UC Davis
San Francisco Estuary and Watershed Science

Title
Will Restored Tidal Marshes Be Sustainable?

Permalink
https://escholarship.org/uc/item/8hj3d20t

Journal
San Francisco Estuary and Watershed Science, 1(1)

Authors
Orr, Michelle
Crooks, Stephen
Williams, Philip B.

Publication Date
2003

DOI
10.15447/sfews.2003v1iss1art5

Copyright Information

Peer reviewed

Introduction

The large scale historic loss of tidal marsh habitat throughout the San Francisco Estuary has prompted major initiatives aimed at restoring extensive areas of diked former tidal marsh (Goals Project 1999; CALFED 2000; Steere and Schaefer 2001). State, federal and local agencies have proposed restoration of between 13,000 and 26,000 ha of tidal marsh in San Francisco Bay (Goals Project 1999; Steere and Schaefer 2001) and 10,000 to 20,000 ha in the Delta (Hastings and Castleberry 2003). Because of the large investment in tidal marsh restoration, the question of whether tidal marsh habitats will persist once restored needs to be addressed.

Tidal marshes are generally resilient but ephemeral landforms (Pethick and Crooks 2000). The natural marshes of the San Francisco Estuary have historically been resilient features of the landscape, behaving dynamically to accommodate slow changes in sea-level and sediment supply (Atwater and others 1979). However just as the estuary's landscape has been transformed in the last 150 years, so have certain key physical processes that affect the character and persistence of vegetated tidal marshes. Looking to the future, rates of sea level rise are expected to increase (IPCC 2001), suspended sediment concentrations decrease (Beeman and Krone 1992; Oltmann and others 1999; Wright and Schoellhamer 2003), and salinity gradients move upstream (Williams 1989). Other processes, such as tidal range, local subsidence, and wind wave climate could also change. Moreover, differences in form and function between restored and natural marshes must be considered. Marshes are often restored on subsided lands, requiring estuarine deposition and/or fill to provide the conditions necessary for marsh re-establishment (Williams and Orr 2002). Restored marshes are often lower in the tidal frame (the vertical range between high and low tides) and subject to higher rates of subsurface consolidation than natural marshes. We use a 100 year planning horizon for this article, with restoration planned within the next 20 to 30 years (CALFED 2000; Steere and Schaefer 2001).

San Francisco Estuary Setting and Changes in Physical Processes

Site Setting

The San Francisco Estuary, in Central California, is one of the largest estuaries in the United States and has undergone a sustained period of anthropogenic modification over the past 150 years (Brown 2003; TBI 1998; Goals Project 1999). The estuary consists of a series of interconnected marine bays which to the northeast give way to the Sacramento-San Joaquin River Delta (Figures 1 and 2). Tides are mixed semi-diurnal with diurnal ranges of approximately 2 m at the Golden Gate and San Pablo Bay, decreasing to 1 to 1.4 m in the Delta. The Delta is a predominantly freshwater tidal system, though salinities can reach 2 ppt in the west Delta during the summer (Conomos and others 1985). San Pablo Bay is a more saline system, with salinities varying seasonally between approximately 4 and 30 ppt (sea water is approximately 32 ppt).
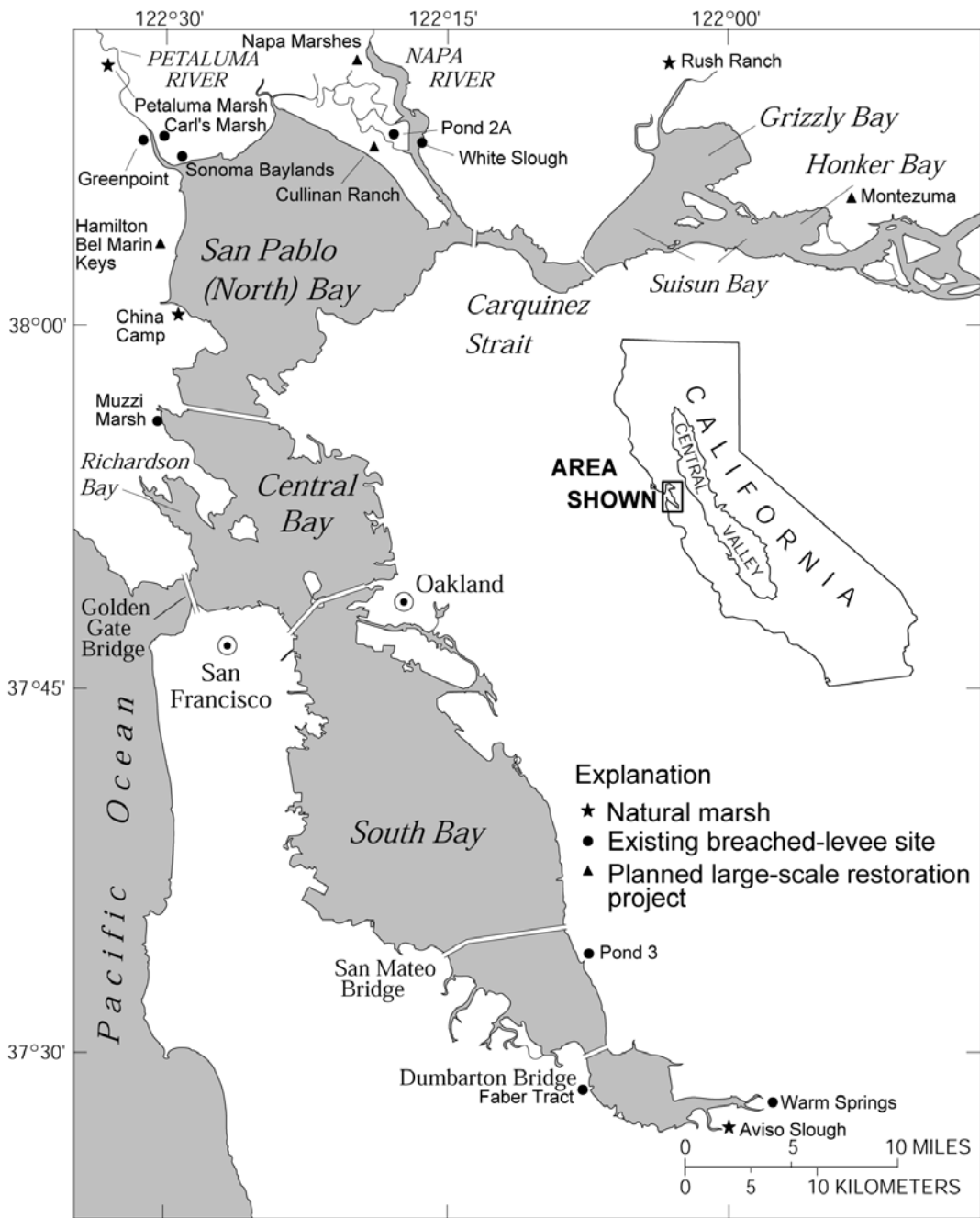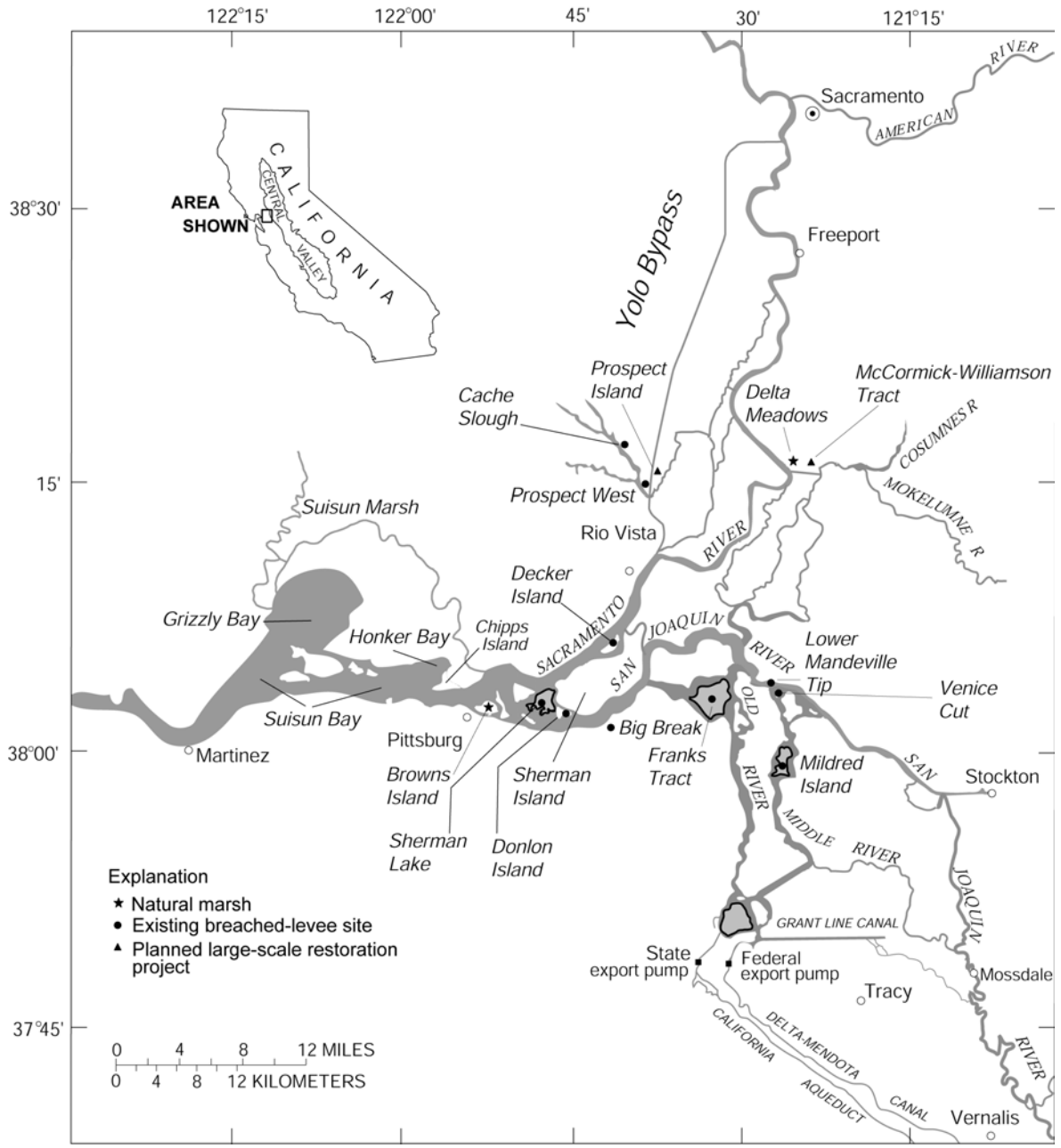
Figure 1 Location map for San Francisco Bay

Figure 2 Location map for Sacramento-San Joaquin Delta. Major flooded islands are indicated by the lighter gray shading.

Of the original 80,000 ha of tidal marsh present in San Francisco Bay in the mid-1800s, less than 10% exist today in pristine condition (Goals Project 1999). An additional 10,000 ha of marsh have been created through levee abandonment, restoration, or evolution from mudflats. In the Delta, ninety-five percent of the original 130,000 ha of tidal wetlands have been lost to reclamation (Atwater and Belknap 1980), and as yet few sites have been successfully restored. In this article we will draw lessons from the northern reaches of the Estuary, concentrating in particular on the tidal salt marshes of San Pablo Bay (see Figure 1) and the tidal fresh water marshes of the Sacramento-San Joaquin River Delta (see Figure 2).

