Sediment Dynamics and Vegetation Recruitment in Newly Restored Salt Ponds: Final Report for Pond A6 Sediment Study

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INTRODUCTION

This report presents the results of the Pond A6 Sediment Study (hereafter, the study) conducted for the South Bay Salt Pond Restoration Project (SBSPRP or the project) by the University of San Francisco and H. T. Harvey & Associates. This work focused on a key uncertainty affecting the South Bay Salt Pond Restoration Project planning and adaptive management process (EDAW et al. 2007), specifically: "*Will sediment accretion in restored tidal areas be adequate to create and to support emergent tidal habitat ecosystems within the 50-yr projected time frame?*" To address this question, our team monitored sediment dynamics at Pond A6 (breached in 2010) and elevations of vegetation establishment at Ponds A6 and A21 (one of the Island Ponds and breached in 2006) between December 2010 and March 2013.

Tidal marsh restoration has accelerated around San Francisco Estuary (SFE) since the 1970s, especially with major restoration planned in the South Bay Salt Pond restoration project, the Napa Marshes, Bair Island, Bahia, and others. Many of these restoration projects have been carried out by breaching levees to restore tidal influence to diked former marshlands that have subsided below elevations suitable for tidal marsh vegetation (Philip Williams and Associates Ltd. and Faber 2004). Following the levee breach, natural accretion through deposition of sediment suspended in tidal waters is required to raise the marsh surface to elevations suitable for vegetation establishment. This method is more cost effective than importing and placing sediment. However, uncertainties exist concerning the rate of natural sedimentation, if sufficient suspended sediment is available in SFE waters to raise the marsh surface to vegetated elevations (Stralberg et al. 2011), or if sediment accumulation in marsh restoration projects will result in the loss of adjacent mudflat habitats (EDAW et al. 2007).

The SBSPRP seeks to restore 15,100 acres of diked former salt ponds around the edge of South San Francisco Bay to a mix of habitats, including tidal marsh. The SBSPRP has commissioned targeted research to address key uncertainties related to the restoration of tidal marsh habitats, including monitoring sediment accumulation rates and elevation thresholds for vegetation establishment in Pond A21, and the effects of re-introducing tidal flow to the "Island Ponds" (Ponds A19, A20 and A21) on adjacent mudflats and tidal marshes (EDAW et al. 2007, Callaway et al. 2009). This report details the findings of the study carried out to identify sediment accumulation rates at Pond A6 and vegetation establishment thresholds at Ponds A6 and A21.

Understanding Sediment Accretion in Restored Marshes

Understanding sediment accretion within restored ponds is critical to on-going salt pond restoration planning. Elevation gain in tidal marshes is driven by deposition of suspended sediment from tidal waters and *in situ* organic matter accumulation (Morris et al. 2002). However, deposition of suspended sediment is critical in subsided restored tidal areas, as organic matter accumulation is negligible until plants have established. Suspended sediment concentrations are generally high in the South Bay relative to other portions of the SFE, and rapid sedimentation has been observed at the Island Ponds (Callaway et al. 2009) and in the vicinity of Pond A6 prior to this study. For example, mudflats and marsh at Calaveras Point and Ogilvie Island (immediately north and northeast of Pond A6 along Coyote Creek) have expanded rapidly via natural sedimentation in recent years (H. T. Harvey & Associates 2012a).

In addition to suspended sediment concentrations, sediment accumulation rates within marshes are affected by elevation and tidal inundation (Krone 1987, French 1993); areas at lower elevations typically accumulate sediment more rapidly due to longer periods of tidal inundation (e.g., see Figure 3 in Williams and Orr 2002). This has important implications for subsided ponds, many of which need to increase elevation substantially before vegetation can establish. Data have been collected from SFE marshes to evaluate this relationship (Williams and Orr 2002); however, this study provides additional data for low elevation locations needed to fine tune models of mudflat-marsh development and vegetation establishment.

Sedimentation rates at a particular location within tidal marshes and other intertidal habitats also are affected by topographic features, including the distance from channels and breaches, and channel size. In general, sedimentation is expected to be highest in and adjacent to larger, high-order marsh channels and lower in smaller, shallower channels (SFEI, 2004). As a result, the presence or absence of a dendritic channel structure in marshes is likely to affect sediment delivery within restored ponds. Borrow ditches, often found adjacent to levees in restored salt ponds, can capture much of the tidal flow to ponds, reducing scour and development of dendritic channel systems and affecting sediments inputs into restored ponds (ESA/PWA et al. 2011).

Another key uncertainty regarding the effect of salt pond restoration is whether the breaching of large ponds will act as new sediment sinks and shift sediment dynamics within the South Bay, potentially reducing sedimentation rates or increasing erosion from adjacent marshes, mudflats, and tidal channels. Mudflats provide habitat for large populations of migratory birds, and their preservation is a concern for the SBSPRP. Results to-date from analyses of sediment dynamics in tidal channels adjacent to the Island Ponds has indicated that rapid sedimentation at the Island Ponds did not come at the expense of adjacent mudflat and tidal marsh habitats. Within Pond A21, sediment accumulated rapidly following levee breaching (~1,500 g/m^2 per two-week interval as compared with 1,000 to 5,000 g/m^2 per year in a typical marsh) and concurrently, sediment accumulated on adjacent mudflats on Coyote Creek and Mud Slough (Callaway et al. 2009, H. T. Harvey & Associates 2012a). Localized erosion has been observed at the breach locations; however, large-scale erosion of the adjacent mudflats and tidal marshes has not been observed (H. T. Harvey & Associates 2012a).

Understanding sedimentation at Pond A6 is particularly important for the design of future tidal restoration phases of the SBSPRP. Restoration design features at Pond A6 included the use of ditch blocks for two of the four breaches to guide tidal flows into remnant slough channels instead of the deeper, borrow ditches. In addition, some external levee lowering was carried out, which also may influence the sedimentation patterns within Pond A6. Pond A21 restoration, by contrast, did not utilize either of these restoration techniques. At Pond A6, ditch blocks were constructed at the two northern external breaches. No ditch blocks were installed at the two southern external breaches. While this study was not designed to directly test the effect of ditch blocks and levee lowering on sedimentation patterns, their incorporation in conjunction with this study allowed us to qualitatively assess the implications for sediment dynamics of installing ditch blocks in tidal restoration projects.

Critical Elevations for Vegetation Establishment

Tidal marsh vegetation is highly sensitive to the frequency and duration of tidal inundation, which is directly affected by elevation relative to tides (Hinde 1954, Atwater and Hedel 1976, Watson and Byrne 2009). Shifts in elevation of 10 cm or less cause changes in dominant plant communities within Pacific Coast tidal marshes (Zedler et al. 1999, Sanderson et al. 2000). Inundation rates at a particular location within the marsh will be affected by the local tidal range (which increases substantially at the south end of the Bay (Philip Williams & Associates 2006)), elevation, the proximity to tidal channels, and any local features such as pans and natural levees that may impede drainage.

Newly establishing vegetation is likely to be even more sensitive to inundation given increased stresses on plants at germination and in early phases of growth. For example, at the Sonoma Baylands Tidal Wetland Demonstration Project, studies have found that while *Spartina foliosa* (Pacific cordgrass) can establish from as low as Mean Sea Level (MSL) to Mean High Water (MHW; Atwater and Hedel 1976, Zedler et al. 1999, Takekawa et al. 2012), most new recruitment of *S. foliosa* has occurred from approximately +0.2 to +0.4 m above MSL, well above the low end of the expected vertical range of *S. foliosa* (ESA/PWA and H. T. Harvey & Associates 2013).

Given these sensitivities, there is considerable variation within species in terms of the elevation relative to tides at which they occur in well-developed marshes (Zedler et al. 1999, H. T. Harvey & Associates 2005, Takekawa et al. 2012, Schile et al. unpublished data from northern San Francisco Estuary). Therefore, while these data provide a good approximation of the elevation at which species are likely to establish in restored marshes, it is essential to measure actual elevations of plant recruitment at nearby locations. As plants establish and spread sediment accretion rates typically increase due to increased trapping of suspended sediment by vegetation (i.e., by slowing water velocity through vegetation) and the contribution of belowground organic matter. Identifying the elevation(s) of vegetation establishment locally (e.g., at Ponds A6 and A21) is important to inform future predictions of vertical accretion and marsh evolution in restored salt ponds.

