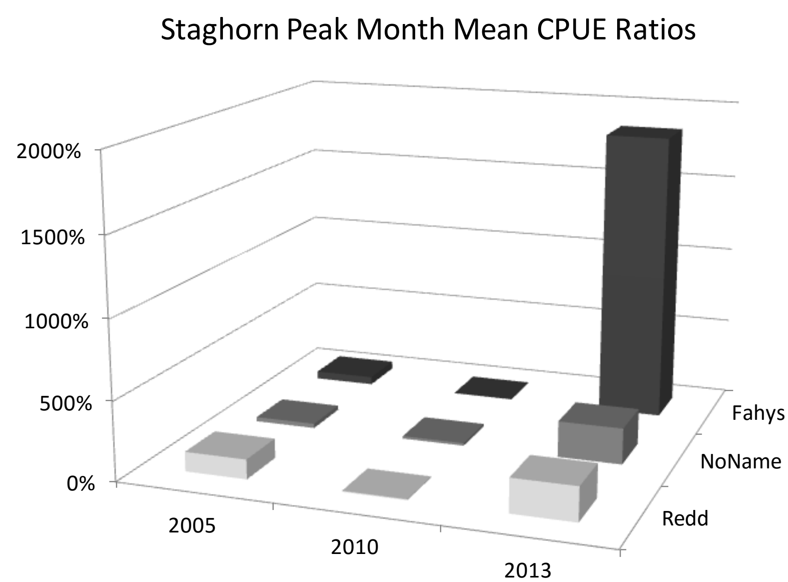

Figure 49. Staghorn sculpin peak month CPUE ratios.

Using BACI analysis, significant restoration effects on CPUE ratio were observed for staghorn (p=0.02; Appendix B, Table B33) but not for age-0 Chinook and three spine stickleback. Significant effects of wood structures were observed for age-0 Chinook (p=0.008; Appendix B, Table B34) and staghorn sculpin (p<0.0001 ; Appendix B, Table B33).

Tidal Migration Patterns

Post-restoration comparison of mainstem river to wetland tidal migration patterns

Peak migration in Fahys increased 354% during post-restoration (Table 6). Before restoration, migration in Fahys was out of the wetland into the tidegated culvert near high slack tide. In comparison, during post-restoration, fish moved out of the wetland early in the flood tide, moving against the current to feed and mill about within the camera transect. The feeding and milling behavior ceased after two hours. Migration was then limited until the ebb tide began at which time fish migrated from the river into the wetland feeding on flushing prey resources being flushed out with the tide. The peak in migration occurred during this early ebb tide feeding period.

Peak migration in NoName increased 200% during post-restoration (Table 6). During pre-restoration, total migration in NoName was very limited, with peak migration being out of the wetland. After restoration, the pattern changed to a peak in-migration, but the number of fish remained limited (Table 6).

Peak migration in Redd increased by 750% during post-restoration (Table 6). Pre-restoration migration in Redd was limited (Table 6). The migration pattern changed during post-restoration with an increase in migration beginning early in the flood tide and continuing at a low rate through the end of the afternoon ebb tide. Peak migration occurred late in the ebb tide. All migration was from inside the wetland out to the Coquille River. We suggest these fish had entered the wetland during a prior low tide, resided in the low tide refugia created by the wood scour holes, and chose to outmigrate during the sampling period.

Pre and post-restoration patterns in the reference sub-basin were different in timing but not in magnitude or in relation to low tide refugia, suggesting limited differences between years. Peak migration in the reference sub-basin was similar (within 1%) during pre and post-restoration periods (Table 6).

Table 6. June pre and post-restoration peak tidal migration estimates and percent increase.

Sub-basin	Pre-restoration Peak Migration	Post-restoration Peak Migration	Increase
Fahys	22	85	386%
NoName	4	8	200%
Redd	4	30	750%
Reference (Shipwreck)	63	64	1%

As described above, we used regression analysis to test for significance between pre- and post-restoration tidal migration estimates. We observed no significant restoration effect for any of the species measured (p=0.21 salmonids; p=0.64 three spine stickleback; p=0.44 shiner perch; p=0.72 staghorn). We found this analysis had limited utility due to the need to pool the three sub-basin migration data sets, which diluted the differences in habitat types.

Post-Restoration targeting of age-1 coho tidal migration

In addition to the pre- and post-restoration June sampling, we sampled Fahys and the Reference sub-basin during May 2013 to allow for some evaluation of age-1 coho typically found in deeper water habitats. The peak estimate for Fahys during May 2013 was 1429 while that of the Reference sub-basin was 29. The Fahys peak occurred during the early ebb tide and was predominantly age-1 coho (>100mm in length) while the reference peak occurred during the flood tide and was predominantly age-0 Chinook (>60mm in length).

Post-restoration comparisons of mainstem river to wetland tidal migration patterns

Additional sampling was conducted during post-restoration to examine the river habitat's role in tidal migration patterns within the four sub-basin tidal channels. Peak river bank migration (migration along the bank of the river) was estimated for three locations: just upstream of the mouth of Fahys, just downstream of the mouth of the reference, and between the mouths of NoName and Redd (Appendix A, Figure A9). River bank migration direction near Fahys and between NoName and Redd was predominantly into the current, while that downstream of the reference sub-basin was a mixture of into and with the current. Peak river bank migration was highest near Fahys and lowest near the reference sub-basin (Table 7). Comparing peak river bank migration to that observed in the wetland sub-basins provides an idea of the percent of local fish using the study sub-basins for tidal migrations (Table 7).

Table 7. Comparison of Coquille River bank peak tidal migration estimates and tidal wetland sub-basins, June 2013.

Sub-basin	River Bank	Marsh Sub-basin	Percentage of River
Fahys	421	85	20%
NoName	352	8	2%
Redd	352	30	9%
Reference (Shipwreck)	238	64	27%

Use of large wood by salmonids

CPUE Analyses

CPUE data from the seine sampling were used with a negative-binomial hurdle model to analyze the effects of wood on fish CPUE. Significant impacts of wood were observed in the count model for age-0 Chinook ($p=0.008$; Appendix B, Table B35) and staghorn sculpin ($p < 0.0001$; Appendix B, Table B36), while an inconclusive effect of wood was observed in the zero count model for age-1 three spine stickleback ($p=0.08$, output not shown).

Tidal migration patterns (within wood structures)

In Fahys, with initiation of the flood tide, juvenile salmonids moved out of the wood habitat and migrated toward the Coquille River. Two thirds of the way through the flood tide, another group of fish began migrating out of the wood habitat and further upstream into the wetland. As the tides switched from flood to ebb, upstream fish migrated back down the channel into the wood habitat. During the same ebb tide period fish began to move from the Coquille River into the wood habitat, feeding on prey flushing out of the wetland channel. Within the first 90 min of the afternoon ebb tide, 96 fish had entered the wood habitat. All these fish remained at the end of the afternoon ebb.

In NoName, fish began to migrate into the wooded section from the downstream end during the flood tide. No early ebb tide pulse migration into the flushing wetland channel was observed. By the end of the tidal sampling period a small reduction in fish present in the wood habitat had occurred.

In Redd fish began to migrate out of the downstream end of the wood section early in the ebb tide. This down-channel migration continued, albeit small, throughout the flood and ebb tides. In addition, a similar down-channel migration occurred from low tide refugia further upstream in the wetland, with fish moving into the wood habitat. A small increase in fish within the wood section had occurred by the end of the tidal sampling period.

Discussion

The relative position of a tidal wetland channel within an estuary and the extent of fresh water flow play a significant role in species use. The downstream extent of the Ni-les'tun marsh, where Fahys Creek spills into the estuary, is positioned near the upper extent of the estuary's salt wedge found during the spring. Daily salinities in Fahys Lower are distributed evenly across oligo-, meso-, and polyhaline conditions whereas salinities in NoName and Redd Lower are more evenly split between oligohaline and lower mesohaline conditions (see **Salmonid habitat opportunity: Salinity** above). Within populations, Chinook salmon exhibit considerable variation in juvenile life history, including different ages at time of migration and duration of freshwater and estuarine residency (Reimers, 1973; Carl and Healey, 1984; Healey, 1991). Juvenile anadromous salmonids undergo a physiological process which allows them to become tolerant to increased salinities (McCormick 1994). Coquille age-0 Chinook and coho use the estuary at different points in their physiological development during the spring and summer seasons, prior to entering the ocean (van de Wetering, unpublished data). Some age-0 Chinook and coho emerge (February - March) from the gravel spawning beds upstream and swim immediately to the estuary. Others may emerge and rear above tide water until June or July. Coho salmon are typically thought of as using small stream freshwater habitats until age-1 at which time they undergo a smolting process and migrate through the estuary and on to the ocean (Hoar 1951, Moser *et al*. 1991). Alternatively, Coquille age-0 coho have also exhibited a life history that includes migration to tidal wetlands the fall of their first year of life after which time they rear throughout the winter in the tidal wetlands prior to salt water entry (Lowe Creek, Coquille River; van de Wetering, unpublished data). Size of fish is related to habitat use in that newly emerged age-0 salmonids will use slower and shallower water than older age cohorts (Levy and Northcote 1982; Beamer *et al*. 2005). These habitats include shallow riffles in tidal channels and mainstem river bank edges. As age-0 salmonids grow during the rearing season, they shift (June-August) to use of deeper water habitats (larger tidal channels and thalweg mainstem river habitats) (Beamer *et al*. 2005). When Coquille Basin upriver smolting age-1 salmonids enter the estuary in April-June they use deeper water habitats (van de Wetering unpublished data). Thus, wetland location within an estuary, species distributions based on life-stage habitat requirements, and salt water intrusion all play significant roles in species use of individual tidal wetlands.

Within the present study, tidal sub-basin daily access and residency for all species was driven by species abundance within the broader estuary, habitat availability and habitat preferences. Our results describe restored tidal wetland use that highlights the need for multi-day rearing habitat as well as daily river-to-wetland feeding corridors.

Occupancy, peak month mean CPUE, and CPUE ratio results suggest age-0 Chinook and staghorn sculpin responded positively to the restoration of all three sub-basins. The data suggest there were specific response groups and uses: early migrant salmonids (predominantly in the 35-60m length range) used shallow water smaller channel networks); and staghorn sculpin used smaller channels for spring spawning and rearing of younger fish followed by larger channels for late season rearing. Although our analysis didn't allow us to examine interspecies competition or predation, the pattern of monthly CPUEs of early migrant salmonids and larger staghorn sculpin in reference and restoration locations suggests there may be predator-prey relationships affecting late spring salmonid CPUEs in the restoration sub-basins.

In addition to our analysis of the standard sites monitored during pre and post-restoration, we observed age-0 coho utilizing a single beaver dam/pond located in a tributary to Redd Creek. This pond habitat was predominantly oligohaline year-round, and was tidally influenced all months of the year. This is an unusual circumstance we have observed at a low frequency in several Oregon estuaries. This small population (<300 fish) were present from post-restoration early winter through late spring at which time they appeared to have transformed into smolts and migrated to the lower estuary. Intertidal beaver ponds are a key habitat that was probably much more common prior to European settlement. In the Skagit River estuary, Hood (2012) found that intertidal beaver ponds tripled habitat capacity for juvenile Chinook compared to herbaceous tidal marsh without beaver pools. Intertidal beaver dams and ponds are also present on the Bandon Marsh Unit reference site (Appendix C, Photos C7 and C8).

June tidal migration results -- evaluated as simple expansions of the raw data -- suggest salmonids responded to the restoration of all three sub-basins with the greatest shifts occurring in Fahys. In addition, our results suggest fish migrations in tidal channels are in large part a response to: 1) channel network location and size; 2) species-life-stage; and 3) temperature and salinity conditions. The influence of these three factors is illustrated by the differences between Fahys *versus* NoName and Redd during June – Fahys was located further down river where salinities and temperatures provided more optimal habitat during the June sampling period, when most age-0 early migrant Chinook had grown to be > 60mm in length and had transitioned to deeper, colder, more saline habitats. June results also suggest that apart from timing, there were limited, if any, differences in the reference sub-basin salmonid migration patterns during the pre-restoration period *versus* the post-restoration period. This was interesting, in that Coquille River CPUEs were lower during June post-restoration than they were during pre-restoration.

May 2013 tidal migration pattern results suggest that larger salmonids (>100mm in length) such as age-1 coho will use larger tidal wetland channels for extensive feeding forays during tidal cycles. When comparing Fahys May 2013 results to those of the smaller reference sub-basin during the same period, it is apparent channel size and depth is related to tidal migration. The magnitude of tidal migration in the wetland channels relative to the mainstem river suggests one additional factor plays a role in migration rates - the "supply" of available fish based on available preferred mainstem habitats. Those fish migrating along the banks of the mainstem in the vicinity of Fahys exhibited an upstream migration during the ebb tide which might have focused their feeding and migration activity in part toward the Fahys sub-basin as they would have intersected the tributary junction during their upstream ebb tide migration. Alternatively, those mainstem migrants between Redd and NoName as well as below the reference sub-basin, migrated against the flood tide current, similar to near Fahys, but a had more limited or no ebb tide response that would have focused their feeding on the flushing sub-basins of Redd and the reference. Although this interpretation is speculative, it provides illustrations of how broader river morphology and flow patterns may play a significant role in tidal migration patterns.

The hurdle model analysis clearly demonstrated the effects of large wood on age-0 Chinook and staghorn sculpin. This was especially obvious to the observer during the early spring sampling of 2013, when age-0 Chinook were highly abundant in NoName and Redd wood treatment locations, as well as the late season abundance in Fahys wood treatment locations. This early spring pattern was less obvious in the larger Fahys channel where deeper water and higher velocities occurred, but might have been similar had we sampled smaller tributary channels within Fahys sub-basin.

As described in Section 2a wood structures placed in the lowest portions of the restored sub-basins resulted in deep scour holes creating low tide refugia. To avoid large variations in CPUE we purposefully chose not to incorporate key scour pool habitats located in the lowest portions of the three restoration sub-basins into our suite of sampling locations. Ancillary sampling of these deep scour pools suggest average monthly densities were 10 to 100 fold greater than those observed at the standard sampling locations. This effect appeared to be even greater for age-1 coho sampled during May 2013 and age-0 Chinook sampled during July 2013.

The tidal migration wood habitat data demonstrate the use of the wood structure scour holes by juvenile salmonids as low tide refugia. Results from Fahys and Redd sub-basins showed juvenile salmonids leaving the section defined as high density wood scour pool habitat at the initiation of the flood tide suggesting the ability of fish to remain in the wetland channel during the morning lower-low tide of the day. There appears to be a gradient of response that is associated with the gradient of channel morphological shifts described above wherein fish in Fahys showed the greatest use of the wood structure as low tide refugia habitat and those in NoName showed the lowest use. In addition, the response in Fahys was stronger during the May 2013 sampling compared to the June 2013 sampling suggesting the response by age-1 coho to the wood structures was greater than that of the age-0 Chinook. This comparison of age-1 coho to age-0 Chinook may be inappropriate, as the absolute number of fish available might have been significantly larger during the May 2013 sampling.

Similar to Reimer's work (1973), Beamer *et al*. (2005) describe the different life history types for Chinook salmon found in the Skagit River Basin but in addition he assigned percentage contributions from life history types to the annual migrant population. Beamer *et al*. (2005) estimated 74% of the population were composed of the *early migrant* life history type. This *early migrant* equates to that life history type observed in Redd, NoName and Fahys sub-basins during March, April, May and June. Beamer *et al*. (2005) found a strong density dependent relationship between total annual migrants and number of estuarine early migrants, suggesting that as total annual fry numbers increased, fresh water habitat became limited, and more fish shifted to an early migrant estuarine rearing strategy. In addition Beamer *et al*. (2005) observed a density dependent relationship for total early migrants and total available estuarine wetland habitat – as early migrants filled available estuarine wetland habitats others began using open ocean nearshore habitats. Bottom *et al*. (2005) and Beamer *et al*. (2005) both calculated average residence periods for juvenile Chinook in estuarine wetlands at 35 days. Beamer *et al*. (2005) furthered their calculation by estimating the number of migrants that depend on the wetland habitats during any given year. This was then used to provide some insight as to the relative value of the *on sight rearing role* these tidal wetland habitats provide. Using a simpler more conservative method we provide similar numbers to stimulate discussion by project sponsors. Using the average densities we observed in each restored sub-basin, during only the peak month of age-0 Chinook early migrant distribution, we estimate the Ni-les'tun marsh produced 6022 Chinook smolts in 2013 – a smolt being defined as a fish that has grown from 35-40mm up to more than 60 mm and is ready to rear in full strength sea water. An inclusion of the three additional months used by other early migrants (recall the estimated residence is 35 days) would only increase this number. If ocean survival is assumed to be 1.5%, a logical conclusion would be that the Ni-les'tun restoration resulted in 90 additional adult Chinook spawners.

Use of the restoration site by other species

The early response to the restoration by juvenile salmonids, staghorn sculpin and three spine stickleback demonstrates the need for a range of habitats across tidal wetlands. Habitats that provided smaller order channels with lower tidal exchange rates were used by specific species and life history stages whereas larger channel habitats with greater tidal exchange were used by others. Habitats that provided low tide refugia *via* deeper scour holes associated with wood structures were used by yet other species and life history stages.

Less common but new observations in the restored wetland channels included northern anchovy (*Engraulis mordax*), Pacific herring (*Clupea pallasii*), surf smelt (*Hypomesus pretiosus*), American shad (*Alosa sapidissima*), pile perch (*Rhacochilus vacca*), surf perch (*Embiotocidae*), bay pipefish (*Syngnathus leptorhynchus*), saddleback gunnel (*Pholis ornate*), a small shrimp (*Crangon crangon*), and dungeness crab (*Metacarcinus magister*) (some illustrated in Figure 50). USFWS is monitoring the use of the restored site and reference site by birds and amphibians (Bill Bridgeland, personal communication). Restoration should incorporate a "portfolio" approach that seeks to enhance development of a variety of wetland habitats within an estuary to accommodate as many species and life history stages as possible.

Figure 50. Other fish and shellfish species observed during 2013 fish monitoring at Ni-les'tun. Clockwise from top left: Northern anchovy, surf smelt, crangon shrimp, bay pipefish, saddleback gunnel, larval Dungeness crab.

3. Resilience to storm-related flooding and climate change

Monitoring objective 3: Measure extent of resiliency to storm-related flooding and climate change

3a. Moderation of storm-related flooding

Monitoring Question 3a: Did restoration improve the site's capacity to moderate storm-related flooding?

Metrics: Restored channel morphology, tidal hydrology, and inundation regime. *(Rationale: Now that dikes are removed, the entire Ni-les'tun site provides floodwater storage during high flow events.)*

Dikes protect specific areas from flooding, but do so by redirecting flood waters to other locations, exacerbating flood damage to those areas. Diking also leads to subsidence of land surfaces, creating areas vulnerable to increased flooding if the dike breaks or water levels rise (e.g. with sea level rise) (Portnoy 1999). Compared to diked systems, which focus flood flows and erosive energy, functioning floodplains that are connected to rivers reduce the risk of flooding by increasing flood storage and conveyance (Hood 2004). Volumetric calculations show that one acre of wetland inundated to a depth of 3 ft can store about one millions gallons of water, while also reducing floodwater speed by the presence of trees and other plants. Reconnected floodplains thus allow for water storage and reductions or delays in flood peaks, as well as potential for evapotranspiration and adsorption of water (Potter 1994). Large scale floodplains, such as the Mississippi River Basin, and small scale floodplains, like Grand Kankakee Marsh in Indiana, showed increased flooding and decreased water quality when floodplain wetlands were converted to agricultural lands (Hey and Philippi 1995; USFWS 1996).

Reconnection of the floodplain at the Ni-les'tun restoration site to the tidally influenced Coquille River estuary has re-established the natural inundation regime, which improves the site's potential to moderate flooding in other areas. This re-establishment of tidal flows will allow the dispersal of water in a high water event, reducing flooding to surrounding areas, similar to the flood mitigation Bandon Marsh Reference Unit is providing. Restoring wetlands has become a best management practice (BMP) for storm-water management and flood mitigation (Opperman *et al*., 2009), and the re-establishment of natural inundation regimes has been shown to moderate flooding in other areas outside of Oregon. For example, at Prairie Wolf Slough in Illinois, restoration led to increased areas of flood protection and moderation of stormwater flows (USFWS 1996); and in Massachusetts, 3800 ha of wetland restoration along the Charles River reduced damage from floods by around $17 million a year (Ramsar 2002).

After restoration, tidal inundation regimes at Ni-les'tun were comparable to those of the Coquille River (see **Tidal hydrology** above). Daily maximum tidal heights within the restoration site matched those of the Coquille River outside of the site (Figure 1); pre-restoration, daily maximum water levels were 1.29 m (4.2 ft), but daily maximum tide rose to 2.09 m (6.9 ft) after restoration. Percent inundation for the Ni-les'tun wetlands increased dramatically from pre-restoration to post-restoration (Figures 3 and 4), indicating that the floodplain has been reconnected to the river. Transects at similar elevations at the restoration and reference sites also had similar percent inundation (Figures 3 and 4). Groundwater data also indicated a restored tidal regime (Figure 10; Appendix A, Figure A16). Prior to restoration, the soils at Ni-les'tun dried out in summer, while post-restoration groundwater levels matched those at the reference site, showing saturation during spring tidal cycles (Figure 10). These data indicate that tidal forcing is fully restored, and is present year-round across the entire restoration site (Figure 4; Appendix A, Figure A15).

In summary, the restoration of natural inundation regimes at Ni-les'tun has improved the site's potential to moderate flooding in other parts of the lower estuary.

3b. Climate change resilience

Monitoring Question 3b: Do post-restoration site conditions show potential for improved resilience to climate change?

Metrics: Plant community composition and extent; soil characteristics (% organic matter, pH, and salinity); groundwater levels. *(Rationale: Native brackish marsh plant communities show higher resilience to climate change compared to non-native pastures, because they are tolerant of increased salinity and flooding. Organic matter in soils is an indicator of carbon sequestration and is a major component of accretion, which allows wetland elevations to rise in equilibrium with sea level rise – a process necessary for wetland climate change resilience. Soil pH and salinity are controlling factors for organic matter accumulation and carbon sequestration, as well as other wetland functions related to moderation of perturbation and habitat resilience. Groundwater levels affect organic matter accumulation and flood storage capability.)*

Post-restoration site conditions at Ni-les'tun showed increased potential for resilience to climate change, compared to pre-restoration site conditions. Native tidal wetland plants increase shoreline stability and aid in sediment accretion (key to wetland equilibration with sea level rise), while also having the capability to retreat with increasing sea-levels (Gardner *et al*. 1992, Morris *et al*. 2002, Turner *et al*. 2004). After restoration, the plant community composition at Ni-les'tun shifted towards that of the reference site (Appendix A, Figure A25). Cover of salt-intolerant non-native species decreased after restoration, and salt-tolerant native species were established on the lower parts of the site (Appendix B, Table B13). These salt-tolerant natives are more likely to withstand the increased inundation and salinity that accompany sea level rise, increasing site's resilience to climate change.

Organic matter accretion in soils also increases climate change resilience by allowing marsh elevations to rise with sea-level, since a high proportion of marsh accretion is due to organic matter accumulation (Turner *et al*. 2004, Cahoon *et al*. 2006). Pre-restoration organic matter content of Ni-les'tun soils was only half that of the reference site, probably due to grazing and drainage (Frenkel and Morlan 1991, MacClellan 2012). After restoration, soil organic matter increased significantly and was not statistically different from the reference site (Figure 9). Controlling factors for organic matter accumulation include groundwater levels, soil salinity, and soil pH. Saturated soils (with high groundwater levels) accumulate organic matter due to the lack of aerobic oxidation processes (Mitsch and Gosselink 1993). Prior to restoration, the soils at Ni-les'tun dried out in summer, while post-restoration groundwater levels matched those at the reference site, showing greater duration and frequency of saturation (Figure 10). After restoration, soil salinity at Ni-les'tun increased to levels comparable to the reference site, while soil pH showed no differences between restoration and reference site, or between pre- and post-restoration (Figure 9). The post-restoration similarities between soils at Ni-les'tun and the reference site suggest that the restoration site is becoming more resilient to climate change and that the marsh soils have improved potential to equilibrate with sea level rise (Callaway *et al*. 2011).

Tidal wetlands are capable of sequestering and storing more carbon per unit area than freshwater wetlands (Whiting and Chanton 2001, Brigham *et al*. 2006, Chmura *et al*. 2003). The restoration of tidal wetlands at Ni-les'tun suggests an improved potential for carbon storage at the restoration site – carbon storage which can provide mitigation for atmospheric carbon dioxide emissions (Crooks *et al*. 2014).

In summary, restoration at Ni-les'tun has improved the site's resilience to climate change through re-establishment of native wetland plant communities, restoration of accretion processes and soil organic matter accumulation, and potential for carbon sequestration.

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Appendix A. Additional figures

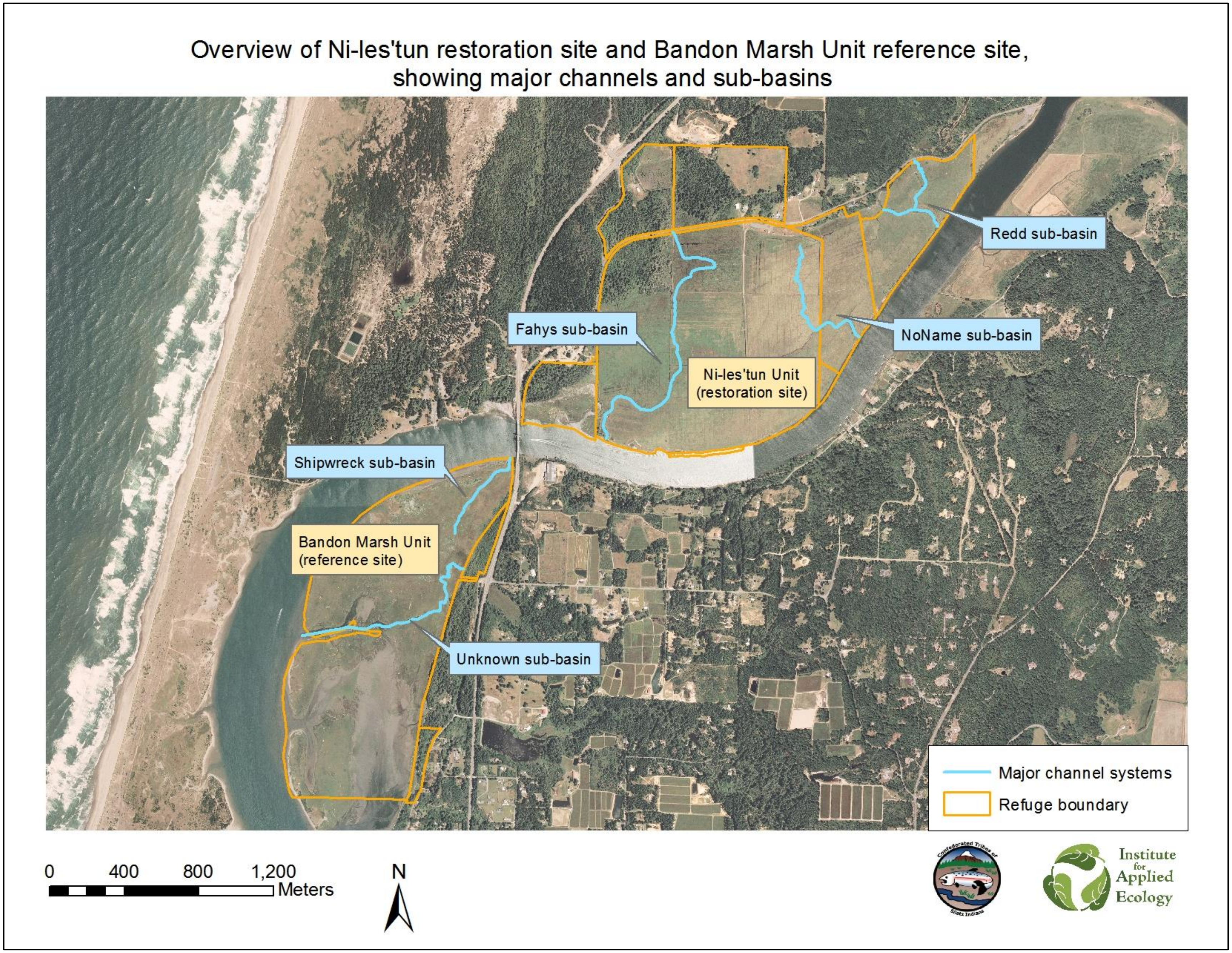


Figure A1. Project area overview showing major channels and sub-basins. Background: 2005 NAIP orthophoto.