The Sacramento and San Joaquin rivers are the most important source of new sediment to the Estuary (Beeman and Krone 1992). Material eroded from the watersheds of the Central Valley rivers is transported to the Estuary during large winter floods. Coarser sediments are deposited upstream on the floodplains, leaving the fine clays and silts to settle out in the shallows of Suisun Bay, San Pablo Bay, and occasionally, the South Bay (Buchanan and Ruhl 2000). The transition from fresh to brackish waters causes the cohesive clays and silts to flocculate and settle from the water column more quickly (Allen 2000). Sediment supply to the Bay is augmented by smaller sediment inflows from local tributaries. In the months following the winter flood deposits, wind waves resuspend muds and silts from the shallows, and tidal currents redistribute them to all parts of the Estuary (Krone 1979; Ruhl and Schoellhamer 2000). The mudflats and shallows act as large sediment reservoirs, where seasonally deposited sediments can be reworked by wave action, then redistributed by tidal currents for eventual deposition on the marsh plains or in deep subtidal channels, or for discharge to the ocean.

Evolution

The Estuary has gradually formed over the last 10,000 years as a rising sea flooded former inland valleys at the mouth of the Sacramento and San Joaquin rivers. The initial rapid rise in sea level (approximately 15 to 20 mm yr^{-1}) probably allowed only a thin, discontinuous fringe of salt marsh to develop along the expanding bay shoreline (Atwater and others 1979). These marshes accreted vertically and retreated landward as marine waters transgressed—a geomorphic process known as coastal or estuarine "roll-over" (Allen 1990c; Pethick 1996). By approximately 6,000 years ago sea level rise had slowed considerably and by 5000 years ago stabilized at about its current rate of 1 to 2 mm yr^{-1} (Atwater and others 1979; Bard 1998). In the estuary's marine embayments, the high availability of reworked sediments and gentle rates of sea-level rise fostered conditions for the accumulation of minerogenic marshes which formed extensive marsh plains capable of accreting with rising sea-level (Atwater and others 1979). Nearby, in the freshwater Delta, sedimentation was very different. Here minerogenic sedimentation was primarily in response to flood flows of the Sacramento and San Joaquin rivers and confined to the margins of their distributary channels. Distal from these internal deltas organic-rich marshes began to accumulate (Atwater 1982).

Changes in Physical Processes

As well as directly impacting marshes through reclamation, human activity has altered the processes that sustained historic marshes. These processes include locally changing rates of relative rise in sea-level, changes in sediment supply, and changes in hydrology which are affecting the salinity regime.

Globally, the rate of eustatic sea level rise is expected to continue along its 20th century global warming induced accelerated trajectory, possibly attaining an average rate of about 3 mm yr^{-1} over the next 50 years (2000 to 2050), rising to an average rate of about 5 mm yr^{-1} over the following 50 years (2050 to 2100) (IPCC 2001). There are some uncertainties in terms of the rate of the rise depending upon the model used and the emissions scenarios analyzed, but the total amount of eustatic sea-level rise is expected to lie between 0.09 and 0.85 meters projected for 2000-2100 with a central value of 0.4 m. This central value equates to a rate of rise some 2 to 4 times that during the 20th century.

The local rate of relative sea-level rise is a combination of the eustatic component with any large-scale tectonic and isostatic (crustal rebound) components and local-scale effects such as groundwater extractions. The tectonics of the Bay Area are complex. Little is known currently of the local rates of tectonic vertical movement though one study has suggested crustal subsidence of 2.7 mm yr^{-1} for the west Delta (Pittsburg) and 0 mm yr^{-1} for San Pablo Bay (at Sonoma Creek) (BCDC 1987). Locally, dewatering and enhanced aerobic decomposition of diked former marshes has caused land surfaces to subside to near or below mean sea level (CDWR 1995; WCC and others 1998). Subsidence is continuing to occur in leveed areas that are dry at least part of the year (Ikehara 1994). Additionally, in certain areas, groundwater extraction and gas field pumping may well be contributing to local and regional subsidence (CDWR 1995).

Within the northern reach of the Estuary sediment dynamics—sediment delivery by the rivers, sediment storage on marsh plains or in deep water, and sediment movement within the estuary—have changed significantly since the mid-1800s. Gilbert (1917) described many of the early land use changes and their affect on the sediment budget. Before colonization, the watershed was largely undisturbed and floodplains intact. This meant that sediment discharge to the Estuary was relatively low. By 1880, the combination of overgrazing, deforestation, floodplain reclamation and, most importantly, hydraulic mining caused huge increases in the amounts of sediment delivered to the Estuary. The balance changed again in the 20th century with the shut down of hydraulic mining, the construction of dams that captured sediment and reduced flood flows, and the elimination of natural marshes that acted as sediment sinks. As of 1990 the amount of sediment supplied to the Estuary had fallen to approximately 30% of its late 1800s peak (Gilbert 1917; Beeman and Krone 1992). Over the next century, these decreases are expected to continue (Beeman and Krone 1992; Oltmann and others 1999; Wright and Schoellhamer 2003). Sediment supply still appears to be higher than under pre-mining

natural conditions, presumably because sediment from previous hydraulic mining and from channel incision downstream of dams (Wright and Schoellhamer 2003) is still being washed into the Estuary.

The impacts of this "pulse" in sedimentation are recognized within the landscape. Studies by Jaffe and others (1998) and Cappiella and others (1999) document mudflat deposition in San Pablo and Suisun Bays to have at first increased in response to the large pulse of hydraulic mining sediments, then decreased as the sediment supply diminished, resulting in current mudflat erosion. These studies are consistent with other studies showing a long-term trend of mudflat lowering and shoreline erosion in Corte Madera and Richardson bays (see Figure 1) in the north Central Bay (PWA 1993).

Suspended sediments in San Pablo Bay typically average between 100 and 150 mg/L, with a wide range of event-driven variations (USGS summaries of suspended sediment concentration data, e.g., Buchanan and Schoellhamer 1998, 1999; Buchanan and Ruhl 2000, 2001; Warner 2000; USGS unpublished data for Hamilton and Sonoma baylands; PWA 2002a). Since suspended sediment concentrations in the water column are sensitive to the extent and elevation of mudflats, lowering of the mudflats will cause suspended sediment concentrations to decrease over time.

Freshwater diversions have significantly reduced spring flows, resulting in the conversion of freshwater tidal wetlands to brackish and salt marshes (Williams 1989). The salinity regime of the Estuary is expected to shift further as sea level rises (Goals Report 1999).

Concept of Geomorphic Sustainability

What does it mean to have a sustainable tidal marsh? In its widest sense, sustainability and sustainable development are defined in terms of protecting the world's resources so that current and future generations enjoy an equitable quality of life (WCED 1987). The basis of this approach is that the foundations of world economies, and societal well being, are dependant upon the health and efficiency of environmental systems and the functions that they perform, e.g., resource production, climate stabilization, waste assimilation (e.g., Pearce and Warford 1993; Turner 1993; Crooks and Turner 1999).

Ecosystems at all geographical scales tend to exist in transitory states and the concept of sustainability is to be distinguished from permanence (e.g., Beeby 1993; Sutherland 1998). In dynamic systems such as coasts or estuaries, sustainable management of resources should not be seen only as preservation of a given landscape but instead as an adaptive process of land-use modification to accommodate long-term changing requirements (Carter 1988). In this way positive aspects of system resilience are reinforced, allowing incremental changes to take place rather than sudden, possibly catastrophic, adjustment from one equilibrium form to another (Knighton 1998; Pethick

and Crooks 2000). These incremental changes may be quite large in response to large-scale perturbation (such as sudden tectonic events) but system resilience will be maintained as long as an equilibrium form is reestablished in a time less than that of the disturbance event return interval (Knighton 1998; Pethick and Crooks 2000).

The importance of replacing those critical functions that have or are being lost is also clear. In wetlands restoration there is a consensus that we need to ensure that a suitable template exists for wetlands to develop with the full range of morphological characteristics that, in turn, support a diverse and rich ecology (SWS 2000; NRC 2001; Zedler 2001; Simenstad and Warren 2002). The complex nature of coastal systems means that it is not always a simple matter to restore intertidal wetlands at a site at which they historically occupied (Atkinson and others 2001; Zedler 2001; Williams and Orr 2002). Over time the shape of the estuary evolves, and to plan for sustainable restoration requires understanding of the dynamic equilibrium processes, both at landscape and local scales, that define the form and function of coastal, estuarine and wetland landform.

Tidal Marshes as Resilient Evolving Systems

Geomorphic Response of Estuarine Systems to Relative Sea-Level Rise and Change in Sediment Delivery

Estuaries are highly dynamic environments and under natural conditions their form reflects an optimal dynamic response to an equilibrium that exists between the morphology (shape) of the estuary and the "forcing factors" or processes of waves, tidal currents and fluvial discharge. In essence, the most fundamental geomorphic function of the estuary is to attenuate wave energy of all periods, from wind waves to tides. Estuaries do this by transferring sediments from high to low energy areas in ways that tend to more evenly attenuate wave energy (Pethick 1996). There is therefore a strong morphodynamic response as wave energy shapes the morphology, which feeds back to alter the local energy climate. The form and function of the coast therefore depends upon the nature of impinging energy climate and the nature and availability of sediment to attenuate that energy (Pethick and Crooks 2000).

Changes in geomorphic factors such as sea-level or sediment supply result in a spatial redistribution of coastal landforms. Increasing water depths at the shore, for example, result in wave and tidal energy being propagated further within the estuary (Carter 1988). In response, estuaries migrate landwards and upwards to maintain their position within the energy gradient. In this way as coastal energy levels increase, salt marshes migrate landwards to a lower energy location and may be eroded and replaced by mudflat and subtidal systems (Pethick 1996). Large-scale coastal landforms, such as estuaries, respond to environmental change in the same way as do small-scale landforms, but the adjustment time is longer. Given adequate space, landward migration might not affect the wider estuarine system directly other than by internal translocation of smaller landforms.

However, with the human development of many coastal floodplains, the space for relocation of the estuary has been lost.

Coastal development means that for some restored sites, there is no relocation space. Generally, perimeter levees border restored bayfront marshes so there is no opportunity for marsh transgression over gently sloped adjacent lands to compensate for marshes eroding at the bayfront edge. Thus, for restored marshes, sufficient sediments and biomass accretion must be available to maintain these marshes in light of sea level rise over the long-term. Additionally, ensuring that a suitable template is in place for functional restored creek systems is critical in sustaining a functional wetland ecosystem.

Dynamic Equilibrium of the Marshplain and Channels

Tidal marshes are sustained vertically by plant growth and sedimentation that tends to maintain marshplain elevations within a narrow band of the high intertidal range even as sea level rises. The long-term sustainability of a marsh at any given site depends on the relative rates of sea level rise, plant productivity, sediment deposition, and subsidence, which interact to maintain the marsh elevation in a dynamic equilibrium with tide levels (Figure 3) (Mitsch and Gosselink 1986; French 1993; Callaway and others 1996; Allen 2000; Morris and others 2002).

Marsh elevation relative to the tides is somewhat self-regulating, since a lowering of relative marsh elevation results in more frequent inundation, which in turn increases sedimentation and decreases further subsidence by slowing the oxidation of soil organic matter (Mitsch and Gosselink 1986). Conversely, the accretionary processes of plant production and sedimentation slow as the marshplain elevation rises relative to the tides (Krone 1985; Morris and others 2002). Changes in the factors that affect marsh maintenance can cause a shift in relative marshplain elevation towards a new dynamic equilibrium (Allen 2000). In response to accelerated sea level rise, for example, high marsh may gradually transition to low marsh with an accompanying shift in marsh vegetation. When the rise in sea level outpaces a marsh's maximum sustainable rate of accretion, the marsh gradually submerges until it can no longer sustain vegetation and becomes mudflat, a fate referred to as "ecological drowning" (French 1993).

