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Early Ecosystem Development Varies With Elevation and Pre-Restoration Land Use/Land Cover in a Pacific Northwest Tidal Wetland Restoration Project

Christopher N. Janousek[1] · **Scott J. Bailey**[2,3] · **Laura S. Brophy**[4,5]

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Abstract

Tidal wetland restoration through dike removal can enhance coastal ecosystem services, such as flood attenuation, fish production, and carbon sequestration. However, landscape-level heterogeneity may influence recovery. For a 169-ha restoration project in Tillamook Bay, Oregon, we hypothesized that areas of more intensive pre-restoration land use/land cover (cropping, grazing) would differ more from reference conditions before restoration than less-intensive uses and that initial post-restoration recovery would vary by land-use/land-cover type and wetland elevation. Before the restoration, the project site overall had higher non-native plant cover, lower elevation and groundwater levels, and lower soil pH than reference high marsh, with some differences by land-use/land-cover type. The cropped and grazed areas were strongly dominated by non-native species, such as *Phalaris arundinacea*, and were 74 and 31 cm lower than reference high marsh. Less intensively managed areas had elevations intermediate to the cropped and grazed areas and a trend towards higher native plant cover. The restoration led to higher dry-season groundwater levels, increased soil salinity to mesohaline conditions, and a 10-fold increase in soil pH at the project site, while reducing total plant cover. The degree of pre- to early post-restoration change for some parameters differed by land-use/land-cover type (total and non-native plant cover) and by wetland elevation (soil salinity, pH, and accretion rate; and total and non-native plant cover). Our results suggest that pre-restoration heterogeneity in elevation and land cover/land use may influence early post-restoration recovery. Restoration planning can incorporate such spatial variability into management targets and interventions for specific outcomes.

Keywords Brackish marsh · Ecosystem services · Estuarine wetlands · Groundwater · Plant composition · Succession · Tidal wetlands

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✉ Christopher N. Janousek
janousec@oregonstate.edu

[1] Department of Fisheries and Wildlife, Oregon State University, Corvallis, OR 97331, USA

[2] Tillamook Estuaries Partnership, Garibaldi, OR 97118, USA

[3] Chelan County Natural Resources Department, Wenatchee, WA 98801, USA

[4] Estuary Technical Group, Institute for Applied Ecology, Corvallis, OR 97333, USA

[5] College of Earth, Ocean and Atmospheric Sciences, Oregon State University, Corvallis, OR 97331, USA

Introduction

About 85% of the historical coverage of tidal wetlands in the western US has been lost to urban development or agriculture through dike construction, drainage, or filling (Brophy et al. 2019). This loss includes about 60–70% of tidal marshes and over 90% loss of forested and shrub-dominated tidal swamps on the outer coast of Oregon (Brophy 2019). Wetland loss has led to a reduction in important estuarine functions and services, such as storm protection, fisheries production, carbon sequestration, nutrient removal, and habitat provision for wildlife (Bottom et al. 2005; Loomis and Craft 2010; Barbier et al. 2011; Bu et al. 2015). Restoring tidal inundation to former tidal wetlands is a common management tool to enhance coastal ecosystem services, increase connectivity between fragmented wetland habitats, protect biodiversity, and improve coastal ecosystem resilience to climate change (Zedler 2000; Davis et al. 2018; Reed et al. 2018; Woo et al. 2018).

Ecologically, the immediate consequence of dike construction is impairment of hydrologic connectivity between the wetland and the remainder of the estuary. Longer-term impacts include land subsidence and changes in groundwater tables (Roman et al. 1984; Portnoy and Giblin 1997; Mossman et al. 2012). Wetland drainage can lead to changes in soil composition and structure, such as acidification, loss of accumulated carbon pools, and reduced salinity and porosity (Portnoy and Giblin 1997; Portnoy 1999; Bu et al. 2015; Spencer et al. 2017; Ewers Lewis et al. 2019). Lower salinity in turn may facilitate establishment of freshwater plants and animals, including non-native species (Roman et al. 1984; Karberg et al. 2018).

Former tidal wetlands are used for a variety of purposes, including crop and livestock agriculture, residential and commercial development, and transportation and other infrastructure (Marcoe and Pilson 2017). These pre-restoration land use practices may affect recovery dynamics upon restoration. For instance, intensive agricultural use, such as annual tillage by heavy machinery could substantially compact organic-rich soils (Drexler et al. 2009) or create seed banks of non-native species if such species are intentionally sown on site (Dawson et al. 2017b). Mowing or livestock grazing may favor certain plant species over others in areas slated to be restored (Roman et al. 1984). Changes to the water table, such as groundwater drawdown, could impact soil oxygen availability which in turn affects soil carbon content. In contrast, some diked areas may not be intensively managed or disturbed by human activities prior to their restoration to tidal habitat; such wet pastures or freshwater wetlands may have higher carbon content in soils, different plant assemblages, and less soil compaction due to lower direct human impacts.

In addition to broader land use/land cover differences, spatial gradients in wetland elevation and salinity could also impact wetland recovery following restoration. For example, post-restoration differences in vegetation colonization and invertebrate abundance have been shown to be a function of wetland elevation (Karberg et al. 2018; Woo et al. 2018). Restoration projects in low-salinity areas may retain freshwater vegetation, including invasive species, longer than projects implemented in more saline areas. While the ultimate goal of many wetland restoration projects may be to mimic least-disturbed ecosystems, rates of ecosystem recovery may vary by attribute (Craft et al. 1999; Zedler 2000) and restored wetlands may or may not reach structural or functional equivalency with reference areas (e.g., Lawrence et al. 2018).

The Southern Flow Corridor project in Tillamook Bay, Oregon (hereafter SFC) is one of the largest tidal wetland restoration projects implemented to date in the Pacific Northwest (PNW) (Sherman et al. 2019). The project was implemented to reduce urban flooding in the estuary floodplain and promote other important ecosystem services within the Tillamook Bay, such as salmonid production. Prior to restoration, land-use/land-cover type varied considerably across the site and included cattle grazing, cropping, a freshwater mitigation wetland, and areas with differing vegetation types which had been relatively unmanaged.

Our objective was to assess early ecosystem change following restoration at SFC and to examine whether differences in elevation and pre-restoration land use/land cover affected post-restoration conditions. We tested the following hypotheses: (1) prior to restoration, soils and vegetation varied by land-use/land-cover class; (2) tidal restoration led to changes in hydrology, soil, and vegetation characteristics in the direction of reference conditions; (3) post-restoration change varied by land-use/land-cover type within SFC, including faster recovery in less intensively managed areas; and (4) the degree of change at SFC between pre- and post-restoration periods was correlated with elevation, with low elevation areas exhibiting greater soil accretion and greater change in soil and vegetation characteristics. This study is part of a broader effort to evaluate the impacts of restoration on biological and physical conditions at SFC and the efficacy of the project in reducing flooding.

Materials and Methods

Site Description and Land-Use History

The SFC project restored about 9% of Tillamook Bay's historical tidal wetland area which had previously been reduced by about 72% of historical area due to diking (Brophy 2019). The primary goal of the project was to reduce the magnitude and duration of episodic flooding along the US Highway 101 corridor in the town of Tillamook that occurred when rivers crested upstream of the highway (often in conjunction with high tides) and floodwaters flowed overland towards the bay. Dikes surrounding the project site exacerbated this flooding by impounding water and impeding drainage.

The project site consisted of 169 ha of former tidal wetlands at the confluence of the Wilson, Tillamook, and Trask Rivers (45.47°N, 123.88°W; Fig. 1). Prior to diking in the middle of the twentieth century, the site likely consisted of higher-elevation brackish tidal emergent marsh and forested tidal wetland (Brown et al. 2016). We divided the SFC site into five zones defined by their land-use/land-cover type during the period approximately 1–2 decades prior to restoration (Fig. 1, Table 1). We describe these zones subsequently based on information in Tillamook County (2018) and field observations.

The north zone ("N") between the Wilson River and Blind Slough was a mixture of freshwater marsh, non-tidal scrub-shrub wetland dominated by *Salix hookeriana*, and non-tidal forested wetland dominated by *Picea sitchensis* (Brown et al. 2016); historically, the eastern portion of this zone was a tidal swamp (Hawes et al. 2018). The middle zone ("M") consisted of extensive freshwater marsh with some patches of woody shrub vegetation (mainly *Salix* spp., *Lonicera involucrata*, and *Rubus*