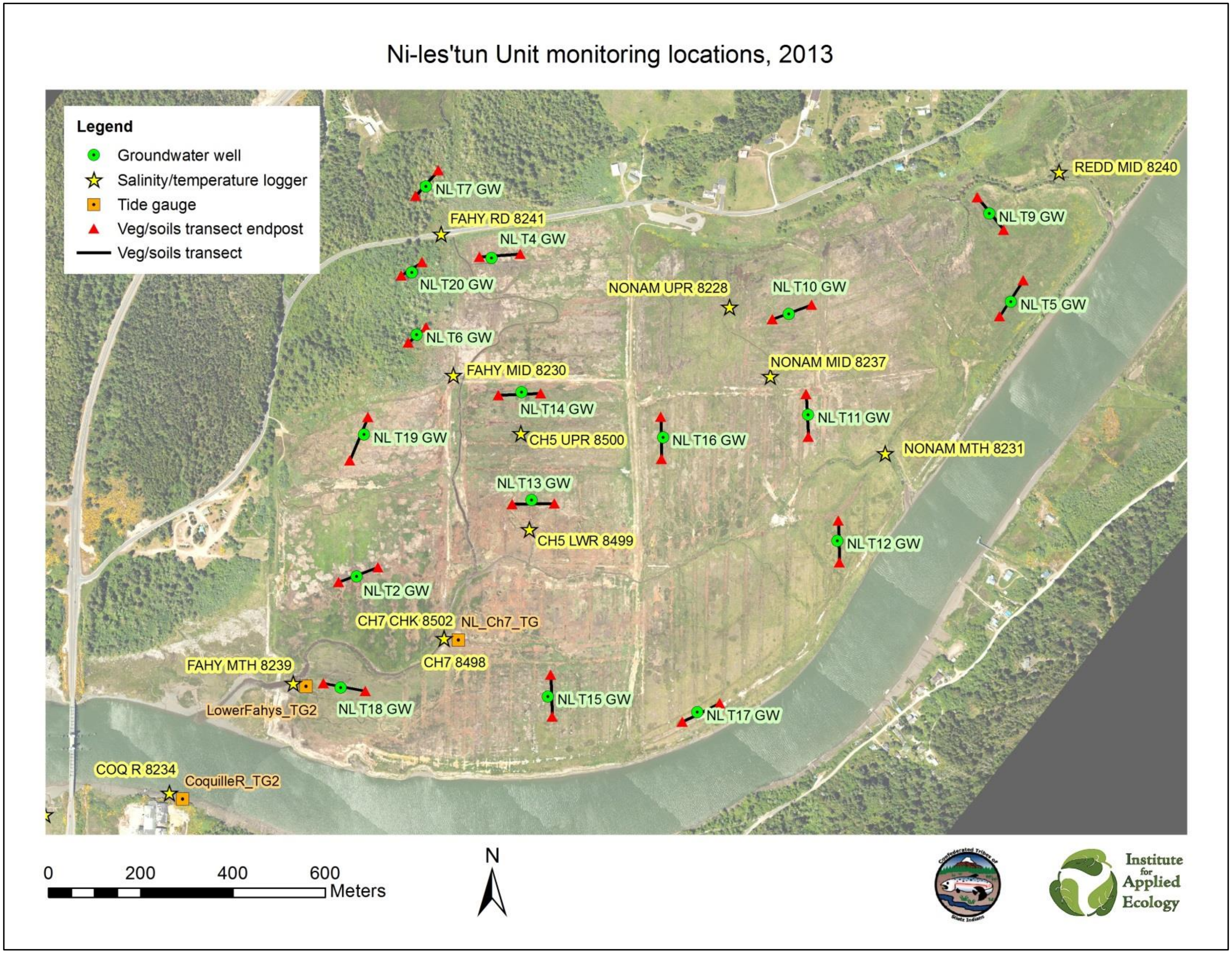


Figure A2. Ni-les'tun restoration site: 2013 sample locations for vegetation, soils, groundwater, tidal hydrology, and surface water temperature and salinity. Background: May 2013 custom aerial photo.

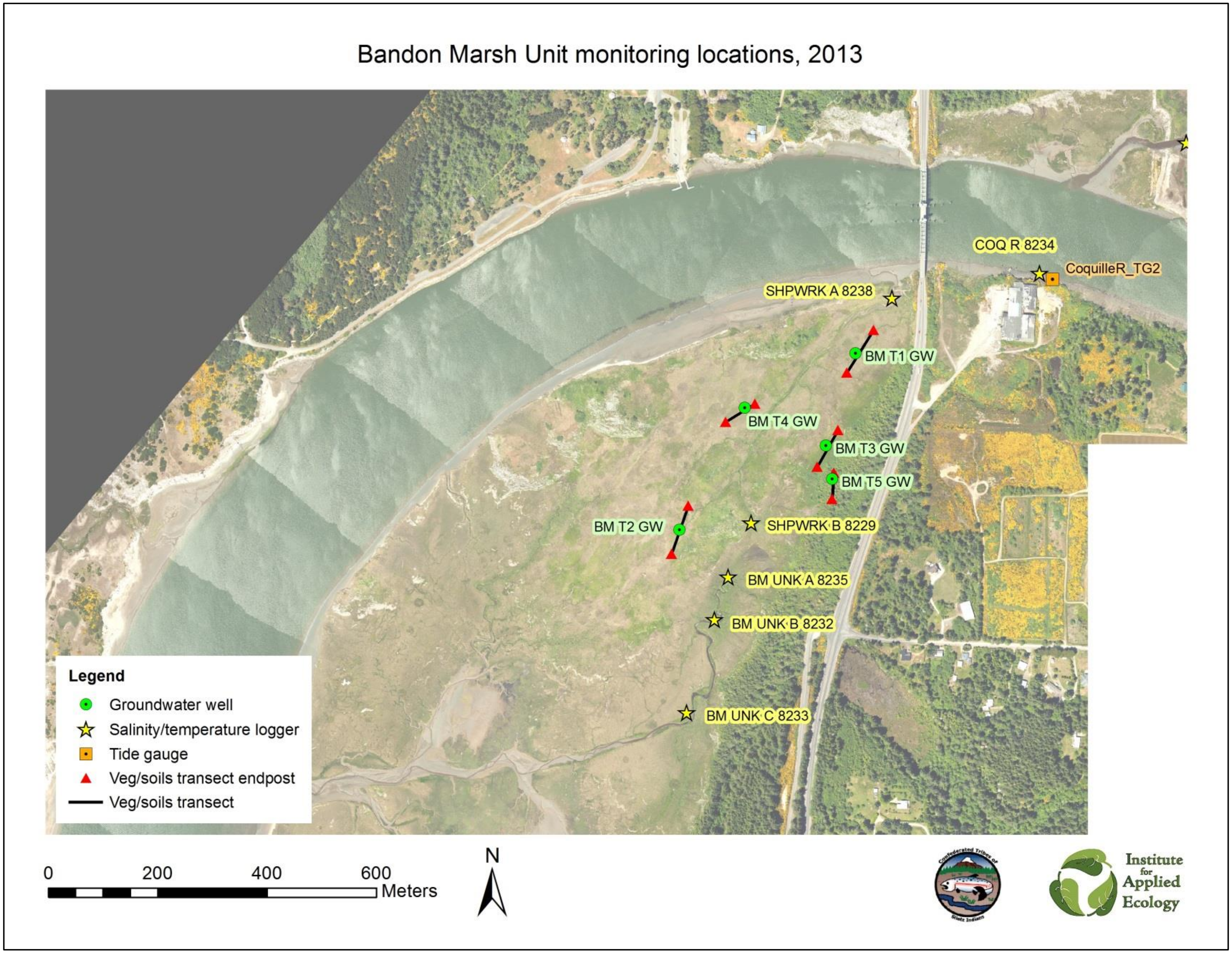


Figure A3. Bandon Marsh Unit reference site: 2013 sample locations for vegetation, soils, groundwater, and surface water temperature and salinity. Background: May 2013 custom aerial photo.

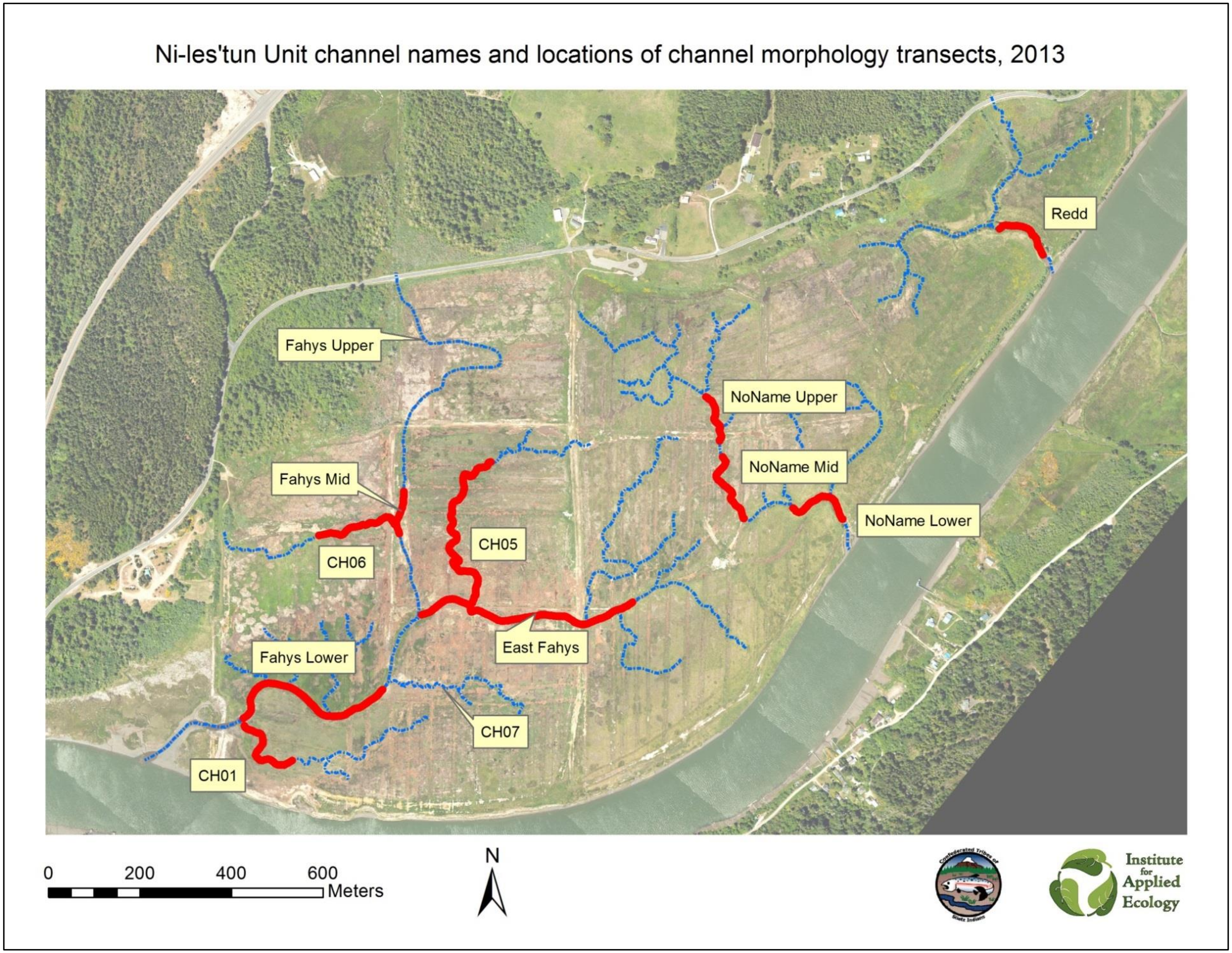


Figure A4. Major channel names at Ni-les'tun restoration site, showing locations of cross-sectional transects in 2013 (red). Cross-sections were measured about every 2 channel widths within the red areas, which included wood and non-wood reaches. For wood reaches, see FigureA33.

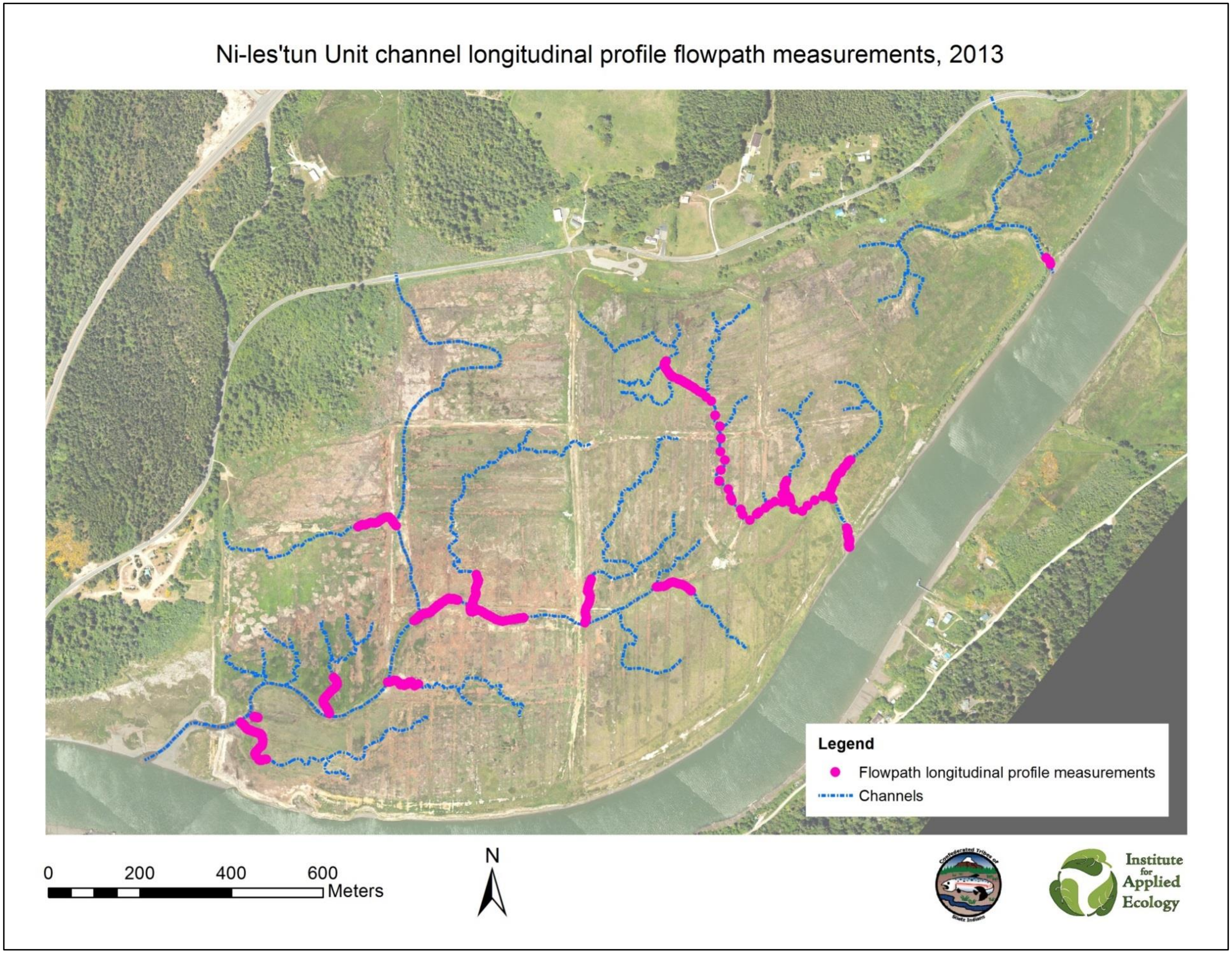


Figure A5. Ni-les'tun restoration site: Locations of RTK-GPS flow path elevation measurements, 2013 (pink).

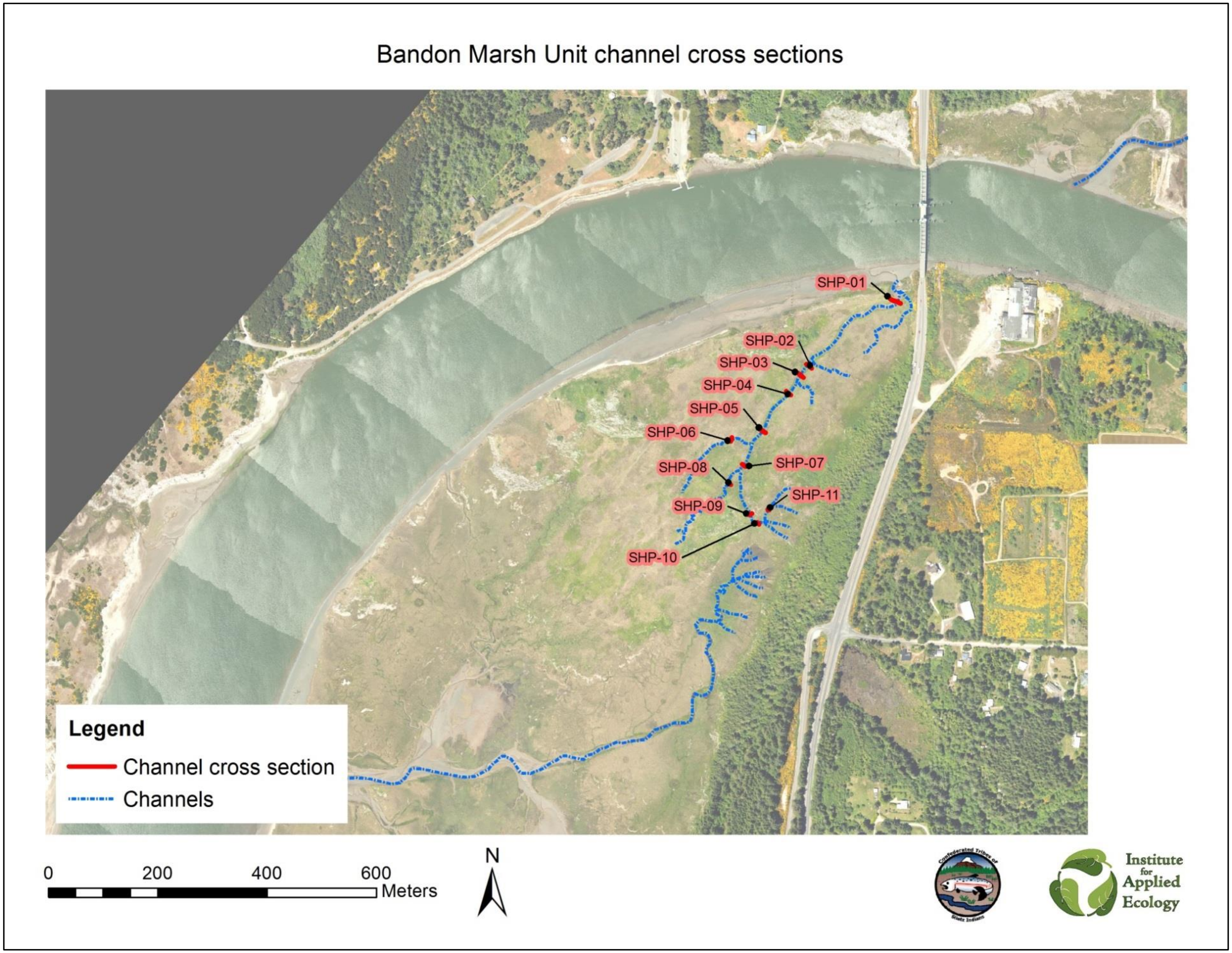


Figure A6. Bandon Marsh Unit reference site: Locations of RTK-GPS cross-sectional transects in 2013.

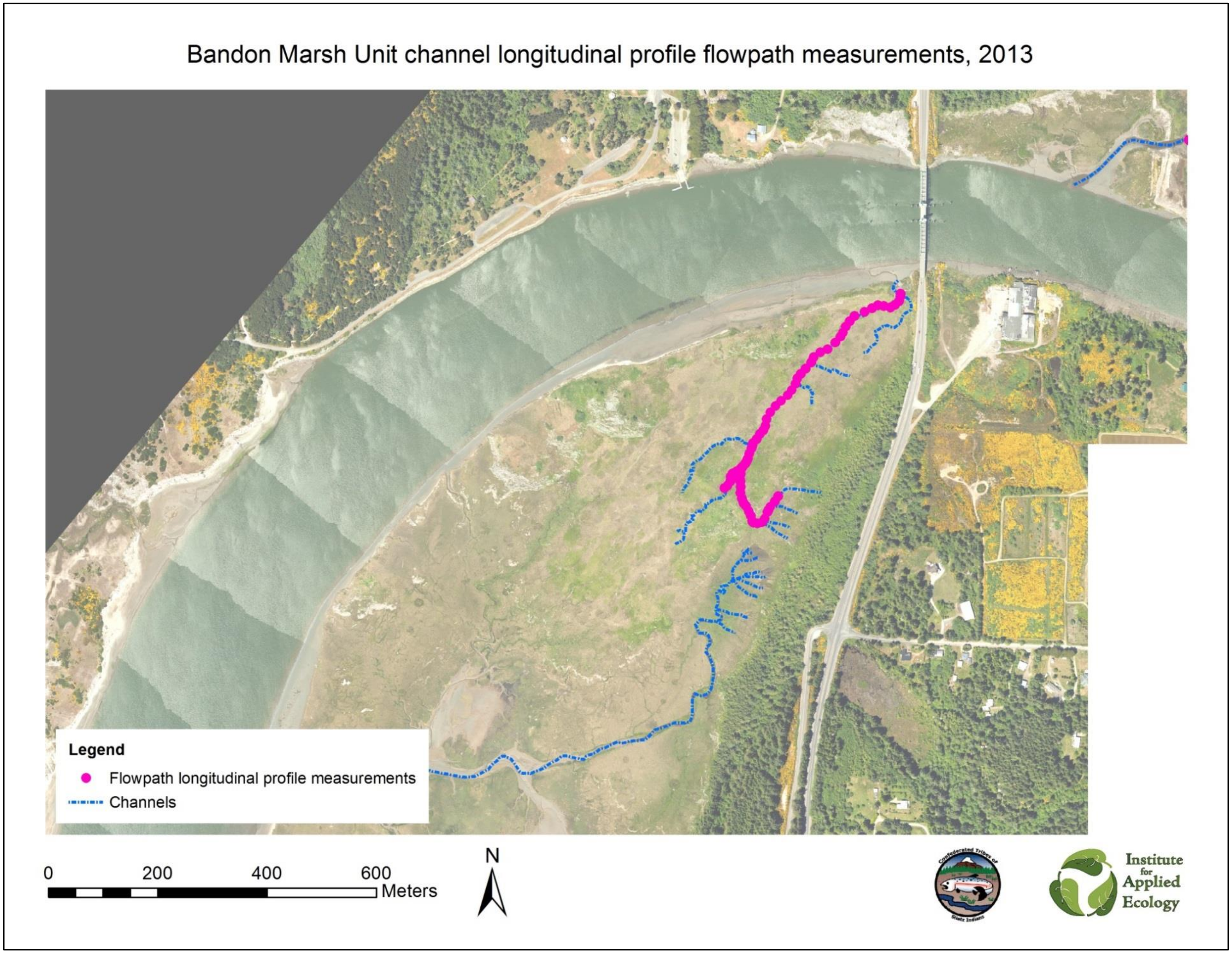


Figure A7. Bandon Marsh Unit reference site: Locations of RTK-GPS flowpath measurements, 2013 (pink).

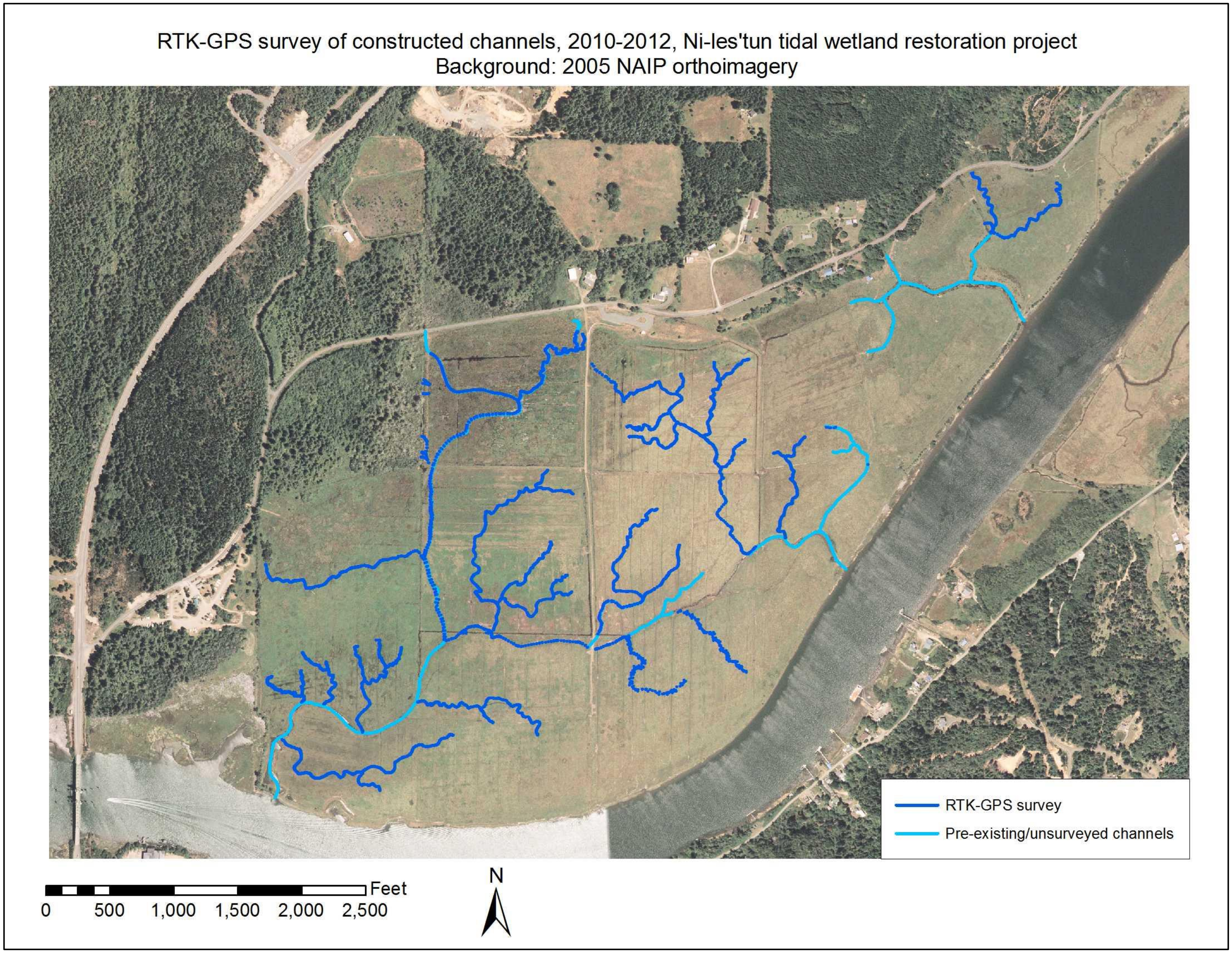


Figure A8. Ni-les’tun restoration site: Pre-restoration “as-built” RTK-GPS survey of restored channels conducted primarily during the baseline period (2010-2011) (from Brophy and van de Wetering 2012). Only the dark blue (excavated) channels were surveyed; the light blue (pre-existing, non-excavated) channels were not surveyed.