Salt marsh vertical accretion under varying conditions of sediment supply and sea-level rise has been quantitatively modeled in a number of studies of minerogenic (sediment-dominated) salt marshes from a range of tidal coastlines (meso-macro) (Randerson 1979; Krone 1987; Allen 1990a, 1990b,1995, 1997; French 1991, 1993; Callaway and others 1996; Temmerman and others 2003) and, less commonly, from studies of organogenic (vegetation-dominated) microtidal marshes (Chmura and others 1992; Morris and others 2002). By extending these models it is possible to look at the effects of increased rates of sea-level rise (and/or sediment influx) on the marsh existence (e.g., French 1993).

The basis of these models is essentially a mass balance calculation reflecting the change in surface elevation over time, relative to the moving tidal frame in response to mineral and organic sedimentation rates, and the rate of sea-level rise. By specifying the sediment supply, tidal and sea-level regimes, and initial elevation, the mass balance equation may be integrated over time to reflect the growth of a young marsh from intertidal flat or the response of a well established wetland to sea-level fluctuations. There are still several gaps in the models that have yet to be accounted for. These include spatial variability in sedimentation, long-term consolidation of sediments, changes in organic sedimentation (root remains etc.) with relative marsh elevation, and the contribution of vegetation to increased sediment trapping efficiency (Eisma and Dijkema 1997; Allen 2000; Morris and others 2002; Reed 2002a). Nevertheless modeling provides a way to integrate the processes that affect marsh accretion and to predict marsh response to future scenarios.
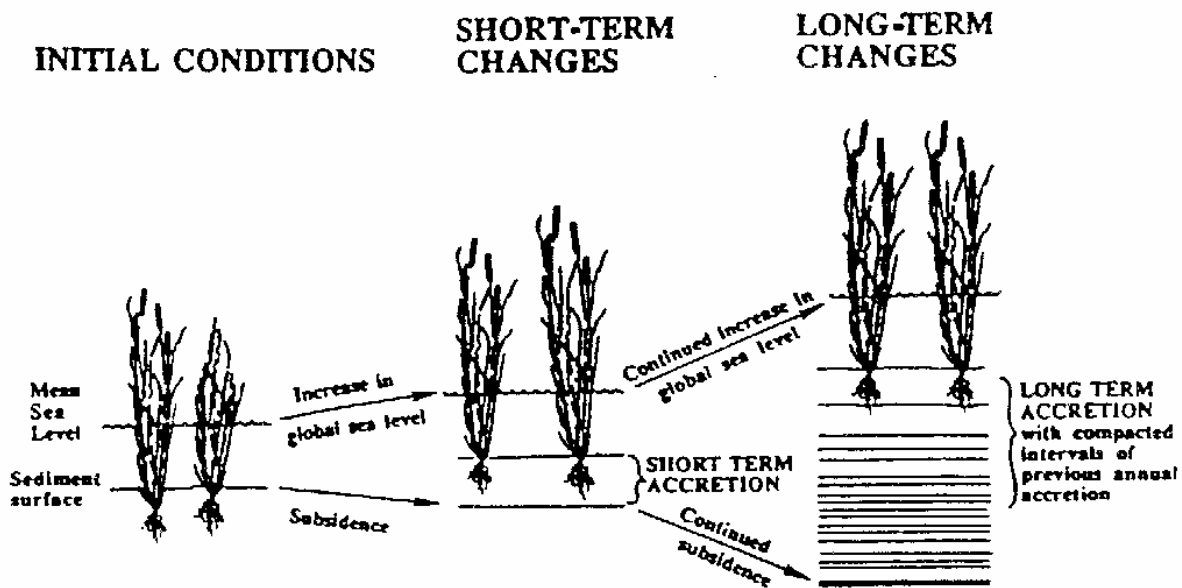

Figure 3 Conceptual model of accretion and autocompaction of coastal marsh sediments and relative elevation over time. From Callaway and others (1996).

Across the marsh surface, channel networks are in a dynamic equilibrium with tidal flows (Steel and Pye 1999; Zeff 1999; Allen 2000). Firstly, studies of hydraulic geometry indicate that bankfull and peak tidal velocities increase only gradually if at all in the downstream directions (Bayliss-Smith and others 1979; French and Stoddart 1992). Channel dimensions increase downstream, indicating an equitable distribution of time-averaged bed shear stress throughout the channel system. Secondly, salt marsh sloughs behave hydraulically much like tidal inlets (O'Brien 1931; Diekmann and others 1988; Hume and Henderdorf 1993) and larger estuarine channels (Pethick 1996; Lowe 2000; Pethick and Crooks 2001) in that there is a strong relationship between the cross-

sectional area of the channel and the volume of the flow passing through that channel over a tidal cycle (known as the tidal prism).

These equilibrium relationships imply that, as the conditions that determine the flow of water on to and off the marsh change with time, the morphology of the sloughs adapts to accommodate those changes. In this way as a tidal marsh accretes from a young low-elevation immature marsh towards high-elevation maturity, the slough network adopted from the precursor mudflat first expands into the marsh and then fills in (Steel and Pye 1999). This slough evolution reflects the changing need for water and sediment to be transferred through a channel system relative to over-marsh flow and the incompetence of weak currents to clear channels of sediment in high back marsh areas.

Characteristics of Restored Marshes

The preponderance of former marshes available for restoration in the estuary are diked and subsided, with many sites subsided below elevations at which emergent vegetation will colonize (Goals Project 1999; Williams and Orr 2002). Restoration must therefore compensate for this legacy of subsidence by building mudflats high enough in the tidal frame for vegetation to colonize, generally relying on natural vegetative recruitment. Because restored marshes may be lower in the tidal frame than natural marshes and subject to additional autocompaction, they can respond differently than natural marshes to the same ongoing physical and vegetative processes.

Restoration Approaches in the Bay

The restoration approach commonly used in San Pablo Bay and other regions of San Francisco Bay is to strategically breach perimeter dikes enabling estuarine sedimentation to build mudflats that accrete to tidal elevations at which marsh vegetation can colonize (Figure 4) (Siegel 1998; Goals Project 1999; Williams and Faber 2001; Williams and Orr 2002). Channels form through differential deposition and tidal scour as the mudflats accrete, and are further defined as vegetation becomes established (French 1993; Beeftink and Rozema 1988; French and Stoddart 1992). Sometimes imported dredged material fill is used to raise site elevations prior to breaching, speeding the restoration process (e.g., PWA and others 1999). However, because overfilling can inhibit channel formation (Williams and Orr 2002), even marshes restored using fill are designed to be lower in the tidal frame than natural marshes. Creating marshes low in the tidal frame, but at elevations at which halophytes can colonize, is necessary to provide the right conditions for slough networks to develop naturally, thus maximizing the ecological benefits of the restoration. Given the reliance on natural processes to restore marsh functions in San Pablo Bay, restoration is a process that occurs gradually, over a time frame of decades (Williams and Orr 2002).

In this article, a restored marsh is defined as one that has completed the transition from mudflat to vegetated plain. For San Pablo Bay, the geomorphic transition between

mudflat and low marsh lies at or about the mean tide level (1 m below mean higher high water, MHHW). The elevation at which this low marsh gives way to high marshplain occurs some 0.8 m above mean tide level (at or about 0.2 m below MHHW) (Atwater and Hedel 1976; Goals Project 1999). These transitions are for established vegetation, though there are some data (Siegel 1998; Williams and Orr 2002; authors' unpublished data) showing that mudflats must reach a higher elevation, near 0.4 m above the mean tide level, before they will colonize rapidly by seed or fragment. Once established, however, they will spread by rhizome extension and persist as low as mean tide level.
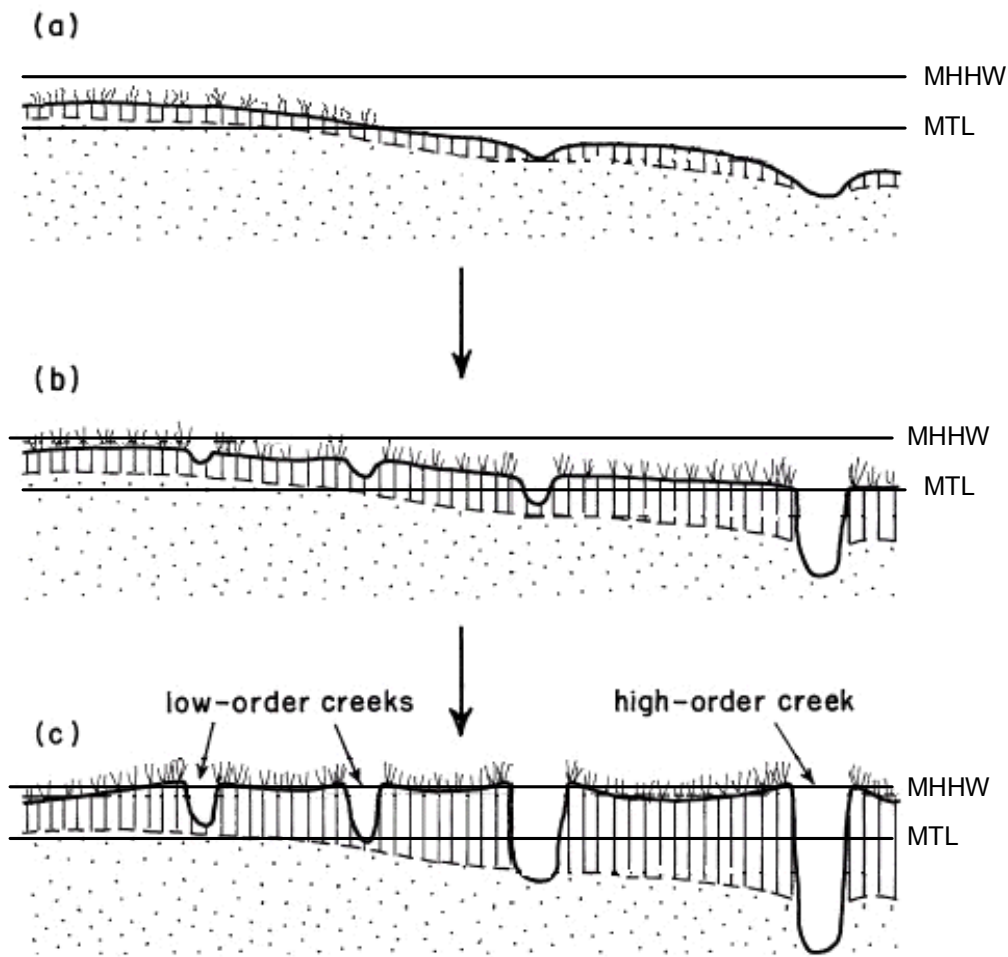

Figure 4 Schematic of youthful marsh development. (a) Young marsh low in the tidal frame with precursor mudflat and incipient channels; (b) Developing marsh showing more extensive vegetation colonization and channel formation; (c) Mature marsh with fully vegetated marsh plain and channel drainage system. MHHW = mean higher high water. MTL = mean tide level. Speckled fill is mud; hatched fill is marsh soil. Adapted from French (1993) and Allen (2000).

Restored marshes lower in the tidal frame than mature marshes are more susceptible to ecological drowning under low sediment or high sea level rise scenarios (Morris and others 2002). Also, creating restored marshes lower in the tidal frame requires that a considerable amount of accommodation space be filled by natural sedimentation as the marsh accretes to a fully developed mature state. Such an evolution requires the accretion of sediment at a rate above that of sea-level rise.

Restoration Approaches in the Delta

Because few tidal marsh restorations are in planning or have been completed in the Delta, there is not a restoration approach that can be characterized as typical. The few restorations or abandoned levee breaches on diked subsided lands have resulted primarily in the creation of shallow subtidal open water (e.g., Big Break, Mildred Island) or islands of vegetated marsh on fill mounds within open water areas (Donlon Island, Venice Cut, Cache Slough mitigation area), rather than vegetated marshplain with channels. A small project (6 ha) on Decker Island where tidal marsh and channels have been restored is atypical because the site was initially filled, not subsided.

