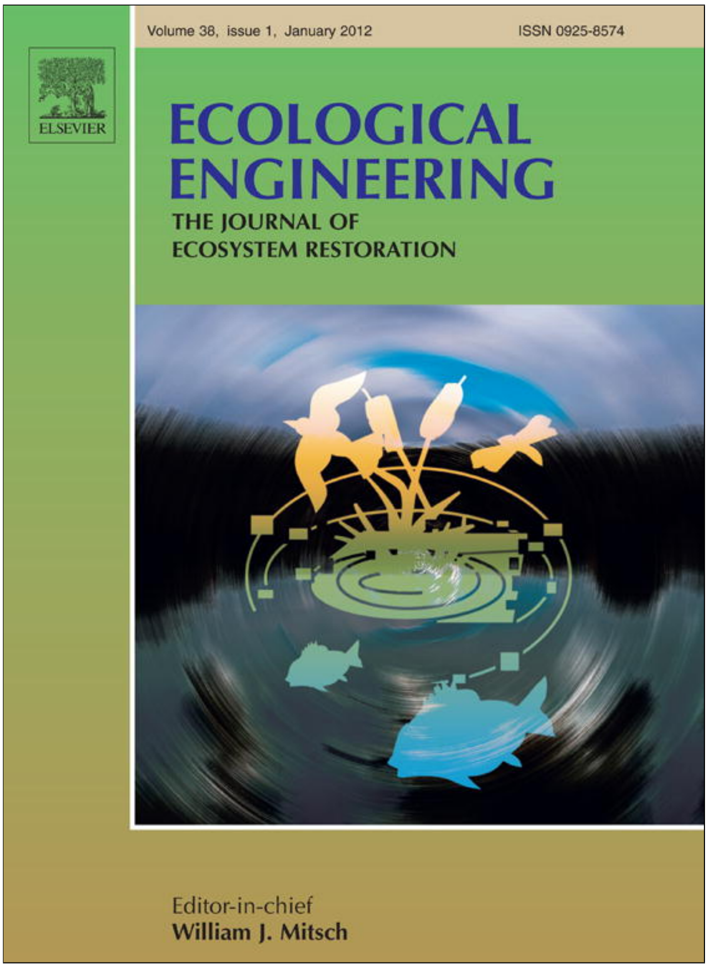
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Trajectory of early tidal marsh restoration: Elevation, sedimentation and colonization of breached salt ponds in the northern San Francisco Bay

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ABSTRACT

Tidal marsh restoration projects that cover large areas are critical for maintaining target species, yet few large sites have been studied and their restoration trajectories remain uncertain. A tidal marsh restoration project in the northern San Francisco Bay consisting of three breached salt ponds (≥300 ha each; 1175 ha total) is one of the largest on the west coast of North America. These diked sites were subsided and required extensive sedimentation for vegetation colonization, yet it was unclear whether they would accrete sediment and vegetate within a reasonable timeframe. We conducted bathymetric surveys to map substrate elevations using digital elevation models and surveyed colonizing Pacific cordgrass (*Spartina foliosa*). The average elevation of Pond 3 was 0.96 ± 0.19 m (mean ± SD; meters NAVD88) in 2005. In 2008–2009, average pond elevations were 1.05 ± 0.25 m in Pond 3, 0.81 ± 0.26 m in Pond 4, and 0.84 ± 0.24 m in Pond 5 (means ± SD; meters NAVD88). The largest site (Pond 3; 508 ha) accreted 9.5 ± 0.2 cm (mean ± SD) over 4 years, but accretion varied spatially and ranged from sediment loss in borrow ditches and adjacent to an unplanned, early breach to sediment gains up to 33 cm in more sheltered regions. The mean elevation of colonizing *S. foliosa* varied by pond (F = 71.20, df = 84, P < 0.0001) and was significantly lower in Ponds 4 and 5 compared with Pond 3 which corresponded with greater tidal muting in those ponds. We estimated 16% of Pond 3, 13% of Pond 4, and 24% of Pond 5 were greater than or equal to the median elevation of *S. foliosa*. Our results suggest that sedimentation to elevations that enable vegetation colonization is feasible in large sites with sufficient sediment loads although may occur more slowly compared with smaller sites.
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1. Introduction

Tidal marshes maintain endemic and endangered vertebrate species and key ecosystem services, but have undergone substantial habitat loss worldwide (Kennish, 2001; Zedler and Kercher, 2005; Greenberg et al., 2006). With growing recognition of their ecological value, numerous tidal marsh restoration projects are underway, such as in temperate coastal estuaries of the northern hemisphere that have lost a substantial proportion of historic distributions (Wolters et al., 2005; Zedler and Kercher, 2005; Greenberg et al., 2006; Konisky et al., 2006). San Francisco Bay is the largest estuary on the Pacific coast of North America, yet >90% of its wetlands has been converted to agriculture, urbanization, and commercial salt production (Nichols et al., 1986; Goals Project, 1999). As part of the effort to reverse these losses, former salt production ponds form the basis of the largest tidal marsh restoration in the western United States. Federal and State agencies have purchased over 11,000 ha across the Bay since 1994 with 50–90% of the total area slated for tidal marsh restoration (Goals Project, 1999; Jones and Stokes, 2004; URS Corporation, 2006; EDAW et al., 2007). Early restoration efforts included site grading, manipulation of water levels through ongoing water management, and extensive plantings of *Spartina* spp. (Williams and Faber, 2001). However, some early restoration efforts lost a substantial proportion of planted vegetation, progressed slowly, or were unfeasible to maintain (Race, 1985; Williams and Faber, 2001). In response to these prior experiences there has been increasing emphasis on the physical processes needed to support tidal marsh (Zedler et al., 1999; Williams and Faber, 2001; Athearn et al., 2010). Recent restoration projects have emphasized ecological engineering, such as natural site evolution post-breach to attain sediment elevations that support marsh vegetation (Teal and Weinstein, 2002; Williams and Orr, 2002; Simenstad et al., 2006).

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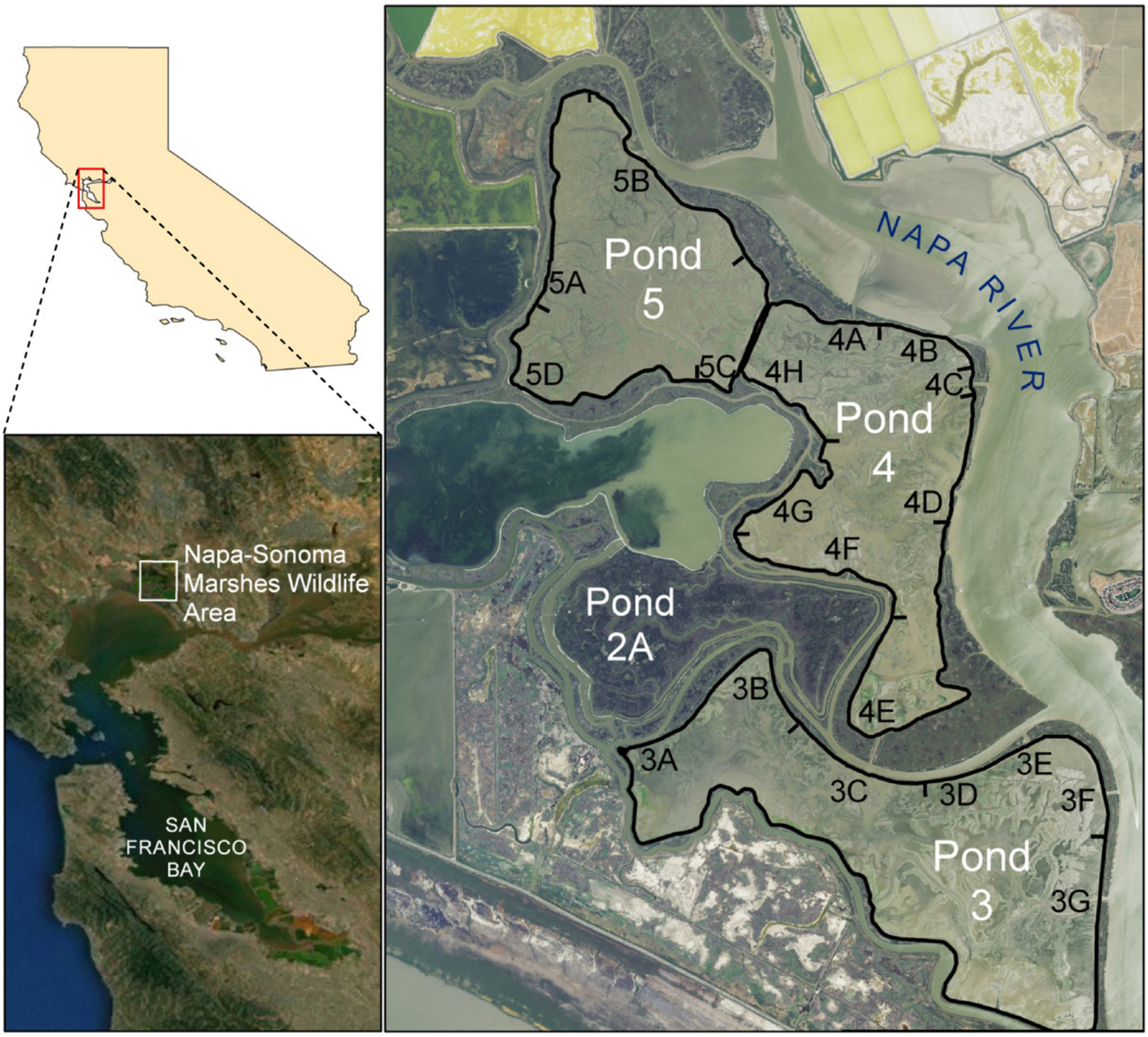

Fig. 1. Ponds 3, 4, and 5 within the Napa-Sonoma Marshes Wildlife Area of the northern reach of San Francisco Bay, California, with 19 pond breaches numbered by pond and letter and ditch blocks marked as rectangles extending from pond levees.