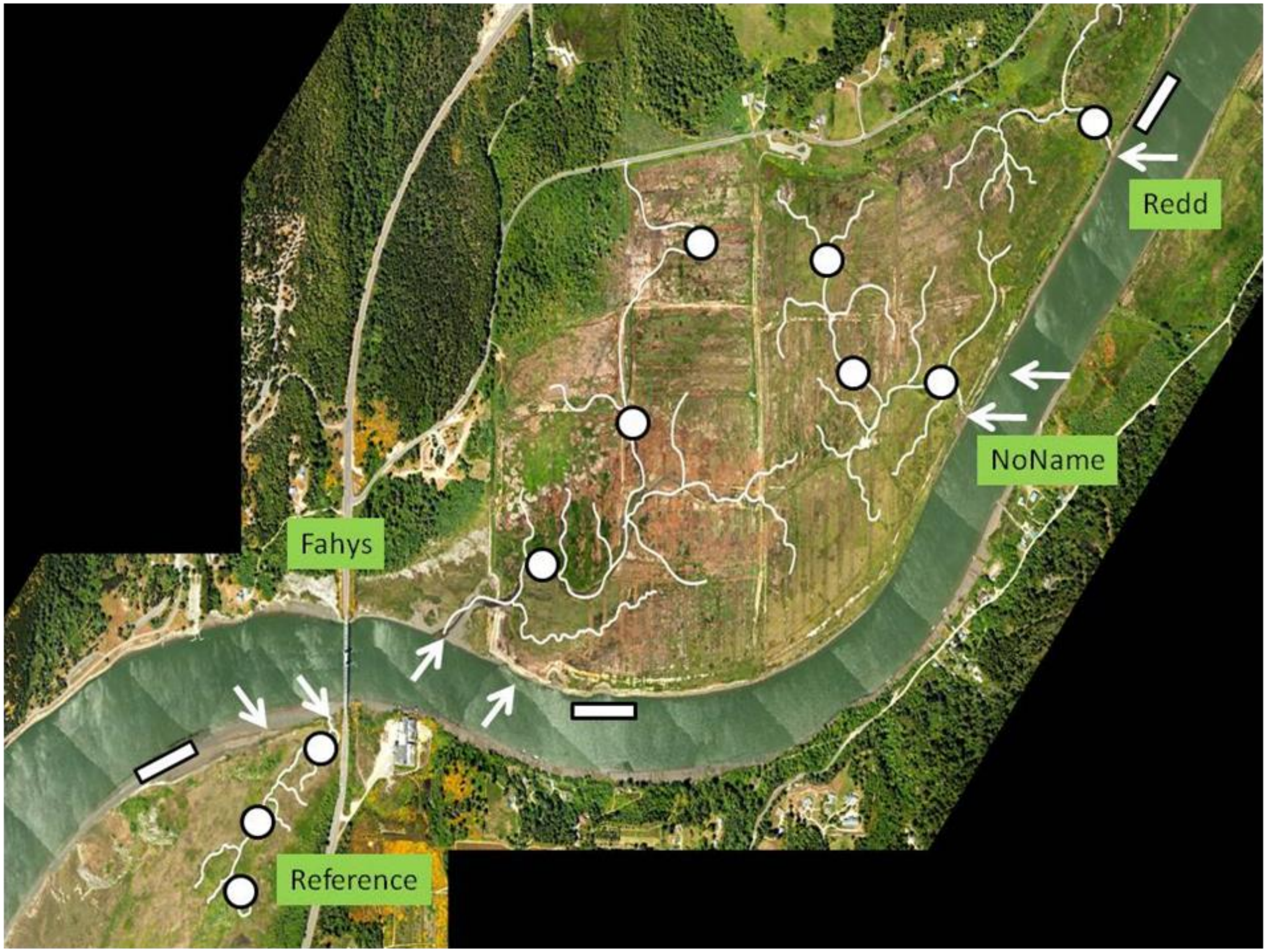


Figure A9. 2013 fish sampling locations: sub-basin reach seine sampling locations (circles, representing the center of each reach); mainstem river reference seine sampling locations (bars); and fish migration sampling locations for the four sub-basins and the mainstem river banks (arrows).

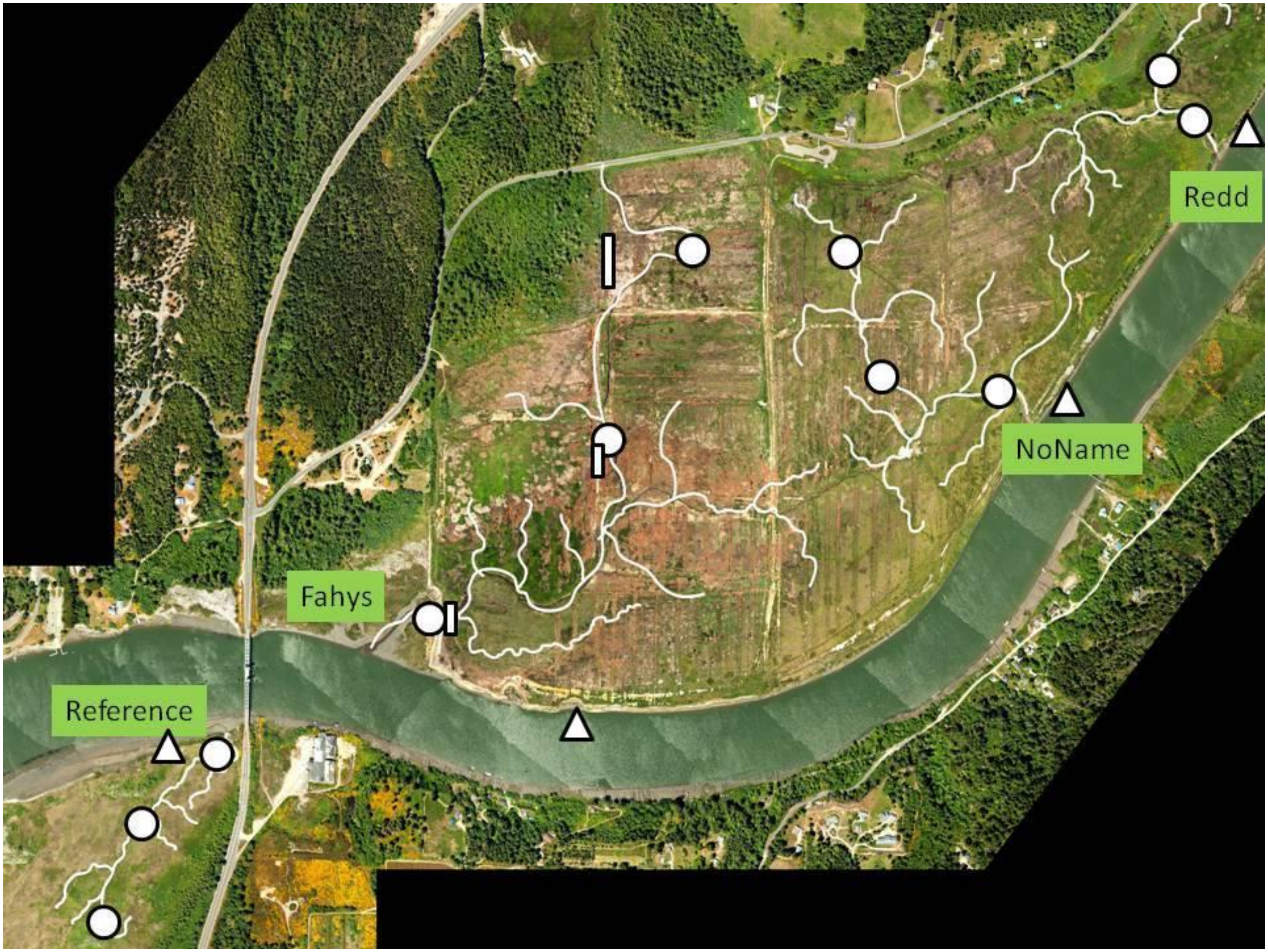


Figure A10. 2013 benthic macroinvertebrate sampling sites in Fahys, NoName and Redd sub-basins (circles on Ni-les'tun restoration site), the reference site (Shipwreck sub-basin), and the mainstem river (triangles). For Fahys, NoName and Reference, the Lower sample is the one closest to the Coquille River; the Upper sample is the farthest from the river; and the Mid sample is between Lower and Upper. Sites sampled in the pre-restoration ditches during 2007 are shown as bars.

Locations of channels used for fish access calculations

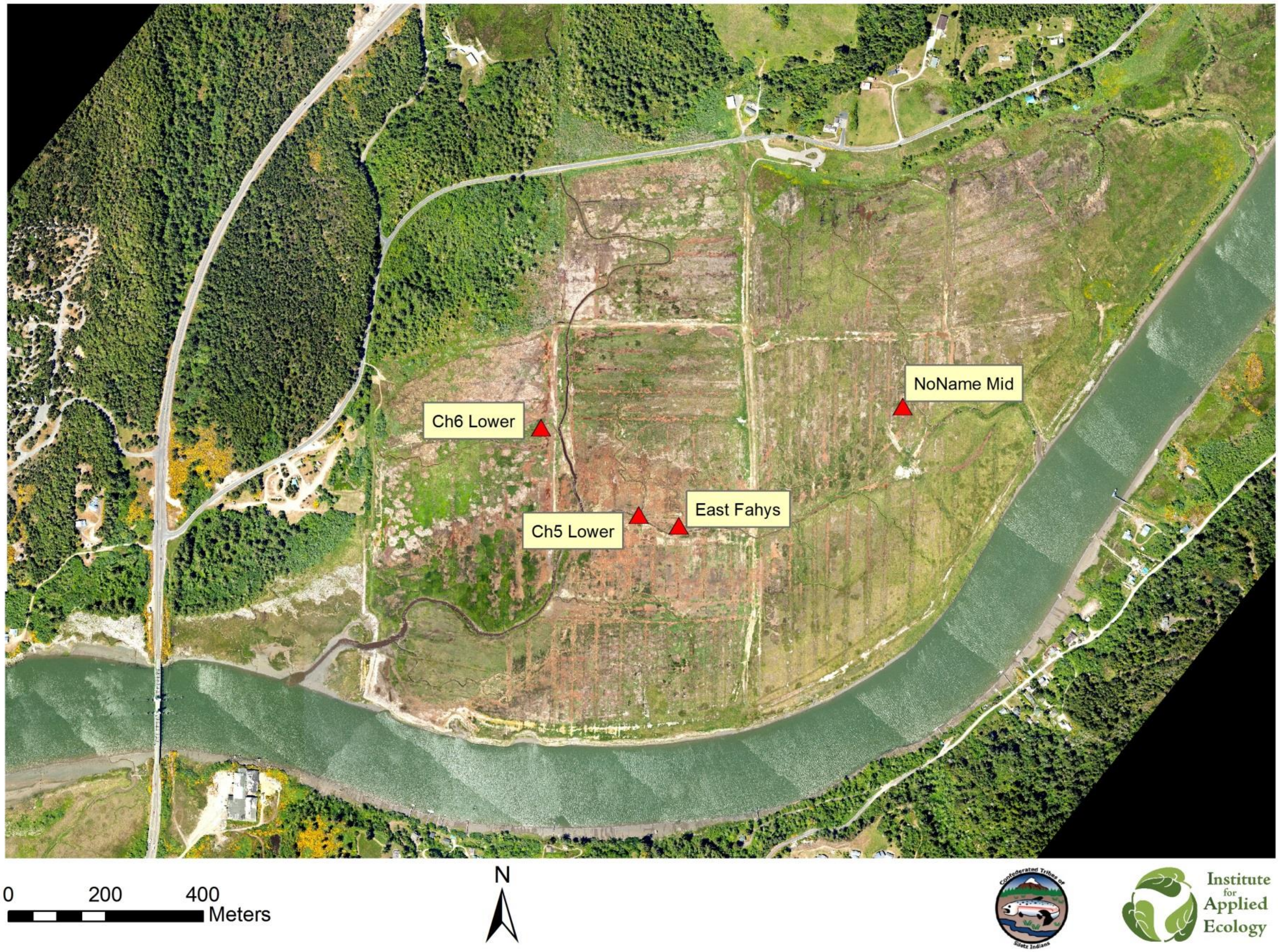


Figure A11. Locations of channels used in calculations of percent inundation for fish habitat availability.

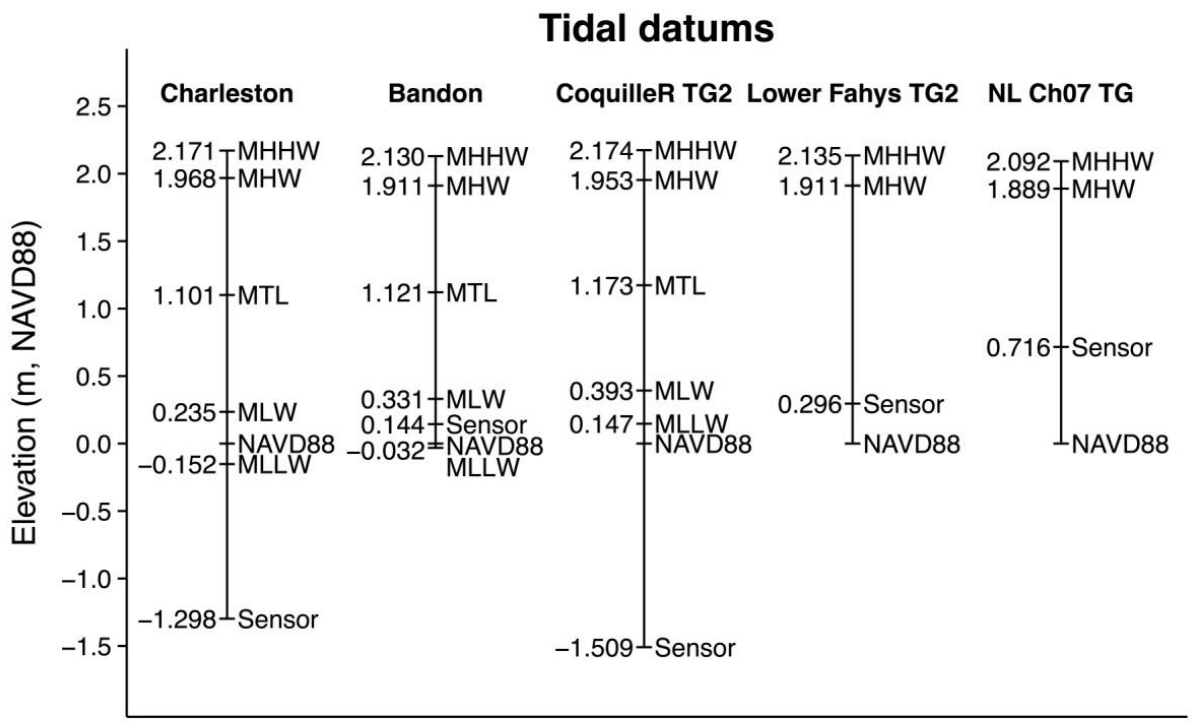


Figure A12. Tidal datums calculated from the three tide gauges installed for the Ni-les'tun effectiveness monitoring program (CoquilleR TG2, Lower Fahys TG2, and NL Ch7 TG), with comparisons to nearby NOAA tide stations (Charleston, station ID 9432780, and Bandon, station ID 9432373). Locations of Ni-les'tun gauges are shown in Figure A2. All data are presented relative to NAVD88.

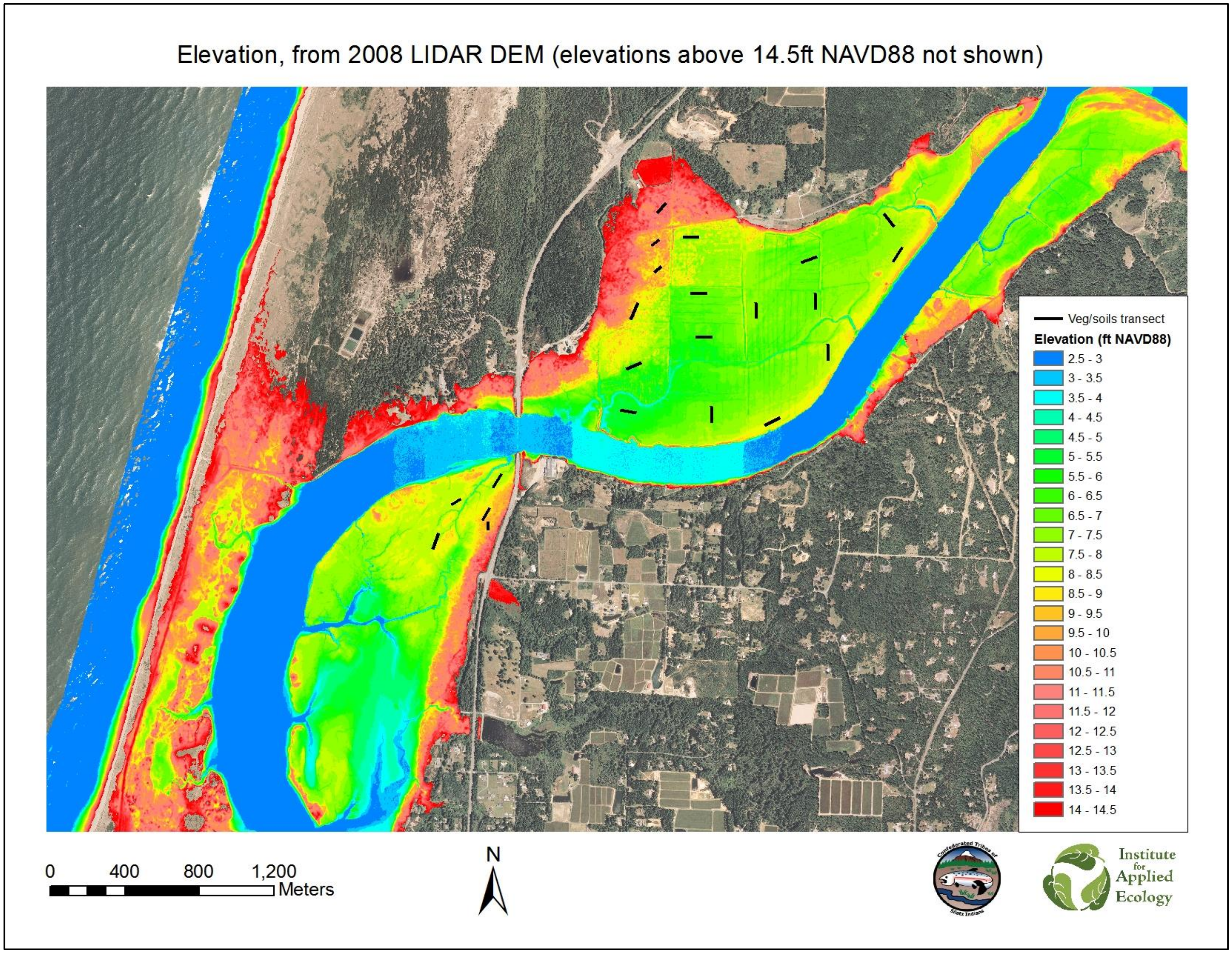


Figure A13. Elevations at the Ni-les'tun restoration site and Bandon Marsh Unit reference site, from 2008 LIDAR DEM. Black lines are vegetation/soils sample transects; see Figures A2 and A3 for transect numbers.

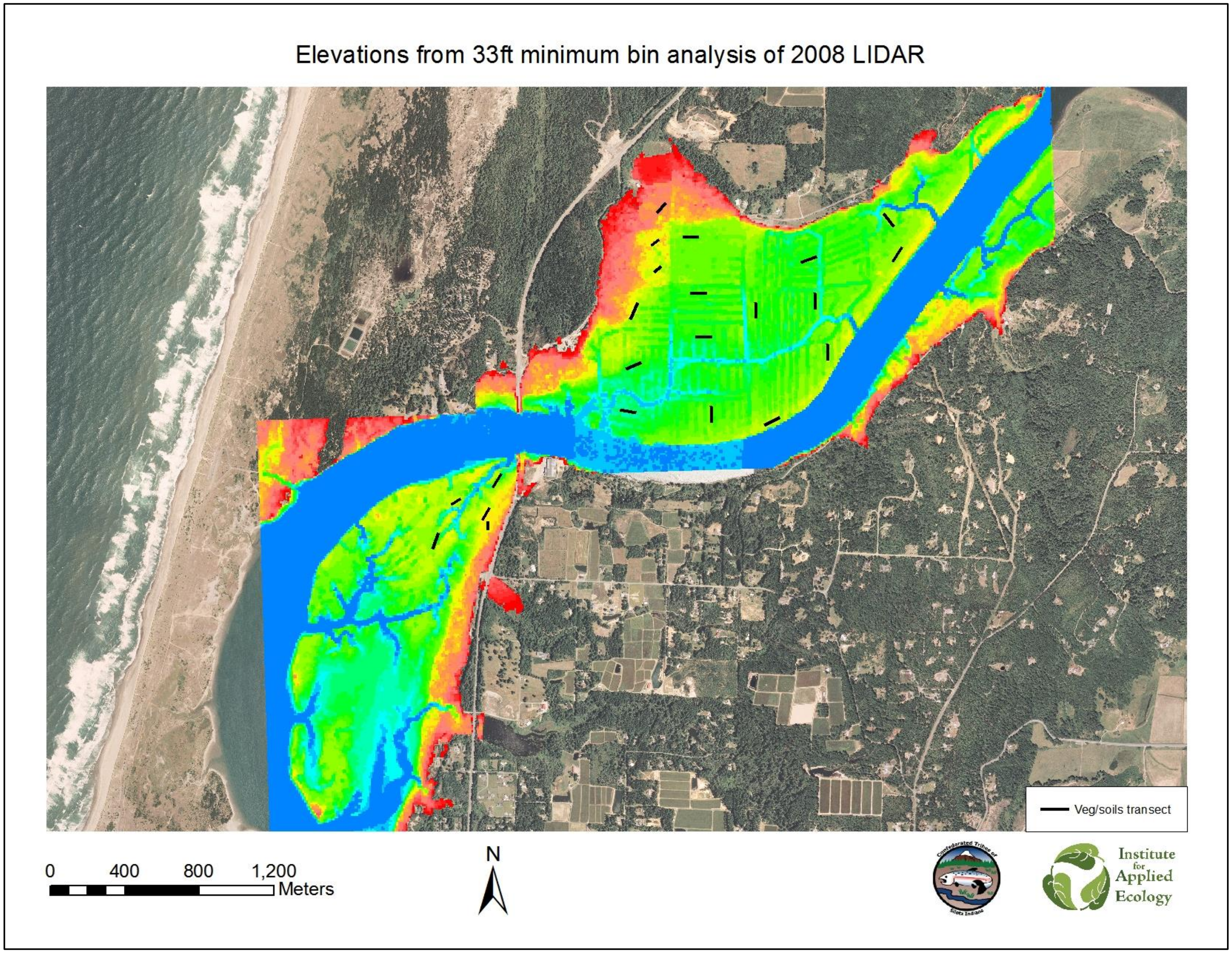


Figure A14. Surface elevation using 33ft (10m) minimum bin analysis of 2008 LIDAR point cloud for the Ni-les'tun restoration site and Bandon Marsh Unit reference site (Ewald 2013). Black lines are vegetation/soils sample transects; see Figures A2 and A3 for transect numbers.

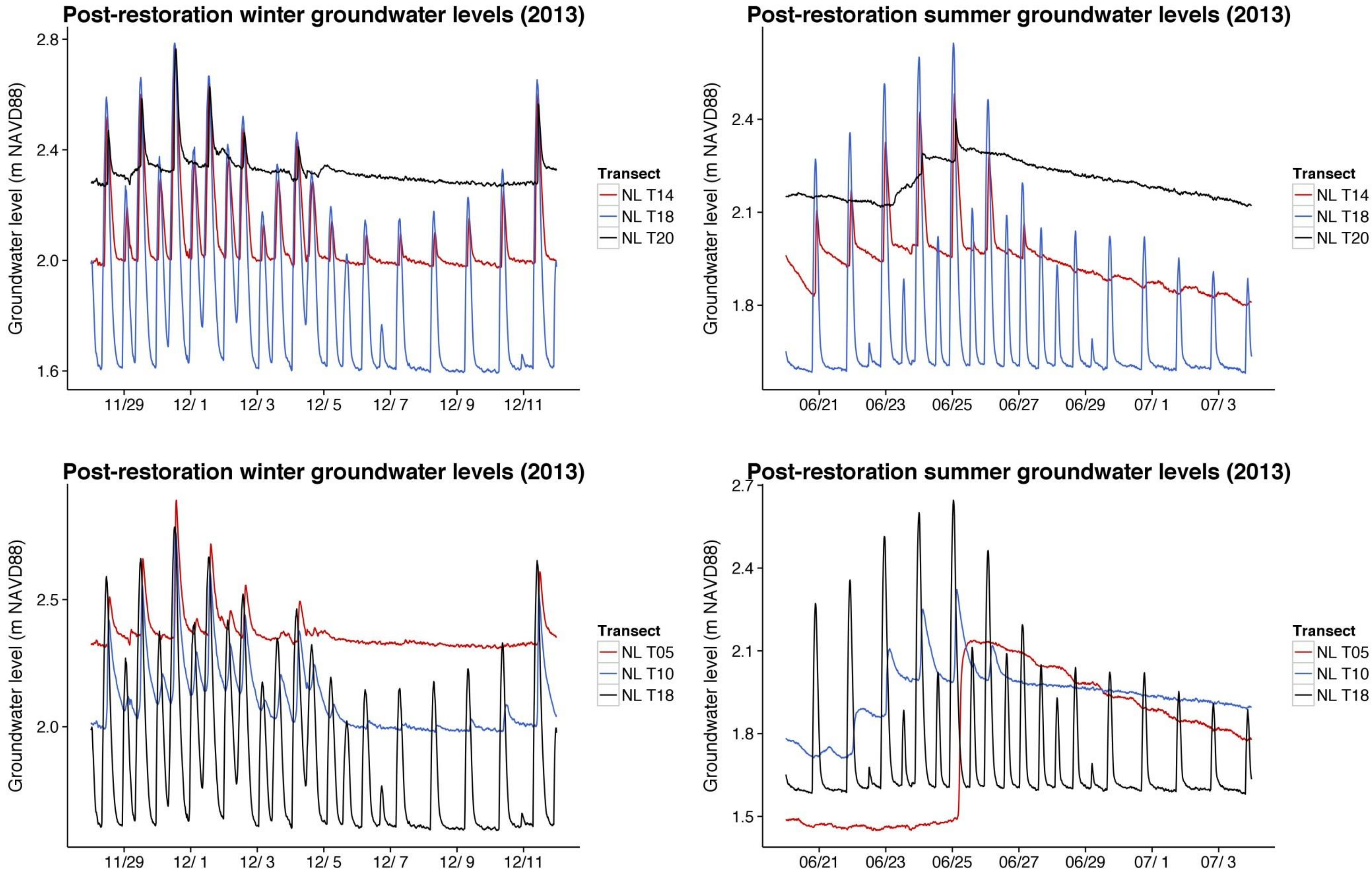
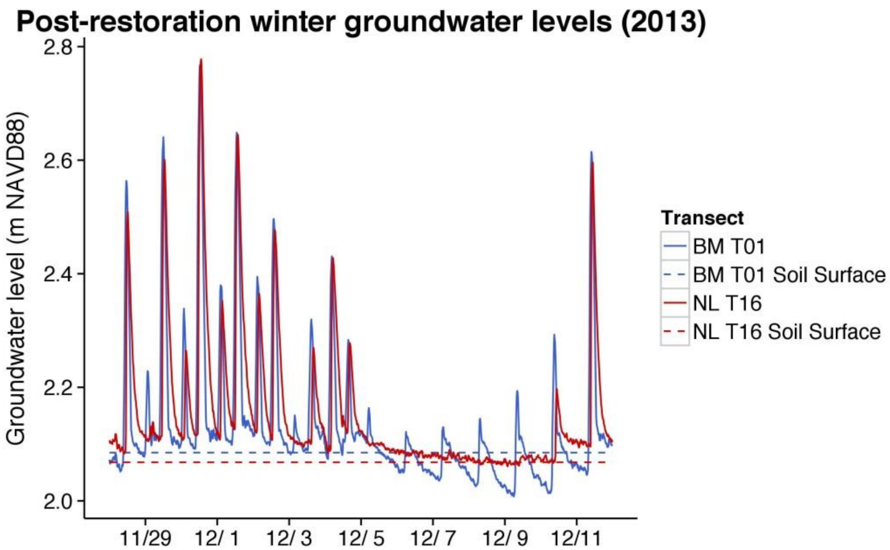


Figure A15. Groundwater data and tide peaks for Ni-les'tun transects at diverse elevations and distances from the Coquille River, showing clear tidal forcing and seasonal changes. NL T20 is a tidal swamp >1 km from the Coquille River, elevation 2.3 m (7.5 ft). NL T5 and NL T10 are at 2.2 m (7.2 ft) and 1.9 m (6.4 ft) respectively. For locations of transects, see Appendix A, Figure A2; for elevations of transects see Appendix B, Table B2.