Lands available for restoration in much of the Delta tend to be deeply subsided, more so than in the Bay, and well below elevations at which emergent vegetation can become established. On natural marshes, freshwater emergent vegetation generally grows within the intertidal zone and slightly below, extending to depths of 0.2 m below mean lower low water (Simenstad and others 2000; Hart and others 2003). Sedimentation rates in much of the Delta are low (Simenstad and others 2000), with particles kept in suspension by the slow settling velocities of the unflocculated fine sediments and high wind wave activity across large flooded sites. Because of this, even on shallowly subsided sites breached nearly 100 years ago in the central and west Delta insufficient sediment had accumulated for vegetation to become re-established (e.g., Sherman Lake and Big Break) (Simenstad and others 2000).

Thus, the restoration approach currently being pursued in the Delta is to use fill to raise the site to elevations at which emergent vegetation can establish prior to breaching (Schmutte and others 1998). With subsidence commonly on the order of 3 to 5 m, sources of fill are extremely limited compared to the volume needed for tidal marsh restoration in the Delta (PWA 2002; SSFBA 1999). Several traditional and alternative sources of fill are being evaluated, including dredged material, in situ biomass accretion, and alternating layers of vegetation and dredged material (SSFBA 1999; PWA 2002; Deverel 2003). Reclaimed areas in the Delta are vulnerable to further consolidation if loaded with fill material because of the low bearing capacity of the peat substrate (PWA 2002b). Because of the degree of reclamation-induced subsidence and the lower bearing capacity of substrate material, engineering the conditions for freshwater tidal marsh restoration in the Delta is considerably more difficult than creating similar template conditions for salt marsh habitat in San Pablo Bay.

To date, only the Decker Island restoration has included the construction of tidal channels. These channels were graded into consolidated fill. Excavation of channels on newly placed, low bearing capacity fill will require the use of specialized methods and equipment.

Anticipated Vertical Response

In defining whether marshes restored in the Estuary will be sustainable into the future a number of lines of evidence must be pulled together. It is clear from historic mapping (Atwater and Belknap 1980; Goals Project 1999) that marshes were well established during the late Holocene at a time when sediment fluxes to the Bay were at a rate lower than at present (Gilbert 1917). This indicates that once marshes have reached a mature state, they can maintain their relative tidal elevations under existing low rates of sea-level rise, even under conditions of low sediment availability.

Assuming that marshes are restored to the required elevation for halophyte colonization, what evidence is there to indicate that accretion would continue to keep pace with sea level rise and viable future salt marsh would result? In the following section we evaluate the vertical response of restored marshes to predicted changes in sea level and sediment supply using empirical data and accretion modeling.

San Pablo Bay

Empirical Evidence

Empirical accretion data are available from monitoring of several restored marshes (Table 1). Since accretion varies with marsh elevation such that low marshes receive more inorganic sedimentation, we present empirical accretion data for low and high marshes separately. Monitoring results from restored marshes suggest low marsh accretion rates on the order of 18 to 70 mm yr^{-1} (Table 1), with wide variations (8 to 80 mm yr^{-1} range). Data for three rapidly accreting high marsh areas suggest accretion rates on the order of 3 (at least) to 10 mm yr^{-1}. The range in data is considerable, 1 to 38 mm yr^{-1}. Other high marsh accretion data are available, but most only indicate that surface sedimentation is occurring at a rate similar to sea-level rise, averaging 1 to 2 mm yr^{-1} (Atwater and others 1979).

Several points must be borne in mind when considering the high accretion rates above. Short-term data such as this are not representative of older, more consolidated marsh deposits. Some of the periods of record are very short, 1 to 6 years. Sedimentation at Greenpoint and other sites at the mouth of the Petaluma River has been observed to be higher than at other marshes in San Pablo Bay. Methods that rely on monitoring accretion above buried marker horizons or surface plates do not account for sub-surface autocompaction and so tend to overestimate true elevation change. Where organic accumulation below ground is significant, sedimentation plate methods may fail to

recognize the importance of this contribution to marsh surface elevation change (see discussion in next section). Overall, empirical data do not tell us whether the marsh is maintaining its elevation relative to rising water levels. That said, these data also do not define the maximum potential for accretion as sea-level rises. At best the data may suggest a steady ability of the marshes to maintain or increase their elevation under existing conditions. Peyton Hill marsh, although a brackish marsh at the time of maximum accretion (3 mm yr^{-1}, approximately 3000 years before present) (Goman and Wells 2000), provides a point of comparison for the San Pablo Bay marshes. Another point of comparison, albeit for a more sediment-rich area (Beeman and Krone 1992), is the apparent ability of marshes in the Alviso Slough area of the South Bay to withstand high rates of submergence. These natural tidal marshes were until recently subject to several decades of high local subsidence on the order of 29 mm yr^{-1} (BCDC 1987). Today, these marshes are observed to support vegetation characteristic of high marsh.

Table 1 Marsh accretion rates for San Pablo Bay

Location	Low/ High Marsh	Dominant Vegetation	Accretion Rates (mm yr^{-1})	Source	Comments
Green Point Marsh, San Pablo Bay	Low	*Scirpus maritimus*	70 average (60 to 80 range)	Simenstad and others 2000	Core to pre-restoration soil horizon. Period = 16 years. N = 3.
White Slough	Low	*Scirpus maritimus*	25 average (20 to 33 range)	Simenstad and others 2000	Core to pre-restoration soil horizon. Period = 25 years. N = 4.
Muzzi Marsh, San Pablo Bay	Low	*Spartina foliosa, Salicornia virginica, Salicornia europea*	18 median (8 to 66 range)	PWA and Faber 2002	Sedimentation plates. Period = 1 to 6 years. N = 10. Dominant vegetation varied by year.
Muzzi Marsh, San Pablo Bay	High	*Salicornia virginica*	10 median (5 to 13 range)	PWA and PFA 2002	Sedimentation plates. Period = 4 to 11 years. N = 8.
Pond 2A, San Pablo Bay	High	*Salicornia virginica*	9 median (8 to 38 range)	MEC and others 2000	Sedimentation plates. Period = 1.6 years. N = 5.
Peyton Hill Marsh, Carquinez Straight	High	*Scirpus americanus* (brackish)	As high as 3 (1 to 3 range)	Goman and Wells (2000)	Historic stratigraphy. 3 mm yr^{-1} rate sustained over 200 years.

Note: Accretion rates for Green Point and White Slough marshes include initial period of mudflat accretion prior to vegetation establishment.

Accretion estimates from other United States and United Kingdom coastal marshes generally range between 0 and 16 mm yr^{-1} (Callaway and others 1996; Bricker-Urso and

others 1989). The range in values reflects the geomorphic setting with higher values recorded on developing immature marshes or in regions of high subsidence such as the Mississippi Delta. Bricker-Urso and others (1989) estimated maximum sustainable accretion rates of 14 to 16 mm yr^{-1} based on a study of Rhode Island salt marshes (*S. alterniflora* and *S. patens*). Nyman and others (1993) measured 10 mm yr^{-1} accretion averaged over nearly three decades for eleven marshes near the lower limits tolerated by the predominant vegetation, *S. alterniflora*. French (1993) estimated a maximum sustainable accretion rate of 4 mm yr^{-1} for minerogenic salt marshes in Norfolk (UK). All these values of maximum accretion potential depend upon the available sediment supply to the marsh.

Modeling Studies

The second line of evidence used to determine whether marshes will maintain their elevation relative to rising tidal waters is developed though predictive modeling. Here, we utilize a mass balance approach to analyze the vertical changes in both high (mature) and low (immature) marshplain elevations in response to sea level rise and under reasonable scenarios for suspended sediment concentrations. The model is a variant of that first developed by Krone (1987), relating mineral sedimentation to hydroperiod and suspended sediment concentration. We have adapted the Krone model to include a constant organic accretion rate, consistent with the approach of French (1993). We used a dry density of inorganic sediment equal to 500 kg m^{-3}, representative of low marsh soils as measured by Pestrong (1965) for South Bay marshes. We assumed a constant organic accretion rate of 1 mm yr^{-1}. This assumption is a general approximation based on soil density and organic content data collected by Collins and others (1987) for the ancient Petaluma marshes, cross-checked against a limited number of estimates for autumnal above ground dry weight biomass (Collins and others 1987; Atwater and others 1979).

The model is run under a range of steady-state sea-level rise conditions (1, 3, 5, 6 and 11 mm yr^{-1}) spanning the low, mid and high estimates of the IPCC (2001) scenarios. Tectonic subsidence, subsurface organic decomposition and autocompaction are incorporated implicitly but simplistically within these relative sea-level rates. Three conditions of sediment availability are modeled. The mid value (100 mg L^{-1}) roughly approximates average sediment currently available in San Pablo Bay (see previous discussion in Site Setting). High (150 mg L^{-1}) and low (50 mg L^{-1}) suspended sediment concentrations are intended to reflect a reasonable range of future variation. The low estimate represents potential future reductions in sediment supply from the declining hydraulic mining sediment pulse and possibly also from extensive marsh restorations sequestering sediment from general circulation. Initiated at both high (MHHW) and low (0.5 m below MHHW) marsh elevations, each combined sea-level rise and sediment availability scenario is run to predict the respective marsh surface after 100 years (Figure 5) and the long-term equilibrium marsh elevation (Figure 6).

The model outputs show that as sediment concentrations fall and/or rates of relative sea-level rise increase, the equilibrium marsh plain elevation falls relative to the tidal

frame. Under present conditions of sediment availability (c.100 mg L^{-1} within the water column) and rates of expected sea-level rise of 3 mm yr^{-1}, the model suggests that the high marsh plain will maintain its elevation at high marsh levels (Figure 5A). At rates of sea-level rise above 3 mm yr^{-1}, the mature marsh plain elevation slowly declines relative to MHHW and over the period of 100 years fall near or below the threshold of low/high marsh. It remains well above the "drowning" elevation, however, which we assume to be the low end of the elevation range for marsh vegetation, or approximately 1 m below MHHW.

Under conditions of greater sediment availability (150 mg L^{-1}), the model predicts that the marsh plain is capable of sustaining relative sea-level rise of up 6 mm yr^{-1} (Figure 6). Conversely, and importantly when considering future restoration actions, under conditions of reduced sediment availability (50 mg L^{-1}) the marsh plain elevation will decline significantly relative to the tidal frame over the next 100 years. While the falling sediment supply will not result in extensive marsh drowning by the year 2100—except under extremely high rates of sea-level rise above those modeled—it will result in greater areas of low marsh being created relative to high marsh. With continued low sediment availability, marshes continue to lower relative to the tidal frame after the year 2100 and drowning is predicted for the highest sea level rise conditions (Figure 6).

