

Fir Island Farms 3rd-year Post-Restoration Tidal Marsh Monitoring Report

9 December 2019

W. Gregory Hood
Skagit River System Cooperative, LaConner, WA

prepared for
Washington Department of Fish and Wildlife



A view of an incipient tidal channel on the Fir Island Farms restoration site. The channel has been eroded by tidal flushing, but constrained in its development by clay hard pan. The view is at low tide from an excavated tidal channel to which the incipient channel is a tributary. The soil auger is for scale.

Contents

Introduction	1
Methods	2
<i>Vegetation</i>	2
<i>Tidal channels</i>	2
<i>Surface elevation change</i>	3
<i>Hard pan sampling</i>	3
Results	4
<i>Vegetation community composition</i>	4
<i>Observed vs. predicted vegetation</i>	8
<i>Experimental seeding</i>	10
<i>SETs and sedimentation</i>	16
Discussion	17
<i>Adaptive management and cattail control</i>	17
<i>Experimental seeding treatments</i>	17
<i>SET measures of sedimentation</i>	18
<i>Hypothesis: tidal channels affect vegetation colonization</i>	18
Recommendations	20
References	22

Figures

Figure 1. Dominant plant species	3
Figure 2. Relative abundance of plant species	4
Figure 3. Non-native cattail 2017 vs. 2018 vs. 2019	5
Figure 4. Mean elevations of plant species	7
Figure 5. Frequency distribution curves of plant species	8
Figure 6. Predictive vegetation maps	9
Figure 7. Predictive vegetation map for <i>Cotula coronopifolia</i>	10
Figure 8. Dominant native sedges	11
Figure 9. Tidal channels	13
Figure 10. Shallow tidal channel erosion	14
Figure 11. Soil texture profiles	14
Figure 12. Surface exposure of clay soils	15
Figure 13. Surface elevation table (SET) locations	16

Introduction

The Fir Island Farms (FIF) tidal marsh restoration site, owned by the Washington Department of Fish and Wildlife (WDFW), was returned to daily tidal inundation three years ago following dike removal at its seaward boundary to allow tidal exchange and dike construction at its landward boundary to protect other properties from the tides. An important goal of this project was to provide habitat for native fish and wildlife, including tidal channels and marsh vegetation. While, some tidal channels were actively excavated, passive channel development through tidal erosion was expected over much of the site. Likewise, most of the site was allowed to recruit plant propagules passively through tidal transport, under the assumption that sufficient seed sources would be available from adjacent marsh. A portion of the site was experimentally seeded with native plant species to accelerate colonization by desirable species and inhibit colonization by non-native species. A particular concern was the possibility that the site might be colonized by the invasive non-native cattail (*Typha angustifolia*). Consequently, vegetation monitoring has been a principal focus of habitat monitoring to evaluate the efficacy of the experimental seeding and to quantify any invasion by cattail. Channel network development and sediment accretion have been secondary considerations due to expectations of slower process rates; these are reported here for the first time.

The first year of monitoring bore out the concern over cattail invasion; much of the site was being colonized by small clumps of non-native cattail. WDFW responded by implementing control measures (spot treatment) during the first year post-restoration to try to rein in this threat. As will be shown below, this effort has been successful in significantly reducing cattail presence two years after restoration and almost completely eliminating cattail within the restoration site three years after restoration. However, large proportions of the site are still either bare mud or sparsely vegetated. Thus, continued monitoring and adaptive management is necessary in the next few years to ensure that cattail is kept under control until desired native sedge species can become firmly established and pre-empt cattail invasion.

This report follows up on the first- and second-year vegetation monitoring reports. It describes vegetation colonization on the site three years after restoration of tidal processes, evaluates the seeding experiment, evaluates the cattail invasion threat, and makes recommendations for future site monitoring to support continued adaptive management. Additionally, changes in planform channel geometry, sediment composition, and accretion rates are reported for the first time. Finally, this report also makes some preliminary comparisons with the nearby zis a ba restoration site in the northern portion of the Stillaguamish Delta, which has shown strongly contrasting rates of vegetation development. An hypothesis is suggested to explain vegetation differences between FIF and zis a ba, which could potentially lead to adaptive management measures to improve vegetation development on the site.

Methods

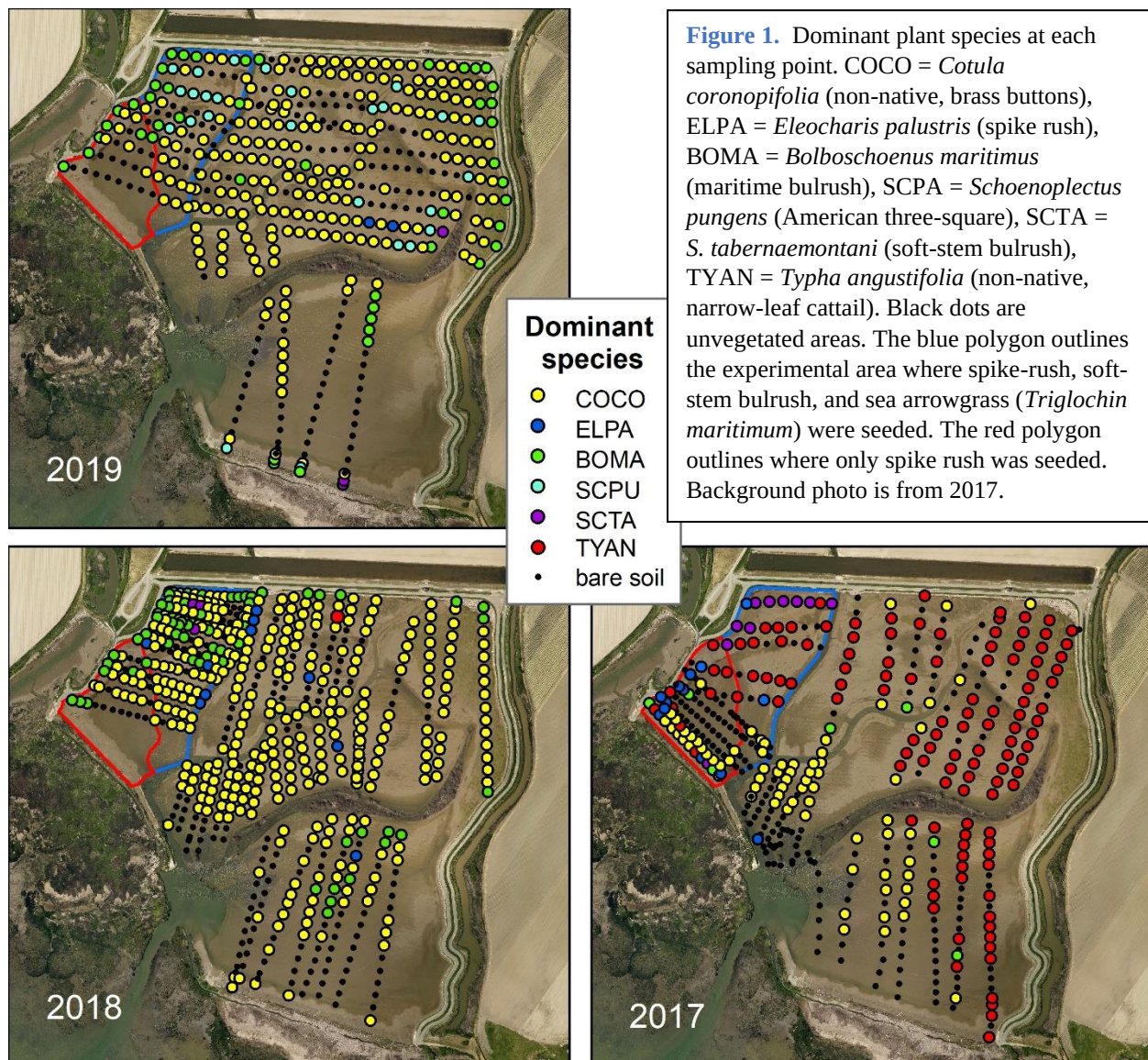
Vegetation

Vegetation was sampled within the restoration site during late summer of 2019, three years after dike removal to restore tidal inundation to the site. The procedure was the same as that used in 2017 and 2018 monitoring. A real-time kinetic global positioning surveyor (RTK-GPS; Leica SmartRover; 3-cm horizontal and vertical resolution) was used to measure the ground surface elevation at points along haphazard transects throughout the site (Fig. 1). Sampling points were spaced at intervals of 30 paces (approximately 22 m) along the transects. At each sample point the dominant vegetation species was estimated visually within a 0.5-m radius of each RTK-GPS survey point. The three most abundant sub-dominant species (if any were present) were also noted. All surveys included checks against an established permanent tidal benchmark to verify instrument accuracy. Elevation data were referenced to the 1988 North American Vertical Datum (NAVD88). A total of 424 points were surveyed in this fashion, following up on the 426 and 589 points surveyed in 2017 and 2018, respectively.