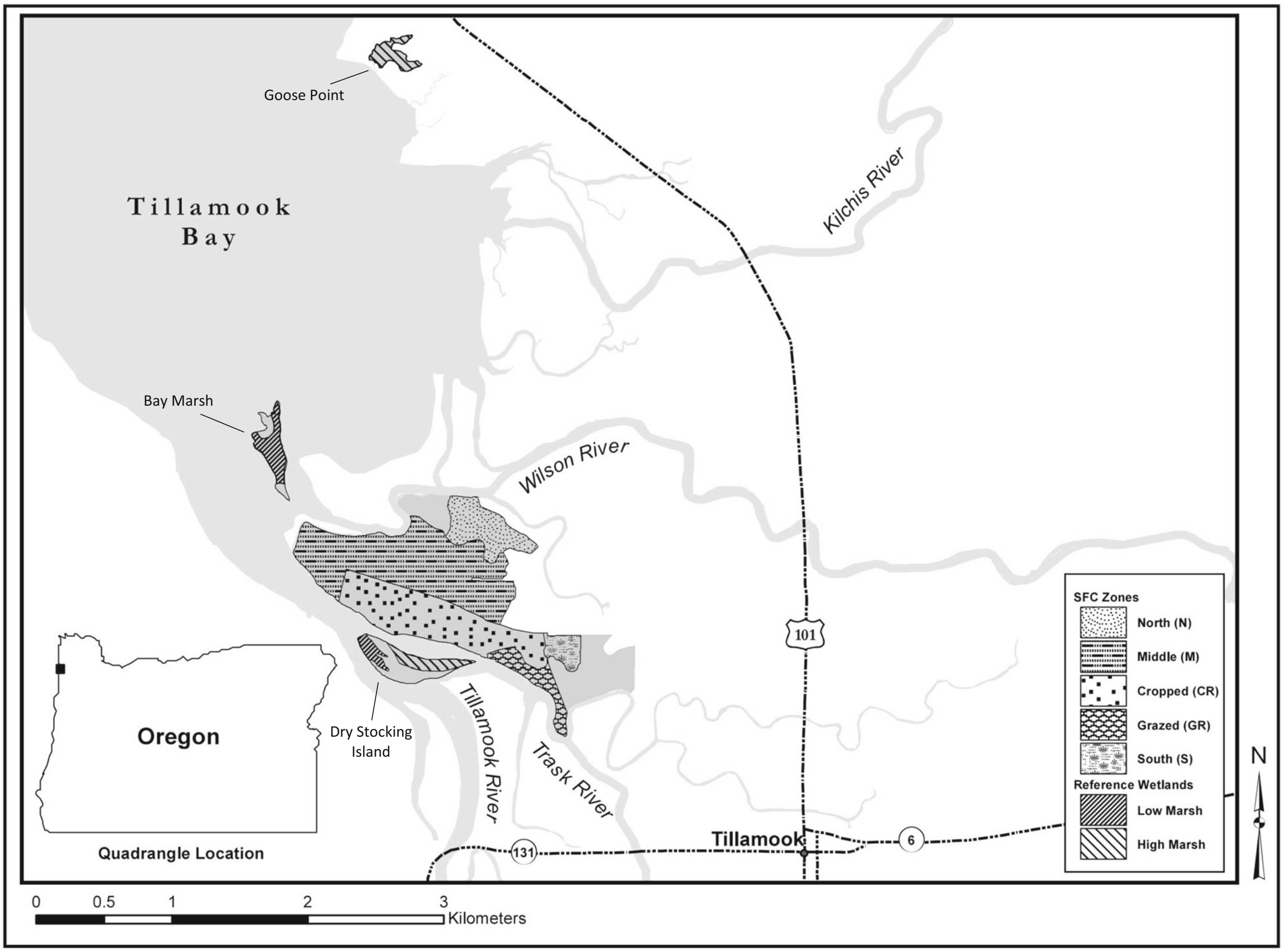

Fig. 1 Map of (**a**) the Southern Flow Corridor (SFC) tidal wetland restoration site and reference wetlands within Tillamook Bay, Oregon, and (**b**) location of pre-restoration land-use zones within the SFC site. LM, reference low marsh; HM, reference high marsh; N, SFC north zone; M, SFC middle zone; S, SFC south zone; CR, SFC cropped zone; GR, SFC grazed zone

spectabilis) and remnant historical tidal channels. Although both the north and middle zones had been used for agriculture at some point in the past, these areas were not intensively used for over a decade before restoration. Several ponds had been excavated for waterfowl use in the middle zone.

North of the bank of the Trask River, a large cropped zone ("CR") was managed for grass hay production, the eastern portion of which appeared to have been at least intermittently plowed before restoration. Except for Nolan Slough to the southeast, remnant tidal channels in this zone had been largely obliterated by agricultural activities and replaced by linear agricultural ditches. Southeast of the cropped zone, there was a grazed zone ("GR") with pasture grasses which were grazed by livestock until just prior to restoration. Additionally, there was a small non-tidal freshwater mitigation wetland (south zone; "S") farmed until about two decades before tidal restoration, but planted with native freshwater wetland plants (mainly woody species) several years before the SFC project.

Restoration actions at SFC began in May 2016. Contractors removed tide gates and most of the dikes surrounding the site. Additionally, crews excavated 17.0 km of new channels throughout the site, especially in the cropped zone. Excavation mostly followed locations of historical (1939) tidal channels. Dike removal and tidal flow restoration was completed during 2016, but channel excavation and minor setback levee adjustments continued until September 2017. Crews avoided work near important monitoring infrastructure, such as accretion plots (see subsequent discussion).

Sampling Design

Our sampling design was similar to a Before After Control Impact (BACI) framework (Stewart-Oaten et al. 1986), although we included multiple reference sites (Underwood 1994). We sampled the five zones within SFC as described previously, focusing on emergent wetlands (not the site's fringing forested wetlands), but if plots included woody vegetation, it was included in cover estimates. We also sampled two types of tidal reference wetlands in Tillamook Bay (Fig. 1, Table 1). These additional zones were two low marsh ("LM") reference sites (Bay

Table 1 Description of wetland zones within the SFC project and reference sites by land use type, and number of groundwater stations, accretion plots, soil composition samples, and vegetation plots per zone. (A range of sample sizes are given for soil samples from several zones because they varied by sampling period or parameter)

Zone	Code	Land use type and impacts during pre-restoration period	Wetland area (ha)	Ground-water stations	Accretion plots	Soil samples	Vegetation plots
Low tidal marsh	LM	Low intertidal least-disturbed brackish marsh. Parcels on Dry Stocking Island and Bay Marsh. Minimal human use, primarily hunting.	8.2	0	5	5	19
High tidal marsh	HM	High intertidal least-disturbed brackish marsh. Separate parcels on Dry Stocking Island and Goose Point. Minimal human use, primarily hunting and some mowing.	8.7	2 (DSI, GP)	5	6	23
SFC North	N	Non-tidal freshwater marsh, scrub-shrub and forested wetland. Abandoned pasture; minimal recent human use (mainly hunting); remnant tidal channels.	16.6	1 (A004)	4	5	19
SFC Middle	M	Non-tidal freshwater marsh. Abandoned pasture; minimal recent human use (mainly hunting); remnant tidal channels.	65.6	3 (A009, A016, A037)	1	7–10	56
SFC South	S	Non-tidal freshwater marsh (mitigation site). Minimal recent human use.	5.5	0	1	2	8
SFC Cropped	CR	Pasture grasses with regular cropping; heavily ditched	42.8	2 (A028, A073)	6	6–10	45
SFC Grazed	GR	Livestock grazing	8.6	0	0	0	8

Marsh northwest of SFC, and the western end of Dry Stocking Island, south of SFC along the Trask River) and two high marsh ("HM") reference sites (the eastern end of Dry Stocking Island and Goose Point marsh, approximately 3 km north of SFC). Low marsh in the PNW is found at elevations below MHHW and is usually inundated by tides at least once daily, while high marsh is found above MHHW and is inundated by monthly spring tides (Janousek and Folger 2014). Reference low marsh allowed comparison of SFC to the wetland type it would most likely restore to in the short term, whereas reference high marsh allowed comparison to SFC's probable long-term target wetland type.

Pre-restoration assessment of wetland surface elevation, channel and groundwater hydrology, soil conditions, vegetation cover and composition, and other parameters were conducted from 2013 to 2014, two years before restoration. Detailed methods and results are described in Brown et al. (2016). We conducted post-restoration sampling between fall 2017 and fall 2018 at the same sampling locations with the same methods. We did not measure a subset of parameters in the grazed zone (e.g., accretion) due to the potential for pre-restoration disturbance by cattle.

Soil Accretion

In October 2013, we established replicate 0.25 m^2 feldspar marker horizon plots randomly across SFC ($n = 27$) and reference sites ($n = 11$) using the method of Cahoon and Turner (1989) to assess rates of soil accretion (Brophy et al. 2018). We sampled these plots in early fall 2018 (4.9 yr post-establishment; accretion rates spanned both pre- and post-restoration periods) by extracting 1–3 soil wedges per plot with a knife. We measured the amount of soil deposited above the feldspar layer on four sides of the wedge, averaged measurements per wedge, and then averaged wedges per plot. To determine an average annual accretion rate, we divided the height of the accumulated soil layer above the top of the marker horizon by the time elapsed. We omitted cores or plots from analysis when a distinct feldspar layer was lacking. Many cores from plots established in areas dominated by *Phalaris arundinacea* within SFC had indistinct layers, probably due to a thick root system that gradually intergraded with surface soils. Comparison of the knife method with liquid nitrogen cryocoring (Knaus and Cahoon 1990) during earlier sampling in 2017 suggested they yielded comparable data (Brophy et al. 2018).

Soil Characteristics

During August 2014 and July 2018, we sampled surface soils near each accretion plot to determine edaphic characteristics. We collected 3–8 cores (to about 15–20 cm depth) near each plot, pooled them, and then sent them for analysis to labs at AgSource Laboratories and Oregon State University for determination of soil pH, conductivity, and organic matter content by the loss on ignition method (Heiri et al. 2001). Organic matter combustion was performed at 360° for 2 h and at 385°

for 5 h by the respective labs in 2014 and 2018. We converted percent organic matter to percent carbon using the relationship developed by Peck (2017) for Oregon tidal wetlands. We converted soil conductivity to salinity based on UNESCO (1981).