Evolution from low marsh elevations requires that considerable sedimentation takes place before an equilibrium marsh plain is attained. At mid-range sediment concentrations and rates of sea-level rise of 5 mm yr^{-1} or less, immature marshes will accrete to high marsh elevations over the next 100 years (Figure 5B). Under conditions of greater sediment availability (150 mg L^{-1}), the model predicts that the marsh plain is capable of rising from low to high marsh within 100 years with a sustained relative sea-level rise of up 6 mm yr^{-1} (Figure 6). The build up of centennial marsh around the margins of the Bay supports these model results. At higher rates of sea-level rise or lower availability of sediment the final elevation is suppressed and may never develop beyond low marsh. Again at low sediment supply the marsh elevation does not stabilize at an equilibrium level but will continue to subside relative to the tidal frame after AD 2100 for the higher sea level rise conditions.
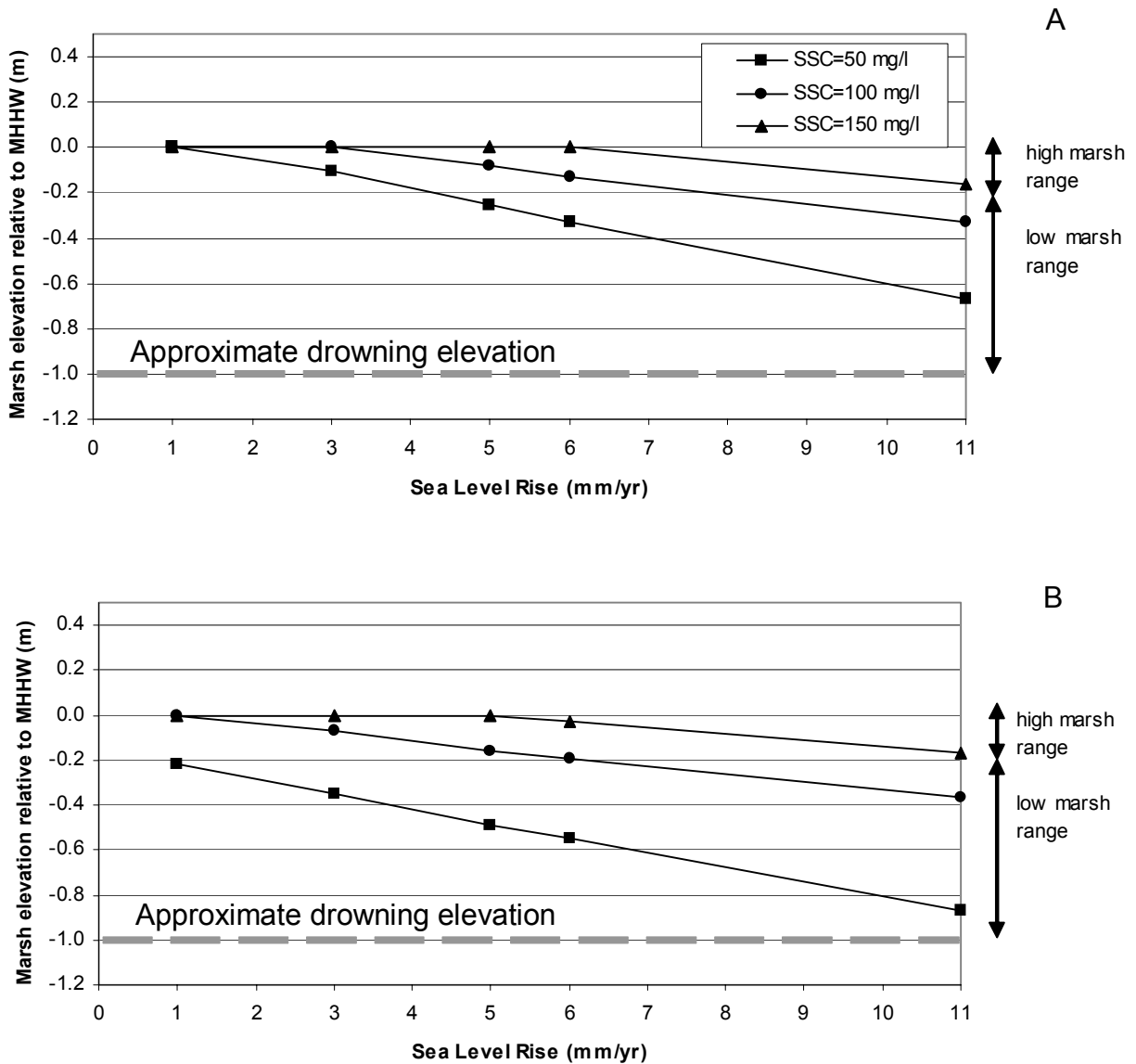
Figure 5 San Pablo Bay restored marsh elevation after 100 years as a function of sea level rise and suspended sediment concentration (SSC). (A) Initially starting at high marsh elevation (MHHW); (B) initially starting at low marsh (–0.5 m MHHW). Calculations based on model by Krone (1987) modified to include constant organic accumulation of 1 mm yr^{-1}. Dry density of inorganic accretion = 500 kg m^{-3}. Tides for Petaluma River mouth.
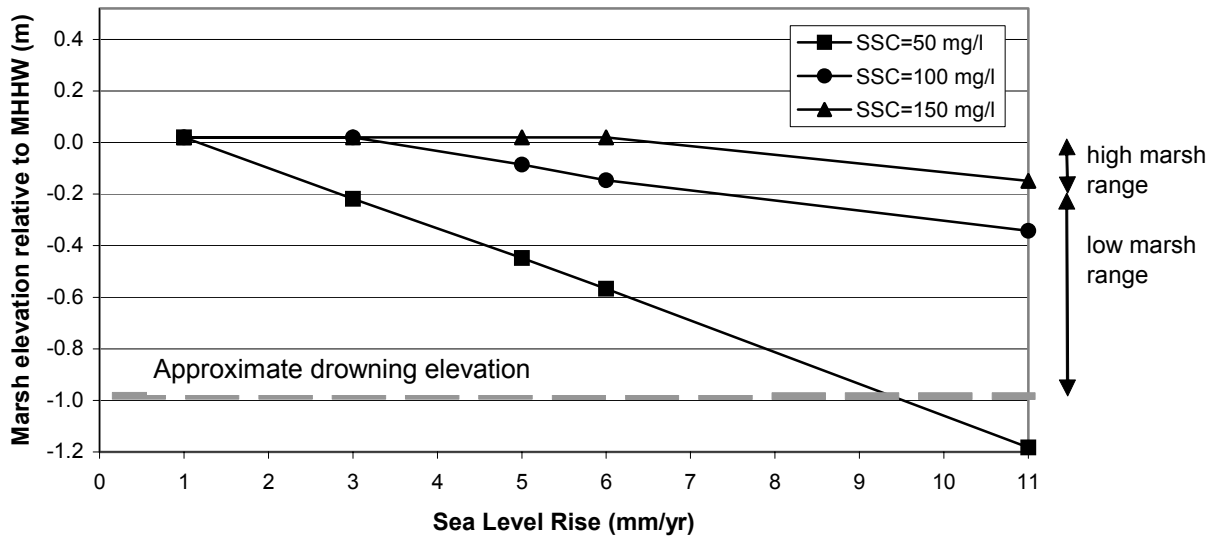
Figure 6 San Pablo Bay equilibrium marsh elevation as a function of sea level rise and suspended sediment concentration (SSC). See notes for Figure 5. Marsh elevations for the low suspended sediment scenario may be overestimated because the marsh did not reach an equilibrium elevation within the 500 years modeled.

When considering the results from this vertical accretion modeling study the role of plant production and other model parameters as uncertain variables should be made clear. Organic sedimentation is an important parameter in maintaining marsh elevations. It is, however, highly regionally variable and is poorly quantified in the Bay area. It is also currently unclear how the contribution of organic sedimentation varies as marshes accumulate from low marsh to high marsh conditions. Here we incorporated a reasonable estimate of constant organic sedimentation rate based upon very limited data of marsh organic accumulation. It is not certain how representative this value is of marshes around the Estuary, and so over- or under-estimations will have some implications for the accretion rate of the marsh surface. Inorganic sediment density is another source of model uncertainty. We selected a value consistent with limited marsh sediment measurements, but additional data are needed to characterize representative values and ranges. Further quantification of marsh carbon cycling and calibration of vertical accretion is required to reduce model uncertainty.

As a final note, the amount of accretion required to keep pace with rising sea level will be increased by the extent of sediment auto-consolidation beneath the marsh surface. All marshes are subject to ongoing consolidation of the underlying substrate (Skempton 1970; Callaway and others 1996; Crooks 1999), but consolidation can be greater for restored marshes than for natural marshes. Loosely consolidated muds that are deposited by estuarine sedimentation or imported to the site consolidate over the first few

years following restoration (Swent 2001; Siegel, personal communication, see "Notes"). Subsidence of fill material has not been included within this model. At the Hamilton Wetland Restoration, dredged material is expected to consolidate by some 0.3 m after placement (WCC and others 1998). Initially this is a rapid process, the rate of which declines with time but may continue for several months or years after tidal reactivation. Similar consolidation effects may well be transferred to underlying material if the site is loaded with significant quantities of material.

The Delta

Marshes in the Delta are sustained primarily by the accumulation of low density organic rich soils—typically 50% organic content (Atwater and Belknap 1980)—derived from surface vegetation growth (Atwater and others 1979). The predominant vegetation type is tule (*Scirpus acutus, S. californicus,* and *S. americanus*), with cattails (*Typha* sp.) and common reed (*Phragmite* spp.) (Atwater 1982). Apart from the north Delta where the Sacramento River supplies high levels of fluvial sediments, and the west Delta, which borders the marine mudflats, mineral sediments are transferred fairly efficiently through the system to Suisun and San Pablo bays. The vertical accumulation of these freshwater marsh plains depends primarily upon the accumulation rate of organic remains determined by a combination of the vegetation growth capacity and the potential for organic matter preservation once buried (Atwater 1982). This latter parameter is currently limiting natural marsh accumulation as organic material deposited above the water table decomposes under oxic conditions. Only after the material has been buried beneath the water table is significant preservation of previous accumulation possible.

The few available empirical data on Delta marsh accretion suggest accretion rates on low marsh of around 9 to 18 mm yr^{-1} (Goman and Wells 2000; D. Reed, personal communication, see "Notes"). Radiometric dating of cores by Reed (personal communication, see "Notes") provides accretion rates of 9 mm yr^{-1} at Sherman Lake (31-year average) and 18 mm yr^{-1} at Lower Mandeville Tip (18 year average). These are both low elevation restored marshes, 0.4 m and 0.6 m below mean higher high water, respectively (Simenstad and others 2000). Historic rates of accretion for natural (high elevation) marsh at Brown's Island have ranged as high as 3 to 4 mm yr^{-1} sustained over several decades in response to rapid sea level rise (Goman and Wells 2000). These historic rates of accretion were limited by the concurrent sea level rise, so do not necessarily represent the potential maximum. Additional short-term accretion data are available, but may not prove representative of longer-term rates. An ongoing study by the USGS to measure accretion in permanently flooded (managed) wetlands has found initial accretion rates of 26 mm yr^{-1} (with a wide variation) over a three year period (Drexler and others 2003). Monitoring of several restored Delta sites using Sediment-Erosion Tables (SETs), which provide elevation change including autocompaction and below ground biomass production (Boumans and Day 1993), suggest initial accretion rates that vary from nearly zero at Sherman Lake and Lower Mandeville Tip, to 6 to 12 mm yr^{-1} at Old Prospect West over a period of approximately two years (Reed 2002b).

The Sherman Lake and Lower Mandeville Tip accretion rates of 9 mm yr^{-1} and 18 mm yr $^{-1}$, respectively, are considered most representative of restored marsh conditions, though they will overestimate accretion of older, higher marshes. It therefore seems likely that accretion rates, sustained in large part by organic accretion at the vegetated surface, are sufficiently high to maintain established marsh elevation against relatively high rates of sea-level rise. If marsh surface lowering does occur, marshes in the Delta will be more resilient to drowning than those in San Pablo Bay because of the greater vertical range over which freshwater emergent vegetation persists. The low end of the vegetation range in the Delta is approximately to1.2 to 1.6 m below MHHW (compared to approximately 1 m below MHHW in San Pablo Bay).

If fill placement is used to restore marshes, consolidation of the fill material and underlying peat soils may be significant. Dredged material mounds placed on Donlon and Venice Cut Islands as part of marsh restoration settled by an average of 0.3 m (range: 0 to 0.6 m) in the two years after placement (analysis of data from USFWS and USACE 1990), which may help explain observed changes in vegetation type from higher to lower marsh species during this time period. A typical wetland restoration in the central Delta might require placement of 3 to 4 m of fill on 3 to 6 m of existing peat (CDWR 1995). If earth fill were used, subsidence would be considerable (Table 2; PWA 2002b). Investigations are currently underway to identify other sources of fill that will be more economical and less likely to cause subsidence (Schmutte and others 1998). Vulnerability to subsidence could result in the drowning of restored freshwater tidal marshes after restoration or in internal breakdown of marshes if differential subsidence occurs.