As part of site restoration, western portions of the site were experimentally seeded with native plants under the hypothesis that this would accelerate colonization of the site by the seeded species and pre-empt colonization by non-native species (Fig. 1). The northern portion of the experimental site was seeded with three species, *Eleocharis palustris* (spike-rush), *S. tabernaemontani* (soft-stem bulrush), and *Triglochin maritima* (sea arrowgrass); the southern portion of the experimental site was only seeded with *E. palustris*. A normal approximation of a binomial test (Z-test), using a correction for continuity, was used to compare the proportion of surveyed sample points colonized by a particular species within the experimental versus the control areas (Zar 1984).

Tidal channels

Tidal channel planform was digitized in GIS primarily from two 2019 aerial photos, one was flown in a small airplane by Pictometry, a commercial provider of high quality aerial orthophotos, while the other was flown in a drone by WDFW staff. The strength of the Pictometry photo was that it is well georeferenced, because Pictometry have flown such photos for the county every two years since 2007 with consistent georeferencing. The strength of the WDFW drone photos is that the photo resolution is 3.5 cm, compared to 23 cm for Pictometry. However, the drone photo had some noticeable errors in georeferencing at small scales, although these small alignment errors had no significant effect on the results. Reference was also made to Pictometry photos from 2015 (pre-restoration; 30-cm pixel resolution) and 2017 (23-cm pixels), as well as 2016 photos of unknown provenance (20-cm pixels).



Surface elevation change

Surface elevations were measured annually from 2016 to 2019 by WDFW personnel at three locations within the restoration site by means of a surface elevation table (SET). SET methodologies have been standardized and described elsewhere (Cahoon et al. 2000; WDFW 2015). Rates of elevation change were estimated by linear regression of surface elevation against time.

Hard pan sampling

Depth to hard pan and the thickness of the hard pan was measured by sampling soils with a soil auger to a depth of up to 1.2 meters, the length of the auger handle. Greater depths

were reached by standing in and sampling from channel bottoms. Soil texture was determined using the ribboning technique (Ritchey et al. 2015). Rust color was also used as an indication of clay presence for surface exposures of hard pan, because positively charged iron ions bind to negatively charged clay particles.

Results

Vegetation community composition

Bare ground was extensive at lower elevations; it had a mean elevation of 1.74 m (NAVD88) in 2019, compared to 1.75 m in 2018 and 1.67 m in 2017. In 2017 the most abundant plant species within the restoration site was the invasive, non-native cattail (*T. angustifolia*), which dominated 52% of the vegetated sampling points, and was present as a dominant or subdominant in 64% (Fig. 2). Following control efforts to reduce the presence of this invasive species, cattail was dominant in only 0.5% of vegetated sampling points in 2018, and was present as a dominant or subdominant in only 8%. In 2019, cattail was not dominant in any portion of the site; it was present in only two spot samples (0.7% of vegetated points), though it was occasionally visible in areas not randomly sampled—along the higher margins of the site, and only as a subdominant presence in areas dominated by maritime bulrush (*Bolboschoenus maritimus*). This striking reduction in the occurrence of cattail was visibly evident when walking through the restoration site (Fig. 3).

In 2019, as in 2018, the dominant species was the non-native forb, brass buttons (*Cotula coronopifolia*), which had been the second-most abundant species in 2017. In 2017, it was dominant in 30% of the vegetated plots; in 2018 it was dominant in 80%; in 2019 in 73%. This spatial increase from 2017 to 2018 and 2019 was accompanied by an increase in the mean elevation at which this species was found, from 1.85 m NAVD88 in 2017 to 1.98 m and 1.99 m in 2018 and 2019, respectively. The 7% decrease in dominance by brass buttons from last year was accompanied by an increase in the presence of three-square sedge (*S. pungens*) from complete absence on the site in 2017 and 2018 to dominance in 9% of the vegetated point samples in 2019 (Fig. 2).

Concomitant with the dramatic decrease in cattail cover from 2017 to 2018 and 2019, there was a significant increase in native sedges (maritime bulrush, soft-stem bulrush, and spike-rush), with three-square sedge being a notable newcomer in 2019 (Figs. 2, 4, 5). Maritime bulrush has remained the most abundant of the sedges. It was found as a dominant species in 16% of the vegetated sampling points in 2019, compared to 15% in 2018 and 3% in 2017. As a dominant or subdominant, it was found in 19% of the vegetated sampling points in 2019, compared to 35% in 2018 and 12% in 2017. This decrease in overall presence was matched by the increased presence of the new colonizer, three-square sedge, which was present in 18% of vegetated points. The elevations at which maritime bulrush was found were higher in 2019 than in 2018 (by 10 cm) and 2017 (by 16 cm) (Fig. 4). Despite being the most

common sedge within the restoration site, maritime bulrush was not intentionally seeded anywhere on the site. Instead it has recruited naturally, presumably through tidal dispersal of seeds from the nearby tidal marsh. The same is true for the newly arrived three-square sedge.

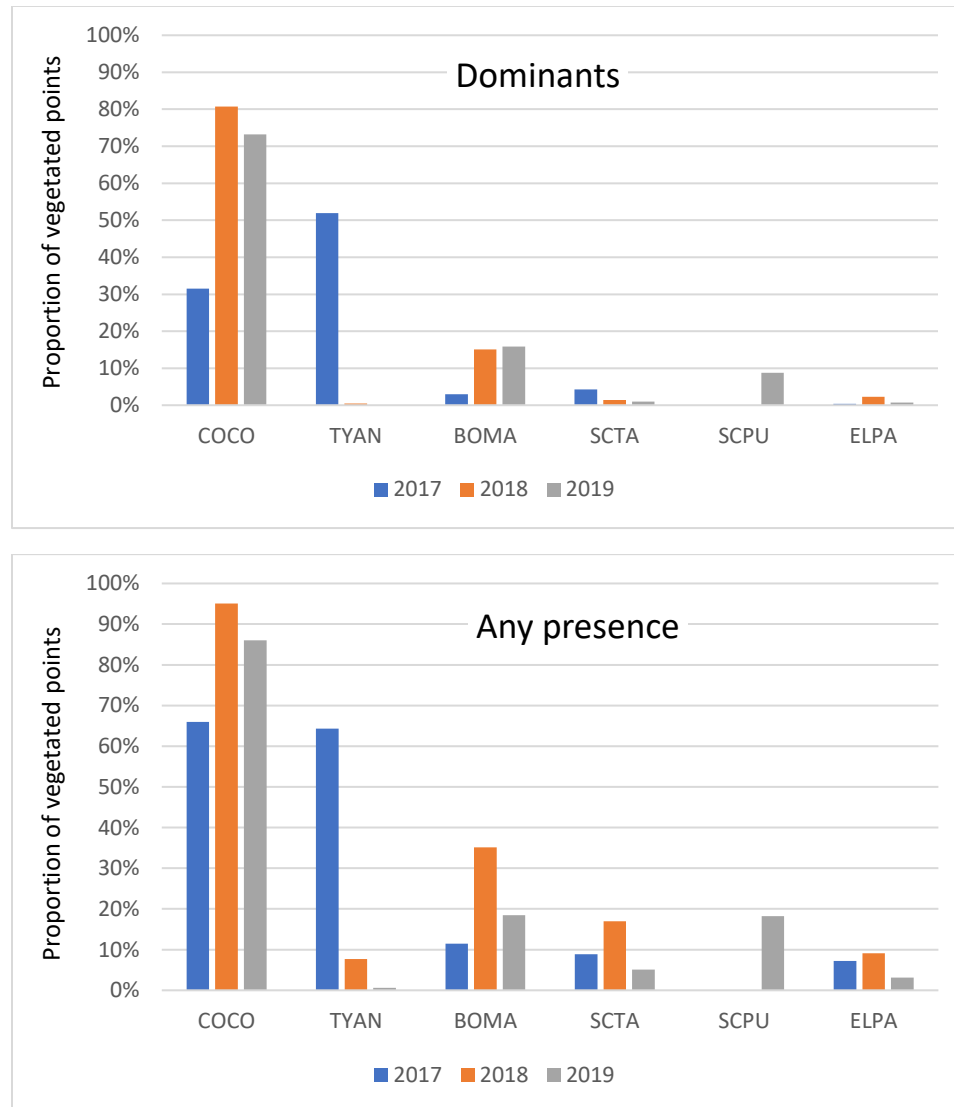


Figure 2. Relative abundance of non-native brass buttons (*Cotula coronopifolia* COCO), non-native cattail (*Typha angustifolia* TYAN), maritime bulrush (*Bolboschoenus maritimus* BOMA), soft-stem bulrush (*S. tabernaemontani* SCTA), American three-square (*S. pungens*), and spike-rush (*Eleocharis palustris* ELPA) amongst vegetated sample points, when the species are dominant (top frame) or merely present in any amount (bottom frame).