Channel Hydrology

We obtained time series of water levels in channels from October 2013 to February 2015 (pre-restoration) and October 2017 to September 2018 (post-restoration) near reference marshes and internal to SFC. At each station, we installed a Hobo U-20 water level logger (Onset Computer Corporation, Bourne, MA) inside a 5 cm diameter PVC stilling well, with the sensor at approximately mean tide level. Sensors recorded absolute pressure (barometric pressure plus water pressure) every 15 min. We cleaned loggers and downloaded data every few months during deployment. We measured barometric pressure with additional sensors placed above the high water mark at a reference site. We compensated for barometric pressure and converted pressure to water levels with Onset's Hoboware software. We used only a subset of channel data here (one near a reference marsh and one internal to SFC) to illustrate pre- and post-restoration inundation patterns.

Groundwater Hydrology

We measured groundwater levels in 4 cm diameter PVC wells at six randomly located stations within SFC and two high marsh reference stations for a year during the pre-restoration period (May 2014–May 2015), and the same post-restoration period that channel conditions were assessed. Wells were similar to those described in USACE (2000) and extended about 1.2 m below the wetland surface, with 20–35 cm risers above the ground. Slots in the well from about 10–15 cm below the ground surface to the well bottom allowed groundwater exchange. At the bottom of each well, we placed a Hobo U-20 water level logger.

As with channel sensors, we recorded water levels every 15 min, cleaned loggers and downloaded data every few months, and corrected for variability in barometric pressure. For each station, we analyzed the time series of water level for wet and dry season periods (Dec–Mar; June–Sept) in 2013–2014 and 2017–2018. Within the wet and dry season periods for both monitoring years, we identified periods of neap and spring tides by defining spring tides as all days on which a new or full moon occurred plus three days before and after that date, and neap tides as all other days (https://aa.usno.navy.mil).

Vegetation Cover and Composition

We assessed vegetation cover and composition in July and August, corresponding with the peak growing season in PNW tidal wetlands (Thom et al. 2002). In 2014, we determined plant cover in 178 1.0 m^2 plots distributed randomly throughout SFC and the reference sites. Sample numbers per zone within SFC and the reference marshes were somewhat proportional to the areal extent of each zone (Brown et al. 2016). We used a handheld GPS to navigate to each plot and then visually estimated cover of all vascular plant species visible from above, as well as other major cover classes (e.g., bare ground, wrack). Total cover summed to 100% because we only determined the upper-most layer, except when taller species (shrubs, trees, or vines) contributed to an additional layer of vegetation. During July 2018, we navigated to the approximate location of each plot with a handheld GPS and re-assessed plant cover and composition using the same methods. We also remeasured plot location and elevation using RTK-GPS; based on those more spatially precise values, we determined that > 90% of the plots were relocated to within 5 m of the original location.

We identified plants to the species level except in rare cases. Nomenclature follows the Oregon Flora Project (Jaster et al. 2017), except for *Salicornia pacifica* which follows Piirainen et al. (2017). We classified species by whether they were native or non-native in Oregon, generally following Jaster et al. (2017). A few taxa we encountered have both native and non-native genotypes in Oregon, or their native status is uncertain (e.g., taxa not identified to species level). For calculation of total native and non-native cover in plots, we considered *Phalaris arundinacea* and *Alopecurus geniculatus* as non-native (Brown et al. 2016; USDA 2019) and *Juncus effusus* as native, but omitted cover of other species of uncertain status, which were infrequent.

Wetland Elevation

During both sampling periods, we determined wetland surface elevations adjacent to vegetation ($n = 176$) and accretion ($n = 22$) plots and measured the elevation of logger installations with Trimble R8 or Spectra Physics survey-grade GNSS rovers using real-time kinematic (RTK) correctors streamed from the Oregon Real Time GPS Network (https://www.oregon.gov/ODOT/ORGN/pages/index.aspx) via cell phone. We periodically checked the rover's accuracy and precision by measuring elevations on benchmarks. In 2018, repeated measurements at a benchmark associated with the Dick Point NOAA tidal station ($n = 11$) indicated an accuracy of 1.3 cm relative to a survey-grade GPS occupation published in the OPUS database (https://www.ngs.noaa.gov/OPUS) and a precision (standard deviation) of 1.2 cm across repeated measurements.

For elevation analyses, we converted geodetic measurements (North American Datum of 1988 or NAVD88, with Geoid 12A) to a standardized, unitless measure that scales elevation relative to total tide range, $z^* = (z - \mathrm{MTL})/(\mathrm{MHHW}\text{-}\mathrm{MTL})$, where z is the measured elevation and MTL and MHHW are local mean tide level and mean higher high water respectively in NAVD88 (Swanson et al. 2014). On the standardized elevation scale, $z^* = 0$ is MTL and $z^* = 1.0$ is MHHW, our boundary between low and high marsh. We used the tidal range computed at the Dick Point NOAA tide gauge (station 9437381, https://tidesandcurrents.noaa.gov/) in southern Tillamook Bay and derived relationships between NAVD88, MTL, and MHHW datums using a published geodetic measurement at the Dick Point benchmark noted previously.

Statistical Analyses

We conducted all analyses with R version 3.5.0 (R Core Team 2018). Although our design was suitable for a two-factor repeated-measures ANOVA for most parameters (e.g., Smokorowski and Randall 2017), interpretation of outcomes, particularly interaction terms, would be challenging because both our reference and restored areas had multiple levels, and one of our main hypotheses of interest was differences between SFC and reference marsh zones before and after restoration. Therefore, for most parameters, we tested for differences among wetland zones (five SFC land-use/land-cover types and two reference marsh types) by conducting separate one-factor ANOVAs on data from pre- and post-restoration sampling periods. We used either parametric ANOVA followed by Tukey's HSD test or non-parametric Kruskal–Wallis tests and Dunn's test of pair-wise differences if the data were heteroscedastic. For the accretion data set, we pooled data for the three less-intensively managed SFC zones (north, middle, and south zones combined into a recently "unmanaged" group, "UM") to achieve adequate replication for analysis. We also analyzed accretion plots from the cropped zone, but did not sample the grazed zone.

To quantify the magnitude of change between pre- to post-restoration sampling periods by zone, we computed differences between paired samples (e.g., soil pH change between 2014 and 2018) and then used 95% confidence intervals to compare whether means deviated significantly from zero, and one-factor ANOVA to test whether mean differences varied between wetland zones. To test potential elevation effects on the degree of change observed within SFC, we used linear regression to compare pre-restoration wetland elevation (z^*) with pre- versus post-restoration change in soil and vegetation parameters, including annual accretion rate, soil carbon content, soil pH, soil conductivity, total plant cover, native and non-native plant cover, and plant species richness.

For groundwater time series, we conducted two types of analyses. First, to evaluate restoration effects, we compiled all groundwater level data (15 min frequency) for each station for wet and dry season periods during pre- and post-restoration sampling periods, computed median level relative to the wetland surface, and plotted the distribution of data using violin plots (package "vioplot"). For each combination of station and season, we compared distributions for pre- versus post-restoration sampling periods with Kolmogorov–Smirnov tests and bootstrapped p values with the package "Matching." Second, we evaluated restoration (as well as seasonal and monthly tide stages) effects on water level variability at each station by computing the daily range of groundwater level (maximum − minimum level) and analyzing data with 3-factor type II ANOVA (package "car") followed by hierarchical partitioning to assess the relative importance of the main factors in the model (Chevan and Sutherland 1991; package "hier.part"). We hypothesized that daily groundwater vertical range would increase after restoration at SFC stations and would be greater during spring versus neap tides and in summer versus winter. For three SFC stations with incomplete dry season data due to water levels falling below the sensor (stations 4, 28, and 73), we analyzed only wet season data with 2-factor ANOVA (restoration and tide phase were main factors). We transformed data ($\log_{10}$ or square-root) prior to analyses to improve homogeneity of variances.

We used multivariate methods to analyze cumulative plant species richness (gamma diversity) and composition. We compared cumulative richness during pre- and post-restoration sampling periods for each wetland zone with species accumulation curves using species presence–absence data from the vegetation plots (package "vegan," 1000 bootstraps to generate 95% confidence intervals; Oksanen 2015). To visualize differences in composition by wetland zone and sampling period, we used non-metric multidimensional scaling (NMDS) based on the percent cover of bare ground and common plant species in the plots ($n = 25$; taxa with < 2% frequency of occurrence across the dataset excluded). The 2-dimensional NMDS analysis was based on square-root-transformed and Wisconsin double standardized percent cover data and a Bray–Curtis matrix of compositional dissimilarity (function "MetaMDS," package "vegan"). To analyze differences in species composition for each zone by sampling period, we tested for differences in centroids using permutational ANOVA and differences in dispersion (beta diversity) with functions "adonis" and "betadisper" in package "vegan" (Anderson and Walsh 2013).

Results

Wetland Inundation and Elevation

Prior to restoration, SFC was not inundated by tides on a regular basis, although dikes were occasionally overtopped during winter floods. Removal of dikes and tide gates restored daily tidal flows inside the site (Fig. 2). During the pre-restoration period, SFC zones and reference marshes differed in median wetland elevation (Kruskal–Wallis test, $\chi^2 = 88.7$, df = 6, $P < 0.0001$), with high marsh having significantly higher elevation than several zones, well above MHHW (Fig. 3), and low marsh having a median elevation below MHHW. The median elevation of all zones within SFC was below MHHW before restoration, but the grazed zone had the highest median elevation at SFC while the cropped zone was the lowest. Relative elevation differences among zones following restoration were similar to patterns during the pre-restoration sampling period (Kruskal–Wallis test, $\chi^2 = 89.5$, df = 6, $P < 0.0001$).