Imagery from Environmental Systems Research Institute, Inc., Redlands, CA and the National Agriculture Imagery Program (2009), Aerial Photography Field Office, Farm Services Agency, US Department of Agriculture.

Many factors affect sediment accretion rates of former diked areas following breaching. In their review of restored salt marshes, Williams and Orr (2002) found that sedimentation rates were influenced by initial site elevations that largely reflect the degree of subsidence from prior land use. Elevation is related with the duration and frequency of tidal inundation that in turn delivers sediment to a site. Local suspended sediment concentrations and flow dynamics that affect scour and resuspension also dramatically influence sedimentation rates (Krone, 1987; Williams and Orr, 2002). In addition, physical factors such as wind and wave erosion and sediment supply may be more important in large than in small restoration sites (Williams and Faber, 2001). Despite the importance of large sites (e.g., ≥300 ha) for maintaining target species, few large tidal marsh restoration sites have been studied (Zedler and Callaway, 2000; Weinstein et al., 2001; Wolters et al., 2005). In San Francisco Bay, over 600 ha of former salt ponds were breached from 1978–2005, yet most sites averaged <100 ha each (Williams and Orr, 2002; Callaway et al., 2009). The largest previously restored site in the northern estuary (Pond 2A; 212 ha) vegetated in approximately 3 years, perhaps in response to its high initial elevation, full tidal regime, high sediment supply, and brackish water inputs (Goals Project, 1999; Williams and Orr, 2002). A number of proposed restoration sites are substantially larger, yet it is unclear whether these large restoration sites will accrete sediment and vegetate within a reasonable timeframe.

The goal of most tidal marsh restoration projects is the development of a mature marsh plain that can support local populations of tidal salt marsh endemic species, such as federally and state endangered California clapper rail (*Rallus longirostris obsoletus*) and salt marsh harvest mouse (*Reithrodontomys raviventris halicoetes*), and state threatened California Black Rail (*Laterallus jamaicensis coturniculus*; Harvey et al., 1992; Goals Project, 1999) in the San Francisco Bay estuary. As sediment accretes in restored salt marshes, sites typically evolve from subtidal mudflats to intertidal marshes. In the northern estuary, the lowest zone of marsh vegetation is comprised primarily of native Pacific cordgrass (*Spartina foliosa*) that helps sequester sediment for development of higher marsh (Josselyn, 1983; Goals Project, 1999; Williams and Orr, 2002; Wallace et al., 2005). Numerous factors affect colonization of *S. foliosa* including proximity to the bay and drainage channels (Zedler et al., 1999; Sanderson et al., 2000), but the most important determinant is the surface elevation of the sediment (Simenstad and Thom, 1996; Zedler et al., 1999; Cornu and Sadro, 2002; Williams and Orr, 2002). However, there is variation in the elevations at which *S. foliosa* colonizes based on the local tidal and inundation regimes, and colonization elevations for particular sites remain difficult to predict (Atwater and Hedel, 1976; Zedler et al., 1999; Williams and Orr, 2002).

Our study focuses on a large-scale wetland restoration project in Ponds 3, 4, and 5 of the Napa-Sonoma Marshes Wildlife Area adjacent to the Napa River in the northern San Francisco Bay. The project area spans more than twice the total salt pond area restored to tidal action in San Francisco Bay to date (1175 ha total) and is one of the largest tidal marsh restoration areas on the west coast of the United States. The goals of the restoration were to restore large areas of formerly subsided, diked salt ponds to vegetated marshplain that initially consists of low salt marsh species *S. foliosa* and that would eventually transition to higher marsh (PWA, 2002). However, the pond bottoms had subsided to elevations too low in the tidal frame for marsh plants to establish or survive, thus substantial sedimentation would be required for vegetation establishment and colonization elevations were uncertain (PWA, 2002).

Our overarching goal was to assess the status of diked, subsided sites that had been breached over time and across sites. Our specific objectives were to (1) assess sedimentation over a 4-year period following breaches in a single pond where repeat surveys were conducted; (2) estimate the current elevations of the pond floors as well as breaches that affect the hydrologic and sedimentation changes for the three ponds; (3) estimate the distribution of elevations for colonizing *S. foliosa* within ponds; and (4) use the elevation data to estimate the area of the ponds available to support *S. foliosa* colonization. Our results also provide insights into use of restoration design elements that were included in the restoration project area.

2. Methods

2.1. Study area

The Napa-Sonoma Marshes Wildlife Area contains 12 former salt production ponds located on the west side of the Napa River in the northern reach of San Francisco Bay (Fig. 1). The project area was reclaimed and diked for grazing and agriculture in the 1870s by removing sediment from borrow ditches in the interior edges to build and maintain levees (Thompson, 1877). The diked sites were flooded with bay waters in the 1950s to form evaporative salt production ponds. Restoration planning was initiated after the purchase of 3828 ha by the California Department of Fish and Game in 1994 (Jones and Stokes, 2004). Pond 2A was breached in 1995 to avoid flood damage, but subsequent plans were developed to breach Ponds 3, 4, and 5 along the Napa River (508, 367, and300 ha, respectively) as part of the effort to restore tidal salt marsh habitat (Fig. 1).

Prior to planned restoration activities in Ponds 3, 4, and 5, however, unknown parties created a “midnight” breach in Pond 3 (Breach 3C; Fig. 1). The midnight breach increased in size from a notch 0.5 m wide in August, 2002 to 24 m wide in January, 2004 and resulted in an increased tidal prism and muted tidal exchange into Pond 3 over that period (Takekawa et al., 2004). The restoration construction in Ponds 3, 4, and 5 was completed in the fall of 2006, and full tidal action was restored with engineered breaches to Pond 3 (7 breaches), Pond 4 (8 breaches), Pond 5 (4 breaches; Fig. 1), and internal breaches on the levee dividing Ponds 4 and 5. Most breaches (3B, 3D–F, 4A–E, and 5C) were excavated to −1.22 m NAVD88, 4F was excavated to −0.76 m, and 5B to −1.52 m (PWA, 2005). Restoration activities on the ponds included the construction of large drainage channels excavated in the footprint of historic channels and adjacent island chains from the contoured dredge spoils. The channels connect to breaches 3G, 4C, 5B, and 5C (Fig. 1; PWA, 2005). Ditch blocks built in Ponds 3, 4, and 5 perpendicular to the borrow ditches were intended to slow the transport of water through the ditches and encourage flow through the natural channels (Fig. 1; Jones and Stokes, 2004; PWA, 2005).

2.2. Elevations

To estimate pond and breach elevations, we conducted a bathymetric survey of Pond 3 from 14 December 2004 to 4 February 2005 (hereafter the 2005 survey). We conducted 31 north–south and 23 east–west transects at 125 m intervals across the interior of the pond that totaled over 79 km (Fig. 2). We also conducted bathymetric surveys of Ponds 3 and 4 from November 2008 to February 2009 and Pond 5 during December 2009 (hereafter referred to as the 2009 survey). These surveys were comprised of 19 north–south and 15 east–west transects at 200 m intervals totaling 53 km in Pond 3; 9 and 18 transects over 46 km in Pond 4; and 10 and 11 transects over 28 km in Pond 5 (Fig. 2). We also surveyed the single breach of Pond 3 in 2005 and the 18 additional breaches in Ponds 3, 4, and 5 in 2009 with 4–12 perpendicular and parallel transects depending on breach dimensions.

Our bathymetric system was comprised of two independent datasets: (1) water depth and (2) water surface elevation (Athearn et al., 2010; Takekawa et al., 2010a). To obtain these data, we used a shallow-water echo-sounding system comprised of an acoustic profiler (Reson, Inc.; Slangerup, Denmark, Navisound 210; 1 cm reported accuracy), global positioning system (GPS) rover unit, and laptop computer mounted on a shallow-draft, portable flat-bottom boat (Bass Hunter, Cabelas, Sidney, NE) equipped with an electric trolling motor. We operated a variable frequency single-beam sonar transducer at a frequency of 200 kHz attached to the front of the boat in >30 cm of water. We calibrated the system prior to our surveys with a bar-check plate suspended below the transducer at a known depth and adjusted the sound velocity for salinity and temperature differences. We tested the system prior to each day of data collection with the bar-check to ensure accurate soundings.