Figure 3. Contrasting presence (red circles) and absence (black dots) of non-native cattail (*Typha angustifolia*) for RTK-GPS survey points within the restoration site in 2019 vs. 2018 vs. 2017. Areas experimentally seeded with native sedges are outlined by red and blue polygons. Background photo is from 2017.



Softstem bulrush declined in relative frequency as a dominant, from 4.3% in 2017 to 1.4% in 2018 and to 1.0% in 2019. As a dominant or subdominant it nearly doubled in frequency from 9% to 18% of vegetated sampling points, before dropping to 5% in 2019. The frequency decline as a dominant may be the consequence of brass buttons becoming more abundant, while softstem bulrush generally continued to occur as isolated stalks and only occasionally as small patches. As a dominant, spike-rush increased in relative frequency from 0.4% in 2017 to 2.3% in 2018, before dropping back to 0.7% in 2019. Its overall presence increased from 7% in 2017 to 9% in 2018, before dropping back to 3% in 2019. This variation could be attributable to small sample size effects for this species, rather than indicating any successional patterns.

The native sedges were found at elevations similar to that previously published for oligohaline marsh in the active South Fork Skagit Delta (Hood 2013), and to that of individuals in

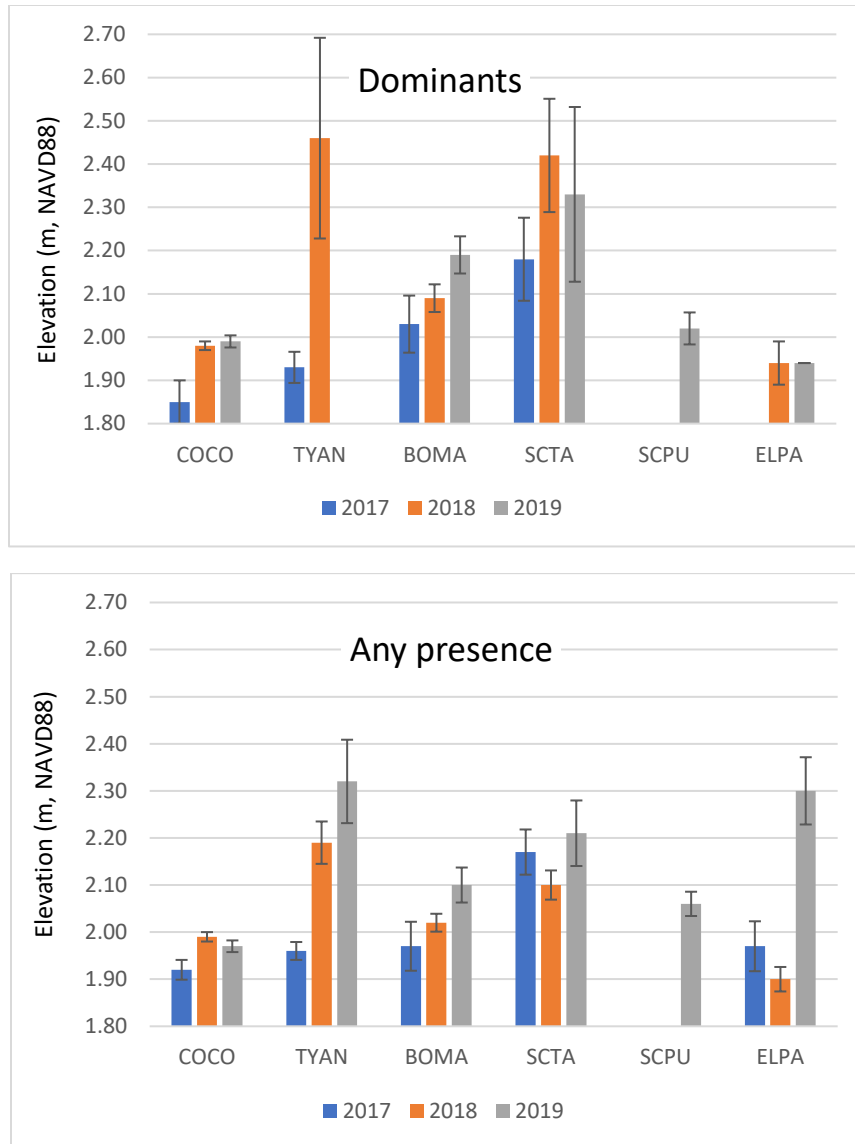


Figure 4. Mean elevations of non-native brass buttons (*Cotula coronopifolia* COCO), non-native cattail (*Typha angustifolia* TYAN), maritime bulrush (*Bolboschoenus maritimus* SCMA), soft-stem bulrush (*S. tabernaemontani* SCTA), American three-square (*S. pungens*), and spike-rush (*Eleocharis palustris* ELPA) when the species are dominant (top frame) or merely present in any amount (bottom frame). Error bars are $\pm$ one standard error.

the adjacent reference marsh seaward of the restoration site, sampled in 2015. For example, softstem bulrush was found at a mean elevation of 2.20 m in the reference marsh adjacent to the restoration site, at 2.30 m in the South Fork reference marshes, and at 2.10 m within the restoration site. Spike-rush was found at a mean elevation of 1.80 m in the South Fork reference marshes, 2.09 in the site-adjacent reference marsh (but $n = 3$), and at 1.90 m within the restoration site. Maritime bulrush was found at 2.10 m in a more distant bay fringe reference marsh, not present in the site-adjacent or South Fork reference marshes, and at 2.02

m within the restoration site. However, there was a notable exception in 2019 with the new appearance of three-square sedge at a mean elevation of 2.02 m in the restoration site ($n = 26$). This contrasts starkly with its mean distribution of 1.40 m in the reference marsh ($n = 114$). Three-square is likely growing at higher than normal elevation because sparse vegetation provides little competition to exclude it from these areas (*viz.* Bertness and Ellison 1987).

Observed vs. predicted vegetation

A predictive vegetation model (PVM) was developed from RTK-GPS transects of the adjacent reference marsh (*cf.* Hood 2013). Probability distributions for dominance by particular species in a series of elevation bins were used to recode lidar rasters in a GIS from representations of elevation to representations of probability of dominance by a particular plant species. Only four species were sufficiently common across a range of elevations to develop interesting distribution curves (Fig. 5). Three-square sedge predominated at lower elevations; it was the only species below 1.36 m (NAVD88) and steadily declined with increasing elevation until it was entirely absent at 2.30 m. Three-square was replaced at higher elevations

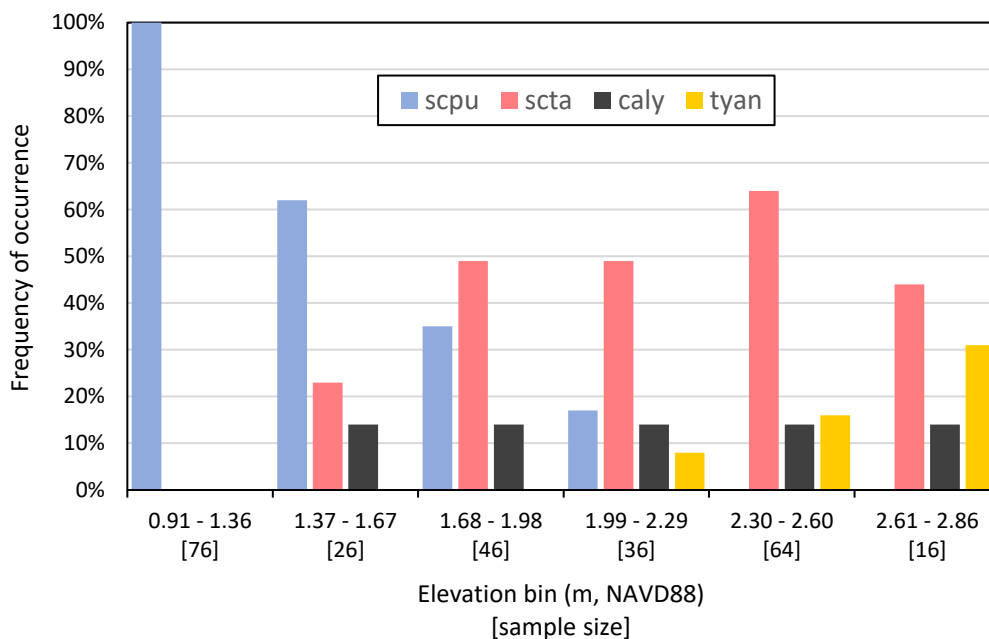


Figure 5. Frequency distribution curves relative to marsh surface elevation for the occurrence of American three-square, *Schoenoplectus pungens* (SCPU); soft-stem bulrush, *S. tabernaemontani* (SCTA); sedge, *Carex lyngbyei* (CALY); and non-native cattail, *Typha angustifolia* (TYAN) as dominants in RTK-GPS sample points in the reference tidal marsh. Sample sizes, i.e., the number of RTK-GPS points in an elevation bin, are shown in brackets.

primarily by soft-stem bulrush, which increased in abundance from 1.37 m to a maximum at 2.30-2.60 m before dipping again at the highest elevation bin. Lyngby's sedge had a relatively constant frequency of occurrence above 1.36 m, ranging from 7% to 19%, with no trend with elevation, for an average frequency of occurrence of 14%. Non-native cattail occurred only at higher elevation, increasing linearly in frequency from 1.99 m to 2.86 m.

