RESEARCH ARTICLE

Floristic Development in Three Oligohaline Tidal Wetlands after Dike Removal

Brenda C. Clifton, W. Gregory Hood and Steve R. Hinton

ABSTRACT

The decline of salmon populations has intensified tidal wetland restoration efforts throughout the Pacific Northwest, but few results are available monitoring the trajectory of these efforts over time. In three oligohaline tidal wetlands, dike removal restored tidal influence to provide juvenile salmon rearing habitat in the South Fork Skagit River Delta, Washington, USA. This study compared up to 13 years of vegetation development in these restoration sites to reference tidal marsh sites using remote sensing and transect surveys. While native emergent plant communities and open water dominated the most recently restored site (41.6% and 39.5% cover), invasive species present prior to restoration dominated the earlier restored sites. *Typha angustifolia* (narrow leaf cattail) overran one site (60.7% cover), and *Phalaris arundinacea* (reed canarygrass) the other (40.0% cover). *Typha angustifolia* also covered 37.5% of the reference sites. Combined elevation distribution of invasive species overlapped that of native species, suggesting direct competition in this environment. Furthermore, the ability of pre-established invasive species to persist in the subsided restoration sites at elevations outside of reference occurrence ranges affected the native species elevation distributions. The authors hypothesize that despite sufficient native propagule dispersal, competition from persistent invasive species resulted in simplified community structures with reduced native herbaceous and scrub-shrub cover. In potential restoration sites dominated by non-native *T. angustifolia* and *P. arundinacea*, managers should consider their control to facilitate native species colonization. In new restoration sites where plant communities are still evolving, they should monitor invasive species cover and composition to keep levels below the that of the reference site condition.

Keywords: estuarine wetland restoration, *Phalaris arundinacea,* Skagit River, *Typha angustifolia,* vegetation elevation distribution

Restoration Recap

- Invasive species present on tidal wetland sites pre-restoration can alter the structure and function of post-restoration vegetation communities.
- Passive revegetation of restored tidal wetland sites may not be viable without aggressive invasive species control, despite the richness of species propagules available in the tidal prism.
- Remote sensing is a cost-effective way to monitor vegetation development on a landscape scale, especially when paired with ground surveys.
- Disturbance factors, such as agricultural subsidence and excess nutrient input, can persist after the reintroduction of tidal influence and favor colonization by invasive plant species.

The continued decline of salmon populations has intensified tidal wetland restoration efforts throughout the Pacific Northwest, especially because threatened juvenile *Oncorhynchus tshawytscha* (Chinook), *O. keta* (chum), and *O. kisutch* (coho) salmon rear in this habitat. Foraging in estuarine habitat while gradually acclimating to increased salinity can increase growth rates and survivorship (Claxton et al. 2013, Craig et al. 2014). Although social and political imperatives for salmon recovery motivate tidal marsh restoration in the Pacific Northwest, there is also interest in providing benefit to other species, such as migratory waterfowl (*Chen caerulescens* [snow geese], *Anas platyrhynchos* [mallards], *A. americana* [american wigeon], *A. acuta* [northern pintail], *A. carolinensis* [Green-winged Teal]), and raptors that feed on the waterfowl (*Haliaeetus leucocephalus* [bald eagles], *Falco peregrinus* [peregrine falcons], *Circus cyaneus* [northern harriers]). Many waterfowl graze

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on native sedges (Cyperaceae) in oligohaline tidal marshes, such as *Carex lyngbyei* (Lyngby's sedge), *Schoenoplectus pungens* (American three-square), *Bolboschoenus maritimus* subsp. *paludosus* (maritime bulrush), and *Eleocharis palustris* (common spikerush; Burgess 1970, Vermeer and Levings 1977), but not on invasive plant species such as *Typha angustifolia* (narrow leaf cattail) or *Phalaris arundinacea* (reed canarygrass). A focus on multi-species and ecosystem benefits of tidal marsh restoration leads to the goal of restoring native tidal marsh vegetation communities and controlling invasive non-native species.

Historically, Puget Sound river deltas contained extensive oligohaline wetlands. However, due to conversion to agricultural and urban use following diking and draining, few estuarine wetlands remained by 1900 (Collins 2000). Seventy-three percent of the tidal wetlands and channels in the Skagit Delta, the largest in Puget Sound, have been lost since 1860 (Beamer et al. 2005). As a result, many regional restoration projects target estuarine wetlands to recover salmon populations and benefit wildlife.

These projects attempt to generate diverse and sustainable plant communities by restoring tidal and riverine flooding blocked by dikes and levees. This approach assumes structural vegetation responses are directly related to hydrology. Passive revegetation relies on tidal inundation to drown existing non-native vegetation and deliver water-borne native plant propagules to revegetate the restoration site with desirable species. However, in oligohaline to fresh tidal wetlands, some vegetation may persist after tidal inundation, and sedges struggle when colonizing sites dominated by established invasive species (Hood 2013). Consequently, ecological priority affects vegetation development, with the established species preempting space and available resources, and impeding new recruits (Belyea and Lancaster 1999, Young et al. 2001).

Little funding is available to monitor restoration projects and monitoring results are rarely published (Zedler 2000b, but see Dawe et al. 2000, Cornu and Sadro 2002, Tanner et al. 2002). Thus, there are few studies comparing reference and post-restoration tidal wetland vegetation. Furthermore, what studies do exist for the Pacific Northwest focus mainly on salt marshes, with little information on oligohaline environments (Burg, et al. 1980, Mitchell 1981, Campbell and Bradfield 1988, Lefstad and Fonda 1995, Cordell et al. 1999, Woo et al. 2011). There is much to learn about the mechanisms and rates of tidal wetland development after restoration—information important for setting benchmarks in adaptive management plans and tracking specific restoration goals.

This article describes vegetation development in three oligohaline tidal wetlands in the Skagit Delta, Washington State, USA, after breaching or removing dikes to restore tidal inundation. We compared up to 13 years of vegetation development, including species composition, community structure, successional relationships, and plant distributions, in restoration and reference sites to evaluate the following hypotheses:

1) Passive revegetation of restored oligohaline wetland sites without additional planting efforts results in species composition and community structure similar to the reference sites over time.
2) Restoring natural hydrologic conditions and tidal regime at oligohaline wetland sites, while eliminating the anthropogenic disturbances, results in the establishment of native vegetation communities and reduces non-native vegetation.
3) Patterns of vegetation distribution with respect to elevation at restored oligohaline wetland sites will progress towards zonation patterns observed at reference sites with similar salinity and tidal ranges.