To obtain x and y coordinates and water surface elevation for the 2005 survey we used a differential global positioning system rover unit (DGPS; Trimble, Ag132) and readings from staff gages in six sections of Pond 3; these were surveyed to project benchmarks with a level and rod. An observer recorded staff gage readings at 10-min intervals to determine water height inside the pond that varied with tide stage. We used linear regression equations to estimate water height between staff gage readings based on the time of depth measurement in SAS 9.1 (SAS Institute, Cary, NC). For the 2009 survey, we updated the system with Leica RX1200 Real Time Kinematic (RTK) Global Positioning System (GPS) rover unit capable of collecting survey-grade elevation and x and y position data from the Leica Smartnet system (±3 cm x, y, and z accuracy; Leica Geosystems Inc., Norcross, GA). The unit averaged ±2.5 cm vertical error at our reference benchmark (X 552 1956 Mare Island), which is within the stated error of the survey unit. Compared with the 2005 survey, the 2009 survey methodology reduced measurement variability from draft, tilt, and waves affecting the boat without adding bias to the average differences.

To process the data, we averaged 20 depth values generated each second by the echosounder with SAS 9.1. We then integrated the water depth and water surface elevation datasets to obtain the final sediment surface elevations by subtraction (sediment

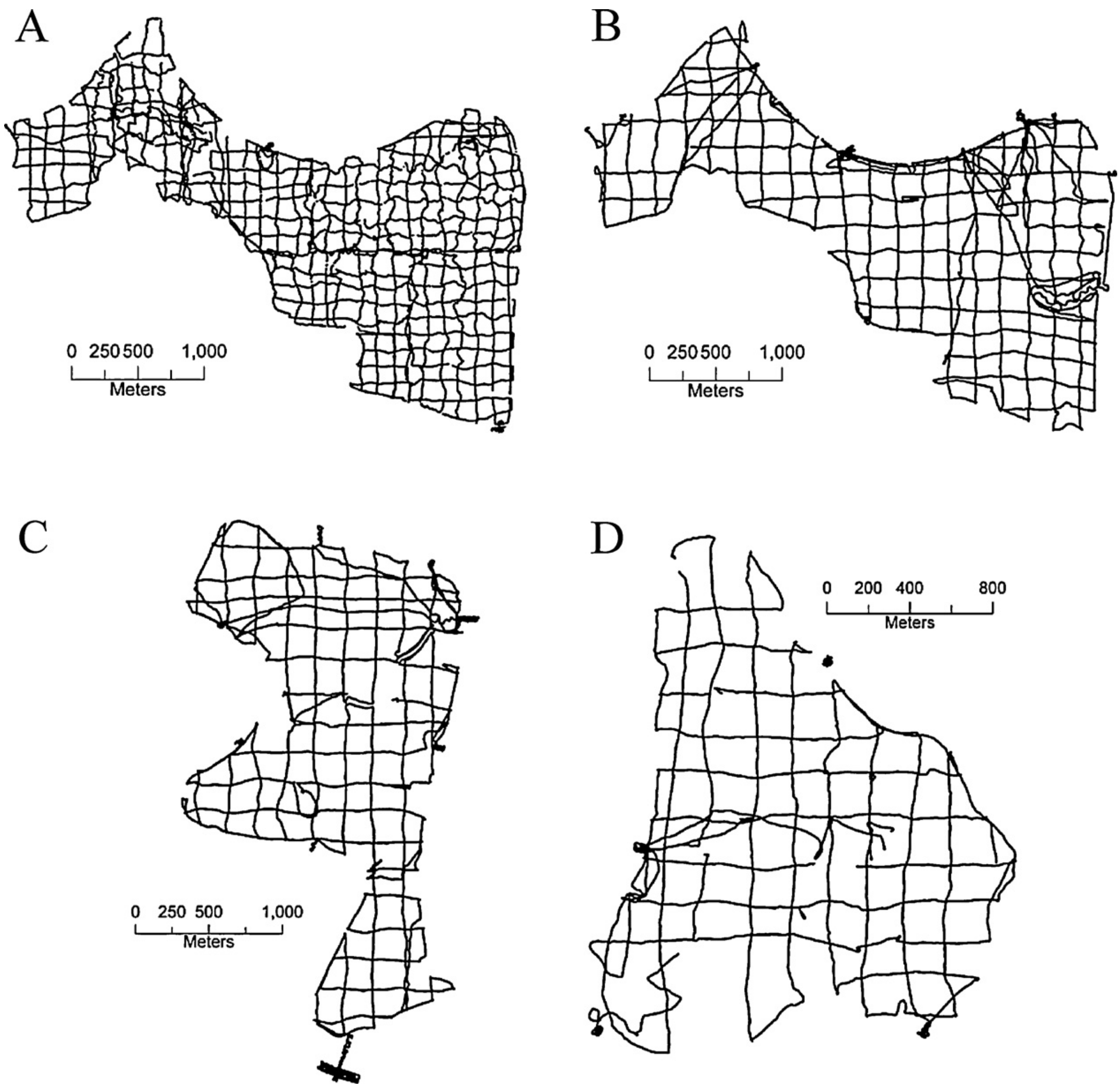

Fig. 2. Transect locations of bathymetric surveys for (A) Pond 3 in 2005; (B) Pond 3 in 2009; (C) Pond 4 in 2009; and (D) Pond 5 in 2009.

surface = water surface elevation − water depth). Because our goal was to assess average elevations resulting from natural sedimentation rather than human activities, we excluded points within constructed ditch blocks, islands, breaches, and channels, based on the construction diagrams provided by Ducks Unlimited (PWA, 2005). This process yielded 115,000 data points for the 2005 Pond 3 survey, and 58,000 (Pond 3), 41,000 (Pond 4), and 28,000 data points (Pond 5) for the 2009 surveys.

We used Spatial Analyst in ArcGIS 9.3 (ESRI, Redlands, CA) to create a digital elevation model with 25 m × 25 m GIS gridcells. We used the Inverse-Distance Weighting (IDW) method to interpolate the elevation point data. IDW allows use of a "barrier" polyline file that forces the interpolation to exclude selected points from gridcell elevation calculations to avoid distortion from nearby features such as deep channels or borrow ditches. For example, if a large channel had a barrier polyline around it, data points from within the channel were not used to calculate the elevation for gridcells in the adjacent pond floor. We created our barrier polylines by mapping known pond features from aerial imagery. We processed and interpolated the breach point data in a similar way, except that the greater point density allowed us to use a 1 m gridcell size in the digital elevation model. We validated our bathymetric and data processing methods by comparing paired elevation estimates at the intersection of our east–west and north–south transects. We found that the average difference between points was <2 cm across all ponds and survey years (Table 1), an accuracy comparable to another study that applied similar methods (e.g. Takekawa et al., 2010a). Unless noted otherwise, all data were collected and reported in meters with horizontal datum UTM NAD83 and vertical datum NAVD88.

To estimate the average and spatial distribution of elevations within ponds, we used Spatial Analyst and geospatial tools in ArcGIS 9.3 (ESRI, Redlands, CA). We first mapped the elevations for each pond and survey by creating the digital elevation models. We then estimated the average elevation and its standard deviation across each pond and breach by survey with the Zonal Statistics tool. For Pond 3, we calculated the total volume of sediment change between the 2005 and 2009 surveys with the Cut/Fill tool and used

Table 1
Comparison of mean elevation differences at intersections of north–south and east–west transects by survey year at Ponds 3, 4, and 5 in the Napa-Sonoma Marshes Wildlife Area as validation of bathymetric and data processing methods.

Variable	Pond 3 2005	Pond 3 2009	Pond 4 2009	Pond 5 2009
Number of paired elevation points	252	150	104	69
Mean distance between points (cm)	41.6	64.3	65.7	64.1
SD (distance between points)	25.3	32.5	34.2	33.3
Mean elevation difference (cm)	0.22	−1.82	−1.92	−0.45
SE (elevation difference)	0.62	0.93	1.33	1.14
SD (elevation difference)	9.88	11.40	13.53	9.43
95% LCL (elevation difference)	−1.01	−3.66	−4.55	−2.72
95% UCL (elevation difference)	1.44	0.02	0.71	1.81

the Raster Calculator tool to estimate the average change in elevation across surveys. We divided the total sediment change by the number of years between surveys to provide an indication of the annual rate of accretion or erosion across the pond. To investigate spatially variable differences, we compared elevation changes over time in five subsections of Pond 3 that corresponded with the highest and lowest elevations in 2009 (Fig. 3B).

2.3. Hydrologic connectivity

To assess the hydrologic context of each pond, we calculated the breach conveyance ability (BCA) as a relative measure of tidal muting. We estimated BCA by summing the cross-sectional area per breach by pond divided by the pond tidal prism: $BCA = \frac{\sum(b_i * h_i)}{P}$ where b is the breach width, h is the breach tidal depth, and P is the unmuted tidal prism of the pond. We calculated h as the difference in elevation between MHHW and MLLW or the breach invert, whichever was greater. We calculated P as the difference between the elevation of MHHW in the adjacent Napa River (the same elevation as in the pond, if the tide is unrestricted) and the pond bottom elevation, multiplied by the pond surface area at MTL. This method compared the maximum possible breach area to the maximum possible volume exchange through the breaches to characterize the ability of the breaches to carry the tidal prism flow in that pond. Larger BCA values suggested better hydrologic connectivity between a pond and the surrounding waters. We calculated the non-dimensional ratio of the BCA in Ponds 4 and 5 versus Pond 3 to estimate the degree of tidal muting in those ponds relative to Pond 3.

