Tidal Wetland Vegetation in the Lower Coos Watershed

Summary:

- *The Coos estuary has lost most of its historic tidal marshes and forested swamps. However, remaining tidal marsh and swamp acreage appears to be relatively unchanged since 1979.*
- *Tidal wetlands in the Coos estuary host diverse plant communities characterized by several dominant species, including pickleweed in marine-dominated marshes and sedges in brackish marshes.*
- *Data from undisturbed marshes in the project area show that the composition of undisturbed marsh plant communities in the Coos estuary appears to be stable.*
- *There are a few non-native, invasive, and endangered tidal wetland plant species of concern in the Coos estuary.*

Evaluation

Most of Coos estuary's original tidal marshes and forested swamps have been converted to other uses; remaining tidal wetland plant communities appear stable.

Evaluation

We do not have enough historic data to fully evaluate the status of scrub-shrub and forested tidal wetlands

DATA GAP

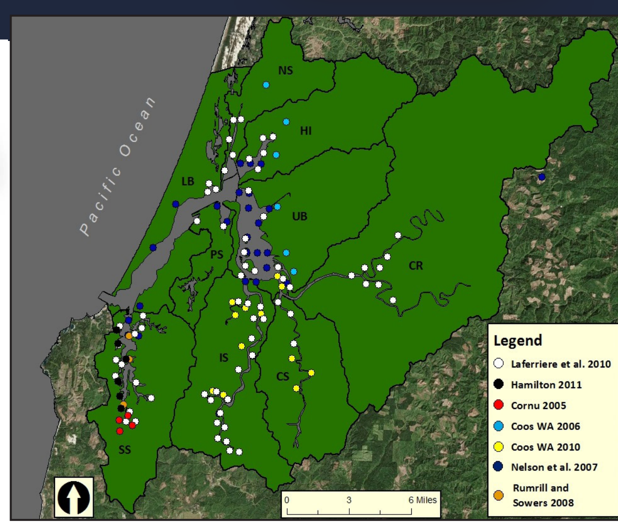


Figure 1. Spatial extent of marsh vegetation studies in the project area.

What's happening?

<u>Change in Tidal Wetland Area</u>

Several studies have quantified the extent of wetland conversion in the Coos estuary since European settlement (ca. 1800). Hofnagle et al. (1976) estimated that almost 90% of Coos Bay's tidal wetlands have been converted to other land uses. Good (2000) estimated the total loss between 1870 and 1970 to be 66%. In a 2006 report, the Coos Watershed Association (CoosWA) examined estuarine sediment and vegetation to determine the historical extent of wetland areas in six lowland streams (CoosWa 2006)(Figures 1 and 2). Their analysis suggests that wetland areas in these basins appear to have decreased by 70-95% of their historic extent (Table 1). Over the past 30 years, some of these converted wetlands have been restored to their original function (see the Marsh Restoration sections below).

This data summary uses the National Wetlands Inventory (NWI) habitat classification (Cowardin et al. 1979)(see sidebar) to analyze recent change to wetland area based on available NWI data from 1979 and from

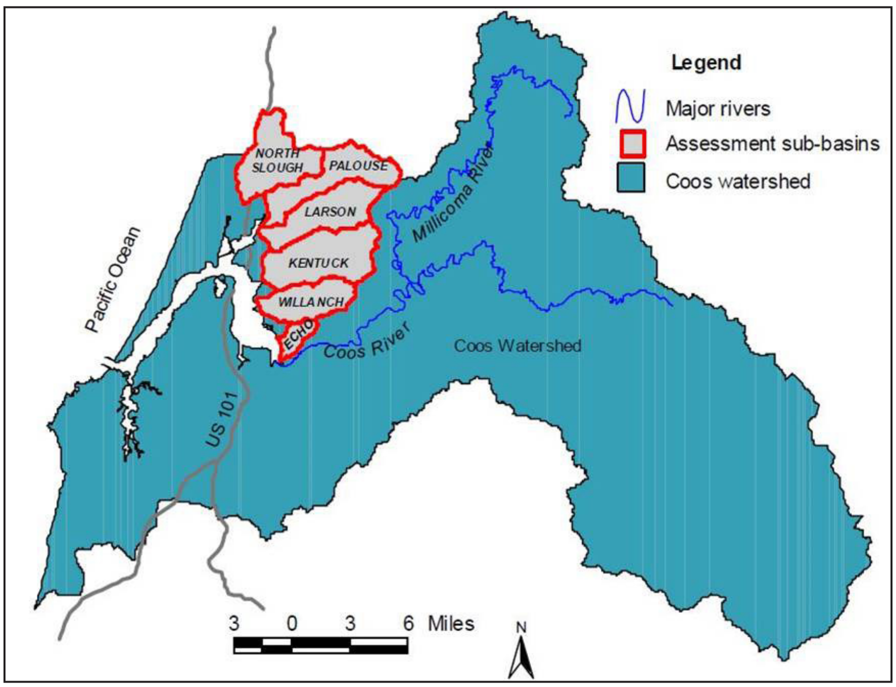


Figure 2. Coos Watershed Association lowland assessment sub-basins . Figure: CoosWA 2006

Lowland Creek	Historic Wetlands (Pre-settlment Acres)	Current Wetlands (2006 Acres)	Long term (% change)
Palouse Creek	555	30	-95%
North Slough	508	165	-68%
Larson Creek	587	46	-92%
Kentuck Creek	608	37	-94%
Willanch Creek	256	17	-93%
Echo Creek	194	60	-69%

Table 1. Estimates of historic wetland habitat loss since European settlement in six lowland sub-basins of the Coos estuary (see Figure 2). Data: CoosWA 2006

National Wetland Inventory

In the late 1970s, the United States Fish and Wildlife Service (USFWS) established a federal standard for wetland and deep water habitat classification. The new system was used to conduct a nationwide wetland habitat inventory to provide information about the distribution of wetlands in the United States and aid in conservation efforts.

This classification scheme defines wetland habitat according to its ecological and physical characteristics, including water-loving plants (hydrophytes), wetland soils (hydric soils), and flooding frequency. At the highest level of the classification hierarchy, "systems" are defined by one of five types: marine, estuarine, riverine, lacustrine, and palustrine. Each system is further classified by substrate material (e.g., unconsolidated gravel bottom), flooding regime (e.g., regularly flooded intertidal habitat) and vegetation (e.g., scrub-shrub wetlands dominated by small trees or shrubs).

Sources: Cowardin et al. 1979, USFWS 2014b

2003 (see 2003 data in Figure 3). The NWI system classifies wetland habitat into several broad "Systems" including Marine, Estuarine, Riverine, Lacustrine, and Palustrine (Table 2 and Figure 4). Each System consists of a series of "Classes", "Subclasses" and "Dominance Types". This data summary focuses on the vegetation classes within NWI's Estuarine system: Aquatic Bed (e.g. seagrasses), Emergent Wetland (rooted vegetation whose leaves and stems extend above the water surface), Scrub-Shrub Wetland, and Forested Wetland (Figure 5). For additional information about the NWI classification system for other wetland types see NWI's online Wetlands Mapper tool (USFWS 2014a). We note here that there is little information available about the historic extent of scrub-shrub and forested tidal wetlands in the Coos estuary which would have been almost entirely converted prior to NWI wetland mapping in 1979. This data summary does not reflect the true loss of those habitats and recognizes the omission as a key information gap.

At the System level, the NWI data suggest that wetlands in the project area have remained relatively unchanged between 1979 and 2003 (Table 3), after years of wetland loss documented in Hofnagle at al. (1976), Good (2000), and CoosWA (2006). At the more refined Class level, the data actually indicate an increase (40%) in vegetated wetland acreage within all Systems (excluding Marine) from 1979 to 2003 (USFWS 1979, 2003)(Table 4). This change is driven by the Aquatic Beds Class in the Estuarine System, which experienced an apparent net increase of approx-