Objectives

This study addressed uncertainties for tidal marsh restoration and for future salt pond restoration projects by testing two hypotheses, listed below, related to sediment dynamics. In evaluating sediment issues, we focused on net sedimentation as measured over month to year time scales, as well as shorter-term (two week), mass-based measurements of sediment accumulation. Measurements at both time scales are important because they can identify temporal variations in sediment dynamics within the pond, especially differences in seasonal patterns of accumulation. In addition, the network of sampling stations will remain in place to permit future long-term evaluation of sediment dynamics. For plant recruitment, we focused on measuring elevations using a Real-Time Kinetic Global Positioning System (RTK GPS) and comparing these to tidal elevations across Ponds A21 and A6 in order to determine threshold elevations for the establishment of dominant species within this region of the South Bay. Our research included the following hypotheses:

Hypothesis 1: Rates of sedimentation are greatest at lowest elevations and decrease asymptotically towards a minimum value at MHHW (e.g., see Figure 3 in Williams and Orr [2002]). Our specific objective is to evaluate this relationship quantitatively so that it can be used as predictive tool in the South Bay. In addressing this hypothesis, we aim to answer the following questions: What are sedimentation rates in newly restored ponds? How do rates vary within the ponds and as a function of elevation?

Hypothesis 2: Absolute elevations for plant recruitment will vary across Ponds A21 and A6; however, elevation ranges relative to tidal levels will remain relatively constant for a given species. In addition, we hypothesize that critical elevations for vegetation establishment will vary significantly among dominant species, with their relative order similar to that found in natural marshes. (Note: Although there is likely very little difference in tidal range from Pond A21 to Pond A6, setting up these data for future comparison with other ponds [e.g., Eden Landing ponds] will help to fine tune the likely timeframe for vegetation recruitment in various ponds across the Bay.)

METHODS

Study Sites

Pond A6. Pond A6 was restored to tidal habitat on 6 December 2010 by breaching and lowering portions of the outboard levee and excavating pilot channels through the fringe marsh outboard of the breaches. Ditch blocks were constructed along the two northern levee breaches to decrease the tidal prism in the perimeter borrow ditch. Pond A6 subsided approximately 1.5 m after it was leveed to create a salt pond. The average elevation of Pond A6 prior to restoration was 0.7 m NAVD88, which is below MSL (approximately 1.0 m NAVD88) and below the elevation at which marsh vegetation typically colonizes emerging mudflats. Because of this low elevation, the reintroduction of tidal action to Pond A6 has initially created large areas of mudflat habitat. Over time, tidal channel and vegetated salt marsh habitats are expected to develop in Pond A6 as tidal channels re-form, sediment accumulates, and vegetation establishes on the emerging mudflats.

Pond A21. Pond A21 was breached in two locations along Coyote Creek in March 2006. Initial elevations at Pond A21 ranged from approximately 1.25 to 1.75 m NAVD88 (Callaway et al. 2009). Following restoration of tidal exchange, ~20 cm of sediment accumulated in the southern half of Pond A21 during 3 years of sediment monitoring, with ~5 cm in the northern half of the pond, which was at slightly higher initial elevation. Sedimentation rates were highest near the levee breaches and increased with initial pond depth (e.g., rates were highest at low elevations). Vegetation, including *Bolboschoenus maritimus* (alkali bulrush), *Salicornia pacifica* (perennial pickleweed), *S. foliosa*, *Distichlis spicata* (saltgrass), and *Salicornia depressa* (annual pickleweed) began to establish shortly after levee breach in portions of Pond A21 (Callaway et al 2009).

Sedimentation Rates at Pond A6

Our sedimentation sampling in Pond A6 focused on the first two years of sediment accumulation post-breach. We established ten sampling stations across Pond A6 (Figure 1). Stations were chosen to be broadly dispersed across the site and were evaluated in the field and established on 5 August 2010 (prior to breaching).

Long-term sediment accumulation was estimated by measuring the burial of sediment pins at the ten sampling stations. Sediment pins were established at the site on 30 August and 10 September, 2010. We pounded PVC pipe (3” diameter, schedule 80) into the sediment approximately 4 m using a modified fence-post driver. We capped each pipe, and approximately 1.5 m of pipe was left standing above the sediment surface. We attached a 1-m staff gauge to each pipe 40-50 cm above the sediment surface, and the vertical distance from the sediment surface to the bottom of the staff gauge on each installed pin was measured on 10 September, 2010 (prior to breaching).

On each subsequent sampling date, we measured the distance from the sediment surface to the bottom of the staff gauge. In some cases, enough sediment accumulated to bury the lower portion of the staff gauge with sediment, especially on the later sampling dates. In these cases, we measured the depth of the sediment above the bottom of the gauge, and this was added to the original distance between the sediment surface and the bottom of the gauge. Post breach, we measured sediment elevation relative to the pins on:

7 April, 2011	(4 months post-breach)
21 June 2011	(6 months post-breach)
19 December 2011	(12 months post-breach)
9 July 2012	(19 months post-breach)
9 November 2012	(23 months post-breach)
27 March 2013	(28 months post-breach)

In order to estimate short-term, mass-based rates of sediment accumulation, we used a modification of the “filter paper” method (Reed 1989). Rather than placing a filter paper on the existing substrate, we used thin rubber disks of approximately 12.5-cm diameter (“handy can openers” with a slight texture on the surface). A two-week period was chosen for deploying the sediment disks/pads to capture a full spring-neap sequence of tidal conditions.

On each sampling date except the first, we placed sediment disks in replicate pairs at all 10 sampling stations, approximately 30 cm apart from each other and within 2 m of the sediment pins. On the first sampling date (5 to 19 December 2011), we did not have time to reach all stations, and we established replicate pairs of disks at stations 1, 2, 3, 4, 5, 6, and 10. For all other sapling dates, sediment disks were established at all stations. For each of the two replicates at a sampling station, we attached two sediment disks to the existing sediment surface using screws that were inserted through both disks. The bottom disk was used simply to provide a clean surface and was not measured. After two weeks, we removed the disks from the sediment surface, and the top disk was carefully separated from the bottom disk so that the only sediment collected had accumulated on the surface of the top disk. The top disk was placed into a ziploc bag, and the disk and sediment were dried and weighed. The weight of the dry, clean disk was subtracted in order to calculate the sediment mass that accumulated over the given surface area of the sediment disk. Sediment mass was averaged across the replicate pairs at each pin and converted to a per m^2 value based on the area of the disk. Sediment disks were established and collected on:

Date of establishment	Date of collection
5 December 2011	19 December 2011
25 June 2012	9 July 2012

26 October 2012	9 November 2012
13 March 2013	27 March 2013

Sediment Sampling at Pond A6

We collected sediment samples from all sampling stations in June and December 2011, and in June/July, and October/November 2012 and delivered samples to Mark Marvin-DiPasquale at the U.S. Geological Survey for his future analysis of mercury in Pond A6.

In addition, we collected sediment samples at all ten sampling stations on 13 March 2013 for analysis of sediment characteristics. We collected four sediment cores (5.1-cm diameter and 5.7-cm depth) at each pin (sampling volume of the approximately 116 cm^3 per core), and these samples were processed separately for bulk density and organic matter content (n=4 per sampling station). Sediment samples were returned to the laboratory at USF, dried at 85°C until they reached a stable weight (approximately 24 hours), and weighed. Bulk density (g/cm^3) was determined based on sediment dry weight and the volume of sediment collected. Dried samples were ground with a mortar and pestle; subsamples of each ground sediment (approximately 15 g) were burned at 450°C for 8 hours and reweighed to determine sediment organic content (loss-on-ignition method; Ball 1964). In addition, subsamples from one core at each of the ten sampling stations were analyzed for particle size (texture) using the hydrometer method (Gee and Bauder 1986).

At Pond A21, we collected individual sediment cores (15-cm diameter and 30-cm depth) at four locations dominated by *S. pacifica* and four locations with no vegetation. Each core was sectioned into 5-cm intervals in the field. Sediment samples were returned to the laboratory at USF, dried at 100°C until they reach a stable weight (approximately 72 hours since samples were larger than those from Pond A6), and weighed. As above, bulk density (g/cm^3) was determined based on sediment dry weight and the volume of sediment collected. Because of the large size of samples from Pond A21, these samples were broken into quarters after drying, and one fourth of the original sample was ground for analysis of organic matter content and texture (methods as above; analysis of these samples is still in process).

Elevations for Vegetation Recruitment at Ponds A6 and A21

Prior to breaching, we measured the elevation of the sediment surface within 0.5 m of each sediment pin across Pond A6 using a RTK GPS unit (±5 cm accuracy) on 5 August, 2010. Additional elevation surveys with the same RTK GPS unit occurred following the breach on 21 June 2011, 25 June and 9 November 2012, and 13 March and 27 March 2013. Not all pins were surveyed on each date, and sometimes RTK errors occurred and measurements could not be taken. Additional pond surface elevations were surveyed between sediment pins on most dates. Although some small patches of *Spartina* and other vegetation have recruited adjacent to existing vegetation on the perimeter of the pond, no plant recruitment has occurred within the central area of the pond since breaching. Because of the lack of recruitment in central pond areas that make up the vast majority of the pond, no data for colonization elevations were collected at Pond A6.