2.4. Vegetation

Elevation at which *S. foliosa* colonizes varies by tidal prism and inundation times specific to a marsh location (Zedler et al., 1999); thus, we sampled *S. foliosa* elevation by pond in the fall of 2010. To minimize measurement variability, we sampled elevation at a given point by taking the average of 2–5 repeat elevation measurements using a RTK Leica Smartpole 1200 GPS unit. *S. foliosa* has been shown to expand to a lower elevation after initial colonization (Williams and Orr, 2002; Wallace et al., 2005). Thus, we measured elevations at different patch sizes within each pond, which we considered to serve as a surrogate to patch age based on the colonial growth pattern of *S. foliosa*. We assumed small plants comprised of ≤25 individual stems to be that year's new colonization and large plants to have been from prior years. We sampled point elevations at the center of 21 large (28.4 ± 14.1 m^2; mean ± SE) and 23 small (0.7 ± 0.2 m^2) patches in Pond 3, 9 large (14.5 ± 4.5 m^2) and 8 small (0.8 ± 0.2 m^2) patches in Pond 4, and 12 large (9.9 ± 2.0 m^2) and 12 small (0.4 ± 0.1 m^2) patches in Pond 5. We used 2-sample t-tests to test whether elevations differed between large and small

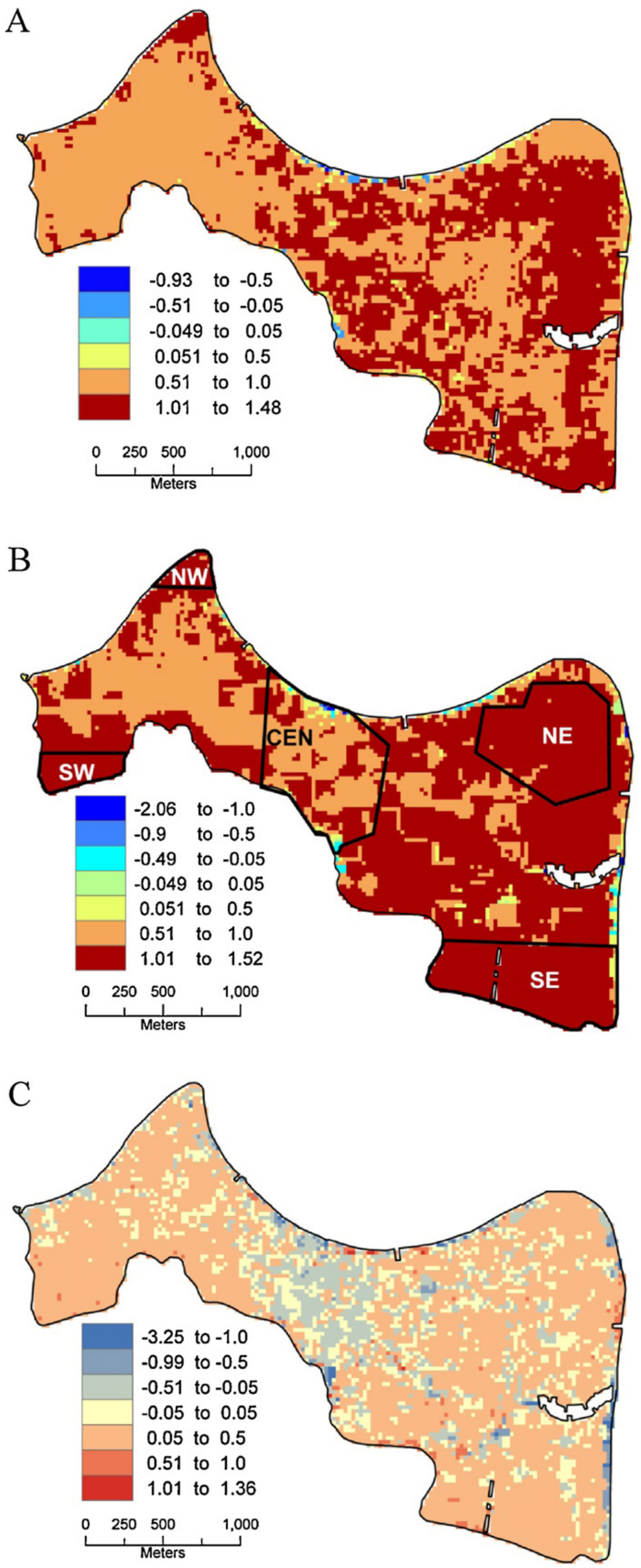

Fig. 3. Elevations for Pond 3 in the Napa Sonoma Marshes Wildlife Area in 25 m × 25 m gridcells (NAVD88 meters) shown for the (A) 2005 survey; (B) 2009 survey with northwest (NW), northeast (NE), southwest (SW), southeast (SE), and central (CEN) sections; and (C) change in elevation between surveys.

Table 2
Tidal datum elevations near the most central breach along the Napa River for Pond 3 (breach 3G), Pond 4 (breach 4D), and Pond 5 (breach 5B) in meters NAVD88.

Tidal datum	Pond 3	Pond 4	Pond 5
MHHW	1.93	1.95	1.95
MHW	1.76	1.77	1.77
MSL	1.07	1.07	1.07
MTL	1.07	1.07	1.04
MLW	0.38	0.37	0.30
MLLW	0.11	0.09	0.11

patches by pond (Stata 11.0; StataCorp, 2009; College Station, TX). Because average elevation did not vary by patch size in Pond 3 (0.03 ± 0.03, mean difference ± SE; P = 0.40), Pond 4 (0.03 ± 0.05; P = 0.63), or Pond 5 (0.01 ± 0.03; P = 0.86), we used all patches to estimate the distribution of colonizing *S. foliosa* elevations by pond using box-and-wisker plots. We tested whether there was a difference among ponds using one-way ANOVA followed by Bonferroni multiple comparisons procedure. We used the 10th percentile to represent the minimum and 50th percentile to represent the average elevations of colonizing *S. foliosa* by pond. We then calculated the proportion of each pond ≥10th and 50th percentiles of *S. foliosa* elevations. While it was not possible to estimate tidal datums for the vegetation elevations directly within the ponds due to lack of data coverage, we estimated tidal datum values in meters NAVD88 (horizontal datum NAD83) near the center breach along the Napa River adjacent to Pond 3 (38.132°N, 122.284°W), Pond 4 (38.161°N, 122.297°W), and Pond 5 (38.179°N, 122.320°W) using V Datum v2.3.3 (NOAA, 2010) to provide context for our reported elevations (Table 2).

3. Results

3.1. Sedimentation

Mean elevation across Pond 3 increased from 2005 to 2009, excluding the breaches and construction areas (Table 3, Fig. 3). During this time, we observed a net gain of 486,600 ± 28,300 m^3 (mean ± SD) in total volume of sediment. This sediment gain corresponds to a 9.5 ± 0.2 cm (mean ± SD) depositional layer of sediment across the pond area or an average accretion rate of 2.4 cm per year between surveys.

While the mean pond elevation increased, there was substantial spatial variation in sediment elevations (Fig. 3). Pond 3 ranged from −0.93 to 1.48 m NAVD88 in 2005, with lowest elevations in borrow ditches along the northern levee and adjacent to the midnight breach, and highest elevations distributed across the pond (Fig. 3). In 2009, we observed an increase in topographic heterogeneity in Pond 3 based on a greater range in values compared with 2005. In the later survey, elevation ranged from −2.06 to 1.52 m NAVD88, with lowest elevations again in borrow ditches along the northern levee and in the central section of the pond, and highest elevations occurring in the four corners of the pond.

Between surveys, the north and south corners of the pond had substantial accretion compared with the mean elevation whereas one area eroded. Mean elevation between the 2005 and 2009 surveys increased an average of 11 cm (1.1 ± 0.13 to 1.21 ± 0.12 meters NAVD88; mean ± SD) in the northeast, 14 cm (1.06 ± 0.13 to 1.20 ± 0.2 m NAVD88) in the northwest, 20 cm (1.04 ± 0.12 to 1.24 ± 0.15 meters NAVD88) in the southeast, and 33 cm (0.90 ± 0.08 to 1.23 ± 0.11 m NAVD88) in the southwest sections of Pond 3, respectively (Fig. 3B). In contrast, sediment eroded an average of 9 cm from the center area adjacent to the midnight breach (0.96 ± 0.26 to 0.87 ± 0.34 m NAVD88; mean ± SD; Fig. 3B). The area within the midnight breach lost an average of 6 cm of sediment between the 2005 and 2009 surveys (Fig. 3C; Table 4).

Table 3
Elevations by pond and survey year with minimum (10th percentile) and median (50th percentile) elevations of *S. foliosa* colonization by pond, and proportion of the ponds above 10th and 50th percentile elevations assumed to support *S. foliosa* growth. Ponds bottoms were subsided and unvegetated prior to breaching.