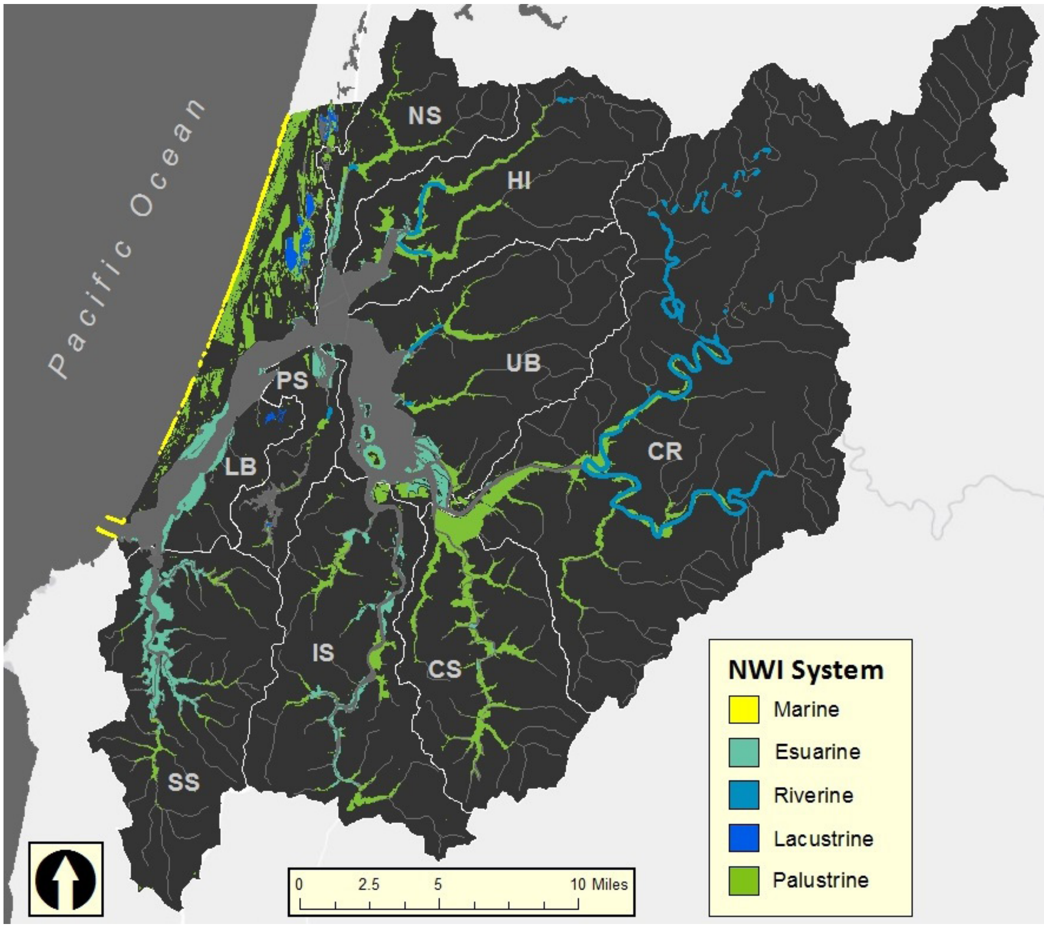


Figure 3. Distribution of wetland habitat at the System level in the lower Coos watershed (project area) in 2003. Map generated from most current National Wetlands Inventory (NWI) data. Data: USFWS 2003

System	Associated Terms	NWI Definition
Marine	ocean, sea, bay, reef	Open ocean overlying the continental shelf and its associated high-energy coastline. Salinities exceed 30 ppt, with little or no dilution.
Estuarine	estuary, marsh, slough, tidal flat, delta	Deepwater tidal habitats and adjacent tidal wetlands that are usutally semienclosed by land but have open, partly obstructred, or sporadic access to the open ocean, and in which ocean water is at least occasionally dilluted by fresh-water runoff from the land.
Lacustrine	lake, pond, reservoir	Wetlands and deepwater habitats with following characteristics : 1) situation in a topographic depression aor a dammed river channel, 2) lacking trees, shrubs, persistent emergents, emergent mosses, or lichens with greater than 30% areal coverage, and 3) total area exceeds 20 acres. Lacustrine waters may be tidal, but ocean-derrived salinity is always less than 5 ppt.
Palustrine	bog, marsh, fen, swamp, prarie, slough	All nontidal wetlands that aredominated by trees, shrubs, persistent emergents, emergent mosses or lichens. Also includes tidal areas were 1) salinity is less than 5 ppt and water depth is less than 2 meters at low water.
Riverine	river, stream, brook	All wetlands and deepwater habitats contained within a channel, with two exceptions: 1) wetland dominated by trees, shrubs, persistent emergents, emergent mosses, or lichens, and 2) habitats with water containing ocean-derived salts

Table 2. Definition of five NWI systems. Data: Cowardin et al. 1979.

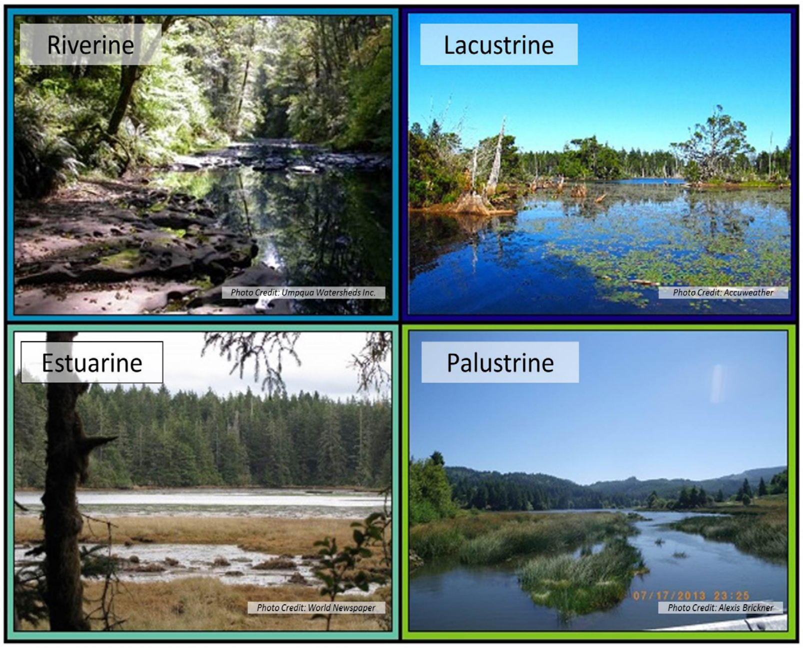


Figure 4. Examples of habitat representing four NWI Systems. See Figure 1 for distribution of Systems within project area and Table 2 for definitions. Top left: Millicoma River, Coos River Subsystem; Top right: Empire Lakes, Lower Bay Subsystem; Bottom left: Hidden Creek marsh, South Slough Subsystem; Bottom right: Matson Creek marsh, Catching Slough Subsystem.

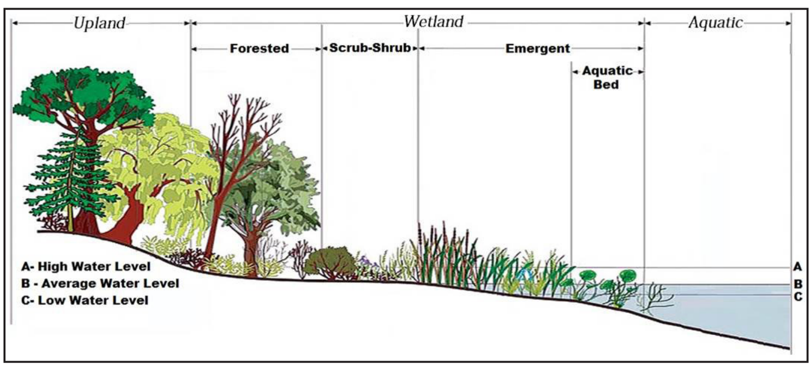


Figure 5. Schematic representation of NWI vegetation classes within a wetland setting. Image modified from Wilcox et al. 2007

System Form	1979 Acreage	2003 Acreage	Percentage Change (1979-2003)
Estuarine	13,349.96	13,381.70	0.2%
Lacustrine	937.21	937.19	0.0%
Palustrine	9,830.30	9,830.17	0.0%
Riverine	412.38	412.37	0.0%
All System Forms	**24,529.85**	**24,561.43**	**0.1%**

Table 3. Summary of wetland habitat change within the lower Coos estuary (1979-2003) based on NWI data aggregated at the System level. Data: USFWS 1979, 2003

Vegetation Class (Estuarine System Form)	1979 Acres	2003 Acres	Percentage Change (1979-2003)
Aquatic Bed	903.8	1476.1	63%
Emergent	1797.9	1797.9	0%
Forested	0.0	0.2	N/A

Table 4. Summary of wetland habitat change within the lower Coos estuary (1979-2003) based on NWI data aggregated at the Class level in the Estuarine System. Data: USFWS 1979, 2003

imately 570 acres (63% increase from 1979 levels). Acreage for all other vegetation Classes decreased slightly; declines in Emergent, Forested, and Scrub-Shrub Wetland Classes collectively accounted for less than 3 acres of lost wetland in the lower Coos estuary from 1979- 2003.