We surveyed the elevation of newly recruited vegetation within Pond A21 on 9 December 2012, and 15 March 2013. On each date, we focused our surveying along the colonizing edge of established vegetation to identify the lower elevation limits of vegetation at this site.

In addition, we compared our data to prior studies that have measured the vertical ranges of established and newly recruited *S. foliosa* and other tidal marsh vegetation in the vicinity of Pond A21.

RESULTS AND DISCUSSION

Sedimentation Rates at Pond A6

Sediment accumulation at Pond A6 has been very rapid since breaching in December 2010, with cumulative depths of sediment erosion and deposition over 28 months ranging from -12 cm (erosion) to 68 cm (deposition) at individual sampling stations (Table 1). Station 2 was the only location that was primarily erosive throughout the entire sampling period (Table 1 and Figure 2). In all sampling periods, there was large-scale spatial variation in accumulation rates across Pond A6; however, patterns across time at individual stations were very consistent, i.e., stations that ranked highly over one period, typically ranked highly in the other sampling periods (Figure 2). Five of the ten stations showed accumulation of more than 50 cm of material over the entire sampling period (Stations 1, 3, 7, 8, and 9), and two stations were just below 50 cm (station 6 and 10). Two stations had slightly lower rates (Stations 4 and 5, both at 33 cm), and Station 2 remained erosive, as noted above. Stations 2, 4, and 5 are located in close proximity to Pond A6 breaches to Guadalupe Slough. The erosion observed at station 2 and the slower rate of sediment accumulation at stations 4 and 5 could be due to the proximity of these stations to the breaches and scour associated with the rapid ebb tide drainage from Pond A6, although there was no significant relationship between accretion/erosion and distance to nearest breach (simple linear regression: $R^2 = 0.0$, $P = 0.4$). We also note that the northern ditch block at the northwestern breach (between stations 4 and 5) along Guadalupe Slough failed and the borrow ditch along the northwestern portion of the pond appears to have captured much of the tidal prism associated with that breach (see Figure 3 and discussion below).

The average deposition across all ten sampling stations was 47 cm over 28 months (Figure 4), which is equal to an average annual accumulation rate of 20.2 cm/yr. There was a slight decrease in sedimentation rates in the last two sampling periods, compared to earlier sampling periods (Figures 2 and 4). Average “annualized” rates of accretion over the individual sampling periods were 19.0, 33.3, 21.5, 23.4, 13.6, and 14.0 cm/yr, reflecting the decrease in rates over the last two periods. Overall, the rate of accretion at Pond A6 was substantially greater than the initial rates of sediment accumulation measured in Pond A21 (Figure 4). By comparison, in the first 12 months post-breach at Pond A21, accretion was 13 cm in the southern (lower elevation) areas of Pond A21 (shown on Figure 4) and 3.3 cm in the northern (higher elevation) areas of Pond A21 (not shown on Figure 4), and accretion at Pond A21 slowed substantially down during the second and third year post-breach, with ~20 cm of material accumulating over 3 years in the southern areas of Pond A21. Changes in elevation documented with the RTK GPS survey equipment agreed well with the sediment pin measurements, showing similar trends across the site and over time (Figures 5 and 6).

At Pond A6, we found a negative relationship, albeit weak, between initial elevation and accretion across all stations (Figure 7A) with a stronger relationship if we excluded station 2, the one station that was erosional over the course of the study (Figure 7B). Combing data from Ponds A21 and A6, there was a significant negative relationship between deposition after 24 months and initial elevation (Figure 8; R^2 = 0.78, P < 0.0001). The weaker relationship within Pond A6 could be due to local variation and the narrower range of initial elevations. These results highlight that elevation is a key driver of deposition rates, although additional factors also affect overall patterns of deposition.

Based on sediment pads that were deployed for two-week periods, average short-term sediment accumulation rates across all stations were: 160 g/m^2/day (Dec. 2011), 365 g/m^2/day (Jun./Jul. 2012), 129 g/m^2/day (Oct./Nov. 2012), and 283 g/m^2/day (Mar. 2013; see Figure 9 for individual stations). In Jun./Jul. 2012 and Oct./Nov. 2012, we were not able to collect all sediment pads; however, of the 74 pads deployed across all sampling periods, only five were not collected (no pads were deployed at stations 7, 8, and 9 in Dec. 2011; a pair of replicate pads was deployed at all stations for each of the other three sampling periods; in Jun./Jul. 2012, no pads were relocated at station 3 and only one pad was relocated at both station 2 and 4; and only one pad was relocated at station 3 in Oct./Nov. 2012). Average short-term accumulation rates from Pond A6 during these sampling periods were slightly higher than rates that were measured at Pond A21 (rates at Pond A21 averaged approximately 160 and 60 g/m^2/day in the lower and higher elevation areas of the pond, respectively; Callaway et al. 2009). Given the small sample size and the large spatial and temporal variation at both Ponds A6 and A21, we did not compare the short-term sediment accumulation rates statistically.

We compared short-term sediment accumulation rates using sediment pads to the longer-term accretion rates measured using the sediment pins at Pond A6. The data from the sediment pins used in this analysis were from the single sampling interval that just preceded the sediment pad data, not the cumulative accretion/erosion (e.g., we compared sediment pads measured in Mar. 2013, to changes in sediment pins from the prior sampling period Nov. 2012 to Mar. 2013). Since the two methods are both measuring sediment accumulation (although on different scales and using mass-based vs. vertical measurements), we expected short-term and long-term rates to be correlated. However, spatial trends in sedimentation patterns from the sediment pad data were not consistently similar to trends from the pin data, with two significant relationships from the four sampling periods, one positive, and one negative (Figure 10). The R^2 value for these data were: 0.43 in Dec. 2011 (positive but not significant relationship, P = 0.11; based on seven of the ten sampling stations), 0.89 in Jun./Jul. 2012 (positive and significant relationship, P < 0.001; based on nine of the ten sampling stations), 0.01 in Oct./Nov. 2012 (not significant relationship, P = 0.74), and 0.45 in Mar. 2013 (negative and significant relationship, P = 0.03). The data point from station 2, which has been consistently erosional based on sediment pin data, was an outlier compared to the data from the other stations for multiple sampling periods. For example, for the data from Mar. 2013, data from this station had strong leverage in creating the negative relationship (high short-term accumulation was measured at this station (> 500g/m^2/day), along with erosion from the sediment pin data).

Sediment Sampling at Pond A6

Sediment characteristics from the ten sampling stations across Pond A6 were similar, with little variation among individual samples. Sediment bulk density in the top 5.7 cm of sediment ranged from 0.77 to 1.0 g/cm^3, with an overall average of 0.86 ±0.02 g/cm^3 (Table 2). The average bulk density for Pond A6 was twice the average bulk density found in sediments from natural salt marshes around SFE (0.43 g/cm^3 in the top 20 cm in “mid” marsh locations; Callaway et al.

2012). Restored salt marshes sampled by Callaway et al. (2012) also had lower bulk density values compared to Pond A6, but these restored marshes closely resemble natural marshes because of their relatively old age (restored in 1976 and 1980). Bulk density values from Pond A21 were intermediate between Pond A6 and tidal marshes, ranging from 0.54 to 0.74 g/cm^3 with an average of 0.59 ±0.04 g/cm^3 (Table 3). Sediment bulk density in mudflat and tidal marsh samples is strongly negatively correlated with sediment organic content (Turner et al. 2000, Callaway et al. 2012), as organic matter has much lower density than mineral matter. The differences in sediment bulk density across A6, A21, and natural marshes are likely due to differences in sediment organic matter content across these sites. Sediment bulk density in restored tidal marshes typically decreases over time as organic matter content increases due to the accumulation of plant roots and rhizomes. Contrary to these expectations, bulk density within Pond A21 was slight higher in vegetated areas than in unvegetated areas (Table 3). This could be due to slight local variations in sediment characteristics or variability in vegetation density (especially belowground biomass) across areas that we characterized as “vegetated”.

Sediment organic and mineral matter content also were consistent across Pond A6 with a range of 4.6 to 5.8% organic matter and 94.2 to 95.4% mineral matter (Table 4; organic matter content + mineral matter content = 100%). We used the quadratic regression from Callaway et al. (2012) to convert organic matter content to carbon content (see equation in Table 4; for natural salt marsh soils), and as expected carbon content was also similar across the site (Table 4). The organic and carbon contents within Pond A6 sediment were less than half those found in natural salt marshes within SFE (Callaway et al. 2012). As above, this is very likely a consequence of the lack of vegetation at Pond A6. We are presently analyzing sediment from Pond A21 for organic, carbon, and mineral matter and will report those comparisons as soon as possible.

