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Carbon Sequestration And Sediment Accretion In San Francisco Bay Tidal Wetlands

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Abstract Tidal wetlands play an important role with respect to climate change because of both their sensitivity to sea-level rise and their ability to sequester carbon dioxide from the atmosphere. Policy-based interest in carbon sequestration has increased recently, and wetland restoration projects have potential for carbon credits through soil carbon sequestration. We measured sediment accretion, mineral and organic matter accumulation, and carbon sequestration rates using ^{137}Cs and ^{210}Pb downcore distributions at six natural tidal wetlands in the San Francisco Bay Estuary. The accretion rates were, in general, 0.2–0.5 cm year^{-1}, indicating that local wetlands are keeping pace with recent rates of sea-level rise. Mineral accumulation rates were higher in salt marshes and at low-marsh stations within individual sites. The average carbon sequestration rate based on ^{210}Pb dating was 79 g C m^{-2} year^{-1}, with slightly higher rates based on ^{137}Cs dating. There was little difference in the sequestration rates among sites or across stations within sites, indicating that a single carbon sequestration rate could be used for crediting tidal wetland restoration projects within the Estuary.

Keywords Brackish marsh · Carbon offsets · Climate change · Mineral matter accumulation · Organic matter

accumulation · Salt marsh · San Francisco Bay Estuary · Sea-level rise · Sedimentation

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Introduction

Climate change directly affects tidal wetlands, primarily through changes in the rate of sea-level rise (SLR; Michener et al. 1997; Morris et al. 2002), and most, but not all, tidal wetlands have accumulated enough sediment to keep pace with recent rates of SLR of 1-3 mm year^{-1} (e.g., see summaries in Stevenson et al. 1986; Allen 2000; Callaway 2001; Friedrichs and Perry 2001). However, if the sea level rises 1 m or more by 2100, as has been predicted by some (Rahmstorf 2007; Vermeer and Rahmstorf 2009), then the ability of tidal wetlands to survive is uncertain (Kirwan et al. 2010). Furthermore, there is potential for substantial variation in sediment dynamics across the range of tidal wetland types and within individual wetlands as the sediment load changes with land use or freshwater discharge and as wetland plant communities shape these dynamics. The end of the Mississippi River delta, for example, has grown and shrunk over the last 200 years as sediment supply changed with land clearing and as dams were built (Tweel and Turner 2012).

Tidal wetlands play a role in buffering climate change by sequestering carbon (Chmura et al. 2003; Bridgham et al. 2006), and they are particularly effective at doing this because of their high primary productivity, ongoing sediment deposition, and relatively low decomposition rates (Connor et al. 2001; Hussein et al. 2004). The few studies evaluating how carbon sequestration rates vary across or within particular wetland complexes describe a wide range of rates (Craft 2007; McLeod et al. 2011), with many factors potentially affecting net carbon sequestration, including local plant

species composition and productivity, decomposition rates, allochthonous sediment imports, salinity, tidal range, and human activities. In particular, a number of studies have found higher rates of carbon sequestration in tidal wetlands with lower salinity (e.g., Craft 2007).

Policy-based interest in carbon sequestration has increased significantly as federal and state governments, as well as international agencies, consider the possibility of incorporating carbon credits for tidal wetland restoration into their plans to develop both voluntary and regulatory carbon markets intended to reduce CO_2 emissions (Freedman et al. 2009; Hansen 2009; Laffoley and Grimsditch 2009; Crooks et al. 2010). Carbon credits would be applied to tidal wetland restoration efforts, as well as to other wetland conservation activities, in order to sequester carbon that would otherwise enter the atmosphere (Emmett-Mattox et al. 2011). These credits likely would be based on carbon removed over approximately 100 years, but not for carbon that could decompose and return CO_2 to the atmosphere over shorter time periods (Crooks et al. 2010). This combined focus on restored wetlands and long-term carbon storage has created a conundrum because there are few, if any, restoration sites of sufficient age to estimate long-term carbon storage. Given this problem, measuring the amount of carbon stored in natural wetlands has been identified as a potential surrogate to establish baseline values for predicting carbon sequestration rates in restored sites. This approach assumes that restored and natural sites will behave in a similar manner over a timescale of centuries.

Tidal freshwater wetlands may have high rates of carbon burial (Craft 2007; Neubauer 2008), but they also can emit substantial amounts of methane, which has 25 times the greenhouse potential as carbon dioxide. This methane release to the atmosphere may be high enough to negate the benefits of carbon stored belowground (Bridgham et al. 2006; Kayranli et al. 2010; Poffenbarger et al. 2011). Saline wetlands produce little methane because the relatively high inputs of sulfate from marine waters promote sulfate reduction and minimize methane production (Bartlett and Harriss 1993; Magenheimer et al. 1996). Poffenbarger et al. (2011) reviewed a wide range of data on methane emissions and found little methane production if the soil pore water salinity was above 18 psu and highly variable emissions below this level. Because of concerns over methane production in freshwater wetlands, much of the interest in developing carbon credits for tidal wetland restoration has focused on saline wetlands. However, some managed freshwater sites may accumulate enough carbon to result in net carbon storage (Miller and Fujii 2010). It is necessary, therefore, to evaluate the variations in carbon sequestration rates across salinity gradients, within wetlands types (i.e., within salt marshes), and within locations to determine what credits might be awarded for future wetland restoration activities.

There is substantial interest in tidal wetland restoration in California, where large-scale restoration projects are underway or planned in the San Francisco Bay Estuary (e.g., Montezuma Wetlands in Suisun Bay, Hamilton Wetlands, the Napa-Sonoma Salt Pond Project, and the South Bay Salt Pond Project) and elsewhere (e.g., Bolsa Chica Wetlands in Huntington Beach and San Deiguito Lagoon in San Diego). The State of California passed Assembly Bill 32: The Global Warming Solutions Act in 2006, which set up the framework for a carbon trading program within the state as a way to accommodate emission reductions for large-scale emitters who will be forced to meet statewide emission reductions over the coming decades. The state is currently working to develop protocols for activities, such as tidal wetland restoration, that will receive carbon credits under the trading program (see http://www.arb.ca.gov/cc/cc.htm). Numerous studies have evaluated accretion rates and carbon dynamics on the Atlantic and Gulf Coasts (see compilations in Stevenson et al. 1986; Turner et al. 2000; Callaway 2001); however, these wetlands are dominated by *Spartina alterniflora* and *Spartina patens* and are quite different from the tidal wetlands in the San Francisco Bay Estuary (hereafter, Estuary) which are mostly dominated by perennial pickleweed, *Sarcocornia pacifica*, formerly *Salicornia virginica* (Emmett et al. 2000; Callaway et al. 2012). Because of these issues and others, it is risky to conclude that what has been learned about sediment and carbon dynamics in *S. alterniflora* and *S. patens* wetlands is directly applicable to Mediterranean-type tidal wetlands like those found in the Estuary and along much of the Pacific Coast. While a number of studies have looked at long-term sediment and carbon dynamics in the Estuary using ^{14}C-based dating methods (e.g., Watson 2004; Drexler et al. 2009), very few data exist for the Estuary or elsewhere on the Pacific Coast on the scale of 50–100 years, which is the timescale of interest for policy needs. Only one prior study used ^{137}Cs profiles to measure sediment accretion rates in San Francisco Bay salt marshes, with one core per site across three sites in South San Francisco Bay (Patrick and DeLaune 1990). No prior measurements of wetland sedimentation have been completed in the Estuary using ^{210}Pb-based dating.

The objectives of our research were to evaluate the ability of salt and brackish tidal wetlands in the Estuary to keep pace with sea-level rise through sediment accretion and to estimate the potential for wetland carbon sequestration. We measured the rates of vertical accretion, as well as mineral matter, organic matter, and carbon accumulation, within six wetlands using two different dating methods. We also compared soil characteristics from natural and restored salt marshes in the Estuary to evaluate the use of natural wetlands as surrogates in studies of the carbon dynamics within restored wetlands. We hypothesized that:

1. Sediment accretion, carbon, and mineral matter accumulation rates are higher in tidal brackish wetlands compared to in salt marshes.
2. Sediment accretion, carbon, and mineral matter accumulation rates within individual wetlands are highest closer to the bay margin or large tidal creeks and lowest at more inland locations.

Methods and Materials

Site Locations

We collected sediment cores from six natural and two restored tidal wetlands in the Estuary (Fig. 1). Four of the natural sites were saline wetlands, including: Whale's Tail (WT), China Camp (CC), Petaluma River Marsh (PRM), and Coon Island (CI). We originally thought that the WT site was a historical wetland like the rest of our natural sites, only to discover later that it was close to 100 years old. Two sites were tidal brackish wetlands: Rush Ranch (RR) and Browns Island (BRI). Additional sediment cores were collected from two restored salt marshes: Cogswell Marsh (CM) in South SF Bay and Muzzi Marsh (MM) in the North Bay. Cogswell Marsh was restored in 1980 and Muzzi Marsh in 1976 (Williams and Faber 2001). Cores were not collected from restored brackish wetlands because there are few existing restored brackish sites in the Estuary. The cores from the two restored wetlands could not be dated using radioisotopic methods because of their relatively young age. The soil characteristics of these restored sites, however, were compared to data from the natural wetlands.

The saline sites are generally dominated by *Spartina foliosa* in the area adjacent to the bay margin and in a narrow margin along tidal creeks and by *S. pacifica* throughout the broad marsh plain. The brackish wetland sites are typically dominated by *Schoenoplectus acutus* or *Schoenoplectus californicus* adjacent to creeks and the bay margin, with the marsh plain areas dominated by *S. pacifica*, *Schoenoplectus americanus*, *Distichlis spicata*, and other species whose relative proportions depend on variations in salinity (Table 1). The Coon Island wetland has a lower salinity range than the other salt marshes and includes a mix of salt and brackish wetland vegetation. Based on vegetation, the Coon Island wetland could be considered a brackish wetland; however, based on soil characteristics and sediment relationships (see "Results"), it is more similar to the other salt marshes than to the brackish wetlands. Two of the salt marsh sites (WT and CC) are located on the edge of the Estuary where there is little local freshwater input. The two other salt marsh sites are on the Petaluma River (PRM) and

Napa River (CI) and are approximately 7.5 and 10 km upriver of the major bay shoreline, respectively. One of the brackish sites (RR) is in northern Suisun Bay and is somewhat removed from the direct influence of the river, while the other brackish site (BRI) is located at the confluence of the Sacramento and San Joaquin Rivers on the western edge of the Sacramento–San Joaquin Delta and the eastern edge of Suisun Bay (Fig. 1).