The PVM is very informative when compared to observed vegetation colonization. For example, three-square sedge is predicted to be present throughout the restoration site, and to be particularly abundant at the lower elevations on the site (60%-70% frequency of dominance). Instead, three-square is absent where it should be most common and present where it should be least common (Fig. 6). The low-elevation areas where three-square should

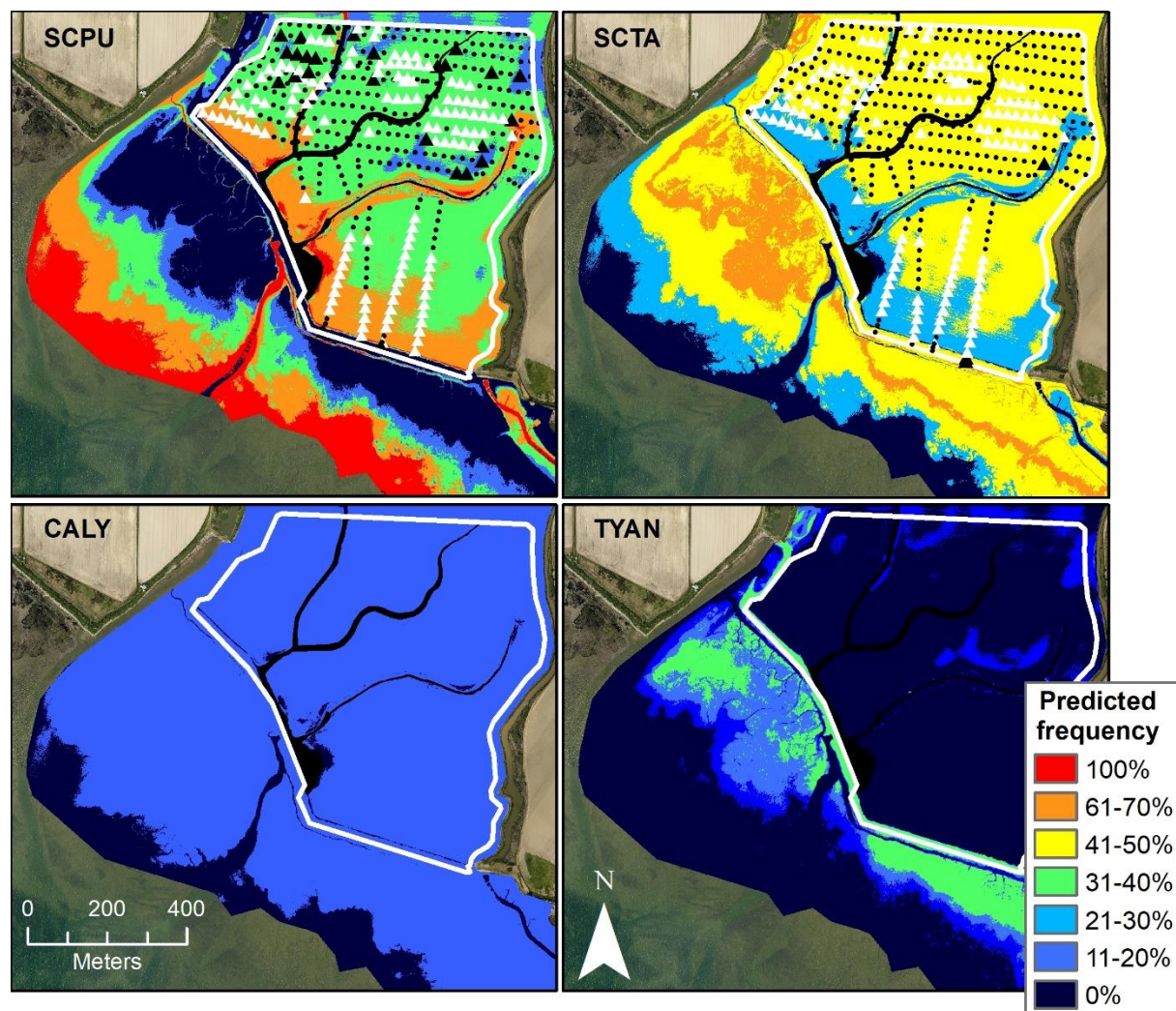


Figure 6. Predictive vegetation maps for areas of dominance by American three-square, *Schoenoplectus pungens* (SCPU); soft-stem bulrush, *S. tabernaemontani* (SCTA); sedge, *Carex lyngbyei* (CALY); and non-native cattail, *Typha angustifolia* (TYAN). Observations of dominance by the modeled species (black triangles), bare ground (white triangle), and other vegetation (black points) are shown for comparison. Neither Lyngby's sedge nor non-native cattail were observed as dominants on the site.

be abundant is entirely unvegetated. This stark contrast between predicted and observed distributions of three-square is an indication that the restoration site is behaving very oddly with regard to vegetation colonization.

The PVM predicts that soft-stem bulrush should also be abundant (40%-50% frequency of dominance) over most of the site, the exceptions being the low-elevation areas where three-square should abound. In the reference marsh soft-stem bulrush forms large mono-specific patches. On the restoration site it was found to be dominant in only three RTK-GPS sample points in 2019, but not yet forming large patches anywhere. Lyngby's sedge has not yet colonized the restoration site, but it is predicted to be a secondary component of the mature vegetation community. Non-native cattail is predicted to be absent on most of the site and have low probability of dominance in only the highest areas of the restoration site. This likely explains why early control efforts have been so successful—site elevations were never favorable for this species.

The non-native species, brass buttons was only encountered twice as a dominant in RTK-GPS sample points in the adjacent reference marsh. Within the elevation bin in which it was found its frequency of dominance in the reference marsh was only 4%. Within this same elevation bin in the restoration site, brass buttons was the most commonly occurring dominant species sampled (Fig. 7). Within this elevation bin, brass buttons appeared to be generally absent in locally low topography (as indicated by temporary ponding in aerial photos and confirmed by field observations) that drained into newly eroding shallow tidal channels. This suggests that these areas may be zones of relatively high scour that prevent seed germination. A 2 x 2 contingency analysis, bare/vegetated by topo-low/topo-high, was used to evaluate this hypothesis within this elevation bin, where the category “vegetated” included the presence of any species, not merely brass buttons. The result indicated a significant association between local topography and vegetation presence/absence ($\chi^2_{[1, n = 225]} = 28.907, p < 0.00001$); local topographic lows were 2.4 times more likely to be bare than vegetated, while local topographic highs were 3 times more likely to be vegetated than bare, and most of the vegetation, as previously mentioned, consisted of brass buttons.

Experimental seeding

For the third year in a row, no evidence was found that experimental seeding facilitated spike-rush or sea arrowgrass colonization (Fig. 8). In the area that was seeded only with spike-rush, no spike rush was encountered in 13 vegetated sampling points. In the area seeded with a mixture of spike-rush, softstem bulrush, and sea arrowgrass, no spike rush was encountered in 44 vegetated sampling points. In the unseeded control area that relied only on natural recruitment, 10 of 217 (4.6%) vegetated sampling points contained spike rush. Experimental seeding also did not facilitate sea arrowgrass establishment. No individuals of sea arrowgrass were found in sampling points in the mixed seeding site, while 2 of 217 (0.9%) vegetated sampling points in the control site contained sea arrowgrass. Seeding also did not facilitate