Sediment texture across Pond A6 was relatively consistent, although the sample from station 5 had higher sand content than the other stations. In general, Pond A6 is dominated by clay particles with average clay content of 62%, and an average sand content of 8% (Table 5), similar to natural and restored marshes across the Estuary (Callaway et al. 2012). Samples from station 5 had the highest sand content, and sediment around this station was also very consolidated (personal observation).

Elevations for Vegetation Recruitment at Ponds A6 and A21

The pre-breach elevation survey on 8 August 2010 of Pond A6 documented surface elevations ranging from 0.31 to 1.13 m NAVD88, with the mean elevation of 0.70 ± 0.01 m NAVD88. Two years following tidal restoration, marsh surface elevations ranged from 0.5 to 1.06 m NAVD88 on the western portion of pond on 25 June, 2012 (no data were available for the eastern portion of the pond) and 0.5 to 1.37 m NAVD88 on 13 March, 2013. The lowest surveyed elevations occurred closest to the breaches, indicating tidal scouring.

An elevation survey conducted at Pond A21 in December 2012 indicated that vegetated elevations in the southern portion of the site ranged from 1.67 to 2.00 m NAVD88. The average elevation for *S. pacifica* was 1.95 ± 0.01 m NAVD88, and the average elevation for *S. foliosa* was slightly lower at 1.79 ± 0.01 m NAVD88 and ranged from 1.67 to 1.96 m NAVD88. Unvegetated areas close to the colonizing edge of *S. foliosa* were an average of 1.79 ± 0.01 m NAVD88, with values between 0.91 and 1.93 m NAVD88. A small number of *Spartina* plants at

Pond A21 were suspected to be *Spartina* hybrids. Locations of these plants were documented, and the location of the most suspect plant was reported to the Invasive *Spartina* Project (ISP) whose staff members visually confirmed it as hybrid *Spartina* (Whitney Thornton, ISP, pers. comm.). A more thorough round of testing by ISP in 2013 will determine genetic makeup of the recruiting *Spartina*. However, it is presumed that this site is almost entirely native *Spartina*, as confirmed by past genetic testing.

H. T. Harvey & Associates (2005) found *S. foliosa* along Mud Slough (bordering the north of Pond A21) ranged in elevation from 1.4 m to 2.0 m NAVD88. A separate analysis of vegetation establishment at Pond A21 using 2010 USGS LIDAR data found *S. foliosa* (interspersed with *S. pacifica*) established as of 2010 from between 1.8 and 2.0 m NAVD88 (H. T. Harvey & Associates 2012b). By contrast, across a range of SFE marshes, Takekawa et al. (2012) found *S. foliosa* establishes from approximately 0.2 to 0.7 m above MSL which corresponds to approximately 1.2 to 1.7 m NAVD88 in the vicinity of the Island Ponds. Measures of tides at the Island Ponds using interpolated tidal datums (Archbald 2010) found MHHW is 2.3 m, MHW is 2.1 m and MSL is 1.0 m NAVD88. Therefore, the initial seedling colonization elevation of *S. foliosa* at Pond A21 is focused in the upper 30% of the elevation range for the mature plant.

Spartina foliosa is known to colonize accreting mudflats slowly by seed. It produces small amounts of pollen (Anttila et al. 1998) and has high seedling mortality (Greer 1998). At Sonoma Baylands in San Pablo Bay, for example, marsh plain average elevations across most of the site reached suitable elevations for *S. foliosa* colonization in 2009, but vegetation mapping in 2012 found only about 15% of the site was vegetated by *S. foliosa*. Similarly, at Ponds A20 and A21, only about 10% of mudflats at suitable elevation had cover of *S. foliosa* in 2012, 6 years after opening the site to tidal exchange, despite that the majority of these ponds were at suitable elevations for marsh vegetation within a few years following restoration of tidal exchange (Callaway et al. 2009, H. T. Harvey & Associates 2012b).

Conclusions and Implications for Salt Pond Restoration Design

Based on results from Ponds A6 and A21, we expect high rates of sedimentation to continue for future restored ponds in the South Bay. Both sites developed very rapidly, with rates of sediment accretion that are orders of magnitude higher than those found in higher elevation, well-developed tidal marshes. Collection of data for suspended sediment concentrations at these and ponds restored in the future would be useful for calibrating models of marsh development, such as Marsh98 (Krone 1987, Stralberg et al. 2011), MEM (Schile 2012), and the WARMER model (Takekawa et al. 2012). Although other factors are also important, elevation affects sedimentation rates, as predicted, with higher accretion rates at low elevations (this is especially apparent across a wide range of elevations). Additional data to further develop this relationship will help to fine tune our understanding and future modeling efforts. Sediment bulk density at Pond A6 is higher than at natural marshes and organic matter is lower, reflecting the typical development of sediment characteristics from mudflat to vegetated marsh.

Despite the rapid increase in elevation at both Ponds, and the presence of large areas of “target” elevations at Pond A21, natural plant recruitment has been patchy, especially for *S. foliosa*. This is likely due in large part to the stochastic nature of seed germination and early plant survival. In addition, plant recruitment appears to occur at slightly higher elevations than the lowest elevation at which vegetation is found in natural, well developed marshes, likely because of the increased sensitivity of newly establishing vegetation to higher rates of inundation at slightly lower elevations. Given these factors, it is likely to take time for plants to recruit widely across the ponds, even after restored ponds reach target elevations of natural marsh vegetation.

Although this monitoring was not specifically designed to test the effectiveness of ditch blocks, they appear to have promoted development of channels. Ditch blocks were installed at both of the northern breaches to facilitate restoration of dendritic, historic slough channel networks; whereas no ditch blocks were installed at the two southern breaches. As noted above, the northern ditch block at the northwestern breach to Guadalupe Slough failed, and Figure 3 (color infrared satellite image taken in June 2012; courtesy of the City of San Jose) appears to indicate that the northwestern borrow ditch is a substantial channel draining to that breach. Similarly, it also appears that borrow ditches are substantial drainage features associated with the southern two breaches, which were installed without ditch blocks (Figure 3). In contrast, slough channel formation around the northeastern breach along Alviso Slough is progressing well where installed ditch blocks appear to be functioning according to the design intent; the historic/sinuous slough channel appears to be a primary drainage feature (Figure 3). A similar positive effect of ditch block installation on historic slough channel re-occupation was also observed during 10 years of monitoring at the Cooley Landing Salt Pond Restoration Project (ESA/PWA et al. 2011). Sediment deposition in the vicinity of the northeastern breach along Alviso Slough is somewhat higher than would be predicted by initial elevation alone; Figure 7A shows that pins 9 and 10 in proximity to the northeastern breach have relatively high sedimentation rates and also have relatively high initial elevations, although statistically not significant.

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Table 1. Cumulative accretion and erosion measured at Pond A6 since the breaching of the levees (from December 2010 through March 2013). Positive values indicate an increase in marsh elevation/accretion, while negative values indicate loss in elevation/erosion. Accretion and erosion were measured based on changes in pin heights. See Figure 1 for sampling station locations at Pond A6.

Cumulative Accretion / Erosion from December 2010 (cm)

Sampling Station	4 months (7 April 2011)	6 months (20 June 2011)	12 months (19 Dec 2011)	19 months (9 July 2012)	23 months (9 Nov 2012)	28 months (27 March 2013)
1	9.0	20.1	34.6	52.3	56.6	63.8
2	0.7	-1.5	-4.9	-4.4	-7.5	-11.6
3	9.0	19.3	31.3	46.5	53.9	58.8
4	1.5	6.3	14.0	21.5	27.4	33.4
5	7.6	9.7	15.7	23.6	27.4	32.5
6	7.3	19.6	27.0	37.8	42.8	48.8
7	6.5	15.7	34.3	53.0	57.0	65.8
8	-0.3	7.3	25.1	43.3	52.0	60.2
9	17.8	27.0	41.3	57.0	60.5	67.5
10	5.3	9.3	20.6	38.3	44.2	48.7
average (±std error)	**6.4 (1.7)**	**13.3 (2.7)**	**23.9 (4.2)**	**36.9 (5.9)**	**41.4 (6.6)**	**46.8 (7.6)**

Table 2. Average sediment bulk density (±standard error) in the top 5.7 cm of sediment at each sampling station at Pond A6 (n=4 for all stations).

Sampling Station	Average Bulk Density (g/cm^3)
1	0.78 (0.02)
2	0.85 (0.02)
3	0.77 (0.02)
4	0.80 (0.02)
5	0.85 (0.02)
6	0.83 (0.05)
7	0.94 (0.05)
8	0.92 (0.03)
9	1.00 (0.04)
10	0.90 (0.03)
overall average	0.86 (0.02)

Table 3. Average sediment bulk density (±standard error) in the top 5 cm of sediment at Pond A21 in locations dominated by *Salicornia pacifica* (SAPA) and in bare locations (n=4).