It's important to note some limitations of the NWI data. While gains or losses may be indicative of actual trends, they could also reflect a reclassification of existing wetland areas. For example, a parcel that was unclassified in 1979 may be classified as aquatic vegetation in 2003. In this case, the data would suggest a net gain of aquatic vegetation habitat, even though this gain may only reflect a reclassification of pre-existing wetlands. Since the data do not account reclassifications, it's possible that the observed increase in aquatic vegetation beds within the estuarine systems of Coos Bay (Table 4) may be exaggerated (see Chapter Summary for data limitations). We are missing data with which to independently check these results since eelgrass has been comprehensively mapped only once in the Coos estuary (in 2005)(see Seagrasses and Algae in the Lower Coos Watershed data summary in this chapter). In general, more data characterizing the current and historic extent of tidal wetlands for the Coos estuary are needed to improve our understanding of local tidal wetland status and trends.

<u>Tidal Wetland Community Composition and Diversity</u>

In 2007, the Western Ecology Division of the United States Environmental Protection Agency (USEPA) published a report that included information about the distribution of emergent marsh vegetation in the extensive estuarine intertidal habitats (tidal wetlands) in California, Oregon and Washington estuaries (Nelson et al. 2007). Sampling did not include high marsh habitats; they focused on characterizing low emergent marshes and intertidal mud and sand flats. In summer 2002, they collected data from 217 sites, with almost half of the 65 sites in Oregon located in the Coos estuary (30 sites). USEPA results indicate that overall low marsh vegetation percent cover for West Coast tidal wetlands is low, underscoring the dominance of intertidal mud and sand flats in the total area of West Coast estuaries (Figure 6). It is important to note that these data are relative to low marsh habitats only; if high marsh habitats were included in the study, mud and sand flat habitat would not have been as dominant. USEPA add that low marsh and intertidal flat vegetative cover from non-native species was very low (8% overall), having encountered only two such species: Japanese eelgrass (*Zos-*

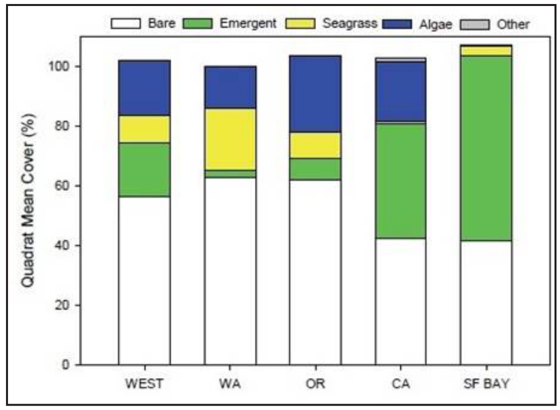


Figure 6. Mean relative abundance of vegetation groups and bare (unvegetated) area in low marsh vegetation plots. Nelson et al. 2007

tera japonica)(estuaries in all three states) and smooth cordgrass (*Spartina alterniflora*) (Washington estuaries only). Japanese eelgrass is commonly found in the low intertidal zones of the Coos estuary (see Seagrasses and Algae in the Lower Coos Watershed data summary in this chapter). Smooth cordgrass is not found in the Coos estuary (see Chapter 18: Non-Indigenous/Invasive Species).

Other studies characterize the emergent marshes (both low and high marshes) of the Coos estuary. Rumrill and Sowers (2008) characterized emergent marsh vegetation along the estuarine gradient in the Coos estuary's South Slough, with one site representing habitats subject to marine-dominated tidal hydrology (full salinity- 33); one site representing habitats subject to polyhaline tidal hydrology (salinity 18-30); and one site representing habitats subject to riverine/mesohaline tidal hydrology (salinity 0-18). All study sites were considered to be "least disturbed", meaning they've not been converted to other land uses and remain relatively undisturbed by other human activities. Not surprisingly, Rumrill and Sowers report that emergent marsh communities displayed "substantial spatial variability" along the estuarine gradient. The greatest species richness was recorded at the riverine/mesohaline site, while the polyhaline and marine-dominated sites

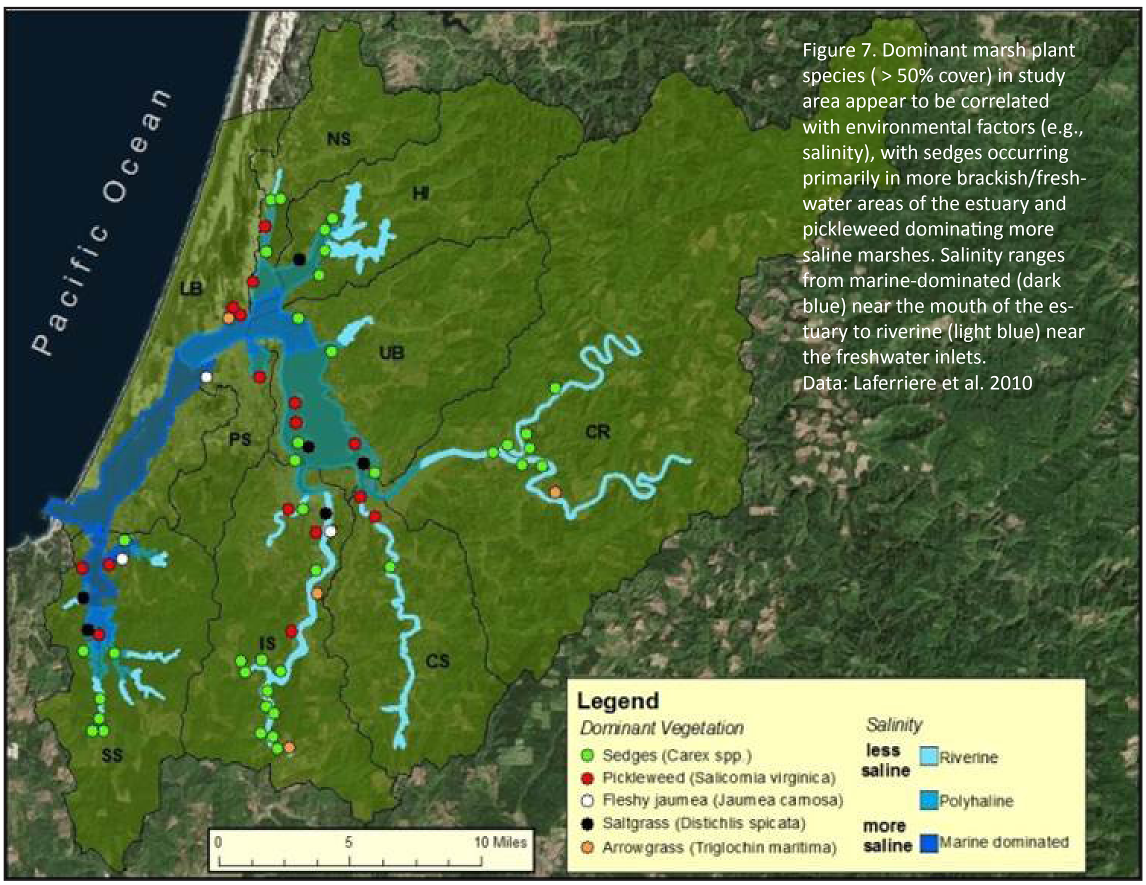


Figure 7. Dominant marsh plant species (> 50% cover) in study area appear to be correlated with environmental factors (e.g., salinity), with sedges occurring primarily in more brackish/freshwater areas of the estuary and pickleweed dominating more saline marshes. Salinity ranges from marine-dominated (dark blue) near the mouth of the estuary to riverine (light blue) near the freshwater inlets.
Data: Laferriere et al. 2010

displayed lower species richness (due to the relatively fewer number of emergent plant species adapted to higher salinity growing conditions). Plant communities in the study sites were generally characterized by a group of four to seven dominant species (>10% cover) with up to eight other sub-dominant species characterized in the study.

Least disturbed intertidal wetlands in the Coos estuary typically host communities comprising "a mixed assemblage of 25-30 common emergent vascular plants." Marshes are often dominated by the relatively high abundance of a few species, including pickleweed (*Salicornia virginica*), fleshy jaumea (*Jaumea carnosa*), salt grass (*Distichlis spicata*), tufted hairgrass (*Deschampsia caespitosa*), creeping bentgrass (*Agrostis stolonifera*), Lyngby's sedge (*Carex lyngbyei*), and arrowgrass (*Triglochin maritimum*)(Hamilton 2011; Nelson et al. 2007; Laferriere et al. 2010; Cornu 2005a; CoosWA 2010; Rumrill and Sowers 2008).