Table 2 Consolidation of underlying peat soils loaded with fill placed under saturated conditions

Thickness of Peat (m)	Depth of Fill (m)				
	1	2	3	4	5
1	0.01	0.17	0.26	0.33	0.38
2	0.02	0.34	0.52	0.65	0.76
3	0.03	0.51	0.79	0.98	1.13
4	0.04	0.68	1.05	1.31	1.51
5	0.05	0.85	1.31	1.64	1.89
6	0.06	1.02	1.57	1.96	2.27
7	0.07	1.20	1.84	2.29	2.64
8	0.08	1.37	2.10	2.62	3.02
9	0.09	1.54	2.36	2.95	3.40
10	0.10	1.71	2.62	3.27	3.78

Assumptions: $Cc/(1 + e_0) = 0.52$; Saturated Unit Weight = 10.2 kNm^2; pre-consolidation pressure at peat surface of 200 pounds per square foot (9.58 kNm^2); sand fill. Sources: K. Tillis and P. Carlston, personal communication, see "Notes." It is important to note the fill described in this table is placed under fully saturated conditions, meaning that the permanent watertable extends throughout the peat substrate to the surface of the deposited fill material. If the watertable were to fall such that the fill were to become partially or fully drained then the increase in effective loading would result in considerably increased amounts of peat consolidation. Table adapted from PWA 2002b.

Thus the primary problem facing wetland restoration in the Delta is not one of sustaining organic accumulation once a suitable marsh elevation has been achieved, but rather providing the elevations for marsh vegetation to initially become established. Large amounts of fill material will be needed to raise deeply subsided lands and, on this peat-rich substrate, may cause extreme substrate consolidation and marsh drowning.

Anticipated Marsh Planform Response

As described above, the resilience of coastal wetland systems is supported by their freedom to relocate in response to changing environmental conditions. Landward expansion of the Estuary's marsh plains was important for maintaining the extent of marsh and mudflat habitats by offsetting bayside losses due to sea-level rise and wind wave erosion (Atwater and others 1979). Today, levees and fills have created a "hard edge" at the upland margin of the Estuary that prevents the landward expansion of the marshes. Yet shoreline erosion is expected to continue, reducing the overall extent of restored and natural marsh. The planform (as seen looking down from above) sustainability of a particular marsh restoration will depend upon its location within the estuary and site specific features such as levee protection from off-shore waves.

On a natural marsh there is generally a gradual slope from high marshes down through mudflats to the subtidal, reflecting the balance between erosion and sedimentation processes. The large mudflats of San Pablo Bay are important for shoreline protection because they absorb wave energy before it reaches the marsh shore. The higher the mudflat elevations, the more wave energy is absorbed and the less marsh erosion occurs (Carter 1988; PWA 1993). Jaffe and others (1998) have estimated a net lowering of 6.5 mm yr^{-1} over the shallows of San Pablo Bay between 1951and 1983. Typical shoreline erosion for San Francisco Bay during this period was on the order of 0.9 m yr^{-1} (SFEP 1991). Acceleration of sea level rise and possible decreases in sediment supply will worsen the potential for future shoreline erosion.

Restoring marshes in sheltered environments behind levees protects the developing site from externally generated wave attack. While this approach of "breach-restoration" provides a sheltered location for a marsh to develop, the down side is that the linkage between the marsh and adjacent mudflat is reduced. In exposed locations, enhanced rates of erosion external to the site may induce a loss of adjacent mudflat habitat. Although shoreline erosion results in direct marsh loss, it also helps sustain the shallow mudflats and adjacent marsh. Some of the sediment released from the eroding marsh contributes directly or indirectly via mudflat storage to future accretion on nearby marsh surfaces (Reed 1988) and some is transferred to the adjacent subtidal mudflat to reduce the impinging wave energy (Pethick 1996).

At Muzzi Marsh (Figure 7) and White Slough, wave action is eroding the outer levee. Indeed, providing sacrificial rather than durable levees may be a desirable design procedure at some locations. For instance, over the first few decades of restoration a

sacrificial levee may provide a protective environment during which time a high marsh can become established. As the levee degrades, this marsh will naturally reconnect to the adjacent open shore where it can take upon those landscape-level geomorphic functions of providing space for estuarine expansion (roll-over), buffering sediment to the mudflat, as well as creating transitional habitat.

Figure 7 Aerial photograph showing erosion of the outer levee at Muzzi Marsh 22 years after restoration (1998 photograph)

Unlike the Bay, the Delta has no single large open water bay over which wind waves build up. However, wind waves can be a significant source of erosion on the downwind side of moderately sized (1,300 ha) flooded islands (or polders) such as Franks Tract and Sherman Lake. It has been observed that waves generated internally on the shallowly subsided flooded island of Frank's Tract (see Figure 2) cause erosion of the levee on the downwind side of the former tract. Beyond this levee, escaping waves have been observed to propagate beyond the island to induce substantial erosion of nearby natural marshes (Fox 1996).

Erosion caused by boat wakes (Bauer and others 2002) is a factor along the channels where many remnant marshes of the Delta are located. The restored Lower Mandeville Tip site (see Figure 2) has eroded at a rate of approximately 0.8 to 1.6 m yr^{-1} since 1937

(Simenstad and others 2000). The marsh plain at this site experiences direct wave attack from boat traffic on the Stockton Deepwater Ship Channel. Marsh areas adjacent to artificial channel cuts (navigation channels and external levee borrow pits) are particularly vulnerable to erosion because there are no natural or human-constructed levees adjacent to the cut to reflect or absorb wave energy. An additional erosion consideration in the Delta is that the relatively narrow and steep mudflats that separate marshes from deep channels offer little protection from steep waves generated by boats (BCDC 1988).

Channels in restored marshes are expected to evolve toward a configuration of dynamic equilibrium with tidal flows. Observations of restored channels that form from mudflat channels or that re-occupy historic marsh channels suggest that they can remain quite stable in their planform locations over time periods of decades. For example, comparison of historic and recent aerial photographs indicates that channels have been stable at the centennial marshes of China Camp, at outer (east) Muzzi Marsh where channels re-occupied historic locations, and Faber Tract where channels formed on placed dredged material. Evidence from mature natural marshes, for which a longer history is available, indicates that channel locations have been quite stable since the mid-1800s (Pestrong 1965; Grossinger 1995).

Restored channels may change rapidly when their initial configuration is out of balance with equilibrium processes. For example, personal observations of an oversized excavated channel at inner (west) Muzzi Marsh indicate that the channel filled in with sediment significantly within the first decade. Hydraulic modeling and equilibrium channel metrics developed for the San Francisco Estuary (e.g., Williams and others 2002; Simenstad and others 2000; PWA and others 1995; Grossinger 1995) can be used to assess the potential for rapid initial change of a restored channel. Not all adjustments towards equilibrium will be rapid, however. Inner Muzzi Marsh, parts of Faber Tract, and Pond 3 (see Figure 1)—all filled to high marsh elevations with minimal initial channel excavation—have yet to develop significant new channels through natural tidal scour over 20 to 30 years (Williams and Orr 2002). This is thought to be a factor of the small scouring flows (small hydraulic duty) at these sites and the high erosion resistance once marsh vegetation becomes established. Actual rates of channel adjustment at a given site will depend on a number of factors such as tidal velocities, sedimentation, and soil erodibility characteristics.

After any initial period of adjustment, changes in channel depth and width are expected to be minimal in restored marshes that keep pace with rising sea level, since tidal prism will remain relatively constant. If the marsh surface lowers relative to the tides, over marsh tidal prism will increase and lead to channel deepening and widening, the extent of which can be estimated from equilibrium channel geometry relationships (e.g., Williams and others 2002; Simenstad and others 2000; PWA and others 1995). One study from the UK indicates that channel density is also a function of relative marsh elevation, with maximum drainage density occurring on marsh plains that are slightly above mean high tide (Steel and Pye 1999). Above this optimal elevation, channel density was

observed to decrease, presumably from reduced tidal scour insufficient to prevent infilling of the smallest channels with sediments. The data set used to develop these marsh drainage-elevation relationships is relatively small, and additional systems should be evaluated for a more thorough test.

Conclusions

1. The sustainability of restored tidal marshes is dependent upon the balance between relative sea-level rise and sediment supply. Vertical accretion modeling of marshes in San Pablo Bay suggests that high marshes will be sustainable for median values of predicted sea level rise (3 to 5 mm yr^{-1}) and moderate sediment availability (c. 100 mg L^{-1}). Salt marshes restored at lower elevations necessary to aid the natural development of channel systems (modeled at 0.5 m below mean higher high water) will gain high marsh elevations within 100 years for these moderate conditions. As sediment concentrations fall (50 mg L^{-1}) and/or relative sea-level rise accelerates (6 to 11 mm yr^{-1}), marsh elevations will become lower relative to the tidal frame and predominant vegetation types may change as a result. Widespread marsh drowning is not expected. Under conditions of greater sediment availability (150 mg L^{-1}), the model predicts that the marsh plain is capable of sustaining relative sea-level rise of up to 6 mm yr^{-1}.

2. Empirical accretion data for restored San Pablo Bay marshes indicate that existing accretion rates are 18 to 70 mm yr^{-1} for low marsh and 9 to 10 mm yr^{-1} for high marsh. These rates are representative of sedimentation sustained over years to decades. If these rates are halved to reflect potential decreases in sediment supply and/or long-term soil consolidation, they still outpace moderate rates of sea level rise (3 to 5 mm yr^{-1}), with low marshes more resilient than high marshes.

3. Fresh water tidal wetlands in the Delta are expected to be more resilient to enhanced rates of sea-level rise and decreases in sediment supply than are the salt marshes of San Pablo Bay because of higher rates of organic production and slightly lower drowning elevations. Empirical data indicate existing accretion rates of roughly 9 to 18 mm yr^{-1} for low elevation restored marshes. Consolidation of fill material, if used, and the underlying peat soils will need to be considered in the restoration design to avoid possible marsh drowning.

4. Restored tidal marshes are dynamic estuarine landforms and their sustainability will depend on landscape scale processes. The ultimate long-term threat to the sustainability of tidal marshes is the presence of perimeter levees and fill which prevent the landward expansion of marshes as the shoreline erodes. The rate of shoreline erosion will likely be accelerated by increasing relative sea level rise and continued lowering of the mudflats.

Acknowledgements

References

Allen JRL. 1990a. The formation of coastal peat marshes under an upwards tendency of relative sea-level. Journal of the Geological Society of London 147:743-745.

Allen JRL. 1990b. Salt-marsh growth and stratification: a numerical model with special reference to the Severn Estuary, southwest Britain. Marine Geology 95:77-96.

Allen JRL. 1990c. The Severn Estuary in southwest Britain: its retreat under marine transgression, a fine sediment regime. Sediment Geology 66:13-28.

Allen JRL. 1995. Salt-marsh growth and fluctuating sea-level: Implications of a simulation model for Holocene coastal stratigraphy and peat-based sea-level curves. Sediment Geology 100:21-45.

Allen JRL. 1997. Simulation models of salt-marsh morphodynamics: Some implications for high-intertidal sediment couplets related to sea-level change. Sediment Geology 113:211-223.

Allen JRL. 2000. Morphodynamics of Holocene salt marshes: a review sketch from the Atlantic and southern North Sea coasts of Europe. Quaternary Science Review 19(12):1155-1231.

Atkinson PW, Crooks S, Grant A, Rehfisch MM. 2001. The success of creation and restoration schemes in producing intertidal habitat for waterbirds. English Nature Research Report 425.

Atwater BF. 1982. Geologic maps of the Sacramento-San Joaquin Delta, California. Menlo Park (CA): U.S. Geological Survey MF-1401.