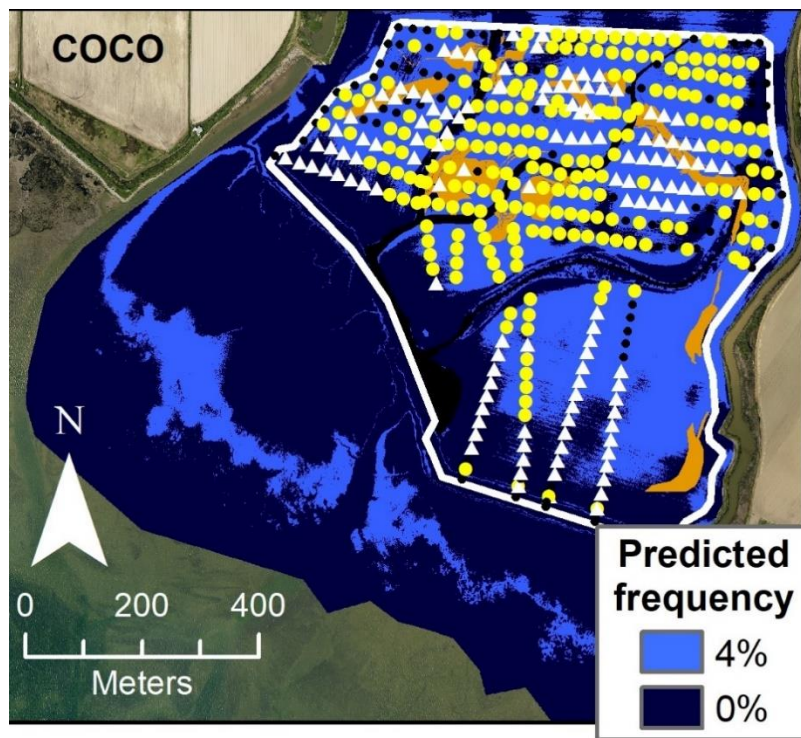
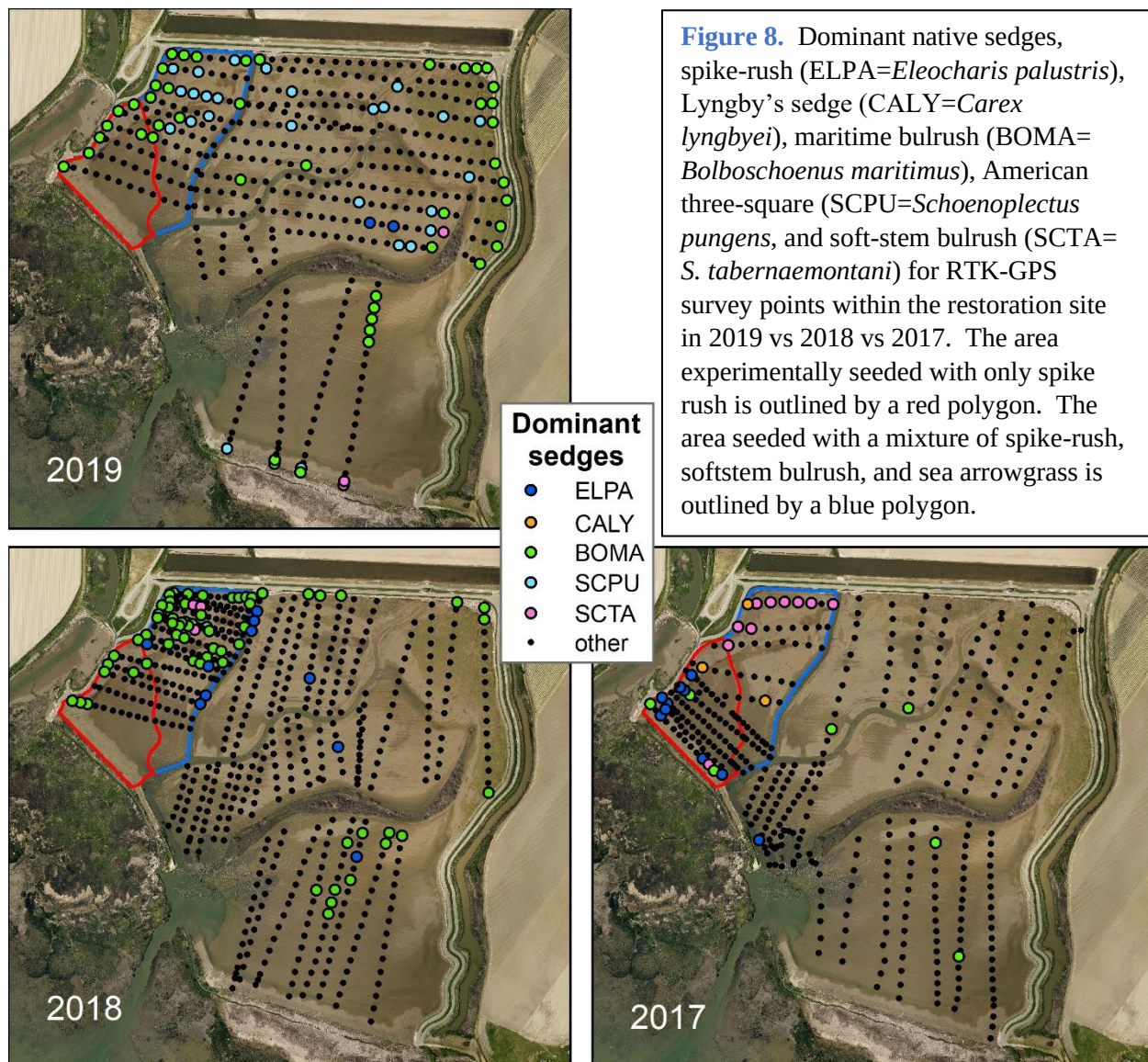


Figure 7. Predictive vegetation maps for areas of dominance by brass button, *Cotula coronopifolia* (COCO). Observations of dominance by the modeled species (yellow circles), bare ground (white triangle), and other vegetation (black points) are shown for comparison.

soft-stem bulrush colonization. Only 2 of 44 vegetated sample points in the treatment area contained this species, while 3 of 317 vegetated sample points in the control area did; there was no statistical difference ($Z = 0.79$, $p = 0.42$).

Cattail was not present in the experimental seeding area nor in the control area in 2019, but as previously reported in 2018, experimental seeding did not appear to inhibit initial colonization by the invasive non-native cattail. In 2018, cattail occurred at 7 of 126 (5.6%) vegetated sampling points of the mixed-species seeded area, while 25 of 275 (9.1%) vegetated points in the control area were colonized by cattail. The difference in cattail colonization between treatment and control areas in 2018 was not statistically significant ($Z = 0.99$; $p = 0.32$). The other abundant non-native invader, brass buttons, was also not affected by the experimental seeding effort. In 2018, it was found at 95% of the vegetated sampling points in the spike-rush only seeding area, at 93% of such points in the mixed seeding area, and at 97% of vegetated points in the control area. In 2019, brass buttons was dominant at 43% of vegetated sampling points in the spike-rush only seeding area, at 50% of such points in the mixed seeding area, and 80% of vegetated points in the control area. The difference with 2018 is likely due to increasing density of native (unseeded) sedges at higher elevations within the experimental seeding areas; these higher elevations comprise a greater proportion of the experimental seeding area than is the case for the control area on the site.



Tidal channels and hard pan

The restoration site inherited two remnant tidal channels (Fig. 9). One was a fork of Dry Slough that in an 1889 US Coast and Geodetic Survey map is shown as the main channel of Dry Slough, but which today appears to be secondary branch. The other is a channel that parallels Browns Slough, and that a map painted on the wall of the Conway Fire Department labels as Claude O. Davis Slough. Engineering plans show that Claude O. Davis Slough (mistakenly labeled as No Name Slough) was dredged about two feet deeper, to a depth that was approximately five feet below the ground surface. Drainage from Fir Island passes into this channel via a culvert and tide gate.