Methods

Site Description

This study focuses on three restoration sites along the South Fork distributary of the Skagit River: Deepwater Slough, Milltown Island, and Wiley Slough and two reference sites within 300 m of the restoration sites (Figure 1). These wetlands are oligohaline, with soil pore water salinity ranging from 0 to 3 ppt during normal to low river flow (225 to 450 m^3/s [W.G. Hood, Skagit River System Cooperative, unpub. Data]). Sediments are organic-rich silt, silty clay, and fine sand. Semi-diurnal tides in the delta have a maximum range of 4.5 m; higher high spring tides can inundate lower portions of the tidal wetland with up to 1.5 m of water. The restoration sites differ in elevation, duration of diking, and historic land uses.

The reference sites were historically at similar elevations and subject to the same tidal inundation range as the restoration sites (Collins et al. 2003). However, recent airborne light detection and ranging (LiDAR) remote sensing of the Skagit Estuary (Grossman and Crump 2012) indicates that Wiley Slough and Deepwater Slough are lower in elevation than the reference sites even though they are both upstream. The reference sites' surfaces ranged from 2.7 to 3.7 m relative to Mean Lower Low Water (MLLW), whereas, at Wiley Slough it ranges from 2.2 to 2.7 m MLLW, and at Deepwater Slough from 2.0 to 3.5 m MLLW. This suggests subsidence due to hydrologic modification and agricultural practices. However, the Milltown Island wetland surface elevation ranges from 3.1 to 4.5 m MLLW. In this case, the wetland surface seems to have subsided less than the other restoration sites, possibly due to a shorter diking duration or reduced agricultural use.

Restoration partners removed the dikes surrounding Deepwater Slough in 2000 (Table 1) to restore 80 ha of wetlands to tidal inundation (Hood 2004). Prior to restoration, the Washington Department of Fish and Wildlife planted portions of the site in agricultural grains to attract

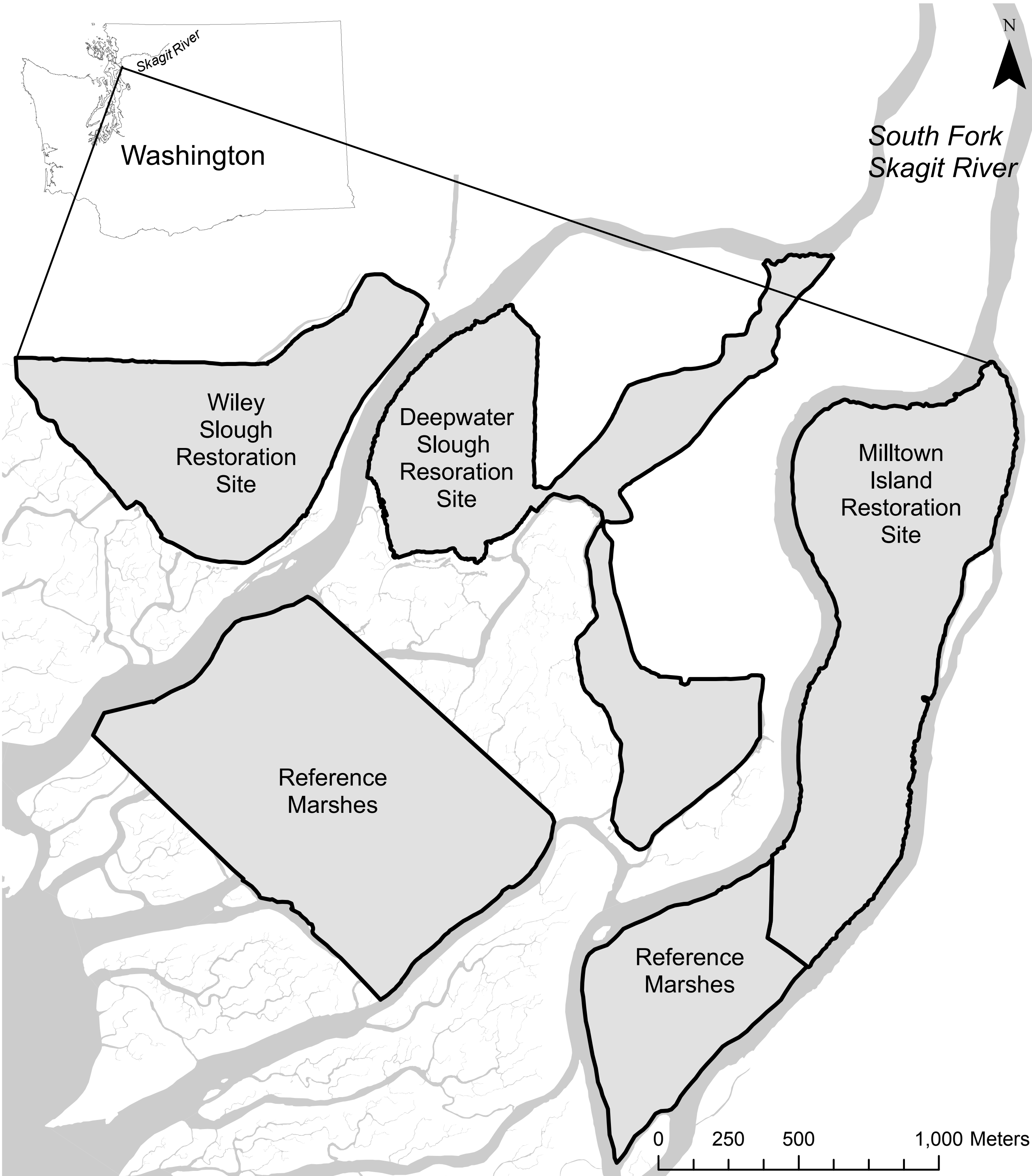

Figure 1. Location of restoration and reference sites within the South Fork Skagit River Delta, Washington, USA (WADNR 2005, WADOE 2011).

Table 1. Site restoration actions, aerial photo, and ground-truth survey timetables for three restorations and two reference sites in the Skagit River Delta, Washington State, USA.

Site	Restoration Action	Year(s) Completed	Aerial Photos Used	Ground Surveys Performed
Reference Sites	N/A	N/A	2000; 2011	2001; 2012
Deepwater Slough	Dike removal	2000	2000; 2011	2001; 2012
Milltown Island	Dike breaching/channel creation	1999; 2006–2009; 2012	2000; 2011	2012
Wiley Slough	Dike removal	2009	2000; 2013	2013

waterfowl for hunters, while invasive *Typha* spp. overran the remainder. Land owners farmed the diked portion of Milltown Island until the 1970s. The island sat fallow, overrun by invasive *P. arundinacea* until 1999, when the first of several phases of restoration began. Dike breaching and 1500 m of channel construction reintroduced tidal hydrology to over 99 ha (Hinton et al. 2010). At Wiley Slough, approximately 1600 m of dike set back in 2009 restored tidal influence to 63 ha. The Washington Department of Fish and Wildlife diked and drained the area between 1959 and 1962, planted fields in cereal grains to attract waterfowl, and then managed it as a public access property for hunting and wildlife viewing (Hinton et al. 2005).