Groundwater Hydrology

During the pre-restoration sampling period, groundwater levels at SFC were much lower and had a different temporal signature, than in reference high marsh (Fig. 4; Supplemental Fig. 2). In the post-restoration sampling period, groundwater rose at SFC stations but changed little in reference high marsh (Supplemental Table 1). Restoration led to an increase in median groundwater levels at SFC ranging from 0.02 to 0.27 m during the wet season, and even greater increases during the dry season as shown by all stations where comparison was possible (increase in median level of 0.71, 0.56, and 0.51 m for stations 9, 16, and 37, respectively). Change in groundwater level at stations 4, 28, and 73 during the dry season could not be quantified because prior to restoration, groundwater often dropped below the well sensor.

Daily groundwater range varied by station, season, tidal phase, and sampling period (Fig. 5a; Supplemental Table 2). Daily groundwater range increased considerably at SFC stations following restoration, with sampling period explaining > 77% of the variability in range at SFC stations 9, 16, and 37, all in the middle zone (Fig. 5b; Supplemental Table 2). At SFC stations 4, 28, and 73 (where only wet season data were examined), sampling period accounted for 54–79.5% of the variability in range. Season and tidal phases also explained some variability in groundwater level, but these effects were considerably less important than sampling period at SFC stations. At the two high marsh stations, season, tidal phase, and sampling period all had statistically significant effects on groundwater range, though tide phase and season explained several times more variability than the sampling period (Fig. 5b; Supplemental Table 2).

Soil Accretion and Characteristics

Annual rates of soil accretion varied (one-way ANOVA; $F_{3,18} = 11.8$; $P = 0.0002$) from an average of 12.2 mm yr^{-1} (95% CL = 9.9, 14.5) in the cropped zone at SFC to only

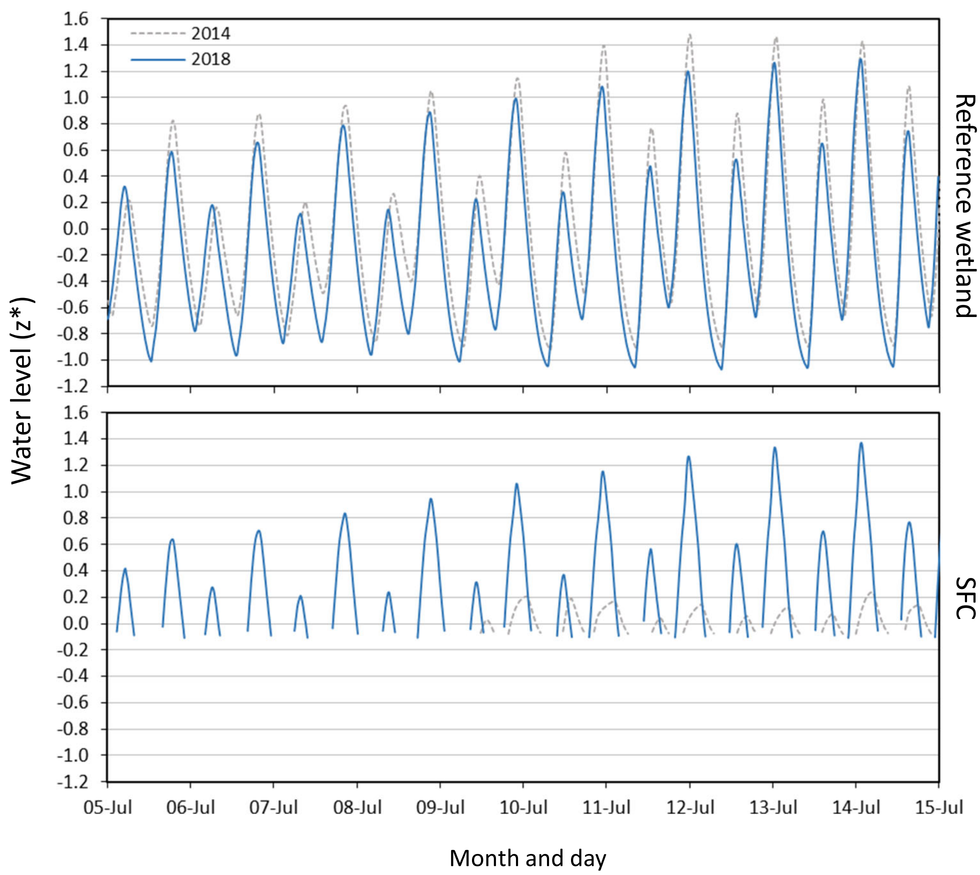

Fig. 2 Example time series of channel water levels recorded in the Trask River near a reference wetland (top panel) and within SFC (bottom panel) for a brief part of the dry season during pre- and post-restoration sampling periods. Water levels are expressed relative to local tide range (z^*) with values of 0.0 = MTL and 1.0 = MHHW. The Trask River reference logger was deployed in the subtidal, so it was always submerged, while the SFC logger was on a tributary channel in the middle of the SFC site (zone M) at about MTL. Pre-restoration (2014) water levels are in gray dashed lines; post-restoration (2018) values are in solid blue

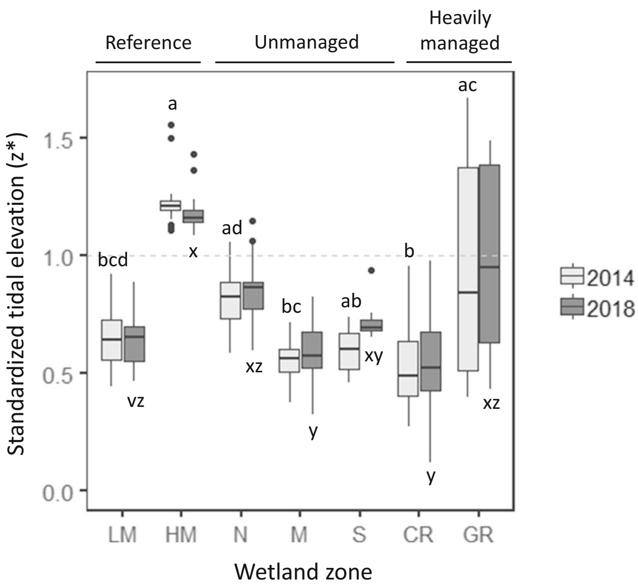

Fig. 3 Boxplots of standardized tidal wetland elevation (z^*) in 2014 and 2018 by wetland zone. LM, reference low marsh; HM, reference high marsh; N, SFC north zone; M, SFC middle zone; S, SFC south zone; CR, SFC cropped zone; GR, SFC grazed zone. Boxes indicate the 25–75% range of values, horizontal lines indicate medians, whiskers represent up to 1.5 times the interquartile range of values, and dots indicate outliers. Letters above box plots indicate significant differences among wetland zones in 2014 (pre-restoration sampling period); letters below boxes indicate significant differences in 2018 (post-restoration sampling period)

2.8 mm yr^{-1} (95% CL = 0.3, 5.3) in reference high marsh (Fig. 6). Reference low marsh (mean = 6.3; 95% CL = 3.8, 8.8) and other zones at SFC (mean = 8.0; 95%CL = 5.7, 10.2) had intermediate rates. There was a negative linear relationship between accretion and wetland elevation at SFC (n = 12; R^2_{adj} = 0.49; P = 0.007; Fig. 7a), but not for reference wetlands (n = 10; R^2_{adj} = 0.17; P = 0.13).

Before restoration, SFC soils were fresh (< 0.5 ppt) during the summer in all zones, but reference marsh soils were in the low mesohaline range (Kruskal–Wallis test, χ^2 = 25.2, df = 5, P = 0.0001; Table 2). Following restoration, soil salinities at SFC rose to the mesohaline range and all zones were similar to reference marshes ($F_{5,24}$ = 1.2, P = 0.32). Change in salinity between pre- and post-restoration sampling periods varied by wetland zone ($F_{5,24}$ = 4.0, P = 0.009), with the north, middle, and cropped zones all having a significant increase in salinity following restoration and the south zone having a nearly significant increase (Fig. 8a). Soil pH also varied by zone during pre-restoration sampling, with SFC zones tending to be more acidic than reference marshes ($F_{5,32}$ = 13.0, P < 0.0001). Following restoration, zones also varied in pH ($F_{5,28}$ = 3.4, P = 0.02), and pre- versus post-restoration change in pH differed by zone ($F_{5,28}$ = 7.8, P = 0.0001). The middle and cropped zones had the largest increases in pH (1.0 and 0.5 units, respectively; Fig. 8b). There were no significant differences in soil carbon content between zones in the pre-restoration sampling period ($F_{5,32}$ = 1.5, P = 0.23) or after restoration ($F_{5,28}$ = 0.7, P = 0.63), and the degree of pre- to post-restoration change did not vary by zone ($F_{5,28}$ = 1.7, P = 0.18; Fig. 8c).

Within SFC, the degree of pre-to-post-restoration change in soil parameters was correlated with wetland elevation. Lower-elevation plots increased more in salinity (R_{adj}^2 = 0.34, P = 0.004; Fig. 7b) and pH (R_{adj}^2 = 0.59, P < 0.0001; Fig. 7c) than higher elevation plots, although there was no relationship between elevation and change in soil carbon content (R_{adj}^2 = −0.01, P = 0.41).

Plant Cover and Richness

During the pre-restoration sampling period, total plant cover was high in reference marshes and at all five SFC zones, ranging from 91 to 119% (Kruskal–Wallis ANOVA, χ^2 = 31.8, df = 6, P < 0.0001; Table 3). Total plant cover remained high in both low marsh and high marsh during the post-restoration sampling period, but varied considerably among SFC zones, from a high of 96% in the north zone to only 35% in the cropped zone (Kruskal–Wallis ANOVA, χ^2 = 82.9, df = 6, P < 0.0001; Table 3). Total plant cover declined in all SFC zones following restoration, but zones varied significantly in their degree of change (Kruskal–Wallis ANOVA, χ^2 = 56.9, df = 6, P < 0.0001; Fig. 9a).