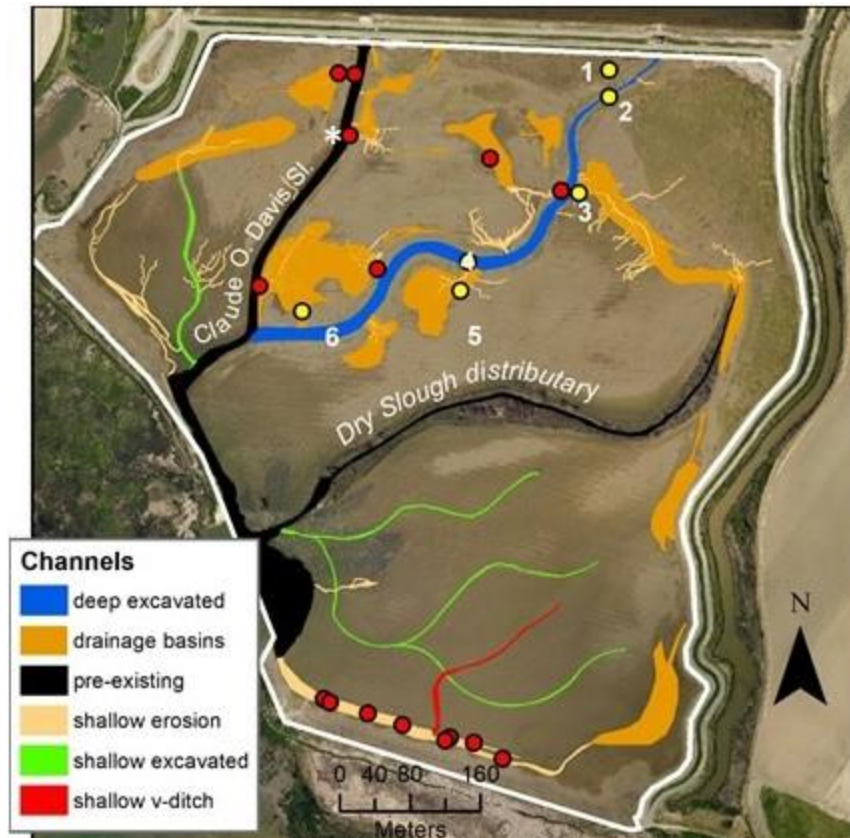
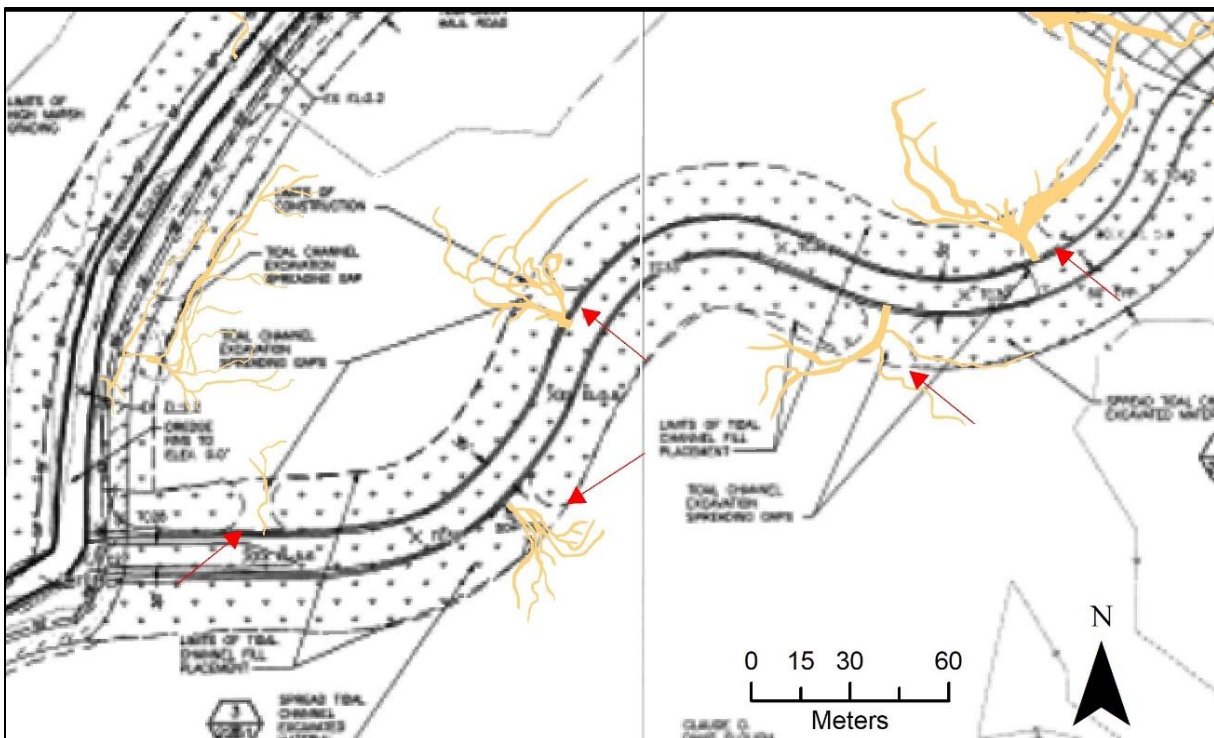


Figure 9. Tidal channels present on the Fir Island Farms restoration site in 2019. Drainage basins for channels developing from shallow tidal erosion were identified from ponded areas in aerial photos. Red dots mark areas where exposed surficial clays were easily observed during vegetation transect surveys; yellow dots are areas where sediment core samples were collected with a soil auger; white asterisk marks location of former bridge. All auger samples contained thick layers of clay from near the surface to depth.

One large channel was excavated to convey agricultural drainage from a pump house. This channel was excavated to a depth that was about three feet below the marsh surface. The dredged sediments were spread along the banks of this newly excavated channel, as well as along the banks of Claude O. Davis Slough, to mimic natural channel levees, but gaps were intentionally left in the artificial levees to allow drainage from the marsh plain into the channels and facilitate erosion of channels through these gaps.

In the newly excavated channel draining from the pump house, five such levee gaps were created, and surface drainage channels developed through tidal erosion in all five gaps (Fig. 10). Only one gap was created along the dredged channel of Claude O. Davis Slough, according to design plans, but no surface drainage developed there. A surface drainage channel did develop where plans called for destruction of an existing bridge across the slough; presumably physical disturbance of the area created topography conducive to surface drainage development.

Two additional shallow channel networks were also excavated, but only to depths of 30 to 40 cm. One is a first-order channel tributary to Claude O. Davis Slough; the other is a second-order tributary to the remnant Dry Slough distributary. These channels were excavated to such shallow depths because they were expected to be pilot channels, that would encourage further channel development through passive tidal erosion. The anticipated development has yet to be observed, likely due to an extensive clay hard pan throughout the site (see below). A third very shallow channel is a re-activated v-ditch, that was excavated by a farmer for winter site drainage the year before restoration occurred. This shallow seasonal drainage ditch empties into a very shallow drainage feature that consists of a remnant borrow ditch that was part of site drainage during agricultural use, but was filled in during site restoration. The fill has either subsided since dike breaching, or has been partially eroded by tidal currents.



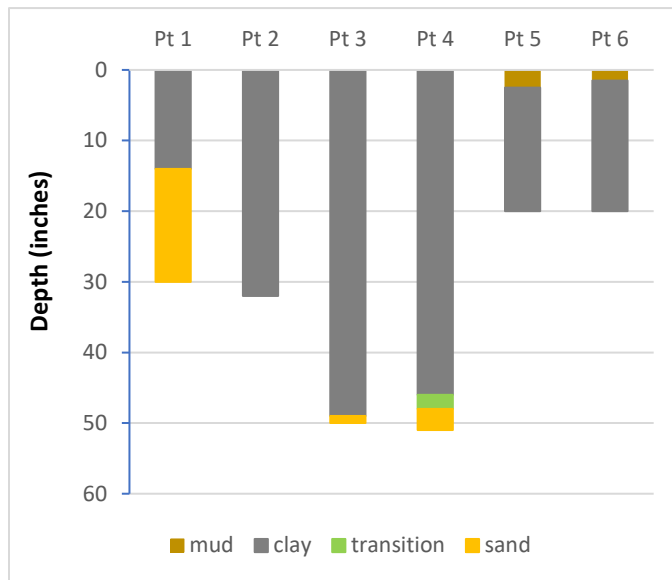


Figure 11. Soil texture profiles from soil auger samples. Deeper cores were sampled in excavated or eroding channel bottoms; shallower cores from the marsh surface, where surface clays were already exposed (Pt 2) or not (Pts 5 and 6). Sample point 1 was located in an area that according to the grading plan was graded to raise its elevation, so this profile likely reflects resorting of sediments by the grading activity. Coring depth was limited by the length of the auger (Pts 1, 3, 4) and by resistance of the clay to coring (Pts 2, 5, 6)



Figure 12. Surface exposure of clay soils in shallow tidal channels showing rusty colors diagnostic of iron-oxides bound to clay particles. Sediment in the auger is not rusty because at depth it is not exposed to oxidation. These tidally-eroded channels have limited depth due to the erosion resistance of the clay.

Other small, shallow, tidal channels that have developed on the site through tidally-driven erosion have depths that are typically less than one foot, because the restoration site is underlain by an extensive and deep layer of clay. This clay layer is resistant to erosion and thus

limits the depth to which channels can erode, thereby forcing much of the tidal exchange to occur over the marsh surface as sheet flow rather than channelized flow, which further limits development of channel networks through tidal erosion. Soil auger samples showed that the clay layer extends from near the surface of the marsh to depths as much as four feet below the marsh surface (Fig. 11). Soil texture, determined using the ribboning technique (Ritchey et al. 2015), indicated the soil was a sandy clay loam (ribbons generally varying in length from 25 to 50 mm, with gritty feel). Surface exposures of clay soils were compact and erosion-resistant, with a rust color that is diagnostic of positively charged iron-oxide particles bound to negatively charged clay particles (Fig. 12).

SETs and sedimentation

SET measurements (Fig. 13) indicated that sediment has been accreting at each site at relatively rapid rates, from 1.2 cm/yr at the lowest elevation location to 2.3 cm/yr at the highest elevation location. For perspective, this translates into gains of 1.2 to 2.3 meters over a century, enough to outpace current projections for sea level rise.