Aerial Photo Analysis

We mapped vegetation change in a geographic information system (ArcGIS v. 10, Environmental Systems Research Institute, Redlands, CA) by digitizing orthophotos taken in 2000, 2011 and 2013: color infra-red orthophoto with 15-cm pixel resolution, taken on 28 August 2000, at a low tide of –0.58 m below MLLW; Pictometry SID orthorectified mosaic tiles with 30-cm pixel resolution, taken on 13 July 2011 at a low tide of –0.6 m below MLLW; 30-cm CIR orthophotos taken on 25 October 2011 at a low tide of –0.2 m below MLLW; and 30-cm CIR orthophotos taken on 19 September 2013 at a low tide of 0.46 m above MLLW. We used the 2011 photos for Deepwater Slough, Milltown Island, and the reference sites and the 2013 photo for Wiley Slough. We drew vegetation patches (as small as 3m diameter) at a mapping scale of 1:1000.

We manually digitized vegetation polygons from aerial photos based on perceived dominant species composition. Interpretation used visual elements (texture, height, density, soil moisture, elevation, and color range) with defined indicators for consistency (Table 2). We classified areas dominated by woody species as shrubs/trees. Common species included *Myrica gale* (sweetgale), *Spiraea douglasii* (hardhack), and *Salix hookeriana* (Hooker's willow). Two easily distinguished nonnative species classifications were *T. angustifolia* and *P. arundinacea*. We classified areas dominated by native, emergent wetland species as emergent herbaceous plants. Common species included *C. lyngbyei*, *Scirpus microcarpus* (small-flowered bulrush), *Schoenoplectus tabernaemontani* (softstem bulrush), *Juncus balticus* (Baltic rush), *Agrostis stolonifera* (creeping bentgrass), *Alisma plantago-aquatica* (water-plantain), and *Oenanthe sarmentosa* (Pacific water-parsley). We classified areas dominated by bare ground or submerged aquatic plants as open water. Common species encountered were *Elodea canadensis* (Canadian waterweed), *Myriophyllum* sp. (watermilfoil), and *Ruppia maritima* (ditch-grass). We classified agricultural fields, parking lots, and roads as anthropogenic modifications. Delineation began with separating out the elements that were easiest to distinguish: anthropogenic modifications and shrubs/trees, followed by, *T. angustifolia*, *P. arundinacea*, open water, and emergent herbaceous plants.

Table 2. Description of indicators used to interpret shrubs and trees (S&T), cattail (CT), reed canarygrass (RCG), emergent herbaceous plant (EHP), open water (OW), and anthropogenic modification (AM) classifications from aerial photos.

	Texture				Height				Density			Moisture			Elevation			Color Range: True Color								Color Range: Infra-red				
	Very Smooth	Smooth	Rough	Very Rough	Short	Medium	Tall	Very Tall	Sparse	Dense	Densest	Very Wet	Moist	Dry	Lowest	Medium	Highest	Yellow	Green	Purple	Blue	Brown	Grey	Black	Dark Red	Brown	Red	Pink	White	Blue
S&T			X	X		X	X	X	X	X	X		X	X		X	X		X				X	X				X	X	X
CT			X	X			X			X	X	X	X		X	X		X		X	X	X				X	X	X		
RCG				X		X	X			X	X		X	X		X	X	X	X									X	X	
EHP			X		X				X	X	X	X				X														
OW	X				X							X			X						X	X							X	X
AM	X				X									X		X	X					X	X				X	X		

Field Surveys

Field surveys took place in 2001, 2012, and 2013. We used the survey data to assess map accuracy; we also used the 2012 and 2013 survey data to determine species composition, richness, diversity, and elevation distribution. The 2001 ground surveys provided baseline data for the Deepwater Slough project monitoring plan as part of a modified before/after/control/impact (BACI) design (Stewart-Oaten et al. 1986, Underwood 1992). We sampled vegetation in 1-m radius circular plots at 15-m intervals along three transects in Deepwater Slough totaling 1,295 m. Plot centers occurred on transects and started haphazardly. We sampled seven additional transects, containing 103 plots over 1,753 m, in an adjoining tidal wetland. Transects ran parallel and perpendicular across the areas to sample all elevations. We used cover classes (Daubenmire 1959) to determine dominant species, as defined by Chappell (2006). A real-time kinematic global positioning system (Leica SmartRover, Leica Geosystems, San Ramon, CA; 3-cm horizontal and vertical resolution; referenced to the 1988 North American Vertical Datum [NAVD88]), recorded center coordinates and plot elevation. In August of 2012, we surveyed ten transects each at Milltown Island, Deepwater Slough, and the reference sites. In August of 2013, we surveyed six transects at Wiley Slough. ArcGIS software randomly selected the transect starting points and each transect extended in a random direction. Transects intersected by impassable channels continued in new, random directions. We established circular plots (1.5-m radius, corresponding with the remote mapping minimum patch diameter) at 3-m intervals along the 200-m transects. We visually estimated dominant species (Daubenmire 1959, Chappell 2006) within the circular plot to assess remote sensing accuracy. In addition, we used the point intercept method (Caratti 2006) to record species at the circle center. Point-intercept data determined species composition, richness, diversity, and elevation distribution. A global navigation satellite systems smart antenna (Leica Viva GS14—GNSS, Leica Geosystems, San Ramon) with a real-time kinematic rover (Leica GS15 RTK, Leica Geosystems, San Ramon) recorded center point coordinates and elevations with 2-cm accuracy. The Washington State Reference Network obtained the Global Positioning System daemon corrections. Plant nomenclature followed WTU Herbarium (2017).

We determined species diversity (Shannon's Diversity Index; Shannon and Weaver 1949) and species richness for each site. Box-and-whiskers plots compare the distribution of species occurrence along elevation gradients. The middle horizontal line separating the upper box (2nd quartile) and lower box (3rd quartile) shows the median elevation for each species. Vertical lines denote two standard deviations; asterisks are points in the distribution tails.