Native species cover dominated low and high reference marshes during both sampling periods but tended to be considerably lower in SFC zones during both periods (Kruskal–Wallis ANOVA, pre-restoration: χ^2 = 67.5, df = 6, P < 0.0001; post-restoration: χ^2 = 86.7, df = 6, P < 0.0001; Table 3). The north, middle, and south zones at SFC all lost a large fraction of native plant cover after restoration, though the change was not statistically significant (χ^2 = 9.0, df = 6, P = 0.18; Fig. 9b). In both sampling periods, non-native plant cover was higher in SFC zones compared to reference marshes (2014: $F_{6,171}$ = 19.5, P < 0.0001; 2018: Kruskal–Wallis ANOVA, χ^2 = 43.4, df = 6, P < 0.0001). Like native plant cover, non-native cover also tended to decline following restoration; loss was highest in the copped and grazed zones at SFC (Kruskal–Wallis ANOVA, χ^2 = 38.2, df = 6, P < 0.0001; Fig. 9c).

In the pre-restoration sampling period, plot-level plant species richness was nearly two-fold higher in high reference marsh and the grazed zone at SFC than in the other wetland zones ($F_{6,171}$ = 19.0, P < 0.0001; Table 3). In 2018, high marsh, and the south and grazed zones within SFC had the highest species richness ($F_{6,171}$ = 9.1, P < 0.0001). Pre- versus post-restoration change in richness varied by wetland zone (Kruskal–Wallis ANOVA, χ^2 = 12.6, df = 6, P = 0.05), with a tendency for species gain in reference high marsh and in the middle and south zones at SFC (Fig. 9d).

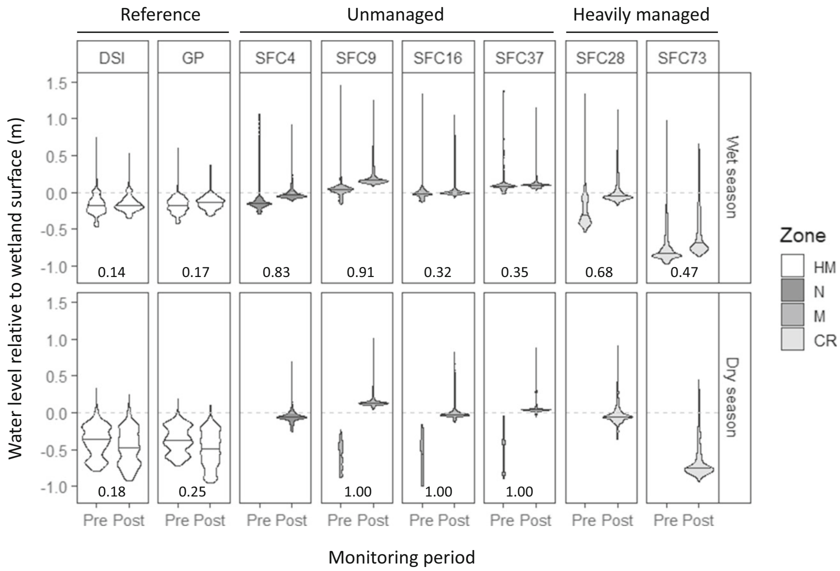

Fig. 4 Violin plots illustrating the distribution of groundwater levels relative to the soil surface (dashed line) at two high marsh reference locations (DSI and GP) and six SFC locations for pre- and post-restoration sampling periods, during the Oregon wet season (Dec–Mar) and dry season (June–Sept). Values above zero indicate water overtopping the wetland surface (e.g., high tide) Horizontal lines indicate median groundwater levels, and the width of the violins are kernel density estimates of the number of observations by water level. Numbers below the violins show Kolmogorov-Smirnov test statistics, D, for pre- versus post-restoration comparisons at each station. D ranges from 0 (no difference in distribution) to 1 (maximum difference in distribution). CR, cropped zone; HM, high marsh zone; M, middle zone; N, north zone

At SFC, there was a greater post-restoration loss of total plant cover at lower wetland elevations (linear regression, $R_{adj}^2 = 0.13$, $P < 0.0001$). There was no relationship between elevation and change in native species cover ($R_{adj}^2 = 0.0$, $P = 0.81$), but non-native species cover tended to decrease more at lower elevations ($R_{adj}^2 = 0.08$, $P = 0.0007$). Change in species richness was not correlated with elevation within SFC ($R_{adj}^2 = 0.0$, $P = 0.98$).

Cumulative plant species richness determined by species accumulation curves was low in reference low marsh and about 5-fold higher in high marsh during both sampling periods (Fig. 10). At SFC, the north, middle, and cropped zones had intermediate levels of plant richness, which increased following restoration, especially in the cropped zone. Except for low marsh, few of the species accumulation curves reached

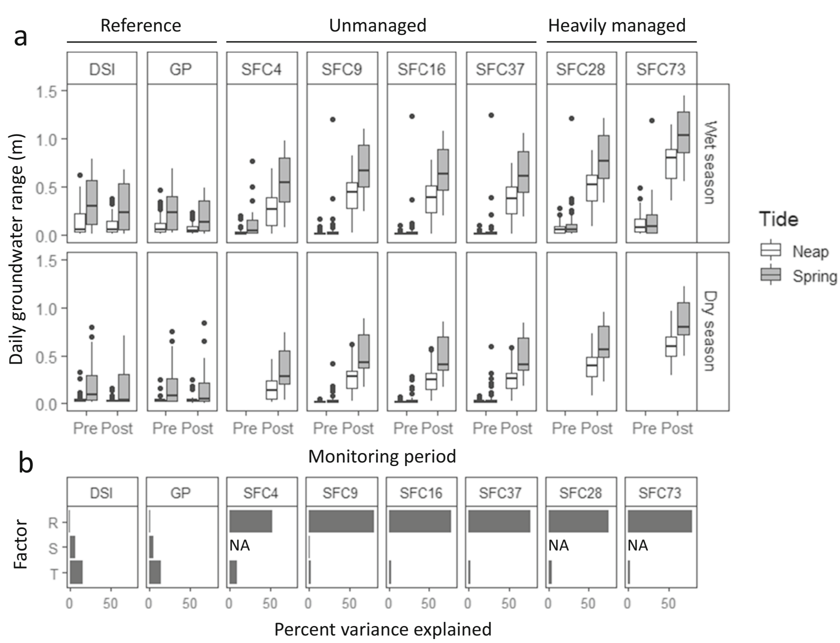

Fig. 5 (**a**) Variability in daily groundwater vertical range at two high reference marsh tidal stations (DSI, GP) and six SFC stations during wet and dry seasons before and after restoration. Data are further separated by neap and spring tide cycles. (**b**) Percent variation in groundwater range for each site explained by model factors. R, restoration (sampling period); S, season; T, monthly tidal phase

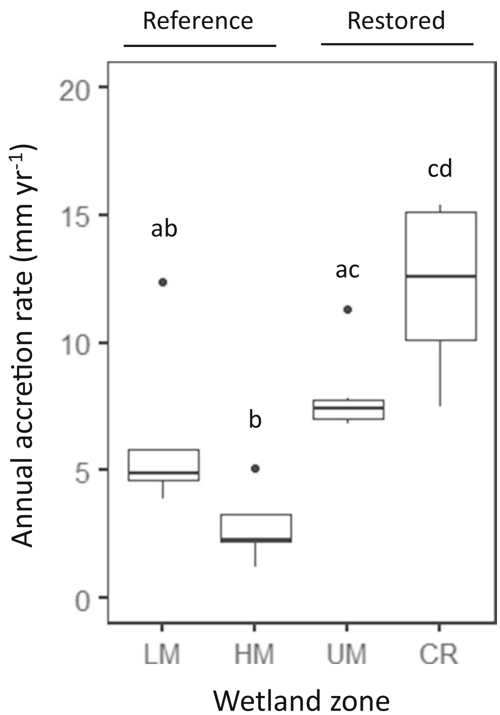

Fig. 6 Annual rates of soil accretion (mm yr^{-1}) measured in feldspar marker horizon plots by wetland zone. Accretion rates are average annual rates between 2013 and 2018. LM, reference low marsh; HM, reference high marsh; UM, SFC unmanaged marsh zones (pooled north, middle, and south zones); CR, SFC cropped zone. Wetland zones with the same letter are not significantly different from each other