Atwater BF, Belknap DF. 1980. Tidal-wetland deposits of the Sacramento-San Joaquin Delta, California. In: Field ME, Bouma AH, Colburn IP, Douglas RG and Ingle JC, editors. Quaternary depositional environments of the Pacific Coast. Pacific Coast Paleogeography Symposium 4. Proceedings of the Society of Economic Paleontologists and Mineralogists. Los Angeles (CA): Society of Economic Paleontologists and Mineralogists. p 89-103.

Atwater BF, Conard SG, Dowden JN, Hedel CW, MacDonald RL, Savage W. 1979. History, landforms, and vegetation of the estuary's tidal marshes. In: Conomos TJ, Leviton AE, Berson M, editors. San Francisco Bay: the urbanized estuary. San Francisco (CA): Pacific Division, AAAS. p 347-385.

Atwater BF, Hedel CW. 1976. Distribution of seed plants with respect to the tide levels and water salinity in the natural tidal marshes of the northern San Francisco Bay estuary, California. Preliminary report. U.S. Geological Survey Open File Report 76-389.

Bard E. 1998. Geochemical and geophysical implications of the radiocarbon calibration. Geochimica et Cosmochimica Acta 62(12):2025-2038.

Bauer BO, Lorang MS, Sherman DJ. 2002. Estimating boat-wake-induced levee erosion using sediment suspension measurements. Journal of Waterway, Port, Coastal and Ocean Engineering 128(4):152-162.

Bayliss-Smith TP, Healey R, Lailey R, Spencer T, Stoddart DR. 1979. Tidal flow in salt marsh creeks. Estuarine and Coastal Marine Science 9:235-255.

[BCDC] San Francisco Bay Conservation and Development Commission. 1987. Sea level rise: predictions and implications for San Francisco Bay. San Francisco (CA): San Francisco Bay Conservation and Development Commission. December 1987. Report revised October 1988.

[BCDC] San Francisco Bay Conservation and Development Commission. 1988. Protecting shoreline property from tidal erosion: an analysis of the effectiveness and environmental impacts of administratively authorized protective structures. San Francisco (CA): San Francisco Bay Conservation and Development Commission. 86+ p.

Beeby A. 1993. Applied ecology. London: Chapman & Hall. 441 p.

Beeftink WG, Rozema J. 1988. The nature and functioning of salt marshes. In: Salomans W, Bayne BL, Duursma EK, Förstner U, editors. Pollution of the North Sea: an assessment. Berlin: Springer. p 59-87.

[Beeman and Krone] Ogden Beeman & Associates, Inc., RB Krone & Associates, Inc. 1992. Sediment budget study for San Francisco Bay. Final report. Prepared for the San Francisco District, U.S. Army Corps of Engineers. Portland (OR): Ogden Beeman & Associates, Inc. 25+ p.

Boumans RM, Day JW. 1993. Measurement of small elevation changes in shallow coastal areas using a sedimentation-erosion table. Estuaries 16:375-80.

Bricker-Urso S, Nixon SW, Cochran JK, Hirschberg DJ, Hunt C. 1989. Accretion rates and sediment accumulation in Rhode Island salt marshes. Estuaries 12:300-317.

Brown LR. 2003. Will tidal wetland restoration enhance populations of native fishes? In: Brown LR, editor. Issues in San Francisco Estuary tidal wetlands restoration. San Francisco Estuary and Watershed Science [online serial publication]. Vol. 1, Issue 1 (2003), Article 2. Available at: http://repositories.cdlib.org/jmie/sfews/vol1/iss1/art2

Buchanan PA, Ruhl CA. 2000. Summary of suspended-solids concentration data, San Francisco Bay, California, water year 1998. U.S. Geological Survey Open File Report 00-88. Prepared in cooperation with the San Francisco Regional Water Quality Control Board and the U.S. Army Corps Of Engineers, San Francisco District. Denver (CO): USGS Information Services. 41 p.

Buchanan PA, Ruhl CA. 2001. Summary of suspended-sediment concentration data, San Francisco Bay, California, water year 1999. U.S. Geological Survey Open File Report 01-100. Prepared in cooperation with the CALFED Bay-Delta Program, the San Francisco Regional Water Quality Control Board, and U.S. Army Corps of Engineers, San Francisco District. District. Denver (CO): USGS Information Services. 40 p.

Buchanan PA, Schoellhamer DH. 1998. Summary of suspended-solids concentration data, San Francisco Bay, California, water year 1996. U.S. Geological Survey Open File Report 98-175. 59 p.

Buchanan PA, Schoellhamer DH. 1999. Summary of suspended-solids concentration data, San Francisco Bay, California, water year 1997. U.S. Geological Survey Open File Report 99-189. 52 p.

[CALFED] CALFED Bay-Delta Program. 2000. Ecosystem restoration program plan final programmatic EIS/EIR. Sacramento (CA): CALFED Bay-Delta Program. Accessed 2003 Aug 28. Available from: http://calwater.ca.gov/CALFEDDocuments/Final_EIS_EIR.shtml

Callaway JC, Nyman JA, DeLaune RD. 1996. Sediment accretion in coastal wetlands: a review and a simulation model of processes. Current Topics in Wetland Biogeochemistry 2:2-23.

Cappiella K, Malzone C, Smith R, Jaffe B. 1999. Sedimentation and bathymetry changes in Suisun Bay, 1867-1990. U.S. Geological Survey Open-File Report 99-563.

Carter RWG. 1988. Coastal environments: an introduction to physical, ecological and cultural systems of coastline. London: Academic Press. 617 p.

[CDWR] California Department of Water Resources. 1995. Sacramento-San Joaquin Delta Atlas. Sacramento (CA): California Department of Water Resources. 121 p.

Chmura GL, Costanza R, Kosters EC. 1992. Modelling coastal marsh stability in response to sea level rise: a case study in coastal Louisiana, USA. Ecological Modelling 64:47-64.

Collins LM, Collins JN, Leopold LB. 1987. Geomorphic processes of an estuarine marsh: preliminary results and hypotheses. In: Gardner V, editor. International geomorphology 1986, part I. New York (NY): J Wiley & Sons. p 1049-1071.

Conomos TJ, Smith RE, Gartner JW. 1985. Environmental setting of San Francisco Bay. In: Cloern JE, Nichols FH, editors. Temporal dynamics of an estuary: San Francisco Bay. Reprinted from Hydrobiologia, vol. 129 (1985). Boston (MA): Dr. W. Junk Publishers. p 1-12.

Crooks S. 1999. Formation of overconsolidated horizons within estuarine alluvium: implications for the interpretation of Holocene sea-level curves. In: Marriot S, Alexander J, Hey R, editors. Floodplains: interdisciplinary approaches. Geological Society Special Publication 163:197-215.

Crooks S, Turner RK. 1999. Coastal zone management: sustaining estuarine natural resources. Advances in Ecological Research 29:241-291.

Deverel SJ. 2003. Reversing the effects of subsidence in the Sacramento-San Joaquin Delta, California [abstract]. In: CALFED Science Conference 2003 Abstracts. CALFED Science Conference 14-16 Jan 2003, Sacramento Convention Center. Sacramento (CA): CALFED Bay-Delta Program. p 39.

Diekmann R, Osterhun M, Psartensky HW. 1988. A comparison between German and North American tidal inlets. Proceedings of the 21st coastal engineering conference, vol. 3. New York (NY): American Society of Civil Engineers. p 2681-2691.

Drexler JZ, Fleck J, Wheeler G, Fujii R. 2003. Carbon storage and sediment accretion in restored wetlands on Twitchell Island in the Sacramento-San Joaquin Delta, California [abstract]. In: CALFED Science Conference 2003 Abstracts. CALFED Science Conference 14-16 Jan 2003, Sacramento Convention Center. Sacramento (CA): CALFED Bay-Delta Program. p 46.

Eisma D, Dijkema KS. 1997. The influence of salt marsh vegetation on sedimentation. In: Eisma D, editor. Intertidal deposits. Boca Raton (FL): CRC Press. p 403-414.

Fox PJ. 1996. Phyllis Fox & Associates. Memorandum to Elain Archibald, Gary Bardini, Grace Chan, Sharon Kramer, BJ Miller and Jud Monroe on Delta shallow water habitat.

French JR. 1991. Eustatic and neotectonic controls on salt-marsh sedimentation. In: Knaus NC, Gingrich KJ, Krieble DL, editors. Coastal sediments 1991. New York (NY): American Society of Civil Engineers. p 1223-1236.

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

French JR, Stoddart DR. 1992. Hydrodynamics of salt marsh creek systems - implications for marsh morphological development and material exchange. Earth Surface Processes and Landforms 17:235-252.

Gilbert GK. 1917. Hydraulic mining in the Sierra Nevada. U.S. Geological Survey Professional Paper 105. Washington, DC: Government Printing Office.

Goals Project. 1999. Baylands ecosystem habitat goals. A report of habitat recommendations prepared by the San Francisco Bay Area Wetlands Ecosystem Goals Project. U.S. Environmental Protection Agency, San Francisco, California; San Francisco Bay Regional Water Quality Control Board, Oakland, California. 209+ p.

Goman M, Wells L. 2000. Trends in river flow affecting the northeastern reach of the San Francisco Bay Estuary over the past 7000 years. Quaternary Research 54:206-217.

Grossinger RM. 1995. Historical evidence of freshwater effects on the plan form of tidal marshlands in the Golden Gate Estuary [MS thesis]. Santa Cruz (CA): University of Santa Cruz, Department of Marine Sciences. 130 p.

Hart JA, Hunter JC, O'Brien J, Fleming HT. 2003. Interspecific differences in growth of *Scirpus* spp. (tules) across a depth gradient [abstract]. In: CALFED Science Conference 2003 Abstracts. CALFED Science Conference 14-16 Jan 2003, Sacramento Convention Center. Sacramento (CA): CALFED Bay-Delta Program. p 72.

Hastings LL, Castleberry D. 2003. An overview of CALFED Ecosystem Restoration Program plans for restoring Delta flooded islands [abstract]. In: CALFED Science Conference 2003 Abstracts. CALFED Science Conference 14-16 Jan 2003, Sacramento Convention Center. Sacramento (CA): CALFED Bay-Delta Program. p 73.

Hume TM, Herdendorf CE. 1993. On the use of empirical stability relationships for characterising estuaries. Journal of Coastal Research 9:413-422.

Ikehara ME. 1994. Global positioning system surveying to monitor land subsidence in Sacramento Valley, California, USA. Hydrological Sciences Journal 39(5):417-429.

[IPCC] Intergovernmental Panel on Climate Change. 2001. Climate change 2001: the scientific basis. Houghton JT, Ding Y, Griggs DJ, Noguer M, van der Linden PJ, Dai X, Maskell K, Johnson CA, editors. Cambridge: Cambridge University Press. 881 p.

Jaffe BE, Smith RE, Torresan L. 1998. Sedimentation and bathymetric change in San Pablo Bay, 1856-1983. U.S. Geological Survey Open-File Report 98-759.

Knighton D. 1998. Fluvial form and processes. London (UK): Arnold. 383 p.

Krone RB. 1979. Sedimentation in the San Francisco Bay system. In: Conomos TJ, Leviton AE, Berson M, editors. San Francisco Bay: the urbanized estuary. San Francisco (CA): Pacific Division, AAAS.

Krone RB. 1985. Simulation of marsh growth under rising sea levels. In: Waldrop WR, editor. Hydraulics and hydrology in the small computer age. Reston (VA): Hydraulic Division, American Society of Civil Engineers. p 106-115.

Krone RB. 1987. A method for simulating historic marsh elevations. In: Kraus NC, editor. Coastal sediments 1987. New York (NY): American Society of Civil Engineers. p 316-323.

Lowe JP. 2000. Geomorphological investigations for the rivers Crouch and Roach. Report to the Environment Agency (Anglian Region). Coastal Geomorphology Partnership, University of Newcastle.