Soil Sampling, Soil Properties, Carbon Analysis, and Dating Techniques

Each site had six stations distributed in the low, middle, and high areas along two parallel transects (Fig. 1). We used ArcGIS to randomly identify multiple potential starting points for transects originating in the upper portion of each wetland (adjacent to upland areas for most sites or the central portion of island sites); two of these potential starting points were randomly chosen as the beginning of each transect. The lower end of the transect was oriented in the direction of the bay or the nearest large tidal creek (>10 m wide); the transect midpoint was halfway to the low-marsh station, which was approximately 20 m from the bay or creek shoreline. We collected 11 sediment cores at the Coon Island wetland along four transects to examine the spatial variation across a single site in more detail than at the other sites. We collected a total of 41 cores for dating and analysis of sediment characteristics (five sites with six cores per site and one site with 11 cores). An additional 12 cores were collected from the restoration sites (two restored sites with six cores per site) that were analyzed for sediment characteristics only.

We collected these cores from December 2008 to February 2011 using a 15-cm diameter sharp-edged, aluminum soil coring tube (Hargis and Twilley 1994). The sample depth ranged from 48 to 54 cm. Compaction during core collection was estimated by measuring the difference in elevation inside and outside of the coring tube to the nearest millimeter. Cores were not used if compaction was >3 cm, and compaction for all cores averaged 0.3 cm. Soil cores were sectioned into 2-cm intervals in the field and put in airtight Ziploc bags. The samples were weighed in the laboratory, dried at 85°C for ~72 h, and reweighed to determine their bulk density (BD). Half of the dried sample was ground with a mortar and pestle for further analysis. The elevation and location of each station were surveyed using a Leica 1200 Series Real-Time Kinematic GPS system (horizontal accuracy, ~1 cm; vertical accuracy, ~3 cm).

We measured the soil organic and mineral matter content of each 2-cm section using the loss-on-ignition (LOI) method (Ball 1964; Craft et al. 1991). Dried and ground subsamples were weighed, burned at 450°C for 8 h, and reweighed. Soil organic matter (OM) content can be used

Fig. 1 Locations of sampling sites in San Francisco Bay estuary, including both natural and restored wetlands. The salt marsh sites were Whale's Tail, Cogswell Marsh (restored), Muzzi Marsh (restored), China Camp, Petaluma River Marsh; Coon Island includes a mix of brackish and salt marsh vegetation, but is classified as a salt marsh based on the sampling locations and soil conditions measured in this project. The brackish wetland sites were Rush Ranch and Browns Island
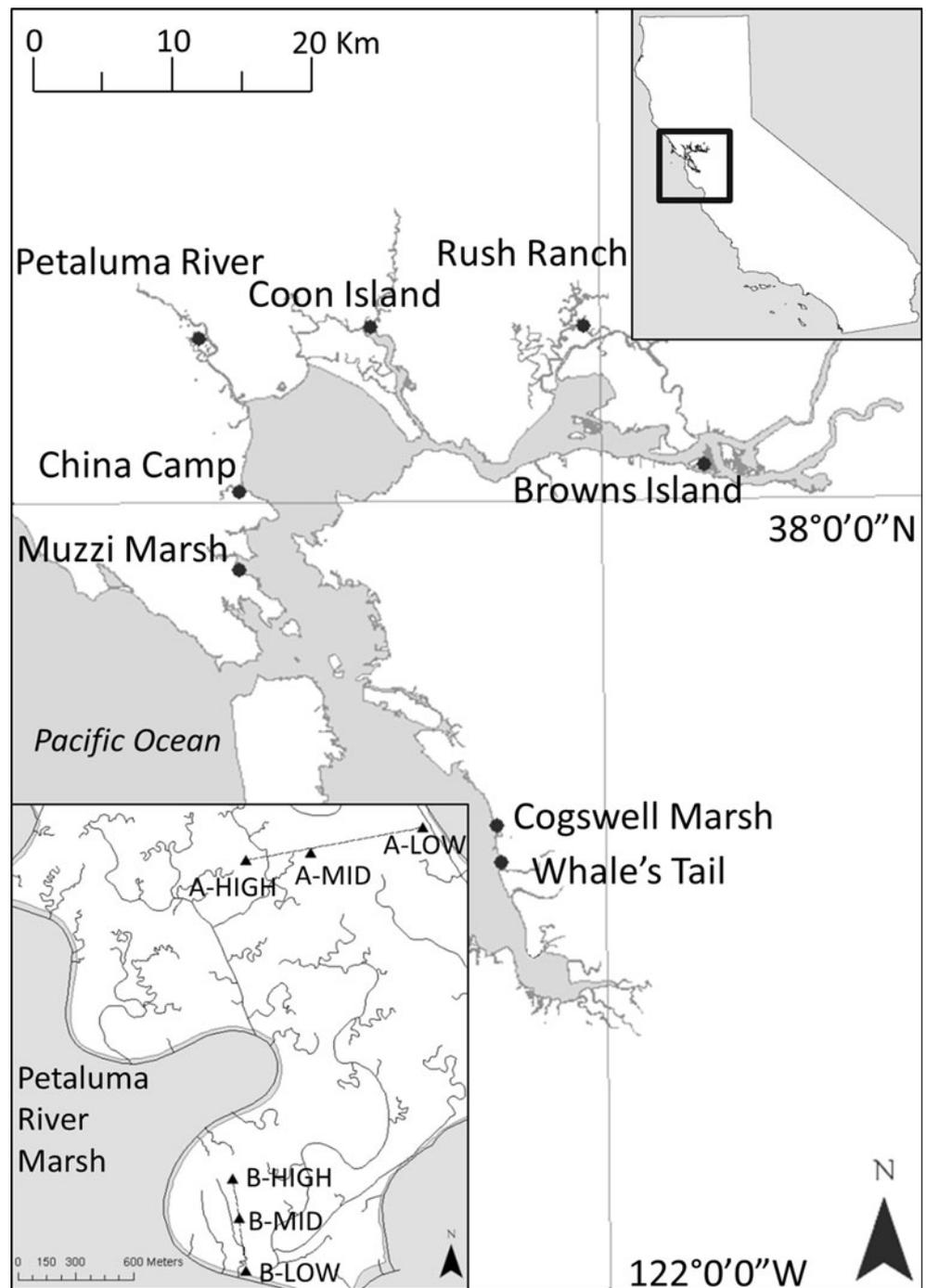
as an indirect measure of carbon content because the concentration of carbon within the OM is relatively constant (Craft et al. 1991). In order to establish a correlation between LOI measurements and carbon content, carbon content was measured directly for a subset of 2-cm sections (97 samples) using a Flash 2000 Organic Elemental Analyzer manufactured by CE Elantech. A standard soil reference material was analyzed every six samples throughout all carbon analyses. The relationship between OM content and carbon content from our samples was compared with

that from tidal wetland sediments made by Craft et al. (1991). We measured the soil texture of composite 10-cm intervals for all cores using the hydrometer method (Gee and Bauder 1986).

The sediments were dated by determining the downcore distribution of ^{210}Pb and ^{137}Cs activity. The radioisotope ^{137}Cs, a residual of nuclear bomb fallout, first appeared in measurable amounts in the early 1950s, peaked in the spring of 1963 with additional large amounts in early 1964, and declined to undetectable amounts after 1986, although the

Table 1 Tidal range, salinity, and dominant vegetation for sampling sites in the San Francisco Bay estuary (abbreviations for each site are included below each site name)

Site	Mean tidal range (m)	Salinity regime	Dominant vegetation
Brackish marshes			
Browns Island (BRI)	0.92	Oligohaline	*Schoenoplectus acutus, Schoenoplectus californicus, Distichlis spicata, Typha domingensis, Triglochin maritima*
Rush Ranch (RR)	1.72	Mesohaline	*S. acutus, S. californicus, Schoenoplectus americanus, Juncus balticus, D. spicata, Sarcocornia pacifica, Grindelia stricta*
Salt marshes			
Coon Island (CI)	1.43	Polyhaline	*Spartina foliosa, S. pacifica, Bolboschoenus maritimus, S. americanus, D. spicata, G. stricta*
Petaluma River Marsh (PRM)	1.49	Polyhaline–hyperhaline	*S. foliosa, S. pacifica, Cuscuta salina, Jaumea carnosa, D. spicata, Frankenia salina*
China Camp (CC)	1.31	Polyhaline–hyperhaline	*S. foliosa, S. pacifica, G. stricta, D. spicata, C. salina*
Whale's Tail (WT)	1.80	Polyhaline–hyperhaline	*S. foliosa, S. pacifica*
Restored salt marshes			
Muzzi Marsh (MM)	1.26	Polyhaline–hyperhaline	*S. foliosa, S. pacifica, D. spicata*
Cogswell Marsh (CM)	1.80	Polyhaline–hyperhaline	*S. foliosa, S. pacifica*

Chernobyl accident in 1986 deposited ^{137}Cs in some locations (Callaway et al. 1996a). The sediments were dated with ^{137}Cs using the peak accretion rate at the 1963/1964 layer. The radioisotope ^{210}P is naturally occurring and is continually deposited by rainfall. The disequilibrium between ^{226}Ra and its long-lived decay product ^{210}Pb (half-life=22.3 years) is widely used to date aquatic and wetland sediments, but with different modeling approaches depending on sediment characteristics, radioisotopic analyses, and assumptions. ^{210}Pb and ^{137}Cs activity was determined using an integrated gamma spectroscopy system consisting of a Princeton Gamma-Tech 60-mm diameter intrinsic germanium "N"-type coaxial detector (40 % efficiency) interfaced to an EG&G Ortec 92X Spectrum Master® multichannel spectrum analyzer. We used Ortec's Gamma-Vision® software to analyze the data. The system was calibrated using a multinuclide (60–1836 keV) standard solution (Isotope Products no. 7600) that was mixed with an estuarine soil and then packed in a 26-cm^3 plastic Petri dish. ^{210}Pb activity was determined using a Canberra Low-Energy Germanium Detector (model GL3825), which was interfaced to a Canberra Digital Spectrum Analyzer (model DSA-100), and Canberra's Genie 2000 software. The calibration of the systems was checked quarterly for ^{210}Pb and ^{137}Cs activity with standard sediment from the International Atomic Energy Agency (IAEA-300). Samples for ^{210}Pb were counted for about 48 h, depending on the sample sediment qualities. The background, with an empty Petri dish, was counted every 6 days for 48 h. The excess (unsupported) ^{210}Pb is calculated by subtracting the supported ^{210}Pb activity from the total ^{210}Pb activity. The supported ^{210}Pb was estimated from the activity of ^{214}Pb (295 and 352 keV).