Variable	Pond 3 2005	Pond 3 2009	Pond 4 2009	Pond 5 2009
Overall pond elevation (mean)[a]	0.96	1.05	0.81	0.84
Overall pond elevation (sd)[a]	0.19	0.25	0.26	0.24
10th percentile of *S. foliosa* elevation[a]	NA	1.09	0.89	0.86
50th percentile of *S. foliosa* elevation[a]	NA	1.24	1.02	0.96
Percent of pond above 10th percentile	NA	47%	38%	55%
Percentof pond above 50th percentile	NA	16%	13%	24%

[a] Elevation in meters NAVD88.

Table 4
Dimensions of breaches in restored Ponds 3, 4, and 5 in the Napa Sonoma Marshes Wildlife Area. Breach locations are shown in Fig. 1. Width and mean elevation in meters (NAVD88) with the standard deviation (SD) and number of sample points (N).

Breach	Width (m)	Inflow source	Elevation	
			Mean ± SD	*N*
3A	29	South Slough	0.0 ± 1.0	480
3B	22	South Slough	0.1 ± 0.9	940
3C (2005)	37	South Slough	−2.0 ± 1.8	1129
3C (2009)	51	South Slough	−2.6 ± 1.6	1074
3D	27	South Slough	−0.1 ± 0.5	538
3E	37	South Slough	−0.8 ± 1.3	3368
3F	30	Napa River	−0.9 ± 1.2	3074
3G	40	Napa River	−1.7 ± 1.0	2508
4A	19	Napa River	−0.8 ± 0.9	4327
4B	19	Napa River	−0.6 ± 0.7	840
4C	33	Napa River	−0.9 ± 1.0	4901
4D	32	Napa River	−1.7 ± 1.1	1948
4E	28	South Slough	−0.8 ± 0.8	6346
4F	21	China Slough	−0.5 ± 1.0	1796
4G	31	China Slough	0.4 ± 0.7	1131
4H	27	China Slough	0.8 ± 1.1	1441
5A	28	Devil's Slough	−0.3 ± 0.8	1110
5B	30	Napa Slough	−1.0 ± 1.2	1173
5C	20	China Slough	−0.6 ± 0.9	1240
5D	28	China Slough	−0.1 ± 0.6	1313

3.2. Elevations and hydrologic connectivity

The mean elevation of Pond 3 was greater than elevations in Ponds 4 and 5 in the 2009 survey (Table 3; Fig. 4). Elevations ranged from −2.06 to 1.52 in Pond 3, −1.41 to 1.64 m in Pond 4, and −1.52 m to 1.64 m in Pond 5. The lowest elevations of all three ponds occurred in the vicinity of the breaches, constructed channels, and borrow ditches adjacent to levees (Fig. 4). The highest elevation for Pond 4 occurred across the middle of the northern half and at the southern end. The highest elevations for Pond 5 were found throughout the pond and particularly in the northern and southern sections (Fig. 4). We found that all 19 breaches surveyed in 2009 had an elevation lower than the pond elevation. Most breaches had an elevation lower than breach construction elevations, indicating scour had occurred since the restoration of tidal action. Breaches adjacent to sloughs generally were at a higher elevation than those adjacent to the river.

The breach cross-sectional area, pond tidal prism, and breach conveyance ability were greatest in Pond 3, intermediate in Pond 4, and least in Pond 5 (Table 5). Pond 4 had about 90% of the Pond 3 conveyance, whereas Pond 5 had 61% of the Pond 3 conveyance (Table 5), indicating that tidal range was more muted in Ponds 4 and 5 relative to Pond 3.

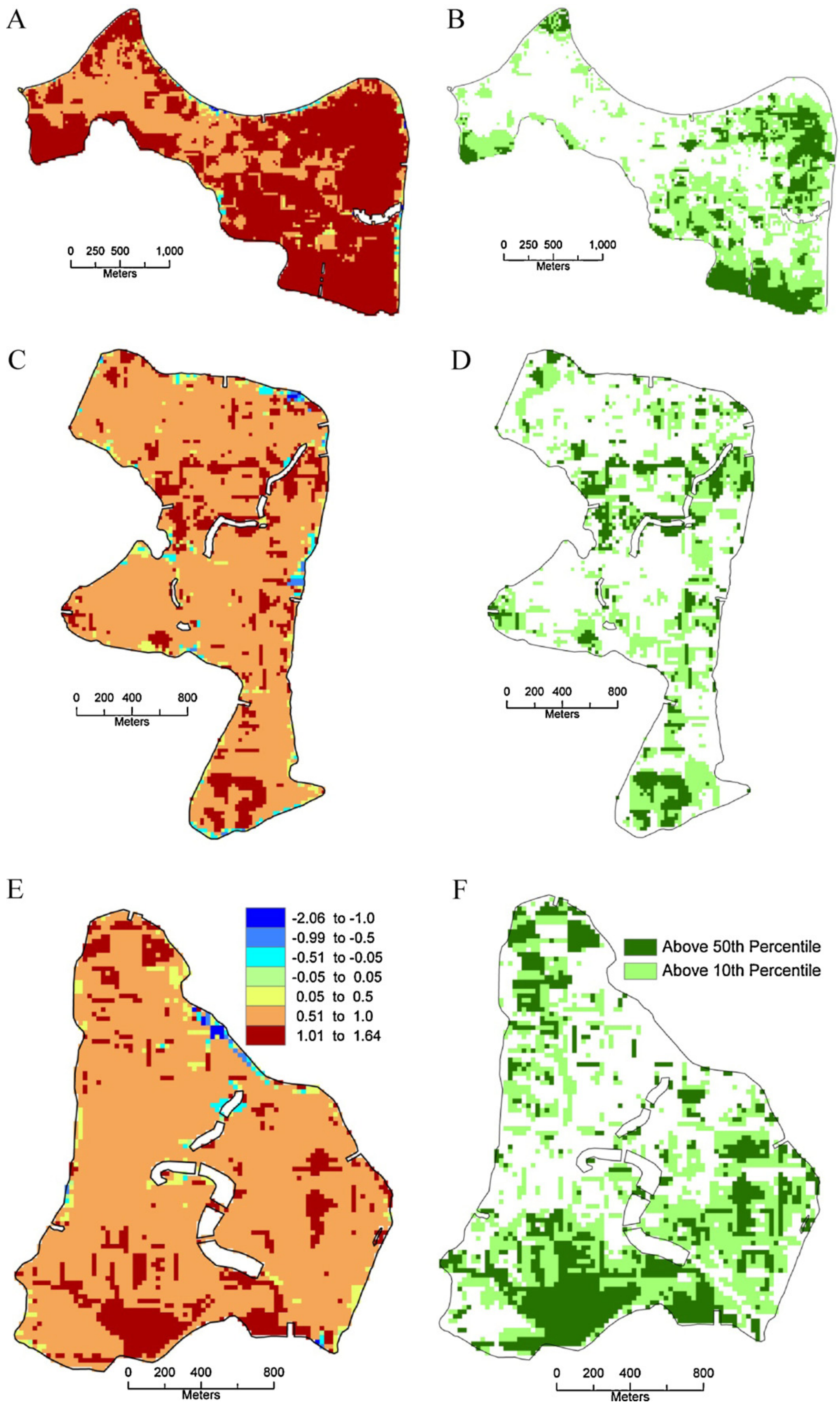

Fig. 4. Elevations in 25 m × 25 m gridcells (NAVD88 meters) during the 2009 survey for (A) Pond 3 substrate elevation; (B) Pond 3 elevations above the 10th and 50th percentiles of measured colonization elevations; (C) Pond 4 substrate elevation; (D) Pond 4 elevations above the 10th and 50th percentiles of measured colonization elevations; (E) Pond 5 substrate elevation; and (F) Pond 5 elevations above the 10th and 50th percentiles of measured colonization elevations.

Table 5
Estimated breach cross-sectional area, tidal prism, and breach conveyance ability by pond. The breach conveyance ratio is a relative measure of the ability of Ponds 4 and 5 to deliver the full tidal range relative to Pond 3.

Pond	Breach cross-sectional area (m^2)	Tidal prism (m^3)	Breach conveyance ability (m^{-1})	Breach conveyance ratio
3	430	4,500,000	9.6×10^{-5}	–
4	362	4,200,000	8.6×10^{-5}	0.90
5	195	3,300,000	5.9×10^{-5}	0.61

3.3. *Vegetation*

Specific regions within each pond had substantially higher elevations than the mean and supported colonizing *S. foliosa* (Table 3; Fig. 5). Median elevations of *S. foliosa* in 2009 corresponded with 0.17 m above MTL in Pond 3, 0.05 m below MTL in Pond 4, and 0.08 m below MTL in Pond 5 (Tables 2 and 3). The average elevation of *S. foliosa* varied significantly by pond ($F = 71.20$, $df = 84$, $P < 0.0001$). Average *S. foliosa* elevation was significantly higher in Pond 3 compared to Ponds 4 and 5 ($P < 0.0001$ for both comparisons), but did not differ significantly between Ponds 4 and 5 ($P = 0.206$; Fig. 5). Over 38% of each pond was ≥ 10th percentile elevation for *S. foliosa*, whereas the percent of the ponds ≥ 50th percentile elevation for *S. foliosa* was 16% in Pond 3, 13% in Pond 4, and 24% in Pond 5 (Table 3; Fig. 4).