Location ID	Average Bulk Density (g/cm^3)
SAPA	0.64 (0.04)
bare	0.55 (0.10)
overall average	0.59 (0.04)

Table 4. Average sediment organic matter, carbon, and mineral matter content (±standard error) in the top 5.7 cm of sediment (118 cm^3 cores) at each sampling station (n=4 for all stations except n=3 for station 8). Organic matter content + mineral matter content = 100%. Carbon content was calculated using relationship from Callaway et al. (2012) [carbon content = $0.001217*\text{organic content}^2 + 0.3839*\text{organic content}$].

Sampling Station	Average Organic Matter Content (%)	Average Carbon Content (%)	Average Mineral Matter Content (%)
1	5.6 (0.2)	2.2 (0.1)	94.4 (0.2)
2	5.5 (0.3)	2.1 (0.1)	94.5 (0.3)
3	5.7 (0.2)	2.2 (0.1)	94.3 (0.2)
4	5.6 (0.1)	2.2 (0.0)	94.4 (0.0)
5	4.6 (0.4)	1.8 (0.1)	95.4 (0.4)
6	5.6 (0.3)	2.2 (0.1)	94.4 (0.3)
7	5.7 (0.1)	2.2 (0.1)	94.3 (0.1)
8	5.8 (0.1)	2.3 (0.0)	94.2 (0.1)
9	5.5 (0.1)	2.2 (0.1)	94.5 (0.1)
10	5.8 (0.1)	2.3 (0.0)	94.2 (0.0)
overall average	5.5 (0.08)	2.2 (0.03)	94.5 (0.08)

Table 5. Sediment texture in the top 5.7 cm of sediment (118 cm^3 cores) at each station (n=1).

Sampling Station	% Sand	% Clay	% Silt
1	9	62	29
2	3	61	36
3	5	63	33
4	5	62	32
5	28	57	15
6	4	68	29
7	4	59	38
8	0	62	37
9	8	60	33
10	15	65	20
overall average	8 (3)	62 (1)	30 (2)

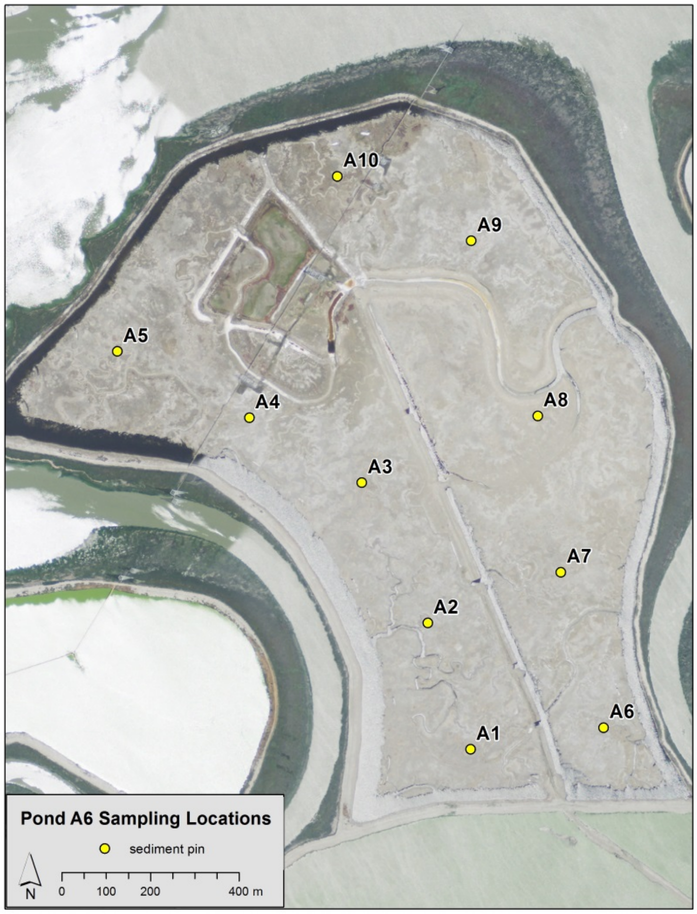

Figure 1. Map of sampling stations at Pond A6 (Note: sampling stations numbers are equivalent to pin numbers).

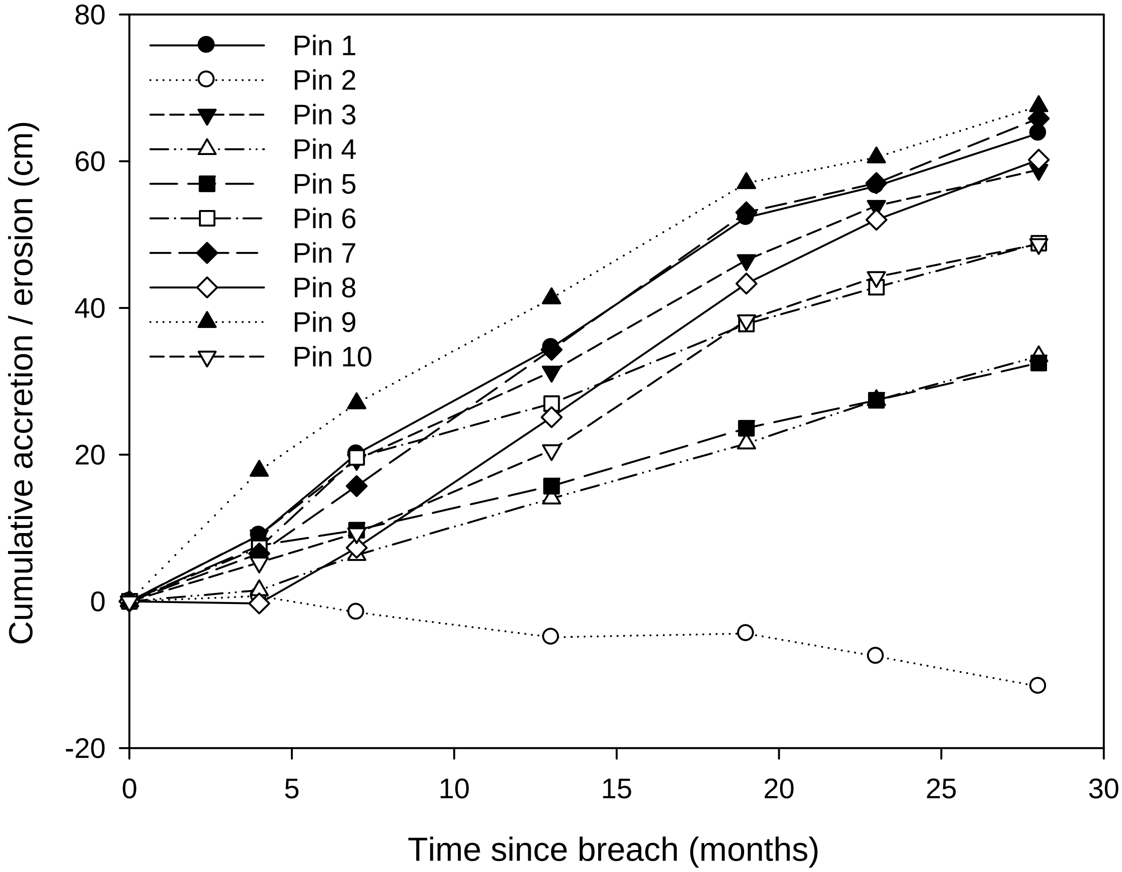

Figure 2. Cumulative accretion and erosion based on changes in pin heights at all stations at Pond A6 through March 2013. See Figure 1 for station locations.