The accretion rates using the downcore distribution of ^{210}Pb were calculated using the constant initial concentration model (Robbins et al. 1978). This model assumes that there is negligible migration of ^{210}Pb and associated radionuclides in sediments, a constant input of ^{210}Pb from the atmosphere, and a constant proportion of ^{210}Pb trapped in sediments by weight. This model is applied under the assumption that the excess ^{210}Pb in sediments deposited at the surface will be the same regardless of the sedimentation rate. This model uses the cumulative sedimentation rate downcore, a decay constant of ^{210}Pb, and the change of excess ^{210}Pb activities with depth to estimate sediment accretion rates. A linear regression of the natural log (ln) of the excess ^{210}Pb activity vs. depth was used to estimate sedimentation rate. The activity of ^{210}Pb in the first few centimeters of the surface can aid in determining how much mixing has occurred. Counting errors were calculated using the counting uncertainties of ^{210}Pb and ^{214}Pb and the formula from Binford (1990). The counting uncertainty ranged from <10 % for surface samples (<10 cm) to 15 %

Table 2 Average sediment organic matter content and bulk density (±standard error) in the top 20 cm of cores from low-, mid-, and high-marsh stations at all sampling sites (n=2 for all stations, except n=1 for Browns Island Low, Petaluma River low, Whale's Tail Mid and High, and n=3 for Coon Island Mid and High)

Site	Organic matter (%)			Bulk density (g cm³)		
	Low	Mid	High	Low	Mid	High
Browns Island	29.3 (n=1)	48.6 (1.40)	59.5 (7.42)	0.29 (n=1)	0.18 (0.001)	0.16 (0.019)
Rush Ranch	18.0 (0.77)	43.9 (4.06)	32.0 (3.28)	0.41 (0.026)	0.25 (0.004)	0.27 (0.051)
Brackish marsh means	23.7	46.2	45.7	0.35	0.22	0.21
Coon Island	13.5 (5.27)	19.2 (4.60)	18.3 (1.48)	0.63 (0.197)	0.43 (0.147)	0.36 (0.037)
Petaluma River	7.6 (n=1)	16.1 (4.13)	25.1 (1.22)	0.82 (n=1)	0.39 (0.093)	0.23 (0.001)
China Camp	11.7 (1.92)	12.7 (3.81)	15.9 (3.33)	0.52 (0.108)	0.46 (0.066)	0.47 (0.082)
Whale's Tail	8.4 (0.02)	13.2 (n=1)	12.8 (n=1)	0.62 (0.023)	0.43 (n=1)	0.40 (n=1)
Salt marsh means	10.3	15.3	18.0	0.65	0.43	0.37
Muzzi Marsh	11.1 (1.29)	11.5 (1.20)	10.8 (0.74)	0.44 (0.025)	0.44 (0.058)	0.68 (0.163)
Cogswell Marsh	9.3 (1.80)	9.0 (0.26)	9.2 (0.34)	0.62 (0.001)	0.55 (0.054)	0.53 (0.026)
Restored marsh means	10.2	10.3	10.0	0.53	0.50	0.60

for deeper samples. The sum of the excess ^{210}Pb was estimated for each core using the interpolated values for core segments not counted and the measured dry weights.

Some cores could not be dated successfully (discussed in Milan et al. 1995) because the local area may experience disturbance, bioturbation, erosion, or rapid sediment accumulation, all of which can compromise dating quality. A set of criteria was developed to determine whether the calculated accretion rate was usable. The criteria used were: (1) agreement between the ^{137}Cs and ^{210}Pb rates, (2) whether there was a significant amount of ^{137}Cs at the surface, and (3) the inventory of ^{137}Cs in the core. The use of ^{137}Cs as a sediment dating tool is based on the assumption that there has been a constant deposition of material to the surface and that sediment has not been removed or added. Because ^{137}Cs has not been deposited from the atmosphere since the late 1980s, the presence of ^{137}Cs at the surface brings into question the validity of the calculated accretion rate. The expected inventory of ^{137}Cs in a core can be calculated based upon the precipitation data and atmospheric fallout records of ^{137}Cs and ^{90}Sr (Milan et al. 1995). Low or high inventories of ^{137}Cs in a core call into question whether the accretion of sediment occurred in an undisturbed manner, and thus the validity of the calculated accretion rate. Based upon these criteria, the cores were ranked as "excellent" if: the accretion rates based on the downcore distribution of ^{137}Cs and ^{210}Pb agreed, there was no ^{137}Cs at the surface, and if the measured ^{137}Cs inventory was close to the estimated inventory. The cores were ranked "good" if the accretion rates from the ^{137}Cs and ^{210}Pb dating methods agreed and if there was ^{137}Cs at the surface and/or a relatively high ^{137}Cs inventory. The cores were classified as "marginal" if the accretion rates from the two dating

methods did not agree or if there was ^{137}Cs at surface and/or a relatively high ^{137}Cs inventory.

We calculated the mineral, organic, and carbon accumulation rates for each core using the accretion rates determined with ^{137}Cs and ^{210}Pb, and the mineral, organic, and carbon densities (the product of bulk density and mineral, organic, or carbon content) for each 2-cm section. The mineral, organic, and carbon densities were summed over a depth appropriate to the accretion rate and time period for a particular core and dating method to provide mass-based accumulation rates for each. The ^{137}Cs-based accretion rates are for 44–46 years (from 1964 to the core collection date), and the ^{210}Pb-based accretion rates are for 100 years. Because the OM accumulation and carbon accumulation rates were strongly correlated and very similar, we primarily present results for carbon accumulation rates.

There was at least one successfully dated core at each of the three marsh stations (low, mid, and high) from each site, although some sites had an unequal numbers of dated cores across stations (ranging from one to three cores/station). In order to avoid excessive influence of stations with more cores, we determined averages for sites by first averaging all cores within each of the three marsh stations and then averaging these three values for each site. We conducted the statistical analyses only using data from those cores that were successfully dated (excellent, good, and marginal). We averaged the soil BD and OM contents over 0–20 cm (a typical rooting depth) and over the entire 0- to 50-cm core for the statistical analyses.

We tested for differences in soil OM content, BD, accretion rates, and mass-based accumulation rates across sites and stations using a two-way ANOVA (SYSTAT 2007), and we used Tukey's honestly significant difference test to make
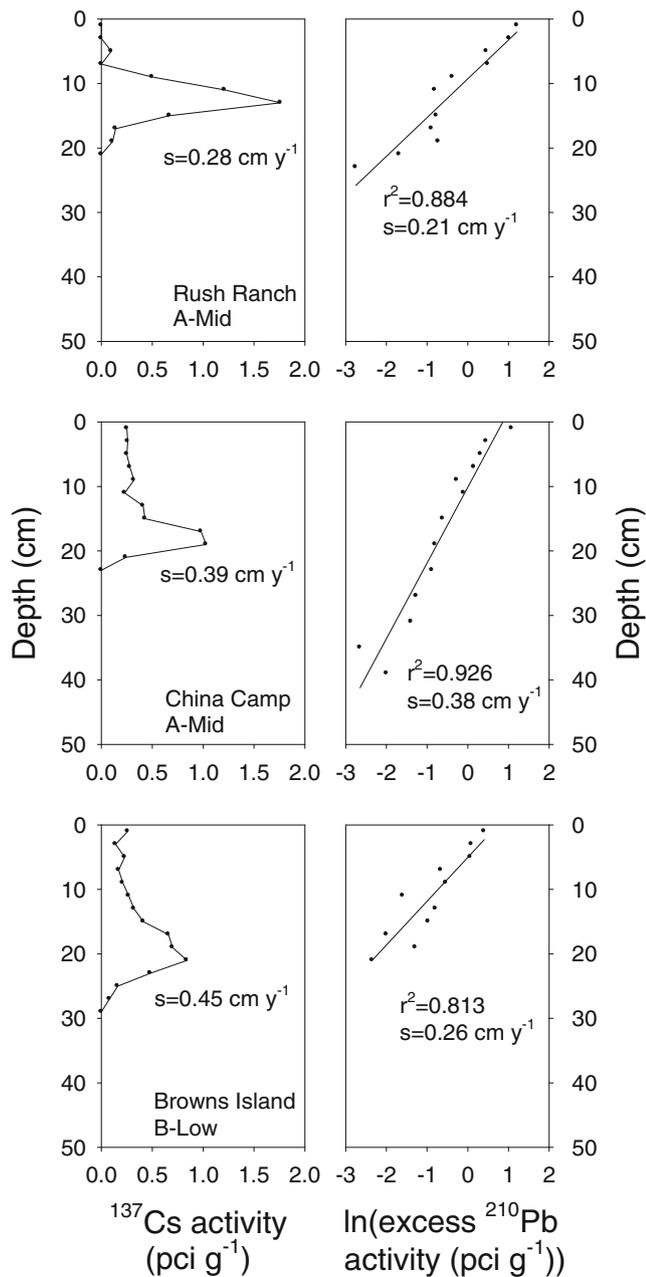
Fig. 2 Sample ^{137}Cs and ^{210}Pb downcore distributions from tidal wetlands in San Francisco Bay estuary. The *top panel* shows data from a core classified as excellent, the *middle panel* from one classified as good, and the *bottom panel* from one classified as marginal

type (e.g., salt vs. brackish marshes), so these results are not included.