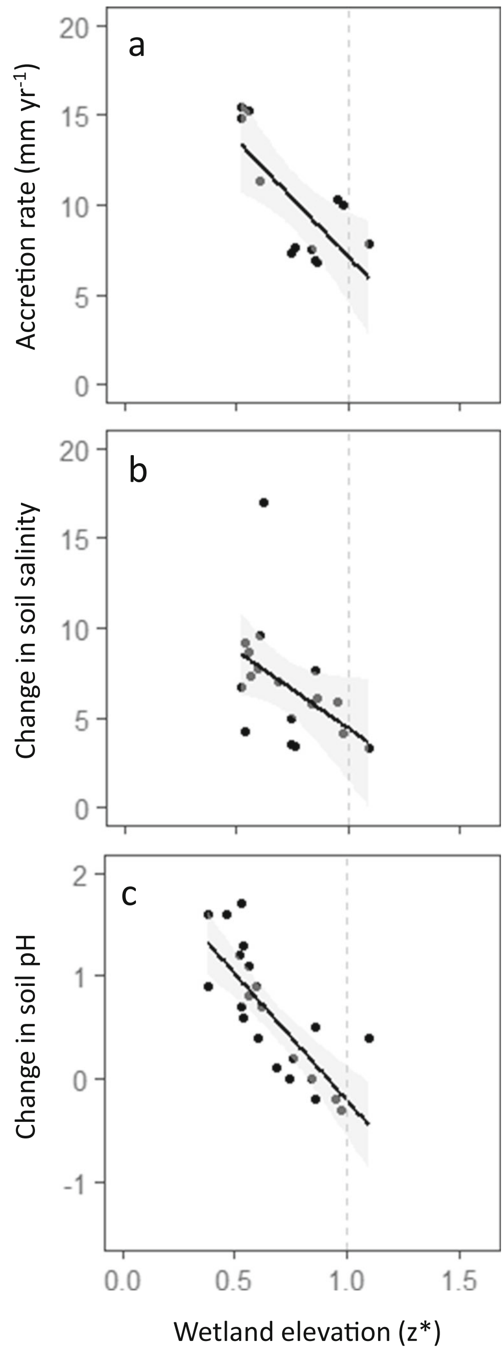

Fig. 7 Magnitude of soil accretion (**a**), change in soil salinity after restoration (**b**), and change in soil pH after restoration (**c**) with wetland elevation (measured in 2013) within SFC. Vertical dashed lines indicate local MHHW that separates low marsh from high marsh. Black lines are linear regressions with 95% confidence intervals in gray

asymptotes, suggesting further sampling would be needed to obtain accurate estimates of total species pools in each zone.

Vegetation Composition

Pre-restoration plant cover at SFC was dominated by *Phalaris arundinacea* (reed canary grass) (Supplemental Table 3). Dominant species in reference low marsh were *Carex lyngbyei* and *Agrostis stolonifera*, while reference high marsh had a greater variety of dominant and sub-dominant species, including *Deschampsia cespitosa* and *Potentilla anserina*. In the early post-restoration period, we observed a variety of native wetland species recruiting to the SFC site, especially in the cropped and middle zones, including *Atriplex prostrata*, *C. lyngbyei*, *D. cespitosa*, *Eleocharis palustris*, and *Eleocharis parvula* as well as the non-native species, *Cotula coronopifolia* (Fig. 11). Benthic mats of the xanthophyte alga, *Vaucheria*, were also relatively common on otherwise unvegetated soils at SFC.

Plant assemblages in reference low marsh were relatively distinct from SFC assemblages in the pre-restoration sampling period while high marsh had some compositional overlap with SFC before and after restoration (Fig. 12). In reference marshes, there was little change in species composition between pre- and post-restoration sampling periods. In contrast, composition changed in the majority of the five SFC zones between sampling periods (Supplemental Table 4). Additionally, beta diversity (manifested as the degree of dispersion among plots in the NMDS space) decreased for the cropped zone and increased for the grazed zone following restoration but did not change significantly for the other zones.

Table 2 Mean (± SE) soil pore water salinity, pH, and carbon content (%) by wetland zone for pre-restoration (2014) and post-restoration (2018) periods. Sample sizes are in Table 1; no samples were collected from the GR zone. Within each year and parameter, zones sharing the same letters or without letters were not significantly different; pore water salinity in 2018 and soil carbon content in 2014 and 2018 did not differ significantly by zone

Wetland type	Zone	Soil salinity		Soil pH		Carbon content (%)	
		2014	2018	2014	2018	2014	2018
Reference	LM	8.0 ± 0.9[a]	10.5 ± 2.3	5.9 ± 0.1[ab]	5.6 ± 0.1[ab]	4.8 ± 0.5	6.8 ± 0.8
Reference	HM	6.3 ± 1.5[a]	7.0 ± 1.4	5.7 ± 0.1[ab]	5.8 ± 0.1[ab]	8.3 ± 1.5	7.4 ± 1.1
SFC	N	0.2 ± 0.0[ab]	5.0 ± 0.9	5.1 ± 0.1[c]	5.3 ± 0.2[a]	7.4 ± 1.1	6.5 ± 1.0
SFC	M	0.2 ± 0.0[ab]	11.1 ± 2.2	5.0 ± 0.1[c]	6.0 ± 0.1[b]	7.6 ± 0.8	7.9 ± 0.5
SFC	S	0.2 ± 0.0[ab]	6.2 ± 1.0	5.4 ± 0.0[bc]	5.5 ± 0.0[ab]	5.7 ± 0.5	6.3 ± 0.1
SFC	CR	0.1 ± 0.0[b]	9.2 ± 2.8	5.3 ± 0.1[ac]	5.8 ± 0.2[ab]	7.7 ± 0.6	6.7 ± 0.2

Discussion

Tidal wetland restoration is an important management tool to mitigate for coastal wetland loss and enhance wetland functions and services in estuaries. In the PNW, restoration may be implemented with the goals of increasing salmonid habitat (Koski 2009) and abatement of urban flooding (this study). However, the development and eventual success of restoration projects may depend in part on estuarine setting, including wetland elevation, salinity regime, and prior land-use/land-cover differences at a site. Overall, restoring natural tidal hydrology at SFC led to substantial change towards reference conditions shortly after restoration for many key ecosystem parameters that we measured (groundwater dynamics, soil pH, soil salinity), while differences between SFC and reference wetlands remained for other metrics (native species cover, plant composition), and additional parameters (soil carbon, total plant cover) at SFC were already similar to reference conditions before dike removal. Some of the changes we observed during early recovery at SFC were consistent with our hypotheses that pre-restoration differences in land use/land cover and wetland elevation were associated both with initial site conditions, and with the magnitude of change following restoration.

Pre-Restoration Land-Use/Land-Cover Differences

We hypothesized that land-use/land-cover differences could affect early restoration dynamics. In tidal wetlands, it has already been documented that channels can affect wetland

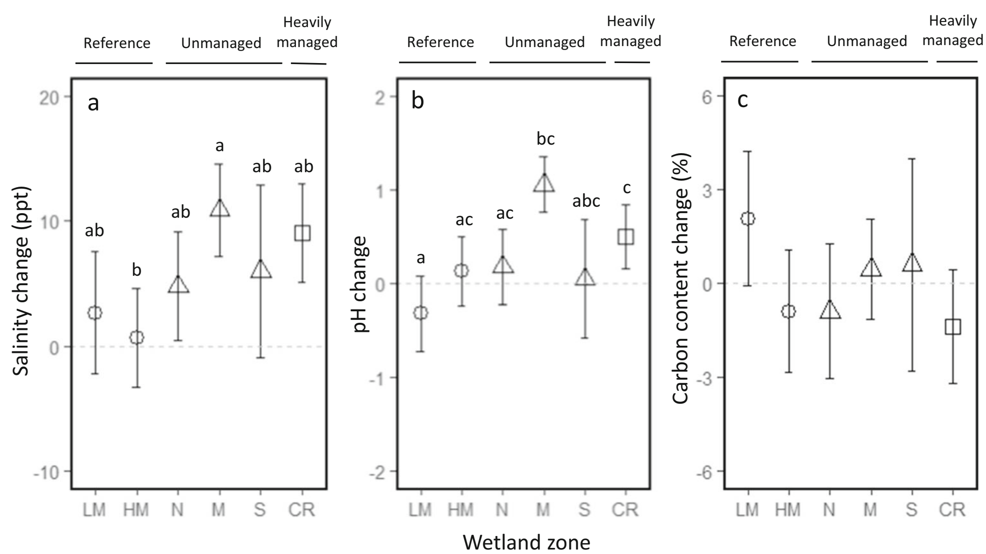

Fig. 8 Pre- to post-restoration sampling period change (means ± 95% confidence intervals) in near-surface soil parameters at SFC and reference marshes. (**a**) Summer pore water salinity, (**b**) pH, and (**c**) carbon content. The dashed line at zero indicates no change between sampling periods; positive values indicate higher values for the post-restoration sampling period relative to the pre-restoration sampling period. Error bars not crossing zero were considered significant differences between sampling periods. Zones sharing the same letters did not show statistically different changes in soil parameters. Reference wetlands are in open circles, unmanaged zones are in open triangles, and heavily managed zones are in open squares

Table 3 Total, native, and non-native plant percent cover and plot-level species richness for pre-restoration (2014) and post-restoration (2018) sampling periods (mean ± SE). Sample sizes are in Table 1. Within each year and parameter, zones sharing the same letters were not significantly different for each parameter per monitoring period. Plot cover > 100% indicates the presence of an overhanging canopy

Wetland type	Zone	Total plant cover		Native plant cover		Non-native plant cover		Species richness	
		2014	2018	2014	2018	2014	2018	2014	2018
Reference	LM	99 ± 0[ab]	95 ± 2[ad]	80 ± 5[a]	85 ± 4[a]	20 ± 5[a]	11 ± 4[a]	1.7 ± 0.2[a]	1.7 ± 0.2[a]
Reference	HM	100 ± 0[ab]	102 ± 3[a]	77 ± 7[a]	84 ± 6[a]	23 ± 7[a]	17 ± 5[a]	4.6 ± 0.4[b]	5.7 ± 0.5[b]
SFC	N	119 ± 9[a]	96 ± 2[ad]	41 ± 13[b]	23 ± 8[b]	78 ± 8[b]	73 ± 9[b]	2.6 ± 0.4[a]	2.2 ± 0.4[a]
SFC	M	102 ± 1[a]	66 ± 4[bc]	25 ± 5[b]	14 ± 4[b]	77 ± 5[b]	50 ± 5[bc]	1.8 ± 0.1[a]	2.6 ± 0.3[a]
SFC	S	106 ± 6[ab]	89 ± 13[ac]	49 ± 17[ab]	10 ± 5[b]	58 ± 13[ab]	78 ± 11[bc]	2.9 ± 0.6[a]	4.3 ± 0.7[ab]
SFC	CR	91 ± 3[b]	35 ± 5[b]	5 ± 2[b]	3 ± 1[b]	85 ± 3[b]	32 ± 5[ac]	2.5 ± 0.2[a]	2.8 ± 0.3[a]
SFC	GR	98 ± 1[ab]	63 ± 16[bcd]	18 ± 7[ab]	25 ± 10[b]	80 ± 7[b]	37 ± 12[ab]	4.9 ± 0.4[b]	4.1 ± 0.7[ab]

structure and processes (Wallace et al. 2005 and references therein), with many restored wetlands having lower channel densities than reference wetlands (Lawrence et al. 2018). Disturbance type and intensity may also influence ecosystem development. For instance, Sobrinho et al. (2016) found differences in tropical forest recovery between different types of land-use disturbance. Moreover, greater floodplain disturbance in Australia was shown to impact native species success relative to non-natives in restoration (Dawson et al. 2017a).