Another study, by Laferriere et al. (2010), surveyed emergent intertidal marsh vegetation at sites throughout the Coos estuary as part of an investigation into the abundance and distribution of *Assiminea parasitologica*, a small, invasive snail native to Japan (Figures 1 and 7)(see more information about the snail in Chapter 18: Non-Indigenous/Invasive Species). Since the sampling design for this study was focused on the invasive snail, the emergent marsh data, while still very useful, include gaps and should not be considered a comprehensive characterization of Coos estuary emergent marsh plant communities. For example, the investigators recorded the presence of generic sedge species (*Carex sp.*), missing the opportunity to distinguish the critical difference between the dominant freshwater sedge (*Carex obnupta*) and the salt-tolerant sedge (*Carex lyngbyei*)(see Data Gaps and Limitations in Chapter Summary).

Laferriere et al. (2010) reports that sedges (*Carex spp*.) are the most abundant plant species in marshes in the upper reaches and freshwater dominated portions of the estuary, while pickleweed dominates marshes in the more marine-dominated (saline) environments. The distribution of other dominant species appears to be patchy throughout the lower watershed. Data from other studies in the South Slough Subsystem support these conclusions (Hamilton 2011)(Figure 8).

Species "richness" or other measures of species diversity (see sidebar) also appear to be a function of ecosystem "drivers", such as tidal inundation period (determined by tidal magnitude and marsh surface elevation- low elevation marshes experience longer periods of tidal inundation than high marshes) and salinity regime (tidewater salinity and resulting marsh soil salinity)(Hamilton 2011)(Figure 9).

Hamilton (2011) compared diversity metrics in multiple years at South Slough's overlapping "biomonitoring" marsh study sites (Figure 1)(Rumrill and Sowers 2008, Hamilton 2011). Both diversity and species richness appear to have increased from 2004-2010 (Figure 10). This trend is most apparent when comparing the average species richness, a

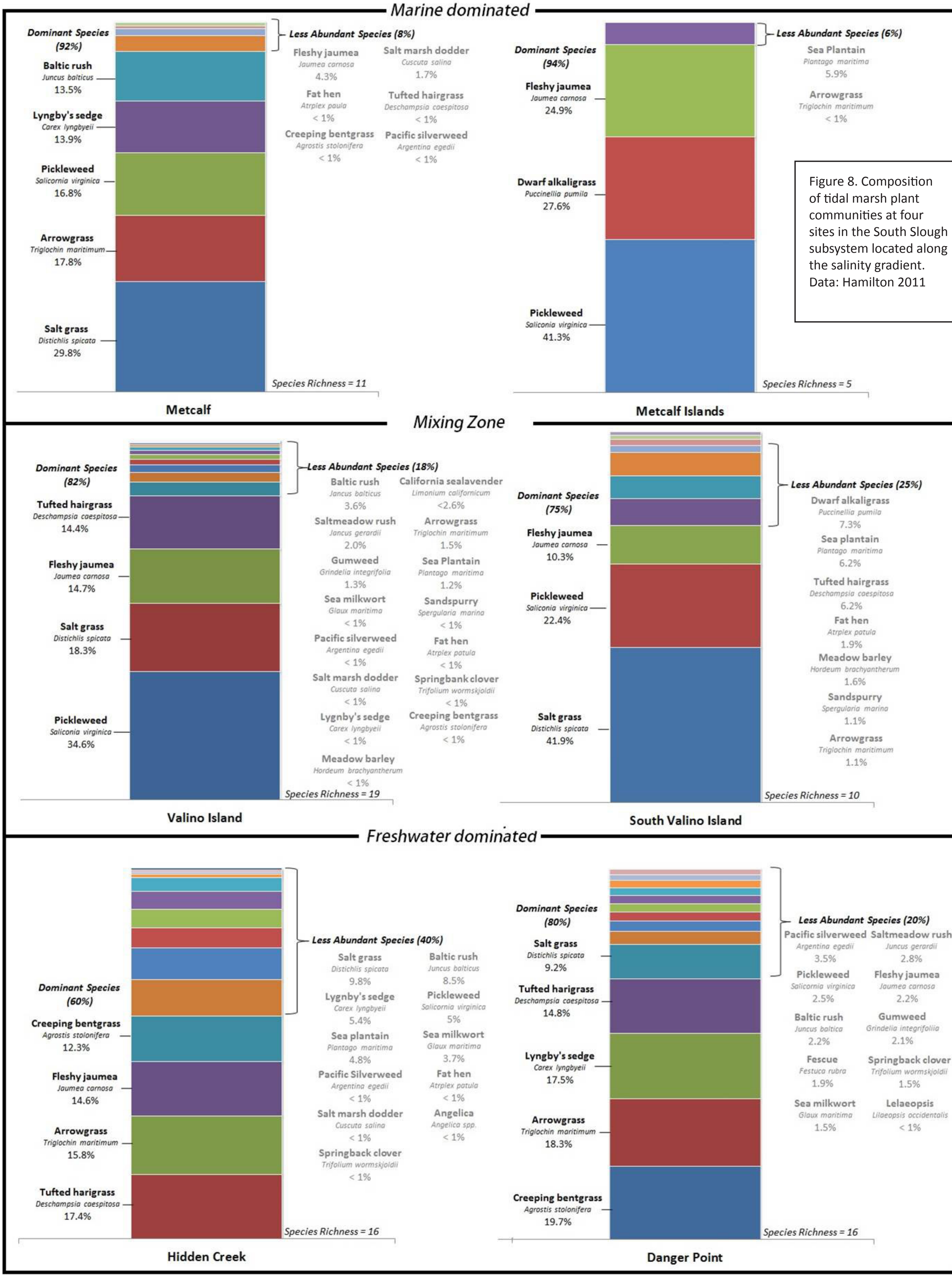


Figure 8. Composition of tidal marsh plant communities at four sites in the South Slough subsystem located along the salinity gradient. Data: Hamilton 2011

Measuring Biodiversity

Because quantifying diversity can be a complex process, several measures or "indices" have been developed. The following list explains a few common indices:

Species Richness: the total number of species in a community.

Species Evenness: a measure of how evenly individual species are distributed within a community, with a value of 1 being perfect parity (i.e., exact same number of each species in community).

Shannon-Weiner Index: a "composite index" that incorporates both richness and evenness. Values range from 1.5-3.5 in most ecological communities, with high values representing greater diversity.

Effective number of species: a measure of species richness based on composite indices calculated by imposing the assumption of perfect evenness and calculating the number of species necessary to achieve a specified diversity value.

Sources: Magurran 2004; Heip et al. 1998; Gotelli and Chao 2013; Kerkhoff 2010

metric that showed substantial increases at both Danger Marsh (33% increase from 2004 levels) and Valino Island (58% increase). It's important to note, however, that while these trends may reflect true ecological changes, they may also be attributable to differences in methods used in the different studies. Data to be collected in the future at the same long-term monitoring sites will shed light on these findings (Hamilton 2011).

Emergent Marsh Restoration

Tidal wetland restoration projects, most commonly focused on emergent marsh habitat classes, have been almost commonplace in the project area over the past 20 years. The most complete record of restored acres in the project area comes from the Oregon Watershed Restoration Inventory (ORWI 2013a, 2013b). The ORWI catalogs all projects funded by the Oregon Watershed Enhancement Board (OWEB), a state agency that is a substantial source of grant funding for wetland restoration projects. These records suggest that OWEB funding has resulted in approximately 268 acres of tidal wetland restoration within the project area since 2002. South Slough National Estuarine Research Reserve (SSNERR) staff have worked with partners since 1996 to restore approximately 80 acres of tidal wetlands within the Reserve (see below). Figures for tidal wetland restoration conducted by others in the project area and for tidal wetlands acreage restored through the compensatory mitigation process were not available at the time of this writing.