We conducted a linear regression analysis to evaluate the relationships between the accretion rates and mass-based accumulations rates of both mineral and organic matter (SYSTAT 2007). We used the OM accumulation rates rather than the carbon accumulation rates for these analyses to facilitate comparisons with previous analyses (Nyman et al. 1993; Callaway et al. 1997; Turner et al. 2000; Neubauer 2008). Regressions were completed for the entire dataset as a whole and also separately for the salt marsh sites and brackish marsh sites.

Results

Soil Characteristics

The sediment BD in the upper 20 cm was higher in low-marsh stations than in mid- or high-marsh stations and was generally highest in salt marshes (Table 2; Appendix 1), although the BD values were only significantly different between BRI and two of the salt marsh sites (see Appendix 2 for the BD values of the entire 0- to 50-cm core). The two restored salt marshes had BD values similar to the natural salt marshes in the top 20 cm, but the BD values from the two restored salt marshes were 30–40 % higher than at the natural salt marshes for samples from 0–50 cm (p=0.017; Appendix 2). The OM content in the upper 20 cm was lowest at the low-marsh stations (Table 2), with most sites showing a gradual increase in OM content across the low, mid, and high stations (Table 2; Appendix 1), and the OM content was higher in both brackish wetlands than in all four salt marsh sites (Table 2; Appendix 1). The differences among salt and brackish sites were similar for OM content from 0–50 cm, although not every salt marsh site was significantly different from the values at RR (Appendices 1 and 3). The OM content in two restored salt marshes were similar to that in the natural marshes (CM=9 %, MM=11 %, and four natural marshes averaged 14 % OM in the upper 20 cm). Most of the cores exhibited the expected trends of increasing BD and decreasing OM with depth, although these patterns were less consistent for the brackish sites (Appendices 4 and 5).

The BD and OM values were strongly and inversely related (Appendix 6; $\log_{10}$ BD=$-0.6922\times\log_{10}$ OM+ 0.4133, r^2=0.80). The OM and carbon content values were also highly correlated, and we used a quadratic regression similar to that developed by Craft et al. (1991) to convert the OM content to carbon content (both expressed as percent of dry weight) and to calculate carbon sequestration: Organic C=(0.001217)×OM^2+(0.3839)×OM (r^2=0.99; Appendix 7). All soils had a fine texture, and there was little sand in most

post-ANOVA comparisons. We tested for the normality and homogeneity of variance using the Kolmogorov–Smirnov test and Levene's test, respectively, and we $\log_{10}$-transformed the accretion and accumulation rates in order to better meet these assumptions. We also evaluated alternative statistical approaches to test for differences between salt and brackish wetlands or between natural and restored salt marshes (e.g., linear mixed models). However, differences were primarily due to individual sites rather than to wetland

cores (<18 % in all cores, except for CI which was 24 % sand) and high levels of clay particles.

Accretion Rates and Mass-Based Accumulation Rates, Including Carbon Sequestration

We successfully dated 34 of the 41 cores collected using the downcore distributions of ^{137}Cs and 28 using the downcore distributions of ^{210}Pb (see Fig. 2 for sample profiles). We classified four of the dated cores as "excellent," 14 as "good," and 16 as "marginal."

The accretion rates measured using the downcore distributions of ^{137}Cs were typically slightly higher than those measured using the downcore distributions of ^{210}Pb for almost all sites and stations (see Appendix 8 for a complete listing of accretion rates and accumulation rates for individual cores). The accretion rates for most cores were between 0.2 and 0.5 cm year^{-1} (26 of 34 cores for ^{137}Cs and 21 of 28 cores for ^{210}Pb, with five of the outliers between 0.1 and 0.2 cm year^{-1}). There were few significant differences in accretion rates across sites. The accretion rate at WT based on ^{137}Cs distributions was the only individual site that was significantly greater than at the others (see Appendix 9 for details), and there were no significant differences among individual sites for the accretion rates based on the ^{210}Pb-determined distribution (Table 3; Appendix 9). Although there were few differences in accretion rates among sites, within a site the accretion rates based on ^{137}Cs distributions at the low-marsh stations were greater than at the mid- or high-marsh stations (Table 3; Appendix 9). There were never significant differences in the accretion rates or mass-based accumulation rates between the mid- and high-marsh stations regardless of the dating method.

The highest rates of mineral accumulation were found at the salt marsh sites, with significant differences across sites regardless of whether the rates were based on the ^{137}Cs or ^{210}Pb distributions (Table 3; Appendix 9). The mineral accumulation rates also were significantly different across stations, with much higher rates at the low-marsh stations (45–450 % greater depending on sites and the dating method) than at mid- or high-marsh stations for both dating methods (Table 3; Appendix 9). These percent differences were much greater than any similar comparisons for vertical accretion rates or carbon accumulation rates.

As with accretion rates, the carbon sequestration rates based on ^{210}Pb dating were slightly lower than those based on ^{137}Cs dating. The average rates for each site that was based on ^{210}Pb dating ranged from 46 to 81 g C m^{-2} year^{-1} at the three salt marshes and from 89 to 117 g C m^{-2} year^{-1} at the two brackish wetlands. The only significant difference in carbon sequestration among individual sites based on ^{137}Cs dating was between BRI and PRM, and there were

no differences in the carbon sequestration rates among sites based on the ^{210}Pb rates (Appendix 9). Within sites, the carbon sequestration rates based on ^{137}Cs dating were higher at the low-marsh stations compared to the mid- and high-marsh stations (Appendix 9), although the magnitude of differences was much smaller than for mineral accumulation rates. In addition, there were no differences across stations for carbon sequestration rates based on ^{210}Pb dating (Appendix 9).

Based on the results from all of the dated 34 cores, there were higher r^2 values for a simple linear regression of the relationship between the OM accumulation and vertical accretion rates than between the mineral matter accumulation and vertical accretion rates for both ^{137}Cs and ^{210}Pb distributions (Fig. 3; Appendix 10). When the dataset was separated into salt and brackish wetland locations, the r^2 values describing the relationship between the OM accumulation and accretion rates were higher, in all cases, than the values describing the relationship between mineral matter accumulation and accretion rates (Appendix 10). The relationships were also consistently stronger for the cores from salt marsh sites compared to those from brackish wetland locations (Appendix 10), with particularly high r^2 values between OM accumulation and the accretion rates in the salt marsh (r^2=0.87 for ^{137}Cs dating; r^2=0.91 for ^{210}Pb dating).

Discussion

Vertical Accretion Rates and Wetland Stability

Based on the large number of stations with vertical accretion rates within the range of 0.2–0.5 cm year^{-1} across the mid- and high-marsh locations, we conclude that tidal wetlands across the Estuary are keeping pace with local SLR, which is very similar to global estimates (Flick et al. 2003; Cayan et al. 2008). The accretion rates from our study are slightly higher than the SLR rates over the last century when this sediment accumulated (IPCC 2007) and are similar to, or slightly higher than, the estimates of SLR rates for the last decade (Beckley et al. 2007). These data, however, primarily indicate that the marsh plain is keeping pace with recent SLR. The historic accretion rates should not be regarded as a maximum or direct predictor of future accretion rates because wetlands may respond to changes in SLR by accumulating additional sediment—or not. Models of wetland sediment dynamics (Callaway et al. 1996b; Temmerman et al. 2003; Rybcyzk and Callaway 2009; Kirwan et al. 2010) calibrated with historic sediment accretion data will be most useful in making realistic predictions of long-term wetland sustainability. The wetlands in South San Francisco Bay have demonstrated rapid rates of short-term accretion during

Table 3 Average rates of vertical accretion, mineral accumulation, and carbon accumulation (±standard error) based on both ^{137}Cs and ^{210}Pb from low-, mid-, and high-marsh stations at all sampling sites ($n=2$ for all stations, except $n=1$ for Browns Island Low (^{137}Cs and ^{210}Pb), Petaluma River Low (^{137}Cs and ^{210}Pb), Petaluma River High (^{210}Pb only), and Whale's Tail Mid and High (^{137}Cs only), and $n=3$ for Coon Island Mid (^{137}Cs and ^{210}Pb) and High (^{137}Cs only))