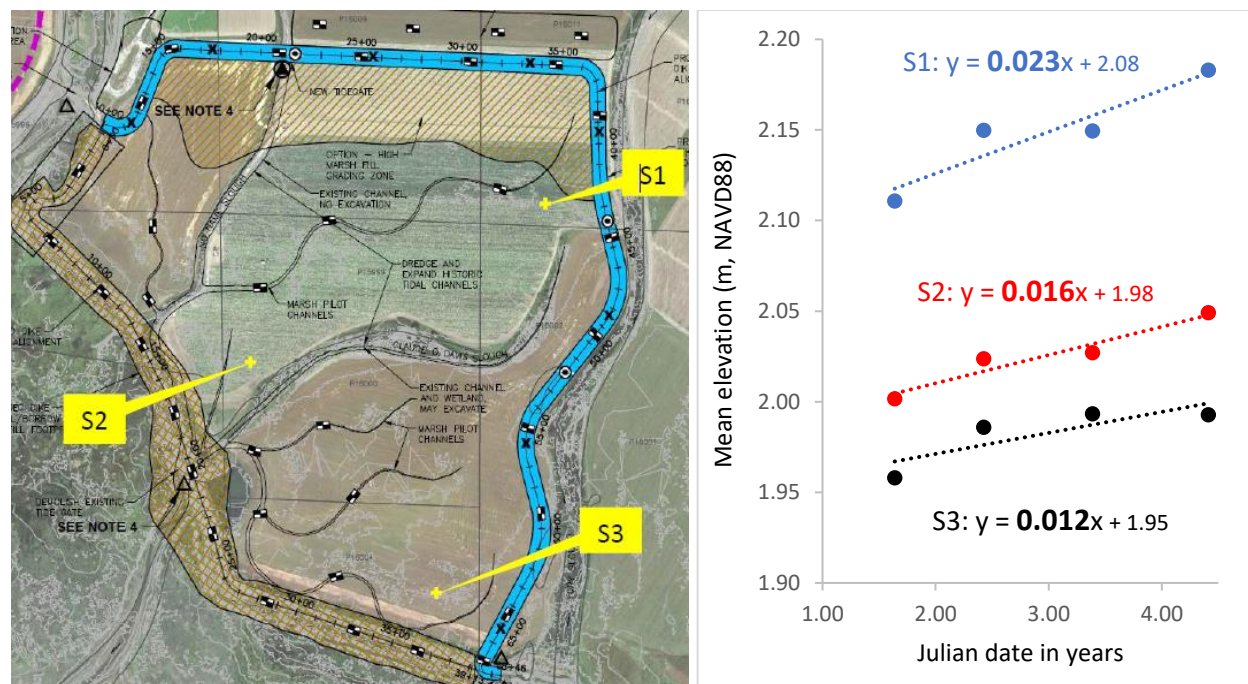


Figure 13. Surface elevation table (SET) locations (left, yellow crosses) and rates of elevation change derived from annual sampling from 2016 to 2019 (right). Note that the slopes of the linear regression lines are in bold for emphasis and represent rates of sediment accretion in meters per year.

Discussion

Adaptive management and cattail control

Site monitoring in 2017 showed that the invasive, non-native cattail was colonizing the restoration site to an alarming extent. WDFW responded to this result by undertaking a cattail suppression effort in a classic example of adaptive management. The 2018 and 2019 monitoring results indicate that this control effort almost completely eliminated cattail from the restoration site, with the exception of a few small occurrences along the higher margins of the site adjacent to the set-back dike. Control of these few patches would be prudent, as would continued monitoring of this invasive species until the native sedge community becomes more firmly established and capable of pre-empting invasion by cattail. Given that site elevations are low relative to reference marsh occurrences of infestation (*cf.* Fig. 6), continued control of non-native cattail should have a high probability of success.

Another non-native species of potential concern is brass buttons, which has extensively colonized the restoration site. While this species is present in the reference marshes of the Skagit Delta, it is relatively uncommon and found only at lower elevations in areas of intermediate salinity. If native sedges become more abundant within the restoration site, they will outcompete brass buttons and displace it. The competitive mechanism would be shading of brass buttons by the taller sedges. In the meantime, brass buttons may be facilitating sedimentation and seed deposition by providing a vegetation canopy throughout the restoration site that can baffle and slow water movement. Thus, in the short to intermediate term brass buttons may benefit the restoration site, before it is replaced by more desirable native sedges. However, this scenario is not guaranteed. An alternative outcome is that brass buttons becomes firmly established such that it pre-empts colonization by native sedges. This uncertainty is another reason why continued monitoring of the site is necessary. If brass buttons becomes problematic, then vegetation control could target this species as well.

Experimental seeding treatments

The 2019 monitoring results provided no reason to change previously reported conclusions about the effectiveness of experimental seeding (Hood 2018). Experimental seeding of native spike-rush, soft-stem bulrush, and sea arrowgrass had no inhibitory effect on colonization by invasive non-native species, nor did it have any benefit for colonization by native species, even those intentionally seeded. The results of this case study, so far, suggest that experimental seeding is not worth the cost. However, conditions on other restoration sites may differ significantly, so that this conclusion may not be universally valid. A larger sample size of restoration projects and target species is necessary to draw more reliable conclusions regarding this issue. For example, rapid colonization by native vegetation of the nearby zis a ba restoration site in the northern end of the Stillaguamish Delta through entirely passive tidal seed recruitment from local marshes indicates that active seeding would have been needless in this case as well. It should be noted that there is currently no good guidance on effectively

planting small seeds in a tidal marsh restoration site, but passive colonization has been anecdotally observed at many sites throughout Puget Sound and the outer coast of Washington State, suggesting that active planting is usually unnecessary.

SET measures of sedimentation

Surprisingly, sedimentation rates were positively correlated with sample point elevation, although it should be cautioned that only three points were sampled. This is contrary to theory, which postulates that given a constant concentration of suspended sediment, lower elevation areas will be subject to greater sedimentation rates, because throughout the tidal cycle they are covered by a greater depth of water compared to higher elevation areas, which are covered by very little water and which during neap tides may not be inundated at all (Pethick 1981, French 1993). However, this model neglects other influences on sedimentation such as hydrodynamics which may vary over the Fir Island Farms restoration site. Due to the absence of an extensive channel network, much of the tidal exchange is forced over the marsh surface rather than through a channel network. Lower elevation sites, which are exposed for a greater length of time to tidal currents due to their greater exposure to inundation, may thus have more unstable sediments, more subject to tidal erosion, resulting in lower accretion rates.

Hypothesis: tidal channels affect vegetation colonization

Lower elevations suitable to colonization by three-square sedge are abundant on the restoration site, but these areas are either entirely bare or sparsely vegetated by brass buttons. This naturally begs the question of why three-square is present at higher elevations than normal and completely absent at normal low elevations. Tidal vegetation species distributions are generally constrained by physical stress (inundation, waves) at low elevations and by competition (canopy shading) at high elevations (Bertness and Ellison 1987). Thus, three-square may be colonizing higher elevations than normal, because vegetation is sparse on the site except along the highest elevation margins, so there little competition to exclude three-square from most high-elevation areas on the site.

The absence of vegetation at lower elevations on the restoration site is puzzling, because a restoration site in the nearby Stillaguamish Delta, zis a ba, has a similar environmental context, salinity around 10 ppt, subsided to similar elevation, a thick clay or hardpan layer of soil, a history of agricultural use, but it is densely vegetated with maritime bulrush only two years after dike breaching and looks like a mature marsh. The only other evident difference between the two sites is in the geometry of the tidal channel network draining each site. Fir Island Farm has only one large tidal channel outlet, and in addition to the mainstem channel, has only two large tributaries, excavated during restoration. The two additional small tributaries there were shallowly excavated, convey minor amounts of flow given their shallow depths, so that most tidal exchange over the site is likely through sheet flow over the marsh surface. In contrast, zis a ba, has seven channel outlets, but is a smaller site. The zis a ba channel network was excavated to match allometric predictions of channel

geometry developed from a reference marsh system. Allometric channel prediction was not used for the Fir Island Farm restoration design; consequently, the site has far too few channel outlets and too few excavated channels (Table 1). Further development of new tidal channels will likely be extremely slow because the site is extensively underlain by a thick hardpan.

Table 1. Allometric predictions for tidal channel network geometry versus restoration site excavation (deep and shallow excavation are distinguished for FIF). The 80% confidence limits of the predictions are in parentheses.

Site	Area	Predicted channel count	Observed channel count	Predicted channel length (m)	Observed channel length (m)
FIF	132 ac (53.4 ha)	22 (11-46)	1	10,965 (4,400 - 27,000)	2,200 deep 3,770 deep +shallow
zis a ba	90 ac (36.2 ha)	17 (8-34)	7	6,240 (2,550 – 15,280)	5,210

Thus, I hypothesize that the two differences between the sites, vegetation development and channel network development, are related. Low channel network development in the FIF site results in high sheet flow energy during flood and ebb tides, as evidenced by the thick sand deposit in a flood tide delta near the channel network opening on the FIF site where the channel network first branches. This high sheet flow energy results in high physical stress (erosion) of seeds or seedlings, preventing significant vegetation establishment at low elevations of the FIF site. Only higher elevation, peripheral areas have sufficiently low shear stress to preclude seed or seedling erosion. In contrast, zis a ba, has low sheet flow energy over most of the site as a result of tidal energy being concentrated in the channels of a well-developed channel network. Consequently, seed deposition and seedling establishment are facilitated in this comparatively low energy environment.