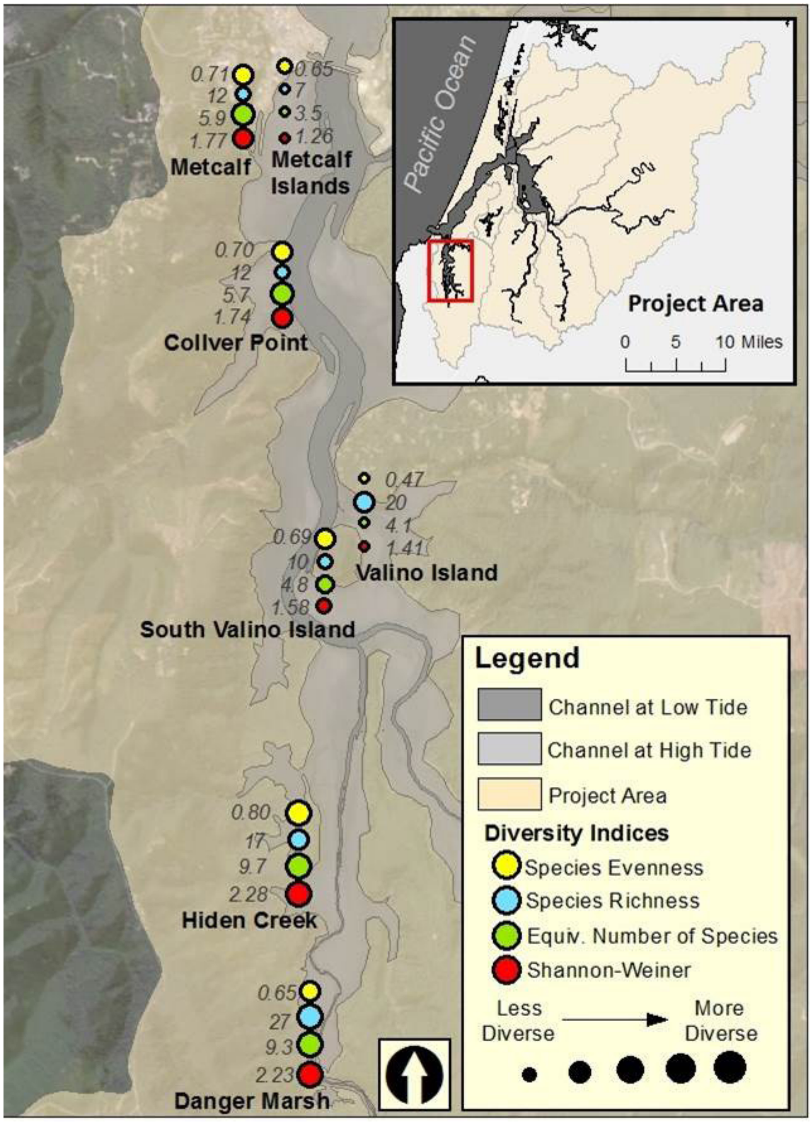


Figure 9. Diversity metrics describing the species evenness and species richness (see "measuring biodiversity" sidebar) of marsh plant communities in South Slough. Data collection occurred at South Slough Reserve "biomonitoring" sites that span South Slough's salinity gradient. More saline sites occur in the north part of the slough (Metcalf); more brackish and freshwater sites occur in the south part of the slough (Danger). Data : Hamilton 2011

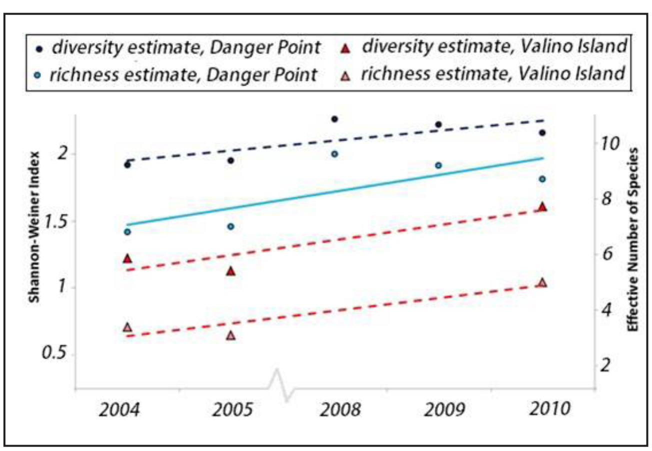


Figure 10. Diversity and species richness of marsh plant communities at Danger Point marsh (blue). Linear trendlines show general increase in both diversity (dashed) and richness (solid) at these two sites. Sampling did not occur in 2006 and 2007. Data: Hamilton 2011

Figure 11. Aerial photo of Kunz Marsh in 1997. The restoration area is divided into four cells by temporary partitions. The elevation of each cell varies with lowest elevation occurring closer to the bottom of the photo and the highest elevation occurring nearer the top. The division of the marsh into cells allowed researchers to examine the effect of marsh elevation on the natural recruitment of emergent salt marsh vegetation and the development of marsh function.

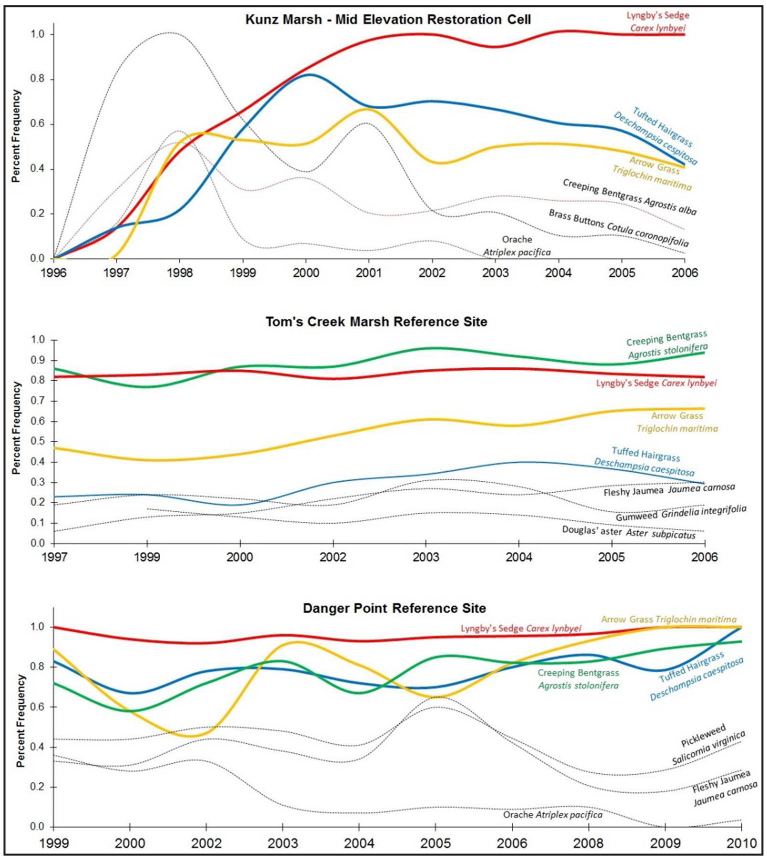


Figure 12. Percent frequency of the most dominant plant species found at Kunz Marsh (restoration site- only mid elevation marsh represented here) as well as Tom's Creek and Danger Point Marshes (reference sites). Data are displayed for both generally abundant species (solid colored lines) as well as less abundant species (black dashed lines). The Kunz Marsh data show a shift in species composition. From 1996 to approximately 2000, the plant community was dominated by early colonizers, including creeping bentgrass (*A. stolonifera*), brass buttons (*C. coronopifolia*), and orache (*A. patula*). In later years, Lyngby's sedge (*C. lyngbyei*), tufted hairgrass (*D. caespitosa*), and arrowgrass (*T. maritima*) were the most dominant species. In comparison, the reference sites show relatively stable plant communities; species composition dominated by the same species that colonized Kunz marsh in the post-2000 years. Data were not collected at the reference sites in 1998, 2001, and 2007. Plant species abundance for these years was interpolated at these sites based on the data that are available for all other years. Data: Cornu 2005a; graphic modified from Cornu et al. 2012.

Figure 13. Excavation of the Anderson Creek high flow channel in the regraded Anderson Creek floodplain.

Several emergent marshes in the project area's South Slough Subsystem (historically converted to agricultural land uses) have been the focus of extensive marsh restoration. These efforts provide insight into the response of brackish marsh plant communities to major "disturbance" events such as re-establishing tidal flooding at a site not flooded for many years. Restoration projects provide insights into the recovery of marsh ecosystems, informing subsequent restoration project planning and design (Cornu 2005a).

Kunz Marsh Restoration Project

In 1997, SSNERR staff began a yearly vegetation monitoring effort at Kunz Marsh, where, in 1996, dike material was used to regrade the subsided marsh to three intertidal elevations and partitioned into four "research cells" (Cornu 2005a)(Figure 11). Kunz marsh presented a unique opportunity to observe the development of a tidal wetland plant community recruited naturally on unvegetated marsh soils at three intertidal elevations (the regraded marsh was not planted). SSNERR staff collected vegetation data (in addition to other data, such as sediment accretion) in each of the Kunz marsh cells to understand the effects of marsh surface elevation on tidal wetland vegetation recruitment and plant community development over time. Kunz marsh vegetation data were compared with vegetation data also collected yearly at adjacent least disturbed marsh sites (Danger Point and Tom's Creek "reference" marshes) to evaluate the rate at which the plant communities developing in the Kunz marsh research cells were progressing towards the "target" marsh plant communities in the reference sites (Cornu 2005a).