Mapping Accuracy

We used the dominant species recorded at all the survey transect points to assess mapping accuracy. Field survey data was limited from 2011, so we delineated an additional area outside the reference and restoration sites on the 2000 map to increase the accuracy assessment sample size. We sampled 155 ground truth points for the 2000 map accuracy assessment (20 for *P. arundinacea*, 18 for shrubs and trees, 30 for emergent herbaceous plants, 87 for *T. angustifolia*, 0 for open water, and 0 for anthropogenic modifications) and 1803 ground truth points for the 2011/2013 map accuracy assessment (213 for *P. arundinacea*, 163 for shrubs and trees, 333 for emergent herbaceous plants, 857 for *T. angustifolia* 237 for open water, and 0 for anthropogenic modifications).

Error matrices determined the producer's, user's, and overall accuracy of the 2000 and 2011/2013 vegetation maps (Congalton 1991). Overall accuracy is the number of image pixels classified correctly divided by the total number of image pixels classified. This accuracy is the easiest to calculate and understand, but it does not analyze accuracy within the different classes. Producer's and user's accuracy analyze accuracy from two perspectives. The producer's accuracy is the map accuracy from the perspective of the map maker. It is the number of accurately classified pixels within a vegetation class divided by the overall number of points within that class in the ground survey. It measures the probability that a feature on the ground is correctly classified on the map. The user's accuracy is the accuracy from the perspective of the map user. It is the number of accurately classified pixels within a class divided by the total number of pixels in that class in the map. It measures the probability that a class on the map represents that category on the ground. Low producer's accuracy results in the underestimation of cover, whereas low user's accuracy results in the overestimation of cover.

We also calculated the Kappa Index (Cohen 1960) and confidence interval. A kappa analysis produces another measure of agreement or accuracy, a KHAT statistic (an estimate of kappa), which is computed as:

$$k = \frac{N\sum_{i=1}^{r} x_{ii} - \sum_{i=1}^{r}\left(x_{i+} \times x_{+i}\right)}{N^2 - \sum_{i=1}^{r}\left(x_{i+} \times x_{+i}\right)}$$

Where r is the number of rows in the error matrix, x_{ii} is the number of observations in row i and column i, x_{i+} and x_{+i} are the marginal totals of row i and column i, respectively, and N is the total number of observations (Bishop et al. 1975). Overall accuracy only incorporates the major diagonal and excludes omission and commission errors. KHAT accuracy, on the other hand, indirectly incorporates the off-diagonal elements as a product of the row and column marginals. A 90% confidence variable was calculated using:

Table 3. Change in reed canarygrass (RCG), shrubs and trees (S&T), emergent herbaceous plant (EHP), cattail (CT), open water (OW), and anthropogenic modification (AM) percent cover in the restoration and reference sites over eleven or thirteen years.

	Reference Marsh Cover (%)		Wiley Slough Cover (%)		Deepwater Slough Cover (%)		Milltown Island Cover (%)	
	2000	2011	2000	2013	2000	2011	2000	2011
RCG	3.6	4.1	1.0	1.9	22.2	4.7	43.4	40.0
S&T	33.1	32.1	44.3	5.8	27.9	15.0	48.9	47.2
EHP	33.3	19.5	0.2	41.6	1.2	12.8	1.6	5.8
CT	22.1	37.5	1.0	12.5	31.0	60.7	5.4	6.6
OW	7.9	6.8	3.0	35.9	6.6	6.8	0.7	0.4
AM	0.0	0.0	50.5	2.3	11.1	0.0	0.0	0.0

$$k \pm t_{1-\frac{0.9}{2},n-1}\sqrt{\frac{N-n}{N}\frac{k(1-k)}{n-1}}$$

Where N is the total number of 3-m patches classified, n is the number of samples taken, and k is the estimated KAPPA accuracy. Microsoft Excel 2010 ran all statistical calculations.

Results

Ground Surveys

Cumulative species richness of the reference sites was 37 and Shannon's Diversity Index was 2.30. This is higher than or equal to all the restoration sites. Deepwater Slough had species richness and diversity of 25 and 1.41; Milltown Island had 26 and 1.96; Wiley Slough had 20 and 2.30. The reference sites were co-dominated by *T. angustifolia* and the nitrogen-fixing shrub, *M. gale* (35.3% and 15.8% relative abundance; Supplementary Table S1; vegetation alliances follow Chappell 2006). *Carex lyngbyei* and *P. arundinacea* were prominent species (11.8 and 6.1% relative abundance). Deepwater Slough was co-dominated by *T. angustifolia* and *P. arundinacea* (61.9% and 19.7% relative abundance). Milltown Island was co-dominated by *P. arundinacea* and *C. lyngbyei* (39.8% and 18.0% relative abundance). *Typha angustifolia* was a prominent species (14.2% relative abundance). Co-dominant species at Wiley Slough included the submerged aquatic plant, *Callitriche heterophylla* (diverse-leaved water-starwort; 23.4% relative abundance), and *A. plantago-aquatica* (15.4% relative abundance). *Typha angustifolia* was a prominent species (14.4% relative abundance).

Remote Mapping

Invasive species increased at both the reference and restoration sites, and the vegetation communities in the restoration sites shifted to reflect tidal influence. From 2000 to 2011, shrub/tree and *P. arundinacea* cover in the reference sites was stable; however, *T. angustifolia* cover increased from 22.1% to 37.5% (Table 3; Figure 2). This increase occurred at the expense of emergent herbaceous cover, which declined from 33.3% to 19.5%. Vegetation change was modest in the oldest and highest elevation restoration site, Milltown Island, where the biggest change from 2000 to 2011 was an increase in emergent herbaceous cover from 1.6% to 5.8%. This change involved native sedges displacing from 43.4% to 40.0% cover. Vegetation change in the newer and lower elevation Wiley Slough and Deepwater Slough restoration sites was more extensive. Most of Deepwater Slough was covered with *T. angustifolia* in 2011, limiting sedge and rush colonization to the eastern lobe. *Phalaris arundinacea* cover declined from 22.2% to 4.7%, while *T. angustifolia* increased from 31.0% to 60.7%. Shrub/tree cover predominantly remained on upland areas, such as the dike remnants, not on the wetland surface. Wiley Slough was still largely in the process of colonization by ruderal wetland species four years after dike removal. Vegetation change from 2000 to 2013 consisted of a large die-off of shrubs/trees following restoration of tidal inundation, and loss of anthropogenic cover, with a concomitantly large increase in emergent herbaceous and open water cover. Open water increased from 3.0% to 35.9% and emergent herbaceous cover from 0.2% to 41.6%. *Typha angustifolia* increased from 1.0% to 12.5%. Vegetation covered only 64.1% of the site, due to ponding from poor development of tidal drainage channels.