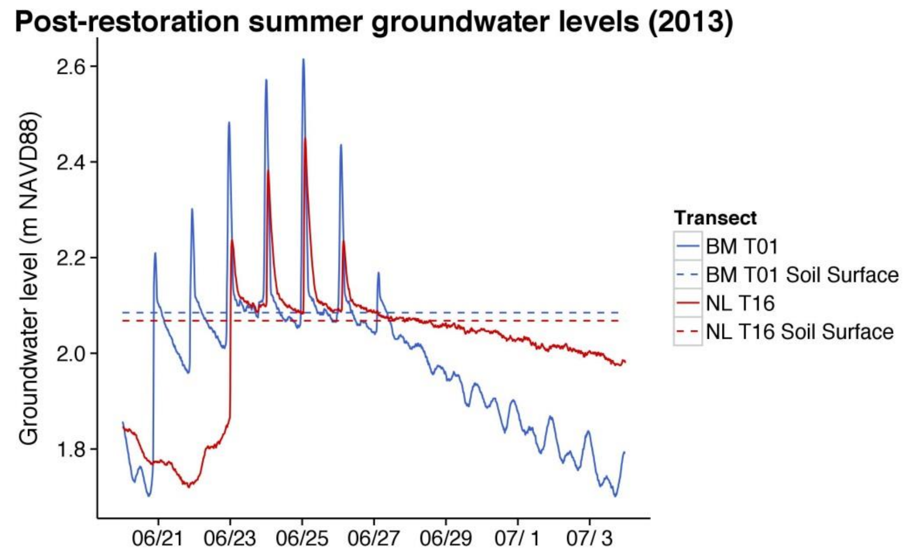


Figure A16. Groundwater data and tide peaks for restoration and reference site transects at similar elevations (NL T16 and BM T1). Spring tide peaks match within 10 cm in winter and 20 cm in summer, illustrating effective restoration of tidal inundation regimes.

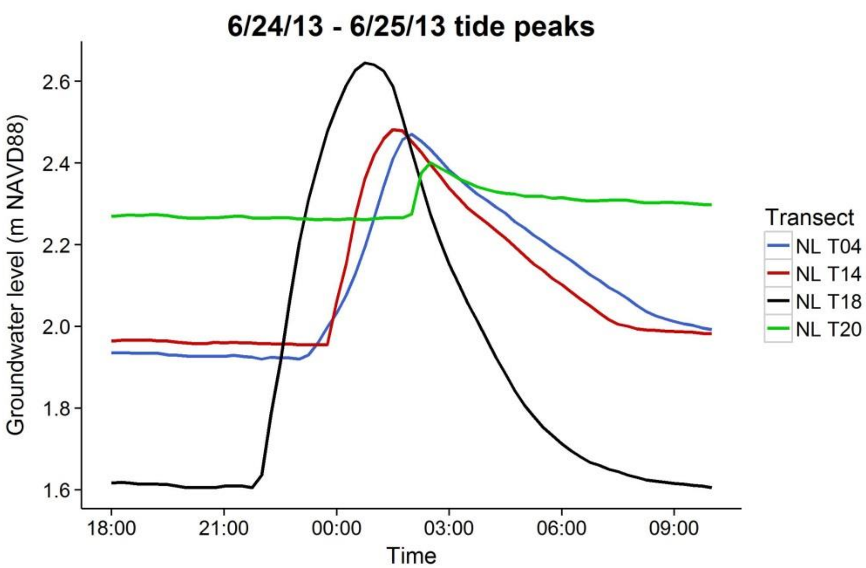


Figure A17. Groundwater peaks from river to hillslope base at Ni-les'tun, illustrating the delay in tide peaks.

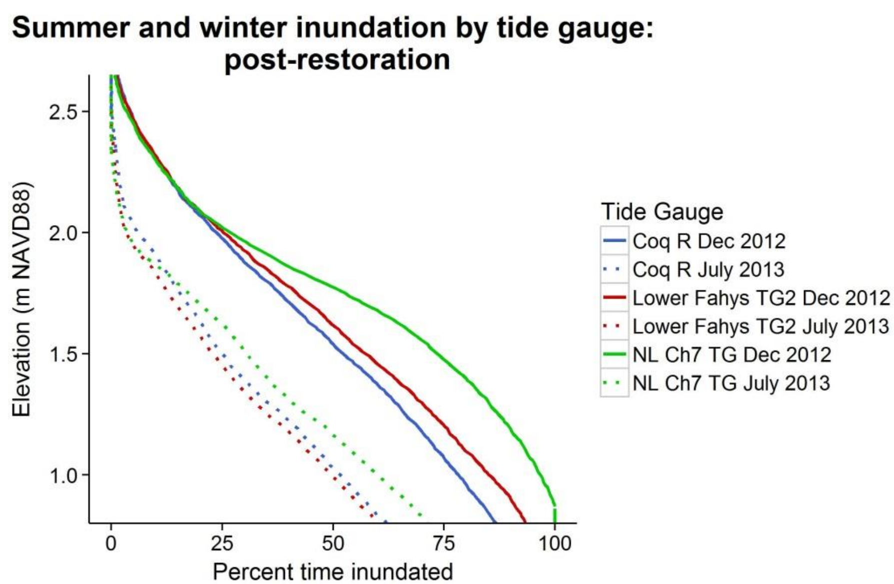


Figure A18. Percent inundation at a range of elevations during winter and summer periods. NL Ch7 TG shows a slight delay in ebb tide drainage.

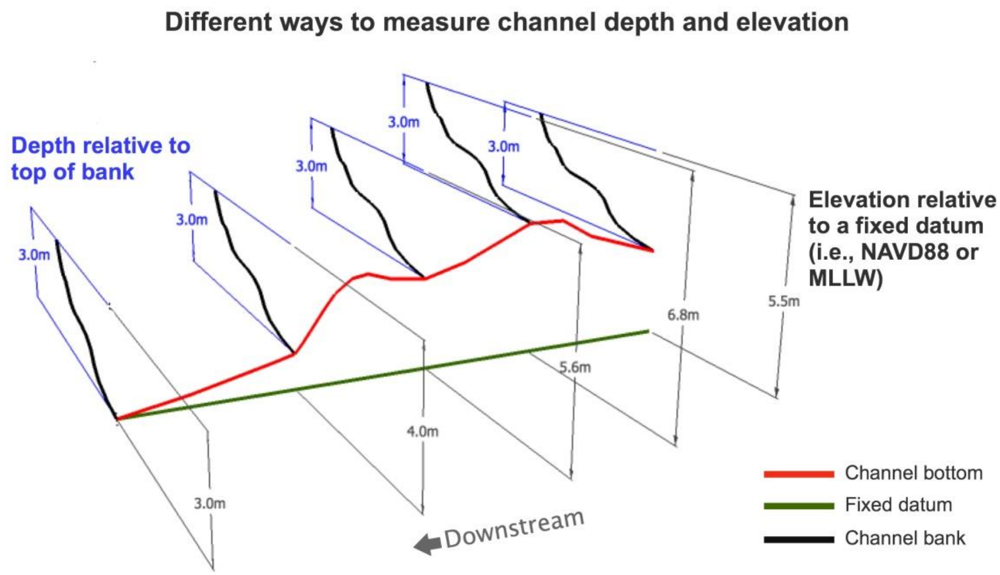


Figure A19. Conceptual illustration of a channel excavated relative to the wetland surface, and the resulting variation in channel bottom elevations relative to a geodetic datum. The channel bottom (red line) in this image is excavated to a depth of 3m below the wetland surface (marked "top of bank"). However, the resulting channel bottom elevation relative to a geodetic datum ranges from 3 to 6.8m, because the top of bank elevation varies.

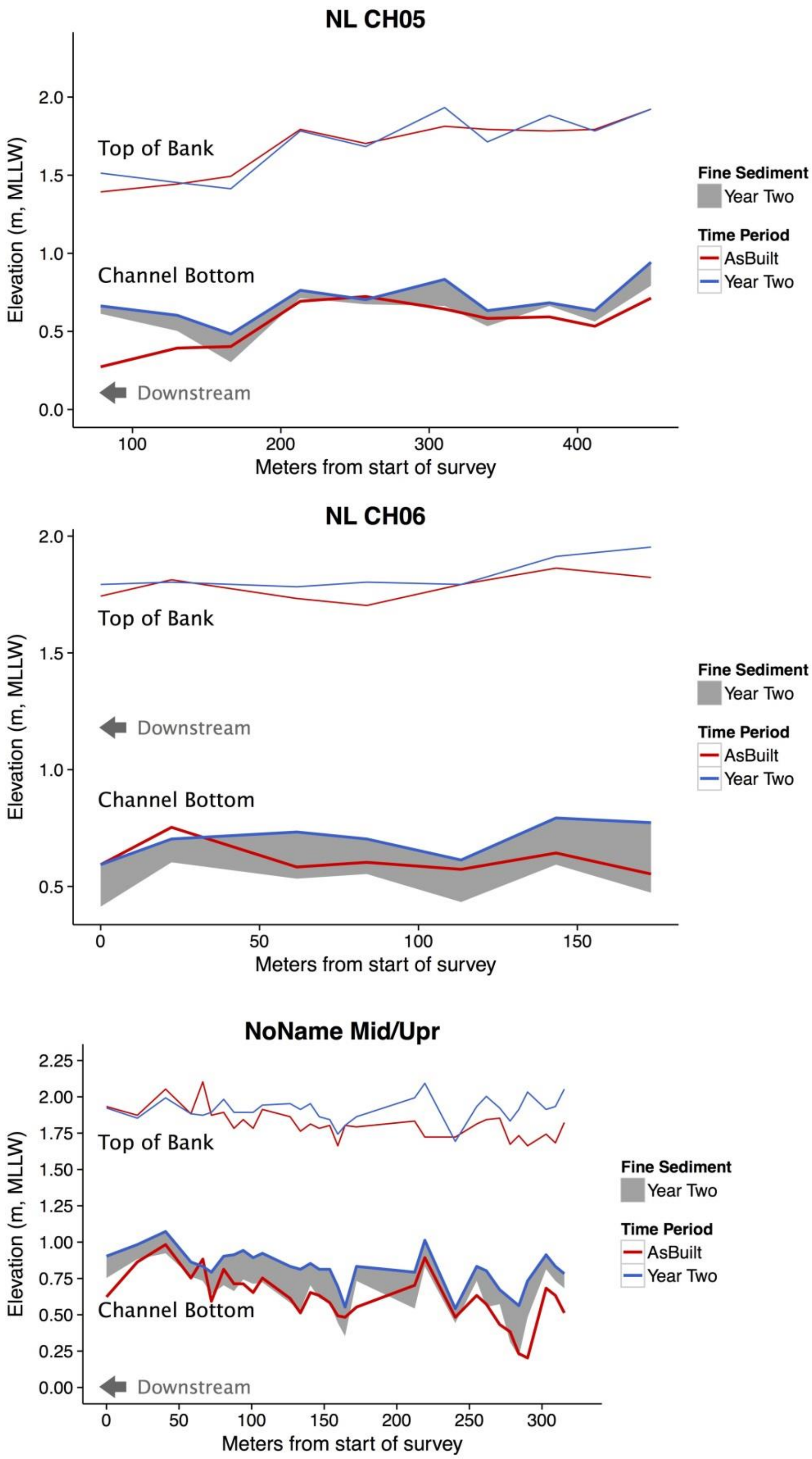


Figure A20. Pre-restoration (red, 2009-2010) and post-restoration (blue, 2013) longitudinal profiles for excavated channels at Ni-les'tun: Ch 5, Ch 6, and NoName Mid/Upper. The gray area represents the thickness of fine sediment. Top of bank represents the wetland surface.

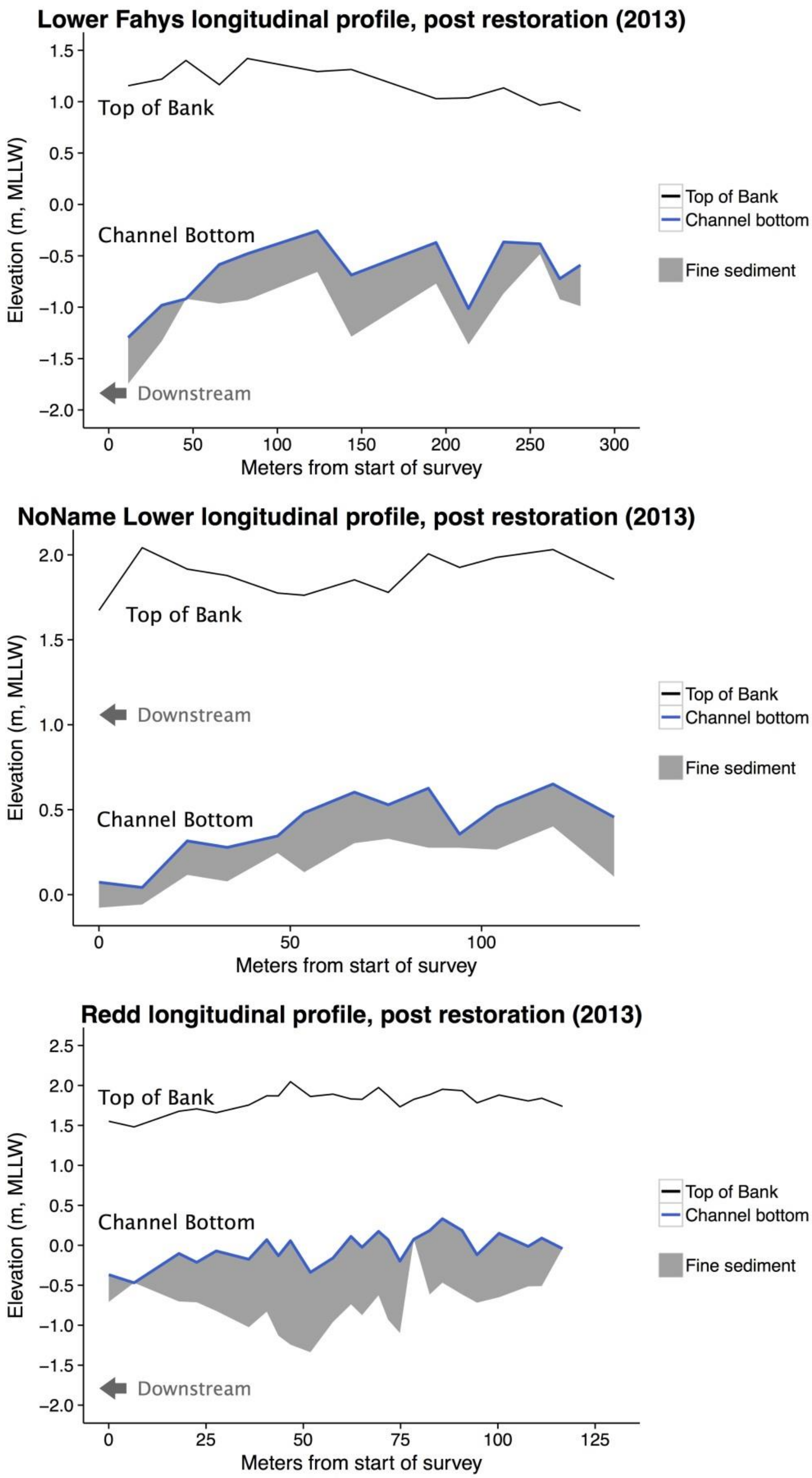


Figure A21. Post-restoration longitudinal profiles for non-excavated, pre-existing channels at Ni-les'tun: Lower Fahys, Redd, and NoName Lower. Blue line = channel bottom (top of fine sediment); gray area = fine sediment.

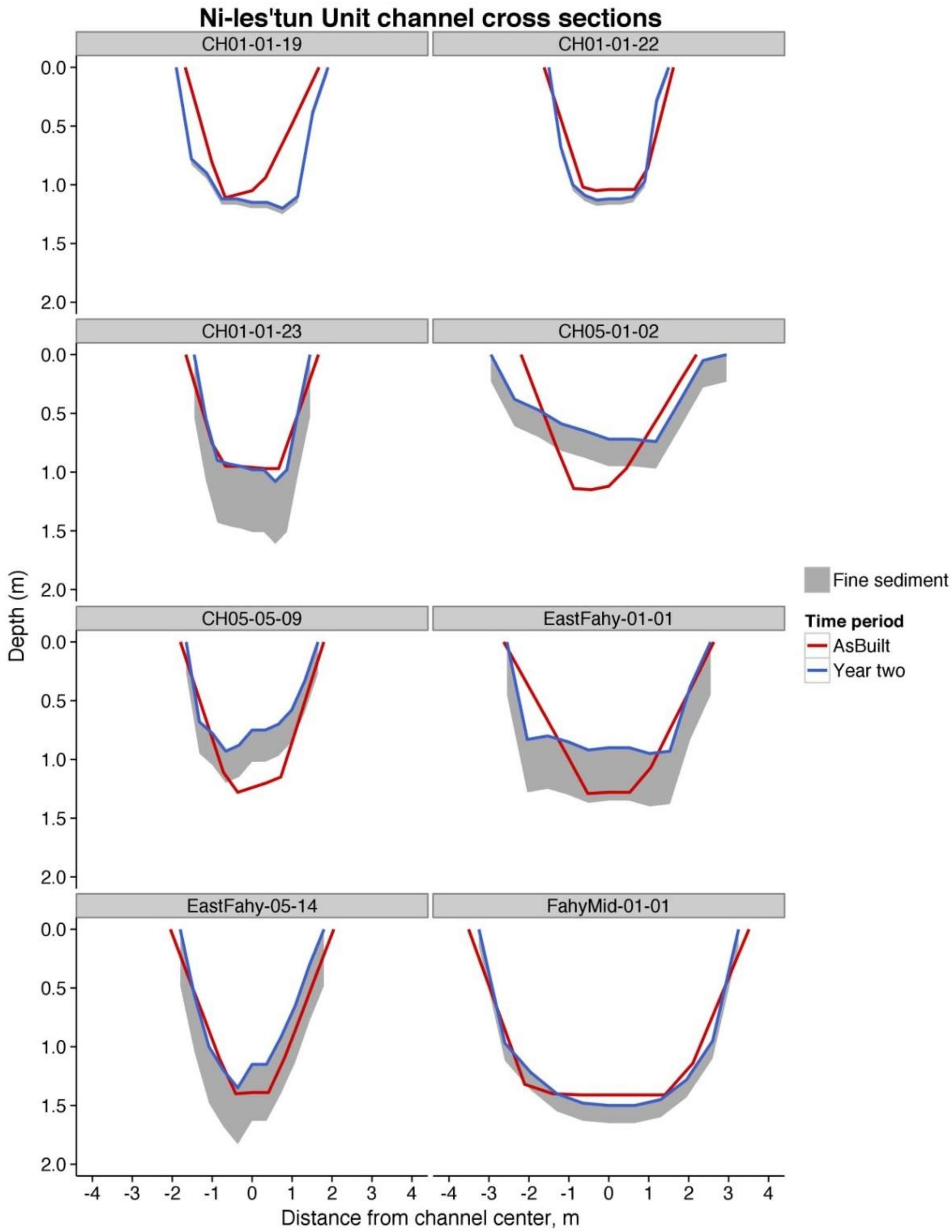


Figure A22. Pre- and post-restoration channel cross-sections in representative excavated channels at Ni-les'tun (locations: see Appendix A, Figure A4). Vertical:horizontal scaling in graphs is 4:1. Blue line = top of fine sediment; bottom of gray band = bottom of channel. Types of changes illustrated:

- Deepening, with little accumulation of fine sediment (i.e., scouring): Top row, left panel (Ch01-01-19)
- Deepening, with fine sediments accumulating: 2nd row, left panel (Ch01-01-23), and 4th row, both panels (EastFahy05-14, FahyMid-01-01)
- Broadening and/or becoming more shallow, and accumulating fines: 2nd row, right panel (Ch05-01-02), and 3rd row, right panel (East_Fahy-01-01)
- In-filling with fines (aggrading), with little change in channel shape: 3rd row, left panel (Ch05-05-09)
- Little change since excavation: Top row, right panel (Ch01-01-22)

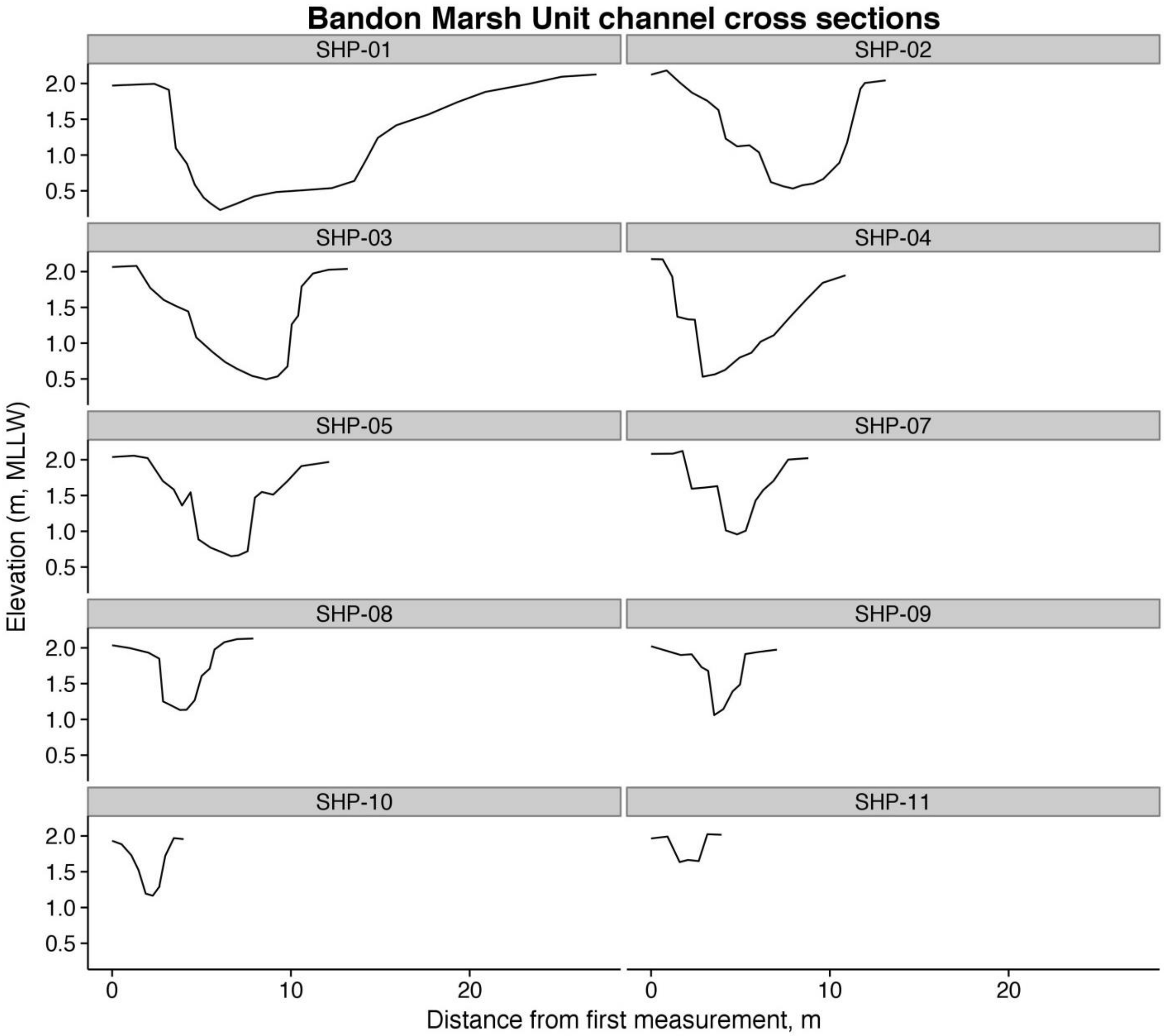


Figure A23. 2013 channel cross-sections at the Bandon Marsh Unit reference site (locations shown in Appendix A, Figure A6). Vertical:horizontal scaling in graphs is 4:1.

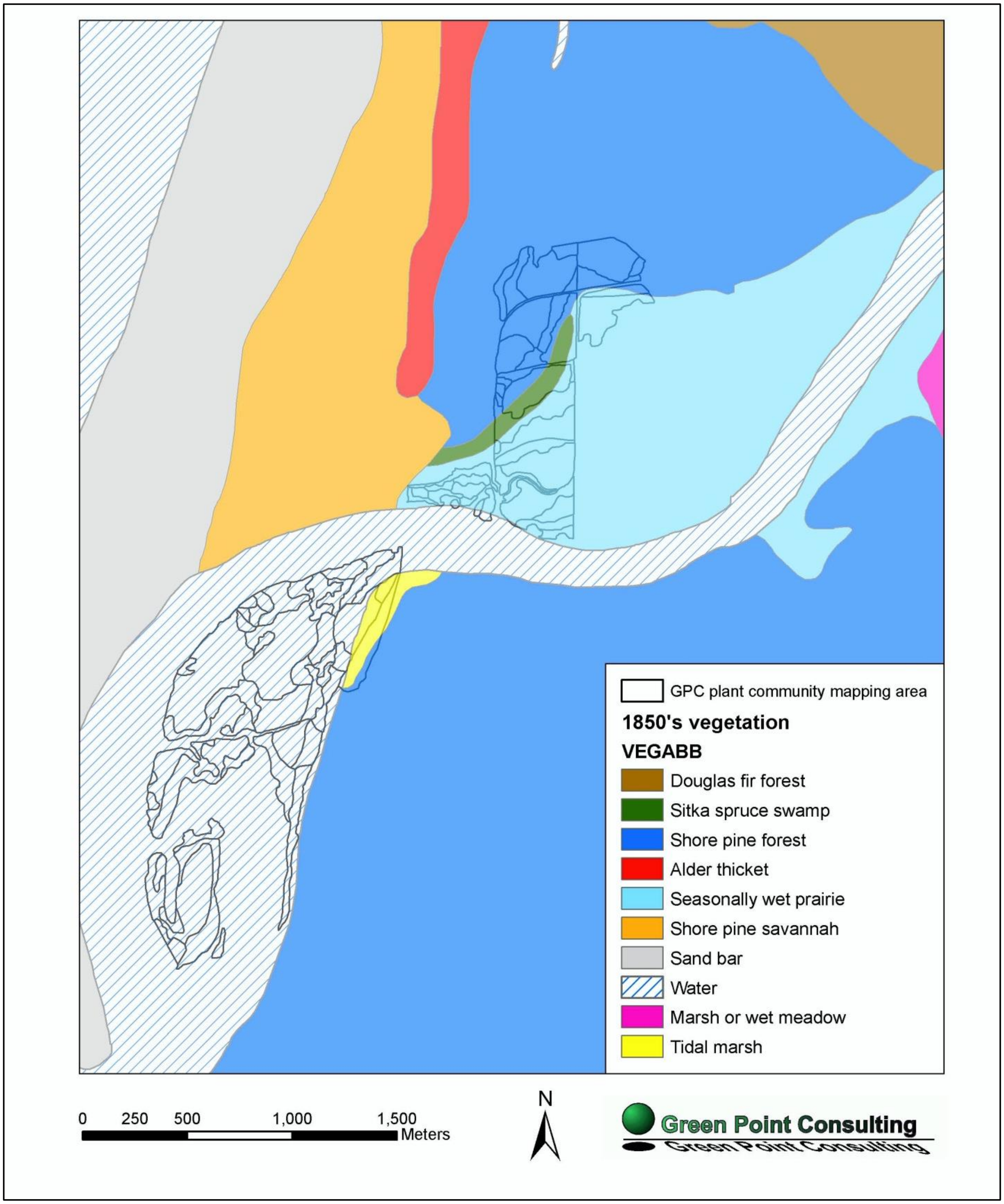


Figure A24. Historic vegetation at the Bandon Marsh NWR (from Hawes *et al*. 2008).
Figure was reproduced from the 2003 monitoring report (Brophy 2005).