4. Discussion

Accretion of sediment to marsh plain elevation is fundamental to the success of marsh restoration efforts in subsided, former diked areas. We estimated an average accretion rate of 9.5 cm over 4 years and 2.4 cm per year across Pond 3 of the Napa Sonoma Marshes Wildlife Area. Prior to breaching, mean suspended sediment concentration (SSC) was 146 mg/L based on measurements taken every 15 min from September 1997 to March 1998 in South Slough adjacent to the midnight breach (3C) of Pond 3 (Warner et al., 1999). Assuming an increase in water volume (mean low tide to mean high tide) post-breach of approximately 1.36 Mm^3, two inundations per day, and a bulk density of 850 kg/m^3 for deposited sediment (Porterfield, 1980; Takekawa et al., 2004), the resulting mean annual sedimentation would be 3.1 cm per year if all sediment transported through the breaches deposited on the restoration area. This is slightly greater than what was observed, perhaps since this calculation does not include erosion, the flux

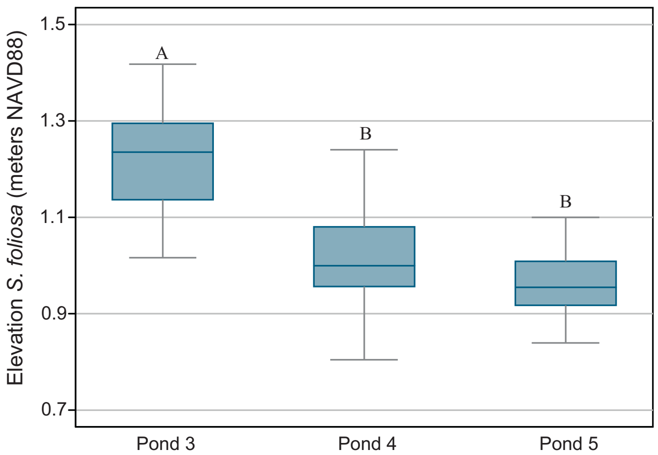

Fig. 5. Box-and-whisker plot of *S. foliosa* elevation by pond. Middle line is the median, upper and lower box limits are the 25th and 75th percentile, and whiskers show the minimum and maximum. Different letters show significant differences in elevation between ponds based on a Bonferroni multiple comparison test.

of suspended sediment does not settle on the restoration area in a single tidal cycle, or the impact of wind-wave re-suspension would decrease sediment deposition (Williams and Orr, 2002). Our estimated average annual sedimentation rate of 2.4 cm per year during early restoration exceeds the recent rate of sea level rise in San Francisco Bay (0.22 cm per year; Flick, 2003) and the upper bound of predicted sea level rise for the 21st century (1.39 cm per year; Cayan et al., 2009). Thus, inorganic sedimentation presently outpaces sea level rise at this location. However, inundation and thus inorganic accretion will likely decrease as the restoration area fills with sediment, and the effect of sea level rise remains a concern in this as in other restoring marshes (Weinstein and Weishar, 2002; Watson, 2004).

Previous studies have questioned whether the evolution of large sites would be feasible within a reasonable timeframe (Williams and Orr, 2002; PWA, 2002). Vertical accretion rates in subsided and formerly diked sites can vary substantially as a function of local sediment supply and overall surface area. Sedimentation at our large sites benefited from barotropic convergence of two sediment sources from the northern San Francisco Bay and Napa River (Warner et al., 2003). The SSC adjacent to Pond 3 (146 mg/L) was about three times higher than the mean SSC continuously measured in San Francisco Bay from 1999 to 2007 (46 mg/L; Schoellhamer, 2011). Nevertheless, the sedimentation rate we observed was generally less than smaller, restored sites adjacent to large sediment loads. At two other restoration sites in the northern estuary, average annual accretion rates ranged from 6.2 cm per year at Guadalcanal to 16.8 cm per year at Tubbs Setback over 8 years (Woo et al., 2008; Takekawa et al., 2010a, 2010b). Pond A21 in the south San Francisco Bay averaged an annual accretion rate of 4.4 cm per year over 4 years (Callaway et al., 2009). These sites receive large sediment supply from adjacent north (Jaffe et al., 1998) or south San Francisco Bay (Brew and Williams, 2010) yet were substantially smaller in surface area than ponds in the Napa-Sonoma Marshes Wildlife Area and thus less likely for sediment to be eroded by wind-wave resuspension (Williams and Faber, 2001; Williams and Orr, 2002). At another relatively large site (200 ha) in the Schelde estuary, Maris et al. (2007) developed a model that estimated an average accretion rate of 2.8 cm per year based on >120 mg/L SSC, similar to our findings. These results suggest that while restoration may proceed more slowly compared to smaller sites, passive sedimentation at large sites is feasible within a reasonable timeframe in areas with adequate suspended sediment supply. Generally, tidal restoration sites with lower initial elevations accrete the most sediment in the first years following breach (French, 1993; Williams and Orr, 2002; Callaway et al., 2009); thus, accretion in Pond 3 will likely continue, but accretion rates may decrease over time. However, large storms tend to increase short-term sedimentation rates (Cahoon et al., 1996; Ward et al., 2003) well above average rates and sedimentation rates could increase with a major flood.

In addition to mean elevation changes across sites, the spatial variation in elevation provides important insights into restoration progress. In Pond 3, we observed increased spatial heterogeneity in elevations between the 2005 and 2009 surveys that reflect differing patterns of deposition and erosion. Generally, sediment deposition occurs on the slack flood tide when water moves slowly, and the greatest accretion rates occur in areas with the lowest elevations that experience longer tidal inundation periods for sediment delivery (Chmura et al., 2001; Williams and Orr, 2002; Callaway et al., 2009). However, areas that have the highest elevation may accrete more rapidly after initial vegetation colonization, since accretion provides a positive feedback to marsh surface elevation and vegetation can act as a filter to trap sediment and prevent erosion (Josselyn, 1983; Cahoon et al., 1996; Ward et al., 2003; Wallace et al., 2005). The areas of greatest accretion in Pond 3 likely reflect both of these processes, with certain low elevation areas likely accreting due to greater tidal exposure at slack tide, and other higher elevation areas accreting perhaps due to vegetation colonization (observed as early as 2008, USGS, unpublished data) or more sheltered conditions.

Restoration to mature marsh also requires development of tidal creeks (Zedler et al., 1999; Sanderson et al., 2000; Wallace et al., 2005) and sedimentation patterns are likely determined by the developing drainage network (de Groot et al., 2011). While the spatial resolution of our analysis was not intended to map the drainage network, our elevation change data indicated that areas of erosion occurred in borrow ditches adjacent to certain breaches. This suggests that water flow through borrow ditches was substantial, in spite of constructed ditch-blocks that were designed to prevent this flow and instead to encourage the re-development of historic channels. Similarly, in a diked former salt marsh that was breached in the Netherlands, tidal water flowed through and scoured ditches that were still present after 10 years of tidal exchange (Verbeek and Storm, 2001). Restoring marshes in the Bay of Fundy, Canada also developed hybrid drainage networks that incorporated both original creeks and reactivated drainage ditches (MacDonald et al., 2010). Further work is needed to investigate and refine restoration design elements that are intended to prevent borrow ditches from forming primary channels (Brand et al., 2010). In addition to borrow ditches, the central area of the pond adjacent to the midnight breach in our study (breach 3C) eroded substantially between surveys. This breach had scoured significantly given that breach width increased from 0.5 m in 2002 to 24 m in 2004 (Takekawa et al., 2004), and during this study from 37 m in 2005 to 51 m in 2009. Loss of sediment in the central area of the pond was not an intended consequence of restoration. However, the transitional mudflat habitats such as those formed by this erosion were heavily used by foraging shorebirds (Brand et al., 2010) but are expected to decline as mudflats transition to marsh (Ward et al., 2003; Brew and Williams, 2010). Further work is needed to investigate potential design elements that could yield a mix of habitats with staged, long-term, or permanent mudflats within restored marsh (Williams and Orr, 2002; Brew and Williams, 2010).

The range of elevations needed for a site to transition from mudflat to low marsh vegetation has been documented in numerous Pacific coast marshes (Patrick and DeLaune, 1990; Zedler et al., 1999; Ward et al., 2003; Watson, 2004). We found that elevations of *S. foliosa* varied among restored marshes as has been found in other studies (Zedler et al., 1999; Silvestri et al., 2005; Beyers and Chmura, 2007). Median elevations of *S. foliosa* in 2009 corresponded with 0.17 m above MTL in Pond 3 but were lower (0.05–0.08 m below MTL) in Ponds 4 and 5, relative to estimated tidal datum values along the adjacent Napa River. These ranges were lower than assumed for Ponds 4 and 5 in the restoration design (PWA, 2002) but were within the lower elevation limit found for *S. foliosa* (0.0–0.3 m below MTL; Atwater and Hedel, 1976). Further work is needed to evaluate the tidal datum values within the ponds directly.