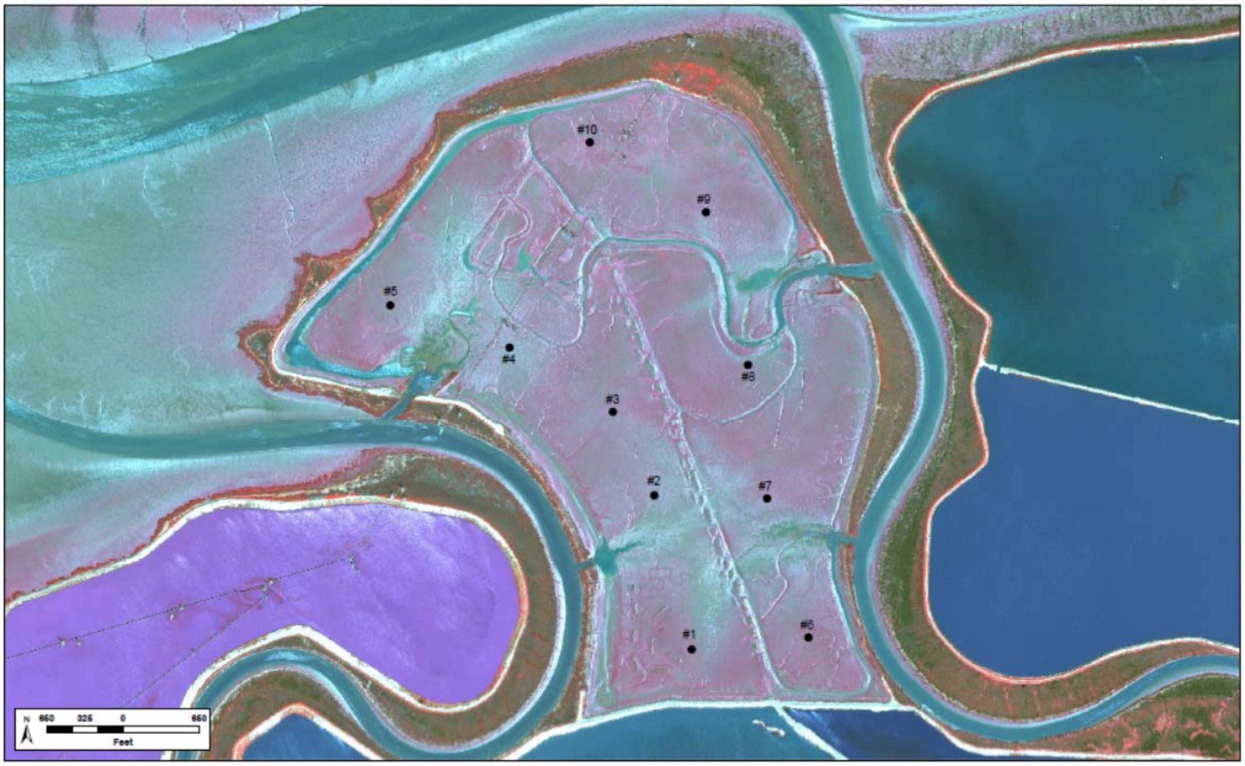

Figure 3. Color infrared satellite image of Pond A6 from June 2012 (courtesy of the City of San Jose.

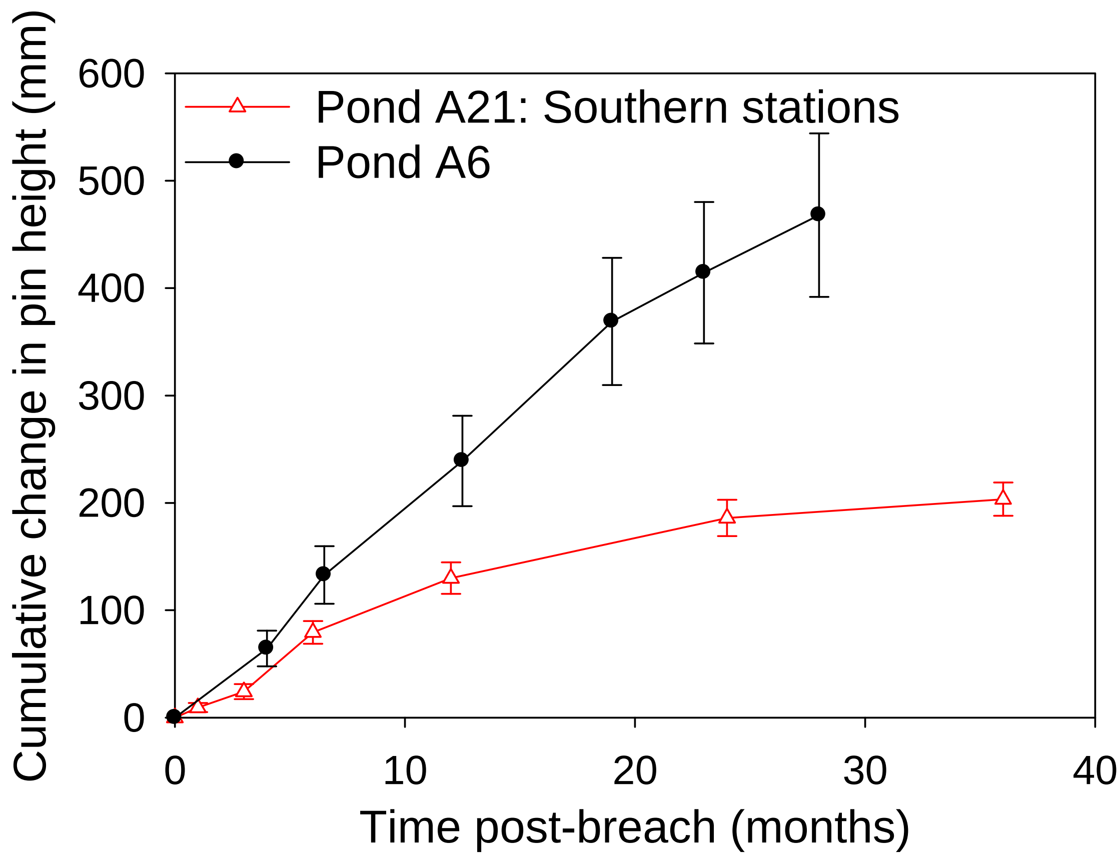

Figure 4. Average cumulative accretion based on changes in pin heights at all stations at Pond A6 (through March 2013) and the southern stations of Pond A21 (one of the three Island Ponds).

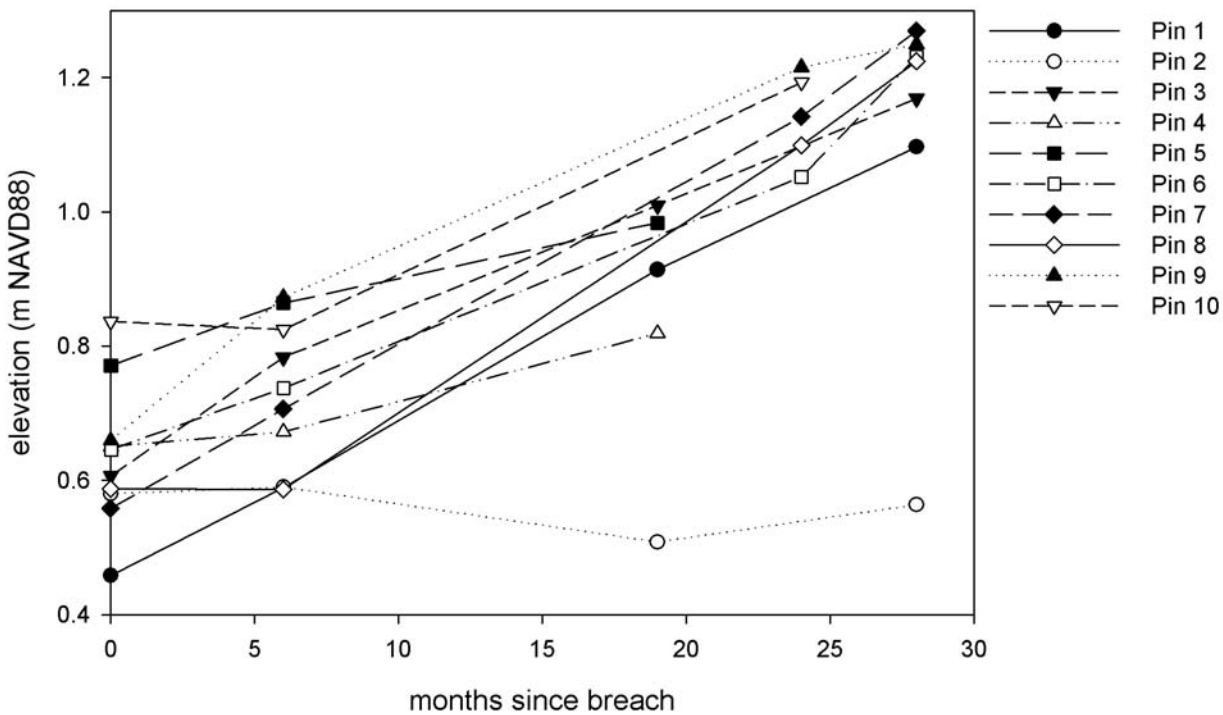

Figure 5. Change in elevation (m NAVD88) over time at each sediment pin as measured by a RTK GPS unit (±5 cm accuracy)