Percent frequency vegetation data (i.e., percent of plots in which individual species are encountered) from the two reference sites indicate relative plant community stability compared with Kunz marsh vegetation data (Figure 12). Kunz marsh data document the

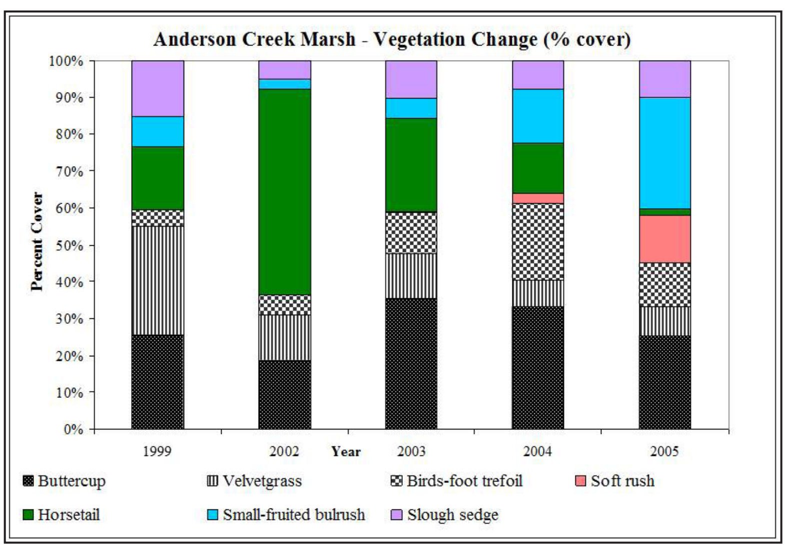


Figure 14. Vegetation change in Anderson Creek floodplain, 1999-2005. Solid colors represent native species. Black and white patterns represent non-native species. Data and graphic: Cornu 2005b

Figure 15. Progression of marsh plant community development in Anderson floodplain 2006-2010 shows continued domination of the native slough sedge (*Carex obnupta*) in areas of slightly less saturated wetland.

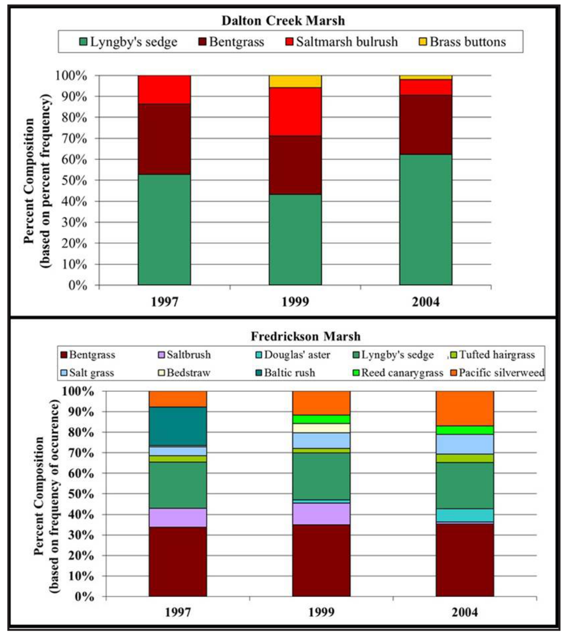


Figure 16. Changes to the brackish plant community at Dalton Creek and Fredrickson marsh restoration sites in the South Slough Subsystem. Data and figure: Cornu 2005c

recruitment of early colonizing vegetation and both the recruitment and establishment of later, permanently colonizing vegetation species, a very dynamic successional process (Figure 12). There are distinct differences in the timing and rate of vegetation recruitment and the colonization trajectories of individual species associated with the Kunz research cell elevations. For more information on monitoring results, see Cornu 2005a, and Cornu and Sadro 2002.

Anderson Creek Marsh Restoration Project

SSNERR staff conducted five years of vegetation monitoring at Anderson Creek marsh, including one year of monitoring in 1999 before restoration began, and four years monitoring after restoration actions were completed in 2002. In 2001, SSNERR staff initiated the restoration of Anderson Creek by regrading the floodplain and generating fill material to eliminate the 850 meter (2,800 ft) deeply downcut ditch that Anderson Creek had become (Cornu 2005b)(Figure 13). Since all existing vegetation was removed during site regrading, and invasive species, including reed canary grass, were among the plant species expected to recruit to the site, Reserve staff re-planted the site aggressively with native wetland and riparian vegetation.

Similar to Kunz Marsh, data from Anderson Creek suggest that the species composition and relative abundance of individual species in the developing plant communities at the site were dynamic. Cornu (2005b) explains that residual pasture grasses and exotic forbs dominated the Anderson floodplain during the growing season in 2003. However, he adds that, by summer 2004, native wetland grasses and forbs had increased in abundance (Figure 14). In subsequent years, the plant community continued to develop, with native vegetation steadily becoming more abundant in the Anderson floodplain, beginning to push out the non-native species by 2005 (Figure 15).

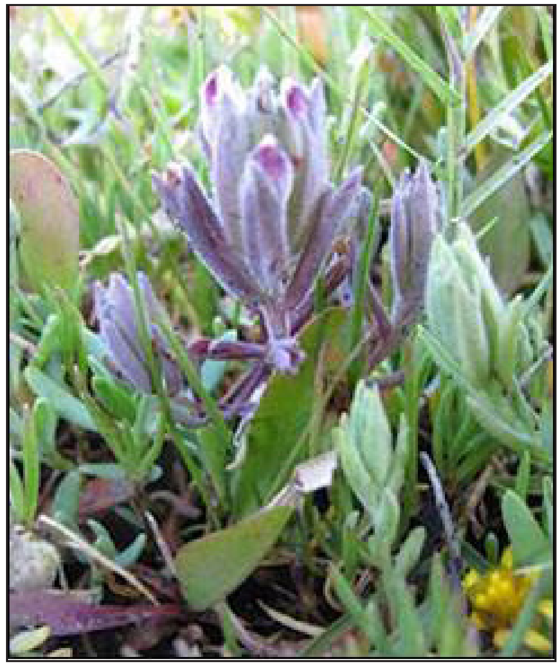


Figure 17. Salt marsh bird's beak. Photo: Institute for Applied Ecology

Figure 18. Pacific reedgrass. Photo: University of California, Berkley

Restoration at Cox, Dalton, and Fredrickson Creek Marshes

Cox, Dalton, and Fredrickson Creek brackish marshes are located near the southern end of the South Slough Reserve boundary. Similar to Kunz and Anderson Creek Marshes, the floodplains of these three sites were diked and converted to agricultural uses in the early 1900's (Cornu 2005c). Restoration began in 1996 for Cox Marsh and 1998 for both Dalton and Fredrickson.

Post-restoration vegetation monitoring revealed very little change to species composition at the Dalton Creek and Fredrickson marsh sites (Figure 16). At the Cox marsh site, a large and now permanent beaver dam was constructed across the mouth of the marsh, raising the freshwater water table all through the marsh. This change is influencing the development of a plant community featuring many more freshwater species than would be represented without the beaver dam, including some non-native (and potentially invasive) species like velvet grass (*Holcus lanatus*), birdsfoot trefoil (*Lotus corniculatus*), and reed canary grass (*Phalaris arundinacea*)(Cornu 2005c).

Plants of Special Concern

The Coos estuary salt marshes are inhabited by a few plant species of special concern. Salt marsh bird's beak (*Chloropyron maritimum ssp. palustre*- formerly known as *Cordylanthus maritimus* ssp. *palustris*) is federally listed as a species of concern and also listed as endangered by the state of Oregon (ORBIC 2013)(Figure 17). This species is found in fringing low marshes in the lower portion of South Slough and in a few locations in the lower Coos estuary (Lower Bay subsystem) (Rumrill and Sowers 2008). SSNERR staff work with partners to keep track of the locations of the plant, which tends to grow in the lower estuary in sandy soils among plants like pickleweed and fleshy jaumea (Cornu et al. 2012).

Pacific reedgrass fen (*Calamagrostis nutkaensis*), a rare native plant, was also found in the South Slough Subsystem in 2005 (Brophy 2005)(Figure 18). For more information about rare and endangered plants in the project area, refer to the Rare and Endangered data summary of this chapter.

In addition to the species mentioned above, the lower Coos watershed also contains non-native and invasive vegetation. The following species are regarded as the biggest non-native or invasive threats to plant communities in the marsh habitats in the project area: reed canary grass, purple loosestrife (*Lythrum salicaria*), knotweeds (*Polygonum spp.*), and cordgrasses (*Spartina spp*.). Grassleaf rush (*Juncus marginatus*), a species considered to be a non-native invasive plant in the Willamette Valley, was also discovered in the South Slough Subsystem (Brophy 2005). Continued monitoring and control of these and other non-native/invasive threats will help ensure the resiliency of native plant communities. For more information about non-native and invasive plants in the project area, refer to Chapter 18: Non-indigenous/Invasive Species.