The elevation range sufficient for vegetation colonization depends in part upon the local tidal regime that determines inundation (Zedler et al., 1999; Williams and Orr, 2002; Silvestri et al., 2005; Pennings et al., 2005). In the case of restored sites, tidal regime varies as a function of the number, size and locations of breaches. Based on the dimensions of breaches in our restoration site relative to the pond tidal prism, we calculated a ratio of tidal conveyance indicating that Ponds 4 and 5 were more muted than Pond 3. While restricted tidal regime may decrease drainage that can delay vegetation establishment lower in the tide range (Williams and Orr, 2002), at higher portions of the tidal range a muted tide reduces the hydroperiod, and in turn, may reduce colonization elevation of low marsh vegetation due to reduced inundation stress (Bakker et al., 2002; Crooks et al., 2002; Beyers and Chmura, 2007). This supports the idea that manipulation of the tidal regime may be used to encourage early vegetation development at restored sites (Maris et al., 2007; Cox et al., 2006). In addition to the tidal regime, soil salinity, soil aeration, nitrogen, competition, and the location of tidal creeks may be important (Mahall and Park, 1975; Zedler et al., 1999; Sanderson et al., 2000; Pennings et al., 2005; Silvestri et al., 2005). Ponds 4 and 5 are farther up the Napa River and thus may have greater brackish water inputs that could lower elevations required for colonization (Mahall and Park, 1975; Williams and Orr, 2002). There may also be spatial variation in drainage within each pond due to the location of channels or varied topography. Regardless of the specific mechanisms, however, the practical implication is that despite the lower absolute elevations of Ponds 4 and 5 relative to Pond 3, we did observe elevations sufficient for colonization by *S. foliosa* for restoration to tidal marsh in all three ponds.

As large restoration projects are implemented, it is important to develop a learning curve that builds on prior experience (Teal and Weinstein, 2002). Before the restoration was implemented, Pond 3 was projected to support 60% vegetation coverage in 20 years and 90% vegetation cover in 50 years, while Ponds 4 and 5 were expected to remain predominantly mudflat for 50 years (PWA, 2002). These projections were based on lower assumed sediment supply across the site, particularly in Ponds 4 and 5, than found by Warner et al. (2003). Our results are relatively close to that expected for Pond 3, though repeat surveys for Ponds 4 and 5 will be needed to validate design assumptions and to assess sedimentation rate in those ponds. Elevation is a key predictor of both sediment accretion and vegetation colonization (Zedler et al., 1999; Williams and Orr, 2002; Callaway et al., 2009), and our finding of a lower colonization elevation than that assumed for Ponds 4 and 5 in the restoration design (PWA, 2002) indicates that the restoration may proceed more rapidly than originally expected. Our estimates of the area available for colonization by *S. foliosa* across the 3 ponds are quite promising, though Ward et al. (2003) found that minimum observed elevations were not sufficient to maintain *S. foliosa* distribution across the Tijuana Estuary in southern California. We suggest that the median elevation likely indicates a better approximation of elevations suitable for further expansion given average conditions, but that the full expansion of *S. foliosa* across restored sites may also be limited by stochastic events such as storm-driven salinity reduction (Ward et al., 2003).

5. Conclusions

The results of our study provide insights into the status of early tidal marsh restoration and can inform future restoration efforts. We found that Pond 3 in the Napa Sonoma Marshes Wildlife Area is on a trajectory toward developing the physical conditions required to establish the plant community on the marsh plain. The emphasis in this restoration has been to enable natural site evolution post-breach, and this approach appears to be successful in this large restoration site. There is more uncertainty surrounding sedimentation of Ponds 4 and 5 due to the lack of repeat elevation surveys, however our results suggest that tidal marsh restoration projects in large (≥300 ha), subsided, formerly diked sites will be feasible given sufficient local sediment loads, despite prior concerns about increased sediment resuspension in large sites due to wind-waves. In addition to areas that accreted sediment, we also observed sections of Pond 3 that eroded. In particular, borrow ditches took the place of primary channels in some locations. Further effort is needed to refine design elements, such as ditch blocks, if the goal is to encourage development of historic and new site-interior channels. We documented a large area of erosion within the middle of Pond 3; whereas this erosion was not expected, this could provide benefits to foraging shorebirds if mudflats become a long-term transitional feature. We have documented the colonization elevations of *S. foliosa* within each pond. Despite lower substrate elevations in Ponds 4 and 5 relative to Pond 3, vegetation colonization in those ponds was perhaps enhanced by tidal muting relative to Pond 3 that may reduce the hydroperiod in higher portions of the tidal range.

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References

Athearn, N.D., Takekawa, J.Y., Jaffe, B., Hattenbach, B.J., Foxgrover, A.C., 2010. Mapping bathymetry of tidal wetland restoration sites in San FranciscoBay: comparing accuracy of aerial LiDAR with a single beam echosounder. J. Coast. Res. 26, 312–319.

Atwater, B.F., Hedel, C.W., 1976. Distribution of seed plants with respect to tide levels and water salinity in the natural tidal marshes of the northern San Francisco Bay estuary, California. Preliminary Report. U.S. Department of the Interior, Geological Survey. Open File Report 76-389, 106 pp.

Bakker, J.P., Esselink, P., Dijkema, K.S., van Duin, W.E., de Jong, D.J., 2002. Restoration of salt marshes in the Netherlands. Hydrobiologia 478, 29–51.

Beyers, S.E., Chmura, G.L., 2007. Salt marsh vegetation recovery on the Bay of Fundy. Estuar. Coast. 30, 869–877.

Brand, L.A., Smith, L.M., Woo, I., Shinn, J., Smith, A., Chan, W., Taylor, K., Gustafson, B., Takekawa, J.Y., 2010. Assessing early tidal marsh restoration at the Napa Salt Marsh and Napa Plant Site restoration projects within the Napa-Sonoma Marshes Wildlife Area. Data Summary Report, U.S. Geological Survey, Western Ecological Research Center, Vallejo, CA, 170 pp.

Brew, D.S., Williams, P.W., 2010. Predicting the impact of large-scale tidal wetland restoration on morphodynamics and habitat evolution in south San Francisco Bay, California. J. Coast. Res. 26, 912–924.

Cahoon, D.R., Lynch, J.C., Powell, A.N., 1996. Marsh vertical accretion in a Southern California Estuary, U. S. A. Estuar. Coast. Shelf Sci. 43, 19–32.

Callaway, J.C., Parker, V.T., Schile, L.M., Herbert, E.R., Borgnis, E.L., 2009. Dynamics of sediment accumulation in Pond A21 at the island ponds. Report to the California State Coastal Conservancy, 66 pp.

Cayan, D.R., Tyree, M., Dettinger, M.D., Hidalgo, H., Das, T., 2009. California climate change scenarios and sea level rise estimates for the California 2008 Climate Change Scenarios Assessment. California Energy Commission Report No. CEC-500-2009-014-F, 62 pp.

Chmura, G.L., Coffey, A., Crago, R., 2001. Variation in surface sediment deposition on salt marshes in the Bay of Fundy. J. Coast. Res. 17, 221–227.

Cornu, C.E., Sadro, S., 2002. Physical and functional responses to experimental marsh surface elevation manipulation in Coos Bay's South Slough. Res. Ecol. 10, 474–486.

Cox, T., Maris, T., De Vleeschauwer, P., De Mulder, T., Soetaert, K., Meire, P., 2006. Flood control areas as an opportunity to restore estuarine habitat. Ecol. Eng. 28, 55–63.

Crooks, S., Schutten, J., Sheern, G.D., Pye, K., Davy, A.J., 2002. Drainage and elevation as factors in the restoration of salt marsh in Britain. Restor. Ecol. 10, 591–602.

de Groot, A.V., Veeneklaas, R.M., Kuijper, D.P.J., Bakker, J.P., 2011. Spatial patterns in accretion on barrier-island salt marshes. Geomorphology 134, 280–296.

EDAW, Williams and Associates, Ltd., H.T. Harvey and Associates, Brown and Caldwell, Geomatrix, 2007. South Bay salt pond restoration project environmental impact statement/report. Final, Volume 1, December 2007. Prepared for U.S. Fish and Wildlife Service, and California Department of Fish and Game, Napa, CA.

Flick, R., 2003. Trends in United States tidal datum statistics and tide range. J. Waterw. Port Coast. Ocean Eng. 129, 155–164.

French, J.R., 1993. Numerical simulation of vertical marsh growth and adjustment to accelerated sea-level rise, North Norfolk, U.K. Earth Surf. Processes Landforms 18, 63–81.

Goals Project, 1999. Bay lands ecosystem habitat goals. A report of habitat recommendations prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project. U.S. Environmental Protection Agency, San Francisco, CA/S.F. Bay Regional Water Quality Control Board, Oakland, CA, 328 pp.

Greenberg, R., Maldonado, J., Droege, S., McDonald, V.M., 2006. Tidal marshes: a global perspective on the evolution and conservation of their terrestrial vertebrates. Bioscience 56, 675–685.

Harvey, T.E., Miller, K.J., Hothem, R.L., Rauzon, M.J., Page, G.W., Keck, R.A., 1992. Status and Trends Report on Wildlife of the San Francisco Estuary. U.S. Fish and Wildlife Service, Sacramento, California.

Jaffe, B.E., Smith, R.E., Torresan, L.Z., 1998. Sedimentation and bathymetric change in San Pablo Bay: 1956–1983. U.S. Geological Survey Open-File Report 98-759.