Several species in the genus, *Cotula*, are known to have a mucilaginous covering on their seeds, although this has not been specifically investigated for brass buttons (*C. coronopifolia*); this mucilaginous coating has been suggested to allow greater adhesion to sediment (Partridge and Wilson 1987). Additionally, seed germination rates are very fast for brass buttons (Costa et al. 2009). Both of these seed traits may explain why brass buttons has been able to colonize hypothesized high shear stress areas in the restoration site that otherwise have minimal colonization by native species, even though those native species are common at those elevations in the reference marsh.

Recommendations

Continued monitoring of the FIF restoration site is recommended to address five issues: [1] the potential threat of resurgent non-native cattail invasion; [2] the fate of the current brass buttons invasion—will there be expansion, contraction, or stasis; [3] whether or at what rate native species will ever spread to their predicted distributions on the site; [4] testing the hypothesis that channel network geometry is impacting native vegetation colonization through its influence on hydrodynamics; and [5] whether or at what rate passive tidal channel erosion will continue to occur.

Potential invasion of the restoration site by non-native cattail, or other non-native species, can continue to be monitored using current methods. The PVM indicates low probability of non-native cattail establishment over the site. Likewise, the nearly complete absence of non-native cattail since initial control of the first year's flush of cattail seedlings also suggests low probability of re-establishment, as well as high efficacy of the control efforts. Nevertheless, the invasion of non-native cattail in the first year following restoration was surprising, because site conditions were thought to be unfavorable, yet extensive invasion occurred anyway. Because this surprise cannot be adequately explained, we risk being surprised again. Thus, continued monitoring would be prudent and would allow quick response to invasion with proven control techniques.

Brass buttons can potentially become a dominant invader under some circumstances (Costas et al. 2009). For example, in Holland a marsh was colonized by brass buttons, which over a three-year span grew into mono-specific patches averaging 2,500 m², i.e., 50m x 50m, or 0.6 acres (Van der Toorn 1980). Thus, given the current abundance of brass buttons on the FIF restoration site, there is potential for this non-native species to become well-established and pre-empt colonization by more desirable native sedges through below-ground competition by roots for space and nutrients. However, if taller native plants can become established, they could outcompete brass buttons by above-ground competition, shading out the low-growing brass buttons. If the brass buttons invasion does not contract naturally over the next few years, it may be necessary to implement control efforts to reduce its coverage and allow native species colonization and expansion. Monitoring of brass buttons to determine its near-term fate and need for control can continue using current methods for vegetation monitoring.

Site colonization by native marsh vegetation is currently limited to the higher margins of the site and a few other patchy areas. This is consistent with the hypothesis that hydrodynamic shear stress is limiting seed germination of native species, if these higher margins are backwater areas with low shear stresses, as seems likely. If the shear stress-germination hypothesis is correct, and if there are limited topographic changes, then hydrodynamics and vegetation distribution are unlikely to change. However, if the hypothesis is incorrect or if topography or hydrodynamics change over time, then native species colonization could increase.

Topography could change if tidal channels erode through the action of tidal currents, but this seems unlikely given the extensive presence of a clay hard pan on the site. Hydrodynamic shear stresses could be reduced by the presence and growth of brass buttons, facilitating colonization by native sedges. Such facilitative interactions between marsh species have been previously documented for local vegetation influences on soil salinity and oxygenation (e.g., Hacker and Gaines 1997) and on sediment stabilization and reduced shear stress (e.g., Bruno 2000). Thus, it is unclear if the spatial distribution of native species will continue to be limited or if it will significantly expand into more of the restoration site. Documentation of significant expansion would provide new and important insights into the colonization dynamics of marsh restoration sites. Absence of such an expansion will suggest the need for active intervention in the site to reduce the abundance of non-native species, such as brass buttons, and increase the abundance of desirable native species.

It is critical to test the hypothesis that the spatial distributions and dynamics of brass buttons versus native plant species are influenced by site hydrodynamics, i.e., tidal fluxes and shear stresses, because hydrodynamics can be optimally manipulated by the proper design and excavation of tidal channel networks, i.e., there can be effective management responses to sub-optimal hydrodynamics, with likely consequent improvements to vegetation. There are several potential approaches to testing this hypothesis: [1] It may be possible to model shear stresses in GIS from lidar data, or perhaps more crudely from planform channel geometry, and then correlate modeled shear stresses with vegetation distributions; [2] Current velocities over the marsh surface on ebb and flood tides could be empirically measured systematically across the restoration site to map high versus low shear stress areas and determine their correlation with vegetation distributions (prior modeling could aid in sampling design); [3] Seeds of various species could be placed in experimental plots and counted after each tide to determine their vulnerability to movement by tidal currents—plots could be located in high versus low shear stress areas; [4] Experimental transplantation of adult three-square into low shear stress, high shear stress, and control areas could be done to confirm that PVM determinations are appropriate if shear stress constraints on seed germination can be bypassed, and to confirm that seed germination is the vulnerable life-history stage that is limiting vegetation distributions.

Finally, while little change in tidal channel planform geometry is expected due to the extensive clay hard pan on the restoration site. It is easy to monitor planform changes through GIS analysis of high resolution air photos. The drone-flown air photos provided by WDFW were particularly useful for this purpose due to their extreme resolution. Future monitoring should also include ground surveys of channel cross-sections. Pre-project and as-built surveys are available for many cross-sections of Claude O. Davis Slough and of the old remnant Dry Slough distributary channel. These could be re-occupied to document potential change over time. The new tributary to Claude O. Davis Slough, that drains the pump house has not been previously surveyed, so new cross-sections would have to be established for this channel. Erosion of channel cross-sections would be surprising given the presence of a clay hard pan, but

sedimentation of the channels (shoaling) is possible. Slow development of the current tidal channel network, which is small relative to reference marsh conditions, might suggest the need for additional channel excavation in the future, particularly if it is proven that hydrodynamic forces are limiting vegetation colonization.

The proposed continuation and elaboration of current monitoring is linked to potential adaptive management for each monitored parameter. These potential management actions could include future control of renewed cattail invasion, control of the current brass buttons invasion, and possible excavation of additional tidal channels. Furthermore, the problems, solutions, and improved understanding that are identified and developed by continued monitoring of the FIF restoration site are likely to inform many similar restoration projects in Puget Sound and elsewhere in the Pacific Northwest. Thus, further investment in three to five more years of site monitoring and hypothesis testing is highly warranted.

References

- Bertness, M and AM Ellison. 1987. Determinants of pattern in a New England salt marsh plant community. *Ecological Monographs* **57**: 129-147.
- Bruno, JF. 2000. Facilitation of cobble beach plant communities through habitat modification by *Spartina alterniflora*. *Ecology* **81**: 1179-1192.
- Cahoon, DR, JR French, T Spencer, D Reed, I Moller (2000). Vertical accretion versus elevation adjustment in UK saltmarshes: an evaluation of alternative methodologies. Coastal and estuarine environments: sedimentology, geomorphology, and geoarchaeology. London, Geological Society of London: 223-238.
- Costa, JC, C Neto, P Arsénio, and J Capelo. 2009. Geographic variation among Iberian communities of the exotic halophyte *Cotula coronopifolia*. *Botanica Helvetica* **119**: 53–61.
- French, JR. 1993. Numerical simulation of vertical marsh growth and adjustment to accelerated sea-level rise, North Norfolk, UK. *Earth Surface Processes and Landforms* **18**: 63-81.
- Hacker, SD and SD Gaines. 1997. Some implications of direct positive interactions for community species diversity. *Ecology* **78**: 1990-2003.
- Hood WG. 2018. Fir Island Farms 2nd-year Post-Restoration Tidal Marsh Monitoring Report. Prepared for Washington Department of Fish and Wildlife. Skagit River System Cooperative, LaConner, WA.
- Hood WG. 2013. Applying and testing a predictive vegetation model to management of the invasive cattail, *Typha angustifolia*, in an oligohaline tidal marsh reveals priority effects caused by non-stationarity. *Wetlands Ecology and Management* **21**: 229-242.

- Partridge, TR and JB Wilson. 1987. Germination in relation to salinity in some plants of salt marshes in Otago, New Zealand. *New Zealand Journal of Botany* **25**: 255-261.
- Pethick, JS. 1981. Long-term accretion rates on tidal marshes. *Journal of Sedimentary Petrology* **51**: 571-577.
- Ritchey, EL., JM. McGrath, D Gehring. 2015. Determining Soil Texture by Feel. Agriculture and Natural Resources Publications. 139. https://uknowledge.uky.edu/anr_reports/139.
- Van der Toorn, J. 1980. On the ecology of *Cotula coronopifolia* L. and *Ranunculus sceleratus*. *Acta Botanica Neerlandica* **29**:385-396.
- Washington State Department of Fish and Wildlife (WDFW). 2015. Fir Island Farm Restoration: Final Monitoring and Adaptive Management Plan. WDFW, LaConner, WA.
- Zar JH. 1984. Biostatistical Analysis, 2nd edition. Prentice Hall, Englewood Cliffs, NJ.