High-intensity land uses prior to restoration, such as tilling, use of heavy machinery, or intensive grazing in agriculture, may particularly impact wetland elevation and soil characteristics. In support of this hypothesis, we found that the heavily managed cropped zone had the lowest elevation within the SFC site prior to restoration (Fig. 3) and tended to have relatively lower groundwater levels (Fig. 4). We cannot definitively attribute lower elevations to land use since elevation data prior to diking are not available, but elevations at the cropped zone were likely to have been similar to nearby reference high marsh at the Dry Stocking Island. Based on this assumption, Brophy et al. (2018) estimated that the cropped zone had subsided 0.71 m below its pre-diking elevation. Regular use of heavy machinery and dewatering of soils—both typical of the cropped zone—would be expected to lead to soil compaction and elevation loss, and diked wetland subsidence of similar magnitude has been observed elsewhere on the Oregon coast (Brophy et al. 2015).

The high-intensity land-use zones at SFC (cropped and grazed zones) had the lowest cover of native plant species and the highest cover of non-native species before restoration

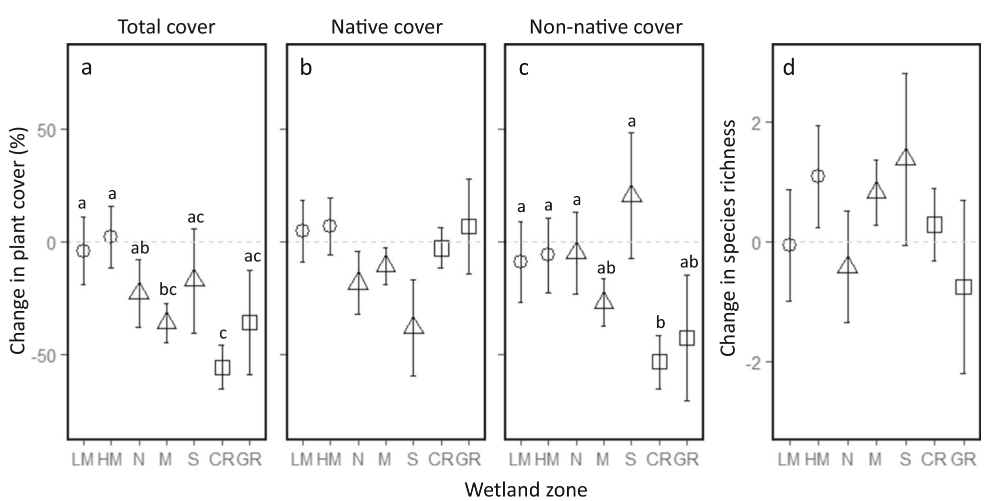

Fig. 9 Pre- to post-restoration sampling period change (means ± 95% confidence intervals) in vegetation cover and species richness at SFC and reference marshes. (**a**) Total plant cover, (**b**) native plant cover, (**c**) non-native plant cover, and (**d**) plot-level species richness. The dashed line at zero indicates no change between sampling periods; positive values indicate higher values for the post-restoration sampling period relative to the pre-restoration sampling period. Error bars not crossing zero were considered significant differences between sampling periods. Zones sharing the same letters did not show statistically different changes (there were no significant pair-wise differences among zones for native cover and total richness so letters are not shown). Reference wetlands are in open circles, unmanaged zones are in open triangles, and heavily managed zones are in open squares

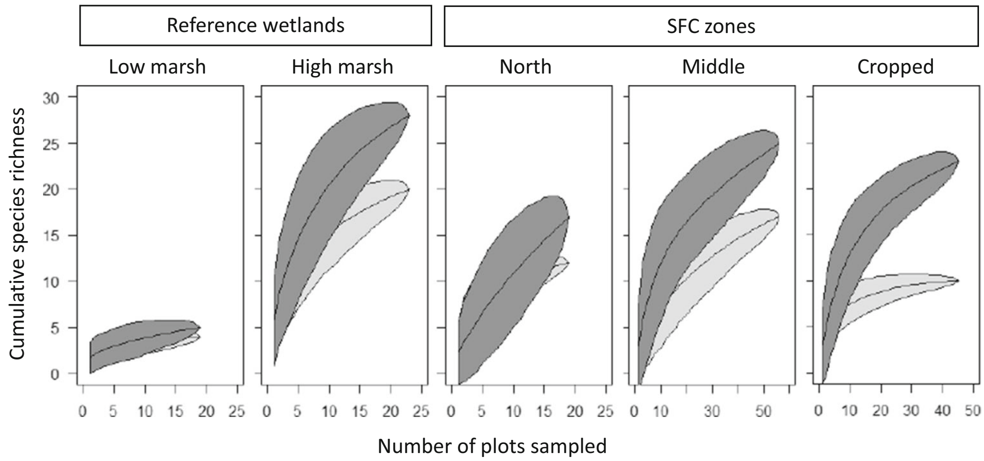

Fig. 10 Cumulative plant species richness in five wetland zones during the pre-restoration sampling period (light gray) and post-restoration sampling period (dark gray) with species accumulation curves (± 95% confidence bands). Only zones with larger sample sizes are shown (S and GR excluded)

(although not statistically significant; Table 3), suggesting potentially greater impacts of high intensity land use on vegetation composition. However, the grazed zone had the highest species richness of all SFC zones before restoration, differing substantially from the cropped zone (Table 3). The grazed zone also had relatively high elevation, unlike the nearby cropped zone (Fig. 3). Grazed sites, especially if grazed only seasonally or intermittently, could have characteristics (such as higher elevation and higher plant diversity) that help them recover faster after restoration than sites with a history of cropping, although this needs to be tested more thoroughly. The south zone was a freshwater wetland mitigation site for a number of year before restoration, and native plantings had been established as part of the mitigation action. Although this zone was relatively low in elevation like the cropped zone (Fig. 3), it had more native species cover than other zones before restoration, perhaps due to plantings. However, it also lost most of this native cover when tidal flow was initially restored, presumably as freshwater-adapted plants died back from salinity intrusion (Table 3) and increased inundation. These findings from SFC suggest that differences in land cover/land use before restoration could imprint unique signatures on recovery of wetland soils and vegetation across a spatially heterogeneous site.

Elevation and Ecosystem Change

Like land-use/land-cover differences, we hypothesized that wetland elevation could influence restoration trajectories. Land use and elevation are intertwined at SFC since elevation likely affected past land-use decisions during the diked period, and conversely, past land uses likely led to change in elevations. Prior to restoration, wetland surface elevation across much of SFC was comparable to reference low marsh, although there was variability in elevation within and between SFC zones (Fig. 3).

Restored tidal wetlands are often characterized by lower wetland surface elevations than their historical type prior to diking, due to soil compaction and organic matter oxidation during the diked period (Frenkel and Morlan 1991; Borde et al. 2012). Our hypothesis was supported by the following several measured parameters: lower elevations had greater increases in salinity and pH (Fig. 7) and a greater loss of total and non-native plant cover compared to less frequently flooded higher-elevation areas. At lower elevations, we observed many newly recruited individuals of native estuarine wetland species, such as *C. lyngbyei*, *D. cespitosa*, and *P. anserina*, and the non-native *C. coronopifolia.* In contrast, at the somewhat higher (and fresher) north zone, freshwater

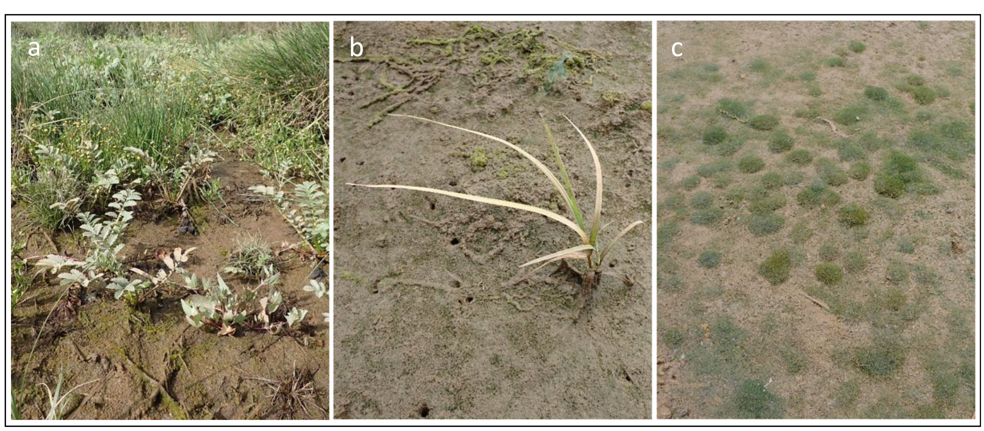

Fig. 11 Examples of vascular plant and algal assemblages at the SFC site in early post-restoration: (**a**) mix of native and non-native species in the M zone including *Potentilla anserina*, *Cotula coronopifolia*, and *Atriplex prostrata*; (**b**) green macroalgae (*Ulva*) and native *Carex lyngbyei* in the S zone; and (**c**) sediment colonization by the xanthophyte alga, *Vaucheria* sp.