Site	Low		Mid		High	
	[137]Cs	[210]Pb	[137]Cs	[210]Pb	[137]Cs	[210]Pb
Vertical accretion (cm year^{-1})						
Browns Island	0.45 ($n=1$)	0.26 ($n=1$)	0.40 (0.03)	0.32 (0.12)	0.35 (0.02)	0.27 (0.01)
Rush Ranch	0.35 (0.65)	0.22 (0.15)	0.26 (0.02)	0.21 (0.00)	0.30 (0.04)	0.27 (0.03)
Brackish marsh means	0.40	0.24	0.33	0.27	0.32	0.27
Coon Island	0.68 (0.09)	0.45 (0.13)	0.18 (0.04)	0.15 (0.02)	0.32 (0.07)	0.22 (0.08)
Petaluma River	0.34 ($n=1$)	0.24 ($n=1$)	0.27 (0.07)	0.26 (0.04)	0.25 (0.05)	0.19 ($n=1$)
China Camp	0.63 (0.22)	0.43 (0.09)	0.36 (0.03)	0.35 (0.04)	0.28 (0.00)	0.32 (0.12)
Whale's Tail	0.77 (0.09)		0.57 ($n=1$)		0.55 ($n=1$)	
Salt marsh means	0.61	0.37	0.35	0.25	0.35	0.24
Mineral accumulation (g m^{-2} year^{-1})						
Browns Island	923 ($n=1$)	513 ($n=1$)	367 (54.8)	298 (108.4)	200 (49.3)	273 (96.8)
Rush Ranch	1,215 (208)	772 (144)	305 (34.5)	320 (7.1)	521 (200)	597 (26.9)
Brackish marsh means	1,069	643	336	309	361	435
Coon Island	3,620 (852)	2,325 (85.1)	362 (52.2)	419 (89.6)	949 (188)	650 (189)
Petaluma River	2,584 ($n=1$)	1,810 ($n=1$)	894 (510)	846 (319)	421 (120)	327 ($n=1$)
China Camp	2,678 (370)	1,960 (12.9)	1,480 (388)	1,465 (451)	1,183 (244)	1,243 (276)
Whale's Tail	4,257 (491)		2,210 ($n=1$)		2,129 ($n=1$)	
Salt marsh means	3,467	1,960	1,845	1,465	1,656	1,243
Carbon sequestration (g m^{-2} year^{-1})						
Browns Island	158.4 ($n=1$)	88.9 ($n=1$)	157.9 (7.7)	116.9 (37.0)	151.8 (22.2)	116.5 (3.7)
Rush Ranch	96.7 (21.8)	62.7 (8.3)	117.9 (11.2)	91.3 (1.4)	100.3 (21.2)	87.6 (0.7)
Brackish marsh means	127.5	75.8	137.9	104.1	126.1	102.0
Coon Island	187.5 (44.9)	111.2 (37.8)	52.3 (8.1)	40.4 (4.9)	89.6 (17.5)	52.7 (17.5)
Petaluma River	87.7 ($n=1$)	57.1 ($n=1$)	66.2 (12.6)	57.5 (7.8)	58.7 (5.1)	48.0 ($n=1$)
China Camp	141.9 (44.3)	95.8 (17.9)	79.8 (6.9)	72.0 (6.2)	87.0 (5.0)	90.8 (35.9)
Whale's Tail	146.7 (14.1)		117.9 ($n=1$)		102.5 ($n=1$)	
Salt marsh means	144.3	95.8	98.9	72.0	94.7	90.8

periods of local subsidence, e.g., Alviso Slough (Patrick and DeLaune 1990) and Triangle Marsh (Watson 2004), indicating that these wetlands may be quite resilient to future increases in SLR.

Multiple low-marsh cores dated using ^{137}Cs dating had accretion rates of 0.6 cm year^{-1} or more, as did the relatively young site, WT (Appendix 8), indicating that wetlands may be able to keep pace with rates of SLR that are higher than the present values. However, the long-term reduction in suspended sediment concentrations in the Estuary over the last 30–50 years (Wright and Schoellhamer 2004) may compromise the ability of local wetlands to survive a future acceleration in SLR. This is of particular concern for salt marshes because they have a higher BD than brackish or

freshwater tidal wetlands (Nyman et al. 1990; Table 2). Schoellhamer (2011) recently hypothesized that the Estuary may have shifted from a transport-regulated to a supply-regulated loading of suspended sediment because the easily erodible sediment pool within the Estuary has diminished. This may explain a recent and precipitous drop of 36 % in suspended sediment concentration over the last 10 years. Both the broad long-term declines in suspended sediment concentration as well as this steep, recent drop could have large-scale implications for the future sediment availability and, therefore, the sustainability of wetlands and other intertidal habitats within the Estuary (Ganju and Schoellhamer 2010). This phenomenon of reduced sediment loads is not unique to San Francisco Bay Estuary, but has been found in
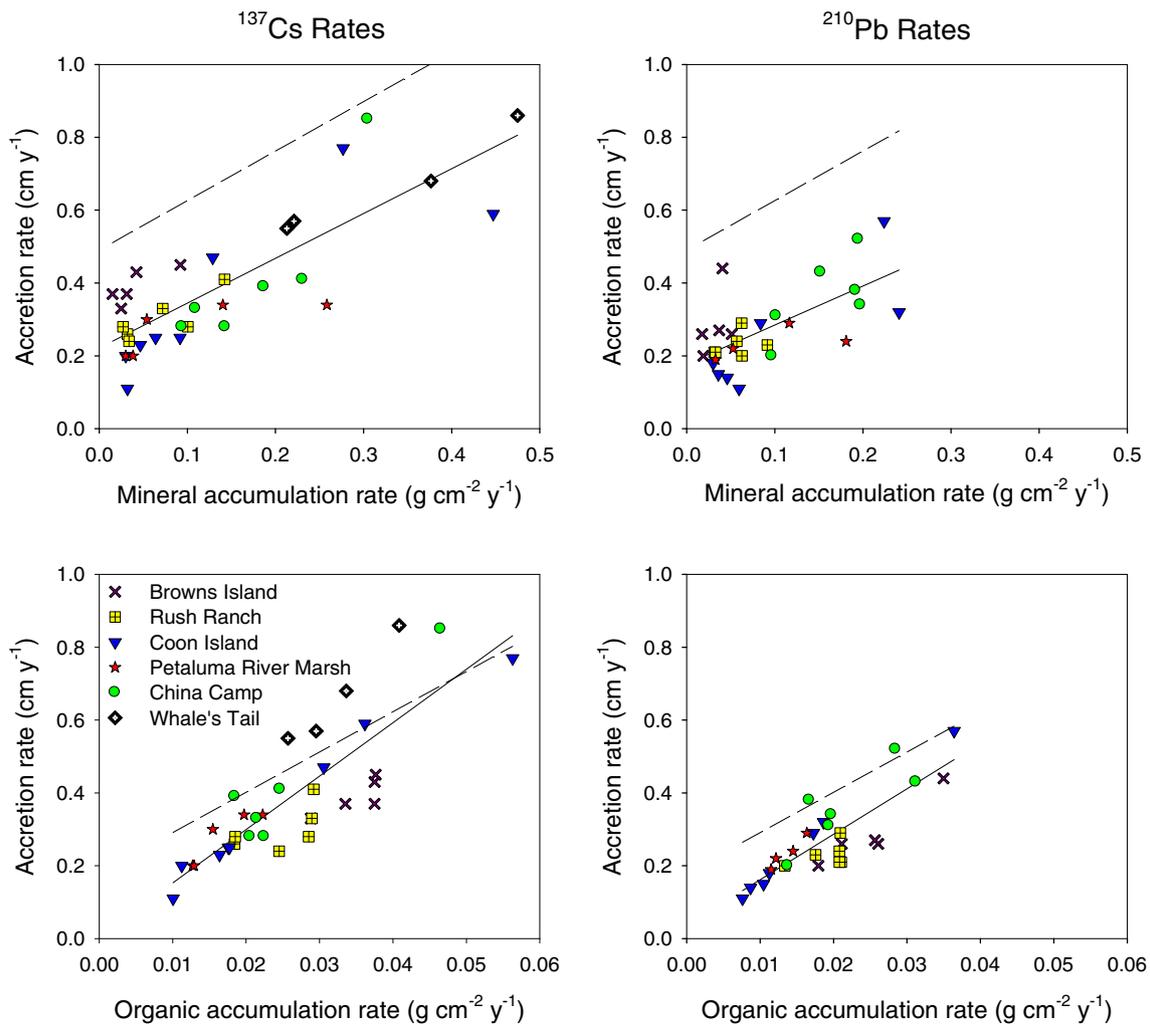
Fig. 3 Relationships between accretion rates and mineral accumulation (*upper graphs*) and carbon accumulation (*lower graphs*) based on both ^{137}Cs and ^{210}Pb downcore distributions. The *solid lines* indicate regression lines for all sites; *dashed lines* indicate regression lines from Turner et al. (2000) based on ^{137}Cs dating. See Appendix 10 for additional information on relationships based on brackish and salt marsh sites

many watersheds because of the proliferation of dams and shifts in upstream land use (Ibàñez et al. 1996; Chen et al. 2001; Horowitz 2010; Tweel and Turner 2012).

There were few estimates of recent wetland sedimentation rates from the San Francisco Bay Estuary before this project, although a number of studies used either ^{14}C for long-term rates (Goman and Wells 2000; Byrne et al. 2001; Watson 2004; Drexler et al. 2009; Drexler 2011) or other methods for very short-term rates, e.g., 1–2 years (Reed 2002; Culberson et al. 2004). We are aware of no previous wetland sediment cores have been dated using ^{210}Pb from the Estuary. In the only published study that used the downcore distributions of ^{137}Cs to date wetlands in the Estuary, Patrick and DeLaune (1990) evaluated three wetlands in South San Francisco Bay and found high rates of accretion (about 4 cm year^{-1}) in one location and rates similar to our findings at the two other locations (0.5 and 0.4 cm year^{-1}).

High rates of short-term sediment accretion have been documented for relatively young sites, including for naturally developing sites at ~1 cm year^{-1} (Watson 2008) and restored sites at 5–10 cm year^{-1} (Williams and Orr 2002). Reed (2002) also measured accretion rates of 1 cm year^{-1} or more in a number of Delta wetlands, although the measurements represent only a 1-year period. It is unlikely that these short-term, very rapid rates of accretion would be sustained over long periods into the future, especially if suspended sediment concentrations continue to decline.