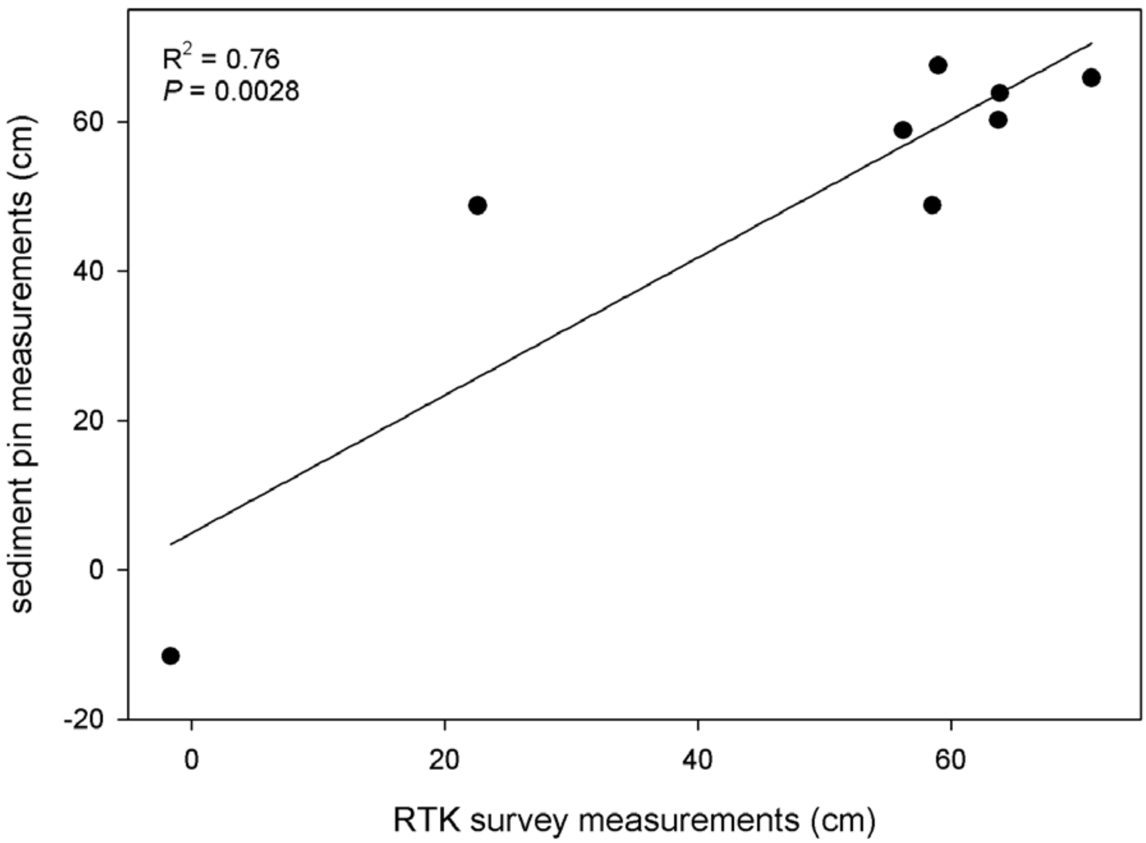

Figure 6. Comparison between marsh accretion/erosion after 28 months post breach using RTK survey and measurements of sediment deposition/erosion at each sediment pin.

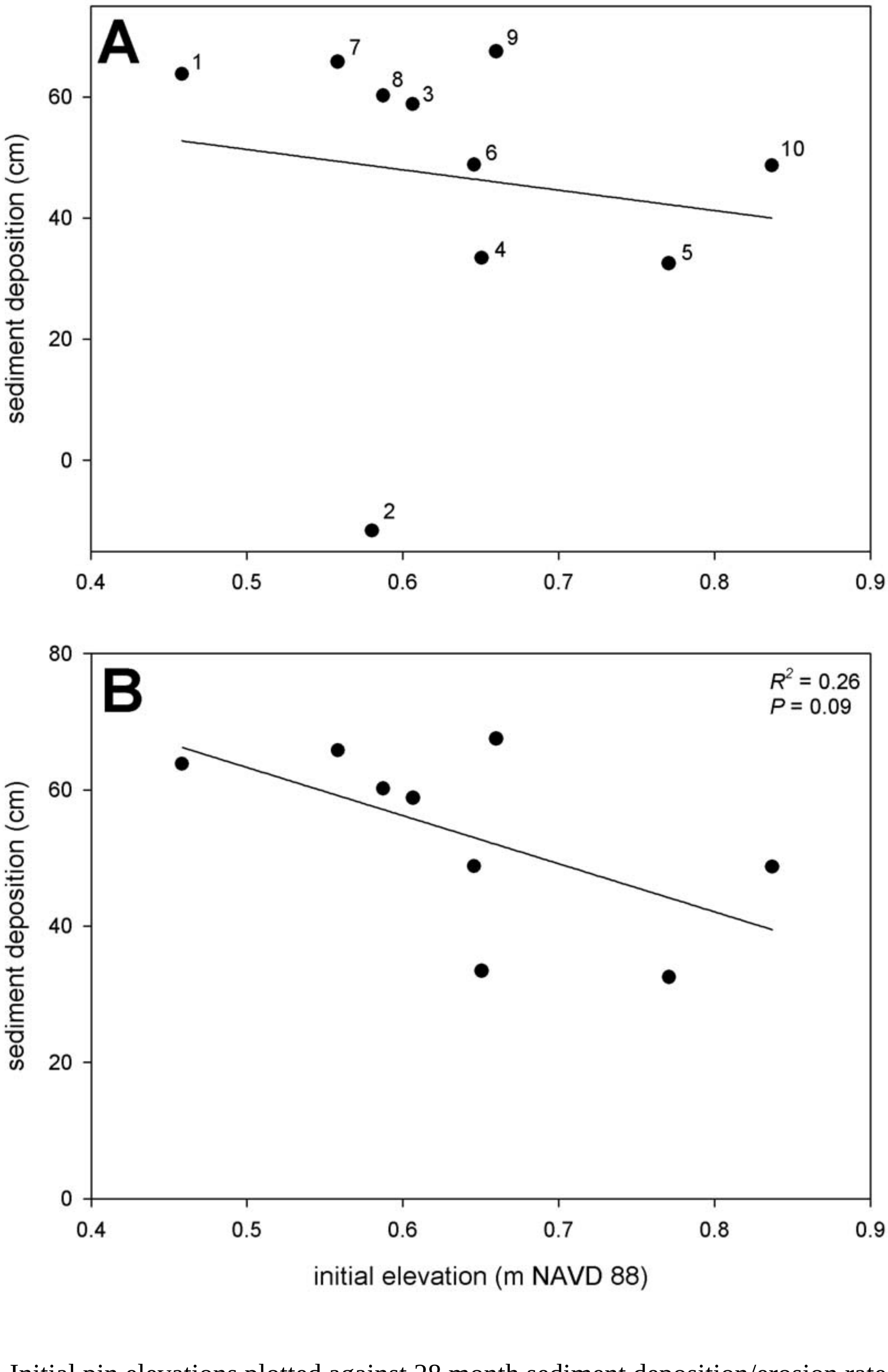

Figure 7. Initial pin elevations plotted against 28 month sediment deposition/erosion rates for A) all sediment pins with pin numbers labeled and B) pins where only deposition is occurring.

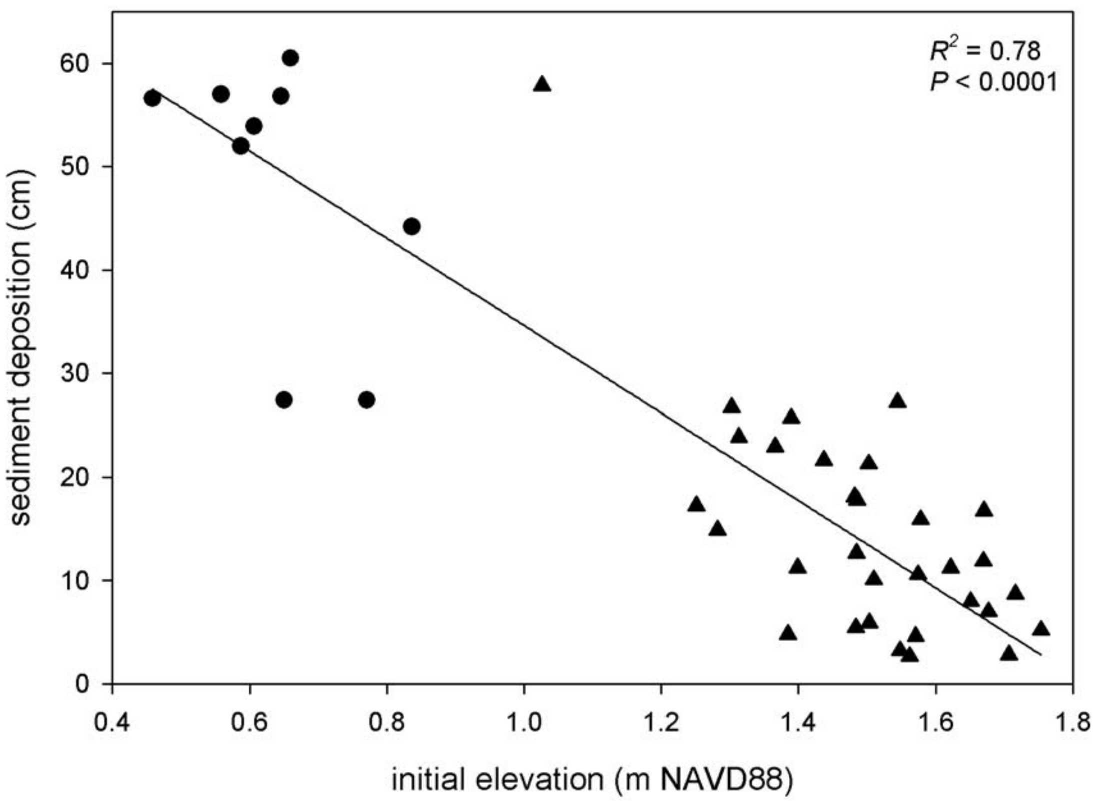

Figure 8. Initial elevation versus deposition rate after 24 months at Ponds A6 (circles) and A21 (triangles).