Jones & Stokes, 2004. Napa-Sonoma Marsh Restoration Project Environmental Impact Report. Final, Volume 1, April (J&S 01-396). Sacramento, CA. Prepared for California State Coastal Conservancy, Oakland, CA and Department of Fish and Game, Napa, CA.

Josselyn, M., 1983. The ecology of San Francisco Bay Tidal Marshes: A Community Profile. U.S. Fish and Wildlife Service, Division of Biological Services, Washington, D.C, FWS/OBS-83/23, 116 pp.

Kennish, M.J., 2001. Coastal salt marsh systems in the U.S. A review of anthropogenic impacts. J. Coast. Res. 17, 731–748.

Konisky, R.A., Burdick, D.M., Dionne, M., Neckles, H.A., 2006. A regional assessment of salt marsh restoration and monitoring in the Gulf of Maine. Res. Ecol. 14, 516–525.

Krone, R.B., 1987. A method for simulating historic marsh elevations. In: Coastal Sediments '87. Proceedings of the Specialty Conference on Quantitative Approaches to Coastal Sediment Processes, New Orleans, LA, pp. 316–323.

MacDonald, G.K., Noel, P.E., van Proosdij, D., Chmura, G.L., 2010. The legacy of agricultural reclamation on channel and pool networks of Bay of Fundy salt marshes. Estuar. Coast. 33, 151–160.

Mahall, B.E., Park, R.B., 1975. The ecotone between *Spartina foliosa* Trin. and *Salicornia virginica* L. in salt marshes of northern San Francisco Bay, Carnegie Institution of Washington, Department of Plant Biology, Publication no. 541, 811–819.

Maris, T., Cox, T., Temmerman, S., De Vleeschauwer, P., Van Damme, S., De Mulder, T., Van den Bergh, E., Meire, P., 2007. Tuning the tide: creating ecological conditions for tidal marsh development in a flood control area. Hydrobiologia 588, 31–43.

Nichols, F.H., Cloern, J.E., Luoma, S.N., Peterson, D.H., 1986. The modification of an estuary. Science 231, 567–573.

NOAA, 2010. Vertical Datum Transformation software version 2.3.3, Available: http://vdatum.noaa.gov/download.html (accessed 01.11).

Patrick, W.H., DeLaune, R.D., 1990. Subsidence, accretion, and sea-level rise in south San Francisco Bay marshes. Limnl. Oceanogr. 35, 1389–1395.

Pennings, S.C., Grant, M., Bertness, M.D., 2005. Plant zonation in low-latitude salt marshes: disentangling the roles of flooding, salinity and competition. J. Ecol. 93, 159–167.

Porterfield, G., 1980. Sediment transport of streams tributary to San Francisco, San Pablo, and Suisun Bays, California, 1909–1966. U.S. Geological Survey Water-Resources Investigations Report, pp. 80–64, 91.

PWA, Ltd., 2005. Napa-Sonoma Marshes Wildlife Area Restoration Ponds 3, 4, & 5. Prepared for Ducks Unlimited, Inc., Rancho Cordova, CA and California Department of Fish and Game, Napa, CA.

PWA, Ltd., 2002. Napa river salt marsh restoration: habitat restoration preliminary design phase 2 stage 2 of the hydrology and geomorphology assessment in support of the feasibility study final report. Prepared for The California State Coastal Conservancy, 37 pp.

Race, M.S., 1985. Critique of present wetland mitigation policies in the United States based on an analysis of past restoration projects in San Francisco Bay. Environ. Manage. 9, 71–82.

Sanderson, E.W., Ustin, S.L., Foin, T.C., 2000. The influence of tidal channels on the distribution of salt marsh plant species in Petaluma marsh, CA, USA. Plant Ecol. 146, 29–41.

Schoellhamer, D.H., 2011. Sudden clearing of estuarine waters upon crossing the threshold from transport- to supply-regulation of sediment transport as an erodible sediment pool is depleted: San Francisco Bay. Estuar. Coasts 34, 885–899.
Silvestri, S., Defina, A., Marani, M., 2005. Tidal regime, salinity, and salt marsh plan zonation. Estuar. Coast. Shelf Sci. 62, 119–130.
Simenstad, C.A., Thom, R.M., 1996. Functional equivalency trajectories of the restored Gog-Le-Hi-Te estuarine wetland. Ecol. Appl. 61, 38–56.
Simenstad, C., Reed, D., Ford, M., 2006. When is restoration not? Incorporating landscape-scale processes to restore self-sustaining ecosystems in coastal wetland restoration. Ecol. Eng. 26, 27–39.
Takekawa, J.Y., Woo, I., Athearn, N.D., Demers, S., Gardiner, R.J., Perry, W.M., Ganju, N.K., Shellenbarger, G.G., Schoellhamer, D.H., 2010a. Measuring sediment accretion in early tidal marsh restoration. Wetl. Ecol. Manage. 18, 297–305.
Takekawa, J.Y., Woo, I., Athearn, N.D., Demers, S., Gardiner, R.J., Perry, W.M., Ganju, N.K., Shellenbarger, G.G., Schoellhamer, D.H., 2010b. Erratum to: Measuring sediment accretion in early tidal marsh restoration. Wetl. Ecol. Manage. 18, 755.
Takekawa, J.Y., Schoellhamer, D.H., Miles, A.K., Shellenbarger, G.G., Athearn, N.D., Spring, S.E., Saiki, M.K., Jannusch, C.A., 2004. Initial biophysical changes after breaching a salt pond levee: final report on the Napa-Sonoma Wildlife Area Pond 3 breach. Unpubl. Progr. Rep., U.S. Geological Survey, Vallejo, CA, 42 pp.
Teal, J.M., Weinstein, M.P., 2002. Ecological engineering, design, and construction considerations for marsh restorations in Delaware Bay, USA. Ecol. Eng. 18, 607–618.
Thompson, T.H., 1877. Historical Atlas Map of Sonoma County, California. T.H. Thompson & Co., Oakland, CA, 121 pp.
URS Corporation, 2006. Napa plant site restoration project draft environmental impact report. Oakland, CA. Prepared for Resources Legacy Fund, Sacramento, CA, and California Department of Fish and Game, Napa, CA.
Verbeek, H., Storm, C., 2001. Tidal wetland restoration in The Netherlands. J. Coast. Res. 27, 192–202.
Wallace, K.J., Callaway, J.C., Zedler, J.B., 2005. Evolution of tidal creek networks in a high sedimentation environment: a 5-year experiment at Tijuana Estuary, California. Estuar. Coast. 28, 795–811.
Ward, K.M., Callaway, J.C., Zedler, J.B., 2003. Episodic colonization of an intertidal mudflat by native cordgrass (*Spartina foliosa*) at Tijuana Estuary. Estuar. Coast. 26, 116–130.
Warner, J.C., Schladow, S.G., Schoellhamer, D.H., 1999. Summary and analysis, hydrodynamic and water-quality data for the Napa/Sonoma Marsh complex. Final report: University of California, Davis, Department of Civil and Environmental Engineering.
Warner, J.C., Schoellhamer, D.H., Schladow, G.S., 2003. Tidal truncation and barotropic convergence in a channel network tidally driven from opposing entrances. Estuar. Coast. Shelf Sci. 56, 629–639.
Watson, E.B., 2004. Changing elevation, accretion, and tidal marsh plant assemblages in a south San Francisco Bay tidal marsh. Estuar. Coast. 27, 684–698.
Weinstein, M.P., Weishar, L.L., 2002. Beneficial use of dredged material to enhance the restoration trajectories of formerly diked lands. Ecol. Eng. 19, 187–201.
Weinstein, M.P., Teal, J.M., Balletto, J.H., Strait, K.A., 2001. Restoration principles emerging from one of the world's largest tidal marsh restoration projects. Wetl. Ecol. Manage. 9, 387–407.
Williams, P.B., Faber, P., 2001. Salt marsh restoration experience in San Francisco Bay. J. Coast. Res. 27, 203–211.
Williams, P.B., Orr, M.K., 2002. Physical evolution of restored breached levee salt marshes in the San Francisco Bay Estuary. Restor. Ecol. 10, 527–542.
Wolters, M., Garbutt, A., Bakker, J.P., 2005. Salt-marsh restoration: evaluating the success of de-embankments in north-west Europe. Biol. Cons. 123, 249–268.
Woo, I., Takekawa, J.Y., Gardiner, R., 2008. Guadalcanal tidal marsh restoration: 2007 annual data summary report, U.S. Geological Survey Western Ecological Research Center, San Francisco Bay Estuary Field Station, Vallejo, CA, 62 pp.
Zedler, J.B., Callaway, J.C., Desmond, J.S., Vivian-Smith, G., Williams, G.D., Sullivan, G., Brewster, A.E., Bradshaw, B.K., 1999. Californian salt-marsh vegetation: an improved model of spatial pattern. Ecosystems 2, 19–35.
Zedler, J.B., Callaway, J.C., 2000. Evaluating the progress of engineered tidal wetlands. Ecol. Eng. 15, 211–225.
Zedler, J.B., Kercher, S., 2005. Wetland resources: status, trends, ecosystem services, and restorability. Annu. Rev. Environ. Resour. 30, 39–74.