We hypothesized that the rate of sediment accretion would be highest in brackish wetlands because of the proximity of these sites to riverine sediment sources, and we were surprised that there was little variability in the accretion rates across sites. Whale's Tail was the one site with higher accretion rates based on ^{137}Cs dating (this could be due in part to its relatively young age), and there were no

differences among sites based on ^{210}Pb distributions (Table 3). We also hypothesized that, within wetlands, the accretion rates would be higher at sampling stations closer to the bay margin, as has been found in a number of previous studies of salt marshes (Pethick 1981; Hatton et al. 1983; Oenema and DeLaune 1988; French et al. 1995; Callaway et al. 1997). The data from this study provide mixed evidence to support this second hypothesis; the accretion rates from the low-marsh stations that were based on ^{137}Cs dating were significantly greater than at the high-marsh stations, but this was not the case for rates based on ^{210}Pb dating (Table 3). The lack of variation in accretion rates based on ^{210}Pb dating across the wetland may be due to the relatively high elevation and uniform conditions across the marsh plain in California tidal wetlands (Zedler et al. 1999). California tidal wetlands are characterized by a narrow band of *S. foliosa* or other low-marsh vegetation at the bay or tidal creek margins, in contrast to the broader expanse of low-marsh vegetation found in many Gulf of Mexico and East Coast *S. alterniflora*-dominated wetlands.

Accretion Vs. Mass-Based Accumulation: What Is Driving Accretion Rates?

Our data are consistent with previous studies which demonstrate a strong link between OM accumulation and vertical accretion rates (Nyman et al. 1993; Callaway et al. 1997; Turner et al. 2000). Although correlation of these values does not identify cause–effect relationships, these analyses have been widely used to gain insight into the relative importance of organic and mineral matter accumulation, both within local areas and across broader regions. The relationship between OM accumulation and the accretion rates from our data (accretion=14.67×OMaccum.+0.006; Fig. 3) was similar to that from the review of 141 salt marshes on the Atlantic and Gulf Coasts (accretion=11.06×OM$_{accum.}$+0.18; Turner et al. 2000), and the addition of our data to their dataset would strengthen that general relationship. The data points for mineral matter accumulation and accretion in our study were consistently below the relationship found for these variables from Turner et al. (2000), which we think is likely due to the high mineral content and the high BD of soils from San Francisco Bay tidal wetlands in comparison to those on the Gulf and Atlantic Coasts. Beyond these general trends, there were large differences in these relationships for salt and brackish wetlands in the Estuary, probably reflecting differences in sediment dynamics across these different wetland types.

In addition, there was more variation in mineral matter accumulation across sites than for either accretion or carbon accumulation. The coefficient of variation (CV) for the average accretion rates across all sites was 30.4 %, while it was 25.0 % for OM accumulation and 58.7 % for mineral accumulation (based on ^{137}Cs dating). The data for CVs based on ^{210}Pb dating showed a similar trend: 20.0 % (accretion), 25.2 % (OM accumulation), and 51.3 % (mineral accumulation). These differences also support the concept of the importance of OM accumulation as a link to vertical accretion: mineral matter varies widely across these relatively similar sites, but both accretion and carbon accumulation were similar and more constant.

There was strong evidence from both dating methods to support our hypothesis that mineral matter accumulation would be greater at the low-marsh stations, but only mixed support for a similar trend in carbon accumulation (Table 3; Appendix 9). On the larger scale, it was surprising that our brackish wetland sites had the lowest rate of mineral sediment accumulation (Table 3), even though they are closer to the major inputs of mineral sediments (and freshwater) to the estuary, i.e., the Sacramento and San Joaquin Rivers. Our results are contrary to those from a number of other studies that have shown high rates of accretion rates and mineral matter accumulation in brackish and freshwater tidal wetlands (e.g., Craft 2007; Ibàñez et al. 2010). One brackish site, RR, is in the upper part of Suisun Bay and is somewhat removed from direct riverine inputs. We expected that it might have a low rate of mineral accumulation. The other brackish wetland, BRI, is very near the confluence of the Sacramento and San Joaquin Rivers, where we expected suspended sediment loads to be quite high; however, the mineral inputs at this site were 50 % less than even the lowest salt marsh sites (Table 3). Darke and Megonigal (2003) found highly variable rates of short-term sediment accumulation in tidal freshwater wetlands near the Upper Chesapeake and attributed some of this variation to the proximity of the turbidity maximum. Similar spatial variation in the sediment dynamics within the Estuary likely affects mineral inputs to brackish wetlands as much of the sediment that enters the Sacramento–San Joaquin Delta is deposited there and does not move to the estuary (Wright and Schoellhamer 2005). Furthermore, the tidal range at BRI is reduced compared to the other sites (Table 1), and this could limit energy to move sediment from the Estuary into the adjacent wetlands.

Carbon Sequestration Rates and Management Implications

Data from our study substantially expand the spatial and temporal scope of carbon sequestration and accretion data across the Estuary and provide one of the most detailed datasets on recent carbon sequestration for a given estuary—something that is needed to evaluate the broader patterns of spatial and regional variability

in carbon sequestration rates (McLeod et al. 2011). Although our average estimates for tidal salt and brackish wetlands in the Estuary (111 g C m^{-2} year^{-1} when based on ^{137}Cs distributions from six sites; 79.3 g C m^{-2} year^{-1} when based on ^{210}Pb distributions from five sites) are below the global average of 220 g C m^{-2} year^{-1} for tidal saline wetlands summarized by Chmura et al. (2003), they are similar to values published recently for other salt and brackish wetlands. Elsey-Quirk et al. (2011), for example, measured 119 and 154 g C m^{-2} year^{-1} for single cores from *J. roemerianus* and *S. alterniflora* salt marshes in Delaware (based on ^{137}Cs distributions). The average sequestration rates based on ^{137}Cs distributions across three estuaries near Sapelo Island, Georgia, were 32, 21, and 108 g C m^{-2} year^{-1} for the Doboy Sound, Sapelo, and Altamaha Rivers, respectively (Craft 2007), and the average rates were 124, 93, and 40 g C m^{-2} year^{-1} for freshwater, brackish, and salt marshes, respectively, along three nearby estuaries (Ogeechee, Altamaha, and Satilla River estuaries; Loomis and Craft 2010).

The average estimate for tidal saline wetlands of 220 g C m^{-2} year^{-1} from Chmura et al. (2003) is likely high because it includes rates that cover time spans from 1 to 100 years, and it includes a large number of data points from areas with exceptionally high values of carbon sequestration. The rates measured over short time periods are likely to be higher than those measured using longer-term dating methods. This has been shown with vertical accretion rates (Callaway et al. 1996b; Mudd et al. 2009) and is also true for carbon sequestration because of ongoing decomposition within the sediment column. The estimates of sequestration rates based on ^{137}Cs distributions (1963/1964 to present) were consistently higher than the estimates based on ^{210}Pb distributions (100+ years) found in our study. In addition, an inordinate number of the samples incorporated into the overall mean by Chmura et al. (2003) come from areas with relatively high rates of local subsidence (e.g., coastal Louisiana) or high-latitude, low-temperature coastal wetlands (e.g., Bay of Fundy), where sequestration rates are quite high. Some consideration of broader spatial and temporal patterns in carbon sequestration data should be incorporated into future estimates of the global averages. Even if previous global estimates of tidal wetland carbon sequestration area are slightly high, tidal wetlands still are likely to be high in terms of overall rates compared with other ecosystems (e.g., see reviews by Duarte et al. 2005; McLeod et al. 2011).

Our data showed little support for our hypothesis that brackish wetlands would have higher rates of OM and carbon accumulation, especially over the long term.

Browns Island (one of the two brackish sites) was the site with the highest carbon accumulation rates (156 and 117 g C m^{-2} year^{-1} based on ^{137}Cs and ^{210}Pb dating, respectively), and it was statistically different from one of the five salt marshes dated using ^{137}Cs distributions. However, it was not statistically different from the other sites for carbon accumulation rates when based on ^{210}Pb distributions. In addition, the second brackish wetland, RR, had a rate of carbon sequestration that was very similar to the other locations (80.5 vs. 69.5 g C m^{-2} year^{-1} from the three salt marsh sites based on ^{210}Pb dating). In contrast to our findings, Loomis and Craft (2010) found that carbon sequestration rates were more than two times higher, on average, in brackish wetlands than in salt marshes (based on ^{137}Cs dating) from tidal wetlands within three estuaries in Georgia. Więski et al. (2010) also found significantly higher stocks of carbon across the same locations. Craft (2007), in a separate study, measured carbon sequestration in three nearby estuaries and found that carbon sequestration rates in tidal wetlands in freshwater-dominated systems were three to five times higher than the rates in marine-dominated wetlands. With the strong salt stress and gradients that are found in Mediterranean-type tidal wetlands, as well as the large differences in plant productivity within the estuary (Atwater et al. 1979), we expected to see strong differences here; further study is needed to explore potential causes for the observed lack of differences.

Given the need for long-term carbon burial to receive credits within the carbon trading program, we recommend that carbon credit accounting be based on sequestration rates obtained from ^{210}Pb or other long-term dating methods. Sequestration rates that are based on short-term accretion are likely to overestimate carbon sequestration over a century timescale because much of the short-term accumulation will be eventually lost to belowground decomposition (e.g., see model results from Callaway et al. 1996a, b; Mudd et al. 2009). However, substantial carbon sequestration data already exist based on ^{137}Cs dating, and these data could be adjusted to compensate for differences in carbon sequestration over 30- to 40-year vs. 100-year time frames. In the case of this study, the ^{210}Pb-based sequestration rates averaged approximately 73 % of the ^{137}Cs rates (based on the five locations with both ^{137}Cs- and ^{210}Pb-dated cores); other studies have shown similar ratios between ^{137}Cs- and ^{210}Pb-based accretion rates, e.g., 75 % from Turner et al. (2006). In cases where ^{210}Pb data are not available, the accretion rates could be reduced by an average ratio such as this or by a regional or local ratio, if appropriate. While it is impossible to obtain long-term carbon sequestration rates from actual tidal wetland restoration sites, the use of natural tidal wetlands as surrogates is justified and is likely to give the best available estimate of the long-term carbon sequestration

in restored tidal wetlands. The initial rates of vertical accretion will be high at restoration sites (Williams and Orr 2002); however, much of this accretion occurs before plant recruitment, and little carbon accumulation may occur under these circumstances. The rates of carbon accumulation based on ^{137}Cs distributions at WT (approx. 100 years old) were not different from the rates from historic salt marshes in the Estuary. We found slightly lower soil OM contents in restored salt marshes compared with natural salt marshes. However, the differences were not significant, and the OM content increased in the surface layers of both natural and restored wetland soils, supporting the hypothesis that natural sites serve as a good predictor of long-term dynamics in restoration sites.