Why is it happening?

Tidal Wetland Alterations and Restoration

Tidal wetland plant communities are sensitive to natural and human-generated alterations, because the manipulation of wetlands can result in complex (and sometimes extreme)

Marsh Subsidence

The soil surface elevation of diked tidal wetlands tends to decrease over time in a process called "marsh subsidence." Subsidence occurs in areas were wetlands have been diked to accommodate alternative land uses. Since wetlands behind dikes are excluded from tidal flooding, they are prevented from the delivery of sediment that helps maintain marsh surface elevation in a healthy, functioning wetland. When diked wetlands dry out, their soils begin to oxidize, decompose, and consolidate. The marsh vegetation, which once added organic material to the soil, is replaced by pasture vegetation that is continuously removed by grazing, and the soil is heavily compacted by livestock and machinery. Over time, the original marsh soil consolidates and subsides, sometimes significantly (e.g., 80 cm [31 in] at South Slough's Kunz Marsh).

Sources: Cornu 2005a; Roman et al. 1984; Frenkel and Morlan 1991; Anisfeld et al. 1999; Weinstein and Weishar 2002

changes to the hydrology of wetland ecosystems (Cornu 2005a, 2005b, 2005c; Hood 2004, Gedan et al. 2009). Historically, tidal wetland alterations occurred throughout the Coos estuary. Tidal marshes, forested tidal swamps, and scrub shrub tidal wetlands were eliminated through wetland filling (often using dredge spoils), or were diked and drained, and their meandering tidal channels filled and replaced with linear drainage ditches, and marsh, forest, and shrub communities converted or removed. All of these changes were made to accommodate other uses (e.g., agriculture, urban, industry, silviculture)(CoosWA 2006).

In most cases, diking and draining tidal wetlands initiates natural processes that can result in significant elevation loss of those lands behind dikes which can be exacerbated by normal agricultural activities such as livestock grazing and transport of heavy equipment across the site (see sidebar). Subsidence makes agricultural activities increasingly difficult over time because the lower the land, the harder it is for the land to drain, especially after sustained wintertime rainfall. Some subsided agriculture lands drain so poorly that their soils remain saturated for most of the year, greatly limiting or eliminating their agricultural productivity. Many of these lands are simply abandoned.

These same lands, however, can be made productive in other ways that benefit human communities. Restoring dikes and drained wetlands to their fully functioning former tidal wetland condition re-establishes beneficial "ecosystem services" such as critical nursery habitat for commercially important fish and shellfish species (e.g., salmon and Dungeness crabs), floodwater retention (tidal wetlands act like sponges and soak up wintertime floodwaters, which reduces flooding in developed areas), and improvements to water quality (sediments are trapped by tidal wetlands, helping clear turbid waters; excess nutrients and many water soluble compounds considered pollutants are taken up by natural biogeochemical processes constantly occurring in tidal wetlands). When restoring wetlands, subsidence is a common issue that must be addressed by restoration practitioners. The Kunz marsh restoration project described above is one approach to be considered for accelerating the recovery of subsided former tidal wetlands.

Figure 19. Development of a low marsh tidal channel network at the Kunz marsh restoration site, South Slough.

Background

Tidal wetlands form over many decades as layers of sediment from both terrestrial and marine sources are slowly but steadily deposited on tidal flats through daily tidal flooding. Eventually, the tide flats reach elevations relative to the tide that allow their colonization by vascular plants. Many saltwater, brackish and freshwater marsh plant species are adapted to colonize and persist in higher elevation tide flats whose flooding frequency and duration (dictated by tide flat elevation) do not exceed maximum thresholds. Different species have different thresholds- for example, pickleweed, a common low marsh plant, is adapted to withstand more frequent and longer tidal flooding than Pacific silverweed (*Argentina egedii*), a high marsh plant. Tide water salinity also plays a significant role in determining which plants are able to colonize which tide flats; more salt tolerant species (e.g., pickleweed, fleshy jaumea, arrowgrass) can colonize tide flats located in the lower portions of estuaries nearer to the ocean (subject to high salinity levels), while less salt tolerant species, (e.g., pacific silverweed, baltic rush [*Juncus balticus*]) will colonize tide flats located in the upper portions of estuaries further away from the ocean and more influenced by river and stream flows (Cornu 2005a).

Once colonized with low marsh vegetation, tide flats continue to build upwards, accelerated by vegetation's tendency to help trap sediments and contribute their own organic material (through yearly senescence), in a process called "vertical accretion (Kerney et al. 1994; Cahoon et al. 1995; Cornu and Sadro 2002). Mature tidal marshes, also known as high marshes, stop growing vertically when they reach an elevation equal to, or a little higher than the mean of all the higher high tides (mean higher high water- MHHW) that flood the marsh. Mature/high marshes still require regular sediment deposits from tidal flooding to maintain their elevation relative to the tide (Cornu and Sadro 2002).

Tidal channels form largely through the same accretion processes that form vegetated tidal wetlands. Tidal wetlands build up around tidal channels, whose steep banks are stabilized by the cohesiveness of the tide flat sediments (mostly clays and silts except near estuary mouths, where sediments have higher sand content), and by the extensive and persistent root systems associated with tidal wetland plant communities (Cornu 2005a)(Figure 19). Tidal channels serve an important function, providing habitat structure and foraging access for animals as well as pathways for the import and export of materials that help sustain life in the marsh, including nutrients, detritus, and propagules (e.g., plant seeds, benthic invertebrate and insect eggs)(Cornu 2005a).

The composition and distribution of vegetation in estuaries depends on several factors:

- On an macro-scale, the hydrology of an estuary determines the characteristics of the brackish plant community. For exam-

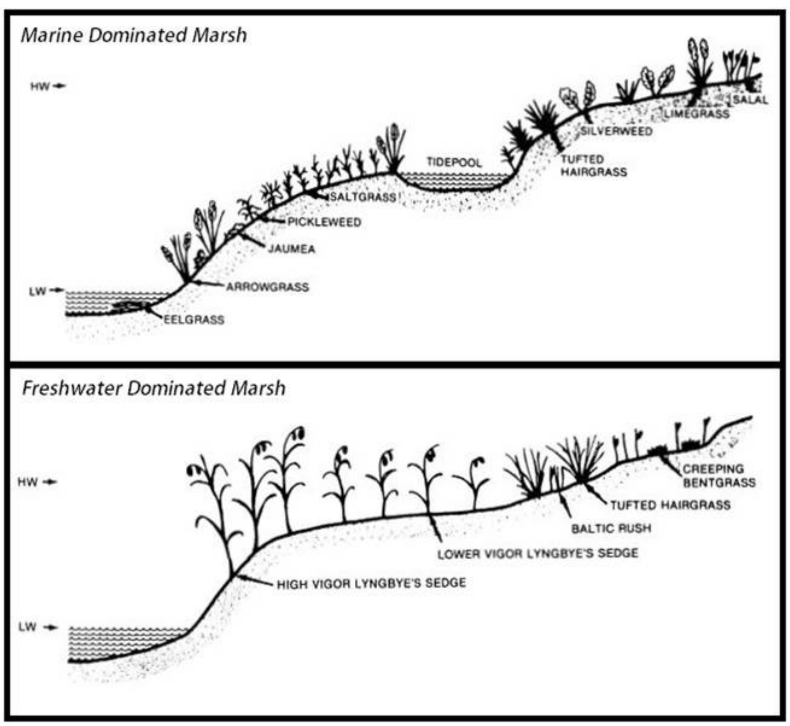


Figure 20. Typical zonation of Pacific Northwest estuarine vegetation showing the distribution of plants relative to marsh elevation. Figure: Seliskar and Gallagher 1983

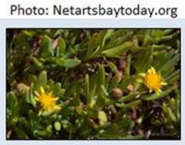
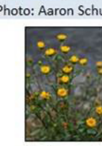
Common Name	Scientific Name	Photo
Angelica	*Angelica spp.*	 Photo: Mdidea.com
Arrowgrass	*Triglochin maritimum*	 Photo: USDA
Baltic rush	*Juncus balticus*	 Photo: Brent Miller
Creeping bentgrass	*Agrostis stolonifera*	 Photo: Zirpe
Douglas Aster	*Symphyotrichum subspicatum*	 Photo: nwplants.com
Fat hen	*Atriplex patula*	 Photo: Netartsbaytoday.org
Fleshy jaumea	*Jaumea carnosa*	 Photo: Aaron Schusteff
Gumweed	*Grindella integrifolia*	 Photo: peardg
Lyngby's sedge	*Carex lyngbyei*	 Photo: Hordur Kristinsson
Pacific silverweed	*Argentina egedii*	 Photo: Jill Weber
Pickleweed	*Saliconia virginica*	 Photo: LTER
Salt grass	*Distichlis spicata*	 Photo: William Skaradek
Salt marsh dodder	*Cuscuta salina*	 Photo: St. Mary College of CA
Tufted hairgrass	*Deschampsia caespitosa*	 Photo: Christian Fischer

Table 5. Commonly occurring native plants in the marshes of the lower Coos estuary.

ple, Nelson et al. (2007) explain that in Washington, where estuaries are dominated by tidal flat habitats, seagrass and macroalgae appear to be more abundant. By contrast, emergent vegetation appears to be abundant in California estuaries, where marsh habitats are more readily available. Since Oregon is a mixture of these two habitat types, the plant communities of its estuaries tend to contain a mixture of macroalgae, seagrass, and emergent vegetation.