[MEC and others] MEC Analytical Systems, Inc., Harding-Lawson & Associates, Chaudhary & Associates, Philip Williams & Associates. 2000. Baseline monitoring of the pond 2A tidal restoration project. Final report July 1996 - July 2000. Prepared for California Department of Fish and Game, Wildlife Management, Region 3, Napa, California. Carlsbad (CA): MEC Analytical Systems. 22 p.

Mitsch WJ, Gosselink JG. 1986. Wetlands. New York (NY): Van Nostrand Reinhold Co. 537 p.

Morris JT, Sundareshwar PV, Nietch CT, Kjerfve B, Cahoon DR. 2002. Response of coastal wetlands to rising sea level. Ecology 83:2869-2877.

[NRC] National Research Council. 2001. Compensation for wetland loss under the Clean Water Act. Washington: National Academic Press. 322 p.

Nyman JA, DeLaune RD, Roberts HH, Patrick WH Jr. 1993. Relationship between vegetation and soil formation in a rapidly submerging coastal marsh. Marine Ecology Progress Series 96:269-279.

O'Brien MP. 1931. Estuary tidal prisms related to entrance areas. Journal of Civil Engineering 1:738-739.

Oltmann RN, Schoellhamer DH, Dinehart RL. 1999. Sediment inflow to the Sacramento-San Joaquin Delta and San Francisco Bay. Interagency Ecological Program Newsletter.12(1):30-33. Available at: http://www.iep.ca.gov/report/newsletter

Pearce DW, Warford JJ. 1993. World without end: economics, environment and sustainable development. Oxford: Oxford University Press. 440 p.

Pestrong R. 1965. The development of drainage patterns on tidal marshes. Geological Sciences, vol X, nr 2. School of Earth Sciences. Stanford (CA): Stanford University Publications. 87 p.

Pethick JS. 1996. The geomorphology of mudflats. In: Nordstrom KF, Roman CT, editors. Estuarine shores: evolution, environment and human health. Chichester, UK: John Wiley. p 185-211.

Pethick JS, Crooks S. 2000. Development of a coastal vulnerability index: a geomorphological perspective. Environmental Conservation 27:359-367.

Pethick JS, Crooks S. 2001. Rivers Crouch and Roach: estuarine modelling. Study 3: Geomorphological implications of managed retreat on to flood compartments. Report to the Environment Agency Specialist Term Consultancy STCG/2000/51. Available from: Centre for Coastal Management, University of Newcastle.

[PWA] Philip Williams & Associates Ltd. 1993. An analysis of the relative effect of ferry wakes on shoreline erosion at the Corte Madera Ecological Reserve. Prepared for the Golden Gate Bridge, Highway and Transportation District. PWA report #847/877. San Francisco (CA): PWA. 45+ p.

[PWA and others] Philip Williams & Associates Ltd., Pacific Estuarine Research Lab, Louisiana Universities Marine Consortium. 1995. Design guidelines for tidal channels in coastal wetlands. Prepared for the U.S. Army Corps of Engineers Waterways Experiment Station. PWA report #934. San Francisco (CA): PWA. 46+ p.

[PWA] Philip Williams & Associates Ltd. 2002a. Napa River salt marsh restoration: habitat restoration preliminary design, phase 2, stage 2 of the hydrology and geomorphology assessment in support of the feasibility study. Prepared for the California State Coastal Conservancy. November. PWA report #1591. In: California State Coastal Conservancy, U.S. Army Corps of Engineers, and California Department of Fish and Game. Napa River Salt Marsh Restoration Project Feasibility Report. Appendix D. Accessed 2003 Aug 25. 25+ p. Available from:
http://www.napa-sonoma-marsh.org/documents/Feasibility/feasibility.html

[PWA] Philip Williams & Associates Ltd. 2002b. Demonstration of techniques for reversing the effects of subsidence in the Sacramento-San Joaquin Delta: assessment of elevation building techniques for large-scale tidal wetland restoration. PWA report #1407. Prepared for California Department of Water Resources. San Francisco (CA): PWA. 30+ p.

[PWA and others] Philip Williams & Associates, Ltd., ENTRIX, Gahagan & Bryant, SP Surveys, Phyllis Faber & Associates, Avocet Research Associates, Stuart Siegel Wetlands Water Resources. 1999. 1997-1998 annual monitoring report Sonoma baylands wetland demonstration project. PWA report #1269. Prepared for the Department of the Army San Francisco District Corps of Engineers, San Francisco, CA. San Francisco (CA): PWA. 57+ p.

[PWA and PFA] Philip Williams & Associates, Ltd., Phyllis Faber & Associates. 2002. Monitoring the evolution of restored tidal marshes in San Francisco Bay: physical and vegetation monitoring data for Warm Springs, Muzzi and China Camp marshes from 1981 to 2002. PWA report #1269. Prepared for The Bay Institute, San Rafael, CA. San Francisco (CA): PWA. 41+ p.

Randerson PF. 1979. A simulation model of salt-marsh development and plant ecology. In: Knights B, Phillips AJ, editors. Estuarine and coastal land reclamation and water storage. Farnborough: Saxon House. p 48-57.

Reed DJ. 1988. Sediment dynamics and deposition in a retreating coastal marsh. Estuarine, Coastal, and Shelf Science 26:201-237.

Reed DJ. 2002a. Sea-level rise and coastal marsh sustainability: geological and ecological factor in the Mississippi delta plain. Geomorphology 48:233-243.

Reed DJ. 2002b. Understanding tidal marsh sedimentation in the Sacramento-San Joaquin Delta, California. Journal of Coastal Research Special Issue 36:605-611.

Ruhl CA, Schoellhamer DH. 2000. Spatial and seasonal variability of suspended sediment concentration in Honker Bay, a shallow subembayment of San Francisco Bay [abstract]. CALFED Bay-Delta Program Science Conference, 3-5 Oct 2000, Sacramento Convention Center. Sacramento (CA): CALFED Bay-Delta Program. p 218.

Schmutte C, MacArthur R, Hall B, Deverel S, Fuji R, Thomas G, Cain J, Orr M, and Williams P. 1998. A learning laboratory for restoring subsided lands in the Delta, phase I. Proposal to the CALFED Ecosystem Restoration Program.

[SFEP] San Francisco Estuary Project. 1991. Status and trends report on wetlands and related habitats in the San Francisco Estuary. Prepared under cooperative agreement #815406-01-0 with the U.S. Environmental Protection Agency by the Association of Bay Area Governments, Romberg-Tiburon Center, and U.S. Fish and Wildlife Service. Oakland (CA): SFEP. 209 p.

Siegel SW. 1998. Petaluma River marsh monitoring report, 1994 to 1998. Prepared for the Sonoma Land Trust, Santa Rosa, CA. San Rafael (CA): Wetlands and Water Resources. 38 p.

Simenstad C, Toft J, Higgins H, Cordell J, Orr M, Williams P, Grimaldo L, Hymanson Z, Reed D. 2000. Sacramento-San Joaquin Delta breached levee wetland study (BREACH). Preliminary report. Seattle (WA): Wetland Ecosystem Team, University of Washington, School of Fisheries. 46 p.

Simenstad CA, Warren RS, editors. 2002. Special issue on dike/levee breach restoration of coastal marshes. Restoration Ecology 10(3).

Skempton AW. 1970. The consolidation of clays by gravitational compaction. Quarterly Journal of the Geological Society of London 125:373-411.

[SSFBA] Save San Francisco Bay Association. 1999. Opportunities for Delta reuse of clean material dredged from San Francisco Bay. Phase II report: conceptual design. Prepared for the California Coastal Conservancy's Dredge Material Reuse Program with support provided by the Pew Charitable Trusts. Oakland (CA): SSFBA. 57 p.

Steel TJ, Pye K. 1999. The development of salt marsh tidal creek networks: evidence from the U.K. Proceedings of the Canadian coastal conference, 1997 May 22-25: University of Guelph, Canada. Canada: University of Guelph. p 267-280.

Steere JT, Schaefer N. 2001. Restoring the estuary: implementation strategy of the San Francisco Bay Joint Venture. Oakland (CA): San Francisco Bay Joint Venture.

Sutherland WJ, editor. 1998. Conservation science and action. London: Blackwell Science. 363 p.

Swent JL. 2001. Stevens Creek tidal marsh restoration monitoring report. Prepared for Paula Bettencourt, Deputy Director, Community Services Department, City of Mountain View, Shoreline Division, Mountain View, CA. Prepared by Janice Leah Swent, Environmental Consultant, Sabastopol, CA.

[SWS] Society of Wetlands Scientists. 2000. Position paper on the definition of wetland restoration. Washington USA.

[TBI] The Bay Institute. 1998. From the Sierra to the sea: the ecological history of the San Francisco Bay-Delta Watershed. San Rafael (CA): The Bay Institute of San Francisco. 240+ p.

Temmerman S, Govers G, Meire P, Wartel S. 2003. Modelling long-term tidal marsh growth under changing tidal conditions and suspended sediment concentrations, Scheldt estuary, Belgium. Marine Geology 193:151-169.

Turner RK, editor. 1993. Sustainable environmental economics and management: principles and practice. London: Belhaven Press. 389 p.

[USFWS and USACE] U.S. Fish and Wildlife Service (Sacramento) and U.S. Army Corps of Engineers (Sacramento District). 1990. Design and biological monitoring of wetland riparian habitats created with dredged materials. Final report. Deepwater Ship Channel monitoring program. Sacramento (CA): Sacramento District Corps of Engineers. 68 p.

Warner JC. 2000. Barotropic and baroclinic convergence zones in tidal channels [PhD dissertation]. Davis (CA): University of California, Davis, Civil and Environmental Engineering. 333 p. Available from: University of California, Davis.

[WCC and others] Woodward Clyde Consultants, Philip Williams & Associates, HT Harvey & Associates, Polson E. 1998. Draft Hamilton Wetlands conceptual restoration plan. Prepared for the California State Coastal Conservancy and the City of Novato. April 24. Oakland (CA): California State Coastal Conservancy. Oakland (CA): California State Coastal Conservancy. 113 p.

[WCED] World Commission on Environment and Development. 1987. Our common future. Oxford (UK): Oxford University Press. 37 p

Williams PB. 1989. The impacts of climate change on the salinity of San Francisco Bay. In: Smith JB, Tirpak DA, editors. The potential effect of global climate change on the United States. Appendix A, Water Resources. Prepared for the United States Environmental Protection Agency, Office of Policy Planning and Evaluation. EPA-230-05-89-051. Washington DC: US EPA. p 3-1 to p 3-29.

Williams PB, Faber PM. 2001. Salt marsh restoration experience in the San Francisco Bay Estuary. Journal of Coastal Research Special Issue 27:203-211.

Williams PB, Orr MK. 2002. Physical evolution of restored breached levee salt marshes in the San Francisco Bay estuary. Restoration Ecology 10:527-542.

Williams PB, Orr MK, Garrity NJ. 2002. Hydraulic geometry: a geomorphic design tool for tidal marsh channel evolution in wetland restoration projects. Restoration Ecology 10:577-590.

Wright SA, Schoellhamer DH. 2003. Trends in the sediment yield of the Sacramento River, 1957-2001 [abstract]. In: CALFED Science Conference 2003 Abstracts. CALFED Science Conference 14-16 Jan 2003, Sacramento Convention Center. Sacramento (CA): CALFED Bay-Delta Program. p 177.

Zedler JB, editor. 2001. Handbook for restoring tidal wetlands. New York (NY): CRC Press. 439 p.

Zeff ML. 1999. Salt marsh tidal channel morphology: applications for wetlands creation and restoration. Restoration Ecology 7:205-211.

Notes

Reed DJ. 2001. University of New Orleans. Unpublished data from ^{210}Pb analysis of cores at seven marshes in the Sacramento-San Joaquin River Delta. For the BREACH project.

Siegel SW. 2001. Telephone conversation with the author regarding site evolution at the Petaluma River Marsh restoration site (aka Carl's Marsh).