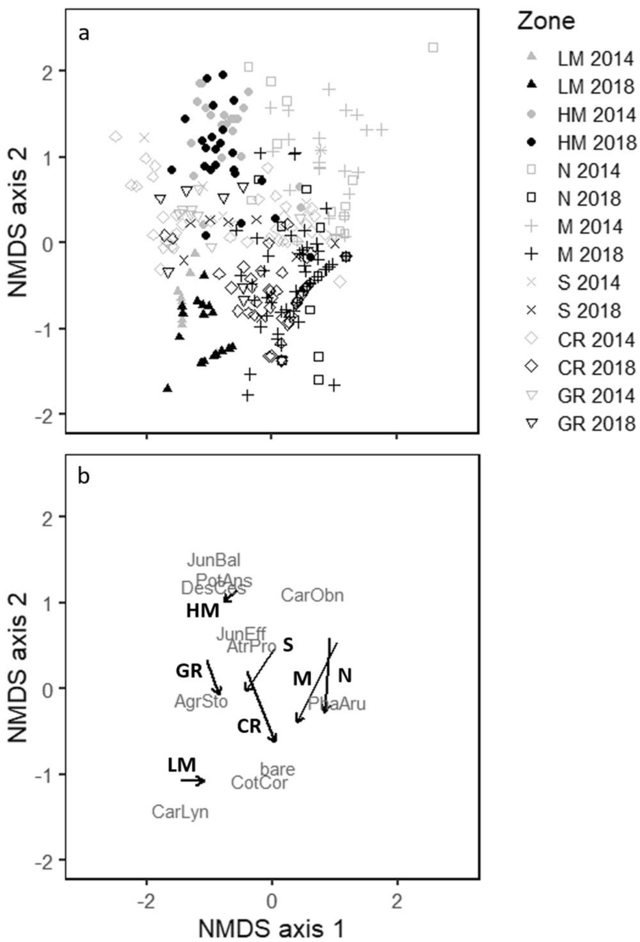

Fig. 12 Non-metric multidimensional scaling plot of differences in plant species composition at SFC and in reference wetlands. (**a**) All vegetation plots, with pre-restoration composition in gray and post-restoration composition in black. (**b**) Arrows on the same NMDS plot show the directional change in NMDS space between pre- and post-restoration composition (difference between centroids) for each of the seven wetland zones. The centroids of 10 select common species and bare space are also shown. LM, low marsh; HM, high marsh; N, north; M, mid; S, south; CR, cropped; GR, grazed. NMDS stress = 0.14

species like invasive reed canary grass tended to persist, suggesting elevation-linked differences in the tempo of vegetation change.

Consistent with our findings, Karberg et al. (2018) documented faster recovery of salt marsh plants in a New England restoration project at lower elevations supportive of emergent vascular plants than at higher elevations where freshwater species persisted. Additionally, during the early post-restoration period at the Ni-les'tun tidal restoration project in southern Oregon, cover of non-native tall fescue declined more at lower elevations early in restoration (Brophy et al. 2014). Together, these studies suggest that under a passive restoration strategy of simple dike removal without other interventions, the vegetation in higher-elevation areas may transition more slowly (if at all) from non-tidal species to the diverse assemblages characteristic of mature high marsh wetlands in the PNW (Janousek and Folger 2014). However, the pace of soil and vegetation recovery along elevation gradients could differ in tidal wetlands in other climates. For example, high elevation areas in drier and warmer regions may rapidly lose non-native vegetation after restoration if high evapotranspiration and low precipitation lead to rapid development of highly saline soils that can only be tolerated by native halophytes.

At SFC, we observed higher soil accretion rates at lower elevations as observed in other studies (Bricker-Urso et al. 1989; Frenkel and Morlan 1991). Some plots within SFC accreted at relatively high rates (15 mm yr^{-1}), greater than observed for instance in the lower elevation areas of a restoring marsh in the Salmon River Estuary in central Oregon (5–7 mm yr^{-1}; Frenkel and Morlan 1991). Further colonization and expansion of plant cover in restored low marsh may facilitate high rates of sediment deposition, a positive feedback that helps the site gain overall elevation and gradually approach historical vegetation composition and wetland functions.

Overall Ecosystem Change at SFC

The removal of dikes surrounding SFC effectively restored hydrologic connectivity and rapidly initiated change towards reference tidal wetland conditions. However, even without substantial spatial heterogeneity due to land use/land cover or elevation differences, various abiotic and biotic components of a restoring site may recover at different rates (Nordström et al. 2014). Surface and sub-surface hydrology may be among the fastest to change once full tidal inundation is restored. About a year after removal of dikes, we found that substantial changes in groundwater hydrology had already occurred at SFC, including large variation in daily groundwater range (Fig. 5a). Spencer et al. (2017) also observed recovery of tidally driven groundwater variability at a restored site in the UK, although it was still more muted than the reference wetland after several decades. Groundwater and surface hydrology did vary spatially within SFC; for instance station SFC-9 was in a location of continuous ponding (Fig. 5), a feature observed in other restoration projects (Lawrence et al. 2018). Ponding could be related to low channel density and/or compacted soils. By contrast, station SFC-73 had dynamic groundwater levels before and after restoration, likely due to groundwater outflow at low tide into the large channel adjacent to this station.

Soil properties may recover slowly or rapidly depending on the restoration project. At SFC, both soil salinity and pH changed rapidly towards reference conditions soon after restoration (Table 2, Fig. 8a,b). Moreover, even though diking can often lead to loss of soil organic matter (Portnoy 1999; Drexler et al. 2009; Spencer et al. 2017), SFC soils already had carbon content similar to reference wetlands before dike removal (Table 2), perhaps due to the high cover of vegetation across all zones at the site. In other tidal wetland restoration sites however, slower development of soil carbon pools and bulk density, and persistent differences in restored soil porosity at depth, have been observed (Spencer et al. 2017). Ballantine and Schneider (2009) observed a chronosequence of restored non-tidal freshwater wetlands and found that even after 50 years, restored sites still differed from reference sites in terms of soil organic matter content, bulk density, and cation exchange capacity. They hypothesized that soil development might proceed more rapidly in estuarine wetlands because of their high rates of hydrologic connectivity.

Following tidal reconnection, we observed rapid loss of pre-restoration vegetation cover across much of SFC (particularly *Phalaris arundinacea*, *Alopecurus pratensis*, and *Carex obnupta* in the cropped and middle zones) and establishment of pioneering tidal marsh vascular plants (such as *Carex lyngbyei* and *Cotula coronopifolia*; Supplemental Table 3) and benthic algae. Benthic algae may colonize marsh soils very rapidly and help stabilize them for vascular plant colonization (Underwood 1997; Janousek et al. 2007; Nordström et al. 2014). Estuarine-adapted herbaceous plants may also re-establish within several years of hydrologic restoration (Underwood 1997), although recovery may vary by project and be slower for overall species composition (Mossman et al. 2012). In the PNW, non-native *C. coronopifolia* rapidly colonizes restored tidal wetlands (Cornu and Sadro 2002). It was observed in the 2018 plots and found in particular abundance in the cropped and grazed zones in 2018 and a year later (pers. observation). Woody species, such as *Picea sitchensis*, may eventually establish and persist in the lower-salinity, higher-elevation areas at SFC (Brophy 2009), but development of this wetland type may take many decades.

Recommendations for Restoration Monitoring

We documented the early phase of recovery of hydrologic processes and several soil and vegetation attributes at SFC, yet long-term monitoring of a range of physical, chemical, and biological parameters at the site is still needed (Frenkel and Morlan 1991; Zedler 2000). Early changes in SFC soils and vegetation were associated, at least in part, with elevation and land use differences present before restoration. Through long-term sampling, it will be possible to evaluate if this pre-restoration heterogeneity has longer-term impacts on a restored wetland's structure and function, and to document the rate at which different aspects of ecosystem structure and function recover.

The pre-restoration land-use/land-cover types across SFC were unreplicated, making causal inference about specific disturbances or land-cover classes difficult, but our results do suggest that consideration of land-cover type and disturbance history in diked areas could be important for restoration design and monitoring. Pre-restoration land use/land cover helps determine initial wetland elevation, soil properties, and vegetation composition, factors which then interact with elevation and salinity gradients upon restoration to determine succession trajectories. Documenting initial spatial heterogeneity across a project site can help inform restoration planning, such as whether to invest resources to manage invasive species or plant natives. Furthermore, consideration of spatial heterogeneity should inform monitoring design, such as stratification of sampling by pre-restoration land-use type or elevation (Bookout and Bruland 2019). As tidal wetland restoration projects become larger in area in order to meet regional and state restoration goals for species recovery, estuarine function, and climate change adaptation, it will be increasingly important to evaluate how spatial heterogeneity across a new project site affects the magnitude and rate of recovery.

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Data availability Data for this study are available at the Knowledge Network for Biodiversity at https://doi.org/10.5063/F1FJ2F5C (Janousek et al. 2020).

Compliance with Ethical Standards

Conflict of Interest The authors declare that they have no conflict of interest.

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