- On a smaller scale, the topography and land-use history of a specific waterway as well as the individual features of each tidal wetland (e.g., tidal channels, large wood or lack thereof, shallow pools, filled or excavated areas) all influence the composition and spatial distribution of specific marsh plant communities (Laferriere et al. 2010).

Tidal wetlands in the Coos estuary are subject to a range of environmental conditions. Many are additionally affected either directly or indirectly by a variety of land use histories. As a result, a diversity of plant communities in various stages of successional development continue to persist or have become established in these habitats (Figure 20). Table 5 presents several native plant species commonly occurring in Coos estuary tidal wetlands.

As mentioned above, tidal wetlands have historically been considered impediments to productive land use, and have commonly been altered to accommodate uses that are traditionally viewed as high-value alternatives (e.g., agriculture, urban development, etc.) (Giannico and Souder 2005). In Europe, this practice began as early as the seventh century, and the continued conversion of low-lying coastal zones throughout the globe is the greatest contributing factor to the destruction of wetlands worldwide (Daiber 1986; Middleton 1999; Giannico and Souder 2005). For more information about human structures in tidal wetlands, including dikes and tide gates in the project area, refer to the Land Use section of Chapter 8: Physical Description.

Public health concerns have provided another historic impetus for altering wetlands. The practice of filling or draining wetlands to control the threat of malaria began over 2,000 years ago in Italy (Doody 2001). In North America, this type of mosquito-control began in the southeastern United States in the early 19th century and was adopted by some northern states with the help of returning soldiers who served in the south during the American Civil War (Doody 2001; Dreyer and Niering 1995). Although the threat of malaria has abated, tidal marsh alteration for the purpose of insect control is still common practice in parts of the U.S. (Montague et al. 1987; Giannico and Souder 2005).

In 2013, the significant nuisance associated with mosquitoes was raised in the Bandon area. After a large-scale (420 acres) tidal wetland restoration project was completed at the Bandon Marsh National Wildlife Refuge's Ni-les'tun site, shallow man-made pools remained. These pools combined with a particularly warm and wet spring to cause a boom in the local salt marsh mosquito (*Aedes dorsalis*) population. While the project was undertaken in 2011 to benefit wildlife, including migratory waterfowl and shorebirds, and commercially important fish species (including the threatened Coho salmon), the unintentional increase occurred during the spring and summer of 2013.

Mosquito populations took advantage of the breeding habitats formed by shallow pools remaining at the restored and recovering Ni-les'tun restoration site. The pools were created inadvertently as filled ditches subsided or as ruts left by equipment collected water.

Though mosquitoes are present in tidal wetlands all along the Oregon coast, and many similar tidal wetland restoration projects have been constructed over the past 30 years, no other tidal wetland restoration effort has experienced this issue at such a large scale.

In 2014, USFWS undertook adaptive management measures to eliminate the pools and reduce mosquito breeding populations at the site with very favorable results. For more information about this mosquito issue, see the Bandon Marsh Wildlife Refuge website: http://www.fws.gov/oregoncoast/bandonmarsh/Mosquito.html .

Understandably, this problem became very controversial among local residents (the local mosquito population was truly unacceptable by anyone's standards). Some called for the re-diking and draining of the site (e.g., Taylor 2014). It should be noted that it's common for habitat restoration projects to require adjustment after restoration construction is completed; adjustments are typically made to redirect natural processes in such a way that ensures the full recovery of the restored habitat without adversely affecting neighboring landowners or local residents.

In the case of the Ni-les'tun project, the mosquito population explosion in 2013 was addressed through adaptive management methods, because local residents were so adversely affected there was not time to wait for natural processes to reduce the mosquito populations. Over time, natural habitat recovery processes will develop and ultimately control mosquito populations on their own (i.e., the shallow pools would have slowly filled with sediments and salt marsh vegetation, eliminating mosquito breeding habitat; populations of aquatic mosquito larvae predators would have grown with the availability of mosquito larvae as prey).

Ecosystem Service	*Example of Human Benefit*	*Avg. Value (adj. 2015 \$ $ha^{-1} yr^{-1}$)*	*Avg. Value in Project Area* (Adj. 2015 \$)
Disturbance regulation	Storm protection and shoreline protection	\$2,912	\$28,944,310
Waste treatement	Nutrient removal and transformation	\$10,605	\$105,410,167
Habitat/refugia	Fish and shrimp nurseries	\$268	\$2,663,831
Food production	Fishing, hunting, gathering, and aquaculture	\$738	\$7,335,474
Raw materials	Fur trapping	\$257	\$2,554,494
Recreation	Fishing, hunting, and birdwatching	\$1,042	\$10,357,133
TOTAL		**\$15,822**	**\$157,265,409**

Table 6. Average annual value of ecosystem services associated with one hectare of tidal wetlands (1 hectare = 2.47 acres). Dollar values were adjusted for inflation from original data, presented in 1994 dollars (Costanza et al. 1997). This calculation was done using the United States Department of Labor Inflation Calculator, which uses the Consumer Price Index to adjust for inflation over time (BLS n.d.). Estimates for the average annual value of services associated with tidal wetlands in the project area are calculated using a total wetland acreage of 24,564, which is the sum of wetland areas of all systems from the 2003 National Wetlands Inventory (USFWS 2003). It should be noted that the valuation methods in Contanza et al. 1997 are not universally accepted by all economists (see Bockstael et al. 2000). Data: Costanza et al. 1997 Table and caption modified from Gedan et al. 2009

There's no evidence to suggest it will be necessary to go to the massive expense of re-diking and re-draining tidal wetland restoration projects to control mosquito populations when natural processes/ecosystem functions occurring in the wetlands themselves maintain acceptable populations at no cost to coastal communities. To our knowledge, the naturally functioning ~300 acre tidal wetland just to the southwest of the Ni-les'tun project site (also part of the Bandon Wildlife Refuge), which local residents have been living next to for many decades, has never been a source of mosquito-related controversy.

Researchers have long recognized the value of tidal wetlands. As mentioned, tidal wetlands provide many important "ecosystem services," including processing and cycling of sediments and nutrients, improving water quality, buffering human communities from floods and destructive waves, providing critical rearing habitat for juvenile fish and crabs, and facilitating recreational activities like hunting, fishing, and wildlife watching (Cornu 2005d; Gedan et al. 2009; Portnoy and Giblin 1997).

In many cases, these ecosystem services have direct ties to high-value economic goods. For example, experts estimate that over 75% of all commercially and recreationally caught fish species depend on estuaries and tidal wetlands at some point in their life cycles (Norse 1993; USEPA 1995; Cornu 2005d). In other cases, these services are "non-market" goods, and although the economic value of these goods are not revealed by a market price, it's widely accepted that they have real (and sometimes substantial) economic value (Cummings et al. 1986; Mitchell and Carson 2013; Champ 2003; Costanza 1997)(Table 6). For example, tidal marshes promote biodiversity (e.g., by providing habitat structure as well as distributing nutrients, detritus, seeds, and eggs)(Cornu et al. 2005a). Although "biodiversity" itself cannot be bought and sold in a marketplace, the persistence of a variety of plants and animals that support economic activity (e.g., hunting, fishing, wildlife watching) must have some value, because if they did not, "consumers" of these non-market goods (e.g., hunters, anglers, and wildlife watchers) would have no reason to purchase goods and services associated with those activities (e.g., fuel, fishing tackle, hunting and wildlife-watching optics, licensing fees, etc.).

In recent decades, as the ecological importance and economic value of wetlands has become increasing recognized, public policies have been put in place to protect these resources. The cornerstone of these policies is wetland "mitigation," which requires that the loss of a wetland be offset by restoration, enhancement, or creation of new wetlands. Early indications suggest that, although these policies have slowed the pace of wetland loss, they have not met their goal of "no net loss" (Ambrose 2000).

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