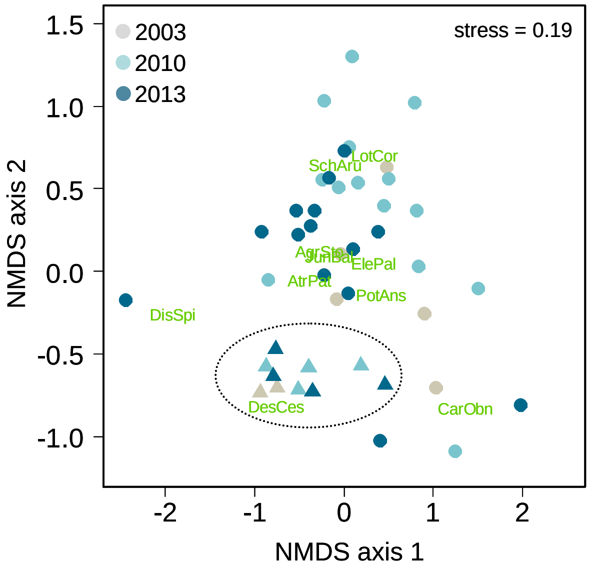


Figure A25. Non-metric multidimensional scaling (NMDS) plot for pre-restoration (2003 and 2010) and post-restoration (2013) plant communities at the Ni-les'tun restoration site and Bandon Marsh reference site. Triangles (enclosed by a dashed ellipse) are from the reference site; circles indicate transects from the restoration site. Each point represents a single transect, with percent cover averaged for all quadrats per transect. Points closer together are more similar compositionally. The centroid positions of 10 common species used in the analysis are also indicated by six letter species codes on the plot. Reference site transects, for instance, are near the DesCes centroid, indicating they have greater cover of *Deschampsia cespitosa* than most Ni-les'tun plots.

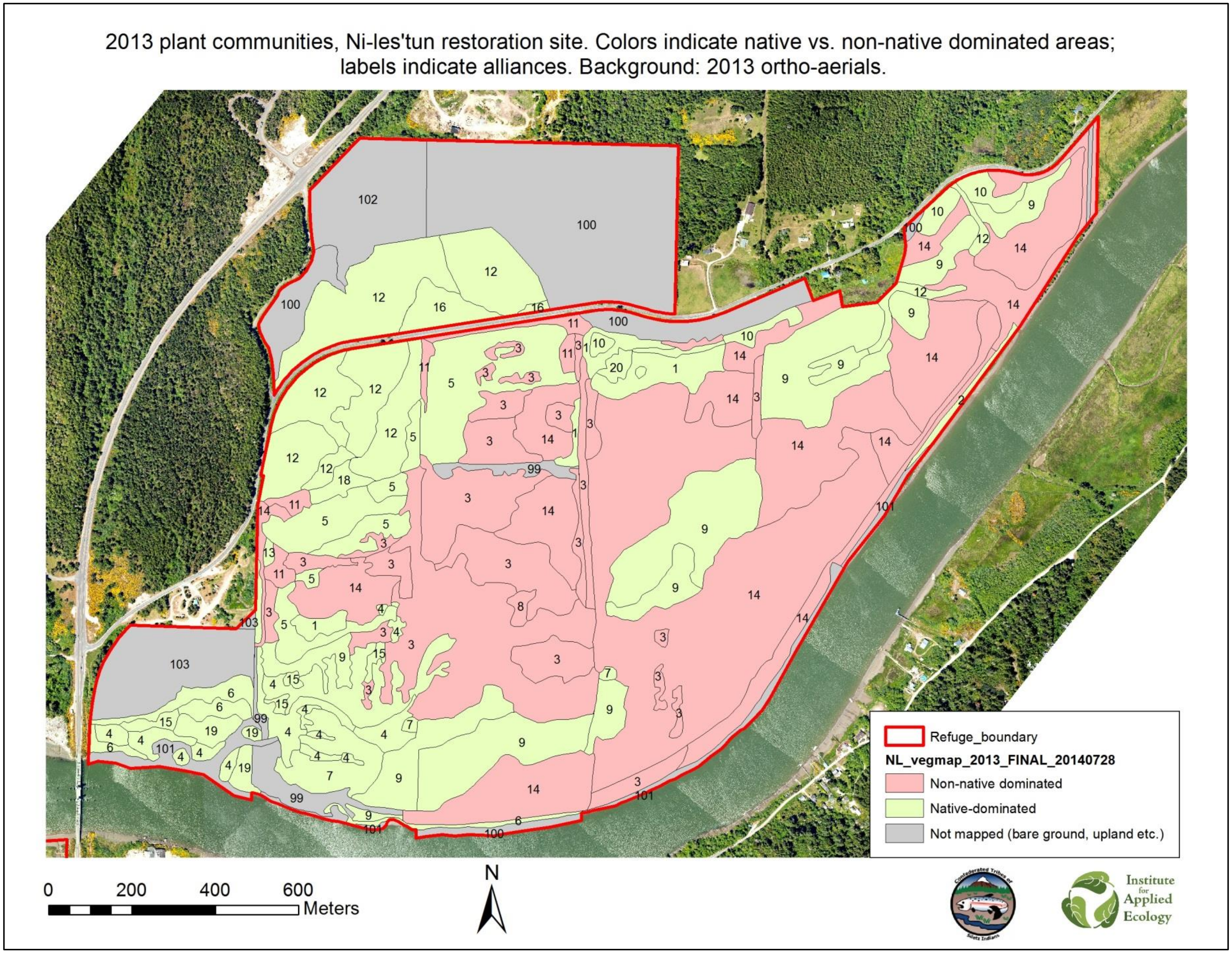


Figure A26. Plant communities at the Ni-les'tun restoration site, July 2013, showing areas dominated by native vs. non-native species. Unmapped areas are outside restoration project area. Labels show alliance numbers (see Appendix B, Table B21).

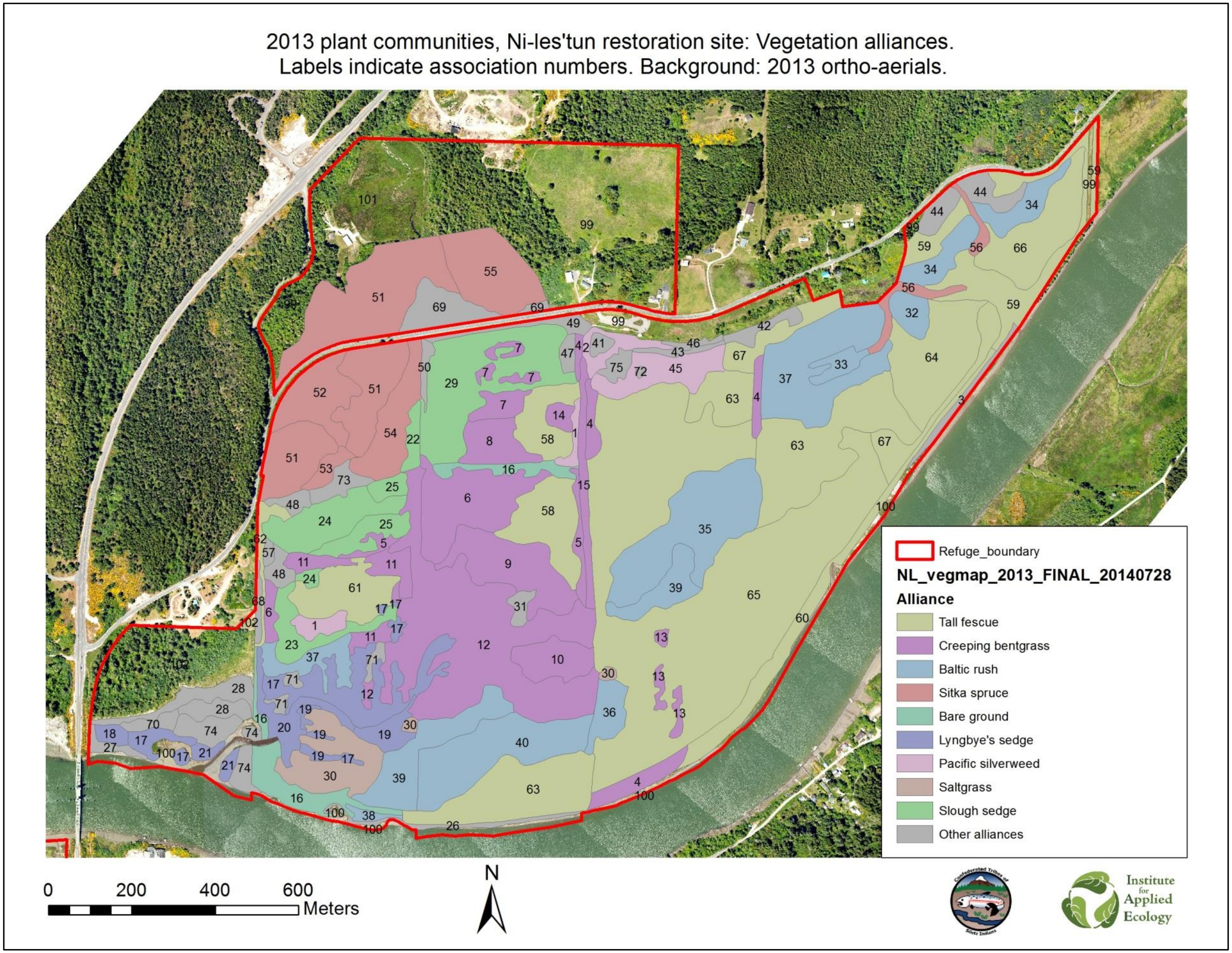


Figure A27. Major vegetation types (alliances) at the Ni-les'tun restoration site, July 2013. Labels show association numbers (map units) (see Appendix B, Tables B22 and B23).

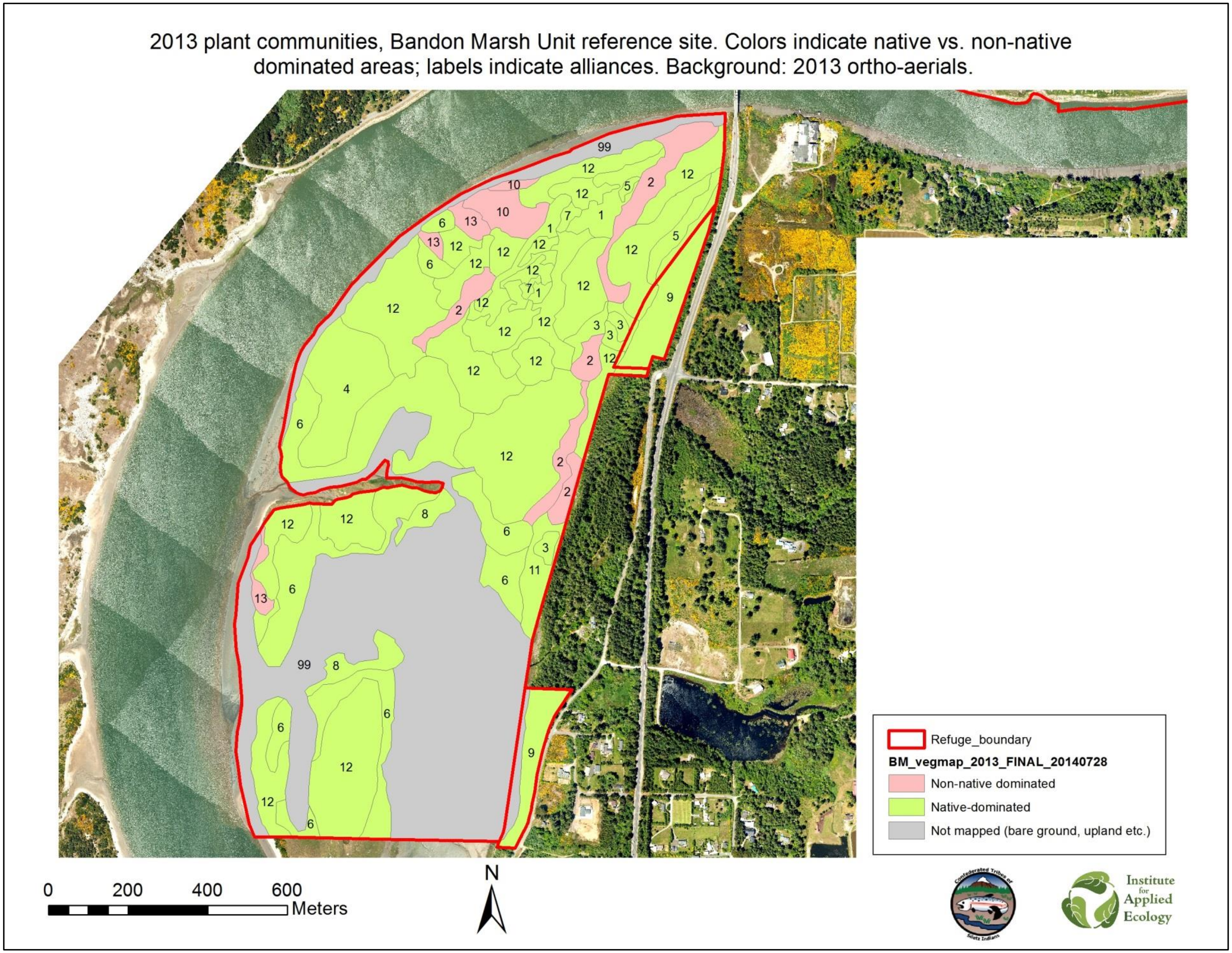


Figure A28. Plant communities at the Bandon Marsh Unit reference site, July 2013, showing areas dominated by native vs. non-native species. Labels show alliance numbers (see Appendix B, Table B25).

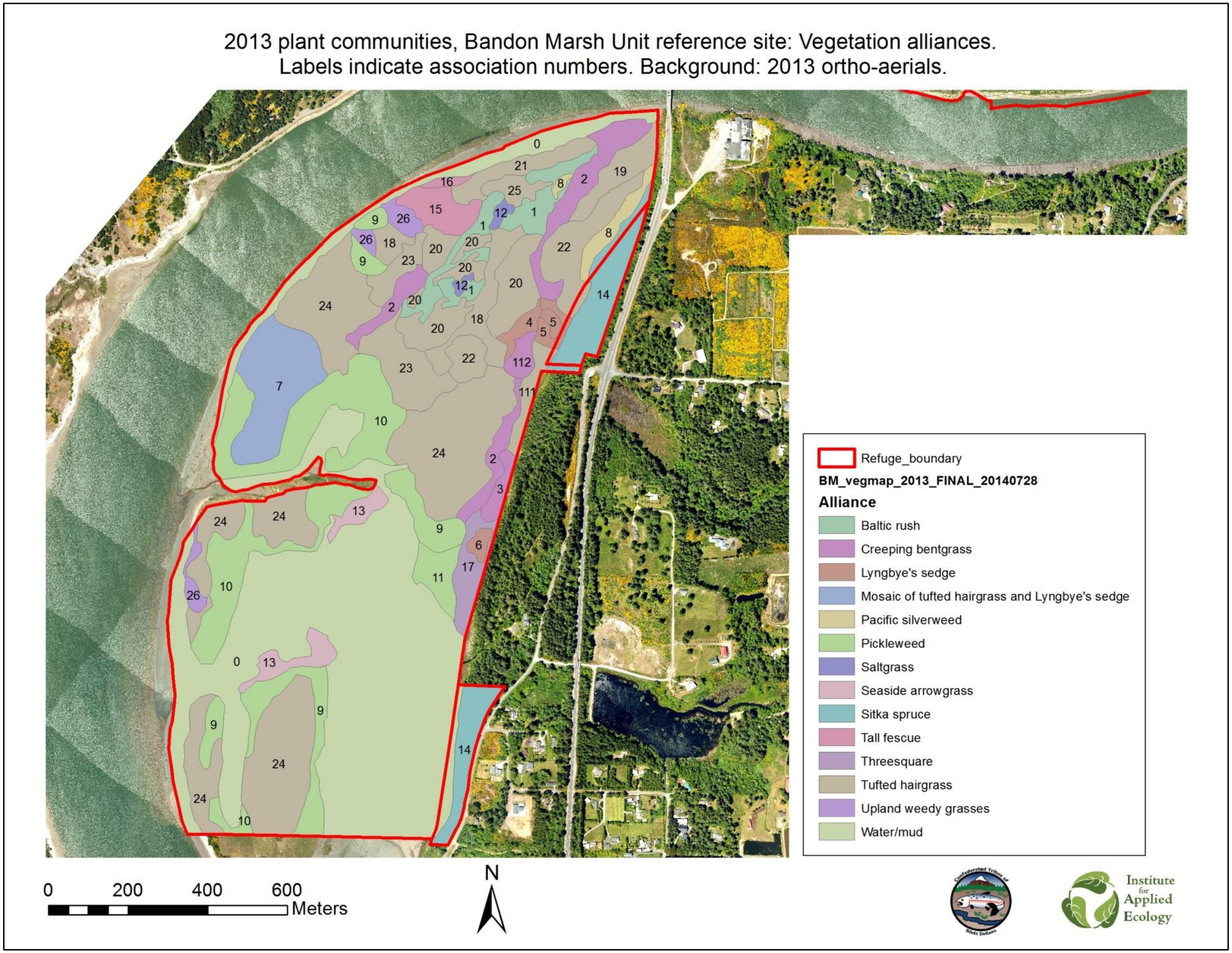


Figure A29. Major vegetation types (alliances) at the Bandon Marsh Unit reference site, July 2013. Labels show association numbers (map units) (see Appendix B, Table B26).

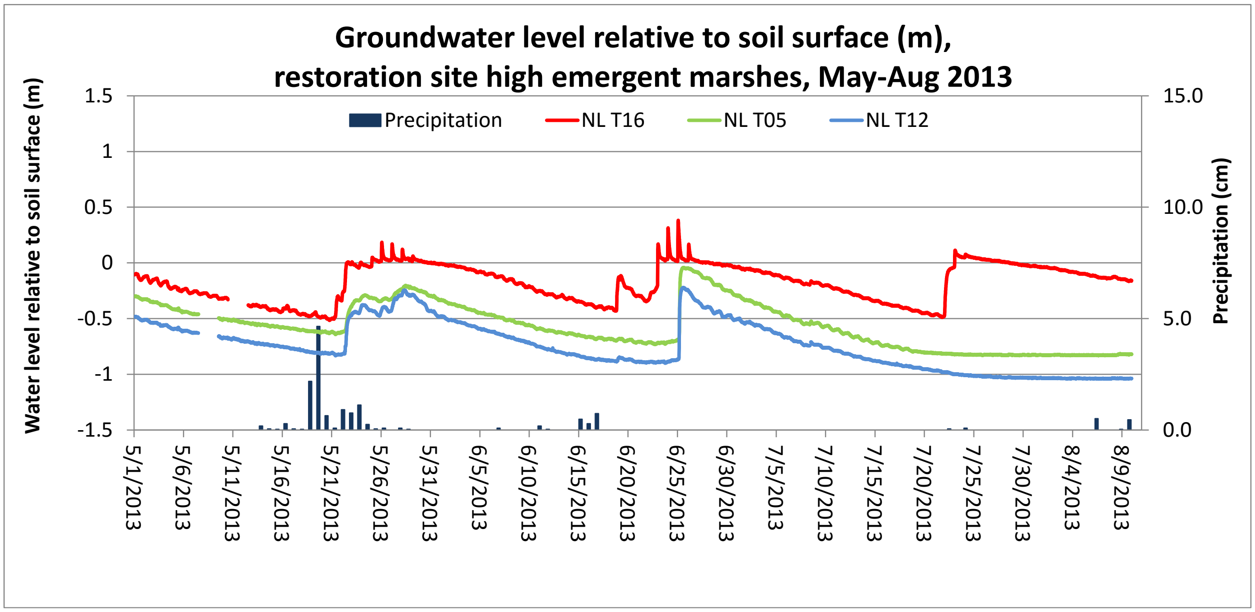


Figure A30. Spring and summer groundwater levels after restoration (relative to the soil surface) for three Ni-les'tun transects representing an elevation gradient from 10 cm below MHHW (NL T16, elevation 2.07 m NAVD88) to 13 cm above MHHW (NL T12, 2.30 m NAVD 88). All show a "spring tide reset" pattern but the higher two dry out in summer.

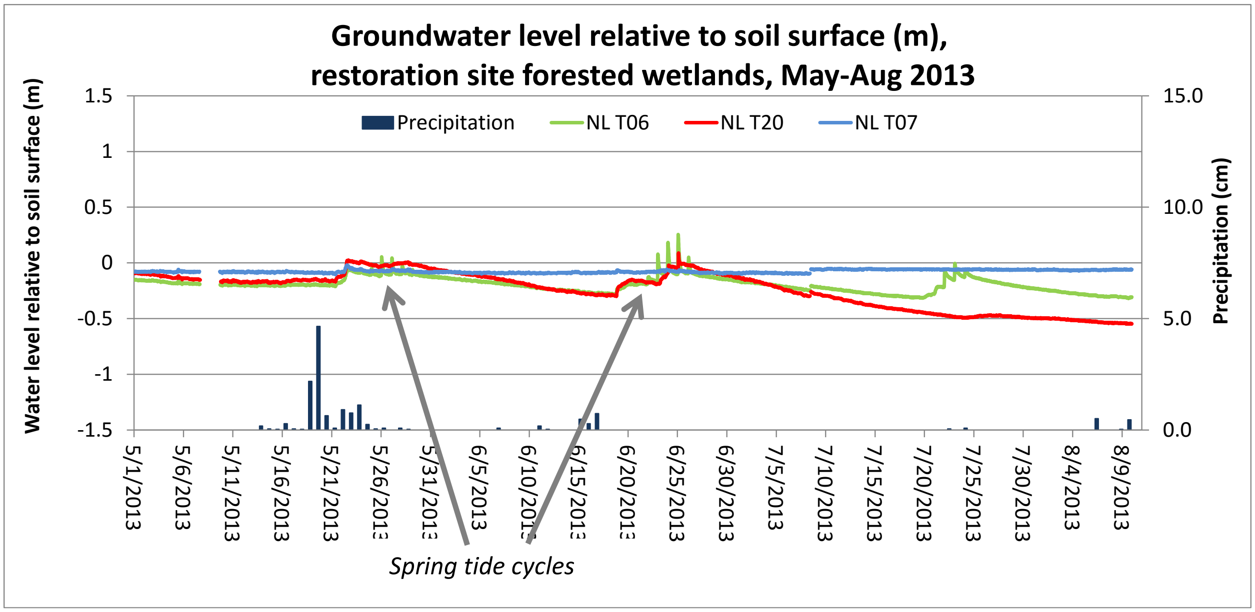


Figure A31. Spring and summer groundwater levels after restoration (relative to the soil surface) at Ni-les'tun forested transects. NL T7 is non-tidal.

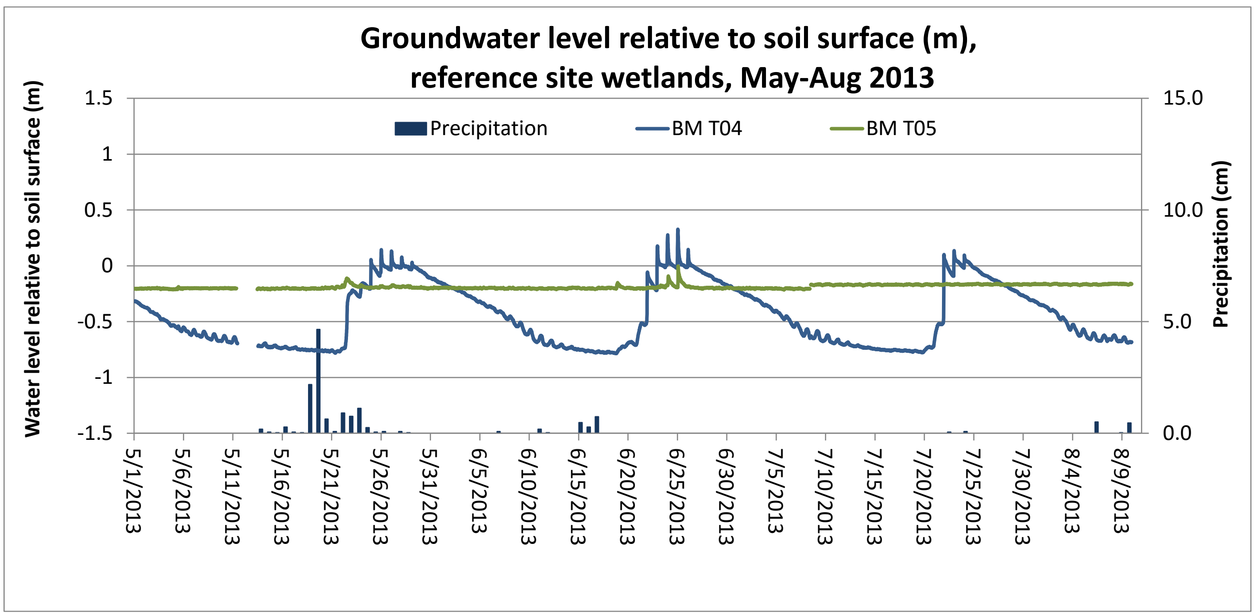


Figure A32. Spring and summer groundwater levels at emergent marsh transect BM T4 and forested wetland transect BM T5, Bandon Marsh Unit reference site.

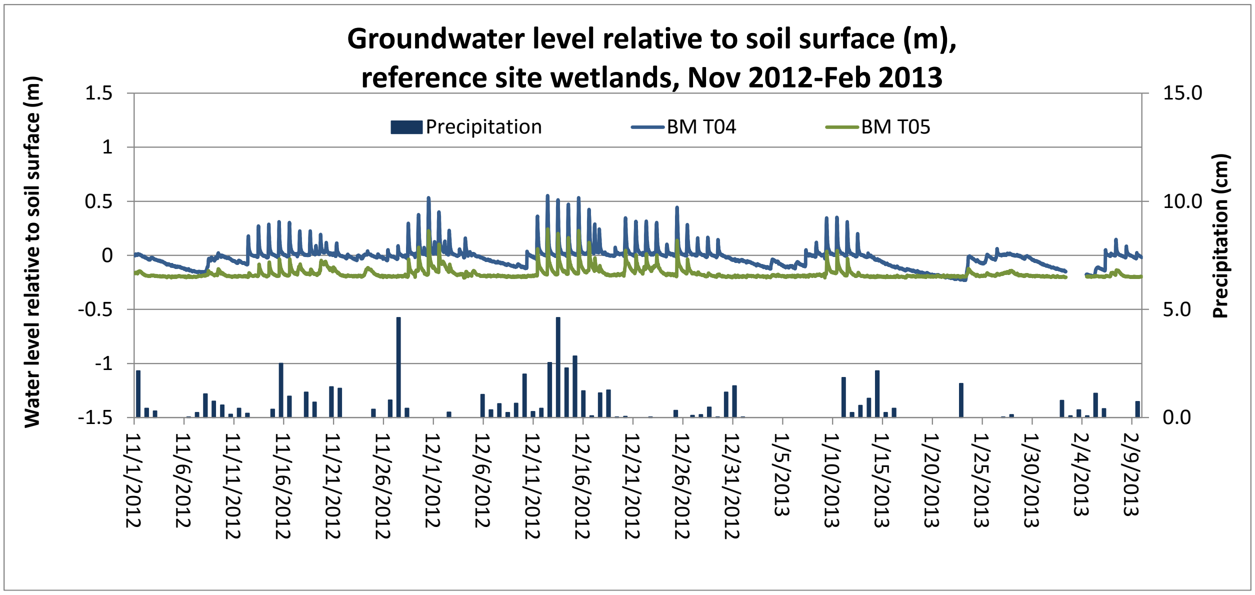


Figure A33. Winter groundwater levels at emergent marsh transect BM T4 and forested wetland transect BM T5, Bandon Marsh Unit reference site.