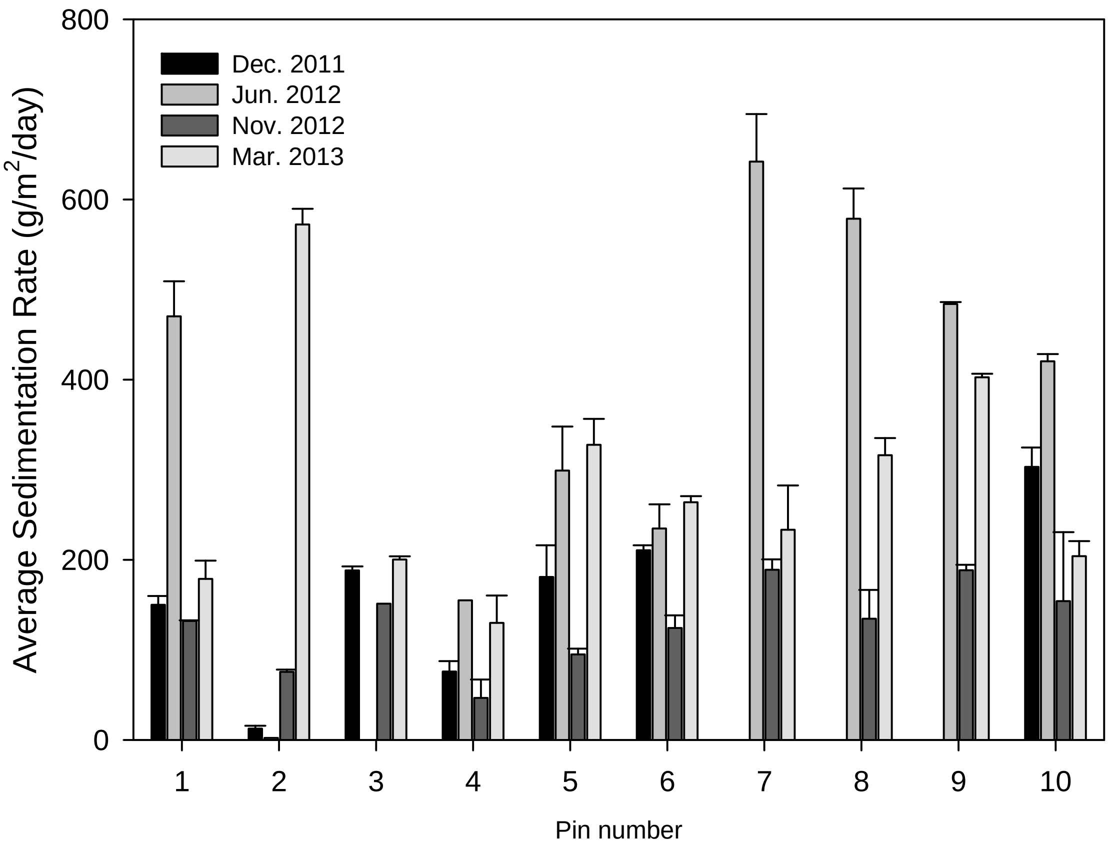

Figure 9. Short-term sediment accumulation rates calculated from sediment deposition over two-week tidal cycles. Where data are missing, either pads were not deployed or none were retrieved after deployment.

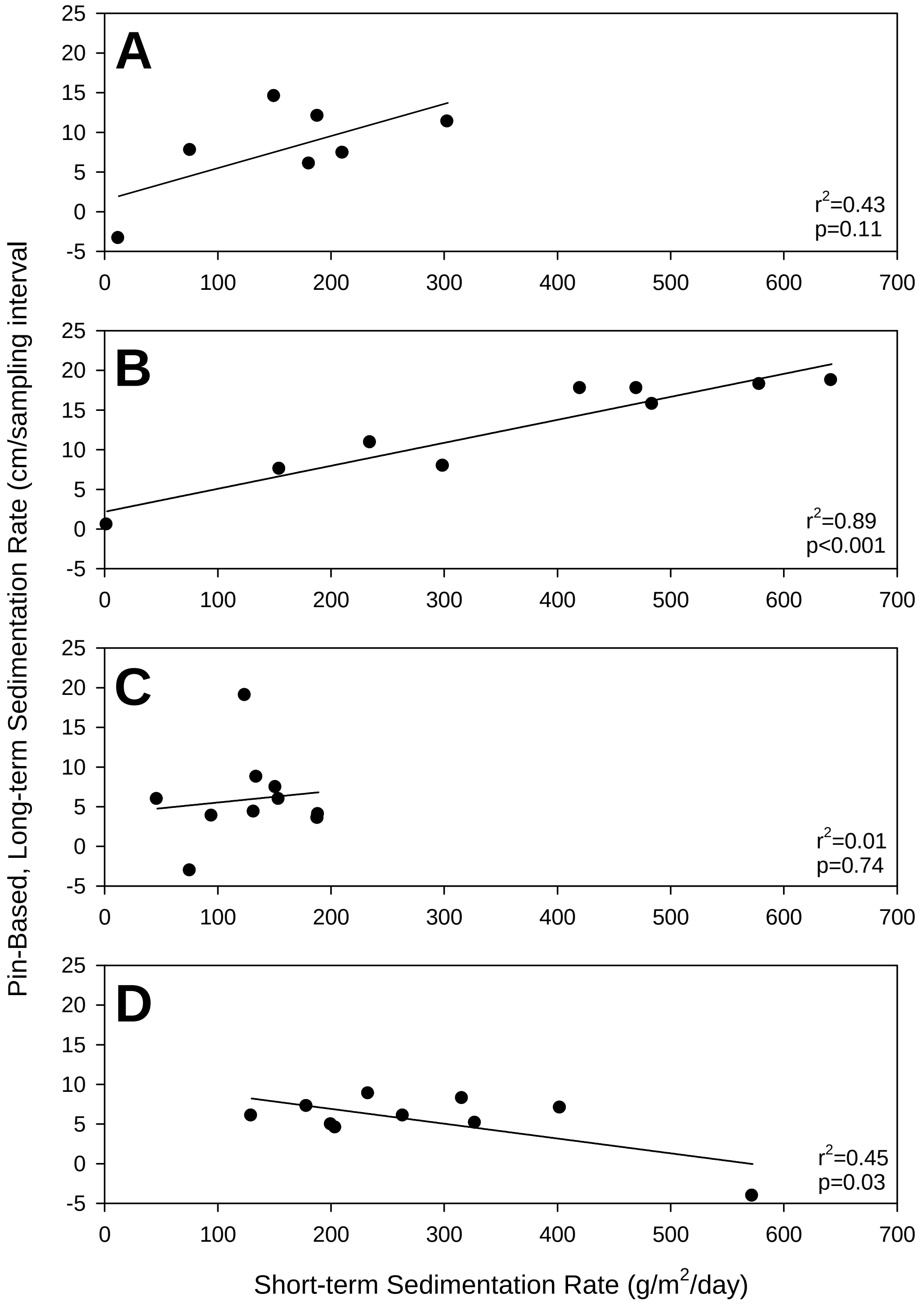

Figure 10. Short-term sedimentation rates (measured using sediment pads) versus long-term rates (measured using sediment pins and for individual sampling intervals, i.e., not cumulative). Sampling intervals were: A): Jun. to Dec. 2011; B) Dec. 2011 to Jul. 2012; C) Jul. 2012 to Nov. 2012; D) Nov. 2012 to Mar. 2013, with short-term sedimentation measured at the end of each sampling interval.