Mapping Accuracy

Overall accuracy of the 2000 map was 89%. The highest producer's and user's accuracies were 100% for emergent herbaceous plants and 94% for shrubs/trees (Table 4). The lowest producer's accuracy was 0% for open water. We mapped ground truth plots dominated by open water as emergent herbaceous in 100% of the image pixels. The next lowest producer's accuracy was shrubs/trees at 65%. We mapped ground truth plots dominated by shrubs/trees as emergent herbaceous and *T. angustifolia* in 4% and 30% of the image pixels. The lowest user's accuracy was 83% for emergent herbaceous plants. We mapped shrubs/trees, *T. angustifolia*, and open water ground truth plots as emergent herbaceous in 3%, 7%, and 7% of the image pixels, respectively. The Kappa Index was 82.8% and the confidence interval was 94.4%.

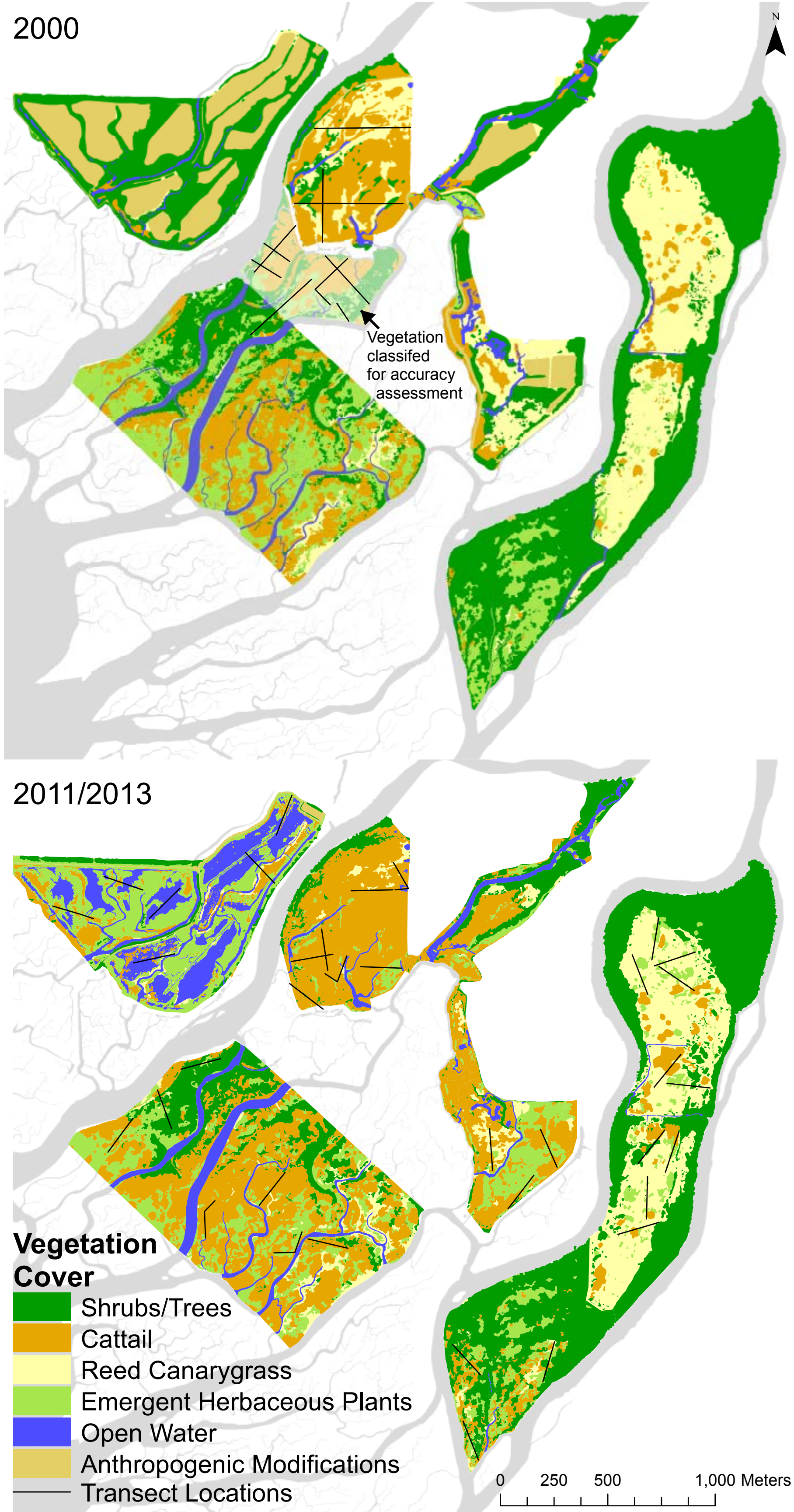

Figure 2. Maps of cover and ground truth transect locations at the restoration and reference sites in 2000 and 2011/2013 within the South Fork Skagit River Delta, Washington, USA.

Table 4. Error matrix for 2000 Deepwater Slough, Milltown Island, Reference Sites, and Wiley vegetation mapping. RCG is the reed canarygrass classification, S&T is the shrubs and trees classification, EHP is the emergent herbaceous plant classification, CT is the cattail classification, OW is the open water classification.

		Classified in Ground Truth as:						
		RCG	S&T	EHP	CT	OW	Total	Accuracy
Classified in Orthophoto as:	RCG	19	0	0	1	0	20	95%
	S&T	0	17	0	1	0	18	94%
	EHP	0	1	25	2	2	30	83%
	CT	2	8	0	77	0	87	89%
	OW	0	0	0	0	0	0	—
	Total	21	26	25	81	2	155	—
	Accuracy	90%	65%	100%	95%	0%	—	89%

The overall accuracy of the 2011/2013 map was 85% (Table 5). The highest producer's and user's accuracies were 94% for open water and 91% for *T. angustifolia*. The lowest producer's accuracy was 76% for emergent herbaceous plants. We mapped ground truth plots dominated by emergent herbaceous points as *P. arundinacea*, *T. angustifolia*, and open water in 7%, 8%, and 7% of the image pixels, respectively. The lowest user accuracy was 73% for *P. arundinacea*. We mapped emergent herbaceous plants and *T. angustifolia* ground truth plots as *P. arundinacea* in 12% and 15% of the image points, respectively. The Kappa Index was 77.9% and the confidence interval was 93.8%.