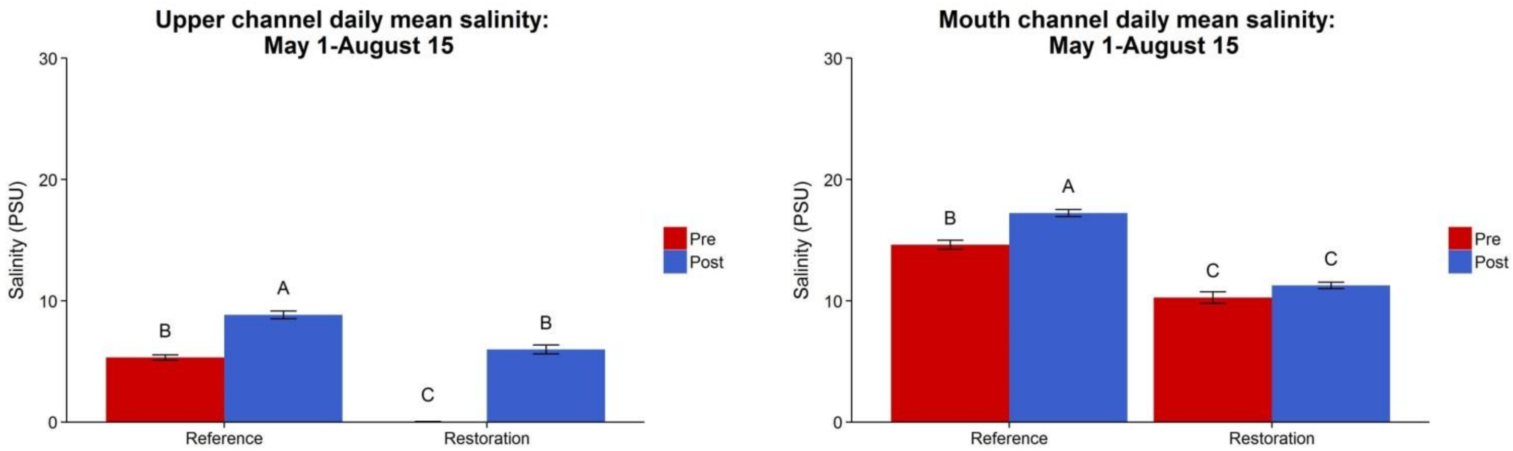


Figure A34. Daily mean salinity at upper channel and channel mouth locations at the restoration and reference sites, pre-restoration (2011) *versus* post-restoration (2013), May 1 - August 15 period. Bars with a letter in common are not significantly different (p > 0.05).

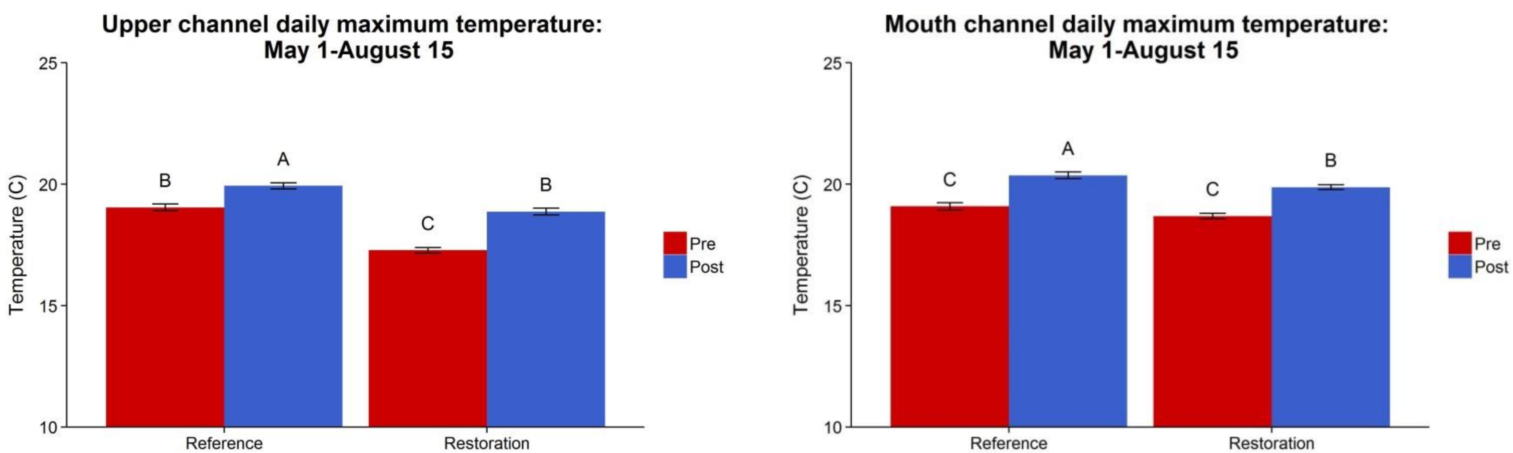


Figure A35. Daily maximum temperature at upper channel and channel mouth locations at the restoration and reference sites, pre-restoration (2011) *versus* post-restoration (2013), May 1 - August 15 period. Bars with a letter in common are not significantly different (p > 0.05).

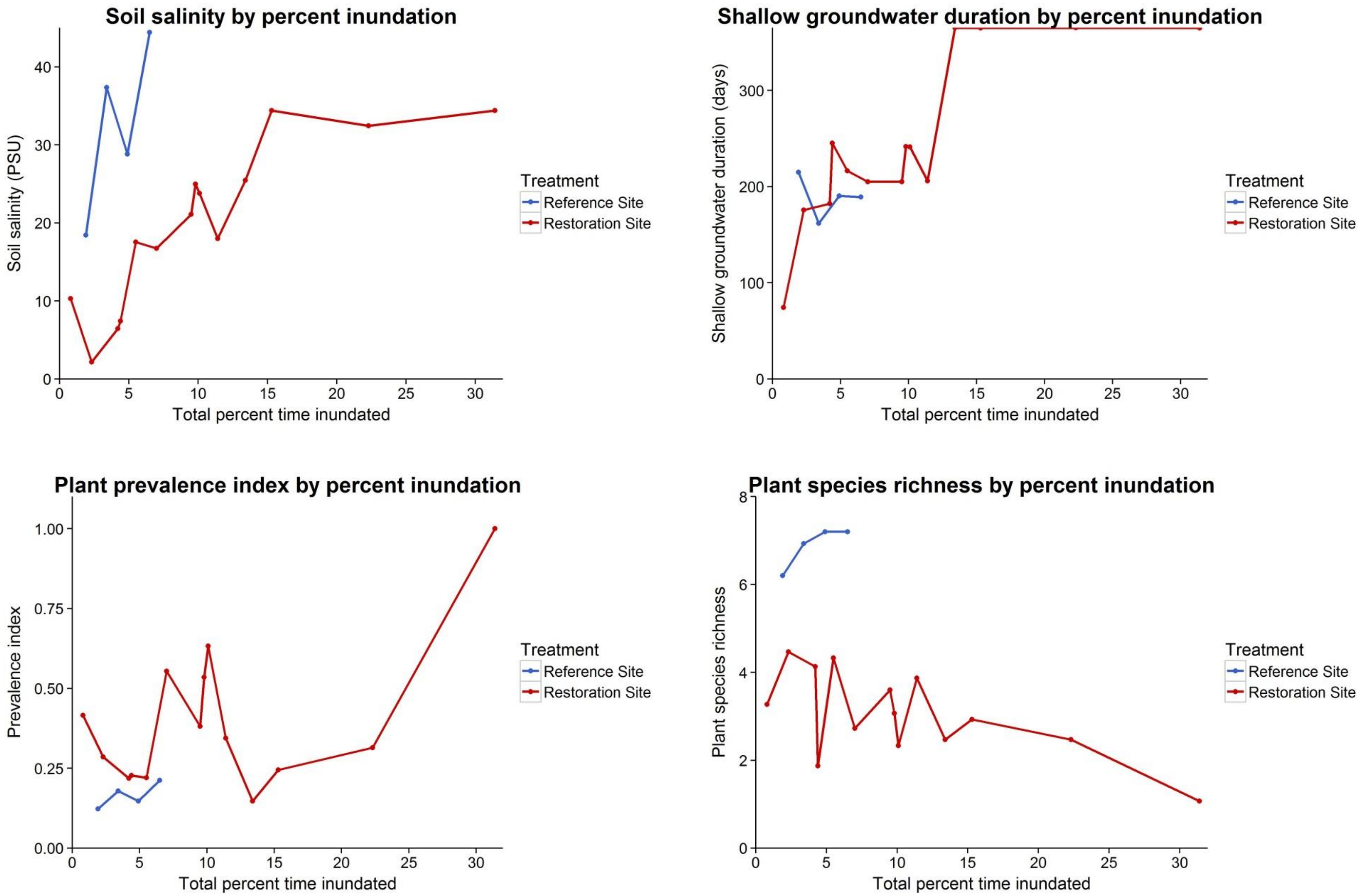


Figure A36. Post-restoration percent inundation *versus* physical and biological site characteristics at 14 emergent wetland transects at the Ni-les'tun restoration site and 4 emergent transects at the Bandon Marsh Unit reference site. Only significant relationships are graphed.

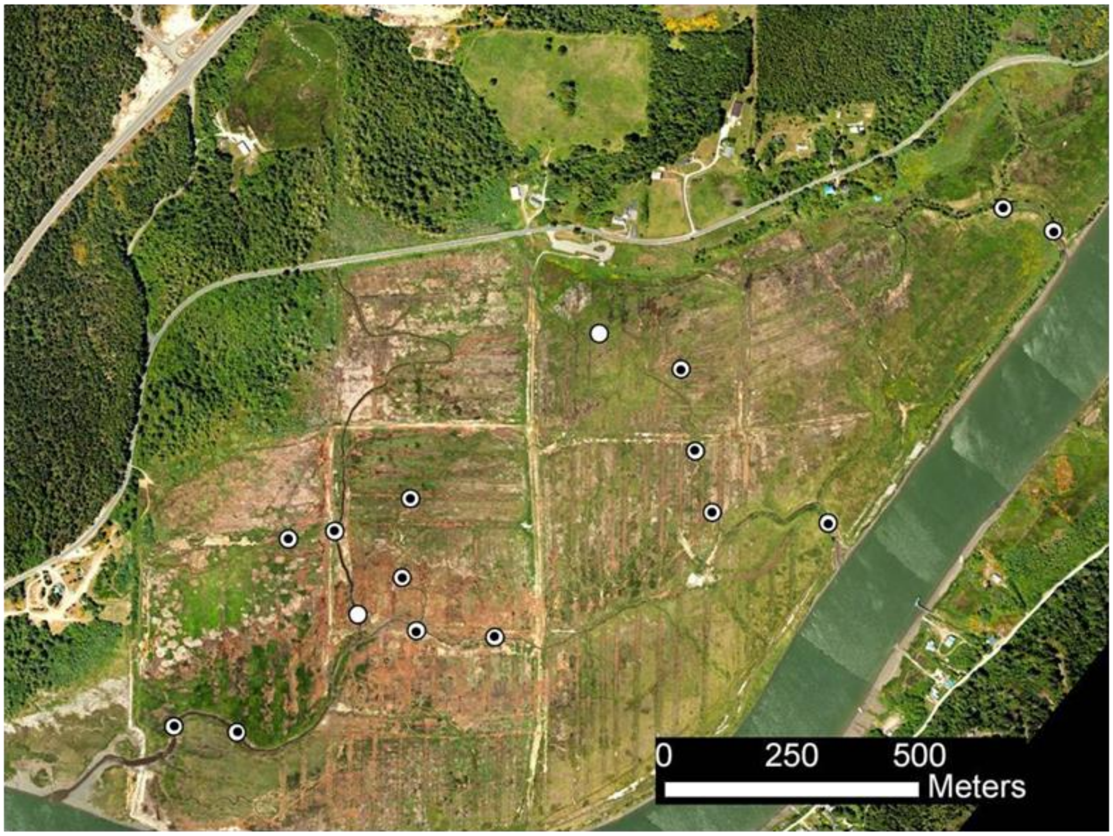


Figure A37. Locations of in-stream large wood placements (white and black-center dots; each dot is at the center of a 100m wood placement reach). Channel cross-sections monitoring occurred in the 100m reaches surrounding the dots with black centers. For a map showing the extent of cross-section monitoring, see Figure A4.

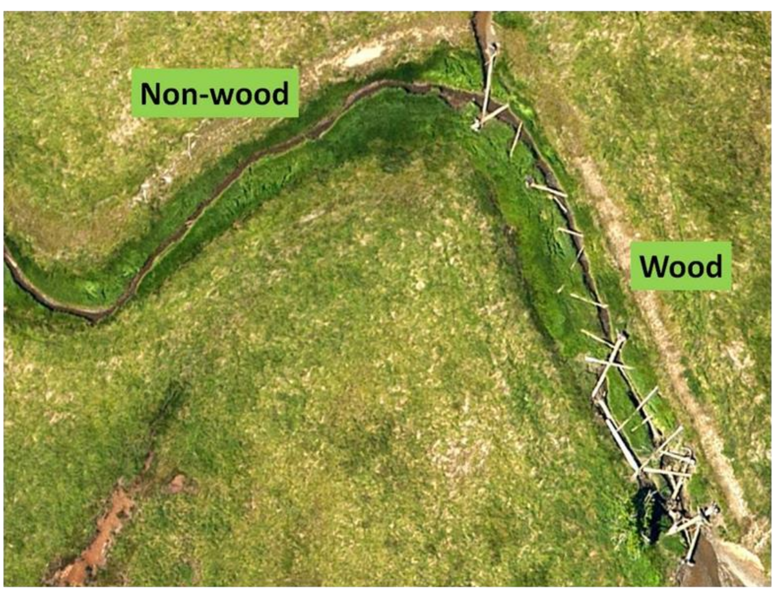
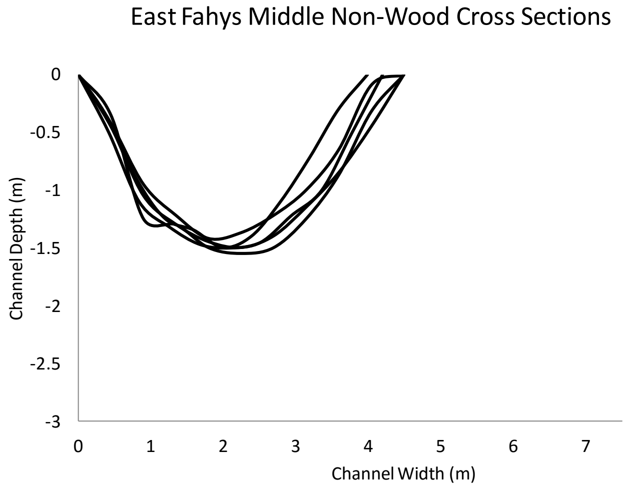


Figure A38. Close-up aerial view of restored in-stream wood habitat (right side of photo) and non-wood habitat in lower NoName channel.



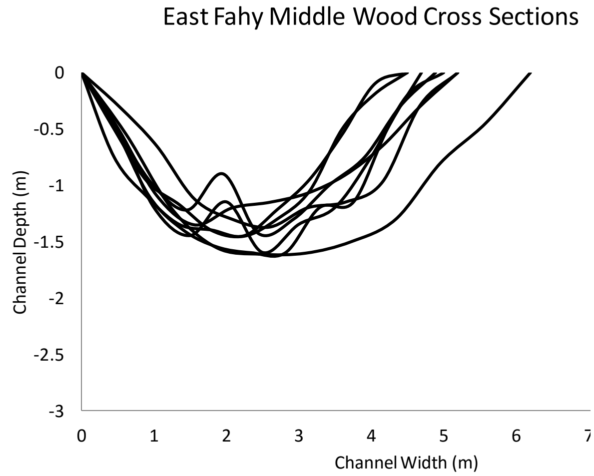


Figure A39. East Fahys Middle cross sections from non-wood reaches (left) and wood reaches (right).

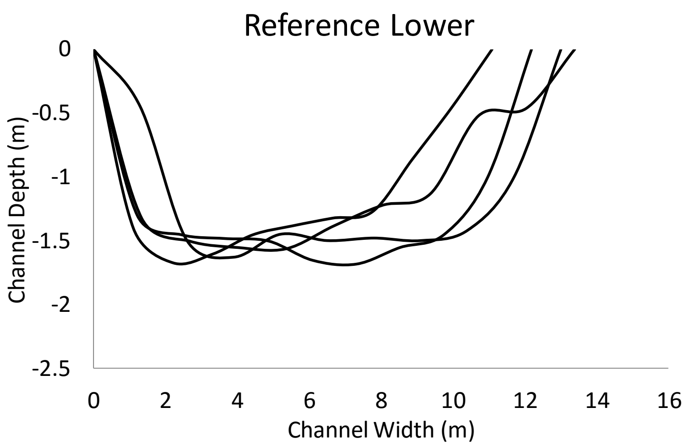


Figure A40. Reference (Shipwreck) cross sections from the single (non-wood) reach.

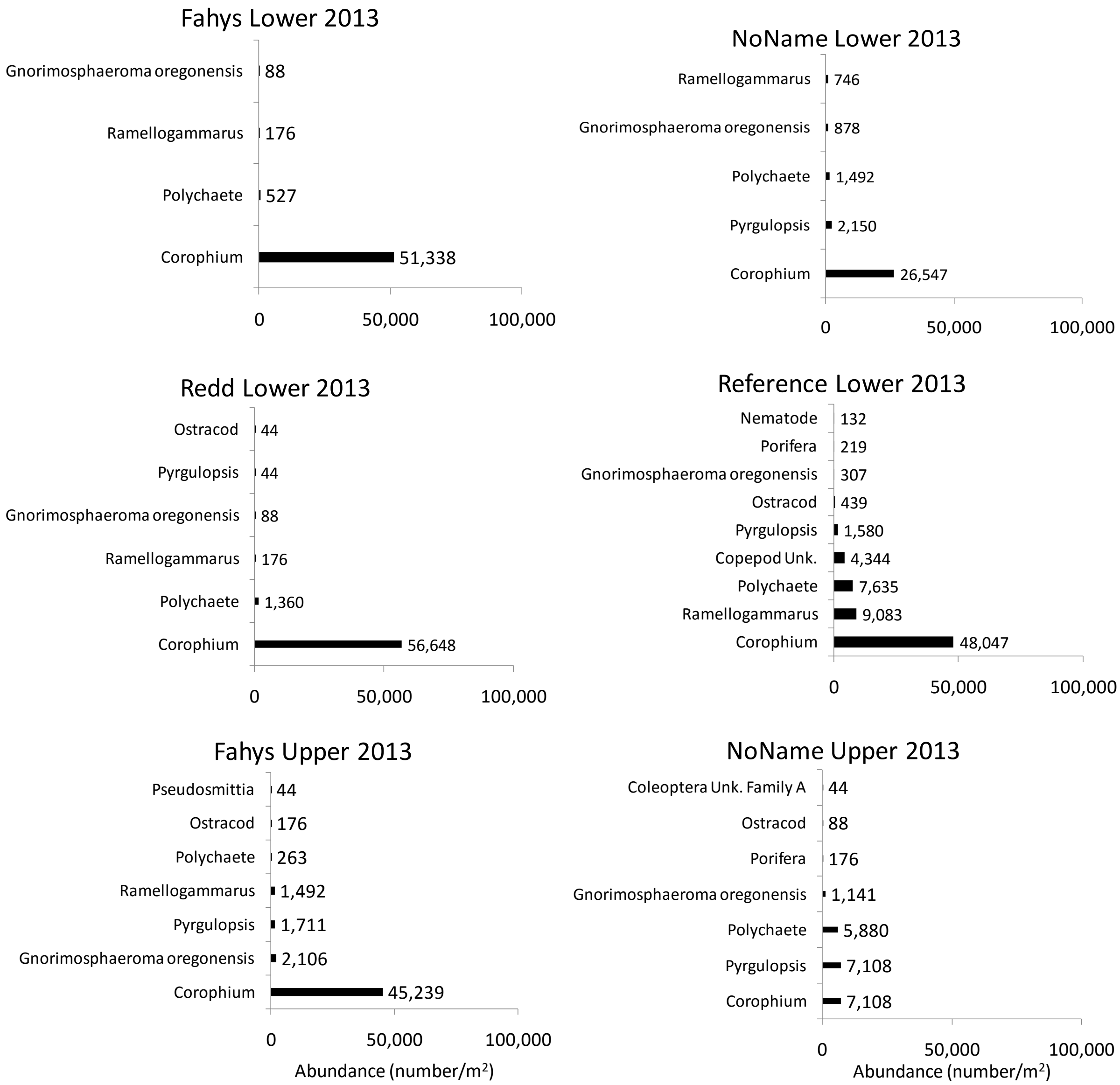


Figure A41. September 2013 benthic macroinvertebrate abundance from Fahys, NoName, Redd and the reference sub-basins.

Appendix B. Additional tables

Table B1. Summary of sampling and analysis methods for monitoring at Ni-les'tun during 2012-2013. "Frequency/timing" shows years for which funding has been obtained. At least 5 years of post-restoration monitoring are recommended; funding is being sought for this work.

Para-meter #	**Parameter**	**Method/equipment**	**Frequency / timing**	**Sample locations***	**Protocol citation**
1	Tidal hydrology	Electronic water level logger	15min interval Duration: 1 yr in 2010-11, 2012-2013; 1mo in summer 2015 & winter 2015	Fahys Cr, Ch 7, and adjacent Coquille River	Roegner *et al.* 2008
2	Channel morphology	Traditional and RTK-GPS survey and leveling; airphoto analysis	1x/yr in 2011, 2013	Stratified random and strategic sampling near permanent plots and wood placements; airphoto analysis of entire site	Roegner *et al.* 2008
3a	Plant community composition – emergent	% cover by species	1x/yr in 2010, 2013, 2015	18 permanent plots (14 restor., 4 ref.) approx. 30X150ft; random sampling within plots	Roegner *et al.* 2008
3b	Plant community composition – forested and scrub-shrub	Stem density (quadrat/transect); diameter tape	1x/yr in 2011, 2013	4 permanent plots (3 restor., 1 ref.) approx 30 by 150ft; random sampling in plots	Roegner *et al.* 2008
3c	Plant community extent	Area of each plant community	1x/yr in 2010, 2013, 2015	Entire restoration site and reference site	Roegner *et al.* 2008
4	Groundwater depth	Electronic water level logger	15min interval, April-Nov. 2010, Sept. 2012 – Sept. 2013	22 shallow observation wells in permanent plots (17 restor., 5 ref.)	Sprecher 2000; Brophy 2009
5	Soil organic matter, salinity and texture	%OM by loss on ignition; pH and salinity (conductivity) by probe.	1x/yr in late summer 2010, 2013	10 soil cores from root zone (upper 30cm) bulked to one sample in each of 22 permanent plots (17 restor., 5 ref.)	Dane and Topp 2002; Sparks 1996
6	Water temperature and salinity	Continuous temperature/salinity datalogger	30min. interval; May-Nov. 2011, Feb.-Sept. 2013	17 stations in tidal channels near permanent plots (10 restoration, 7 reference)	Roegner *et al*. 2008; OPSW 2001
7a	Low tide salmonid density and distribution Peak Use (June)	Pole seine	June 2010, 2013	9 standard sites for each of Fahys and NoName, 6 in Redd; 13 large wood habitat; 9 reference	Roegner *et al*. 2008

Para-meter #	Parameter	Method/equipment	Frequency / timing	Sample locations*	Protocol citation
7b	Low tide salmonid distributions (non-peak months)	Pole seine	May, July, Aug Sept 2010, 2013	9 standard sites for each of Fahys and NoName, 6 in Redd; 13 large wood habitat; 9 reference	Roegner *et al.* 2008
8	Salmonid tidal migration Peak Use (May and June)	Underwater videography	June 2010, 2013	Mouth Fahys, Mouth NoName, Mouth Redd, Mouth Reference, mainstem at Fahys, mainstem at Reference, mainstem below Redd; Mouth Fahys and Reference	Van de Wetering *et al.* 2007.
9	In-stream habitat	Tape measure, measuring rod, GPS, Existing RTK monuments	Winter 2010, 2013	Cross sections every two channel widths across 100m study sections: Fahys 12 non-wood reaches, 15 wood reaches; NoName 3 wood, 3 non-wood; Redd 1 wood 1 non-wood	Roegner *et al.* 2008
10a	Wood habitat use	Pole seine	May, June, July Aug 2013	13 of the 25 standard sites	Roegner *et al.* 2008
10b	Wood habitat use	Underwater videography	June 2010, 2013	Lower Fahys, lower NoName, and lower Redd	Van de Wetering 2007
11	Macroinverte-brate density and composition	Channel core samples	Summer 2010, 2013	4 habitat zones in the reference and lower Fahys (matching pre-treatment samples) – 20 cores	Gray 2005

* Sampling is conducted at Ni-les'tun restoration site and Bandon Marsh Unit reference site, unless otherwise described

Table B2. Elevations of study transects at Ni-les’tun and Bandon Marsh Units, from 2010 RTK-GPS and total station survey. (2013 RTK-GPS survey did not show changes in elevation.) Each elevation is the average of several surveyed points; the number of points is listed in the far right column. Elevations are relative to NAVD88, GEOID03.

Site	**Transect**	**Transect elevation (m)**	**Transect elevation (ft)**	**Transect elevation relative to MHHW (m)**	**Transect elevation relative to MHHW (ft)**	**Vegetation type**	**Habitat description**	**# of survey points**
Ni-les'tun restoration site	NL T2	1.69	5.54	-0.484	-1.593	emergent	diked pasture	14
Ni-les'tun restoration site	NL T4	1.89	6.20	-0.284	-0.933	emergent	diked pasture	8
Ni-les'tun restoration site	NL T5	2.18	7.15	0.006	0.017	emergent	diked pasture	8
Ni-les'tun restoration site	NL T6	2.23	7.32	0.056	0.187	forested	forested wetland	1
Ni-les'tun restoration site	NL T7	2.89	9.48	0.716	2.347	forested	forested wetland	1
Ni-les'tun restoration site	NL T9	2.12	6.96	-0.054	-0.173	emergent	diked pasture	16
Ni-les'tun restoration site	NL T10	1.94	6.37	-0.234	-0.763	emergent	diked pasture	14
Ni-les'tun restoration site	NL T11	1.98	6.50	-0.194	-0.633	emergent	diked pasture	13
Ni-les'tun restoration site	NL T12	2.30	7.55	0.126	0.417	emergent	diked pasture	14
Ni-les'tun restoration site	NL T13	1.84	6.04	-0.334	-1.093	emergent	diked pasture	13
Ni-les'tun restoration site	NL T14	1.98	6.50	-0.194	-0.633	emergent	diked pasture	12
Ni-les'tun restoration site	NL T15	1.99	6.53	-0.184	-0.603	emergent	diked pasture	16
Ni-les'tun restoration site	NL T16	2.07	6.79	-0.104	-0.343	emergent	diked pasture	14
Ni-les'tun restoration site	NL T17	2.46	8.07	0.286	0.937	emergent	diked pasture	15
Ni-les'tun restoration site	NL T18	1.50	4.92	-0.674	-2.213	emergent	diked pasture	14
Ni-les'tun restoration site	NL T19	2.17	7.12	-0.004	-0.013	emergent	diked pasture	17
Ni-les'tun restoration site	NL T20	2.31	7.58	0.136	0.447	forested	forested wetland	2
Bandon Marsh Unit ref. site	BM T1	2.09	6.86	-0.084	-0.273	emergent	high tidal marsh	7
Bandon Marsh Unit ref. site	BM T2	2.15	7.05	-0.024	-0.083	emergent	high tidal marsh	11
Bandon Marsh Unit ref. site	BM T3	2.34	7.68	0.166	0.547	emergent	high tidal marsh	10
Bandon Marsh Unit ref. site	BM T4	2.21	7.25	0.036	0.117	emergent	high tidal marsh	10
Bandon Marsh Unit ref. site	BM T5	2.50	8.20	0.326	1.067	forested	forested tidal wetland	2

Table B3. Elevations of monitoring instruments at Ni-les'tun restoration site and Bandon Marsh Unit reference site during 2012-2013, from RTK-GPS and total station survey. Locations of loggers are keyed to map codes shown in Appendix A, Figures A2 and A3. Elevations are relative to NAVD88, GEOID03.