Given the lack of statistical differences in ^{210}Pb-based carbon sequestration rates, both across sites and within sites, and the need for long-term carbon storage, we recommend that a single value be used in awarding credits for carbon sequestration for tidal wetland restoration projects within the Estuary: 79 g C m^{-2} year^{-1} (based on ^{210}Pb distributions). Our data did indicate slightly higher carbon sequestration rates at the one low-salinity brackish site (BRI). If future data analyses suggest that there are higher rates at similarly low-salinity sites, then the use of different credits for low-salinity brackish wetland restorations may be justified. However, methane emissions are likely to be a much greater issue at low-salinity sites (Poffenbarger et al. 2011), and the effect of methane emissions on net carbon sequestration would have to be incorporated into any consideration of local carbon credits. Similarly, there was a difference in the short-term rates with higher sequestration rates at low-marsh stations; however, the data based on rates over a 100-year timescale using ^{210}Pb distributions do not justify using different rates across a wetland. Furthermore, differences in the sequestration rates between mid- and high-marsh stations were minimal and never significant for either dating method. In most cases, the mid- and high-marsh stations had similar dominant plants, indicating that in other locations, it may be possible to stratify and reduce the sampling effort to determine carbon sequestration based on dominant vegetation.

Appendix 1

Table 4 Summary of ANOVA results for average soil bulk density and organic matter content from 0–20 cm and from 0–50 cm at natural wetland sites (data are *p* values for each ANOVA as well as significant pairwise comparisons)

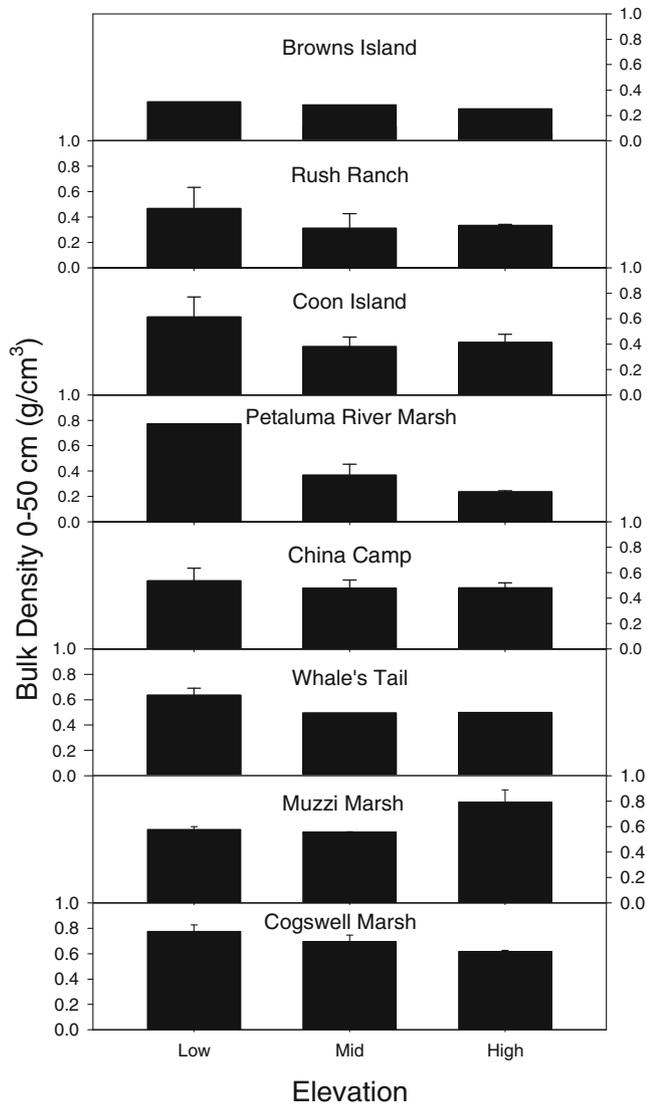|	Bulk density			Organic matter content		
	Site	Station	Interaction	Site	Station	Interaction
0–20 cm (*n*=34)	0.0177 CC,CI>BRI	0.0037 Low>mid and High	0.6643	<0.0001 RR>WT,CC,PRM,CI;BRI>WT,CC,PRM,CI;	0.0002 Mid and high>Low	0.0368
0–50 cm (*n*=34)	0.1740	0.0295 Low>high	0.8400	0.0005 RR>WT,CC,BRI>WT,CC,PRM,CI	0.0386 High>low	0.6767

Appendix 2

Fig. 4 Average sediment bulk density from 0 to 50 cm for cores from low-, mid-, and high-marsh stations at all sampling sites. Sample size for each location is indicated in Table 2. *Error bars*, ±1 SE
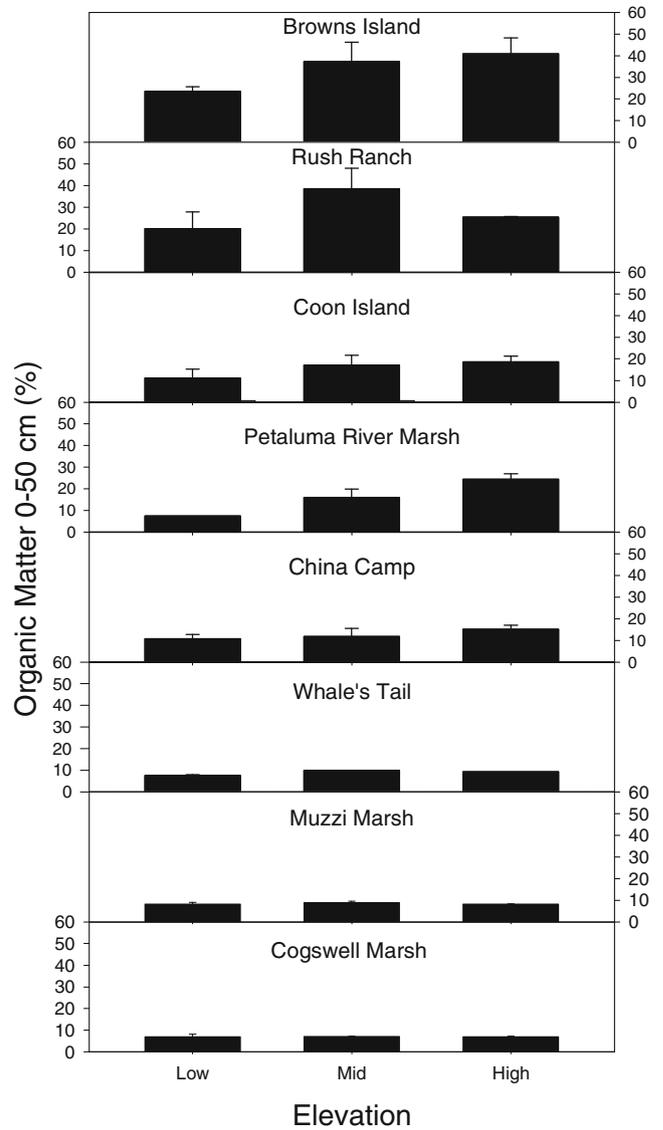
Appendix 3

Fig. 5 Average sediment organic matter content from 0 to 50 cm for cores from low-, mid-, and high-marsh stations at all sampling sites. Sample size for each location is indicated in Table 2. *Error bars*, ±1 SE
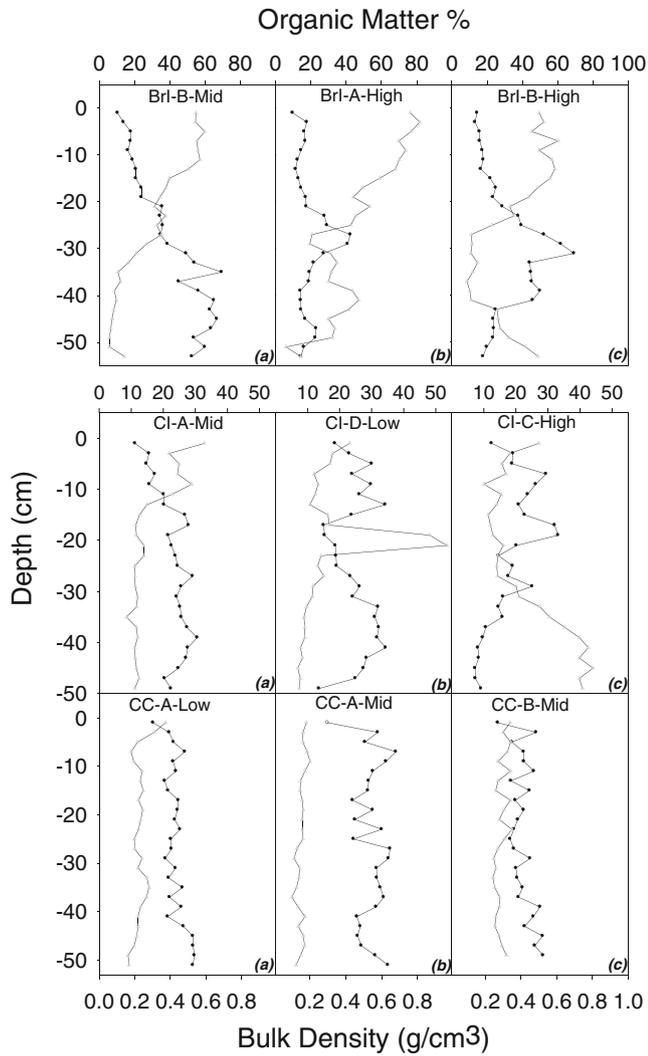
Appendix 4