Elevation Distribution

Wiley Slough transect points occupied the lowest elevations, followed by Deepwater Slough, then the reference sites. Milltown Island sample points congregated at the highest elevations. In the reference sites, the interquartile range of *T. angustifolia* and *C. lyngbyei* spanned the lowest elevations, where they were the dominant species; *S. tabernaemontani, P. arundinacea,* and *S. microcarpus* were subdominants at these elevations (Figure 3). The interquartile range of *M. gale* spanned the highest elevations. The combined interquartile occurrence of the invasive species, *T. angustifolia* and *P. arundinacea* almost completely overlapped the interquartile elevation distributions of all the native species, suggesting direct competition between native and invasive species. Vegetation distributions at Milltown Island were the most like the reference sites. The interquartile ranges of *T. angustifolia*, *C. lyngbyei*, and *S. tabernaemontani* spanned the lowest elevations; the interquartile ranges of *S. microcarpus* and *P. arundinacea* spanned the highest elevations. For each species, elevation ranges at Deepwater Slough were notably lower than those of the reference sites. The interquartile ranges of *T. angustifolia* and *C. lyngbyei* spanned the lowest elevations in the Deepwater Slough site; the interquartile ranges of *P. arundinacea* and *M. gale* were found in the middle elevations; and the interquartile range of *S. microcarpus* occupied the highest elevations. This differed from the reference sites, where *M. gale* was found at the highest elevations; however, the sample size was low for *M. gale* at Deepwater Slough. Wiley Slough had the greatest differences with the reference sites. The interquartile ranges of *C. lyngbyei* and *S. tabernaemontani* spanned the lowest elevations; the interquartile range of *P. arundinacea* occupied the highest elevations; and *T. angustifolia* was in the middle elevations. The sample size was small for all species at Wiley Slough due to the large expanses of open water.

Discussion

We found that vegetation development in the three restored oligohaline tidal wetlands compared to the reference sites did not support the stated hypotheses during the study period:

Table 5. Error matrix for 2011 Deepwater Slough, Milltown Island, and Reference Sites and 2013 Wiley vegetation mapping. RCG is the reed canarygrass classification, S&T is the shrubs and trees classification, EHP is the emergent herbaceous plant classification, CT is the cattail classification, OW is the open water classification.

		Classified in Ground Truth as:						
		RCG	S&T	EHP	CT	OW	Total	Accuracy
Classified in Orthophoto as:	RCG	155	2	25	31	0	213	73%
	S&T	3	142	8	10	0	163	87%
	EHP	18	4	272	33	6	333	82%
	CT	26	20	29	777	5	857	91%
	OW	0	3	26	31	177	237	75%
	Total	202	171	360	882	188	1803	—
	Accuracy	77%	83%	76%	88%	94%	—	85%

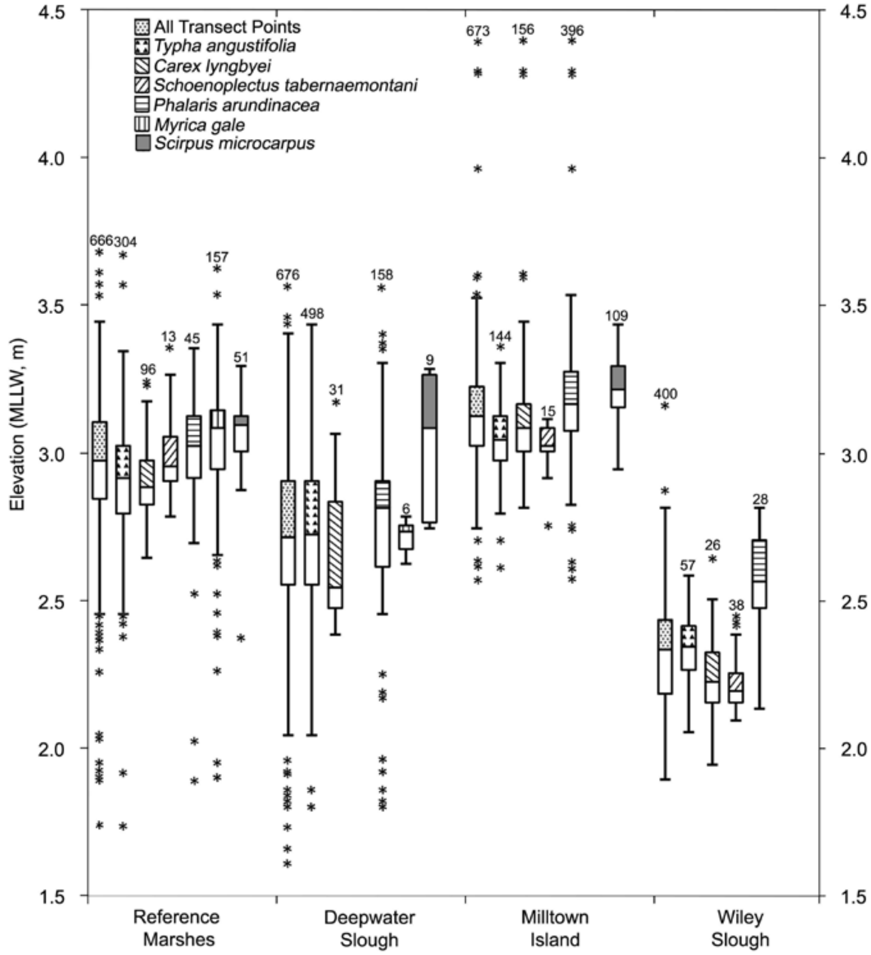

Figure 3. Box-and-whiskers plot of key species vegetation elevation distributions along survey transects throughout the restoration and reference sites. Sample sizes are written above the outliers. The sum of the species points differs from the total number of transect points due to open water (no species) or overlapping species at the same point. The median elevation for each species is shown by the middle horizontal line separating the upper box (2nd quartile) and lower box (3rd quartile). Asterisks denote outlying individuals.

1) Passive revegetation of restored oligohaline wetland sites without additional planting efforts resulted in a different species composition and community structure than the reference sites due to the history of disturbance and invasive species present prior to restoration.
2) Restoring natural hydrologic conditions and tidal regime at oligohaline wetland sites, while eliminating the anthropogenic disturbances, resulted in native vegetation establishment in available niches, but non-native vegetation persistence and expansion limited native plant colonization.
3) Patterns of vegetation distribution with respect to elevation at restored oligohaline wetland sites differed from the zonation patterns observed at reference sites due to competitive pressure from invasive species.