Site/location	Instrument type	Map code	Location description	Sensor elevation (m NAVD88)	Sensor elevation (ft NAVD88)
Coquille River	tide gauge	CoquilleR_TG2	Coquille River, opposite mouth of Fahys Cr	-1.51	-4.95
Ni-les'tun	tide gauge	Lower Fahys TG2	Lower Fahys Creek, inside restoration site	0.30	0.98
Ni-les'tun	tide gauge	NL_Ch7_TG	Mid-Fahys Creek, inside restoration site	0.72	2.36
Coquille River	salinity/temp. logger	COQ R 8234	Coquille River outside Fahys Cr tide gates	-0.80	-2.62
Ni-les'tun	salinity/temp. logger	FAHY MTH 8239	Fahys Creek mouth, inside tide gate	0.37	1.21
Ni-les'tun	salinity/temp. logger	FAHY MID 8230	Fahys Creek, midway to N Bank Road	0.47	1.54
Ni-les'tun	salinity/temp. logger	FAHY RD 8241	Fahys Creek at N Bank Road	1.00	3.28
Ni-les'tun	salinity/temp. logger	CH5 LWR 8499	Center of restoration site, Channel 5, lower	0.66	2.17
Ni-les'tun	salinity/temp. logger	CH5 UPR 8500	Center of restoration site, Channel 5, upper	0.99	3.25
Ni-les'tun	salinity/temp. logger	CH7 8498	South-center of restoration site, Channel 7	0.72	2.36
Ni-les'tun	salinity/temp. logger	CH7 CHK 8502*	South-center of restoration site, Channel 7	0.72	2.36
Ni-les'tun	salinity/temp. logger	NONAM MTH 8231	NoName channel mouth, inside tide gate	0.51	1.67
Ni-les'tun	salinity/temp. logger	NONAM MID 8237	NoName channel, middle	1.03	3.38
Ni-les'tun	salinity/temp. logger	NONAM UPR 8228	NoName channel, upper	1.06	3.48
Ni-les'tun	salinity/temp. logger	REDD MID 8240	Redd Creek, at tributary junction	0.49	1.61
Bandon Marsh Unit	salinity/temp. logger	SHPWRK A 8238	Shipwreck channel at mouth	0.30	0.98
Bandon Marsh Unit	salinity/temp. logger	SHPWRK B 8229	Shipwreck channel (upper)	1.07	3.51
Bandon Marsh Unit	salinity/temp. logger	BM UNK A 8235	Unnamed tidal channel, upper	1.03	3.38
Bandon Marsh Unit	salinity/temp. logger	BM UNK B 8232	Unnamed tidal channel, middle	0.91	2.99
Bandon Marsh Unit	salinity/temp. logger	BM UNK C 8233	Unnamed tidal channel, lower	0.88	2.89

* CH7 CHK 8502 was a backup salinity/temperature logger installed at the same elevation and location as CH7 8498

Table B4. Summary of BACI ANOVA results for mean daily maximum water level (pre-restoration, 2009) or mean daily maximum tide height (post-restoration, 2013) at Ni-les'tun restoration site and Bandon Marsh Unit reference site, during January through September.

	F-value (df)	**p-value**
Site	11.35 (1)	<0.0001
Year	0.79 (1)	0.38
Site*Year	743.33 (1)	<0.0001

Table B5. Means comparison from BACI ANOVA for mean daily maximum water level (pre-restoration, 2009) or mean daily maximum tide height (post-restoration, 2013) at Ni-les'tun restoration site and Bandon Marsh Unit reference site during January-September. Means with a letter in common are not significantly different (p > 0.05).

Site	**Year**	**Mean maximum tide height (m NAVD88)**	**SE**	**Group**
Restoration	Pre-restoration (2009)	1.29	0.01	c
Reference	Pre-restoration (2009)	2.14	0.01	ab
Restoration	Post-restoration (2013)	2.09	0.01	b
Reference	Post-restoration (2013)	2.15	0.01	a

Table B6. Summary of ANOVA results for change in mean channel depth from pre-restoration (2010) to post-restoration (2013) at monitored, excavated channels at Ni-les'tun restoration site. Channels measured are listed in Table B7. "Wood" indicates wood *versus* non-wood reaches.

	F-value (df)	**p-value**
Channel	3.49 (5)	0.005
Wood	0.61 (1)	0.44
Channel*Wood	3.37 (5)	0.006

Table B7. Means comparison from ANOVA for change in mean channel depth from 2010 (pre-restoration) to 2013 (post-restoration) in excavated channels, Ni-les'tun restoration site. Means with a letter in common are not significantly different (p > 0.05).

Channel	Channel type	Change in mean depth (m)	SE	df	Group
CH05	Excavated	-0.119	0.016	209	a
Fahys Mid	Excavated	-0.139	0.032	209	ab
East Fahys	Excavated	-0.153	0.016	209	a
CH01	Excavated	-0.184	0.042	209	abc
CH06	Excavated	-0.236	0.021	209	bc
NoName Mid/Upr	Excavated	-0.300	0.020	209	c

Table B8. Summary of ANOVA results for fine sediment depth in 2013 at monitored channels at Ni-les'tun restoration site. See Table B6 header for ANOVA factors.

	F-value (df)	p-value
Channel	11.45	<0.0001
Wood	0.15	0.71
Channel*Wood	1.84	0.07

Table B9. Means comparison from ANOVA for fine sediment depth in 2013 at monitored channels at Ni-les'tun restoration site. Means with a letter in common are not significantly different (p > 0.05).

Channel	Channel type	Fine sediment depth (m)	SE	DF	Group
CH01	Excavated	0.050	0.067	291	ab
CH05	Excavated	0.109	0.025	291	a
Fahys Mid	Excavated	0.164	0.048	291	ab
NoName Mid/Upr	Excavated	0.170	0.031	291	ab
CH06	Excavated	0.188	0.034	291	ab
East Fahys	Excavated	0.224	0.023	291	bc
NoName Lower	Non-excavated	0.378	0.048	291	cd
Fahys Lower	Non-excavated	0.497	0.039	291	de
Redd	Non-excavated	0.636	0.043	291	e

Table B10. Summary of BACI ANOVA for plant community metrics at Ni-les'tun restoration site and Bandon Marsh Unit reference site. "Year" indicates pre restoration (2010) *versus* post-restoration (2013).

	Native cover (%)		Non-native cover (%)		Total plant cover (%)		Species richness	
	F-value	**p-value**	**F-value**	**p-value**	**F-value**	**p-value**	**F-value**	**p-value**
Site	5.82	0.02*	2.35	0.14	3.32	0.08	27.06	<0.0001***
Year	0.007	0.93	0.07	0.79	0.48	0.49	0.16	0.69
Site*Year	0.08	0.78	0.24	0.63	2.47	0.13	4.94	0.03*

Table B11. Summary of plant community metrics (native cover, non-native cover, total cover, and species richness) at Ni-les'tun restoration site and Bandon Marsh Unit reference site transects. Means with a letter in common are not significantly different ($p > 0.05$).

Site	Year	Average native plant cover (%)	Average non-native plant cover (%)	Average total plant cover (%)	Average plant species richness
Restoration	Pre-restoration (2010)	56.61 a	58.78 a	115.8 a	5.00 a
Reference	Pre-restoration (2010)	94.28 a	17.74 a	112.1 ab	6.52 a
Restoration	Post-restoration (2013)	47.14 a	39.92 a	87.1 b	3.04 b
Reference	Post-restoration (2013)	92.34 a	11.68 a	104.0 ab	6.88 a

Table B12. Emergent wetlands, Ni-les'tun restoration site and Bandon Marsh Unit reference site: Changes in percent cover across all transects, 2010 to 2013. Native species are highlighted in green, non-native species in orange. "Year" indicates pre restoration (2010) *versus* post-restoration (2013). Results of ANOVA are shown at right; significant effects are marked with asterisks indicating p-values in footnote. Table includes only species with more than 5% average cover in any single transect.

		Percent cover						
		Restoration site		Reference site		Factor effects ($F_{1,16}$)		
Common name	Scientific name	Pre-restoration (2010)	Post-restoration (2013)	Pre-restoration (2010)	Post-restoration (2013)	Site	Year	Site:Yr
Baltic rush	*Juncus balticus*	17.3	19.0	36.0	33.7	2.4	0.1	0.4
Tall fescue	*Schedonorus arundinaceus*	31.2	21.2	0	0	2.8	3.2*	0.9
Creeping bentgrass	*Agrostis stolonifera*	11.3	16.6	17.7	10.2	0.0	0.5	2.2
Pacific silverweed	*Potentilla anserina*	12.3	4.4	22.0	16.1	1.6	3.3*	0.0
Slough sedge	*Carex obnupta*	10.8	11.2	0	0	0.7	0.2	0.0
Seashore saltgrass	*Distichlis spicata*	6.0	7.2	11.9	9.5	0.1	0.1	1.7
Birdsfoot trefoil	*Lotus corniculatus*	11.7	0.6	0	0.2	2.8	8.3*	2.4
Tufted hairgrass	*Deschampsia cespitosa*	0	0	11.5	16.5	13.5**	2.5	9.0**
Creeping spikerush	*Eleocharis palustris*	4.3	1.7	0	0	0.6	1	0.3
Fleshy jaumea	*Jaumea carnosa*	0	0	5.0	3.6	16.4***	0.3	1.1
Pickleweed	*Sarcocornia perennis*	0	0	3.1	4.8	30.1***	4.5*	15.9**

* $P < 0.1$; ** $P < 0.01$; *** $P < 0.001$

Table B13. Composition of plant communities by transect in emergent wetlands (percent cover by species), Ni-les'tun restoration site, July 2013. Native species are highlighted in green, non-natives in orange. Species at each transect that changed more than 10% since pre-restoration (2010) are marked with an upward arrow (indicating >10% increase in cover) or downward arrow (indicating >10% decrease).

		Average Percent Cover[1]													
Common name	**Scientific name**	**NL T2**	**NL T4**	**NL T5**	**NL T9**	**NL T10**	**NL T11**	**NL T12**	**NL T13**	**NL T14**	**NL T15**	**NL T16**	**NL T17**	**NL T18**	**NL T19**
Creeping bentgrass	*Agrostis stolonifera*	20.4	0.0	5.7	9.8	18.4 ↑	15.9	6.0	32.7	78.7 ↑	19.8 ↑	↓ 20.2	4.7	↓ 0.1	0.0
Fat hen	*Atriplex patula*	1.0	3.7	0.0	5.5	0.3	3.5	0.0	8.8	11.4 ↑	4.7	1.3	0.0	0.0	0.0
Slough sedge	*Carex obnupta*	0.0	67.6 [2]	13.2	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	75.6
Seashore saltgrass	*Distichlis spicata*	0.0	0.0	0.0	0.0	2.4	0.0	0.0	0.0	0.0	0.8	0.0	0.0	97.4 ↑	0.0
Creeping spikerush	*Eleocharis palustris*	↓ 11.5	0.0	0.0	0.8	0.0	0.0	0.3	8.2	0.0	2.5	0.0	0.0	0.0	0.0
Purple leaved willowherb	*Epilobium ciliatum*	0.0	0.0	8.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.1	0.0	0.0
Baltic rush	*Juncus balticus*	4.0	11.2	↓ 33.1	54.1 ↑	↓ 49.9	21.0	15.4 ↑	0.4	1.2	44.7 ↑	21.9	8.5	0.0	0.0
Reed canarygrass	*Phalaris arundinacea*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	10.7
Pacific silverweed	*Potentilla anserina*[3]	↓ 1.6	0.1	14.5	24.7	0.0	0.0	0.1	↓ 0.6	↓ 11.0	0.0	0.0	0.0	0.0	9.4
Tall fescue	*Schedonorus arundinaceus*	0.8	0.0	28.1	10.0	18.2	↓ 11.8	↓ 82.4	0.4	3.7	↓ 5.3	55.7	80.9	0.0	0.0

[1] Table includes only species with more than 5% cover in any single transect.

[2] The majority of the slough sedge cover (about 40%) in NL T4 was dead or dying.

[3] Pacific silverweed was formerly referred to as *Argentina egedii.*

Table B14. Composition of plant communities in emergent wetlands (percent cover by species), Bandon Marsh Unit reference site, July 2013. Native species are highlighted in green, non-natives in orange.

Common name	Scientific name	Average Percent Cover*			
		BM T1	BM T2	BM T3	BM T4
Creeping bentgrass	*Agrostis stolonifera*	31.6	6.2	0.0	3.1
Fat hen	*Atriplex patula*	0.1	0.7	0.0	6.7
Tufted hairgrass	*Deschampsia cespitosa*	22.7	40.3	2.6	0.5
Seashore saltgrass	*Distichlis spicata*	13.7	0.3	0.0	23.9
Sea milkwort	*Glaux martima*	1.3	1.2	0.0	5.4
Fleshy jaumea	*Jaumea carnosa*	3.8	5.3	0.0	5.1
Baltic rush	*Juncus balticus*	16.1	27.3	42.9	48.3
Pacific silverweed	*Potentilla anserina***	0.0	16.6	47.7	0.1
Perennial pickleweed	*Sarcocornia perennis***	5.0	5.3	0.0	8.8

* Table includes only species with more than 5% average cover in any single transect

** Pacific silverweed was formerly referred to as *Argentina egedii*; perennial pickleweed was formerly *Salicornia virginica.*

Table B15. Tree density by species in forested wetlands, Ni-lestun restoration site (NL T6, NL T7, NL T20) and Bandon Marsh Unit reference sit e (BM T5), pre-restoration (2011) and post-restoration (2013). All trees in the table are native to Oregon.

Common name	Scientific name	Density (stems/hectare)							
		Restoration site						Reference site	
		NL T6		NL T7		NL T20		BM T5	
		2011	2013	2011	2013	2011	2013	2011	2013
Red alder	*Alnus rubra*	0	0	271	271	155*	155	1914	2691
Pacific crabapple	*Malus fusca*	478	1258	0	0	621*	485	0	0
California waxmyrtle	*Myrica californica*	0	0	0	0	0	0	1734	3229
Sitka spruce	*Picea sitchensis*	41	103	319	335	155	175	1495	2332
Cascara	*Rhamnus purshiana*	0	21	80	16	58	78	2332	3947
Scouler's willow	*Salix scouleriana*	0	0	0	0	0	39	0	0
Western hemlock	*Tsuga heterophylla*	0	0	0	0	0	0	60	60
Total		519	1382	670	622	989	931	7535	12259

*estimated

Table B16. Tree basal area by species, pre-restoration (2011) and post-restoration (2013), forested wetlands, Ni-les'tun restoration site (NL T6, NL T7, NL T20) and Bandon Marsh Unit reference sit e (BM T5). All trees in the table are native to Oregon.

		Basal area (m^2/hectare)							
		Restoration site						Reference site	
		NL T6		NL T7		NL T20		BM T5	
Common name	**Scientific name**	**2011**	**2013**	**2011**	**2013**	**2011**	**2013**	**2011**	**2013**
Red alder	*Alnus rubra*	0.0	0.0	7.0	6.1	4.7*	4.3	8.1	7.7
Pacific crabapple	*Malus fusca*	0.0	0.9	0.0	0.0	4.7*	4.3	0.0	0.0
California waxmyrtle	*Myrica californica*	0.0	0.0	0.0	0.0	0.0	0.0	7.8	7.8
Sitka spruce	*Picea sitchensis*	12.7	23.0	22.6	19.0	28.7	33.7	7.5	14.0
Cascara	*Rhamnus purshiana*	0.0	0.0	1.2	0.2	0.2	0.2	1.7	2.2
Scouler's willow	*Salix scouleriana*	0.0	0.0	0.0	0.0	0.0	0.5	0.0	0.0
Western hemlock	*Tsuga heterophylla*	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
Total		12.7	24.0	30.9	25.2	38.2	43.1	25.0	31.8

*estimated

Table B17. Shrub density by species in forested wetlands, July 2011 and July 2013, Ni-les'tun restoration site (NL T6, NL T7, NL T20) and Bandon Marsh Unit reference site (BM T5). All shrubs in the table are native to Oregon.

		Density (stems/hectare)							
		Restoration site						Reference site	
		NL T6		NL T7		NL T20		BM T5	
Common name	**Scientific name**	**2011**	**2013**	**2011**	**2013**	**2011**	**2013**	**2011**	**2013**
Salal*	*Gaultheria shallon*	0	0	6817	40,903	0	0	0	837
Black twinberry	*Lonicera involucrata*	0	0	598	0	0	0	478	478
Pacific crabapple	*Malus fusca*	478	120	0	0	0	837	0	0
Salmonberry	*Rubus spectabilis*	2870	6817	5621	30,139	4066	2990	0	0
Red elderberry	*Sambucus racemosa*	478	21	0	0	39	78	0	0
Huckleberry*	*Vaccinium* spp.**	0	0	17,461	42,219	3827	3827	2033	2153
Total		5837	8971	32,508	115,274	9943	9745	4522	5481

* Salal and huckleberry (upland shrubs) grew almost exclusively on fallen logs, not in the soil.

** Consisted largely of *V. ovatum* and *V. parviflorum*, but some plants were not identified to species level.

Table B18. Percent cover of herbaceous species in forested wetlands, pre-restoration (2011) and post-restoration (2013), Ni-les'tun restoration site (NL T6, NL T7, NL T20) and Bandon Marsh Unit reference site (BM T5). All species in the table are native to Oregon.

		Average % cover							
		Restoration site						Restoration site	
		NL T6		NL T6		NL T6		NL T6	
Common name	**Scientific name**	**2011**	**2013**	**2011**	**2013**	**2011**	**2013**	**2011**	**2013**
Pacific lady-fern	*Athyrium filix-femina*	4	0	3	7	0	0	0	0
Deer fern	*Blechnum spicant*	0	0	2	0	0	0	0	0
Slough sedge	*Carex obnupta*	72	96	21	17	75	75	0	0
Giant horsetail	*Equisetum telmateia*	0	0	1	2	0	2	0	0
Soft rush	*Juncus effusus*	0	0	0	3	0	0	0	0
Skunk cabbage	*Lysichiton americanum*	28	1	60	28	12	14	6	16
Pacific water parsley	*Oenanthe sarmentosa*	0	1	0	0	0	0	10	2
Common sword fern	*Polystichum munitum*	0	0	0	0	0	3	3	0

Table B19. Area of native and non-native dominated plant communities at Ni-les'tun during pre-restoration (2010) and post-restoration (2013).

	Area (ha)	
Plant community type	**Pre-restoration (2010)**	**Post-restoration (2013)**
Native dominated	91.2	86.2
Non-native dominated	104.5	103.0
Not mapped (upland/offsite or water/mud)	38.3	44.8
Grand Total	**234.0**	**234.0**

Table B20. Area of dominant vegetation types (alliances) at Ni-les'tun during pre-restoration (2010) and post-restoration (2013). Native species are highlighted in green, non-natives in orange. Data include the "Osprey Site" west of the main Ni-les'tun pasture (13 ha).

Alliance number	Scientific name	Common name	2010 area (ha)	2013 area (ha)
14	*Schedonorus arundinaceus*	Tall fescue	95.14	68.98
3	*Agrostis stolonifera*	Creeping bentgrass	0.33	31.40
9	*Juncus balticus*	Baltic rush	4.54	26.99
12	*Picea sitchensis*	Sitka spruce	20.42	20.81
5	*Carex obnupta*	Slough sedge	9.40	12.37
4	*Carex lyngbyei*	Lyngbye's sedge	7.38	7.78
1	*Potentilla anserina*	Pacific silverweed	30.71	3.72
7	*Distichlis spicata*	Saltgrass	4.22	3.17
6	*Deschampsia cespitosa*	Tufted hairgrass	2.15	2.74
10	*Juncus effusus*	Soft rush	2.10	2.48
11	*Phalaris arundinacea*	Reed canarygrass	8.73	2.29
16	*Salix hookeriana*	Hooker willow	2.23	1.98
19	*Sarcocornia perennis*	Pickleweed	1.61	1.46
15	*Schoenoplectus americanus*	Threesquare	0.74	0.89
18	*Scirpus microcarpus*	Small-fruited bulrush	1.33	0.70
2	*Alnus rubra*	Red alder	1.09	0.56
20	*Typha latifolia*	Common cattail	0.28	0.40
8	*Eleocharis palustris*	Creeping spikerush	0.40	0.34
13	*Rhamnus purshiana*	Cascara	0.00	0.14
n/a		Minor alliances in 2010	0.68	0.00
99		Bare ground and water/mud	2.87	7.16
100-102		Not mapped (upland/offsite)	37.68	37.66
		Total	234.02	234.02

Table B21. Major plant communities (associations) and map units at Ni-les'tun in 2013. Associations occupying less than 2 ha are lumped here but are listed separately in Table B22.

Association number (map unit)	Association	Area (ha)
65	Tall fescue - Baltic rush - creeping bentgrass - orache - Pacific silverweed	30.46
12	Creeping bentgrass - orache - creeping spikerush - brass buttons	12.82
63	Tall fescue - Baltic rush - creeping bentgrass	12.49
51	Sitka spruce - red alder / slough sedge - skunk cabbage	11.02
37	Baltic rush - creeping bentgrass - orache - creeping spikerush	6.67
29	Slough sedge (dead/dying) - reed canarygrass (dead/dying) - orache	6.00
35	Baltic rush - creeping bentgrass	5.95
40	Baltic rush - saltgrass - creeping bentgrass - orache	4.72
58	Tall fescue - creeping bentgrass - orache - Pacific silverweed	4.61
59	Tall fescue - common velvetgrass - colonial bentgrass - birdsfoot trefoil	4.59
39	Baltic rush - saltgrass - creeping bentgrass	4.47
17	Lyngbye's sedge	4.42
64	Tall fescue - Baltic rush - creeping bentgrass - Pacific silverweed	4.41
6	Creeping bentgrass - Pacific silverweed - orache	4.36
66	Tall fescue - Baltic rush - birdsfoot trefoil - Pacific silverweed	3.47
55	Transition: Sitka spruce / slough sedge - skunk cabbage to Sitka spruce / evergreen huckleberry - deer fern	3.46
9	Creeping bentgrass - orache	3.42
30	Saltgrass	3.17
67	Tall fescue - Baltic rush - birdsfoot trefoil - creeping bentgrass - Pacific silverweed - common velvetgrass	3.01
60	Tall fescue - common velvetgrass - creeping bentgrass	2.83
16	Bare ground	2.79
24	Slough sedge - reed canarygrass - Pacific silverweed	2.79
61	Tall fescue - common velvetgrass - creeping bentgrass - Pacific silverweed - Baltic rush	2.53
34	Baltic rush - Pacific silverweed - creeping bentgrass - creeping spikerush	2.45
52	Sitka spruce - red alder / Hooker willow - salmonberry / slough sedge - skunk cabbage	2.44
54	Sitka spruce / slough sedge	2.36
45	Pacific silverweed - common cattail - creeping bentgrass - birdsfoot trefoil - small-fruited bulrush - creeping spikerush	2.05

Association number (map unit)	Association	Area (ha)
4	Creeping bentgrass	2.00
n/a	Other associations <2 ha each (47 associations)	36.24
99	Upland - not mapped	24.63
100	Water/mud	4.37
101	Wetland/upland - not mapped	6.76
102	Wetland - not mapped	6.28

Table B22. Minor plant communities (associations occupying <2 ha) at Ni-les'tun in 2013. Major associations are listed in Table B21.

Association number (map unit)	Common name	Area (ha)
1	Pacific silverweed	0.89
2	Pacific silverweed - birdsfoot trefoil	0.78
3	Red alder - Hooker willow	0.56
5	Creeping bentgrass - Pacific silverweed	0.79
7	Creeping bentgrass - Pacific silverweed - orache - creeping spikerush	1.26
8	Creeping bentgrass - Pacific silverweed - orache - creeping spikerush - Baltic rush	1.51
10	Creeping bentgrass - orache - saltgrass - creeping spikerush	1.56
11	Creeping bentgrass - orache - creeping spikerush	1.65
13	Creeping bentgrass - saltgrass	0.55
14	Creeping bentgrass - creeping spikerush	0.37
15	Creeping bentgrass - common velvetgrass - birdsfoot trefoil - perennial ryegrass	1.09
18	Lyngbye's sedge - creeping bentgrass - seaside arrowgrass - saltgrass - pickleweed	0.33
19	Lyngbye's sedge - saltgrass	1.22
20	Lyngbye's sedge - Baltic rush - creeping bentgrass	1.35
21	Lyngbye's sedge - seaside arrowgrass	0.46
22	Slough sedge	0.50
23	Slough sedge - Pacific silverweed	1.48
25	Slough sedge (dead/dying)	1.60

Association number (map unit)	Common name	Area (ha)
26	Tufted hairgrass - creeping bentgrass	0.59
27	Tufted hairgrass - saltgrass - pickleweed - creeping bentgrass	0.34
28	Tufted hairgrass - Baltic rush - silverweed - creeping bentgrass	1.81
31	Creeping spikerush - brass buttons	0.34
32	Baltic rush - Pacific silverweed	0.81
33	Baltic rush - Pacific silverweed - orache	0.63
36	Baltic rush - creeping bentgrass - orache	1.07
38	Baltic rush - creeping bentgrass - orache - meadow barley	0.24
41	Soft rush	0.23
42	Soft rush - slough sedge (both dead/dying)	0.54
43	Soft rush - creeping spikerush	0.28
44	Soft rush - common velvetgrass - creeping buttercup - birdsfoot trefoil	1.43
46	Reed canarygrass	0.21
47	Reed canarygrass - Pacific silverweed	0.30
48	Reed canarygrass - slough sedge - Pacific silverweed	0.81
49	Reed canarygrass - slough sedge - soft rush - birdsfoot trefoil	0.69
50	Reed canarygrass - common velvetgrass - soft rush - birdsfoot trefoil	0.29
53	Sitka spruce - red alder / small-fruited bulrush - soft rush - slough sedge - skunk cabbage	0.50
56	Sitka spruce / Hooker willow / tall fescue - Baltic rush - creeping bentgrass	1.04
57	Cascara - red alder - willow	0.14
62	Tall fescue - common velvetgrass - birdsfoot trefoil	0.59
68	Hooker willow - Sitka willow	0.27
69	Hooker willow - Sitka willow / slough sedge - skunk cabbage	1.72
70	Threesquare - saltgrass - Lyngbye's sedge	0.38
71	Three square	0.50
72	Small-fruited bulrush	0.05
73	Small-fruited bulrush - soft rush - slough sedge - Pacific silverweed	0.65
74	Pickleweed - saltgrass - jaumea (seaside arrowgrass - Lyngbye's sedge)	1.46
75	Common cattail	0.40

Table B23. Area of native and non-native dominated plant communities at the Bandon Marsh Unit reference site during pre-restoration (2010) and post-restoration (2013).