Fig. 6 Profiles of sediment bulk density and organic matter from 0 to 50 cm for selected cores from natural wetland sites

Appendix 5

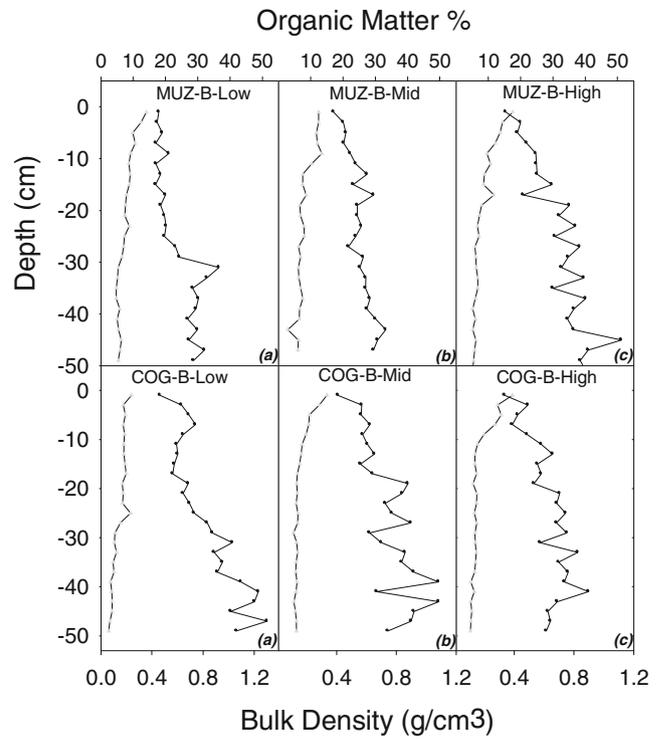**Fig. 7** Profiles of sediment bulk density and organic matter from 0 to 50 cm for selected cores from restored wetland sites

Appendix 6

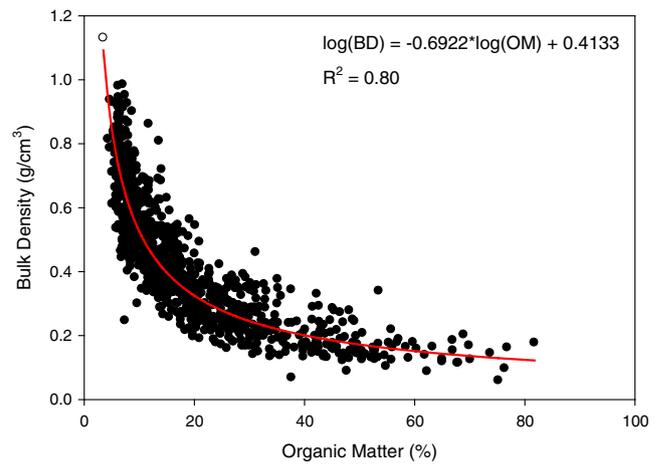$$\log(BD) = -0.6922*\log(OM) + 0.4133$$
$$R^2 = 0.80$$

Fig. 8 Relationship between sediment organic matter content and bulk density based on sections from all cores
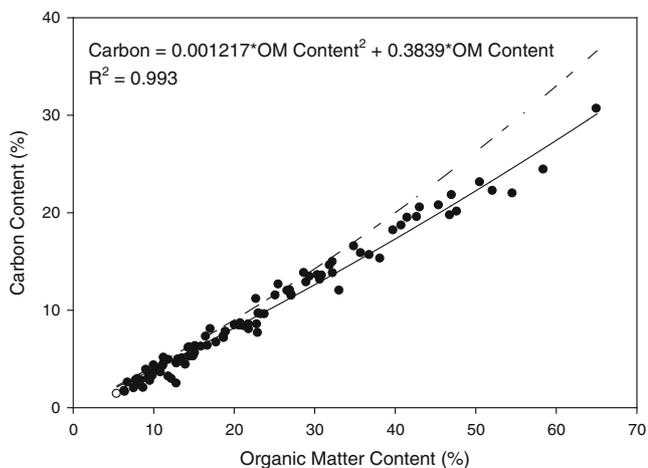
Appendix 7

Fig. 9 Relationship between sediment organic matter content and carbon content based on data from 97 core sections. The *solid line* indicates the best fit regression for these data, and the *dashed line* indicates the regression equation from Craft et al. ([1991](#))

Appendix 8

Table 5 Table of accretion rates, mineral matter, and carbon accumulation rates for all individual cores, with dating using downcore distributions of both ^{137}Cs and ^{210}Pb

Site	Station	Transect	Rates based on ^{137}Cs			Rates based on ^{210}Pb		
			Accretion (cm year^{-1})	Mineral accum. (g m^{-2} year^{-1})	Carbon accum. (g m^{-2} year^{-1})	Accretion (cm year^{-1})	Mineral accum. (g m^{-2} year^{-1})	Carbon accum. (g m^{-2} year^{-1})
Whale's Tail	Low	A	0.86	4,747.3	160.8	–	–	–
Whale's Tail	Low	B	0.68	3,765.7	132.6	–	–	–
Whale's Tail	Mid	A	0.57	2,210.5	117.9	–	–	–
Whale's Tail	High	A	0.55	2,129.3	102.5	–	–	–
China Camp	Low	A	0.85	3,047.6	186.2	0.52	1,947.1	113.8
China Camp	Low	B	0.41	2,308.2	97.5	0.34	1,972.9	77.9
China Camp	Mid	A	0.39	1,867.3	72.9	0.38	1,915.8	65.8
China Camp	Mid	B	0.33	1,092.0	86.7	0.31	1,014.3	78.2
China Camp	High	A	0.28	938.9	92.0	0.43	1,518.8	126.7
China Camp	High	B	0.28	1,427.0	82.1	0.20	966.5	54.9
Petaluma River	Low	A	0.34	2,584.5	87.7	0.24	1,810.1	57.1
Petaluma River	Mid	A	0.2	383.7	53.6	0.22	527.8	49.7
Petaluma River	Mid	B	0.34	1,403.7	78.8	0.29	1,165.0	65.4
Petaluma River	High	A	0.2	301.4	53.5	0.19	327.0	48.0
Petaluma River	High	B	0.3	541.3	63.8	–		
Coon Island	Low	A	0.59	4,472.9	142.6	0.32	2,410.2	73.4
Coon Island	Low	D	0.77	2,768.0	232.4	0.57	2,240.1	149.0
Coon Island	Mid	A	0.23	466.2	68.2	0.15	361.6	43.2

Table 5 (continued)

Site	Station	Transect	Rates based on [137]Cs			Rates based on [210]Pb		
			Accretion (cm year⁻¹)	Mineral accum. (g m⁻² year⁻¹)	Carbon accum. (g m⁻² year⁻¹)	Accretion (cm year⁻¹)	Mineral accum. (g m⁻² year⁻¹)	Carbon accum. (g m⁻² year⁻¹)
Coon Island	Mid	B	0.11	320.1	41.7	0.11	595.0	30.9
Coon Island	Mid	C	0.2	301.1	47.0	0.18	301.4	47.1
Coon Island	High	A	0.25	640.2	72.3	0.29	839.0	70.2
Coon Island	High	B	0.47	1,288.7	124.7	–	–	–
Coon Island	High	C	0.25	918.3	71.9	0.14	460.3	35.3
Rush Ranch	Low	A	0.41	1,423.1	118.4	0.23	916.3	71.0
Rush Ranch	Low	B	0.28	1,006.4	74.9	0.20	628.2	54.4
Rush Ranch	Mid	A	0.28	270.9	129.1	0.21	312.7	92.7
Rush Ranch	Mid	B	0.24	340.0	106.7	0.21	326.8	90.0
Rush Ranch	High	A	0.26	321.1	79.1	0.29	623.8	88.2
Rush Ranch	High	B	0.33	720.3	121.5	0.24	569.9	86.9
Browns Island	Low	B	0.45	922.9	158.4	0.26	513.3	88.9
Browns Island	Mid	A	0.43	421.9	165.5	0.44	406.4	153.9
Browns Island	Mid	B	0.37	312.3	150.2	0.2	189.7	79.9
Browns Island	High	A	0.37	151.1	174.1	0.26	176.1	120.2
Browns Island	High	B	0.33	249.6	129.6	0.27	369.7	112.9

Appendix 9

Table 6 Summary of ANOVA results for accretion rates and mass-based accumulation rates (data are p values for each ANOVA as well as significant pairwise comparisons)

	Accretion rates			Mineral accumulation			Carbon accumulation		
	Site	Station	Interact.	Site	Station	Interact.	Site	Station	Interact.
[137]Cs (n=34)	0.0104 WT> PRM, CI,RR	0.0036 Low>mid and high	0.1681	<0.0001 WT>PRM,CI,RR,BRI;CC> RR,BRI; CI>BRI	<0.0001 Low> mid and high	0.0556	0.0059 BRI>PRM,CI	0.0150 Low> mid and high	0.0683
[210]Pb (n=28)	0.1662	0.3110	0.2431	0.0002 CC>RR,BRI;PRM>BRI;CI> BRI	0.0003 Low> mid and high	0.0944	0.0437 None<0.05	0.7160	0.2089

Appendix 10

Table 7 Regression results (r^2 values) for comparisons between accretion rates and mineral and organic matter accumulation rates based on both ^{137}Cs and ^{210}Pb data (from all sites and separately from salt marsh and tidal brackish wetland sites)

	Mineral accumulation vs. accretion		Organic accumulation vs. accretion	
	[137]Cs	[210]Pb	[137]Cs	[210]Pb
All sites	0.682 (n=34)	0.443 (n=28)	0.700 (n=34)	0.702 (n=28)
Salt	0.758 (n=23)	0.618 (n=17)	0.872 (n=23)	0.914 (n=17)
Brackish	0.124 (n=11)	<0.001 (n=11)	0.681 (n=11)	0.768 (n=11)

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