Species Composition, Distribution, and Structure

Vegetation composition, distribution, and structure differed between the restoration and reference sites for the period studied. Even though the colonization diversity at Wiley Slough suggests sufficient propagule dispersal to support reference plant communities, passive revegetation resulted in different dominant species and a reduced tidal shrub community in the restoration sites. Furthermore, restoring natural hydrologic conditions and tidal regime, and reducing anthropogenic disturbances, did not reduce non-native vegetation. Invasive species persisted at Deepwater Slough and Milltown Island, impeding native plant community development. Thirteen years after restoration, vegetation distribution patterns relative to elevation do not appear to be progressing towards the reference sites. Differences between restoration and reference conditions may be due to pre-restoration vegetation composition, land subsidence, excessive nutrient inputs, and underdeveloped tidal channels that impede hydraulic exchange.

Four years post-restoration, many species colonizing Wiley Slough are absent from the reference sites. However, vegetation communities and physical conditions are still developing. The dominant emergent species, *A. plantago-aquatica*, is a facultative ruderal species (Abernethy and Willby 1999) that should decline as sedimentation fills in ponded areas. Similar palustrine emergent wetland species occurred after the reintroduction of tidal influence at both Fisher Slough, a subsided, oligohaline site in the Skagit River Delta (Boyd and Clifton 2015) and Spencer Island, an oligohaline tidal wetland in the nearby Snohomish River estuary (Tanner et al. 2002). *T. angustifolia* cover has increased, but it remains below reference site levels.

Zonation by elevation is a pattern of plant species distribution in tidal marshes (Jefferson 1974, Pennings and Callaway 1992, Janousek and Folger 2014). Although many additional factors influence vegetation distribution (including salinity, flooding frequency, soil texture, soil organic matter, competition, facilitation, herbivory, disturbance, tidal range, and climate, among others [Ewing 1983, Snow and Vince 1984, Bertness and Leonard 1997, Crain et al. 2004]), elevation has been an effective predictor of vegetation development within the oligohaline zone of the South Fork Skagit Delta (Hood 2013).

Vegetation distribution relative to elevation at Deepwater Slough and Milltown Island differed from the zonation patterns observed at the reference sites. Interspecific competition plays a large role in structuring the vegetation of low salinity marshes (Crain et al. 2004), and persistent invasive species may be forcing native species recruitment to areas above and below their interquartile elevations in the reference sites. At Deepwater Slough, *T. angustifolia* occurrence overlapped most of the sampled elevations and native *S. microcarpus* and *C. lyngbyei* interquartile occurrence extended above and below reference marsh ranges, to elevations outside of the *T. angustifolia* interquartile occurrence. Similarly, *P. arundinacea* occurrence covered most of the sampled elevations at Milltown Island, and *S. microcarpus* occurred at higher elevations than in the reference marsh. This may be the result of competitive pressure from *P. arundinacea*, the higher elevation range sampled at Milltown Island, or *S. microcarpus* colonizing elevations normally occupied by *M. gale*.

Pre-established non-native species, such as *T. angustifolia* and *P. arundinacea*, may persist outside of their normal niche after tidal reintroduction because established plants are often more resistant to environmental stresses than new propagules (see example in Hood 2013). Additionally, non-natives may persist or spread because native species are colonizing the restoration sites slowly, producing low competition stress. In the reference sites, competition is higher because *T. angustifolia* and *P. arundinacea* are invading established wetland.

Phalaris arundinacea persistence at Milltown Island may be correlated with the missing scrub-shrub component; in higher elevations, *P. arundinacea* establishment is limited by the shade from trees and shrubs. The *M. gale* shrub community—that covered over 30% of the reference sites—is absent from the restoration sites. Milltown Island historically consisted of equal parts estuarine and palustrine shrub vegetation (Collins et al. 2003), but *M. gale* establishes on nurse logs, which are slow to recruit after diking and agricultural use (Hood 2007a).

Contributing Factors

Subsidence of the restored sites influenced the composition of colonizing plant communities. Farming causes subsidence through soil compaction and microbial oxidation, while dikes prevent sediment deposition (Taylor 1983, Ingebritsen et al. 2000, Williams and Orr 2002.) Subsided and poorly drained areas are susceptible to *T. angustifolia* invasion, which has greater germination and establishment rates than *Carex* species in prolonged flooding (Hall and Zedler 2010, Boers et al. 2007, Tanner et al. 2002). *Carex lyngbyei* prefers regularly drained areas for establishment in tidal wetlands (Eilers 1975, Bradfield and Porter 1982, Bradfield and Campbell 1986). Subsided areas may build up sedimentation as the restoration sites develop: subsided areas disappeared after three years in the restored oligohaline tidal wetland at Spencer Island (Tanner et al. 2002). However, once established, *T. angustifolia* alter the wetland's structure, reduce plant community diversity, and influence a range of ecosystem functions (Mitchell et al. 2011).

Excess nutrient input may also be affecting community structure by promoting the spread of invasive species in the restoration and reference sites. Total phosphorous levels measured upstream, in the city of Mount Vernon, were moderate to high in 15 of 22 monitored years (WADOE 2015). WADOE (2014) classified portions of Puget Sound as impaired due to dissolved oxygen levels, correlated with anthropogenic dissolved inorganic nitrogen (Ahmed et al. 2014). Nutrient availability is an important determinant of community invasibility (Davis et al. 2000). Invaders often have adaptions, such as large leaves and high growth rates, which are advantageous when excess nutrients are available (Daehler 2003). Nutrient addition reduces *T. angustifolia* stress and increases its height in low saline environments (Smith et al. 2015), where competition for light is high (Pidwirny 1990). *P. arundinacea* also increases yields with nutrient enrichment (Wetzel and van der Valk 1998, Green and Galatowitsch 2001).

Incomplete hydrologic connections may also be impeding vegetation community development. Underdeveloped tidal channel networks affect vegetation community composition (Bradfield and Porter 1982, Zedler et al. 1999, Sanderson et al. 2000), and tidal marsh restoration projects in the Pacific Northwest average five-fold fewer channel outlets than do reference marshes (Hood 2015). Deepwater Slough and Milltown Island monitoring (E. Mickelson, Skagit River System Cooperative, unpub. data) show that tidal channel length was 86% and 33%, respectively, of the length predicted by an empirical model based on the marsh island area (Hood 2007b). The persistence of *P. arundinacea* at Milltown Island may also suggest an incomplete hydrologic connection. In low elevation oligohaline wetland restorations, including Deepwater Slough and Spencer Island, *P. arundinacea* cover typically decreases with restored tidal inundation (Tanner et al. 2002).