	Area (ha)	
Plant community type	**Pre-restoration (2010)**	**Post-restoration (2013)**
Native-dominated	80.84	81.43
Non-native dominated	9.51	8.92
Not mapped (upland/offsite or water/mud)	42.75	42.75
Grand Total	133.11	133.11

Table B24. Area of dominant vegetation types (alliances) at the Bandon Marsh Unit reference site during pre-restoration (2010) and post-restoration (2013), in descending order by area. Native species are highlighted in green, non-natives in orange. 2010 data were very similar (not shown).

Alliance number	**Scientific name**	**Common name**	**2013 area (ha)**
12	*Deschampsia cespitosa*	Tufted hairgrass	42.61
6	*Sarcocornia perennis*	Pickleweed	18.31
2	*Agrostis stolonifera*	Creeping bentgrass	5.78
9	*Picea sitchensis*	Sitka spruce	5.50
4	*Deschampsia cespitosa/Carex lyngbyei*	Mosaic of tufted hairgrass and Lyngbye's sedge	5.07
1	*Juncus balticus*	Baltic rush	3.41
8	*Triglochin maritima*	Seaside arrowgrass	1.86
10	*Schedonorus arundinacea*	Tall fescue	1.85
3	*Carex lyngbyei*	Lyngbye's sedge	1.63
11	*Schoenoplectus americanus*	Threesquare	1.52
13	n/a	Upland weedy grasses	1.30
5	*Potentilla anserina*	Pacific silverweed	1.15
7	*Distichlis spicata*	Saltgrass	0.37
99		Water/mud	42.75
		Total	133.11

Table B25. Plant communities (associations) and map units at the Bandon Marsh Unit reference site in 2013, in descending order by area. Associations and their areas were very similar in 2010 (not shown).

Association number (map unit)	Association	Area (ha)
24	Tufted hairgrass - saltgrass - pickleweed - jaumea	22.37
10	Pickleweed - saltgrass - jaumea - (seaside arrowgrass - Lyngbye's sedge)	11.48
9	Pickleweed - saltgrass - jaumea	5.55
14	Sitka spruce - red alder - California wax myrtle	5.50
7	Mosaic of tufted hairgrass-saltgrass-pickleweed-jaumea and Lyngbye's sedge-seaside arrowgrass	5.07
20	Tufted hairgrass - Baltic rush - Pacific silverweed	5.02
2	Creeping bentgrass	4.51
22	Tufted hairgrass - Baltic rush - Pacific silverweed - creeping bentgrass	3.77
23	Tufted hairgrass - Baltic rush - Pacific silverweed - pickleweed - saltgrass - jaumea	3.70
1	Baltic rush - saltgrass	3.41
18	Tufted hairgrass - Baltic rush	2.84
13	Seaside arrowgrass - pickleweed	1.86
15	Tall fescue - common velvetgrass - creeping bentgrass - Pacific silverweed	1.62
21	Tufted hairgrass - Baltic rush - Pacific silverweed - (Douglas aster - yarrow - sea-watch angelica)	1.60
19	Tufted hairgrass - Baltic rush - creeping bentgrass	1.59
17	Threesquare - saltgrass - Lyngbye's sedge - pickleweed	1.52
26	Upland weedy grasses	1.30
11	Pickleweed - saltgrass - jaumea - threesquare	1.29
8	Pacific silverweed - Baltic rush	1.15
111	Tufted hairgrass - Baltic rush - creeping bentgrass - Lyngbye's sedge	0.98
5	Lyngbye's sedge - Baltic rush - threesquare	0.76
25	Tufted hairgrass - saltgrass - pickleweed - jaumea - Lyngbye's sedge	0.74
3	Creeping bentgrass - Lyngbye's sedge - saltgrass - seaside arrowgrass	0.65
112	Creeping bentgrass - Lyngbye's sedge	0.61

Association number (map unit)	Association	Area (ha)
4	Lyngbye's sedge - Baltic rush - seaside arrowgrass - creeping bentgrass - tufted hairgrass	0.54
12	Saltgrass	0.37
6	Lyngbye's sedge - threesquare - pickleweed - jaumea - Baltic rush	0.33
16	Tall fescue - European beachgrass - American dunegrass	0.23
0	Water/mud	42.75
	Grand Total	133.11

Table B26. Summary of BACI ANOVA for soil characteristics in emergent wetlands at Ni-les'tun restoration site and Bandon Marsh Unit reference site. "Year" indicates pre restoration (2010) *versus* post-restoration (2013).

	Soil salinity		% OM		% C		pH	
	F-value (df)	p-value	F-value (df)	p-value	F-value (df)	p-value	F-value (df)	p-value
Site	11.25 (1)	0.002	15.75 (1)	0.0004	15.73 (1)	0.0004	1.76 (1)	0.19
Year	4.75 (1)	0.04	0.00 (1)	0.97	0.00 (1)	0.97	0.08 (1)	0.77
Site*Year	0.78 (1)	0.38	3.10 (1)	0.09	3.13 (1)	0.09	0.02 (1)	0.89

Table B27. Means comparison for soil characteristics in emergent wetlands, Ni-les'tun restoration site and Bandon Marsh Unit reference site, before (2010) and after restoration (2013). Letter codes indicate significant differences (p<0.05); means with a letter in common are not significantly different.

Site	Year	Soil salinity	% OM	% C	pH
Restoration	Pre-restoration (2010)	3.66 b	9.28 b	6.61 b	5.24 a
Reference	Pre-restoration (2010)	15.70 a	17.59 a	11.96 a	5.54 a
Restoration	Post-restoration (2013)	19.66 a	13.78 a	9.37 a	5.29 a
Reference	Post-restoration (2013)	32.28 a	18.59 a	12.64 a	5.61 a

Table B28. Soil characteristics in emergent and forested wetlands, Ni-les'tun restoration site, pre-restoration (2010, unless otherwise noted) and post-restoration (2013).

		Average Soil salinity		**Average % OM**		**Average % C**		**Average pH**	
Transect	**Vegetation type**	**2010**	**2013**	**2010**	**2013**	**2010**	**2013**	**2010**	**2013**
NL T02	emergent	1.50	32.46	10.00	14.29	6.80	9.72	5.57	5.63
NL T04	emergent	1.26	25.45	8.14	13.05	5.53	8.87	4.90	5.09
NL T05	emergent	0.38	6.46	4.88	7.52	3.32	5.11	5.95	5.04
NL T06*	forested	0.44	15.25	45.40	25.67	30.87	17.45	4.60	4.47
NL T07*	forested	0.75	0.62	65.33	51.62	44.42	35.10	4.30	4.90
NL T09	emergent	0.96	17.56	6.99	12.81	4.75	8.71	5.33	5.02
NL T10	emergent	3.06	18.01	10.25	14.93	6.97	10.16	5.20	5.29
NL T11	emergent	6.30	23.81	9.11	16.24	6.20	11.04	5.15	5.31
NL T12	emergent	4.69	2.16	7.76	12.05	5.28	8.19	5.50	5.24
NL T13	emergent	1.80	34.42	9.41	15.97	6.40	10.86	5.13	5.73
NL T14	emergent	4.03	24.98	19.44	19.19	13.22	13.05	4.76	4.95
NL T15	emergent	1.93	21.11	9.25	12.21	6.29	8.30	5.05	4.97
NL T16	emergent	4.03	16.73	9.48	12.84	6.45	8.73	5.11	5.09
NL T17	emergent	1.56	10.29	6.16	16.00	4.19	10.88	5.81	5.27
NL T18	emergent	19.29	34.42	8.74	14.65	5.94	9.96	4.78	6.77
NL T19	emergent	0.38	7.44	10.28	11.14	6.99	7.58	5.11	4.62
NL T20*	forested	0.27	1.22	20.12	26.42	13.68	17.97	5.00	4.81
Average		3.10	17.20	15.34	17.45	10.43	11.86	5.13	5.19
Average - emergent		3.66	19.66	9.28	13.78	6.31	9.37	5.24	5.29
Average - forested		0.49	5.70	43.62	34.57	29.66	23.51	4.63	4.73

* Pre-restoration soil samples were collected in 2003 for NL T6 and NL T7, and 2011 for NL T20 (see text for details).

Table B29. Soil characteristics in emergent and forested wetlands, Bandon Marsh Unit reference site, pre-restoration (2010, unless otherwise noted) and post-restoration (2013).

		Average soil salinity		**Average % OM**		**Average % C**		**Average pH**	
Transect	**Vegetation type**	**2010**	**2013**	**2010**	**2013**	**2010**	**2013**	**2010**	**2013**
BM T1	emergent	22.88	44.44	11.65	16.12	7.92	10.96	5.64	5.56
BM T2	emergent	20.73	28.85	20.87	11.86	14.19	8.06	5.62	5.99
BM T3	emergent	5.62	18.46	22.00	32.22	14.96	21.91	5.23	5.19
BM T4	emergent	13.56	37.38	15.84	14.17	10.77	9.63	5.65	5.71
BM T5	forested	1.38	2.41	33.00	54.05	22.44	36.75	4.90	4.94
Average		12.84	26.31	20.67	25.68	14.06	17.46	5.41	5.48
Average - emergent		15.70	32.28	17.59	18.59	11.96	12.64	5.54	5.61
Average - forested		1.38	2.41	33.00	54.05	22.44	36.75	4.90	4.94

* “Pre-restoration” soil samples were collected in 2011 for BM T5 (at the reference site).

Table B30. Summary of BACI ANOVA results for May-August daily mean and daily maximum salinity at channel mouth locations (FAHYMTH8239, NONAMMTH8231, REDDMID8240, BMUNKC8233, SHPWRKA8238), Ni-les'tun restoration site *versus* Bandon Marsh Unit reference site. "Year" indicates pre restoration (2010) *versus* post-restoration (2013).

	Daily mean salinity		**Daily maximum salinity**	
	F Value (DF)	**P-value**	**F Value (DF)**	**P-value**
Site	56.2 (1)	< 0.0001	430 (1)	< 0.0001
Year	20.0 (1)	< 0.0001	13.3 (1)	< 0.0001
*Site*Year*	3.76 (1)	0.05	126 (1)	< 0.0001

Table B31. Summary of BACI ANOVA results for May-August daily mean and daily maximum salinity at upper channel locations (FAHYMID8230, FAHYRD8241, BMUNKA8235, BMUNKB8232, SHPWRKB8229), Ni-les'tun restoration site *versus* Bandon Marsh Unit reference site. "Year" indicates pre restoration (2010) *versus* post-restoration (2013).

	Daily mean salinity		**Daily maximum salinity**	
	F Value (DF)	P-value	F Value (DF)	P-value
Site	192 (1)	< 0.0001	395 (1)	< 0.0001
Year	78.1 (1)	< 0.0001	27.6 (1)	< 0.0001
*Site*Year*	17.5 (1)	< 0.0001	157 (1)	< 0.0001

Table B32. May-August daily mean and maximum salinity before (2011) and after restoration (2013) at channel mouth locations. Means with a letter in common are not significantly different (p > 0.05).

Channel Position	**Site**	**Year**	**Daily mean salinity**	**Daily maximum salinity**
Mouth	Restoration	Pre-restoration (2011)	10.3 c	12.8 c
Mouth	Reference	Pre-restoration (2011)	14.6 b	24.0 b
Mouth	Restoration	Post-restoration (2013)	11.3 c	23.6 b
Mouth	Reference	Post-restoration (2013)	17.2 a	25.9 a

Table B33. May-August daily mean and maximum salinity before and after restoration at upper channel locations. Means with a letter in common are not significantly different (p > 0.05).

Channel Position	**Site**	**Year**	**Daily mean salinity**	**Daily maximum salinity**
Upper	Restoration	Pre-restoration (2011)	0.05 c	0.1 c
Upper	Reference	Pre-restoration (2011)	5.3 b	12.5 b
Upper	Restoration	Post-restoration (2013)	6.0 b	15.4 a
Upper	Reference	Post-restoration (2013)	8.8 a	15.9 a

Table B34. Summary of BACI ANOVA results for May-August daily mean and daily maximum temperature at channel mouth locations, Ni-les'tun restoration site *versus* Bandon Marsh Unit reference site. "Year" indicates pre restoration (2010) *versus* post-restoration (2013).

	Daily mean temperature		**Daily maximum temperature**	
	F Value (DF)	**P-value**	**F Value (DF)**	**P-value**
Site	69.4 (1)	< 0.0001	4.58 (1)	0.03
Year	47.1 (1)	< 0.0001	43.4 (1)	< 0.0001
*Site*Year*	30.2 (1)	< 0.0001	0.12 (1)	0.72

Table B35. Summary of BACI ANOVA results for May-August daily mean and daily maximum temperature at upper channel locations, Ni-les'tun restoration site *versus* Bandon Marsh Unit reference site, before restoration (2011) and after restoration (2013).

	Daily mean temperature		**Daily maximum temperature**	
	F Value (DF)	**P-value**	**F Value (DF)**	**P-value**
Site	17.3 (1)	< 0.0001	84.1 (1)	< 0.0001
Year	41.4 (1)	< 0.0001	24.6 (1)	< 0.0001
*Site*Year*	1.78 (1)	0.18	6.48 (1)	0.01

Table B36. Daily mean and maximum temperature before and after restoration at channel mouth locations (FAHYMTH8239, NONAMMTH8231, REDDMID8240, BMUNKC8233, SHPWRKA8238; see Appendix A, Figures A2 and A3 for locations). Means with a letter in common are not significantly different (p > 0.05).

Channel Position	**Site**	**Year**	**Daily mean temperature**	**Daily maximum temperature**
Mouth	Restoration	Pre-restoration (2011)	16.3 a	18.7 c
Mouth	Reference	Pre-restoration (2011)	15.1 b	19.1 c
Mouth	Restoration	Post-restoration (2013)	16.2 a	19.9 b
Mouth	Reference	Post-restoration (2013)	16.1 a	20.4 a

Table B37. Daily mean and maximum temperature before and after restoration at upper channel locations (FAHYMID8230, FAHYRD8241, BMUNKA8235, BMUNKB8232, SHPWRKB8229; see Appendix A, Figures A2 and A3 for locations). Means with a letter in common are not significantly different (p > 0.05).

Channel Position	**Site**	**Year**	**Daily mean temperature**	**Daily maximum temperature**
Upper	Restoration	Pre-restoration (2011)	15.6 c	17.3 c
Upper	Reference	Pre-restoration (2011)	16.2 b	19.0 b
Upper	Restoration	Post-restoration (2013)	16.2 b	18.9 b
Upper	Reference	Post-restoration (2013)	17.1 a	19.9 a

Table B38. Linear relationships between monitoring results (soils, groundwater, and vegetation) and percent inundation, from linear regression of data from 14 emergent wetland transects at the Ni-les'tun restoration site. Significant relationships are indicated with asterisks (* represents p<0.05, *** represents p<0.001).

Parameter	Adjusted R2	F-value	P-value
Soil carbon content	0.01	1.099	0.32
Soil salinity	0.70	31.38	0.0001***
Shallow groundwater duration[1]	0.65	24.66	0.0003***
Prevalence index[2]	0.24	5.058	0.04*
Total % cover by plants	0.13	3.027	0.11
% native plant cover	-0.01	0.83	0.38
Plant species richness	0.38	8.96	0.01*

[1] Shallow groundwater duration is defined in **Groundwater levels: Methods** above

[2] Prevalence index is a measure of the hydrophytic (wetland-tolerant) status of vegetation at a specific location.

Table B39. Frequency of fish habitat access opportunity (percent of time that fish could access the channel) for typical channels at Ni-les'tun during the peak use month of May and year-round, before (2009) and after restoration (2013).

			% of time fish could access channel			
			May		Full year	
Location	Channel bottom elevation (m NAVD88)	Elevation needed for fish access (m NAVD88)	Pre-restoration (2009)	Post-restoration (2013)	Pre-restoration (2009)	Post-restoration (2013)
E Fahys	0.52	0.97	42.4	54.1	41.9	58.7
CH5 Mouth	0.77	1.23	11.4	41.9	13.3	45.6
CH6 Lower	0.91	1.37	4.5	34.6	6.7	38.2
NoName Mid	1.13	1.59	1.5	22.1	1.8	26.8
Shipwreck Mouth	0.33	0.78	64.0	64.0	67.9	67.9

Table B40. Count model and zero model coefficients for staghorn sculpin CPUE ratio analysis.

Count Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
(Intercept)	6.4314	0.9272	6.937	4.02E-12	***
mo	-0.6394	0.1399	-4.569	4.90E-06	***
trt	-0.3149	0.4028	-0.782	0.4343	
after	-0.7578	0.4704	-1.611	0.1072	
trt:after	1.4423	0.6056	2.381	0.0172	*
Log(theta)	-0.3894	0.237	-1.643	0.1003	

Zero Hurdle Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
(Intercept)	4.684	1.6877	2.775	0.00551	**
mo	-0.4602	0.2282	-2.017	0.04369	*
trt	-2.0493	0.6392	-3.206	0.00135	**
after	2.3159	0.6228	3.719	0.0002	***

Table B41. Count model and zero model coefficients for age-0 chinook CPUE ratio analysis.

Count Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
(Intercept)	2.1155	0.3084	6.86	6.90E-12	***
trt	-1.3216	0.4324	-3.056	0.00224	**
after	-0.4302	0.5096	-0.844	0.39852	
trt:after	0.748	0.6857	1.091	0.27534	
Log(theta)	-0.5533	0.3380	-1.637	0.10159	

Zero Hurdle Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
(Intercept)	12.8182	2.4712	5.187	2.14E-07	***
mo	-1.4612	0.3078	-4.748	2.06E-06	***
trt	-4.1292	0.9863	-4.187	2.83E-05	***
after	-0.9392	1.1800	-0.796	0.426	
trt:after	1.9788	1.3429	1.474	0.141	

Table B42. Count and zero hurdle model coefficients with level of significance for age-0 chinook wood structure analysis.

Count Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
(Intercept)	1.5369	0.355	4.329	1.50E-05	***
wood.ind	1.0058	0.3805	2.643	8.21E-03	**
Log(theta)	-1.2648	0.5059	-2.5	1.24E-02	*

Zero Hurdle Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
Redd	2.8432	0.9974	2.851	0.004362	*
Fahys	1.8426	0.9343	1.972	0.048581	.
NoName	1.3242	0.9257	1.43	0.152586	
Ref. 1	2.4094	1.047	2.301	0.021383	
Ref. 2	4.9974	1.4209	3.517	0.000436	
Ref. 3	4.2166	1.2267	3.437	0.000587	
mo	-0.3241	0.1281	-2.529	0.011429	

Table B43. Count and zero hurdle model coefficients with level of significance for staghorn sculpin wood structure analysis.

Count Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
(Intercept)	5.3111	0.8334	6.373	1.86E-10	***
mo	-0.535	0.1218	-4.394	1.11E-05	***
wood.ind	1.2848	0.293	4.385	1.16E-05	***
Log(theta)	-0.5117	0.2651	-1.93	0.0536	.

Zero Hurdle Model Coefficients

	Estimate	Std. Error	z value	Pr(>|z|)	Signficance
(Intercept)	2.9171	1.1832	2.465	0.0137	*
mo	-0.3113	0.1753	-1.775	0.0758	.

Appendix C. Additional photos

Photo C1. Dillon Blacketer (L) and Isaac Kentta (R) conducting channel cross-sectional survey with survey chain and depth pole.

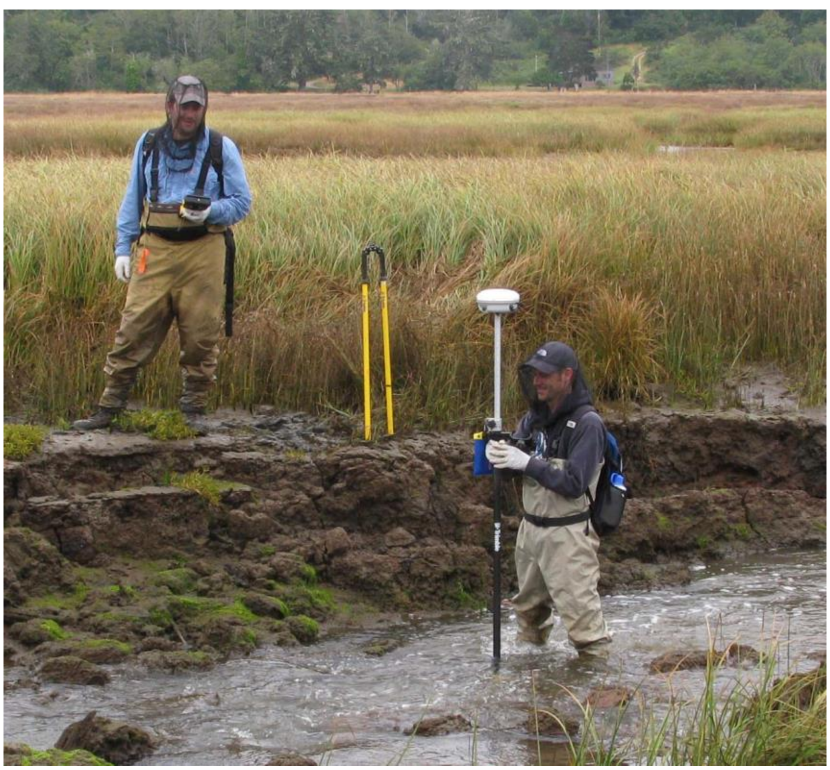


Photo C2. Michael Ewald (L) and Chris Janousek (R) conducting flowpath survey with RTK-GPS.

Photo C3. Headcut on lower Fahys Creek. This prominent headcut is located where the creek crosses the mudflats, which are higher than the flow path of the restored channel within the restoration site.

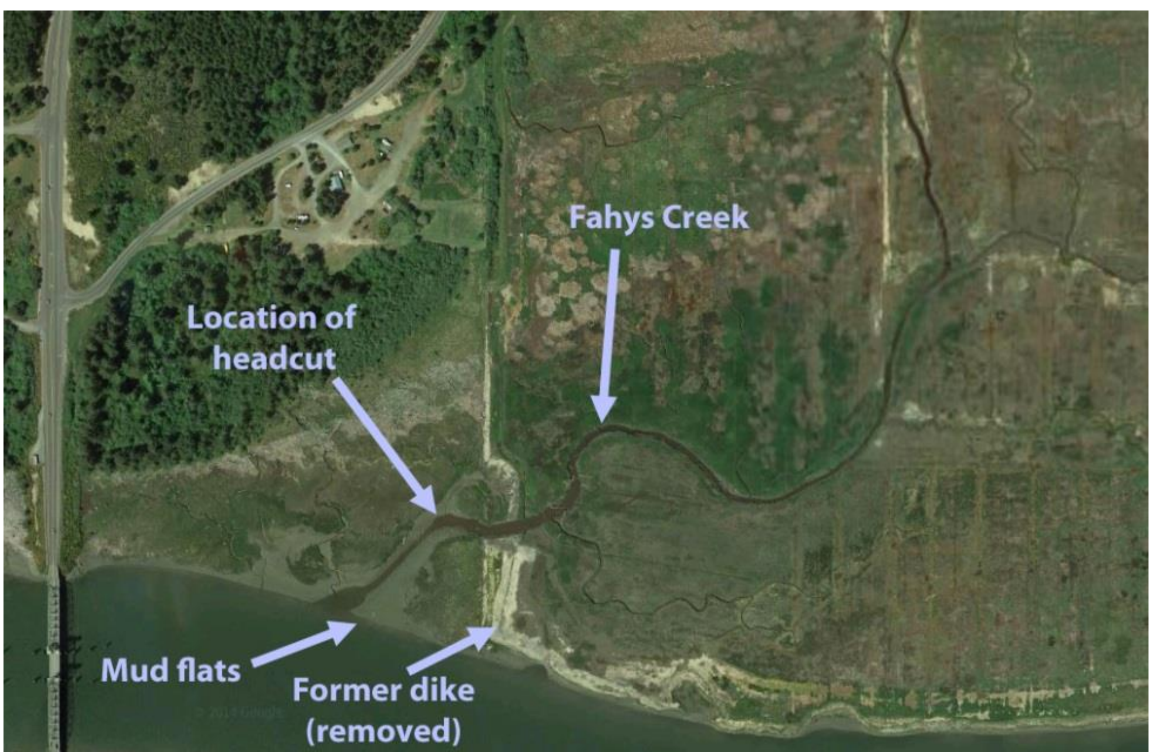


Photo C4. Aerial view of location of headcut in Photo C1. Aerial photo ©Google 2014.

Photo C5. Rapidly-changing vegetation at Ni-les'tun, due to restored tidal inundation and salinity: orache is growing on clumps of dying tall fescue. Tall fescue in the background is healthier, but deteriorating.

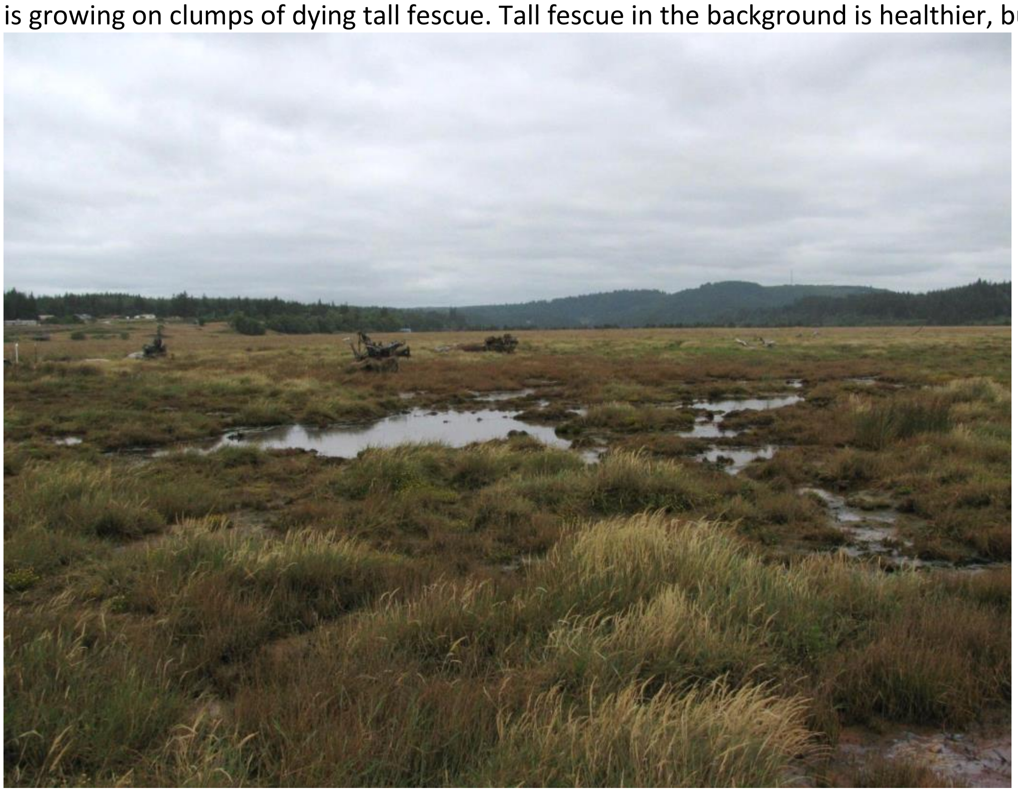


Photo C6. Rapidly changing vegetation at Ni-les'tun due to restored tidal inundation and salinity: creeping bentgrass and creeping spikerush are growing amidst areas of bare ground and ponded water.

Photo C7. Groundwater well with tall riser to exclude surface tidal inundation (white PVC pipe). Tidal wrack is deposited around the well; metal t-posts helped protect the well from floating debris.

Photo C8. Large intertidal beaver dam and pool at the Bandon Marsh Unit reference site, looking upstream. The dam and surrounding high marsh inundate on spring tide cycles. Smaller intertidal beaver dams and pools were also present nearby.


Photo C9. The same intertidal beaver dam and pool as in Photo C8, looking downstream.