Implications for Restoration

Remote sensing is a cost-effective way to monitor vegetation development on a landscape scale that requires few resources and minimal field time. In this study, remote sensing analyzed revegetation and invasive species abundance in 420 ha of oligohaline tidal wetland more efficiently than the field surveys. We used manual digitization based on our skill set and available software. Future tidal wetland monitoring could incorporate automated and hybrid automated-user image classification, which is increasingly accessible and accurate (Tuxen et al. 2008, Klemas 2011, Zhang et al. 2011). Object-based image analysis has shown promise in classifying wetland habitat (Dronova 2015, Guo et al. 2017), especially in overcoming pixel-based difficulties in separating the within-class spectral variations of higher resolution imagery (Yu et al. 2006, Liu and Xia 2010). Incorporating automated image classification techniques in monitoring reduces analysis time on large projects and could increase mapping frequency and scale.

Established invasive species impacted passive revegetation and post-restoration vegetation communities despite the richness of propagules in the tidal prism. Integrating different techniques to control of non-native species for several years prior to restoration could have increased cover, richness, and diversity of native marsh species. For *P. arundinacea*, repeated mowing reduces productivity (Geber 2002) and removes litter, a feedback mechanism that suppresses competing species (Annen 2011). Repeated mowing also reduces *T. angustifolia* abundance (Hood 2013). Applying herbicide (alone [Annen et al. 2005] and with disking [Annen 2010]) reduces *P. arundinacea* biomass and interplanting native shrubs, such as *Salix* spp., and can shade it out over time (Kee et al. 2006, Seebacher 2008). Sowing native short-lived cover crops and perennial target species also impedes reestablishment (Iannone and Galatowitsch 2008).

These options can be impractical and labor intensive after tides return. Access is complicated, mowing is restricted to hand-held brush cutters or specialized heavy equipment, certain herbicides are prohibited (grass-specific sethoxydim and fluazifop), disking is muddy and compacts soils, and sown seed can float away. Preparing the site prior to dike removal saves effort and future expense.

Reference sites provide the template to design restoration models and set performance standards (Callaway et al. 2000), but in this study it remains uncertain if a functional system comparable to the restoration sites is an attainable target (even with invasive species removal and greater hydrologic connectivity). There is little information available about tidal wetland restoration pathways and endpoints (Simenstad and Thom 1996, Simenstad and Cordell 2000, Zedler 2000b, Borja et al. 2010), including the rebuilding of marsh surface elevations after subsidence (Simenstad et al. 1999). Restoration sites can take decades even centuries to reach functional equivalency (Zedler and Callaway 1999, Zedler 2000a, Zedler 2000b, Borja et al. 2010), which is longer than the typical monitoring study. During this period, pressures affecting the site (level of disturbance, on-going nutrient loading, invasive species recruitment, etc.) can alter recovery pathways. More monitoring data on the long-term development of restoration sites would clarify restoration trajectories given specific disturbance regimes.

This study indicates that removing ecosystem process barriers may be insufficient to restore highly disturbed tidal wetlands. Land managers need to take an active approach that addresses invasive species prior to dike removal and establishes an adaptive management plan to monitor the sites and intervene if invasive species reach levels greater than the reference sites. As a result of this study, the Washington Department of Fish and Wildlife began mowing *P. arundinacea* and *T. angustifolia* at all three restoration sites and project managers began studying the feasibility of increasing hydrologic connectivity (through increased dike removal and channel creation), and planting *M. gale* on Milltown Island. On a watershed scale, policies need to reduce nutrient pollution to slow the spread of non-native species like *T. angustifolia* and *P. arundinacea* in both the restored and reference sites (Deegan et al. 2012, Smith et al. 2015).

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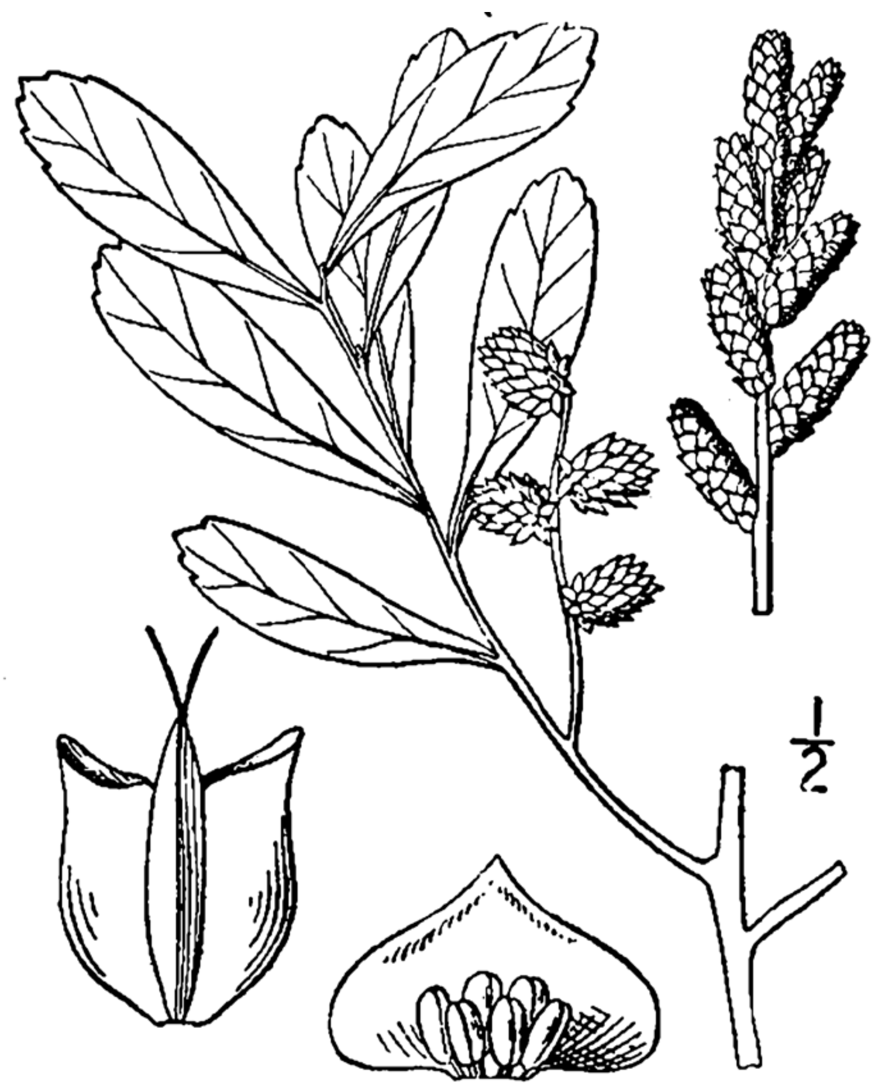

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